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RESEARCH AND APPLICATION TO WATER SUPPLY

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Management of Biesbosch Reservoirs for quality control with special reference to eutrophication

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The three Biesbosch Reservoirs are pumped storage reservoirs, fed with rather polluted and highly eutrophic water from the River Meuse. Air injection at the bottom of the reservoirs prevents thermal stratification, which would otherwise result in serious water quality deterioration. Reservoir mixing also serves as an economic algal control measure; mixing over sufficient depth causes light to play the role of limiting factor and this, combined with zooplankton grazing, keeps the biomass of phytoplankton at acceptable levels. Special problems are caused by benthic, geosmin-producing *Oscillatoria* species growing on the inner embankment. Rooting up the bottom with a harrow is used as the method of control, based on underwater observations by biological staff trained as SCUBA-divers.

With regard to pollutant behaviour the three reservoirs act as a series of fully mixed reactors. This enables the application of kinetic models to describe their behaviour and allows the use of a selective intake policy, e.g. for suspended solids with associated contaminants, ammonia and polynuclear aromatic hydrocarbons. A combination of selective intake and self-purification processes – enhanced by the compartmentalisation of the storage volume in three reservoirs – leads to a striking improvement for many water-quality parameters. Unfortunately this does not apply to some herbicides, like atrazin, which require extra treatment.

Introduction

The Biesbosch Reservoirs are man-made pumped storage reservoirs (Table 1), fed with moderately polluted but highly eutrophic water from the River Meuse. The reservoirs were constructed during the 1970s and serve as the raw water supply for several waterworks in the southern and western parts of the Netherlands, e.g. Rotterdam and its wide surroundings. The delivery of water in 1991 amounted to 166 million m^3 $year^{-1}$, about one-eighth of the total supply in the Netherlands.

The three reservoirs are connected in series as shown in Figure 1. Honderd en Dertig and Petrusplaat are designed as process reservoirs to guarantee a minimum residence time for quality improvement, whereas De Gijster reservoir really serves the storage function of the project and can be nearly emptied, ensuring a safe yield of about 2 months at the present throughput.

Table 1. *Characteristics of the three Biesbosch reservoirs*

Variable	Units	De Gijster	Honderd en Dertig	Petrusplaat
Surface area	ha	305	210	100
Volume	10^6 m^3	40	33	13
Maximum depth	m	27	27	15
Mean depth	m	13	15	13
Retention time	weeks	11	9	4

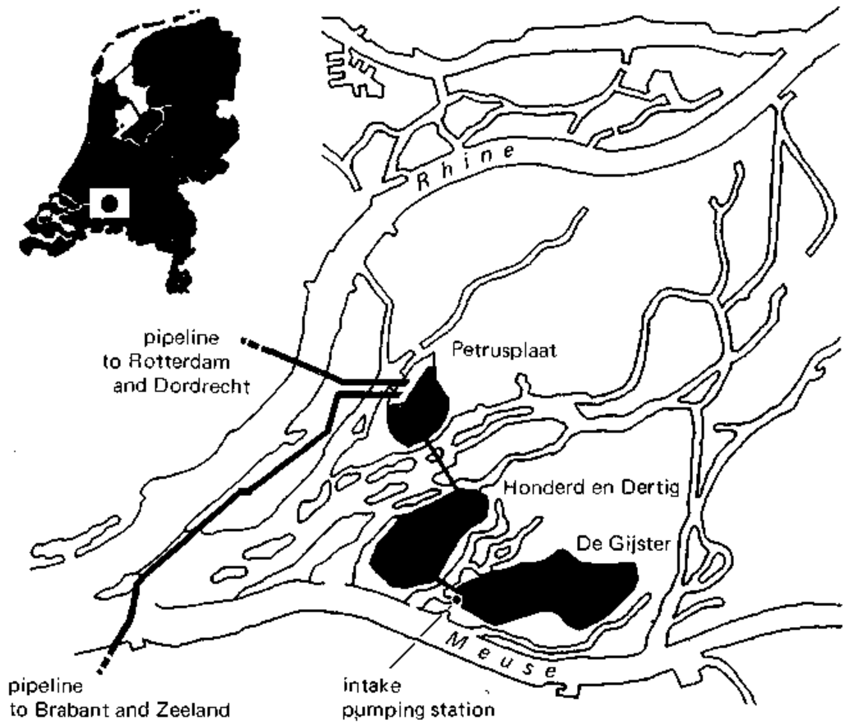

Figure 1. Location of the Biesbosch Reservoirs, in the Netherlands.

Judged by Dutch standards the Biesbosch reservoirs are very deep, varying from 15 m to 27 m locally. Based on the experience with the Thames Valley reservoirs, thermal stratification had to be expected (Ridley *et al.* 1966). When a eutrophic reservoir stratifies, deoxygenation of the hypolimnion water often leads to anaerobic conditions in the bottom layers and this initiates several undesirable processes. Particulate $Fe(OH)_3$ and MnO_2 will be reduced chemically to dissolved Fe^{2+} and Mn^{2+}, also resulting in solution of associated phosphates. Biological processes are denitrification ($NO_3^- \rightarrow N_2$), sulphate reduction ($SO_4^{2-} \rightarrow H_2S$), formation of NH_4^+, and putrefaction of organic substances. As a result the quality of the hypolimnion water becomes unacceptable for treatment to drinking water.

The Meuse water is rich in nutrients, although phosphorus concentrations have decreased because of the replacement of phosphates in detergents. At present the average concentrations of total phosphorus and nitrogen amount to 0.3 and 5 mg l^{-1} respectively. When storing such water the mass development of algae will occur, causing a lot of problems for water supply, like formation of taste and odour compounds, impairment of coagulation, clogging of filters, filter passage of algae, etc. As the objective of storage is also quality improvement and not deterioration, the control of possible adverse effects is the first aim of the management of the reservoirs. In this the air injection system plays an essential role.

Air injection for reservoir mixing

The destratification system, developed for the reservoirs, uses air injection at the bottom for maintaining isothermal conditions. Air injection takes place in so-called injection units, the number of units per reservoir varying from three to six, depending on the size of the reservoir. Figure 2 shows an injection unit, which is very compact and consists of a 1 × 3 metres steel

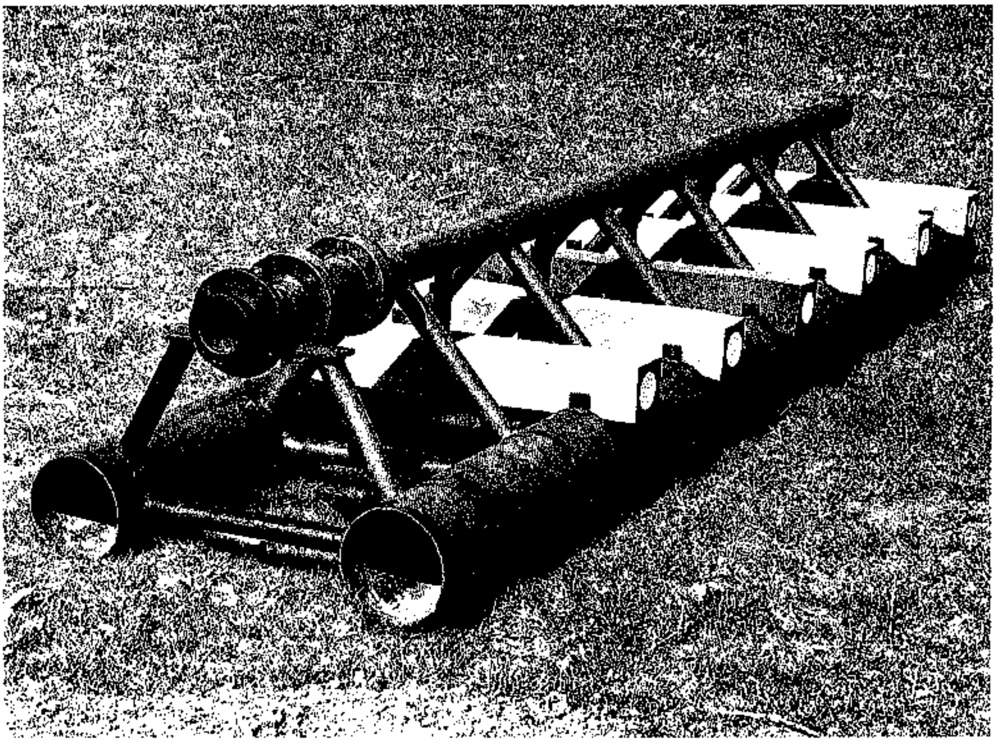

Figure 2. An air-injection unit.

frame on which six Flygt (polyethylene) foam plastic blocks are installed. The land-based compressors feed the injection units by means of plastic pipes, forcing air through the foam blocks and producing a bubble column of very finely divided air, thus entraining large amounts of water.

The system has several advantages. Maintenance cost is very low, because the foam blocks do not clog. Materials deposited during long periods of non-use (autumn and winter) crumble, because the foam block expands when air injection is started again. Installation is easy and very flexible as the position of the units can be easily altered, if their original configuration does not prove to be optimal. The overall investment is moderate.

Table 2. *Characteristics of the air injection system in the three Biesbosch reservoirs*

Variable	Values
Total surface area	615 ha
Total volume	86 10^6 m^3
Water flow	166 10^6 m^3 $year^{-1}$
Injection units	3–6 per reservoir
Injection depths	15–25 m
Air flow	0.8 m^3 s^{-1}
Compressor power	250 kW
Compressor working hours	2,200 h $year^{-1}$
Annual energy consumption	550,000 kWh $year^{-1}$
Price per kWh	0.10 Hfl

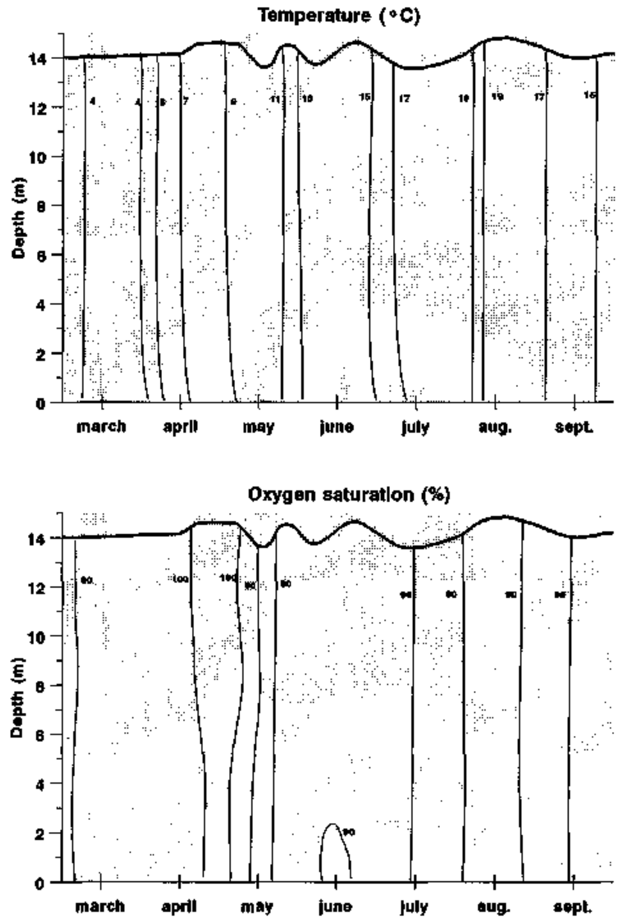

Figure 3. Temperature and oxygen isopleths for Petrusplaat Reservoir.

Table 2 gives the operational characteristics of the air injection systems in the three Biesbosch reservoirs. The total operating cost of the reservoir mixing system amounts to about Hfl. 80,000. The capital cost is Hfl. 150,000 (total investment is Hfl. 1.5 million), so the total cost can be calculated at less than 0.15 cents per cubic metre of delivered water and is thus very moderate.

The air injection is very effective in maintaining homogeneous conditions in the reservoirs. Measurements have shown that under optimal conditions, 1 m^3 of air entrains up to 250 m^3 of water, thus ensuring efficient vertical and horizontal mixing. This is illustrated in Figure 3, where the temperature and oxygen isopleths in Petrusplaat are shown. It can be calculated that the mixing time of the reservoirs is small compared to the average residence time, so the reservoirs can be considered as a series of fully mixed reactors.

Control of planktonic algal growth

Algal control in the Biesbosch reservoirs is based on the principle of light limitation through reservoir mixing over sufficient depth. Artificial mixing as an algal control method was first recognized and applied in the management of the deeper Thames Valley reservoirs, which are circulated by means of jet-type inlet pumps (Steel 1975). Whether or not light can operate as a limiting factor depends mainly on the available mixing depth in relation to the optical properties of the stored water. Simple models of light-limited algal growth have been developed, which consider net algal growth as the balance between gross photosynthesis and respiration. Photosynthesis is confined to the upper layers, whereas respiration occurs in the whole water column. With a large ratio of mixing depth to euphotic depth, respiration will dominate the algal metabolism and thus restrict growth. The algal concentration itself of course affects light penetration. This self-shading phenomenon leads to the concept of a maximum algal concentration for every given situation of mixed depth and extinction of the water, assuming that nutrients and sunlight conditions are optimal.

In very simplified form the expression for the maximum algal biomass (C_{max}, mg Chl m^{-3}) is (Oskam 1978):

$$C_{max} = \frac{1}{\varepsilon_c}\left(\frac{27}{Z_m} - \varepsilon_w\right)$$

where ε_c is specific extinction per unit of algal concentration (m^2 mg Chl^{-1}), ε_w is the extinction coefficient of the water without algae (m^{-1}), Z_m is the mixing depth (m), and 27 is a numerical factor incorporating approximate light intensity and duration.

Figure 4 illustrates the influence of water extinction, ε_w, on the maximum algal biomass for different mixing depths. In the Biesbosch reservoirs, ε_w varies from 0.5 to 1.0 m^{-1} (the corresponding Secchi disc transparency is 2–4 m), so that with average depths of 13–15 m a maximum chlorophyll concentration between 40 and 70 mg m^{-3} can be expected.

Figure 5 shows the course of algal development in Petrusplaat during 1986 to 1991, together with zooplankton numbers. Up to 1988 the pattern was the same every year, with a moderate spring pulse of diatoms (30–60 mg Chl m^{-3}) and small algal crops during summer and autumn. These low levels during most of the growing season were caused by the grazing pressure of the zooplankton, mostly consisting of large cladocerans such as *Daphnia longispina*, *D. magna* and *D. pulex*.

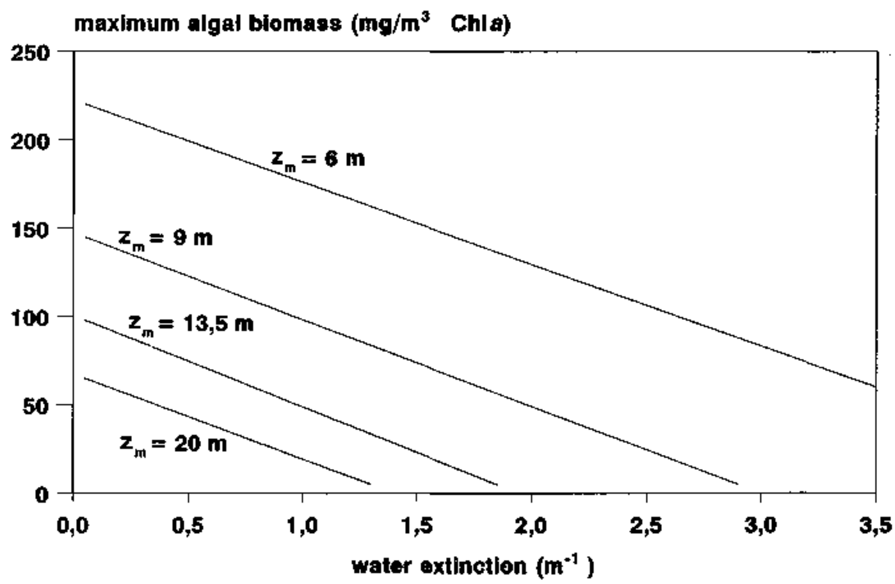

Figure 4. Relationship between maximum algal biomass (mg Chl*a* m^{-3}) and water extinction (m^{-1}) for different mixing depths (specific extinction ε_c = 0.02 m^2 mg Chl^{-1}).

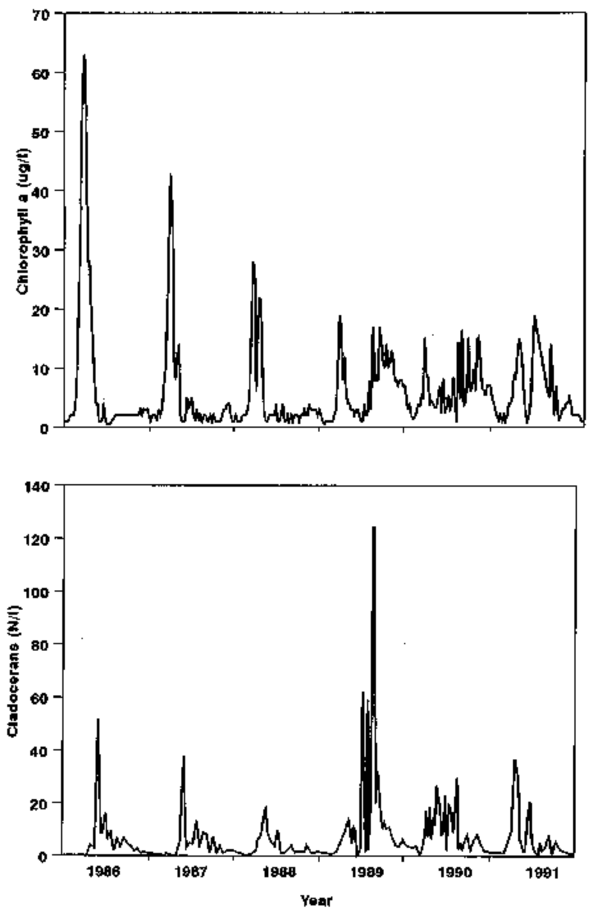

Figure 5. Concentrations of chlorophyll-*a* (mg l^{-1}) and planktonic cladocerans (numbers per litre) in Petrusplaat Reservoir during 1986-1991.

Only in early spring, with water temperatures under 10°C, the grazing pressure of the zooplankton was low and the amount of diatom growth was determined by the light climate. During the rest of the growing season, mixing over sufficient depth retarded the algal growth rate to levels where the zooplankton could "keep up" with the algae, suppressing them to low levels. This controlling effect of the herbivorous zooplankton was possible because predation pressure on the zooplankton was low, as each year the year-class 0 of planktivorous fish was eaten by pike-perch, perch and eels.

Since 1988 the pattern of events seems to have changed. The spring pulse of diatoms decreased, because the mild winters allowed an early appearance of the zooplankton. But also a shift is occurring from predominantly cladoceran to mixed cladoceran and copepod zooplankton populations. The herbivorous copepods are less efficient grazers. Consequently there is a trend to higher algal levels during the summer period. This shift to copepod dominance could well be connected to the presence of the carnivorous zooplankter *Bythotrephes longimanus*. After its first appearance in 1987, it has now become well-established with levels of about 1–2 individuals per litre of water during the summer months. This development gives cause for some concern and merits close attention. Nevertheless, the present levels of algal biomass in the reservoirs are still five to ten times lower than in

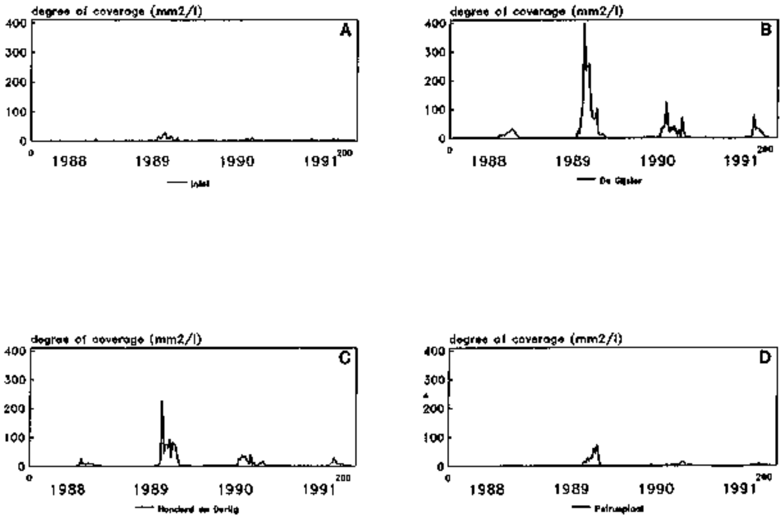

Figure 6. Second distribution of planktonic cyanobacteria expressed as relative aerial units (mm^2 l^{-1}) in the Biesbosch Reservoirs during 1988-1991. A, river water from the Meuse; B, de Gijster Reservoir; C, Honderd en Dertig Reservoir; D, Petrusplaat Reservoir.

comparable water-bodies, but which are shallow and have large populations of herbivorous fish like bream and roach.

In spite of the growth-control by vertical mixing and grazing, some algal species are less easily controlled. These are benthic species and planktonic algae with buoyancy by gas vacuoles, such as the planktonic blue-green algae *Microcystis aeruginosa*, *Aphanizomenon flos-aquae* and *Anabaena* spp. These cyanobacteria reflect a horizontal and vertical inhomogeneity in large water-bodies, induced by wind. Therefore it is necessary to take samples at several locations in the reservoir to get a real impression of the abundance of the cyanobacteria. A mixture of these samples represents the average biomass in the whole reservoir.

Figure 6 shows the abundance of the cyanobacteria in the period 1988–1991, in the inlet and the three reservoirs, expressed as relative aerial units (mm^2 l^{-1}). The small inlet concentrations can be explained by the turbulent and turbid nature of the river water. The growth period in the reservoirs starts in June–July and ends in September–October, with the maximum biomass in August–September. Although biomass varies greatly in different years, the constant pattern is that the highest biomass is found in the first reservoir, De Gijster. Some time-shift can be seen in the starting point of the growth period in each reservoir. This could indicate that the appearance of the planktonic cyanobacteria in the last two reservoirs is caused by successive inoculation.

The comparatively high biomass of cyanobacteria in De Gijster might be explained by its different morphometry. All around its perimeter the reservoir has a shallow area (width 100 m, depth 3–7 m). This has a negative influence on the mixing characteristics of the reservoir and favours the growth of cyanobacteria. *In situ* measurements are being planned to confirm this hypothesis.

It has long been known that cyanobacteria can produce toxins. These substances are held responsible for the deaths of animals (wild and livestock), birds and fish, and have sometimes been associated with human illness after skin contact and ingestion. Especially *Microcystis*, but also *Anabaena*, *Aphanizomenon* and *Oscillatoria* species have been implicated in producing toxins. A recent overview of the problems encountered has been given by Lawton & Codd (1991). From their data it appears that 44 to 75% of the blooms tested showed toxicity in a mouse bioassay. A recent Dutch study showed that the toxins of *Microcystis* were present intracellularly as well as in the water (Hoekstra *et al.* 1991). For this reason the reduction of cyanobacterial numbers in the process reservoirs Honderd en Dertig and Petrusplaat is considered to be very important.

Control of geosmin-producing benthic cyanobacteria

Objectionable tastes and odours in drinking water, especially earthy-musty odours, are a worldwide problem. Two organic compounds, geosmin and 2-methylisoborneol, have been indicated as the main cause of earthy-musty odours in water (Gerber & Lechevalier 1965; Rosen *et al.* 1970). These compounds are produced by actinomycetes (Gerber 1983; Wood *et al.* 1983) and cyanobacteria (Persson 1983).

In April–May 1984 an earthy-musty odour developed in Petrusplaat. Geosmin was detected at a concentration of 33 ng l^{-1}. One of the receiving waterworks had no facilities for sufficient geosmin removal and many consumer complaints were received as the geosmin concentration of the drinking water exceeded the 5 ng l^{-1} threshold odour concentration. The high concentration of geosmin could not be explained by the composition of the biota in the pelagial zone of the reservoir. Algal biomass was low (2 μg Chl*a* l^{-1}) and consisted mainly of diatoms (*Stephanodiscus hantzschii*, *Melosira islandica* and *Asterionella formosa*) which are not recorded as possible geosmin producers (Persson 1983, 1988). Also the number of actinomycetes (maximum 0.4 CFU per ml) had not increased compared to previous years.

Water samples taken adjacent to the inner embankment showed higher taste (dilution) numbers. Therefore an intensive sampling campaign of the bottom was started with an Eckman–Birge mud-sampler. Microscopic examination showed the presence of benthic cyanobacteria, *Oscillatoria limosa*, *O. tenuis*, *O. brevis* and *Pseudoanabaena catenata*. Cyanobacteria of the genus *Oscillatoria* in general, but also benthic *Oscillatoria*, are recorded as producers of geosmin and 2-methylisoborneol (Izaguirre *et al.* 1982; Persson 1983). This information led to the suggestion that these *Oscillatoria* were responsible for the production of geosmin. This was confirmed by *in situ* measurements and laboratory experiments, where pure culture isolates of *Oscillatoria splendida* and *O. limosa* were shown to produce geosmin but not 2-methylisoborneol (van Breemen *et al.* 1991, 1992).

As it was considered impossible to monitor *Oscillatoria* development by conventional sampling it was decided to train biologists as SCUBA-divers, so that *in situ* observations would allow control measures at the right moment (Means *et al.* 1984). In the spring of 1985 a team of three divers was operational. The underwater observations showed that *Oscillatoria* were growing in a narrow band on the inner embankment at a depth of 3–6 m. This confined area of growth offered two possible methods of control: dosing with copper sulphate pellets, as already successfully applied in California (McGuire *et al.* 1983) and a newly-developed method, in which cyanobacteria are stirred up by rooting up the bottom with a harrow-like device (van Breemen *et al.* 1991)

Experiments on controlling the cyanobacteria were started in May 1985. Four areas (each approximately 2,000 m^2) with a benthic cyanobacterial mat were chosen, based on underwater observations. Two locations served as controls and two were experimental areas where copper sulphate was applied in doses of 250 and 500 kg ha^{-1} respectively. Elsewhere, rooting up the bottom was done with a harrow consisting of 21 chains, dragged by a boat. Figure 7 shows the harrow in its original form.

The results of this experiment (Fig. 8) indicated that both methods of control had the same effect: within one week there was a sharp decrease in the area of cyanobacterial growth. After

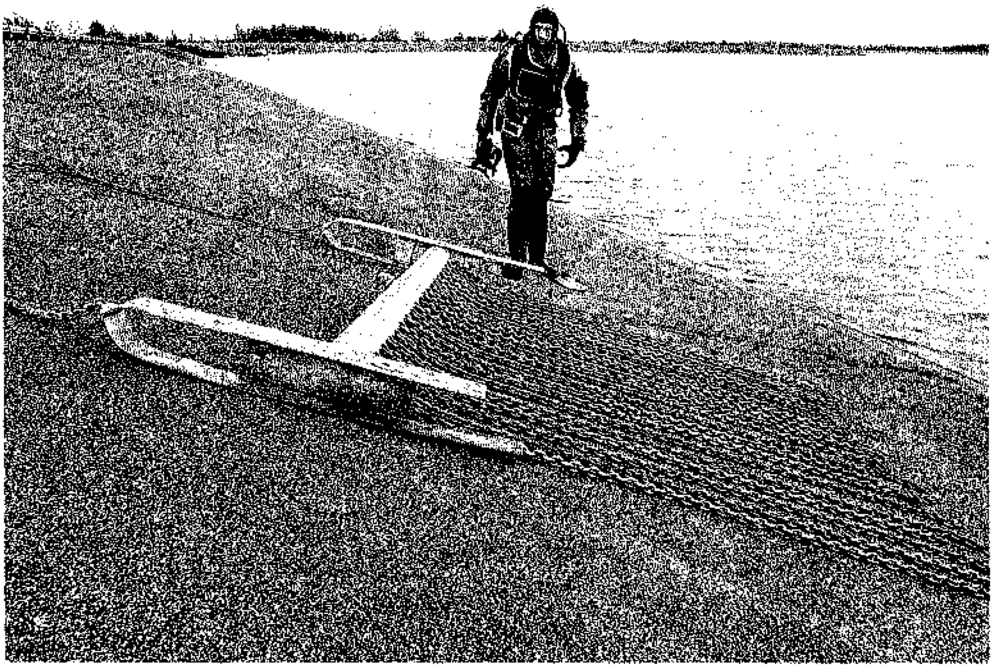

Figure 7. A harrow used for rooting up the bottom of the reservoirs.

two weeks, recolonisation was observed in the area where the bottom was rooted up and in the area where copper sulphate was dosed at about 250 kg ha^{-1}. In the area where 500 kg ha^{-1} was applied, regrowth was not observed, indicating that this method is successful in eliminating benthic cyanobacteria.

Copper sulphate not only had a great impact on the cyanobacteria, it appeared to be lethal for all living benthos at 500 kg ha^{-1} and killed more than 50% at 250 kg ha^{-1}. Because of this harmful effect of copper sulphate on an essential part of the aquatic ecosystem, it was decided to continue the experiment in 1986 by only rooting up areas of cyanobacterial growth. The results (Fig. 9) show that in the untreated area, cyanobacterial growth was comparable to that in 1985, and the growth period lasted from April to July. In the treated areas, hardly any growth occurred, showing that a frequency of once every fortnight is sufficient to control the growth of benthic cyanobacteria.

In the final reservoir, Petrusplaat, various *Oscillatoria* species produced geosmin from February until June. Underwater observations indicate that growth of benthic cyanobacteria in the first two reservoirs can be neglected. To understand to what extent production or reduction of taste and odour compounds occurs in the Biesbosch reservoirs a survey was started in 1988. River water and outlets of Honderd en Dertig and Petrusplaat were analysed routinely for geosmin and 2-methylisoborneol. The results for the period 1988–1990 are shown in Figure 10.

The concentrations of both compounds in the river, with maxima of 10–50 ng l^{-1}, are higher than after storage in the first two reservoirs. The seasonal variations of geosmin and 2-methylisoborneol in the river water are the result of processes in the river and catchment area; both compounds increase during periods of high river flow. In the first two reservoirs a substantial decrease of geosmin and 2-methylisoborneol concentrations takes place; at the outlet of Honderd en Dertig, maximum values of about 3-6 ng l^{-1} were found.

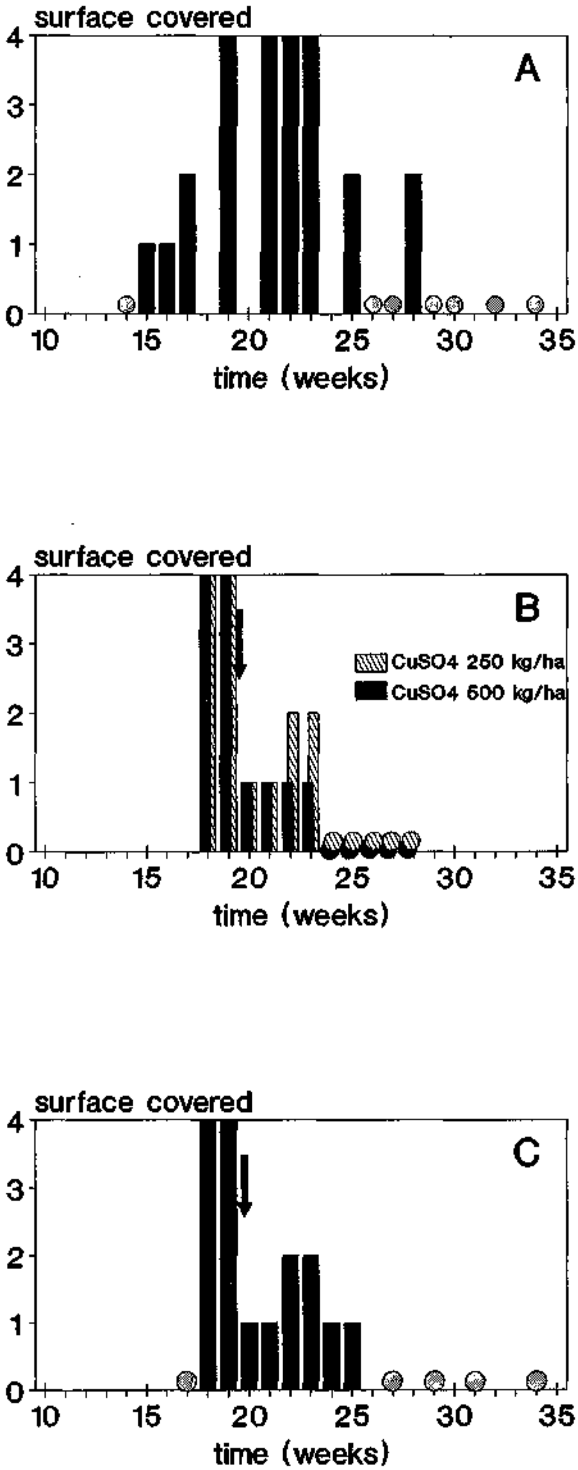

Figure 8. Percentage of the bottom covered with benthic cyanobacteria in the growth-control experiment in Petrusplaat Reservoir in 1985. A, untreated area; B, area treated with $CuSO_4$; C, area rooted up with a harrow. ●, no cyanobacteria observed, and 1 = 0-25%, 2 = 25-50%, 3 = 50-75%, 4 = 75-100% of the bottom covered with cyanobacteria. ↓:time of $CuSO_4$ dosage or start of the rooting up treatment.

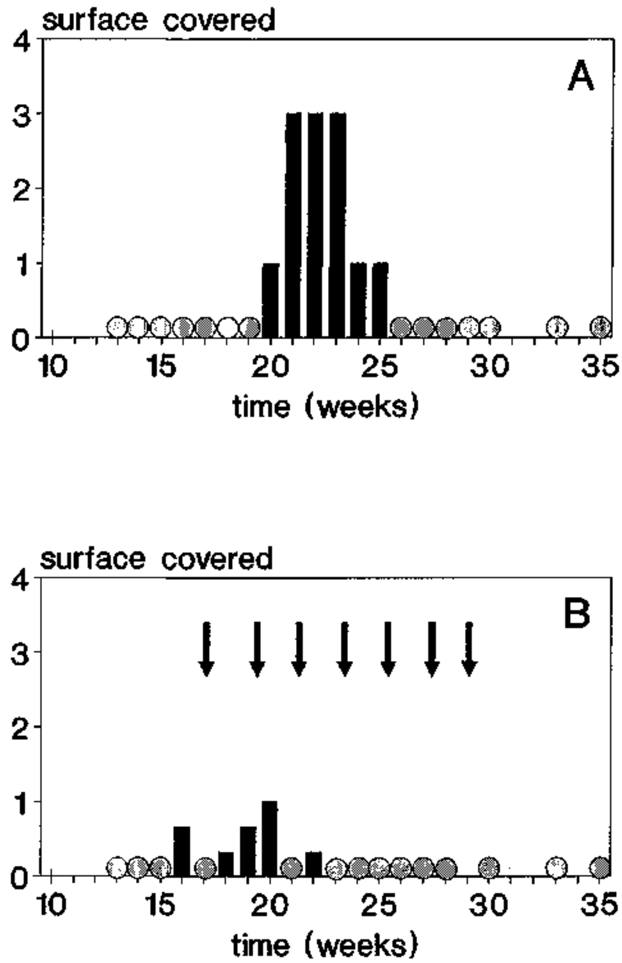

Figure 9. Percentage of the bottom covered with benthic cyanobacteria in the growth-control experiment in Petrusplaat Reservoir in 1986. A, untreated area; B, area rooted up with a harrow. 0-4 on the ordinate represent proportions covered with cyanobacteria, as in Fig. 8. ↓ start of the rooting up treatment.

As a result of cyanobacterial control in Petrusplaat, maximum geosmin concentrations sharply decreased from 33 ng l^{-1} in 1984 to 6.4 ng l^{-1} in 1985, and to 2–3 ng l^{-1} in 1986-1988. In 1989 and 1990, geosmin concentrations in Petrusplaat increased, while those of 2-methylisoborneol did not change. Underwater observations showed that the growth of the benthic cyanobacteria started earlier in 1989 (March) and much earlier in 1990 (January–February) than in previous years. These observations, and the fact that the maximum geosmin concentrations in those years were found 5-8 weeks earlier, indicates that the growth of the benthic cyanobacteria may be influenced by water temperature.

In 1990 cyanobacterial growth started in January. Because geosmin concentrations were low it was decided not to start rooting up the bottom. However, the rather sharp increase of the benthic cyanobacteria in March resulted in an increase of the geosmin concentrations. Although the bottom was then rooted up daily it was not possible to prevent a rise in the geosmin concentrations (maximum of 16.3 ng l^{-1}). This experience in 1990 shows that for a successful control of growth it is necessry to start rooting up the bottom as soon as the

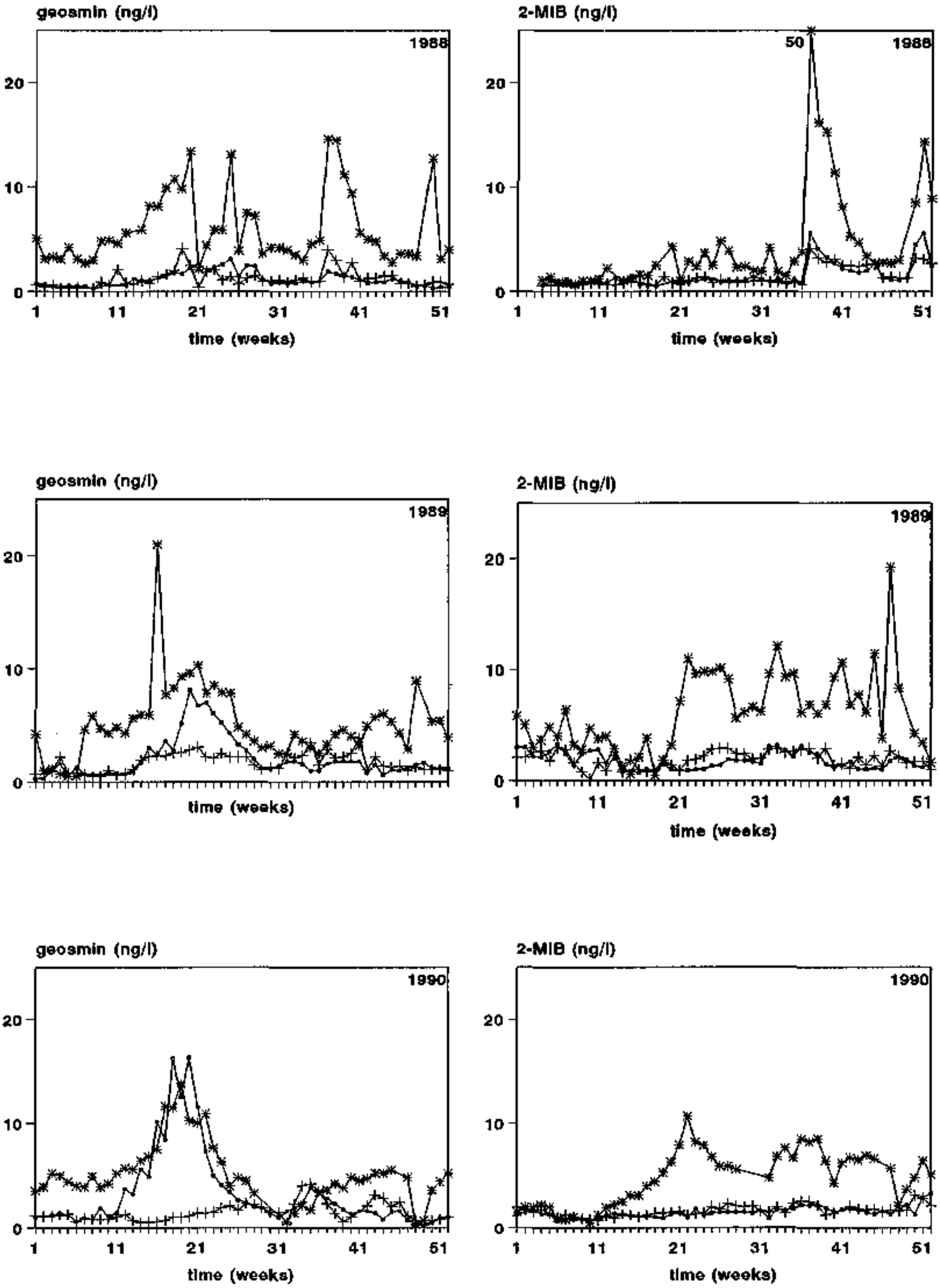

Figure 10. Concentrations (ng l⁻¹) of geosmin and 2-methylisoborneol in the River Meuse (∗), the outlet of Honderd en Dertig Reservoir (+) and the outlet of Pertusplaat Reservoir (■) in 1988, 1989 and 1990.

cyanobacteria are observed. The general conclusion of this survey is that storage of water from the River Meuse in open reservoirs, with a retention time of 4–5 months, reduces geosmin and 2-methylisoborneol concentrations by 60–70%. Autochthonous production of geosmin by benthic cyanobacteria, however, may interfere with this water quality improvement.

Quality improvement in the reservoirs

Theoretical considerations

Quality considerations already formed an integral part of the design stage of the Biesbosch project. For this reason the total volume is divided over three reservoirs. Even when the first storage reservoir is empty, the other two process reservoirs will guarantee sufficent residence time to reach the quality objectives for the abstracted water. The most important processes that contribute to quality improvement in storage reservoirs are outlined in Table 3.

Many degradation processes can be described by first-order reaction kinetics. Assuming steady-state conditions, it can be shown that the removal effect of a series of completely mixed reservoirs can be expressed as (Oskam 1982):

$$\frac{C_{out}}{C_{in}} = \left[1 + k\left(\frac{T}{n}\right)\right]^{-n}$$

where C_{out} is the concentration in outlet water (mass per volume), C_{in} is the concentration in inlet water (mass per volume), k is the rate constant (day^{-1}), T is the total residence time (days) and n is the number of reservoirs.

Table 4 gives calculated C_{out}/C_{in} ratios as a function of compartmentalisation and reaction rates (k) for a total residence time of 180 days. From Table 4 it can be concluded that slow reactions do not benefit much from compartmentalisation, but at higher reaction rates the effect becomes quite large. Apart from planological requirements this was the main reason for dividing the process volume of the Biesbosch reservoirs into two: Honderd en Dertig and

Table 3. *Quality improvement processes in reservoirs*

Type	Process
Biological	Biodegradation of organic substances
	Nitrification of ammonia to nitrate
	Die-off of faecal bacteria and viruses
Physical	Equalizing of water quality
	Gas exchange of oxygen and carbon-dioxide
	Evaporation of volatile substances
	Settling of suspended solids and absorbed material
Chemical	Oxidation of divalent iron and manganese
	Hydrolysis of organic esters and polyphosphates
	Photolysis of humic substances and polynuclear aromatic hydrocarbons.

Table 4. *Ratios of oulet and inlet concentrations for three different k-values and compartments (T = 180 days)*

	Number of compartments		
k	1	2	3
0.01	0.36	0.28	0.24
0.05	0.10	0.033	0.016
0.15	0.036	0.0048	0.001

Petrusplaat (Knoppert & Vreedenburgh 1982). It was also decided to include the storage volume of De Gijster as much as possible in attaining maximum benefit from quality improvement processes.

Results from practice

This section describes typical results for, respectively, a slow, intermediate and fast removal process, with rate constants for nitrification of ammonia in winter ($k \sim 0.018$ day^{-1}), photolysis of benzo(a)pyrene ($k \sim 0.07$ day^{-1}), and die-off of enteroviruses ($k \sim 0.14$ day^{-1}).

In order to eliminate the need for breakpoint chlorination, the delivered water from the Biesbosch reservoirs should not contain more than 0.2 mg l^{-1} ammonia in the critical winter period. A model description of the ammonia behaviour in the reservoirs – based on nitrification ($k \sim 0.018$ day^{-1}) and equalization – shows that the intake should be closed when ammonia in the river exceeds 1.2 mg l^{-1}. Figure 11 shows the results of this ammonia control policy for average and maximum values in river water, inlet water and successive reservoirs. The winter maximum in the delivered water usually amounts to 0.15 mg l^{-1}. If only one large reservoir with the same total residence time of about 170 days had been available, the reduction would have been only 4-fold. This would result in a maximum ammonia concentration of about 0.3 mg l^{-1}, at which level treatment problems have been experienced in the past, due to nitrite formation.

Figure 12 illustrates the effect of storage for the carcinogenic polynuclear aromatic hydrocarbon (PAH), benzo(a)pyrene. The removal process of PAHs is photolysis. The inlet concentration of 21 ng l^{-1} is reduced to 0.3 ng l^{-1} through the successive reservoirs. This 70-fold reduction would have been only 10-fold if only one large reservoir with the same total residence time of 170 days had been available. Figure 12 also shows that the concentration in the river itself was nearly double the inlet concentration. This is also caused by the application of selective water intake, when the quality does not meet the requirements. This also applies

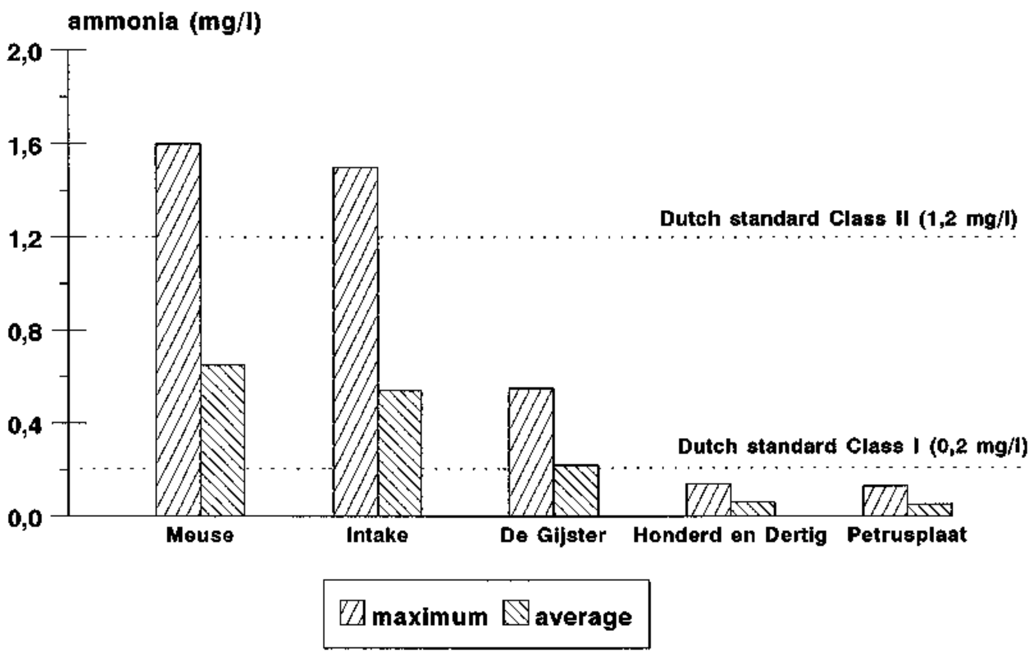

Figure 11. The effects of a selective intake policy and nitrification in the Biesbosch reservoirs.

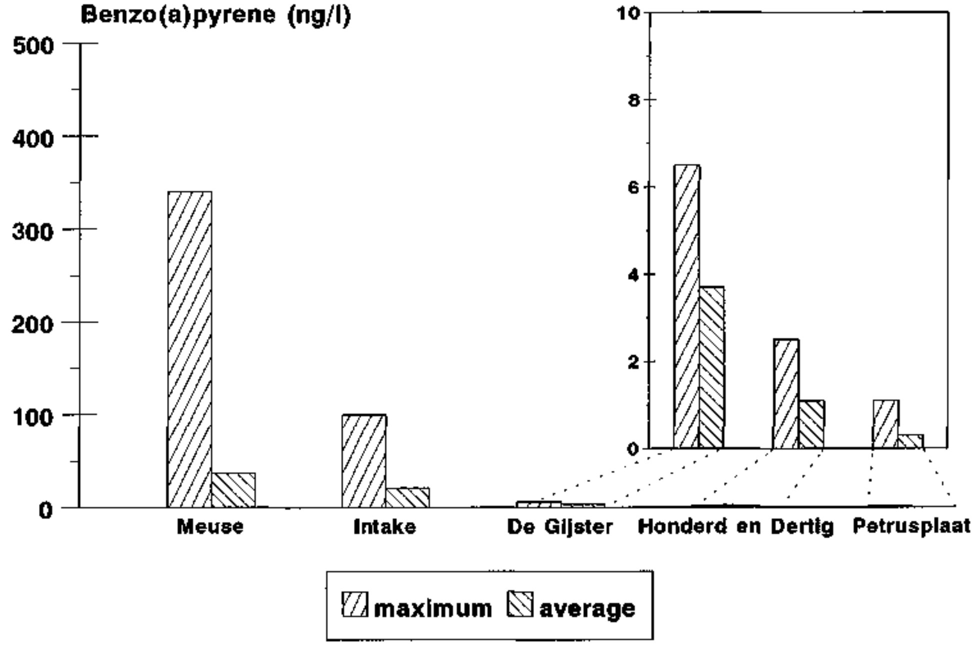

Figure 12. The effects of a selective intake policy and photolysis in the Biesbosch reservoirs.

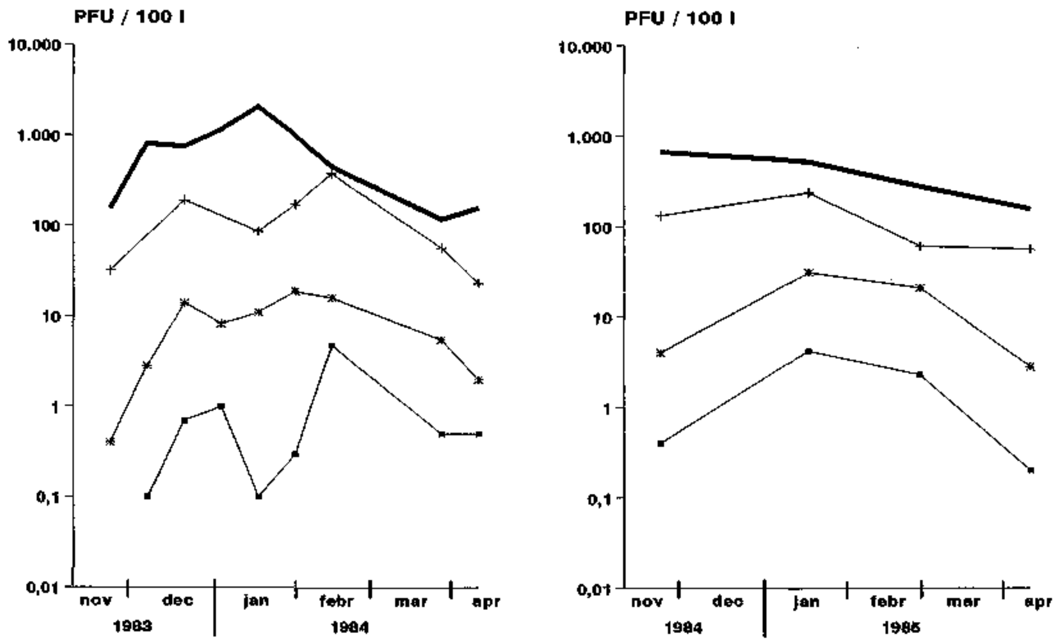

Figure 13. Decreasing numbers (PFU per 100 litres) of enteric viruses in water from the River Meuse (heavy solid line, top) and the outlets of De Gijster Reservoir (+), Honderd en Dertig Reservoir (∗) and Petrusplaat Reservoir (■), during November 1983 to April 1984.

when, under spate (flood) conditions, the river is highly loaded with suspended solids and associated impurities like hydrocarbons and heavy metals.

During the winters of 1983–84 and 1984–85, a study of virus removal was carried out. The winter period was chosen as the critical period, virus survival being better at low temperatures. The results are shown in Figure 13. The number of enteroviruses in the river water varied from 1 to 20 per litre. The overall reduction in the reservoirs was nearly 1000-fold, even while some short-circuiting between inlet and outlet water of De Gijster reservoir could be noted. An average k-value of 0.14 day^{-1} can be calculated, indicating that virus die-off is a fast process in the kinetic sense, which greatly benefits from compartmentalisation. In a single large reservoir the reduction would have been less than 30-fold and the production of virologically safe drinking water would need more elaborate treatment.

The overall quality improvement is very important. Table 5 gives an idea for some selected parameters. From these results it is apparent that the Biesbosch storage system, through its quality improvement potential, may be regarded as an integral part of the treatment of river water to provide a safe drinking water. Unfortunately the herbicides atrazine and simazine are hardly removed in the reservoirs. Due to equalizing, the river peaks of 0.5 to 1.0 μg l^{-1} in summer are reduced to maxima of 0.2 to 0.3 μg l^{-1} in the delivered water, but on average the drinking water limit of 0.1 μg l^{-1} is exceeded for both herbicides. In the meantime, all receiving waterworks are equipped with granular activated carbon filtration to remove these substances.

Table 5. *Average quality of inlet and reservoir waters, in 1991.*

Variables	Units	Inlet	De Gijester	Honderd en Dertig	Petrusplaat
Suspended solids	mg l^{-1}	0.71	0.22	0.08	0.02
Aluminium	mg l^{-1}	15	5.2	2.5	3.4
Iron	mg l^{-1}	0.60	0.20	0.05	0.01
Lead	μg l^{-1}	5.9	3.3	1.5	1.1
DOC	mg l^{-1}	4.0	3.7	3.5	1.1
COD	mg l^{-1}	17	13	11	11
Colour (Pt-Co scale)	mg l^{-1}	14	11	8	8
Chlorophyll	μg l^{-1}	27	12	6	5
Faecal coli	no. ml^{-1}	1.6	0.31	0.20	0.03
Atrazine	μg l^{-1}	0.16	–	–	0.15
Simazine	μg l^{-1}	0.13	–	–	0.11

Conclusions

(1). In "optically deep" reservoirs, artificial mixing provides an efficient and economic way of maintaining the development of planktonic algae at moderate levels.

(2). The development of geosmin-producing benthic cyanobacteria can be effectively controlled by rooting up the bottom with a harrow.

(3). Modelling approaches can be applied to management for water-quality control in storage reservoirs.

(4). With fully-mixed reservoirs, compartmentalisation of the reservoir volume leads to increased water quality improvement during storage.

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