RESERVOIRS AND THEIR MANAGEMENT: A REVIEW OF THE LITERATURE SINCE 1990

Report to the
Water Research Commission

by

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Report No.: KV173/06
ISBN No.: 1 -77005-418-9

FEBRUARY 2006

This Report is obtainable from:

Water Research Commission
Private Bag X03
Gezina Pretoria
0031

This report emanates from the WRC research consultancy K8/612 titled:
REVIEW OF RESERVOIR MANAGEMENT

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EXECUTIVE SUMMARY

Chapter 1. General introduction.

South Africa is hugely dependent on impounded water, captured and stored in river-reservoirs or "dams" as they are colloquially known. These man-made lake ecosystems are complex entities that exhibit broad parallels with natural lakes, but also show fundamentally distinctive attributes that are themselves subject to local and regional particularities and pecularities.

Any realistic prospect for the sustainable management of the renewable fresh water resource that they store depends on an adequate and locally relevant underpinning of scientific limnological comprehension. Although available in good measure several decades ago, such operational functionality is now gravely deficient in South Africa. Some historical back-ground to the origin of the presently untenable, illogical and incomprehensible deficiency of locally informed research-based expertise to manage this most limiting resource rigorously is briefly sketched. This deficiency certainly merits growth of the nascent perception, recognition and concern evident by informed members of society. This very real need for scientifically informed limnological management of reservoirs serves as the point of departure for this report, which considers and addresses current and emerging issues in respect of reservoir management. Based on a desk-top search of primary international literature (principally but not exclusively since 1990), this report provides a systematic overview and review of current and emerging concerns in respect of reservoir management, drawn from inspection and consideration of approximately 1450 publications (as detailed in the Reference List and Bibliography – Chapter 6).

Chapter 2. Reservoirs as ecosystems – reservoir limnology with respect to water quality management.

It is axiomatic that rational management presumes relevant comprehension which in turn builds on objective basic knowledge. This section provides a concise overview of the functional and operational attributes of reservoirs that variously influence, modify, or determine the quantity, quality and/or sustainability of the impounded resource. This treatment is intended to provide a contextually factual and objective basis of understanding to facilitate rational management. It serves as a first pointer to factors that underpin issues of concern and relevance to reservoir management, and the implicit role for functional process-based limnologists to serve in that regard.

The physical, chemical and biological structures, functions, and interactions that underpin the natural ecology and utilitarian operation of reservoir systems are outlined. The importance of an holistic and integrated comprehension is stressed. As provision of a coherent self-standing summary of the material content addressed in this chapter is impractical, the interested reader is requested to examine the relevant full text account given in the chapter.

Chapter 3. Factors affecting water quality of reservoirs, their consequences and their management.

This section systematically considers the major categories of factors recognized by contemporary aquatic scientists as having influence or impact on reservoir water quality (in its widest sense) and its management, and outlines contemporary perspectives on the treatment or mitigation of such factors.

As for the preceding chapter, readers are advised of the impracticality of providing a stand-alone summary, and are refered to the relevant full text content in the chapter.

Chapter 4. Management tools and aids.

This section outlines various tools and aids of contextual relevance to reservoir management, and sketches prospects and limitations relating to their use, efficacy and efficiency.

Chapter 5. Looking ahead – perceived future needs.

Having sketched the array of issues that underpin reservoir management in preceding chapters, this section presents various possibilities to facilitate the active revitalization of reservoir limnology in South Africa, thereby attempting to re-initiate professional competence to assist in an informed, scientifically-based rational management of the country’s most precious resource. As the proposals suggested are essentially individualistic, input and consideration by a wider spectrum of interested and affected parties is required to ensure a better balance of opinion in advancing this ‘cause’.

Appendix.

Chapter 6. References and Bibliography.

ACKNOWLEDGMENTS

A review like this relies heavily on the assistance of library staff. Various members of the University of KwaZulu-Natal Library in Pietermaritzburg were especially accommodating in ensuring this support. Mr Nazim Gani in the Inter-Library Loans Division fielded a barrage of requests for literature acquisition, and Desk staff and the Subject Librarian at the Life Sciences Library were always obliging. Thanks also to Mr Arthur Baijnath in the university's Information Technology Division for streamlining the delivery and accessibility of a large number of literature papers that were supplied in electronic format.

Dr Jeff Thornton kindly provided a copy of a basic technical manual on "Managing Lakes and Reservoirs" (Holdren et al., 2001).

Reservoir limnology has incontestable strategic relevance in the context of securing and sustainably managing the storage vessels of South Africa's indubitably most limiting natural resource – fresh water. By soliciting this review, Dr Steve Mitchell has demonstrated an active concern about an indefensible deficiency in South African water resource management – the demise of process-based basic and applied reservoir research, on which rests prospects for ensuring continued provision and sound operational security of freshwater. The country at large owes special thanks to Dr Mitchell, for initiating a resurgence of competent engagement in this vital field of scientific endeavour. We sincerely hope that the report that follows will provide some tangible ammunition on which to take this concern forward, in order to secure and sustainably manage these critical resource storage vessels – our 'dams', reservoirs, impoundments, or man-made lakes, as they are variously called.

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1. GENERAL INTRODUCTION

1.1 Background to the report - general contextualization

The damming of rivers to create standing water-bodies – variously termed reservoirs, impoundments, man-made lakes, or "dams" in colloquial South African English[1] – represents the largest primary source of fresh water for agricultural, industrial and domestic use for the South African nation at large. Some 520 large dams intercept and capture more than 50% of the country's mean annual runoff (Allanson 2004). The resource they collect and hold in store is indubitably the nation's most limiting natural resource. In every sense it is "liquid gold". Surprisingly, however, the internationally acknowledged expertise in scientifically based limnological management of this resource that developed locally, grew, and was accumulating strongly in South Africa during the 1970's and 1980's was totally marginalized dissipated by implementation of an anonymous executive committee decision in the late 1980's to divert all active scientific limnological investigations into river research. However well intentioned this decision – recognizing that rivers are the primary natural freshwater ecosystems of South Africa – it effectively sabotaged reservoir limnology. Thereafter, no national stimulus or incentive existed to maintain any active engagement in reservoir or lake research.

The particular expertise in reservoir research that had been spawned and nurtured by the Inland Water Ecosystems division of the national CSIR, through a powerful and productive National Programme for Environmental Sciences (NPES) effectively dissipated. Four major limnological groupings that had been encouraged and grown to high, expert and authoritative levels of proficiency in reservoir limnology – namely Rhodes University's Institute for Fresh

[1] **Terminology.** These terms are often used interchangeably. In an attempt to standardize terminology and avoid ambiguity or inconsistency, Straškraba & Tundisi (1999) restrict use of "**impoundment**" to describe off-river or side-bank storage bodies, and refer to bodies of water captured by the damming of a river channel as "**dam reservoirs**" (or "**river-reservoirs**" by other authors), analgous to the colloquial South African term "**dam**", where strictly, however, 'dam' refers to the fabricated retaining structure – the wall. We defer to retain use of 'impoundment' as a synonym for 'reservoir' – a long-standing practice in the South African context in which off-channel storages are virtually unknown. In this report, "reservoir" is used in the context of an in-channel storage body – with the prefix 'river' omitted for simplicity and convenience. We consider that any potential ambiguities in our text can be reconciled and interpreted correctly in the context of their usage.

Water Studies (P K le Roux Dam, Buffalo River reservoirs, and southern African coastal lakes, inter alia), the National Institute for Water Research in Pretoria (Hartbeespoort Dam, other impoundments in the former 'Transvaal' province), Natal University (Midmar Dam and the Phongola floodplain), and the University of the Orange Free State (Wuras Dam) fragmented and/or altered direction. By necessity, its practitioners either re-aligned with river research, or redeployed in other new ventures, often totally unrelated to limnology or aquatic science. It is no exaggeration to state that thematic reservoir limnology has been extinct in the country for at least the past two decades. Fewer limnologists than exist on the fingers of one hand have persisted in fundamental reservoir research to pursue and advance rather specialist and individualistic interests.

The incongruity of this situation is quite unbelievable: that the coherent development of regionally appropriate and relevant information to undertake objective scientific management of **the store** of the nation's most limiting resource relies on a palm-full of practicing aquatic ecologists and a greater number of individuals with a principally technical rather than scientific mandate whose primary responsibility involves basic routine monitoring functions in various regional water agency laboratories. This scenario is frankly quite incredible – but indubitably true. Despite being plausibly well-intentioned, the instigator(s) of this promulgation is (are) accountable for limnological and natural resource myopia and sabotage – arguably grounds for at least figurative lynching.

This deficiency has been recognized and articulated at various forums, by academics, water supply agency biologists and a few senior managers in various agencies with freshwater-related responsibilities. However, active intervention or remediation responses by responsible management agencies were limited, allegedly on account of cited resource limitations. NEMP – the National Eutrophication Monitoring Programme (Department of Water Affairs and Forestry 2002), a joint DWAF and WRC initiative that commenced in 2002, represented the first salvo at breaking this log-jam. But its principal mandate of pure monitoring – as reflected in its title, is very restrictive. Little scope exists for open and objective process-related scientific investigation of regionally appropriate peculiarities and particularities of the eutrophication process, its consequences, and hence prospects for corrective management in the regional biophysical context.

1.2 Aims and objectives

The present report arises from the firm recognition of the need for a broader basis to reservoir limnology than eutrophication alone, despite the ubiquity of this phenomenon as a central water resource concern. Dr Steve Mitchell, Director of the Water Research Commission's Key Strategic Area – Water-Linked Ecosystems (KSA 2), an individual long and acutely aware of the critical urgency to revitalize reservoir research to ensure effective management into the future of the nation's potable water supplies, solicited the present literature review to pre-empt and serve as a point of departure for development of a reservoir limnology thrust within KSA 2. The following excerpt from the official WRC web page is directly relevant.

P5: Impoundments
Research within this programme will cover ecological functions and processes within impoundments with a view to improving our ability to manage these.
No current projects

http://wrc.org.sa/ (excerpted on 18 January 2006)

The brief and terms of reference for the present literature review were rather broad and general. The present report arises from a contract (K8/612) that required the compilation of a literature 'review of reservoir management', with the agreement and understanding that this should focus particularly on information and developments in the field after 1990. What follows below is an attempt to contextualize and summarize the relevant published knowledge base and its application in management. However, some explanation and clarification of underlying philosophy may help to contextualize this product.

1. Management is interpreted and has indeed been formally defined as a process of continuous decision taking. Decisions are required on an ongoing basis to ensure that pre-determined aims, objectives and outcomes are accomplished.

2. The adage "You cannot manage what you do not understand, and you cannot understand what you do not know" suggested that a rational review of reservoir

management cannot stand alone. Some basic contextual understanding of the structural attributes and functional processes of and within reservoirs was considered necessary if not essential. This is perceived to be especially relevant given that by present-day default, the intended revitalization of reservoir limnology would need to begin from a sorely depleted base of local expertise.

3. Aquatic ecology at large, and its constituent sub-discipline of Limnology that deals with inland waters, share much common ground, perceptively summarized by Barnes in his lead chapter on 'The Unity and Diversity of Aquatic Systems' in the synthetic introductory volume on aquatic ecosystems by Barnes & Mann (1980). For this reason, and despite their distinctiveness in certain important respects, any dichotomy between natural lakes and impoundments is in many ways an artificial one. While this review attempts to restrict its consideration to reservoirs, reference to natural lakes (and other inland waters where relevant) is made where appropriate – especially in regard to new and emerging issues involving transferable knowledge and understanding.

4. While synthesis of more recent (post-1990) literature was the primary focus of the review, the nature of the review topic precluded exclusive reliance on this later material. Much if not most of the current understanding of the topic is founded in earlier work, which accordingly required consideration to achieve any adequate contextual coherence in the present report. As reasoned above (point 2), this also appeared pertinent and relevant for any attempt to restore capacity and expertise.

5. Reservoir limnology is an extremely broad topic. The quality of water flowing in any river, and then stored in a reservoir ultimately hinges on the natural biophysical characteristics of its catchment and the associated land-uses, as formalized by Hynes (1975). The boundaries between catchment management and reservoir management are accordingly greatly blurred, and optimal reservoir management depends greatly on firm and diligent application of Integrated Catchment Management (ICM) policies, procedures and protocols (e.g. DWAF & WRC 1996), despite their attendant difficulties and complexities (van Zyl 1995). As far as possible, however,

this review is confined to addressing management issues and options within the impounded water body itself, and not those associated with or related to the drainage basin from whence its water is drawn.

6. It is clearly evident that sustainable reservoir management into the future relies not purely on scientific needs, but also technological and human resources and cultural and organizational transformations that are taking place nationally (Maaren & Dent 1995) and globally (Kennedy 2003). These parallel requirements are entirely apposite, but fall beyond the explicit scope of this review and the competence of its authors. Recommendations regarding these latter aspects accordingly require consideration and craftsmanship by respective experts and authorities in these areas.

2 RESERVOIRS AS ECOSYSTEMS – RESERVOIR LIMNOLOGY WITH RESPECT TO WATER QUALITY MANAGEMENT

2.1 Introduction

Based on the well-known adage that "it is not possible to manage what is not understood, and it is not possible to understand what is not known," we commence with an overview of reservoir ecosystem structure and functioning. This is especially important in South Africa, where available expertise on reservoir limnology all but totally dissipated with the almost exclusive shift from lentic to lotic research in the latter years of the 1980s decade.

Reservoir systems comprise typical communities e.g. plankton, benthos, nekton and fringing communities which develop and change both temporally and spatially in response to factors such as reservoir hydrology and climate. Thus, although this section is discussed under a series of headings, the need for a holistic, integrated ecological comprehension of these systems is emphasised.

2.2 The Physical Subsystem

2.2.1 Light, heat and stratification

Solar radiation is of fundamental importance to the dynamics of aquatic ecosystems. Most of the energy that controls the metabolism of water bodies is derived directly from the solar energy utilized in photosynthesis (Wetzel, 2001). The attenuation of light in water is of great limnological significance as it determines the maximum depth of photosynthesis, and the depth of direct heating (Townsend et al., 1996).

Absorption of solar energy, and its dissipation as heat, has profound effects on the thermal structure, stratification and circulation patterns in lakes and reservoirs (Kimmel et al., 1990; Mwaura, 2003). Nutrient cycling, distribution and concentration of dissolved gases (Wetzel, 2001), distribution of biota (Reynolds, 1992; Watanabe, 1992), and behavioural adaptations of organisms, are all markedly influenced by the thermal structure and stratification patterns (Coveney and Wetzel, 1995; Lewis, 1996; Kimmel et al., 1990).

Vertical stratification occurs when surface warming increases the temperature difference (and resulting density difference) between the surface and deeper layers, to the point where resistance to mixing becomes greater than the mixing power of wind-driven turbulence (Kalff, 2002). In reservoirs, the mixing caused by the unidirectional flow of water from the inflow to the dam wall also plays a significant role (Straškraba and Tundisi, 1999). Water temperatures take on the added importance of generally determining the depth at which an inflowing river enters the water column, affecting the resulting longitudinal thermal and trophic structure of the ecosystem, and potentially the dynamics of the outflowing river too (Groeger and Bass, 2005).

Stratification separates the water in lakes and reservoirs into three vertical zones, where permitted by climatic conditions and depth, and in addition by flows in reservoirs (Straškraba and Tundisi, 1999; Kalff, 2002; Wetzel, 2001). (See Figure 1)

The open water zone of a deep reservoir can be divided (vertically) into 3 sub-zones.

1 The mixing zone (z_{mix}) or upper mixed layer extends to the depth at which the thermocline is located. Also at the surface is the epilimnion or euphotic zone (z_{eu}), the depth at which 1% of surface illumination is received. The mixing zone and the euphotic zone may extend to the same depth, or one or the other may be deeper or shallower. Daily mixing due to wind and temperature differences tends to homogenize any vertical differences.
2 The metalimnion is a narrow zone (maximally only a few metres) in which the thermocline is usually located.
3 The hypolimnion is characterized by greatly reduced vertical mixing and is where most of the decomposition process takes place.

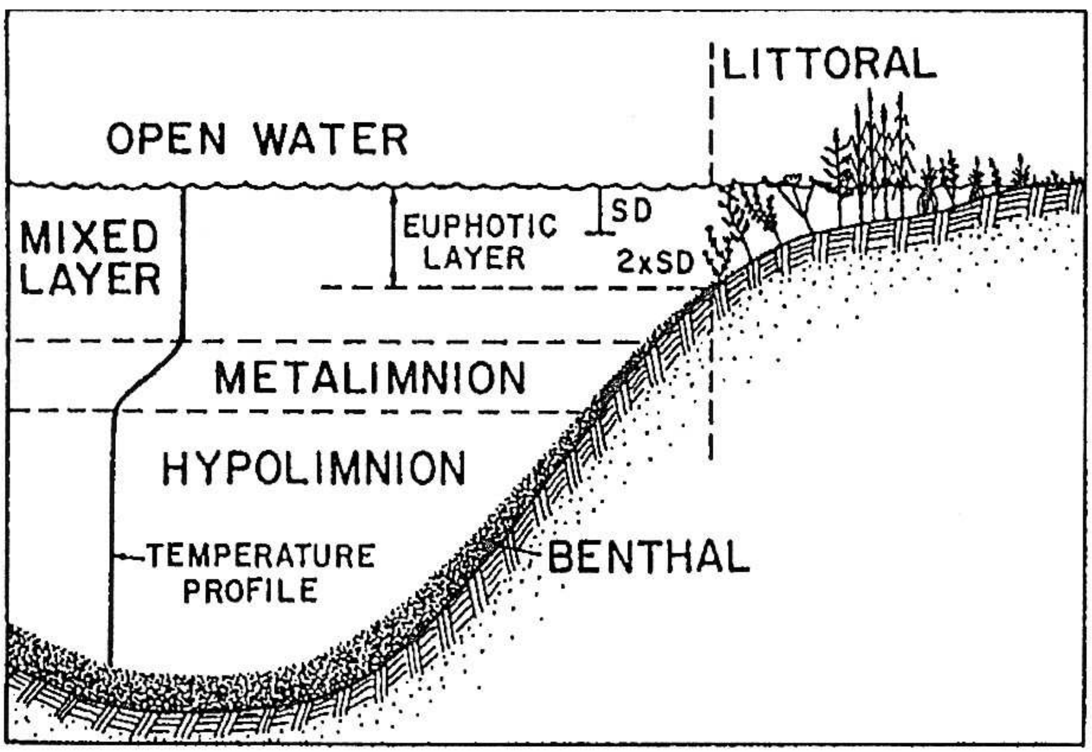


Figure 1. Major vertical regions in a reservoir with an indication of the distinction between the mixing zone and the euphotic zone. (After Straškraba and Tundisi, 1999).

The boundaries between the zones are determined by density differences (Straškraba and Tundisi, 1999). The principal factors influencing the formation, strength and extent of thermal stratification are the density of the water, solar radiation, energy transfer at the air-water interface, reservoir morphometry, and the mixing resulting from advections and wind-induced phenomena (Ford, 1990; Han et al., 2000).

In natural lakes, advective transport (inflows and outflows) is often ignored in heat budgets because it tends to be insignificant in systems with long residence times (Ragotzkie, 1978; Imboden and Wuest, 1995, both in Groeger and Bass, 2005). Conversely, in reservoirs, where residence time tends to be much shorter, advective gains and losses of heat may be quite important in influencing water temperatures (and consequently, stratification) (Ara, 1973; Straškraba, 1973, in Groeger and Bass, 2005; Straškraba and Hocking, 2002). This is particularly true in a reservoir with a deep or mid-water column release, where, during the period of stratification, warm water enters the reservoir while much cooler water is being lost downstream (Groeger and Bass, 2005).

Understanding and being able to predict particular temperatures of inflow, near-surface, deep and outflow waters within a reservoir is important in understanding how that ecosystem functions over time. Such information helps determine how the reservoir heat content varies and how the reservoir may respond to future climatic conditions (Groeger and Bass, 2005; Han et al., 2000).

In reservoirs the depth distribution of temperature is additionally dependent on theoretical retention time (Straškraba and Tundisi, 1999; Han et al., 2000).

- If the retention time exceeds 300 days, conditions in a reservoir are the same as those in a natural lake of similar geographic position and size.
- With shorter retention times the bottom water temperature increases with decreasing retention time until there is no difference between surface and bottom temperature in reservoirs that have a very short retention time.

Horizontal variations in temperature also occur and result from differential heating, inflow or mixing. Differential heating takes place where the smaller volume of water in shallow areas such as littoral zones and headwater regions, warms or cools more rapidly than the water of the open-water regions. Similarly, rivers flowing into a reservoir may be of different temperatures and can cause horizontal variations (Ford, 1990).

2.2.2 Transport processes

In the open water zone there are multiple water movement and mixing processes, classified into two groups (Straškraba and Tundisi, 1999).

- Those related to heat and momentum exchange processes at the water surface. In natural lakes these are the only factors which create vertical and horizontal differentiation of water masses
- Those related to flows. In reservoirs with intense through flow the dominant factor is the unidirectional flow of water to the dam wall. In reservoirs with a retention time of less than 300 days this factor is also important but not dominant

This overall movement creates complicated flows and longitudinal and vertical differentiation of reservoir water (Straškraba and Tundisi, 1999). (See Figure 2). Because these physical transport mechanisms influence the environment (temperature, light and chemical regimes) in which aquatic organisms exist, an understanding of reservoir transport processes is essential to understanding reservoir limnology (Ford, 1990).

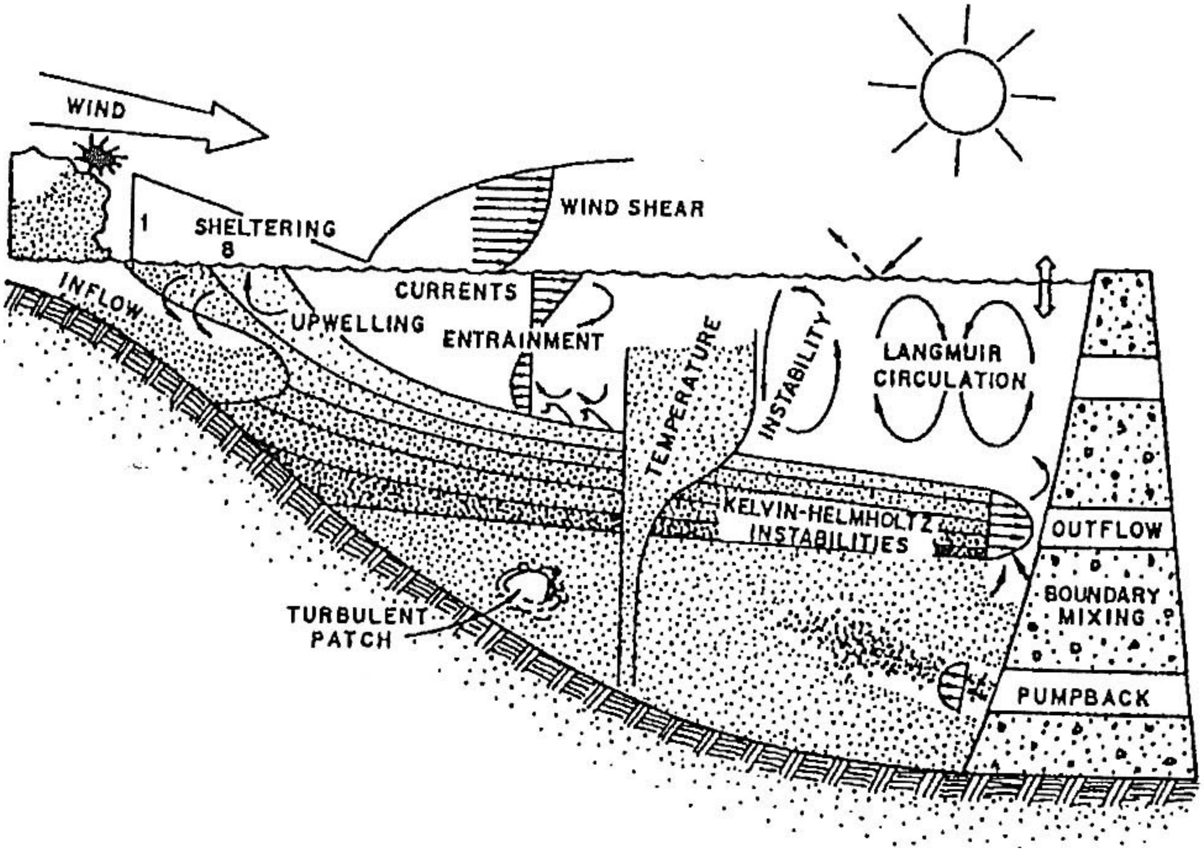


Figure 2. The mixing processes and water circulation in a reservoir. (From Straškraba and Tundisi (1999) redrawn from Ford (1987)).

Based on the conventions of Fischer et al. (1979) and Ford and Johnson (1986), in Ford (1990), the major transport mechanisms in reservoirs (and lakes) are the following:

Advection. Advection is transport by an imposed current system such as river inflows, outflows and wind-shear at the air-water interface.

Convection. Convection is vertical transport induced by density instabilities.

Turbulence. Turbulent flows are irregular (random), diffusive (produce mixing), rotational (overturning motions), time-varying and dissipative (decay rapidly without a continual source of energy). They can be generated by wind, inflows, outflows, convections, boundaries etc.

Diffusion. Diffusion is a mechanism where differences in mean concentrations are always reduced.

Shear. Shear is the advection of a fluid at different speeds at different positions and therefore requires a velocity gradient. It can be generated at the air-water interface by inflow currents, and internally by density currents.

Dispersion. Diffusion is the combined effects of shear and diffusion.

Entrainment. Entrainment is an advective-type mechanism where the thickness of the stirred layer grows by entraining (trapping) fluid from the unstirred layer.

Mixing. Mixing is any mechanism or process that causes a parcel of water to blend with or be diluted by another.

Settling. The sinking of particles with densities different from the surrounding fluid is called settling.

The time scales for these mechanisms vary from fractions of a second for turbulence to months for basin-wide motion. The length scales of motion vary from fractions of a centimetre to the size of the basin (Ford, 1990).

2.2.3 Meteorological Forcing

The type and magnitude of meteorological forcing is dependent on the reservoir location (latitude, longitude and elevation) (Talling, 1990). For the most part, meteorological forcing acts on the air-water interface and is therefore also dependent on the size and shape of the water surface and on the surrounding terrain. Of all the meteorological factors influencing movement and mixing in reservoirs, solar radiation and wind are probably the most important (Ford, 1990).

The energy available to warm the waters of a reservoir ultimately comes from solar radiation, which varies seasonally. In addition diurnal cycles occur. Water temperatures respond to both these cycles with a slight delay (Ford, 1990; Talling, 1990).

In most reservoirs, wind is the major source of energy for mixing. Mixing results from the interaction and cumulative effects of wind-induced shear at the air-water interface (e.g. currents, surface waves, seiches and entrainment). Like solar radiation, the wind is highly variable with seasonal, synoptic and diurnal cycles. Synoptic cycles correspond to the passage of major weather systems and commonly have a period of 5-7 days. Wind also has a diurnal cycle (Ford, 1990; Findikakis and Law, 1999).

The meteorological phenomena responsible for mixing and movement in reservoirs are highly variable. A reservoir is always in a state of flux and never in equilibrium with regard to forcing functions (Ford, 1990).

2.2.4 Inflows

An understanding of reservoir physics and consequential water quality conditions is not possible without knowledge of horizontal changes within a reservoir (Straškraba and Tundisi, 1999).

In accordance with differences in density between inflow water and reservoir water, water can spread to different depths as it enters the reservoir, and can enter the epilimnion, metalimnion or hypolimnion. Because the inflow density usually differs from the density of the reservoir surface water, inflows enter and move through reservoirs as density currents. (See Figure 3)

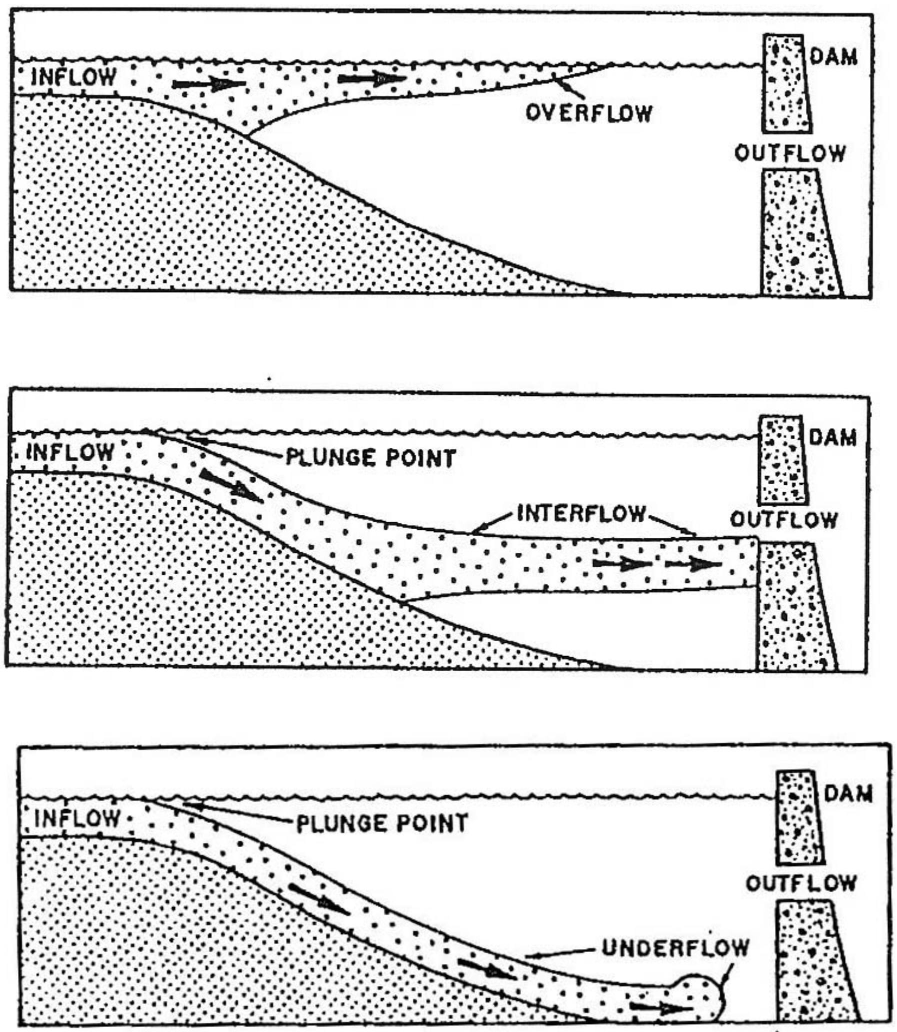


Figure 3. Density inflows to impoundments. (From Straškraba and Tundisi, 1999).

Density differences can be caused by temperature, total dissolved solids and by suspended solids. Because inflow densities (temperatures) are continuously changing, the level at which an intrusion moves through a reservoir will also change, as will its impact on vertical structure (Ford, 1990; Alavian et al., 1992; Lee and Yu, 1997; Morris and Fan, 1998; Straškraba and Tundisi, 1999; Ahlfeld et al., 2003).

If the inflow density is greater than the density of the surface water, the inflow will plunge beneath the water surface. The location of the plunge point is determined by a balance of the stream momentum (advective force), and the pressure gradient across the interface separating the river and reservoir waters (buoyancy force), and the resisting shear forces (wind, bed friction). The position of the plunge point is highly dynamic and changes with both flow and density. During a storm event it can move ten kilometres or more in a single day (Ford and Johnson, 1983, in Ford, 1990).

2.3 Spatial variations within a reservoir

The degree of horizontal and vertical heterogeneity within a reservoir is decisively influenced by reservoir morphometry, flow, and stratification conditions (Straškraba and Tundisi, 1999).

2.3.1 Vertical stratification

This was dealt with in section 2.2.

2.3.2 Horizontal zones

Interactions between river inflows, and reservoir basin and water column characteristics, lead to the establishment of longitudinal gradients in physical, chemical and biological characteristics (Kennedy 1999; Armengol et al., 1990). Thornton, 1981 (in Kennedy, 1999), identified riverine, transitional and lacustrine zones extending from headwater to dam (Davis and Reeder, 2001; Melcher et al., 1997; Armengol et al., 1999) (Figure 4).

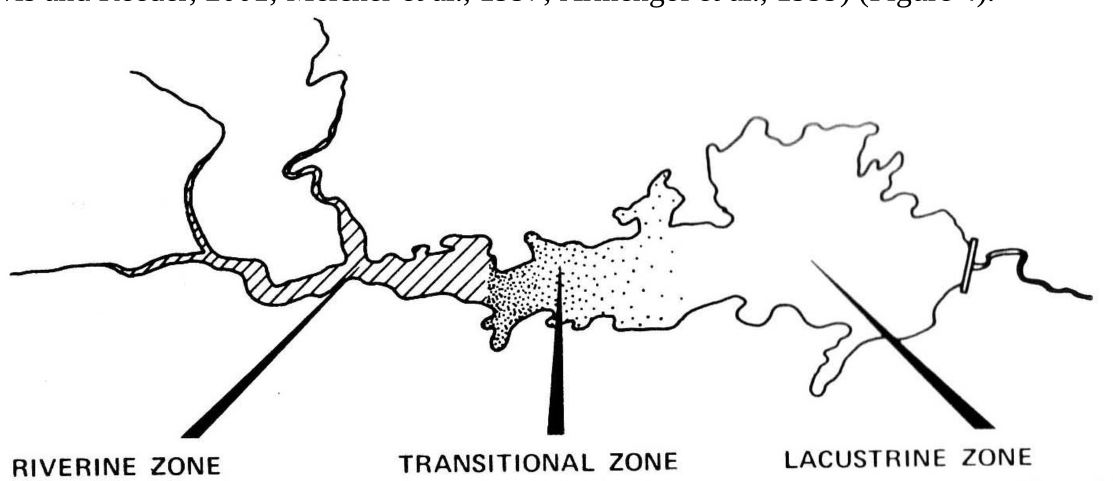


Figure 4. Horizontal zones in a typical reservoir. (After Kimmel and Groeger, 1984).

The characteristics of each reflect the diminishing influence of the inflowing river (Kimmel and Groeger, 1984, in Kennedy, 1999; Marzolf, 1984; Kimmel et al., 1990.)

The RIVERINE ZONE experiences relatively high flows due to the proximity of the river inflow. Water residence time is relatively short. Despite high nutrient concentration, algal production is often low due to inorganic turbidity and advective loss of cells (Kimmel et al., 1990; Melcher et al., 1997; Kennedy, 1999).

The TRANSITION ZONE, intermediate in location between river inflow and dam, receives nutrients by advective transport, while sedimentary loss of suspended material increases light availability. This zone often exhibits high algal production (Kennedy, 1999; Straškraba, 1999). The exact position of the transition zone in the reservoir depends on flow rates and therefore retention time (Straškraba, 1999). This zone is often associated with the plunge point if one occurs (Kimmel et al., 1990).

The LACUSTRINE ZONE, located in the broader, deeper downstream reach of the reservoir, is little influenced by river flows (Kennedy, 1999), and usually has the longest water retention time, higher water transparency and a deeper photic layer (Kimmel et al., 1990). Turbidity and nutrient concentrations are low due to the effects of sedimentation. Primary productivity is often nutrient limited, and is supported primarily by in situ nutrient cycling rather than by advected nutrients (Melcher et al., 1997).

The size of each successive horizontal zone varies within and between individual reservoirs, and is affected by such factors as watershed runoff, flow velocity, reservoir drawdown (Worth, 1994; Melcher et al., 1997), morphometry (Armengol et al., 1990), retention time, thermal stratification, season and geographical location (Straškraba and Tundisi, 1999; Straškraba, 1999). Lind (1984) in Kennedy (1999) recognized the importance of water residence time in the establishment of longitudinal gradients and hypothesized that reservoirs with intermediate water residence times would exhibit greatest spatial heterogeneity. In the case of a reservoir with a short retention time (<10 days) the whole reservoir may become a riverine zone, whereas with a retention time of >200 days, the riverine zone is short and most of the reservoir is lacustrine (Straškraba and Tundisi, 1999; Straškraba, 1999). Depending on watershed and inflow characteristics, and flushing rates, all three zones may not always be distinguishable within a single reservoir. In fact the dynamic nature of reservoir inflow and discharge greatly influences the differences observed between individual reservoirs (Kimmel et al., 1990). (Figure 5)

Allochthonous inputs of both detrital organics and nutrients are delivered to the impoundment by the inflowing river. The larger particulate material settles out and dissolved nutrients are incorporated into the algal biomass. Thus, seston of both allochthonous and autochthonous origin contributes to the total organic sediment load of a reservoir. Gradients in the nutrient content of the water, and resulting deposition rate of organic sediments can have important ecological consequences from the standpoint of oxygen distribution, nutrient cycling and reservoir metabolism. In larger reservoirs these gradients can be a primary factor controlling ecological conditions (Morris and Fan, 1998).

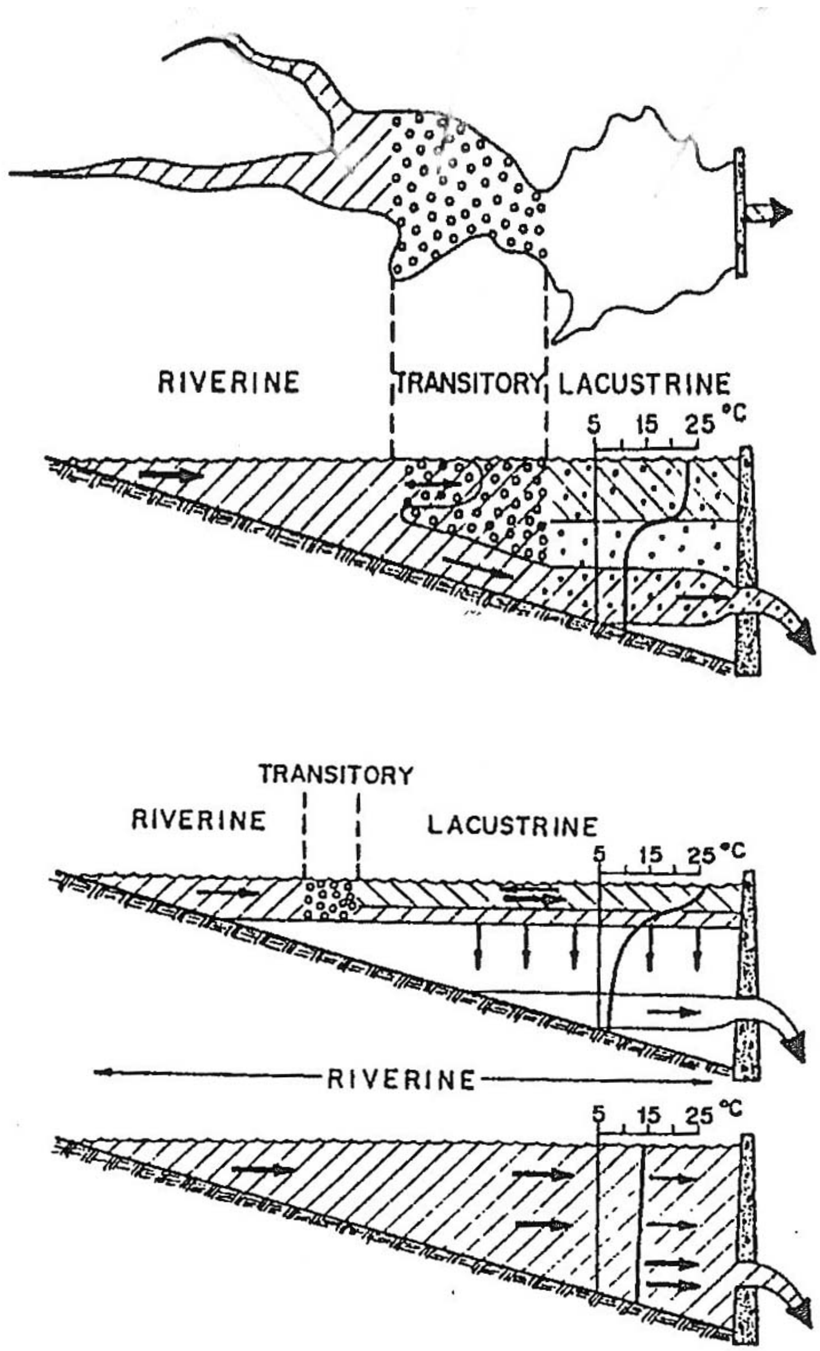


Figure 5. Longitudinal zoning in a reservoir, and changes in the zone extension, flow and mixing pattern for different reservoir conditions with regards retention time (R). Top panel, R>10 but <100days, middle panel, R>100days, bottom panel, R<10 days. (From Straškraba and Tundisi 1999).

Many examples of reservoirs with spatial gradients exist in the literature, e.g. Jurumirim Reservoir in Brazil (Nogueira et al., 1999); Sau Reservoir in Spain (Simek et al., 1998; 1999; Comerma et al., 1999; Armengol et al., 1999); Saidenbach Reservoir in Germany (Uhlmann et al., 1995); Lake Kariba in Zambia/Zimbabwe (Marshall et al., 1982); Lake le Roux in South Africa (Hart, 1990).

2.3.3 Other examples of horizontal spatial heterogeneity

The dendritic nature of reservoir basins, tributaries of differing water quality and embayments having limnological characteristics which differ from main-channel areas, all result in lateral variations in productivity and thereby further contribute to the spatial heterogeneity of reservoir ecosystems. Embayments may be important in providing food resources and nursery areas for fish, and macrophyte beds may function as isolated pockets of highly concentrated primary and secondary production, high-quality habitat and intensive predator-prey interactions (Kimmel et al., 1990).

In the Guri Reservoir in Venezuela, a tropical blackwater reservoir, differences in water colour, turbidity and mineralization led to the distinction of at least three different pelagic environments. These are reflected in the phytoplankton abundance and diversity (Vegas-Vilarrubia, 1995).

2.3.4 Management implications

Management goals in reservoirs are typically geared towards factors that control primary productivity (Davis and Reeder, 2001). “Trophic state” (as reflected by Secchi depth, nutrients (Kennedy and Walker, 1990), primary productivity (Kimmel et al., 1990), or hypolimnetic dissolved oxygen depletion (Cole and Hannan, 1990) or other indices based on these parameters) usually shifts from more eutrophic to more oligotrophic conditions along the riverine- transition-lacustrine gradient (Kimmel et al., 1990). Furthermore, these gradients are affected by the physical processes that predominate in each zone as well as by the type and degree of human activity in and around the reservoir (Davis and Reeder, 2001).

Davis and Reeder (2001) maintain that although many authors have shown the presence and causes of water quality gradients in reservoirs, none has considered the use of such information in reservoir water quality monitoring and trophic state assessments. Based on their work on water quality in seven reservoirs in Kentucky, USA, they proposed an improved monitoring system. Whereas the current most common approach to reservoir trophic state assessment was to monitor only lacustrine, near-dam sites, they proposed that these should be combined with a longitudinal profile of Secchi depth along the reservoir gradient during high and low flow periods. This was prompted by their results, which indicated that water transparency increased significantly downstream in accordance with water quality parameters. They maintained that this would be a logistically simple and inexpensive way to monitor the influence of the watershed, residence time and the advective movements of nutrients down-reservoir, on phytoplankton production.

The phenomenon of changes in physical and chemical composition contributing to the differentiated establishment of biological communities is characteristic of reservoirs

(Armengol et al., 1999; Zanata (1999) in Zanata and Espindola, 2002). The spatial distribution of phytoplankton and zooplankton in reservoirs will be discussed in section 2.6.

2.4 Temporal variations within a reservoir

2.4.1 Reservoir ageing and reservoir evolution

Reservoir ageing refers to the large-scale limnological changes that take place during the years immediately following filling. This is also referred to as ‘trophic upsurge’ because higher biological production takes place during this period (Straškraba et al., 1992; Straškraba and Tundisi, 1999). Reservoir evolution refers to a much longer process, lasting for decades or centuries (Straškraba et al., 1992). These changes that occur subsequent to reservoir ageing (i.e. reservoir evolution) are largely due to the impact of man’s activities (Straškraba and Tundisi, 1999).

2.4.2 The process of reservoir ageing

Based on natural lake geomorphological progression, cultural eutrophication and land-water interactions, Kimmel and Groeger (1986) in Holz et al. (1997), hypothesised that a period of high biological productivity, termed the “trophic upsurge”, should occur soon after reservoir filling. Internal nutrient loading, plankton productivity and fish productivity reach maximum rates during this initial phase and then decline during the next phase, “trophic depression”. Following the trophic depression, biological productivity remains at a lower level if external loading remains constant. However, productivity may increase or decrease if the external loading rate changes during reservoir evolution (Holz et al., 1997; Straškraba et al., 1993; Goldyn et al., 2003). (See Figures 6 and 7)

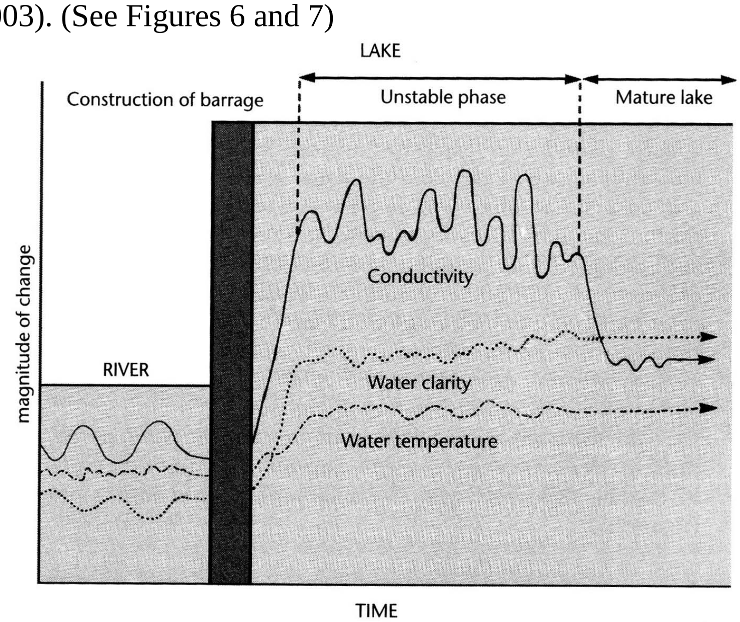


Figure 6. Differences between some physical and chemical conditions in a river, a newly developing reservoir, and a more mature reservoir. (After Davies and Day, 1998).

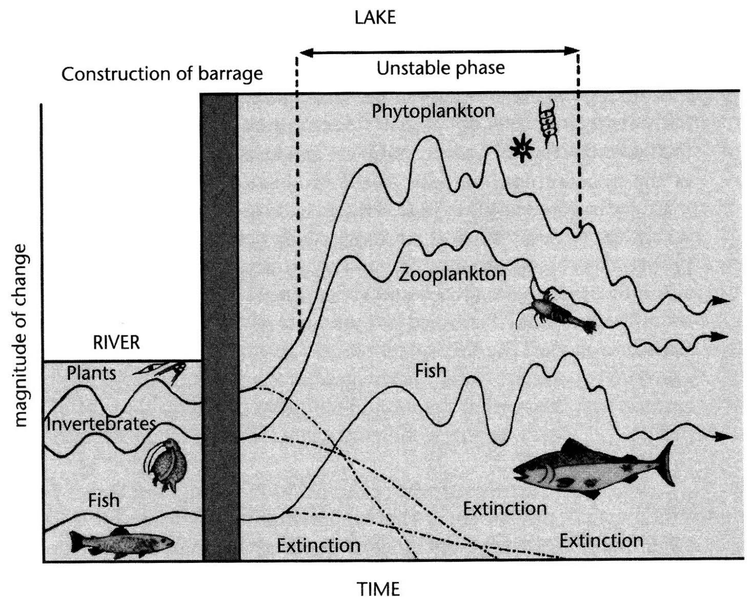


Figure 7. Differences between some biotic features in a river, and a newly developing reservoir. With time, the various biotic communities stabilise in a new and steadier state. (After Davies and Day, 1998).

Water quality problems that commonly occur during reservoir ageing, and their respective causes, are given in Table 1 that indicates that most problems are derived from the increased amount of organic matter in the reservoir. These problems manifest in different ways for different reasons.

INCREASED CONCENTRATIONS OF ORGANIC MATTER	Leached from the soil Decomposition of drowned vegetation
HIGH NUTRIENT CONCENTRATIONS	Leached from disturbed soil Decomposition of organic matter
EXCESSIVE GROWTH OF ALGAE AND MACROPHYTES	Increased nutrient concentrations from internal and external sources.
LOW OXYGEN CONCENTRATIONS (ESPECIALLY IN THE HYPOLIMNION)	Oxygen is consumed during decomposition of dissolved and particulate organic matter that enters via the inflow and is released from disturbed soil and decaying vegetation. Autochthonous organic matter increases with increased plant growth.
INCREASED COLOUR	Colour indicates the concentration of resistant organic matter. Colour changes occur very slowly.

Table 1. Water quality problems that commonly occur during reservoir ageing (modified from Straškraba and Tundisi, 1999).

2.4.3 Factors affecting ageing of reservoirs (and time required for stabilization).

Hydrological and operational processes influence the ageing process in various ways. For example, slow and intermittent filling prolongs ageing (Straškraba and Tundisi, 1999; Chang and Weng, 1997). Ageing is slower with longer retention time and faster in systems with rapid through flow (Straškraba and Tundisi, 1999). Prolongation of retention time affects conditions during filling, due to the leaching of nutrients from disturbed soil and vegetation as well as biotic interactions of different groups of organisms in relation to their development times (Straškraba, 1999).

Geographical location may be significant from the point of view of reservoir ageing. It is possible, but not verified in the literature, that more rapid ageing takes place in the tropics due to higher temperatures (Straškraba et al., 1993). However, the tropical forests that are drowned by developing reservoirs in these regions take a long time to decompose, lengthening the ageing process (Straškraba and Tundisi, 1999).

Forty years after its creation, Lake Kariba is still unstable as far as large-scale trends are concerned (Moreau, 1997).

2.4.4 Reservoir ageing and management

The process of reservoir ageing is extremely important from a management perspective since it occurs in the first few years of the existence of every new reservoir, with water quality inevitably deteriorating during this period (Straškraba and Tundisi, 1999; Straškraba et al., 1993; Scharf, 2002; Chang and Weng, 1997; Tundisi et al, 1998; Holz et al., 1997). The length of ageing differs between reservoirs, the average time span ranging from 4 to 10 years (Straškraba and Tundisi, 1999; Chang and Weng, 1997). Management strategies to deal with reservoir ageing fall into two distinct groups, those that attempt to prevent or reduce the "trophic upsurge", and those which attempt to manage the results of the "trophic upsurge" after it has occurred (i.e. eutrophication).

2.4.4.1 Eutrophication

Algal blooms and/or excessive growth of aquatic macrophytes often occur as a result of the increase in nutrients in a filling or newly filled reservoir (Straškraba and Tundisi, 1999; Goldyn et al., 2003).

Management issues include whether the terrestrial vegetation must be cleared completely, partially or not at all prior to filling, and what the consequences will be and how long the impact will last if the vegetation is not cleared (Scharf, 2002). Tundisi et al. (1998) maintain that the decrease of forest biomass prior to filling is essential although costly in the management of South American reservoirs. They are of the opinion that this problem has been ignored in the reservoirs built so far but must be considered in the planning of future reservoirs.

2.4.4.2 Hypolimnetic anoxia and internal loading

As a result of eutrophication, hypolimnetic anoxia is a common phenomenon in newly flooded reservoirs, with expensive implications related to the reducing potential of the water. Sulphide produced by bacteria under anaerobic conditions corrodes power plant machinery. (Tundisi et al., 1998).

Internal loading resulting from increased phosphorus release under anoxic conditions in the hypolimnion is a related problem. During the refilling of Brucher Reservoir in Germany (Scharf, 2002), anoxic conditions developed in the deep waters, resulting in internal nutrient enrichment. In this situation, artificial mixing was used to manage the anoxic water conditions, the reduction in the need for this process after a few years, indicating the end of the period of “trophic upsurge”. In this case the decision not to clear the basin of terrestrial vegetation prior to refilling proved to be cost efficient.

2.4.4.3 Gas releases

Methane and carbon dioxide emissions from reservoirs, due to drowned vegetation during filling, is recognized as a significant problem (Matvienko et al., 2000; Lima et al., 2000, 2002, 2005; Bergstrom et al., 2004; Venkiteswaran and Schiff, 2005). Latitude is the predominant factor influencing emissions from reservoirs, mainly because it determines the ambient temperature (Duchemin et al., 1995, 2000; Galy-Lacaux et al., 1999), but also because that temperature influences the specific type of vegetation which grows and thus determines the type and density of the biomass which is eventually flooded (Matvienko et al., 2000).

Recent studies have shown that methane emissions from hydroelectric reservoirs may comprise a considerable fraction of the anthropogenic contribution to greenhouse gases in the atmosphere. (Keller and Stallard, 1994; Novo and Tundisi, 1994; Duchemin et al., 1995; Fearnside 1995, 2001, 2002; Galy-Lacaux et al., 1997; St Louis et al., 2000). Estimates suggest that, per unit of energy produced, greenhouse-gas flux to the atmosphere from some hydroelectric reservoirs may be significant compared to greenhouse-gas emission by fossil-fuelled electricity generation (Rudd et al., 1993). To what degree hydropower and construction of hydroelectric reservoirs, in comparison with fossil fuel combustion, contributes to climatic change is still a matter of debate (Bergstrom et al., 2004). The issue in the assessment of methane emission is to determine the type of power-generating system that will produce a cleaner environment in the future (Lima et al., 2000).

2.4.4.4 Sedimentation

With the slowing of the flow rate of the river, sedimentation rates increase (Straškraba et al., 1993; Tundisi et al., 1998; Vorosmarty et al., 1997). Turbidity increases, affecting light penetration, which may or may not be a problem. Low light can moderate the effect of nutrient loading in turbid reservoirs (Holz et al., 1997).

2.5 The Chemical Subsystem.

2.5.1 Oxygen

Besides the water itself, oxygen is the most fundamental parameter of aquatic systems as it is obviously essential to the metabolism of all aquatic organisms that possess aerobic respiratory biochemistry (Wetzel, 2001).

Oxygen conditions in a reservoir depend on a number of processes (Straškraba and Tundisi, 1999), involving both supply and consumption (Henry, 1999; Bellanger, 2004).

- the rate of phytoplankton production (enrichment of water with oxygen during photosynthesis by day) and respiration (consumption oxygen by night)
- oxygen concentration and temperature of reservoir inflow
- the rate of oxygen exchange at the air-water interface
- the rate of phytoplankton sedimentation and decay in the deeper strata
- the organic matter content of the sediments and resulting oxygen consumption
- vertical and horizontal mixing conditions in the reservoir .

2.5.1.1 Vertical distribution of dissolved oxygen concentration.

In reservoirs that stratify during spring and summer, the vertical distribution of dissolved oxygen changes accordingly. In eutrophic waters there is usually a sharp drop in dissolved oxygen at or slightly below the thermocline, the hypolimnetic waters becoming increasingly anaerobic as the summer proceeds. Oligotrophic reservoirs tend to show no, or only slight, oxygen depletion below the seasonal thermocline (Allanson, 1995).

The role of phytoplankton and bacteria

Phytoplankton production is restricted to the illuminated epilimnion and decomposition mostly takes place in deeper waters. Oxygen concentrations are therefore vertically differentiated with a higher concentration at the surface and a deficit in deeper waters (Straškraba and Tundisi, 1999). The process of bacterial decay is dependent upon dissolved oxygen in its initial stages; thereafter, different assemblages of bacteria continue the heterotrophic decomposition, drawing upon the oxygen in dissolved sulphate and nitrate ions, with the concomitant liberation of hydrogen sulphide and ammonia (Allanson, 1995).

Temperature

In South African reservoirs, the mean water temperature is relatively high, resulting in substantial bacterial growth and activity. Deoxygenation of the hypolimnion of nutrient and algal rich systems is therefore rapid (Allanson, 1995), but is not limited to eutrophic systems. Townsend (1995) recorded two tropical reservoirs of LOW trophic status in Australia, which also showed rapid oxygen depletion rates and long periods of anoxia (maximum of 20 weeks) in response to thermal stratification. Deoxygenation was attributed primarily to water temperature (25-30^{o} C) and its effect on microbial metabolism. Due to the temperature dependence of hypolimnetic oxygen depletion, the trophic classification of lakes and reservoirs based on hypolimnetic deoxygenation and anoxia is not globally applicable without qualification. However, it is applicable to water bodies of similar hypolimnetic temperatures, especially when morphometric influences are also taken into account (Townsend, 1999).

Many South African reservoirs of low nutrient status also show considerable hypolimnetic oxygen depletion For example, Midmar (Hart, 1996), VanderKloof (Allanson and Jackson, 1984), Spioenkop (Hart, 1999) and Wagendrift (Hart, 2001). It is also evident in Lake Kariba (Klautsky et al., 1997).

Reservoirs in tropical regions are characterized by high surface temperatures, low temperature differences between the surface and bottom waters, and larger actual oxygen deficits than temperate lacustrine environments (Henry 1999). In sufficiently deep reservoirs, limited water mixing and long residence time lead to apparently persistent thermal stratification. If water renewal is not sufficient, stratified tropical freshwater systems may be subject to severe dissolved oxygen depletion conditions extending to shallower water, comparable to those observed in deeper sections of temperate eutrophic water bodies (Bellanger et al., 2004). River discharge is the principal destratification determinant in many deep tropical reservoirs (Kalff, 2002)

Mixing

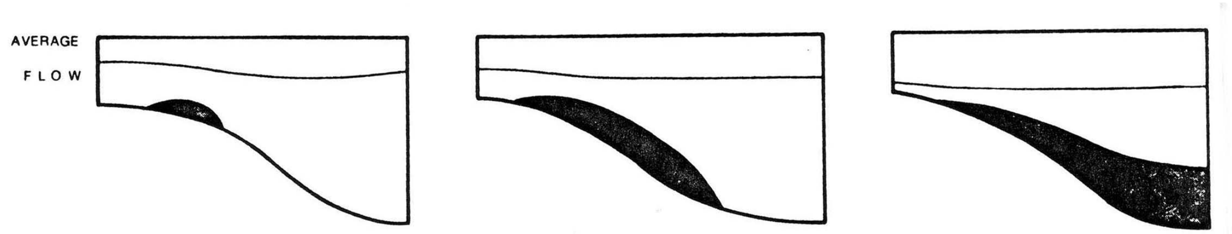
During periods of mixing, oxygen concentration is usually consistent throughout the water column (Straškraba and Tundisi, 1999; Henry 1999; Allanson, 1995; Cole and Hannan, 1990).

Oxygen stratification is less pronounced in rapidly through-flowing reservoirs (Straškraba and Tundisi, 1999; Cole and Hannan, 1990).

2.5.1.2 Horizontal distribution of dissolved oxygen concentration

Though the seasonal vertical stratification of water quality constituents has been well studied in lacustrine systems, the horizontal stratification of water quality in many reservoirs has not been widely appreciated until recently (Romero and Imberger, 1999).

Longitudinal patterns of oxygen depletion in reservoirs result from an interaction of flow and morphology. Small-sized allochthonous organic matter is deposited in the transition zone due to a decrease in flow velocity, causing a high rate of oxygen uptake. Increase in light penetration due to sedimentation, facilitates the build-up of phytoplankton populations. The volume of the hypolimnion in reservoirs is smaller in the transition zone, compared to the lacustrine zone, so has a lower total DO supply to meet hypolimnetic oxygen demands. The morphology of the reservoir results in generally higher hypolimnetic temperatures in the transition zone than the lacustrine zone, causing higher respiration rates. These higher temperatures also result in lower DO concentrations due to the relationship between oxygen solubility and temperature. The result of all these factors is that the highest biochemical oxygen demand occurs in the hypolimnion of the transition zone, causing the first anoxic zone to form there (Cole and Hannan, 1990). (See Figure 8)



Early summer Midsummer Late summer

Figure 8. The initial location and the extent of longitudinal development of the anoxic zone in a deep storage reservoir. (After Cole and Hannan, 1990).

Once formed, the anoxic zone develops and expands both upstream and downstream. The upwards development is influenced primarily by inflow from the river. Any flood event which breaks down thermal stratification, will dissipate the anoxic zone. The downstream anoxic zone develops more slowly and with less fluctuation. Concurrently, the anoxic zone develops upwards and laterally out of the thalweg (the deepest part of the channel of the original river).

2.5.1.3 Management

From a management perspective, hypolimnetic anoxia can have far reaching implications in reservoirs. This is primarily due to its effect on chemical exchanges at the sediment water interface (Townsend, 1999; Hamilton and Schladow, 1994). It also needs to be considered in the management of outflow from reservoirs where low outflow gates release deoxygenated water to the river below the dam.

2.5.2 Organic carbon

Organic matter in reservoirs is either allochthonous (it enters the reservoir from the watershed) or autochthonous (it is produced within the reservoir by photosynthesis by algae or aquatic macrophytes (Straškraba and Tundisi, 1999), or by bacterial chemosynthesis of dissolved organic carbon). It is a mixture of plant, microbial and animal products in various stages of decomposition. Nonhumic or labile substances are easily degraded by micro organisms (hours). Humic substances, designated as ‘refractory’ are more recalcitrant to degradation and constitute 70% to 80% of the organic matter of water and soils.

Nearly all the organic carbon of natural waters consists of dissolved organic carbon (DOC) and particulate organic carbon (POC). The ratio of DOC to POC is commonly between 6:1 and 10:1.

The rate of decomposition of organic substances is greatly dependent on solubility; however the energetic transformations of organic carbon are similar whether in the dissolved or particulate form.

Much of the decomposition of organic compounds in reservoirs occurs in aerobic waters prior to sedimentation. However, the benthic region is the dominant site of decomposition of most POC, and DOC that has aggregated into insoluble particles or co precipitated with particulate matter and settled to the bottom sediments (Wetzel, 2001).

It is increasingly recognized that much of the energy involved in the food webs of reservoirs and lakes derives from the catchment, rather than being fixed *in situ.* Traditional perceptions of standing water bodies as autotrophic systems are being replaced by recognition of their frequently heterotrophic nature linked to the carbon input (Reynolds, 2004).

2.5.3 Nutrients

The importance of nutrients in reservoirs is due to their effect on biological production (Straškraba and Tundisi, 1999). Their distribution and availability are influenced by nutrient loading (internal and external), sedimentation, flow, mixing and discharge (Kennedy and Walker, 1990). From a management perspective, it is critical to characterize sources of nutrients and their respective quantities, in order to manage their inputs to freshwater systems (Nowlin et al., 2005).

2.5.3.1 Phosphorus

2.5.3.1.1 Sources and losses of phosphorus in reservoirs

Sources of phosphorus in reservoir waters include

- precipitation (Miranda and Matvienko, 2003)
- groundwater (Miranda and Matvienko, 2003)
- surface runoff (Wetzel, 2001; Brzakova et al., 2003)
- sediment resuspension (James and Barko, 1997; Istvanovics and Somlyody, 1999)
- chemical release from sediments (Ting and Appan, 1996; James and Barko, 1997; Istvanovics et al., 1997; Istvanovics and Somlyody, 1999)

Losses of phosphorus from reservoir waters occur by

- outflow from the reservoir
- sedimentation (Wetzel, 2001; James and Barko, 1997; Havens and Schelske, 2001).
- storage as polyphosphates by meroplanktonic algae (algae which have resting stages in the sediments) (Carrick et al., 1993)

2.5.3.1.2 **The phosphorus cycle** (See Figure 9)

The phosphorus cycle is considered the most critical process in organic production within reservoirs, phosphorus-uptake by phytoplankton being more significant than that by pelagic bacteria (Wetzel, 2001). Soluble reactive phosphorus is taken up to such an extent that the concentration of this nutrient in surface waters can be as low as a few micrograms per litre (Straškraba and Tundisi, 1999). Total phosphorus concentration in most nonpolluted natural waters is between 10 – 50ug l^{-1} (Wetzel, 2001).

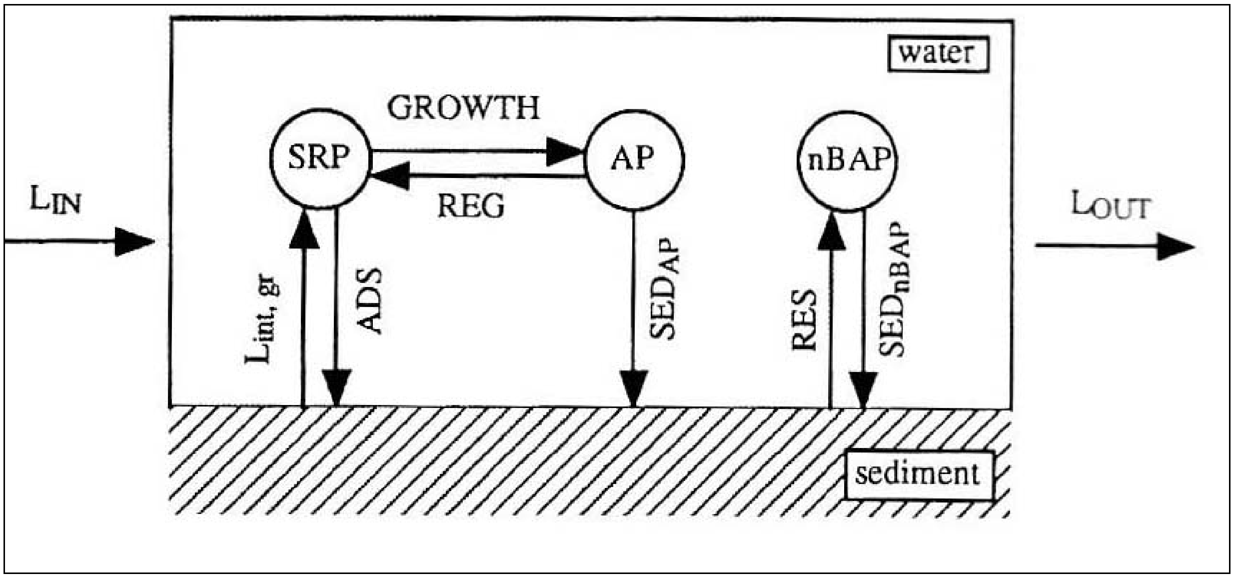


Figure 9. A simplified P cycle. (SRP, soluble reactive P; AP, algal-P; nBAP, non-biologically-available P; L_{in} and L_{out} , input and output TP loads; $L_{int.gr.}$, gross internal P load; ADS, phosphate adsorption; GROWTH, algal growth; REG, regeneration of SRP during the mineralization of algae in the water; $SED_{AP,}$ algal sedimentation; SED $_{nBAP,}$ sedimentation of nBAP; RES, resuspension of nBAP). (After Istvanovics and Somlyody, 1999).

The only significant form of inorganic phosphorus in lake systems is orthophosphate (PO_4^{3-}). A very large proportion of the phosphorus in lake water (>90%) is bound in organic phosphates or variously associated with, or adsorbed to, inorganic, and dead particulate organic materials (Wetzel, 2001). Considerable amounts of phosphorus can be bound and cycled in sediment-living bacteria. Shifts in bacterial biomass in combination with the abiotic sorption characteristics of the sediments may therefore play an essential role in the phosphorus dynamics of lake sediments (Waara et al., 1993).

Reservoirs function as traps for phosphorus (Straškraba and Tundisi, 1999; Miranda and Matvienko, 2003). Particulate phosphorus brought in from the watershed, starts settling to the bottom of a reservoir immediately, but dissolved phosphorus entering the reservoir has to be incorporated into particles before it can sink to the sediments. Phosphorus uptake by particles is by physical and chemical processes, such as adsorption onto particle surfaces or precipitation (Brzakova et al., 2003). The dissolved phosphorus taken up by phytoplankton eventually accumulates in the sediments in large amounts when the cells die, especially in eutrophic conditions (Porcalova, 1990). In some cases, the phosphorus content of the upper millimetres of the sediment can be greater than that in the whole overlying water column. The amount of phosphorus that leaves the reservoir is thus much lower than the amount that enters the system (Straškraba and Tundisi, 1999).

It is still not certain whether settling particles constitute a sink, or serve as a source of phosphorus for the surrounding water (Brzakova et al., 2003). Significant, short-term P-uptake capacity was demonstrated by suspended and settling seston in the epilimnion of Rimov Reservoir in the Czech Republic during periods of high P-deficiency of phytoplankton. This was apparently caused by the presence of phytoplankton cells with depleted cellular P-quota within the seston particles, which removed a significant amount of dissolved orthophosphate from the water column. No release of phosphorus from suspended or entrapped seston particles was observed during this study.

2.5.3.1.3 Horizontal distribution of phosphorus in a reservoir

Reservoirs are typically elongate resulting in a definite spatial variation in nutrient concentration, primary production and the factors controlling them (Kimmel and Groeger, 1984, in Istvanovics and Somlyody, 1999; Straškraba, 1996). The relative importance of the various pathways of phosphorus cycling varies accordingly. (See Section 2.3).

Sedimentation patterns along the longitudinal axis of a reservoir influence the distribution of phosphorus accumulated in the sediments. Sedimentation measurements along the Rimov Reservoir revealed that half of the phosphorus input was deposited within the first 3km of the length of the reservoir which represented only 20% of the whole surface area of the reservoir (Porcalova, 1990).

2.5.3.1.4 Management

Elevated phosphorous concentrations are significant in the problem of eutrophication which will be discussed in Section 3.1.2.

2.5.3.2 Nitrogen

2.5.3.2.1 Sources and losses of nitrogen in reservoirs

Sources of nitrogen in reservoir waters include

- precipitation on the water surface
- nitrogen fixation in the water and in the sediments
- inputs from surface and groundwater drainage

Losses of nitrogen from reservoir waters occur by

- effluent outflow from the reservoir
- reduction of nitrate to N_2 by bacterial denitrification with the subsequent return of gaseous N_2 to the atmosphere (Straškraba and Tundisi, 1999; Tomaszek and Czerwieniec, 2000; Tomaszek and Koszelnik, 2003).

- permanent loss of inorganic and organic nitrogen-containing compounds to the sediments (Alaoui-Mhamdi et al., 1996; Tomaszek and Czerwieniec, 2000; Tomaszek and Koszelnik, 2003).
- uptake by aquatic organisms (Tomaszek and Czerwieniec, 2000; Tomaszek and Koszelnik, 2003).

2.5.3.2.2 The nitrogen cycle (See Figure 10)

This cycle differs from the phosphorus cycle because bacteria actively participate in the cycling process by means of nitrogen fixation (Wetzel, 2001; Straškraba and Tundisi, 1999). (See Figure 10)

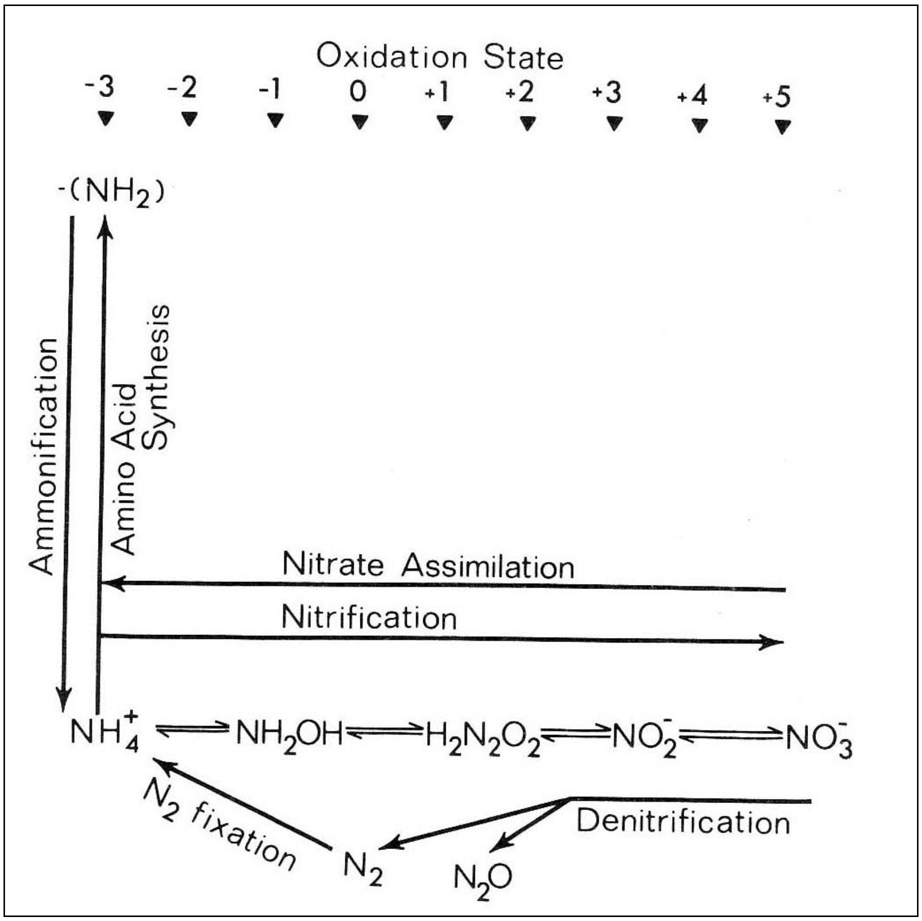


Figure 10. A simplified nitrogen cycle showing biochemical reactions that influence the distribution of nitrogen compounds in water (after Stadelmann (1971) and Kuznetsov (1970) in Wetzel, 2001).

Nitrogen retention is one of the important issues of nitrogen management for different water bodies and basins (Galicka et al., 1994; Nielsen et al., 1995, 2001; Seitzinger et al., 2002, all in Tomaszek and Koszelnik, 2003; Windolf et al., 1996; Josefson and Rasmussen, 2000; Saunders and Kalff, 2001).

The predominant opinion is that nitrogen (and phosphorus) retention in lakes and reservoirs is directly proportional to hydraulic retention time (Kawara et al., 1998; Straškraba, 1998; Saunders and Kalff, 2001; Nielsen et al., 1995 in Tomaszek and Koszelnik, 2003). Nevertheless, some shallow and unstratified water bodies with short hydraulic retention time, show greater capacity to retain nitrogen than the larger and deeper ones (Howarth et al., 1995), especially when they have substantial loads of nitrogen (Fleisher et al., 1994). This is due to the water column/bottom sediment interfacial contact in shallow water bodies being conducive to nitrogen retention by benthic assimilation and denitrification or by coupled nitrification-denitrification processes (Josefson and Rasmussen, 2000; Tomaszek and Czerwieniec, 2000).

2.5.3.2.3 Denitrification

A major mechanism removing excess nitrogen from the aquatic system, denitrification acts as a controller of eutrophication (Abe et al., 2002). Denitrifying bacteria are naturally encountered in diverse types of environments due to their ability to use different compounds as a carbon source and final electron acceptors (oxygen in aerobic conditions vs. nitrate, nitrite, nitric or nitrous oxide in anaerobic conditions) (Knowles, 1982, in Bednarek et al., 2002).

In anoxic waters, heterotrophic, anaerobic bacteria utilize NO_3 or NO_2 as terminal electron acceptors during oxidation of organic matter. Autotrophic bacteria use sulphur, thiosulphate or hydrogen as electron donors (Tomaszek and Czerwieniec, 2003). Because the final products of denitrification are gaseous compounds (N_2O, NO, N_2) the process represents a sink in the overall N budget of many aquatic ecosystems (Seitzinger, 1988, 1990; Tomaszek, 1991; Law et al., 1991, 1992; Nielson, 1992; Nielson and Glud, 1996; Nielson et al., 1996; Mengis et al., 1997 in Tomaszek and Czerwieniec, 2003). Sediment nitrification can be a major source of nitrates for denitrification (Tomaszek and Czerwieniec, 2003).

Rates and regulations of the denitrification process are of increasing interest due to eutrophication problems caused by anthropogenic inputs of combined nitrogen (Seitzinger, 1990, in Tomaszek and Czerwieniec, 2000). Many attempts to quantify denitrification have been made, however the lack of suitable methods has been a major obstacle in denitrification studies. Denitrification rates in bottom sediments are difficult to measure accurately, as several reactions e.g. nitrification, denitrification, N-fixation, occur simultaneously in an aquatic environment. (Tomaszek and Czerwieniec, 2000, 2003).

A number of environmental factors are capable of influencing the rate of the denitrification process in bottom sediments. These include water temperature, concentrations of nitrate in overlying water and organic matter in the sediment, the depth of the sediment and conditions therein including redox potential and dissolved oxygen. The positive influence of light on denitrification rate suggests that photosynthetic oxygen production stimulates the process and consequently increases the diffusion of nitrate into the bottom sediments (Tomaszek and Czerwieniec, 2003). Preliminary results on denitrification rates in Sulejow Reservoir in Poland prove the positive correlation between the denitrification rate and the content of organic carbon in the bottom sediments (Bednarek et al., 2002). A positive correlation was also recorded between the content of organic carbon and the percentage of denitrifying bacteria.

2.5.3.2.4 Management issues

Tomaszek and Koszelnik (2003) developed an empirical model suitable for predicting nitrogen seasonal dynamics in a reservoir. Models described in the literature, capable of predicting nitrogen retention with a relatively high degree of accuracy were selected and adapted (e.g. Jensen et al., 1992; Arheimer and Wittgren, 1994; Windolf et al., 1996). All models related nitrogen retention to simple parameters such as hydraulic retention time, volume, surface area, average depth, temperature and relative area hydraulic loading.

Depending on the species of bacteria involved, accumulation of nitrite to toxic levels may occur in the sediments during the denitrification process (Xu et al., 1996).

It has been suggested that gaseous sub products of denitrification such as N_2O and NO, have increased in the atmosphere, leading to ozone depletion and the greenhouse effect (Davidson, 1991, in Abe et al., 2002).

2.5.4 The role of sediments in reservoir chemistry

Sediments are an important source of nutrients to freshwater ecosystems (Nowlin et al., 2005). Historically, phosphorus has received more attention than nitrogen because of the complex nature of nitrogen transformations in lake sediments, and the fact that phosphorus is often the limiting nutrient in phytoplankton productivity (Dillon and Rigler, 1974; Smith, 1979 both in Nowlin et al., 2005).

2.5.4.1 Internal loading

Attempts at lake restoration through the reduction of external phosphorus inputs often fail because of large phosphorus inputs from sediments, even after external loads have been substantially reduced (Sondergaard et al., 1990; Martinova, 1993; Straškraba and Tundisi, 1999; Istvanovics and Somlyody, 1999; Heidenreich and Kleeberg, 2003). In some lakes sediment nutrient release may represent a more important process ecologically, than inputs from external nutrient sources because phosphorus released from sediments often contains a larger proportion of immediately available phosphorus (Peters, 1981; Nurnberg and Peters, 1984; Nurnberg, 1985, all in Nowlin et al., 2005).

2.5.4.2 Factors regulating release of nutrients from the sediments

Processes leading to phosphorus release to the water column from underlying sediments include

- desorption and dissolution of phosphorus bound in precipitates and inorganic material (Moore and Reddy, 1994, in Nowlin et al., 2005).
- microbial mineralization of organic matter (Gachter and Meyer, 1993).
- diffusion of dissolved phosphorus from sediment pore waters (Moore et al., 1998, in Nowlin et al., 2005).

Under oxic conditions, phosphorus is retained in the sediments beneath an oxidized micro zone (Wetzel, 2001). Much of the phosphorus is inorganic, derived from the watershed, and includes phosphate adsorbed onto clays and ferric hydroxides. In addition, phosphate co-precipitates with iron, manganese and carbonates (Mackereth, 1966; Harter, 1968; Wentz and Lee, 1969 in Wetzel, 2001). The oxidized layer forms a trap for iron and manganese as well as for phosphate, thereby greatly reducing transport of materials into the water, whilst scavenging materials such as phosphate from the water.

In an area of calcareous sedimentary rock in Morocco, it was demonstrated that Ca was responsible for binding a large fraction of the phosphate in the sediments as apatite, preventing the phosphate release into the overlying water (Alaoui-Mhamdi et al., 1996).

However, in productive lakes, after the onset of summer stratification, biological activity in sediments and the hypolimnion decreases hypolimnetic DO concentration and pH, lowering the redox potential at the sediment-water interface. The environmental variables that may regulate the release rate of dissolved phosphorus from sediments are temperature, dissolved oxygen, pH and redox potential (Nowlin et al., 2005; Schladow and Hamilton, 1995; Martinova, 1993; Gachter and Meyer, 1993).

Iron exists in solution in either the ferrous form (Fe(II)) or the ferric form (Fe(III)). As the redox potential drops, the Fe(III) in the sediments which is in the form of ferric hydroxide with adsorbed phosphates, is reduced to Fe(II), leading to the release of $PO^{3-}{}_{4}$. The mobility of phosphorus is thus determined by the supply of iron in the sediments, and the resulting capacity for phosphorus sorption and subsequent release. (Heidenreich and Kleeberg, 2003). The total phosphorus concentration in the sediments and the relative concentration of various phosphorus fractions, also influence sediment phosphorus release rates (Nowlin et al., 2005).

Biotic processes also play a part in phosphorus release rates. Under anoxic conditions, high concentrations of nitrate (NO_3) in waters overlying the sediments can raise the redox potential by providing denitrifying bacteria with an alternate electron acceptor, thereby enhancing Fe oxidation and sediment sorption of $PO^{3-}{}_{4}$ (Gachter and Meyer, 1993). On the other hand, the growth of Fe reducing bacteria may be stimulated by high NO_3 levels, leading to higher decomposition rates of organic matter and enhancement of $PO^{3-}{}_{4}$ release from sediments.

Thus the release of $PO^{3-}{}_{4}$ from reservoir sediments into overlying water during summer hypolimnetic anoxia is a result of the complex interaction between biotic and abiotic processes in the water column and sediment (Nowlin et al, 2005).

2.5.4.3 Exposed reservoir sediments

Large fluctuations in reservoir water levels create an additional source of phosphorus release from reservoir sediments into the overlying water. The difference between the littoral sediments of a reservoir and the permanently inundated sediments is that the former are periodically exposed, experiencing cycles of wetting and drying. This can have a profound effect on the sediment's ability to adsorb or desorb nutrients (Watts, 2000a; Mitchell and Baldwin, 1998). More phosphorus can be released from reflooded dried sediments than from sediments that remain inundated (De Groot and Van Wijck, 1993; Baldwin, 1996; Mitchell and Baldwin, 1998; Qui and McComb, 1994; 1995 both in Watts, 2000b).

Because such sediments are potentially, the largest sink for, and the largest source of, many nutrients (particularly phosphorus), sediment changes induced by exposure may have a marked effect on the nutrient budget of the reservoir on their rewetting (Baldwin, 1996). Twinch (1987) in Watts (2000b) showed experimentally, a three-fold increase in phosphate equilibrium concentration in dried sediments, which increased potential phosphorus P release by 127%.

In submerged sediments, there is continuous redox cycling, resulting in the production of ferric hydroxide (Fe(OOH)), which has a high affinity for phosphorus. When the sediments are exposed, this process of redox cycling is adversely affected, resulting in reduced phosphorus affinity in the dried sediments. Consequently, there is a greater potential for the phosphorus in the sediments to be released to the water column on their rewetting (De Groot and Van Wijck, 1993). The reflooding of sediments therefore has the potential to create conditions conducive to aerobic sedimentary phosphorus release.

From a management perspective, this is highly significant where large expanses of sediment are exposed, either during dry seasons or as a result of reservoir drawdown (Watts, 2000a, 2000b). Implications for management would depend on the type of drawdown, severity of drying and conditions of refilling. Ideally, water levels should be maintained at as high a level as possible to reduce phosphorus release, but this is not often possible, especially in dry climates. During the drought of 1997-1998, the volume of the Carcour Dam in Australia fell to 19%, exposing 70.5% of the total sediment surface (Watts, 2000b).

It has been suggested that, especially where there is rapid inundation and short residence times, phosphorus flushing could be used as a management tool, where phosphorus could be repeatedly flushed from the sediment into the water column, the ultimate aim being to reduce sedimentary phosphorus levels and thus internal loading (Baldwin, 1996). Flushing as a management tool will be discussed fully in the relevant section. Where littoral sediments are exposed for long periods, it may be possible to harvest these sediments for use as a nutrient resource such as top dressing (Watts, 2000b).

Historically, when describing phosphorus release from inundated sediments, it has been the top few centimetres or even millimetres that are considered important for most chemical and biological activity (Mortimer, 1971; Fillos and Swanson, 1975; Bostrom et al., 1982, in Watts, 2000b). De Groot and Van Wijck (1993), however, found chemical and physical changes in sediments during desiccation that affected layers to a depth of 40cm. An increase in total phosphorus was observed in the top layer, with a decrease in the deeper layers, indicating an upward flux of phosphorus. The fact that sediment 10cm deep may be active in the processes of sedimentary phosphorus release has implications for the prediction of internal loading to a reservoir (Watts, 2000b).

2.5.4.4 Temporal and spatial scales

Sediment nutrient release also involves some important temporal and spatial scales. Following the release of phosphorus, phytoplankton growth in the hypolimnion is light limited (and oxygen limited). In the epilimnion it is phosphorus limited. It is only following overturn that the majority of the phosphorus released from the sediments becomes available in the photic zone. From a management perspective, there is a lag, corresponding to the time required for the nutrients to be assimilated and the phytoplankton to increase to sufficient concentration to constitute a bloom (Schladow and Hamilton, 1995).

Because of the strong longitudinal gradient of physical and chemical conditions in a reservoir, nutrient release is affected accordingly (Nowlin et al., 2005). Relatively shallow areas near inflows may lack thermal stratification throughout the year, while deeper areas near the dam may develop summer stratification (Thornton, 1990). This longitudinal development of thermal stratification can lead to variation in DO concentration in waters above sediments, thus affecting sediment nutrient release rates. The composition of sediments may also vary along the longitudinal axis of the reservoir, with sediments near inflows containing a larger proportion of allochthonous material, and sediments near the dam containing a larger proportion of autochthonous material (Thornton, 1990). Various phosphorous-containing fractions within sediments may vary with the spatial heterogeneity of sediment characteristics within a reservoir, potentially affecting the P-binding capacity of sediments and sediment phosphorus release rates (Nowlin et al., 2005).

2.6 The Biological Subsystem

2.6.1 Bacteria

2.6.1.1 Bacterioplankton

Microbial organisms, commonly overlooked or disregarded (largely because of their unseen nature and technical difficulties of analysis) play an especially important role in reservoirs. Heterotrophic elements utilize the often plentiful allochthonous carbon sources, while autotrophic cyanobacteria are commonly dominant primary producers.

2.6.1.1.1 Heterotrophic Bacteria

The abundance of free-living planktonic bacteria normally oscillates between 10^5 and 10^6 cells ml^{-1} in all but ultraoligotrophic or hypertrophic waters (Kalff, 2002). Heterotrophic bacterioplankton are predominantly free-living, but in reservoirs where concentrations of suspended particles are high, bacterial attachment to particles can predominate (Wetzel, 2001; Lind and Davalos-Lind, 1991). The attached bacteria tend to be larger and contribute more to the community biomass (Kalff, 2002). Numbers, biomass and productivity of bacterioplankton generally increase with increasing photosynthetic productivity (Wetzel, 2001; Voros et al., 2003).

- **Vertical distribution**

  In thermally stratified lakes, heterotrophic bacterioplankton numbers and biomass are usually high in the epilimnion, decrease to a minimum in the metalimnion and upper hypolimnion, and increase in the lower hypolimnion, especially if it is anoxic (Wetzel, 2001; Tietjen and Wetzel, 2003).

  It is well known that phytoplankton is negatively affected by high light intensities at the water surface, but there are few reports of UV effects on bacteria and extra cellular enzymes (Sieracki and Sieburth, 1986; Herndl et al., 1993, in Lindell and Edling, 1996). Algae may respond to light stress by decreased metabolism, protective pigments or migration to deeper waters (Karentz et al., 1994, in Lindell and Edling, 1996). Heterotrophic bacteria, on the other hand, have very limited capacity to migrate, thus may be severely affected by UV radiation.

  Lindell and Edling (1996) demonstrated that bacteria living in surface waters (0 to 3-4 metres) in the tropical Lake Kariba are negatively affected by solar radiation, with inhibition of bacterial production and enzyme activity probably being due to both UV-A and UV-B light. Bacterial production at the surface was nearly four times less than the production at 10m, although bacterial numbers were only 26% less.

  Bacterioplankton from anoxic hypolimnia are often abundant and of unusually large cell size (Cole et al., 1993; Lind et al., 2001, Christian et al., 2002; Kalff, 2002). However, anoxia is not always associated with large cells (Lind et al., 2001). Possible reasons for the large cell size suggested by Cole et al. (1993) are organic enrichment, sediment nutrient release, species shifts and lack of grazing. Christian et al. (2002) showed that larger cells in low oxygen and anoxic hypolimnia are a function of increased supply of organic substrate and/or inorganic nutrients.

- **Longitudinal distribution**

The longitudinal gradients of chemical and biological parameters, nutrient and substrate availability, and consequently the development of pelagic grazers in canyon-shaped reservoirs have been shown to influence microbial activity and bacterial community composition (Simek et al., 2001a; Masin et al., 2003; Jezbera et al., 2003; Tietjen and Wetzel, 2003; Gasol et al., 2002).

In the riverine zone of eutrophic reservoirs, primary producers often have a limited role due to water turbidity with resulting light limitation (Hejzlar and Vyhnalek, 1998; Comerma et al. 1999). Most matter transformation and nutrient cycling thus occurs through enhanced activity of heterotrophic microbial communities (Armengol et al., 1999; Comerma et al., 1999; Simek et al., 2001). A continuous supply of substrates by a river results in a gradient of nutrient and substrate concentration and microbial succession downstream (Hejzlar and Vyhnalek, 1998; Armengol et al., 1999; Kalff, 2002).

Masin et al., (2003) and Jezbera, et al., (2003) found that two reservoirs of different trophic status and with different inflow loading showed different specific longitudinal changes in bacterial community composition, which were related mainly to allochthonous input and longitudinal patterns of primary production and nutrient availability. In the riverine zone, the main source of carbon that fuels microbial processes is in the allochthonous organic matter, while downstream, through the transition and into the lacustrine zones, autochthonously produced organic carbon becomes more important (Masin et al. 2003).

The pattern of longitudinal distribution of phytoplankton and zooplankton communities in reservoirs has been known for some time (e.g. Urabe, 1989; Pinel-Alloul, 1995; Masin et al. 2003). However, recent studies carried out in the Sau Reservoir in Spain mapped longitudinal changes in the microbial food web in relation to other biological and chemical variables (Comerma et al., 1999; Simek et al., 2001; Gasol et al., 2002), illustrating that the communities of protistan bacterial grazers displayed a clear longitudinal pattern; bacterivory of heterotrophic nanoflagellates dominated the upper inflow area of the reservoir, while ciliates were the most important bacterivores in the lacustrine area (Comerma et al., 1999; Simek et al., 2001; Simek et al., 1999a).

The important role of protistan bacterivory in shaping longitudinal distribution of bacterial community composition was confirmed in experiments conducted in the Sau Reservoir (Simek et al., 1998; Gasol et al., 2002). Lindstrom (2000) also confirmed the important impacts of micro zooplankton grazing on shaping bacterial community composition. It may also be affected by nutrient supply (Pace and Cole, 1994; Thingstad and Lignell, 1977), or by host-specific viral lysis (Fuhrman, 1999 in Masin et al., 2003).

Recent focus on carbon flux dynamics has raised new interest in detailed studies of longitudinal changes in reservoirs (Simek et al., 1998; Armengol et al., 1999; Comerma et al., 2001)

- **Environmental factors affecting heterotrophic bacterioplankton**

The availability of organic substances, a common correlate with bacterial activity and biomass (Garnier and Benest, 1990; Balogh and Voros, 1997) has been identified as much more significant than predatory controls on the bacterial community (Pace and Cole, 1994). A correlation between temperature and bacterial biomass has been reported by many workers (Findlay et al., 1991; White et al., 1991; Wikner and Hagstrom, 1991; Coveney and Wetzel, 1995; Tietjen and Wetzel, 2003) and the correlation between phytoplankton and bacterial community dynamics has been made in many previous studies (Tietjen and Wetzel, 2003). These correlations support the hypothesis that both bacterio- and phytoplankton are influenced by similar environmental parameters, especially temperature and nutrients.

Many studies imply that the bacterial community also depends upon the phytoplankton for organic matter (Tietjen and Wetzel, 2003). However, this view is undergoing revision (Kalff, 2002) with the recognition that bacteria, characterized by a ten-fold lower C:P ratio than phytoplankton (Fagerbakke et al., 1996, in Kalff, 2002) compete successfully with algae for limiting inorganic nutrients when the organic matter supply is not their principal constraining factor.

It has been widely accepted that predators can change bacterial community composition by feeding on larger more active cells (e.g. del Giorgio et al., 1995), and/or specific groups of bacteria (Simek et al., 1997; Jurgens et al., 1999). Factors that induce changes in bacterial community composition have become one of the central issues in aquatic microbial ecology (Simek et al. 1999). Protozoan grazing on bacteria is size selective (Chrzanowski and Simek, 1990; Gonzalez et al., 1990a), thus inducing a strong morphological shift in bacterial communities towards cells that are resistant to grazing (Sibbald and Albright, 1988; Bianchi, 1989; Gude, 1989, in Simek et al., 1999a). Different bacterial strains are consumed with varying growth efficiency for the consumer (Mitchell et al., 1988; Gonzalez et al., 1990a; Irriberi et al., 1994, in Simek et al., 1998). Finally, through feedback mechanisms, grazing is known to affect substrate availability for bacteria, resulting in a tight relationship between protistan grazing and the metabolic and physiological activities of bacterioplankton (Simek et al., 1990).

Simek and Straskrabova (1992) obtained data that showed the existence of significant seasonal changes in grazer control of bacterioplankton in the mesotrophic Rimov Reservoir in the Czech Republic. During the spring period and during the clear-water phase, bacterial growth was not regulated by protistan grazing alone (only about 20% of bacterial production was consumed by protozoa). On the other hand, during the summer-fall period, equilibrium between bacterial production and protozoan grazing was evident. They concluded from their study that a steady top down control of bacterial by protozoa did not exist, and that bottom up control might be effective during the spring period. Other losses such as grazing by meso- and macro zooplankton could be more significant than protozoan control.

- **Seasonal distribution and viability**

A close correlation often exists between seasonal changes in phytoplankton biomass and resulting increased photosynthetic productivity, and heterotrophic bacterioplankton biomass and productivity (Wetzel, 2001; Tietjen and Wetzel, 2003; Voros et al., 2003; Di Siervi et al., 1995). An increase in summer bacterial biomass has also been recorded by Coveney and Wetzel, 1995; Ochs et al., 1995; Jugnia et al., 2000; Tietjen and Wetzel, 2003.

Tietjen and Wetzel (2003), using a staining procedure that provides information on the physiological state of bacteria, found the bacterial biomass was highest during the late summer in a small reservoir in Alabama, with 50-70% of the cells being viable. This was associated with the mixing of large accumulations of hypolimnetic nutrients, hypolimnetic DOC and dissolved oxygen to the entire water column. Bacterial numbers also increased following a >10-fold increase in turbidity associated with a major precipitation event, but only about 10% of these cells were viable. This was probably the result of an influx of large numbers of soil bacteria, dead cells and cells compromised by the sudden introduction to an aquatic environment, associated with the turbid clay inflow associated with the storm.

- **Humic lakes**

Humic lakes generally have higher DOC concentrations than clear-water systems, which results in a higher carrying capacity for bacterioplankton in humic lakes. In this way, the DOC of allochthonous origin in humic lakes can constitute an input to the aquatic biota, counteracting its negative effect on primary production due to light attenuation (Balogh and Voros, 1997).

In humic waters, the relative abundance of larger bacteria is significant (Pinel-Alloul and Letarte, 1993) and these larger bacteria contribute disproportionately to the total bacterioplankton production (Kalff, 2002).

- **Role in the microbial food web**

Historically, less attention has been paid to the role of pelagic microbial food webs in reservoirs than in marine ecosystems and natural lakes (Masin et al., 2003). However, during the last two decades, microbial studies have demonstrated the existence of a diverse microbial community in aquatic planktonic ecosystems where microbial production is integrated in the pelagic food web at all levels (Gaedke et al., 1995, in Comerma et al., 2003; Kalff, 2002).

In terms of food web structure and energy flow, the heterotrophic bacteria and phytoplankton occupy the same trophic level (Jones, 1992); in the sense that both nourish the primary consumers. From a food web perspective, heterotrophic bacterial production can be considered as primary production in that it converts dissolved to particulate matter or synthesises organic matter by chemosynthetic autotrophy (Kalff, 2002; Kim et al., 2000).

In systems with significant autochthonous input (e.g. humic lakes or the riverine zone of some reservoirs), the bacteria may act as a primary source of organic carbon, passing organic matter imported from external sources up the food web (Tranvik, 1988, in Balogh and Voros, 1997).

The "microbial loop" is simply a model of the pathways of carbon and nutrient cycling through microbial components of pelagic aquatic communities (Wetzel, 2001). In addition to bacterial uptake of nonliving organic matter, many direct links exist among algae, bacteria and other heterotrophic microbes. The "microbial loop" complements the classical food web leading from nanoplankton to the macro zooplankton and up (Kalff, 2002). (See Figure 11)

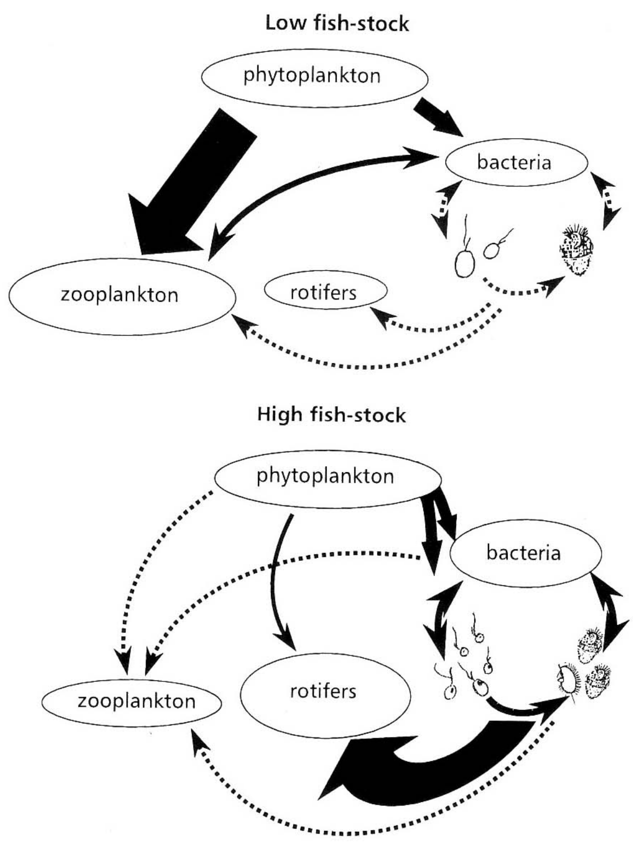


Figure 11. A simplified scheme of the contrasting importance of the major carbon fluxes among large zooplankton, rotifers, phytoplankton, and the most important members of microbial food webs (bacteria, HNF and ciliates) under high fish stock versus low fish stock. Black arrows and their thickness indicate relative importance of the principal carbon fluxes; dotted line arrows indicate the carbon fluxes of minor importance (after Simek et al., 1999)

Bacterivory by protists (especially nanoflagellates) and other zooplankton is a dominant mortality factor in natural bacterioplankton communities (usually about 50%) (Wetzel, 2001; Kalff, 2002; V-Balogh and Voros, 1997; Comerma et al., 2001). Protozoans are in turn preyed upon by micro zooplankton such as rotifers (Arndt, 1993; Ooms-Wilms, 1997) and small crustaceans, or by macro zooplankton such as the larger filter-feeding cladocerans. Some of the large herbivorous cladocerans can and do graze directly on larger bacteria

(Kalff, 2002; Thouvenot et al., 1999b). Paterson et al., (1997) suggested that metazoan zooplankton in a newly flooded reservoir could use bacteria as an energy source.

In fact, unless a high fraction of bacterial carbon flows directly into large filter feeding cladocerans that are fed upon by fish, the principal role of the microbial food web lies in the degradation (respiration) of organic matter (Kalff, 2002). The reason for this is the loss of energy at each level of the food web. The conversion efficiency of the bacterivores and their predators is not high enough to ensure the transfer of bacterial carbon through more than about a three-step chain. Three to five steps is more likely for the microbial loop and it is evident that little would remain for transfer to macro zooplankton and then on to fish. Only when there is a major one-step transfer from bacteria to large bacterivorous macro zooplankton, is it likely that appreciable amounts of bacterial carbon would become available to fish (Wylie and Currie, 1991).

The microbial food web, therefore, appears to be primarily an energy "sink" in the plankton, but, in the process serves as a crucially important nutrient "link" (Kalff, 2002). Where it serves as a loop, it is largely responsible for recycling critical nutrients within the euphotic zone (Nakano, 1994, in Kalff, 2002). In the process it greatly reduces losses by sedimentation of particulate nitrogen and phosphorus from the site of primary production in the water column.

2.6.1.1.2 Autotrophic Cyanobacteria

Cyanobacteria exhibit extraordinary structural and functional heterogeneity, which is reflected in their diversity of form and habitat (Kalff, 2002). Many different factors determine the dominance of particular species in different environments – light, nutrients, CO_2,, pH, temperature, turbulence, competition and selective grazing by predators (Kalff, 2002). In addition, some cyanobacteria can regulate the density of their individual cells (Oliver, 1994) and thus their buoyancy (Brookes et al., 1999).

- **Buoyancy**

  When $z_{mix}/z_{eu} = 1$, cells are constantly illuminated and photosynthesis is continuous during the daylight period. If the mixing depth becomes greater than the depth of light penetration, then phytoplankton spend a proportion of their daylight period in the dark where photosynthesis cannot take place but respiration can (Bormans et al., 2005). In stratified systems, buoyant cyanobacteria have a clear advantage over other phytoplankton groups to stay in the well-lit surface mixed layer.

  Brookes et al. (1999) observed that *Anabaena circinalis* filaments accumulated at the surface of Chaffey Dam in New South Wales, Australia, as diurnal surface layer thermal stratification developed with many previously buoyant, homogenously distributed colonies accumulating in the top 2 metres. It was found that colonies suspended at low light aggregated together, increasing the size of the biomass unit, which contributed to an increased flotation velocity, a strategy to increase light capture. This attribute, shared by *A. circinalis* and *Microcystis aeruginosa* would appear to make these species well suited to an environment which is either shallow and stratifies daily, or deeper water bodies with a permanent deep-water thermocline well below the euphotic zone overlain by diurnal surface layer stratification.

- **Nitrogen uptake**

Some cyanobacteria are able to fix atmospheric nitrogen. They belong to the filamentous genera that have heterocysts, specialized heavy-walled cells that provide the anoxic conditions required (Kalff, 2002). Nitrogen fixing cyanobacteria require the energy of the sun to reduce N_2 to the organic nitrogen of protoplasm with the result that the process varies diurnally and typically declines with depth. It is energetically expensive and consequently cyanobacteria able to fix nitrogen, stop when nitrate or ammonia become readily available. However, this ability gives cyanobacteria a great advantage over other phytoplankton in the seasonal succession of species. The cyanobacteria able to fix nitrogen often gain the upper hand, when it becomes limiting to other species. This is especially relevant in conditions of eutrophication, when N:P stoichiometry is altered.

In the Marcali Reservoir, a shallow hypertrophic system, Presing et al. (1997) found that despite conditions, including the N:P ratio, being favourable for nitrogen fixing cyanobacteria, they did not occur. It was suggested that although they are capable of using nitrate as a nitrogen source, they are not able to compete with other phytoplankton for it. According to Blomqvist et al., (1994) this may be attributed to differences in the enzymatic processes by which nitrogen is assimilated. Cyanobacteria are not only uncompetitive for nitrate, but above a certain level it reduces their growth. The general rule that ammonium is preferred over nitrate as a nitrogen source for phytoplankton holds true, especially for cyanobacteria (Presing et al., 1997).

- **Resistance to herbivory**

Cyanobacteria are resistant to herbivory because of their low nutritional value for zooplankton, aggregation into large, indigestible colonies or filaments and their ability to synthesize toxins. These traits may partly explain their dominance in many freshwater systems (Paerl, 1988, in Panosso et al., 2003). However, some copepods have high ingestion rates on cyanobacteria filaments or colonies (Koski et al., 2002), as they are able to capture and handle large particles (DeMott, 1990). In addition, possibly through physiological adaptations, some herbivores are able to graze on cyanobacteria and develop resistance to their cyanotoxins (Fulton and Paerl, 1987).

The problem of cyanobacterial blooms and their management is discussed in Section 3.1.2

2.6.1.1.3 Photosynthetic Bacteria

Green and purple photosynthetic bacteria, unlike planktonic heterotrophic bacteria that normally require dissolved oxygen but need no light, require anoxic conditions and need light. They are found in abundance at the bottom of illuminated oxyclines in either the water column or the sediments. The anoxygenic photosynthesis of green and purple bacteria, in contrast to the oxygenic photosynthesis of the eukaryotic algae and cyanobacteria, is primarily dependent on the availability of reduced forms of sulphur.

Purple sulphur bacteria usually dominate closer to the water surface than green bacteria. The green bacteria have highly efficient light-harvesting units allowing them to grow photosynthetically deeper in the water column where they benefit from higher concentrations of H_2S at the chemocline.

Many of the purple sulphur bacteria have flagellae, allowing them to migrate vertically in chemoclines in response to phototaxic or chemotaxic cues. Most planktonic species of green, and some of the purple sulphur bacteria contain gas vacuoles that impart buoyancy control and allow them to position themselves in the non-turbulent metalimnia to optimize light and substrate availability. (Kalff, 2002)

2.6.1.2 Bacteria in the sediments

The sediment layer is notoriously difficult to sample and most methods of identifying bacterial density and species composition in sediment samples are unsatisfactory, and very laborious. The composition of the microbial communities in sediments is therefore largely unknown (Kalff, 2002; Roske et al., 2002; Wobus et al., 2003). Recent studies involving the application of molecular biological methods have enhanced the knowledge of bacterial diversity in different ecosystems (Head et al., 1998; Torsvik and Ovreas, 2002, in Wobus et al., 2003).

2.6.1.2.1 Distribution

The sediment bacterial abundance, expressed per unit volume of surface sediment is, depending on the water column thickness (depth), typically between 2 and 1000 times greater per unit volume than in the water column above (Schallenberg and Kalff, 1993). The reasons for high densities may be low grazing pressure compared to the overlying water, low temperature, and the presence of particulate organic material and associated microbial species from erosional processes in the catchment (Uhlmann et al., 2000). Their size is typically several times larger than the free-living bacteria in the water column (Kalff, 2002). Numbers decrease rapidly with increasing depth in the sediments (Wetzel, 2001).

2.6.1.2.2 Structure and dynamics of microbial communities

Knowledge of the structure and dynamics of microbial communities in lake sediments is still very limited (Uhlmann et al., 2000).

Wobus et al. (2003) in a study of sediment bacteria from reservoirs of different trophic status, found that the species composition was affected to a larger extent than bacterial density, and the diversity of the microbial sediment communities seemed to be hardly influenced by the trophic state.

2.6.1.2.3 Rates of production

Organic matter and associated nutrients play a major role in determining sediment-production rates (Bostrom et al., 1989; Sander and Kalff, 1993; Wetzel, 2001). Microbial decomposition usually exhibits its highest intensity in the uppermost sediment layer (Roske et al., 2002).

Sediment-water content is the single best indicator of bacterial **abundance** in the sediments, with highest numbers at intermediate water content levels (Kalff, 2002). Bacterial heterotrophic **production** is a function of the organic content of sediments, and flushing rate (Schallenberg and Kalff, 1993). Increased flushing results in greater export of organic matter and inorganic nutrients from catchments that is reflected in enhanced phytoplankton and bacterioplankton production rates.

2.6.1.2.4 Importance

The bottom sediment of a reservoir is the compartment with the highest concentration of most chemical constituents as well as the highest microbial biomass (Wobus et al., 2003). Therefore abiotic and microbial processes in the sediment have a major influence on the nutrient balance in the water body. The relative importance of sediments as a site for organic matter decomposition and bacterial production increases with decreasing water depth. Rapid sedimentation of particles from shallow water columns decreases the time available for decomposition in the water column. (Kalff, 2002)

In extremely oligotrophic systems, total numbers of bacteria in the sediments can be less than the cumulative total of the entire overlying water column, which indicates that most of the readily decomposable organic matter has been mineralized before it reaches the sediments (Kalff, 2002). Roske et al. (2002), however, found high bacterial numbers in the sediments of oligotrophic reservoirs, from which they concluded that they could tolerate long periods of starvation.

The role of bacteria in the mobilization and fixation of the important nutrients is discussed in Section 2.5.

2.6.1.2.5 Anaerobic decomposition in sediments

Bacterial metabolism rapidly produces anoxic reducing conditions where various substances e.g. nitrates, manganese and iron oxides and sulphate replace molecular oxygen as electron acceptors (Wetzel, 2001). Accordingly, methane generation can be very intense in anaerobic sediments. At the aerobic-anaerobic interface (sediment water interface or in the metalimnion overlying an anoxic hypolimnion), much of the methane is converted to CO_2 by aerobic methane oxidizing bacteria.

Most of the very high bacterial production of sediments is mineralized and respired as CO_2 and CH_4 within the microbial food web and eventually enters the atmosphere. Less than 20% of this rich bacterial production is expected to reach higher trophic levels in the lake ecosystem.

2.6.2 Phytoplankton

2.6.2.1 Reservoir primary producers

Primary producers in reservoirs (as in rivers and lakes) occur in four categories – phytoplankton (algae), planktonic phototrophic bacteria (which includes cyanobacteria, and purple and green bacteria), attached algae (periphyton) and rooted macrophytes (Kimmel et al., 1990; Straškraba and Tundisi, 1999).

Turbidity due to suspended silts and clays eroded from the watershed imposes significant light limitations on primary productivity. However, large water-level fluctuations often restrict the development of attached algal and rooted macrophyte communities in reservoirs and thus ensure a maximal contribution of the planktonic rather than benthic primary producers (Kimmel and Groeger, 1984, in Kimmel et al., 1990). Although photosynthetic bacteria are probably common in clear, stably stratified reservoirs and contribute to the development of chlorophyll peaks, their contribution to reservoir primary production is certainly small in comparison to that of the phytoplankton (Knowlton and Jones, 1989; in Kimmel et al., 1990).

The relative quantities of the different categories result from their competition for resources (mostly light and nutrients) as well as their ability to withstand environmental variations (water level and turbulence) or biological interactions (Thomas et al., 2000). In the case of water level fluctuations, the success of sessile (rooted or attached) plants depends on their ability to colonise new substrates to avoid desiccation or alternatively, insufficient light, when water levels change (Thomas et al., 2000). However, established, attached micro algal communities may tolerate moderate changes in water level more easily than the colonizing of new substrates (Thomas et al., 1999, in Thomas et al., 2000). Some strategies which epilithic algae use for survival are variability in life cycles (the organisms change their habitats in the adult phase), and physiological adaptations (they develop thickened mucilaginous sheaths to avoid desiccation) (Casco, 1997).

In shallow waters, attached primary producers can reduce the flow of mineral nutrients from the bottom sediments to the water column. Epipelon may retain part of newly mineralized nutrients at the sediment surface (Carlton and Wetzel, 1988; Hansson. 1988, 1989), as can macrophytes (Thomas et al., 2000), therefore having a competitive advantage over phytoplankton.

2.6.2.2 Environmental factors controlling primary production in reservoirs

Although the growth and biomass of phytoplankton in reservoirs and lakes are controlled by similar parameters, marked differences between these two types of aquatic ecosystems have been reported (Wetzel, 1990; Komarkova and Hejzlar, 1996; Menshutkin and Klekowski, 2001; Uhlmann, 1998; Carpenter and Pace, 1997; Timchenko and Oksiyuk, 2002).

Reservoir phytoplankton productivity and biomass levels are dependent on interrelated physical, chemical and biological factors that are themselves functions of the climatic and hydrological regimes, the size and nature of the watershed, reservoir basin morphology, nature and volume of river inflow, and the reservoir food-web structure (Kimmel et al., 1990; Komarkova and Hejzlar, 1996; Naselli-Flores and Barone, 2000; Horn, 2003). In particular, factors including precipitation, evaporation, flow rate, stratification, irregular water use (including drawdown), concentrations of suspended solids and floral and faunal diversity may influence the growth of phytoplankton (Albay and Akcaalan, 2003).

Basically, algal growth is determined by the relative availability of some limiting factor, and increases as that factor increases, until another factor becomes limiting (Kimmel et al., 1990). The fundamental factors controlling phytoplankton productivity are temperature (Perry et al., 1990; Kureyshevich, 2004), light (low light is often due to turbidity) (Kureyshevich, 2004; Arfi et al., 2001; Perry et al., 1990; Menshutkin and Klekowski, 2001), and nutrient availability (Strojsova et al., 2003; Naselli-Flores and Barone, 1994; Arfi et al., 2001; Horn, 2003; Menshutkin and Klekowski, 2001). However, the concept of a complex of environmental factors controlling algal growth might be more appropriate than that of control by a single limiting factor (O'Brien, 1972, 1974; Harris, 1978, 1980a, 1988, all in Kimmel et al. 1990). In reservoirs, water retention time is also important factor controlling phytoplankton productivity (Perry et al., 1990; Komarkova, 1993).

2.6.2.2.1 Nutrients

The availability of a particular nutrient may determine the presence or absence of species of algae or cyanobacteria within the phytoplankton (Sommer, 1993). Phosphorus is the most commonly limiting nutrient in fresh waters although the P requirement of phytoplankton is relatively small (Graham and Wilcox, 2000). Only orthophosphate can be taken up in substantial amounts (Taft et al., 1977; Heath and Edinger, 1990; Cotner and Wetzel, 1991), therefore orthophosphate depletion occurs periodically in the epilimnion of most natural waters as a consequence of phytoplankton growth (Wetzel, 2001).

One of the possible mechanisms which enables phytoplankton to overcome P limitation is the production of extra cellular phosphatases. Most phosphorus-deficient algae and cyanobacteria produce extra cellular, mostly cell-attached phosphatases, presumably to make ambient organically- bound phosphorus available. However, the distribution of phosphatase activity among natural phytoplankton populations and its ecological significance is largely unknown (Strojsova et al., 2003). These authors are of the opinion that this new information (obtained by means of the new enzyme labelled fluorescence technique – ELF) has extended freshwater studies on P-limited phytoplankton and offers new areas for future research).

2.6.2.2.2 Perturbations

The effect of environmental variations can be measured as changes in the phytoplankton dynamics, and according to the Intermediate Disturbance Hypothesis (IDH), perturbations of intermediate intensities or frequencies support high values of diversity in phytoplankton communities. A disturbance can be defined as a non-biotic, stochastic event that results in distinct and abrupt changes in the composition (of phytoplankton) and that interferes with internally driven progress towards self-organization and ecological equilibrium (Reynolds et al., 1993).

An autogenic sequence of different phytoplankton associations has been described by Reynolds (1997, 2001, and 2002) in temperate lakes. However, many aquatic systems, including many reservoirs, are highly perturbed. In these cases, allogenic factors change the expected trajectory, and other functional groups of algae adapted to those perturbations can appear (Reynolds, 1993). Phytoplankton shows patterns of a strong relationship between physical changes in the environment and biological processes, especially within the euphotic zone. These interactions usually prevent equilibrium conditions (Sousa, 1984 in Leitao et al., 2003), affecting the growth and persistence of phytoplankton populations and preventing the attainment of a steady state community (Reynolds, 1994, 1997).

Some causes of perturbation recorded in the literature are storms (Barbiero et al., 1999), floods and droughts (Arfi et al., 2000; Thomas et al., 2000; Hart, 2006), turbulence (Peterson and Hoagland, 1990), destratification and mixing (Arfi et al., 2000; Matsumura-Tundisi and Tundisi, 1997; Hart, 2006), as well as reservoir operation (Leitao et al., 2003; Oksiyuk et al., 2001; Bertrand et al., 2004).

The magnitude of water level fluctuations may be so great in some reservoirs that they can never be considered to be in a steady state (Ford, 1990), and are often described as "river-lake hybrids" (Kimmel et al., 1990). In Mediterranean climates, reservoirs often become so shallow that they can no longer accommodate a stable thermocline (Calvo et al., 1993 in Barone and Naselli-Flores, 1994). Such unstable conditions tend to affect the dynamics of plankton communities (Barone et al., 1991; Barone and Naselli-Flores, 1994). In Lake Arancio, a hypertrophic reservoir in Sicily, the seasonal cycle of the phytoplankton was strongly related to the water level fluctuations (Barone and Naselli-Flores, 1994).

In regions of the world which experience monsoons, the intensity of the perturbations is often extreme (An and Park, 2002), resulting in distinct limnological differences between pre- and post monsoon seasons (Zafar, 1986; Lohman et al., 1988; Khondker and Kabar, 1995; Silva and Davies, 1987, all in An and Park, 2002). Such a disturbance or reorganization is, however, not a random phenomenon but a consistent cyclic pattern which influences functional processes in affected aquatic systems (An and Park, 2002).

Similarly, the drastic impact of weather associated with El Nino events, on catchment and hydrological processes can cause unexpected changes in physical, chemical and biological properties of freshwater aquatic ecosystems (Bouvy et al., 2003; Gerten and Adrian, 2001). Changes in climatic conditions linked to the 1997-1998 El Nino phenomenon in the Brazilian Northeast, have been responsible for modifications of the ecological conditions of many reservoirs (Bouvy et al., 2000).

2.6.2.2.3 Retention time

In general, when retention times are short, higher flushing rates cause increased export of plankton from the reservoir. When the ratio of flushing rate: specific growth rate is >1, the growth rate is insufficient to compensate for losses by flushing. This can lead to changes in community structure associated with species differences in intrinsic growth rates. Shorter retention times also alter local limnological conditions through the introduction of nutrients, or water of a different temperature or turbidity, affecting growth rates and species composition of phytoplankton. When retention times are longer, losses of algal cells associated with sedimentation to the bottom, and with grazing become more important factors influencing cell density and type (Perry et al., 1990).

Numerous run-of-the-river mainstream impoundments have retention times of less than 7 days, which influences phytoplankton production and species composition in such systems (Kimmel et al., 1990).

2.6.2.2.4 Gain and loss processes determining primary production

The phytoplankton standing crop (or algal biomass) in reservoirs is determined by both the rate of biomass production and the rate of biomass loss (Kimmel et al., 1990). Physical and chemical factors influence the availability of light and nutrients for photosynthesis, and temperature dependent metabolic rates. On the other hand they influence phytoplankton losses due to cell sedimentation (Galvez et al., 1993), and washout. Biotic factors influence photosynthetic efficiency, extra cellular excretion of photosynthate, and phytoplankton losses resulting from predation or parasitism.

2.6.2.3 Temporal and spatial variation in primary production

The quantification of temporal and spatial phytoplankton distribution in reservoirs is of major concern to limnologists (Gurbuz et al., 2003). Temporal and spatial distribution is influenced by both loading from the watershed and by internal processes within the reservoir ecosystem (Wetzel, 1990)

2.6.2.3.1 Short term variation

Short-term variations of phytoplankton biomass, primary productivity and species diversity often occur as a result of perturbations (discussed above) (Calijuri and Dos Santos, 1996). The frequency of disturbances has a critical influence on the diversity of phytoplankton and on the establishment of equilibrium (Sommer et al., 1993). A number of studies have shown that fluctuations in the stability of surface water in scales of time of about 10 days, are responsible for changes in algal composition and diversity, and that the biomass and taxonomic composition of phytoplankton can be controlled by changes in the mixing layer over a few days (Sommer et al., 1993; Reynolds et al., 1987; Capblanq and Catalan, 1994).

Evaluation of the response of a phytoplankton community to the variability of a water body would provide further understanding of the processes in stressed environments (Calijuri and Dos Santos, 1996).

2.6.2.3.2 Seasonal variation

Distinct seasonal patterns and successions of algal populations and biomass are observed in phytoplankton communities (Wetzel, 2001).

In temperate reservoirs, growth is greatly reduced in winter when both light and temperature are low. Phytoplankton numbers and biomass normally increase greatly in the spring under improved light conditions and often consist predominantly of diatoms adapted to low temperatures. This spring maximum is generally short-lived (less than 3 months) and is often followed by a period of low numbers and biomass that may extend through the summer.

In more eutrophic waters with high phosphorus loading, silica concentrations are commonly reduced in the spring to levels inadequate to support large diatom populations. Summer populations of green algae often then flourish until concentrations of nitrogen are severely reduced. Under these conditions, nitrogen-fixing bacteria have competitive advantage and often proliferate.

Independently of taxonomic affiliation, ecological strategies of phytoplankton also play a part in their succession in aquatic systems. Following Grime (1979), Reynolds, (1988a) described phytoplankton strategies in relation to intensity of environmental stress (availability of light and nutrients) and disturbance (Padisak, 2003).

If both stress and disturbance are low, rich resources favour species with high intrinsic growth rates. These are C-strategists (exploitative colonists), typically small green algae, cryptophytes and centric diatoms. If the stress factor is high, but disturbances insignificant, only effective competitors can survive. These are stress tolerant S-strategists, typically cyanobacteria, both colonial and filamentous forms, some chrysophytes and large colonial green algae. The third type, the ruderal or disturbance- tolerant R-strategists are adapted to conditions of high disturbance, reduced light, low temperatures and plentiful nutrients. Typically they are diatoms (which need the disturbance to keep them suspended in the water column), and some green algae (Padisak, 2003). This is a very simplified account; these strategies are not wholly exclusive and individual species may possess intermediate characteristics. Put simply, the species composition of phytoplankton at any given time, depends on the ability of some species to do things more effectively or faster than others when conditions are perceived to be near-ideal, or on their relative abilities to function adequately under conditions that are markedly sub-ideal with respect to resource availability or energy supply (Reynolds, 1999).

In tropical reservoirs total phytoplankton biomass is more constant and greater than in temperate waters, with the maximum often in winter (Wetzel, 2001), a trend observed locally by Hart (2006).

If the system is not perturbed by outside influences, the seasonal periodicity of phytoplankton biomass is reasonably constant from year to year (Wetzel, 2001). By their very nature, however, reservoirs tend to be modified continuously, with correspondingly inconsistent patterns or trends in phytoplankton composition and biomass.

2.6.2.4 Vertical and horizontal gradients influencing the distribution and amount of primary production in reservoirs

Lewis (1995) maintains that the (vertical) stratification regime is the most important hydroclimatic factor regulating biotic processes in lake ecosystems generally. Given the wide variability in stratification intensity and duration in reservoirs, hydrodynamics plays a correspondingly large part in their biotic structure and function, as emphasized by Allanson (2004).

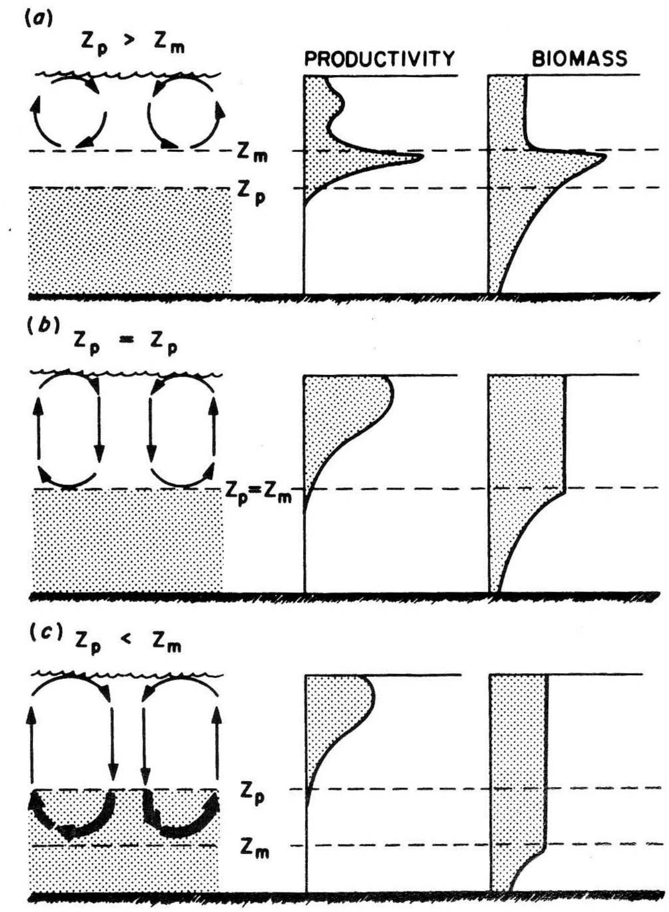


Figure 12. The influence of photic-layer depth (Z_{eu}) and mixed-layer depth (Z_{mix}) on the vertical distribution of phytoplankton productivity and biomass (after Kimmel et al., 1990).

In highly transparent reservoirs, the depth of the photic layer (Z_{eu}) may exceed that of the mixed layer (Z_{mix}) and phytoplankton production can occur in both the epilimnion and in euphotic portions of the metalimnion and hypolimnion (Figure 12a). Commonly by midsummer the mixed layer becomes nutrient depleted, and metalimnion or hypolimnion peaks in phytoplankton biomass and productivity, the so-called deep chlorophyll maxima, develop. In stratified reservoirs, deep chlorophyll peaks can be formed by the accumulation of viable cells settled from the mixed layer, by active growth of cells adapted to the low light/high nutrient environment of the deeper layers and/or by the subsurface transport of phytoplankton entrained from more productive surface waters upreservoir (Kimmel et al., 1990).

If Z_{eu}=Z_{mix}, phytoplankton circulate through a vertical light gradient ranging from high, potentially photo inhibiting light intensities at the surface, to the compensation level (where photosynthesis=respiration at Z_{eu}. (See Figure 12b) In turbid, well mixed reservoirs Z_{mix} often exceeds Z_{eu} and planktonic algae are exposed not only to rapidly fluctuating light but intermittently to aphotic conditions Figure 12c. Because phytoplankton P<R in the aphotic

zone, primary production must be limited to some extent depending on the relative amount of time cells spend at depths >Z_{eu} (Kimmel et al, 1990).

Vertical water movements

Studies of the dynamics of plankton organisms and the interpretation of primary production measurements require thorough quantitative knowledge of vertical (and horizontal) advection (Schanz et al., 2001).

Vertical displacement causes changes of layer thickness which influences phytoplankton abundance at various depth levels (Thomas 1950, in Schanz et al., 2001). The most pronounced vertical movements occur around the thermocline and are seiche-driven (Gaedke and Schimmele, 1991).

Short-term mixing events in the euphotic zone are of considerable quantitative importance as a controlling mechanism of the primary production of phytoplankton. These short-term events may last for only minutes or hours and also change on a daily basis (Horn and Goldman, 1994). They indicate unstable conditions and introduce variable situations in the light climate and nutrient availability. Furthermore, in reservoirs, the nature of mixing is complicated by the changing patterns of flow rate, which determine retention time (Matsumura-Tundisi and Tundisi, 1997).

The vertical movement of water plays an important role in controlling the distribution of nutrients in the euphotic zone, photosynthetic processes and the distribution of phytoplankton. This constant change in the compartmentalization of the system produces intermediate disturbances as discussed above (Matsumura-Tundisi and Tundisi, 1997). Vertical mixing regulates the distribution of ions and suspended particles with respect to depth. Water column mixing processes are undoubtedly one of the dominant controls on algal community structure and succession (Reynolds, 1984).

Longitudinal gradients

The horizontal zones along the axis of a typical reservoir were described in Section 2.3, but will be discussed here with reference to primary production.

The riverine zone is characterised by higher flow, shorter water residence time, higher levels of available nutrients and suspended solids, and greater light extinction relative to the downstream zones of the reservoir. Abiotic turbidity often limits light penetration and thereby limits the thickness of the photic layer; consequently primary productivity is often light-limited (Kimmel et al., 1990).
The transition zone is characterised by higher phytoplankton productivity and biomass occurring in conjunction with increasing basin breadth, decreasing flow velocity, increased water residence time, sedimentation of silt and clay particles from near-surface waters, and increased light penetration. Because light and nutrients are available for algal photosynthesis, this can be the most fertile region in the reservoir (Kimmel et al., 1990; Armengol et al., 1999). Noguiera (2000) recorded higher phytoplankton diversity in the transition zone of the Jurumirim Reservoir in Brazil.

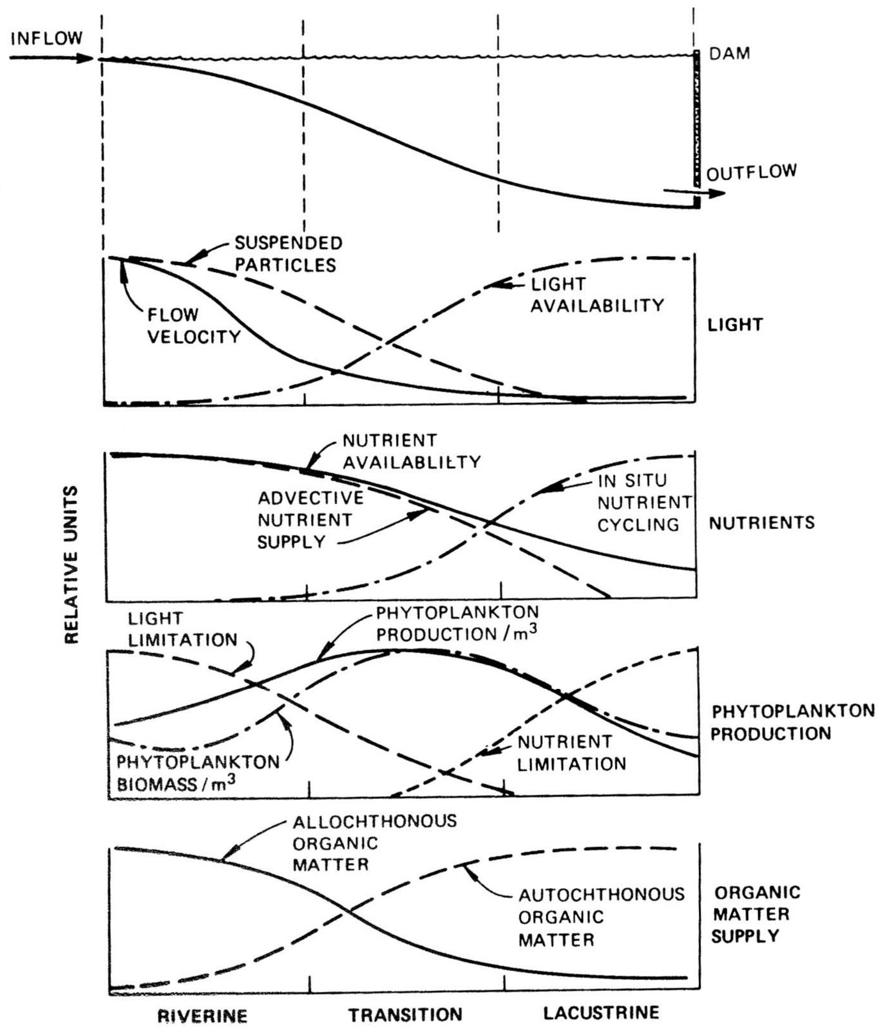


Figure 13. A cross-sectional view of gradients in environmental factors affecting phytoplankton productivity and biomass, and of the relative importance of allochthonous and autochthonous organic matter along the longitudinal axis of an idealized reservoir. (After Kimmel et al., 1990).

The lacustrine zone usually has a longer water residence time, lower concentrations of dissolved nutrients and suspended abiogenic particles, higher water transparency and a deeper photic layer. However, the phytoplankton productivity is reduced (often nutrient limited) during most of the growing season and is supported primarily by in situ nutrient cycling rather than by advected nutrients (Kimmel et al., 1990). (Figure 13)

The photic layer/mixed layer relationship discussed above is also relevant to the horizontal zones of a reservoir. All three patterns described can occur within a single reservoir as photic layer depth increases and nutrient availability decreases along the riverine to lacustrine zone gradient (Kimmel et al., 1990).

2.6.2.5 Importance and quantification

Properly collected and analysed phytoplankton from any reservoir can be beneficially utilized in the identification process of its trophic level and primary productivity (Reynolds et al., 2002). Artificial neural network approaches employ easily and precisely measurable environmental parameters to quantify temporal and spatial phytoplankton distribution in reservoirs. Utilization of neural network approaches can be considered a remedy to eliminate the difficulties of the classical approach of arduous counting and volumetric calculations (Gurbuz et al., 2003).

Early pioneering works have utilized the neural network approach beneficially, either for predicting the primary productivity in reservoirs as a function of environmental variables (Karul et al., 1999), or for predicting algal blooms (Recknagel et al., 1997).

2.6.3 Macrophytes

Four groups of macrophytes can be distinguished on the basis of morphology and physiology (Wetzel, 2001), namely:

- emergent macrophytes which occur on saturated or submersed soils (upper littoral),
- floating-leaved macrophytes which are rooted in submersed sediments (middle littoral),
- submersed macrophytes which occur at all depths within the photic zone, and
- freely floating macrophytes which are not rooted and are usually restricted to protected areas.

As mentioned above, fluctuating water levels in many reservoirs prevent the development of macrophytes, particularly submerged and floating-leaved rooted species. Floating species are not affected to the same extent by drawdown, although wave action on an unprotected shoreline is a major factor limiting their distribution.

2.6.3.1 Effects of reservoir formation on macrophyte abundance and diversity

The formation of a reservoir strongly influences the conditions that determine the diversity and abundance of macrophytes (Agostinho, 1999). If there is an increase in spatial heterogeneity during flooding (for example a contained river may expand into a shallower riparian zone), macrophyte habitat suitability will be enhanced as exemplified by the Itaipu Reservoir in Brazil. On the other hand, if the inundated terrestrial habitat has high habitat diversity with wetlands, lagoons etc., simplification of the environment by impoundment can lead to reduction of diversity of habitat suitable for macrophyte growth (Thomaz and Bini, 1998a in Agostinho et al., 1999).

Other significant factors associated with reservoir formation that affect aquatic macrophyte abundance and diversity include

- reduction in water velocity which increases favourable conditions for attachment of macrophytes;

- increase in sedimentation which increases nutrient concentration in the water and sediments, but decreases light penetration, and may result in unconsolidated substrate that hampers rooting;
- alteration in hydrological regime, often with severely fluctuating water levels;
- an increase in nutrient availability (Thomaz and Bini, 1998b in Agostinho et al., 1999; Mazzeo et al., 1995).

Independently of immediate or intermediate effects, reservoir formation results in the creation of lentic environments that will inevitably undergo the ecological succession typical of lakes. With time, natural or artificial eutrophication and reduction of depth will lead to the development of various species of aquatic macrophytes (Thornton, 1990; Agostinho et al., 1999). Although considered a nuisance when their colonization interferes with multiple uses, macrophytes are important for the functioning and biodiversity conservation in reservoirs (Smart et al., 1996; Martinez et al., 2000; Thomaz et al., 2003). Submerged and floating water plants serve a number of important functions (Brendonck et al. 2003). A well-developed macrophyte community provides shelter from predation for vulnerable prey species (Diehl, 1992; Batzer, 1998). Macrophytes are usually covered with epiphytes (Kiss et al., 2003) that are grazed upon by invertebrates (van den Berg et al., 1997), many of which are important in the diet of fish and birds (Batzer and Wissinger, 1996). Generally, lakes and reservoirs with a well-developed macrophyte community have a more diverse community of zooplankton (Timms and Moss, 1984), zoobenthos (Munro, 1966) and fish (Olson et al., 1994; Maceina et al., 1992). Vegetated sites usually support a greater diversity of macroinvertebates than open water sites (Olson et al., 1994; Savage and Beaumont, 1997).

Other determinants of aquatic macrophyte diversity include water chemistry and trophic state (Murphy et al., 1990; Bornette et al., 2001; Lougheed et al., 2001), reservoir morphometry including depth, slope, shoreline development ratio (Duarte, 1986), degree of exposure to wind (Chambers, 1987; Hudon et al., 2000) and degree of connectivity to rivers (Amoros and Bornette, 1999). These characteristics enable the prediction, within certain limits, of the intensity of development of aquatic macrophytes after impoundment (Agostinho et al., 1999).

Several studies have dealt with the effect of impoundment on the mid- to long term dynamics of aquatic macrophyte communities in temperate regions (e.g. Krahulec and Kaplan, 1994; Rorslett and Johansen, 1996; Krolikowska, 1997), but there is a relative lack of information for tropical and subtropical regions (Agostinho et al., 1999). However massive development of free-floating macrophytes have been recorded in the early stages of filling of many tropical reservoirs such as Tucurui in the Amazon River Basin (Tundisi, 1994) and Kariba on the Zambezi River (Mitchell et al., 1990). Such extensive development of aquatic macrophytes depends on conditions such as absence of strong winds, low water turbulence and availability of propagules or other sources of dispersion occurring simultaneously with nutrient increases typical of filling phases (Thomaz and Bini, 1998a in Agostinho et al., 1999). These conditions favour free-floating species, especially *Eichhornia crassipes, Pistia stratiotes* and *Salvinia auriculata* whose populations can experience huge increases during and soon after filling, constituting potential threats to reservoir uses. The largest problems are usually associated with invasive species (Brendonck et al., 2003). (See Section 3.4).

2.6.3.2 Periphyton associated with macrophytes

The functional role of macrophytes as regulators of water quality (through nutrient sequestration, for instance) is enhanced by the periphyton that develops on the submerged parts of the macrophytes (Lalende and Downing, 1991; Lakatos et al. 1999; King et al., 2000; Roberts et al., 2003). In addition, the structure and composition of periphyton reflect habitat differences relating to ecological and water quality factors, while any qualitative or quantitative transformation indicates the ecological state of the environment as well as the changes occurring (Round, 1991; McCormick and Stevenson, 1998). In their study on littoral macrophyte-periphyton complexes in the Kis-Balaton Protection System and the Kiskore Reservoir in Hungary, Kiss et al. (2003) stated that this is a particularly important and required area of investigation in studies on water quality.

2.6.3.3 Macrophytes in reservoir cascades

In the case of reservoir cascades, position in the series may also be an indicator of potential for the development of submerged macrophytes (Agostinho et al., 1999). Suitable environments seem to be characteristic of reservoirs located at the downstream end of the cascade. These reservoirs receive water with reduced suspended loads and consequently light penetration into the water column is high. This observation is corroborated by the excessive development of *Egeria* in the Paulo Afonso Reservoir (Sao Francisco River) and the Jupia Reservoir (Upper Parana River), Brazil, both of which are situated at or near the downstream end of reservoir cascades.

Management of macrophytes in reservoirs is discussed in Section 3.1.2.

2.6.4 Zooplankton

2.6.4.1 Role in secondary production

The study of secondary production has an important place in biological research. Assessing the balance between input and output of matter and energy of populations is vital for an understanding of the ecosystem's dynamics. Basic knowledge of the life history and biomass of different zooplankton species provides the necessary data for the calculation of secondary production in aquatic environments, as well as information about the competitive strategies responsible for their success in a given environment (Melao and Rocha, 2000, 2004).

2.6.4.2 Zooplankton community structure and dynamics in reservoirs

Change in size-structure, species composition and biomass of zooplankton in relation to trophic state have been reported repeatedly in freshwater environments (Peters, 1986 in De Manuel and Jaume, 1993), but there is a scarcity of information about these relationships in reservoirs.

From the point of view of zooplankton, reservoirs are very different from lakes. Since zooplankton populations are not characteristically a component of rivers, the impoundments constructed in the last 50 years have established new lentic habitats, which have been colonized by zooplankton (Marzolf et al., 1990). However, reservoirs can be considered favourable environments for the development of zooplankton communities, which may establish diversified assemblages in relatively short periods of time after impoundment (Rocha et al. 1999).

The colonization process differs greatly among reservoirs because the initial conditions in recently filled water bodies are variable (Rocha et al., 1999; Schmid-Araya and Zuniga, 1992). Depending on morphometry, geology and vegetation of the flooded basin, the water chemistry can be significantly different, providing a “selective filter” for the organisms that potentially can establish populations (Rocha et al., 1999).

The typical zooplankton assemblage of reservoirs is commonly comprised of 10-20 species of protozoa, 20-60 species of Rotifera, 5-10 species of Copepoda and 10-20 species of Cladocera. In addition there are usually some species of Ostracoda, Gastrotricha, Diptera larvae and planktonic Turbellaria. However, within these guidelines, species diversity varies greatly, between individual reservoirs, locations within each reservoir, geographical regions, and also with time (Rocha et al., 1999).

2.6.4.3 Factors influencing zooplankton community structure in reservoirs

2.6.4.3.1 Turbidity

Introduction

The susceptibility of reservoirs to high sediment loading has been emphasized. The high level of sediment-related or mineral turbidity that results in reservoirs has significant impacts on zooplankton, and the resulting food-web structure of affected reservoirs. The scientific investigation of this topic was largely pioneered in limnological studies of various South African reservoirs, commencing with Lake le Roux (Vanderkloof Dam).

Suspended sediment impacts on small particle-feeding (‘herbivorous’) zooplankton are essentially three-fold. First, light-limitation of autotrophic photosynthetic production reduces their primary food resource base. This impact has been discussed/explored in Section 2.6.2, and is not further re-considered here. Second, astronomical densities (10^6-10^7 ml^{-1}) of non-nutritive particles differentially compromise the food-collecting abilities of different component taxa. And lastly, suspended particles provide a ‘visual screen’ for zooplankton that effectively serves as a refuge against visual zooplanktivores – fish. The present section concentrates on these two attributes.

Additional indirect impacts largely involve the lake heat budget. High light attenuation results in a shallower heated surface layer and somewhat warmer **surface** temperatures than expected for locality latitudes (Van Schalkwyk & Walmsley 1984). Conversely, this tends to translate into slightly cooler water column **average** temperatures. Despite unexpectedly rapid **surface** warming (Van Schalkwyk & Walmsley 1984), seasonal heating of the entire surface **mixed layer** can be slower, and lessened, accordingly delaying and shortening high summer conditions, even while stratification intensity may be stronger. These heat budget influences principally affect the seasonal dynamics of zooplankton, and are not explored in further detail here. Contextually relevant information is given in Section 2.2.

Impacts of turbidity on feeding biology as a determinant of zooplankton species composition/community structure.

Total zooplankton abundance is inversely related to sediment turbidity (Hart 1986a), in line with associated reductions of primary food resources. But in addition, striking reductions in the proportional contribution of daphnids to total zooplankton biomass during years of extremely high turbidity (SD < 30cm) observed in L. le Roux (Hart 1986a) clearly signaled the disproportionately severe impact of suspended sediments on daphniid cladocerans (Figure 14), and the experimental elucidation of underlying reasons (Hart 1988, 1992). In short, the contrast reflects fundamental differences between food collecting mechanisms of cladocerans like *Daphnia*, and copepods. *Daphnia* is an automatic filter-feeder, relying on continuous filtration of its ambient fluid surroundings. Inter-setal and inter-setular spacing on the filtering thoracopods varies between species, setting the effectively aperture of the filter-combs. Very fine filters retain a large proportion of the fine suspended sediment particles. Once filtered, the bolus of collected particles is moved to the mouth and essentially 'tasted' for the first time, invoking a general rejection of the bolus. Filtration is recommenced, and the cycle of collection-rejection repeated. The energy and nutrient deficiency created by this 'empty' feeding reduces the species fitness, leading to *Daphnia*'s displacement from the community assemblage. However, some species of *Daphnia* are more tolerant of elevated turbidity than others (Hart 1992), presumably depending on differences in filter-mesh size and/or sensitivity (or responsiveness) to the 'taste' cues imparted by the food bolus. Species that commonly inhabit turbid waters include *D. gibba* and *D. barbata*, whereas *D. pulex* and *D. longispina* appear to be essentially 'clean-water' taxa.

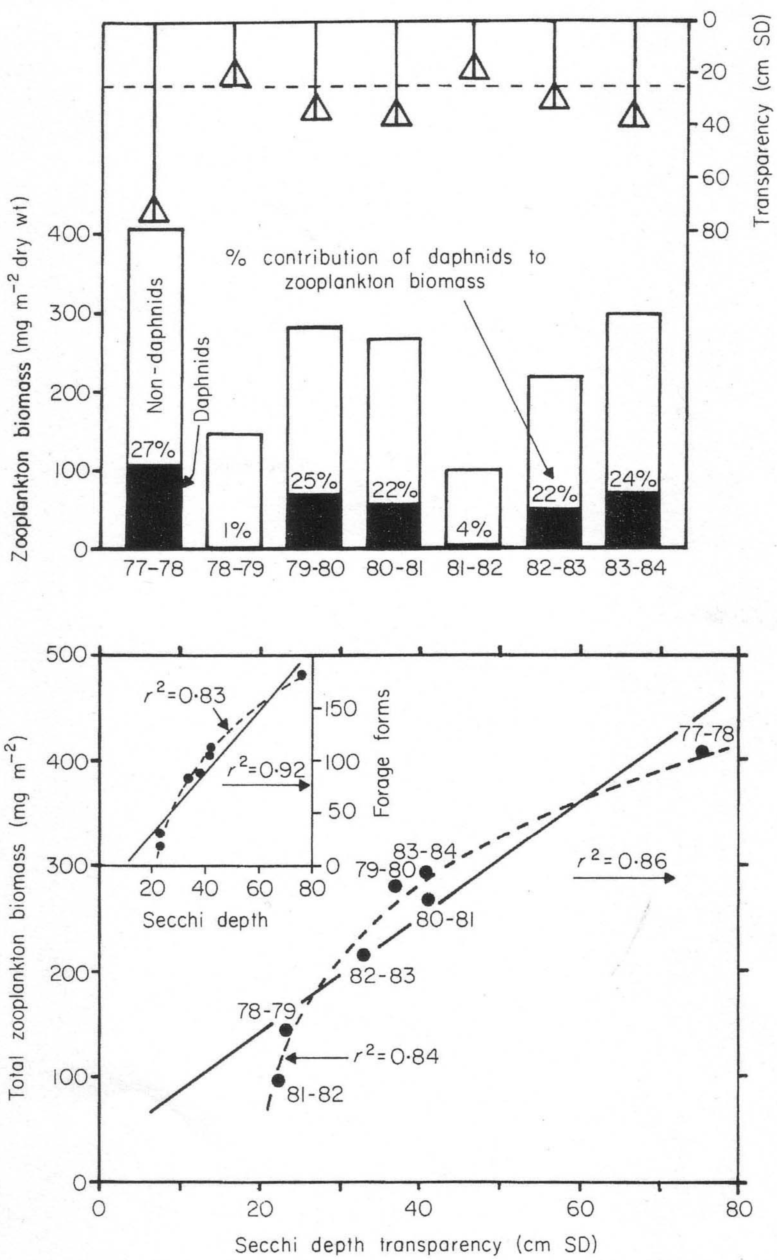


Figure 14. Weighted annual mean zooplankton standing stock levels in relation to corresponding Secchi depth transparency values in Lake le Roux between 1977 and 1984. Forage zooplankton include all daphnids and large predatory copepod *Lovenula excellens* which are important dietary items for planktivorous yellowfish *Barbus aeneus.* Note the severe depression of daphnids in most turbid years (upper panel). (After Hart, 1986b).

Copepods, on the other hand, are essentially 'raptorial' feeders, that identify and capture individual food particles. 'Herbivorous' calanoids and 'herbivorous' instars of cyclopoids selectively identify and collect uni-cellular and other algae as food; predatory calanoids like *Lovenula*, and the carnivorous instars of many cyclopoid taxa select and capture appropriate live animal prey. The high particle selectivity exemplified by copepods accordingly facilitates their occupancy of turbid waters, within which the plethora of nutritionally poor or valueless sediment particles translates into ‘white noise’ rather than the serious ‘contamination’ experienced by daphnids.

In contrast to the depressive effect of sediment particles on *Daphnia*’s feeding ability described above, small-bodied cladoceran taxa such as *Moina* is a common and sometimes abundant occupant of turbid waters (Hart 1986a, 1987b, 1999, 2001), implying some fundamental difference in its feeding biology and ecology. Paradoxically, even *Diaphanosoma* – that Geller & Müller (1981) functionally classified as a ‘high-efficiency bacteria feeder’ in view of its extremely fine-meshed filter – occurs in moderately (Hart 1996, 2001, 2004) as well as very turbid reservoirs (Hart 1999), suggesting potential duality or plasticity in its feeding mode that merits scrutiny.

Impacts of turbidity on visual predation, and zooplankton community composition.

In general, large-bodied zooplankters are selectively predated by visual zooplanktivores, commonly leading to their local extirpation and exclusion (Brooks & Dodson 1965), with differences in escape probability rendering cladocerans generally more vulnerable than copepods to this exclusion (Drenner et al., 1978). Co-existence of very large-bodied zooplankton and (facultative) visual zooplanktivores in turbid reservoirs is accordingly unanticipated. In L. le Roux, *Daphnia gibba* (> 3.5 mm) and *Lovenula excellens*, an equally large predatory copepod persist despite visual zooplanktivory by *Barbus aeneus* (Allanson & Jackson 1984, Hart 1986a).

2.6.4.3.2 Food quality and quantity/Trophic status

Eutrophication causes great changes in the structure of zooplankton communities (Rocha et al., 2002). Generally an increase in eutrophication causes increased zooplankton numbers and biomass (Rocha et al., 1999; Ostojic, 2000), unless changes in phytoplankton composition lead to the dominance of unfavourable forms (Rocha et al., 1999). Elenbaas and Grundel (1994), comparing a more eutrophic with a less eutrophic impoundment in Zimbabwe found the relative abundance of main zooplankton groups to be similar, but the density in the more eutrophic reservoir was nearly seven times higher during the spring.

However, many species disappear as a consequence of algal toxins or the clogging of filter-feeding apparatus during algal blooms, especially blooms of cyanobacteria which have both effects (Infante, 1982; Matsumura-Tundisi et al., 1986, in Rocha et al., 2002). Some species such as *Brachionus calyciflorus* and *Bosmina longirostris* have the ability to utilize colonial cyanobacteria as food, so become abundant under such conditions (Fulton and Paerl, 1987).

In non-eutrophic waters in temperate regions, cladoceran species usually dominate. However, in tropical regions, Rotifera have been observed to be dominant irrespective of the level of eutrophication, thus suggesting that other factors (particularly competition and predation) may be more important (Rocha et al., 2002). Rotifera total biomass is usually low, but this is compensated for by a short generation time, thus a fast renewal of populations (Hutchinson 1967).

Among Copepoda, the Cyclopoida are considered a more successful group than Calanoida in eutrophic systems (Tundisi et al., 1988, in Rocha et al., 1999), by virtue of their ability to capture large particles such as colonial and filamentous algae which usually become dominant in eutrophic systems (Magrin and Matsumura-Tundisi, 1997). Even the colonial cyanobacteria might be consumed by them, either directly or after the decaying of blooms, as detritus enriched by bacteria (Rocha et al., 1999). Otherwise, the shift to a microbial food web provides an abundant supply of micro zooplankton which serves as food for predatory cyclopoid species or life stages.

2.6.4.3.3 Meteorological factors (precipitation and wind)

Long-term studies on zooplankton composition have indicated that in tropical regions, precipitation and wind are important physical factors affecting zooplankton community structure (Matsumura-Tundisi and Tundisi, 1976; Noguiera and Matsumura-Tundisi, 1996, in Rocha et al., 2002). In general, the effect of precipitation is predominant during the rainy season (summer) and the wind during the dry winter. This situation has been reported for a number of Brazilian reservoirs such as Barra Bonita (Fonseca, 1990 in Fonseca, 1997), Broa Reservoir (Oishi 1990 in Fonseca, 1997), and Bariri Reservoir (Sandes 1990 in Fonseca, 1997). The rainfall results in allochthonous fluxes into the reservoirs, increasing suspended matter and lowering water transparency

2.6.4.3.4 Reservoir ageing

The age of a reservoir is an important factor in zooplankton community development, older reservoirs tending to display higher species richness unless this is prevented by pollution and contamination. However, many factors besides reservoir age influence the diversity of the system, making it difficult to establish a direct relationship between reservoir age and community diversity (Rocha et al., 1999). Some important studies on the development of zooplankton communities in the Amazonian reservoirs have been published (Magrin and Matsumura-Tundisi, 1997; Falotico (1993) and Moreno (1996) in Rocha et al., 1999). Conditions in the early stages promote the development of R-strategist pioneers, such as species of Protozoa and Rotifera. Later the Cladocera and the K-strategists such as Copepoda appear (Rocha et al., 1999).

2.6.4.3.5 Reservoir operations e.g. residence time, pulses

There is a need to understand how reservoir ecosystems couple with hydrological processes. This is often ignored in lake studies because of their more stable hydrological regimes, but becomes very important in environments subject to pronounced and short-term changes in water level (Straškraba et al., 1993). This is discussed in Section 3.2.

Bledzki and Ellison (2000) found zooplankton abundance and community structure to be significantly dependent on water retention time (and its converse, flow rate). Results obtained from Wloclawek Reservoir in Poland and Upper Lake in Massachusetts, USA, suggested that a retention time of 6 days represented a threshold. At this retention time, Cladocera comprised the highest percent of total zooplankton biomass, while its relative importance declined at both higher and lower retention times. These authors hypothesized that because cladocerans remove phosphorus from water through herbivorous grazing activity and transformation of smaller particles into larger ones (faster sedimentation) (Ejsmont-Karabin et al. 1993), the 6 day retention time could result in maximal depletion of phosphorus. Under such conditions, zooplankton community composition could ameliorate declines in water quality.

Naselli-Flores and Barone (1997) recorded a dominance of cladocerans in Lake Arancio, a hypertrophic reservoir in Sicily, associated with strong fluctuations in water level. The dominance of Cladocera resulted from the influence of water level fluctuations on populations of phytoplankton (Padisak et al. 1993; Naselli-Flores and Barone, 1996) and fish (Soranno et al., 1993; Kubecka, 1993), thus affecting bottom-up and top-down forces acting on the zooplankton community. Naselli-Flores and Barone (1994) also recorded a dominance of large cladocerans in reservoirs in Sicily that were subjected to extreme fluctuations in water level.

High retention time was found to favour rotifers in Spanish reservoirs (Guiset and De Manuel, 1993), but diversity (of rotifers) was relatively higher in reservoirs with low retention time (De Manuel and Armengol, 1993).

The zooplankton communities of two Chilean reservoirs, Penuelas and Rungue were investigated by Schmid-Araya and Zuniga (1992). Rungue Reservoir undergoes extreme water level fluctuations on a regular basis with the water of the reservoir being renewed annually. At the lowest water level during their study, the number of species and their abundance decreased drastically, even to the point of disappearing completely. The observed community was therefore in an early successional stage with only r-strategists present. In contrast, Penuelas, which undergoes moderate fluctuations, has a zooplankton community that follows the seasonal changes during the year.

Reservoirs can be viewed as “pulsed systems” in which abrupt changes (pulses) can be of either natural or artificial origin (Straškraba et al., 1993). They affect physical, chemical and biological conditions in the reservoir and can be induced by natural causes such as precipitation or by man during reservoir operation (Barbiero et al., 1997). Communities are sensitive to the operational regime. Great changes in species composition, density and production can occur as a consequence of variations in water outflow. Braga et al., (1998) in Rocha et al. (1999), recorded decreases in algal and zooplankton populations in Barra Bonita Reservoir in Brazil after strong water outflow. Fonseca (1996) in Rocha et al., (1999), showed that the loss was selective in Balbina Reservoir, inducing changes in the zooplankton community structure. Small organisms such as Protozoa, Rotifera and Copepod nauplii were more vulnerable than Cladocera and Copepoda, which occupied the deeper waters. Thus the effect of outflow depends on a variety of factors such as the depth of the outlet, and the vertical distribution of the plankton. Such removal of organisms may prevent the occurrence of competitive exclusion, allowing congeneric associations of organisms as indicated for the genera *Notodiaptomus* and *Mesocyclops* in Barra Bonita (Espindola (1994) and Rietzler (1995) in Rocha et al., 1999).

The littoral zone tends to have a higher biodiversity than the limnetic zone, thus contributing to the total diversity of the system (Okano (1995) in Rocha et al., 1999). Macrophytes growing in the littoral zone affect zooplankton by providing increased food availability and refuges from predation. Rocha et al. (1982) in Rocha et al., 1999, observed that *Argyrodiaptomus furcatus*, a calanoid copepod occurred in a vegetated zone of the littoral zone at densities ten times higher than areas without macrophytes. Changes in water level as a result of the operational regime may prevent or inhibit the establishment of macrophytes in the littoral, or change the species composition of such plants, thereby indirectly influencing the zooplankton assemblages (Rocha et al., 1999).

2.6.4.3.6 Morphometry

The morphometry of a reservoir is possibly the most marked distinction to be made among reservoirs. They vary enormously in area, volume, depth and shape, each factor influencing the community structure of zooplankton, as spatial heterogeneity increases and the possibility of colonization by a larger number of species occurs. Many reservoirs are extremely dendritic, the complexity of the numerous drained valleys increasing the length of the shoreline and the extent of the littoral zone, and introducing potential chemical novelty and distinctiveness. These peripheral zones often differ greatly from the open water in the central body of the reservoir (Rocha et al., 1999).

2.6.4.3.7 Stoichiometry

In recent years, several lines of evidence have indicated that the stoichiometry of carbon, nitrogen and phosphorus may be a major factor controlling zooplankton communities (Sterner and Hessen, 1994; Hessen, 1997). It is known that lacustrine systems demonstrate a wide range in their C: N and C: P ratios (Hecky et al., 1993) and that they differ from marine systems (Elser and Hassett, 1994). It is also known that zooplankton demonstrate an interspecific variation in their C: N and C: P ratios (Andersen and Hessen, 1991), and thus they presumably also differ in stoichiometric requirements for these elements. Individual zooplankton species have been described as “homeostatic” meaning that their C: N: P ratios remain relatively constant in spite of the wide variation of food they consume (Hessen and Lyche, 1991).

There is evidence in the literature that stoichiometry does in fact have an effect on zooplankton communities (Sterner et al., 2000). Data from Europe (Gulati et al., 1991; Hessen, 1992) and North America (Hassett et al., 1997) indicate that the high P herbivore, *Daphnia* is generally not favoured at high C:P ratios resulting in growth penalties when the seston food base has a high C:P ratio. Sterner et al. (2000) pose the question: is there a consistent pattern of species replacements in zooplankton arranged along a seston C: P ratio gradient. If so, great inroads to another predictive axis for zooplankton communities would have been found.

2.6.4.3.8 The reservoir food web

See Section 2.6.7 for a contextually relevant consideration.

2.6.4.4 Zooplankton migration

2.6.4.4.1 Vertical migration

Diel vertical migration (DMV) of zooplankton has been described by many authors (e.g. Wojtal et al., 2003; Johnsen and Jacobsen, 1987; Lampert, 1993; Horppila, 1997). The results of long-term studies on diurnal changes in zooplankton distribution indicate that several factors are responsible for this phenomenon (Wojtal et al., 2003). According to the generally accepted hypothesis, migrations of zooplankton are the consequence of a foraging rate-predation risk trade-off (Desmarius and Tessier, 1999). By means of DVM zooplankton avoid visually orienting predators like fish (Stich and Lampert, 1981; Brancelj and Blejec, 1994) and invertebrate predators (Lampert, 1993). Migrations can also be the effect of vertical distribution of nutrients in the water column (Kitchell et al., 1979) and/or the location of available food (Pijanowska and Dawidowicz, 1987).

2.6.4.4.2 Horizontal migration

DVM occurs mostly in deep stratified lakes and reservoirs that can provide refuges for herbivores (Wright and Shapiro, 1990). In shallower waters, lack of deep water refuges result in higher planktivory pressure and lower biomass of zooplankton (Desmarius and Tessier, 1999). Many authors have suggested that in shallow lakes with sufficient development of macrophytes in the littoral zone, these plants provide *Daphnia* sp. with spatial refuges from fish during the day (Timms and Moss, 1984; Davies, 1985; Vuille, 1991; Lauridsen and Buenk, 1996; Lauridsen et al., 1997). Consequently, submerged macrophytes may indirectly improve water transparency, as increased zooplankton survival will enhance their grazing pressure on phytoplankton both within the vegetation during the day and in the pelagic zone at night (Jeppesen, 1998, in Wojtal et al., 2003). This is the basis for the much vaunted management prospect of bio- or food web manipulation (e.g. Moss, 1998).

In the Sulejow Reservoir in Poland, which has a sparsely vegetated littoral zone due to high water level fluctuations, large-bodied zooplankton were shown to migrate towards the open water at dusk, and towards submerged macrophytes at dawn, answering the question posed by Wojtal et al. (2003) whether horizontal migration of zooplankton, described for lakes, also occurs in reservoirs with a sparsely vegetated littoral zone.

2.6.4.5 Longitudinal distribution of zooplankton in reservoirs

The longitudinal distribution of zooplankton in reservoirs is influenced by many of the same abiotic and biotic factors as those discussed above. When these factors change along the longitudinal gradient, so do the zooplankton community structure and dynamics. The longitudinal gradients of a variety of parameters in reservoirs have been discussed in Section 2.3.

Various factors have been cited as being responsible for the horizontal heterogeneity of zooplankton in various types of water bodies, including, the effects of advective physical forces resulting from river inflows (Patalas and Salki, 1990; Ford, 1990), turbidity gradients (Zettler and Carter, 1986; Hart, 1990), temperature regime (Johannson et al., 1991); food availability and trophic conditions (Peters, 1983; Carter et al., 1995) or predation and competition (Jakobsen and Johnsen, 1987; Urabe, 1990; Gliwicz and Rykowska, 1992; Pont and Amrani, 1990). On the other hand there are at least two known mechanisms for reducing zooplankton patchiness and randomizing zooplankton distribution (Seda and Devetter, 2000). These are wind induced surface turbulence and advective processes in epilimnetic layers (Pinel-Alloul et al., 1988; Patalas, 1990; Pinel-Alloul and Pont, 1991).

Marzolf (1990) presented a theoretical model that described the abundance distribution of zooplankton along the longitudinal axis of reservoirs, according to which it is determined by two main factors. These are the velocity of the current and the exportation of material (clay, nutrients, DOC and micro flora). (Figure 14) The model postulates that if the velocity of the current is the factor that exerts the greatest influence on the distribution of zooplankton, there is an increase in zooplankton density towards the dam. If the exportation of material exerts the greatest influence, the zooplankton density is greatest towards the riverine zone. If both factors are acting, the density along the longitudinal axis of the reservoir (in the direction river-dam) resembles a frequency distribution with positive asymmetry (Figure 15).

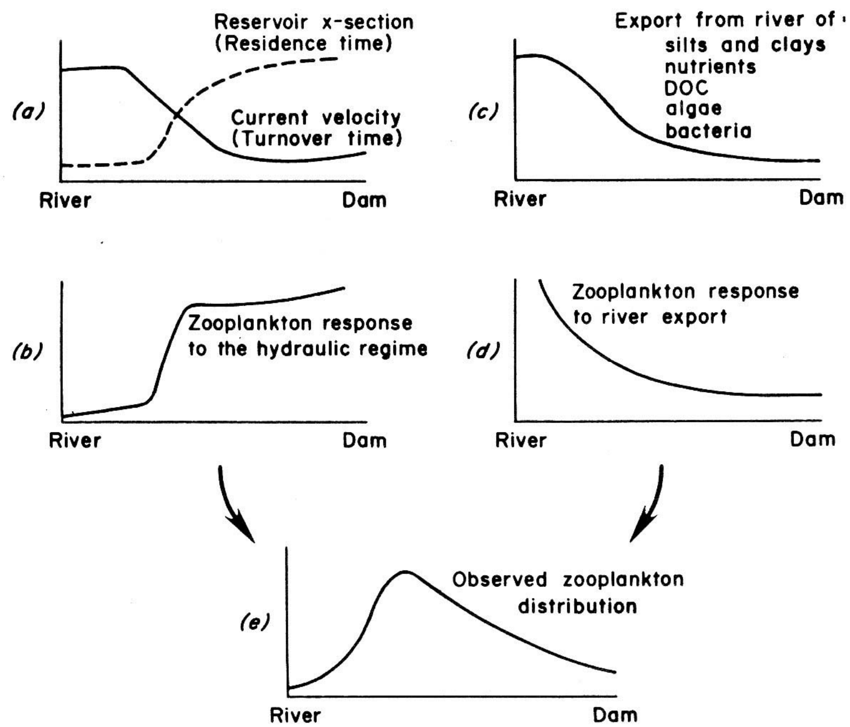


Figure 15. An illustration of the several trends that develop along the longitudinal axis of an impoundment, which when taken together result in the observed distribution of zooplankton shown in the bottom panel (e). (From Marzolf, 1990).

Confirmation of some features of Marzolf's model is available. Seda and Machacek (1998) found that the spring flood in Rimov Reservoir in the Czech Republic removed much of the zooplankton from the upstream region but did not affect the mean zooplankton densities for the reservoir as a whole. The recovery of the upstream zooplankton took just one week, and was mediated by the animals that had been displaced downstream, as was proven by allozyme analysis. This is in agreement with Seda (1994) who showed that there is no evidence for a unidirectional flow of zooplankton from the upstream to the downstream end of Rimov Reservoir

Pinel-Alloul (1995), in a comprehensive overview of zooplankton spatial heterogeneity, concluded that at large spatial scales (>1km) the abiotic factors are more important for the maintenance of spatial heterogeneity, while at smaller scales (1-10m) biological processes are more important. However she maintained that both are linked interactively and the resulting heterogeneity can be perceived as a functional interaction between the organisms and their environment

Bini et al. (1997), investigating spatial gradients in Broa Reservoir in Brazil found that the copepods (all age classes) increased towards the riverine zone, while Cladocera increased towards the lacustrine zone, although limnological variables such as temperature, dissolved oxygen, pH and suspended material varied very little. In the eutrophic Salto Grande Reservoir in Brazil, Zanata and Espindola (2002) found that rotifers were more abundant in the riverine zone, decreasing towards the dam, whereas in this case, copepods were more abundant in the lacustrine zone, decreasing towards the river inflow. Cladocera did not display a significant density gradient although they were slightly more abundant in the lacustrine zone. In the Kenyir Reservoir in Malaysia, cladocerans formed the major group in the lacustrine and transitional zones, whereas copepods were the main group in the riverine zone. Copepod nauplii and copepodids contributed >50% of the total copepods at all stations, probably indicating that the adults were kept at low levels by predation (Yusoff et al., 2002).

2.6.5 Zoobenthos in reservoirs

2.6.5.1 Introduction

The zoobenthos is the animal community living in association with any substrate-water interface. It is divided on the basis of size (Kalff, 2002) into:

- the mega benthos, e.g. large molluscs (>1000µm) (Etim and Sankare, 1998; Kharchenko and Zorina-Sakharova, 2002; Jurkiewicz-Karnkowska, 2005; Yu and Culver, 1999; 2000; Kiibus and Kautsky, 1996; Blay, 1990),
- the relatively well studied macro benthos e.g. most insect larvae, small molluscs (<1000-400µm) (Koskenniemi, 1994; Tavcar, 1993; Petridis and Sinis, 1993; Grzybkowska and Dukowska, 2002; Real et al., 2000; Armitage et al., 1995; Hamburger et al., 1995; Heinis and Crommentuijn, 1992; Koskenniemi, 1992; Lindegaard, 1995; Pelegri and Blackburn, 1996; Svensson and Leonardson, 1996; Szito and Botos, 1993)
- the meiobenthos e.g. rotifers, copepods, young chironomids, small oligochaetes and nematodes (400-100µm) (Mashina, 2003; 890) and
- the relatively little studied micro benthos e.g. protozoans and juveniles of larger forms (<100µm) (Kalff, 2002). The zoobenthos is numerically dominated by small species and is species rich compared to the invertebrates of the open water (zooplankton) (Kalff, 2002).

2.6.5.2 Distribution

2.6.5.2.1 Littoral zone

The littoral zone, when not adversely affected by reservoir drawdown, is often dominated by macrophytes, and is widely recognized as the most productive region of lakes and reservoirs. It generally has the largest number of animal species, contains the highest animal biomass and density, and has the highest secondary production (Brinkhurst, 1974 in Kalff, 2002). For convenience, three subdivisions of this zone can be recognized:

- the upper littoral reaching from the shoreline to where emergent macrophytes (if present) disappear (approx. 1-1.5m),
- the middle littoral extending from this depth to where rooted submerged macrophytes disappear, and
- the sublittoral zone from the lower level of the former, to the lower level of the euphotic zone (Wetzel, 2001; Kalff. 2002).

The largest numbers of macrobenthic species occupy the structurally diverse and often macrophyte-dominated middle littoral zone, where they live on the macrophytes or in the organic sediments below. However, fluctuating water levels in reservoirs can severely alter the habitat and survival of benthic invertebrates as macrophytes; submerged species in particular are often unable to survive marked fluctuations of water level resulting in almost complete elimination of benthic invertebrate habitats in some reservoirs (Wetzel, 2001).

2.6.5.2.2 Profundal zone

The zoobenthos of the profundal zone inhabits the zone of sediment accumulation, where the sediments are generally fine silts and clays. The sediment-water content is typically high (75-90% water) allowing for easy burrowing (Kalff, 2002). However, increased productivity and organic loading to reservoirs often induces hypolimnetic anoxia and related toxic products of fermentative metabolism (Wetzel, 2001; Kalff, 2002; Prat et al., 1992), leading to the disappearance of many benthic invertebrate taxa and the reduction in biomass and production of invertebrates commonly found in reservoirs (Siegfried, 1984 in Wetzel, 2001; Popp and Hoagland, 1995).

2.6.5.3 Zoobenthos as indicators of water quality

Benthic macroinvertebrates are found in large numbers in rich organic sediment of eutrophic reservoirs. Composition of the benthic fauna changes in response to physical and chemical changes in the catchment and in the water, both during reservoir ageing, and in subsequent changes in the trophic state of a water body (Real et al, 2000). Jonasson (1996) showed that hypolimnion oxygen content, food quantity and quality and water temperature are the main factors influencing the presence and biomass of benthic species in lakes and reservoirs. Zoobenthic composition is therefore a good indicator of trophic status (Popp and Hoagland, 1995; Lyashenko and Protasov, 2003; Bren, 2001; Ogbeibu and Oribhabor, 2002, as exemplified in the following two cases.

> Example 1
> Chironomidae (Diptera) present a wide range of feeding behaviour and lifestyles, and collectively inhabit all freshwater habitats (Grzybkowska and Dukowska, 2002). As a result of the relationship between specific species and specific environmental factors, different species of the genus *Chironomus* have been used for many years as indicators of trophic conditions of lakes and reservoirs (Lindegaard, 1995, in Real et al., 2000; Popp and Hoagland, 1995).
>
> Example 2
> Because they filter a great volume of water containing soluble and particulate substances, bivalve molluscs are effective accumulators of metals and organic micropollutants. After being metabolized, these pollutants are selectively concentrated in the soft tissue or the shell. In addition, bivalve molluscs are resistant to several pollutants (e.g. organochlorines), which are dangerous for other animals such as crustaceans and insects. They are sedentary at the adult stage, often widespread and abundant and sometimes long-lived. These factors make them useful in evaluating the levels and distribution of many pollutants (Ravera, 2001). For example, *Dreissena polymorpha* is commonly used to monitor freshwater ecosystems (Doherty et al., 1993; Sures et al., 1999).

2.6.5.4 Ecosystem impact

The ecological role of mussels in tropical Lake Kariba was investigated by Kiibus and Kautsky (1996), who found that they had a significant effect on the nutrient dynamics. A volume of water corresponding to the total epilimnion of the lake was found to be filtered by the mussels annually. It was also estimated that they remineralized 25% of the total phosphate load, and 8 times the total nitrogen load in a year. Approximately 5% of the annual phytoplankton production was channelled through the mussels, which tend to be limited to a narrow inshore zone of gently shelving water of 3-9 m depth in Kariba (Kenmuir, 1980), with a similarly restricted depth distribution in Lake McIlwaine (Kenmuir, 1980; Marshall, 1982). In Lake Kariba 95.8% of the benthic biomass consists of mussels, 4.1% of snails, and only 0.1% of insect larvae (Machena and Kautsky, 1988 in Kiibus and Kautsky, 1996).

2.6.6 Fish

2.6.6.1 The development of fish communities in reservoirs

The filling of reservoirs can be considered a catastrophic event for the impounded riverine fish community; however some riverine fishes show various adaptive abilities in newly formed lentic environments (Kubecka, 1993). The composition of the fish fauna of a dammed river is thus the first of the factors influencing the final fish composition.

Abiotic conditions in reservoirs such as temperature regime, retention time, pH, oxygen conditions, presence of toxins, water level fluctuations and turbidity, are fundamentally important to ichthyofauna, and may be responsible for the presence or absence of particular species in particular reservoirs. When a species in unable to survive in an impounded water body it is often for lack of

- suitable water temperature and/or dissolved oxygen;
- habitat diversity;
- spawning sites;
- sufficient prey for a particular stage in the life cycle;
- refuge from predators (O'Brien, 1990);
- seasonally appropriate spawning cues

However, reservoirs are aquatic ecosystems created by humans and most riverine fish species tend to disappear with time (Quiros and Boveri, 1999; Encina and Rodriguez-Ruiz, 2002; Paller and Gladden, 1991), while some do adapt to the new environment (Kubecka, 1993). The change from high flow to low flow tends to select riverine fish species previously adapted to floodplain habitats. The reduction in the number of species is usually high (Arcifa and Northcote, 1997; Araujo and Santos, 2001). The impoundment of most Brazilian rivers resulted in several modifications of their original fish fauna including a decrease in the number and diversity of species, as well as shifts in their dominance and trophic relationships (Arcifa and Northcote, 1997). Most reservoirs in Brazil have less than 75 species whereas most influent rivers have more than 125 species (Araujo-Lima et al., 1995).

After the damming of rivers, reservoirs are usually stocked with fish (Quiros and Boveri, 1999). Fish introductions and stocking, though usually justified on socio-economic grounds, are only occasionally evaluated in their outputs, and are rarely supported by sound ecological science (Quiros, 1999). Fish assemblages in most reservoirs include both native and introduced fish in a non-naturally occurring habitat that is subjected to manipulation by man. Such artificial assemblages have occurred only briefly in evolutionary time relative to those in naturally occurring habitats (Araujo and Santos, 2001).

2.6.6.2 Factors affecting sustainable fish growth and reproduction in reservoirs

2.6.6.2.1 Spawning

For any nest building fish, or fish which attach their eggs to a particular substrate, the nature of the substrate is important for successful spawning (Clark et al., 1998). Vegetation may also be important, but soil type, wave action, unconsolidated shorelines and high levels of turbidity often preclude the development of substantial littoral zone vegetation and are thus not suitable for fish dependent on vegetation for successful spawning (O'Brien, 1990) or as food sources and/or refuges for juveniles. Many reservoirs are characterized by moderate to heavy turbidity and siltation (Quiros and Boveri, 1999). Silt deposition on eggs increases egg mortality and thus decreases spawning success (O'Brien, 1990).

A common attribute of reservoirs is the water level fluctuation (Encina and Rodriguez-Ruiz, 2002; Clark et al., 1998). This may have both positive and negative effects on spawning. Because most gravel beds occur in the upper few meters of the littoral zone where wave action keeps gravel beds clear of silt, drawdown below this level may leave muddy areas which are poor for spawning. Drawdown also minimizes the amount of vegetation available for spawning, especially in turbid reservoirs where light sufficient for macrophyte growth may not penetrate very far into the water. Increase in water level may have the opposite effect. Spawning success has been found to increase either as reservoirs fill and flood vegetation, or in years of high water. High water sometimes floods terrestrial vegetation and could inundate gravel areas on the reservoir shore, providing a diversity of substrates needed for spawning. (O'Brien, 1990). This phenomenon was observed by Zalewski et al. (1990) in the Sulejow Reservoir in Poland when high water levels were recorded and large areas of terrestrial vegetation were flooded. The increased diversity of "littoral" habitats enabled the coexistence of large densities of perch (the dominant species) and cyprinids. The highly significant correlation between water level, its fluctuations, and the reproductive success of dominant species might be used as a predictive or biomanipulating tool for steering the reproductive success of fish in reservoirs, but data from other reservoirs are needed to allow verification (Zalewski et al., 1990).

2.6.6.2.2 Larval fish and fingerlings

Survival of larval fish is dependent primarily on acquiring food and avoiding predation. Almost all larval fish feed on zooplankton, the very youngest being gape-limited and able to feed only on very small zooplankton such as small cladocerans and rotifers. However, this stage of gape limitation is brief (a few weeks). Survival of larval fish is probably influenced more by predation than by feeding, these very small fish being vulnerable to virtually every other predator in lakes and reservoirs. Not only visual-feeding fish, but also predaceous copepods may have considerable influence on larval fish densities. Protective cover such as aquatic macrophytes is especially critical in minimizing fish and avian predation on small fish. Any factors that reduce vegetation cover, such as turbidity or wave action could therefore also minimize larval fish survival. Drawdown of water levels below the vegetation zone would be especially detrimental to larval fish (O'Brien, 1990).

2.6.6.3 Distribution of fish in reservoirs

An important area of study in fisheries and fish ecology is the estimation of the number of fish in a population (Encina and Rodriguez-Ruiz, 2002). The patterns and degree of patchiness in time and space are therefore of fundamental importance because of their relationship to abundance estimates (Levin, 1992; Margalef, 1993; Schael et al., 1995), and their importance for the understanding of fish biology and therefore its management.

In a newly flooded reservoir most of the fish are found close to the shore and at the riverine end of the reservoir. The pelagic zone and deep waters are little used, probably due to low efficiency in exploiting food resources (Granado-Lorencio, 1992). Encina and Rodriguez-Ruiz (2002) studied the seasonal distribution of fish in the newly filled Zahara Reservoir in Spain, where the native species were unable to exploit the planktonic resources. Because detritus formed part of their natural diet in the lotic environment, the reservoir shoreline and the river above the reservoir were suitable feeding zones (Encina and Granado-Lorencio, 1994; Magalhaes, 1992; 1993). The spatial distribution could also be related in some species to seasonal reproductive migration up the river (Granado-Lorencio, 1991). Other species can modify their reproductive tactics and associate them to the littoral zones of a reservoir (Fisher and Zale, 1991). Spanish reservoirs generally have little stability as they are built for power generation, irrigation and water supply and are subject to large water level fluctuations, hypolimnetic spillage and other disturbances. Consequently, only those species with generalist and detritivorous feeding habits and reproductive migrations out of the reservoir are capable of existing in the reservoir. (Encina and Rodriguez-Ruiz, 2002).

The spatial distribution of fish assemblages was investigated in Lajes Reservoir in Brazil (Araujo and Santos, 2001) to detect patterns of available habitat use by fish. Lajes is unusual compared to most Brazilian reservoirs in that there is no marked longitudinal gradient due to small tributary contributions, forming a more restricted habitat with no available lotic environment to allow for fish migrations into and out of the lake. Three zones were distinguished – an upper zone near the tributary inflows, and middle and lower zones progressively nearer the dam. Fish abundance, number of species and biomass were higher in the upper zone (probably due to higher food availability and habitat diversity) but only two species were more abundant there, all other species showing no difference in their abundance among the zones. Seasonal environmental variables of temperature, pH, transparency and water level did not show an association with fish occurrence. Most fish used the different zones of the reservoir with no sign of spatial separation.

The pelagic zone of reservoirs has been considered by some authors to be a vacant habitat for fish, and not inhabited by riverine species (Gaedke et al., 1998). Although pelagic fish are scarce in some reservoirs (Straile and Geller, 1998), in some others fish may be quite abundant (Adrian et al., 1999). In Lake Kariba, tigerfish have adapted to inshore pelagic zones, while in Lake Hume, a warm monomictic lake in Australia (Matveev et al., 2002), the presence of significant numbers of small planktivorous fish, mostly juvenile introduced European perch and Australian smelt were determined concordantly in hydro acoustic surveys and direct catches from pelagic water.

The inflow zone situated at the boundary between the river and the reservoir is often an important migration path. All the main types of fish migration as defined by Lucas and Baras (2001) (feeding, refuge-seeking, spawning migrations, post-disturbance movements, recolonization and exploratory migration can be expected to happen in this ecotone (Hladik and Kubecka, 2003). Upstream and downstream migration of fish between the Rimov Reservoir and its only tributary, the Malse, in the Czech Republic was studied by Hladik and Kubecka (2003). Due to many of the fish species being of riverine origin, they require access to the river habitat during crucial periods. In addition, the river fish may use the reservoir as a refuge or feeding ground. This study was carried out to provide a detailed understanding of fish migration events in the river/reservoir ecotone and to estimate the importance of the tributary for the functioning and management of the fish stock. It was found that the reservoir tributary zone was an extremely important area for fish, a significant proportion of the total fish stock migrating through it. Some species (asp, bleak, chub and white bream), seemed to be strictly dependent on the tributary zone (obligate tributary spawners) as they were never observed reproducing in the reservoir, while others (roach, bream, pike, perch, ruffe). were facultative tributary users or generalists, spawning in suitable places both in the reservoir and in the tributary (roach, bream, pike, perch, ruffe). A third group (carp, pikeperch, catfish and eel) spawned out of the tributary (i.e. in the reservoir). Six periods of migration succession were identified during spring and summer, which differed in species dominance, gonadal status and migration rates.

Allanson and Jackson (1983), in Paxton (2004) investigated the effects of flow reduction resulting from the construction of the Vanderkloof and Gariep dams on the Orange River (South Africa) on reproduction, growth, dispersal and mortality of freshwater fish in the river between the two dams. Regulation by the Gariep Dam was found to have significant biological consequences for the populations of two species of yellowfish (*Barbus)* in the Vanderkloof Dam downstream. Both species migrate upstream from the lower reservoir to the lotic region between the reservoirs where they spawn in gravel-bed riffles. The entire population of yellowfish in the Vanderkloof Dam is believed to depend on recruitment from this region. In addition, these *Barbus* spp. generally spawn during the first floods of spring. Allanson and Jackson (1983) suggested that continuous flows from the Gariep Dam through the year resulted in spawning events being triggered primarily by temperature (Paxton, 2004).

2.6.6.4 The effect of fish on reservoir trophic relationships

Current theory holds that, in lentic systems, the pelagic trophic level biomass is controlled both from below by producers (bottom up) and from above by consumers (top down) (see Section 2.6.7). The evidence supporting this assertion derives from studies of individual lakes and mesocosm and enclosure experiments (e.g. Vanni et al., 1990; Lazzaro, et al., 1992; Meijer et al., 1994, and many others). Trophic cascade hypotheses were mostly developed from north temperate lake data but Lazzaro (1997) has reviewed these data in order to analyse their application to tropical lakes and reservoirs. He concluded that it remains unclear whether hypotheses developed for temperate systems are appropriate for the tropics. Food webs of tropical systems are typically more complex and for practical reasons, the relevance of alternative models should be explored

It has been suggested that the effects of particular fish assemblages might cascade down the food web in both lakes and reservoirs, but that bottom-up and top-down effects may be asymmetrical (Quiros and Boveri, 1999). Top-down trophic interactions at the macrozooplankton-phytoplankton level have been queried by DeMelo et al., (1992), who concur with other studies that question the validity of the biomanipulation/trophic cascade/top-down model.

Quiros and Boveri (1999) examined the differences between reservoirs supporting different fish assemblages. In 31 reservoirs in Argentina, they differentiated between Type I reservoirs with planktivorous fish but no piscivores, and Type II reservoirs with both piscivores and planktivores and found that in Type I reservoirs, there was a huge increase in macrozooplankton biomass with decreased body size. In Type II reservoirs they found that both biomass and size of macro-zooplankton had decreased or remained unchanged. They interpreted this to mean that planktivores had not suppressed macrozooplankton biomass, but they may control macrozooplankton size. For both types of reservoirs, bottom-up effects from nutrients to algae were strong, but top-down effects on phytoplankton increased in the planktivore dominated Type I reservoirs.

These results are coincident with the view that in reservoirs where facultative zooplanktivores are dominant, such fish play a central role in reservoir ecosystem function by controlling zooplankton size, and also with the hypothesis (Pace, 1986) that zooplankton size structure, not biomass influences nutrient-phytoplankton relationships (Quiros and Boveri, 1999). For reservoirs with both piscivores and planktivores it would be expected that efficiency of nutrient transfer would diminish from the bottom up (Quiros, 1998a). These patterns were not entirely displayed in the above results. Speculations as to the reasons for this might be that human actions influence both bottom-up (nutrient loading) and top-down (fish stocking) mechanisms. In reservoirs fish assemblage is usually dependent on human action through fish introductions and stocking. The dependence of facultative planktivorous fish on benthic resources may also explain some of the results (Quiros and Boveri, 1999).

During recent years there has been increasing interest in determining the role played by perch *(Perca fluviatilis* L.), the dominant species in temperate lakes and reservoirs, especially in relation to their pressure on the zooplankton communities (Zalewski et al., 1990). These authors, however, are of the opinion that insufficient attention has been paid to the fish fry communities as a biomanipulating tool. Despite the usually lower biomass of fry than mature fishes, production of juveniles of fish such as Percidae and Cyprinidae has been found to be mostly on the level of 60-70% of the total production of a fish community. In addition adult fish from the above two groups consume a much smaller amount of zooplankton than the juveniles (Persson, 1986, in Zalewski et al., 1990). In a five-year study on the Sulejow Reservoir in Poland (Zalewski et al., 1990) the feeding strategy of perch and the strong correlation between the density of the dominating perch fry and their growth rate, indicated their influence on the density of large filter-feeding zooplankton, and thus water quality. These authors believe that efforts should be focused on the elaboration of the optimal fry density from the point of view of water quality and fisheries management. Their data indicate that the level of 500 fry $100m^{-2}$ in mid-July would be optimal for Sulejow Reservoir to maintain populations of large Cladocera at a high and balanced level. On the other hand, sufficient optimal food would accelerate growth of fry to increase their winter survival and reduce their vulnerability to predators, thus increase amount and quality of fish yield.

Fish can alter both the internal cycling of nutrients and the primary producer biomass in aquatic systems through a variety of mechanisms (Schaus et al., 1997). These include incorporating nutrients into body tissue (Deegan, 1993) and the converse release by decomposition of fish tissue (Parmenter and Lamarra, 1991), as well as by excretion (Braband et al., 1990). Excretion by fish has been shown to affect phytoplankton biomass, productivity and community structure, and serves to recycle nutrients within the water column or transport nutrients from the benthos to the epilimnetic phytoplankton depending on the feeding habits and movement patterns of the fish. The release of nutrients from benthic feeders can be viewed as a source of new nutrients that is fundamentally different from nutrients recycled by pelagic feeding fish or zooplankton. Schaus et al. (1997) quantified nutrient release by gizzard shad, a detritivorous fish in a eutrophic reservoir, Acton Lake, Ohio, USA. Their results indicated that nutrient excretion by detrivorous fish can be an important source of nutrients to open waters, especially when other sources of nutrients are reduced.

A study on the shallow, hypertrophic Bautzen Reservoir in Germany showed that diet shifts in under yearling fish may change the patterns of P recycling and distribution. In the case of the dominant perch, this bottom-up impact of nutrient regeneration by fish was connected to shifts in zooplankton density. Especially during the midsummer decline of *Daphnia* when the under yearling fish fed mainly on benthic prey, excretion of P by fish was higher than removal by feeding of P stored in pelagic prey. In comparison, during periods of high zooplanktivory, in spring and autumn, pelagic P consumption dominated and P release by fish was comparatively lower (Mehner et al., 1998b).

Despite the importance of internal nutrient loading by gizzard shad, nutrient transport by other fish species and nutrient recycling by planktivorous fish, many lake nutrient budgets have either downplayed or ignored the role of fish. Although fish may not regulate major nutrient fluxes in lakes, inclusion of their roles may help to balance nutrient budgets and depict more accurately the cycling of nutrients within these systems. Inclusion of fish is especially warranted in systems with a high fish biomass or where detritivorous or benthic feeding fish are abundant. In addition population biomass and size structure of fish may alter the impact that fish populations and communities have on nutrient dynamics and ecosystem productivity. Approaches that integrate fisheries biology and ecosystem ecology will foster insight into the processes underlying the structure and function of aquatic ecosystems.

2.6.6.5 Management

In South Africa there have been relatively few studies on the consequences of reservoir construction on indigenous fish populations and its effect on recruitment. In many cases knowledge of the relationship between the hydrological regime and the life-histories of the SA fish fauna is sparse or non-existent. The causal links between river regulation and the demographic decline of freshwater fish populations are therefore inadequately understood (Paxton, 2004).

Management of fisheries involving introduced species in South African reservoirs is a specialized study which will not be discussed in this review.

2.6.7 The reservoir food web

2.6.7.1 Introduction

Species composition and biomass of biota in any water body depend on a number of factors. Resources for growth, such as nutrients (which occur at the “bottom” of the stylized food chain) are referred to as bottom-up factors, and controlling factors such as grazing and predation (at the "top" of the food chain) are known as top-down factors (Komarkova et al., 1995; Horn and Horn, 1995; Mehner et al., 1998a; Lazzaro et al., 2003; Matveev, 2003; Horn, 2003; Korponai et al., 2003). (Figure 16)

For management purposes, the more that is known about the food web of a reservoir – the organisms that occur there and their interrelationships – the more possible it is to manage the system in an informed, productive and sustainable manner.

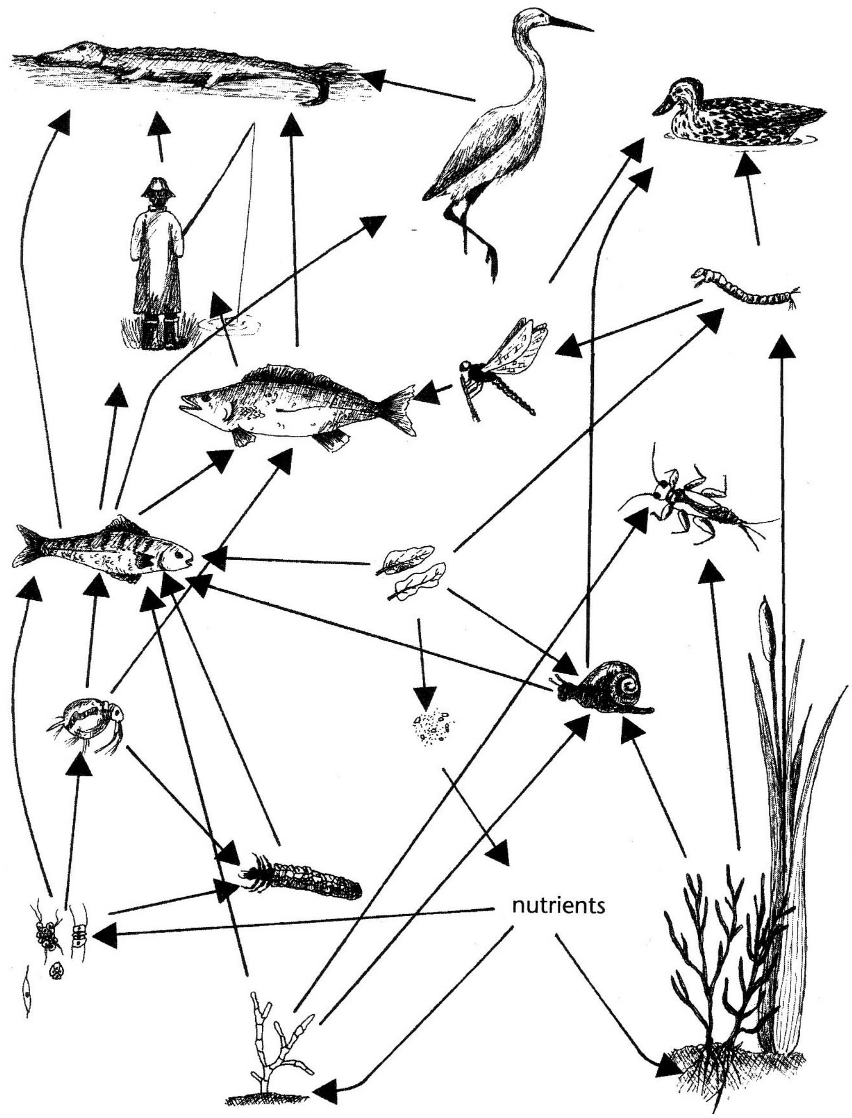


Figure 16. A hypothetical food web of a typical sub-tropical African lake. (After Davies and Day, 1998).

In addition, microbial studies in the last two decades have demonstrated the existence of a diverse microbial community in aquatic planktonic ecosystems where microbial production is integrated into the pelagic food web at all levels (Gaedke et al., 1995 in Comerma et al., 2003). The microbial loop mediates carbon and energy flow from bacterioplankton to zooplankton (Thouvenot et al., 1999a; Comerma et al., 2003). Sommaruga and Robarts (1997) concluded that bacteria are relevant members of the limnetic planktonic food web, in terms of both biomass and production, and Riemann and Christoffersen (1993) suggested the increasing importance of the microbial food chain compared to the classical food chain (Figure 17) along a gradient of increased productivity.

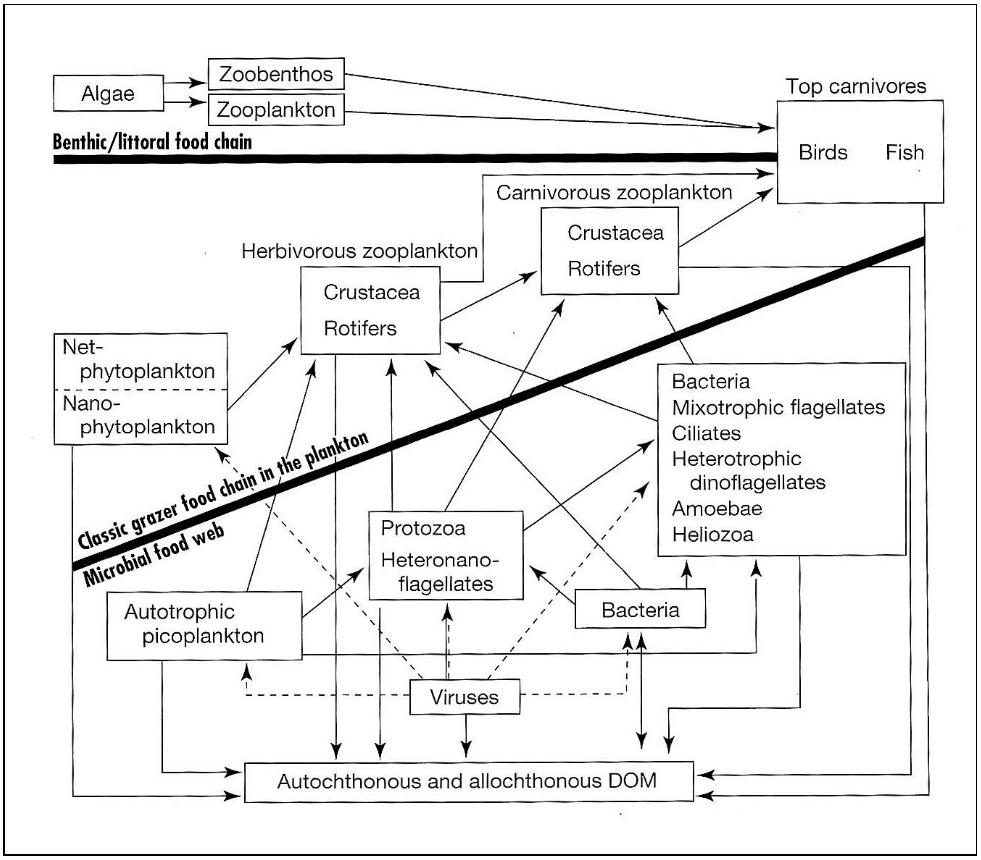


Figure 17. The contemporary view of microbial food web structure (below the lower thick black line) in relation to the "classic" grazer food chain in plankton (above the thick line). Full lines and arrows indicate feeding interactions, broken arrows indicate viral infections. The pool of DOM used as a substrate by the bacteria is replenished by various release processes (excretion, exudation, cell lysis, sloppy feeding) from each compartment and from the catchment (Modified from Weisse and Stockner (1993) in Kalff (2002)).

2.6.7.2 Bacterivory and Picoplanktivory

Protozoans are consumers of bacterioplankton, phytoplankton and organic matter (Fenchel, 1987, in Gomes and Godinho, 2003; Laybourn-Parry, 1992; Simek and Straskrabova, 1992), and thus are an important link in the transfer of energy from bacteria to the higher trophic levels since they are a common staple diet for microcrustaceans and fish larvae (Porter et al., 1985, in Gomes and Godinho, 2003), and reach very high population densities. Gomes and Godinho (2003) emphasise that protozooplankton research is essential for the understanding of the dynamics of aquatic ecosystems.

An increasing number of studies have underlined the importance of flagellate and ciliate protozoans in the trophic web (Sanders et al., 1992; Riemann and Christoffersen, 1993; Pace and Vaque, 1994; Burns and Schallenberg, 1996; Thouvenot et al., 1999b). Heterotrophic flagellates are generally thought to be the main consumers of bacteria in fresh waters (Riemann, 1985; Gude 1986, both in Thouvenot et al., 1999), but phagotrophic phytoflagellates (photosynthetic flagellates which are also particle feeders) have grazing rates that are similar to heterotrophic flagellates (Bird and Kalff 1986 in Thouvenot et al., 1999b). In addition ciliates can be important consumers of bacteria in fresh water (Simek et al., 1995). To date very few reports are available in which bacterivory or picoplanktivory of freshwater pelagic ciliates have been well documented in situ (Sherr et al., 1991; Simek and Straskrabova, 1992; Sommaruga and Psenner, 1993; Simek et al., 1995). On the other hand, increasing evidence from marine systems indicates that some ciliate taxa are voracious consumers of bacteria (Sherr and Sherr 1987; E.B. Sherr et al., 1989; B.F. Sherr et al., 1989, all in Simek et al., 1995). In Rimov Reservoir in Czechoslovakia, Simek et al. (1995) found that ciliates consumed a significant proportion of the bacterial production in the reservoir (20% as opposed to 70% consumed by heterotrophic flagellates). However, some taxa were found to consume mostly picoplankton in natural conditions where availability of organic carbon in picoplankton is almost equal to bacteria. Most ciliate species preferred larger picoplankton, as has been reported for freshwater heterotrophic flagellates (Simek and Chrzanowski, 1992). This size-selective grazing may have ecological impacts on natural bacterial assemblages (Gonzalez et al., 1990; Simek et al., 1994).

Although protists are thought to be the main consumers of bacteria, some metazoans can nevertheless play an important role in regulating bacterial communities (Jurgens et al., 1994). Among these, Cladocera rather than Copepoda are thought to be effective consumers of bacteria (Geller and Muller, 1981; Pace et al., 1983; Gude, 1988; Jurgens et al., 1994) that can be an important food source during the summer in some temperate waterbodies (Pace et al., 1983). Although some rotifers consume heterotrophic bacteria (Boon and Shiel, 1990; Ooms-Wilms et al., 1995; Ooms-Wilms, 1997; Thouvenot et al., 1999), their contribution to regulating the bacterial plankton is thought to be modest (Sanders et al., 1989; Pace et al., 1990; Pernie et al., 1990).

In the newly flooded Sep Reservoir in France, Thouvenot et al. (1999a) measured bacterial consumption in the epilimnion (1m) and metalimnion (7m) during the period of thermal stratification. They found that the main consumers were the cladocerans *Daphnia longispina* and *Ceriodaphnia quadrangula,* accounting for, on average, 72% of the potential total predation of bacteria at 1m and 56% at 7m, especially during May, June and August. Heterotrophic nanoflagellates accounted for 12% at 1m and 13% at 7m, and ciliates, 4% at 1m. They concluded that in a newly flooded reservoir, metazoan zooplankton seemed to be the main consumers of bacteria, and that predation of heterotrophic nanoflagellates and ciliates by zooplankton Crustacea could account for the low contribution of these organisms to bacterial consumption.

Straskrabova and Simek (1993) considered the possibility that protozooplankton and metazooplankton might compete with one another as microbial grazers, and concluded that the protozooplankton were more advantaged than the metazooplankton as follows:

1 They are capable of faster growth and thus a faster adaptation to changing food resources. This favours protozooplankton after spring overturn until the slower

growing cladoceran grazers increase and control both phytoplankton and protozooplankton (Simek et al., 1990; Arndt and Nixdorf, 1991; Simek and Straskrabova, 1992). The capability for faster growth is also an advantage for protozooplankton in areas of low and heterogeneous food resources, for example during winter in temperate lakes (Arndt and Nixdorf, 1991). The low temperature itself also seems to favour the occurrence of larger protozoans (Sime-Ngando and Hartmann, 1991).

2. They survive better under high planktivorous fish predation pressure.
3. In some adverse conditions, the abundance of metazooplankton decreases sharply and then the pelagic assemblage consists entirely of microbes (e.g. in oligotrophic mountain clear-water lakes with low alkalinity (Simek and Straskrabova, 1992)). Similarly, in anoxic layers of eutrophic and dystrophic lakes, protozooplankton usually prevails over metazooplankton (Simek and Straskrabova, 1992; Sime-Ngando and Hartmann, 1991).

2.6.7.3 Phytoplankton-zooplankton-fish interrelationships

Zooplankton – representing an important trophic level and a deciding factor for the flow of matter and energy – can show a conspicuous seasonal and annual variability of its biomass caused by availability of food sources (Horn and Horn, 1990; Urabe, 1990; Mason and Abdul-Hussein, 1991; Matsumura-Tundisi et al., 2002; Roue et al., 2002; Korponai et al., 2003; Horn, 2003), predator structures (Urabe, 1990; Mason and Abdul-Hussein, 1991; Ketelaars and van Breemen, 1993; Seda and Duncan, 1994; Gliwicz, 1994; Korponai et al., 1997; Mehner et al., 1998a; Pichlova and Brandl, 2003; Vasek et al., 2003; Matveev, 2003; Horn, 2003; Lazzaro et al., 2003; Hrbacek et al., 2003; Korponai et al., 2003; Tatrai et al., 2003) and environmental conditions (Urabe, 1990; Schram and Marzolf, 1993; Hulsmann and Weiler, 2000; Benndorf et al., 2001; Horn, 2003; Hart, 2004). Its dynamic and long-term changes are of particular interest in terms of its great importance in the issue of water quality. This is the basis of biomanipulation in lakes and reservoirs (Horn, 2003). Biomanipulation as a management method will be discussed in Section 3.1.2. However reference will be made to it in this section where relevant.

2.6.7.3.1 Herbivory

Numerous investigations on the interactions between phytoplankton and zooplankton have been undertaken. Mostly they have been based on short-term experiments in enclosures, laboratory or in situ experiments. Long-term observations of these relationships are scarce (Horn and Horn, 1990, 1995).

Different indices have been suggested for assessing grazing effects at the level of whole lakes (Matveev and Matveeva, 1997). Lampert (1988) used total zooplankton biomass to predict declines of phytoplankton in lakes due to grazing. He suggested a threshold of 1.5g m-2 of zooplankton biomass to be sufficient to cause a clear-water phase. Pace (1984) in Matveev and Matveev (1997) suggested that zooplankton composition and mean body size (not absolute biomass) influences the P-Chl a relationship in lakes. Zooplankton body length was found to be a good predictor of the intensity of grazing and variance in phytoplankton biomass in 25 North American lakes as well as in whole-lake experiments (Carpenter et al., 1991, 1996). The ratios of biomass of zooplankton to phytoplankton (McCauley and Kalff, 1981; Jeppesen et al., 1990; Boon et al., 1994) or Cladocera to phytoplankton (Schriver et al., 1995) have also been used as indices of grazing. Zooplankton community grazing rates have also been estimated for different lakes (Hart, 1986; Jarvis, 1986) and compared with phytoplankton growth (Reynolds, 1984 and Sterner, 1989, both in Matveev and Matveeva, 1997).

Phosphorus and chlorophyll a relationships have been used for the prediction of lake trophic state and thus lake restoration measures. However, because of logarithmic scale and high residual variance, this relationship predicts a significant decline in algal abundance only when P-reduction is strong. Chl *a* can vary as much as two orders of magnitude for a given constant P-value for inter-lake comparisons (Mazumder, 1994; Baigun and Marinone, 1995), or one order of magnitude for intra-lake comparisons (Ferris and Tyler, 1985 in Matveev and Matveeva, 1997). Zooplankton grazing may determine a considerable part of this residual variance in Chl *a* (Carpenter et al., 1995). Inter-lake comparisons revealed that when grazers are not under intensive fish pressure, Chl *a* increases with total P at a much slower rate than when they are (Hansson, 1992; Sarnelle, 1992; Mazumder, 1994).

Population dynamics of algae can be affected directly, by grazing (or conversely by non-grazing), or indirectly, for example, by elimination of competitors or changes in nutrient conditions by the zooplankton (Horn and Horn, 1995). Colony forming green algae can take advantage of the zooplankton suppression of other competing algae, and of nutrient uptake via gut passage (Horn and Horn, 1990). Those with gelatinous sheaths are protected from an herbivore's digestive enzymes and are therefore hardly digestible, but they are permeable to certain nutrient ions (Porter, 1976, in Horn and Horn, 1995). Cyanobacteria can also profit from Crustacea that suppress their competitors (Horn and Horn, 1995). Certain species may inhibit zooplankton mechanically or by toxins (see Section 2.6.1) (Benndorf and Henning, 1989; de Bernardi and Giussani, 1990; Fulton and Jones, 1991; Nauwerck, 1991; Lathrop and Carpenter, 1992). Filamentous cyanobacteria reduce the growth rate of *Daphnia* species, but their effect is smaller on small species. When the filament density reaches a critical concentration, zooplankton is no longer able to control it (Gliwicz, 1990a). Gliwicz (1990b) showed that at high levels of abundance, *Aphanizomenon* sp reduces the growth rate of large *Daphnia* species (*D. magna* and *D. pulicaria*), whereas small species (*D. cucullata and D. hyalina*) tend to grow faster.

Horn and Horn (1990, 1995) investigating the diatom *Asterionella formosa* in the Saidenbach Reservoir in Germany, found that it was ingested by *Daphnia galeata* and the dominant cyclopoid (*Cyclops vicinus*). However, their analysis revealed that the dynamics of this diatom was essentially influenced by other factors such as stratification conditions (good mixing in the epilimnion) and sedimentation. A further important fact for diatoms is the elimination of nanoplankton competitors by *Daphnia*, thus improving the nutrient conditions by removing P-competition. Because diatoms are important for phosphorus export to the sediment, and therefore for self-purification of the water column (Grim, 1967, in Horn and Horn, 1995), this possible shift as a result of *Daphnia* grazing can serve a desired effect (Horn and Horn, 1995).

It is widely accepted that nanoplankton is a good ingestible food source and is therefore intensively grazed upon by filter feeding crustaceans, particularly cladocerans. On the other hand copepods are not primarily 'micro-filter feeders' and are not able to graze on particles as small as those filtered by cladocerans (Horn and Horn, 1990, 1995).

Studies of the role of zooplankton grazing in the Southern Hemisphere are less numerous than the Northern hemisphere (Matveev and Matveeva, 1997). However, for 97 Argentinean lakes and reservoirs, Quiros, (1990) found that mean zooplankton size was a significant predictor of residual variance in the TP-Chl *a* relationship, suggesting that grazing was important. In spite of this, various authors have claimed that neither TP-Chl *a* models of the Northern Hemisphere (Baigun and Marinone, 1995) nor the concept of biomanipulation/top-down control (Boon et al., 1994) are likely to be applicable to regions in the Southern hemisphere. Boon et al. (1994) maintained that in Australian inland waters, grazing on phytoplankton is unlikely to be enhanced by biomanipulation because efficient Cladoceran grazers are rare. They also speculated that cyanobacteria might have detrimental effects on zooplankton, restraining grazers' growth.

Testing the above contradictions, Matveev and Matveeva (1997) studied grazer control and nutrient limitation of phytoplankton biomass in two Australian reservoirs, Lake Hume and Lake Dartmouth. Their results showed that, in fact, zooplankton grazing might be sufficient to affect the residual variance in TP-Chl a regressions as described for other regions of the world (Quiros, 1990; Sarnelle, 1992; Mazumder, 1994; Carpenter et al., 1995). Their findings also suggested that the observed grazing effects were sufficient to make the crustacean length model for biomanipulation potential applicable to the conditions of the two reservoirs studied. Matveev and Matveev (1997) also hypothesized that Australian water bodies containing large daphnids have a higher potential for biomanipulation than other lakes in the world. They qualified this with the caution that the hypothesis may be limited to the situations where total grazing impacts are not confounded by ungrazability of phytoplankton, or not overridden by positive food web interactions such as stimulation of phytoplankton by fish (Matveev et al., 1994a), by invertebrate predators (Matveeva and Matveev, 1995) or by copepods (Cruz-Pizarro and Carrillo, 1991; Lyche et al., 1996).

There are studies showing that southern hemisphere Cladocera (Matveev and Balseiro, 1990) and Copepoda (Burns and Xu, 1990) can benefit from feeding on cyanobacteria at a population level. However, as Australian zooplankton differ taxonomically from other continents, conclusions about grazer-cyanobacteria interactions based on northern hemisphere studies should be transferred with considerable caution (Matveev and Matveeva, 1997).

2.6.7.3.2 Predation

The effects of predation on zooplankton

Predation is considered to be a major driving force in shaping zooplankton communities (Gliwicz, 1994; Seda and Kubecka, 1997). Planktonic herbivores are extremely vulnerable to predation, particularly by visually oriented planktivorous fish that are forced to feed in the most illuminated strata of the waterbody where food is most abundant, but where fish can visually locate their prey at light intensities as low as 10 lux (Gliwicz, 1994). Large-bodied herbivores such as *Daphnia* are most conspicuous, resulting in their being the first to be decimated or even locally driven to extinction (Hrbacek et al., 1961; Brooks and Dodson, 1965; Hall et al., 1976; Lazzaro, 1987, all in Gliwicz, 1994).

Mortality is the most apparent direct top-down effect of predation. However, there are other top-down and bottom-up indirect effects of predation on prey density that are transmitted through individual growth and reproduction (Gliwicz, 1994; Seda and Kubecka, 1997). The effect of predation is always combined with food limitation, since each prey individual has to compromise between the two conflicting demands – to eat well in order to grow and reproduce, and the imperative to survive long enough to reproduce. Because of this trade-off, it is often difficult to determine whether a decrease observed in a prey population exposed to strong predation is really a direct effect (predator induced mortality) or an indirect effect (a decline in the prey's reproduction caused by reduced foraging associated with predator avoidance). When a visually oriented predator selects for large prey body size or body mass (optimal foraging), it also selects for other prey body characteristics that would make prey more conspicuous and less evasive. This is evident in increased clutch size in cladocerans and copepods. Larger clutch size also results in larger body mass making a prey individual even more desirable by an optimally foraging predator. When a predator selects for an ovigerous female, it also causes a decrease in reproduction, thus the birth rate and death rate in the prey population are simultaneously affected (Gliwicz, 1994).

A sudden decline in the population density of a planktonic herbivore (or even its local extinction) should not be regarded automatically as a direct effect of predation due to increased mortality rate. It should rather be considered as a result of both effects working simultaneously: a direct effect of predation through mortality and an indirect effect of predation that has resulted from a decrease in growth and reproduction (Gliwicz, 1994).

2.6.7.3.3 The effect of fish communities on freshwater ecosystems

The role of fish in regulating the structure and function of freshwater ecosystems (top-down control) either directly or indirectly is now generally accepted in limnological literature. Nevertheless, several authors have cast some doubts on the validity of top-down regulation and its application for management of water quality (biomanipulation) (Evans, 1990; McQueen, 1990; McQueen et al., 1992; De Melo et al., 1992; Seda and Kubecka, 1997; Hart, submitted). Wetzel (2001) even cautions against the use of the terms bottom-up and top-down, claiming that they are not only highly ambiguous but that they promulgate concepts that are not founded in thermodynamic relationships of nutrient storage and recycling and are used without recognition of the true complexities of the regulation of food web interactions.

It has been observed in many studies that with a decrease in the biomass of planktivorous fish, the biomass of zooplankton increases (Carpenter et al., 1987; Luecke et al., 1992, both in Korponai et al., 1997; Riemann et al., 1990; Christoffersen et al., 1993, Korponai et al., 1997; Sarvala et al., 1998), resulting in a decrease in algal biomass (Jeppesen et al., 1997; Bergman et al., 1999; Drenner and Hambright, 1999). Within the zooplankton community, the relative abundance of cladocerans, as well as the average size of daphnids increases with decrease in predation pressure by fish (Giussani and Galanti, 1992). With an increase in the biomass of planktivorous fish, the biomass of the cladoceran plankton decreases (Jeppesen et al., 1997; Matyas et al., 2003), and in turn, that of the phytoplankton increases (Carpenter et al., 1987, in Korponai et al., 1997; Riemann et al., 1990; Vanni et al., 1990). At the same time, the size of adult daphniid females gets smaller in response to an increase in the predation pressure by positively size selective fish (Machacek, 1991, 1993; Stibor, 1992; Taylor and Gabriel, 1992; Tatrai et al., 2003; Korponai et al., 2003), imposing the reduced fecundity associated with the general direct relationship of body size and brood size in copepods and cladocerans. Small-bodied cladocerans like *Bosmina* spp and *Chydorus* spp are generally more abundant than large-bodied species when the predation by fish is intense (Ramcharan et al., 1996; Vijverberg and Boersma, 1997). Besides small cladocerans, copepods dominate the zooplankton under such conditions (Vijverberg et al., 1990).

Several enclosure and whole-lake experiments have shown that the reduction in stocks of planktivorous fish may cause decreases in both algal biomass and the proportion of cyanobacteria (Reinertsen et al., 1990; Jeppesen et al., 1997; Bergman et al., 1999; Drenner and Hambright, 1999; Tatrai et al., 2003). Reduction of planktivorous fish biomass often does produce clear water conditions initially, but the probability of sustained effects depends on nutrient loading and the development of stabilizing feedback mechanisms (Benndorf, 1990; Lammens, 1999). Field studies have also shown that fish community composition could have profound effects on phytoplankton productivity by several indirect effects and interactions that affect nutrient availability (Persson et al., 1999). Many fish species use resources from different habitats and trophic levels that may either suppress or promote phytoplankton productivity (Braband et al., 1990; Schindler et al., 1993). If shallow lakes are eutrophied, the fish community typically shifts from percids to cyprinids such as bream, carp and roach, which are omnivorous under certain circumstances (Simonian et al., 1995; Tatrai et al., 1998). When feeding on bottom dwelling organisms, these species potentially translocate nutrients from the sediments to the water column (Tatrai et al., 1990; Breukelaar et al., 1994; Persson, 1997). Populations of these species are in turn dominated by small young individuals many, of which are planktivores. This results in high predation pressure on zooplankton, and consequently a low predation pressure on phytoplankton.

The removal of about 50% of cyprinids and their replacement by predatory fish, from a shallow lake in the Kis-Balaton Reservoir System (Tatrai et al., 2003) had an impact on almost all of the observed parameters. Summer mean chlorophyll *a* values decreased by 43%, the phytoplankton community changed from being dominated by filamentous bloom-causing cyanobacteria to a much more diverse community with green algae, dinoflagellates and cryptomonads in the year following fish removal. However, the zooplankton community showed the composition typical of lakes with high predatory pressure of omnivorous cyprinid fish leading to a decrease in mean body length and fecundity of daphnids. The authors concluded that the reduction of biomass of omnivorous fish cascaded down to phytoplankton via changes in nutrient availability rather than grazing pressure from crustaceans.

The critical fish biomass below which herbivore cladoceran species can dominate the zooplankton differs depending on which species dominate the fish community (Benndorf, 1995). Effects on the phytoplankton of different fish communities were studied by Matyas et al. (2003) in three shallow Hungarian reservoirs - Casette, the outer reservoir of the Kis-

Balaton Water Protection System, and the Marcali Reservoir. Possible interactions between nutrient concentrations and phytoplankton biomass in these reservoirs were also examined. Considerable differences in the phytoplankton population dynamics and the proportions of different nutrient forms were observed between the three sites, which could be explained by the presence of different fish stocks in these reservoirs. The predominance of Prussian carp, which can consume either planktonic or benthic organisms (Korponai et al., 2003) controlled the zooplankton in the outer Kis-Balaton reservoir. In the Casette, a massive fish kill decimated most of the Prussian carp leaving roach (grazers of larger-bodied species) (Paulovits et al., 1998) and young Prussian carp as the characteristic fish species that were not able to control the zooplankton biomass. In the Marcali Reservoir, silver carp grazing affected the quality rather than the quantity of phytoplankton.

To ascertain whether an investigation of the size composition of the cladoceran and copepod fractions of the zooplankton could supply important information on the impact of fish on zooplankton, Hrbacek et al. (1986) introduced a (then) new parameter, namely the percentage of biomass of large cladocerans in the total cladoceran biomass (%LCla). This method was used by Hrbacek et al. (2003) to investigate the long-term changes in zooplankton biomass in Slapy Reservoir in the Czech Republic in relation to changes in the composition of the fish stock. By monitoring changes in this parameter, information was obtained on changes in the feeding activity of individual groups within the fish stock, both in the long-term and during the year.

2.6.7.3.4 Effects of fish on the longitudinal distribution of zooplankton in reservoirs.

Among studies focusing on longitudinal biotic trends in deep elongated reservoirs, a relatively large number have dealt with causes of heterogeneity in zooplankton abundance and community structure (Hart, 1990; Betsill and van den Avyle, 1994; Thys et al., 1998; Seda and Machacek, 1998; Seda and Devetter, 2000). However, less attention has been paid to the spatial distribution of fish (Fernando and Holcik, 1991; Kubecka and Wittengerova, 1998; Brosse et al., 1999), and very few studies have addressed the longitudinal distribution patterns of both groups simultaneously (Pont and Amrani, 1990; Urabe, 1990). Vasek et al. (2003) maintained that until their study on Rimov Reservoir in the Czech Republic, no apparent attention had been paid to spatial heterogeneity in fish diet in the open waters of an elongated reservoir. The objectives of their study were (i) to assess the longitudinal distributions of fish and zooplankton in the Rimov Reservoir, (ii) to describe diet spectra and diet overlap for the three most abundant fish species i.e. roach (*Rutilus rutilus*), bleak (*Alburnus alburnus*) and bream (*Abramis brama*), and (iii) to evaluate size-selective predation on *Daphnia* by these cyprinids at the scale of the entire longitudinal transect.

(i) Previous work carried out on the nutrient and phytoplankton concentrations (Hejzlar and Vyhnalek, 1998; Seda and Devetter, 2000) and zooplankton (Seda and Machacek, 1998; Seda and Devetter, 2000) had found that small-bodied zooplankton were more dominant in upstream than downstream sites. Vasek et al. (2003) found higher fish abundance in the riverine zone which could have been related to spawning in May, and its suitability as a nursery habitat in August (Vasek et al., 2002, in Vasek et al., 2003). Other authors have also recorded higher fish abundance in the upper part of elongated reservoirs during thermal stratification (Urabe, 1990; Pont and Amrani, 1990; Fernando and Holcik, 1991; Brosse et al., 1999; Swierzovski et al., 2000). Thus the distinct longitudinal gradient in fish distribution seems to be common within elongated reservoirs having natural tributaries, although further investigations of the importance of the tributary zone for reservoir fish stocks are required (Vasek et al., 2003).

(ii) The three fish species, roach, bream and bleak foraged almost exclusively on crustacean zooplankton. This was apparently the result of scarcity of alternative food sources as the fauna of the littoral zone is impoverished due to steep reservoir sides and water level fluctuations (Duncan and Kubecka, 1995) forcing the fish to forage offshore. In addition, most of the reservoir bottom lies in the anoxic hypolimnion during the warm months and is thus unavailable for fish (Hejzlar and Vyhnalek, 1998) making the epilimnion the only available habitat for fish feeding. Such planktivory in large roach and bream has been observed elsewhere as a consequence of food availability and competition (Michelsen et al., 1994; Garcia-Berthou, 1999; Persson and Hansson, 1999), although the diet of larger roach and bream usually consists of benthic invertebrates, detritus, and in the case of roach, also molluscs, filamentous algae and macrophytes. In May, *Daphnia galeata* was the highly preferred prey item of all fish except the smaller length class in the riverine zone that consumed mainly bosminids. In August all bleak and roach and large bream preyed on *D. galeata* and *Leptodora kindtii,* with large bream also feeding significantly on *Diaphanosoma brachyurum*. Higher proportions of *L. kindtii* occurred in the transition and lacustrine sites. All bream preyed on cyclopoid copepods in the riverine zone, but this was insignificant in bleak and roach. One reason for the observed differences in predation might be the fact that roach and bleak forage by visual particulate feeding (Lammens and Hoogenboezem, 1991, in Vasek et al., 2003) while bream use mainly filter feeding (van den Berg et al., 1994b, in Vasek et al., 2003).

(iii) A decreasing mean size of *D. galeata* was observed from dam to river site in May and August. This indicates stronger fish predation in the upstream part of the reservoir. Lower diet overlaps of the three fish species in the riverine site compared to the transition and lacustrine sites suggested intense competition for food at the upstream station.

Since it is widely recognised that zooplankton composition and biomass can be strongly influenced by planktivorous fish, there is a great need for more detailed information on longitudinal variation of fish predation on zooplankton to better understand the complex ecology of reservoirs having persistent environmental gradients (Vasek et al., 2003).

2.6.7.3.5 Non-predatory mortality of zooplankton

Non-predatory mortality (also sometimes designated as ecological, realized or non-consumptive mortality) includes natural death due to senescence, disease, limiting physical or chemical factors (Gladyshev et al., 2003). Estimations of non-predatory mortality are important for the study of detritus food webs and fluxes of matter and energy in aquatic ecosystems (Velimirov, 1991; Wetzel, 1995). Top-down biomanipulation theory is based on increasing the abundance of large daphnids by decreasing their predation (consumptive mortality), by reducing the abundance of planktivorous fish. If non-predatory mortality of daphnids predominates in a particular water body, biomanipulation will be ineffective (Gladyshev et al., 2003; Dubovskaya et al., 2003).

A number of attempts to estimate non-predatory mortality have been made (Gladyshev and Gubanov, 1996; Dubovskaya et al., 1999; both in Gladyshev et al., 2003; Gries and Gude, 1999). Gladyshev et al. (2003) and Dubovskaya et al. (2003) evaluated non-predatory mortality of *Daphnia cucullata, D. longispina* and *Cyclops vicinus* in Bugach Pond, a reservoir in Russia, finding it to be considerable, and often the determinant component of the zooplankton population dynamics. This is in agreement with other authors who demonstrated a low contribution of fish predation to *Daphnia* mortality (Boersma et al., 1996; Mehner et al., 1998a; Hulsmann et al., 1999; Romare et al., 1999; Hulsmann and Weiler, 2000). Comparatively high mortality at the beginning of the growing season was recorded in agreement with Boersma et al. (1996), Mehner et al. (1998a) and Hulsmann and Weiler (2000). Sharp fluctuations of mortality within and between sampling seasons could not be explained but may have been caused by a species-specific infection (Gries and Gude, 1999), poisoning by cyanobacterial toxins such as microcystin (Rohrlack et al., 1999) or low food quality (Muller-Navarra, 1995; Gulati and DeMott, 1997; Elser et al., 2000). More studies are necessary to understand non-predatory mortality fluctuations (Gladyshev et al., 2003).

A midsummer decline of large-bodied cladocerans is a common phenomenon in temperate and some sub-tropical lakes and reservoirs (Hulsmann and Weiler, 2000; Benndorf et al., 2001; Hart, 2004). Many studies have analysed *Daphnia* population dynamics during midsummer declines but explanations for the observed mortality patterns have remained speculative and contradictory (Threlkeld 1979; De Stasio et al., 1995; Wu and Culver, 1994; Boersma et al., 1996; Werner et al., 1996; Hansen et al., 1993; Whiteside and Hatch, 1997; Cerny and Bytel, 1991; Hulsmann and Mehner, 1997). Hulsmann and Weiler (2000) analysed the dynamics and structure of a *Daphnia galeata* population in Bautzen Reservoir in Germany, prior to, and during a midsummer decline of this species. They concluded the following progression of events:

1. A quick increase in *Daphnia* abundance leads to the formation of a strong 'peak cohort' of about the same age.
2. During the clear-water phase, food conditions deteriorate, fecundity declines and hence recruitment is low; juvenile mortality is low but present.
3. Adult mortality increases when the 'peak cohort' reaches its mean life span which is reduced due to interactions between age-specific and starvation-induced mortality.
4. At this point, *Daphnia* population dynamics can no longer be explained without the onset of size-selective predation.

Hence the timing between enhanced mortality due to senescence on the one hand and predation on the other hand, may be decisive. Timing of both sources of mortality and hence the onset of a midsummer decline is thus controlled by the water temperature in the preceding winter and early spring (Hulsmann and Weiler, 2000; Benndorf et al., 2001).

2.6.7.3.6 Predatory invertebrates and their effect on the food web

Vertebrate and invertebrate predators have a widely recognized impact on zooplankton through size selective predation (Caramujo et al., 1997). The selective removal of large prey individuals by vertebrate predators is counteracted by the selective removal of smaller individuals by invertebrate predators. Gliwicz and Lampert (1993) found a total depletion of small cladoceran prey species in the presence of natural densities of the predatory copepod *Acanthocyclops robustus*. In the same enclosure experiment they found that in the large bodied prey species, old individuals were dominant and neonates nearly absent. The feeding rate of *Acanthocyclops robustus* has been shown to decrease with increasing prey body size (Gliwicz and Umana, 1994).

The predatory cladoceran, *Leptodora kindtii* is reported as being able to influence the zooplankton community significantly (Herzig and Auer, 1990; Lunte and Luecke, 1990; Branstrator et al., 1991; Arndt et al., 1993; Herzig, 1994). It consumes a great variety of mainly cladoceran species and also feeds on rotifers and nauplii (Herzig and Auer, 1990; Lunte and Luecke, 1990; Branstrator et al., 1991). It is size selective in its prey choice, feeding on smaller species or smaller individuals of larger species. The limiting factor for the capture of prey is the size of the 'feeding basket', a structure formed by five pairs of thoracic appendages (Herzig and Auer, 1990; Branstrator, 1998). Pichlova and Brandl (2003) studied the predatory impact of *L. kindtii* on the zooplankton community of Slapy Reservoir (Czech Rep) and concluded that it does influence the zooplankton in this reservoir, but only in limited periods during the summer, with the most pronounced impact in August.

Other morphological and behavioural features unrelated to body size may also be of extreme importance in regulating invertebrate predation (Li and Li, 1979). In the presence of waterborne cues produced by *Chaoborus*, cladocerans produce progeny with neckteeth or enlarged crests (helmets) (Havel, 1985; Havel and Dodson, 1984). Another common response is the presence of elongated tail spines in juveniles (Luning, 1992) and the production of larger newborns (Luning, 1995). Cyclopoid copepods also have the ability to induce tail spine elongation in daphnids (Caramujo et al. 1997).

As a result of enclosure experiments in Maranhoa, a mesotrophic reservoir in Portugal, Caramujo et al. (1997) concluded that the only important predation impact controlling the *Daphnia* population seemed to have originated from vertebrate predators. Elongation of the tail spines of juvenile daphnids seemed to have enabled them to coexist with the invertebrate predator, Acanthocyclops *robustus.* These authors emphasize that further evidence is needed, as is information on the role of fish kairomones on tail spine induction in *Daphnia.*

3. FACTORS AFFECTING WATER QUALITY OF RESERVOIRS, THEIR CONSEQUENCES AND THEIR MANAGEMENT

3.1 Pollution (Externally derived)

3.1.1 Suspended Sediments

3.1.1.1 Introduction

Although not a typical pollutant, sediment accumulation in reservoirs is conveniently treated in this context i.e. as a substance occurring or accumulating in unnatural or excessive quantities.

3.1.1.2 The need for sediment management in reservoirs

Most natural river reaches are approximately balanced with respect to sediment inflow and outflow. Dam construction alters this balance, creating an impounded river reach characterized by low flow velocities and efficient sediment trapping. The impounded reach will accumulate sediment and lose storage capacity until a balance is again achieved which would, if left unmanaged, occur after the impoundment has “filled up” with sediment (Morris and Fan, 1998; US Army Corps of Engineers, 1977; Salas and Shin, 1999).

As reservoirs age and sediments continue to accumulate, sediment-related problems will increase in severity and more sites will be affected. At any dam or reservoir where sustainable long-term use is to be achieved, it will be necessary to manage sediments as well as water (Morris and Fan, 1998; De Cesare et al., 2001).

3.1.1.3 Factors affecting sedimentation of reservoirs

Some of the most important factors affecting reservoir sedimentation are (Salas and Shin, 1999):

- Quantity of streamflow
- Quantity of sediment inflow into a reservoir
- Sediment particle size
- Reservoir operation

- **Quantity of streamflow**

  In general, the stream order is higher and the drainage basin bigger for reservoirs than for lakes (Thornton et al. 1981, in Thornton et al. 1990). The larger drainage basins associated with reservoirs may result in greater annual flows entering reservoirs than lakes. These factors indicate the potential for greater sediment loads to reservoirs. However, there is a finite amount of energy in rainfall that determines the rate of erosion and transport of particulate matter from the watershed to the stream (Thornton et al., 1990).

  Sediment (as well as particulate organic matter and adsorbed constituents) is transported primarily during storm events and elevated flows (Bilby and Likens, 1979; Johnson et al., 1976; Kennedy et al., 1981; Sharpley and Seyers, 1979; Verhoff and Melfi, 1978, all in Thornton et al., 1990). Rainfall frequency, intensity and duration as well as the amount of rain determine the rate of erosion and transport of sediment (Straškraba and Tundisi, 1999). Lohman et al., 1988, in An and Jones, 2000, reported that the volume in Nepalese lakes can be replaced fifteen times by monsoon runoff. River flow in South Africa tends to be in the form of seasonal high-flood peaks of short duration (Allanson, 1995).

- **Quantity of sediment inflow into a reservoir**

  Sediment export from a catchment is the result of the interaction between the weathering process acting on the parent material, and the hydraulic properties of flowing water which detach sediment particles, take them into suspension, and transport them to a point further downstream (Allanson, 1995).

  Sediment yield to water is determined by a variety of interactions namely rainfall erosivity, soil erodibility, slope steepness, slope length and vegetation cover (Allanson 1995; Pacini et al., 1993; Straškraba and Tundisi, 1999). Areas of intermediate precipitation appear to be the most susceptible to sediment production for in very arid areas, runoff is insufficient to move large amounts of sediment while in wet, humid areas, good vegetation cover stabilizes soil loss (Allanson, 1995). Erosion is generally higher in semi-arid and arid regions (Straškraba and Tundisi, 1999). Chanson (1998) found that reservoir sedimentation has been a serious problem in Australia, several reservoirs having become fully silted due to the designers not taking the sediment transport processes into account, and thus not introducing appropriate soil conservation practices.

  Agriculture is a major source of increased erosion, most agricultural soil remaining fallow and often barren or with insufficiently developed vegetation for long periods of time. Other sources of erosion include road construction, and any other building activities, that leave the soil barren for prolonged periods (Straškraba and Tundisi, 1999). The Masinga dam in Kenya, the top reservoir in a cascade of five, is described as a sediment-dominated impoundment (Pacini et al., 1993). The upper catchment is among the most fertile and productive regions in the country. Intensive agricultural practices with loss of soil cover, erosion by cultivation of steep slopes and fertilizer application dominate the physiography of the catchment, increasing erosion and leaching processes.

- **Sediment particle size**

  The finite energy of rainfall is reflected in differential transport of particulate matter (Thornton, 1990). Fluvial sediments generally contain finer clay and silt particles relative to the contributing watershed soils since less energy is required to transport the finer particles (Dendy, 1981; Duffy et al., 1978; Rhoten et al., 1979, all in Thornton, 1990). This fine material is then readily transported downstream to the reservoir, particularly during storm events or elevated flows.

  Fine material such as the colloidal and silt fractions tend to move through the system in a single flood event, whereas coarser particles are not taken into suspension but "leapfrog" from point to point down the river bed (Allanson, 1995). These coarser fractions constitute the bed material after the passage of the flood event.

  Sediment particle size has an important impact on the patterns of sedimentation within a reservoir, as discussed below.

- **Reservoir operation**

  Hydrological management of stored water, with its attendant consequences on water levels and depth stability, has major impacts on sedimentation. These are elaborated below (Section 3.2.4).

3.1.1.4 Distribution of sediments in reservoirs

The spatial distribution of sediment loading differs fundamentally in lakes and reservoirs. In lakes there is generally an equitable distribution of inflow around the periphery of the system. Reservoirs on the other hand, generally receive the majority of their inflow from one or two major tributaries located a considerable distance from the outflow. This promotes the development of pronounced physical and chemical gradients within the reservoirs that have important consequences for biological productivity and water quality (Morris and Fan, 1998).

Sediment deposition is the main problem affecting the useful life of reservoirs. Knowledge of both the rate and pattern of sediment deposition in a reservoir is accordingly required to predict the types of service impairments that will occur, the time frame in which they will develop, and the types of remedial strategies that may be practicable (Morris and Fan, 1998).

3.1.1.4.1 Longitudinal deposition zones

A reservoir changes the hydraulics of flow by forcing the energy gradient to approach zero. This results in a loss of transport capacity with resulting deposition of suspended sediment

(Thornton, 1990; US Army Corps of Engineers, 1997; Morris and Fan, 1998; Tarela and Menendez, 1999; Yu et al., 2000; Morgui et al., 1993).

Longitudinal deposition in reservoirs may be divided into three main zones (Morris and Fan, 1998). (See Figure 18)

1. Topset beds correspond to delta deposits of rapidly settling sediment. The downstream limit corresponds to the downstream limit of bed material transport in a reservoir.
2. Foreset deposits represent the face of the delta advancing into the reservoir and are differentiated from the topset beds by an increase in slope and decrease in grain size.
3. Bottomset beds consist of fine sediments that are deposited beyond the delta by turbidity currents or nonstratified flow.

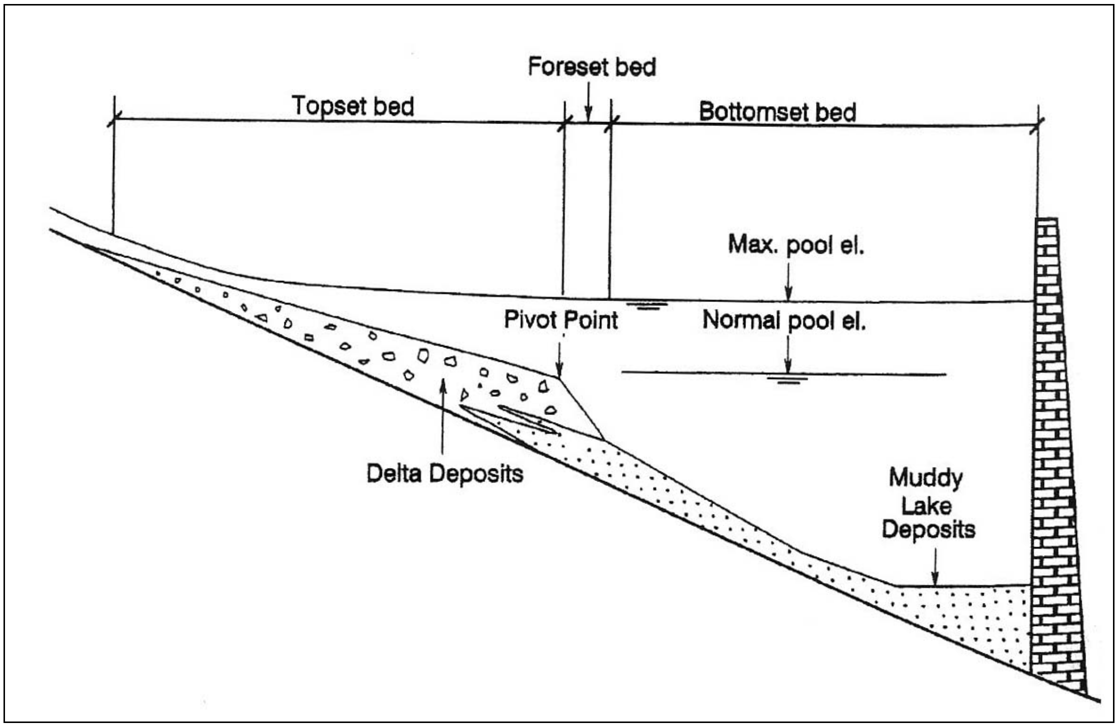


Figure 18. Generalized depositional zones in a reservoir (from Morris and Fan, 1998).

Where rivers enter the broad, open expanses of a lake or an estuary, the flow velocity, sediment load and particle size, vary both laterally and longitudinally, generally resulting in a fan-shaped delta (Sundborg 1967 in Thornton, 1990). The longitudinal dimension is the more important factor in reservoir delta formation. The river is generally confined to the old channel in the reservoir headwater so that the lateral dimension remains constant while the velocity, sediment load and particle size vary longitudinally (Thornton, 1990; Morris and Fan, 1998).

The longitudinal dimension is also important in sediment distributions within the reservoir. River inflow and its constituent sediment load generally follow the thalweg (river channel); therefore sediment deposition is initially greatest in the old channel (Thornton, 1990; Morris and Fan, 1998).

Longitudinal gradients in sediment deposition also reflect the occurrence of longitudinal size sorting of particulate matter (McHenry et al., 1982; Olness and Rausch, 1977; Pharo and Carmack, 1979, all in Thornton 1990). The larger heavier sands and coarse silts settle in the delta areas, and then as flow velocities and turbulence continue to diminish, additional particle size-sorting occurs along the longitudinal axis of the reservoir. The silts and coarse clays are the next particle size class to settle with the fine clays and colloidal material settling very slowly (Morris and Fan, 1998; Tarela and Menendez, 1999; US Army Corps of Engineers, 1997). In South Africa, the inorganic suspensoids are primarily fine clay particles (Allanson, 1995), explaining why so many of its reservoirs have high turbidity.

Stratified flow occurs frequently in reservoirs because of density differences caused by differences in temperature, dissolved solids, turbidity or combinations thereof. Because of the high densities imparted by high turbidity levels, turbidity currents may plunge and flow along the bottom of a reservoir, regardless of the temperature regime (Morris and Fan, 1998; De Cesare et al., 2001). Turbidity currents are often the governing process in reservoir sedimentation by transporting fine materials over long distances through the impoundment to the vicinity of the dam (De Cesare et al., 2001; Yu et al., 2000).

The zone where the inflowing turbid water entering a reservoir plunges beneath the clear water is called the plunge point or plunge line. In a narrow reservoir the plunging flow will form a line across the width of the reservoir, with turbid surface water upstream of the line and clear water downstream. As the turbidity current travels downstream, it will deposit the coarser part of its sediment along the bottom, filling the thalweg first and eventually producing a relatively flat layer of deposits. If enough sediment is deposited, the current will dissipate (Morris and Fan, 1998; Yu et al., 2000).

If the current reaches the dam, it will accumulate to form a submerged muddy lake. This occurs when the forward velocity of the current causes it to rise up against the face of the barrier and subsequently fall back to initiate formation of a muddy lake with a nearly horizontal profile upstream from the dam (Morris and Fan, 1998; Yu et al., 2000) (Figure 19).

Yu et al. (2000) drew attention to deposition behaviour in the quasi-homogenous flow region which they maintain is not usually mentioned in the literature, and without which the description of the deposition system is not complete. This is the non-stratified area, which occurs between the downstream end of the deposition delta and the plunge section. It is shorter in cases of smaller discharge and/or higher sediment concentration and longer in cases of larger discharge and/or lower sediment concentration (Figure 20). Hydraulic sorting of particles exists in this region, becoming less significant with distance downstream.

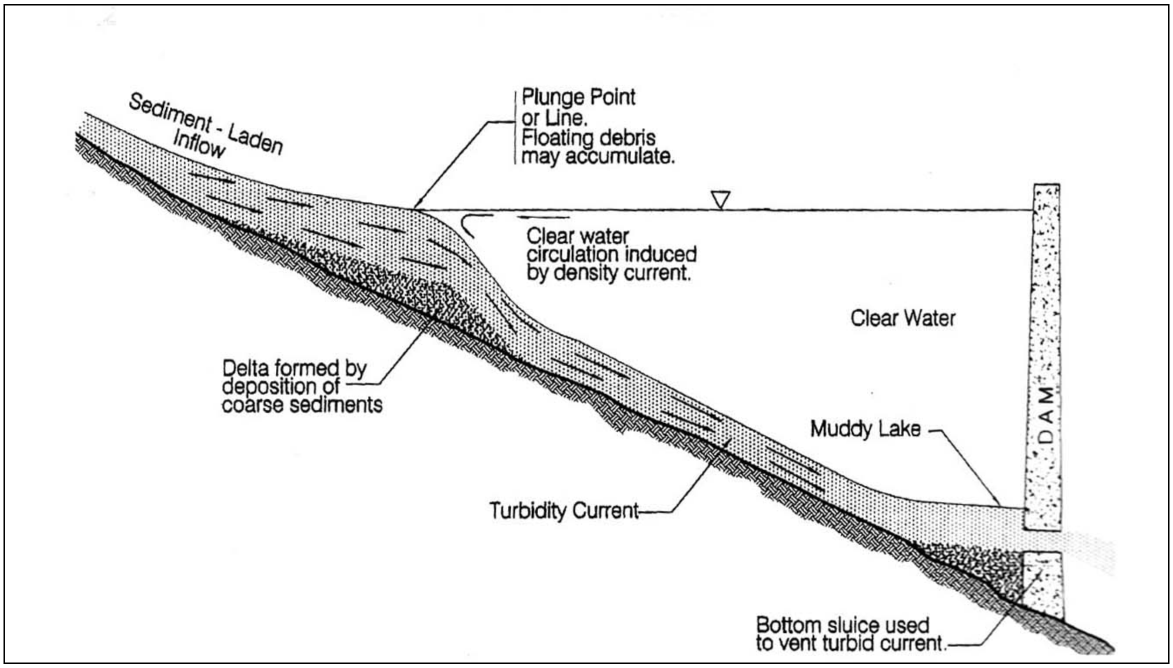


Figure 19. Schematic diagram of the passage of a turbidity current through a reservoir and being vented through a low-level outlet (after Morris and Fan, 1998).

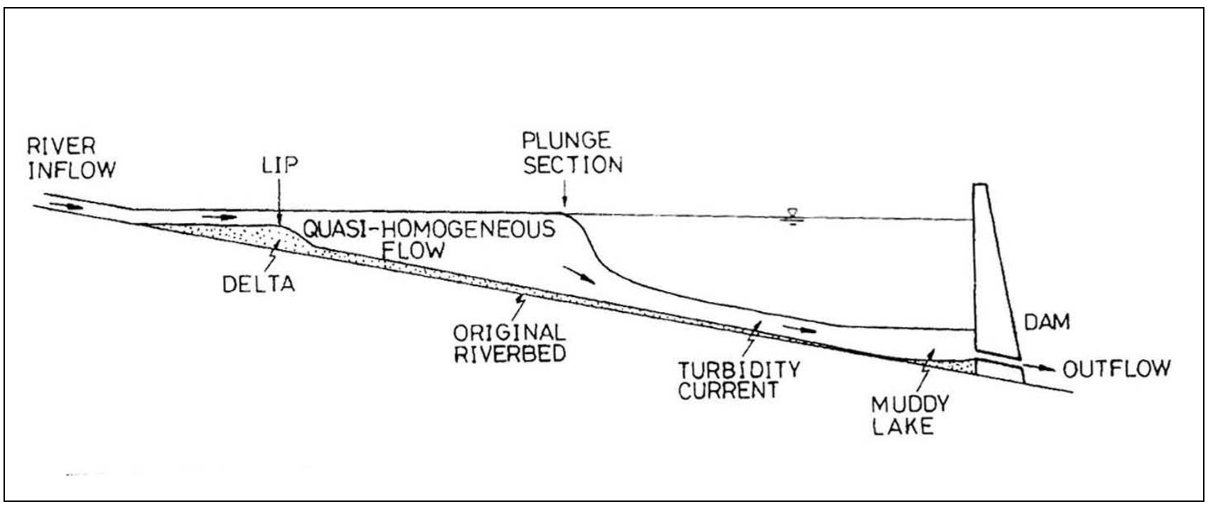


Figure 20. Schematic diagram of flow of sediment-laden water in a reservoir showing the position of the region of quasi-homogenous flow (after Yu et al., 2000).

3.1.1.4.2 Lateral deposition patterns

Sediment deposition is initially focused in the thalweg or deepest part of each cross section, creating deposits with a near horizontal surface regardless of the original channel cross section. Variations from this pattern can usually be explained by phenomena such as local sediment inflow from a tributary, scour at bends or channel erosion during drawdown and flushing (Morris and Fan, 1998).

3.1.1.4.3 Reservoir operation

Water control represents a major difference between lakes and reservoirs. It not only affects the zone and depth of withdrawal, but also sediment distributions within reservoir (Thornton, 1990; Morris and Fan, 1998).

Reservoirs used for flood control, hydroelectric power generation, irrigation and municipal and industrial water supply commonly exhibit significant fluctuations in water level. Lake Red Rock in the USA can increase from 10 to 25km in length, 10 to 22m in depth and 1.2 to 9.6 x $10^8 m^3$ in volume during flood control operations. Fluctuations such as these can influence sedimentation patterns by altering reservoir morphometry (length, depth or volume), mixing regime, water exchange between coves and the main pool, or water residence time.

Periods of elevated flow may result in delta formation and sedimentation upstream. Then, as the pool returns to normal seasonal levels, sediment deposited, sorted and processed upstream may be resuspended and transported downstream into the pool. Similar particle resuspension and transport may occur during withdrawal (Thornton, 1990). The rapid advancement of the delta deposits towards Tarbela Dam in Pakistan is caused by seasonal drawdown – one of the management strategies considered at that site is to limit drawdown to focus delta deposition further upstream, thereby delaying adverse impacts on the low-level intakes at the dam (Morris and Fan 1998).

3.1.1.5 Consequences of sedimentation in reservoirs

3.1.1.5.1 Storage loss:

The most direct consequence of siltation in reservoirs is the corresponding decrease in reservoir capacity (Straškraba and Tundisi, 1999; Tarela and Menendez, 1999; Morris and Fan, 1998). Sediment deposition in the reservoir pool reduces and eventually eliminates usable storage capacity, making the reservoir useless for either water supply or flood control (Morris and Fan, 1998; Yang et al., 2003).

There are nearly 45 000 large dams (over 15m tall) worldwide, the majority of which were constructed after 1950 (ICOLD 1988; Schnitter, 1994, in Palmieri et al., 2001), for a variety of purposes. Until now the common engineering practice has been to design and operate reservoirs to fill with sediment slowly. The consequences of sedimentation and project abandonment have been left to be taken care of by future generations. For many dams "this future has arrived, and in some cases earlier than anticipated." (Palmieri et al., 2001).

The majority of dams in the world are designed and operated to trap sediment continuously, without provisions for sustained long-term use. Neither current nor projected levels of population and economic activity can be sustained if storage reservoirs are lost to sedimentation. Demands on the services provided by dams are increasing, with reservoir-dependent societies ranging from technically advanced urban and agricultural systems in the western United States to village irrigators in India.

Sudden loss of the world's reservoir capacity would be a catastrophe of unprecedented magnitude, yet their gradual loss due to sedimentation receives little attention or corrective action (Morris and Fan, 1998).

The average rate of sediment accumulation in reservoirs can be expressed in volumetric units, but for comparing reservoirs of different sizes, it is convenient to express the rate of sedimentation as the percent of the original storage volume lost per year. Sedimentation rate can also be expressed in terms of the reservoir half-life – the time required to infill half the original capacity. Because the efficiency of sediment trapping declines as reservoir capacity is reduced, the half-life does not represent half the time required to lose all storage capacity (Lajczak, 1995). The ability of a storage reservoir to fulfil its design function will already be severely compromised by the time half the storage capacity has been lost (Dendy et al., 1973; Murthy, 1977). Thus the concept of half-life is a much more realistic indicator of the "life" of a conventional storage reservoir than the time required for complete sedimentation (Morris and Fan, 1998).

Crowder (1987) estimated the rate of storage loss in the United States as 0.22% per year. In other areas of the world, however sedimentation rates are often significantly higher. Van Den Wall Bake (1986) suggested that about half the reservoirs in Zimbabwe are losing capacity at the rate of more than 0.5% per year. Data examined from 16 reservoirs in Turkey (Gogus and Yalcinkaya, 1992) indicated that the annual rate of storage loss ranged from 0.20% to 2.40%, with a mean of 1.2%. Annual storage loss rates of 0.7% in Morocco and 2.3% in Tunisia are recorded by Abdelhadi (1995). Storage loss in China's 82000 reservoirs is reported at an average annual rate of 2.3%, the highest rate of loss of any country in the world (Zhou, 1993; Morris and Fan, 1998), while an annual sedimentation rate of 0.5% has been recorded in India by Varma et al., 1992 (All in Morris and Fan, 1998).

Loss of reservoir capacity can be extreme and may decrease the life span of reservoirs to just a few decades (Straškraba and Tundisi, 1999). Mahmood (1987) summarized the situation of worldwide reservoir sedimentation as follows: "…the average age of man-made storage reservoirs in the world is estimated to be around 22 years. The loss of capacity due to siltation is already being felt at a number of structures. It is entirely possible that, unless ingenious solutions are developed, we will lose the struggle to enhance the available water resources." (Palmieri et al., 2001).

China has 82 000 reservoirs which are losing storage capacity at an average annual rate of 2.3%, the highest rate of loss of any country in the world (Zhou, 1993; Morris and Fan, 1998).

3.1.1.5.2 Delta deposition:

The coarser portion of the inflowing sediment load is deposited where rivers enter reservoirs, forming delta deposits which not only deplete reservoir storage, but can also cause channel aggradations extending many kilometres upstream from the reservoir pool (Morris and Fan, 1998).

Channel aggradation can:

- **increase flooding of infrastructure, communities and agricultural lands on floodplains.**
- **increase groundwater levels, create waterlogging and soil salinization.**
- **reduce navigational clearance beneath bridges and submerge upstream intakes.**
- **further enhance sediment entrapment promoting further aggradation in cases where delta areas become vegetated.**
- **significantly increase water losses from the reservoir by evapotranspiration from large areas of phreatophytic vegetation in delta areas, especially in arid zones.**

In Lake Nassar located on the Nile River, a major problem is the continuous deposition of sediments, resulting in a bed-level increase, a decrease in lake capacity and the creation of a delta at the lake entrance that might eventually hinder navigation. Although the expansion of the delta is multidirectional, its advance towards the High Aswan Dam is driven by the frequent occurrence of high floods following extended drought periods (El-Manadely et al., 2002).

3.1.1.5.3 Mechanical Abrasion

In hydropower facilities, sediment coarser than 0.1mm greatly accelerates the erosive abrasion of turbine runners and Pelton wheel nozzles. This reduces power generation efficiency and requires removal of generating units for repair. Sediment sizes up to boulders can often pass through bottom outlets. Abrasion can damage gate seals, outlet works, aprons and spillways (Morris and Fan, 1998).

3.1.1.5.4 Intakes and outlets

Sediments can block or clog intakes and low-level outlets at dams, and clog or otherwise damage gates not designed for sediment passage (De Cesare et al., 2001; Boillat and De Cesare, 1994; Schleiss et al., 1996 in De Cesare et al., 2001). During extreme floods, deposition of many meters of material can occur in a few hours e.g. sediment and debris 17m deep were deposited in front of Valdesia Dam in the Dominican Republic during the passage of Hurricane David in 1979, clogging the power intakes for 6 months (Morris and Fan, 1998).

3.1.1.5.5 Ecology

Consideration of the physical accumulation of sediment in reservoirs alone would indicate its potential importance in ecosystem structure and function (Thornton, 1990). In addition, changes in sediment loading and sediment accumulation within the pool can dramatically alter reservoir ecology, affecting species composition and both recreational and subsistence fishing. (Thornton, 1990; Morris and Fan, 1998).

Sediment is not only the major water pollutant by weight and volume; it is also a major carrier and catalyst for pesticides, organic residues, nutrients and pathogenic organisms (Bachmann, 1980; Ogg et al., 1980; Sharpley et al., 1980, in Thornton, 1990). Fine silt and clay particles, in particular, have a high sorptive capacity for phosphorus, dissolved organic acids and other nutrients and contaminants. Contaminated sediments which have accumulated on the floor of the reservoir can be remobilized by sediment resuspension or removal (Morris and Fan, 1998).

In areas where sedimentation continues unabated, open water habitat will be transformed into wetland and eventually into dry land because of continued sediment deposition (Morris and Fan, 1998).

3.1.1.5.6 Energy loss:

Loss of storage eliminates the potential to capture high flows for subsequent energy generation (Morris and Fan, 1998).

3.1.1.5.7 Navigation:

Both commercial and recreational navigation can be severely impaired by sediment accumulation, especially in delta areas and in the vicinity of locks. Recreational access can be impaired as sediment accumulates at marinas and boat ramps (Morris and Fan, 1998).

3.1.1.5.8 Landslides and debris flows:

These can partially or completely fill reservoirs e.g. landslides totalling 177 Mm^3 collapsed into Sanmenxia Reservoir in China during the first 16 months of impoundment (Qian, 1982 in Morris and Fan, 1998). These events can also cause catastrophic daybreak-type floods. The 265m Vaiont thin arch dam in Italy was filled by a 240 Mm^3 rock slide in 1963, displacing water in the reservoir which overtopped the wall to a depth of 100m and created a flood which killed 2600 people downstream (Jansen 1983, in Morris and Fan, 1998).

3.1.1.5.9 Earthquake hazard:

Sediment deposits have a greater mass than water, and some research indicates that the presence of sediment against the dam wall can significantly increase the force of an earthquake shaking against the structure (Chen and Hung, 1993). Sediments accumulating near the dam may be liquified by earthquake shaking so that they flow towards, and bury bottom outlets, entering and clogging any conduits that are open. At the Tarbela dam on the Indus River in Pakistan, it was estimated that it would take 6 to 12 months to restore irrigation and hydropower service after an event of this nature (Lowe and Fox, 1995, in Morris and Fan, 1998).

3.1.1.5.9 Air pollution:

In seasonally empty irrigation reservoirs, desiccated deposits of fine sediment can be eroded and transported by wind, creating a nuisance and health hazard to nearby communities (Danielevsky, 1993; Tolouie, 1993, in Morris and Fan, 1998), especially where toxic pollutants such as heavy metals are sediment bound.

3.1.1.6 Management of sedimentation in reservoirs

Conversion of sedimenting reservoirs into sustainable resources, which generate long-term benefits, requires fundamental changes to the way they are designed and operated. It requires that the prevailing concept of a reservoir life limited by sedimentation be replaced by a concept of managing both water and sediment to sustain reservoir function (Morris and Fan, 1998).

The cost and applicability of each management strategy will vary from one site to another and also as a function of sediment accumulation. However, even the largest reservoirs will eventually be reduced to small reservoirs by sedimentation and sooner or later will require sediment management (Morris and Fan, 1998).

The actual choice of the most convenient strategy is a complex process involving hydrology, hydrogeology, morphology and dam engineering. The results will always be site specific and no standardization is possible apart from the basic principles on which the solution is based (Palmieri et al., 2001).

3.1.1.6.1 Reduction of sediment inflow

Sediment delivery to a reservoir can be reduced by techniques such as soil conservation measures (erosion control, reforestation or revegetation), and upstream sediment trapping (Morris and Fan, 1998; Palmieri et al., 2001).

3.1.1.6.2 Routing of sediments

Some or all of the inflowing sediment may be hydraulically routed beyond the storage pool by various techniques (Morris and Fan, 1998; Palmieri et al., 2001). The sediment may be passed through or around storage or intake areas while minimizing objectionable deposition (Morris and Fan, 1998). Generally, routing is environmentally the most benign sediment management strategy, while flushing is potentially the most damaging. This is because sediment routing partially preserves the natural sediment transport characteristics of the river, whereas flushing usually changes them dramatically. Unlike sediment routing which attempts to prevent deposition during flood events, flushing uses drawdown and emptying to scour and release sediment after it has been deposited.

Seasonal drawdown

The reservoir is partially or completely emptied during the flood season at a predetermined time each year. This serves to increase flow velocity with a corresponding decrease in retention time and sediment trapping. Under appropriate conditions, a sediment input-output balance can be achieved with year-round uninterrupted reservoir operation by using partial drawdown. An example of partial drawdown is the Three Gorges Project on the Yangtze River in China (Morris and Fan, 1998).

Sluicing

This is an operational technique aimed at reducing the trap efficiency of the reservoir by releasing most of the sediment load with the flow through the dam before the sediment particles settle. This is usually accomplished by operating the reservoir at a lower level during the flood season in order to maintain sufficient sediment transport capacity through the reservoir. After the flood season the pool level is raised to store relatively clear water. A disadvantage is that storage capacity is limited to a fraction of the annual runoff and reservoir operation is limited to part of the year (Palmieri et al., 2001), clearly not an optimal option in arid or semi-arid areas...

Sediment bypass

When topographic conditions are favourable, a large capacity channel or tunnel can be constructed to bypass sediment-laden flow around an instream storage reservoir. Nagle Dam in South Africa uses this method (Annandale, 1987, in Morris and Fan, 1998) (See Figure 21).

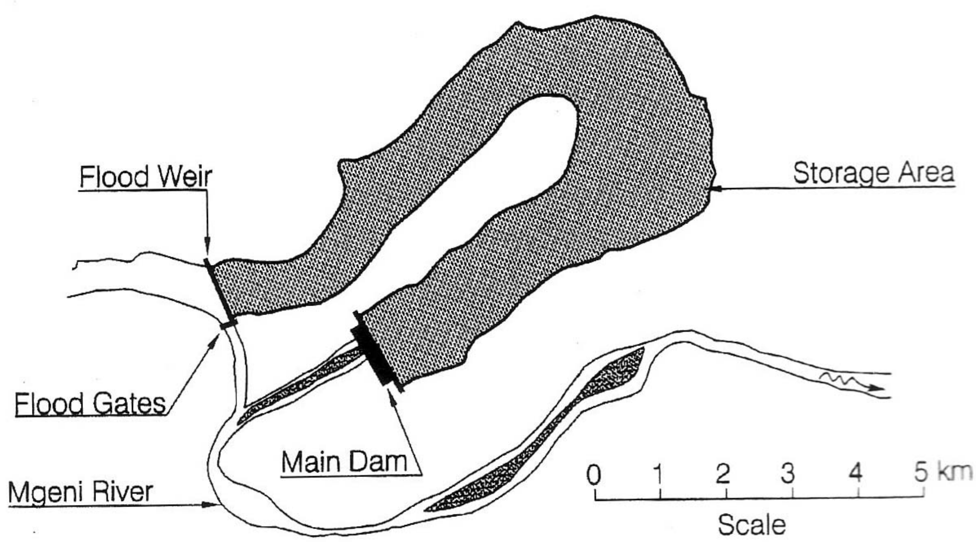


Figure 21. Sediment bypass configuration at Nagle Reservoir in South Africa (after Annandale, 1987).

Offstream reservoirs:

Since inflow can be controlled at the diversion point, an off-stream reservoir can be prevented from receiving sediment-laden water in various ways. An advantage is that downstream transport of bed material can be maintained. Of

the 33 major reservoirs in Taiwan, 9 are designed to bypass sediment (Morris and Fan, 1998) (See Figure 22).

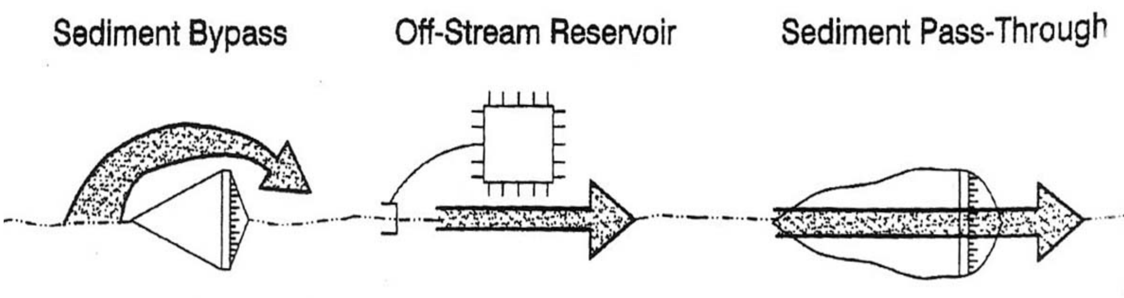


Figure 22. Methods of sediment routing, generally the most environmentally benign management strategy (after Morris and Fan, 1998).

Venting of turbid density currents:

Turbidity currents can be vented from reservoirs by opening a low-level outlet at the dam. In some reservoirs it has been possible to release more than half the total sediment load in an individual flood by venting the turbidity current. Successful venting depends on properly located low-level outlets that are opened in time to release the current, using a discharge rate that matches the turbidity current flow. However, the behaviour of turbid underflows in reservoirs is not easy to quantify (Morris and Fan, 1998).

3.1.1.6.3 Removal of sediments

Deposited sediments may be removed periodically (Morris and Fan, 1998; Palmieri et al., 2001) by various means:

Hydraulic flushing

Flow velocities in a reservoir are increased to such an extent that deposited sediments are remobilized and transported through bottom outlets (Shen, 1999, a review). Unlike sediment routing which attempts to prevent deposition during flood events, flushing uses drawdown and emptying to scour and release sediment after it has been deposited. However, many reservoirs cannot be removed from service for flushing and in many cases flushing cannot restore or maintain the original reservoir volume. Flushing also releases large amounts of sediment downstream creating potentially serious problems including interference with water intakes, increased sediment loading on downstream reservoirs and adverse impacts on fisheries, the environment and recreational uses. The Cachi Hydropower Reservoir in Costa Rica, built in 1966 has been emptied for flushing 14 times. It is considered to demonstrate the effectiveness of regular flushing for sediment removal (Morris and Fan, 1998). (Figure 23).

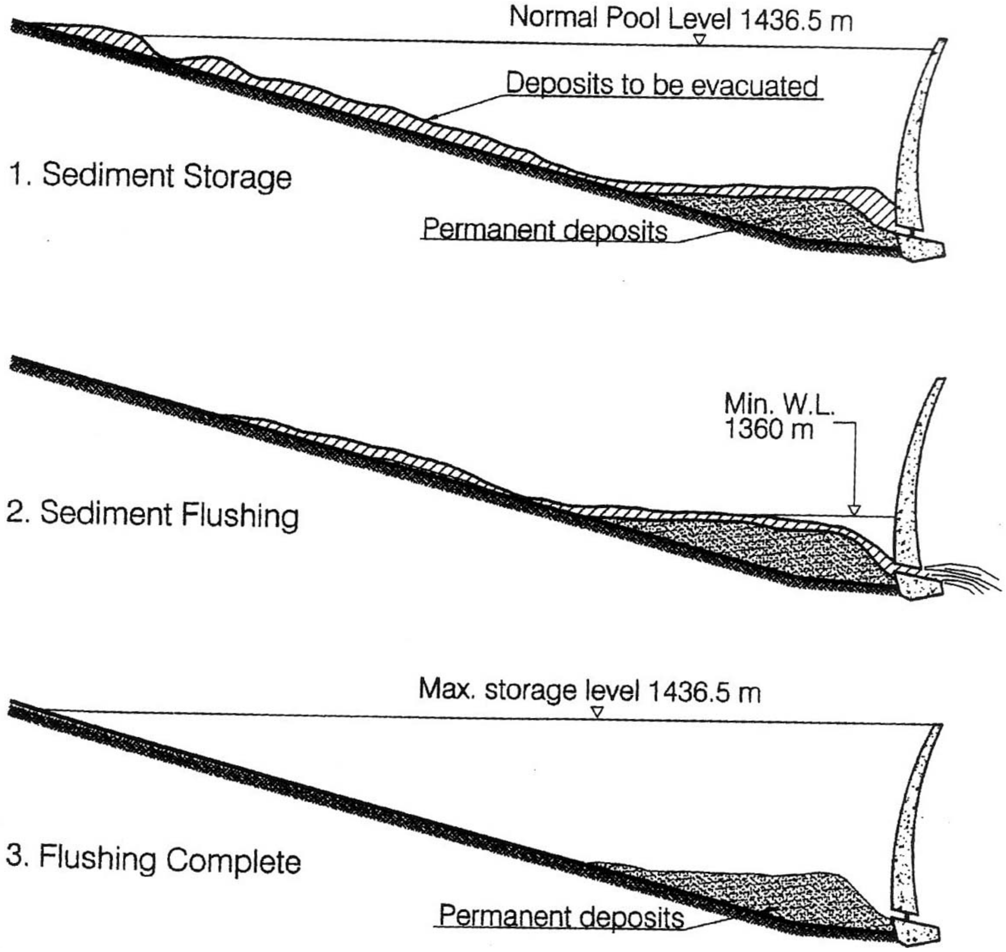


Figure 23. Pressure flushing sequence in reservoir management (after Ullmann, 1970).

Hydraulic dredging and dry excavation

To retain capacity, dredging must continue for as long as the reservoir is to remain in service, creating recurrent costs and long-term disposal problems. For example, if Lake Powell on the Colorado River were fully sedimented, it would cost $83 billion at current rates to fully restore its original capacity, assuming that a suitable disposal site could be found for 33km^3 of sediment (Morris and Fan, 1998; Yang et al., 2003).

Hydrosuction sediment removal systems (HSRS)

This system holds promise as a method of removing deposited or incoming sediments from reservoirs using the energy represented by the difference between water levels upstream and downstream from the dam (Hotchkiss and Huang, 1995). There are two types of hydrosuction sediment removal, (i) hydrosuction dredging in which deposited sediment is dredged and transported to either a downstream receiving stream or to a holding treatment basin

(Figure 24a), and (ii) hydrosuction bypassing in which incoming sediment is transported without deposition, past the dam to the downstream receiving stream (Figure 24b).

Conventional methods of hydraulic dredging use a mechanical pump to supply the energy to remove deposited sediment. Hydrosuction dredging uses the hydraulic head represented by the different levels. No external energy is required to transport the sediment through pipelines from the intake point to the point of discharge. Two variations of the method have been used: bottom discharge when the pipeline passes through the low level outlets, and siphon dredging when the discharge pipeline is passed over the top of the wall. Both methods may employ a floating barge which moves the pipeline inlet around the reservoir to access a larger area.

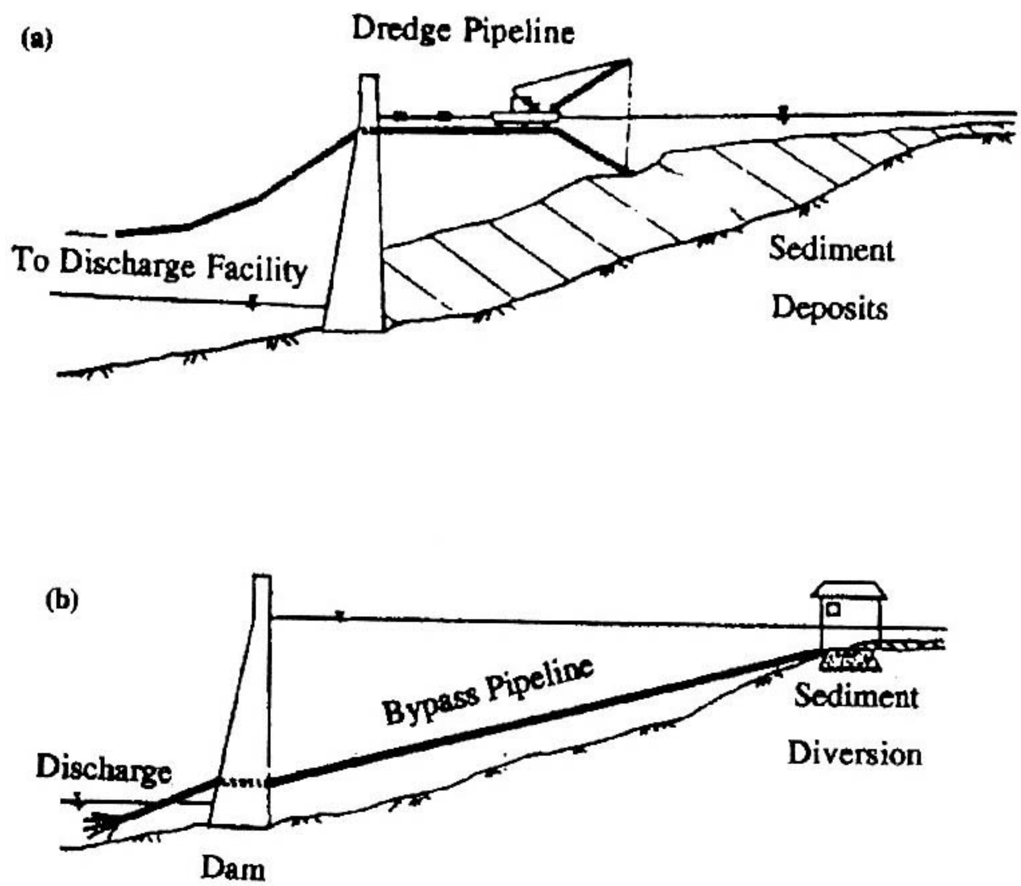


Figure 24. Hydrosuction sediment removal systems (HSRS). (a) Hydrosuction dredging; (b) Hydrosuction bypassing (after Eftekharzadeh, 1991, in Hotchkiss and Huang, 1995).

Hydrosuction bypassing would employ the same principle but would feature a permanent inlet station upstream from the reservoir deposition zone to collect the sediment into a pipeline. Presently, this is a theoretical concept since no such system is known to have been constructed yet.

3.1.1.6.4 Provision of large storage volume

Reservoirs may be considered sustainable if the storage volume provided exceeds the volume of the sediment supply in the watershed. The required sediment storage may be included within the reservoir pool or in one or more upstream impoundments (Morris and Fan, 1998). But this option too appears theoretical rather than practical.

3.1.1.6.5 Sediment placement

Sediment deposition may be focused in areas where its subsequent removal is facilitated or where it minimizes interference with reservoir operation. Intakes and other facilities should be configured to minimize interference from transported or deposited sediments.

3.1.1.6.6 Sediment Management Tools

Modelling approaches

Many modelling studies have been carried out, and various computer models are available to assist in the prediction of sedimentation in reservoirs, its measurement, its management and its economics (Tarela and Menendez, 1999; Goodwill et al., 1995; Chang et al., 1996; Nicklow and Mays, 2000; De Cesare et al., 2001; Salas and Shin, 1999; Ziegler and Nisbet, 1995; Nicklow and Mays, 2001; Palmieri et al., 2001).

Empirical remote sensing

The conventional technique of reservoir sediment deposition quantification using hydrographic surveys and the inflow-outflow methods is cumbersome, costly and time-consuming. With the availability of high-resolution satellite data, capacity surveys of reservoirs by remote sensing are gaining recognition and acceptance. In conjunction with GIS, the temporal change in waterspread area can be analysed to evaluate the sediment deposition pattern in a reservoir (Goel et al., 2002; Jain and Singh, 2002). There are limitations to this method, however. For example, remote sensing techniques provide information on reservoir capacities only within the zone of water level fluctuation.

3.1.1.7 Turbidity of reservoirs

The most obvious effect of suspensoids in water is to increase turbidity (Allanson, 1995). Waters in areas with fine silt and clay tend to have persistent turbidity that lasts long after the eroding waters have reached their destination. Sedimentation rates are low and ordinary turbulence in the water is enough to keep these fine particles suspended for months (Straškraba and Tundisi, 1999). In South Africa, turbid waters occur naturally. It is when the suspensoid load increases beyond the natural limit that significant impairment of the aquatic ecosystem occurs (Allanson, 1995).

Suspended clay, an attribute of reservoirs situated in certain geographic regions has numerous physical, chemical and biological impacts on the limnological processes of these ecosystems (Lind and Davalos-Lind, 1999). The most obvious effect of suspended clay in reservoirs is reduction of phytoplankton photosynthesis due to competition for light. This may be desirable in reservoirs with high nutrient loading, where eutrophication is a potential problem.

This suspended material is a significant component of aquatic ecosystems as most interactions between solutes and suspensoids occur at the surface of the particles. Their surface areas, which relate to particle size, is thus of particular importance (Figure 25). Organic and inorganic compounds adsorbed onto the surface of the particles include plant nutrients, pesticides, heavy metals and organic molecules, all of which are of concern to the water manager. Adsorbed nitrate and ammonium are weakly bonded and are readily available for algal growth. Phosphate complexes form phosphate pools of varying availability to phytoplankton (Allanson, 1995).

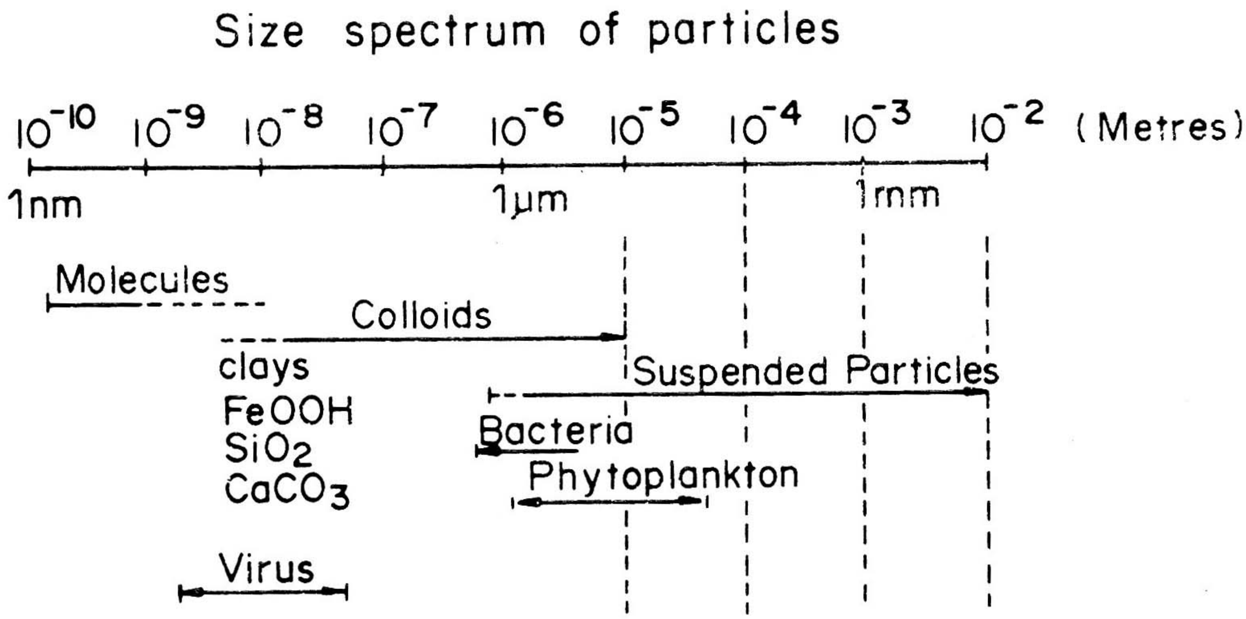


Figure 25. Size spectrum of particles dissolved or suspended in natural waters expressed as particle diameter in metres[12] (after Allanson, 1995, from Bruton, 1984).

Marzolf (1984) in Lind and Davalos-Lind, 1999, drew attention to the adsorptive properties of clays as competitor with phytoplankton for nutrients and with bacterioplankton for organic substrate. One of the best-studied adsorptive associations with clay is that of phosphorus because of its importance as a growth-regulating nutrient. Clays can deprive algae of phosphorus by competing in the water column, and ultimately by carrying it out of the photic zone in sedimentation. Temperature, pH, ionic strength of the solution and the presence of adsorbed organics on the clay surface, each affect the phosphorus adsorption kinetics. Nevertheless, clays can also provide an accessible reservoir of phosphorus to phytoplankton.

Suspended clay can adsorb large quantities of dissolved organic materials and concentrate them on the clay surface.

Based on studies of turbid sub-tropical reservoirs (Lind and Davalos, 1990), it was found that clays often bind with dissolved organic carbon to form large amorphous aggregates (up to 65µm). The formation of such aggregates varies greatly and what causes the differences in unknown. In Waco Reservoir, Texas, clays remain independent, while in nearby Lake Brazos (a reservoir), aggregates that are colonized by bacteria are formed.

Bacterioplankton are possibly the most intimately related to suspended clays of any of the limnetic communities (Lind and Davalos-Lind, 1999). This is in part because much dissolved organic matter (DOM), which provides bacterioplankton nutrition is adsorbed to the clay. Lind and Davalos (1990) proposed that the organic component of interest to microbial ecologists in clay turbid waters should be the particulate and not the dissolved fraction. This is not to suggest that phytoplankton photosynthesis is not important to bacterioplankton. These authors found that the highest rates of bacterioplankton production in Waco Reservoir (Texas) occurred at the transition zone of the reservoir where turbidity was intermediate and algal biomass was greatest.

Clay adsorbed organic matter is an important food source for microcrustaceans and copepods (Arruda and Foulk, 1983; Gliwicz, 1986, both in Lind and Davalos-Lind, 1999). This use of ingested clay-organic matter is seen as compensatory for the loss of phytoplankton due to light limitation. Although the quantity and quality of organic matter available by this means varies greatly, depending on both clay and environmental factors, it appears that for many reservoirs the ambient clay turbidity is sufficient to provide food at least in excess of the starvation level.

As a consequence of the clay facilitation of bacterial production, Lind et al., 1997, proposed that greater microheterotrophy can replace part of the lost food web base caused by light limitation of photosynthesis (Figure 26). This food web incorporates the role of suspended clay with the shift from photosynthetic autotrophy to microheterotrophy. Features of this model include the concentration of DOM on the clay with the formation of COBA (clay-organic-bacteria-aggregates) that increases bacterial use efficiency and the use of the COBA by macrozooplankton thus by-passing the microzooplankton link of nanoflagellates and ciliates. It also suggests the possibility of direct fish consumption of the COBA with even greater trophic level efficiency. Lind et al., 1994, in Lind and Davalos-Lind, 1999, used this model to explain the relatively large fishery on Lake Chapala (Mexico), which was in excess of that predicted by any of the usual photosynthesis or algal biomass based fishery models.

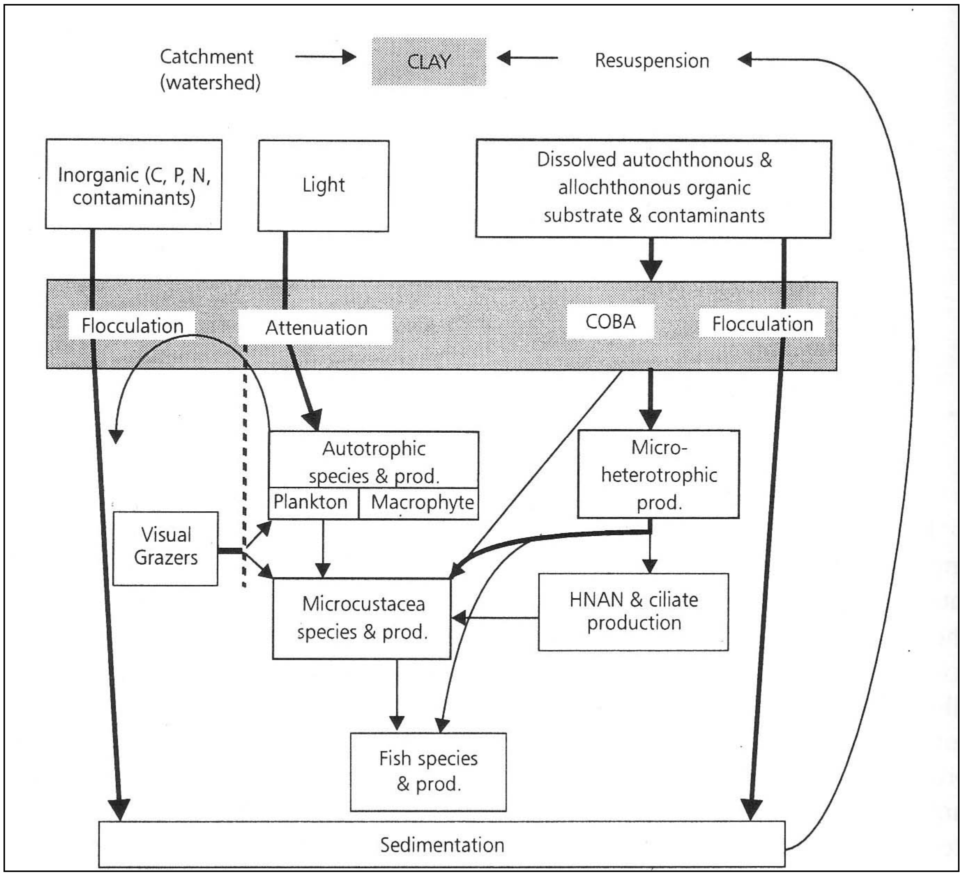


Figure 26. A portrayal of some clay–mediated processes in lakes and reservoirs with particular reference to production and trophic relations. (After Lind and Davalos-Lind, 1999). Heterotrophic nanoflagellates are designated as HNAN.

3.1.1.8 Sustainable use of reservoirs with respect to sediments

Concepts of long-term sustainability and sediment management have rarely been incorporated into reservoir projects. The design strategy for China's Three Gorges Project represents a significant departure in this respect. Sustained use was established as an engineering criterion, and the project incorporated the structural and operational features required to achieve a long-term sediment balance with no loss in project benefits from the early planning stage. (Alternatively it could be stated that project benefits were limited to those that could be sustained, as opposed to focusing on high initial benefits alone). A balance between sediment inflow and outflow is expected to be achieved after about 100 years of operation (Morris and Fan, 1998; Palmieri et al., 2001).

The sustainability criteria suggested for new reservoirs is to design for a minimum of 1000 years of operation. This may be accomplished by balancing sediment inflow and outflow, by providing 1000 years of sediment storage capacity, or some combination thereof.

Many benefits from existing reservoirs may not be sustainable as sedimentation progresses. For existing reservoirs, sustainable sediment management should seek to balance sediment inflow and outflow across the impounded reach while maximizing long-term benefits.

This may involve strategies to minimize sediment inflow, enhance sediment release, or a combination of both.

However, sustainable sediment management encompasses the entire fluvial sediment system. It is not achieved without cost. At a minimum it involves better information and improved management. It may also involve large operational and capital costs for watershed management, construction, and temporary removal of the dam from service or the release of increased volumes of water downstream. Nevertheless continued operation with reduced net benefits is preferred to project abandonment (Morris and Fan, 1998).

3.1.2 Nutrient Pollution - Eutrophication

3.1.2.1 Introduction

Eutrophication - the enrichment of water bodies with plant nutrients, typically nitrogen and phosphorus, and the subsequent effects on water quality and biological structure and function – is a process rather than a state (Rast and Thornton, 1996). It represents the ageing process of lakes whereby external sources of nutrients and organic matter of terrestrial origin accumulate in a lake basin, gradually decreasing the depth and ultimately filling it completely. Under natural conditions (without human interference) this process takes place over geological time. However, "cultural eutrophication" can accelerate the enrichment process to a few decades, especially in the case of reservoirs, which usually have disproportionally large catchment areas.

Plant nutrients which enter the water body as solutes or bound to organic and inorganic particles result in enhanced growth and increased abundance of aquatic plants, both algae and macrophytes, often resulting in reduction of water quality (UNEP, 1999; Baker et al., 2001).

Eutrophication of inland waters ranks as one of the most widespread environmental problems and has many significant and negative ecological, health and economic problems. In both developed and developing countries everywhere in the world, with a few local exceptions, it is recognized to be an acute problem (UNEP, 1999).

3.1.2.2 Factors that contribute to eutrophication

3.1.2.2.1 The reservoir ecosystem

Many different limnological aspects are important to the understanding of eutrophication and thus its management. These were discussed in detail in Section 2 and will therefore be only briefly revisited in this section.

Physical factors

Solar radiation and its dissipation as heat are of fundamental importance in the reservoir ecosystem and have a great influence on the development of eutrophication if other conditions are favourable. Light is essential for photosynthesis, while temperature differences drive the vertical stratification process.

In reservoirs, transport processes, either related to heat and momentum exchanges at the water surface, or to flow, play a significant role in eutrophication. Nutrient transport, distribution and circulation are influenced by water movements, while hydraulic residence time influences the time available for plant growth.

Chemical factors

Chemical conditions in reservoirs are influenced by biogeochemical and hydrological processes in the watershed, as well as by ecological and chemical processes within the water and sediments of the waterbody. Eutrophication, by definition is the result of the increase in the amount of available phosphorus and nitrogen and other nutrients in the water, but dissolved oxygen, organic and inorganic carbon are also essential elements in the many and complex interactions which occur. The role of sediments in the chemistry of reservoirs is vitally important and the exchange of nutrients between the sediments and the water column plays a very significant part in the eutrophication process.

Biotic factors

Biotic communities in reservoirs can be divided into those in the pelagic zone, those in the profundal zone and those in the littoral zone. Contributions to, and the effects of eutrophication vary from zone to zone, and among different biota. Pelagic organisms include fish, zooplankton, phytoplankton and free-living and attached bacteria. Organisms in the profundal zone include a wide variety of invertebrates and microbes, the abundance and species composition of which is strongly influenced by the presence or absence of dissolved oxygen. Emergent, floating and submerged macrophytes occur in the littoral zone of reservoirs when conditions are favourable. They provide a habitat for attached algae and bacteria (periphyton) and for free swimming fish and invertebrates.

Food webs

Interactions among trophic levels can modulate the effects of eutrophication. Factors such as predation and grazing pressure, and their effects on community structure and size range within populations, are significant.

3.1.2.2.2 External loading

The role of the **watershed** in the eutrophication of reservoirs is the subject of much research, and is of great significance with respect to its management and control. However, in spite of its considerable importance, it is not within the scope of this report.

3.1.2.2.3 Internal loading

Internal recycling of nitrogen and phosphorus from sediments of reservoirs can sustain eutrophic conditions for long periods after external loading is reduced (Horn, 2003).

Ammonium is produced by the decomposition of organic matter in reservoir sediments. A portion of the ammonium may be nitrified to nitrate, which may be partially nitrified to nitrous oxide and gaseous nitrogen depending on the oxidation-reduction status of the sediments. High concentrations of dissolved organic nitrogen in the surface sediments typically diffuse into the overlying water.

3.1.2.2.4 Limiting factors

Light and nutrients, respectively functioning as rate- and yield-limiting factors, determine the growth of algae and aquatic macrophytes in reservoirs. These resources are therefore considered to be limiting factors when they are in "short supply." Although one factor seldom consistently limits plant growth under the varying and interactive conditions of the aquatic ecosystem, dominant control at a particular time and in a particular place can often be attributed to a single factor (UNEP, 1999; Sherman et al., 1998; Hernandez-Aviles et al., 2001).

Light availability plays a key role in the growth of submerged macrophytes which are usually rooted and can therefore access the sediments for nutrients. Turbidity caused by suspended sediments or algal blooms, or shading by floating plants would thus be instrumental in limiting light. Floating plants, on the other hand, are easily able to receive light, but depend on nutrients in the water.

Phytoplankton abundance and species composition changes with different ratios of nutrients in the water and underwater light conditions. Some cyanobacteria can regulate their buoyancy and often become dominant under turbid conditions. Different ratios of, for example, nitrogen to phosphorus (Ahn et al., 2002) or phosphorus to silicon can alter competitive relationships among some algal species.

3.1.2.3 Effects of Eutrophication

3.1.2.3.1 Algal blooms

A pervasive result of nutrient enrichment of reservoirs is increased algal growth. Some species of green algae (Chlorophyta), cyanobacteria and dinoflagellates can cause off-flavours and odours in water (Baker et al., 2001; Wang et al., 2002; van Ginkel et al., 2001; Izaguirre et al., 1999; Yamada et al., 1998; Fukuju et al., 1998) and fish, and/or clog filters in water treatment and industrial facilities (UNEP, 1999; van Ginkel et al., 2001). Some diatom species e.g. *Asterionella formosa* are known to interfere with the treatment of drinking water supplies (Bertrand et al., 2003).

Cyanobacteria are an especially problematic group. They can form unsightly surface scums due to the fact that the buoyant cells of some species float to the surface (UNEP, 1999;

Olsen et al., 2000; Wang et al., 2002), generating “hyperscums” in hypertrophic reservoirs such as Hartbeespoort (Robarts and Zohary, 1984; Zohary and Robarts, 1989). This buoyancy also causes water quality problems associated with the local accumulation of algae downwind (Piyasiri, 2001), causing attenuation of light and oxygen depletion (Olsen et al., 2000). Fish mortalities may occur on die-off of some species of cyanobacteria. (UNEP, 1999).

Dinoflagellates often form blooms (van Ginkel et al., 2001) and develop red tides that can include toxic strains (Yamada et al., 1998; Fukuju et al., 1998).

A by-product of dense algal blooms is high concentrations of dissolved organic carbon. When water with high DOC is chlorinated, potentially carcinogenic and mutagenic trihalomethanes are formed.

3.1.2.3.2 Algal toxins

Freshwater toxins are produced almost exclusively by cyanobacteria (UNEP, 1999). The most common genera of potentially toxic cyanobacteria in water storage reservoirs are *Microcystis*, *Anabaena* and *Cylindrospermopsis* (Vieira et al., 2003). Toxic cyanobacterial blooms in reservoirs pose a potential health risk to humans, livestock and other biota (Chellappa and Costa, 2003; Caleffi et al., 1994), whether the water is used for irrigation, recreation or potable purposes (Wang et al., 2002). In 1988, 88 deaths in 42 days in Brazil were attributed to cyanobacterial toxins in the drinking water from the Itaparica Dam, while, in China, high incidence of primary liver cancer was attributed to cyanobacterial toxin contaminated drinking water (Wang et al., 2002).

Various genera and species produce different toxic compounds generally classified as neurotoxins, hepatotoxins, cytotoxins and endotoxins. Neurotoxins are highly toxic but their degradation in the water is rapid. Hepatotoxin removal from reservoirs containing toxic cyanobacteria is difficult because some forms are stable and resistant to chemical hydrolysis or oxidation and can easily pass through traditional water treatment processes (Wang et al., 2002). They may persist for months or even years, and remain potent even after boiling (McGregor and Fabbro, 2000).

The majority of the approximately 50 known microcystins have been isolated from species and strains of *Microcystis*. They are extremely toxic and can cause death in laboratory mice in 1-3 hours. Microcystins are hepatotoxins that can accumulate in vertebrates and invertebrates, including fish, mussels and zooplankton. Because of their potential for bioaccumulation and their slow rate of degradation, microcystins have considerable potential for toxicity in aquatic food chains.

The alkaloid cylindrospermopsin is considered a cytotoxin because it attacks cells throughout the body. Gastroenteritis, renal malfunction and hepatitis have been observed in animals and humans poisoned by water containing cylindrospermopsin producing cyanobacteria (McGregor and Fabbro, 2000).

Several environmental factors i.e. light, temperature, nutrient concentration or pH can influence the degree of toxin production but the genetic structure of the bloom seems to be the major factor determining its toxicity. Typically about half all blooms tested are toxic and the occurrence of toxic blooms is becoming more frequent. (Hummert et al. (2001) *Bleiloch Reservoir, Germany;* Oudra et al. (2002) *Lalla Takerkoust Reservoir, Morocco*; Vieira et al. (2003) *Utinga Reservoir, Brazil*, where microcystins were recorded from *Radiocystis fernandoi* for the first time; Blaha and Marsakek (2003) *Czech Republic*; van Ginkel et al. (2000) *South Africa*; McGregor and Fabbro (2000): Baker et al., (2001) *Australia*; Pizzolon et al. (1999) *Argentina;* Wang et al. (2002) *Bang Phra Reservoir, Thailand;* Panosso et al. (2003) *Funil Reservoir, Brazil*).

Toxin content is highest within actively growing cyanobacterial cells and release to the water appears to occur during cell senescence. Since algal scums commonly accumulate against dam walls in proximity to water off-take sites, this problem assumes especially great significance in reservoirs.

3.1.2.3.3   Growth of macrophytes

Dense mats of floating aquatic plants e.g. *Eichhornia crassipes* (Magadza, 2003, Lake Chivero, Zimbabwe; Fernandes and Crisman, 1994, Lake Paranoa, Brazil), *Salvinia molesta* (Moreau, 1997; Mhlanga, 2001, Lake Kariba) and Pistia *stratiotes* (Sommaruga et al., 1993, Cisne Reservoir, Uruguay) develop and often cover large areas, especially in wind sheltered areas within reservoirs. These mats prevent light from reaching submerged macrophytes and phytoplankton.

They produce large quantities of organic detritus that can lead to anoxia and emission of gases such as methane and hydrogen sulphide. The material derived from these plants is usually of low nutritional value and not usually an important component of the diet of zooplankton or fish.

Accumulations of aquatic macrophytes may restrict access for fishing or recreational areas and can block irrigation or navigation channels as well as hydroelectric power plant intakes.

3.1.2.3.4   Anoxia

A by-product of increase in abundance of algae or macrophytes is generation of more organic matter. As this decomposes in the water column or in the sediments, the concentration of dissolved oxygen decreases due to bacterial activity. This can lead to deoxygenation of the sediments and deeper waters, threatening the survival of fish and invertebrates, especially benthic organisms.

Under anoxic conditions, ammonia, iron, manganese and hydrogen sulphide concentrations can rise to levels deleterious to the biota and to hydroelectric power facilities. Phosphate and ammonium are released into the water from anoxic sediments, further enriching the reservoir.

3.1.2.3.5 Species changes

Shifts in the abundance and species composition of aquatic organisms are frequently a consequence of eutrophication. Algal blooms or floating macrophytes reduce light penetration, reducing or eliminating submerged macrophytes. Changes in food quality associated with shifts in algal or macrophyte composition, and decreases in oxygen concentration often alter the species composition of fish.

3.1.2.3.5 Hypereutrophy

Hypereutrophic lakes and reservoirs represent the ultimate stage of the eutrophication process. Unlike other eutrophic systems where reductions in nutrient loading may reverse the process, these measures are seldom feasible in hypereutrophic conditions. They usually have uncontrollable nutrient inputs, from diffuse and non-point sources.

3.1.2.3.6 Enhanced internal recycling of nutrients

Internal recycling of nutrients into a reservoir from the sediments can be significant and can exceed external loading. Once a system has reached a eutrophic or hypereutrophic state its dependence on external sources of nutrients is diminished.

3.1.2.3.7 Elevated nitrate concentrations

High concentrations of nitrate resulting from nitrate-rich runoff or nitrification of ammonium within the waterbody can cause public health problems. Methylhaemoglobinaemia in infants results from nitrate levels above 10mg/l in drinking water. By interfering with the oxygen carrying capacity of the blood, high nitrate levels can lead to life-threatening oxygen deficiency.

3.1.2.3.8 Increased fish yields

Yields of fish tend to increase as primary productivity increases - up to a cut-off level, when oxygen depletion or elevated pH and ammonia levels impose limits to, or even decrease fish yields. Assuming that the fish whose yields are improved are edible and marketable, the increase in primary productivity associated with nutrient enrichment can have a positive result. Partly as a consequence of this response, distinct cultural differences may be evoked by eutrophication. Developing nations show greater tolerance to the consequences of eutrophication than industrialized societies. However, it is relevant to stress that the human health impacts of eutrophication are commonly greater for developing than developed societies, for a variety of reasons.

3.1.2.4 Assessment of Eutrophication

3.1.2.4.1 Limiting nutrients

The following factors determine the relative role of phosphorus or nitrogen in limiting the productivity of phytoplankton in reservoirs.

the ratio of nitrogen to phosphorus in hydrologic and meteorologic inputs, and in the vertical fluxes of dissolved nutrients within the reservoir.

- preferential losses from the euphotic zone by processes such as denitrification, sorption of phosphorus to particles and differential settling of particles with different nitrogen to phosphorus ratios.
- the relative magnitude of the external supply to internal recycling and redistribution.
- the contribution from nitrogen fixation.

Unfortunately, coordinated measurements of these processes have been undertaken in very few lakes and reservoirs. Instead, inferences from several indicators of nutrient limitation must be made.

3.1.2.4.2 Evaluatory modelling approaches

Many empirical models have been developed to predict the concentration of total phosphorus in lakes and reservoirs as a function of annual phosphorus loading. Extensions of such models offer predictions of chlorophyll concentrations in phytoplankton, of Secchi disc visibility or of dissolved oxygen levels. However, these models often have very high uncertainty levels and usually require modification for different regions.

Dynamic simulation models incorporate mathematical descriptions of physical, chemical and biological processes in lakes and reservoirs (Olsen et al., 2000, *Eglwys Reservoir, Wales*; Easthope and Howard, 1999, Howard and Easthope, 2002, *Farmoor Reservoir, UK*; Bonnet and Poulin, 2002, 2004, *Lake Villerest, France, Burrinjuck Reservoir, Australia*; Lavrik et al., 2002, *Dnieper Reservoirs, Russia*; Wu et al., 2004, *Shihmen Reservoir, Taiwan*; Soyupak et al., 1997, *Keban Dam, Turkey).* If properly designed and calibrated these models can assist with management decisions that require the consideration of alternative scenarios. They often offer sufficient spatial and temporal resolution to model algal blooms and other responses to eutrophication. However, the data requirements and process-level understanding demanded by dynamic models can be formidable.

3.1.2.5 Management of Eutrophication

The primary step in the reduction of eutrophication in reservoirs is to limit, divert or treat inputs of nutrients and associated particles i.e. preventative intervention. In many north temperate lakes and reservoirs, such control of external phosphorus supplies has improved water quality. However, in other regions, nitrogen, which is much more difficult to control, is sometimes the primary nutrient.

Because reservoirs can trap and recycle nutrients and organic matter, reduction in loading from the watershed may not rapidly reverse the impacts of eutrophication. Hence, it may be necessary to modify internal physical, chemical or biological processes, as described below under ‘in-lake-management’.

- **Management and Restoration**
  Theoretically, a reservoir will respond to “management” in accordance with the impact of the forcing functions or external variables. Management therefore has no

meaning unless the forcing functions are changed (UNEP, 1999). From a management point of view, however, it may take much too long before the response of the reservoir can be observed after forcing functions are changed, especially if there is a long retention time In such situations, restoration methods may significantly speed up the response to the changed forcing functions.

In-lake (ecotechnological) management

Many techniques are available for the improvement of water quality within the body of a reservoir. Information in this section is taken from Straškraba and Tundisi (1999) unless otherwise indicated.

3.1.2.5.1 Mixing and oxygenation

The aim of artificial mixing is oxidation of either a deoxygenated hypolimnion or the entire waterbody and/or inhibition of phytoplankton growth. Figure 27 shows the basic types of mixing.

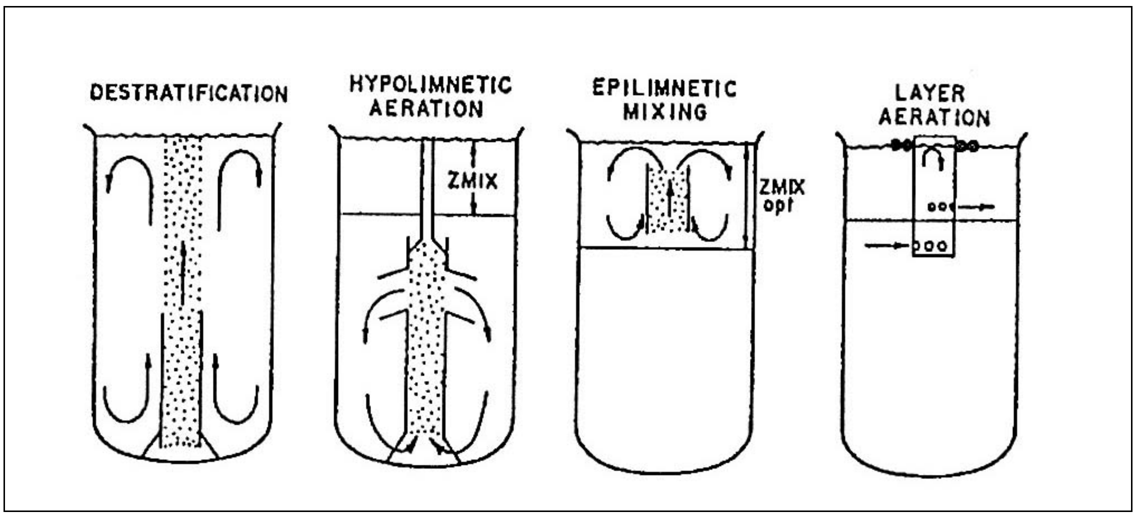


Figure 27. Four basic types of mixing (after Straškraba and Tundisi, 1999).

- **Destratification – artificial circulation**

  Destratification is accomplished by injection of compressed air from a diffuser into the water at the reservoir bottom. (See Figure 28)

  The aims of such artificial circulation are firstly to reduce algal biomass by means of light limitation of cells, which is achieved due to their spending less time in the illuminated zone and, secondly, to maintain oxygenated conditions at the sediment-water interface in order to prevent internal loading of nutrients from the sediment. Thirdly, shifts from nuisance cyanobacteria to populations of green algae or diatoms may be achieved through increased CO_2 and thus lowered pH, loss of effectiveness of buoyancy controlling mechanisms, particularly in the case of subsurface populations such as *Oscillatoria*, and reduced sinking of heavier species such as diatoms (Barbiero et al., 1996b).

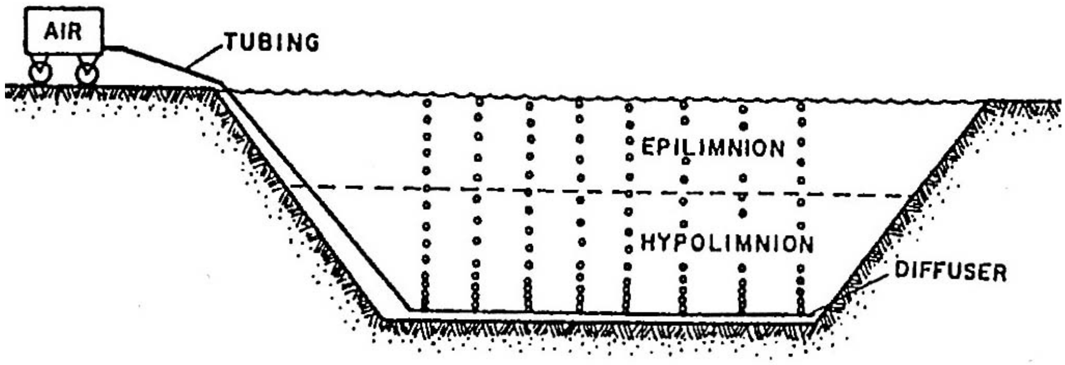


Figure 28. Destratification with a compressor and diffuser (after Straškraba and Tundisi, 1999).

This method is not inevitable successful however. Insufficient mixing can have extremely negative effects and water quality may become degraded rather then improved (Schladow, 1993). Compressed air caused supersaturation of dissolved nitrogen, causing downstream fish kills (Fast and Hulquist, 1982, both in Straškraba and Tundisi (1999).

Case studies

1 From results obtained in East Sidney Lake, a small eutrophic reservoir in USA, Barbiero et al. (1996a) concluded that, in and of themselves, changes in the physical structure of the water column should not be expected to alter phytoplankton community makeup unless they effect changes in the underlying physical or chemical factors that determine successional sequences. Of the factors examined in this study, these included TN: TP, temperature and flushing rate, none of which were sufficiently altered by artificial circulation to affect phytoplankton community size or structure.

2 The effects of artificial circulation in a small tropical reservoir, Solomon Dam, on Palm Island, off the coast of Queensland, Australia, were reported by Hawkins and Griffiths (1993). In this study, the general tendency was for total chlorophyll to be higher during periods of artificial stratification than during unmanaged periods. This was probably related to extensive drawdown during the latter part of the destratification period which reduced the ratio $Z_{m:}$ Z_{eu} from 2.0 to 1.2. The method was nevertheless effective in changing the structure of the phytoplankton community, switching dominance from bloom-forming cyanobacteria to diatoms.

3 A long term data set from North Pine Dam in Brisbane, Australia (Antenucci et al., 2005), proved invaluable in assessing both the long-term impact of

destratification on water quality, and the factors affecting the dominant cyanobacterial species, *Cylindrospermopsis raciborskii.* Although destratification was successful in reducing chlorophyll a in some areas of the reservoir, the method was not successful in avoiding the dominance by *Cylindrospermopsis.* This indicates that the adaptive features of phytoplankton must be considered when implementing amelioration strategies in reservoirs, and that attempts to reduce internal loading should also be matched with concurrent efforts to reduce external nutrient loading.

4 The use of an intermittent destratification strategy (Simmons, 1998) was used to prevent stratification from becoming established in Hanningfield Reservoir (UK). It was anticipated that the constant deselection of any one type of environment would promote competition among the algae, leading to lower overall biomass. The results showed a 66% reduction in mean annual phytoplankton biomass after three years, but no change in type or succession of phytoplankton. However, the author cautions that even though the results are encouraging, the observed outcomes cannot be conclusively attributed to the intervention used.

5 Problems of manganese release from the bottom sediments in Teddington Weir, (Queensland, Australia) prompted the installation of an automatic stratification device, controlled by temperature sensors (Burns, 1998). By controlling mixing automatically, to take place only when stratified conditions are detected, stratification can be kept under control at minimum cost, and the resulting intermittent mixing has been found to have advantages over continuous mixing for algal control (Steinberg and Gruhl, 1991; Burns, 1994, both in Burns, 1998).

Burns' design of air-diffusion mixing is based on the assumption that localized mixing creates the unstable condition of having a column of uniform density water surrounded by non-uniform water stratified into various densities. By providing sufficient energy to maintain the uniform density column of water, convection eddies are generated which destratify the whole water body. (Figure 29). The successful application of these design methods on many projects (Burns, 1996 in Burns, 1998; Daldorph, 1998) and the speed at which destratification happens even in very large water bodies with no visible disturbance other than at the aerator (Burns, 1995a in Burns, 1998) gives support to the theory (Burns, 1998).

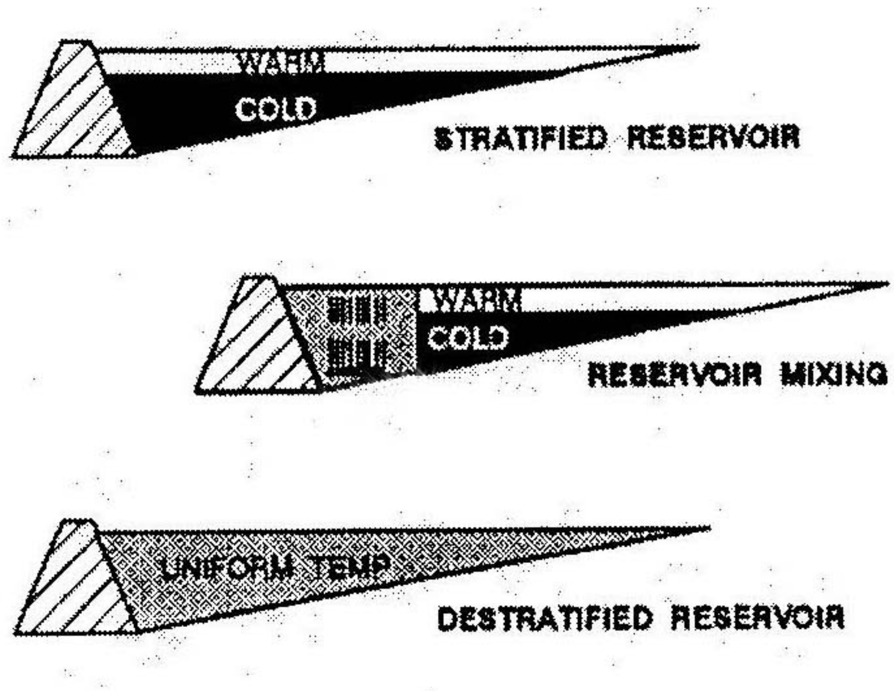


Figure 29. Thermal stratification and its control (after Burns 1998).

Some of the complex consequences of destratification are shown in Figure 30. This complexity is the reason that knowledge of limnological processes is necessary, although it may appear to be a mechanically simple method.

The cost of the procedure is relatively low, being limited to the price of the compressor and corresponding energy demands, and the installation costs. In an isolated area in Australia, solar panels were used to drive the compressor instead of disturbing the environment by the construction of a power line. Paradoxically, the only attempt at artificial mixing known for a South African reservoir (Inanda Dam) was apparently physically unsuccessful.

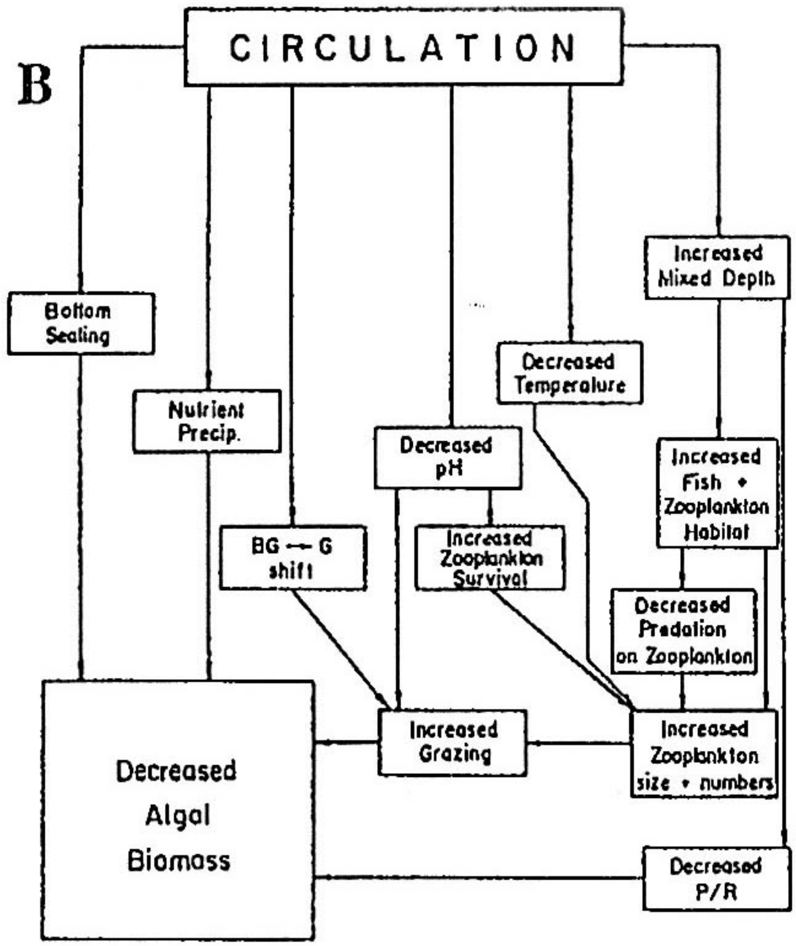


Figure 30. Consequences of circulation for water quality according to Pastorok et al. (1981) in Straškraba and Tundisi (1999).

- **Hypolimnetic aeration**

  Several types of commercial aerators are available that can aerate the hypolimnion without destroying the thermocline. (Figure 31). This is appropriate as a corrective technique in cases of hypolimnetic oxygen deficits, but is not applicable for shallow lakes or narrow, near-bottom anoxic layers.

  Theoretically, the advantages of this method are that (i) aeration is accomplished without transferring elements (e.g. nutrients) from the hypolimnion to the epilimnion, and thus algal growth is not enhanced, (ii) water quality is improved by the decrease of iron, manganese and taste and odour problems in drinking water supplies, (iii) improved oxygen conditions in the hypolimnion permit cultivation of sensitive fish, and (iv) damage by corrosion to turbines and other structures is reduced thus improving downstream water quality and the cost of repair (Straškraba and Tundisi, 1999).

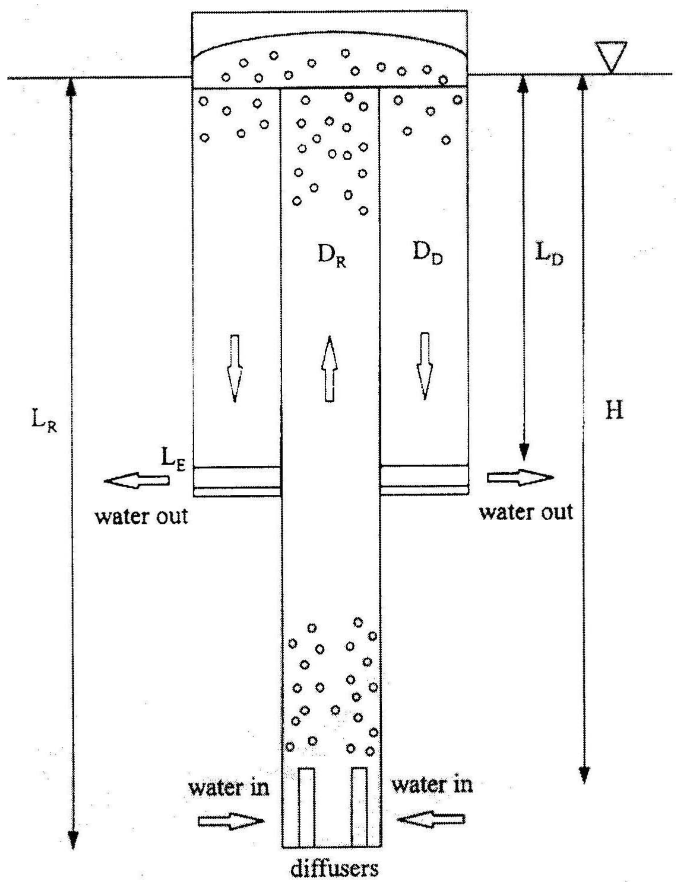


Figure 31. Schematic representation of a hypolimnetic aerator (after Burris and Little, 1998).

The investment cost is higher than destratification because of specialized equipment requirements. The operating costs depend on the areas of the hypolimnion, rate of oxygen consumption of the reservoir and the degree of thermal stratification. These costs can be estimated by a method devised by Cooke et al. (1993).

A hypolimnetic aerator operation was tested in Lake Prince, a water supply reservoir in Norfolk, USA by Burris and Little (1998). Improvements in hypolimnetic water quality had been recorded since the installation of the aerator, including lowered concentrations of iron, manganese, sulphide and phosphorus. However, steady oxygen loss from the hypolimnion, found during aerated periods, lead to increased releases of iron and manganese from lake sediments with corresponding deterioration in water quality. A model that had been developed for a Speece Cone (McGinnis and Little, 1998) was modified to conform to the conditions of the hypolimnetic aerator. By varying a single parameter (the initial bubble size) the model was able to show that a doubling in oxygen transfer could be achieved if initial bubble size was reduced from 5mm to 2.5mm.

- **Epilimnetic mixing**

This method is recommended because, as opposed to other mixing methods that consist of corrective actions, epilimnetic mixing seeks to prevent or reduce the formation of phytoplankton biomass. Surface waters are mixed to an optimum depth $z_{mix}opt$ (the depth corresponding to z_{eu}). In shallower impoundments (where the depth corresponds to $z_{mix}opt$) the formation of a hypolimnion with resulting deoxygenation is prevented.

No negative environmental impacts are evident but if the technology fails, rapid growth of algae follows, with the associated costs.

- **Layer aeration**

This approach is based on detailed knowledge of the stratification conditions of a given reservoir and the consequences of these in terms of water quality. Heat and oxygen in a stratified reservoir are redistributed into discrete layers. Manipulation of the thermal structure can create desirable physical and chemical (in particular oxygen) conditions.

Detailed limnological knowledge is required for successful implementation of this method.

- **Speece cone**

The Speece Cone (Speece et al., 1982; Speece, 1994) is a highly technical device designed to oxygenate water with little mixing. The device works by releasing water supersaturated with oxygen into the hypolimnion by means of diffusers (Figure 32).

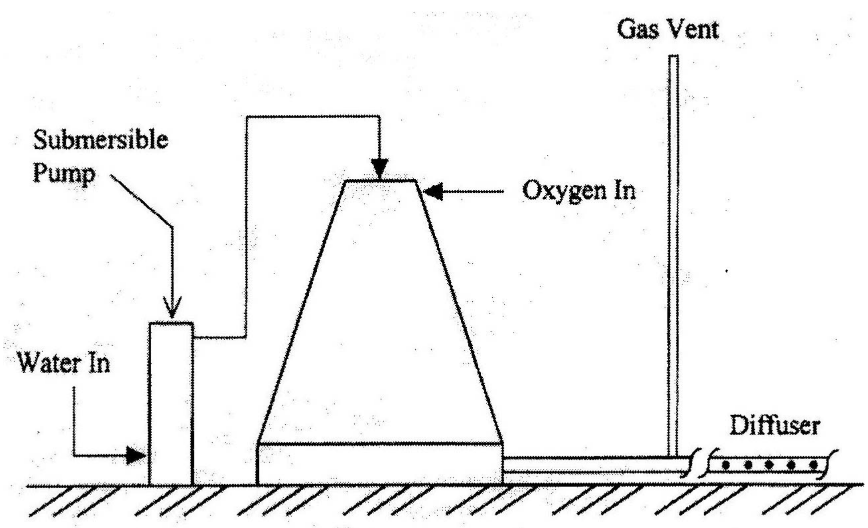


Figure 32. Diagram of a Speece Cone (after McGinnis and Little, 1998)

A model that predicts bubble dynamics and oxygen transfer within a Speece Cone has been developed by McGinnis and Little (1998). Kennedy et al. (1995) report that the investment costs for a large reservoir are approximately 5 million $US.

- **Propeller mixing and oxygenation**

  This method differs from all those already discussed in terms of how the mixing is achieved. The previously described methods are based on mixing by raising air bubbles from below. This method involves mixing from above by the use of a propeller that is attached to the bottom of a pontoon and can be used in many different applications (Figure 33).

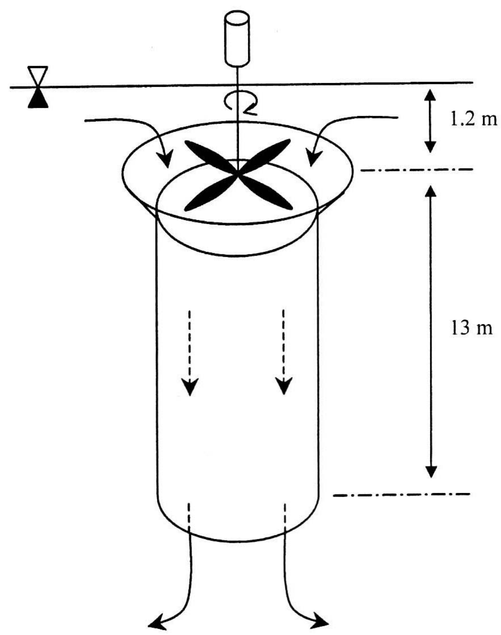


Figure 33. Schematic diagram of one type of surface mixer (after Lewis et al., 2003).

A laboratory study by Stephens and Imberger (1993) examined the efficiency of energy conversion from mechanical energy to buoyant flux as a function of impeller diameter, the speed of rotation and the degree and type of stratification. The authors concluded that it remains to be seen whether mechanical mixers are a viable alternative stratification technique to the above methods. Baines and Leitch (1992) examined experimentally, the mechanics of liquid transport by bubbles in a stratified environment. A stream of bubbles carries with it an upward liquid flow. In an unstratified environment, the liquid rises in an expanding turbulent jet until it is expelled at the surface. However, when the environment is stratified, both the upflow in the liquid plume and the return motion are affected by buoyancy. All the fluid in the liquid plume has been entrained from lower in the stratification, so will always be denser than its surroundings. Any of this fluid that is detrained will flow downward to the elevation of static equilibrium. Although this experiment was performed in a small tank, these results can be used to evaluate the use of a bubble plume to mix a stratified reservoir.

3.1.2.5.2 Sediment removal

- **Dredging the sediments**

  This method involves removing the phosphorus-rich upper layers of the sediment using various types of dredging equipment (See Fig 34).

  The main advantage of this method is that the results are long-lasting, as long as the external loading of nutrients from the watershed in minimized (UNEP, 1999).

  A disadvantage is that extensive areas are needed to store dredged slurry while it is drying and before it can be used as a fertilizer (only when it has a low heavy metal content) or otherwise disposed of (UNEP, 1999). In many countries the management of dredged material is an issue of growing concern because the volumes are so large and the contamination may be very high. Ecological risk or hazard assessment approaches are frequently used as decision support tools (Babut et al., 2003).

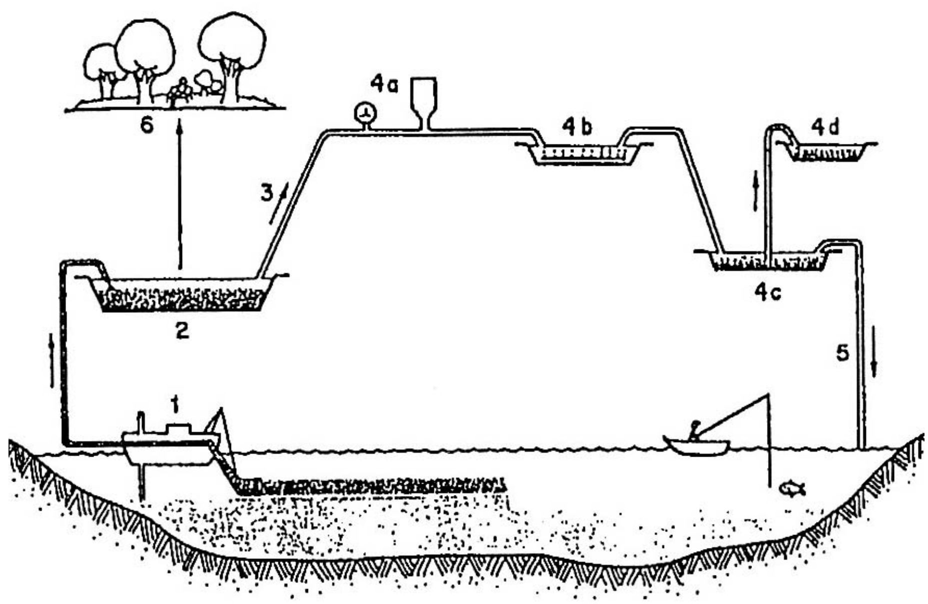


Figure 34. Schematic representation of sediment removal. 1- The bottom mud suction dredger, 2-Settling pond for sediment drying, 3-Runoff water to the aluminium sulphate automatic dosing instrument 4a and its aeration basin 4b. The overlying water is returned from the sedimentation basin 4c to the lake through tube 5, while the dried treated sediment is used as fertilizer in agriculture 6. (Redrawn from Eiseltova (1994) in 818).

3.1.2.5.3 Sediment aeration and oxidation - sediment injection

The goal of this method is to decrease phosphorus release from the sediments. This may be achieved by the application of ferric chloride to sediments that are low in iron to decrease phosphorus release. Lime may be added to create a pH level that is optimal for denitrification, or calcium nitrate is injected into the top 30cm of sediments to oxidize and break down organic matter, and denitrify the elements. This procedure must be specifically adapted for each application in accordance with existing chemical conditions in the sediments. However, this method requires special equipment that can be used on flat and shallow bottoms only.

3.1.2.5.4 Sediment capping - covering sediments with inert matter

An alternative to sediment aeration or oxidation is simply to cap the sediments with foil, raw ash, crushed bricks, sand, clay or other inert material. This method is relatively inexpensive and effective.

3.1.2.5.5 Phosphorus inactivation

- **Alum precipitation**

Spreading alum ($AlSO_4$) over the lake surface is a procedure used to precipitate phosphorus from the lake and seal the bottom against phosphorus release. Alum forms gelatinous flocs that sorb dissolved phosphorus and accumulate on the bottom where they sorb phosphorus that leaches out of the sediments.

The hypolimnion, only, of Eau Galle Reservoir, Wisconsin, USA (James et al., 1991) was treated with alum as it was assumed that only anoxic sediments contributed to the internal loading of phosphorus. Theoretically, the dose applied was sufficient to control sediment phosphorus release for 5 years. However, the effectiveness of the treatment was greatly diminished after1 year. Several possible factors may have contributed to this, including burial and inactivation of alum by sedimentation, phosphorus release from the untreated littoral zone, and phosphorus inputs from ungauged hydraulic sources (groundwater inputs).

Advantages of this method are that no special equipment is needed and the effect is long lasting (up to 14 years). Negative environmental impacts can occur in connection with toxicity of aluminium at pH less than 6. However, Al concentrations below 50 $ug.l^{-1}$ are not considered harmful to organisms. In addition, this method is not feasible in waterbodies overgrown with macrophytes or where there is intensive resuspension, a serious constraint in many reservoirs where marked drawdown occurs.

Phosphate can also be precipitated by the addition of ferric sulphate, ferric chloride or calcium hydroxide. These chemicals are added either to the pumped input, the water column or the sediment (Perkins and Underwood. 2002).

3.1.2.5.6 Biomanipulation

The principle of biomanipulation as a means of eutrophication management is the manipulation of the food chain. Control is theoretically achieved by maintaining low feeding pressure on zooplankton by fish, so that large species of zooplankton predominate that are capable of keeping phytoplankton biomass under control (Moss, 1998; Straškraba and Tundisi, 1999). (Figure 35)

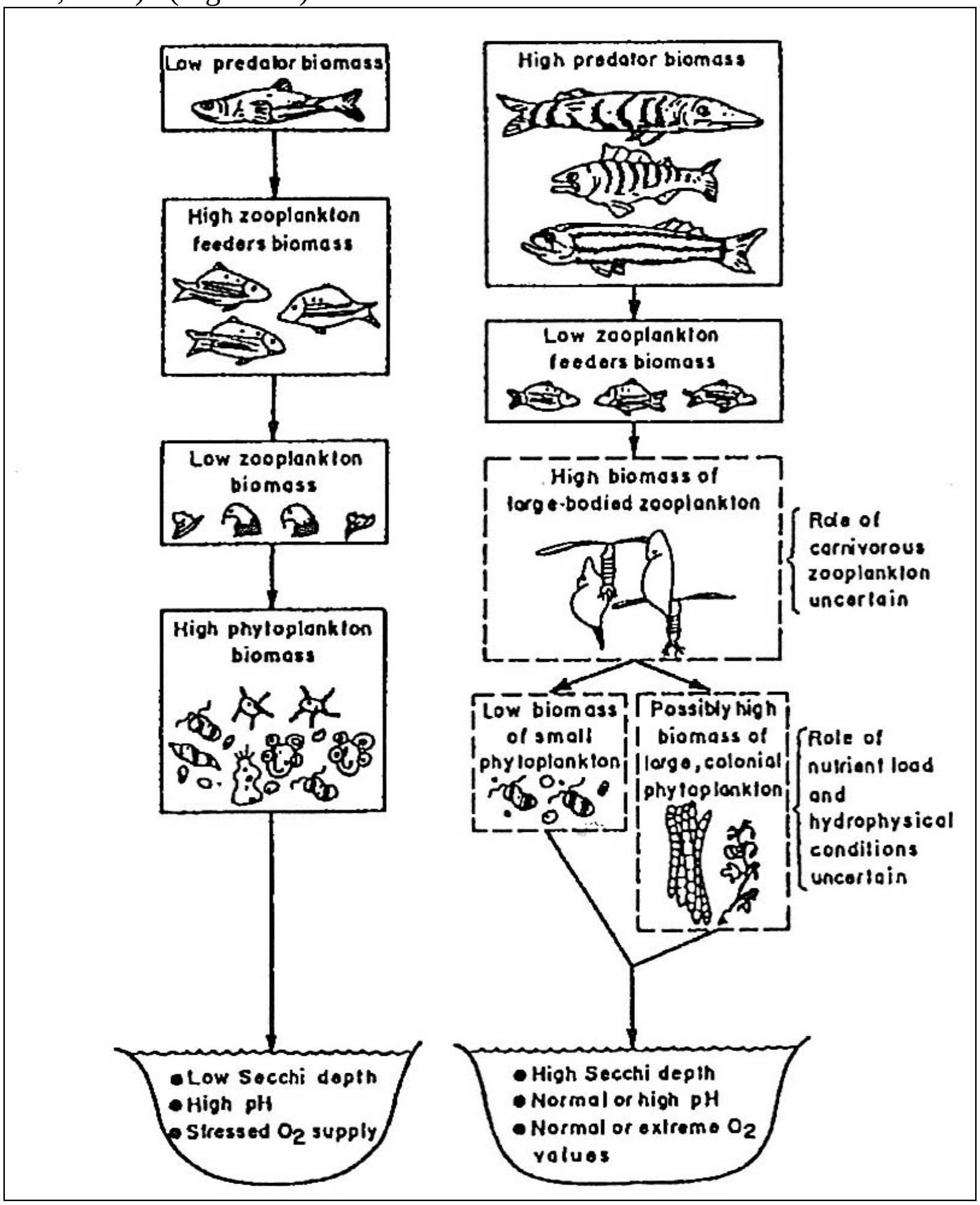


Figure 35. A schematic representation of biomanipulation. The left panel shows the consequence of low predatory fish biomass on the composition of zooplankton and phytoplankton, and consequently low transparency, high pH and low hypolimnetic oxygen concentration. The right panel shows the consequences of increasing the predator biomass thus decreasing the biomass of zooplanktivores and allowing an increase in zooplankton which decreases the phytoplankton biomass resulting in high transparency, lower pH and sufficient oxygen in the hypolimnion (from Benndorf et al., 1984, in Straškraba and Tundisi, 1999).

3.1.2.5.6.1 Manipulation of fish populations

Development of fish populations to maintain a desirable zooplankton/phytoplankton ratio can be achieved in various ways (Straškraba and Tundisi, 1999).

- The eradication of overcrowded, stunted fish populations, which due to their high numbers and greater food requirements grow very slowly, can be managed by means of Rotenone poisoning followed by predator stocking.
- The continuous introduction of predatory fish and net harvesting of non-predatory fish has been utilized in many cases (Seda and Kubecka, 1997). Collaboration with local sport fishery organizations and commercial fisheries is useful, if not necessary.
- Reservoir drawdown during reproductive periods of undesirable fish species, by exposing the eggs on shore vegetation or gravel is another method of controlling populations (Seda and Kubecka, 1997).

Recent reviews of biomanipulation (Straškraba and Tundisi, 1999; Gulati et al., 1990; DeBernardi and Giussani (Eds), 1995; Shapiro, 1995; DeMelo et al., 1992) have shown that this method is only partially successful and only under certain conditions. A thorough evaluation of biomanipulation experiments published prior to 1992 (22 whole lake experiments and 10 enclosure experiments) is summarized in the box below (after Straškraba and Tundisi, 1999). With one exception, these were all performed in the well-watered regions of the north temperate zone.

1) Success, measured by decreased phytoplankton biomass, was achieved in 2/3 of the 25 observations, whereas in 28% the results were ambiguous and, undesirable results were obtained only in two cases.

2) When phosphorus concentrations were very high, no successes were observed. Therefore, in very eutrophic waterbodies, it is necessary to combine biomanipulation with nutrient reduction by using other methods.

3) Success is generally more pronounced in shallow and small water bodies, more specifically because control of fish populations is easier under such conditions.

4) A shallow waterbody or shallows of deeper waterbodies may switch to a macrophyte dominated state and it is to be decided which consequences this has for the respective use of the water body.

5) Creation and stabilization of a strong population of piscivorous, predatory fish population is difficult and time consuming, unless achieved by commercial net fisheries. The use of rotenone poisoning is often required to start the conversion.

6) The extent of planktivorous fish that must be removed depends considerably upon the species and size composition of the fish community.

7) Due to very different organism turnover times, a stable new equilibrium may require several years to develop. However, this is also true for other eutrophication processes.

8) Biomanipulation procedures cannot be considered a routine method because use of this method depends on a number of special circumstances and can only be performed with the participation of skilled limnologists.

9) Criteria for successful application of biomanipulation must be based upon local and waterbody type specifics.

Some case studies in temperate regions since 1992

- **Rimov Reservoir Czech Republic (Seda and Kubecka, 1997; Komarkova et al., 1995)**

Biomanipulative management of planktivorous fish involved 3 strategies, (i) control of spawning success of cyprinids by water level manipulation, (ii) capture and removal of undesirable fish, and (iii) enhancement of predatory fish populations.

The only parameter influenced by the changes in fish stock was the zooplankton body size structure. The biomass and density of large-bodied *Daphnia* were negligible during periods of high fish stock and increased up to 10% of total zooplankton when fish stocks were reduced. No significant differences in total phytoplankton biomass were found. These authors maintain that the assumption of managed control of fish stock in lakes and reservoirs becomes more and more fragile with the increase in size of the water body. Planktivorous fish cannot practically be removed by poisoning or by draining, any substantial lowering of the impact of planktivory requiring a massive stocking programme of adult fish predators. The cost-effectiveness of such stocking is often questionable, especially because of the uncertainty of its stability.

The authors posed the question whether a threshold level of fish predation or zooplankton status exists above which the top-down effects on the lowering of phytoplankton biomass become evident. Based on this study, the biomanipulation experiment on Hubenov Reservoir in the Czech Republic (Hrbacek et al., 1978; 1986, in Seda and Kubecka, 1997), and the work of Seda and Duncan (1994) in the London reservoirs, they concluded that this critical limit relies on a total zooplankton community biomass comprising between 20 and 30% of large daphnids in total biomass.

- **Maltanski Reservoir Poland (Goldyn et al., 1997)**

This reservoir was emptied in September, 1992, and all fish were harvested. After refilling the following year, it was stocked with fry of eel, pike and wels, and a year later with adult pike in spring, pike fry in summer and pike and perch-pike fry in autumn. However, high water transparency was only retained in the first year when large-bodied *Daphnia* were present. A changed composition of cladocerans then resulted in uncontrolled growth of the colonial cyanobacterium *Aphanizomenon flos-aquae.* Similar situations were reported by Benndorf (1990), Gliwicz (1990), Sondergaard et al. (1990, 1997) and Kasprzak et al. (1993). It was concluded that insufficient predatory fish had been introduced in this experiment, with a resultant inadequate control of planktivorous fry and cascading effects. According to Sondergaard et al. (1997) and Prejs et al. (1994) stocking of 1000 piscivorous fry ha^{-1} is needed to achieve a strong effect on lower trophic levels, while introduction in the Maltanski Reservoir was only 236 ha^{-1} in the first year, and 156 ha^{-1} and 413 ha^{-1} subsequently.

3.1.2.5.6.2 Food web manipulation in tropical and subtropical waterbodies

Theory concerning lake and reservoir food webs, and in particular regarding the key role of fish, has been based largely on research in temperate lakes of Europe and North America (Drenner et al., 1996). Research is still needed to assess whether these theories apply to tropical and subtropical systems (Lazzaro, 1997; Arcifa and Northcote, 1997; Lazzaro et al., 2003).

Lazzaro (1997) listed the following "peculiarities" of tropical lakes compared with temperate lakes.

(i) In many tropical lakes, visual feeding zooplanktivores coexist with filter feeding omnivores (clupeids, cyprinids and cichlids, occurring naturally or as a consequence of stocking) that feed on several trophic levels. These omnivores are neither limited by zooplankton (because they can feed alternatively on phytoplankton and detritus) nor controlled by fish predators (because of fast growth enabling them to quickly reach a size larger than predator mouth gape). Systems dominated by omnivory have complex web-like structures in addition to weaker trophic links and their dynamics are poorly predictable from the "trophic cascade model" (Lazzaro, 1997; Lazzaro et al., 2003).

(ii) Invertebrate predation by e.g. *Chaoborus* and *Culex* species (analogous to *Leptodora* and *Neomysis* species in temperate waters) is often strong and occurs all year round. It may prevent the increase in abundance of large herbivores even in the absence of zooplanktivorous fish.

(iii) Large-bodied herbivorous crustaceans (mainly cladocerans) are relatively less abundant and/or species reach smaller sizes. This weakens any possible control of phytoplankton by zooplankton.

(iv) Contrary to the typical seasonality of fish reproduction in temperate systems, in tropical systems fish species may spawn at different times. Therefore juvenile fish that are predominantly planktivorous, regardless of their feeding habits as adults, maintain a quasi-permanent size selective predation pressure on zooplankton. This favours small zooplankton species and reduces even more the grazing control of phytoplankton by zooplankton.

(v) Due to high temperature regimes in tropical waters, zooplankton generation times are shorter. As a result, food requirements, which are higher, may occasionally not be met. Both circumstances favour smaller species. (This is questionable in terms of the "Size Efficiency Hypothesis" (SEH)).

(vi) Large open-water piscivores that are the dominant fish predators in North America and Europe often have no equivalent in tropical lakes or reservoirs. They are replaced by generalists, predominantly carnivorous, fish, often small-sized and sedentary (e.g. carnivorous cichlids and erythrinids in South America).

The above characteristics of tropical lakes and reservoirs weaken the top-down links between piscivores and planktivores, and between zooplankton and phytoplankton. These links must be strong for trophic cascades to occur, thus, the weaker the links, the weaker the chance of successful biomanipulation based on trophic cascades alone (Lazzaro, 1997).

Arcifa and Northcote (1977), reviewing their work on the Americana reservoir in Brazil and the fish fauna in other Brazilian water bodies, suggested that the complex food web structure would not allow regulation by simple chain linkages typical of trophic cascades in northern temperate lakes, a conclusion also reached for lower latitude reservoirs in the United States by Stein et al., 1995. Thus in prospective biomanipulation experiments in Brazilian reservoirs there is a great need for holistic approaches in designing the experiments, interpreting their results and applying their findings to practical management measures (Arcifa and Northcote, 1997). This need arises for the following reasons:

(i) The origin of the rich fish community is riverine, has been subjected to marked changes in habitat and food web structure, and is often not fully stabilized after the multiple effects of impoundment.

(ii) The strictly zooplanktivorous fish species are of lesser importance, while omnivores, insectivores, detritivores, piscivores and iliophagous forms are of greater importance.

(iii) Turbidity levels are higher, especially in the wet season, combined with nocturnal vertical migration of zooplankton.

(iv) Small zooplankton species are dominant, even under low predation pressure or in the absence of zooplanktivores.

(v) There is a great opportunity for strong experimental method interactions with physical, chemical and biological factors during the course of the experiments.

The control of phytoplankton biomass through the direct exploitation by a filter feeding fish (e.g. *Hypophthalmus edentatus)* able to feed also on large inedible algae (Arcifa et al., 1995) seems to be the most suitable method for Brazilian reservoirs, in addition to bottom-up control (Arcifa and Northcote, 1997).

The potential use of food web manipulation for controlling cyanobacteria has also been addressed for Australian waters (Boon et al., 1994; Gehrke and Harris, 1994; Matveev et al., 1994; all in Hunt et al., 2003; Matveev. 1998, 2003). Arguments are largely unfavourable, mainly on the basis of resistance of cyanobacteria to zooplankton grazing due to uningestible size, grazing resistant coverings and toxicity (Boon et al. 1994). However, results from food web manipulation studies suggest that both grazing and nutrient recycling can affect cyanobacteria (Elser, 1999). Substantial declines of cyanobacteria biomass have been observed in response to reductions or removal of planktivorous fish. This has been attributed to increased zooplankton grazing (Sanni and Waevagen, 1990; Christoffersen et al., 1993; Borics et al., 2000), to different rates of nutrient regeneration (Vanni et al., 1990) or to both (Lyche et al., 1990; Sondergaard et al., 1990). However, the positive correlation of cyanobacteria with planktivorous fish does not always apply. Reduction or removal of such fish has resulted in increased cyanobacteria biomass (Lyche, 1989, in Hunt et al., 2003), and has shifted the dominance to a different species of cyanobacteria (Carpenter et al., 1995). Hence the complex interactions occurring in lake and reservoir food webs result in unpredictability in the response of a system to manipulation (Hunt et al., 2003).

The subtropical reservoirs of Australia present novel environments for food web manipulation in terms of their biota, climate, physical limnology, age, and nutrient fluxes (Banens and Davis, 1998; Matveev, 2003).

Hunt et al. (2003) experimentally evaluated the short-term response of phyto- and zooplankton to complete removal, reduced density and an ambient density of the filter-feeding Australian gudgeon (*Hypseleotris* sp), a zooplanktivore, in Lake Maroon, a subtropical reservoir, during a summer period when the phytoplankton was dominated by the cyanobacterium *Anabaena circinalis.* The results revealed that the density of *Hypseleotris* can have strong effects on the biomass and community structure of phytoplankton, this being particularly evident in the response of cyanobacteria. The possibility that the N: P ratios of nutrient excretion could explain the relative abundance of grazer-resistant nitrogen-fixing bacteria and chlorophytes at different densities of *Hypseleotris* implies that nutrient excretion is an important mechanism in determining phytoplankton community structure. These authors propose that nutrient regeneration becomes increasingly important in determining phytoplankton response to food web manipulation as the degree of nutrient limitation increases.

As previously mentioned the response of subtropical lakes to food web manipulations has been shown to deviate from expectations based on temperate lakes (Crisman and Beaver, 1990; Havens et al., 1996). One of the deviations of subtropical and tropical freshwater environments in Australia is an observed tendency for nitrogen to exceed phosphorus as the predominant limiting nutrient (Hunt et al., unpublished data; Udy et al., 1999, in Hunt et al., 2003). Consequently such systems may not fit with models developed within the "phosphorus paradigm" from studies conducted in temperate environments (Hunt et al., 2003).

3.1.2.5.6.3 Assessment of biomanipulation as a method of eutrophication management.

The main advantage of this method is that it is environmentally friendly, with no machinery or chemicals required (as long as Rotenone is not used). It is not desirable to use Rotenone in drinking water supplies. In addition it may result in kills of desirable species, although it is not toxic to invertebrates or phytoplankton.

It combines the requirements of fisheries with those of water quality, but in this regard, education of fishermen is necessary. Continuous control of fish populations is important especially where sport fishery selectively removes predatory rather than planktivorous fish.

Expert limnological knowledge is essential and each system needs to be thoroughly investigated before restorative action is taken. Continuous control of fish populations is necessary, especially where sport fishery selectively removes predatory rather than planktivorous fish.

Logically, however, the prospects of biomanipulation as a tool for eutrophication management in southern African reservoirs have also been cautioned on the following grounds (Hart, submitted).

(i) High mineral turbidity counters the development of large-bodied cladoceran grazers (even though light-limitation serves as a primary constraint on algal growth under such conditions).

(ii) Large-bodied cladoceran grazers are mostly ineffective feeders on large colonial/filamentous cyanobacteria that are fundamentally more favoured by the N: P ratios and light climates expressed in hypertrophic reservoirs, like Hartbeespoort Dam.

(iii) The absence of obligate planktivores in reservoirs limits the impact of predation by facultative zooplanktivores largely to littoral of inshore margins. Zooplankton grazing control in offshore waters is naturally not constrained by planktivores.

3.1.2.5.7 Hydraulic regulation

Selective off-takes

Flushing of either the epilimnion or the hypolimnion will depend on the goal – either a decrease of algal mass in the surface waters, or flushing of the deoxygenated bottom waters with high nutrient concentration and/or other undesirable pollutants. Another function is to pass water, with high nutrient input or pollutants, through the reservoir as quickly as possible with the least possible mixing with other layers and before it can settle. (Figure 36)

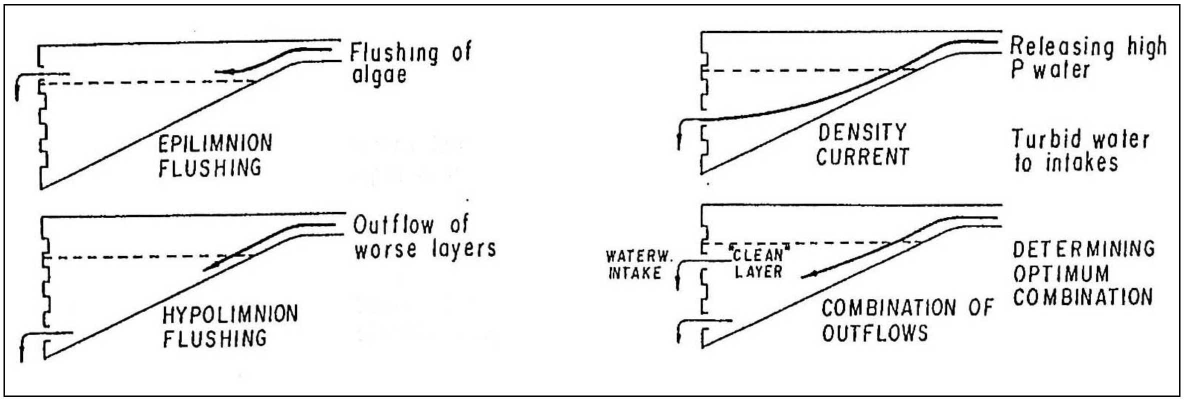


Figure 36. Function of selective off-takes. From top left – withdrawal through the surface outlet is used for the flushing of excessive algae. Hypolimnion flushing is used to release anoxic and iron, phosphorus and manganese rich hypolimnetic water. By creating density currents, a peak of turbid or pollution-rich water (during floods) can pass rapidly through the reservoir before it has time to sink.

Several scenarios are possible provided the limnological regime of the reservoir is understood.

The best scenario is that a reservoir is built with multiple outlets, with outlet pipes placed at about 5m intervals. A minimum provision is to include a surface and a bottom water discharge. Reservoirs with multiple off takes afford much better control in that good quality water can always be abstracted by selecting the appropriate outlet depth in relation to thermal stratification. (Klapper, 2002).

The disadvantages of this method are that knowledge is required of the inflow water quality and depth distribution in the reservoir. Dynamic changes in the quality of a specific water layer can take place as a result of selective offtakes and these are not easily understood unless hydrodynamic models are used. Possible negative downstream impacts caused by releases must be considered.

Hypolimnetic siphoning

This simple method consists of siphoning water from the hypolimnion of a lake or reservoir. An estimate of the character of the water that will replace what is siphoned off is needed (Straškraba and Tundisi, 1999).

Curtains

The use of plastic curtains to modify outflow depth can substitute to some degree for multiple outlets. Near surface or near bottom outlets are more feasible than those at intermediate depths.

Amongst other functions, the Terauchi Reservoir in Japan supplies water for paddy field irrigation. Because paddy field irrigation requires water warmer than 17^{o}C, destruction of the thermocline is not an option in the management of eutrophication (Priyantha et al., 1997; Asaeda et al., 1996; 2001). As a means of reducing algal blooms in the reservoir, two curtains with the same depth as the epilimnion were suspended vertically from the surface. This method reduced algal growth in the down-stream zone of the reservoir (Asaeda et al., 1996). (See figure 37).

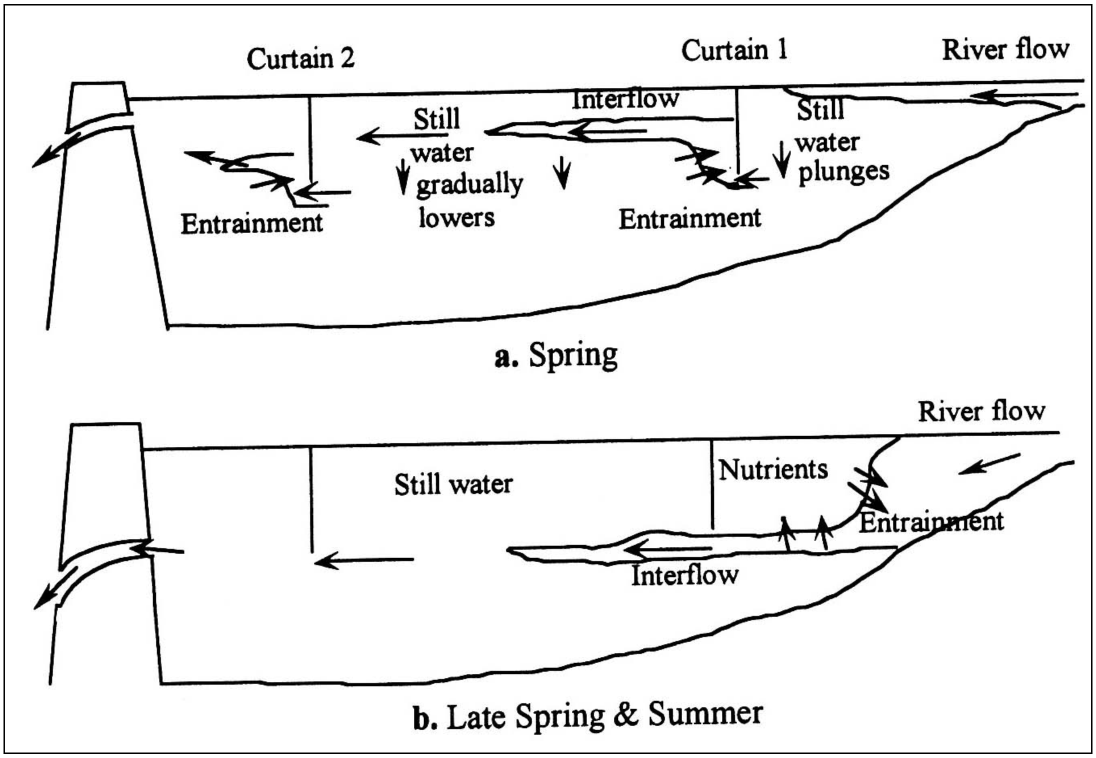


Figure 37. Illustration of possible mechanisms after installing curtains in the Terauchi Dam Reservoir in Japan (after Priyantha et al. 1997).

A submerged flexible curtain 160m long and 6m deep was installed in the Saidenbach Reservoir, Germany (Paul et al., 1998). The results were (i) an interruption of hydraulic short-circuits between inlet and main reservoir basin, (ii) a longer retention time of the inflowing water, and (iii) a 30-40% increase in the elimination of soluble reactive phosphorus in the mouth region during summer stratification. It is proposed that flexible curtains may be cost effective substitutions for conventional pre-dams of thermally stratified lakes or reservoirs (Paul et al., 1998).

3.1.2.5.8 Use of chemicals

According to Lam and Prepas (1997), chemical treatments to reduce phytoplankton biomass in freshwater ecosystems can be categorized into two groups. Chemicals in the first group (e.g. copper sulphate, Reglone A, chlorine and potassium permanganate) disrupt cell functions and induce cell lysis. Chemicals in the second group (e.g. lime and alum) precipitate phytoplankton and leave cells essentially intact The second group of chemicals are recommended for treating toxic phytoplankton blooms because they induce minimal release of toxins into the water (Naselli-Flores et al., 2003).

The addition of copper sulphate has been used often as an emergency measure in controlling excessive algal growth (Fernandes and Crisman, 1994). Its only advantage is that it works quickly. The disadvantages include the short duration of its effects. In alkaline waters or waters with high organic content, a chelated form must be used otherwise Cu is rapidly lost from solution. The method has negative environmental impacts as $CuSO_4$ is toxic to fish, zooplankton and other organisms, and the application of copper leads to long-term accumulation in the sediments (Straškraba and Tundisi, 1999).

The quality of Sicilian waters has deteriorated to a critical level, often posing a threat to human health (Naselli-Flores et al., 2003). As an emergency measure, the application of lime was tested as a possible short-term solution. The results indicated that the water bodies were suitable for lime treatments in spite of the high content of sulphate of geological origin, and an extreme hydrological regime. The recommendation was that the timing and method of lime treatment needed to be carefully planned.

In the case of algicide use, addition of a toxic chemical to drinking water, even in low concentrations is undesirable. Costs depend on the dosage and frequency of application.

3.1.2.5.9 Manipulation of underwater light regime

Light limitation can be imposed in two ways.

- The quantity of light that reaches the water surface is decreased. This can be achieved to some degree by shading the waterbody by means of surrounding tree growth, but this has obvious limitations. The placement of soot on the water surface has been suggested but is easily displaced by wind.

- The light absorption capacity of the water is decreased. The light attenuation coefficient may be increased by artificially colouring small waterbodies.

A decrease in light availability to the algal populations can be achieved by deep mixing of these populations with some of the techniques described above.

3.1.2.5.10 Macrophyte control

The need to control aquatic macrophytes

While it is advantageous to have healthy, stable macrophyte communities in aquatic ecosystems, overgrowth of macrophytes occurs in reservoirs as a result of changes in the physical, chemical and biological conditions brought about by the uncontrolled flow of nutrients from urban, agricultural and industrial centres and in silt eroded from the watersheds.

These "aquatic weeds" cause water loss through evapotranspiration, deterioration of water quality, displacement of native species and the consequent loss of biodiversity in water bodies, public health risks, obstruction of channels and drains in irrigation systems and intakes to hydroelectric plants, restriction of tourist, recreational and fishing activities and increased sedimentation with subsequent shortening of the useful life of the reservoir (Gutierrez et al., 1994).

- **Biological control**

  Biological control is defined as the use of one living organism to control another. The controlling agent may be bacterial, viral, animal or vegetable and may debilitate, eat or compete with the species targeted for control (Gutierrez et al., 1994; Riemer, 1988; Grodowitz, 1998; Chikwenhere, 1994; Pieterse et al., 2002). As a management strategy biological control is possibly the most economical in the long run, however, the controlling agent should be extremely well researched and preferably host specific to prevent it becoming an invasive alien (Gutierrez et al., 1994).

  Approximately 70 species of arthropods have been cited by authors in 12 countries as possible controlling agents of *Eichhornia crassipes* (Gutierrez et al., 1994). In South Africa a weevil *Neochetina eichhorniae* has been imported from South America where it feeds naturally on the plant, boring into the stem and leaves and severely damaging it. It has already proved successful in Namibia and has also been introduced into Lake Victoria (Davies and Day, 1998). The grasshopper *Paulinia acuminata* (also from South America) was used successfully to control *Salvinia molesta* on Lake Kariba after long and careful screening to ensure it would not attack agricultural crops (Davies and Day, 1998; Moreau, 1997).

  Grass carp (*Ctenopharygodon idella*) have been used successfully in many parts of the USA and Europe to control submersed macrophytes (Kalff, 2002; Pieterse and Murphy, 1990). A plant eating (phytophagous) fish native to northeast Asia, it is capable of consuming large amounts of vegetation (Maceina et al., 1992). However in many areas waterbodies are, by law, stocked with infertile triploid individuals (which display the same feeding behaviour as diploid fish), in order to prevent their overpopulation (Maceina et al., 1992), or accidental dispersal to other waterbodies.

  Another similar species (*Hypophthalmichthys molithrix*) is successfully used in other regions for the reduction of macrophytes (Maceina et al., 1992). This species is also capable of feeding on large colonies of Cyanobacteria, thus contributing to another positive water quality effect. As it is edible, it provides a food source for human consumption, which can offset management costs to some extent.

- **Mechanical control**

  Macrophyte harvesting is usually accomplished by the use of specially adapted boats, the design of which depends on the type of plants to be harvested and whether they are floating and softly-rooted or tightly-rooted submerged forms. Mechanical control is used to destroy them in situ or after the plants are transported to a disposal site (Gutierrez et al., 1994). This is generally a high cost method. If the harvested plants can be used as fertilizers or fodder, the cost may be recovered to some degree. However, in addition, the process needs to be repeated at regular intervals and has little or no long-term effect on biomass or areas covered by macrophytes. In addition it may actually stimulate the spread of nuisance species as many are able to propagate from small fragments produced during the cutting. It can also remove large numbers of small fish and invertebrates (Kalff, 2002). This system is mostly used selectively on small areas as a means of clearing channels, canals or outlets.

- **Chemical control**

  Chemical control of macrophytes by means of pesticides is the most commonly used and widespread method (Kalff, 2002; Gutierrez et al., 1994). These toxins are easy to apply, are usually effective and act rapidly. However, their use is expensive over large areas, and requires repeating after regrowth. The biggest issue is the increasing unacceptability of chemical controls involving the introduction of toxins to waterbodies (Kalff, 2002).

- **Habitat manipulation**

  The most often used method of habitat manipulation in reservoirs is water level management, which is obviously limited by operational policies and seasonal variation (Gutierrez et al., 1994). Water level is lowered; the resulting stranded macrophytes die off and are burnt. This may indeed eliminate macrophytes but depending on the timing and duration may simply select for species able to cope (Kalff, 2002). The nutrients contained may also simply be recycled upon reinundation.

  Other methods of habitat manipulation include the use of artificial coverings to limit sunlight and plant growth. However the cost is prohibitive and it is usually only applicable on a small scale in recreational areas or small dams

NB Emphasis should be placed on methods that do not require the use of toxic chemicals, the use of ecotechnological methods based on the use of the natural enemies of plants being obvious. However, extreme care and thorough investigation and evaluation are necessary before organisms are introduced into areas outside their natural ranges. There is great danger that they might eat other plants as well such as agricultural crops, and desirable indigenous species.

Precautions in macrophyte control

Based on experience with Hartbeespoort Dam, very careful assessment of trade-offs should be made before macrophyte control is implemented. Here, chemical eradication of *Eichhornia* (Ashton et al. 1980) actually invoked a much more serious problem of algal blooms which were precluded by light limitation when this macrophyte was prolific. Despite mechanical damage to boats, and inconvenience to recreational boaters, *Eichhornia* was the lesser of two evils in this hypertrophic reservoir.

3.1.5.2.11 Barley straw

The use of decomposing barley straw to inhibit the growth of algae (both green algae and cyanobacteria) is a technique that is becoming more widely known (Barrett et al., 1996; Ridge and Pillinger, 1996; Newman and Barrett, 1993).

3.1.5.2.12 Pre-reservoirs

Pre-reservoirs may or may not be considered to be a "within reservoir" water quality treatment method; however, they will be included in this report. Pre-reservoirs are comparatively small reservoirs with an average water retention time of a few days. They are normally situated immediately above the larger main reservoir whose water quality they are designed to improve (Putz, 1995; Putz and Benndorf, 1998).

Water quality is improved by reduction in loads of suspended matter and dissolved nutrients (especially soluble reactive phosphorus) (Paul, 2003). Chemical binding or adsorption of the orthophosphates in solution can take place in the inflowing waters, but uptake by the algae is more important in the pre-reservoir, particularly in the pH range 6.0 – 8.0. Algal sedimentation is enhanced by natural precipitants and flocculants, and by an appropriate phytoplankton community in the pre-reservoir. Algae that have a high sedimentation velocity, especially diatoms are favourable, whereas cyanobacteria are highly undesirable. Mass developments of zooplankton, especially filter feeders must be avoided in a pre-reservoir because of the high loss by grazing of phytoplankton and the associated nutrient remineralization. Both the desired phytoplankton structure and the absence of planktonic crustaceans can be achieved by an optimal water retention time within the pre-reservoir, which allows diatoms and other fast growing algae to grow but flushes away or precludes the development of slow-growing cyanobacteria and zooplankton. No predatory fish should be present in pre-reservoirs, assuring the control of zooplankton by planktivores.

Other important prerequisites for the optimal management of pre-reservoirs are:

(i) The mean depth should not exceed the depth of the euphotic zone z_{eu}. If it does, which is the case in most existing pre-reservoirs, surface release is an urgent necessity (Figure 38).
(ii) The bottom sediment must be removed at time intervals of 5-10 years.

The reduction of inorganic nitrogen by phytoplankton uptake and denitrification is low in pre-reservoirs because of their short retention times. Rapid siltation may also reduce the nutrient elimination capacity of pre-reservoirs (Paul, 2003).

Although pre-reservoirs are an important tool for reservoir water quality management, especially when the water entering a reservoir is of poor quality, they can be no substitute for remedial actions in the catchment area (Salvia-Castellvi et al., 2001).

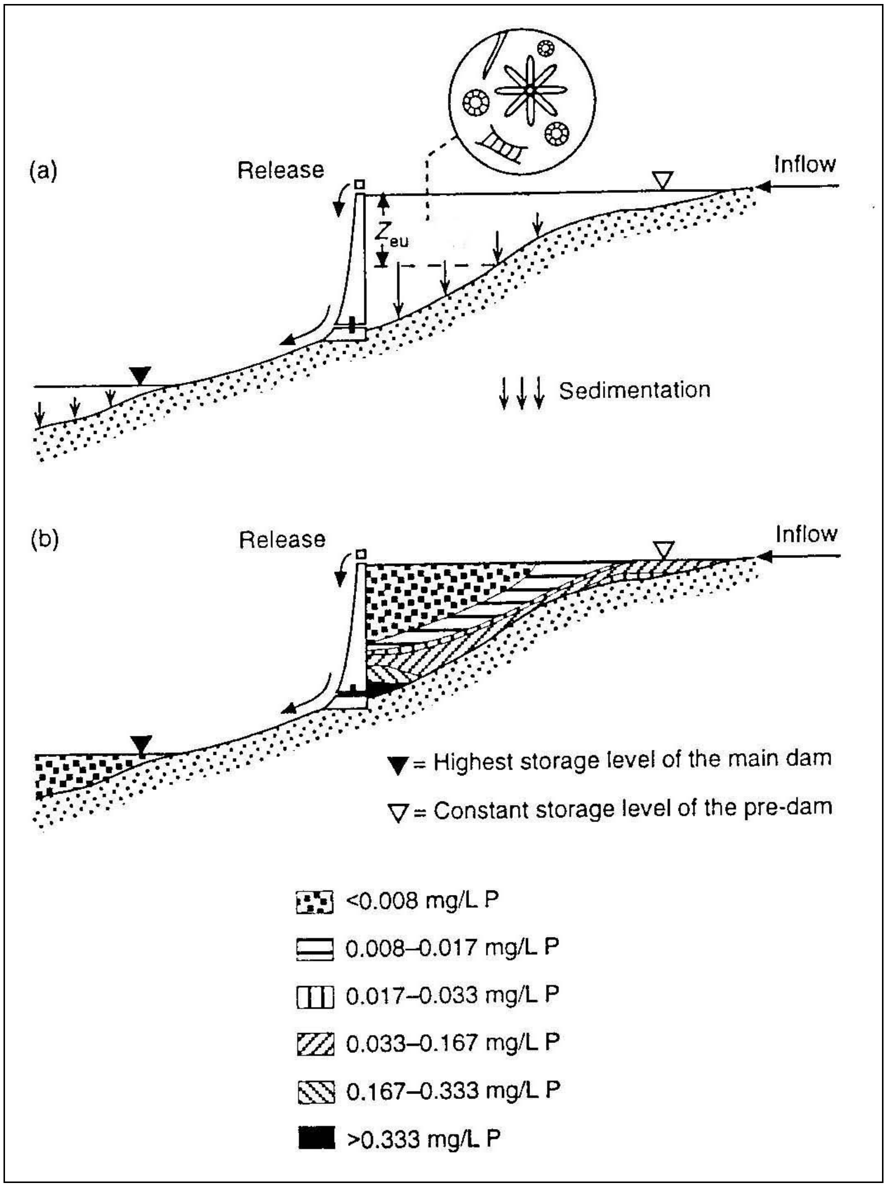


Figure 38. Schematic representation of the function of pre-reservoirs. (a) Orthophosphate uptake by phytoplankton within the euphotic zone, and sedimentation in the pre-reservoir and in the shallow inlet section of the main reservoir. (b) The typical resultant orthophosphate distribution (after Putz, 1995).

3.1.5.2.13 Modelling as a management tool

Modelling was discussed as an aid to prediction and management of eutrophication in Section 3.1.2.5. Modelling is used for many other purposes including the simulation of the effects of different management methods e.g. Priyantha et al., 1997 (the curtain method in Terauchi Dam Reservoir in Japan); Imteaz and Asaeda, 2000 (artificial mixing by the bubble plume method in Lake Calhoun, USA); Sahoo and Luketina, 2003; Abdelwahed et al., 2002; Schladow, 1993 (bubble plume design); Lewis et al., 2003 (artificial mixing and copper sulphate dosing in Myponga reservoir, Australia); McGinnis and Little, 1998 (Speece Cone).

3.1.3 Pollution by toxic substances

3.1.3.1 Introduction

"The solutions to the toxic chemical problems are much more complex and intractable than those associated with eutrophication and acidifying precipitation. Better wastewater treatment and agricultural practices can reverse eutrophication. Emission controls on power plants and vehicles are resulting in major reductions in H^+ or acid precursors emitted into the atmosphere of Western countries. In contrast, the number and variety of toxic chemicals that are subject to release in the environment is vast – nearly 80,000 synthetic organic chemicals plus a small number of toxic metals are now in daily use (Stumm et al., 1982)." (Kalff, 2002)

Toxic management of reservoirs is aimed at the protection of aquatic life and human health from impacts caused by the release of toxic substances into surface waters. The increasing release of toxicants into reservoirs (and lakes) is of great concern because of the unique physical and chemical characteristics of confined water. Toxic substances are retained longer in water and sediments than in flowing water, which increases the risk of exposure of toxicants in terms of concentration and duration, to both aquatic organisms and humans who depend on drinking water and food from lakes and reservoirs (Matsui, 1992).

3.1.3.2 Toxicity

Toxicity may be defined or classified in various ways, e.g. (Table 2)

TOXIC SUBSTANCES have one or more of three properties:
- Toxicity
- Bioaccumulation
- Persistence

TOXICITY	→ EFFECT
• Acute	→ Quick death.
• Sublethal/Chronic	→ Impairment of growth, breeding success, etc.
• Carcinogenicity	→ Impairment, death.
• Mutagenicity • Teratogenicity	→ Effects on future generations.
• Genotoxicity	→ Possible effects on future generations.

BIOACCUMULATION
- Capacity to enter food chain

PERSISTENCE
- Resistance to degradation under environmental conditions

VOLATILITY
- Ease of evaporation and atmospheric transport

Table 2. Properties of toxic substances (modified from Kaiser (1984) in Kalff (2002))

3.1.3.3 The fate of toxic substances in reservoirs

The chemical properties of the toxic substance and the physical and chemical properties of the lake or reservoir will determine the fate of a toxic substance. A continuing interaction between the toxic substance and the receiving system may lead to storage or concentration of toxic substances, producing conditions that may present a hazard. In addition the dynamics of reservoirs, specifically shorter retention times and greater transport potential, will lead to dilution and transport as a major mechanism of concentration change (Herricks, 1992). This will be discussed in more detail under the separate headings of inorganic and organic contaminants. Table 3 shows the processes that control the fate of toxic substances, and their effects.

Physical Characteristic	Primary Process or Processes That Control Toxic Substance Concentration	Effect Anticipated
Surface area	Photodegradation/activation	Concentration change, chemical transformation
Depth (maximum, average)	Photodegradation, Partitioning, Dilution, Storage	Concentration change, chemical transformation accumulation / bioconcentration
Volume	Dilution	Concentration change
Shoreline development, Shoreline length	Partitioning, Biotransformation, Storage	Concentration change, accumulation / bioconcentration
Temperature/Heat Budget	Chemical Reaction Rate, Biotransformation Rate	Concentration change, chemical transformation, accumulation / bioconcentration
Stratification/Mixing	Dilution, Transport, Isolation	Concentration change (seasonal cycles)
Retention Time	Dilution, Transport	Concentration change, distribution in basin
Currents	Dilution, Transport	Concentration change, distribution in basin
Color/Light Penetration	Photodegradation/activation	Concentration change, chemical transformation

Table 3. Physical characteristics of lakes and reservoirs, control processes and effects (after Herricks, 1992).

3.1.3.4 Inorganic toxic pollutants – heavy metals.

3.1.3.4.1 Types and sources

The heavy metals occurring in natural waters may be divided into two groups (Bren, 2001) (Figure 39). Bio metals of predominantly natural origin include Fe, Mn, Cu, Co, Zn, Mo, and V. Contaminants of predominantly anthropic origin include Hg, Cd, Cr, Pb, Sn, As and Ni. Both groups exist in water bodies in various forms. Heavy metals, depending on their chemical properties are found as suspensions and colloids, simple and complex hydrated cations and anions, mono- and polynuclear hydroxycomplexes, low and high-molecular complex compounds with inorganic and organic ligands varying in their structure and strength. The sources of contamination are numerous, diverse and difficult to identify (Jackson, 1992).

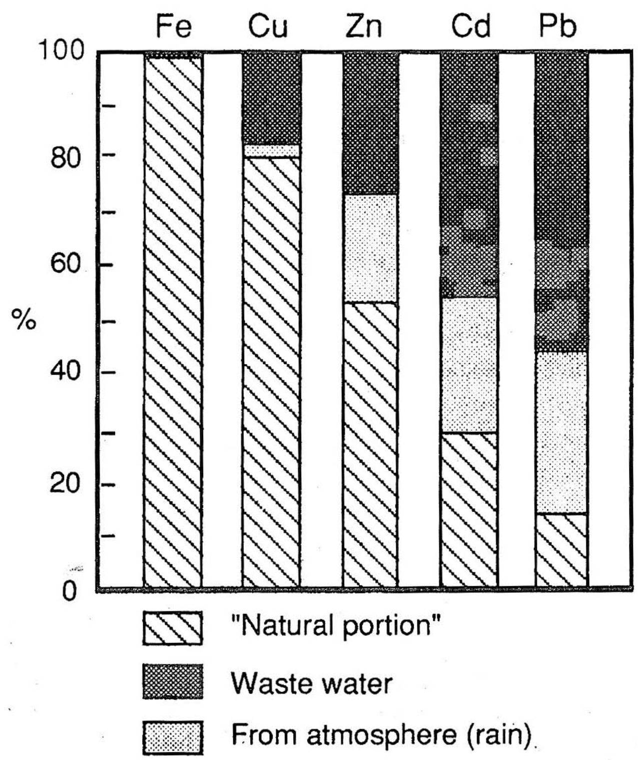


Figure 39. Proportions of natural and anthropogenic portions in the heavy metal load of a prealpine lake (from Stumm and Baccini, 1978, in Jackson 1992).

Natural processes

Natural physical and chemical processes such as weathering and leaching of soils and rocks are some of the main sources of "background" levels of metals in fresh waters. Surface waters in certain areas have relatively high concentrations of individual elements as a result of contact with particular soils or rocks (e.g. fluoride or selenium), which may be associated with health problems (Jackson, 1992).

Human activities

The main sources of heavy metals in water bodies other than natural processes are industrial wastes, irrigation system wastewaters, urban, industrial and domestic wastewater/sewage, stormwater runoff from urban areas (Tkalitch et al., 1996), mining and smelting of metals, use of pesticides containing heavy metals, fossil fuel combustion processes (Crutchfield, 2000; Lemley, 1997), pre-mixes to cattle fodders, active silts as fertilizers, and recirculation of solid waste (Bren, 2001; Jackson, 1992). (Table 4).

Average concentrations in μg/l

Industry	Cu	Cr	Ni	Zn	Cd
Meat processing	150	150	70	460	11
Fat rendering	220	210	280	3,890	6
Fish processing	240	230	140	1,590	14
Bakery	150	330	430	280	2
Miscellaneous foods	350	150	110	1,100	6
Brewery	410	60	40	470	5
Soft drinks and flavourings	2,040	180	220	2,990	3
Ice cream	2,700	50	110	780	31
Textile dyeing	37	820	250	500	30
Fur dressing and dyeing	7,040	20,140	740	1,730	115
Miscellaneous chemicals	160	280	100	800	27
Laundry	1,700	1,220	100	1,750	134
Car wash	180	140	190	920	18

Table 4. Metals in industrial wastewaters in New York (after Klein et al., 1974 in Jackson, 1992).

Many studies of heavy metal contamination of reservoir water and sediments due to human activities have been made These include the following: Avila-Perez et al., 1999, 2002 (Jose Antonio Alzate Reservoir, Mexico); Wang et al., 2003 (Guanting Reservoir, People's Republic of China); Linnik and Zubenko, 2000; Linnik, 2000, 2001 (Dnieper Reservoirs, Ukraine); Carvalho and Zanardi, 1997 (Billings Reservoir, Brazil); Arnason and Fletcher, 2003 (Patroon Reservoir, NY, USA); Audry et al., 2004 (Lot River Reservoirs, France); Moalla et al., 1998 (Lake Nasser, Egypt); Tkalitch et al., 1996 (Kranji Reservoir, Singapore); Zakova and Kockova, 1999 (Vranov and Nove Mlyny Reservoirs, Czech Republic); Juracek and Mau, 2003 (Tuttle Creek Lake, Kansas, USA); Loska and Wiechula, 2003 (Rybnik Reservoir, Poland).

In many areas, the atmosphere is also a significant route of transport of heavy metals entering a water body.

3.1.3.4.2 The fate of inorganic toxic pollutants in aquatic systems

- **Physical and chemical factors**

The **transport and distribution** of heavy metals in lakes and reservoirs are dependent on the physical and chemical characteristics of the receiving water body and the nature of the contaminants and their sources (Jackson, 1992). Unlike organic contaminants, which can be degraded to various degrees, heavy metals do not undergo such transformations. They are always present in aquatic ecosystems and redistribute only among different components (Linnik and Zubenko, 2000; Bren, 2001).

Physical characteristics such as temperature, colour, light penetration, flow dynamics and retention time of reservoirs may produce conditions that lead to the **storage or transformation** of metals.

The chemical form of metals is of great importance in determining their behaviour and impact in fresh waters (Jackson, 1992; Linnik. 2000; Rudd, 1995). **Speciation** is of special significance for biological availability and toxicity, different chemical species of the same element often having quite different biological effects (Chen et al., 1996; Schintu et al., 2000). Free (hydrated) metal ions are regarded as the most toxic form (Gouvea et al., 2005). Bound in complexes with inorganic, and particularly with naturally occurring organic substances, their toxicity is often decreased or completely suppressed (Linnik, 2000; Flemming and Travers, 1989). Many factors influence the chemical speciation of heavy metals (Wang et al., 2003) – sediment particle size, organic matter content, salinity, pH and redox potential. These factors are not completely understood and simple relationships are seldom found. Zhang et al. (2002) showed that the total concentrations of heavy metals in sediments were significantly linked to the particle size distribution. The finer particles showed higher concentrations due to increased surface area, higher clay minerals and organic matter content, and the presence of Fe-Mn oxide phases.

Complexation of metals with inorganic and organic ligands is of importance in transforming metals in aquatic systems (Gouvea et al., 2005). A significant amount is in association with either dissolved organic matter or with organic or inorganic particulates (Jackson, 1992). In particular, studies relating phytoplankton metabolic products to trace metals in natural aquatic systems are of interest due to the important role of phytoplankton as the first link in the aquatic food chain (Gouvea et al., 2005; Lombardi and Vieira, 1998).

Forstner and Wittmann (1983) in Jackson (1992) identified **complexation** and **flocculation** with organic matter, **co-precipitation** with hydrous iron and manganese oxides, and **uptake by biota** as being of particular importance for the **sedimentation** of heavy metals. Once in the sediment, redox reactions are important in determining their form and mobility, and may result in release of metals from particulates, resulting in renewed availability to biota and the potential for resuspension into the water column (Munk and Faure, 2004; Brandenberger et al., 2004; Carignan and Lean, 1991; Linnik and Zubenko, 2000). Most redox reactions tend to be biologically mediated by micro organisms (Jackson, 1992). Movements of metals back into the water column from the sediment may also occur as a result of **sediment resuspension** by physical influences (currents, waves), biological action (benthic animals) or human action (dredging, boating etc). Deposited sediments typically contain particulate and colloidal materials which exhibit high sorption affinity for trace metals (Elbaz-Poulichet et al., 1997; Canavan et al., 2000; Kneebone et al., 2002).

Water bodies with longer retention time (lakes and some reservoirs) accumulate heavy metals in their bottom sediments in considerable quantities (Linnik and Zubenko, 2000; Chen et al., 1996). This has both positive and negative implications. On the one hand, the bottom sediments promote self-purification of the overlying water because of heavy metal accumulation. On the other hand, this accumulation can be reversed and under certain conditions, bottom sediments can be a strong source of secondary pollution (Linnik, 2000). The release of heavy metals from the bottom sediments is promoted by a deficit in dissolved oxygen, a decrease in pH and redox-potential, and an increase in mineralization and dissolved organic matter concentration (Chen et al., 1996).

The mobility of heavy metals depends on the form in which they occur in the solid substrates and the pore solutions of the bottom sediments, as well as on the physico-chemical conditions that arise on the boundary of solid and liquid phases. Heavy metal flow from pore solutions is one of the most important methods of exchange between bottom sediments and water (Linnik and Zubanko, 2000).

Field data and laboratory experiments conducted by Linnik and Zubenko (2000) indicated that complexes of heavy metals with relatively low molecular weight play the principal role in secondary water pollution.

- **Biological factors**

Biota are central to the study of, and control of heavy metals and other inorganic toxic substances in reservoirs. Transport of metals is influenced by

  - uptake or adsorption by biota
  - binding to particles of biogenic origin, or
  - complexing by dissolved organic matter, especially of plant origin (Gouvea et al, 2005)

Transfer to, and storage in, sediment, is influenced by settling of biological particles with which the metals are associated (such as dead phytoplankton cells or zooplankton faeces) and biological action in sediments such as bioturbation or microbial action, which change mobility (Jackson, 1992).

The uptake and accumulation of trace elements in aquatic consumers is important in assessing the fate and effect of contaminants and food web dynamics (Klochenko and Medved, 1999; Allinson et al., 2002; Vinot and Pihan, 2005; Martinez-Tabche et al., 2001; Bolotova and Konovalov, 2002; Park and Curtis, 1997; Veinott et al., 2001; Tremblay et al., 1998a; Porvari, 1998; Ikingura and Akagi, 2003; Wong and Pak, 2004).

For many aquatic invertebrates, trophic transfer accounts for a major portion of total trace element accumulation (Luoma et al., 1992). In the field, the ecotoxicological approach is very difficult for evaluation of the impact of heavy metals on the aquatic environment due to the complexity of interrelationships between organisms and the ecosystem. However, field studies can enable assessment of the long term effect on organisms of heavy metals like copper; Vinot and Pihan (2005), for example, showed the bioavailability, bioaccumulation, biotransference and long-term toxicity of copper in the food chain of a freshwater ecosystem, the Mirgenbach Reservoir in France

Mercury in water may be converted by bacteria in the sediments into organic mercurials (Matilainen, 1995) particularly the highly toxic alkylmercurials, methyl- and dimethylmercury, which may, in turn be concentrated by fish and other organisms (Park and Curtis, 1997; Bodaly and Fudge, 1999; Becker and Bigham, 1995).

- **The problem of mercury in newly flooded reservoirs**

A large amount of literature is available on the problem of mercury in newly flooded reservoirs (most often those built for hydroelectric power generation), and its uptake in the form of methyl mercury (CH_3Hg^+), by aquatic organisms. Recent research has shown that there are three important sources of methyl mercury to lakes and reservoirs – internal production, inputs from watersheds that contain wetlands, and atmospheric inputs, with internal production being very important in reservoirs (Rudd, 1995).

Methyl mercury is a neurotoxin that damages the central nervous system in humans (Clarkson, 1990). Its uptake by fish is therefore a significant problem as this is the dominant pathway of Hg to humans. As of 2002, there were 2140 water bodies in 45 states of the USA that had advisories recommending against human consumption of Hg-contaminated fish (USEPA, 2003). The US Environmental Protection Agency (USEPA) has indicated that of all pollutants mentioned in the Clean Air Act, mercury has the greatest potential to impact on human health (Gray et al., 2005).

The creation of reservoirs, with the associated flooding of soils and vegetation, results in elevated concentrations of mercury in the newly formed water bodies. Natural and anthropogenic mercury is widely dispersed via atmospheric transport and accumulates in the vegetation and soils following wet and dry deposition (Grondin et al., 1995; Lucotte et al., 1995; Caldwell et al., 2000). The inundated soils and vegetation then release the mercury into the water column during flooding (Therriault and Schneider, 1998; Plourde et al., 1997; Tremblay et al., 1996; Porvari, 1998; Tremblay et al., 1998a; Montgomery et al., 2000).

Flooding also releases organic matter, providing nutrients for bacterial communities that methylate inorganic mercury (Therriault and Schneider, 1998; Porvari, 1998). Flooding of both podzolic and peat soil has been found to increase mercury methylation (Porvari and Verta, 1994; Rudd, 1995; Morrison and Therien, 1995).

Methylation by bacteria, followed by bioaccumulation via the food web (Hall et al. 1997) is the primary pathway for mercury accumulation in fish (Therriault and Schneider, 1998; Tremblay et al., 1998b). Invertebrates, which represent up to 90% of the diet of non-piscivorous fish, play a key role in their contamination (Tremblay et al., 1998b; Spry and Wiener, 1991; Plourde et al., 1997), with bottom feeding benthic insect larvae (Tremblay et al., 1998) and zooplankton feeding on periphyton and suspended particulate matter (Grondin et al., 1995) constituting the link between flooded soils and fish (Hall et al., 1997; Plourde et al., 1997; Tremblay and Lucotte, 1997; Tremblay et al., 1996; Bodaly and Fudge, 1999). Mercury concentrations have been found to be higher in larger, older fish (Therriault and Schneider, 1998), and also in species at the highest trophic levels of new reservoirs (Potter et al., 1975), reflecting the phenomenon of biomagnification.

Typically fish mercury levels rise rapidly within the first few years after impoundment, and remain high for periods ranging from 10-20 years for non-piscivorous species and for 20-30 years in piscivores (Schetagne et al., 2000; Therriault and Schneider, 1998; Tremblay et al., 1998a; Porvari, 1998). Depending on species and reservoir characteristics the maximum total mercury concentrations in reservoir fish reach levels 3-6 times higher than in fish found in natural surrounding lakes (Schetagne et al., 2000).

Whereas the phenomenon of elevated mercury levels in fish has been demonstrated in hydroelectric reservoirs in many different geographical regions, there is a lack of information from reservoirs in Africa. Four reservoirs in two different geographical areas in Tanzania were investigated (Ikingura and Akagi, 2003). In general, the fish from these reservoirs had very low mercury concentrations compared to those in reservoirs of similar age and temperature in other regions. This suggested a reservoir environment that has not been significantly impacted by mercury contamination from natural or anthropogenic sources.

3.1.3.4.3 The effect of inorganic contaminants on the biota

The effects and uptake of metals by aquatic organisms depends on a number of factors that influence the availability of the substance to the organism. The concentration of the metal ion is a basic factor, but not necessarily in absolute terms because different chemical forms will have different availabilities.

- **Phytoplankton**

The influence of heavy metals on phototrophic organisms, including algae, is registered at the level of cells, organisms and populations, manifesting in the disruption of photosynthetic activity, respiration and synthesis of proteins, nucleic acids, lipids, sugars and pigments (Klochenko and Medved, 2001).

Munawar et al. (1988) in Jackson (1992), group the effects of contaminants on algae as (i) community structure responses (changes in size and species composition) (ii) physiological responses related to biomass production with possible effects on food chains, and (iii) ultrastructural responses.

Figure 40 gives an indication of the physiological effect of metals on phytoplankton, showing decreased carbon assimilation (primary production) with the addition of metals.

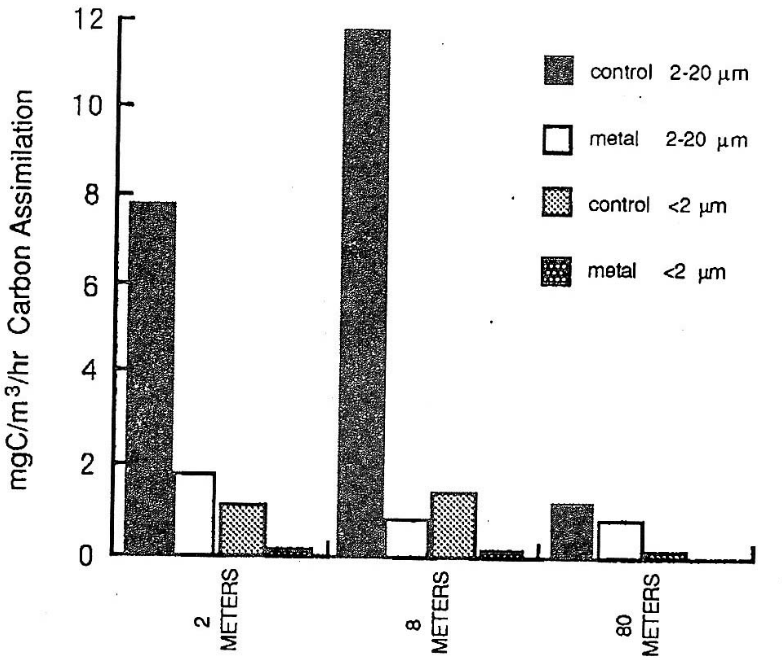


Figure 40. Effects of metal mixture additions in algal size fractions at different depths in Lake Ontario during a stratified period (from Munawar et al., 1998 in Jackson, 1992).

- **Zooplankton and zoobenthos**

Reduction in biomass production has been observed in zooplankton and zoobenthos as a result of exposure to heavy metals, both in terms of growth and reproduction (Vinot and Pihan, 2005), but there are considerable difficulties in working with natural communities and in identifying the effects of heavy metals as phenomena distinct from natural fluctuations (Jackson, 1992). Many effects have, however, been observed in the laboratory.

The causes of morphological abnormalities in microcrustaceans have not been sufficiently investigated (Elmoor-Loureiro, 2004), however, environments contaminated by industrial and urban waste have been associated with the occurrence of morphological anomalies in copepods and ostracods (Dias, 1999) and bioassays have shown that some environmental toxicants can produce morphological abnormalities in cladocerans (Shurin and Dodson, 1997; Otha et al., 1998).

- **Fish**

Sublethal effects of heavy metals on fish have been well described as a result of laboratory studies, fieldwork on direct responses to metal pollution being time consuming and complex. However, measurements of tissue metal concentrations in natural populations are made in many cases (Jackson, 1992). The concentrations of toxicants in fish tissue correlate with morphological and pathological-anatomical features. The initial manifestations are morphological changes in the size of vital organs (heart, spleen, liver, kidneys, gills and visceral organs), followed by specific pathologies (Bolotova and Konovalov, 2002).

This phenomenon of the effects of heavy metals on the biota will be discussed in relation to biomonitoring of pollutants in reservoirs.

3.1.3.4.4 Management

Aquatic ecosystems in general receive continuously increasing levels of heavy metals. In most cases, decreasing the impact of heavy metal pollution follows the transformation of toxic forms to less active forms (Linnik, 2000). The degree of detoxification depends on the intensity of several processes to reduce the concentration of the heavy metal in its most toxic form – free ions.

The processes are:

Adsorption on suspended particles.
Complexing (mainly with ligands of natural organic origin).
Sedimentation and co-precipitation
Hydrolysis and formation of low solubility compounds.
Accumulation and adsorption by biota.

It is difficult to determine the most important of these processes because each can play a dominant role in the transformation of heavy metal compounds according to the type of water body studied (Linnik, 2000).

Sediment analysis is very important in investigations of heavy metal burdens in aquatic systems (Chen et al., 1996), different methods being applied for different reasons (Scancar et al., 2000). Total concentrations (Garcia-Miragaya and Sosa, 1994; Ravanelli et al., 1997; Facetti et al., 1998) are determined by decomposition with strong acid solutions (Kada et al., 1994) or *aqua regia* digestion, (Muller et al., 1994; Ure, 1996; Gupta et al., 1996; Sterckemann et al., 1996) while for investigation of heavy metal bioavailability, various single extraction procedures are used (Houba et al., 1996; Tack and Verloo, 1996; McGrath, 1996). Sequential extraction is used in the assessment of heavy metal mobility (Cordos et al., 1995; Quevauviller et al., 1997; Maiz et al., 1997; Bodog et al., 1997) and chemical partitioning of sediments is also used to deduce the source and pathways by which natural and anthropogenic heavy metals have entered the environment (all the above in Scancar et al., 2000).

In general, the management of heavy metals and other toxic substances in reservoirs is combined in practice with the management of other problems, as part of an integrated approach to water quality (Jackson, 1992). The complexity of metal compounds and their interactions in aquatic systems makes the planning of practical management strategies very difficult.

- **Reduction of external loading**

As in the case of eutrophication management, this aspect of heavy metal management involves, to a large extent, watershed management, which is not within the scope of this report. However, the importance of source control cannot be overemphasised – the effects of remedial action are limited without it.

- **Intervention in the reservoir ecosystem**

Because of the multiple causes of water quality deterioration, the selection of techniques to improve the water quality is not as straightforward as it might seem. For example, aeration of the water column as a management technique is carried out primarily to prevent thermal stratification. The immobilization of metals would be only one of the aims – others would include the prevention of sulphide formation, inhibition of cyanobacterial blooms, elevation of oxygen to prevent fish kills, and reduction of internal phosphorus recycling, but other consequences would include the reduction of transparency, elimination of cold water habitats and the ruling out of depth-selective withdrawals as a hydrological management option. Other aeration techniques which were discussed for eutrophication management could also be used, always with the proviso that the characteristics of each problem would be vital in determining the effectiveness of the techniques chosen.

Sediment removal may be used to reduce the release the accumulated metals back into the water column. It may be successful at one level but dredging has the disadvantage of resuspending sediments, risking secondary pollution of the water column, and temporarily destroying benthic communities (Davidson et al., 2005). Disposal of heavy-metal containing dredged sediment also presents a problem. Again, sediment removal techniques discussed for eutrophication management may be applicable in certain cases.

- **Modelling**

Models such as mass balances aid in an understanding of the structure of a system and in estimating the effectiveness of management action. Residence time models which use physical and chemical data are of use in predicting the behaviour of toxic metals in lakes – the time that any given input takes to be flushed from the system or immobilized in the sediments. Thomas et al. (1988) in Jackson (1992) draw attention to the difficulty of associating observed effects with particular contaminants in the modelling of complex lake and reservoir systems. They suggest that for highly persistent substances such as heavy metals, the time lag between input and observed results may be so long as to prevent effective management action.

3.1.3.5 Organic toxic pollutants

3.1.3.5.1 Types of organic toxic pollutants

- **Pesticides/herbicides**

  These chemicals are, by definition, toxic, because they are used to control or kill plants and/or animals. (Allan, 1992)

  Three main types of pesticides have been, or are still being widely used:

  - Organochlorine pesticides - These are persistent toxins that bioaccumulate and are widely used. Being lipophilic, with slow chemical and biological degradation, they may be concentrated in organisms and biomagnified along the food chain. They are the most serious organic pesticides in terms of impact on aquatic ecosystems

  - Organophosphorus pesticides – Mainly insecticides, this group is relatively immobile in comparison with the organochlorines, are rapidly degraded in the environment, and are seldom detectable in aquatic ecosystems except locally after application and immediate runoff. However, several studies have demonstrated their toxicity, especially their ability to inhibit acetylcholinesterase.

  - Carbamates – Like organophosphorus pesticides, these are relatively immobile, are rapidly degraded and are seldom detected in aquatic ecosystems.

- **Other organic toxic pollutants (persistent organic pollutants - POPs)**

  It is estimated that there are more than 4,000,000 known organic compounds, a high proportion of them being synthetic, and less than 60,000 being frequently used. Three groups with important ecotoxic and mutagenic effects and widespread distribution (Catoggio, 1992) are:

  - Polycyclic aromatic hydrocarbons (PAHs) - These are the most toxic hydrocarbons. They are formed during the distillation of coal and are also found as minor components of exhaust gases from diesel motors and cigarette smoke. Several are proven carcinogens in humans.

    They are all solid and scarcely soluble in water. Almost immediately after being formed and released into the air, they adsorb onto airborne particulate matter where they are much more stable and resistant to oxidation and nitration reactions to which they would otherwise be quite sensitive due to photochemical processes in the atmosphere.

- Polychlorinated biphenyls (PCBs) – These have been used until recently in developed countries as isolating fluids in condensers for electric power transmission lines. Unfortunately they remain in use in underdeveloped and developing countries.

  Unavoidably, they slowly volatilize, sorbing on airborne particulate matter. This is deposited by rainfall whereby they enter aquatic ecosystems where, because of their hydrophobicity, they sediment out, and may be ingested. They are selectively accumulated and bioconcentrated, especially by benthic fauna, thus entering the food chain.

- Chlorinated dioxins – There are 75 dibenzo-p-dioxins whose chlorinated derivatives (PCDDs) are known. Highly toxic, as are their homologues - chlorinated benzofurans (PCDFs) - they are responsible for some serious pollution accidents in modern times.

3.1.3.5.2 Sources of contaminants

- **Pesticides/herbicides**

Organic pesticides enter freshwater systems through a variety of sources, particularly agricultural runoff. Domestic and municipal use of organic pesticides and the associated urban runoff is another important source. In addition, it has been increasingly realised that the atmosphere can be a major source of organic pesticides to large waterbodies by means of wet or dry deposition (Van Dijk and Guicherit, 1999; Guicherit et al., 1999). They may also be applied directly to aquatic systems for the control of weeds and insects, for example (Allan, 1992). Herbicides, extensively employed in agriculture for the control of weeds contaminate ground and surface waters. They represent about 50% of all pesticides used in many countries.

- **Other organic toxic substances**

Organic pollutants other than pesticides/herbicides, originate from a large variety of sources, too numerous to mention. They are all anthropogenic, thus differing from inorganic toxicants, and are produced, for example, by industry (Guzzella, 1997; Bodzek et al., 1998), agricultural activities, wastewater treatment plants (Pham et al., 1999) and urban runoff. Their input may be continuous (e.g. sewers), discontinuous (e.g. spillages or other accidents (Unlu and Demirekler, 2000) or periodical (e.g. emissions from stacks or effluents from technological processes). They may be toxic in themselves, or in their immediate products or by-products, products of their degradation (physical, chemical or biological), or as final waste forms (Catoggio, 1992).

3.1.3.5.3 The fate of organic toxic pollutants in aquatic systems

- **Transport and transformation**
  When organic toxic substances enter an aquatic system, they are subject to natural transport processes that result in their partition and dispersion among various components of the system (Figure 41) and to chemical transformation that alters them into other substances. Transportation and transformation are the sum of the following processes; votalization, photolysis, hydrolysis, biological degradation, exchange with the sediments and dispersion (Matsui, 1992; Galiulin et al., 2005).

  - Volatility, as characterised by vapour pressure, is important in controlling certain types of toxic substances (e.g. DDT and PCBs), which migrate significantly into lakes and reservoirs from the surface of the water where they are deposited by rainfall. Aqueous solubility is important because it provides an upper limit to the extent of incorporation of a substance into an aquatic environment, and also indicates hydrophobicity and thus the potential for transfer into lipid phases of aquatic organisms (Matsui, 1992).

  - Photolysis takes place with chemical molecules which can absorb light energy, while

  - Hydrolysis of chemical molecules usually results in the introduction of a hydroxyl radicle (-OH) into the molecules and depends on pH.

  - Biological degradation includes
    (i) microbial degradation (Galiulin et al., 2005),
    (ii) breakdown by photoautotrophs and
    (iii) metabolic transformation by higher aquatic organisms.

    Information on (ii) and (iii) is very limited (Matsui, 1992).

  - Sediment exchanges involving sorption of organic substances onto or into sediments is an important phenomenon in aquatic environments (Guzzella, 1997). The fate of persistent organic pesticides (which have a low solubility) in aquatic ecosystems is highly dependent on sorption to particulates (Galiulin, 2005), the key process involved in their physical transport and determining their degree of bioavailability, and thus eventual bioaccumulation and effects.

    Sediments can act as a sink for sorbed materials, removing them from the water, and certain types are retained in the sediments for long periods of time. However, others are often desorbed for various reasons and can thus become a renewed source of pollution.
  - Dispersion is an extremely important factor in the consideration of toxic substances in aquatic systems. In the past, mistakes have been made with regards pollution control due to lack of knowledge of the biological factors influencing the transmission of toxic substances in food chains.

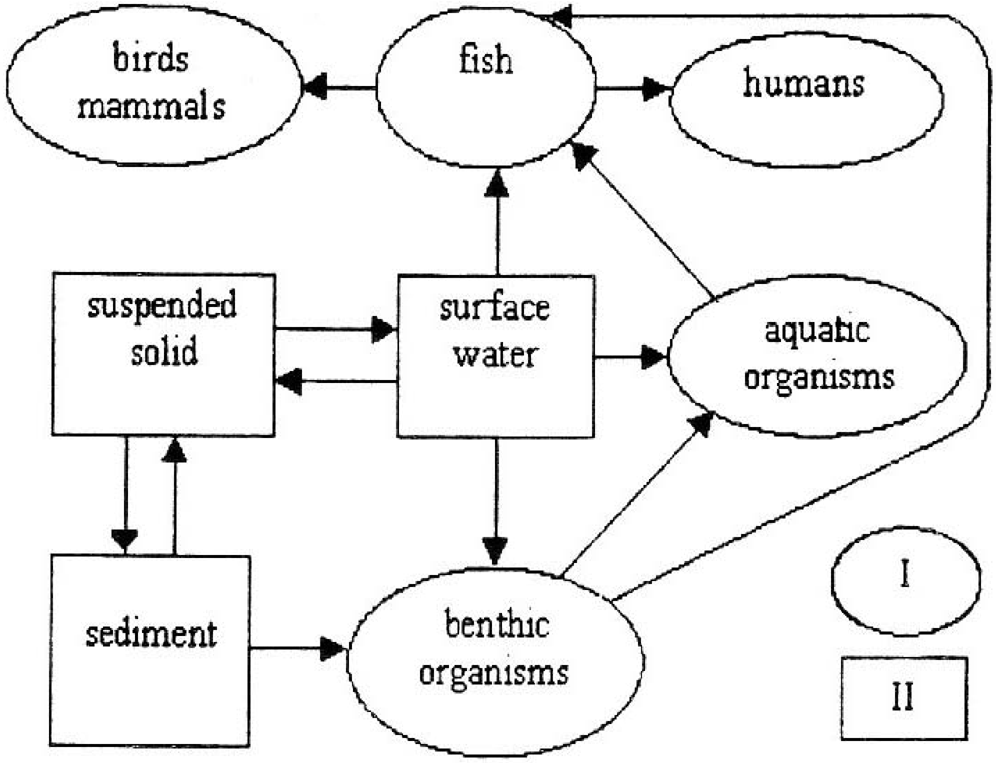


Figure 41. Simplified scheme of POC (persistent organochlorinated compounds) transformation in a biogeochemical food web in an aquatic ecosystem. (I= receptor, II = compartment) (From Galiulin et al., 2005).

Predicting the fate of organic pollutants associated with fine-grained particulates is far more difficult than when they are in solution because the dynamics of small particles are governed by a host of physical, chemical and biological processes. Once associated with a particulate, the fate of an organic pollutant is influenced by many and often interacting processes (Allan, 1992).

- Sorption/desorption
- Flocculation/deflocculation
- Precipitation and co-precipitation
- Degradation and transformation
- Bioaccumulation and biomagnification
- Short-term transport events (floods, wind)
- Sedimentation rates
- Resuspension episodes
- Mixing
- Pore-water release
- Mineralization
- Burial

- **Bioaccumulation, Bioconcentration and Biomagnification**

  Bioaccumulation is the process by which a substance is taken up by aquatic organisms, both from the water and through food, whereas **bioconcentration** is the process by which a substance is absorbed from water through gills or epithelial tissues and is concentrated in the body. Bioaccumulation of substances through food chains exhibits increasing concentration in organisms related to their trophic status. **Biomagnification** results in organisms of higher trophic levels (predatory water birds or fish) potentially being at higher risk. However, biomagnification depends to a great extent on toxicokinetics, which is the metabolic rate and excretion rate of a toxic substance in organisms. Some persistent substances such as organochlorine pesticides do not universally increase along food chains, the hazardous effects depending on both the species and the toxic chemical due to toxicokinetics (Matsui, 1992).

3.1.3.5.4 Management

Undoubtedly the best way of managing the release of toxic organic substances into aquatic ecosystems is through the reduction of their use, or through the use of technologies that do not generate them. This is however idealistic as the elimination of most toxic substances would be extremely problematic if not impossible.

Finding ways of reducing the dangerous use and effects of organic pesticides and/or most other toxic pollutants is the responsibility of governments, agricultural, industrial and environmental engineers and the like. Limnologists have a role to play in producing scientific evidence of the effects on toxic pollutants on freshwater ecosystems.

3.1.3.6 Biomonitoring as a management tool for inorganic and organic toxic pollution

This is discussed under the general heading of biomonitoring in water quality assessments, in Section 4.1.

3.1.4 Non-toxic organic pollution

3.1.4.1 Introduction

To some degree, the effects of organic pollution and eutrophication on reservoirs are similar. In both cases the stimulating agent is organic matter, the major difference being that in eutrophic waterbodies, this is mostly produced within the system (predominantly by algae), whereas in organically polluted waters, the source is external (predominantly untreated sewage from human settlements), or less commonly, metabolic products from aquaculture (Straškraba and Tundisi, 1999).

From a management perspective it is important to differentiate between the causes of organic pollution because different control measures are employed to solve the accompanying problems.

3.1.4.2 Sources of organic pollutants in reservoirs

Two main potential sources of organic pollution can be distinguished.

3.1.4.2.1 Human settlements

A primary large-scale consequence of population growth and urbanization is the high input of untreated sewage into rivers (e.g. the uMsinduze River in KZN) and thence reservoirs, resulting in an increase in decomposable organic matter and nutrients. In addition, other ingredients including microbial contaminants, heavy metals and pesticides are often present in urban wastewater, which includes domestic wastes and urban runoff. Depending on the sources and degree of prior treatment, if any, organic matter of this origin may be only partially degradable (Straškraba and Tundisi, 1999).

3.1.4.2.2 Aquaculture

Enrichment and degradation of aquatic ecosystems often occur in the vicinity of fish farms (Diaz et al., 2001). Wastes include metabolic products (faecal and excretory material) and uneaten food, which enter the aquatic environment directly. The nutrient content of the artificial feeds, the type of sediment, the existing trophic state and the morphometry of the reservoir together with retention time, are some of the variables that are required to predict the impact of fish farming on water quality (Diaz et al., 2001).

In addition to the expected consequences of organic loading (i.e. reduction in dissolved oxygen concentration and increased BOD, TSS and nutrients), the use of therapeutic agents and chemicals for aquacultural purposes are also causes for concern (Fidalgo, 2002). There is also an increase in numbers of indicator bacteria as well as the presence of antibiotic resistant bacteria, due to the treatment of diseased fish with antimicrobial drugs (Niemi and Taipalinen, 1982).

3.1.4.3 Fate and effects of non-toxic organic pollutants

Bacterial and viral contaminants

Outbreaks of water-borne diseases via public water supplies continue to be reported in developed countries even though there is increased awareness of, and treatment for, pathogen contamination. (Gibson et al., 1998; Herwaldt et al., 1992; Hawkins et al., 2000; Lisle and Rose, 1995; MacKenzie et al., 1994; Moore et al., 1994; Payment et al., 1997) To obtain a true assessment of the overall pathogen risk, it is necessary to understand the critical variables controlling pathogen fate and distribution in each part of the water supply system including the reservoirs (Brookes et al., 2004).

The processes of dispersion, dilution, horizontal and vertical transport determine the distribution of pathogens in reservoirs, the settling of pathogen particles operating in conjunction with these hydrodynamic processes (Brookes et al., 2004). The riverine inflow is considered to be the major source of pathogens, the behaviour of these inflows thus being of particular importance (Walker and Stedinger, 1999).

Detailed knowledge regarding the persistence of pathogens in reservoirs is necessary to estimate their contribution to health risk (Meays et al., 2004). This is a function of both survival and transport. Different pathogens persist for different lengths of time, the major mode of inactivation or mortality varying significantly (Brookes et al., 2004). The main factors controlling inactivation are light (visible and UV radiation) and temperature, others being salinity, hydrostatic pressure and predation (Simek et al., 2001). The grazing of pathogens by aquatic invertebrates has several implications. It may change the settling behaviour if excreted intact in a faecal pellet (Turner, 2002); however the passage through the gut of a predator may render it nonviable (Weislo and Chrost, 2000). Filter-feeding molluscan shellfish can accumulate waterborne pathogens in their tissue with possible significant human health implications (Graczyk et al., 1999; Fayer et al., 1998).

The prolonged survival and accumulation of microorganisms in sediments and the likelihood of being desorbed by dilution or water turbulence indicates that sediments as well as surface waters should be assessed when estimating potential health risks (Davies et al., 1995).

3.1.4.4 Management

3.1.4.4.1 Reservoirs as barriers to pathogen transport

The first control point for pathogen risk management is in minimizing the concentrations entering reservoirs from rivers and catchments (Venter et al., 1997). Sewage pollution is potentially the most easily managed of urban problems, the provision of sanitary sewers being one of the first actions taken to manage urban catchments. In developed countries, pollutants originating from settlements are, for the most part, reduced to acceptable limits. However, recreational activities may be a major uncontrolled source, the mushrooming of holiday homes on lake and reservoir shores, with accompanying waste disposal problems becoming an ever-increasing problem (Straškraba and Tundisi, 1999). For example, overflow from septic tanks enters surface waters via groundwater, which produces a separate challenge to catchment managers and ecologists. Proposals for high-density housing developments on reservoir margins (e.g. Midmar Dam in KZN) are clearly irrational and illogical.

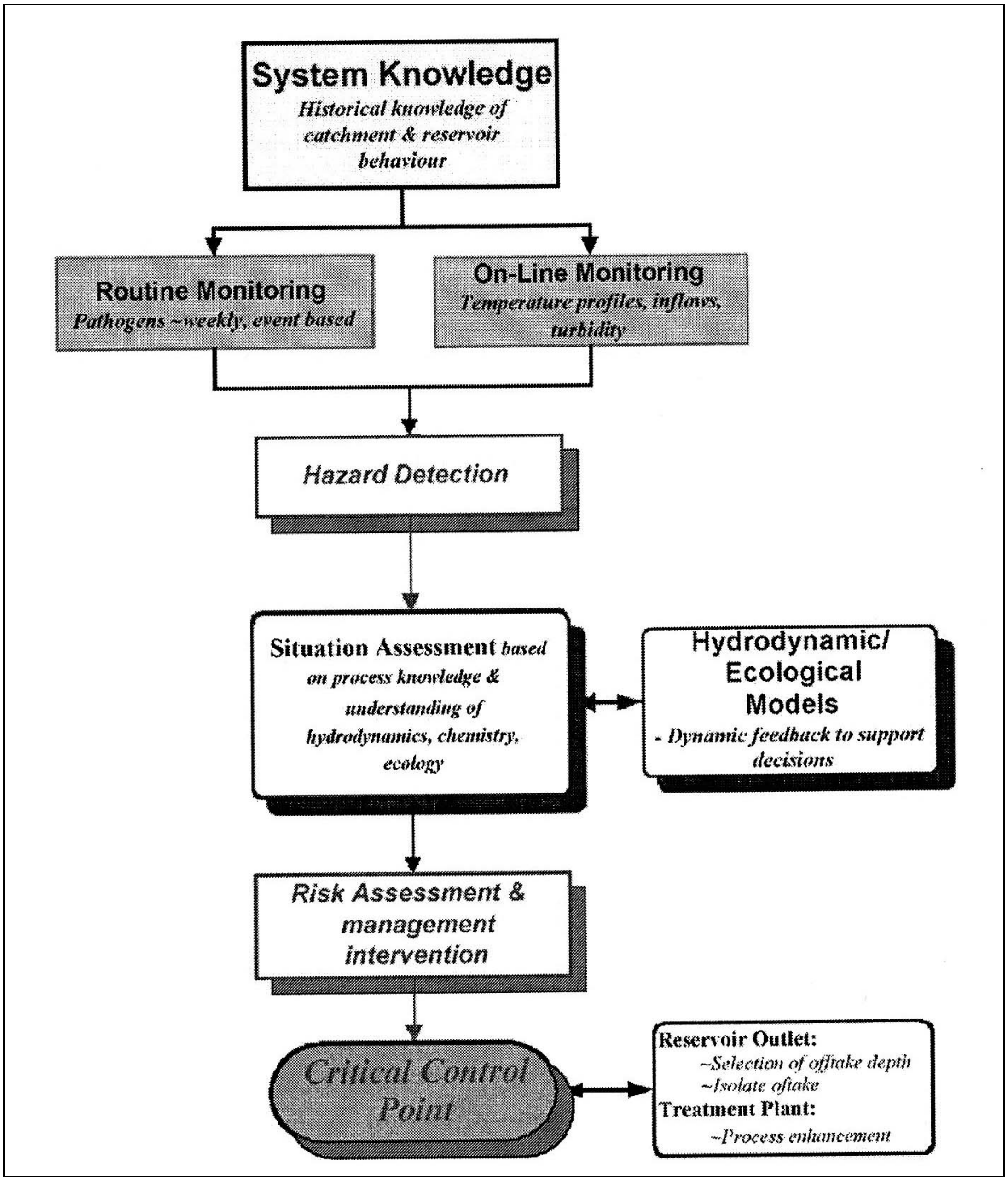


Figure 42. Conceptual framework for pathogen monitoring and risk assessment in reservoirs. (After Brookes et al., 2004).

The reservoir itself is the second control point, acting as it does, as a barrier to pathogen transport (Brookes et al., 2004). Of primary concern is *Cryptosporidium* because of its longevity (Medema et al., 1997) and the resistance of oocysts to treatment processes (Robertson et al., 1992). However, Van Breeman et al. (1998) and Bertolucci (1998) both reported significant decreases in *Giardia* and *Cryptosporidium* concentrations during water storage in reservoirs.

Time scales of the major processes in pathogen inactivation are of paramount importance in determining risk reduction in reservoirs. These are:

- the time taken for the inflow to reach the off-take point after entering the reservoir,
- the time taken for particles in the surface layer to sink to the bottom of the reservoir,
- the time taken for pathogens (e.g. oocysts) to become inactive after exposure to water of a certain temperature, and
- the time taken for pathogens (e.g. oocysts) to receive a fatal dose of UV radiation (Clancy et al., 2000; Craik et al, 2001) (Figure 42).

3.1.4.4.2 Use of surrogates to estimate pathogen risk

Indicator bacteria such as faecal coliforms, faecal streptococci and *E. coli* are themselves usually not pathogenic. They are used because they are much easier and less costly to detect and enumerate than the pathogens themselves (Meays et al., 2004). Traditionally, bacterial indicators of faecal contamination, notably faecal coliforms and enterococci, have been used to assess the microbial quality of water sources (Thurman et al., 1998). However, it is now apparent that these bacterial indicators are not suitable for assessing the risk posed by protozoan pathogens (e.g. *Cryptosporidium* and *Giardia*) and some enteric viruses. Several outbreaks of cryptosporidiosis have been documented where the water quality met microbiological standards based on bacterial indicators (MacKenzie et al., 1994; Lisle and Rose, 1995).

Various bacteriophages have been proposed as indicator organisms for enteric viruses in freshwater (Havelaar et al., 1993; Armon and Kott, 1995), and it has also been suggested that spores of *Clostridium perfringens* are good indicators of human faecal contamination (Payment and Franco, 1993; Ferguson et al., 1996). However these two methods need to be used with caution as they are not suitable in some circumstances. Other potential surrogates for pathogens include particle counting and turbidity, but these also have limitations, especially in waters of high mineral turbidity.

Thus, while no single water quality indicator can reliably assess the bacterial, protozoan and viral contamination of aquatic environments in all circumstances, it is feasible that a suite of surrogates may be identified that will estimate levels of microbial contamination within defined circumstances (such as within a storage reservoir with well characterised inputs) (Brookes et al., 2004).

It is widely accepted that the potabilization of river water requires the application of multiple barriers to ensure microbiologically safe drinking water (Lambert et al., 1998). Storage reservoirs are an important preliminary treatment step because of self-purification - however, additional barriers are needed. It is necessary to determine the elimination rate of each consecutive water treatment step e.g. storage, coagulation, filtration, ozonation etc., with suitable surrogate parameters.

Bacterial source tracking includes several methodologies used to determine sources of faecal bacteria (wildlife (mostly waterfowl), humans, and domestic stock) from environmental samples (Meays et al., 2004). These fall into 3 main groups, molecular, biochemical and chemical. There is currently no standard method which has been adopted for source tracking but it is a rapidly growing area of research and technology development.

3.1.5 Salinization

3.1.5.1 Introduction

Salinization is the process that increases the salinity of inland waters. Two types, natural and anthropogenic can be distinguished (Williams, 1999).

Natural (primary) salinization

This is essentially restricted to closed (endorheic) drainage basins in semi-arid and arid regions of the world (Williams, 1999). This occurs when the salts deposited in rainwater (mainly sodium, chloride and sulphate) accumulate following solar evaporation (Davies and Day, 1998). High concentrations of salts in freshwater may also be caused by a number of other factors such as wind-borne sea-spray, groundwater stores of "fossilised" sea water, sea salt stored in rocks, and easily weathered rocks that naturally contain high concentrations of mineral ions e.g. the Malmesbury Shales of the Western Cape (Davies and Day, 1998).

Over geological time, a natural equilibrium has been established in inland surface waters between salt inputs, salt retained for shorter or longer periods, and salt ultimately lost by drainage and it is likely that human activities have not disturbed this equilibrium to any great extent (Williams, 1999).

Secondary salinization also occurs mostly in arid and semi-arid regions and is caused by human activity. This is discussed in this section.

The increasing salinity of saline natural lakes will not be considered in this report.

3.1.5.2 Causes

Major causes of salinization of freshwaters are catchment activities that alter hydrological balances and mobilize underground salt, the most significant and devastating of these resulting from agricultural activities.

Long term irrigation carried out in arid areas and/or in areas where the rocks and soils have high concentrations of minerals, result in human-induced salinization (Straškraba and Tundisi, 1999; Davies and Day, 1998). During irrigation, particularly spray-irrigation, much of the water is lost by evaporation before it enters the soil, leaving behind the salts it contained. This accumulates with time, affecting agriculture, and leaching out into rivers and thus lakes and reservoirs. In regions where the water has a naturally higher salt load, the process is exacerbated (Davies and Day, 1998).

The felling of large eucalyptus trees in Australia caused serious terrestrial salinization in some areas (Williams, 1999; Davies and Day, 1998). Felling led to a rise in the water table as the deep-rooted vegetation was replaced with shallow rooted crops with lower rates of transpiration. When the water table is not far below the surface, capillary action increases the effects of evaporation, leaving a layer of salts on the soil surface after evaporation. As with irrigation, the soil becomes less and less able to support crops, eventually leading to desertification.

Other human activities may also increase the salinity of freshwater to varying degrees. Sewage purification subjects the water to evaporative concentration, particularly significant in efforts to recycle treated effluent. Saline industrial and mine effluents are a problem in many areas (Davies and Day, 1998).

Examples of salinization have been documented in a number of arid areas, predominantly Australia (Boulton and Brock, 1999; Macumber, 1991), but also Egypt (Saad, 1999), Ethiopia (Gebre-Marium, 1999) and South Africa (Davies and Day (1998).

3.1.5.3 Effects

The extent and importance of salinization as a global threat has been greatly underestimated (Williams, 1999). The effects are usually deleterious and often irreparable.

The effects of even small rises in the salinity of freshwater lakes and reservoirs can be profound because the halotolerance of the freshwater biota is more limited than that of salt lakes, frequently leading to an initial disappearance of macrophytes and riparian trees. The overall biological response is a decrease in biodiversity – the replacement of the halosensitive biota with a halotolerant one.

The salinization of lakes and reservoirs is closely associated with the salinization of rivers and streams whose salt load is discharged into these waterbodies. Several authors have drawn attention to the hazards posed by salinization of streams and rivers to lakes, reservoirs and wetlands (Williams, 1987; Davies and Day, 1998).

Economic losses include a loss in the ability of these affected waters to serve as supplies for domestic, agricultural and other uses (Williams, 1999). A series of impacts on human health when water of increased salinity is used for drinking, have been observed. One consequence of large-scale salinization is the increase in blood pressure rates and consequent renal disease (Straškraba and Tundisi, 1999).

3.1.5.4 Management

Prevention of further salinization is the most important management priority, as rehabilitation and management of waters already salinized is much more difficult and often not possible. Integrated catchment management is recommended as the only really viable method of addressing the problem (Williams, 1999).

Because freshwater salinization is taking place at an alarming rate, especially in arid and semi-arid areas where freshwater is in short supply anyway, effective, informed water resource management is essential. In some dryland countries, salinization is regarded as the single most important threat to the natural character of freshwater lakes and the viability of water resources (Davies and Day, 1998).

Climatic change will impact lakes worldwide, but its greatest impacts are likely to be on dryland aquatic ecosystems (Williams, 2000).

3.1.5.5 Salinisation of inland waters in South Africa

South Africa, being arid or semi-arid over large areas, has a serious salinization problem in many regions. In fact, salinization of rivers is recognised as one of the major threats to South African water resources: the water quality of many rivers e.g. the Berg and Breede rivers in the western Cape and Sundays and Fish rivers in the eastern Cape, is declining rapidly as a result of irrigation-induced salinization (Davies and Day, 1998).

Human manipulation of river flow is adding to salinization in some areas. For example, the Pongolapoort Dam, by preventing normal floods in the Phongola River below the dam results in accumulating salts in the floodplain pans. In addition, the Makatini Flats Irrigation Scheme in the Phongola river valley overlies substrata containing trapped and "fossilised" sea-water. The probability exists that salinization will seriously impact on agriculture in the region sometime in the future.

Although agricultural activities have the greatest effect on salinization worldwide, urban and industrial effluents may also have a serious impact on water resources. Large effluent loads are disposed of into the Vaal Dam where the concentration of TDS (Total Dissolved Solids) is increasing every year. About 60% of the salt load entering the Vaal Barrage is produced by only four mines. In addition, the Buffalo, Mkhuze, Phongola, Wasbank, Mfolozi and Tugela rivers are also receiving saline mine effluents to a greater or lesser degree (Davies and Day, 1998).

3.1.6 Acidification

3.1.6.1 Introduction

The dominant sources of air pollution are connected with the use of fossil fuels for energy production. However, the situation is complicated with many "primary pollutants " reacting in the atmosphere to produce new "secondary pollutants". Pollution of the atmosphere has a series of well-documented effects ranging from the local phenomenon of smog, to the regional effects of acidification caused by "acid rain", and the global effects of the "greenhouse effect" which may, in time, totally change the living conditions on earth (Fenger, 1997).

3.1.6.2 Causes of lake/reservoir acidification

Air pollution by sulphates and nitrates (which form sulphuric and nitric acid, and which may be transported over thousands of kilometres), leads to general acidification of precipitation. The emission rates of these two gases into the atmosphere exceed natural rates on a global scale by about 4 and 8 times, respectively (Galloway, 1996; 1998). In some regions of the world, rates are 100 times greater than the natural rates (Galloway, 2001). Unpolluted rain has a pH of 5-6, while in Europe, typical values are about 4.5 and occasional values as low as 3 have been registered (Fenger, 1997).

3.1.6.3 Effects of acidification on lake/reservoir ecosystems

3.1.6.3.1 Effects on water chemistry

A low pH results when hydrogen carbonate is converted into free CO_2, as shown by the following equation

$$\mathbf{CO_3^{2-} + H^+ = HCO^{3-} + H^+ = H_2O + CO_2}$$

This change leads to a suite of alterations in water chemistry, particularly increases in solubility of metal ions. Elevated concentrations of many metal ions, especially aluminium, iron and manganese (but also more toxic ions such as cadmium, copper, zinc and mercury) are therefore observed. This impacts widely on aquatic biota, since the free ions are generally more toxic to aquatic organisms than complex ions. (Jorgensen, 1997).

3.1.6.3.2 Effects on biota

All biological processes have a pH-optimum, which is usually in the range 6-8. This implies that plant growth, microbiological decomposition, nitrification and denitrification are all influenced by pH (Jorgensen, 1997). The general trend with decreasing pH is a decrease in species diversity, reduction in algal species diversity being one of the most immediate consequences of declining pH.

Several studies have indicated that cladoceran species amongst the zooplankton are generally sensitive to acute acid conditions, *Bosmina*, being more acid-tolerant, often replacing *Daphnia* species (Jorgensen, 1997). Copepods are less sensitive than cladocerans and are often representative of acidic waters (Brett, 1989). Effects on zooplankton physiology are reduced filtering, haemoglobin loss, reduced oxygen and sodium uptake and gill tissue damage. Long-term effects are reduced brood size and egg fertility, and delayed reproduction (Jorgensen, 1997).

In contrast to fish, some macroinvertebrates are able to tolerate acid exposure for a few days, and are accordingly often used as bioindicators for acidity. Many insect larvae show a wide range of tolerance, some being able to survive at pH 3.5 for long periods. Some mayfly larvae have been found to be sensitive to acidity, while some species of dragonfly, damselfly, caddis fly and stone fly larvae are typically well represented in acidic waters. Low pH affects macroinvertebrates through several toxic mechanisms such as reduced gas exchange, impaired calcium metabolism (significant in molluscs), element deficiencies and toxic effects of aluminium ions (Hermann 1987).

3.1.6.3.3 Effects on fish

The tolerance of fish to pH conditions is summarized in Table 5.

pH-range	Effect
3.0 - 3.5	Unlikely that any fish can survive more than a few hours
3.5 - 4.0	Lethal to salmonoids. Some other fish species might survive in this range, presumably after a period of acclimation to slightly higher pH
4.0 - 4.5	Harmful to salmonoids, bream, goldfish and carp, although the resistance to this pH increases with the size and age
4.5 - 5.0	Likely to be harmful to eggs and fry of salmonoids. Harmful also to adult salmonoids and carp at low calcium, sodium and/or chloride concentration
5.0 - 6.0	Unlikely to be harmful, unless concentration of free CO_2 is greater than 20 mg l^{-1} or the water contains freshly precipitated $Fe(OH)_3$
6.0 - 6.5	Harmless unless concentration of free CO_2 > 100 mg l-1

Table 5. Effects of pH on fish (from Alabaster and Lloyd, 1980 in Jorgensen, 1997).

A characteristic of damaged fish populations in acidified lakes is not just the absence of fish, but the nature of the population structure (Harvey, 1975). Annual age and size classes are sometimes grossly depleted or even entirely absent.

3.1.6.4 Management

3.1.6.4.1 Liming

Liming is the most commonly applied ecotechnological method for reduction and neutralization of acidification in lakes and reservoirs. (Jorgensen 1997). The efficiency of the liming process is dependent on the contact time and the contact surface; the distribution method of the lime in relation to the surface may be important in maximizing efficiency. Liming may be performed in the lake/reservoir, in the tributaries or in the entire catchment area.

3.1.6.4.2 Other ecotechnological methods

Other in situ methods may be introduced in conjunction with liming to obtain a higher efficiency of biological recovery. Rhode (1981) in Jorgensen (1997) introduced the concept of a biological buffering system, the idea being to introduce acid tolerant and acid resistant algal species to increase the primary production in the ecosystem. Theoretically this would raise the pH, however, practically, it is difficult to obtain a higher level of photosynthesis when there is a lack of hydrogen carbonate ions in the water.

The introduction of acid tolerant fish species has been used successfully to accelerate biological recovery when used in conjunction with liming (Erikson et al 1980, 1987).

The application of land use/management strategies has been explored, particularly those associated with coniferous trees. Conifers have been replaced with broadleaf species, and cutback, burning and ploughing have been used to reduce coniferous growth (Ormerod et al., 1988: Welsh et al., 1987). This has been successful when the low pH is caused by natural processes, but has little effect if the acidification is caused by acid rain.

Hydrological management has also been used to counter acidification where a source of less acidic groundwater is available as a diluent, or retention time can be increased to enhance sediment generation of alkalinity in cases where the sediment itself is calcium-rich

Ideally, the whole spectrum of methods should be used to obtain the best and fastest results.

- Only by changing the forcing functions, i.e. the pH of the rainwater, is it possible to restore the ecosystem. Otherwise a return to acidic conditions is inevitable.

- Liming ensures a higher pH but only for a limited period (dependent mainly on the retention time of the water in the lake/reservoir).

3.1.6.5 Time scales of acidification and recovery

Atmospheric acidification occurs relatively quickly (hours to days), once the gaseous acid precursors are emitted (Galloway, 2001). Depending on the nature of the catchment soils, headwater acidification can take years if the deposited sulphate and/or nitrate is not retained strongly by the soil, but decades if it is (Galloway et al., 1983).

Once surface waters have been acidified, biological responses can also be rapid or take years. Episodic acidification can result in fish fry mortality over the course of a single storm (Bulger et al., 2000) or it can take years for adult fish to suffer mortality. In combination, short and long term effects on aquatic biota have resulted in the disappearance of species from acidified surface waters over time (Galloway, 2001).

The recovery follows the same general trend (in reverse), but it takes place more slowly (Figure 43). The atmosphere will recover quickly once emissions are reduced. (Galloway, 2001). Soils will recover slowly due to their large capacity (Stoddard et al., 1999).

Surface waters will only recover chemically after soils recover, and fish recovery may not occur until 5-10 years after zooplankton recovery which themselves may take up to ten years to recover (Galloway, 2001).

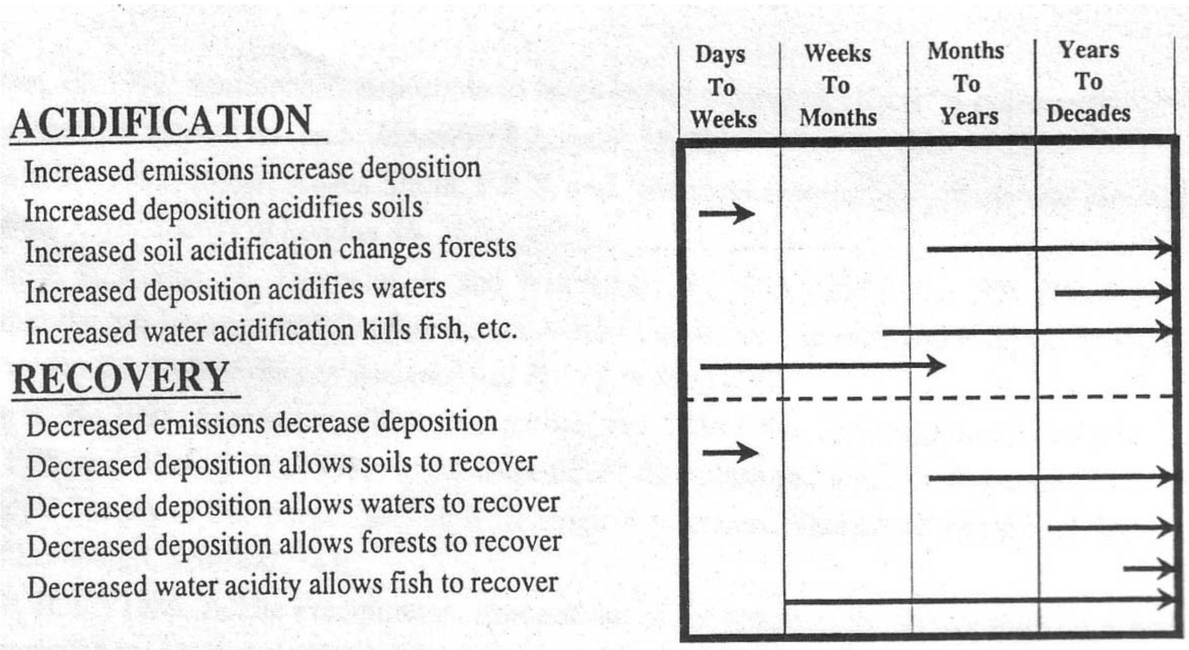


Figure 43. Timescales of acidification and recovery (from Galloway, 2001)

3.1.6.6 Acidification of freshwaters in South Africa due to atmospheric pollution

Air pollution in South Africa matches or exceeds that in some of the more industrialised nations of Europe, mostly because abundant low-grade coal deposits provide a relatively cheap source of power. Acid rain has been recorded in several regions, especially the eastern Highveld where there are many large coal-burning power stations and where the lowest recorded pH value for rain is 2.9 (Davies and Day, 1998).

There is very little information available in SA on the effects of air pollution on the environment. A management plan is needed which will incorporate rehabilitation of acidifying waters before they become so badly damaged that recovery is much more difficult.

3.1.6.7 Prospects for the future

The easiest projection to make is that in the future there will be more people needing more food and using more energy. Increase in per capita resource use will mean that food and energy production will rise faster than the population, resulting in an increase of emissions into the atmosphere that will in turn, result in increased deposition, and acidification of ecosystems (Galloway, 1998).

3.1.6.8 Acid mine drainage

Acid mine drainage (AMD) is drainage that emanates from both surface and underground mine workings, waste and development rock, and tailings piles and ponds (Durkin and Herrmann, 1994). After being removed from the ground, and having had the ore removed, waste rock is stored above ground in large free-draining piles. The extracted ore is then processed to separate the target mineral from the valueless portion, and this valueless portion becomes another form of mining waste called “tailings.” Tailings are usually stored above ground in containment areas or ponds or they are pumped into the excavated space from where they were mined (Johnson and Hallberg, 2005). Subsurface mining often progresses below the water table, in which case, water must be constantly pumped out to prevent flooding. When a mine is abandoned, the water table returns to its original level, flooding the mine. The introduction of water in all these situations is the initial step in most acid mine drainage situations (http//en.wikipedia.org).

In 1989 it was estimated that approximately 19,300 km of streams and rivers, and 72,000 ha of lakes and reservoirs worldwide, had been seriously damaged by mine effluents (now much greater), although the true scale of the environmental pollution caused by mine water discharges is difficult to assess accurately (Johnson and Hallberg, 2005). The formation of AMD and its associated contaminants have been described as the largest environmental problem facing the mining industry worldwide (Evangelou, 1995; Harris et al., 2003). It is a serious problem in South Africa, pollution of streams and rivers due to discharges of contaminated water from mines and dumps being one of the severest problems facing the future of South Africa.

3.2 Hydrological factors

3.2.1 Introduction

The hydrodynamic processes associated with reservoir operations may be considered to be an abiotic component contributing to the water quality of reservoirs (Dubnyak and Timchenko, 2000). Hydrodynamic processes play an important part in the functioning of reservoir ecosystems, contributing, as they do, to water movements, water level fluctuations, and their consequences. These processes have been discussed elsewhere in this report but are briefly reconsidered in this section..

3.2.2 The river downstream from the dam

Reservoir operations impact significantly (physically, chemically and biologically) on the river downstream of the dam. The large amount of literature available on this topic is not within the scope of this report, EXCEPT in the case of reservoir cascades where the quality of the outflowing water impacts on the water of a downstream water body. This is discussed in Section 4.4.

3.2.3 The reservoir ecosystem

The effects of reservoir operations within a reservoir water body are an integral part of the functioning of the reservoir ecosystem, and are interlinked in the consequences that they exert on water quality.

(1) Water movements

In the open waters of a reservoir, there are multiple water movement and mixing processes. Many of these are related to the hydrodynamics of the reservoir operation. The flow through a reservoir is of considerable importance in determining transport processes of e.g. suspended sediments and nutrients, as well as influencing the temperature and density gradients and thus stratification of the water column. Longitudinal gradients are often evident from the riverine end of a reservoir to the dam, as discussed in detail in Sections 2.1 and 2.3.

(2) Retention time

Retention time is directly related to (1) above. In reservoirs with short retention time and more intense through flow, the effect of the flow through the reservoir will be more pronounced, whereas in reservoirs with long retention time, the effects are much less pronounced. This is often a direct result of withdrawal rate or volume and the withdrawal schedule

(3) Water level fluctuation

Water level fluctuations are linked to both (1) and (2) with the associated dependence on the inflow and outflow of water into the system.

3.2.4 Reservoir operations

Reservoirs are designed and operated to achieve two basic goals – water storage and its subsequent withdrawal for a variety of purposes. Their operation to accomplish these two aims plays a critical role in the limnological character of the impounded reservoir (Kennedy, 1999). Water withdrawal depth, the rate and volume of withdrawal, and the withdrawal schedule are of particular importance in terms of their impact on the waterbody. (Table 6). The effects that water withdrawal has on thermal stratification and material balances in reservoirs may be the most limnologically significant influence of the operation of dams (Kennedy, 1999; Ford, 1990).

Operational goals	Operational parameters	Direct reservoir effects
Water withdrawal	Withdrawal depth Withdrawal rate or volume Withdrawal schedule	Storage or dissipation of heat and material Thermal stability and resistance to mixing Flushing rate and residence time Sedimentation and material retention Inflow mixing and material transport
Water storage	Water volume Surface elevation Storage schedule	

Table 6. Limnological implications of reservoir operation (After Kennedy, 1999)

3.2.4.1 Water withdrawal depth

From the point of view of reservoir management, it is important to know the possible effects of the water extraction level on the reservoir (Casamitjana et al., 2003). When multiple outlets exist, each different through flow option can produce a suite of positive and negative effects, with the overall result depending on the particularities of the reservoir in question (Straškraba 1996). Hydraulic regulation by means of selective off-takes and hypolimnetic siphoning were discussed as eutrophication control measures in Section 3.1.2, while hydraulic flushing through bottom outlets was considered as a means of siltation control in Section 3.1.1.

A knowledge of temperature distribution is fundamental to the understanding of the performance and functioning of reservoir ecosystems (Kimmel et al., 1990); the effects of water withdrawal have been found to be important in determining thermal stratification in reservoirs (Ford 1990). The release of water from the surface results in removal of strata most directly influenced by solar heating, resulting in the preservation of cooler, denser hypolimnetic water. Materials that settle below the effective withdrawal zone are retained as are those released from bottom sediments into the overlying water. On the other hand, the release of low-temperature water from the hypolimnion results in the storage of heat through the expansion of the epilimnion, vertical temperature differences are reduced (particularly at high withdrawal rates), and resistance to wind mixing is reduced. Materials accumulated in bottom sediments are removed due to flushing (Kennedy, 1999).

Casamitjana et al. (2003) used a numerical one-dimension model to predict the thermal structure of Boadella Reservoir (Spain) and then to investigate various possible water withdrawal scenarios. The model was found to predict the basic trends in thermal stratification of the reservoir correctly, and could also be used to predict the stratification pattern in other water withdrawal scenarios. These authors concluded that stratification plays a crucial role in determining the reservoir’s water quality and that prediction of its thermal evolution is essential for its management by means of selective withdrawal.

3.2.4.2 The withdrawal schedule

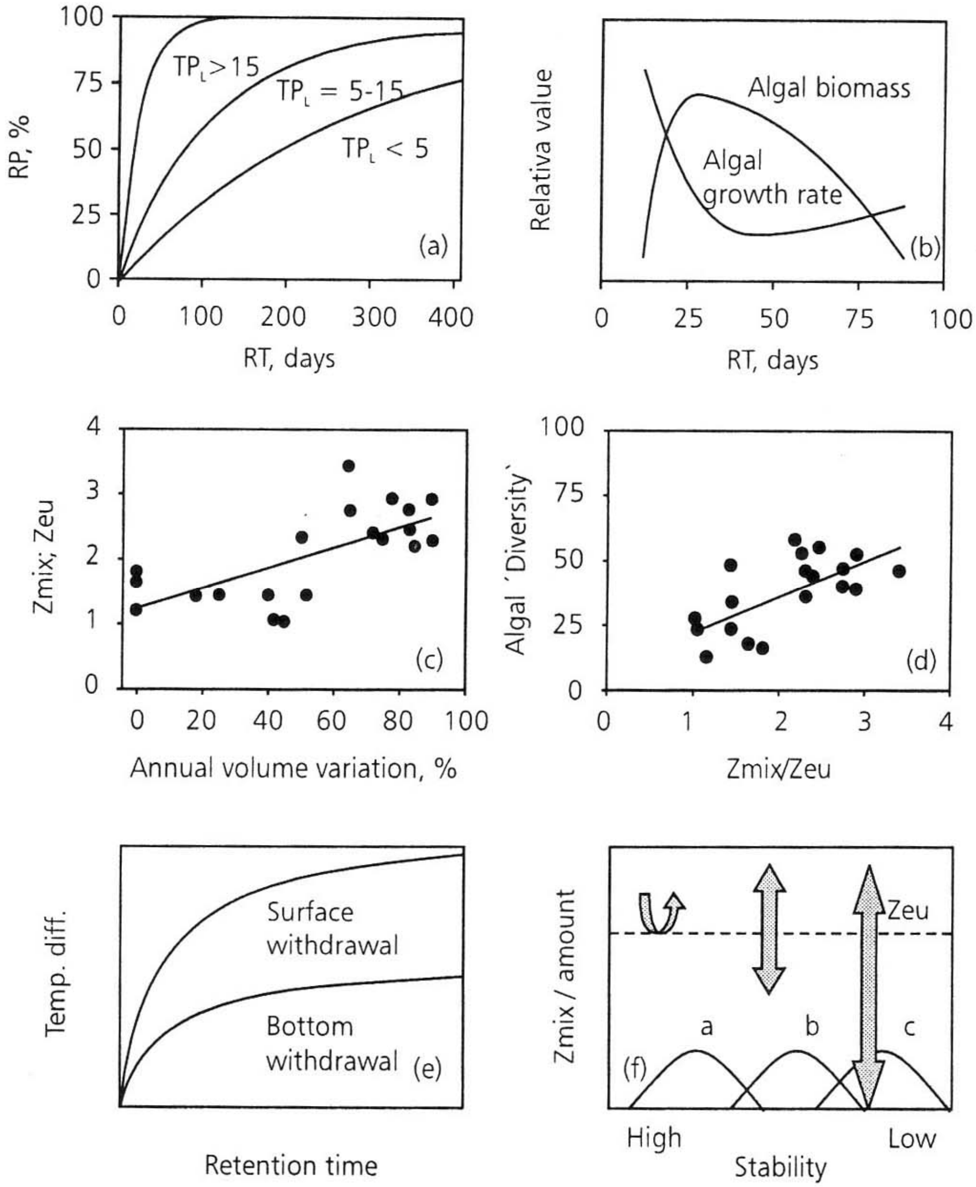
The water withdrawal schedule of a reservoir operates on different levels of magnitude, and thus impacts on the aquatic ecosystem in different ways.

3.2.4.2.1 Retention time

The withdrawal schedule is one of the main factors determining the retention time of water in a reservoir. The retention time, together with the concentrations of various chemical compounds in the inflowing water, is among the most important factors that determine the water quality of reservoirs (Ambrosetti et al., 2002; Kawara et al., 1998; Jorgensen, 2002; Townsend and Luong-Van; Perry et al., 1990).

Jorgensen (2002), using eutrophication models, attempted to answer two questions; (i) What is the role of the residence time in determination of water quality? (ii) Is it possible to manipulate residence time to improve water quality? Applying the models to three case studies he demonstrated that the residence time is an important forcing function that in many cases would have a great influence on the resulting water quality of a lake or reservoir. He also concluded that manipulation of the residence time to improve water quality was a possibility.

Figure 44 summarizes the influences of hydraulic retention time on a reservoir ecosystem.



(a) Influence of hydraulic retention time (R_T) and area phosphorus loading rate (TP_L; gm/m^2/yr) on total phosphorus retention (R_P; from Kennedy, in press). (b) Influence of hydraulic retention time (R_T) on algal growth and biomass (based on Straškraba *et al.*, 1993). (c) Correlation between variations in reservoir volume and the ratio of mixed depth to euphotic depth, and (d) between the ratio of mixed depth to euphotic depth and algal diversity as measured by percent rare species (from Naselli-Flores, 1998). (e) Effects of outlet depth and hydraulic retention on water column stability (as inferred from vertical temperature gradients). (f) The influence of water stability and mixing events (arrows indicate relative mixing depth) on abundance of algal associations (based on Reynolds, 1997).

Figure 44. A graphic summary of the influences of hydraulic retention time on a reservoir ecosystem (Kennedy, 1999).

3.2.4.2.2 The littoral zone

Under hydrodynamically stable conditions, littoral zone areas, occupying a buffer position between land and water have considerable biodiversity and play a role of natural biofilters. However, hyrodynamic processes in the shallow areas of reservoirs are often limiting factors for the development of biological communities (e.g. macrophytes and their associated biota) (Dubnyak and Timchenko, 2000; Casco, 1997). Hill et al. (1998) studying regulated and unregulated systems in Nova Scotia, Canada, found that shoreline plant communities of dammed systems were less diverse, contained more exotic species and were generally devoid of rare plants.

3.2.4.2.3 The rate and volume of withdrawal

Pronounced fluctuations of water level over a short period of time are an important characteristic of many reservoirs (Naselli-Flores and Barone, 1997). These fluctuations may affect phytoplankton biomass and species composition by influencing the underwater light climate (Barone and Naselli-Flores, 1994; Naselli-Flores and Barone, 1996) and nutrient dynamics (Kimmel et al., 1990). With respect to some fish species, water levels play an important role in breeding success. This is sometimes found to improve as reservoirs fill because flooded vegetation offers better sites for spawning (O'Brien. 1990). However in other cases increased siltation associated with reservoir filling, damages spawned eggs. Drawdown may seriously damage nursery areas and lead to desiccation of the eggs (Kubecka, 1993).

Little has been written on water quality changes associated with rapid drawdown of reservoirs and lakes (Boland, 1995; James et al., 2001). This was addressed in Manton River Reservoir, a small tropical reservoir in northern Australia by Boland (1995), who found a marked deterioration in water quality due to turbulence associated with the rapid drawdown, which destroyed the vertical density structure and caused elevated chlorophyll a and total phosphorus concentrations and higher turbidity and colour. All these water quality parameters improved following recharge of the reservoir in the following wet season, but most remained higher than their pre-drawdown values.

3.2.5 Management possibilities

Often viewed as impairments to water quality, reservoirs may also offer management opportunities. Within the design and operation parameters defined by engineering considerations, it may be possible to reach a consensus of cooperative use. For example, the timing and rates of release, including the use of selective withdrawal outlets and the seasonal progression in reservoir volume, may be modified to affect changes in water column thermal structure, and water and material retention, as a means of accomplishing water quality management objectives (Kennedy, 1999a).

If the requirement of water control (e.g. flood reduction, power generation,, navigation, etc) can be met while at the same time encouraging a desired effect on water quality (e.g.

reduced phytoplankton biomass or shifts in species composition), then cost efficient management strategies can be realized (Kennedy, 1999a).

3.3 Climate Change

3.3.1 Introduction

There is no doubt that a shift in long-term global climate is taking place as a result of the intensification of the “greenhouse effect” (IPCC, 1992). The greenhouse effect (a natural phenomenon) occurs as a result of the earth’s outgoing long-wave radiation being unable to escape the lower atmosphere owing to the presence of certain gases, notably carbon dioxide, methane and water vapour. Without this natural phenomenon the earth would be approximately 33^{o} C colder and largely uninhabitable (Bruce, 1991). There has been a sharp increase in the amount of carbon dioxide, methane and nitrous oxide released into the atmosphere since the middle of the 20^{th} century due to increased industrial activity, and in addition, chlorofluorocarbons (CFCs), since they were introduced in the 1950s (Adeloye et al., 1999). These may also hinder the release of long-wave radiation, although by depleting the ozone layer, the greenhouse effects of CFCs are partially neutralized (IPCC, 1992). The temperature of the earth’s surface has increased by 0.3 to 0.6^{o} C over the last century with the greatest increases in the last few decades.

Such increases in temperature are anticipated to affect reservoir ecosystems significantly through a combination of direct and indirect influences involving hydrological, physical, chemical and biological process mechanisms and interactions.

3.3.2 Modelling global warming

Changes in climate are projected by means of General Circulation Models or Climate System Models, which mathematically simulate the climate system - atmosphere, oceans, hydrologic cycle and ice. Put very simply, three models were run with a forcing equivalent of a doubled carbon dioxide concentration in the atmosphere until the system reached a new equilibrium (IPCC, 1990). The projected increase in global mean temperature from these three models was 3.5^{o} – 4^{o} C. When ocean lags are taken into account, the IPCC estimates a 0.2 - 0.5^{o} C increase per decade over the next century under a “business as usual” scenario (i.e. no major controls of greenhouse gas emissions except those already agreed to internationally to reduce CFCs for protection of the ozone layer) (Bruce, 1991).

These estimates may be conservative since the aggregate of present national projections of carbon dioxide and methane emission yields a global projection 10-20% higher than “business as usual”. In addition, these mathematical climate system models do not take feedbacks in the system into account, especially the biological feedbacks. The Second World Climate Conference, Geneva, 1990, concluded that the feedbacks would be likely to increase rather than decrease greenhouse gases.

Sea level is expected to rise 3-10cm per decade under the "business as usual" scenario. Agreement between models on precipitation changes is not close, but all models show greater global precipitation and increased vigour of the hydrological cycle with rising temperatures (Bruce, 1991).

3.3.3 Effects of climate change on reservoirs and their management

A warmer climate will put phenomenal pressures on water resources in many parts of the world (Adeloye et al., 1999). For reservoir quality management, three types of consequences of global climate change can be expected, as shown in Figure 45 (Straškraba and Tundisi, 1999):

(i) direct effects on air temperature and flow;
(ii) indirect effects due to changes in natural and agricultural vegetation; and
(iii) increased water demands on reservoirs

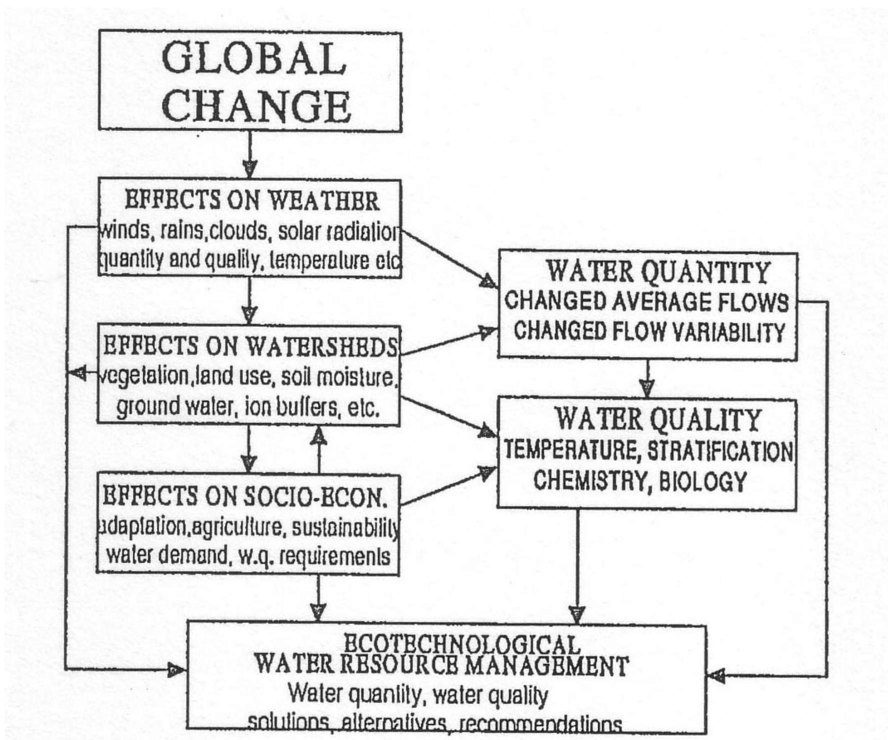


Figure 45. Expected consequences of global changes on reservoir water quality (after Straškraba and Tundisi, 1999)

3.3.3.1 The hydrological cycle

Global warming is accelerating the hydrological cycle. Evaporation from land and water will increase, resulting in more precipitation on average, although regional precipitation patterns will continue to be complex and variable. Reviews of state-of–the-art climate models suggest that average global evaporation and precipitation may increase by 3 – 15%

for an equivalent doubling of atmospheric carbon dioxide concentration – the greater the warming, the larger the increases (Gleick, 1997).

Improvements in modelling of the climate have begun to permit more realistic estimates of regional evaporation and precipitation patterns. With a doubling of atmospheric CO_2 concentrations, models show increases in humidity and greater precipitation at high latitudes and tropics throughout the year and in mid-latitudes in winter. In many model estimates, summer rainfall decreases slightly over much of the northern mid-latitude continents. Other changes in mid-latitudes remain highly variable and ambiguous; information on changes in precipitation in subtropical and arid regions is scanty, but even small changes in arid zones can have significant implications (Gleick, 1997; Williams, 1999).

In fact, the occurrence or influence of global warming is not globally uniform, temporally or spatially. It is occurring twice as fast as overall warming during the winter and at night, with winter warming occurring faster at high latitudes than near the tropics. Enhanced evapotranspiration dries out soils in some regions whereas the warmer atmosphere holds more water vapour (6% increase for 1^{o} C), fuelling more intense violent downpours elsewhere. Prolonged droughts and intense precipitation can be equally punishing (Epstein, 2004).

3.3.3.2 Runoff

Many estimates of changes in runoff due to climatic change have been produced using detailed regional hydrologic models of specific river basins and a variety of climate scenarios (Arnell, 1996). Results show that significant changes in the timing and magnitude of runoff are likely to result from quite plausible changes in climatic variables. Some studies suggest that temperature increases of $2\text{-}4^{o}$ C, with no change in precipitation, can result in runoff decreases of 20% or more (Lettenmaier and Gan, 1990; Burns, 1994; Mimikou et al., 1991; Gleick, 1987a,b). Increases or decreases in precipitation of 10% - 20% tend to change runoff by about the same amount. In areas where demands for water are close to the limit of reliable supplies, such changes will have enormous management implications (Gleick, 1997).

There is also a risk of increased flooding. The authors of the 1996 IPCC report concluded that flood related consequences of climate change might be as serious and widely distributed as the adverse impacts of drought.

3.3.3.3 Soil moisture

Changes in temperature and precipitation patterns will have direct effects on soil moisture. In regions where precipitation increases, increases in evaporation driven by higher temperatures may be even greater, resulting in a net drying of the land surface (Gleick, 1997). Incidence of droughts in the USA, measured by soil moisture conditions, is likely to increase as temperatures increase even where precipitation increases, because of increased evaporation (Rind et al., 1990).

All climate model results simulate a general increase in soil moisture at the northern high latitudes in winter where precipitation increases exceed evaporation increases. Most models also simulate large-scale drying of the Earth's surface over northern mid-latitude continents in summer owing to higher temperatures and either insufficient precipitation increases or actual reduction in rainfall (Gleick, 1997).

3.3.3.4 Water quality

An increase in the water temperature of lakes or reservoirs will be a significant factor affecting their ecological balance (Hosomi, et al., 1996; Bruce, 1991). The growth rate and standing crops of algae will increase, and, in addition, the period of thermal stratification will be longer, the thermocline deeper, and depletion of dissolved oxygen in the hypolimnion more severe. This will accelerate eutrophication as nutrients will be released from bottom sediments (Kalff, 2002). The change in water retention time and hence, nutrient loading, could have significant consequences on the trophic state of reservoirs. Vegetation changes in the watershed could contribute to hydrological changes and could cause major changes in nutrient and pollution loads (Straškraba and Tundisi, 1999).

3.3.3.5 Public health

The warming of lakes and reservoirs inferred by global warming projections would also result in pole ward movement of water-borne or water-connected diseases which are temperature limited as water is the principal transmission agent for 80% of tropical diseases. Mosquito-transmitted diseases are the prime example, but there are many other such as Bilharzia as the snails which transmit schistosomiasis can only survive in temperatures between 18 and 35^{o} C (Epstein, 2004).

3.3.4 Implications for reservoir management

Impacts of climate change on water resources will be felt in virtually every aspect of natural resources management. These include all decisions about long-term water planning, design and construction of new water-supply infrastructure, the type and acreage of crops to be grown, urban water allocations and rate structures, reservoir operation and water-supply management.

Major uncertainties still exist surrounding the impact of climate change on lakes and reservoirs. Indeed the most important effect of climate change for water managers will be the overall uncertainty that they face. There are three possible approaches that managers might take at this time:

- the "wait and see" or "do nothing" approach which is cheapest in the short term but could have enormous medium- and long-term risks,
- the "no regrets" approach which includes evaluating management and operational options under a broader range of scenarios than managers have traditionally

considered, and designing and building new infrastructure to deal with climatic uncertainly, and
- going “overboard” and making expensive but incorrect decisions.

The flow of information from the scientific global change community to the public and water management community needs to be developed and expanded (Gleick, 1997).

3.3.5 Other factors

It must be emphasised that climate change is only one factor that will affect the behaviour of lakes and reservoirs in the future. Cutting of forests and construction of reservoirs result in increases in streamwater temperature of a similar order of magnitude to that proposed by global warming. The dominant factor controlling erosion and sedimentation will continue to be the presence or absence of vegetation and management of soils in the watershed. Water levels will continue to be influenced by upstream water withdrawals for irrigation and direct withdrawals for human use. It is important to place the changes that may arise from global warming, in the context of all the other changes that will affect lakes and reservoirs in the future (Bruce, 1991).

3.4 Invasive Species

3.4.1 Introduction

Species invasion is one of the leading mechanisms of global environmental change, particularly in freshwater ecosystems (Garcia-Berthou et al., 2005). After loss of habitat, invasive species are the second leading cause of biodiversity loss (Vitousek et al., 1997; Mack et al., 2000). Thousands of exotic species have been introduced to aquatic habitats (Cohen and Carlton, 1998), and besides imposing huge economic costs, invasive species cause extinction of native species and reduction of genetic diversity through a variety of mechanisms (Rahel, 2000). This form of biological pollution also has the potential to alter the transfer of energy and matter within an ecosystem (Yan and Pawson, 1997).

3.4.2 Stages in the invasion process

1. The intentional or unintentional introduction to the wild of a species reaching a region beyond its native range
2. The establishment of self-sustaining reproductive populations
3. Population growth and spreading of the species (Richardson et al., 2000; Kolar and Lodge, 2001)

3.4.3 Approaches to the study of invasion biology

The issue of species introductions and range expansions is of growing importance as ecosystems become more disturbed (Work and Gophen, 1999). However, the factors that favour invasions, and the effect of invasions are often difficult to predict (Williamson and Fitter, 1996).

The two most frequent approaches to the study of invasion biology are the identification of species traits that that might predict invasion success (Garcia-Berthou et al., 2005; Williamson and Fitter, 1996; Mack et al., 2000; Kolar and Lodge, 2001) and the identification of attributes of communities (and ecosystems) that affect their susceptibility or resistance to invasion (Levine and D'Antonio, 1999). (Table 7)

Characteristics relating to 'invasiveness' of a species

1. The source and invaded environments are far apart geographically.
2. They have rapid dispersal rates.
3. They possess colonising abilities superior to those of native species.
4. They exhibit r-selected breeding attributes, many offspring, little parental care.
5. They have some form of protection for eggs and young larvae.
6. They lack natural enemies in the invaded environments.
7. They are generalist, rather than specialist, in their feeding preferences.
8. Invasives with specialist feeding habits (e.g. blackflies) occur when there is an abundance of food.
9. They breed throughout the year.

Characteristics of ecosystems that are susceptible to invasion

1. They are degraded, disturbed or modified.
2. They exhibit reduced species diversity.
3. They have anthropogenically modified flow regimes or water quality.
4. Periurban environments with many artificial water bodies are often exploited.
5. Invasion is enhanced when the climates of source and invaded environments are similar.

Table 7. Characteristics relating to 'invasiveness" of a species and characteristics of ecosystems that are susceptible to invasion. (After Davies and Day, 1998).

Moyle and Light (1996) suggested that the suitability of abiotic factors predetermines the success of an invasion, regardless of the biotic interactions in the new habitat. Success is strongly influenced by a species' ability to adapt to critical environmental variables such as salinity (Thompson, 1991) and temperature (Garton et al., 1990; Hall et al., 1992; Stelzer, 1998; Sanford, 1999; Lodge, 1993). Other studies have documented the influence of biological characteristics such as high rates of dispersal and reproduction, good competitive ability and predation resistance on the success of the invader (Garton and Berg, 1990; Lehman, 1991; Mackie, 1991). However, unfavourable abiotic conditions preclude a successful invasion, and therefore a consideration of both the native and the new environment is necessary to answer the question of why a species is able to invade an ecosystem (Work and Gophen, 1999).

However, some results of these approaches have been inconclusive (Williamson, 1996; 1999; Smith et al., 1999). More recent suggestions are the identification of predominant donor regions and invasion pathways (Jenkins, 1996; Ricciardi and Rasmussen, 1998; MacIsaac et al., 2001) particularly for aquatic invasions (Ricciardi and MacIsaac, 2000).

3.4.4 Invasive plants

In Southern Africa, a number of introduced "water weeds" are major economic pests. They include *Salvinia molesta* (Kariba weed), a free floating fern, *Eichhornia crassipes* (water hyacinth), and *Myriophyllum aquaticum* (parrot's feather), a floating or rooted water plant, all from South America and all proclaimed noxious weeds. *Azolla filiculoides* (also a floating fern), and *Pistia stratiotes* (Nile Cabbage), possibly originating from Australia, have not yet been declared undesirable aliens.

These plants all exhibit the potential for explosive growth, particularly in eutrophic conditions, and may cover vast areas of standing and slow-running waters. They are widespread throughout South Africa (Figure 46) and have already cost the country tens of millions of rand to eradicate. (Macrophyte control is discussed in Section 3.1.2.5).

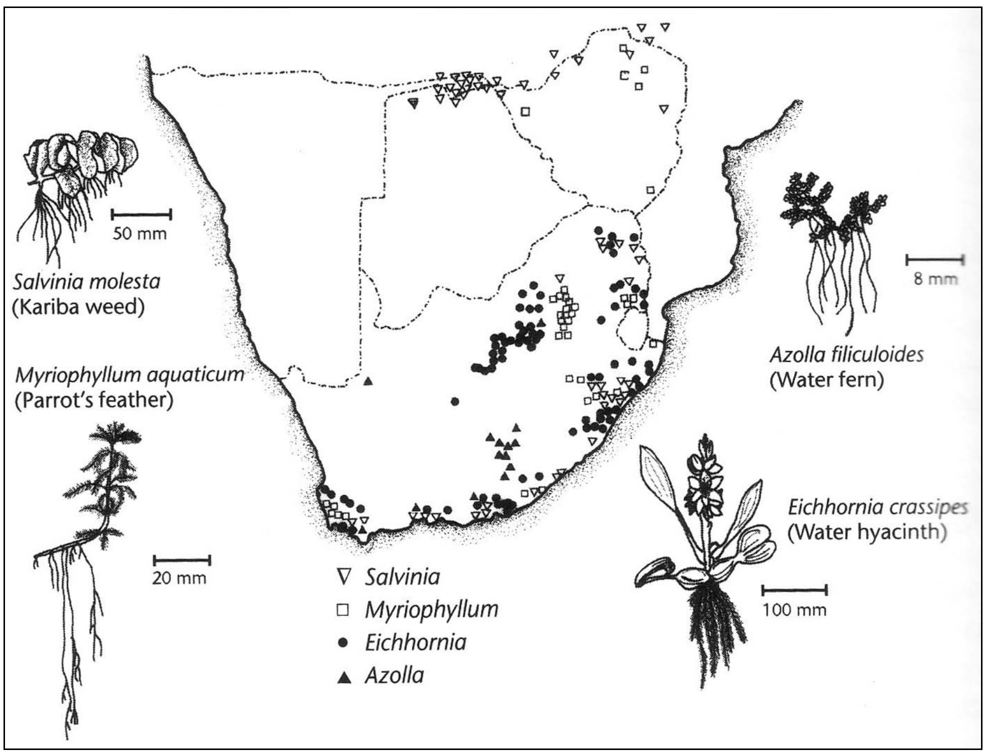


Figure 46. Distribution of the four major alien aquatic plants in Southern Africa. (After Davies and Day, 1998).

Eichhornia crassipes

Ever since its introduction to Egypt in the late 19th century, *Eichhornia crassipes* has spread through Africa's lakes and reservoirs (Brendonck et al., 2003). Its prolific growth causes considerable economic problems and affects fisheries, boat traffic, water supply and the whole ecology of affected lakes (Ogutu Ohwayo et al., 1997). It tends to invade waterbodies where hydrological or nutrient conditions have been altered by human activity (Barret, 1989). Biological, chemical and mechanical control measures are expensive and hampered by re-infestation from its long-lived seeds (Brendonck et al., 2003).

In spite of the detrimental effects of *Eichhornia crassipes*, Rommens et al. (2003) in their study on the impact of this macrophyte on the water quality of Lake Chivero, a reservoir in Zimbabwe, pointed out that it does have some positive attributes. Its capacity for accumulating heavy metals and organic contaminants and its wide tolerance of environmental conditions makes it suitable for treating waste water. In addition, its rapid growth and multiplication has led its use as an animal food, in paper and other products, and as compost (Mehra et al., 1999; Navarro and Phiri, 2000). Its presence can also prevent the development of algal blooms, by direct light limitation. A workshop on "The Control of Africa's Floating Water Weeds" in 1991, stressed the importance of conducting ecological studies on aquatic systems affected by *Eichhornia* in order to estimate its affect on biodiversity (Greathead and DeGroot (1993) in Rommens et al., 2003). However, to date, little attention has been paid to the potential importance of water hyacinth mats for maintaining the structural complexity of a lake and its species diversity. Rommens et al. (2003) suggest that this is probably due to the emphasis given to the many environmental and socio-economic problems caused by its infestations, with its eradication being given priority.

Salvinia molesta

This invader became famous as "Kariba weed" soon after the filling of Lake Kariba in the 1960s. At one stage it covered more than 2200km^2 of the surface of Lake Kariba with mats often more that half a metre thick. This situation created ideal conditions for mosquitoes and snails, the transmission agents for malaria and Bilharzia (Davies and Day, 1998). A discussion of the "Kariba weed" as an alien invader can be found in Moreau (1997).

3.4.5 Invasive animals

Foreign invaders, including fish (e.g. Moyle and Light, 1996), molluscs (e.g. Strayer, 1999; Yu and Culver, 1999; 2000) and crustacean zooplankton (e.g. Lennon et al., 2001; Work and Gophen, 1999; Pattinson, 2993; Ketelaars and van Breeman, 1993) are threatening the integrity of freshwater ecosystems worldwide 288.

3.4.5.1 Fish

It has been hypothesised that the construction of reservoirs and the accompanying hydrological manipulation has been instrumental in increasing the susceptibility of river systems in SA to invasions by introduced fish (De Moor 1996; Gehrke and Harris, 2001), as well as range extensions to native species e.g. the movement of *Clarias* from the Orange to the Fish River.

Many of South Africa's freshwater fish populations are undergoing a demographic decline. Of the 97 species occurring in SA 30% have been listed by the IUCN as threatened (Skelton 2002). This is probably due to a variety of reasons – deteriorating water quality, habitat degradation and introduction of exotic species (Paxton, 2004). Flow regulation and river fragmentation are also likely to have had an influence in reducing recruitment by obstructing movement, degrading instream habitat and facilitating invasion by exotic species (De Moor, 1996).

3.4.5.2 Molluscs

Dreissena polymorpha Pallas (the zebra mussel) is the most often cited example of an invasive mollusc species. Initially found in the USA in the 1980s, it has now spread to all the Great Lakes and many connected waters. These animals attach themselves easily to the hulls of boats and thus spread quickly along transport routes. Larvae remain planktonic for several weeks during which they are transported by currents, and in bilge water and bait buckets. These factors contribute to their successful invasive potential (Kalff, 2002).

The effect of zebra mussels on the ecosystem has been of great concern to ecologists and fisheries managers since their appearance as invaders. MacIsaac et al. (1992) estimated that they could theoretically filter a 7m water column between 3.5 and 18.8 times daily. Bunt et al. (1993) estimated that even the small-bodied zebra mussels (2-11mm) were theoretically capable of filtering between 39% and 96% of the water column daily in western Lake Erie. This filtering activity significantly decreases algal abundance and increases water clarity (Reeders and de Vaate, 1990; Lei et al., 1996; MacIsaac et al., 1992; Bunt et al., 1993; Fanslow et al., 1995; Nalepa et al., 1995). These changes potentially affect the aquatic food web and thus the whole lake ecosystem (Yu and Culver, 1999). Thermal structure can also be influenced by water transparency by increased epilimnion thickness and metalimnion heating rate (Mazumder and Taylor, 1994). Yu and Culver (1999, 2000) investigated the possibility that zebra mussels could change stratification patterns. They concluded that indirect impacts on a small reservoir stratification pattern may have much broader implications then the direct trophic interactions to the whole ecosystem.

Predators on zebra mussels are most often not able to control populations, resulting in the long-term effect of shifting invertebrate production away from zooplankton in the water column, and thereby away from zooplanktivorous fish and their predators to the benthos. The effect on pelagic fish stocks is an issue of great economic importance and scientific interest (Kalff, 2002). Effects of a shift from planktivory to benthivory on pollutant mobilization have not been explicitly considered, but can be anticipated.

3.4.5.3 Crustacean zooplankton

Studies of freshwater crustacean zooplankton have revealed a variety of successful invaders (Havel and Hebert, 1993; Reid and Pinto-Coelho, 1994; Lee, 1999; Ricciardi and MacIsaac, 2000). Cladoceran zooplankton possess many of the predicted traits of a successful colonizer. Adults and ephippia can be introduced by bait bucket releases, intentional transfers of water or fish, and by natural passive transport of the ephippia by wind, birds, mammals and fish. The ability of cladocerans to reproduce parthenogenetically not only ensures that they can reproduce quickly, but also reduces the size of the propagule required for successful colonization (Work and Gophen, 1999).

An example of a cladoceran invader, *Daphnia lumholtzi* (Sars) was first collected in North America in 1990 in a small lake in Texas, USA (Sorenson and Sterner, 1992). Since that date it has been found in at least 125 lakes and reservoirs in the US from Florida to Arizona. Features that may have enhanced its ability to invade North America from Australia, south-east Asia and Africa are:

(i) The native and new habitats have similar maximum temperatures. The tropical/subtropical origin may explain its ability to withstand higher temperatures than the native North American zooplankton (Moore et al., 1996). This factor may present a constraint on its ecological success as an invasive species in colder waters of North America (Lennon et al., 2001), although climate warming is likely to modify this.

(ii) Invasion theory predicts that a species adapted to disturbed habitats is more likely to invade successfully (Orians, 1986). *Daphnia lumholtzi* has been recorded primarily in reservoirs, rivers, ephemeral lakes and other disturbed systems in its native range (Mishra and Saksena, 1990; King and Greenwood, 1992).

(iii) The extremely long head and tail spine may reduce its vulnerability to predation by fish and invertebrates (Havel and Dodson, 1984; Havel, 1985; Parejko, 1991; Sorenson and Sterner, 1992).

3.4.6 Management

Introduction prevention should be regulated at all organizational levels, of particular importance being the introduction of species to new continents. However, introductions to new countries within a continent (or new basins within a country), although difficult to control, should also be legislated against, with implementation being internationally coordinated (Garcia-Berthou et al., 2005). Previous success, and taxonomy, to a lesser extent, are relatively good predictors of invasive potential, species and families with previous invasive histories clearly being of special concern. (Mack, 1996; Williamson, 1999).

4. MANAGEMENT TOOLS AND AIDS

Management can be functionally and operationally defined as a process of continuous decision-making to achieve specified aims and objectives. Reservoir management hinges largely on hydrological, chemical and biological objectives. Rapid monitoring of conditions, and outcomes of management activities is pivotal to effective management. Various methods and approaches to such monitoring exist, and new and novel techniques and instrumentation are finding welcome application. A brief outline of certain established and emerging approaches follows.

4.1 Biomonitoring, bioindicators, etc.

4.1.1 Procedure

A wide range of testing approaches are available to determine the effect of polluting and toxic substances on lakes and reservoirs, the abundance of which makes it extremely difficult to select a single approach or test battery that will meet all management needs. Simple tests with indigenous organisms, particularly tests that use species from multiple trophic levels known to occur in the system, can provide a screening-level assessment of pollution hazard. The next level of investigation should consider use of standardized microcosms or mesocosms. Accompanying any experimental testing programme should be a well-designed, field-based assessment to place test results in the context of the local environment. (Herricks, 1992).

4.1.2 Effects of pollutants and toxic substances on aquatic biota

The presence of harmful substances, of natural substances in excess, and the changes in the aquatic environment that result from them, or by physical alteration of the habitat, result in, or induce, a variety of effects in organic organisms. Some of the most common effects are:

- changes in the species composition of aquatic communities
- changes in the dominant groups of organisms in a habitat
- impoverishment of species
- high mortality of sensitive life stages e.g. eggs, larvae
- mortality of the whole population
- changes in behaviour of the organisms
- changes in physiological metabolism
- histological changes
- morphological deformities

4.1.3 Methods of biomonitoring and bioassessment

The biological assessment of water, waterbodies and effluents is based on five main approaches (Friedrich et al., 1992), and is summarized in Table 9.

4.1.3.1 Ecological methods

- Community studies: Analysis of biological communities of a water body i.e. changes in species composition or community structure e.g. Lyashenko and Protasov, 2003; Dombrovskiy, 2003; Kozlovskaya and Bakanov, 2003; Ogbeibu and Oribhabor, 2002 (macrozoobenthos); Magadza, 1994; Branco et al., 2002 (zooplankton), Harding et al., 2005; Taylor et al., 2005 (diatoms), Adams and Greeley, 2000. Analysis of communities developing on artificial substrates is especially convenient.

- Indicator species: Presence or absence of specific species e.g. Geetanjali et al., 2002 (nematode parasites in catfish); Passy and Bode, 2004; Harding et al., 2005; de la Rey et al., 2004; Taylor et al., 2005 Kitner and Poulickova, 2003; (diatoms); Fleituch, 1992 (macroinvertebrates); Canosa and Pinilla, 1999 (bacteria).

4.1.3.2 Physiological and biochemical methods

- Respiration and/or growth measurement of organisms e.g. Klochenko and Medved, 2001; Shehata et al., 1997, (algal cultures).

- Use of biomarkers to assess sub-lethal effects of pollutants, mostly on free-living organisms e.g. Sherry, 2003; Rocha-e-Silva et al., 2004 (fish); Favari et al., 2002 (zooplankton and fish); Martinez-Tabche et al. 2001 (oligochaetes); Adams and Greeley, 2000).

4.1.3.3 Use of organisms in controlled environments

- Assessment of toxic effects of samples on organisms under defined laboratory conditions – toxicity tests or bioassays e.g. Shcherban et al., 2001; Tisler and Zagorc-Koncan, 1999; Martinez-Tabche et al., 2000; Wong and Pak, 2004; Mangas-Ramirez et al., 2001 (zooplankton); Shehata et al., 1997 ; Bednarz, 1995 (algae); Sladeckova, 1990 (periphyton).

4.1.3.4 Assessment of biological accumulation

- Bioaccumulation by organisms living in the environment (passive monitoring) e.g. Cenci, 2000; Zakova and Kockova, 1999 (mosses); Zurayk et al., 2001; Aula et al.1995 (macrophytes); Farkas et al., 2003 (zooplankton); Favari et al., 2002 (zooplankton and fish); Bren, 2001 (macrozoobenthos); Martinez-Tabche et al., 2001 (oligochaetes); Veinott et al., 2001(molluscs); Bolotova and Konovalov, 2002; Allinson et al., 2002; Svobodova, 1999; Rehulka, 2002; Bakre et al., 1990 (fish); Ravera, 2001;
- Bioaccumulation by organisms deliberately exposed in the environment (active monitoring).

4.1.3.5 Histology and morphology

- Observation of histological and morphological changes e.g. Bolotova and Konovalov, 2002 (fish); Elmoor-Loureiro, 2004 (zooplankton).
- Embryological development or early life stages e.g. Wong and Pak, 2004 (zooplankton).

4.1.3.6 Rapid algal profiling

- Heterotrophic bacteria and viruses excepted, microbial phytoplankters (cyanophytes and algae) show the most rapid response to in situ changes in conditions in aquatic ecosystems. Really effective monitoring of changes in this guild of autotrophs requires profiling at a very short temporal scale – sampling on the order of one or two day intervals. Manual collection and subsequent conventional analysis by microscopic examination is unwieldy, enormously costly, and impractical. In its place (or at least as an alternative option), new generation technology has stimulating the development and deployment of more automated monitoring methods. In this regard, considerable benefit has been achieved with various new-generation probes, particularly the so-called fluoro-probe (Dubelaar et al. 2004). While not yet capable of species-level discrimination, broad characterization of autotrophs on the basis of proportional representation of different chlorophyll types and mixes provides an extremely powerful tool for rapid biomonitoring. Such equipment can be deployed in situ, to provide continuous and continuing information useful for water quality management purposes (Dubelaar et al. 2004).

4.1.3.7 Genetic molecular markers

- Partly linked to the former, concern regarding growing incidences of toxic 'algae' particularly cyanobacteria, and recognition of their important implications to human health has led to attempts to develop rapid diagnostic and monitoring capabilities for toxic algae. As toxic and non-toxic strains of several important taxa are not distinguishable using conventional morphological screening, the development and introduction of molecular characterisation methods has been an emerging tool (Sivonen 2004).

4.1.4 Indicator organisms

A wide variety of pollutants may be detected and monitored by means of bioindicators, different organisms generally being more suitable as indicators of different pollutants (Table 8).

- An indicator species is one that is commonly present or absent under specified environmental conditions and "indicates" environmental quality by its presence or absence.

- A target species is, in a general sense, an ecological indicator that is maintained in a lake or reservoir. It is assumed that meeting the environmental requirements of the target species or group will protect the supporting ecosystem, affording general environmental protection.

- A pollutant accumulator ("scavenger") is an organism which can accumulate relatively large amounts of pollutants without apparent noxious effects (Ravera, 2001). The concentration of a pollutant in an organism is the result of many variables i.e. the concentration of the pollutant in the water, the physical-chemical form of the pollutant, the membrane permeabilities of the organism, the type and quantity of food and its degree of contamination, the physiological state of the organism and the characteristics of the physical environment influencing the organism as well as the pollutant.

- The most commonly used species aggregate is the community (an interacting group of organisms in a specified habitat). Also used is the guild (species that exploit the same class of environmental resources in a similar way). The aggregation of species depending on the same resource may often provide a convenient base of extrapolation (Herricks, 1992).

4.1.4.1 Bacteria

Most suitable as indicators of faecal pollution, also used as indicators of eutrophication (Canosa and Pinilla, 1999).

Organisms	Advantages	Disadvantages
Bacteria	Routine methodology well developed. Rapid response to changes, including pollution. Indicators of faecal pollution. Ease of sampling.	Cells may not have originated from sampling point. Populations recover rapidly from intermittent pollution. Some special equipment necessary.
Protozoa	Saprobic values well known. Rapid responses to changes. Ease of sampling.	Good facilities and taxonomic ability required. Cells may not have originated from sampling point Indicator species also tend to occur in normal environments.
Algae	Pollution tolerances well documented. Useful indicators of eutrophication and increases in turbidity.	Taxonomic expertise required. Not very useful for severe organic or faecal pollution. Some sampling and enumeration problems with certain groups.
Macroinvertebrates	Diversity of forms and habits. Many sedentary species can indicate effects at site of sampling. Whole communities can respond to change. Long-lived species can indicate integrated pollution effects over time. Qualitative sampling easy. Simple sampling equipment. Good taxonomic keys.	Quantitative sampling difficult. Substrate type important when sampling. Species may drift in moving waters. Knowledge of life cycles necessary to interpret absence of species. Some groups difficult to identify.
Macrophytes	Species usually attached, easy to see and identify. Good indicators of suspended solids and nutrient enrichment.	Responses to pollution not well documented. Often tolerant of intermittent pollution. Mostly seasonal occurrence.
Fish	Methods well developed. Immediate physiological effects can be obvious. Can indicate food chain effects. Ease of identification.	Species may migrate to avoid pollution.

Table 8. The advantages and disadvantages of different organisms as indicators of water quality. (After Friedrich et al., 1992).

4.1.4.2.1 Phytoplankton/Algae

A wide variety of uses as indicators for e.g. eutrophication, organic pollution, acidification, pesticides, heavy metals (Bednarz, 1995; Shehata et al., 1997; Becker and Bigham, 1995; Klochenko and Medved, 2001; Kitner and Poulickova, 2003; Passy and Bode, 2004; Bertrand et al., 2003; Harding et al. 2005; de la Rey et al., 2004; Taylor et al., 2005; Sladeckova, 1990.

4.1.4.3 Zooplankton

Indicators for e.g. eutrophication, heavy metals (Magadza, 1994; Mangas-Ramirez et al., 2001; Branco et al., 2002; Shcherban et al., 2001; Wong and Pak, 2004; Tisler and Zagorc-Koncan, 1999; Elmoor-Loureiro, 2004; Martinez-Tabche et al., 2000; Becker and Bigham, 1995; Farkas et al. 2003; Favari et al., 2002).

4.1.4.4 Macroinvertebrates/macrozoobenthos

Indicators for e.g. eutrophication, heavy metals (Lyashenko and Protasov, 2003; Ogbeibu and Oribhabor, 2002; Fleituch, 1992; Bren, 2001).

2.1.4.5 Aquatic macrophytes

Indicators for e.g. eutrophication, heavy metals (Zakova and Kockova, 1999; Aula et al., 1995; Zurayk et al., 2001).

4.1.4.6 Mosses

Indicators of heavy metals (Cenci, 2000; Zakova and Kockova, 1999).

4.1.4.7 Nematodes

Indicators of water quality (Geetanjali, 2002 (nematode parasites in catfish)).

4.1.4.8 Fish

Indicators of eutrophication, inorganic and organic pollutants (Becker and Bigham, 1995; Rehulka, 2002; Park and Curtis, 1997; Bakre et al., 1990; Favari et al., 2002)

	Ecological methods		Microbiological methods	Physiological and biochemical methods	Bioassays and toxicity tests	Chemical analysis of biota
	Indicator species[1]	Community studies[2]				
Principal organisms used	Invertebrates, plants and algae	Invertebrates	Bacteria	Invertebrates, algae, fish	Invertebrates, fish	Fish, shellfish, plants
Major assessment uses	Basic surveys, impact surveys, trend monitoring	Impact surveys, trend monitoring	Operational surveillance, impact surveys	Early warning monitoring, impact surveys	Operational surveillance, early warning monitoring, impact surveys	Impact surveys, trend monitoring
Appropriate pollution sources or effects	Organic matter pollution, nutrient enrichment, acidification	Organic matter pollution, toxic wastes, nutrient enrichment	Human health risks (domestic and animal faecal waste), organic matter pollution	Organic matter pollution, nutrient enrichment, toxic wastes	Toxic wastes, pesticide pollution, organic matter pollution	Toxic wastes, pesticide pollution, human health risks (toxic contaminants)
Advantages	Simple to perform. Relatively cheap. No special equipment or facilities needed	Simple to perform. Relatively cheap. No special equipment or facilities needed. Minimal biological expertise required	Relevant to human health. Simple to perform. Relatively cheap. Very little special equipment required	Usually very sensitive. From simple to complex methods available. Cheap or expensive options. Some methods allow continuous monitoring	Most methods simple to perform. No special equipment or facilities needed for basic methods. Fast results. Relatively cheap. Some continuous monitoring possible	Relevant to human health. Requires less advanced equipment than for the chemical analysis of water samples
Disadvantages	Localised use. Knowledge of taxonomy required. Susceptible to natural changes in aquatic environment	Relevance of some methods to aquatic systems not always tested. Susceptible to natural changes in aquatic environment	Organisms easily transported, therefore, may give false positive results away from source	Specialised knowledge and techniques required for some methods	Laboratory based tests not always indicative of field conditions	Analytical equipment and well trained personnel necessary. Expensive

[1] e.g. biotic indices

[2] e.g. diversity or similarity indices

Table 9. The principal biological approaches to water quality assessment, their uses, advantages and disadvantages. (After Friedrich et al., 1992).

4.2 Remote sensing

4.2.1 Introduction

Traditionally, reservoir water quality has been monitored and evaluated using field data. However, the collection and analysis of data are expensive and time consuming and it is often queried whether a limited number of field data can characterize the spatial variation of parameters within a large water body (Chen, 2003; Cheng and Lei, 2001). To improve traditional data collection methods, utilization of remote sensing data for water quality assessment has been investigated by many scientists.

Early studies focused on the relationship between remote sensing data and *in situ* measurements. To move the application of remote sensing to a forecasting and thus more practical level, it is necessary to incorporate it with traditional water quality modelling (Kondratyev et al., 1996). Moreover, integrating a GIS system with the monitoring and forecasting systems can improve the display of water quality information (Yang et al., 1999). A shortcoming of this technique is that it does not allow recognition of the vertical distribution of variables (Straškraba and Tundisi, 1999).

Imagery from both high altitude (satellite) and low altitude (aircraft) remote sensing systems may be used.

4.2.2 Information that can be obtained by remote sensing of reservoirs

2.1 The hydrodynamics e.g. upwelling zones, internal waves, warm and cold surface layers of large water bodies (Kondratyev et al., 1996 (Lake Ladoga, Russia); Baban, 1994).

2.2 Areas of high and low turbidity, indicating suspended inorganic and organic matter (Silva et al., 1997 (Tucurui Reservoir, Brazil); Choubey and Subramanian, 1992 (Tawa Reservoir, India); Nelson et al., 2002 (Michigan lakes, USA); Allee and Johnson, 1999 (Bull Shoals Reservoir, USA); Fraser, R.N., 1998 (Nebraska lakes, USA); Arenz et al., 1996 (Colorado Reservoirs, USA)).

2.3 Horizontal displacement of river plumes that may carry pollutants and/or suspended material into the reservoir (Straškraba and Tundisi, 1999).

2.4 Spatial distribution of total phosphorus and nitrogen can be depicted with careful analysis and reservoir-specific correlations with other variables such as chlorophyll a, temperature and transparency or adsorption coefficients (Bilge et al., 2003 (Porsuk Dam Reservoir, Turkey)).

2.5 Horizontal distribution of chlorophyll a and location of low and high concentrations of phytoplankton (Allee and Johnson, 1999 (Bull Shoals Reservoir, USA); Arenz et al. 1996 (Colorado reservoirs, USA); Bilge et al., 2003 (Porsuk Dam Reservoir, Turkey); Novo et al., 1993 (Barra Bonita Reservoir, Brazil); Mayo et al., 1995 (Lake

Kinneret); Yacobi et al., 1995 (Lake Kinneret); Thiemann and Kaufmann, 2000 (Mecklenburg Lake District, Germany); Gitelson et al., 1993b (Lake Kinneret); Mittenzwey et al., 1992).

2.7 Horizontal distribution of cyanobacterial blooms (Jupp et al., 1994)

2.8 Detection of macrophyte abundance (Malthus and George, 1997 (Cefni Reservoir, UK); Noernberg et al., 1999 (Tucurui Reservoir, Brazil); Duarte and Kalff, 1990); Penuelas et al., 1993)

2.9 Location of mass fish kills

2.10 Trophic status and water quality (Baban, 1996 (Norfolk Broads, UK); Chen, 2003 (Yeong-Her-Shan Reservoir, Taiwan); Cheng and Lei, 2001 (Te-Chi Reservoir, Taiwan); Yang et al., 1999 (Te-Chi Reservoir, Taiwan); Gitelson et al., 1990; 1993a.)

2.11 Complimentary data for EIAs – before, during and after construction (Nagarajan, 2000 (Dudhganga Dam, India).

4.3 Modelling

The former biomonitoring approaches generally assess what has happened in the immediate or recent past. These essentially address consequences. Modelling approaches on the other hand attempt to pre-empt adverse effects by providing some predictive fore-casting of potential problems.

4.3.1 Uses of mathematical models

Unless otherwise indicated, the information on modelling is based on Straškraba and Tundisi (1999).

Mathematical models are now extensively used to solve a wide variety of problems in water quality management, with an increasing number of models becoming available. Some problems for which modelling is useful are outlined in Table 10

4.3.2 Groups of models useful for water quality management.

1 Simple static calculation models consisting of algebraic equations or graphs.

Models in this group are predominantly based on statistical elaboration of large datasets. Table 11 lists a variety of simple models and their respective uses and applications.

BEFORE RESERVOIR CONSTRUCTION To estimate budgets of major water components of rivers that will enter the reservoir, of the reservoir itself, and of the reservoir inflow. To provide reasonable estimates between several alternate construction sites, dam heights and outflow and outlet structures so that decisions are supported.
IN THE WATERSHED To estimate pollution sources in the watershed by means of simple calculation models. To predict conditions in future reservoirs and the consequences of different management options on water quality in the watershed **FOR EXISTING RESERVOIRS** To predict possible future reservoir quality conditions when environmental conditions in the watershed are altered by human activities. To provide estimates for decisions between different water quality management options for use in long term planning. To support short-term operational management decisions regarding water quantity and quality. To optimize sampling schedules, investigations and controls of water quality.

Table 10. Some uses of mathematical models (modified from Straškraba, 1994).

MODEL	AUTHOR	APPLICABLE REGION OF MODEL USE
* Nitrogen retention by shallow reservoirs	Kelly *et al.* 1987, extended by Howarth *et al.* 1996	N. Hemisphere
* Phosphorus retention by stratified reservoirs	Straškraba *et al.* 1995	N. Hemisphere
* Retention of organic matter by reservoirs	Straškrabová 1976	Central Europe
* Hypolimnetic oxygen demand by reservoirs	Staufer 1987	United States
* Lake Number Model - specification of stratification	Imberger & Patterson 1990	Australia
* Temperature stratification model RESTEMP	Straškraba & Gnauck 1985	Central Europe
* Model of DO and P in stratified lakes	Chapra & Canale 1991	United States
* Model of end-of summer oxygen profiles in lakes	Molot *et al.* 1992	United States
* Model of hypolimnion discharge scenarios	Horstman *et al.* 1983	United States

Table 11. Simple models that are most useful for the estimation of various reservoir features and prediction of responses to management plans.

2 Complex dynamic models that provide temporal analysis of aspects of water quality conditions.

For purposes of modelling in rivers and reservoirs, Ambrose et al., (1982) in McCutcheon, (1989), distinguished four levels of increasing complexity in this group of complex dynamic models, based on the criteria specified below.

Level 1: Steady state solution, simple kinetics

e.g.LAKE, produced by ILEC (Jorgensen, 1992).

Level 2: Steady hydrodynamics, specified or handled empirically, steady or time variable water quality.

e.g. Lung Phosphorus Model (Lung and Canale, 1977).
AQUAMOD 3 (Dvorakova and Kozerski, 1980; Straškraba and Gnauck, 1985)
SALMO (Benndorf and Recknagel, 1982)
SALMOSED (Recknage et al., 1995)
CE-QUAL-RIVI (Bedford et al, 1983)
WASP4 (Ambrose et al., 1988)
MINLAKE (Riley and Stefan, 1988)
BLOOMII (Los, 1991)
MIKE12 (Ecological Modelling Centre, 1992)

Level 3: Unsteady hydrodynamics, but simplified solutions, simplified reservoir solutions, dynamic water quality.

e.g. Fortran (HSPF) Hydrologic Simulation Programme

Level 4: Unsteady hydrodynamics, dynamic water quality, ability to handle backwater and stratified reservoirs.

e.g. DYRESM, the most widely used reservoir hydrodynamics model (Imberger and Patterson, 1981: Imberger, 1982)
DYRESM-WQ (Hamilton and Schladow, 1995).
ASTER and MELODIA (Salencon and Thebault, 1994).

3 Geographical information systems (GIS) that provide computer software for problems that require spatial resolution.

e.g. IIASA (Fedra et al., 1990), and RAISON (Lam et al., 1994) are used when mapped or tabular data exists.

4 Prescriptive models that calculate water quality conditions but do not directly indicate appropriate management options for a given situation, and:

5 Management or optimization models that incorporate selection procedures to choose the most suitable option according to a set of criteria that is appropriate for a given situation.

This group comprises 4th generation models that are very useful for management but are very complicated and difficult to use. See Table 12

* Dynamic optimization of eutrophication by phosphorus removal. Used for a Japanese lake (Matsumura & Yoshiuki 1981)
* Optimal control by selective withdrawal (Fontane *et al.* 1981)
* Optimizing reservoir operation for downstream aquatic resources. Applied on Lake Shelbyville, Illinois (Sale *et al.* 1982).
* GIRL OLGA for cost minimization of eutrophication abatement using time dependent selection from five management options (Schindler & Straškraba 1982). Applied on several reservoirs in the Czech Republic
* Stochastic optimization of water quality (Ellis 1987)
* COMMAS for prediction of environmental multi-agent system (Bouron 1991)
* DELWAG-BLOOM-SWITCH for management of eutrophication control of shallow lakes (van der Molen *et al.* 1994)
* GFMOLP, a fuzzy multi-objective program for the optimal planning of reservoir watersheds (Chang *et al.* 1996)

Table 12. Models for use in water quality optimization

6 Expert systems that use qualitative or quantitative expressions to guide the user towards relevant answers to complex water quality questions.

7 Decision support systems (DSS) represent a further extension of expert systems. They incorporate other computer software products relevant for a specific water quality decision problem.

Only a few DSS (listed in Table 13) are currently available for water quality management decisions

MODEL NAME	PURPOSE	AUTHOR
-----	selection of control strategy for lake eutrophication	Grobler *et al.* 1987
-----	DSS for environmental decisions	Fedra 1990
-----	analysis of environmental catchment policies	Davis *et al.* 1991
MASAS	evaluation of micropollutants	Ulrich *et al.* 1995
AQUATOOL	water resources management	Andreu *et al.* 1991
HEC-3	multipurpose quantitative operation of reservoir systems	Haestad Methods 1993

Table 13. Decision Support Systems for water quality management.

The interactive DSS function (Figure 47) allows the user to try various versions of decisions under different possible situations.

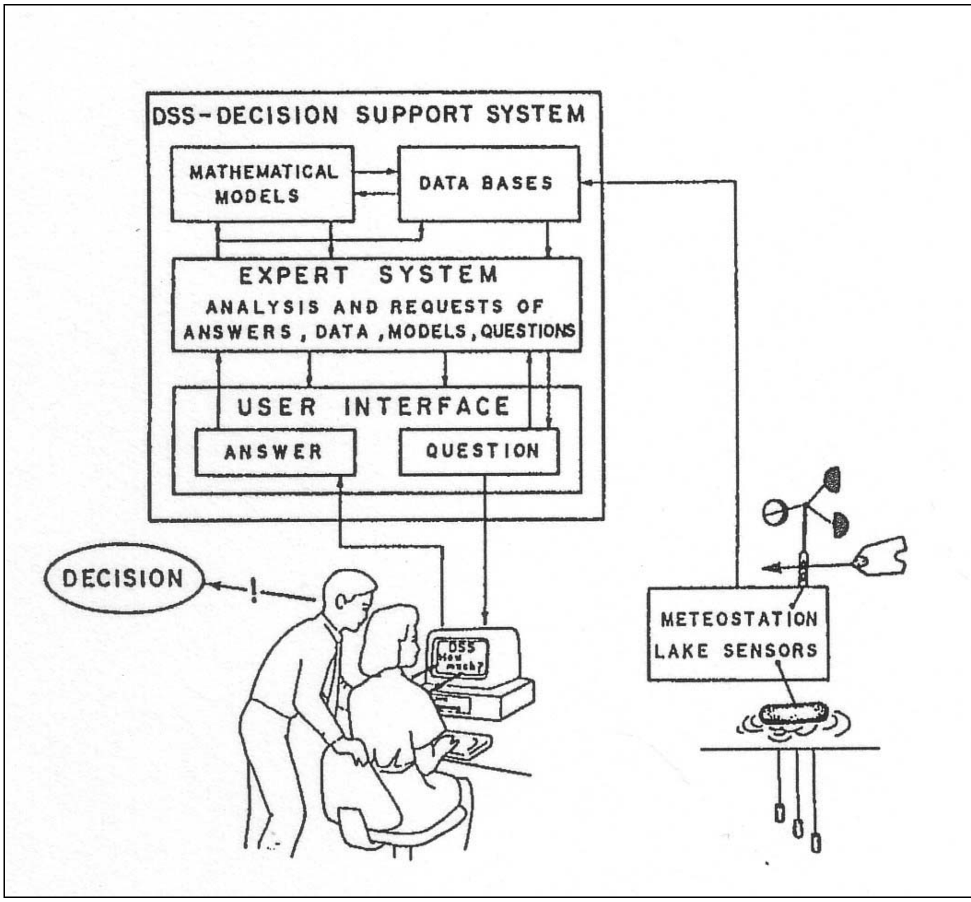


Figure 47. The structure of a Decision Support System for water quality control.

4.3.3 Selection of an appropriate model

Appropriate model selection requires balance between the importance of the problem, cost, available time, available personnel and the availability of the adequate models.
N.B. Models only produce a gross simplification of reality and caution is always necessary when considering model results.

4.4 Reservoir cascades – a special case in terms of management

4.4.1 Introduction

Management of reservoir cascades presents a unique range of problems and options. In a series of reservoirs, if no additional sources of pollution, other than those present in the inflowing river of the highest reservoir, are introduced, considerable improvement in water quality is obtained. This is particularly so if the reservoirs are stratified, and the retention times exceed specific limits. However, where local pollution is introduced downstream of the uppermost reservoir, successive improvement of water quality only occurs if the local pollution does not exceed improvements achieved upstream by particular operations such as selective off-take.

Complex management problems can arise when different uses are required for individual reservoirs in the cascade. In these cases it is necessary to find a balanced optimized solution for the overall multi-goal performance of the system. Models that address water quality and quantity demands are required to integrate parameters such as flow, water levels, pollution, and stratification.

Changes that occur in reservoirs downstream of the first reservoir of a cascade tend to remain operative throughout the entire system. This is the basis for the Cascading Reservoir Continuum Concept (CRCC), which has been proposed for handling the ecological processes at a system level (Barbosa et al., 1999).

4.4.2 Theoretical effects of upstream reservoirs on reservoirs downstream

Some theoretical considerations of the effects of upstream reservoirs on those downstream have been grouped by Straškraba (1990).

4.4.2.1 Physical factors

Compared with the temperature of the upstream reservoirs, those downstream manifest the following:

- (i) Decrease of surface temperatures in spring (max. temperatures nearly identical)
- (ii) Increase in bottom temperatures
- (iii) Increase in mixing depth
- (iv) Increase in Birgean heat budget (the amount of heat required to raise temperature from annual minimum to annual maximum)
- (v) Increase in depth of inflow stream in spring
- (vi) Increase in intensity of mixing of inflow with reservoir water

4.4.2.2 Chemical factors

Provided additional pollutants are insignificant, compared with chemical factors of upstream reservoirs, downstream reservoirs show:

(i) Decrease in turbidity
(ii) Decrease in organic load and colour (with consequences for light conditions)
(iii) Decrease in P-concentration
(iv) Decrease in oxygen concentration at inflow to lower reservoir

4.4.2.3 Indirect effects of upstream reservoirs on downstream reservoirs

(i) Vertical distribution of conservative chemical substances is more uniform due to increased mixing, but this does not apply to biologically affected chemical variables.
(ii) Decrease in primary production due to increased mixing depth and decreased P.
(iii) Phytoplankton composition tends to shift towards more oligotrophic assemblages
(iv) Decrease in oxygen concentration of deeper strata.

4.4.3 Limnological investigations of reservoir cascades

Worldwide, numerous cascading reservoir systems have been constructed on large rivers. These include the Dnieper (Ukraine), Volga (Russia), Ebro (Spain), Zambezi (Southern Africa), Missouri, Colorado (USA), Paranaiba, Tiete, Paranapanema (Brazil) and Parana (Brazil/Paraguay). Cascades on smaller rivers are plentiful. Some South African examples are the uMgeni River, Thukela River, Orange River, and Buffalo River.

Limnological investigations of whole cascade systems are not very numerous despite their obvious hydrological and suspected functional connectivity ((Barbosa et al., 1999; Litvinow and Roschchupko, 1994; Korneva and Solovyova, 1999). Studies involve mostly transport mechanisms of, for example natural suspended matter (Ibanez et al., 1996), or toxic substances (Iskra and Linnik, 1994; Linnik, 1995a, 1995b; Gapeeva et al., 1997). Transport of radionuclides, especially ^{137}Cs, and its biological effects along the Dnieper reservoir cascade have been documented in detail (Klenus et al., 1992; Sirenko et al., 1992; Rogal and Dobrynskij, 1994; Shevchenko, 1995).

Various aspects of the reservoir cascade of the Tiete River in Brazil have been investigated by a number of authors (Abe et al., 2002, 2003; Silva and Matsumura-Tundisi, 2002; Padisak et al., 2000; Tundisi et al., 1991), as has the reservoir cascade of the Dnieper River in the Ukraine. Other systems which have been studied are e.g. the Colorado River (USA) (Shannon et al., 2001), the Tennessee Valley Authority reservoir cascade on the Columbia River (USA) (Higgins and Brock, 1999), the Guadiana River (Spain) (Armengol et al. 1990) and the Volga River (Russia) (Pautova and Nomokonova, 2004; Gapeeva et al., 1997; Litvinow and Roschchupko, 1994).

4.4.4 Management of water quality of reservoir outflows

Three basic options are currently in use:

4.4.4.1 Management of the water quality in the reservoir watershed and reservoir.

This is obviously the best option and has been discussed at length in this report.

4.4.4.2 Use of selective offtakes

Selective offtakes use an outflow depth that corresponds to the layer of best quality water in the reservoir. This is important for river outflow considerations. Hydrological changes within a reservoir may change this optimal depth, a factor which has to be taken into account by water managers, water quality models using hydrodynamic parameters being of considerable assistance in this regard.

4.4.4.3 Management of reservoir outflows

Possibilities of improving water quality at the location of the dam are limited and consist mainly of aeration at the outflow.

4.4.5 Reservoir multisystems

Reservoir multisystems differ from cascades in that reservoirs are constructed on different rivers, but the water is used in a central location. (e.g. the Lesotho Highlands Water Scheme). This is particularly common in dry areas where a single watershed does not produce enough water. Production of sufficient water is often the dominant management goal in these systems, water quality being of secondary importance. Optimal coordination of the many facets of a reservoir system requires the assistance of computer modelling to provide information for rational operational decisions. The reader is referred to Labadie (2004) for a review of optimal operation of multireservoir systems.

5. LOOKING AHEAD – PERCEIVED FUTURE NEEDS

In this section, an individual's perspective is given on prospective needs and priorities to revitalize impoundment research in South Africa, to secure an adequate contemporary structural, functional and operational understanding of these ecosystems under the particular and spatially variable biophysical conditions applicable in the subcontinent, on which their sustainable management into the future can be based.

Reservoir research and management is a broad cross-cutting domain, spanning a wide array of topics which tend largely to be site-specific. Explicit general recommendations are accordingly difficult to formulate except in commensurately broad and nebulous terms. However, there are certain underlying general issues and prospects that require or warrant enunciation. These range from the need to highlight and variously address and redress severe deficiencies of scientific expertise, the need to attempt integrated synthetic analysis of large (and largely un-worked) databases of relevant information to identify regional particularities and peculiarities in reservoir structure and functioning, and to ensure that reservoir water resources attain adequate national prominence and remain at the forefront of limnological consideration and endeavour. While such suggestions are easy to dismiss as impractical academic ramblings, they are in accord with growing international experiences and concerns about ensuring sustainable fresh water resources globally, and international recognition of the particular susceptibility of developing nations located in water-scarce regions.

The following explicit suggestions are offered. Their numbering largely reflects a potential chronology, rather than implying any ranking of priority or importance. The reality that all ecosystems, including impoundments, are holistic, integrated entities, revitalization of reservoir limnology must be recognized holistically. Any reductionist approach to this endeavour would be logically and practically untenable.

1. Regional and national work-shopping of existing and emerging concerns.

The present literature review is precisely and exclusively that - a review, principally derived from primary literature sources. Internal reports and other 'grey literature' do not feature prominently in its contents. Given the deficiency of recent (post 1990) original research on South African reservoirs, it was inevitable that little primary literature of direct local relevance was encountered in this review. Accordingly, a danger exists that major local issues and concerns have been overlooked or disregarded. For this reason, an adequate 'take' on national issues and concerns requires informed inputs from regional water authorities and limnologists, to deliberate whether or not the present review captures the spectrum of issues and concerns that such practitioners face in their day to day operations, perhaps prioritize these concerns, and explicitly identify additional concerns.

2. Integrated synthesis and analysis of existing baseline information in record data-bases held by agreeable/willing regional water authorities, a) to interrogate and quantify the expressed concerns, b) to identify potential unformulated and/or cryptic issues of relevance, and c) ultimately, to achieve firmer factual foundations on reservoir structure and functioning, at different spatial scales – local catchment, regional, and national levels.

Various agencies in South Africa have explicit responsibilities for water resource monitoring and management at different spatial scales – from individual catchments upwards through water management regions to overall national level. Such authority is vested in water supply agencies – various water boards (such as Umgeni Water, Rand Water, Umhlatuze Water, etc), as well as sectoral divisions within the over-arching governmental Department of Water and Environment Affairs. Collectively, these agencies and organizations hold an enormous limnological data base on rivers and reservoirs, collected and used by these agencies for their specific, generally rather circumscribed purposes. However, apart from one exception related to eutrophication (Department of Water Affairs and Forestry 2002), we are unaware of any directed and concerted attempt to coherently interrogate these information sources with a view to identifying important local, regional or national patterns, trends and peculiarities. An earlier initial compilation by Walmsely & Butty (1980) served to catalogue impoundment-specific circumstances, but made no real attempt to provide any integrated analytical synthesis.

We suggest that relevant data-holding agencies be approached to release or make available their information to facilitate a coherent integrated synthesis to quantify issues relevant to the development and implementation of strategic reservoir management for South Africa. Various central topics merit attention within such a synthesis. Listed in no order of priority, and particularly emphasizing their common if not invariable inter-dependence, these include:-

- Type and intensity of stratification; effects of rainfall-linked seasonal/annual and operationally controlled (i.e. pumped storage) draw-downs on stratification and mixing. Oxygen depletion, internal nutrient loading, changes in underwater light climate and associated/consequential phytoplankton periodicity response patterns.
- General features of reservoir water chemistry – ionic composition, salinization issues, zeta potentials and prospective reduction or removal of light-limitation for phytoplankton growth.
- Phytoplankton community structure, and its prospective control by stratification/mixing, by *in situ* nutrient status, by underwater light climate (mineral/biogenic turbidity), and (where data exists) by grazer intensity. Problem algae, Harmful Algal Blooms (HAB).
- Reservoir food-webs structures, to identify and evaluate prospects for biomanipulation as a potential eutrophication control tool.
- Pathogenic and toxic biota – incidence, severity, and actuating/determinant factors or circumstances.
- Recreational fisheries in reservoirs, and their bearing on prospective biomanipulation pathways and the exotic species problem.

3. **Plan strategic research ventures to address identified primary major concerns, with commensurate commitment to underpin potential prospects for regional and local capacity development**.

Resumption of deeper process orientated reservoir research in South Africa (Allanson 2004) is essential to recapture any fundamental national proficiency in sustainable reservoir management. Traditionally, three process levels have been employed – the physical, the chemical, and the biological. While these cannot be effectively treated entirely independently, they serve as convenient pigeon-holes to identify specialist entities within the integral discipline of river, lake or reservoir limnology. All three components require revitalization. Various specific thrusts can be identified at the outset as clear priority needs.

- Water quality is largely biological (Wetzel 2001), and this reality is manifest most obviously and rapidly in the phytoplankton assemblage. As phytoplankton contributes most strongly to problems of water filtration, problems associated with tastes and odours, and cyanotoxins, it is logical that a greatly improved understanding of phytoplankton periodicity and abundance will offer prospects for better fore-casting of such problems. At the present time, however, most attention to this pivotal taxonomic assemblage is subsumed in determinations of chlorophyll. Improved comprehension of the determinants of phytoplankton composition and abundance in reservoirs, encompassing information regarding the identity of their primary limiting factor(s) – light, temperature, vertical circulation, nutrients (P, N, Si), major ions, salinity, grazing predators and parasites or disease – largely encompassed within their functional determination in terms of C-S-R strategists (Reynolds 1996) – as recently elaborated by Hart (2006) in respect of Mgeni reservoir assemblages, offers a useful and potentially powerful means to fore-cast prospective adverse 'seasonal' changes in water quality, allowing rapid instigation of appropriate management interventions. Monitoring development of toxic cyanophytes may be effectively and conveniently linked to this broad thrust. New tools that promise to assist in these respects have been outlined (see Section 4.1.3).

- Trophic status transitions from oligotrophy and mesotrophy to eutrophy and hypereutrophy that are associated with the process of cultural eutrophication are commonly accompanied by switches from direct grazing food webs to detritus-dominated pathways (based largely on the inability of zooplankton to consume large cyanophytes directly). Microbial loop pathways can accordingly be anticipated to be prominent in the resulting food webs. Local reservoirs that exhibit extended warm-waters conditions during relatively prolonged 'summers' can be expected to manifest unusual characteristics of hypolimnetic oxygen depletion. Associated processes and patterns have not been investigated quantitatively (or qualitatively), but have direct and significant relevance in reservoir management issues (oxygen depletion, internal nutrient loading, food web efficiency, etc). Fundamentally different trophic transfer processes can be anticipated, and merit directed research attention.

- The role and impacts of zooplanktivory by fish on food-web structures. Obligate zooplanktivorous fish are essentially lacking in reservoirs. Facultative zooplanktivory, by mostly littoral-inhabiting juveniles of typically riverine fish, is believed to occur, but has not been quantified or analyzed. Such investigations have direct bearing on vaunted prospects for biomanipulation as a tool to manage eutrophication in South Africa (e.g. Anonymous, 2004). Hard scientific evidence is awaited as yet, however.

4. Mount and execute training courses to assist capacity development in South African and SADC nationals.

Loss of capacity in the discipline of reservoir limnology in South(ern) Africa has reached a critical level. Many, perhaps most, of its few experienced practitioners that remain in South Africa today are approaching or have passed retirement age – heralding a commensurate disappearance of local expertise. A remarkably short window of time remains in which to strategically utilize their expertise, and instill some 'new generation' human capacity to resuscitate competence in process-based reservoir limnology and its application in management.

5. Instill stimulating and encouraging incentives and initiatives to facilitate realistic professional practitioner career paths aligned to reservoir limnology and ecohydrology.

Regarding reservoirs, Tundisi (1993) has poignantly stated: "The protection, recovery and optimization of multiple uses of these artificial ecosystems, can only be achieved if a sound ecological basis is constructed with long term observations, experimental studies and theoretical approaches in which seasonal changes, the aging process of the reservoir under anthropogenic actions and the interactions of the reservoirs with the watershed are followed up". Similar sentiments are embedded in the very incisive short preface to the most recent International Conference on Reservoir Limnology and Water Quality (Kennedy et al. 2003) that is appended below in full (our shading emphases) for ease of reference. Realization of this need will demand a core of adequately trained initiates competent to develop regionally appropriate levels of understanding to achieve the outcomes summarized by Tundisi (1993), and Kennedy et al (2003).

6. Encourage multipurpose use of reservoirs

Apart from its obvious utilitarian value, water is widely recognized as having important impacts on the spiritual and emotional well-being of humankind, whose early progenitors have even been named *limnetic people* (Allanson, 2004) in recognition of their origin. Given the scarcity of this resource in South Africa, encouraging the multipurpose use of reservoirs can be deemed to offer benefits to their sustainable management, although will inevitably also introduce some contradictory "tensions" and disbenefits too. Recreational use of these water-bodies for leisure swimming, angling and boating, alongside their various primary functions of potable water supply, flood control, hydro-power generation, etc., is likely to generate greater public awareness of their importance, and wider vested interests in their "condition". Careful resistance to potential dangers – of pressures to introduce exotic alien fish species, to develop high-density shore-side housing, and dangers of careless, ignorant or blatant pollution of the reservoirs, can be anticipated and therefore pre-empted from the outset.

7. At a cross-cutting domain between engineering and limnology, greater appreciation of issues relating to long-term prospects of reservoir sustainability (multi-decades to centuries).

Based on international perspectives and experiences, time-frames envisaged in the context of sustainable reservoir operation and management planning have been greatly extended. Formal consideration is now being given to time frames of centuries rather than single or multi-decades as formally contemplated in regard to anticipated reservoir life-spans in particular. This perspective appears especially relevant in South(ern) Africa, where economically viable and hydrologically productive sites for new reservoir placement and construction are diminishing almost as rapidly as existing reservoirs are experiencing rapid capacity loss due to siltation. Reservoir de-commissioning appears not to have been contemplated in the South(ern) African region, but features importantly in strategic water resource management agendas in the Northern Hemisphere.

8. Accelerate and legislate implementation of CMA-based ICM strategies – the only way to prevent the onset and/or stem further degradation of reservoirs.

In keeping with the inescapable reality that reservoirs are integral parts of the catchment-level waterscape, with an inevitable inter-dependence with the biophysical environment and land use practices in the drainage basin, it is equally inescapable that reservoir conditions will largely mirror their catchments. For this reason, assured sustainable maintenance of adequate reservoir quality will intimately and irrevocably pivot around Integrated Catchment Management. Promulgation and the functional operation of incipient Catchment Management Areas assumes very great urgency.

9. The vast majority of impoundments are small farm dams, the limnological features, attributes, and impacts on water quality remain essentially unexplored in the regional context. Methodical and systematic investigation of these reservoirs remains a necessary adjunct to accomplish the requisite holistic consideration of impounded water on a regional or national scale.

Although the individual total storage capacity of small farm dams and similar waterbodies is relatively limited, their collective capacity is significant. Their location immediately at or close to the interface between particular forms of land-use and stream drainages is likely to render them especially vulnerable to disturbance-related impacts or perturbations of local (sub)catchments. Their potential functional role as pre-impoundments to major reservoirs is one feature that warrants particular scrutiny.

10. Specific attention to reservoir cascades is necessary, since many of the country's major supply rivers are serially impounded. While this need is envisaged particularly for major impoundments, it has relevance in the context of small farm dams as well.

Sequential changes in important biophysical attributes along reservoir cascades impart significant impacts on the limnological structure and function of the impounded water bodies. Local applicability of the Cascading Reservoir Continuum Concept (Barbosa et al., 1999) that has been proposed for handling the ecological processes at a system level along such cascades requires scrutiny and possible refinement.

ADDENDIX
(Highlighting by present authors)

Hydrobiologia **504**: xi-xii, 2003.

V. Straškrábová, R. H. Kennedy, O. T. Lind, J. G. Tundisi & J Hejzlar (eds),

Reservoir Limnology and Water Quality. *xi*

Reservoirs and the limnologist's growing role in sustainable water resource management

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The need for water and the building of dams have long been inexorably linked. Reservoirs provide dependable supply of water to meet a range of social, economic and environmental needs, including water for drinking, irrigation, power generation and waste assimilation, as well as the benefits of protection from flooding, sustained flows for navigation, habitat for fish harvesting and aesthetics. The recent global trend in the construction of dams, now amounting to nearly 45,000 large dams in over 140 countries (Icold, 1998), began with modest construction rates during the first half of the 20*th* century but greatly accelerated rates were prompted by the demands of growing economies following World War II. However, rates have declined markedly since the early 1980s, especially in developed countries, as suitable construction sites have been exploited and as environmental concerns over adverse environmental impacts grew.

Despite this downward trend in dam construction, geographical differences in population growth, a new urgency for dependable water supplies and the potential for resource development suggest resurgence in dam construction may be inevitable (World Commission on Dams, 2000). High population growth rates for South America (0.6 to 1.9%), Asia (-0.1 to 2.2%) and Africa (0.9 to 3.0%) foretell of growing and urgent demands for water on these continents in the coming decades. The desire for industrial development in less industrialized countries, the need for relatively 'clean' sources of energy and expanding demands for irrigation water will drive efforts to finance and build more dams. Based on water availability and engineering considerations, these same regions offer the greatest remaining potential for hydropower development (Anonymous, 1998), an observation clearly linked to apparent lending patterns of the World Bank and other global financial institutions. Unfortunately, regions of anticipated future dam construction are, in general, regions where our understanding of the dynamics of limnological events is most limited (Rast & Thornton, 1993). Since much of our understanding of the limnology and ecology of reservoirs is structured by our decades-long study of natural lakes, especially those located in North Temperate regions of North America and Europe, renewed dam construction will raise questions about our ability to effectively manage these resources.

Our understanding of reservoirs as a unique class of lakes is growing but incomplete. Initially, comparisons of characteristics of reservoirs and natural lakes, prompted by a growing appreciation for potential differences, especially with regard to the importance of hydrologic and engineering influences on reservoirs (Thornton et al., 1990), led to general concepts with relevance for management. For instance, construction on higher order streams and in relatively large watersheds, often at their downstream boundary, leads to higher material loads and more rapid flowing for reservoirs and the promotion of marked longitudinal gradients in water quality. Recognition of the importance of relationships between reservoir design and operation (e.g., location on the landscape, depth of release and flushing rate), and limnological consequences provided a basis for ongoing efforts to link engineering and limnology (Kennedy, 1999). Still lacking, however, is a catchment or system-level understanding of the role reservoirs play in the dynamics of hydrologic landscapes, the scale at which we must begin to manage water resources (Winter, 2000).

The realization that sustaining water resources requires the integration of social and economic, as well as environmental considerations offers a challenging opportunity for limnologists (Forsberg, 2002). Can we be effective partners in crafting a new, integrated approach to water management? To do so, we must broaden our understanding of basic limnological processes and help define linkages between ecological outcomes, engineering requirements and societal pressures on aquatic resources. And we must find ways to effectively transfer scientific understanding to those who will guide the future development of the world's water resources (Robarts & Wetzel, 2000).

In this regard, the 4th International Conference on Reservoir Limnology and Water Quality, and the three preceding conferences, provided an international forum for limnologists and water scientists to share experiences and ideas concerning reservoir limnology and water quality, and the opportunity to explore new research directions. The papers included in this special issue add to the knowledge base upon which to build the understanding to effectively guide the management of reservoirs and the water resource landscape of which they are an integral part. Topics discussed include physical limnology; nutrient cycling; the dynamics of microbial food webs and carbon flow; the structure, interactions and long-term changes of pelagic environments; fisheries dynamics; and human impacts and management.

References

Anonymous, 1998. World atlas and industry guide. International Journal of Hydropower and Dams, Aqua Media International Ltd., Surrey, UK.

Forsberg, C., 2002. SIL and the International Water Academy as actors for sustainable water resources. Verhandlungen der Internationalen Vereinigung fur Limnologie 28: 632-634.

Icold, 1998. World Registry of Dams. Computer Database, International Commission on Large Dams, Paris.

Kennedy, R., 1999. Reservoir design and operation: limnological implications and management opportunities. In Tundisi, J.G. & Straškraba, M. (eds), Theoretical Reservoir Ecology and its Application. Backhuys Publishers, The Netherlands: I-28.

Rast, W. & Thornton J. A., 1993. A test of hypotheses relating to the comparative limnology and assessment of eutrophication in semi-arid man-made lakes. In Straškraba, M., Tundisi, J.G. & Duncan, A. (eds), Comparative Reservoir Limnology and Water Quality Management. Kluwer Academic Publishers, The Netherlands: 1-24.

Robarts, R. & Wetzel, R.G., 2000. The looming global water crisis and the need for international education and cooperation. SIL News 29: 1-3.

Thornton, K. W., Kimmel, B.L. & Payne, F.E., 1990. Reservoir Limnology: Ecological Perspectives. John Wiley and Sons. New York. 246 pp.

Winter, T., 2001. The concept of hydrologic landscapes. Journal of the American Water Resources Association. 37: 335-349. World Commission on Dams, 2000. Dams and Development: a New Framework for Decision-Making. Earthscan Publication, London. 404 p.

6. REFERENCES AND BIBLIOGRAPHY.

(Entries marked with an asterisk are articles of direct relevance to the topic of this review, but which are not explicitly cited in this report.)

Abdelwahed, M., Dabaghi, F. and Ouazar, D. (2002). A virtual numerical simulator for aeration effects in lake eutrophication. International Journal of Computational Fluid Dynamics, 16(2): 119-128.

Abe, D.S., Rocha, O., Matsumura-Tundisi, T. and Tundisi, J.G. (2002). Nitrification and denitrification in a series of reservoirs in the Tiete River, south-eastern Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 877-880.

Abe, D.S., Matsumura-Tundisi, T., Rocha, O. and Tundisi, J.G. (2003). Denitrification and bacterial community structure in the cascade of six reservoirs on a tropical river in Brazil. Hydrobiologia, 504: 67-76.

*Abraham, J., Allen, P.M., Dunbar, J.A. and Dworkin, S.I. (1999). Sediment type distribution in reservoirs: sediment source versus morphometry. Environmental Geology, 38(2). 101-110.

Adams, S.M. and Greeley, M.S. (2000). Ecotoxicological indicators of water quality: using multi-response indicators to assess the health of aquatic ecosystems. Water, Air and Soil Pollution, 123: 103-115.

Adeloye, A.J., Nawaz, N.R. and Montaseri, M. (1999). Climate change water resources planning impacts incorporating reservoir surface net evaporation fluxes: A case study. Water Resources Development, 15(4): 561-581.

Adrian, R., Walz, N., Hintze, T., Hoeg, S. and Rusche, R. (1999). Effects of ice duration on plankton succession during spring in a shallow polymictic lake. Freshwater Biology, 41: 621-632.

Agostinho, A.A., Vazzoler, A.E.A. and Thomaz, S.M. (1995). The High Parana River Parana Basin: limnological and ichthyological aspects. In: Tundisi, J.G., Bicudo, C.E.M. and Matsumura-Tundisi, T. (eds). Limnology in Brazil,* Brazilian Academy of Science/Brazilian Society of Limnology, Rio de Janeiro; pp. 59-104.

Agostinho, A.A., Miranda, L.E., Bini, L.M., Gomes, L.C., Thomaz, S.M. and Suzuki, H.I. (1999). Patterns of colonization in Neotropical reservoirs, and prognosis on aging. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 227- 265.

Ahlfeld, D., Joaquin, A., Tobiason, J. and Mas, D. (2003). Case study: Impact of reservoir stratification on interflow travel time. Journal of Hydraulic Engineering, 129(12): 966-975.

Ahn, C.-Y., Chung, A.-S. and Oh, H.M. (2002). Rainfall, phycocyanin, and N:P ratios related to cyanobacterial blooms in a Korean large reservoir. Hydrobiologia, 474: 117-124.

*Akbay, N., Anul, N., Yerli, S., Soyupak, S. and Yurteri, C. (1999). Seasonal distribution of large phytoplankton in the Keban Dam Reservoir. Journal of Plankton Research, 21(4): 771-787.

Alaoui-Mhamdi, M., Aleya, L.and Bahhou, J. (1996). Nitrogen compounds and phosphate of the Driss I reservoir (Morocco): input, output and sedimentation. Hydrobiologia, 335(1): 75-82.

Alavian, V., Jirka, G.H., Denton, R.A., Johnson, M.C. and Stefan, H.G. (1992). Density currents entering lakes and reservoirs. Journal of Hydraulic Engineering, 118(11): 1464-1489.

Albay, M. and Akcaalan, R. (2003). Factors influencing the phytoplankton steady state assemblages in a drinking-water reservoir (Omerli Reservoir, Istanbul). Hydrobiologia, 502: 85-95.

Allan, R.J. (1992). Organic pesticides in aquatic environments with emphasis on sources and fate in the Great Lakes. In: *Guidelines of Lake Management.* Volume 4. International Lake Environment Committee, Kusatsu.

Allanson, B.R. (1995). An introduction to the management of inland water ecosystems in South Africa. *Water Research Commission Report TT 72/95*.

Allanson, B.R. (2004). Limnology in South Africa: Past and present status and future needs. In: Gopal, B. and Wetzel, R.G. (Eds). *Limnology in Developing Countries 4.* International Association of Limnology, pp 1-116.

Allanson, B.R. and Jackson, P.B.N. (1983). Limnology and fisheries potential of Lake le Roux. *South African National Scientific Programmes Report No. 77,* 182pp.

Allee, R.J. and Johnson, J.E. (1999). Use of satellite imagery to estimate surface chlorophyll a and Secchi disc depth of Bull Shoals Reservoir, Arkansas, USA. International Journal of Remote Sensing, 20(6): 1057-1072.

Allinson, G., Nishikawa, M., De Silva, S.S., Laurenson, L.J.B. and De Silva, K. 2002. Observations on metal concentrations in Talapia (*Oreochromis mossambicus*) in reservoirs of South Sri Lanka. Ecotoxicology and Environmental Safety, 51: 197-202.

*Amarasinghe, U.S., Duncan, A., Moreau, J., Scheimer, F., Simon, D. and Vijverberg, J. (2001). Promotion of sustainable capture fisheries and aquaculture in Asian reservoirs and lakes. Hydrobiologia, 458: 181-190.

Ambrose, R.B., Wool, P.E.T.A., Connolly, J.P. and Schanz, R.W. (1988*). WASP4, A hydrodynamic and water quality model – Model theory, user's manual, and programmer's guide*. Environmental Research Laboratory. Office of Research and Development, USEPA, Athens, Georgia 30613.

Ambrosetti, W., Barbanti, L. and Sala, N. (2003). Residence time and physical processes in lakes. Journal of Limnology, 62 (Suppl.1): 1-15.

*Amoros, C. and Bornette, G. (1999). Antagonistic and cumulative effects of connectivity: a predictive model based on aquatic vegetation in riverine wetlands. Archiv fur Hydrobiologie, 145: 311-327.

An, K.-G. and Jones, J.R. (2000). Temporal and spatial patterns in salinity and suspended solids in a reservoir influenced by the Asian monsoon. Hydrobiologia, 436: 179-189.

*An, K.-G. and Jones, J.R. (2000). Factors regulating bluegreen dominance in a reservoir directly influenced by the Asian monsoon. Hydrobiologia, 432: 37-48.

An, J.-G. and Park, S.S. (2002). Indirect influence of the summer monsoon on chlorophyll-total phosphorus models in reservoirs: a case study. Ecological Modelling, 152 (2-3): 191-203.

Andersen, T. and Hessen, D.O. (1991). Carbon, nitrogen and phosphorus content of freshwater zooplankton. Limnology and Oceanography, 36: 807-814.

Andreu, J., Capilla, J. and Sanchis, E. (1991). AQUATOOL, A computer-assisted support system for water resources research management including conjunctive use. In: Loucks, D.P. and DaCosta, J.R. (eds). *Decision Support Systems: Water Resources Planning.* Nato ASI Series, Vol. G26. Springer-Verlag, Berlin.

*Angelini, R. and Petrere, M. (2000). A model for the plankton system of the Broa reservoir, Sao Carlos, Brazil. Ecological Modelling, 126: 131-137.

*Angradi, T.R. and Kubly, D.M. (1995). Concentration and transport of particulate organic matter below Glen Canyon Dam on the Colorado River, Arizona. Journal of the Arizona-Nevada Academy of Science, 28: 12-22.

*Anne, I., Lelek, A. and Tobias, W. (1991). African man-made lakes: critical notes on their ecology and economic contribution. Natural Resources and Development, 33: 7-19.

Anonymous (2004). New hope for troubled waters: The Hartbeespoort Dam test case. The Water Wheel, 3(1): 16-19.

Antenucci, J.P., Ghadouani, A., Burford, M.A. and Romero, J.R. (2005). The long-term effect of artificial destratification on phytoplankton species composition in a subtropical reservoir. Freshwater Biology, 50: 1081-1093.

*Araujo, F.G. and Santos, L.N. (2001). Distribution of fish assemblages in Lajes Reservoir, Rio de Janeiro, Brazil. Brazilian Journal of Biology, 61(4): 563-576.

Araujo-Lima, C.A.R.M., Agostinho, A.A. and Fabre, N. (1995). Trophic aspects of fish communities in Brazilian rivers and reservoirs. In: Tundisi, J.G., Bicudo, C.E.M. and Matsumura-Tundisi, T. (Eds): *Limnology in Brazil.* Academia Brasileira de Ciencias, Rio de Janeiro, Brazil:

*Arauzo, M. and Cobelas, M.A. (1994). Phytoplankton strategies and time scales in a eutrophic reservoir. Hydrobiologia, 291: 1-9.

*Arcifa, M.S., Gomes, E.A.T. and Meschiatti, A.J. (1992). Composition and fluctuations of the zooplankton of a tropical Brazilian reservoir. Archiv fur Hydrobiologie, 123: 479-495.

*Arcifa, M.S. and Meschiatti, A.J. (1993). Distribution and feeding ecology of fishes in a Brazilian Reservoir, Lake Monte Alegre. Interciencia, 18(6): 302-313.

Arcifa, M.S., Starling, F.L.R.M., Sipauba-Tavares, L.H. and Lazzaro, X. (1995). Experimental Limnology. In Tundisi, J.G., Bicudo, C.E.M. and Matsumura-Tundisi, T. (Eds): *Limnology in Brazil.* Academia Brasileira de Ciencias, Rio de Janeiro, Brazil: pp. 257-281.

Arcifa, M.S. and Northcote, T.G. (1997). Need for holistic approaches in food web experiments and biomanipulation in tropical lakes: a Brazilian reservoir experience. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 661-665.

Arenz, R.F., Lewis, W.M. and Saunders, J.F. (1996). Determination of chlorophyll and dissolved organic carbon from reflectance data for Colorado reservoirs. International Journal of Remote Sensing, 17(8): 1547-1566.

*Arfi, R. (2003). The effects of climate and hydrology on the trophic status of Selingue Reservoir, Mali, West Africa. Lakes and Reservoirs: Research and Management, 8: 247-257.

Arfi, R., Bouvy, M., Cecchi, P., Pagano, M. and Thomas, S. (2001). Factors limiting phytoplankton productivity in 49 shallow reservoirs of North Cote d'Ivoire (West Africa). Aquatic Ecosystem Health and Management, 4: 123-138.

*Arfi, R., Bouvy, M., Cecchi, P., Corbin, D. and Pagano, M. (2003). Environmental conditions and phytoplankton assemblages in two shallow reservoirs of Ivory Coast (West Africa). Archiv fur Hydrobiologie, 156: 511-534.

Arheimer, B. and Wittgren, H.B. (1994). Modelling the effects of wetlands on regional nitrogen transport. Ambio. 23: 378-386.

Armengol, J., Sabater, F., Riera, J.L. and Morgui, J.A. (1990). Annual and longitudinal changes in the environmental conditions in three consecutive reservoirs of the Guadiana River (W. Spain). Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 33: 679-6876.

Armengol, J., Garcia, J.C., Comerma, M., Romero, M., Dolz, J., Roura, M., Han, B.H., Vidal, A. and Simek, K. (1999). Longitudinal processes in canyon-type reservoirs: The case of Sau (N.E. Spain). In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 313-345.

Armitage, P.D., Cranston, P.S. and Pinder, L.C.V. (eds) (1995). *The Chironomidae: Biology and Ecology of Non-Biting Midges.* Chapman and Hall. London.

Armon, R. and Kott, Y. (1995). Distribution comparison between coliphages and phages of anaerobic bacteria (*Bacteriodes fragilis*) in water sources, and their reliability as faecal pollution indicators in drinking water. Water Science and Technology, 31(5-6): 215-222.

Arnason, J.G. and Fletcher, B.A. (2003). A 40+ year record of Cd, Hg, Pb and U deposition in sediments of Patroon Reservoir, Albany County, NY, USA. Environmental Pollution, 123: 383-391.

Arndt, H. (1993). Rotifers as predators on components of the microbial web (bacteria, heterotrophic flagellates, ciliates)- a review. Hydrobiologia, 255-256: 231-246.

Arndt, H. and Nixdorf, B. (1991). Spring clear-water phase in a eutrophic lake: control by herbivorous zooplankton enhanced by grazing on components of the microbial web. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 879-883.

Arndt, H. Krocker, M., Nixdorf, B. and Kohler, A. (1993). Long term annual and seasonal changes of metazooplankton and protozooplankton in Lake Muggelsee (Berlin): effects of eutrophication, grazing activities and the impact of predation. International Review of Hydrobiology, 78: 379-402

Arnell, N.W. (1996). *Global Warming, River Flows and Water Resources*. Chichester,Wiley.

Asaeda, T., Nimal Priyantha, D.G., Saitoh, S. and Gotoh, K. (1996). A new technique for controlling algal blooms in the withdrawal zone of reservoirs using vertical curtains. Ecological Engineering, 7: 95-104.

Asaeda, T., Pham, H.S., Nimal Priyantha, D.G., Manatunge, J. and Hocking, G.C. (2001). Control of algal blooms in reservoirs with a curtain: a numerical analysis. Ecological Engineering, 16: 395-404.

Ashton, P.J., Scott, W.E. and Steyn, D.J. (1980). The chemical control of the water hyacinth [*Eichhornia crassipes* (Mart.) Solms]. Progress in Water Technology, 12: 865-882.

Audry, S., Schafer, J., Blanc, G. and Jouanneau, J.-M. (2004). Fifty-year sedimentary record of heavy metals pollution (Cd, Zn, Cu, Pb) in the Lot River reservoirs (France). Environmental Pollution, 132: 413-426.

Aula, I., Braunschweiler, H. and Malin, I. (1995). The watershed flux of mercury examined with indicators in the Tucurui Reservoir in Para, Brazil. The Science of the Total Environment, 175: 97-107.

*Averhoff, O.R.L. (1999). Fish yields in Cuban reservoirs and relationships with some morphometric and edaphic parameters. Lakes and Reservoirs: Research and Management, 4: 75-83.

Avila-Perez, P., Balcazar, M., Zarazua-Ortega, G., Barcelo-Quintal, I. and Diaz-Delgado, C. (1999). Heavy metal concentrations in water and bottom sediments of a Mexican reservoir. The Science of the Total Environment, 234: 185-196

Avila-Perez, P., Garcia-Aragon, J.A., Diaz-Delgado, C., Tejeda-Vega, S. and Reyes-Gutierrez, R. (2002). Heavy metal distribution in bottom sediments of a Mexican reservoir. Aquatic Ecosystem Health and Management, 5(2): 205-216.

*Aydin, A. and Yurdun, T. (1999). Residues of organochlorine pesticides in water sources of Istanbul. Water, Air and Soil Pollution, 111: 385-398.

Baban, S.M.J. (1996). Trophic classification and ecosystem checking of lakes using remotely sensed information. Hydrological Sciences Journal, 41(6): 939-957.

Babut, M.P., Garric, J., Camusso, M and den Besten, P.J. (2003). Use of sediment quality guidelines in ecological risk assessment of dredged materials: Preliminary reflections. Aquatic Ecosystem Health and Management, 6: 359-367.

Baigun, C. and Marinone, M.C. (1995). Cold-temperate lakes of South America: do they fit northern hemisphere models? Archiv fur Hydrobiologie, 135: 23-51.

Baines, W.D. and Leitch, A.M. (1992). Destruction of stratification by bubble-plume. Journal of Hydraulic Engineering, 118(4): 559-577.

*Bakanov, A.I., Skalskaya, I.A. and Flerov, B.A. (2004). Peculiarities of the initial formation of the communities of benthos and periphyton in reservoirs of the upper reaches of the Volga River. Hydrobiological Journal, 40(2): 23-29.

*Baker, D., Young, J. and Arocena, J.M. (2000). An integrated approach to reservoir management: the Williston Reservoir case study. Environmental Management, 25(5): 565-578.

Baker, P.D., Steffensen, D.A., Humpage, A.R., Nicholson, B.C., Falconer, I.R., Lanthois, B., Fergusson, K.M. and Saint, C.P. (2001). Preliminary evidence of toxicity associated with the benthic cyanobacterium *Phormidium* in South Australia. Environmental Toxicology, 16(6): 506-511.

Bakre, P.P., Misra, V. and Bhatnagar, P. (1990). Residues of organochlorine insecticides in fish from Mahala Water Reservoir, Jaipur, India. Bulletin of Environmental Contamination and Toxicology, 45: 394-398.

Baldwin, D.S. (1996). Effects of exposure to air and subsequent drying on the phosphate sorption characteristics of sediments from a eutrophic reservoir. Limnology and Oceanography, 41: 1725-1732.

Balogh, K.V. and Voros, L. (1997). High bacterial production in hypertrophic shallow reservoirs rich in humic substances. Hydrobiologia, 342-343: 63-70.

Banens, R.J. (1990). The limnology of Dumaresq Reservoir, a small monomictic upland lake in northern New South Wales, Australia. Archiv fur Hydrobiologie, 117(3): 319-355.

*Banens, R.J., and Davis, J.R. (1998). Comprehensive approaches to eutrophication management: the Australian example. Water, Science and Technology, 37: 217-225.

Barbiero, R.P., Ashby, S.L. and Kennedy, R.H. (1996a). The effects of artificial circulation on a small north-eastern impoundment. Water Research Bulletin, 32: 575-584.

Barbiero, R.P., Speziale, B.J. and Ashby, S.L. (1996b). Phytoplankton community succession in a lake subjected to artificial circulation. Hydrobiologia, 331: 109-120.

Barbiero, R.P., James, W.F. and Barko, J.W. (1997). The effects of a change in withdrawal operations on phytoplankton and nutrient dynamics in Eau Galle Reservoir, Wisconsin (USA). Internationale Revue der Gesamten Hydrobiologie, 82: 531-543.

Barbiero, R.P., James, W.F. and Barko, J.W. (1999). The effects of disturbance events on phytoplankton community structure in a small temperate reservoir. Freshwater Biology, 42: 503-512.

Barbosa, F.A.R., Padisack, J., Espindola, E.L.G., Borics, G. and Rocha, O. (1999). The Cascading Reservoir Continuum Concept (CRCC) and its application to the Tiete River basin, Sao Paulo State, Brazil. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 425-438.

Barnes, R.S.K. (1980). The unity and diversity of aquatic systems. In: Barnes, R.S.K. and Mann, K.H. (Eds). *Fundamentals of Aquatic Ecosystems.* Blackwell Scientific, Oxford, pp 5-23.

Barone, R., Naselli-Flores, L. (1994). Phytoplankton dynamics in a shallow, hypertrophic reservoir (Lake Arancio, Sicily). Hydrobiologia, 289: 199-214.

Barone, R., Naselli-Flores, L. and Calvo, S. (1991). Plankton communities in the artificial lakes of Sicily (Italy). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 1409-1414.

Barret, S.C.H. (1989). Waterweed invasions. Scientific American, 261: 90-97.

Barrett, P.R.F., Curnow, J.C. and Littlejohn, J.W. (1996). The control of diatom and cyanobacterial blooms in reservoirs using barley straw. Hydrobiology, 340(1-3): 307-312.

*Bass, D. (1992). Colonization and succession of benthic macroinvertebrates in Arcadia Lake, a South-Central USA reservoir. Hydrobiologia, 242: 123-131.

*Battaglin, W.A. and Goolsby, D.A. (1998). Regression models of herbicide concentrations in outflow from reservoirs in the midwestern USA, 1992-1993. Journal of the American Water Resources Association, 34(6): 1369-1390.

Batzer, D.P. (1998). Trophic interactions among detritus, benthic midges and predatory fish in a freshwater marsh. Ecology, 79: 1688-1698.

Batzer, D.P. and Wissinger, S.A. (1996). Ecology of insect communities in nontidal wetlands. Annual Review of Entomology, 41: 75-100.

*Baumann, P.C. (1992). The use of tumours in wild populations of fish to assess ecosystem health. Journal of Aquatic Ecosystem Health, 1: 135-146.

*Bayley, R., Ta, C.T., Sherwin, C.J. and Renton, P.J. (2001). Traditional and novel reservoir management techniques to enhance water quality for subsequent potable water treatment. Water, Science and Technology: Water Supply, 1(1): 9-16.

Becker, D.S. and Bigham, G.N. (1995). Distribution of mercury in the aquatic food web of Onandaga Lake, New York. Water, Air and Soil Pollution, 80: 563-571.

Bedford, K.W., Sykes, R.M. and Libicki, C. (1983). Dynamic advective water quality model for rivers. Journal of Environmental Engineering Division, ASCE 109: 535-554.

Bednarek, A., Zalewski, M., Blaszczyk, M., Dabrowska, E., Czerwieniec, E. and Tomaszek, J. (2002). Estimation of denitrification rate in diversified bottom sediments of the Sulejow Reservoir. Ecohydrology and Hydrobiology, 2(1-4): 297-304.

Bednarz, T. (1995). Assessment of the trophic state of the water of the Dobczyce dam reservoir and its selected tributaries (southern Poland) by the method of the *Chlorella pyrenoidosa* Chick. Biotest. Acta Hydrobiologica, 37(2): 77-85.

*Beklioglu, M., Ince, O. and Tuzun, I. (2003). Restoration of the eutrophic Lake Eymir, Turkey, by biomanipulation after a major external nutrient control. Hydrobiologia, 489: 93-105.

*Bell, S.G. and Codd, G.A. (1994). Cyanobacterial toxins and human health. Review of Medical Microbiology, 5: 256-264.

Bellanger, B., Huon, S., Steinmann, P., Chabaux, F., Velasquez, F., Valles, V., Arn, K., Clauer, N. and Mariotti, A. (2004). Oxic-anoxic conditions in the water column of a tropical freshwater reservoir (Pena-Larga dam, NW Venezuela). Applied Geochemistry 19: 1295-1314.

Benndorf, J. (1990). Conditions for effective biomanipulation: conclusions derived from whole-lake experiments in Europe. Hydrobiologia, 200/201: 187-203.

Benndorf, J. (1995). Possibilities and limits for controlling eutrophication by biomanipulation. International Review of Hydrobiology, 80: 519-534.

Benndorf, J. and Recknagel, F. (1982). Problems of the application of the ecological model SALMO to lakes and reservoirs having various trophic status. Ecological Modelling, 17: 129-145.

Benndorf, J. and Henning, M. (1989). *Daphnia* and toxic blooms of *Microcystis aeruginosa* in Bautzen Reservoir (GDR). International Review of Hydrobiology, 74: 233-248.

Benndorf, J., Kranich, J., Mehner, T. and Wagner, A. (2001). Temperature impact on the midsummer decline of *Daphnia galeata*: an analysis of long-term data from the biomanipulated Bautzen Reservoir (Germany). Freshwater Biology, Vol. 46: 199-211.

*Benndorf, J., Boing, W., Koop, J. and Neubauer, I. (2002). Top-down control of phytoplankton: the role of time scale, lake depth and trophic state. Freshwater Biology, 47: 2282-2295.

*Bennion, H., Juggins, S. and Anderson, N.J. (1996). Predicting epilimnetic phosphorus concentrations using improved diatom-based transfer function and its application to lake management. Environmental Science and Technology, 30: 2004-2007.

Bergman, E., Hamrin, S. and Romare, P. (1999). The effect of cyprinid reduction on the fish community. Hydrobiologia, 404: 65-75.

Bergstrom, A.-K., Algesten, G., Sobek, S., Tranvik, L. and Jansson, M. (2004). Emission of CO_2 from hydroelectric reservoirs in northern Sweden. Archiv fur Hydrobiologie, 159(1): 25-42.

*Bernardi, R.D. and Giussani, G. (1990). Are blue-green algae a suitable food for zooplankton? An overview. Hydrobiologia, 200/201: 29-41.

*Bernhardt, H. (1995). Reservoirs and drinking-water supply – a global perspective. Aqua: 44: 2-17.

Bertolucci, G.C., Gilli, G., Carraro, E., Giacosa, D. and Puppo, M. (1998). Influence of raw water storage on *Giardia*, *Cryptosporidium* and nematodes. Water, Science and Technology, 37(2): 261-267.

Bertrand, C., Fayolle, S., Franquet, E. and Cazaubon, A. (2003). Responses of the planktonic diatom *Asterionella formosa* Hassall to abiotic environmental factors in a reservoir complex (south-eastern France). Hydrobiologia, 501: 45-58.

Bertrand, C., Franquet, E., Chomerat, N. and Cazaubon, A. (2004). An approach to the Intermediate Disturbance Hypothesis at the landscape scale: the effects of hydrodynamic disturbance on phytoplankton communities. Archiv fur Hydrobiologie, 161(3): 351-369.

Betsill, R.K., van den Avyle, M.J. (1994). Spatial heterogeneity of reservoir zooplankton: a matter of timing? Hydrobiologia 277(1): 63-70.

*Beyruth, Z., Caleffi, S., Zanardi, E., Cardosa, E. and Rocha, A.A. (1997). Water quality of Guarapiranga Reservoir, Sao Paulo, Brazil, 1991-1992. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 675-683.

*Bhade, C., Unni, K.S. and Bhade, S. (2001). Limnology and eutrophication of Tawa Reservoir, M.P. State, India. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3632-3635.

Bilge, F., Yazici, B., Dogeroglu, T. and Ayday, C. (2003). Statistical evaluation of remotely sensed data for water quality monitoring. International Journal of Remote Sensing, 24(24): 5317-5326.

*Bini, L.M. (1997). Spatial variation of some limnological parameters in Barra Bonita Reservoir (Sao Paulo, Brazil): a geostatistical approach. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 229-231.

Bini, L.M., Tundisi, J.G., Matsumura-Tundisi, T. and Matheus, C.E. (1997) Spatial variation of zooplankton groups in a tropical reservoir (Broa Reservoir, Sao Paulo State-Brazil). Hydrobiologia, 357: 89-98.

*Bini, L. M., Thomaz, S. M., Murphy, K. J. and Camargo, A. F. M. (1999) Aquatic macrophyte distribution in relation to water and sediment conditions in the Itaipu Reservoir, Brazil. Hydrobiologia, 415: 147-154.

Blaha, L. and Marsalek, B. (2003). Contamination of drinking water in the Czech Republic by microcystins. Archiv fur Hydrobiologie, 158(3): 421-429.

*Blais, J.M. and Kalff, J. (1995). The influence of lake morphometry on sediment focusing. Limnology and Oceanography, 40: 582-588.

Blay, J. (1990). Fluctuations in some hydrological factors and the condition index of *Aspatharia sinuata* (Bivalvia, Unionaceae) in a small Nigerian reservoir. Archiv fur Hydrobiologie, 117(3): 357-363.

Bledzki, L.A. and Ellison, A.M. (2000). Effects of water retention time on zooplankton of shallow rheolimnic reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2865-2869.

Blomqvist, P., Petter, A. and Hyenstrand, P. (1994). Ammonium nitrogen: A key regulatory factor causing dominance of non-nitrogen-fixing cyanobacteria in aquatic systems. Archiv fur Hydrobiologie, 132: 141-164.

Bodaly, R.A. and Fudge, R.J.P. (1999). Uptake of mercury by fish in an experimental boreal reservoir. Archives of Environmental Contamination and Toxicology, 37: 103-109.

Bodzek, D., Luks-Betlej, K. and Janoszka, B. (1998). Occurrence of PAHs in various elements of environment in Zabrze (Upper Silesia, Poland). Water, Air and Soil Pollution, 103: 91-100.

Boersma, M., van Tongeren, O.F.R. and Mooij, W.M. (1996). Seasonal patterns in the mortality of *Daphnia* species in a shallow lake. Canadian Journal of Fisheries and Aquatic Sciences, 53: 18-28.

Boland, K.T. (1995). Some effects on water quality of rapid draw-down in a small, tropical reservoir; Manton River Reservoir, Northern Territory, Australia. In: Timotius, K.H. and Goltenboth, F. (eds). *Tropical Limnology, Vol II.* Satya Wacana Christian University, Salatiga, Indonesia.

Bolotova, N.L. and Konovalov, A.F. (2002). Morpho-pathologic analysis of zander (*Stizostedion lucioperca* L.) in Beloe Lake. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 1609-1612.

*Bonnet, M.P., Poulin, M. and Devaux, J. (2000). Numerical modelling of thermal stratification in a lake reservoir. Methodology and case study. Aquatic Sciences, 62: 105-124.

Bonnet, M.P. and Poulin, M. (2004). DyLEM-1D: a 1D physical and biochemical model for planktonic succession, nutrients and dissolved oxygen cycling. Application to a hyper-eutrophic reservoir. Ecological Modelling, 180: 317-344.

Bonnet, M.P. and Poulin, M. (2002). Numerical modelling of the planktonic succession in a nutrient-rich reservoir: environmental and physiological factors leading to *Microcystis aeruginosa* dominance. Ecological Modelling, 156: 93-112.

Boon, P.I. and Shiel, J.J. (1990). Grazing on bacteria by zooplankton in Australian billabongs. Australian Journal of Marine and Freshwater Research. 41: 247-257.

Boon, P.I., Bunn, S.E., Green, J.D. and Shiel, R.J. (1994). Consumption of cyanobacteria by freshwater zooplankton: implications for the success of “top-down” control of cyanobacteria in Australia. Australian Journal of Marine and Freshwater Research, 45: 875-887.

*Booth, W.E. and Choy, S.C. (1995) Phototrophic bacteria in the Benutan Reservoir, Brunei Darussalem. In: Timotius, K.H. and Goltenboth, F. (eds). *Tropical Limnology, Vol II*. Satya Wacana Christian University, Salatiga, Indonesia, pp. 171-177.

*Borgstrom, R., Braband, A. and Solheim, J.T. (1992). Effects of siltation on resource utilization and dynamics of allopatric brown trout, *Salmo trutta,* in a reservoir. Environmental Biology of Fishes, 34: 247-255.

Borics, G., Grigorsky, I., Szabo, S. and Padisak, J. (2000). Phytoplankton associations in a small hypertrophic fishpond in East Hungary during a change from bottom-up to top-down control. Hydrobiologia, 424: 79-90

*Bormans, M., Sherman, B.S. and Webster, I.T. (1999). Is buoyancy regulation in cyanobacteria an adaptation to exploit separation of light and nutrients? Marine and Freshwater Research, 50: 897-906.

Bormans, M., Ford, P.W. and Fabbro, L. (2005). Spatial and temporal variability in cyanobacterial populations controlled by physical processes. Journal of Plankton Research, 27: 61-70.

Bornette, G., Piegay, H., Citterio, A., Amoros, C. and Godreau, V. (2001). Aquatic plant diversity in four river floodplains: a comparison at two hierarchical levels. Biodiversity and Conservation, 10: 1683-1701.

Bostrom, B., Pettersson, A.K. and Ahlgren, I. (1989). Seasonal dynamics of a Cyanobacteria-dominated microbial community in surface sediments of a shallow eutrophic lake. Aquatic Sciences, 51: 153-178.

Boulton, A.J. and Brock, M.A. (1999). *Australian Freshwater Ecology: Process and Management*. Gleneagles Publishing, Adelaide.

Bouron, T. (1991). COMMAS: A communication and environment model for multi-agent systems. In: Mosekilde, E. (ed). *Modelling and Simulation 1991*, European Simulation Multiconference, June 17-19, 1991. The Society for Computer Simulations International, Copenhagen, Denmark; pp220-225.

*Bouvy, M., Molica, R, De Oliveira, S., Marinho, M. and Beker, B. (1999). Dynamics of a toxic cyanobacterial bloom *(Cylindrospermopsis raciborskii)* in a shallow reservoir in the semi-arid region of Northeast Brazil. Aquatic Microbiology Ecology, 20: 285-297.

Bouvy, M., Falcao, D., Marinho, M., Pagano, M. and Moura, A. (2000). Occurrence of *Cylindrospermopsis* (Cyanobacteria) in 39 Brazilian tropical reservoirs during the 1998 drought. Aquatic Microbiology Ecology, 23: 13-27.

Bouvy, M., Nascimento, S.M., Molica, R.J.R., Ferreira, A., Huszar, V. and Azevedo, S.M.F.O. (2003). Limnological features in Tapacura Reservoir (northeast Brazil) during a severe drought. Hydrobiologia, 493: 115-130.

Brancelj, A. and Blejec, A. (1994). Diurnal vertical migration of *Daphnia hyalina* Leydig, 1860 (Crustacea: Cladocera) in Lake Bled (Slovenia) in relation to temperature and predation. Hydrobiologia, 284: 125-136.

Brabrand, A., Faafeng, B.A.. and Nilssen, J.P.M. (1990). Relative importance of phosphorus supply to phytoplankton production: Fish excretion versus external loading. Canadian Journal of Fisheries and Aquatic Sciences, 47: 364-372.

*Branco, C.W.C. and Senna, P.A.C. (1996). Relations among heterotrophic bacteria, chlorophyll *a*, total phytoplankton, total zooplankton and physical and chemical features in the Paranoa Reservoir, Brasilia, Brazil. Hydrobiologia, 337: 171-181.

Branco, C.W.C., Rocha, M.-I., Pinto, G.F.S., Gomara, G.A. and De Filippo, R. (2002). Limnological features of Funil Reservoir (R.J., Brazil) and indicator properties of rotifers and cladocerans of the zooplankton community. Lakes and Reservoirs: Research and Management, 7: 87-92.

Brandenberger, J. Louchouarn, P. Herbert, B. and Tissot, P. (2004). Geochemical and hydrodynamic controls on arsenic and trace metal cycling in a seasonally stratified US sub-tropical reservoir. Applied Geochemistry, 19: 1601-1623.

*Brandl, Z. (1994). The seasonal dynamics of zooplankton biomass in two Czech reservoirs. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 40: 127-135.

Branstrator, D.K. (1998). Predicting diet composition from body length in the zooplankton predator *Leptodora kindtii.* Limnology and Oceanography, 43: 530-535.

Branstrator, D.K. and Lehman, J.T. (1991). Invertebrate predation in Lake Michigan regulation of *Bosmina longirostris* by *Leptodora kindtii.* Limnology and Oceanography, 36: 483-495.

Bren, N.V. (2001). Use of invertebrates in monitoring of aquatic ecosystems contaminated with heavy metals (A review). Hydrobiological Journal, 37(2): 66-79.

Brendonck, L., Maes, J., Rommens, W., Dekeza, N., Nhiwatiwa, T., Barson, M., Callebaut, V., Phiri, C., Moreau, K., Gratwicke, B. et al. (2003). The impact of water hyacinth (*Eichhornia crassipes*) in a eutrophic subtropical impoundment (Lake Chivero, Zimbabwe). II. Species diversity. Archiv fur Hydrobiologie, 158(3): 389-405.

Brett, M.T. (1989). Zooplankton communities and acidification processes (a review). Water, Air and Soil Pollution, 44: 387-414.

Breukelaar, E., Lammens, E.H.R.R., Klein Breteler, J.G.P. and Tatrai, I. (1994) Effects of benthivorous bream (*Abramis brama* L.) and carp on sediment resuspension and the concentrations of nutrients and chlorophyll *a*. Freshwater Biology, 32: 113-121.

*Brodeur, J.C., Sherwood, G., Rasmussen, J.B. and Hontela, A. (1997). Impaired cortisol secretion in yellow perch *(Perca flavescens)* from lakes contaminated by heavy metals: in vivo and in vitro assessment. Canadian Journal of Fisheries and Aquatic Sciences, 54: 2752-2758.

Brookes, J.D., Ganf, G.G., Green, D. and Whittington, J. (1999). The influence of light and nutrients on buoyancy, filament aggregation and flotation of *Anabaena circinalis.* Journal of Plankton Research, 21: 327-341.

*Brookes, J.D. and Ganf, G.G. (2001). Variations in the buoyancy response of *Microcystis aeruginosa* to nitrogen, phosphorus and light. Journal of Plankton Research, 23(12): 1399-1411.

*Brookes, J.D., Lewis, D., Linden, L. and Burch, M.D. (2002). On-line monitoring of reservoirs for risk management: linking reservoir data and treatment for improved management. Water, 29(5): 20-27.

Brookes, J.D., Antenucci, J., Hipsey, M., Burch, M.D., Ashbolt, N.J. and Ferguson, C. (2004). Fate and transport of pathogens in lakes and reservoirs. Environment International, 30: 741-759.

*Brosse, S., Lek, S. and Dauba, F. (1999). Predicting fish distribution in a mesotrophic lake by hydroacoustic survey and artificial neural networks. Limnology and Oceanography, 44(5): 1293-1303.

Brosse, S., Dauba, F., Oberdorff, T. and Lek, S. (1999). Influence of some topographical variables on the spatial distribution of lake fish during summer stratification. Archiv fur Hydrobiologie, 145: 359-371.

Bruce, J.P. (1991). Impact of climate change on lakes and reservoirs. In: *Guidelines of Lake Management. Volume8.* International Lake Environment Committee, Kusatsu.

*Brugger, A., Reitner, B., Kolar, I., Queric, N. and Herndl, G.J. (2001). Seasonal and spatial distribution of dissolved and particulate organic carbon and bacteria in the bank of an impounding reservoir on the Enns River, Austria. Freshwater Biology, 46: 997-1016.

*Bruner, K.A., Fisher, S.W. and Landrum, P.F. (1994). The role of the zebra mussel, *Dreissena polymorpha* in contaminant cycling. 1. The effect of body size and lipid content on the bioconcentration of PCBs and PAHs. Journal of Great Lakes Research, 20: 725-734.

Brzakova, M., Hejzlar, J. and Nedoma, J. (2003). Phosphorus uptake by suspended and settling seston in a stratified reservoir. Hydrobiologia, 504: 39-49.

*Bucka, H. (1989). Ecology of selected planktonic algae causing water blooms. Acta Hydrobiologica, 31(3-4): 207-258.

*Bucka, H., Zurek, R. and Kasza, H. (1993). The effect of physical and chemical parameters on the dynamics of phyto- and zooplankton development in the Goczalkowice Reservoir (southern Poland). Acta Hydrobiologica, 35(2): 133-151.

Bulger, A.J., Cosby, B.J. and Webb, J.R. (2000). Current, reconstructed past and protected future status of brook trout (*Salvelinus fontinalis*) streams in Virginia. Canadian Journal of Fisheries and Aquatic Sciences, 57: 1515.

Bunt, C.M., MacIsaac, H.J. and Sprules, W.G. (1993). Pumping rates and projected filtering impacts of juvenile zebra mussels *(Dreissena polymorpha)* in Western Lake Erie. Canadian Journal of Fisheries and Aquatic Sciences, 50: 1017-1022.

Burns, C.W. and Xu, Z. (1990). Utilization of colonial bacteria and algae by freshwater calanoid copepods: survivorship and reproduction of adult *Boeckella* spp. Archiv fur Hydrobiologie, 177: 257-270.

Burns, C.W. and Schallenberg, M. (1996). Relative impacts of copepods, cladocerans and nutrients on the microbial food web of a mesotrophic lake. Journal of Plankton Research, 18: 683-714.

Burns, D.H. (1994). Hydrologic effects of climatic change in west-central Canada. Journal of Hydrology, 160: 53-70.

Burns, F.L. (1998). Case study: automatic reservoir aeration to control manganese in raw water Maryborough Town water supply Queensland, Australia. Water Science and Technology, 37(2): 301-308.

Burris, V.L. and Little, J.C. (1998). Bubble dynamics and oxygen transfer in a hypolimnetic aerator. Water Science and Technology, 37(2): 293-300.

*Cairns, J., Mc Cormick, P.V. and Niederlehner, B.R. (1993). A proposed framework for developing indicators of ecosystem health. Hydrobiologia, 263: 1- 44.

Caldwell, C.A., Canavan, C.M. and Bloom, N.S. (2000). Potential effects of forest fire and storm flow on total mercury and methylmercury in sediments of an arid-lands reservoir. Science of the Total Environment, 260: 125-133.

Caleffi, S., Zanardi, E. and Beyruth, Z. (1994). Trophic state of Guarapiranga Reservoir in 1991/92. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1306-1310.

Calijuri, M.C. and Dos Santos, A.C.A. (1996). Short-term changes in the Barra Bonita reservoir (Sao Paulo, Brazil): emphasis on the phytoplankton communities. Hydrobiologia, 330 (2): 163-175.

Canavan, C.M., Caldwell, C.A. and Bloom, N.S. (2000). Discharge of methylmercury-enriched hypolimnetic water from a stratified reservoir. Science of the Total Environment, 260: 159-179.

Canosa, A. and Pinilla, G. (1999). Bacteriological eutrophication indicators in four Colombian water bodies (South America). Lakes and Reservoirs, Research and Management, 4: 23-27.

Capblancq, J. and Catalan, J. (1994). Phytoplankton: which, and how much? In: Margalef, R. (Ed). *Limnology Now: a Paradigm of Planetery Problems.* Elsevier Science B.V. Amsterdam, The Netherlands; 9-13.

Caramujo, M.-J., Crispim, M.C. and Boavida, M.-J. (1997). Assessment of the importance of fish predation versus copepod predation on life history traits of *Daphnia hyalina*. Hydrobiologia, 360: 243-252.

Carignan, R. and Lean, D.R.S. (1991). Regeneration of dissolved substances in a seasonally anoxic lake: the relative importance of processes occurring in the water column and in the sediments. . Limnology and Oceanography, 36: 683-707.

Carlton, R.G. and Wetzel, R.G. (1988). Phosphorus flux from lake sediments: Effects of epipelic algal oxygen production. Limnology and Oceanography, 33: 562-570

*Carmichael, W. (1994). The toxins of cyanobacteria. Scientific American, 270: 64-72.

Carpenter, S.R., Frost, T.M., Kitchell, J.F., Kratz, T.K., Schindler, D.W., Shearer, J., Sprules, W., Vanni, M.J. and Zimmerman, A.P. (1991). Patterns of primary production and herbivory in 25 North

American lake ecosystems. In: Cole, J., Lovett, G. and Findlay, S. (eds). *Comparative Analyses of Ecosystems*. Springer-Verlag, New York; pp. 67-96.

Carpenter, S.R. and Kitchell, J.F. (eds). (1993): *The Trophic Cascade in Lakes*. Cambridge University Press, New York, USA.

Carpenter, S.R., Christensen, D.L., Cole, J.J., Cottingham, J.J., He Xi., Hodgson, K.L., Kitchell, J.F., Knight, J.F., Pace, S.E., Post, M.L., Schindler, D.M. & N. and Voichick, D.E. (1995). Biological control of eutrophication in lakes. Environmental Science and Technology, 29: 784-786.

Carpenter, S.R., Kitchell, J.F., Cottingham, K.L., Schindler, D.E., Christensen, D.L., Post, D.M. and Voichick, N. (1996). Chlorophyll variability, nutrient input and grazing: evidence from whole-lake experiments. Ecology, 77: 725-735.

Carpenter, S.R. and Pace, M.L. (1997). Dystrophy and eutrophy in lake ecosystems: implications of fluctuating inputs. Oikos, 78: 3-14.

Carrick, H.J., Aldridge, F.J. and Schelske, C.L. (1993). Wind influences phytoplankton biomass and composition in a shallow productive lake. Limnology and Oceanography, 38: 1179-1192.

*Carrick, H.J., Worth, D. and Marshall, M.L. (1994). The influence of water circulation on chlorophyll-turbidity relationships in Lake Okeechobee as determined by remote sensing. Journal of Plankton Research, 16: 1117-1135.

*Carron, J.C. and Rajaram, H. (2001). Impact of variable reservoir releases on management of downstream water temperatures. Water Resources Research, 37(6): 1733-1743.

Carter, J.C.H., Taylor, W.D. and Tudorancea, C. (1995). A horizonal gradient in zooplankton community structure, and its implications for the relationships among mesozooplankton, microzooplankton and phytoplankton. Archiv fur Hydrobiologie, 133: 197-222.

*Carvalho, M.C., Coelho-Botelho, M.J., Lamparelli, M.C., Roquetti-Humaita, M.H., Salvador, M.E.P., Souza, R.C.R. and Truzzi, A. (1997). Spatial and temporal variations of chlorophyll *a*, plankton and some physico-chemical factors at Billings Complex, Sao Paulo, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 452-457.

Carvalho, P.S.M. and Zanardi, E. (1997). Sediment heavy metal contamination at Billings Reservoir, Sao Paulo, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 684-688.

Casamitjana, X., Serra, T., Colomer, J., Baserba, C. and Perez-Losada, J. (2003). Effects of the water withdrawal in the stratification patterns of a reservoir. Hydrobiologia 504: 21-28.

Casco, M.A. (1997). Epilithic algal strategies in a reservoir with periodic water-level changes. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 458-462.

Catoggio, J.A. (1992). Other organic toxic substances. In: *Guidelines of Lake Management. Volume 4.* International Lake Environment Committee, Kusatsu.

*Celma, A.V. and Tubau, J.O. (1993). The eutrophication process in Sau reservoir (NE Spain): A long term study. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1247-1256.

Cenci, R.M. (2000). The use of aquatic moss *(Fontinalis antipyretica)* as monitor of contamination in standing and running waters: limits and advantages. Journal of Limnology, 60 (Suppl.1): 53-61.

Cerny, M. and Bytel, J. (1991). Density and size distribution of *Daphnia* population at different fish predation levels. Hydrobiologia, 25: 199-208.

*Cetin, A.K. (2004). Seasonal distribution of phytoplankton in Orduzu Dam Lake (Malatya, Turkey). Turkish Journal of Botany, 28: 279-285.

Chambers, P.A. (1987). Nearshore occurrence of submerged aquatic macrophytes in relation to wave action. Canadian Journal of Fisheries and Aquatic Sciences, 44: 1666-1669.

Chang, H.H., Harrison, L.L., Lee, W. and Tu, S. (1996). Numerical modelling for sediment-pass-through reservoirs. Journal of Hydraulic Engineering, 122(7): 381-388.

Chang, N.-B., Weng, C.G., Chen, Y.L. and Yong, Y.C. (1996). A grey fuzzy multiobjective programming approach for the optimal planning of a reservoir watershed. Part A: Theoretical development. Water Research, 30: 2329-2334.

Chang, S.-P. and Wen, C.-G. (1997). Changes in water quality in the newly-impounded subtropical Feitsui Reservoir, Taiwan. Journal of the American Water Resources Association, 33(2): 343-357.

*Chang, S.-P. and Wen, C.-G. (1998). Nutrient release from inundated land of a tropical reservoir (Nanhua Reservoir, Taiwan). Water, Science and Technology, 37(2): 325-332.

Chanson, H. (1998). Extreme reservoir sedimentation in Australia: A review. International Journal of Sediment Research, 13(3): 55-63.

Chapra, S.C. and Canale, R.P.(1991). Long-term phenological model of phosphorus and oxygen for stratified lakes. Water Research, 25: 707-715.

Chellappa. N.T. and Costa, M.A.M. (2003). Dominant and co-existing species of Cyanobacteria from a eutrophicated reservoir of Rio Grande do Norte State, Brazil. Acta Oecologia, 24: S3-S10.
.
Chen, B.F. and Hung, T.K. (1993). Dynamic pressure of water and sediment on rigid dam. ASCE Journal of Mechanical Engineering, 19(7): 1411-1433.

Chen, L. (2003). A study of applying genetic programming to reservoir trophic state evaluation using remote sensor data. International Journal of Remote Sensing, 24(11): 2265-2275.

Chen, W., Tan, S.K. and Tay, J.H. (1996). Distribution, fractional composition and release of sediment-bound heavy metals in tropical reservoirs. Water, Air and Soil Pollution, 92: 273-287.

*Chen, X., Brimicombe, A.J., Whiting, B.M. and Wheeler, C. (1998). Enhancement of three-dimensional reservoir quality modelling by Geographic Information Systems. Geographical and Environmental Modelling, 2(2): 125-139.

Cheng, K.-S. and Lei, T.-C. (2001). Reservoir trophic state evaluation using Landsat TM images. Journal of the American Water Resources Association, 37(5): 1321-1334.

Chikwenhere, G.P. (1994). Biological control of water lettuce in various impoundments of Zimbabwe. Journal of Aquatic Plant Management, 32: 27-29.

*Chitale, S.V., Sinha, S. and Mishra, P.K. (1998). Estimation of delta profile in the Indravati Reservoir. Journal of Hydraulic Engineering, 124: 109-113.

Choubey, V.K. and Subramanian, V. (1992). Estimation of suspended solids using Indian Remote Sensing Satellite-1A data: a case study from Central India. International Journal of Remote Sensing. 13(8): 1473-1486.

Christian, B.W., Early, C.N. and Lind, O.T. (2002). Factors affecting bacterioplankton size and abundance in anoxic hypolimnia. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 705-709.

*Christoffersen, K., Riemann, B., Hansen, L.R., Klysner, A. and Sorensen, H.B. (1990). Qualitatiave importance of the microbial loop and plankton community structure in a eutrophic lake during a bloom of cyanobacteria. Microbial Ecology, 20: 253-272.

Christoffersen, K., Riemann, B., Klysner, A. and Sondergaard, M. (1993). Potential role of fish predation and natural popuations of zooplankton in structuring a plankton community in eutrophic lake water. Limnology and Oceanography, 38: 561-573.

Chrzanowski, T.H. and Simek, K. (1990). Prey-size selection by freshwater flagellated protozoa. Limnology and Oceanography, 35: 1429-1436.

*Chung, S. and Gu, R.R. (2003). Estimating time-variable transformation rate of atrazine in a reservoir. Advances in Environmental Research, 7: 933-947.

Clancy, J.L., Bukhari, Z., Hargy, T.M., Bolton, J.R., Dussert, B.W. and Marshall, M.M. (2000). Using UV to inactivate *Cryptosporidium.* Journal of the American Water Works Association, 92(9): 97-104.

Clark, M.E., Rose, K.A., Chandler, J.A., Richter, T.J., Orth, D.J. and van Winkle, W. (1998). Simulating smallmouth bass reproductive success in reservoirs subject to water level fluctuations. Environmental Biology of Fishes, 51: 161-174.

Clarkson, T.W. (1990). Human health risks from methylmercury in fish. Environmental Toxicology and Chemistry, 9: 957-961.

*Clement, A., Somlyody, L. and Koncsos, L. (1998). Modelling the phosphorus cycle in the Kis-Balaton Upper Reservoir. Water, Science and Technology, 37: 113-120.

*Codd, G.A. (1995). Cyanobacterial toxins: Occurrence, properties and biological significance. Water, Science and Technology, 32: 149-156.

Cohen, A.N. and Carlton, J.T. (1998). Accelerating invasion rate in a highly invaded estuary. Science, 279: 555-557.

Cole, G.A. (1994). *Textbook of Limnology (4th ed.)* Waveland Press, Inc., Prospect Heights, IL.

Cole, J.J., Pace, M.L., Caraco, N.F. and Steinhart, G.S. (1993). Bacterial biomass and cell size distributions in lakes: More and larger cells in anoxic waters. Limnology and Oceanography, 38: 1627-1632.

Cole, T.M. and Hannan, H.H. (1990). Dissolved oxygen dynamics. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds*).* *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 71-10515-41.

*Colvin, M.A. (2000). Criteria and procedures for evaluating the quality of fish populations in reservoirs. Environmental Science and Policy, 3: S127-S132.

Comerma, M., Garcia, J., Armengol, J., Romero, M. and Simel, K. (2001). Planktonic food web structure along the Sau Reservoir (Spain) in summer 1997. International Review of Hydrobiology, 86: 195-209.

Comerma, M., Garcia, J.C., Romero, M., Armengol, J. and Simek, K. (2003). Carbon flow dynamics in the pelagic community of the Sau Reservoir (Catalonia, NE Spain). Hydrobiologia 504: 87-98.

*Connel, D.B. and Sanders, J.G. (1999). Variation in cadmium uptake by estuarine phytoplankton and transfer to the copepod *Eurytemora affinis.* Marine Biology, 133: 259-265.

Cooke, G.D., Welch, E.B., Peterson, S.A. and Newroth, P.R. (1993). *Restoration and Management of Lakes and Reservoirs.* 2nd edition., Lewis Publishers, Boca Raton, Florida, 548pp.

*Copertino, V.A., Molino, B. and Telesca, V. (1998). Spatial and temporal evolution of water quality in reservoirs. Physics and Chemistry of the Earth, 23(4): 475-478.

Cotner, J.B. and Wetzel, R.G. (1991). 5'-Nucleotidase activity in a eutrophic lake and an oligotrohic lake. Applied Environmental Microbiology, 57: 1306-1312.

Coveney, M.F. and Wetzel, R.G. (1995). Biomass, production and specific growth rate of bacterioplankton and coupling to phytoplankton in an oligotrophic lake. Limnology and Oceanography, 40: 1187-1200.

*Cox, J.D., Padley, M.B. and Hannon, J. (1998). Use of computational fluid dynamics to model reservoir mixing and destratification. Water, Science and Technology, 37(2): 27-234.

Craik, S.A., Weldon, D., Finch, G.R., Bolton, J.R. and Belosevic, M. (2001). Inactivation of *Cryptosporidium parvum* oocysts using medium and low-pressure ultraviolet radiation. Water Research, 35(6): 1387-1398.

Crisman, T.L. and Beaver, J.R. (1990). Applicability of planktonic biomanipulation for managing eutrophication in the subtropics. Hydrobiologia, 200-201: 177-185.

*Crispim, M.C. and Watanabe, T. (2001). What can dry reservoir sediments in a semi-arid region in Brazil tell us about cladocera? Hydrobiologia, 442: 101-105.

Crowder, B. M. (1987). Economic costs of reservoir sedimentation: A regional approach to estimating cropland erosion damages. Journal of Soil and Water Conservation, 42(3): 194-197.

Crutchfield, J.U. (2000). Recovery of a power plant cooling reservoir system from selenium bioaccumulation. Environmental Science and Policy, 3: S145-S163.

Cruz-Pizarro, L. and Carillo, P. (1991). Top-down regulation under different species-specific and size-structure grazer assemblages in an oligotrophic lake. Memorie dell'Istituto Italiano di Idriobiologia Milano, 48: 23-37.

*Da Graca, M., Melao, G. and Rocha, O. (2004). Life history, biomass and production of two planktonic cyclopoid copepods in a shallow subtropical reservoir. Journal of Plankton Research, 26(8): 909-923.

Daldorph, P.W.G. (1998). Management and treatment of algae in lowland reservoirs in eastern England. Water Science and Technology, 37(2): 57-63.

Davidson, G.R., Bennett, S.J., Beard, W.C. and Waldo, P. (2005). Trace elements in sediments of an ageing reservoir in rural Mississippi: Potential for mobilization following dredging. Water, Air and Soil Pollution, 163: 281-292.

Davies, B. and Day, J. (1998). *Vanishing Waters*. University of Cape Town Press, Cape Town, South Africa.

*Davies, B.R., Beilfuss, R.D. and Thoms, M.C. (2000). Cahora Bassa retrospective, 1974-1997: effects of flow regulation on the Lower Zambezi River. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2149-2157.

Davies, C.M., Long, J.A.H., Donald, M. and Ashbolt, N.J. (1995). Survival of faecal microorganisms in marine and freshwater sediments. Applied and Environmental Microbiology, 61(5): 1888-1896.

Davies, J. (1985). Evidence for a diurnal horizontal migration in *Daphnia hyalina lacustris* Sars. Hydrobiologia, 120: 103-105.

Davis, J.R., Nanninga, P.M., Biggins, J and Laut, P. (1991). Prototype decision support system for analyzing impact of catchment policies. Journal of Water Resources Planning and Management, ASCE, 117: 399-414.

Davis, S.E. and Reeder, B.C. (2001). Spatial characterization of water quality in seven Eastern Kentucky reservoirs using multivariate analyses. Aquatic Ecosystem Health and Management, 4: 463-477.

*Dawidowicz, P. (1990). Effectiveness of phytoplankton control by large-bodied and small-bodied zooplankton. Hydrobiologia, 200-201: 43-47.

De Bernardi, R.D. and Giussani, G. (1990). Are blue-green algae a suitable food for zooplankton? An overview. Hydrobiologia, 200/201: 29-41.

De Bernardi, R. and Giussani, G. (Eds). (1995). *Biomanipulation in Lakes and Reservoirs Management. Guidelines of Lake Management.* Volume 7. International Lake Environment Committee, Kusatsu.

*De Ceballos, B.S.O., Konig, A. and De Oliveira, J.F. (1998). Dam reservoir eutrophication: a simplified technique for a fast diagnosis of environmental degradation. Water Research, 32(11): 3477-3483.

De Cesare, G., Schleiss, A. and Hermann, F. (2001). Impact of turbidity currents on reservoir sedimentation. Journal of Hydraulic Engineering, 127(1): 6-16

Deegan, L.A. (1993). Nutrient and energy transport between estuaries and coastal marine ecosystems by fish transportation. Canadian Journal of Fisheries and Aquatic Sciences, 50: 74-79.

De Groot, J.C. and van Wijck, C. (1993). The impact of desiccation of a freshwater marsh (Garines Nord, Carmargue, France) on sediment-water-vegetation interactions, Part I. Sediment chemistry. Hydrobiologia, 252: 83-94.

*De la Torre, F.R., Salibian, A. and Ferrari, L. (2000). Biomarkers assessment in juvenile *Cyprinus carpio* exposed to waterborne cadmium. Environmental Pollution, 109: 277-282.

Del Giorgio, P.A. and Scarborough, G. (1995). Increase in the proportion of metabolically active bacteria along gradients of enrichment in freshwater and marine plankton: implications for estimates of bacterial growth and production rates. Journal of Plankton Research, 17: 1905-1924.

*Dekker, A.G. and Peters, S.W.M. (1993). The use of the Thematic Mapper for the analysis of eutrophic lakes: a case study in the Netherlands. International Journal of Remote Sensing, 14: 799-821.

De la Rey, P.A., Taylor, J.C., Laas, A., van Rensburg, L. and Vosloo, A. (2004). Determining the possible application value of diatoms as indicators of general water quality: A comparison with SSAS 5. Water SA, 30(3): 325-332.

De Manuel, J. and Armengol, J. (1993). Rotifer assemblages: a contribution to the typology of Spanish reservoirs., Hydrobiologia, 255-256: 421-428.

De Manuel, J. and Jaume, D. (1993). Zooplankton of reservoirs from the River Ebro basin (Spain): Relationships with some physical, chemical and biological features. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1236-1241.

De Melo, R., France, R. and McQueen, D.J. (1992). Biomanipulation: Hit or myth? Limnology and Oceanography, 37; 192-207.

*Demir, N. (2005). Zooplankton of two drinking water reservoirs in central Anatolia: composition and seasonal cycle. Turkish Journal of Zoology, 29: 9-16.

De Moor, I.J. (1996). Case studies of the exotic species by four alien fish species (*Cyprinus carpio, Micropterus salmoides, Oreochromis macrochir and O. mossambicus)* of freshwater ecosystems in Southern Africa. Transactions of the Royal Society of South Africa, 51: 233-255.

DeMott, W.R. (1990). Retention efficiency perceptual bias, and active choice as mechanisms of food selection by suspension –feeding zooplankton. In: Hughes, R.N. (ed). *Behavioural Mechanisms of Food Selection.* NATO ASI Series, Series G: Ecological Series. Springer, New York; pp. 569-594.

*DeMott, W.R., Gulati, R.D. and Siewertsen, K. (1998). Effects of phosohorus-deficient diets on the carbon and phosphorus balance of *Daphnia magna*. Limnology and Oceanography, 43: 147-161.

*Den Besten, P.J. (1998). Concepts for the implementation of biomarkers in environmental monitoring. Marine Environmental Research, 46: 253-256.

Dendy, F.E., Champion, W.A. and Wilson, R.B. (1973). Reservoir sedimentation surveys in the United States. In: Ackermannn, W.C., White, G.F. and Worthington, E.B. (Eds). *Man-made Lakes: Their Problems and Environmental Effects*. American Geophysical Union, Washington D.C.

Desmarius, K.H. and Tessier, A.J. (1999). Performance trade-off across a natural resource gradient. Oecologia, 120: 137-146.

De Stasio, B.T., Rudstam, L.G., Haning, A., Soranno, P. and Allen, Y.C. (1995). An *in situ* test of the effect of food quality on *Daphnia* population growth. Hydrobiologia, 307: 221-230.

*Dettmers, J.M. and Stein, R.A. (1996). Quantifying linkages among gizzard shad, zooplankton and phytoplankton in reservoirs. Transactions of the American Fisheries Society, 125: 27.

Dias, C.O. (1999). Morphological abnormalities of *Acartia lilljeborgi* (Copepoda, Crustacea) in Espirito Santo Bay (ES, Brazil). Hydrobiologia, 394: 249-251.

Diaz, M.M., Temporetti, P.F. and Pedrozo, F.L. (2001). Response of phytoplankton to enrichment from cage fish farm waste in Alicura Reservoir (Patagonia, Argentina). Lakes and Reservoirs: Research and Management 6(2): 151-158.

*Dibble, E.D., Killgore, K.J. and Harrel, S.L. (1996). Assessment of fish-plant interactions. In: Miranda, L.E. and DeVries, D.R. (Eds). *Multidimensional Approaches to Reservoir Fisheries Management.* American Fisheries Society, Bethesda, Maryland, pp. 357-372.

Diehl, S. (1992). Fish predation and benthic community structure: the role of omnivory and habitat complexity. Ecology, 73: 16-46.

*Di Giovanni, M. V., Goretti, E. and Tamanti, V. (1996) Macrobenthos in Montedoglio Reservoir, central Italy. Hydrobiologia, 321(1): 17-28.

*Dionne, D. and Therien, N. (1997). Minimizing environmental impacts of hydroelectric reservoirs through operational control: a generic approach to reservoirs in northern Quebec. Ecological Modelling, 105: 41-63.

Di Siervi, M.A., Mariazzi, A.A. and Donadelli, J.L. (1995). Bacterioplankton and phytoplankton production in a large Patagonian reservoir (Republica Argentina). Hydrobiologia, 297(2): 123-129.

*Dixit, S.S., Smol, J.P., Kingston, J.C. and Charles, D.F. (1992). Diatoms: Powerful indicators of environmental change. Environmental Science and Technology, 26: 23-33.

*Dobosiewicz, K., Luks-Betlej, K. and Bodzek, D. (2000). Concentration of polycyclic aromatic hydrocarbons and base cations in the air and drinking water of the Silesian Rehabilitation Centre, Poland. Water, Air and Soil Pollution, 118: 101-113.

Dubovskaya, O.P., Gladyshev, M.I., Gubanov, V.G. and Makhutova, O.N. (2003). Study of non-consumptive mortality of Crustacean zooplankton in a Siberian reservoir using staining for live/dead sorting and sediment traps. Hydrobiologia, 504: 223-227.

Dombrovskiy, K.O. (2003). Influence of anthropic contamination on macrozoobenthos of the upper reaches of the Kakhovka Reservoir. Hydrobiological Journal. 39(4): 92-96.

*Donaldson, E.M. (1990). Reproductive indices as measures of the effects of environmental stressors in fish. American Fisheries Society Symposium, 8: 109-122.

Drenner, R.W., Smith, J.D. and Threlkeld, S.D. (1996). Lake trophic state and the limnological effects of omnivorous fish. Hydrobiologia, 319: 213-223

Drenner, R.W. and Hambright, K.D. (1999). Biomanipulation of fish assemblages as a lake restoration technique. Archiv fur Hydrobiologie, 146: 129-165

Duarte, C.M. and Kalff, J. (1986). Littoral slope as a predictor of the maximum biomass of submerged macrophyte communities. Limnology and Oceanography, 31: 1072-1080.

Dubelaar G.B.J., Geerders P.J.F. and Jonker R.R. (2004) High frequency monitoring reveals phytoplankton dynamics. Journal of Environmental Monitoring 6: 946-952.

Dubnyak, S. and Timchemko, V. (2000). Ecological role of hydrodynamic processes in the Dnieper reservoirs. Ecological Engineering, 16: 181-188.

*Dubovskaya, O.P., Gladyshev, M.I., Gubanov, V.G. and Makhutova, O.N. (2003). Study of non-consumptive mortality of Crustacean zooplankton in a Siberian reservoir using staining for live/dead sorting and sediment traps. Hydrobiologia, 504: 223-227.

Duchemin, E.; Lucotte, M.; Canuel, R. and Chamberland, A. (1995). Production of greenhouse gases CH_4 and CO_2 by hydroelectric reservoirs of the boreal region. Global Biogeochemical Cycles, 4: 529-539.

Duchemin, E.; Lucotte, M.; Canuel, R.; Queiroz, A.G.; Almeida, D.C.; Pereira, H.C.; Dezincourt, J. (2000). Comparison of greenhouse gas emissions from an old tropical reservoir with those from other reservoirs worldwide. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 1391-1395.

*Dumestre, J.F., Guezennec, J., Galy-Lacaux, C., Delmas, R., Richard, S. and Labroue, L. (1999). Influence of light intensity on methanotrophic bacterial activity in Petit Saut Reservoir, French Giuana. Applied and Environmental Microbiology, 65(2): 534-539.

*Dumnicka, E. (1993). Profundal macrofauna of the Dobczyce Reservoir (southern Poland) in the fifth year after its filling. Acta Hydrobiologica, 35(4): 329-340.

Duncan, A. and Kubecka, J. (1995). Land/water ecotone effects in reservoirs on the fish fauna. Hydrobiologia, 303: 11-30.

Durkin, T.V. and Herrmann, J.G. (1994). Focusing on the problem of mining wastes: An introduction to acid mine drainage. EPA Seminar Publication No. EPA/625/R-95/007.

*Dutova, N.V. and Kozlova, O.M. (2000). Bacterial periphyton of the Bratsk Reservoir. Hydrobiological Journal, 36(5): 37-42.

Easthope, M.P. and Howard, A. (1999). Simulating cyanobacterial growth in a lowland reservoir. Science of the Total Environment, 241: 17-25.

*Echevarria, F. and Rodriguez, J. (1994). The size structure of plankton during a deep bloom in a stratified reservoir. Hydrobiologia, 284: 113-124.

Ecological Modelling Centre (EMC). (1992). *MIKE12, a short description*. Report from the Ecological Modelling Centre, Horsholm, Denmark.

*Effler, S.W. and O'Donnell, D.M. (2001). Resolution of spatial patterns in three reservoirs with rapid profiling instrumentation. Hydrobiologia, 450: 197-208.

*Egborge, A.B.M. (1990). Eutrophication of Nigeria's Lake Asejire. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 793-802.

Ejsmont-Karabin, J., Weglenska, T. and Wisniewski, R.J. (1993). The effect of water flow rate on zooplankton and its role in phosphorus cycling in small impoundments. Water, Science and Technology, 28: 35-43.

Elbaz-Poulichet, F., Nagy, A. and Cserny, T. (1997). The distribution of redox sensitive elements (U, As, Sb, V and Mo) along a river-wetland-lake system (Balaton Region, Hungary). Aquatic Geochemistry, 3: 267-282.

Elenbaas, P.F.M. and Grundel, C. (1994). Zooplankton composition and abundance in two impoundments in Zimbabwe. Hydrobiologia, 272(1-3): 265-275.

Ellis, J.H. (1987). Stochastic water quality optimization using imbedded chance contraints. Water Resources Research, 23: 2227-2238.

El-Manadely, M.S., Abdel-Bary, R.M., El-Sammany, M.S. and Ahmed, T.A. (2002). Characteristics of the delta formation resulting from sediment deposition in Lake Nasser, Egypt: Approach to tracing lake delta formation. Lakes and Reservoirs: Research and Management, 7(2): 81-86.

Elmoor-Loureiro, L.M.A. (2004). Morphological abnormalities in the cladoceran *Ilyocryptus spinifer* (Apipucos Reservoir, Pernambuco State, Brazil). Brazilian Journal of Biology, 64(1): 53-58.

Elser, J. J. (1999). The pathway to noxious Cyanobacteria blooms in lakes: The food web as the final turn. Freshwater Biology, 42: 537-544.

Elser, J.J. and Hassett, R.P. (1994). A stoichiometric analysis of the zooplankton-phytoplankton interaction in marine and freshwater ecosystems. Nature. 370: 211-213.

Elser, J.J., Sterner, R.W., Galford, A.E., Chrzanowski, T.H., Findlay, D.L., Mills, K.H., Paterson, M.J., Stainton, M.P. and Schindler, D.W. (2000). Pelagic C:N:P stoichiometry in a eutrophied lake: responses to a whole-lake food-web manipulation. Ecosystems, 3: 293-307.

Encina, L. and Granado-Lorencio, C. (1994). Gut evacuation in barbel (*Barbus sclateri* G. 1868) and nase (*Chondrostoma willkommi* S. 1866). Ecology of Freshwater Fishes, 23: 1-8.

Encina, L. and Rodriguez-Ruiz, A. (2002). Patterns of seasonal distribution of the fish assemblage in a reservoir of recent construction. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 641-650.

Epstein, P.R. (2004). Climate change and public health: emerging infectious diseases. In: Encyclopaedia of Energy, Vol. 1. Elsevier Inc.

Eriksson, M.O.G., Henrikson, H., Nilsson, B.I. and Nyman, G. (1980). Predator-prey relations important for biotic changes in acidified lakes. Ambio, 9: 248-249.

Eriksson, M.O.G. and Tengelin, B. (1987). Short-term effect on perch (*Perca fluvialis*) in acidified lakes in southwest Sweden. Hydrobiologia, 146: 187-191.

Etim, L. and Sankare, Y. (1998). Growth and mortality, recruitment and yield of the fresh-water shrimp, *Macrobrachium vollenhoveenii*, Herklots 1851 (Crustacea, Palaemonidae) in the Fahe reservoir, Cote d'Ivoire, West Africa. Fisheries Research, 38: 211-223.

Evangelou, V.P, (1995). *Pyrite oxidation and its control.* Boca Raton. CRC Press, Inc. 293pp.

*Fabian, C., Soria, J.M. and Vicente, E. (2002). Evaluation of fish density in some Spanish reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 560-563.

*Faithful, J.W. and Griffiths, D.J. (2000). Turbid flow through a tropical reservoir (Lake Dalrymple, Queensland, Australia): responses to a summer storm event. Lakes and Reservoirs: Research and Management, 5: 231-247.

*Falconer, I.R. (1996). Potential impact on human health of toxic cyanobacteria. Phycologia, 35: 6-11.

*Falconer, I.R. (1997). Blue-green algae in lakes and rivers: their harmful effects on human health. Australian Biologist, 10: 107-110.

Fanslow, D.L., Nalepa, T.F. and Lang, G.A. (1995). Filtration rates of the zebra mussel (*Dreissena polymorpha*) on natural seston from Saginaw Bay Lake Huron. Journal of Great Lakes Research, 21: 489-500.

Farkas, A., Salanki, J. and Varanka, I. (2003). Crustaceans as biological indicators of heavy metal pollution in Lake Balaton (Hungary). Hydrobiologia, 506-509: 359-364.

Favari, L., Lopez, E., Martinez-Tabche, L. and Diaz-Pardo, E. (2002). Effect of insecticides on plankton and fish of Ignacio Ramirez Reservoir (Mexico): A biochemical and biomagnification study. Ecotoxicology and Environmental Safety, 51: 177-186.

*Fayer, R. and Nerad, T. (1996). Effects of low temperature on viability of *Cryptosporidium parvum* oocysts. Applied Environmental Microbiology, 62(4): 1431-1433.

Fayer, R., Graczyk, T.K., Lewis, E.J., Trout, J.M. and Farley, C.A. (1998). Survival of infectious *Cryptosporidium parvum oocysts* in seawater and eastern oysters (Crassostrea virginica) in the Chesapeake Bay. Journal of Applied and Environmental Microbiology, 64(3): 1070-1074.

Fearnside, P.M. (1995). Hydroelectric dams in the Brazilian Amazon as sources of greenhouse gases. Environmental Conservation, 22: 7-19.

Fearnside, P.M. (2001). Environmental impacts of Brazil's Tucurui Dam: Unlearned lessons for hydroelectric development in Amazonia. Environmental Management, 27(3): 377-396.

Fearnside, P.M. (2002). Greenhouse gas emissions from hydroelectric reservoir (Brazil's Tucurui dam) and the energy policy implications. Water, Air and Soil Pollution, 133: 69-96.

Fedra, K. (1990). Interactive environmental software: Integration, simulation and visualization. In: Pillman, W. and Jaeschke, A. (Eds). Proceedings: *Informatik fur den Umweldschutz,* 5 Symposium, 19-21 September 1990. Wien, Osterreich; pp. 735-744.

Fedra, K., Weigkricht, E. and Winkelbauer, L. (1990). Models, GIS and Expert Systems for Environmental Impact Analysis. In: Pillman, W. (ed). *Computer applications for Environmental Impact Analysis.* International Society for Environmental Protection (ISEP), Envirotech Vienna; pp. 13-22.

Fenger, J. (1993). Sources, Pollutants and abatement possibilities. In: *Guidelines of Lake Management. Volume 5, Acidification.* International Lake Environment Committee, Kusatsu.

*Ferguson, A. J. D. (1997). The role of modelling in the control of toxic blue-green algae. Hydrobiologia, 349:1-4.

Ferguson, C.M., Coote, B.G., Ashbolt, N.J. and Stevenson, I.M. (1996). Relationships between indicators, pathogens and water quality in an estuarine system. Water Research, 30(9): 2045-2054.

Fernandes, C.A. and Crisman, T.L. (1994). Lake Paranoa, management approaches for a tropical urban reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1301-1305.

Fernando, C.H. and Holcik, J. (1991). Fish in reservoirs. International Review of Hydrobiology, 76: 149-167.

*Fernando, H.J.S., Johnstone, H. and Zangrando, F. (1991). Interfacial mixing caused by turbulent buoyant jets. Journal of Hydraulic Engineering, 117(1): 1-20

*Ferris, J.M. and Tyler, P.A. (1992). The effects of inflow and outflow on the seasonal behaviour of a stratified reservoir in temperate Australia – a 20 year analysis. Archiv fur Hydrobiologie, 126: 129-162.

Fidalgo, M.L. (1993). Response of Alto Rabagao Reservoir to trout farm operations. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1200-1203.

Fidalgo, M.L. (2002). Fish-farm effluents in the River Baceiro, effects on water quality. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 1345-1349.

Findikakis, A.N. and Law, A.W.K. (1999). Wind mixing in temperature simulations for lakes and reservoirs. Journal of Environmental Engineering, 125(5): 420-428.

Findlay, S., Pace, M.L., Lints, D., Cole, J.J., Caraco, N.F. and Peierls, B. (1991). Weak coupling of bacterial and algal production in a heterotrophic ecosystem: the Hudson River estuary. Limnology and Oceanography, 36: 268-278.

Fisher, W.L. and Zale, A.V. (1991). Effect of water level fluctuations on abundance of young of the year largemouth bass in a hydropower reservoir. Proceedings Annual Conference SEAFWA, 45: 422-431.

Fleischer, S., Gustafson, A., Joelsson, A., Pansar, J. and Stibe, L. (1994). Nitrogen removal in created ponds. Ambio, 23: 349-357.

Fleituch, T.M. (1992). Evaluation of the water quality of future tributaries to the planned Dobczyce reservoir (Poland) using macroinvertebrates. Hydrobiologia, 237: 103-116.

Flemming. C.A. and Travers, J.T. (1989). Copper toxicity and chemistry in the environment: A review. Water Air and Soil Pollution, 44: 143-158.

*Foissner, W. and Berger, H. (1996). A user-friendly guide to the ciliates (Protozoa, Ciliophora) commonly used by hydrobiologists as bioindicators in rivers, lakes and waste waters, with notes on their ecology. Freshwater Biology, 35: 375-482.

Fonseca, C.P. (1997). Plankton community dynamics over short periods at Jacare-Pepira Reservoir, State of Sao Paulo, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26; 472-477.

Fontane, D.G., Labadie, J.W. and Loftis, B. (1981). Optimal control of reservoir discharge quality through selective withdrawal. Water Resources Research, 17: 1594-1604.

Ford, D.E. (1990). Reservoir transport processes. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds*).* *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 15-41.

Fraser, R.N. (1998a). Hyperspectral remote sensing of turbidity and chlorophyll a among Nebraska Sand Hills lakes. International Journal of Remote Sensing, 19: 1579-1589.

Fraser, R.N. (1998b) Multispectral remote sensing of turbidity among Nebraska Sand Hills lakes. International Journal of Remote Sensing, 19: 3011-3016.

*French, K.J., Scruton, D.A., Anderson, M.R. and Schneider, D.C. (1999). Influence of physical and chemical characteristics on mercury in aquatic sediments. Water, Air and Soil Pollution, 110: 347-362.

Friedrich, G., Chapman, D. and Beim, A. (1992). The use of biological material. In: UNESCO/WHO/UNEP (1992). *Water Quality Assessments – A Guide to the Use of Biota, Sediments and Water in Environmental Monitoring,* 2nd Ed.

*Friis, J.C., Holm, C. and Sorensen, H. (1998). Evaluation of elemental composition of algal biomass as toxical endpoint. Chemosphere, 37: 2665-2676.

Fukuju, S., Takahashi, T. and Kawayoke, T. (1998). Statistical analysis of freshwater red tide in Japanese reservoirs. Water Science and Technology, 37(2): 203-210.

*Fulton, M.H. and Key, P.B. (2001). Acetylcholinesterase inhibition in estuarine fish and invertebrates as an indicator of organophosphorus insecticide exposure and effects. Environmental Toxicology and Chemistry, 20: 37-45.

Fulton, R.S. and Jones, R.C. (1991). Growth and reproductive responses of *Daphnia* to cyanobacterial blooms on the Potomac River. International Review of Hydrobiology, 76: 5-19.

Fulton, R.S. and Paerl, H.W. (1987). Effects of colonial morphology on zooplankton utilization of algal resources during blue-green algal (*Microcystis aeruginosa*) blooms. Limnology and Oceanography, 32(3): 634-644.

Gachter, R. and Meyer, J.S. (1993). The role of microorganisms in mobilization and fixation of phosphorus in sediments. Hydrobiologia, 253: 103-121

Gaedke, U. and Schimmele, M. (1991). Internal seiches in Lake Constance: influence on plankton abundance at a fixed sampling site. Journal of Plankton Research, 13: 743-754.

Gaedke, U., Ollinger, D., Bauerle, E. and Straile, D. (1998). The impact of the interannual variability in hydrodynamic conditions on the plankton development of Lake Constance in spring and summer. Archiv fur Hydrobiologie: Advances in Limnology, 53: 565-585.

Galiulin, R.V., Bashkin, V.N. and Galiulina, R.A. (2005). Ecological risk assessment of riverine contamination in the Caspian Sea Basin: A conceptual model for perisistent organochlorinated compounds. Water, Air and Soil Pollution, 163: 33-51.

*Gallie, E.A. and Murtha, P.A. (1992). Specific absorption and backscattering spectra for suspended minerals and chlorophyll a in Chilko Lake, British Columbia. Remote Sensing of the Environment, 39: 103-118.

*Galloway, J.N. (1996). Anthropogenic mobilization of sulphur and nitrogen: Immediate and delayed consequences. Annual Review of Energy and the Environment, 21: 261

Galloway, J.N. (1998). The global nitrogen cycle: Changes and consequences. Environmental Pollution, 102(S1), 15.

Galloway, J.N. (2001). Acidification of the world: natural and anthropogenic. Water, Air and Soil Pollution, 130: 17-24.

Galloway, J.N., Norton, S.A. and Church, M.R. (1983). Freshwater acidification from atmospheric deposition of sulphuric acid: A conceptual model. Environmental Science and Technology, 17: 541

Galvez, J.A., Niell, F.X. and Lucena, J. (1993). Sinking velocities of principal phytoplankton species in a stratified reservoir: ecological implications. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1228-1231.

*Galvez, J.A. and Niell, F.X. (1993). Sedimention and mineralization of seston in a eutrophic reservoir, with a tentative sedimentation model. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 19-126.

Galy-Lacaux, C., Delmas, R., Jambert, C., Dumestre, J.F., Labroue, L., Richard, S, and Gosse, P. (1997). Gaseous emission and oxygen consumption in hydroelectric dams: a case study in French Guyana. Global Biogeochemical Cycles, 11(4): 471-483.

Galy-Lacaux, C., Delmas, R., Kouadio, G., Richard, S, and Gosse, P. (1999). Long-term greenhouse gas emissions from hydroelectric reservoirs in a tropical forest region. Global Biogeochemical Cycles, 13: 503-517.

*Gamito, S. and Erzini, K. (2005). Trophic food web and ecosystem attributes of a water reservoir of the Ria Formosa (south Portugal). Ecological Modelling, 181: 509-520.

Gapeeva, M.V., Zakonnov, V.V. and Gapeev, A.A. (1997). Localization and distribution of heavy metals in bottom sediments of the upper Volga reservoirs. Water Research, 24: 174-180.

Garcia-Berthou, E. (1999). Spatial heterogeneity in roach (*Rutilus rutilus*) diet among contrasing basins within a lake. Archiv fur Hydrobiologie, 146: 239-256.

Garcia-Berthou, E., Alcaraz, C., Pou-Rovira, Q., Zamora, L., Coenders, G. and Feo, C. (2005). Introduction pathways and establishment rates of invasive aquatic species in Europe. Canadian Journal of Fisheries and Aquatic Sciences, 62: 453-463.

Garnier, J. and Benest, D. (1990). Seasonal coupling between phytoplankton and bacterioplankton in a sand pit lake (Creteil Lake, France). Hydrobiologia, 207: 71-77.

*Garnier, J., Billen, G., Sanchez, N. and Leporcq, B. (2000). Ecological functioning of the Marne Reservoir (Upper Seine Basin, France). Regulated Rivers: Research and Management, 16: 51-71.

Garton, D.W. and Berg, D.J. (1990). Occurrence of *Bythotrephes cederstroemi* (Schoedler 1877) in Lake Superior, with evidence of demographic variation within the Great Lakes. Journal of Great Lakes Research, 16: 148-152.

Garton, D.W., Berg, D.J. and Fletcher, R.J. (1990). Thermal tolerances of the predatory cladocerans *Bythotrephes cederstroemi and Leptodora kindtii:* relationships to seasonal abundance in western Lake Erie. Canadian Journal of Fisheries and Aquatic Sciences, 47: 731-738.

*Garvey, E., Tobiason, J.E., Hayes, M., Wolfram, E., Reckhow, D.A. and Male, J.W. (1998). Coliform transport in a pristine reservoir: modelling and field studies. Water, Science and Technology, 37: 137-144.

Gasol, J.M., Comerma, M., Garcia, J.C., Armengol, J., Casamayor, E.O., Kojecka, P. and Simek, K. (2002). A transplant experiment to identify the factors controlling bacterial abundance, activity, production and community composition in a eutrophic canyon-shaped reservoir. Limnology and Oceanography, 47: 62-77.

*Gayoso, A.M. (1993). Nuisance blooms in Argentina reservoirs. Harmful Algae News, 5: 2.

Gebre-Marium, Z. (1999). Contrasting changes in the chemical and biological features of seven Ethiopian Rift Valley Lakes. *Proceedings of the 8th International Conference on the Conservation and Management of Lakes*, 17-21 May, 1999, Copenhagen. S3-6.

Geetanjali, Malhotra, S.K., Malhotra, A., Ansari, Z. and Chatterji, A. (2002). Current Science, 82(5): 505-507.

Gehrke, P.C. and Harris, J.H. (1994). The role of fish in cyanobacterial blooms in Australia. Australian Journal of Marine and Freshwater Research, 45: 175-185.

Gehrke, P.C. and Harris, J.H. (2001). Regional-scale effects of flow regulation on lowland rivierine fish communities in New South Wales, Australia. Regulated Rivers: Research and Management, 17: 369-391.

Geller, W. and Muller, H. (1981). The filtration apparatus of cladocera: filter mesh sizes and their implications on food selectivity. Oecologia (Berlin), 49: 316-321.

*Geraldes, A.M. and Boavida, M.J. (2003). Distinct age and landscape influence on two reservoirs under the same climate. Hydrobiologia, 504: 277-288.

*Geraldes, A.M. and Boavida, M.J. (2003). Do distinct water chemistry, reservoir age and disturbance make any difference on phosphatase activity? Journal of Limnology, 62(2): 163-171.

Gerten, D. and Adrian, R. (2001). Differences in the persistency of the North Atlantic Oscillation signal among lakes. Limnology and Oceanography, 46: 448-455.

Gibson, C.J., Haas, C.N. and Rose, J. (1998). Risk assessment of waterborne protozoa: current status and future trends. Parasitology, 117: s205-212.

Gitelson, A., Garbuzov, G., Szilagy, F., Mittenzwey, K.-H., Karnieli, A. and Kaiser, A. (1993). Quantitative remote sensing methods for real-time monitoring of inland water quality. International Journal of Remote Sensing, 14: 1269-1295.

Gitelson, A., Mayo, M., Yacobi, Y.Z., Parparov, A. and Berman, T. (1994). The use of high-spectral-resolution radiometer data for detection of low chlorophyll concentrations in Lake Kinneret. Journal of Plankton Research, 16: 993-1002.

Giussani, G. and Galanti, G. (1992). Experience in eutrophication recovery by biomanipulation. Memorie dell'Istituto Italiano di Idrobiologia Milano, 50: 397-416.

Gladyshev, M.I., Dubovskaya, O.P., Gubanov, V.G. and Makhutova, O.N. (2003). Evaluation of non-predatory mortality of two *Daphnia* species in a Siberian reservoir. Journal of Plankton Research, 25(8): 999-1003.

Gleick, P.H. (1987a). The development and testing of a water balance model for climate impact assessment: modelling the Sacramento Basin. Water Resources Research, 23(6): 1049-1061.

Gleick, P.H. (1987b). Regional hydrologic consequences of increases in atmospheric carbon dioxice and other trace gases. Climatic Change, 10(2): 137-161.

Gleick, P.H. (1997). *The World's Water 1998-1999*. Island Press, Washington D.C.

Gliwicz, M. (1990a). *Daphnia* growth at different concentrations of blue-green filaments. Archiv fur Hydrobiologie, 120: 51-65.

Gliwicz, M. (1990b). Why do cladocerans fail to control algal bloom? Hydrobiologia, 200/201: 83-97.

Gliwicz, Z.M. (1994). Relative significance of direct and indirect effects of predation by planktivorous fish on zooplankton. Hydrobiologia, 272(1-3): 201-210.

*Gliwicz, Z.M. and Pijanowska, J. (1989). The role of predation in zooplankton succession. In: Sommer, U. (ed). *Plankton Ecology*. Springer-Verlag, Berlin; pp. 253-296.

Gliwicz, Z.M. and Rykowska, A. (1992). Shore avoidance in zooplankton: A predator induced behaviour or a predator-induced mortality. Journal of Plankton Research, 14: 1331-1342.

Gliwicz, Z.M. and Lampert, W. (1993). Body-size related survival of cladocerans in a trophic gradient: An enclosure study. Archiv fur Hydrobiologie, 129: 1-23.

Gliwicz, Z.M. and Umana, G. (1994). Cladoceran body size and vulnerability to copepod predation. Limnology and Oceanography, 39: 419-424.

*Godlewska, M. and Swierzowski, A. (2003). Hydroacoustical parameters of fish in reservoirs with contrasting levels of eutrophication. Aquatic Living Resources, 16: 167-173.

*Godlewska, M., Mazurkiewicz-Boron, G., Pociecha, A., Wilk-Wozniak, E. and Jelonek, M. (2003). Effects of flood on the functioning of the Dodczyce reservoir ecosystem. Hydrobiologia, 504: 305-313.

*Goedkoop, W. and Johnson, R.K. (1996) Pelagic-benthic coupling: profundal benthic community response to spring diatom deposition in mesotrophic Lake Erken. Limnology and Oceanography, 41: 636-647.

Goel, M.K., Jain, S.K. and Agarwal, P.K. (2002). Assessment of sediment deposition rate in Bargi Reservoir using digital image processing. Hydrological Sciences Journal, 47: S81-S92.

Goldyn, R., Kozak, A. and Romanowicz, W. (1997). Food-web manipulation in the Maltanski Reservoir. Hydrobiologia, 342-343: 327-333.

*Goldyn, R., Kozak, A., Romanowicz, W., Grzeskowiak, K. and Starczewska-Porada. (1997). Phytoplankton-zooplankton interaction in the newly restored Maltanski Reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 483-486.

Goldyn, R., Joniak, T., Kowalczewska-Madura, K. and Kozak, A. (2003). Trophic state of a lowland reservoir during 10 years after restoration. Hydrobiologia, 506-509: 759-765.

*Golovko, T.V., Yakushin, V.M. and Tronko, N.I. (2003). Bacterioplankton of the Kanev Reservoir and its productional characteristics. Hydrobiological Journal, 39(6): 55-69.

Gomes, E.A.T. and Godinho, M.J.L. (2003). Structure of the protozooplankton community in a tropical shallow and eutrophic lake in Brazil. Acta Oecologia, 24: S153-S161.

*Gomes, L.C. and Miranda, L.E. (2001). Riverine characteristics dictate composition of fish assemblages and limit fisheries in reservoirs of the Upper Parana River Basin. Regulated Rivers: Research and Management, 17: 67-76.

*Gomes, L.C., Miranda, L.E. and Agostinho, A.A. (2002). Fishery yield relative to chlorophyll a in reservoirs of the Upper Parana River, Brazil. Fisheries Research, 55: 335-340.

*Gomez, N. (1995). Changes in the phytoplankton of the reservoir Embalse Rio Tercero (Prov. Cordoba, Argentina) as a result of the nuclear power plant operating there. Acta Hydrobiologica, 37(3): 129-139.

*Gonzalez, E.J. (1998). Natural diet of zooplankton in a tropical reservoir (Embalse El Andino, Venezuela). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 1930-1934.

*Gonzalez, E.J., Ortaz, M., Penaherrera, C. and de Infante, A. (2004). Physical and chemical features of a tropical hypertrophic reservoir permanently stratified. Hydrobiologia, 522: 301-310.

Gonzalez, J.M., Iriberri, J., Egea, L. and Barcina, I. (1990b). Differential rates of digestion of bacteria by freshwater and marine phagotrophic protozoa. Applied and Environmental Microbiology, 56: 1851-1857.

Gonzalez, J.M., Sherr, E.B. and Sherr, B.F. (1990a). Size-selective grazing on bacteria by natural assemblages of estuarine flagellates and ciliates. Applied and Environmental Microbiology, 56: 583-589.

*Goodland, R. (1990). The World Bank's new environmental policy for dams and reservoirs. Water Resources Development, 6(4): 226-239.

Goodwill, P., Rowan, J.S. and Greco, M. (1995). Sediment routing through reservoirs, Wyresdale Park Reservoir, Lancashire, UK. Physics and Chemistry of the Earth, 20(2): 183-190.

*Gophen, M. (1990). Biomanipulation: retrospective and future development. Hydrobiologia, 200-201: 1-11.

*Gopchenko, E.D. and Loboda, N.S. (An evaluation of possible changes in the water resources of Ukraine under global warming conditions. Hydrobiological Journal, 37(5): 105-117.

*Gorniak, A., Grabowska, M., Jekatierynczuk-Rudczyk, E., Zielinski, P. and Suchowolec. (2003). Long-term variations of phytoplankton production in a shallow, polyhumic reservoir. Hydrobiologia, 506-509: 305-310.

Gouvea, S.P., Vieira, A.A.H. and Lombardi, A.T. (2005) Copper and cadmium complexation by high molecular weight materials of dominant microalgae and of water from a eutrophic reservoir. Chemosphere (in press).

*Grabowska, M., Gorniak, A., Rudczyk, E.J. and Zielinski, P. (2003). The influence of hydrology and water quality on phytoplankton community composition and biomass in a humoeutrophic reservoir, Siemianowka Reservoir (Poland). Ecohydrology and Hydrobiology, 3(2): 185-196.

Graczyk, T.K., Fayer, R., Lewis, E.J., Trout, J.M. and Farley, C.A. (1999). *Cryptosporidium* oocysts in Bent mussels (*Ischadium recurvum*) in the Chesapeake Bay. Parasitology Research, 85(7): 518-521.

*Graczyk, T.K., Marcogliese, D.J., de Lafontaine, Y., Da Siva, A.J., Mhangami-Ruwende, B. and Pieiazek, N.J. (2001). *Cryptosporidium parvum* oocysts in zebra mussels (*Dreissena polymorpha*): evidence from the Lawrence River. Parasitology Research, 87(3): 231-234.

Graham, L.E. and Wilcox, L.W. (2000*).* *Algae*. Prentice Hall, Inc. Upper Saddle River, USA.

Granado-Lorencio, C. (1991). Fish communities of Spanish reservoir systems: a non-deterministic approach. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 2428-2431

Granado-Lorencio, C. (1992). Fish species ecology in Spanish freshwater ecosystems. Limnetica, 8: 255-261.

Gray, J.E., Fey, D.L., Holmes, C.W. and Lasorsa, B.K. (2005). Historical deposition and fluxes of mercury in Narraguinnep Reservoir, southwestern Colorado, USA. Applied Geochemistry, 20: 207-220.

*Greve, G.D., Ter Laak, T.L., Boivin, M. and Reede, T. (2000). The use of the benthic cladoceran *Chydorus sphaericus* for toxicity testing. *Abstracts of the NABS Annual Meeting*. Keystone, Colorado, 2000.

Gries, T. and Gude, H. (1999). Estimates of the nonconsumptive mortality of mesozooplankton by measurement of sedimentation losses. Limnology and Oceanography, 44: 459-465

*Grobler, D.C. (1985). Phosphorus budget models for simulating the fate of phosphorus in South African reservoirs. Water S.A., 11: 219-230.

Grobler, D.C., Rossouw, J.N., van Eeden, P. and Oliveira, M. (1987). Decision Support System for selecting eutrophication control strategies. In: Beck, M.B. (ed). *Systems Analysis in Water Quality Management*. Proceedings of a Symposium held in London, UK., 30 June-2 July 1987. Pergamon Press, Great Britain; pp. 219-230.

Grodowitz, M.J. (1998). An active approach to the use of insect biological control for the management of non-native aquatic plants. Journal of Aquatic Plant Management, 36: 57-61.

Groeger, A.W. and Bass, D.A. (2005). Empirical predictions of water temperatures in a subtropical reservoir. Archiv fur Hydrobiologie, 162: 267-285.

Grondin, A., Lucotte, M., Mucci, A. and Fortin, B. (1995). Profiles of mercury and lead and their burdens in Quebec soils (Canada) before and after flooding. Canadian Journal of Fisheries and Aquatic Sciences, 52: 2493-2505.

*Ground, T.A. and Groeger, A.W. (1994). Chemical classification and trophic characteristics of Texas reservoirs. Lake and Reservoir Management, 10(2): 189-201.

Grzybkowska, M. and Dukowska, M. (2002). Communities of Chironomidae (Diptera) above and below a reservoir on a lowland river: long-term study. Annales Zoologici, 52(2): 235-247.

Gude, H. (1988). Direct and indirect influences of crustacean zooplankton on bacterioplankton of Lake Constance. Hydrobiologia, 159: 63-73.

Guicherit, R., Bakker, D.J., de Voogt, P., van den Berg, F., van Dijk, H.F.G. and van Pul, W.A.J. (1999). Environmental risk assessment for pesticides in the atmosphere: The results of an international workshop. Water Air and Soil Pollution, 115: 5-19.

Guiset, A. and De Manuel, J. (1993). Distribution of planktonic rotifers in Spanish reservoirs: changes after 15 years. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1213-1217.

Gulati, R.D., Lammers, E.H.R.R., Meijer, M.-L. and Van Donk, E. (1990*). Biomanipulation. Tool for Water Management.* Kluwer Acad. Publishers, Dordrecht.

Gulati, R.D. (1990). Structural and grazing responses of zooplankton community to biomanipulation of some Dutch water bodies. Hydrobiologia, 200-201: 99-118.

Gulati, R.D., Lammens, E.H.R., Meijer, M. and van Donk, E. (Eds) (1990). Biomanipulation, tool for water management. First International Conference, 8-11 August, 1989, Amsterdam, The Netherlands. Kluwer Academic Publishers, Hydrobiologia 200/201: 1-628.

Gulati, R.D., Siewetsen, K and van Liere, L. (1991). Carbon and phosphorus relationships of zooplankton and its seston food in Loosdrecht lakes. Memorie dell' Istituto Italiano di Idrobiologia Milano, 48: 279-298.

Gulati, R.D. and DeMott, W.R. (1997). The role of food quality for zooplankton: remarks on the state-of-the-art, perspectives and priorities. Freshwater Biology, 38: 753-768.

*Gunatilake, H.M. and Gopalakrishnan, C. (1999). The economics of reservoir sedimentation: A case study of Mahaweli Reservoirs in Sri Lanka. Water Resources Development, 15(4): 511-526.

*Gunduz, O., Soyupak, S. and Yurteri, C. (1998). Development of water quality management strategies for the proposed Isikli Reservoir. Water, Science and Technology, 37(2): 369-376.

*Gunkel, G., Lange, U., Walde, D. and Rosa, J.W.C. (2003). The environmental and operational impacts of Curua-Una, a reservoir in the Amazon region of Para, Brazil. Lakes and Reservoirs: Research and Management, 8: 201-216.

Gurbuz, H., Kivrak, E., Soyupak, S. and Yerli, S.V. (2003). Predicting dominant phytoplankton quantities in a reservoir by using neural networks. Hydrobiologia 504: 133-141.

Gutierrez, E., Arreguin, F., Huerto, R. and Saldana, P. (1994). Aquatic weed control. Water Resources Development, 10(3): 291-312.

Guzzella, L. (1997). PCBs and organochlorine pesticides in Lake Orta (Northern Italy) sediments. Water, Air and Soil Pollution, 99: 245-254.

Haestad Methods. (1993). *HEC-B, SEDIMOT-II and STORM. Haestad Methods, Softwater Reference Handbook*, Haestad, Waterbury, Connecticut.

Hakanson, L. (2001). Assessment of critical loading of lakes as a basis for remedial measures: A review of fundamental concepts. Lakes and Reservoirs: Research and Management, 6: 1-20.

Hakanson, L., Ostapenia, A., Parparov, A., Hambright, K.D. and Boulion, V.V. (2003). Management criteria for lake ecosystems applied to case studies in nutrient loading and climate change. . Lakes and Reservoirs: Research and Management, 8: 141-155.

Hall, B.D., Bodaly, R.A., Fudge, R.J.P., Rudd, J.W.M. and Rosenberg, D.M. (1997). Food as the dominant pathway of methylmercury uptake by fish. Water, Air and Soil Pollution, 100: 13-24.

Hall, C.A.S., Stanford, J.S. and Hauer, F.R. (1992). The distribution and abundance of organisms as a consequence of energy balances along multiple environmental gradients. Oikos, 65: 377-390.

*Hall, R.I., Leavitt, P.R., Dixit, A.S., Quinian, R. and Smol, J.P. (1999). Limnological succession in reservoirs: a palaeolimnological comparison of two methods of reservoir formation. Canadian Journal of Fisheries and Aquatic Sciences, 56: 1109-1121.

*Hamblin, P.F. and McAdam, S.O. (2003). Impoundment effects on the thermal regimes of Kootenay Lake, the Arrow Lakes Reservoir and Upper Columbia River. Hydrobiologia 504: 3-19.

Hamburger, K., Dall, P.C. and Lindegaard, C. 1995. Effects of oxygen deficiency on survival and glycogen content of *Chironomus anthracinus* (Diptera, Chironomidae) under laboratory and field conditions. Hydrobiologia, 297: 187-200.

Hamilton, D.P. and Schladow, S.G. (1994). Modelling the sources of oxygen in an Australian reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1282-1285.

Hamilton, D.P. and Schladow, S.G. (1995). Controlling the indirect effects of flow diversions on water quality in an Australian reservoir. Environment International, 21(5): 583-590.

Hamilton, D.P. and Schladow, S.G. (1997). Prediction of water quality in lakes and reservoirs. Part I – Model description. Ecological Modelling, 96: 91-110.

Han, B.-P., Armengol, J., Garcia, J.C., Comerma, M., Roura, M., Dolz, J. and Straškraba, M. (2000). The thermal structure of Sau Reservoir (NE Spain): a simulation approach. Ecological Modelling, 125(2 - 3): 109-122.

*Han, L. (1997). Spectral reflectance with varying suspended sediment concentrations in clear and algae-laden waters. Photogrammetric Engineering and Remote Sensing, 63: 701-708.

*Han, L. and Rundquist, D.C. (1998). The impact of a wind-roughened water surface on remote measurements of turbidity. International Journal of Remote Sensing, 19(1): 195-201.

*Handy, R.D. and Depledge, M.H. (1999). Physiological responses: their measurement and use as environmental biomarkers in ecotoxicology. Ecotoxicology, 8: 329-349.

*Hanna, N.S. and Scheimer, F. (1993). The seasonality of zooplanktivorous fish in an African reservoir (Gebel Aulia Reservoir, White Nile, Sudan). Part 1: Limnological cycle and the fish community dynamics. Hydrobiologia, 250: 173-185.

*Hanna, N.S. and Scheimer, F. (1993). The seasonality of zooplanktivorous fish in an African reservoir (Gebel Aulia Reservoir, White Nile, Sudan). Part 2: Spatial distribution and resource partitioning in zooplanktivorous fish assemblages. Hydrobiologia, 250: 187-199.

Hansen, M.J., Boisclair, D., Brandt, S.B., Hewatt, S.W., Kitchell, J.F., Lucas, M.C. and Ney, J.J. (1993). Applications of bioenergetics models to fish ecology and management: Where do we go from here? Transactions of the American Fisheries Society, 122: 1019-1030.

Hansson, L.A. (1988). Effects of competitive interactions on the biomass development of planktonic and periphytic algae in lakes. Limnology and Oceanography, 33: 121-128

Hansson, L.A. (1989). The influence of a periphytic biolayer on phosphorus exchange between substrate and water. Archiv fur Hydrobiologie, 115: 21-26.

Hansson, L.A. (1992). The role of food chain composition and nutrient availability in shaping algal biomass development. Ecology, 73: 241-247.

Harding, W.R., Archibald, C.G.M. and Taylor, J.C. (2005). The relevance of diatoms for water quality assessment in South Africa: A position paper. Water SA, 31(1): 41-46.

*Harrington, J.A., Schiebe, F.R. and Nix, J.F. (1992). Remote sensing of Lake Chicot, Arkansas: monitoring suspended sediments, turbidity and Secchi depth with Landsat MSS data. Remote Sensing of the Environment, 39, 15-27.

Harris, D.L., Lottermoser, B.G. and Duchesne, J. (2003). Ephemeral acid mine drainage at the Montalbion silver mine, north Queensland. Australian Journal of Earth Sciences, 50(5): 797-809.

*Harris, G.P. (1986). *Phytoplankton Ecology: Structure, Function and Fluctuation.* Chapman and Hall, New York. 384 pp.

*Harris, G.P.; Baxter, G. (1996). Interannual variability in phytoplankton biomass and species composition in a subtropical reservoir. Freshwater Biology, 35(3): 545-560.

*Hart, B.T., van Dok, W. and Djuansih, N. (2002). Nutrient budget for Saguling reservoir, West Java, Indonesia. Water Research, 36: 2152-2160.

Hart, R.C. (1984). Zooplankton community grazing in silt-laden Lake le Roux, Orange River, South Africa. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 22: 1602-1607.

Hart, R.C. (1986a). Zooplankton abundance, community structure and dynamics in relation to inorganic turbidity, and their implications for a potential fishery in subtropical Lake le Roux, South Africa. Freshwater Biology, 16: 351-371.

Hart, R.C. (1986b). Aspects of the feeding ecology of turbid water zooplankton. *In situ* studies of community filtration rates in silt-laden Lake le Roux, Orange River, South Africa. Journal of Plankton Research, 8: 401-426.

Hart, R.C. (1987a). Observations on calanoid diet, seston, phytoplankton-zooplankton relationships, and inferences on calanoid food limitation in a silt laden reservoir. Archiv fur Hydrobiologie, 111: 67-82.

Hart, R.C. (1987b). Population dynamics and production of five crustacean zooplankters in a subtropical reservoir during years of contrasting turbidity. Freshwater Biology, 18: 287-318.

Hart, R.C. (1988). Zooplankton feeding rates in relation to suspended sediment content: potential influences on community structure in a turbid reservoir. Freshwater Biology, 19: 123-139.

Hart, R.C. (1990). Zooplankton distribution in relation to turbidity and related environmental gradients in a large subtropical reservoir: patterns and implications. Freshwater Biology, 24(2): 241-263.

*Hart, R.C. (1991). Food and suspended sediment influences on the naupliar and copepodid durations of freshwater copepods: comparative studies on *Tropodiaptomus* and *Metadiaptomus.* Journal of Plankton Research, 13: 645-660.

Hart, R.C. (1992). Experimental studies on food and suspended sediment effects on growth and reproduction of six planktonic cladocerans. Journal of Plankton Research, 14: 1425-1448.

*Hart, R.C. (1995). Comparative studies of plankton ecology in warm-water reservoirs: aspects of relevance to tropical limnology. In: Timotius, K.H. and Goltenboth, F. (Eds). *Tropical Limnology. Vol II. Tropical Lakes and Reservoirs*. Salatiga, Indonesia; pp.229-245.

Hart, R.C. (1996). Comparative ecology of plankton in cascading warm-water reservoirs: aspects of relevance to tropical limnology. In: Scheimer, F. and Boland, K.T. (Eds). *Perspectives in Tropical Limnology*. SPB Academic Publishing bv, Amsterdam, pp. 113-130.

Hart, R.C. (1999). On the limnology of Spioenkop, a turbid reservoir on the upper Thukela River, with particular reference to the structure and dynamics of its plankton community. Water SA, 25(4): 519-528

Hart, R.C. (2001). A baseline limnological study of Wagendrift Dam (Thukela basin, KwaZulu.Natal). Water SA, 27(4): 507-516.

Hart, R.C. (2004). Cladoceran periodicity patterns in relation to selected environmental factors in two cascading warm-water reservoirs over a decade. Hydrobiologia, 526: 99-117.

Hart, R.C. (2006). Phytoplankton dynamics and periodicity in two cascading warm-water reservoirs from 1989 to 1997 – taxonomic and functional (C-S-R) patterns, and determining factors. Water SA, 32: 81-92.

Hart, R.C. and Jarvis, A.C. (1993). *In situ* determinations of bacterial selectivity and filtration rates by five cladoceran zooplankters in a hypertrophic subtropical reservoirs. Journal of Plankton Research, 15: 295-315.

Harvey, H.H. (1975). Fish populations in a large group of acid stressed lakes. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 19: 2401-2417.

*Hassanzadeh, Y. (1995). The removal of reservoir sediment. Water International, 20: 151-154.

Hassett, R.P., Cardinale, B., Stabler, L.B. and Elser, J.J. (1997). Ecological stoichiometry of N and P in pelagic ecosystems: comparison of lakes and oceans with emphasis on the zooplankton-phytoplankton interaction. Limnology and Oceanography, 42: 648-662.

Havel, J.E. (1985). Predation of common invertebrate predators on long- and short-featured *Daphnia retrocurva*. Hydrobiologia, 124: 141-149.

Havel, J.E. and Dodson, S.I. (1984). Chaoborus predation on typical and spined morphs of Daphnia pulex: behavioural observations. Limnology and Oceanography, 29: 487-494.

Havel, J.E. and Hebert, P.D.N. (1993). *Daphnia lumholtzi* in North America: another exotic zooplankter. Limnology and Oceanography, 38: 1823-1827.

*Havel, J.E., Mabee, W.R. and Jones, J.R. (1995). Invasion of the exotic cladoceran *Daphnia lumholtzi* into North American reservoirs. Canadian Journal of Fisheries and Aquatic Sciences, 52: 151-160.

Havelaar, A.H., Vanolphen, M. and Drost, Y.C. (1993). F-specific RNA bacteriophages are adequate model organisms for enteric viruses in fresh water. Applied Environmental Microbiology, 59(9): 2956-2962.

*Havens, K.E. (1994a). An experimental comparison of the effects of two chemical stressors on a freshwater zooplankton assemblage. Environmental Pollution, 84: 245-251.

*Havens, K.E. (1994b). Structural and functional responses of a freshwater plankton community to acute copper stress. Environmental Pollution, 86: 259-266.

Havens, K.E. and Schelske, C.L. (2001). The importance of considering biological processes when setting total maximum daily loads (TDML) for phosphorus in shallow lakes and reservoirs. Environmental Pollution, 113: 1-9.

Havens, K.E., East, T.L. and Beaver, J.R. (1996). Experimental studies of zooplankton- phytoplankton-nutrient interactions in a large subtropical lake (Lake Okeechobee, Florida, USA). Freshwater Biology, 36: 579-597

Hawkins, P.R. and Griffiths, D.J. (1993). Artificial destratification of a small tropical reservoir effects upon the phytoplankton. Hydrobiologia, 254: 169-181.

Hawkins, P.R., Swanson, P., Warnecke, M., Shanker, S.R. and Nicholson, C. (2000). Understanding the fate of *Cryptosporidium* and *Giardia* in storage reservoirs: a legacy of Sydney's water contamination incident. Aqua (Oxford), 49(6): 289-306.

*Hayward, R.S. and van den Avyle, M.J. (1986). The nature of zooplankton spatial heterogeneity in a nonriverine impoundment. Hydrobiologia, 131: 261-271.

Head, I.M., Saunders, J.R. and Pickup, R.W. (1998). Microbial evolution, diversity and ecology: a decade of ribosomal RNA analysis of uncultivated microorganisms. Microbiology Ecology, 35: 1-21.

Heath, R.T. and Edinger, A.C. (1990). Uptake of ^{32}P-phosphoryl from glucose-6-phosphate by plankton in an acid bog lake. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 210-213.

Hecky, R.E., Campbell, P. and Hendzel, L.L. (1993). The stoichiometry of carbon, nitrogen and phosphorus in particulate matter of lakes and oceans. Limnology and Oceanography, 38: 709-724.

*Hedger, R.D., Atkinson, P.M. and Malthus, T.J. (2001). Optimizing sampling strategies for estimating mean water quality in lakes using geostatistical techniques with remote sensing. Lakes and Reservoirs: Research and Management, 6: 279-288.

Heidenreich, M. and Kleeberg, A. (2003). Phosphorus-binding in iron-rich sediments of a shallow reservoir: spatial characterization based on sonar data. Hydrobiologia, 506-509: 147-153.

Heinis, F. and Crommentuijn, T. (1992). Behavioural responses to changing oxygen concentrations of deposit feeding chironomic larvae (Diptera) of littoral and profundal habitats. Archiv fur Hydrobiologie, 124: 173-185.

Hejzlar, J. and Vyhnalek, V. (1998). Longitudinal heterogeneity of phosphorus and phytoplankton concentrations in deep-valley reservoirs. International Review of Hydrobiology, 83: 139-146.

*Henry, R. (1992). The oxygen deficit in Jurumirim Reservoir (Paranapanema River, Sao Paulo, Brazil). Japanese Journal of Limnology, 153: 379-384.

*Henry, R. (1993). Thermal regime and stability of Jurumirim Reservoir (Paranapanema River, Sao Paulo, Brazil). Internatonal Review of Hydrobiology, 78: 501-511.

Henry, R. (1999). Heat budgets, thermal structure and dissolved oxygen in Brazilian reservoirs. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 125-151.

Hernandez-Aviles, J.S., Bernal-Brookes, F., Velarde, G., Ortiz, D., Lind, O.T. and Davalos-Lind, L. (2001). The algal growth potential and algae growth-limiting nutrients for 30 of Mexico's lakes and reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3583-3588.

Herricks, E.E. (1992). General Principles of Toxicology. In: *Guidelines of Lake Management. Volume 4.* International Lake Environment Committee, Kusatsu.

Herrmann, J. (1987). Aluminium impact of freshwater invertebrates at low pH: a review. In: Lander, L. (ed). *Speciation of Metals in Water, Sediment and Soil Systems.* Springer-Verlag, Berlin. pp 157-175

Herwaldt, B.L., Craun, G.F., Stokes, S.L. and Juranek, D.D. (1992). Outbreaks of waterborne disease in the United States: 1989-90. Journal of the American Water Works Association, 64(4):129-135.

Herzig, A. (1994). Predator-prey relationships within the pelagic community of Neusiedler See. Hydrobiologia 276: 81-96.

Herzig, A. and Auer, B. (1990). The feeding behaviour of *Leptodora kindtii* and its impact on the zooplankton community of Neusiedler See (Austria). Hydrobiologia 198: 107-117.

Hessen, D.O. (1992). Nutrient element limitation of zooplankton production. American Naturalist, 140: 799-814.

Hessen, D.O. (1997). Stoichiometry in food webs – Lotka revisited. Oikos, 79: 196-200.

Hessen D.O. and Lyche, A. (1991). Inter- and intraspecific variations in zooplankton element composition. Archiv fur Hydrobiologie, 121: 355-363.

Higgins, J.M. and Brock, W.G. (1999). Overview of reservoir release improvements at 20 TVA dams. Journal of Energy Engineering, 125(1):1-17

*Higgitt, D.L. and Lu. X.X. (2001). Sediment delivery to the three gorges: 1. Catchment controls. Geomorphology, 41: 143-156.

Hill, N.M., Keddy, P.A. and Wisheu, I.C. (1998). A hydrological model for predicting the effects of dams on the shoreline vegetation of lakes and reservoirs. Environmental Management, 22(5): 723-736.

*Hjorth, P. (2001). From insight to action in reservoir management – lessons from Asia. Natural Resources Forum, 25: 313-320.

Hladik, M. and Kubecka, J. (2003). Fish migration between a temperate reservoir and its main tributary. Hydrobiologia, 504: 251-266.

*Hoehn, E. and Schmidt-Halewicz, S. (1995). The impact of high flood-nutrient loading and *Daphnia* grazing on plankton development in the Kleine Kinzig reservoir. Aqua: 44(1): 102-107.

*Holdren, C., Jones, W. and Taggart, J. (2001). Managing lakes and reservoirs. North American Lake Management Society and Terren Institute, in cooperation with the Office of Water, Assessment and Watershed Protection Division of the U.S. Environmental Protection Agency. 3rd Edition. 382 pp.

Holz, J.C., Hoagland, K.D., Spawn, R.L., Popp, A. and Andersen, J.L. (1997). Phytoplankton community response to reservoir ageing, 1968-92. Hydrobiologia, 346: 183-192.

Horn, H. (2003). The relative importance of climate and nutrients in controlling phytoplankton growth in Saidenbach Reservoir. Hydrobiologia, 504: 159-166.

Horn, W. (2003). Long-term development of the crustacean plankton in the Saidenbach Reservoir (Germany) – changes, causes, consequences. Hydrobiologia, 504: 185-192.

Horn, W. and Horn, H. (1990). Long-term relationships between phyto- and zooplankton in the meso-eutrophic reservoir Saidenbach. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 749-762.

Horn, W. and Horn, H. (1995). Interrelationships between crustacean zooplankton and phytoplankton: results from 15 years of field observations at the mesotrophic Saidenbach Reservoir (Germany). Hydrobiologia, 307: 231-238.

*Horn, H., Uhlmann, D. (1995). Competitive growth of blue-greens and diatoms (Fragilaria) in the Saidenbach Reservoir, Saxony. Water Science and Technology, 32(4): 77-88.

Horne, A. and Goldman, C. (1994). *Limnology*. 2nd Edition. MacGraw Hill, 576pp.

Horppila, J. (1997). Diurnal changes in the vertical distribution of cladocerans in a biomanipulated lake. Hydrobiologia, 345: 215-220.

Horstman, H.K. Copp, R.S. and Browne, F.X. (1983). Use of predictive phosphorus model to evaluate hypolimnetic discharge scenarios of Lake Wallenpupack. *Lake and Reservoir Management*, EPA, Washington DC; pp. 165-170.

Hosomi, M., Saigusa, T., Yabunaka, K., Okubo, T. and Murakami, A. (1996). Development of a water temperature – ecological model to simulate global warming effects on lake ecosystem. Water, Science and Technology, 34(7-8): 237-244.

Hotchkiss, R.H. and Huang, X. (1995). Hydrosuction sediment-removal systems (HSRS): Principles and field test. Journal of Hydraulic Engineering, 121(6): 479-489.

*Howard, A. and Easthope, M.P. (2002). Application of a model to predict cyanobacterial growth patterns in response to climatic change at Farmoor Reservoir, Oxfordshire, UK. Science of the Total Environment, 283-283: 459-469.

Howarth, R.W., Billen, G. and Swaney, D. (1996). Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochemistry, 35: 75-139.

Hrbacek, J., Brandl, Z. and Straškraba, M. (2003). Do the long-term changes in zooplankton biomass indicate changes in fish stock? Hydrobiologia, 504: 203-213.

*Huckabee, J.W., Elwood, J.W. and Hildenbrand, S.G. (1979). Accumulation of mercury in freshwater biota. In: Nriagu, J. (ed). *The Biogeochemistry of Mercury in the Environment*. Elsevier/North Holland Biomedical Press, New York; pp. 277-301.

*Hudon, C., Lalonde, S. and Gagnon, P. (2000). Ranking the effects of site exposure, plant form growth, water depth and transparency on aquatic plant biomass. Canadian Journal of Fisheries and Aquatic Sciences, 57: 31-42.

*Hudson, C., Lalonde, S. and Gagnon, P. (2000). Ranking the effects of site exposure, plant growth form, water depth, and transparency on aquatic plant biomass. Canadian Journal of Fisheries and Aquatic Sciences, 57: 31-42.

Hulsmann, S. and Mehner, T. (1997). Predation by underyearling perch (*Perca fluviatilis*) on a *Daphnia galeata* population in a short-term enclosure experiment. Freshwater Biology, 37: 209-219.

Hulsmann, S., Mehner, T., Worischka, S. and Plewa, M. (1999). Is the difference in population dynamics of *Daphnia galeata* in littoral and pelagic areas of a long-term biomanipulated reservoir affected by age-0 fish predation? Hydrobiologia, 408-409: 57-63.

Hulsmann, S. and Weiler, W. (2000). Adult, not juvenile mortality as a major reason for midsummer decline of a *Daphnia* population. Journal of Plankton Research, 22: 151-168.

Hummert, C., Reichelt, M., Weiss, J., Liebert, H.-P. and Luckas, B. (2001). Identification of microcystins in cyanobacteria from the Bleiloch former drinking-water reservoir (Thuringia, Germany). Chemosphere, 44: 1581-1588.

Hunt, R.J., Matveev, V., Jones G.J. and Warburton, K. 2003. Structuring of the cyanobacterial community by pelagic fish in subtropical reservoirs: experimental evidence from Australia. Freshwater Biology 48: 1482-1495

*Hwang, S.J., Heath, R.T. (1997). The distribution of protozoa across a trophic gradient, factors controlling their abundance and importance in the plankton food web. Journal of Plankton Research, 19: 491-518.

Hynes, H.B.N. (1975). The stream and its valley. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 19: 1-15.

Ibanez, C., Prat, N. and Canicio, A. (1996). Changes in hydrology and sediment transport produced by large dams on the lower Ebro River and its estuary. Regulated Rivers: Research and Management, 12: 51-62.

Ikingura, J.R. and Akagi, H. (2003). Total mercury and methylmercury levels in fish from hydroelectric reservoirs in Tanzania. Science of the Total Environment, 304: 355-368.

Imberger, J. (1982). Reservoir dynamics modelling. In: O'Loughlin, E.M. and Cullen, P. (eds). *Prediction in Water Quality*. Proceedings of a Symposium on the Prediction of Water Quality, Canberra 1982. Australian Academy of Science, Canberra; pp. 223-248.

Imberger, J. and Patterson, J. (1981). A dynamic reservoir simulation model - DYRESM5. In: Fischer, H.B. (ed). *Transport Models for Inland and Coastal Waters*. Academic Press, New York; pp. 310-361.

Imberger, J. and Patterson, J.C. (1990). Physical limnology. Advances in Applied Mechanics, 27: 303-475.

Imteaz, M.A. and Asaeda, T. (2000). Artificial mixing of lake water by bubble plume and effects of bubbling operations on algal bloom. Water Research, 34(6): 1919-1929.

International Commission on Large Dams (ICOLD). (1988). *World Register of Dams, 1988. Update.* Paris, France, ICOLD.

International Commission on Large Dams (ICOLD). (1989). *Sedimentation control of reservoirs. Guidelines.* ICOLD Bulletin 67. Paris, France, ICOLD.

*Ioriya, T., Inoue, S., Haga, M. and Yogo, N. (1998). Change of chemical and biological water environment at a newly constructed reservoir. Water, Science and Technology, 37(2): 187-194.

IPCC (1992). Climatic Change 1992. Houghton, J.T., Callander, B.A. and Varney, S.K. (Eds).*The Supplementary Report to the IPCC Scientific Assessment.* Cambridge, Cambridge University Press.

Iskra, I.V. and Linnik, P.N. (1994). Content and migration forms of cadmium in the Dnieper water reservoirs. Hydrobiological Journal, 30: 72-81.

Istvanovics, V. and Somlyody, L. (1999). Changes in the cycling of phosphorus in the Upper Kis-Balaton Reservoir following external load reduction. Freshwater Biology, 41: 147-165

*Istvanovics, V. and Somlyody, L. (2001). Factors influencing lake recovery from eutrophication – the case of Basin 1 of Lake Balaton. Water Research, 35(3): 729-735.

Istvanovics, V., Kovacs, A., Voros, L., Herodek, S. and Pomogyi, P. (1997). Phosphorus cycling in a large, reconstructed wetland, the lower Kis-Balaton Reservoir (Hungary). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 323-329.

*Itagaki, N., Nagafuchi, O., Takimoto, K. and Okada, M. (2000). Fate of pesticides in a shallow reservoir. Water, Science and Technology, 42(7-8): 217-222.

Izaguirre, G., Taylor, W.D. and Pasek, J. (1999). Off-flavor problems in two reservoirs, associated with planktonic *Pseudanabaena* species. Water Science and Technology, 40(6): 85-90.

Jackson. J. (1992). Heavy metals and other inorganic toxic substances. In: *Guidelines of Lake Management. Volume 4.* International Lake Environment Committee, Kusatsu.

Jain, S.K. and Singh, P. (2002). Assessment of sedimentation in Bhakra Reservoir in the western Himalayan region using remotely sensed data. Hydrological Sciences Journal, 47(2): 203-212.

Jakobsen, P.J. and Johnsen, G.G. (1987). The influence of predation on horizontal distribution of zooplankton species. Freshwater Biology, 17: 501-507.

James, W. F., Barko, J. W. and Taylor, W.D. (1991). Effects of alum treatment on phosphorus dynamics in a north-temperate reservoir. Hydrobiologia, 215: 231-241.

James, W. F. and Barko, J. W. (1997). Net and gross sedimentation in relation to the phosphorus budget of Eau Galle Reservoir, Wisconsin. Hydrobiologia, 345: 15-20.

James, W. F., Barko, J. W., Eakin, H.L. and Helsel, D.R. (2001). Changes in sediment characteristics following drawdown of Big Muskego Lake, Wisconsin. Archiv fur Hydrobiologie, 151(3): 459-474.

*Jang, M., Ha, K. and Joo, G. (2003). Toxin-mediated interaction between cyanobacteria and native fishes in the eutrophic Hoedong Reservoir, South Korea. Journal of Freshwater Ecology, 18(4): 639-646.

*Jannson, M.B. and Erlingsson, U. (2000). Measurement and quantification of a sedimentation budget for a reservoir with regular flushing. Regulated Rivers: Research and Management, 16: 279-306.

Jarvis, A.C. (1986). Zooplankton community grazing in a hypertrophic lake (Hartbeespoort Dam, South Africa). Journal of Plankton Research, 8: 1065-1078.

*Jenkins, P.T. (1996). Free trade and exotic species introductions. Conservation Biology, 10: 300-302.

*Jennings, M.J., Fore, L.S. and Karr, J.R. (1995). Biological monitoring of fish assemblages in Tennessee Valley Reservoirs. Regulated Rivers: Research and Management, 11: 263-274.

Jensen, J.P., Jeppesen, E., Kristensen, P., Christensen, P.B. and Sondergaard, M. (1992). Nitrogen loss and denitrification as studied in relation to reductions in nitrogen loading in a shallow hypertrophic Lake Sobygard, Denmark. International Review of Hydrobiology, 77: 29-42.

Jeppesen, E., Sondergaard, M., Mortensen, E., Kristensen, P., Riemann, B., Jensen, H.J., Muller, J.P., et al. (1990). Fish manipulation as a lake restoration tool in shallow, eutrophic temperate lakes 1: cross analysis of three Danish case studies. Hydrobiologia, 200-201 205-218.

Jeppesen, E., Jensen, J.P., Sondergaard, M., Lauridsen, T., Pedersen, L.G. and Jensen, L. (1997). Top-down control in freshwater lakes: the role of the nutrient state, submerged macrophytes and water depth. Hydrobiologia 342-343: 151-164.

Jezbera, J., Nedoma, J. and Simek, K. (2003). Longitudinal changes in protistan bacterivory and bacterial production in two canyon-shaped reservoirs of different trophic status. Hydrobiologia, 504: 115-130.

*Jin, L., Hui, Y. and Xu, X. (1999). Predictive model for mercury accumulation in carp *(Cyprinus carpio)* of reservoirs in China. Water, Air and Soil Pollution, 115: 363-370.

*Jochimsen, E.M., Carmichael, W.W., Cardo, D.M., Cookson, S.T., Holmes, C.E.M., Azevedo, S.M.F.O. and Jarvis, W.R. (1998). Liver failure and death after exposure to microcystin at a haemodialysis centre in Brazil. New England Journal of Medicine, 338: 873-878.

Johannson, O.E., Mills, E.L. and O'Gorman, R. (1991). Changes in the nearshore and offshore zooplankton communities in Lake Ontario: 1981-88. Canadian Journal of Fisheries and Aquatic Sciences, 48: 1546-1557.

Johnsen, G.H. and Jacobsen, P.J. (1987). The effect of food limitition on vertical migration in *Daphnia longispina*. Limnology and Oceanography, 32: 873-880.

*Johnson, B.M., Saito, L., Anderson, M.A., Weiss, P., Andre, M. and Fontane, D.G. (2004). Effects of climate and dam operations on reservoir thermal structure. Journal of Water Resources Planning and Management, 130(2): 112-122.

Johnson, D.B. and Hallberg, K.B. (2005). Acid mine drainage remediation options: a review. Science of the Total Environment, 338(1-2): 3-14.

*Johnson, R.K. and Goedkoop, W. (1992). Interactions of *Monoporeia affinis* (Lindstrom) (Amphipoda) with sedentary Chironomidae. Netherlands Journal of Aquatic Ecology, 26: 491-497.

Jonasson, P.M. (1996). Limits for life in the lake ecosystem. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 1-33.

*Jones, G.J. and Korth, W. (1995). In situ production of volatile odour compounds by river and reservoir phytoplankton populations in Australia. Water Science and Technology, 31(11): 145-151.

*Jones, G.J. and Poplawski, W. (1998). Understanding and management of cyanobacterial blooms in sub-tropical reservoirs of Queensland, Australia. Water Science and Technology, 37(2): 161-168.

*Jones, J.R., Perkins, B.D., Witt, A., Kaiser, M.S., Thamasara, S., Siriworakul, M, and Benyasut, P. (2000). Limnological characteristics of some reservoirs in Thailand. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2158-2166.

Jones, R.I. (1992). The influence of humic substances on lacustrine planktonic food chains. Hydrobiologia, 229: 73-91.

Jorgensen, S.E. (2002). The application of models to find the relevance of residence time in lake and reservoir management. Journal of Limnology, 62(Suppl 1): 16-20.

Jorgensen, S.E. (1992). *Lake Model for IBM PC.* International Lake Environment Committee (ILEC), Kusatsu, Japan.

Jorgensen, S.E. (1997). Effects of lake acidification. In: *Guidelines of Lake Management. Volume5.* International Lake Environment Committee, Kusatsu.

Josefson, A.B. and Rasmussen, B. (2000). Nutrient retention by benthic macrofaunal biomass of Danish estuaries: importance of nutrient load and retention time. Estuarine and Coastal Shelf Science, 50: 205-216.

*Jugnia, L.-B., Sime-Ngando, T., Debroas, D., Tadonleke, R.D. and Devaux, J. (2001). Bacteria-phytoplankton relationships in a recently formed reservoir (Sep, France). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3820-3824.

Jugnia, L.-B., Richardot, M., Debroas, D., Sime-Ngando, T. and Devaux, J. (2000). Variations in the number of active bacteria in the euphotic zone of a recently flooded reservoir. Aquatic Microbial Ecology, 2: 251-259.

Jupp, D.L.B., Kirk, J.T.O. and Harris, G.P. (1994). Detection, identification and mapping of Cyanobacteria – using remote sensing to measure the optical quality of turbid inland waters. Australian Journal of Marine and Freshwater Research, 45: 801-828

Juracek, K. E.; Mau, D. P. (2003). Metals, trace elements, and organochlorine compounds in bottom sediment of Tuttle Creek Lake, Kansas, USA. Hydrobiologia 494: 277-282

Jurgens, K., Arndt, H. and Rothaupt, K.O. (1994). Zooplankton-mediated changes of bacterial community structure. Microbiology Ecology, 27: 27-42.

Jurkiewicz-Karnkowska, E. (2005). Some aspects of nitrogen, carbon and calcium accumulations in molluscs from the Zegrzynski Reservoir ecosystem. Polish Journal of Environmental Studies, 14(2): 173-177.

*Kagawa, H., Fukuroda, Y. and Hirotani, H. (2000). Nutritive elements in sediment in relation to chlorophyll concentration in the epilimnion of a reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2198-2202.

*Kalajzic, T., Bianchi, M., Muntau, H. and Kettrup, A. (1998). Polychlorinated biphenyls (PCBs) and organochlorine pesticides (OCPs) in the sediments of an Italian drinking water reservoir. Chemosphere, 36(7): 1615-1625.

Kalff, J. (2002). *Limnology. Inland Water Ecosystems.* Prentice-Hall, Inc.

Karul, C., Soyupak, S. and Yurteri, C. (1999). Neural network models as a management tool in lakes. Hydrobiologia, 408-9: 139-144.

Kasprzak, P., Krienitz, L. and Koschel, R. (1993). Changes in the plankton of Lake Feldberger Haussee (Germany, Mecklenburg-Vorpommern) in response to biomanipulation. Archiv fur Hydrobiologie, 128: 149-168.

*Kawabata, Z. and Hirano, Y. (1995). Growth pattern and cellular nitrogen and phosphorus contents of the dinoflagellate *Peridinium penardii* (Lemm.) causing a freshwater red tide in a reservoir. Hydrobiologia, 312: 115-120.

Kawara, O., Yura, E., Fujii, S. and Matsumoto, T. (1998). A study on the role of hydraulic retention time in eutrophication of the Asahi River Dam reservoir. Water Science and Technology, 37(2): 245-252.

Keller, M. and Stallard, R.F. (1994). Methane emission by bubbling from Gatun Lake, Panama. Journal of Geophysical Research, 99(D4): 8307-8319.

Kelly, C.A., Rudd, J.W.M., Hesslin, R.H., Schindler, D.W., Dillon, P.J., Driscoll, C.T., Gherini, S.A. and Heskey, R.H. (1987). Prediction of biological acid neutralization in acid sensitive lakes. Biogeochemistry, 3: 129-140.

Kenmuir, D.H.S. (1980). Aspects of the biology and population dynamics of freshwater mussels in Lake Kariba and Lake McIlwaine. Ph.D. thesis, University of Natal.

Kennedy, R.H. (1999). Reservoir design and operation: Limnological implications and management opportunities. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 1-28.

Kennedy, R.H. and Walker, W.W. (1990). Reservoir nutrient dynamics. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 109-131.

Kennedy, R.H., Carroll, J.N., Hains, J.J., Jabour, W.E. and Ashby, S.L. (1995). Water quality management at a large hydropower reservoir: design, operation and effectiveness of an oxygenation system. In: *Harmonizing Human Life with Lakes. 6th International Conference on the Conservation and Management of Lakes – Kasumigaura 95.* ILEC, pp. 517-520.

Ketelaars, H.A.M. and van Breemen, L.W.C.A. (1993). The invasion of the predatory cladoceran *Bythotrephes longimanus* Leydig, and its influence on the plankton communities in the Biesbosch reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1168-1175.

*Kettunen, J. (1993). Design of limnological observations for detecting processes in lakes and reservoirs. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 139-146.57-67.

Kharchenko, T.A. and Zorina-Sakharova, E.F. (2002). The ecological consortia of bivalve molluscs in littoral zones of a flatland reservoir, as a structural and functional community of hydrobionts. Hydrobiological Journal, 38(1): 3-14.

*Khondker, M. and Kabar, M.A. (1995). Plankton primary production in a mesotrophic pond in sub-tropical Bangladesh. Hydrobiologia, 304: 39-47.

Kiibus, M. and Kautsky, N. (1996). Respiration, nutrient excretion and filtration rate of tropical freshwater mussels and their contribution to production and energy flow in Lake Kariba, Zimbabwe. Hydrobiologia, 331: 25-32.

Kim, H.-W., Hwang, S.-J. and Joo, G.-J. (2000). Zooplankton grazing on bacteria and phytoplankton in a regulated large river (Nakdong River, Korea). Journal of Plankton Research, 22: 1559-1577.

*Kim, S.D., Ma, H., Allen, H.E. and Cha, D.K. (1999). Influence of dissolved organic matter on the toxicity of copper to *Ceriodaphnia dubia:* effect of complexation kinetics. Environmenal Toxicology and Chemistry, 18: 2433-2437.

Kimmel, B.L., Lind, O.T. and Paulson, L.J. (1990). Reservoir Primary Production. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 133-178.

King, C.R. and Greenwood, J.G. (1992). The productivity and carbon budget of a natural population of *Daphnia lumholtzi* Sars. Hydrobiologia, 231: 197-207.

*King, C.R. and Greenwood, J.G. (1992). The seasonal population changes and carbon budget of the calanoid copepod *Boeckella minuta* Sars in a newly formed sub-tropical reservoir. Journal of Plankton Research, 14(3): 329-342.

King, L., Jones, R.I. and Barker, P.A. (2000). Periphyton response to nutrient manipulation in an oligotrophic lake (Wastwater, English Lake District). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3154-3158.

Kiss, M.K., Lakatos, G., Borics, G., Gido, Z and Deak, C. (2003). Littoral macrophyte-periphyton complexes in two Hungarian shallow waters. Hydrobiologia 506-509: 541-548.

Kitchell, J.F., O'Neill, R.V., Webb, D., Gallepp, G.W., Bartell, S.M., Koonce, J.F. and Ausmus, B.S. (1979). Consumer regulation of consumer cycling. Bioscience, 29: 28-34.

Kitner, M. and Poulickova, A. (2003). Littoral diatoms as indicators for the eutrophication of shallow lakes. Hydrobiologia, 506-509: 519-524.

*Kivrak, E. and Gurbuz, H. (2005). The benthic algal flora of Demirdoven Dam Reservoir (Erzurum, Turkey). Turkish Journal of Botany, 29: 1-10.

*Klapper, H. (1998). Water quality problems in reservoirs of Rio de Janeiro, Minas Gerais and Sao Paulo. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 83: 93-102.

Klapper, H. (2002). Technologies for lake restoration. Journal of Limnology, 62(Suppl. 1): 73-90.

*Kleeberg, A. and Heidenreich, M. (2004). Release of nitrogen and phosphorus from macrophyte stands of summer dried out sediments of a eutrophic reservoir. Archiv fur Hydrobiologie, 159(1): 115-136.

Klenus, V.G., Kuzmenko, M.I. Matviyenko, L.P., Kaglyan, A.I. and Nasvit, O.I. (1992). Radionuclide concentrations in water and suspended matter of the Dnieper and its reservoirs. Hydrobiological Journal, 28: 34-39.

Klochenko, P.D. and Medved, V.A. (2001). The influence of lead and copper on some indices of vital activity of green and blue-green algae. Hydrobiological Journal, 37(2): 39-52.

*Knapp, C.W., de Noyelles, F., Graham, D.W. and Bergin, S. (2003). Physical and chemical conditions surrounding the diurnal vertical migration *of Cryptomonas* spp. (Cryptophyceae) in a seasonally stratified midwestern reservoir (USA). Journal of Phycology, 39: 855-861.

Kneebone, P.E., O'Day, P.A., Jones, N. and Hering, J.G. (2002). Deposition and fate of arsenic in iron- and arsenic-enriched reservoir sediments. Environmental Science and Technology, 36: 381-386.

*Knowlton, M.F. and Jones, J.R. (1996b). Experimental evidence of light and nutrient limitations of algal growth in a turbid Midwest reservoir. Archiv fur Hydrobiologie, 135: 321.

*Koelmans, A.A., van der Heijde, A., Knijff, L.M. and Aalderink, R.H. (2001). Integrated modelling of eutrophication and organic contaminant fate and effects in aquatic ecosystems. A review. Water Research, 35(15): 3517-3536.

*Kohler, J. (1992). Influence of turbulent mixing on growth and primary production of *Microcystis aeruginosa* in the hypertrophic Bautzen Reservoir. Archiv fur Hydrobiologie, 123: 413-429.

Kolar, C.S. and Lodge, D.M. (2001). Progress in invasion biology: predicting invaders. Trends in Ecology and Evolution, 16: 199-204.

Komarkova, J. (1993). Cycles of phytoplankton during long-term monitoring of Rimov and Slapy Reservoirs (Czech Republic). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1187-1191

*Komarkova, J. (2002). Do cyanobacterial picoplankton exist in eutrophic reservoirs? Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie. 28: 497-500.

Komarkova, J., Vyhnalek, V. and Kubecka, J. (1995). Impact of fishstock manipulation on the composition of net phytoplankton in the Rimov Reservoir (Czech Republic). Water Science and Technology, 32(4): 207-216.

Komarkova, J. and Hejzlar, J. (1996). Summer maxima of phytoplankton in the Rimov Reservoir in relation to hydrologic parameters and phosphorus loading. Archiv fur Hydrobiologie, 136: 217-236.

*Komarkova, J., Komarek, O. and Hejzlar, J. 2003. Evaluation of the long term monitoring of phytoplankton assemblages in a canyon-shaped reservoir using multivariate statistical methods. Hydrobiologia 504 (1-3): 143-157.

*Kondratyev, K.Y. and Pozdnyakov, D.V. (1990). Passive and active remote sensing of phytoplankton in inland waters. Journal of Photogrammetry and Remote Sensing, 44: 257-294.

Kondratyev, K.Y.; Bobylev, L.P., Pozdnyakov, D.V., Melentyev. V.V., Naumenko, M.A., Mokievsky, K.A., Korotkerich, O.E., Zaitsev, L.V., Karetnikov, S.G., Beletsky, D.V. and Litvinenko, A.V. (1996). Combined application of remote sensing and *in situ* measurements in monitoring environmental processes. Hydrobiologia, 322: 227-232.

Korponai, J., Paulovits, G., Matyas, K. and Tatrai, I. (2003). Long-term changes of cladoceran community in a shallow hypertrophic reservoir in Hungary. Hydrobiologia, 504: 193-201.

Korponai, J., Matyas, K., Paulovits, G., Tatrai, I. and Kovacs, N. (1997). The effect of different fish communities on the cladoceran plankton assemblages of the Kis-Balaton Reservoir, Hungary. Hydrobiologia, 360: 211-221.

*Korneva, L.G. and Mineeva, N.M. (1996). Phytoplankton composition and pigment concentrations as indicators of water quality in the Rybinsk Reservoir. Hydrobiologia, 322: 255.

Korneva, L.G. and Solovyova, V.V. (1999). Spatial organization of phytoplankton in reservoirs of the Volga River. International Review of Hydrobiology, 83: 163-166.

Koskenniemi, E. (1992). The role of Chironomids (Diptera) in the profundal macrobenthos in Finnish reservoirs. Netherlands Journal of Aquatic Ecology, 26: 503-508.

Koskenniemi, E. (1994). Colonization, succession and environmental conditions of the macrozoobenthos in a regulated, polyhumic reservoir, Western Finland. International Review of Hydrobiology, 79(4): 521-555.

Koski, M., Schmidt, K., Engstrom-Ost, J., Viitasalo, M., Jonasdottir, S., Repka, S. and Sivonen, K. (2002). Calanoid copepods feed and produce eggs in the presence of toxic cyanobacteria *Nodularia spumigena.* Limnology and Oceanography 47: 878-885

*Kosolapov, D.P. and Namsarayev, B.B. (2001). The microbial metabolism of organic carbon in bottom sediments of the Rybinsk Reservoir. Hydrobiological Journal, 37(4): 109-116.

*Kovalchuk, A.A. (2000). Species composition and quantitative development of benthic ciliates in the reservoirs of the Dnieper River. Hydrobiological Journal, 36(2): 27-39

*Kovalchuk, A.A. (2001). Epiphytic Ciliata of the Dnieper Reservoirs. Hydrobiological Journal, 37(6): 45-64.

*Kovalchuk, A.A. (2003). Role of benthic protozoa and microinvertebrates of the shallow-water zone of the Kremenchug Reservoir in the processes of production and decomposition. Hydrobiological Journal, 39(4): 26-44

Kozlovskaya, V.I. and Bakanov, A.I. (2002). Effect of industrial and agricultural pollution on macrozoobenthos of the Sheksninskiy Reach of the Rybinsk Reservoir and its tributaries. Hydrobiological Journal, 38(3): 41-52.

Krahulec, F. and Kaplan, Z. (1994). Diversity of *Potamogeton* species during 21 years of succession in a new reservoir. Prestia, Praha, 66: 237-241.

*Krzyzanek, E. and Kasza, H. (1995). Formation of bottom macrofauna in the Goczalkowice Reservoir (southern Poland) against the background of changing selected physico-chemical properties of the water. Acta Hydrobiologica 37(2): 103-111.

Kubecka, J. (1993). Succession of fish communities in reservoirs of Central and Eastern Europe. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 153-168.

*Kubecka, J. and Duncan, A. (1994). Low fish predation pressure in the London reservoirs: Species composition, density and biomass. International Review of Hydrobiology, 79: 141-153.

Kubecka, J. and Wittingerova, M. (1998). Horizontal beaming as a crucial component of acoustic fish stock assessment in freshwater reservoirs. Fisheries Research, 35: 99-106.

*Kureyshevich, A.V. and Zhuravleva, L.A. (2000). Relationship between the content of chlorophyll *a* and concentration of nutrients in water of the Dnieper Reservoirs. Hydrobiological Journal, 36(6): 27-35.

Kureyshevich, A.V. (2004). Relationship between long-term dynamics of chlorophyll a content in plankton of the Dnieper Reservoirs and the total solar radiation and its activity. Hydrobiological Journal, 40(5): 15-25.

*Kwandrans, J. (1992). Characteristics of the phytobenthos communities in the Goczalkowice Reservoir (southern Poland). Acta Hydrobiologica, 34(1-2): 115-128.

Labadie, J.W. (2004). Optimal operation of multireservoir systems: state-of-the-art review. Journal of Water Resources Planning and Management, 130(2): 93-111.

*Lafforgue, M., Walker, D. and Simmons, J. (1993). Hanningfield Lake: Functioning and restoration processes. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1162-1167.

Lajczak, A. (1995). Proposed model for long-term course of mountainous reservoir sedimentation and and estimating the useful life of dams. In: *6th International Symposium on River Sedimentation.* Central Board of Irrigation and Power, New Delhi, pp. 699-711.

Lakatos, G., Kiss, M. and Meszaros, I. (1999). Heavy metal content of common reed (*Phragmites australis* Cav. Trin. ex Steudel) and its periphyton in Hungarian shallow standing waters. Hydrobiologia, 415: 47-53.

Lalende, S. and Downing, J.A. (1991). Epiphyton biomass is related to lake trophic status, depth, and macrophyte architecture. Canadian Journal of Fisheries and Aquatic Sciences, 48: 2285-2291.

Lam, A. K. Y. and E. E. Prepas. (1997) . In situ evaluation of options for chemical treatment of hepatotoxic cyanobacterial blooms. Canadian Journal of Fisheries and Aquatic Sciences, 54(8):1736-1742.

Lam, D.C.L., Mayfield, C.I., Swayne, D.A. and Hopkins, K. (1994). A prototype information system for watershed management and planning. Journal of Biological Systems, 2: 499-517.

*Lammens, E.H.R.R. and Hoogenboezem, W. (1991). Diets and feeding behaviour. In: Winfield, I.J. and Nelson, J.S. (eds). *Cyprinid Fishes – systematics, biology and exploitation*. Chapman and Hall, London; pp. 353-376.

Lammens, E. H. R. R.(1999). The central role of fish in lake restoration and management. Hydrobiology, 395-396: 191-198.

*Lamparelli, M.C., Carvalho, M.C. and de Souza, R.C.R. (2000). Water and sediment quality as a response to nutrients and metals (Al, Fe and Cu) in Guarapiranga Reservoir, Sao Paulo, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3199-3205.

Lampert, W. (1988). The relationship between zooplankton biomass and grazing. A review. Limnologica (Berlin), 19: 11-20.

Lampert, W. (1993). Ultimate causes of diel vertical migration of zooplankton: new evidence for the predator-avoidance hypothesis. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 39: 79-88.

*Lane, M.E., Kirshen, P.H. and Vogel, R.M. (1999). Indicators of impacts of global climate change on US water resources. Journal of Water Resources Planning and Management, 125(4): 194-204.

Lathrop, R.C. and Carpenter, S.R. (1992). Zooplankton and their relationship to phytoplankton. In: Kitchell, J.F. (ed). *Food Web Management. A case study of Lake Mendota.* Springer-Verlag, New York; pp. 127-150.

*Latour, D., Sabido, O., Salencon, M.-J. and Giraudet, H. (2004). Dynamics and metabolic activity of the benthic cyanobacterium *Microcystis aeruginosa* in the Grangent Reservoir (France). Journal of Plankton Research, 26(7): 719-726

*Latufo, G.R. (1998). Bioaccumulation of sediment – associated fluoranthene in benthic copepods: uptake, elimination and biotransformation. Aquatic Toxicology, 44: 1-15.

Lauridsen, T.L., Jeppesen, E., Sondergaard, M. and Lodge, D.M. (1997). Horizontal migration of zooplankton: predator mediated use of macrophyte habitat. In: Jeppesen, E., Sondergaard, M. and Christoffersen, K. (eds). *The Structuring Role of Submerged Macrophytes in Lakes.* Ecological Studies, New York, Springer Verlag; pp. 233-239.

Lauridsen, T.L. and Buenk, I. (1996). Diel changes in the horizontal distribution of zooplankton in two shallow eutrophic lakes. Archiv fur Hydrobiologie, 137: 161-176.

*Lavandier, P. (1990). Dynamics of bacterioplankton in a mesotrophic French reservoir (Paraloup). Hydrobiologia, 207: 79-86.

Lavrik, V.I., Dobrynskiy, V.A. and Rogal, I.V. (2002). Application of simulation mathematical modelling to the problems of management of eutrophication processes in lakes and reservoirs. Hydrobiological Journal, 38(6): 78-87.

Laybourn-Parry, J. (1992). *Protozoan Plankton Ecology*. Chapman and Hall, London 231pp.

Lazzaro, X. (1997). Do the trophic cascade hypothesis and classical biomanipulation approaches apply to tropical lakes and reservoirs? Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 719-730.

Lazzaro, X., Drenner, R.W., Stein, R.A. and Smith, J.D. (1992). Planktivores and plankton dynamics: effects of fish biomass and planktivore type. Canadian Journal of Fisheries and Aquatic Sciences, 49: 1466-1473.

Lazzaro, X., Bouvy, M., Ribeiro-Filho, R.A., Oliviera, V.S., Sales, L.T., Vasconcelos, A.R.M. and Mata, M.R (2003). Do fish regulate phytoplankton in shallow eutrophic Northeast Brazilian reservoirs? Freshwater Biology 48: 649-668.

Lecointe, C., Coste, M. and Prygiel, J. (1993). “OMNIDIA”: Software for taxonomy, calculation of diatom indices and inventories management. Hydrobiologia, 269-270: 509-513.

Lee, C. (1999). Rapid and repeated invasions by the freshwater copepod *Eurytemora affinis.* Evolution, 53: 1423-1434.

Lee, H.-Y. and Yu, W.-S. (1997). Experimental study of reservoir turbidity current. Journal of Hydraulic Engineering, 123(6): 520-528.

Lehman, J.T. (1991). Causes and consequences of cladoceran dynamics in Lake Michigan: implications of species invasions by *Bythotrephes.* Journal of Great Lakes Research, 17: 437-445.

Lei, J., Payne, B.S. and Wang, S.Y. (1996). Filtration dynamics of the zebra mussel, *Dreissena polymorpha*. Canadian Journal of Fisheries and Aquatic Sciences, 53: 29-37.

*Leipnik, M.R., Kemp, K.K. and Loaiciga, H.A. (1993). Implementation of GIS for water resources planning and management. Journal of Water Resources Planning and Management, 119(2): 184-205.

*Leitao, M. and Leglize, L. (2000). Long-term variations of epilimnetic phytoplankton in an artificial reservoir during a 10-year survey. Hydrobiologia, 424: 39-49.

Leitao, M., Morata, S.M., Rodriguez, S. and Vergon, J.P. (2003). The effect of perturbations on phytoplankton assemblages in a deep reservoir (Vouglans, France). Hydrobiologia, 502: 73-83.

*Leite, R.L. and Watanabe, T. (1997). Seasonal variation of the physical and chemical condition of the Gramame and Mamuabe Reservoirs, Alhandra, Paraiba. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 350-354.

Lemly, A.D. (1997). Ecosystem recovery following selenium contamination in a freshwater reservoir. Ecotoxicology and Environmental Safety, 36: 275-281.

Lennon, J.T., Smith, V.H. and Williams, K. (2001). Influence of temperature on exotic *Daphnia lumholtzi* and implications for invasion success. Journal of Plankton Research, 23(4): 425-434.

Lettenmaier, D.P. and Gan, T.Y. (1990). Hydrologic sensitivities of the Sacramento-San Joaquin River Basin, California, to global warming. Water Resources Research, 26: 69-86.

*Leventer, H. and Teltsch, B. (1990). The contribution of silver carp (*Hypophthalmichthys molitrix* to the biological control of Netofa reservoirs. Hydrobiologia, 191: 47-55.

*Levich, A.P. (1996). The role of nitrogen-phosphorus ratio in selecting for dominance of phytoplankton by cyanobacteria or green algae and its application to reservoir management. Journal of Aquatic Ecosystem Health, 5: 55-61.

Levin, S.A. (1992). The problem of pattern and scale in ecology. Ecology, 73: 1943-1967.

Levine, J.M. and D'Antonio, C.M. (1999). Elton revisited: a review of evidence linking diversity and invasibility. Oikos, 87: 15-26

*Lewis, A.F. and Mitchell, A.C. (1995). Effectiveness of water release as mitigation for hydroelectric impacts to fish. Journal of Energy Engineering, 121(2): 81-88.

*Lewis, D.M., Elliott, J.A., Lambert, M.F. and Reynolds, C.S. (2002). The simulation of an Australian reservoir using a phytoplankton community model: PROTECH. Ecological Modelling, 150: 107-116.

Lewis, D.M., Elliott, J.A., Brookes, J.D., Irish, A.E., Lambert, M.F. and Reynolds, C.S. (2003). Modelling the effects of artificial mixing and copper sulphate dosing on phytoplankton in an Australian reservoir. Lakes and Reservoirs: Research and Management, 8(1): 31-40.

Lewis, W.M. (1996). Tropical Lakes: How latitude makes a difference. In: Scheimer, F. and Boland, K.T. (eds): *Perspectives in tropical limnology*. SPB Academic Publishing, Amsterdam; pp. 43-64.

Li, J.L. and Li, H.W. (1979). Species-specific factors affecting predator-prey interactions of the copepod *Acanthocyclops vernalis* with its natural prey. Limnology and Oceanography, 24: 613-626.

*Lieberman, D.M. (1995). Nutrient limitation in a southwestern desert reservoir: eutrophication of Las Vegas Bay, Lake Mead, Nevada. Journal of Freshwater Ecology, 10(3): 241-253.

*Lieschke, J.A. and Closs, G.P. (1999). Regulation of zooplankton composition and distribution by a zooplanktivorous fish in a shallow, eutrophic floodplain lake in south east Australia. Archiv fur Hydrobiologie, 146: 397-412.

Lima, I.B.T., Victoria, R.L., Novo, E.M.L.M., Feigl, B.J., Ballester, M.V.R. and Ometto, J.P. (2002). Methane, carbon dioxide and nitrous oxide emissions from two Amazonian reservoirs during high water table. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 438-442.

Lima, I.B.T. (2005). Biogeochemical distinction of methane releases from two Amazon hydroreservoirs. Chemosphere, in press.

Lima, I.B.T., Novo, E.M.L.M., Ballester, M.V.R. and Ometto, J.P. (2000). Role of the macrophyte community in CH4 production and emission in the tropical reservoir of Tucurui, Para State, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 1437-1440.

*Lind, O. (2002). Microbial production and reservoir zone trophic states. Lake and Reservoir Management, 18: 129-137.

Lind, O.T. and Davalos. (1990). Clay, dissolved organic matter, and bacterial interactions in two reservoirs. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 34: 119-125.

Lind, O.T. and Davalos-Lind, L. (1991). Association of turbidity and organic carbon with bacterial abundance and cell size in a large, turbid tropical lake. Limnology and Oceanography, 36: 1200-1208.

Lind, O.T., Terrel, T. and Kimmel, B.L. (1993). Problems in reservoir trophic state classification and implications for reservoir management. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 57-67.

Lind, O., Davalos-Lind, L., Chrzanowski, T. and Limon, J. (1994). Inorganic turbidity and the failure of fishery models. International Review of Hydrobiology, 79: 7-16.

Lind, O., Chrzanowski, T. and Davalos-Lind, L. (1997). Clay turbidity and the relative production of bacterioplankton and phytoplankton. Hydrobiologia, 353, 1-18.

Lind, O.T. and Davalos-Lind, L. (1999). Suspended Clay: Its role in reservoir productivity. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 85-97.

Lind, O.T., Davalos-Lind, L.O. and Rutherford, K.B. (2001). Hypolimnion oxygen concentration and the abundance and size of bacteria. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3381-3384.

*Lind. O.T. and Barcena, E. (2003). Response of riverine and transition zone bacterioplankton communities to a pulsed river inflow. Hydrobiologia 504: 79-85.

Lindegaard, C. (1995). Classification of water–bodies and pollution. In: Armitage, P.D., Cranston, P.S. and Pinder, L.C.V. (eds): *The Chironomidae: Biology and Ecology of Non-biting Midges*. Chapman and Hall. London; pp. 385-404.

Lindell, M. and Edling, H. (1996). Influence of light on bacterioplankton in a tropical lake. Hydrobiologia, 323(1): 67-73.

Lindstrom, E.S. (2000). Bacterioplankton community composition in five lakes differing in trophic status and humic content. Microbiology Ecology, 40: 104-113.

Linnik, P.N. (1995a). Concentrations and forms of chromium in the Dnieper reservoirs and the Dnieper-Bug lagoon. Hydrobiological Journal, 30: 97-104.

Linnik, P.N. (1995b). Heavy metals in the rivers Dnieper and Danube and the role of intra-reservoir processes in the migration and transformation of their compounds. Archiv fur Hydrobiologie, 113: 1-4.

Linnik, P.N. (2000). Zinc, lead and cadmium in Dnieper water-bodies. Lakes and Reservoirs: Research and Management, 5: 261-270.

Linnik, P.N. (2001). Bottom sediments of reservoirs as a potential source of secondary pollution of the aquatic environment by heavy metal compounds. Hydrobiological Journal, 37(1): 73-86.

Linnik, P.N. and Zubenko, I.B. (2000). Role of bottom sediments in the secondary pollution of aquatic environments by heavy-metal compounds. Lakes and Reservoirs: Research and Management, 5: 11-21.

Linnik. P.N. and Vasilchuk, T.A. (2002). The role of humic substances in the processes of complexation and detoxication (by the example of the Dnieper Reservoirs). Hydrobiological Journal, 38(5): 82-97.

Lisle, J.T. and Rose, J.B. (1995). *Cryptosporidium* contamination of water in the USA and UK – a mini review. Aqua, 44(3): 103-117.

Litvinow, A.S. and Roschchupko, V.P. (1994). Changes of the Volga River thermal regimen after constructing the Volga River reservoir cascade. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 40: 41-44.

*Liu, C.W. and Narasimhan, T.N. (1994). Modelling of selenium transport at the Kesterson Reservoir, California, USA. Journal of Contaminant Hydrology, 15: 345-366.

Lodge, D.M. (1993). Biological invasions: Lessons for ecology, TREE, 8: 133-141.

Lohman, K., Jones, J.R., Knowlton, M.F., Swar, B.D., Pamperl, M.A. and Brazos, B.J. (1988). Pre- and post-monsoon limnological characteristics of lakes in the Pokhara and Kathmandu Valleys, Nepal. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 23: 558-565.

Lombardi, A.T. and Vieira, A.A.H. (1998). Copper and lead complexation by high molecular weight compounds produced by *Synura* sp. (Chrysophyceae). Phycologia, 37: 34-39.

*Lombardi, A.T. and Vieira, A.A.H. (2000). Copper complexation by Cyanophyta and Chlorophyta exudates. Phycologia, 39: 118-125.

*Lopez, P. and Morgui, J.A. (1993). Factors influencing fractional phosphorus composition in sediments of Spanish reservoirs. Hydrobiologia, 253: 73-82.

Los, F.J. (1991). *Mathematical simulation of algal blooms by the model BLOOM II* Version 2. Documentation Report, Delft Hydraulics, The Netherlands.

Loska, K. and Wiechula, D. (2003). Application of principal components analysis for the estimation of source of heavy metal contamination in surface sediments from the Rybnik Reservoir. Chemosphere, 51: 723-733.

*Lotti, C. (1991). Management of reservoirs. Water Resources Development, 7(2): 117-121.

Lougheed, V.L., Crosbie, B. and Chow-Fraser, P. (2001). Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes Basin: latitude, land use and water quality effects. Canadian Journal of Fisheries and Aquatic Sciences, 58: 1603-1612.

*Lu, X.X. and Higgitt, D.L. (2001). Sediment delivery to the Three Gorges. 2: Local response. Geomorphology, 41: 157-169.

Lucas, M.C. and Baras, E. (2001). *Migration of Freshwater Fishes.* Blackwell Science Ltd., Oxford. 420pp.

Lucotte, M., Mucci, A., Hillaire-Marcel, C., Pichet, P. and Grondin, A. (1995). Anthropogenic mercury enrichment in remote lakes of northern Quebec (Canada). Water, Air and Soil Pollution, 80: 467-476.

*Luecke, C., Rudstam, L.G. and Allen, Y. (1992). Interannual patterns of planktivory 1987-89: An analysis of vertebrate and invertebrate planktivores. In: Kitchell, J.F. (ed). *Food Web Management. A case study of Lake Mendota*. Springer-Verlag, New York; pp. 275-301.

Lung, W.S. and Canale, R.P. (1977). Projection of phosphorus levels in White Lake. Journal of Environmental Engineering Division, ASCE, 103: 663-667.

Luning, J. (1992). Phenotypic plasticity of *Daphnia pulex* in the presence of invertebrate predators: morphological and life-history responses. Oecologia, 92: 383-390.

Luning, J. (1995). Life-history responses to *Chaoborus* of spined and unspined *Daphnia pulex*. Journal of Plankton Research, 17: 71-84.

Lunte, C.C. and Luecke, C. (1990). Trophic interactions of *Leptodora* in Lake Mendota. Limnology and Oceanography, 35: 1091-1100.

Luoma, S.N., Johns, C., Fisher, N.S., Steinberg, C.E., Oremland, R.S. and Reinfelder, J.R. (1992). Determination of selenium bioavailability to a benthic bivalve from particulate and solute pathways. Environmental Science and Technology, 26: 485-491.

*Lusk, S. (1995). Influence of valley dams on the changes in fish communities inhabiting streams in the Dyje River drainage area. Folia Zoologica, 44: 45-56.

Lyashenko, A.V. and Protasov, A.A. (2003). Use of the indices of macrozoobenthos diversity as indicators of the state of aquatic ecosystems. Hydrobiological Journal, 39(4): 16-25.

*Lyche, A. (1989). Plankton community response to reduction of planktivorous fish populations: a review of 11 case studies. Aqua Fennica, 19: 59-66.

Lyche, A., Faafeng, B.A. and Braband, A. (1990). Predictability and possible mechanisms of plankton response to reduction of planktivorous fish. Hydrobiologia 200/201: 251-261

Lyche, A., Anderson, T., Christoffersen, K., Hessen, P.H. and Klysner, A. (1996). Mesocosm tracer studies. 1. Zooplankton as sources and sinks in the pelagic phosphorus cycle of a mesotrophic lake. Limnology and Oceanography, 41: 460-474

*Maassen, S., Roske, I. and Uhlmann, D. (2003). Chemical and microbial composition of sediments in reservoirs with different trophic state. International Review of Hydrobiology, 88: 508-518.

*Macek, L. and Grunwald, A. (1996). Influence of acid deposition on water quality in the Sous Reservoir. Water, Science and Technology, 33(4-5): 285-290.

Machacek, J. (1991). Indirect effects of planktivorous fish on the growth and reproduction of *Daphnia galeata*. Hydrobiologia, 225: 193-197.

Machacek, J. (1993). Comparison of the response of *Daphnia galeata* and *Daphnia obtusa* to fish-produced chemical substrate. Limnology and Oceanography, 38: 1544-1550.

*Machacek, J. (2001). *Daphnia galeata* life history response to heterogeneous food conditions and dissolved chemicals in the Rimov Reservoir. Hydrobiologia, 442: 215-222.

*Machacek, J. and Seda, J. (1998). Spatio-temporal changes of morphological and life-history parameters in *Daphnia galeata* in a canyon-shaped dam lake. International Review of Hydrobiology, 83: 171-178.

Maceina, M.J., Cichra, M.F., Betsill, R.K. and Bettoli, P.W. (1992) Limnological changes in a large reservoir following vegetation removal by grass carp. Journal of Freshwater Ecology, 7(1): 81-95

MacIsaac, H.J., Sprules, W.G., Johannsson, , O.E. and Leach, J.H. (1992). Filtering impacts of larval and sessile zebra mussels *(Dreissena polymorpha)* in western Lake Erie. Oecologica, 92: 30-39.

MacIsaac, H.J., Grigorovich, I.A. and Ricciardi, A. (2001). Reassessment of species invasion concepts: the Great Lakes basin as a model. Biological Invasions, 3: 405-416.

Mack, R.N. (1996). Predicting the identity and fate of invaders: emergent and emerging approaches. Biological Conservation, 78: 107-121.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M. and Bazzaz, F.A. (2000). Biotic invasions: causes, epidemiology, global consequences and control. Ecological Applications, 10: 689-710.

*Mackay, N.A. and Elser, J.J. (1998a). Factors potentially preventing trophic cascades: food quality, invertebrate predation, and their interaction. Limnology and Oceanography, 43: 339-347.

MacKenzie, W.R., Hoxie, N.J., Procter, M.E., Gradus, M.S., Blair, K.A. and Peterson, D.E. (1994). A massive outbreak in Milwaukee of *Cryptosporidium* infection transmitted through the public water supply. New England Journal of Medicine, 331(3): 161-167.

Mackie, G.I. (1991). Biology of the exotic mussel, *Dreissena polymorpha* in relation to native bivalves and its potential impact on Lake St.Clair. Hydrobiologia, 219: 251-268.

Macumber, P.G. (1991). *Interaction between Groundwater Surface Systems in Northern Victoria*. Department of Conservation and Environment, Melbourne.

Magadza, C.H.D. (1994). Evaluation of eutrophication control in Lake Chivero, Zimbabwe, by multivariate analysis of zooplankton. Hydrobiologia, 272: 277-292.

Magadza, C.H.D. (2003). Lake Chivero: A management case study. Lakes and Reservoirs: Research and Management, 8(2): 69-81.

Magalhaes, M.F. (1992). Feeding ecology of the Iberian cyprinid *Barbus bocagei* Steindachner, 1865 in a lowland river. Journal of Fish Biology, 40: 123-133.

Magalhaes, M.F. (1993). Feeding of an Iberian stream cyprinid assemblage: seasonality of resource use in a highly variable environment. Oecologia, 96: 253-260.

Magrin, A.G.E. and Matsumura-Tundisi, T. (1997). Phytoplankton assemblages in the Samuel Reservoir (Rondonia State, Brazil) during the filling phase. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 529-534.

Mahmood, K. (1987). Reservoir sedimentation: impact, extent and mitigation. World Bank Technical Paper Number 71, Washington DC.

Malthus, T.J. and George, D.G. (1997). Airborne remote sensing of macrophytes in Cefni Reservoir, Anglesey, U.K. Aquatic Botany, 58: 317-332.

Mangas-Ramirez, E., Sarma, S.S.S. and Nandini, S. (2001). Acute and chronic toxicity of ammonium chloride to the cladoceran *Daphnia pulex* Leydig in relation to algal food density. Bulletin of Environmental Contamination and Toxicology, 67(6): 834-840.

*Mangas-Ramirez, E. and Elias-Gutierrez, M. (2004). Effect of mechanical removal of water hyacinth (*Eichhornia crassipes)* on the water quality and biological communities in a Mexican reservoir. Aquatic Ecosystem Health and Management, 7(1): 161-168.

Margalef, R. (ed) (1993). *Limnologia.* Omega. Barcelona, Spain. pp720.

*Mariazzi, A., Conzonno, V., Echenique, R. and Labollita, H. (1991). Physical and chemical characters, phytoplankton and primary production of Ezequiel Ramos Mexia Reservoir (Argentina). Hydrobiologia, 209: 107-116.

*Mariazzi, A., Di Siervi, M.A. and Donadelli, J.L. (1991). Bacterial secondary production and its relation with primary production in the Embalse del Rio III Reservoir, Argentina. Hydrobiologia, 211: 57-64.

Markosova, R. and Jezek, J. (1994). Indicator bacteria and limnological parameters in fish ponds. Water Research, 28(12): 2477-2485.

*Marques, R.T. and Boavida, M.J. (1993). Effects of the water level fluctuation on the forms of phosphorus in mesotrophic Maranhao Reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1207-1209.

Marshall, B.E. (1982). The benthic fauna of Lake McIlwaine. In: Thornton, J.A. (Ed). *Lake McIlwaine. The Eutrophication and Recovery of a Tropical Man-made Lake.* Dr W. Junk Publishers, The Hague, pp. 144-155.

Marshall, B.E., Junor, F.J.R. & Langerman, J.D. (1982). Fisheries and fish production of the Zimbabwean side of Lake Kariba. Kariba Studies 10: 175-231.

*Marshall, T.R. and Lee, P.F. (1994). Mapping aquatic macrophytes through digital image-analysis of aerial photographs: an assessment. Journal of Aquatic Plant Management, 32: 61-66.

Martinez-Tabche, L., Gomez-Olivan, L., Martinez, M.G., Castillo, C.R. and Santiago, A.M. (2000). Toxicity of nickel in artificial sediment on acetylcholinesterase activity and haemoglobin concentration of the aquatic flea, *Moina macrocopa*. Journal of Environmental Hydrology, 8: 1-10.

Martinez-Tabche, L., Ortega, M.D.A.G., Mora, B.R., Faz, C.G., Lopez, E. and Martinez, M.G. (2001). Haemoglobin concentration and acetylcholinesterase activity of oligochaetes in relation to lead concentration in spiked sediments from Ignacio Ramirez Reservoir. Ecotoxicology and Environmental Safety, 49: 76-83.

Martinez, J.C.C., Canesin, A. and Bonecker, C.C. (2000). Species composition of rotifers in different habitats of an artificial lake. Mato Grosso do Sul State, Brazil. Acta Scientiarum, 2: 343-346.

Martinova, M.V. (1993). Nitrogen and phosphorus compounds in bottom sediment: Mechanisms of accumulation, transformation and release. Hydrobiologia, 252: 1-22.

Marzolf, G.R (1984). Reservoirs in the great plains of North America. In: Taub, F.B. (Ed). *Ecosystems of the World, 23, Lakes and Reservoirs.* Elsevier; pp 291-302.

Marzolf, G.R. (1990). Reservoirs as environments for zooplankton. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (eds*). Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 195-208.

Mashina, V.P. (2003). Structure of communities of free-living benthic Nematoda of the Kremenchug Reservoir. Hydrobiological Journal, 39(3): 111-115.

Masin, M., Jezbera, J., Nedoma, J., Straskrabova, V., Hejzlar, J. and Simek, K (2003). Changes in bacterial community composition and microbial activities along the longitudinal axis of two canyon-shaped reservoirs with different inflow loading. Hydrobiologia 504: 99-113.

*Mastin, B.J., Rodgers, J.H. and Deardorff, T.L. (2002). Risk evaluation of cyanobacteria-dominated algal blooms in a North Louisiana reservoir. Journal of Aquatic Ecosystem Stress and Recovery, 9: 103-114.

Mason, C.F. and Abdul-Hussein, M.M. (1991). Population dynamics and production of *Daphnia hyalina* and *Bosmina longirostris* in a shallow, eutrophic reservoir. Freshwater Biology, 25: 243-260.

*Masundire, H.M. (1994). Seasonal trends in zooplankton densities in Sanyati basin, Lake Kariba: multivariate analysis. Hydrobiologia, 272: 211-230.

*Masundire, H.M. (1997). Spatial and temporal variations in the composition and density of crustacean plankton in the five basins of Lake Kariba, Zambia-Zimbabwe. Journal of Plankton Research, 19(1): 43-62..

Matilainen, T. (1995). Involvement of bacteria in methylmercury formation in anaerobic lake waters. Water, Air and Soil Pollution, 80: 757-764.

Matsui, S. (1991). *Toxic Substances Management in Lakes and Reservoirs. Guidelines of Lake Management.* Volume 4. International Lake Environment Committee, Kusatsu

Matsumura, T. and Yoshiuki, S. (1981). An optimization problem related to the regulation of influent nutrient in aquatic ecosystems. International Journal of Systematic Sciences, 12: 565-585.

Matsumura-Tundisi, T. and Tundisi, J.G. (1997). Mixing processes affecting primary production of phytoplankton in Barra Bonita Reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 536-541.

*Matsumura-Tundisi, T. and Tundisi, J.G. (2003). Calanoida (Copepoda) species composition changes in the reservoirs of Sao Paulo State (Brazil) in the last twenty years. Hydrobiologia, 504: 215-222.

Matsumura-Tundisi, T., Tundisi, J.G. and Rocha, O. (2002). Zooplankton diversity in eutrophic systems and its relation to the occurrence of cyanophycean blooms. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 671-674.

*Mattos, S.P., Estuqui, V.R. and Cavalcanti, C.G.B. (1997). Lake Paranoa (Brazil): Limnological aspects with emphasis on the distribution of the zooplankton community (1982-1994). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 542-547.

Matveev, V. (1998). Evidence for biotic control of Australian reservoir phytoplankton communities and the potential for applied biomanipulation. Lakes and Reservoirs: Research and Management, 3: 105-111.

Matveev, V. (2003). Testing predictions of the lake food web theory on pelagic communities of Australian reservoirs. Oikos, 100: 149-161.

Matveev, V. and Balseiro, E.G. (1990). Contrasting responses of two cladocerans to changes in the nutritional value of nanoplankton. Freshwater Biology, 23: 197-204.

Matveev, V., Matveeva, L. and Jones, G.L. (1994). Study of the ability of *Daphnia carinata* King to control phytoplankton and resist cyanobacterial toxicity: implications for biomanipulation in Australia. Australian Journal of Marine and Freshwater Research, 45: 889-904.

Matveeva, L.K. and Matveev, V.F. (1995). The stimulation of phytoplankton growth by an invertebrate predator. Russian Journal of Aquatic Ecology, 4: 1-7.

Matveev, V. and Matveeva, L. (1997a). Foodweb interactions and biomanipulation in Australian reservoirs. Freshwater Forum, 8: 65-68.

Matveev, V. and Matveeva, L. (1997b). Grazer control and nutrient limitation of phytoplankton biomass in two Australian reservoirs. Freshwater Biology, 38: 49-65.

*Matveev, V. and Matveeva, L. (1998). Trophic level interactions in Australian reservoirs: Evidence from plankton and fisheries acoustics. International Review of Hydrobiology, 83: 443-450.

*Matveev, V., Matveeva, L. and Jones, G.J. (2000) Relative impacts of Daphnia grazing and direct stimulation by fish on phytoplankton abundance in mesocosm communities. Freshwater Biology 44: 375-386

Matveev, V., Closs, G., Lieschke, J.A. and Shirley, M.J. (2002). Are pelagic fish important in the food webs of Australian reservoirs? Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 116-119.

Matvienko, B., Sikar, E., Pinguelli Rosa, L., dos Santos, M.A., De Filippo, R. and Cimbleris, A.C.P. (2000). Gas release from a reservoir in the filling stage. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 1415-1419.

Matyas, K., Oldal, I., Korponai, J., Tatrai, I. and Paulovits, G. (2003). Indirect effect of different fish communities on nutrient chlorophyll relationship in shallow hypertrophic water quality reservoirs. Hydrobiologia 504: 231-239.

*Matyas, K. Tatrai, I., Paulovits, G. and Korponai, J. (2001). Effect of different fish stocks on phytoplankton dynamics in shallow ecosystems. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3300-3303.

Mayo, M., Gitelson, A., Yacobi, Y.Z. and Ben-Avraham, Z. (1995). Chlorophyll distribution in Lake Kinneret determined from Landsat Thematic Mapper data. International Journal of Remote Sensing, 16: 175-182.

Mazumder, A.. (1994). Patterns of algal biomass in dominant odd vs even-link lake ecosystems. Ecology, 75: 1141-1149.

Mazumder, A. and Taylor, W.D. (1994). Thermal structure of lakes varying in size and water clarity. Limnology and Oceanography, 39: 968-976.

McCauly, E. and Kalff, J. (1981). Empirical relationships between phytoplankton and zooplankton biomass in lakes. Canadian Journal of Fisheries and Aquatic Sciences, 38: 458-463.

*McCord, S.A. (1997). Lake and Reservoir Management. Water Environment Research. 69(4): 737-749.

McCormick, P.V. and Stevenson, R.J. (1998). Periphyton as a tool for ecological assessment and management in the Florida Everglades. Journal of Phycology, 34: 726-733.

McCutcheon, S.C. (1989). *Water Quality Modelling Vol. 1 Transport and Surface Exchanges in Rivers*. CRC Press, Boca Raton, Florida.

McGinnis, D.F. and Little, J.C. (1998). Bubble dynamics and oxygen transfer in a Speece Cone. Water Science and Technology, 37(2): 285-292.

McGregor, G.B. and Fabbro, L.D. (2000). Dominance of *Cylindrospermopsis raciborskii* (Nostocales, Cyanoprokaryota) in Queensland tropical and subtropical reservoirs: implications for monitoring and management. Lakes and Reservoirs: Research and Management, 5(3): 195-205.

*McQueen, D.J., Johannes, M.R.S., Post, J.R., Stewart, T.J. and Lean, D.R.S. (1989). Bottom-up and top-down impacts on freshwater pelagic community structure. Ecological Monographs, 59: 289-309.

McQueen, D.J. (1990). Manipulating lake community structure: Where do we go from here? Freshwater Biology, 23: 613-620.

McQueen, D.J., France, R. and Kraft, C. (1992). Confounded impacts of planktivorous fish on freshwater biomanipulations. Archiv fur Hydrobiologie, 125: 1-24.

Meays, C.L., Broersma, K., Nordin, R. and Mazumder, A. (2004). Source tracking faecal bacteria in water: a critical review of current methods. Journal of Environmental Management, 73: 71-79.

Medema, G.J., Bahar, M. and Schets, F.M. (1997). Survival of *Cryptosporidium parvum, Escherichia coli,* faecal enterococci and *Clostridium perfringens* in river water: influence of temperature and autochthonous microorganisms. Water Science and Technology, 35: 249-252.

*Mehner, T., Schultz, H. and Herbst, R. (1995). Interaction of zooplankton dynamics and diet of 0+ perch (*Perca fluviatilis*) in the top-down manipulated Bautzen Reservoir (Saxony, Germany) during summer. Limnologica, 25: 1-9.

Mehner, T., Hulsmann, S., Worischka, S., Plewa, M. and Benndorf, J. (1998a). Is the midsummer decline of *Daphnia* really induced by age-0 fish predation? Comparison of fish consumption and *Daphnia* mortality and life history parameters in a biomanipulated reservoir. Journal of Plankton Research, 20(9): 1797-1811.

Mehner, T., Mattukat, F., Bauer, D., Voight, H. and Benndorf, J. (1998b). Influence of diet shifts in underyearling fish on phosphorus recycling in a hypertrophic biomanipulated reservoir. Freshwater Biology 40: 759-769.

*Mehner, M., Benndorf, J., Kasprzak, P. and Koschel, R. (2002). Biomanipulation today: Successful applications and expanding complexity in science. Freshwater Biology, 47: 2453-2295.

Mehra, A., Farago, M.E., Banerjee, D.K. and Cordes, K.B. (1999). The water hyacinth: An environmental friend or pest? A review. Resources in Environmental Biotechnology, 2: 225-281.

Meijer, M., Jeppesen, E., van Donk, E., Moss, B., Scheffer, M., Lammens, E., Vanes, E., et al., (1994). Long-term responses to fish-stock reduction in small shallow lakes: Interpretation of five-year result of four biomanipulation cases in The Netherlands and Denmark. Hydrobiologia, 275-276: 457-466.

Melao, M.D. G. and Rocha, O. (2000). Productivity of zooplankton in a tropical oligotrophic reservoir over short periods of time. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2879-2887.

Melao, M.D.G. and Rocha, O. (2004). Life history, biomass and production of two planktonic cyclopoid copepods in a shallow subtropical reservoir. Journal of Plankton Research, 26(8): 909-923.

Melcher, J., Soracco, R.J. and Boylen, C.W. (1997). Spatial delineation of biotic and abiotic gradients in a rural New York reservoir. Journal of the American Water Resources Association, 33(1): 35-45

*Mengis, M.R.G., Wehrli, B. and Bernasconi, S. (1997). Nitrogen elimination in two deep eutrophic lakes. Limnology and Oceanography, 42(7): 1530-1543.

Menshutkin, V.V. and Klekowski, R.Z. (2001). Optimal management of the dam reservoir ecological system. Ecohydrology and Hydrobiology, 4: 435-440.

Mhlanga, L. (2001). Water quality of the Eastern basin of Lake Kariba, Zimbabwe in relation to the encroachment of *Eichhornia crassipes.* Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3595-3598.

Michelsen, K., Pedersen, L., Christoffersen, K. and Jensen, F. (1994). Ecological consequences of food partitioning for the fish population structure of a eutrophic lake. Hydrobiologia, 291: 35-45.

Mimikou, M., Hadjisavva, P.S., Kouvopoulos, Y.S. and Afrateos, H. (1991). Regional climate change impacts. Hydrological Sciences Journal, 36: 259-270.

Miranda, S.A. and Matvienko, B. (2003) Rain and groundwater as phosphorus sources of a small reservoir. Lakes and Reservoirs: Research and Management 8 (1): 27-30.

Mishra, S.R. and Saksena, D.N. (1990). Seasonal abundance of the zooplankton of waste water from the industrial complex at Birla Nagar (Gwalior), India. Acta Hydrochimica et Hydrobiologia, 18: 215-220.

Mitchell, A., and Baldwin, D.S. (1998). Effects of desiccation/oxidation on the potential for bacterially mediated P release from sediments. Limnology and Oceanography, 43: 481-487.

Mitchell, D.S., Pieterse, A.H. and Murphy, K.J. (1990). Aquatic weed problems and management in Africa. In: Pieterse, A.H. and Murphy, K.J. (1990). *Aquatic Weeds. The Ecology and Management of Nuisance Aquatic Vegetation.* Oxford University Press, Oxford, pp. 341-354.

Mittenzwey, K.H., Ullrich, S., Gitelson, A.A. and Kondratiev, K.Y. (1992). Determination of chlorophyll a of inland waters on the bsis of spectral reflectance. Limnology and Oceanography, 37: 147-149.

Moalla, S.M.N., Awadallah, R.M., Rashed, M.N. and Soltan, M.E. (1998). Distribution and chemical fractionation of some heavy metals in bottom sediments of Lake Nasser. Hydrobiologia, 364: 31-40.

*Moffett, J.W. and Brand, L.E. (1996). Production of strong, extracellular Cu chelators by marine cyanobacteria in response to Cu stress. Limnology and Oceanography, 41: 388-395.

Molot, L.A., Dillon, P.J., Clark, B.J. and Neary, B.P. (1992). Predicting end-of-summer oxygen profiles in stratified lakes. Canadian Journal of Fisheries and Aquatic Sciences, 49: 2363-2372.

Montgomery, S., Lucotte, M. and Rheault, I. (2000). Temporal and spatial influences of flooding on dissolved mercury in boreal reservoirs. Science of the Total Environment, 260: 147-157.

Moore, A.C., Herwaldt, B.L., Craun, G.F., Calderon, R.L., Highsmith, A.K. and Juranek, D.D. (1994). Waterborne disease in the United States, 1991 and 1992. Journal of the American Water Works Association, 86(2): 87-99.

Moore, M.V., Folt, C.L. and Stemberger, R.S. (1996). Consequences of elevated temperatures for zooplankton assemblages in temperate lakes. Archiv fur Hydrobiologie, 135: 289-319.

Moreau, J. (1997). *Advances in the Ecology of Lake Kariba.* University of Zimbabwe Publications, Harare.

Morgui, J.A., Quiles, R., Real, M., Riera, J.L., Lopez, P and Armengol, J. (1993). Sediment particle size patterns in Spanish reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1210-1212.

Morris, G.L. and Fann, J. (1998). *Reservoir Sedimentation Handbook. Design and Management of Dams, Reservoirs and Watersheds for Sustainable Use.* McGraw-Hill

Morrison, K. and Therien, N. (1995). Mercury release and transformation from flooded vegetation and soils: Experimental evaluation and simulation modelling. Water, Air and Soil Pollution, 80: 819-828.

*Mossman, D.J. and Mulki, N.A. (1996). One-dimensional unsteady flow and unsteady pesticide transport in a reservoir. Ecological Modelling. 89: 259-267.

*Moya, G., Bennasar, G., Frau, C., Garcia, L., Gomez, M. and Ramon, G. Long term changes (after twelve years) in the composition of phytoplankton communities in the Gorg Blau Reservoir (Majorca, Spain). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1257-1261.

Moyle, P.B. and Light, T. (1996). Fish invasions in California: Do abiotic factors determine success? Ecology, 77: 1666-1670.

*Mozeto, A.A., Silverio, P.F. and Soares, A. (2001). Estimates of benthic fluxes of nutrients across the sediment-water interface (Guarapiranga Reservoir, Sao Paulo, Brazil). Science of the Total Environment, 266: 135-142.

Muller-Navarra, D. (1995). Evidence that highly unsaturated fatty acid limits *Daphnia* growth in nature. Archiv fur Hydrobiologie, 132: 297-307.

*Muller-Navarra, D. and Lampert, W. (1996). Seasonal patterns of food limitation in *Daphnia galeata*: separating food quantity and food quality effects. Journal of Plankton Research, 18: 1137-1157.

*Munger, C., Hare, L., Craig, A. and Charest, P.M. (1999). Influence of exposure time on the distribution of cadmium with the Cladocerans *Ceriodaphnia dubia.* Aquatic Toxicology, 33: 195-200.

Munk, L. and Faure, G. (2004). Effects of pH fluctuations on potentially toxic metals in the water and sediment of the Dillon Reservoir, Summit County, Colorado. Applied Geochemistry, 19: 1065-1074.

Munro, J.L. (1966). A limnological survey of Lake McIlwaine, Rhodesia. Hydrobiologia, 28: 281-308.

Murphy, K.J., Rorslett, B. and Springuel, I. (1990). Strategy analysis of submerged lake macrophyte communities: an International example. Aquatic Botany, 36: 303-323.

Murthy, B.N. (1977). *Life of Reservoirs*. Central Board of Irrigation and Power, New Delhi.

Mwaura, F. (2003). The spatio-temporal characteristics of water transparency and temperature in shallow reservoirs in Kenya. Lakes and Reservoirs: Research and Management, 8: 259-268.

*Mwaura, F., Mavuti, K.M. and Wamicha, W.N. (2002). Biodiversity characteristics of small high-altitude tropical man-made reservoirs in the Eastern Rift Valley, Kenya. Lakes and Reservoirs: Research and Management, 7: 1-12.

Nagarajan, R. (2000). Environmental impact analysis of Dudhganga Dam in India – a multi-temporal remote sensing approach. International Journal of Remote Sensing, 21(3): 483-497.

*Naimo, T.J. (1995). A review of the effects of heavy metals in freshwater mussels. Ecotoxicology, 4: 341-362.

*Nakamoto, N., Tundisi, J.G., Marins, M.A. and Godinho, M.J.L. (1997). Longitudinal distribution of plankton in a shallow reservoir of Broa in Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 553-557.

Nalepa, T.F., Wojcik, J.A., Fanslow, D.L. and Lang, G.A. (1995). Initial colonization of the zebra mussel (Dreissena polymorpha) in Saginaw Bay, Lake Huron: Population recruitment, density and size structure. Journal of Great Lakes Research, 21: 417-434.

*Nandalal, K.D.W. and Bogardi, J.J. (1995). Optimal operation of a reservoir for quality control using inflows and outflows. Water, Science and Technology, 31(8): 273-280.

*Naselli-Flores, L. (2003). Man-made lakes in Mediterranean semi-arid climate: the strange case of Dr Deep Lake and Mr Shallow Lake. Hydrobiologia 506-509: 13-21.

Naselli-Flores, L. and Barone, R. (1994). Relationship between trophic state and plankton community structure in 21 Sicilian dam reservoirs. Hydrobiologia, 275-276: 197-205.

Naselli-Flores, L. and Barone, R. (1996). Phytoplankton and underwater light climate in a hypertrophic reservoir (Lake Arancio, Sicily). Giornale Di Botanica Italiana., 129: 1288-1292.

Naselli-Flores, L. and Barone, R. (1997). Importance of water-level fluctuation on population dynamics of cladocerans in a hypertrophic reservoir (Lake Arancio, south-west Sicily, Italy). Hydrobiologia, 360: 223-232.

Naselli-Flores, L. and Barone, R. (2000). Phytoplankton dynamics and structure: a comparative analysis in natural and man-made water bodies of different trophic state. Hydrobiologia, 438: 65-74.

*Naselli-Flores, L. and Barone, R. (2003). Steady-state assemblages in a Mediterranean hypertrophic reservoir. The role of *Microcystis* ecomorphological variability in maintaining an apparent equilibrium. Hydrobiologia, 502: 133-143.

Naselli-Flores, L., Barone, R., Mosello, R. (2003). Eutrophication control by lime addition: A preliminary approach in Sicilian reservoirs. Hydrobiologia, 504: 297-303.

Nauwerck, A. (1991). Zooplankton changes in Mondsee. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 974-979.

Navarro, L. and Phiri, G. (2000*). Water hyacinth in Africa and the Middle East. A survey of problems and solutions*. International Development Research Centre, Canada; 120pp.

*Nedoma, J.; Porcalova, P.; Komarkova, J.; Vyhnalek, V. (1993). A seasonal study of phosphorus deficiency in a eutrophic reservoir. Freshwater Biology, 30(3): 369-376.

*Nellis, M.D., Harrington, J.A. and Wu, J. (1998). Remote sensing of temporal and spatial variations in pool size, suspended sediment, turbidity and Secchi depth in Tuttle Creek Reservoir, Kansas, 1993. Geomorphology, 21: 281-293.

Nelson, S.A.C., Soranno, P.A., Cheruvelil, K.S., Batzli, S.A. and Skole, D.L. (2002). Regional assessment of lake water clarity using satellite remote sensing. Journal of Limnology, 62(Suppl. 1): 27-32.

Newman, J.R. and Barrett, P.R.F. (1993). Control of *Microcystis aeruginosa* by decomposing barley straw. Journal of Aquatic Plant Management, 31: 203-206.

*Nicholls, K.H. and Hopkins, G.J. (1993). Recent changes in Lake Erie (North Shore) phytoplankton: cumulative impacts of phosphorus loading reductions and zebra mussel introduction. Journal of Great Lakes Research, 19: 637-647.

Nicklow, J.W. and Mays, L.W. (2000). Optimization of multiple reservoir networks for sedimentation control. Journal of Hydraulic Engineering, 126(4): 232-242.

Nicklow, J.W. and Mays, L.W. (2001). Optimal control of reservoir releases to minimize sedimentation in rivers and reservoirs. Journal of the American Water Resources Association, 37(1): 197-211.

Niemi, M. and Taipalinen, I. (1982). Faecal indicator bacteria at fish farms. Hydrobiologia, 86: 171-175.

*Niimi, A.J. (1990). Review of biochemical methods ofn other indicators to assess fish health in aquatic ecosystems containing toxic chemicals. Journal of Great Lakes Research, 16: 529-541.

*Nimal Priyantha, D.G., Asaeda, T., Saitoh, S. and Gotoh, K. (1997). Modelling effects of curtain method on algal blooming in reservoirs. Ecological Modelling, 98: 89-104.

Noernberg, M.A., Novo, E.M and Krug, T. (1999). The use of biophysical indices and coefficient of variation derived from airborne synthetic aperture radar for monitoring the spread of aquatic vegetation in tropical reservoirs. International Journal of Remote Sensing, 20(1): 67-82.

Nogueira, M.G. (2000). Phytoplankton composition, dominance and abundance as indicators of environmental compartmentalization in Jurumirim Reservoir (Paranapanema River), Sao Paulo, Brazil. Hydrobiologia, 431: 115-128.

Nogueira, M.G., Henry, R. and Maricatto, F.E. (1999). Spatial and temporal heterogeneity in the Jurumirim Reservoir, Sao Paulo, Brazil. Lakes and Reservoirs: Research and Management, 4: 107-120.

*Nor, Y. (1987). Ecotoxicity of copper to aquatic biota: A review. Environmental Research, 43: 274-282.

*Northcote, T.G, Arcifa, M.S. , Munro, K.A. (1990). An experimental study of the effects of fish zooplanktivory on the phytoplankton of a Brazilian reservoir. Hydrobiologia, 194(1): 31-45.

Novo, E.M.L.M. and Tundisi, J.G. (1994). Contribution of remote sensing techniques to the assessment of methane emission from large tropical reservoirs. Remote Sensing Review, 10: 143-153.

Nowlin, W.H., Evarts, J.L. and Vanni, M.J. (2005). Release rates and potential fates of nitrogen and phosphorus from sediments in a eutrophic reservoir. Freshwater Biology, 50: 301-322.

O'Brien, W.J. (1990). Perspectives on fish in reservoir limnology. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp. 209-225.

Ochs, C.A., Cole, J.J. and Likens, G.E. (1995). Population dynamics of bacterioplankton in an oligotrophic lake. Journal of Plankton Research, 17: 365-391.

*Oertel, N. (1998). Molluscs as bioindicators of heavy metals in a side-arm of the River Danube disturbed by engineering activities. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 2120-2124.

Ogbeibu, A.E. and Oribhabor, B.J. (2002). Ecological impact of river impoundment using benthic macro-invertebrates as indicators. Water Research, 36: 2427-2436.

OgutuOhwayo, R., Hecky, R.E., Cohen, A.S. and Kaufman, L. (1997). Human impacts on the African Great Lakes. Environmental Biology of Fishes, 50: 117-131.

Oksiyuk, O.P., Davydov, O.A., Melenchuk, G.V. and Karpezo, Y.I. (2001). Peculiarities of phytoplankton of the Kiev section of the Kanev Reservoir depending on the operating condition of the Kiev hydro-electric power station. Hydrobiological Journal, 37(2): 122-135.

Oliver, R.L. (1994). Floating and sinking gas-vacuolate cyanobacteria. Journal of Phycology, 30: 161-173.

*Oliveira, H.T. (1997). Primary production in a dentritic tropical reservoir: spational and seasonal heterogeneity (Barra Bonita Reservoir, Brazil). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 569-573.

Olsen, N.R.B., Hedger, R.D. and George, D.G. (2000). 3D numerical modeling of *Microcystis* distribution in a water reservoir. Journal of Environmental Engineering, 126(10): 949-953.

Olson, E.J., Engstrom, E.S., Doringsfeld, M.R. and Bellig, D.R. (1994). Abundance and distribution of macroinvertebrates in relation to macrophyte communities in a prairie marsh, Swan Lake, Minnesota. Journal of Freshwater Ecology, 1014: 325-335.

Ooms-Wilms, A.L. (1997). Are bacteria important food source for rotifers in eutrophic lakes? Journal of Plankton Research, 19: 1125-1141.

Ooms-Wilms, A.L., Postema, G. and Gulati, R.D. (1995). Evaluation of bacterivory of Rotifera based on measurements of *in situ* ingestion of fluorescent particles, including some comparisons with Cladocera. Journal of Plankton Research, 17: 1057-1077.

Orians, G.H. (1986). Site characteristics favouring invasions. In: Mooney, H.A. and Drake, J.A. (Eds). *Ecology of Biological Invasions of North America and Hawaii.* Springer-Verlag, Berlin: pp. 133-148.

Ormerod, S.J., Weatherley, N.S., Varallo, P.V. and Whitehead, P.G. (1988). Preliminary estimates of the historical and future impact of acidification on the ecology of Welsh streams. Freshwater Biology, 20: 127-140.

Oskam, G. (1995). Main principles of water-quality improvement in reservoirs. Aqua, 44 (Suppl. 1): 23-29.

Ostojic, A.M. (2000). Effect of eutrophication on changes in the composition of zooplankton in the Grosnica Reservoir (Serbia, Yugoslavia). Hydrobiologia, 436: 171-178.

Otha, T., Tokishita, S., Shiga, Y., Hanazato, T. and Yamagata, H. (1998). An assay system for detecting environmental toxicants with cultured cladoceran eggs in vitro: malformations induced by ethylenethiourea. Environmental Research, section A, 77: 43-48.

Oudra, B., Loudiki, M., Sbiyyaa, B., Sabour, B., Martins, R., Amorim, A. and Vasconcelos, V. (2002). Detection and variation of microcystin contents of *Microcyctis* blooms in eutrophic Lalla Takerkoust Lake, Morocco. Lakes and Reservoirs: Research and Management, 7(1): 35-44.

*Pace, M.L. (1984). Zooplankton community structure but not biomass influences the phosphorus-chlorophyll a relationship. Canadian Journal of Fisheries and Aquatic Sciences, 41: 1089-1096.

Pace, M.L. (1986). An empirical analysis of zooplankton community size structure across lake trophic gradients. Limnology and Oceanography, 31: 45-55.

Pace, M.L., Porter, K.G. and Feig, Y.S. (1983). Species- and age-specific differences in bacterial resource utilization by two co-occurring cladocerans. Ecology, 64: 1145-1156.

Pace, M.L., McManus, G.B. and Findlay, S.E.G. (1990). Plankton community structure determines the fates of bacterial production in a temperate lake. Limnology and Oceanography, 35: 795-808.

Pace, M.L. and Cole, J.J. (1994). Primary and bacterial production in lakes: are they coupled over depth? Journal of Plankton Research, 16: 661-672.

*Pace, M.L. and Cole, J.J. (1994). Comparative and experimental approaches to top-down and bottom-up regulation of bacteria. Microbiology Ecology, 28: 181-193.

Pace, M.L. and Vaque, D. (1994). The importance of Daphnia in determining mortality rates of protozoans and rotifers in lakes. Limnology and Oceanography, 39: 985-996.

Pacini, N., Harper, D.M. and Mavuti, K.M. (1993). A sediment-dominated tropical impoundment: Masinga Dam, Kenya. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1275-1279.

Padisak, J. (2003). Phytoplankton. In: In O'Sullivan, P.E. and Reynolds, C.S. (Eds). *The Lakes Handbook, Volume 1. Limnology and Limnetic Ecology*. Blackwell Science Ltd., pp. 251-308.

Padisak, J., Reynolds, C.S. and Sommer, U. (Eds). (1993). *Intermediate Disturbance Hypothesis in Phytoplankton Ecology*. Dev. Hydrobiol. 81. Kluwer Academic Publishers, Dordrecht, 199pp.

Padisak, J., Barbosa, F.A.R., Borbely, G., Borics, G., Chorus, I., Espindola, E.L.G., Heinze, R., Rocha, O., Torokne. A.K. and Vasas, G. (2000). Phytoplankton composition, biodiversity and a pilot survey of toxic cyanoprokaryotes in a large cascading reservoir system (Tiete basin, Brazil). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 2734-2742.

Paller, M.H. and Gladden, J.B. (1991). Development of the fish community in a new South Carolina reservoir. American Midland Naturalist, 128: 95-114.

Palmieri, A., Shah, F. and Dinar, A. (2001). Economics of reservoir sedimentation and sustainable management of dams. Journal of Environmental Management, 61: 149-163.

Panosso, R., Carlsson, P., Kozlowsky-Suzuki, B., Azevedo, S.M.F.O. and Graneli, E. (2003). Effect of grazing by a neotropical copepod, *Notodiaptomus*, on a natural cyanobacterial assemblage and on toxic and non-toxic cyanobacterial strains. Journal of Plankton Research, 25(9): 1169-1175.

*Paolini, J. (1994). Dissolved organic carbon in a tropical blackwater reservoir, Guri Lake (Venezuela). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1291-1294.

Parejko, K. (1991). Predation by chaoborids on typical and spined *Daphnia pulex*. Freshwater Biology, 25: 211-217.

Park, J.-G. and Curtis, L.R. (1997). Mercury distribution in sediments and bioaccumulation by fish in two Oregon reservoirs: Point-source and nonpoint-source impacted systems. Archives of Environmental Contamination and Toxicology, 33; 423-429.

*Park, S., Muller-Navarra, D.C. and Goldman, C.R. (2003). Seston essential fatty acids and carbon to phosphorus ratios as predictors for *Daphnia pulex* dynamics in a large reservoir, Lake Berryessa. Hydrobiologia, 505: 171-178.

*Parkhurst, B.R. and Olem, H. (1991). Lake and Reservoir Management. Research Journal WPCF, 63(4). 543-550.

Parmenter, R.R. and Lamarra, V.A. (1991). Nutrient cycling in a freshwater marsh: The decomposition of fish and waterfowl carrion. Limnology and Oceanography, 36: 976-987.

*Pashkova, O.V. (2004). Stages and peculiarities of long-term succession of zooplankton of the pelagic zone of the Kanev Reservoir. Hydrobiological Journal, 40(2): 39-50.

Passy, S.I. and Bode, R.W. (2004). Diatom model affinity (DMA), a new index for water quality assessment. Hydrobiologia, 524: 241-251.

Patalas, K. (1990). Patterns in zooplankton distribution and their causes in North American Great Lakes. In: Tilzer, M.M. and Serruya, C. (eds). *Large Lakes Ecological Structure and Function*. Springer-Verlag, Berlin: pp. 440-458.

Patalas, K. and Salki, A. (1992). Crustacean zooplankton in Lake Winnipeg: variation in space and time as a function of lake morphology, geology and climate. Canadian Journal of Fisheries and Aquatic Sciences, 49: 1035-1059.

Paterson, M.J., Findlay, D., Beaty, K., Findlay, W., Schindler, E.U., Stainton, M. and McCullough, G. (1997). Changes in the planktonic food web of a new experimental reservoir. Canadian Journal of Fisheries and Aquatic Sciences, 54(5): 1088-1102.

Pattinson, K.R., Havel, J.E. and Rhodes, R.G. (2003). Invasibility of a reservoir to exotic *Daphnia lumholtzi*: experimental assessment of diet selection and life history responses to cyanobacteria. Freshwater Biology, 48: 233-246.

*Pattinson, V.A., Butcher, D.P., Labadz, J.C. and Shacklock, J. (1995). Water discolouration and the role of the reservoir. Physics and Chemistry of the Earth, 20(2): 175-181.

Paul, L. (2003). Nutrient elimination in pre-dams: Results of long term studies. Hydrobiologia, 504: 289-295.

Paul, L., Schruter, K. and Labahn, J. (1998) . Phosphorus elimination by longitudinal subdivision of reservoirs and lakes. Water Science and Technology, 37(2): 235-244.

Paulovits, G., Tatrai, I., Matyas, K., Korponai, J. and Kovats, N. (1998). Role of prussian carp (*Carassius auratus gibelio* Bloch) in the nutrient cycle of the Kis-Balaton Reservoir. International Review of Hydrobiology, 83: 467-470.

Pautova, V.N. and Nomokonova, V.I. (2004). On seasonal dynamics of the biomass of phytoplankton in the Volga Reservoirs. Hydrobiological Journal, 40(4): 39-45.

Paxton, B.R. (2004). Catchment-wide movement patterns and habitat utilization of freshwater fish in rivers: Implications for dam location, design and operation. A review and methods development for South Africa. *Water Research Commission Report. KV145/04*, 69pp.

Payment, P. and Franco, E. (1993). *Clostridium perfringens* and somatic coliphages as indicators of the efficiency of drinking water treatment for viruses and protozoan cysts. Applied Environmental Microbiology, 59: 2418-2424.

Payment, P., Siemiatycki, J., Richardson, L., Renaud, G., Franco, E. and Provost, M. (1997). A prospective epidemiological study of gastrointestinal health effects due to consumption of drinking water. International Journal of Environmental Health Research, 7: 5-31.

Pelegri, S.P. and Blackburn, T.H. (1996). Nitrogen cycling in lake sediments bioturbated by *Chironomus plumosus* larvae under different degrees of oxygenation. Hydrobiologia, 325: 231-238

Penuelas, J., Gamon, J.A., Griffin, K.L. and Field, C.B. (1993). Assessing community type, plant biomass, pigment composition and photosynthetic efficiency of aquatic vegetation from spectral reflectance. Remote Sensing of the Environment, 25: 1-9

*Perez-Fuentetaja, A., McQueen, D.J. and Ramcharan, C.W. (1996b). Predator-induced bottom-up effects in oligotrophic systems. Hydrobiologia, 317: 163.

*Perez-Martinez, C. and Cruz-Pizarro, L. (1993). Changes in the phytoplankton structure and biomass after experimental fertilization and zooplankton manipulations in a mesotrophic reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1232-1235.

*Perkins, R.G. and Underwood, G.J.C. (2000). Gradients of chlorophyll a and water chemistry along a eutrophic reservoir with determination of the limiting nutrients by in situ nutrient addition. Water Research, 34: 713-724.

*Perkins, R.G. and Underwood, G.J.C. (2001). The potential for phosphorus release across the sediment-water interface in a eutrophic reservoir dosed with ferric sulphate. Water Research, 35(6): 1399-1406.

Perkins, R.G. and Underwood, G.J.C. (2002). Partial recovery of a eutrophic reservoir through managed phosphorus limitation and unmanaged macrophyte growth. Hydrobiologia, 481: 75-87.

Pernie, G.L., Scavia, D., Pace, M.L. and Carrick, H.J. (1990). Micrograzer impact and substrate limitation of bacterioplankton in Lake Michigan. Canadian Journal of Fisheries and Aquatic Sciences, 47: 1836-1841.

Perry, S.A., Perry, W.B. and Simmons, G.M. (1990). Bacterioplankton and phytoplankton populations in a rapidly-flushed eutrophic reservoir. International Review of Hydrobiology, 75(1): 27-44.

Persson, A. (1997). Phosphorus release by fish in relation to external and internal load in a eutrophic lake. Limnology and Oceanography, 42(3): 577-584

Persson, A., Barkman, A. and Hansson, L.-A. (1999). Simulating the effects of biomanipulation on the food web of Lake Ringsjon. Hydrobiologia, 404: 131-144.

Persson, A. and Hansson, L.-A. (1999). Diet shift in fish following competitive release. Canadian Journal of Fisheries and Aquatic Sciences, 56: 70-78.

Peters, R.H. (1983). Size structure of the plankton community along the trophic gradient of Lake Memphremagog. Canadian Journal of Fisheries and Aquatic Sciences, 40: 1035-1059.

Peterson, C.G. and Hoagland, K.D. (1990). Effects of wind-induced turbulence and algal mat development on epilithic diatom succession in a large reservoir. Archiv fur Hydrobiologie, 118(1): 47-68.

Petridis, D. and Sinis, A. (1993). Benthic macrofauna of Tavropos reservoir (central Greece). Hydrobiologia, 262: 1-12.

*Pham, T., Proulx, S., Brochu, C. and Moore, S. (1999). Composition of PCBs and PAHs in the Montreal urban community wastewater and in the surface water of the St. Lawrence River (Canada). Water, Air and Soil Pollution, 111: 251-270.

Pichlova, R. and Brandl, Z. (2003). Predatory impact of *Leptodora kindtii* on zooplankton community in the Slapy Reservoir. Hydrobiologia, 504: 177-184.

Pieterse, A.H. and Murphy, K.J. (1990). *Aquatic Weeds. The Ecology and Management of Nuisance Aquatic Vegetation.* Oxford University Press, Oxford, 593pp.

Pieterse, A.H., Hellsten, S.K., Janauer, G.A., Dieme, C., Diouf, S. and Exler, N. (2002). Management of aquatic vegetation in the lower Senegal River basin. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 549-555.

Pijanowska, J. and Dawidowicz, P. (1987). The lack of vertical migration in *Daphnia*: the effect of homogeneously distributed food. Hydrobiologia, 148: 175-181.

Pinel-Alloul, B. (1995). Spatial heterogeneity as a multiscale characteristic of a zooplankton community. Hydrobiologia, 300-301: 17-42.

Pinel-Alloul, B., Downing, J.A., Perusse, M. and Codin-Blumer, G. (1988). Spatial heterogeneity in freshwater zooplankton, variation with body size, depth and scale. Ecology, 69: 1393-1400.

Pinel-Alloul, B. and Pont, D. (1991). Spatial distribution patterns in freshwater zooplankton: variation with scale. Canadian Journal of Zoology, 69: 1557-1570.

Pinel-Alloul, B. and Letarte, Y. (1993). Relationships between small and large bacterioplankton and primary producers in Quebec lakes. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 321-324.

*Pinto-Coelho, R.M. (1998). Effects of eutrophication on seasonal patterns of mesozooplankton in a tropical reservoir: a 4-year study in Pampulha Lake, Brazil. Freshwater Biology 40: 159-173.

*Pinto-Coelho, R.M., Amorim, M.K. and da Costa, A.R. (1997). Temporal dynamics of lipids in the zooplankton of two tropical reservoirs of different trophic status. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 584-587.

*Pinto-Coelho, R.M., Pinel-Alloul, B., Methot, G. and Havens, K.E. (2005). Crustacean zooplankton in lakes and reservoirs of temperate and tropical regions: variation with trophic status. . Canadian Journal of Fisheries and Aquatic Sciences, 62: 348-361.

*Pinto, R.M. and Greco, M.K.B. (1999). The contribution of water hyacinth (Eichhornia crassipes) and zooplankton to the internal cycling of phosphorus in the eutrophic Pampulha Reservoir, Brazil. Hydrobiologia, 411: 115-127.

Piyasiri, S. (2001). Increasing eutrophy in Kotmale Reservoir, Sri Lanka: a 5-year study. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3604-3607.

Pizzolon, L., Tracanna, B., Prosperi, C. and Guerrero, J.M. (1999). Cyanobacterial blooms in Argentinean inland waters. Lakes and Reservoirs: Research and Management, 4: 101-105.

*Plenet, S. (1995). Freshwater amphipods as bioindicators of metal pollution in surface and interstitial aquatic systems. Freshwater Biology, 33: 127-137.

Plourde, Y., Lucotte, M. and Pichet, P. (1997). Contribution of suspended particulate matter and zooplankton to MeHg contamination of the food chain in mid-northern Quebec (Canada) reservoirs. . Canadian Journal of Fisheries and Aquatic Sciences, 54: 821-831.

Pont, D. and Amrani, J. (1990). The effects of selective fish predation on the horizontal distribution of pelagic Cladocera in a southern French reservoir. Hydrobiologia, 207: 259-267.

Popp, A. and Hoagland, K D. (1995). Changes in benthic community composition in response to reservoir aging. Hydrobiologia, 306 (2): 159-171.

*Popp, A., Hoagland, K D. and Hergenrader, G.L. (1996). Zooplankton community response to reservoir ageing. Hydrobiologia, 339: 13-21.

Porcalova, P. (1990). Phosphorus losses from the epilimnion in Rimov Reservoir. International Review of Hydrobiology, 75(3): 273-279.

*Porter, K.G., Sherr, E.B., Sherr, B.F., Pace, M. and Sanders, R.W. (1985). Protozoa in planktonic food webs. Journal of Protozoology, 32: 409-415.

Porvari, P. (1998). Development of fish mercury concentrations in Finnish reservoirs from 1979 to 1994. Science of the Total Environment, 231: 279-290.

Porvari, P. and Verta, M. (1994). Methylmercury production in flooded soils: A laboratory study. Water Air and Soil Pollution, 80: 765-773.

Potter, L. Kidd, D., Standiford, D. (1975). Mercury levels in Lake Powell: bioamplification of mercury in man-made desert reservoir. Environmental Science and Technology, 9: 41-46.

*Pozdnyakov, D.V., Kondratyev, K.Y., Bukata, R.P. and Jerome, J.H. (1998). Numerical modelling of natural water colour: implications for remote sensing and limnological studies. International Journal of Remote Sensing, 19: 1913-1932

Prat, N., Real,M. and Rieradevall, M. (1992). Benthos of Spanish lakes and reservoirs. Limnetica, 8: 221-229.

Prejs, A., Martyniak, A., Boron, S., Hliwa, P. and Koperski, P. (1994). Food web manipulation in a small eutrophic Lake Wirbel, Poland: effect of stocking with juvenile pike on planktivorous fish. Hydrobiologia, 275-276: 65-70.

Presing, M., Balogh, K.V., Voros, L., and Shafik, H.M.K. (1997). Relative nitrogen deficiency without occurrence of nitrogen fixing blue-green algae in a hypertrophic reservoir. Hydrobiologia 342/343 55-61.

Priyantha, D.G., Asaeda, T., Saitoh, S. and Gotoh, K. (1997). Modelling of curtain method, a new technique for controlling algal blooming in withdrawal zone of reservoirs. Ecological Modelling, 98: 89-104.

Putz, K. (1995). The importance of pre-reservoirs for the water-quality management of reservoirs. Aqua – Journal of Water Supply Research and Technology, 44 (Suppl 1): 50-55.

Putz, K. and Benndorf, J. (1998). The importance of pre-reservoirs for the control of eutrophication of reservoirs. Water, Science and Technology, 37(2): 317-324.

*Quibell, G. (1992). Estimating chlorophyll concentrations using upwelling radiance from different freshwater algal genera. International Journal of Remote Sensing, 13: 2611-2621.

*Quiros, R. (1990). Predictors of relative fish biomass in lakes and reservoirs of Argentina. Canadian Journal of Fisheries and Aquatic Sciences, 47, 928-939.

Quiros, R. (1990). Factors related to variance of residuals in chlorophyll a-total phosphorus regressions in lakes and reservoirs of Argentina. Hydrobiologia, 200-201, 343-355.

Quiros, R. (1998). Fish effects on trophic relationships in the pelagic zone of lakes. Hydrobiologia, 361: 101-111.

Quiros. R. (1999). The relationship between fish yield and stocking density in reservoirs from tropical and temperate regions. In: Tundisi, J.G. and Straškraba, M. (eds). *Theoretical Reservoir Ecology and its Applications*. International Institute of Ecology, Brazilian Academy of Sciences and Blackhuys Publishers, Sao Carlos, SP., Brazil; pp. 67-83.

Quiros. R. and Boveri, M.B. (1999). Fish effects on reservoir trophic relationships. In: Tundisi, J.G. and Straškraba, M. (eds). *Theoretical Reservoir Ecology and its Applications*. International Institute of Ecology, Brazilian Academy of Sciences and Blackhuys Publishers, Sao Carlos, SP., Brazil; pp. 529-546.

Rahal, F.J. (2000). Homogenization of fish faunas across the United States. Science (Wash., D.C.), 288: 854-856.

Ramcharan, C.W., France, R.L. and McQueen, D.J. (1996). Multiple effect of plantivorous fish on algae through a pelagic trophic cascade. Canadian Journal of Fisheries and Aquatic Sciences, 53: 2819-2828.

*Rapport, D.J. (1992b). Evaluating Ecosystem Health. Journal of Aquatic Ecosystem Health, 1: 15-24.

Rast, W. and Thornton, J.A. (1996). Trends in eutrophication research and control. Hydrological Processes, 10: 295-313.

Ravera, O. (2001). Monitoring of the aquatic environment by species accumulator of pollutants: a review. Journal of Limnology, 60 (Suppl.1): 63-78.

*Ravera, O. and Giannoni, L. (1995). Plankton as an indicator of variation over time of the Chernobyl fall-out. Science of the Total Environment, 172: 19-125.

Real, M., Rieradevall, M. and Prat, N. (2000). *Chironomus* species (Diptera: Chironomidae) in the profundal benthos of Spanish reservoirs and lakes: factors affecting distribution patterns. Freshwater Biology, 43: 1-18

Recknagel, F., Hosomi, M., Fukushima, T. and Kong, D.-S. (1995). Short- and long-term control of external and internal phosphorus loads in lakes – A scenario analysis. Water Research, 29: 1767-1779.

Recknagel, F., French, M., Harkonen, P. and Yabunaka, K. (1997). Artificial neural network approach for modelling and prediction of algal blooms. Ecological Modelling, 96: 11-28.

*Recknagel, F., Burch, M., Jablonskas, G., Minney, J. and Schoofs, S. (1998). Combined effects of organic pollution and eutrophication in the South Para Reservoirs, South Australia. Water, Science and Technology, 37(2): 113-120.

Reeders, H.. and Bij de Vaate, A. (1990). Zebra mussels (*Dreissena polymorpha):* a new perspective for water quality management. Hydrobiologia, 200-201: 437-450.

Rehulka, J. (2002). Content of inorganic and organic pollutants in the fish from the Slezska Harta Reservoir. Czech Journal of Animal Science, 47(1); 30-44.

Reinertsen, H., Jensen, A., Koksvik, J.I., Langeland, A. and Olsen, Y. (1990). Effects of fish removal on the limnetic ecosystem of a eutrophic lake. Canadian Journal of Fisheries and Aquatic Sciences, 47: 166-173.

*Renton, P.J., Duncan, A., Kubecka, J. and Seda, J. (1995). The management implications of low fish stocks in the London reservoirs. Aqua – Journal of Water Supply Research and Technology, 44(Suppl 1): 72-79.

Reynolds, C.S. (1984). *The Ecology of Freshwater Phytoplankton*. Cambridge University Press, Cambridge.

Reynolds, C.S. (1984). Phytoplankton periodicity: the interaction of form, function and environmental stability. Freshwater Biology, 14: 111-142.

*Reynolds, C.S. (1989). Physical determinants of phytoplankton succession. In: Sommer, U. (ed). *Plankton Ecology: Succession in Plankton Communities.* Springer Verlag, Berlin; pp. 9-56.

*Reynolds, C.S. (1992). Eutrophication and the management of planktonic algae: what Vollenweider couldn't tell us. In Sutcliffe, D.W. and Jones, J.G. (Eds). *Eutrophication: Research and Application to Water Supply*. Freshwater Biological Association, Ambleside, pp 5-29.

*Reynolds, C.S. (1992). Dynamics, selection and composition of phytoplankton in relation to vertical structure in lakes. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 35: 13-31.

Reynolds, C.S. (1993). Scales of disturbance and their role in plankton ecology. Hydrobiologia, 249: 157-171.

Reynolds, C.S. (1994). The long, the short and the stalked: on the attributes of phytoplankton selected by physical mixing in lakes and rivers. Hydrobiologia, 289: 9-21.

*Reynolds, C.S. (1994). The ecological basis for the successful biomanipulation of aquatic communities. Archiv fur Hydrobiologie, 130: 1-33.

Reynolds, C.S. (1996) The plant life of the pelagic. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 97-113.

*Reynolds, C.S. (1997). Vegetation processes in the pelagic: a model for ecosystem theory. In: Kinne, O. (ed). *Excellence in Ecology series.* Ecology Institute, Oldendorf/Luhe. Germany. 371pp.

Reynolds, C.S. (1999). Phytoplankton assemblages in reservoirs. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 439-456.

Reynolds, C.S. (2001). Emergence in pelagic communities. Scientia Marina, 65 (Suppl 2): 5-30.

Reynolds, C.S. (2004). Organization and energetic partitioning of limnetic communities. In O'Sullivan, P.E. and Reynolds, C.S. (Eds). *The Lakes Handbook, Volume 1. Limnology and Limnetic Ecology*. Blackwell Science Ltd. 230-250.

Reynolds, C.S., Oliver, R.L. and Walsby, A.E. (1987). Cyanobacterial dominance: the role of buoyancy regulation in dynamic lake environments. New Zealand Journal of Marine and Freshwater Research, 21: 379-390.

Reynolds, C.S., Padisak, J., and Sommer, U. (1993). Intermediate disturbance in the ecology of phytoplankton and the maintenance of species diversity: a synthesis. Hydrobiologia, 249: 183-188.

Reynolds, C.S. and Irish, A.E. (1997). Modelling phytoplankton dynamics in lakes and reservoirs: the problem of in-situ growth rates. Hydrobiologia, 349: 5-17.

*Reynolds, C.S., Irish, A.E. and Elliott. J.A. (2001). The ecological basis for simulating phytoplankton responses to environmental change (PROTECH). Ecological Modelling, 140: 271-291.

Reynolds, C.S., Huszar, V., Kruk, C., Naselli-Flores, L. and Melo. S. (2002). Review towards a functional classification of the freshwater phytoplankton. Journal of Plankton Research, 24: 417-428.

Ricciardi, A.. and Rasmussen, J.B. (1998). Predicting the identity and impact of future biological invaders: a priority for aquatic resource management. Canadian Journal of Fisheries and Aquatic Sciences, 55: 1759-1765.

Ricciardi, A. and MacIsaac, H.J. (2000). Recent mass invasion of the North American Great Lakes by Ponto-Caspian species. Trends in Ecology and Evolution, 15: 62-65.

Richardson, D.M., Pysek, P., Rejmanek, M., Barbour, M.G., Panetta, F.D. and West, C.J. (2000). Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distribution, 6: 93-107.

Ridge, I. And Pillinger, J.M. (1996). Towards understanding the nature of algal inhibitors from barley straw. Hydrobiologia, 340: 301-305.

Riemann, B., Christoffersen, K., Jensen, H.J., Muller, J.P., Lindegaard, C. and Bosselmann, S. (1990). Ecological consequences of a manual reduction of roach and bream in a eutrophic temperate lake. Hydrobiologia, 200/201: 241-250.

Riemann, B. and Christoffersen, K. (1993). Microbial trophodynamics in temperate lakes. Marine Microbial Food Webs, 7: 69-100.

Riemer, M.D. (1988). *Introduction to Freshwater Vegetation.* Westport. CT, AVI, 207pp.

*Riera, J.L., Jaume, D., de Manuel, J., Morgui, J.A. and Armengol, J. (1992). Patterns of variation in the limnology of Spanish reservoirs: a regional study. Limnetica, 8: 111-123.

Riley, M.J. and Stefan, H.G. (1988). Development of the Minnesota Lake Quality Management Model, “MINLAKE”, Lake and Reservoir Management, 4: 73-84.

Rind, D., Goldberg, R., Hansen, J. Rosenzweig, C. and Ruedy, R. (1990). Potential evapotranspiration and the likelihood of future drought. Journal of Geophysical Review, 95: 9983-10004.

*Robarts, R.D. (1988). Heterotrophic bacterial activity and primary production in a hypertrophic African lake. Hydrobiologia, 162: 97-107.

Robarts, R.D. and Zohary, T. (1984). *Microcystis aeruginosa* and underwater light attenuation in a hypertrophic lake (Hartbeespoort Dam, South Africa). Journal of Ecology, 72: 1001-1017.

Robarts, R.D. and Zohary, T. (1992). The influence of temperature and light on the upper limit of *Microcystis aeruginosa* production in a hypertrophic reservoir. Journal of Plankton Research, 14(2): 235-247.

*Robarts, R.D., Zohary, T., Jarvis, A.C., Pais-Madeira, C.M., Sephton, L.M. and Combrink, S. (1992). Phytoplankton and zooplankton population dynamics and production of a recently formed African reservoir. Hydrobiologia, 237: 47-60.

Roberts, E., Kroker, J., Korner, S. and Nicklisch, A. (2003). The role of periphyton during the recolonization of a shallow lake with submerged macrophytes. Hydrobiologia, 506-509: 525-530.

Robertson, L.J., Campbell, A.T. and Smith, H.V. (1992). Survival of *Cryptosporidium parvum* oocysts under various environmental pressures. Applied Environmental Microbiology, 58(11): 3494-3500.

Rocha, O., Matsumura-Tundisi, T., Espindola, E.L.G., Roche, K.F. and Rietzler, A.C. (1999). Ecological theory applied to reservoir zooplankton. . In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 29-51.

Rocha, S.E.V., Matsumura-Tundisi, T. and Tundisi, J.G. (2002). Composition and abundance of zooplankton in the limnetic zone of seven reservoirs of the Paranapanema River, Brazil. Brazilian Journal of Biology, 62(3): 525-545.

Rocha-e-Silva, T.A.A., Rantin, F.T., Matsumura-Tundisi, J.E., Matsumura-Tundisi, T., Tundisi, J.G. and Degterev, I.A. (2004). Dynamics of cytochrome P450 inducers in polluted sites of Sao Paulo city reservoirs. Ecotoxicology and Environmental Safety, 59: 109-115.

*Roche, K. F., Sampaio, E. V., Teixeira, D., Matsumura-Tundisi, T., Tundisi, J. G. and Dumont, H. J. Impact of *Holoshestes heterodon* Eigenmann (Pisces: Characidae) on the plankton community of a subtropical reservoir: the importance of predation by *Chaoborus* larvae. Hydrobiologia, 254 (1); 7-20.

*Rodriguez-Ruiz, A. (1998). Fish species composition before and after construction of a reservoir on the Guadalete River (SW Spain). Archiv fur Hydrobiologie, 142: 353-369.

*Rodriguez-Ruiz, A. and Granado-Lorencio, C. (1992). Spawning period and migration of three species of cyprinids in a stream with Mediterranean regimen (SW Spain). Journal of Fish Biology, 41: 545-556.

Rogal, I.V. and Dobrynskij, V.A. (1994). Imitation-mathematical model of radionuclides migration in the water of the Dnieper Reservoir cascade. Hydrobiological Journal, 30: 90-99.

Rohrlack, T., Dittmann, E., Henning, M., Borner, T. and Kohl, J.-G. (1999). Role of microcystins in poisoning and food ingestion inhibition of *Daphia galeata* caused by cyanobacterium *Microcystis aeruginosa*. Applied Environmental Microbiology, 65: 737-739.

Romare, P., Bergman, E. and Hansson, L.-A. (1999). The impact of larvae and juvenile fish on zooplankton and algal dynamics. Limnology and Oceanography, 44: 1655-1666.

Romero, J.R. and Imberger, J. (1999). Seasonal horizontal gradients of dissolved oxygen in a temperate Austral reservoir. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 211-226.

*Romero, J.R. and Imberger, J. (2003). Effect of a flood underflow on reservoir water quality: Data and three-dimensional modelling. Archiv fur Hydrobiologie, 157(1): 1-25.

*Romero, J.R., Antenucci, J.P. and Imberger, J. (2004). One- and three-dimensional biogeochemical simulations of two differing reservoirs. Ecological Modelling, 174: 143-160.

Rommens, W., Maes, J., Dekeza, N., Inghelbrecht, P., Nhiwatiwa, T., Holsters, E., Ollevier, F., Marshall, B. and Brendonck, L. (2003). The impact of water hyacinth (*Eichhornia crassipes*) in a eutrophic subtropical impoundment (Lake Chivero, Zimbabwe). I. Water quality. Archiv fur Hydrobiologie, 158(3): 373-388

*Rorslett, B. (1991). Principal determinants of aquatic macrophyte richness in northern European lakes. Aquatic Botany, 39: 173-193.

Rorslett, B. and Johansen, S.W. (1996). Remedial measures connected with aquatic macrophytes in Norwegian regulated rivers and reservoirs. Regulated Rivers: Research and Management, 12: 509-522.

Roue, A., Richardot, M., Thouvenot, A., Debroas, D., Devauz, J and Poulin, M. (2002). Functioning of a lacustrine ecosystem in a newly flooded reservoir (Sep, Puy de Dome, France): modelling of carbon flux between the different food web compartments. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 1253-1257.

Round, F.E. (1991). Diatoms in river-monitoring studies. Journal of Applied Physiology, 3:129-145.

Rudd, J.W.M., Harris, R., Kelly, C.A. and Hecky, R.E. (1993). Are hydroelectric reservoirs significant sources of greenhouse gases? Ambio. 22(4): 246-248.

Rudd, J. (1995). Sources of methyl mercury to freshwater ecosystems: A review. Water, Air and Soil Pollution, 80: 697-713.

*Ryding, S.O. and Rast, W. (Eds) (1989). *The Control of Eutrophication of Lakes and Reservoirs.* The Parthenon Publishing Company, Park Ridge, N.J.

Saad, M.A.H. and Hemeda, E.I. (1999). Environmental characteristics of Lake Qarun, an Egyptian closed basin receiving huge amounts of brackish drainage waters from agricultural lands. *Proceedings of the 8th International Conference on the Conservation and Management of Lakes*, 17-21 May, 1999, Copenhagen. S1-2.

*Sabri, A.W. (1990). Local and seasonal variation of the epipelic algae in Samarra Impoundment, Iraq. Limnologica, 21(1): 275-279.

Sahoo, G.B. and Luketina, D. (2003). Modelling of bubble plume design and oxygen transfer for reservoir restoration. Water Research, 37: 393-401.

St. Louis, V.L., Kelly, A.C., Duchemin, E., Rudd, J.W.M. and Rosenberg, D.M. (2000). Reservoir surfaces of greenhouse gases to the atmosphere: a global estimate. Bioscience, 50: 766-775.

*Saito, L., Johnson, B.M., Bartholow, J. and Hanna, R.B. (2001). Assessing ecosystem effects of reservoir operations using food web – energy transfer and water quality models. Ecosystems, 4: 105-125.

*Saker, M.L. and Eaglesham, G.K. (1999). The accumulation of cylindrospermopsin from the cyanobacterium *Cylindrospermopsis raciborskii* in tissues of the redclaw crayfish *Cherax quadricarinatus*. Toxicon, 37: 1065-1077.

*Sakevich, A.I. and Usenko, O.M. (2003). Phenol compounds in the water of the Dnieper Reservoirs. Hydrobiological Journal, 39(5): 110-119.

Salas, J.D. and Shin, H.-S. (1999). Uncertainty analysis of reservoir sedimentation. Journal of Hydraulic Engineering, 125: 339-350.

Sale, M.J., Brill, E.D. and Herricks, E.E. (1982). An approach to optimizing reservoir operation for downstream aquatic resources. Water Resources Research, 18: 705-712.

Salencon, M.-J. and Thebault, J.-M. (1996). Simulation model of a mesotrophic reservoir (Lac de Pareloup, France): MELODIA, an ecosystem reservoir management model. Ecological Modelling, 84: 163-187.

Salvia-Castellvi, M., Dohet, A., Vander Borght, P. and Hoffman, L. (2001). Control of the eutrophication of the reservoir of Esch-sur-Sure (Luxembourg): evaluation of the phosphorus removal by predams. Hydrobiologia, 459: 61-71.

Sander, B.C. and Kalff, J. (1993). Factors controlling bacterial production in marine and freshwater sediments. Microbiology Ecology, 26: 79-99.

Sanders, R.W., Porter, K.G., Bennett, S.J., and DeBiase, A.E. (1989). Seasonal patterns of bacterivory by flagellates, ciliates, rotifers and cladocerans in a freshwater planktonic community. Limnology and Oceanography, 34: 673-687.

Sanders, R.W., Caron, D.A. and Berninger, U.G. (1992). Relationship between bacteria and heterotrophic nanoplanktonic marine and fresh waters: an inter-ecosystem comparison. Marine Ecology Progress Series, 86: 1-4.

Sanford, E. (1999). Regulation of keystone predation by small changes in ocean temperature. Science, 283, 2095-2097.

Sanni, S. and Waevagen, S.B. (1990). Oligotrophication as a result of planktivorous fish removal with rotenone in the small eutrophic Lake Mosvatn, Norway. Hydrobiologia, 200-201: 263-274.

*Santos, G.B., Maia-Barbosa, P.M., Vieira, F. and de Lopez, C.M. (1994). In: Pinto-Coelho, R.M., Giani, A. and von Sperling, E. (Eds). *Ecology and Human Impact on Lakes and Reservoirs in Minas Gerais with Special Reference to Future Development and Management Strategies.* SEGRAC: Belo Horizonte (MG): pp. 115-132.

*Santos-Wisniewski, M.J., Rocha, O., Guntzel, A.M. and Matsumura-Tundisi, T. (2002). Diversity of Cladocera Chydoridae in lakes and reservoirs from the high portion of Tiete River, SP, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 881-886.

Sarnelle, O. (1992). Nutrient enrichment and grazer effects on phytoplankton in lakes. Ecology, 73: 551-560.

Sarvala, J., Helminen, H., Saarikari, V., Salonen, S and Vuorio. (1998). Relations between planktivorous fish abundance, zooplankton and phytoplankton in three lakes of differing productivity. Hydrobiologia, 363: 81-95.

Saunders, D.L. and Kalff, J. (2001). Nitrogen retention in wetlands, lakes and rivers. Hydrobiologia, 443: 305-312.

Savage, A.A. and Beaumont, D.L. (1997). A comparison of the benthic macroinvertebrate communities of a lowland lake, Oak Mere, in 1980 and 1994. Archiv fur Hydrobiologie, 139: 197-206.

Scancar, J., Milacic, R. and Horvat, M. (2000). Comparison of various digestion and extraction procedures in analysis of heavy metals in sediments. Water, Air and Soil Pollution, 118: 87-99.

Schael, D.M., Rice, J.A. and Degan, D.J. (1995). Spatial and temporal distribution of threadfin shad in a south-eastern reservoir. Transactions of the American Fisheries Society, 124: 804-812.

Schallenberg, M. and Kalff, J. (1993). The ecology of sediment bacteria in lakes and comparisons with other aquatic ecosystems. Ecology, 74: 919-934.

Schanz, F., Borer, A. and Kiefer, B. (2001). Factors influencing the time scale of vertical phytoplankton distribution in a mountain lake (Lake Waegital, Switzerland). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3766-3769.

Scharf, W. (2002). Refilling, ageing and water quality management of Brucher Reservoir. Lakes and Reservoirs: Research and Management, 7: 13-23.

Schaus, M.H., Vanni, M.J., Wissing, T.E., Bremigan, M.T., Garvey, J.E., Stein, R.A. (1997). Nitrogen and phosphorus excretion by detritivorous gizzard shad in a reservoir ecosystem. Limnology and Oceanography, 42(6): 1386-1397.

Schetagne, R., Doyon, J.-F. and Fournier, J.-J. (2000). Export of mercury downstream from reservoirs. Science of the Total Environment, 260: 135-145.

*Schindler, D.E. (1992). Nutrient regeneration by sockeye salmon (*Oncorhynchus nerka*) fry and subsequent effects on zooplankton and phytoplankton. Canadian Journal of Fisheries and Aquatic Sciences, 49: 2498-2506.

Schindler, D.E., Kitchell, J.F., He, X., Carpenter, S.R., Hodgson, J.R. and Cottingham, K.L. (1993). Food web structure and phosphorus cycling in lakes. Transactions of the American Fisheries Society, 122: 756-722.

Schintu, M., Meloni, P. and Contu, A. (2000). Aluminium fractions in drinking water from reservoirs. Ecotoxicology and Environmental Safety, 46: 29-33.

Schladow, S.G. (1993). Lake destratification by bubble-plume systems: design methodology. Journal of Hydraulic Engineering, 119(3): 350-368.

Schladow, S.G. and Hamilton, D.P. (1995). Effect of major flow diversion on sediment nutrient release in a stratified reservoir. Marine and Freshwater Research, 46: 189-195.

Schladow, S.G. and Hamilton, D.P. (1997). Prediction of water quality in lakes and reservoirs: Part II – Model calibration sensitivity analysis and application. Ecological Modelling, 96: 111-123.

Schmidt-Araya, J.M. and Zuniga, L.R. (1992). Zooplankton community structure in two Chilean reservoirs. Archiv fur Hydrobiologie, 123(3): 305-335.

Schram, M.D. and Marzolf, G.R. (1993). Seasonal variation of crustacean zooplankton in Kentucky Lake, USA. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1158-1161.

Schriver, P., Bogestrand, J., Jeppesen, E. and Sondergaard, M. (1995). Impact of submerged macrophytes on fish-zooplankton-phytoplankton interactions: large scale enclosure experiments in a shallow eutrophic lake. Freshwater Biology, 33: 255-270.

*Schwaiger, J., Wanke, R., Adam, S., Pawert, M., Honnen, W. and Triebskorn, R. (1997). The use of histopathological indicators to evaluate contaminant-related stress in fish. Journal of Aquatic Ecosystem Stress and Recovery, 6: 75-86.

Seda, J. (1994). The flow of the plankton community along a canyon-shaped reservoir. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 40: 97-106.

Seda, J. and Duncan, A. (1994). Low fish predation pressure in London reservoirs: II. Consequences to zooplankton community structure. Hydrobiologia, 291: 179-191.

Seda, J and Kubecka, J. (1997). Long term biomanipulation of the Rimov Reservoir, Czech Republic. Hydrobiologia, 345: 95-108.

Seda, J. and Machacek, J. (1998). The effect of flow-through regimes on zooplankton densities in a canyon-shaped dam reservoir. International Review of Hydrobiology, 83: 477-484.

Seda, J. and Devetter, M. (2000). Zooplankton community structure along a trophic gradient in a canyon-shaped dam reservoir. Journal of Plankton Research, 22(10): 1829-1840.

*Senogles, P., Shaw, G. and Smith, M. (2000). Degradation of the cyanobacterial toxin cylindrospermopsin, from *Cylindrospermopsis raciborskii*, by chlorination. Toxicon, 38: 1203-1213.

Shannon, J.P., Blinn, D.W., McKinney, T., Benenati, E.P., Wilson, K.P. and O'Brien, C. (2001). Aquatic food base response to the 1996 test flood below Glen Canyon Dam, Colorado River, Arizona. Ecological Applications, 11(3): 672-685.

Shapiro, J. (1995). Lake restoration by biomanipulation – a personal view. Environmental Review, 3: 83-93.

Shcherban, E.P., Konovets, I.N. and Arsan, O.M. (2001). Assessment of cuproxyl toxicity by the method of bioassay using Cladocera. Hydrobiologia, 37(6): 15-23.

Shehata, S.A., El-Dib, M.A. and Abou Waly, H.F. (1997). Effect of certain herbicides on the growth of freshwater algae. Water, Air and Soil Pollution, 100: 1-12.

Shen, H.W. (1999). Flushing sediment through reservoirs. Journal of Hydraulic Research, 36(6): 743-757.

Sherman, B.S., Webster, I.T., Jones, G.J. and Oliver, R.L. (1998). Transitions between *Aulocoseira* and *Anabaena* dominance in a turbid river weir pool. Limnology and Oceanography, 43(8): 1902-1915.

*Sherr, E.B. and Sherr, B.F. (1988). Role of microbes in pelagic food webs. Limnology and Oceanography, 33: 1225-1227.

Sherr, E.B., Sherr, B.F., Berman, T. and Hadas, O. (1991). High abundance of picoplankton-ingesting ciliates during late fall in Lake Kinneret, Israel. Journal of Plankton Research, 13: 789-799.

Sherry, J.P. (2003). The role of biomarkers in the health assessment of aquatic ecosystems. Aquatic Ecosystem Health and Management, 6(4):423-440.

Shevchenko, T.F. (1995). Accumulation of ^{137}Cs by communities of periphyton algae in the Dniepr reservoirs. Hydrobiological Journal, 29: 84-91.

Shurin, J.B. and Dodson, S.I. (1997). Sublethal toxic effects of cyanobacteria and nonylphenol on environmental sex determination and development in *Daphnia*. Environmental Toxicology and Chemistry, 16(6): 1269-1276.

*Silva, E.I.L. and Davies, D.W. (1987). The seasonality of monsoonal primary productivity in Sri Lanka. Hydrobiologia, 150: 165-175.

Silva, O.F., Novo, E.M. and Krug, T. (1997). Total suspended solid distribution in time and space derived from in situ and TM/Landsat data. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26, 388-391.

Silva, W.M. and Matsumura-Tundisi, T. (2002). Distribution and abundance of Cyclopoida populations in a cascade of reservoirs of the Tiete River (Sao Paulo State, Brazil). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 667-670.

Simek, K., Macek, M., Seda, J. and Vyhnalek, V. (1990). Possible food chain relationships between bacterioplankton, protozoans and cladocerans in a reservoir. International Review of Hydrobiology, 75(5): 583-596.

Simek, K. and Chrzanowski, T.H. (1992). Direct and indirect evidence of size-selective grazing on pelagic bacteria by freshwater nanoflagellates. Applied and Environmental Microbiology, 58: 3715-3720.

Simek, K. and Straskrabova, V. (1992). Bacterioplankton production and protozoan bacterivory in a mesotrophic reservoir. Journal of Plankton Research, 14: 773-787.

Simek, K., Vrba, J. and Hartman, P. (1994). Size-selective feeding by *Cyclidium* sp on bacterioplankton and various sizes of cultured bacteria. FEMS Microbiology Ecology, 14: 157-168.

Simek, K., Bobkova, J., Macek, M., Nedoma, J. and Psenner, R. (1995). Ciliate grazing on picoplankton in a eutrophic reservoir during the summer phytoplankton maximum: a study at the species and community level. Limnology and Oceanography, Vol. 40: 1077-1090.

Simek, K., Hartman, P., Nedoma, J., Pernthaler. J., Vrba, J., Springman, D. and Psenner, R. (1997). Community structure, picoplankton grazing and zooplankton control of heterotrophic nanoflagellates in a eutrophic reservoir during the summer phytoplankton maximum. Aquatic Microbial Ecology, 12: 49-63.

Simek, K., Armengol, J., Comerma, M., Garcia, J.C., Chrzanowski, T.H., Macek, M., Nedoma, J. and Straskrabova, V. (1998). Characteristics of protistan control of bacterial production in three reservoirs of different trophy. International Review of Hydrobiology, 83: 485-494.

Simek, K., Armengol, J., Comerma, M., Garcia, J.C., Chrzanowski, T.H., Kojecka, P., Macek, M., Nedoma, J. and Straskrabova, V. (1999a). Impacts of protistan grazing on bacterial dynamics and composition in reservoirs of different trophy. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 267-282.

Simek, K., Kojecka, P., Nedoma, J., Hartman, P., Vrba, J. and Dolan, J.R. (1999b). Shifts in bacterial community composition associated with different microzooplankton size fractions in a eutrophic reservoir. Limnology and Oceanography, 44(7): 1634-1644.

*Simek, K., Jurgens, K., Nedoma, J., Comerma, M. and Armengol, J. (2000). Ecological role and bacterial grazing of *Halteria* spp.: small freshwater oligotrichs as dominant pelagic ciliate bacterivores. Aquatic Microbial Ecology, 22: 43-56.

Simek, K., Armengol, J., Comerma, M., Garcia, J.C., Kojecka, P., Nedoma, J. and Hejzlar, J. (2001a). Changes in the epilimnetic bacterial community composition, production and protists-induced mortality along the longitudinal axis of a highly eutrophic reservoir. Microbiology Ecology, 42: 359-371.

Simek, K., Pernthaler, J., Weinbauer, M.G., Hornak, K., Dolan, J.R., Nedoma, J., Masin, N. and Amann, R. (2001b). Changes in bacterial community composition and dynamics and viral mortality rates associated with enhanced flagellate grazing in a mesotrophic reservoir. Applied Environmental Microbiology, 67: 2723-2733.

Sime-Ngando, T. and Hartmann, H.J. (1991). Short-term variations of the abundance and biomass of planktonic ciliates in a eutrophic lake. European Journal of Protistology, 27: 249-263.

Simmons, J. (1998). Algal control and destratification at Hanningfield Reservoir. Water Science and Technology, 37(2): 309-316.

Simonian, A., Tatrai, I., Biro, P., Paulovits, G. amd Toth, L.G. (1995). Biomass of planktonic crustaceans and the food of young cyprinids in the littoral zone of Lake Balaton. Hydrobiologia, 303: 39-48.

*Singhai, S., Ramani, G.M.A. and Gupta, U.S. (1990). Seasonal variations and relationships of different physico-chemical characteristics in newly made Tawa Reservoir. Limnologica, 21(1): 293-301.

Sirenko, L.A., Kostikova, L.Y., Shevchenko, T.F. and Kostikov, I.Y. (1992). Change of *Microcystis aeruginosa* populations of the Dnieper Reservoirs after the Chernobyl nuclear accident. Hydrobiological Journal, 28: 40-45.

*Sirenko, L.A., Kureyshevich, A.V. and Medved, V.A. (2000). Peculiarities of the development of phytoplankton in the upper and lower reaches of the regulated river (on the example of the Dnieper River). Hydrobiological Journal, 36(4): 3-14.

Sivonen, K (2004). Toxic cyanobacteria research, pp123-131. In Eloranta, P (ed.) Inland and coastal waters of Finland. (Commemorative volume for XXIX Congress of SIL, Finland.) University of Helsinki/Palmenia Centre for Continuing Education; Saarijärvi.

Skelton, P.H. (2002). An overview of the challenges of conserving freshwater fishes in South Africa. In: Collares-Pereira, M.J., Cowx, I.G. and Coelho, M.M. (Eds). *Conservation of freshwater fishes: options for the future.* Blackwell Science, 21, pp 21-236.

Sladeckova, A. (1990). Periphyton as indicator of the reservoir water quality. III. Biomonitoring techniques. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 775-782.

*Sladeckova, A. (1998). Technological and hygienic consequences of reservoirs eutrophication in the Czech Republic. Water, Science and Technology, 37: 195-202.

*Smakulska, J. and Gorniak, A. (2004). Morphological variation in *Daphnia cucullata* Sars with progressive eutrophication of a polymictic lowland reservoir. Hydrobiologia, 526: 119-127.

Smart, R.M., Doyle, R.D. and Madsen, J.D. (1996). Establishing native submersed aquatic plant communities for fish habitat. American Fisheries Society Symposium, 16: 347-356.

Smith, C.S., Lonsdale, W.M. and Fortune, J. (1999). When to ignore advice: invasion predictions and decision theory. Biological Invasions, 1: 89-96

*Smol, J.P. (1992): Palaeolimnology: An important tool for effective ecosystem management. Journal of Aquatic Ecosystem Health, 1: 49-58.

*Somlyody, L. (1998). Eutrophication modelling, management and decision making: the Kis-Balaton case. Water, Science and Technology, 37: 165-175.

Sommer, U. (1993). Phytoplankton competition in Plusssee: A field test of the resource-ratio hypothesis. Limnology and Oceanography, 38: 838-846.

Sommer, U., Padisak, J., Reynolds, C.S. and Juhasz-Nagy, P. (1993). Hutchinson's heritage: the diversity-disturbance relationships in phytoplankton. Hydrobiologia, 249: 1-7.

Sommaruga, R. and Psenner, R. (1993). Nanociliates of the order Prostomatidae: Their relevance in the microbial food web of a mesotrophic lake. Aquatic Sciences, 55: 179-187.

Sommaruga, R. and Robarts, R.D. (1997). The significance of autotrophic and heterotrophic picoplankton in hypereutrophic ecosystems. FEMS Microbiology Ecology, 24: 187-200.

Sondergaard, M., Jeppesen, E., Mortensen, E., Dall, E., Kristensen, P. and Sortjaer, O. (1990). Phytoplankton biomass reduction after planktivorous fish reduction in a shallow eutrophic lake: a combined effort of reduced internal P-loading and increased zooplankton grazing. Hydrobiologia, 200/201: 229-240

Sondergaard, M., Jeppesen, E. and Berg, S. (1997). Pike (*Esox lucius* L.) stocking as a biomanipulation tool. 2. Effects on lower trophic levels in Lake Lyng (Denmark). Hydrobiologia, 342-343: 319-325.

*Sondergaard, M., Jensen, J.P. and Jeppesen, E. (2003). Role of sediment and internal loading of phosphorus in shallow lakes. Hydrobiologia, 506-509: 135-145.

Soranno, P.A., Carpenter, S.R. and Elser, M.M. (1993). Zooplankton community dynamics. In: Carpenter, S.R. and Kitchell, J.F. (eds). *The Trophic Cascade in Lakes* Cambridge University Press, Cambridge; pp. 116-152.

Sorenson, K.H. and Sterner, R.W. (1992). Extreme cyclomorphosis in *Daphnia lumholtzi*. Freshwater Biology, 28: 257-262.

Soyupak, S., Mukhallalati, L., Yemisen, D., Bayar, A. and Yurteri, C. (1997). Evaluation of eutrophication control strategies for the Keban Dam reservoir. Ecological Modelling, 97: 99-110.

Speece, R.E. et al. (1982). Hypolimnetic oxygenation studies in Clark Hill Lake. ASCE Journal of Hydraulics Engineering Division, 108: 225-244.

Speece, R.E. (1994). Lateral thinking solves stratification problems. Water Quality International, 3: 12-15.

Spry, D. and Wiener, J.G. (1991). Metal bioavailability and toxicity to fish in low-alkalinity lakes: A critical review. Environmental Pollution, 71: 243-304.

*Starling, F.L.R.M. and Lazzaro, X. (1997). Experimental investigation of the feasibility of improving water quality by controlling exotic planktivore overpopulation in the eutrophic Paranoa Reservoir (Brasilia, Brazil). Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 789-794.

Stauffer, R.E. (1987). Effects of oxygen transport on the areal hypolimnetic oxygen deficit. Water Resources Research, 23: 1887-1892.

*Steel, J.A. (1997). Scope and limitation in algal modelling – an example from the Thames Valley reservoirs. Hydrobiologia, 349: 27-37.

*Steffensen, D., Burch, M., Nicholson, B., Drikas, M. and Baker, P. (1999). Management of toxic blue-green algae (Cyanobacteria) in Australia. Environmental Toxicology, 14(1): 183-195.

*Stein, R.A., DeVries, D.R. and Dettmers, J.M. (1995). Food-web regulation by a planktovore: exploring the generality of the trophic cascade hypothesis. Canadian Journal of Fisheries and Aquatic Sciences, 52: 2518-2526.

*Stein, L.Y., Jones, G., Alexander, B., Elmund, K., Wright-Jones, C. and Nealson, K.H. (2002). Intriguing microbial diversity associated with metal-rich particles from a freshwater reservoir. FEMS Microbiology Ecology, 42: 431-440.

Stelzer, P.C. (1998). Population growth in planktonic rotifers. Does temperature shift the competitive advantage for different species? Hydrobiologia, 387/388: 349-353.

*Stephen, D., Moss, B. and Phillips, G. (1997). Do rooted macrophytes increase sediment phosphorus release? Hydrobiologia, 342-343:27-34.

*Sterner, R.W. (1989). The role of grazers in phytoplankton succession. In: Sommer, U. (ed). *Plankton Ecology. Succession in Plankton Communities.* Springer-Verlag, Berlin; pp. 107-171.

*Sterner, R.W., Hagemeier, D.D., Smith, W.L. and Smith, R.F. (1993). Phytoplankton nutrient limitation and food quality for *Daphnia*. Limnology and Oceanography, 38: 857-871.

Sterner, R.W. and Hessen, D.O. (1994). Algal nutrient limitation and the nutrition of aquatic herbivores. Annual Review of Ecology and Systematics, 25: 1-29.

Sterner, R.W., Schampel, J.H., Schulz, K.L., Galford, E.E. and Elser, J.J. (2000). Joint variation of zooplankton and seston stoichiometry in lakes and reservoirs. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3009-3014.

Stephens, R. and Imberger, J. (1993). Reservoir destratification via mechanical mixers. Journal of Hydraulic Engineering. 119(4): 438-457.

*Steynberg, M.C., Adam, K. and Pieterse, A.J.H. (1998). An algal monitoring protocol: the strategic link between reservoir and treatment process. Water, Science and Technology, 37(2): 153-159.

Stibor, H. (1992). Predator induced life-history shifts in a freshwater cladoceran. Oecologia, 92: 162-165.

Stich, H.B. and Lampert, W. (1981). Predator evasion as an explanation of diurnal vertical migration by zooplankton. Nature, 293: 396-398.

Stoddard, J.L., Jeffries, D.S., Lukeville, A., Clair, T.A. and Dillon, P.J. (1999). Regional trends in aquatic recovery from acidification in North America and Europe. Nature, 401: 575.

Straile, D. and Geller, W. (1998). The response of *Daphnia* to changes in trophic status and weather patterns: a case study from Lake Constance. ICES Journal of Marine Science, 55: 775-782.

Straškraba, M. (1990). Limnological particularities of mulyiple reservoir series. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 677-678.

Straškraba, M, (1994). Ecotechnological models for reservoir water quality management. Ecological Modelling, 74: 1-38.

Straškraba, M. (1995). Models of algal blooms. *Harmonizing Human Life with Lakes, 6th International Conference on the Conservation and Management of Lakes – Kasamigaura '95.* International Lake Environment Committee; pp. 838-842.

Straškraba, M. (1996). Lake and reservoir management. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 193-209.

Straškraba, M. (1998). Linmological differences between deep valley reservoirs and deep lakes. International Review of Hydrobiology, 83: 1-12.

Straškraba, M. (1999). Retention time as a key variable of reservoir limnology. In: Tundisi, J.G. and Straškraba, M. (Eds). *Theoretical Reservoir Ecology and its Applications.* International Institute of Ecology, Brazilian Academy of Science, Backhuys Publishers: pp. 385-410.

Straškraba, M. and Gnauck, A. (1985). *Freshwater ecosystems. Modelling and Simulation.* Elsevier, Amsterdam.

Straškraba, M., Tundisi, J.G. and Duncan, A. (1993a). State-of-the-art of reservoir limnology and water quality management. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (Eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 213-288.

*Straškraba, M., Blazka, P. and Brandl, Z. (1993b). Framework for investigation and evaluation of reservoir quality in Czechoslovakia. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds).

Comparative Reservoir Limnology and Water Quality Management. Kluwer Academic Publishers, Dordrecht; pp. 169-212.

Straškraba, M. and Tundisi, J.G. (1999). *Reservoir water quality management*. International Lake Environment Committee Foundation, Shiga, Japan.

Straškraba, M. and Hocking, G. (2002). The effect of theoretical retention time on the hydrodynamics of deep river valley reservoirs. International Review of Hydrobiology, 87: 61-83.

Straskrabova, V. (1976). Self-purification of impoundments. Water Research, 9: 1171-1177.

Straskrabova, V. and Simek, K. (1993). Microbial loop in lakes and reservoirs related to trophy and metazooplankton development. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1183-1186.

*Straskrabova, V., Kennedy, R.H., Lind, O.T., Tundisi, J.G. and Hejzlar, J. (Eds) (2003). *Reservoir Limnology and Water Quality*. Kluwer Academic Publishers.

Strayer, D.L. (1999). Effects of alien species on freshwater mollusks in North America. Journal of the North American Benthological Society, 18: 74-98.

Strojsova, A., Vrba, J., Nedoma, J., Komarkova, J. and Znachor, P. (2003). Seasonal study of extracellular phophatase expression in the phytoplankton of a eutrophic reservoir. European Journal of Phycology, 38: 295-306.

*Sullivan, A.B., Jager, H.I. and Myers, R. (2003). Modelling white sturgeon movement in a reservoir: the effect of water quality and sturgeon density. Ecological Modelling, 167: 97-114.

Svensson, J.M. and Leonardson, L. (1996). Effects of bioturbation by tube-dwelling chironomid larvae on oxygen uptake and denitrification in eutrophic lake sediments. Freshwater Biology, 35: 289-300.

Svobodova, Z., Dusek, L., Hejtmanek, M., Vykusova, B. and Smid, R. (1999). Bioaccumulation of mercury in various fish species from Orlik and Kamyk water reservoirs in the Czech Republic. Ecotoxicology and Environmental Safety, 43: 231-240.

Swierzovski, A., Godlewska, M. and Poltorak, T. (2000). The relationship between the spatial distribution of fish, zooplankton and other environmental parameters in the Solina Reservoir, Poland. Aquatic Living Resources, 13: 373-377.

Szito, A. and Botos, M. (1993). Macrozoobenthos in the shallow Hungarian Kiskore Reservoir on the River Tisza. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1196-1199.

*Ta, C.T. and Brignal, W.J. (1998). Application of computational fluid dynamics technique to storage reservoir studies. Water, Science and Technology, 37(2): 219-226.

Taft, J.L., Loftus, M.E. and Taylor, W.R. (1977). Phosphate uptake from phosphomonoesters by phytoplankton in the Chesapeake Bay. Limnology and Oceanography, 22: 1012-1021.

Talling, J.F. (1990). Diel and seasonal energy transfer, storage and stratification in African reservoirs and lakes. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 33: 651-660.

*Tanik, A., Baykal, B.B. and Gonenc, I.E. (1999). The impact of agricultural pollutants in six drinking water reservoirs. Water, Science and Technology, 40(2): 11-17.

Tarela, P.A. and Menendez, A.N. (1999). A model to predict reservoir sedimentation. Lakes and Reservoirs: Research and Management, 4: 121-133.

*Tatrai, I., Toth, G.L., Istvanovics, V. and Zlinszky, J. (1990). Interactions between different trophic levels in enclosures in Lake Balaton. Hydrobiologia, 191 ; 307-313.

Tatrai, I., Toth, G., Ponyi, J.E., Zlinszky, J. and Istvanovics, V. (1990). Bottom-up effects of bream (*Abramis brama* L.) in Lake Balaton. Hydrobiologia, 200-201: 167-175.

Tatrai, I., Paulovits, G., Matyas, K., Korponai, J. and Pomogyi, P. (1998). Long-term food web interactions in Kis-Balaton Reservoir. International Review of Hydrobiology, 83: 515-522.

Tatrai, I., Matyas, K., Korponai, J., Paulovits, G., Pomogyi, P. and Heri, J. (2003). Regulation of plankton by omnivore cyprinids in a shallow lake in the Kis-Balaton Reservoir System. Hydrobiologia, 504: 241-250.

*Tavares de Lima, I.B., Novo, E.M.L.M., Ballester, M.V.R. and Ometto, J.P. (2000). Role of the macrophyte community in CH4 production and emission in the tropical reservoir of Tucurui, Para State, Brazil. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 1437-1440.

Tavcar, V. (1993). Chironomid (Diptera) larvae of the Butoniga Reservoir, Istria, Croatia, and its feeder rivers. Hydrobiologia, 262: 89-96.

Taylor, B.E. and Gabriel, W. (1992). To grow or not to grow: optimal resource allocation for *Daphnia.* American Naturalist, 139: 248-266.

Taylor, J.C., Harding, W.R., Archibald, C.G.M. and van Rensburg, L. (2005). Diatoms as indicators of water quality in the Jukskei-Crocodile river system in 1956 and 1957, a re-analysis of diatom count data generated by BJ Cholnoky. Water SA, 31(2): 1-10.

*Temporetti, P.F., Alonso, M.F., Baffico, G., Diaz, M.M., Lopez, W., Pedrozo, F.L. and Vigliano, P.H. (2001). Trophic state, fish community and intensive production of salmonids in Alicura Reservoir (Patagonia, Argentina). Lakes and Reservoirs: Research and Management, 6: 259-267.

*Tetteh, I.K., Frempong, E. and Awuah, E. (2004). An analysis of the environmental health impact of the Barakese Dam in Kumasi, Ghana. Journal of Environmental Management, 72: 189-194.

*Thebault, J.-M. (2004). Simulation of a mesotrophic reservoir (Lake Pareloup) over a long period (1983-1998) using ASTER2000 biological model. Water Research, 38: 393-403.

*Theiss, J., Zielinski, K. and Lang, H. (1990). Biomanipulation by introduction of herbivorous zooplankton. A helpful shock for eutrophic lakes? Hydrobiologia, 200-201: 59-68.

Therriault, T. W. and Schneider, D.C. (1998). Predicting change in fish mercury concentrations following reservoir impoundment. Environmental Pollution: 101: 33-42.

Thiemann, S. and Kaufmann, H. (2000). Determination of chlorophyll content and trophic state of lakes using field spectrometer and IRS-IC satellite data in the Mecklenburg Lake District, Germany. Remote Sensing of Environment, 73: 227-235.

Thingstad, T.F. and Lignell, R. (1997). Theoretical models for the control of bacteria growth rate, abundance, diversity and carbon demand. Aquatic Microbial Ecology, 13: 19-27.

*Thomas, G. (1993). Nutrient removal processes in freshwater submerged macrophyte systems. Ecological Engineering, 2: 1-30.

Thomas, S., Cecchi, P., Corbin, D., Lemoalle, J. (2000). The different primary producers in a small African tropical reservoir during a drought: temporal changes and interactions. Freshwater Biology, 45: 43-56.

Thomaz, S.M., Souza, D.C. and Bini, L.M. (2003). Species richness and beta diversity of aquatic macrophytes in a large subtropical reservoir (Itaipu Reservoir, Brazil): the influence of limnology and morphometry. Hydrobiologia, 505: 119-128.

Thompson, J.D. (1991). The biology of an invasive plant. Bioscience, 41: 393-401.

*Thornton, J.A. (1987). Aspects of eutrophication management in tropical-subtropical regions: a review. Journal of the Limnological Society of South Africa, 13: 25-43.

*Thornton, J.A. and Rast, W. (1993). A test of hypotheses relating to the comparative limnology and assessment of eutrophication in semi-arid man-made lakes. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (Eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp. 1-24.

Thornton, K.W. (1990a). Perspectives on reservoir limnology. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp.1-13.

Thornton, K.W. (1990b). Sedimentary processes. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (Eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp.43-69.

Thouvenot, A., Debroas, D., Richardot, M. and Devaux, J. (1999a). Impact of natural metazooplankton assemblage on planktonic microbial communities in a newly flooded reservoir. Journal of Plankton Research, 21(1): 179-199.

Thouvenot, A, Richardot, M., Debroas, D. and Devaux, J. (1999b). Bacterivory of metazooplankton, ciliates and flagellates in a newly flooded reservoir. Journal of Plankton Research, 21: 1659-1679.

*Thouvenot, A, Richardot, M., Debroas, D. and Devaux, J. (2003). Regulation of the abundance and growth of ciliates in a newly-flooded reservoir. Archiv fur Hydrobiologie, 157(2): 185-193.

Threlkeld, S.T. (1979). The midsummer dynamics of two *Daphnia* species in Wintergreen Lake, Michigan. Ecology, 60: 165-179.

Thurman, R., Faulkner, B., Veal, B., Cramer, G. and Meiklejohn, M. (1998). Water quality in rural Australia. Journal of Applied Microbiology, 84(4): 627-632.

Thys, I., Dohet, A., Cauchie, H.M. and Hoffmann, L. (1998). Multivariate analysis of the longitudinal variation of the zooplankton community in a riverine reservoir. International Review of Hydrobiology, 83: 191-198.

Tietjen, T.E. and Wetzel, R.G. (2003). Seasonal and spatial distribution of bacterial biomass and the percentage of viable cells in a reservoir of Alabama. Journal of Plankton Research, 25(12): 1521-1534.

*Timchenko, V.M. and Dubnyak, S.S. (2001). Ecological aspects of the hydrological regime of the Kiev section of the Kanev Reservoir. Hydrobiological Journal, 37(5): 67-78.

Timchenko, V. and Oksiyuk, O. (2002). Ecosystem condition and water quality control at impounded sections of rivers by the regulated hydrological regime. Ecohydrology and Hydrobiology, 1-4: 259-264.

Timms, R.M. and Moss, B. (1984). Prevention of growth of potentially dense phytoplankton populations by zooplankton grazing, in the presence of zooplanktivorous fish, in a shallow wetland ecosystem. Limnology and Oceanography, 29: 472-486.

*Timotius, K.H. and Goltenboth, F. (Eds) (1995). *Tropical Limnology, Vol. II.* Satya Wacana Christian University, Salatiga, Indonesia.

Ting, D.-S. and Appan, A. (1996). General characteristics and fractions of phosphorus in aquatic sediments of two tropical reservoirs. Water, Science and Technology, 34(7-8): 53-59.

Tisler, T. and Zagorc-Koncan, J. (1999). A simple flow-through system for toxicity testing with daphnids. Water, Air and Soil Pollution, 111: 327-336.

Tkalitch, P.V., Chen, W. and Tay, J.H. (1996). Modelling of heavy metals in a reservoir with diffusive bottom layer. Water, Science and Technology, 34(7-8): 117-123.

*Toja, J. (1990). Longitudinal differentiation according to environmental factors and phytoplankton in Aracena and La Minilla reservoirs (Sevilla, Spain). Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 733-747.

*Toja, J., Basanta, A. and Fernandez Ales, R. (1992). Factors controlling algal biomass in the complex of water supply reservoirs of Seville (Spain). Limnetica, 8: 267-277.

*Tolk, B.L., Han, L. and Rundquist, D.C. (2000). The impact of bottom brightness on spectral reflectance of suspended sediments. International Journal of Remote Sensing, 21: 2259-2268.

*Tomaszek, J.A., Gardner, W.S. and Johengen, T.H. (1997). Denitrification of sediments of Lake Erie coastal wetland (Old Woman Creek, Huron, Ohio, USA). Journal of Great Lakes Research, 23: 403-415.

*Tomaszek, J.A. and Czerwieniec, E. (1998). Sedimentation, sediment oxygen demand and eutrophication in the Rzeszow Reservoir. International Review of Hydrobiology, 83: 261-268.

Tomaszek, J.A. and Czerwieniec, E. (2000). In situ chamber denitrification measurements in reservoir sediments: an example from southeast Poland. Ecological Engineering, 16: 61-71.

Tomaszek, J.A. and Koszelnik, P. (2003). A simple model of nitrogen retention in reservoirs. Hydrobiologia, 504: 51-58.

Tomaszek, J.A. and Czerwieniec, E. (2003). Denitrification and oxygen consumption in bottom sediments: factors influencing rates of processes. Hydrobiologia 504: 59-65.

Torsvik, V. and Ovreas, L. (2002). Microbial diversity and function in soils: from genes to ecosystems. Current Opinion in Microbiology, 5, 240-245.

*Town, R.M. and Filella, M. (2000). Dispelling the myths: is the existence of L1 and L2 ligands necessary to explain metal ion speciation in natural waters? Limnology and Oceanography, 45: 1341-1357.]

Townsend, S.A. (1995). Metalimnetic and hypolimnetic deoxygenation in an Australian tropical reservoir of low trophic status. In: Timotius, K.H. and Goltenboth, F. (Eds). *Tropical Limnology. Vol II. Tropical Lakes and Reservoirs*. Salatiga, Indonesia; pp. 255-264.

*Townsend, S.A. (1998). The influence of retention time and wind exposure on stratification and mixing in two tropical Australian reservoirs. Archiv fur Hydrobiologie, 141: 353-371.

Townsend, S.A. (1999). The seasonal pattern of dissolved oxygen, and hypolimnetic deoxygenation, in two tropical Australian reservoirs. Lakes and Reservoirs: Research and Management, 4: 41-53.

Townsend, S.A., Luong-van, J.T. and Boland, K.T. (1996). Retention time as a primary determinant of colour and light attenuation in two tropical Australian reservoirs. Freshwater Biology, 36: 57-69.

*Townsend, S.A., Boland, K.T. and Luong-van, J.T (1997). Wet and dry season heat loss in two tropical Australian reservoirs. Archiv fur Hydrobiologie, 139: 51-68.

Townsend, S.A. and Luong-van, J.T. (1998). Phytoplankton biomass and composition in Manton River Reservoir, a mesotrophic impoundment in the Australian wet/dry tropics. International Review of Hydrobiology, 83: 113-120.

Tremblay, A., Lucotte, M., Rheault, I. (1996). Methylmercury in a benthic food web of two hydroelectric reservoirs and a natural lake. Water, Air and Soil Pollution, 91: 255-269.

Tremblay, A. and Lucotte, M. (1997). Accumulation of total mercury and methylmercury in insect larvae of hydroelectric reservoirs. Canadian Journal of Fisheries and Aquatic Sciences, 54: 832-841.

Tremblay, A., Lucotte, M. and Schetagne, R. (1998a). Total mercury and methylmercury accumulation in zooplankton of hydroelectric reservoirs in northern Quebec (Canada). Science of the Total Environment, 213: 307-315.

Tremblay, A., Cloutier, L. and Lucotte, M. (1998b). Total mercury and methylmercury fluxes via emerging insects in recently flooded hydroelectric reservoirs and a natural lake. Science of the Total Environment, 219: 209-221.

*Tsaplina, Y.N. (2003). Functioning of communities of submerged plants within the river section of the Kanev Reservoir. Hydrobiological Journal, 39(1): 76-86.

*Tsujimura, S. and Okubo, T. (2003). Development of *Anabaena* blooms in a small reservoir with dense sediment akinete population, with special reference to temperture and irradiance. Journal of Plankton Research, 25(9): 1059-1067.

*Tufford, D.L. and McKellar, H.N. (1999). Spatial and temporal hydrodynamic and water quality modelling analysis of a large reservoir on the South Carolina (USA) coastal plain. Ecological Modelling, 114: 137-173.

Tundisi, J.G. (1994). Tropical South America: present and perspectives. In: Margalef, R. (Ed). *Limnology Now: A paradigm of Planetary Problems.* Elsevier, Amsterdam.

Tundisi, J.G., Matsumura-Tundisi, T., Calijuri, M.C. and Novo, E.M.L. (1991). Comparative limnology of five reservoirs in the middle Tiete River, S. Paulo State. . Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 1489-1496.

*Tundisi, J.G., Matsumura-Tundisi, T. and Calijuri, M.C. (1993). Limnology and management of reservoirs in Brazil. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp.25-55.

Tundisi, J.G., Bicudo, C.E.M. and Matsumura-Tundisi, T. (Eds). (1995). *Limnology in Brazil.* Brazilian Academy of Sciences, Rio de Janeiro, Brazil.

Tundisi, J G; Matsumura-Tundisi, T. (1990). Limnology and eutrophication of Barra Bonita Reservoir, Sao Paulo State, Southern Brazil. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 53: 661-676.

Tundisi, J.G., Rocha, O., Matsumura-Tundisi, T. and Braga, B. (1998). Reservoir Management in South America. Water Resources Development, 14(2): 141-155.

*Tundisi, J G; Matsumura-Tundisi, T. 2003. Integration of research and management in optimizing multiple uses of reservoirs: the experience in South America and Brazilian case studies. Hydrobiologia, 500(1-3): 231-242.

Turner, J.T. (2002). Zooplankton faecal pellets, marine snow and sinking phytoplankton blooms. Aquatic Microbial Ecology, 27: 57-102

*Uchmanski, J., Szeligiewicz, W. and Loga, M. (1993). Modelling of physical, chemical and biological processes in Polish lakes and reservoirs. In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp.99-118..

Uhlmann, D. (1998). Reservoirs and ecosystems. International Review of Hydrobiology, 83: 13-20.

*Uhlmann, D., Hupfer, M. and Paul, L. (1995). Longitudinal gradients in the chemical and microbial composition of the bottom sediment in a channel reservoir (Saidenbach R., Saxony). International Review of Hydrobiology, 80: 15-25.

*Uhlmann, D., Hupfer, M. and Appelt, C. (1997). Composition of sediments in drinking water reservoirs as a basis for the assessment of potential changes in water quality. Aqua, 46(2): 84-94.

Uhlmann, D., Appelt, C., Ulrich, K.U. and Paul, L. (2000). Microbial and chemical composition of the upper sediment layers in a man-made lake. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 866-870.

*Uhlmann, D., Maassen, S. and Roske, I. (2002). The chemical structure of the sediment-water interface in reservoirs as a reflection of microbial activity and trophic conditions. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 628-631.

UNEP (1999). *Planning and Management of Lakes and Reservoirs: An Integrated Approach to Eutrophication.*. Technical Publications Series 11.

UNESCO/WHO/UNEP (1992). Water Quality Assessments – *A Guide to the Use of Biota, Sediments and Water in Environmental Monitoring*, 2nd Ed.

Unlu, K. and Demirekler, E. (2000). Modelling water quality impacts of petroleum contaminated soils in a reservoir catchment. . Water, Air and Soil Pollution, 120: 169-193.

*Urabe, J. (1989). Relative importance of temporal and spatial heterogeneity in zooplankton community of an artificial reservoir. Hydrobiologia, 184: 1-6.

Urabe, J. (1989). Relative importance of temporal and spatial heterogeneity in zooplankton community of an artificial reservoir. Hydrobiologia, 184: 1-6.

Urabe, J. (1990). Stable horizontal variation in the zooplankton community structure of a reservoir maintained by predation and competition. Limnology and Oceanography, 35(8): 1703-1717.

*Urabe, J., Clasen, J. and Sterner, R.W. (1997). Phosphorus limitation of Daphnia growth: is it real? Limnology and Oceanography, 42: 1436-1443.

US Army Corps of Engineers (1977). *Sedimentation investigations of rivers and reservoirs.* Engineering Manual No. 1110-2-4000, Washington D.C.

USEPA (2003). Update: National listing of fish and wildlife advisories. US Environmental Protection Agency Fact Sheet, EPA-823-F-03-003.

Van Breeman, L.W.C.A.., Ketelaars, H. A. M.; Hoogenboezem, W.; Medema, G. (1998). Storage reservoirs - a first barrier for pathogenic micro-organisms in the Netherlands. Water Science and Technology, 37(2): 253-260.

*Van de Bund, W.J. and Davids, C. (1993). Complex relations between freshwater macro- and meiobenthos: interactions between *Chironomus riparius* and *Chydorus piger*. Freshwater Biology, 29: 1-6.

*Van den Berg, C., van den Bogaart, J.G.M., Sibbing, F.A. and Osse, J.W.M. (1994b). Zooplankton feeding in common bream (*Abramis brama*), white bream (*Blicca bjoerkna*) and roach (*Rutilus rutilus*): experiments, models and energy intake. Netherlands Journal of Zoology, 44: 15-42.

Van den Berg, M.S., Coops, H., Noordhuis, R., van Schie, J. and Simons, J. (1997). Macroinvertebrate communities in relation to submerged vegetation in two *Chara* dominated lakes. Hydrobiologia, 342-343: 143-150.

Van der Molen, D.T., Los, F.J., van Ballegooijen, L. and van der Vat, M.P. (1994). Mathematical modeling as a tool for management in eutrophication control of shallow lakes. Hydrobiologia, 275-276: 479-492.

van Dijk, H.F.G. and Guicherit, R. (1999). Atmospheric dispersion of current-use pesticides: a review of the evidence from monitoring studies. Water, Air and Soil Pollution, 115: 21-70.

*Van Donk, E., Gulati, R.D., Iedema, A. and Meulemans, J.T. (1993). Macrophyte-related shifts in the nitrogen and phosphorus contents of the different trophic levels in a biomanipulated shallow lake. Hydrobiologia, 251: 19-26

*Van Donk, E. and Gulati, R.D. (1995). Transition of a lake to turbid state six years after biomanipulation: mechanisms and pathways. Water, Science and Technology, 32: 197-206.

Van Ginkel, C.E., Silberbouer, M.J. and Vermaak, E. (2000). The seasonal and spatial distribution of cyanobacteria in South African surface waters. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 871-878.

Van Ginkel, C.E., Hohls, B.C. and Vermaak, E. (2001). A *Ceratium hirundinella* (O.F.Muller) bloom in Hartbeespoort Dam, South Africa. Water SA, 27(2): 269-276.

*Vanni. M.J. and Findlay, D.L. (1990). Trophic cascades and phytoplankton community structure. Ecology, 7: 921-937.

Vanni. M.J., Luecke, C., Kitchell, J.F., Magnuson, J.J. (1990). Effects of planktivorous fish mortality on the plankton community of Lake Mendota, Wisconsin: implications for biomanipulation. Hydrobiologia, 200/201: 329-336.

*Vanni. M.J. and Layne, C.D. (1997). Nutrient recycling and herbivory as mechanisms in the “top-down” effect of fish on the phytoplankton in lakes. Ecology, 78: 21-40.

Van Schalkwyk, D.J. and Walmsley, R.D. (1984). Prediction of surface water temperatures of South African impoundments. Journal of the Limnological Society of South Africa, 10: 57-61.

*Vaque, D. and Pace, M.L. (1992). Grazing on bacteria by flagellates and cladocerans in lakes of contrasting food-web structure. Journal of Plankton Research, 14: 307-321.

*Varis, O. (1992). Impacts of growth factors on competitive ability of blue-green algae analysed with whole-lake simulation. . In: Straškraba, M., Tundisi, J.G. and Duncan, A. (eds). *Comparative Reservoir Limnology and Water Quality Management.* Kluwer Academic Publishers, Dordrecht; pp.127-137.

*Vasconcelos, V. (1990). Seasonal fluctuation in the zooplankton community of Azibo Reservoir (Portugal). Hydrobiologia, 196: 183-191.

*Vasconcelos, V. (1991). Species composition and dynamics of the phytoplankton in a recently-commissioned reservoir (Azibo – Portugal). Archiv fur Hydrobiologie, 121(1): 67-78.

*Vasconcelos, V. (1994). Seasonal fluctuation of planktonic rotifers in Azibo Reservoir (Portugal). Hydrobiologia, 294: 177-184.

Vasek, M., Kubecka, J. and Seda, J. (2003). Cyprinid predation on zooplankton along the longitudinal profile of a canyon-shaped reservoir. Archiv fur Hydrobiologie, 156(4): 535-550.

Vegas-Vilarrubia, T. (1995). Distribution of the phytoplankton from the Guri Reservoir (Venezuela). Hydrobiologia, 310(1): 33-46.

Veinott, G., Anderson, M.R., Sylvester, P.J. and Gani, D.O. (2001). Metal concentrations in bivalves living in and around copper mine tailings released after a tailings dam breach. Bulletin of Environmental Contamination and Toxicology, 67: 282-287.

Velimirov, B. (1991). Detritus and the concept of non-predator loss. Archiv fur Hydrobiologie, 121: 1-20.

Venkiteswaran, J.J. and Schiff, S.L. (2005). Methane oxidation: isotopic enrichment factors in freshwater boreal reservoirs. Applied Geochemistry, 20: 683-690.

Venter, S.N., Steynberg, M.C., de Wet, C.M.E., Hohls, D., du Plessis, G. and Kfir, R. (1997). A situational analysis of microbial water quality in a peri-urban catchment in South Africa. Water, Science and Technology, 35(11-12): 119-124.

*Verdon, R., Brouard, D., Demers, C., Lalumiere, R., Laperle, M. and Schetagne, R. (1991). Mercury evolution (1978-1988) in fishes of the La Grande hydroelectric complex, Quebec, Canada. Water, Air and Soil Pollution, 1991, 405-417.

*Victor, R. and Victor, J.R. (2002). Zooplankton of an arid zone mountain reservoir in the Sultanate of Oman, Arabia. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 507-512.

Vieira, J.M., Azevedo, M.T., Azevedo, S.M.F., Honda, R.Y. and Correa, B. (2003). Microsystin production by *Radiocystis fernandoi* (Chroococcales, Cyanobacteria) isolated from a drinking water reservoir in the city of Belem, PA, Brazilian Amazonia region. Toxicon, 42: 709-713.

Vijverberg, J., Boersma, M., van Densen, W.L.T., Hoogenboezem, W., Lammens, E.H.R.R. and Mooij, W.M. (1990). Seasonal variations in the interactions between piscivorous fish, planktovorous fish and zooplankton in a shallow eutrophic lake. Hydrobiologia, 207: 279-286.

Vijverberg, J. and Boersma, M. (1997). Long-term dynamics of small-bodied and large-bodied cladocerans during the eutrophication of a shallow reservoir, with special attention for *Chydorus sphaericus*. Hydrobiologia, 360: 233-242.

*Vikhristyuk, L.A. (1996). Hydrochemical state of the Kuibyshev Reservoir. Water Resources Development, 12(4): 547-559.

*Vila, I., Contreras, M. and Pizzaro, J. (1997). Eutrophication and phytoplankton selective responses in a temperate reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 26: 798-802.

*Vilarrubia, T.V., Ibanez, A.M. and Riehl, W. (1994). Aspects of the phytoplankton of the Guri Reservoir, Venezuela, and the importance of its relationship with water chemistry. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1295-1300.

Vinot, I. And Pihan, J.C. (2005). Circulation of copper in the biotic compartments of a freshwater dammed reservoir. Environmental Pollution, 133: 169-182.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. and Melillo, J.M. (1997). Human domination of Earth's ecosystems. Science (Wash., D.C.). 277: 494-499.

*Visser, P.M., Ibelings, B.W., van der Veer, B., Koedood, J. and Mur, L.R. (1996a). Artificial mixing prevents nuisance blooms of the cyanobacterium *Microcystis* in Lake Nieuwe Meer, the Netherlands. Freshwater Biology, 36: 435.

*Visser, P.M., Ketelaars, H.A.M., Vanbreemen, L.W.C. and Mur, L.R. (1996b). Diurnal buoyancy changes of *Microcystis* in an artificially mixed storage reservoir. Hydrobiologia, 331: 131.

*Voigt, H. and Hulsmann, S. (2001). Do fast increasing food conditions promote the midsummer decline of *Daphnia galeata* ? Hydrobiologia, 442: 253-259.

Voros, L., Balogh, K.V., Koncz, E. and Kovacs, A. (2003). Phytoplankton and bacterioplankton production in a reed-covered water body. Aquatic Botany, 77: 99-110.

Vorosmarty, C.J., Sharma, K.P., Fekete, B.M., Copeland, A.H., Holden, J., Marble, J. and Lough, J.A. (1997). The storage and ageing of continental runoff in large reservoir systems of the world. Ambio, 26(4): 210-219.

Vuille, T. (1991). Abundance, standing crop and production of microcrustacean populations (Cladocera, Copepoda) in the littoral zone of Lake Biel, Switzerland. Archiv fur Hydrobiologie, 123: 165-185.

*Vyhnalek, V. Seda, J. and Nedoma, J. (1993). Fate of the spring phytoplankton bloom in Rimov Reservoir (Czechoslovakia): grazing, lysis and sedimentation. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 25: 1192-1195.

*Wagner, I. and Zalewski, M. (2000). Effect of hydrological patterns of tributaries on biotic processes in a lowland reservoir – consequences for restoration. Ecological Engineering, 16: 79-90.

Walker, F.R. and Stedinger, J.R. (1999). Fate and transport model of *Cryptosporidium.* Journal of Environmental Engineering, 125(4): 325-333.

*Walter, M., Recknagel, F., Carpenter, C. and Bormans, M. (2001). Predicting eutrophication effects in the Burrinjuck Reservoir (Australia) by means of the deterministic model SALMO and the recurrent neural network model ANNA. Ecological Modelling, 146: 97-113.

*Wang, N. and Eckmann, R. (1994). Distribution of perch (*Perca fluviatilis* L.) during their first year of life in Lake Constance. Hydrobiologia, 277: 135-143.

Wang, S.X., Zhang, S.Z. and Shan, X.Q. (2003). Fractionation of heavy metals in different particle-size sediments and its relationship with heavy metal pollution. Bulletin of Environmental Contamination and Toxicology, 71: 873-880.

Wang, X., Parkpian, P., Fujimoto, N., Ruchirawat, K.M., DeLaune, R.D. and Jugsujinda, A. (2002). Environmental conditions associating microcystins production to *Microcystis aeruginosa* in a reservoir of Thailand. Journal of Environmental Science and Health: Part A, 37(7): 1181-1207.

*Wang, X. T., Chu, S.G. and Xu, X.B. (2003). Organochlorine pesticide residues in water from Guanting Reservoir and Yongding River, China. Bulletin of Environmental Contamination and Toxicology. 70: 351-358.

*Wang, Y., Xia, H., Fu, J. and Sheng, G. (2004). Water quality changes in reservoirs of Shenzhen, China: detection using LANDSAT/TM data. Science of the Total Environment, 328: 195-206.

Watanabe, Y. (1992). Effects of thermal stratification on trophic linkages among plankton communities in eutrophic lakes. Archiv fur Hydrobiologie Beiheft Ergebnisse der Limnologie, 35: 1-12.

Watts, C.J. (2000a). Seasonal phosphorus release from exposed re-inundated littoral sediments of two Australian reservoirs. Hydrobiologia, 431: 27-39

Watts, C.J. (2000b). The effect of organic matter on sedimentary phosphorus release in an Australian reservoir. Hydrobiologia, 431: 13-25.

Wcislo, R. amd Chrost, R.J. (2000). Survival of *Escherichia coli* in fresh water. Polish Journal of Environmental Studies, 9(3): 215-222.

*Weiss, P., Schweitzer, B., Amann, R. and Simon, M. (1996). Identification *in situ* and dynamics of bacteria on limnetic organic aggregates. Applied Environmental Microbiology, 62: 1998-2005.

Welsh, W.T. and Burns, J.C. (1987). The Loch Dee Project: Run-off and surface water quality in an area subject to acid precipitation and afforestation in SW Scotland. Transactions of the Royal Society of Edinburgh. Earth Sciences, 78: 249-260.

Werner, R.G., Jonckheere, B.V., Clapsadl, M.D. and Farrell, J.M. (1996). A bioenergetic exploration of piscivory and planktivory during the early life history of two species of freshwater fishes. Marine and Freshwater Research, 47: 113-121.

Wetzel, R.G. (1990a). Land-water interfaces: metabolic and limnological regulators. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 24: 6-24.

*Wetzel, R.G. (1990b). Reservoir Ecosystems: Conclusions and Speculations. In: Thornton, K.W., Kimmel, B.L. and Payne, F.E. (eds). *Reservoir Limnology: Ecological Perspectives.* John Wiley and Sons, New York; pp 27-239.

Wetzel, R.G. (1995). Death, detritus and energy flow in aquatic ecosystems. Freshwater Biology. 33: 83-89.

Wetzel, R.G. (2001): *Limnology: Lake and River ecosystems*. Academic Press, San Diego; pp. 1-1006.

*Whipple, W. (1993). Buffer zones around water-supply reservoirs. Journal of Water Resources Planning and Management, 119(4): 495-500.

White, P.A., Kalff, J., Rasmussen, J.B. and Gasol, J.M. (1991). The effect of temperature and algal biomass on bacterial production and specific growth rates in freshwater and marine habitats. Microbiology Ecology, 21, 99-118.

Whiteside, M.C. and Hatch, J.T. (1997). Quantitative sampling techniques for age-0 fish from diverse lake habitats. Archiv fur Hydrobiologie Special Issue Advances in Limnology, 49: 99-116.

*Wiechula, D., Loska, K. and Korus, I. (2005). Lead partitioning in the bottom sediment of Rybnik Reservoir (southern Poland). Water, Air and Soil Pollution, 164: 315-327.

*Wik, S.J. (1995). Reservoir drawdown: case study in flow changes to potentially improve fisheries. Journal of Energy Engineering, 121(2): 89-96.

Wikner, J. and Hagstrom, A. (1991). Annual study of bacterioplankton community dynamics. Limnology and Oceanography, 36: 1313-1324.

*Wilk-Wozniak, E. (2003). Phytoplankton formation reflecting variation of trophy in dam reservoirs. Ecohydrology and Hydrobiology, 3(2): 213-219.

Williams, W.D. (1987). Salinisation of rivers and streams: An important environmental hazard. Ambio, 16: 180-185.

Williams, W.D. (1999). Salinisation: A major threat to water resources in the arid and semi-arid regions of the world. Lakes and Reservoirs: Research and Management, 4: 85-91.

Williams, W.D. (2000). Dryland Lakes. Lakes and Reservoirs: Research and Management, 5: 207-212.

Williamson, M. (1996). *Biological Invasion.* Chapman and Hall, London.

Williamson, M. (1999). Invasions. Ecogeography, 2: 5-12.

Williamson, M. and Fitter, A. (1996). The varying success of invaders. Ecology, 77: 1661-1666.

Windolf, J., Jeppesen E., Jensen, J.P. and Kristensen, P. (1996). Modelling of seasonal variation in nitrogen and in-lake concentration: a four years mass balance study in 16 shallow Danish lakes. Biogeochemistry, 33: 25-44.

Wobus, A., Bleul, C., Maassen, S., Scheerer, C., Schuppler, M., Jacobs, E. and Roske, I. (2003). Microbial diversity and functional characterization of sediments from reservoirs of different trophic state. FEMS Microbiology Ecology, 46: 331-347.

Wojtal, A., Frankiewicz, P., Izydorczyk, K. and Zalewski, M. (2003). Horizontal migration of zooplankton in a littoral zone of the lowland Sulejow Reservoir (Central Poland). Hydrobiologia 506-509: 339-346.

Wong, C.K. and Pak., A.P. (2004). Acute and subchronic toxicity of the heavy metals copper, chromium, nickel and zinc, individually and in mixture, to the freshwater copepod *Mesocyclops pehpeiensis.* Bulletin of Environmental Contamination and Toxicology, 73: 190-196.

Work, K.A. and Gophen, M. (1999). Factors which affect the abundance of an invasive cladoceran, *Daphnia lumholtzi*, in U.S. reservoirs. Freshwater Biology, 42: 1-10.

Worth, D.F. (1994). Gradient changes in water quality during low flows in run-of-the-river and reservoir impoundments, Lower Snake River, Idaho. Lake and Reservoir Management, 11: 217-224.

Wright , D.I. and Shapiro, J. (1990). Refuge availability: a key to understanding the summer disappearance of *Daphnia*. Freshwater Biology, 24: 43-62.

Wu, L. and Culver, D.A. (1994). *Daphnia* population dynamics in Western Lake Erie: Regulation by food limitation and yellow perch predation. Journal of Great Lakes Research, 20: 537-545.

Wu, R.-S., Liu, W.-C. and Hsieh, W.-H. (2004). Eutrophicication modelling in Shihmen Reservoir, Taiwan. Journal of Environmental Science and Health, A39(6): 1455-1477.

Wylie, J.L. and Currie, D.J. (1991). The relative importance of bacteria and algae as food sources for crustacean zooplankton. Limnology and Oceanography, 36: 708-728.

Xu, B. and Enfors, S.O. (1996). Influence of nitrate starvation on nitrite accumulation during denitrification by *Pseudomonas stutzeri*. Applied Microbiology and Biotechnology, 45(1-2): 229-235.

*Xue, H. and Sigg, L. (1993). Free cupric ion concentration and Cu(II) speciation in a eutrophic lake. Limnology and Oceanography, 38: 1200-1213.

Yacobi, Y.Z., Gitelson, A. and Mayo, M. (1995). Remote sensing of chlorophyll in Lake Kinneret using high-spectral-resolution radiometer and Landsat TM: spectral features of reflectance and algorithm development. Journal of Plankton Research, 17: 2155-2173.

Yamada, M., Ono, Y. and Somiya, I. (1998). Accumulation of freshwater red tide in a dam reservoir. Water Science and Technology, 37(2): 211-218.

Yan, N.D. and Pawson, T.W. (1997). Changes in the crustacean zooplankton community of Harp Lake, Canada, following invasion by *Bythotrephes cederstroemi* . Freshwater Biology, 37: 409-425.

Yang, M.-D., Merry, C.J. and Sykes, R.M. (1999). Integration of water quality modelling, remote sensing and GIS. Journal of the American Water Resources Association, 35(2): 253-263

Yang, X., Li, S. and Zhang, S. (2003). The sedimentation and dredging of Guanting Reservoir. International Journal of Sediment Research, 18(2): 130-127.

*Yarmoshenko, L.P. (2004). Influence of the intra-daily regime of the operation of the Kiev hydro-electric power station on the structure of microphytobenthos of the Kiev section of the Kanev Reservoir. Hydrobiological Journal, 40(2): 30-38.

Yu, W.-S., Lee, H.-Y. and Hsu, S.M. (2000). Experiments on deposition behaviour of fine sediment in a reservoir. Journal of Hydraulic Engineering, 126(12): 912-920.

Yu, N. and Culver, D.A. (1999). Estimating the effective clearance rate and refiltration by zebra mussels, *Dreissena polymorpha*, in a stratified reservoir. Freshwater Biology, 41: 481- 492.

Yu, N. and Culver, D.A. (2000). Can zebra mussels change stratification patterns in a small reservoir? Hydrobiologia, 431: 175-184.

*Yurista, P.M., Rice, G.T. and White, D.S. (2000). Long-term establishment of *Daphnia lumholtzi* Sars in Kentucky Lake, USA. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3102-3106.

*Yusoff, F.M. and Lock, M.A. (1995). Thermal stratification and its role in controlling eutrophication in a tropical reservoir, Malaysia. In: Timotius, K.H. and Goltenboth, F. (eds). *Tropical Limnology. Vol II. Tropical Lakes and Reservoirs.* Salatiga, Indonesia; pp. 277-285.

*Yusoff, F.M., Happey-Wood, C.M. and Anton. A. (1998). Vertical and seasonal distribution of phytoplankton in a tropical reservoir, Malaysia. International Review of Hydrobiology, 83: 121-134.

Yusoff, F.M., Matias, H.M. and Khan, N. (2002). Changes of water quality, chlorophyll a and zooplankton along the river-lacustrine continuum in a tropical reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 28: 295-298.

Zakova, Z. and Kockova, E. (1999). Biomonitoring and assessment of heavy metal contamination of streams and reservoirs in the Dyje/Thaya River basin, Czech Republic. Water, Science and Technology, 39(12): 225-232.

*Zafar, A.R. (1986). Seasonality of phytoplankton in some South Indian lakes. Hydrobiologia, 138: 177-187.

Zalewski, M., Brewinska-Zaras, B. and Frankiewicz, P. (1990). Fry communities as a biomanipulation tool in a temperate lowland reservoir. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie, 33: 763-774.

*Zalewski, M., Wagner-Lotkowska, I. And Tarczynska, M. (2000). Ecohydrological approaches to the elimination of toxic algal blooms in a lowland reservoir. Verhandlungen Internationale Vereinigung fur Theoretische und Angewandte Limnologie, 27: 3176-3183.

Zanata, L.H. and Espindola, E.L.G. (2002). Longitudinal processes in Salto Grande Reservoir (Americana, SP, Brazil) and its influence in the formation of compartment system. Brazilian Journal of Biology, 62(2): 347-361.

Zettler, E.R. and Carter, J.C.H. (1986). Zooplankton community and species responses to a natural turbidity gradient in Lake Temiskaming, Ontario-Quebec. Canadian Journal of Fisheries and Aquatic Sciences, 43: 665-673.

Zhang, S.Z., Wang, S.X. and Shan, X.Q. (2002). Distribution and speciation of heavy metals in surface sediments from Guanting Reservoir, Beijing. Journal of Environmental Science and Health, A37: 465-478.

Zhou, Z. (1993). Remarks on reservoir sedimentation in China. In: Fan, S.S. and Morris, G. (Eds). *Notes on Sediment Management in Reservoirs: National and International Perspectives.* Federal Energy Regulatory Commission, Washington DC, pp. 153-160.

Ziegler, C.K. and Nisbet, B.S. (1995). Long-term simulation of fine-grained sediment transport in large reservoir. Journal of Hydraulic Engineering, 121(11): 773-781

*Zlinszky, J. and Herodek, S. (1990). Biologically available phosphorus retention by the Kis-Balaton reservoir. Archiv fur Hydrobiologie Beiheft Ergebnisse Limnologie 33: 703-707.

Zohary, T. and Robarts, R.D. (1989). Diurnal mixed layers and the long-term dominance of *Microcystis aeruginosa.* Journal of Plankton Research, 11, 25-48.

*Zoumis, T., Schmidt, A., Grigorova, L. and Calmano, W. (2001). Contaminants in sediments: remobilisation and demobilization. Science of the Total Environment, 266: 195-202.

*Zubkova, Y.I. (2000). Monitoring of trace elements in the Kuchurgansk Reservoir. Hydrobiological Journal, 36(1): 140-151.

Zurayk, R., Sukkariyah, B. and Baalbaki, R. (2001). Common hydrophytes as bioindicators of nickel, chromium and cadmium pollution. Water, Air and Soil Pollution, 127: 373-388.