BALTIC SEA ENVIRONMENT PROCEEDINGS

No. 30

SECOND SEMINAR ON WASTEWATER TREATMENT IN URBAN AREAS

6-8 September 1987
Visby, Sweden

BALTIC MARINE ENVIRONMENT PROTECTION COMMISSION
– HELSINKI COMMISSION –

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April 1989

This publication has been technically edited by the host country
of the Seminar, Sweden.

For bibliographic purposes this document should be cited as:
Baltic Marine Environment Protection Commission
– Helsinki Commission –
1989
Second Seminar on Wastewater Treatment in Urban Areas
6-8 September 1987, Visby, Sweden
Baltic Sea Environment Proceedings No. 30

ISSN 0357-2994

Helsinki 1989 – Government Printing Centre

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PREFACE

Within the framework of the Helsinki Commission, a Second Seminar on Wastewater Treatment in Urban Areas was held in Visby, Sweden, 6-8 September 1987.

The Seminar was hosted by Sweden and organized jointly by the National Environmental Protection Board and the Swedish Water and Wastewater Works Association.

Experts of Contracting Parties to the Helsinki Commission, both from the administrative and from the scientific fields participated in the seminar. In order to broaden the base for the seminar, participants from countries bordering to the North Sea were also invited.

After the opening of the seminar by Mr. Bertil Hawerman, Sweden, a statement was made by Professor Harald Velner, Executive Secretary of the Helsinki Commission.

Mr. Ragnar Elmgren held a general lecture on the issue nitrogen/phosphorus and Mr. J. Bryan Ellis presented a state-of-the-art document on Quality of Stormwater.

After these opening lectures, the seminar continued in two different sub-groups.
- Sub-group I - Combined Sewers; Stormwater - Separate Systems and Industrial Discharges,
- Sub-group II - Phosphorus and Nitrogen Removal in Municipal Wastewater Treatment.

This report contains all documents presented at the Seminar as well as the paper by Prof. Dr. I. Sekoulov "The State of Advanced Biological Wastewater Treatment in the Federal Republic of Germany" presented at the First Visby Seminar but not included in BSEP No. 25.

The authors are considered to be responsible for the contents of their papers. The Helsinki Commission is not responsible for any statements made or opinions expressed in this publication.

STATEMENT BY PROFESSOR HARALD VELNER, EXECUTIVE SECRETARY OF THE BALTIC MARINE ENVIRONMENT PROTECTION COMMISSION (HELSINKI COMMISSION) - HELD AT THE SECOND SEMINAR ON WASTEWATER TREATMENT IN URBAN AREAS, VISBY, SWEDEN, 6-8 SEPTEMBER 1987 AND IN KARLSKRONA, SWEDEN, 7 JUNE 1988

Mr. Chairman, Ladies and Gentlemen,

The Convention on the Protection of the Marine Environment of the Baltic Sea Area was signed by all the seven Baltic Sea States in 1974 and has been in force since 1980. It is the first international instrument to deal with all different sources of pollution in the Baltic Sea. The Convention has led to close cooperation between the Baltic Sea States in the protection of the Baltic Sea marine environment coordinated by the Helsinki Commission and its executive body - the Secretariat. Close cooperation was established at first with the Paris and Oslo Commissions protecting the North Sea, and with ICES, UNEP and some other intergovernmental organizations.

The principle of the Helsinki Convention is that the Contracting Parties take all appropriate measures to prevent and abate the pollution of the Baltic Sea (Article 3). The main task is to reduce (or stop, if possible) discharges into the Baltic Sea of toxic and other harmful substances (Paragraph 1 of Article 6). The Contracting Parties agree to apply effective treatment of municipal and industrial wastes aiming at the reduction of discharges of harmful substances, organic matter and nutrients to the marine environment (Paragraphs 1 and 3 of Annex III). During 1980-88 the Helsinki Commission adopted 24 Recommendations prepared by its subsidiary bodies working in the scientific and technological field with the aim of minimizing the load of pollution entering the Baltic Sea from different land-based sources.

The First Periodic Assessment of the State of the Marine Environment of the Baltic Sea covering the five-year period from 1980 to 1985, published, in 1986 and adopted by the Commission, showed that there had been progress in the state of the marine environment compared with the state of the Baltic evaluated in

1980. Concentrations of toxic substances like DDT, PCBs, and mercury in living organisms and seawater were decreased. Unfortunately it has not been possible to reduce remarkably the load of phosphorus and nitrogen compounds discharged into the marine environment from municipalities, industries and agriculture. Increase of nitrogen and phosphorus compounds was reported in the Baltic Sea Environment Proceedings No. 17, 1986,1987 (BSEP). The greatest increase of nutrients has been found in the Gulf of Finland, Kattegat, Danish Sounds and Bay of Gdansk. During the last years exceptional bloomings of plankton algae has been reported from the Southern Baltic. For the first time in many years it was found that oxygen deficiency had developed not only in the deep parts of the Baltic but also in the shallow waters. Along the coast there are areas where the larger benthic fauna has been distroyed by pollution.

The first Baltic Sea pollution load compilation was published by the Commission in 1987 (BSEP No. 20). The pollution load is considered in three categories according to the source: municipalities, industry, rivers. The load carried via the atmosphere is considered in this document only to give an overall picture of the amount of pollutants (Figures 1 and 2). The sources may not be strictly defined due to overlapping in some cases. Considering that the values are often based on very rough background data, especially pollution from atmosphere, the document should be seen as a first step. The load of pollution was calculated for 20 sub-areas of the Baltic Sea and the results presented clearly indicated to the most affected areas (Figures 3 and 4). The total load of phosphorus (P) and nitrogen (N) into the Baltic Sea is estimated to be ca. 50 000 tons and 1 000 000 tons respectively. The load of pollution of BOD make ca. 1 7000 000 tons. The consideration of the atmospheric fallout is a task of an expert group (EGAP) which has given its first estimations for the use of the Commission.

Investigations made revealed an urgent need to take all appropriate measures to treat municipal sewage and industrial wastewater with special emphasis on the reduction of discharges of nutrients and a recommendation was adopted by the Commission as a first step (HELCOM Recommendation 6/7).

During the seventh meeting of the Helsinki Commission in 1986 further steps were adopted by the Commission. According to HELCOM Recommendation 7/3 the sewerage systems should be renewed in such a way that infiltration and exfiltration be minimized, a separated or a semi-separated sewerage system should be selected for new developments. The Commission adopted also HELCOM Recommendation 7/5 concerning reduction of discharges from urban areas by the treatment of stormwater, where it was indicated that measures should be taken already at the source and the first flush of stormwater especially from heavily polluted industrialized urban areas be purified.

Recommendation concerning measures aimed at the reduction of discharges from urban areas by the use of effective methods in wastewater treatment were discussed and adopted by the Commission in two steps: in 1987 (HELCOM Recommendation 8/3) and in 1988 (HELCOM Recommendation 9/2). It was declared that:

a) urban wastewater deriving from households or idustrial enterprises should be collected and treated before being discharged into water bodies; by-passes may only be used in emergency cases

b) domestic sewage or wastewater of similar type which is collected in a central sewerage system and treated in wastewater treatment plants, loaded with more than 10 000 person equivalents, should be treated as soon as possible and not later than 1998 by biological methods or other methods giving equivalent results, so that the treatment should result in:
   - at least 90% reduction of BOD_5; and
   - at most a concentration of BOD_5 in the effluent of the treatment plant of 15 mg/l;
   - annual average values of total phosphorus below 1,5 mg P/l

It was also decided that each Contracting Party should start research and evaluation projects as soon as possible with the purpose to give a basis for further recommendations for nitrogen removal within three years. The project should include studies of process technology and costs for nitrogen removal to reach the targets:

- as a first step 12 mg total nitrogen/l in the effluent water or 50% reduction of total incoming nitrogen;
- as the second step 8 mg total nitrogen/l in the effluent water or 75% reduction of total incoming nitrogen.

The results of these investigations could be discussed by experts at the seminars organized annually within the auspices of the Commissions. This Karlskrona seminar arranged in cooperation with Paris Commission is an important step in the necessary meetings of experts arranged for the purpose of exchanging relevant information.

The reduction of discharges from agriculture as a source of pollution of the marine environment was discussed during the seventh and ninth meetings of HELCOM (HELCOM Recommendations 7/2 and 9/3).

This year 1988 the Commission decided inter alia that:

a) the farming practice with regard to manure should be in accordance with the following principles:
   - the capacity of the storage facilities should be large enough to ensure that manure need only be brought out when the plants can use the nutrients.
   - the application of manure on frozen or bare soil should be restricted

b) farming practice should be managed so as to favour the efficient use of the nutrients that are available in the agriculture system.

The Commission decided also to reduce drastically the industrial discharges from land and by air where, inter alia, harmful substances, nutrients or organic matter are the main pollution problem. The discharges should be reduced as soon as possible and not later than 1995 using the best available technology (HELCOM Recommendation 9/8) At first restrictions of discharges e.g. from the pulp and paper industry and minings should be applied.

The great importance and need for further steps to reduce the substances most harmful to the ecosystem of the Baltic Sea was underlined in the Declaration on the Protection of the Marine Environment of the Baltic Sea Area adopted and signed during the ninth meeting of HELCOM in February 1988 by the Ministers of the Baltic Sea States responsible for the environmental protection. In this environmental-political declaration the ministers declare, inter alia, their firm determination to reduce radically heavy metals and toxic or persistent organic substances and nutrients in the order of 50% out of the total discharges of each of them as soon as possible but not later than 1995. The best available technology should be used. Therefore, the exchange of information and knowledge on environmental technologies including preferential treatment is of great importance.

During the second North Sea Conference in London 24-25 November 1987 the similar principles and ways to reduce the pollution entering into the North Sea were adopted in the ministerial declaration. A task was given to the Paris Commission to elaborate possible measures to substantially reduce nutrient inputs to the North Sea. I am sure that the present joint symposium will thus speed up the elaboration of new effective methods to protect the marine environment of the North Sea and the Baltic Sea.

Some argue that we do not have enough proof of the harmful effects of the discharges of nutrients into the sea. They say that we must wait for science to provide that proof.

1 would like to finish my statement with the words expressed by HRH the Prince of Wales at the Second North Sea Conference

"If science has taught us anything it is that the environment is full of uncertainties. It makes no sense to test it to destruction. While we wait for the doctor's diagnosis, the patient may die."

Thank you.

FIGURE 1

POLLUTION LOAD COMPILATION
Area abbreviation: TOTAL
THE BALTIC SEA AREA

PARAMETER SUBSTANCE		UNIT	FROM MUNICIPAL.	THROUGH RIVERS	FROM INDUST.	TOTAL SUM
NITROGEN	TOT-N	T/A	67652	449150	11549	528351
PHOSPHORUS	TOT-P	T/A	11801	28321	8397	48519
B O D	BOD	T/A	196610	1212400	295440	1704450
MERCURY	HG	KG/A	1093	3707	268	5068
CADMIUM	CD	KG/A	3216	46340	9320	58876
ZINC	ZN	T/A	460	6709	1765	8934
LEAD	PB	T/A	18	239	8	265
COPPER	CU	T/A	133	3962	101	4196
OIL	OIL	T/A	8977	26021	576	35574
ARSENIC	AS	T/A	4	72	101	177
NICKEL	NI	T/A	10	0	96	106
VANADIUM	v	T/A	0	0	290	290
CHROMIUM	CR	KG/A		127	78	205

Figure from Baltic Sea Environment Proceedings No. 20 (1987)

FIGURE 2

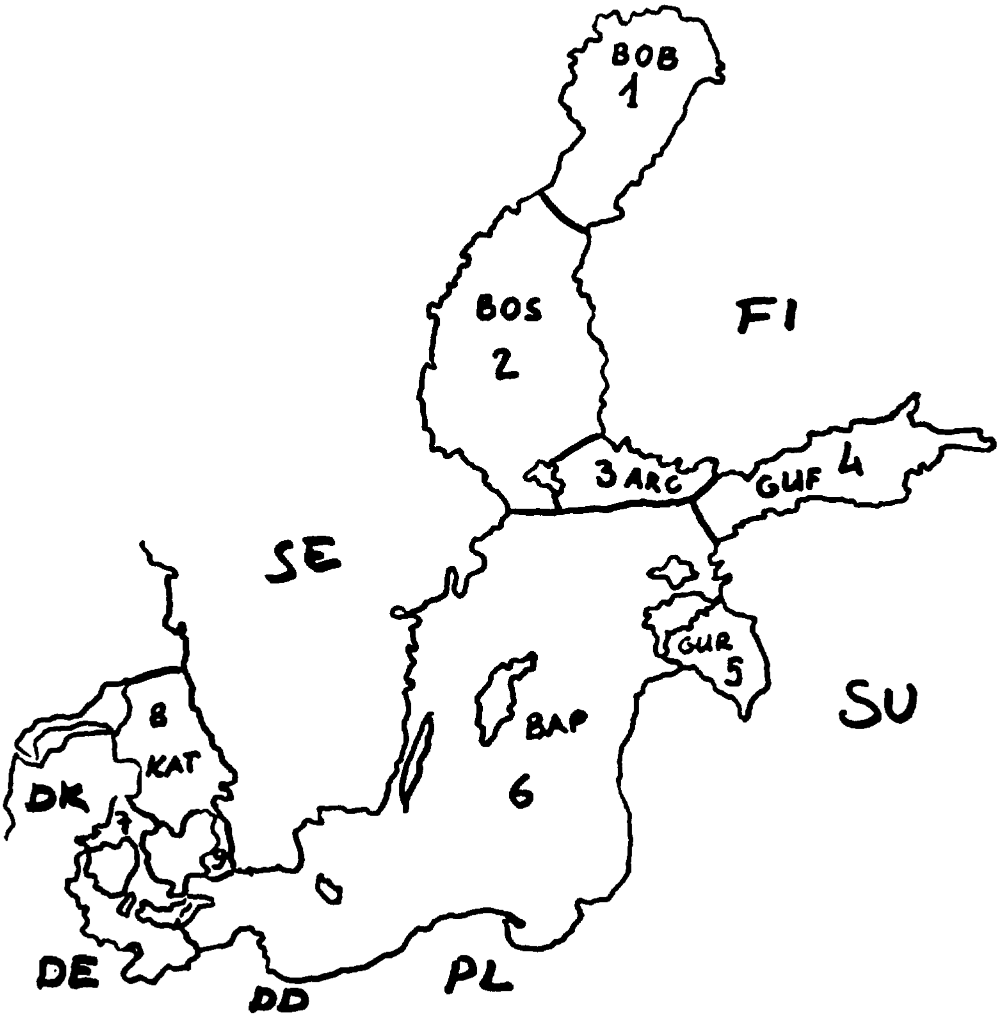
THE TOTAL POLLUTION LOAD

		INPUT FROM LAND		ATMOSPHERIC INPUT (EGAP 3/8)
NITROGEN	TOT-N	530 000 t/a		413 000 t/a
PHOSPHORUS	TOT-P	42 000 t/a		6 000 t/a
B O D		1 640 000 t/a		
MERCURY	Hg	5 t/a	(*	
CADMIUM	Cd	60 t/a		80 t/a
ZINC	Zn	9 000 t/a		3 200 t/a
LEAD	Pb	300 t/a	(*	2 900 t/a
COPPER	cu	4 200 t/a		380 t/a
OIL		36 000 t/a	(*	
ARSENIC	AS	180 t/a	(*	
NICKEL	Ni	110 t/a	(*	
VANADIUM	V	290 t/a	(*	
CHROMIUM	Cr	0.2 t/a	(*	

(* THE DATASET OF THESE SUBSTANCES IS NOT COMPLETE; THEREFORE, THE GIVEN VALUES ARE VERY PRELIMINARY AND NOT REPRESENTATIVE.

Figure from Baltic Sea Environment
Proceedings No. 20 (1987)

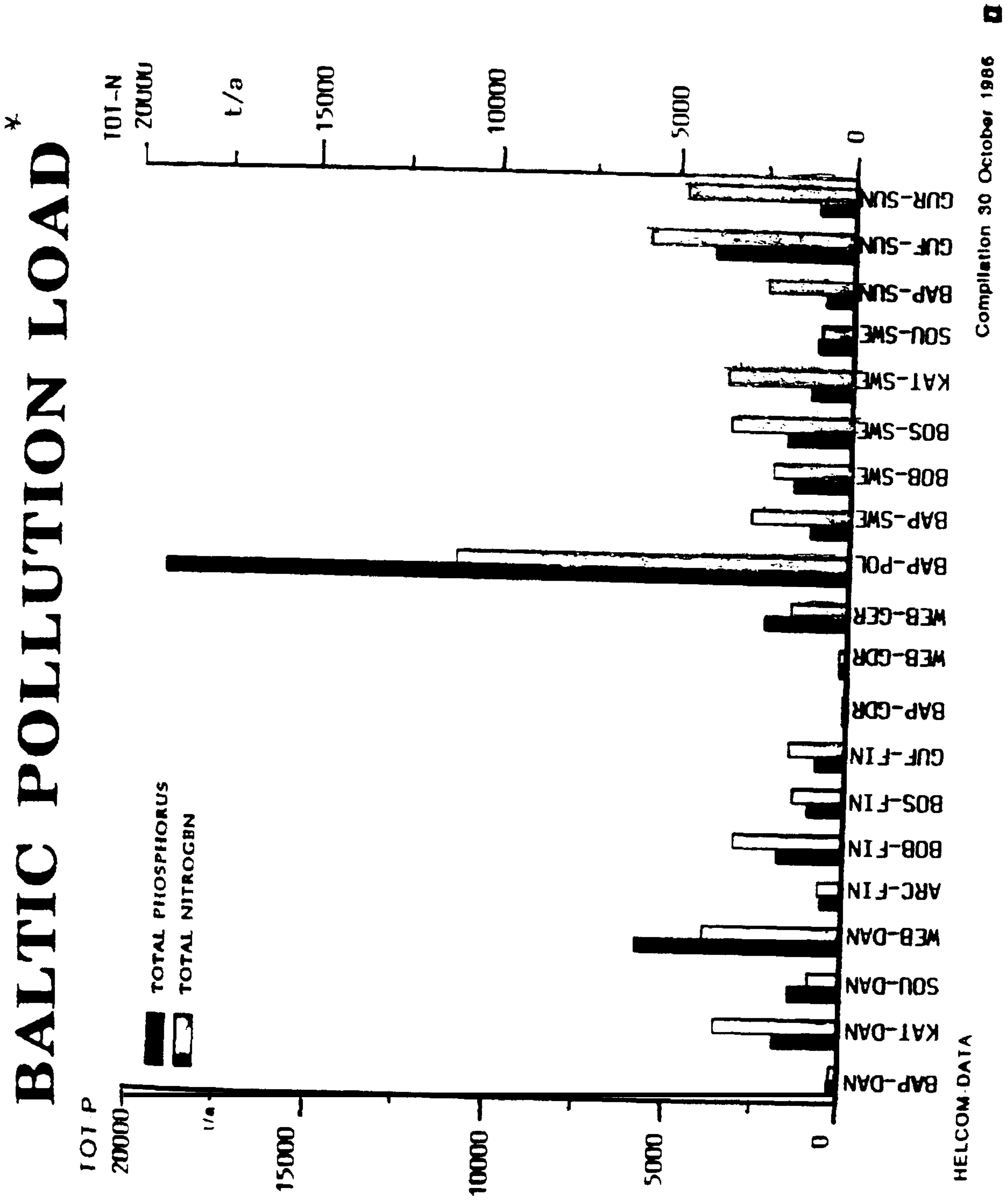
FIGURE 3



SUB-BASINS OF THE BALTIC SEA.

1. BOTHNIAN BAY
2. BOTHNIAN SEA
3. ARCHIPELAGO SEA
4. GULF OF FINLAND
5. GULF OF RIGA
6. BALTIC PROPER
7. THE BELT
8. KATTEGAT
9. THE SOUND

FIGURE 4



*) Figure from Baltic Sea Environment
Proceedings No. 20 (1987)

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Baltic Marine Environment
Protection Commission,
Second Seminar on Wastewater
Treatment in Urban Areas,
Visby, Sweden, Sep 6-8, 1987

THE EUTROPHICATION STATUS OF THE BALTIC SEA: INPUT OF NITROGEN AND PHOSPHORUS, THEIR AVAILABILITY FOR PLANT PRODUCTION, AND SOME MANAGEMENT IMPLICATIONS.

INTRODUCTION

The Baltic Sea has shown clear symptoms of nutrient enrichment in recent decades. This has been discussed in a large number of scientific publications, and in two international assessments, within the framework of the Helsinki Convention on the protection of the marine environment of the Baltic Sea (Melvasalo et al. 1981, HELCOM 1987a, Lassig 1987). What follows is a short summary of this literature, focussed mainly on the Baltic Sea in a restricted sense, inside the Belt Sea and the Sound (= the Danish straits), with some preliminary consideration of likely future developments and of the management options available.

BACKGROUND

The Physical setting

The Baltic Sea is one of the world's largest brackish water areas, and shows remarkably small variations in salinity at any given point and depth. This is the result of a large volume (Table 1), a positive water balance (Table 2) and a semi-enclosed topography (Fig 1). Surface salinity varies from a mere 2-3 o/ooS in the northernmost Bothnian Bay, via 5-6 o/ooS in the Bothnian Sea, 6-8 o/ooS in most of the Baltic proper and rapidly increasing values in the Danish sounds to about 15- 20 o/ooS at the borderline to the Kattegatt. The Baltic Sea is shallow, with an average depth less than 60 m.

	km^3/a
River inflow	-440
Seawater inflow	-470
Precipitation less evaporation	-50
Seawater outflow	-950

Table 1. Baltic water balance (after sources in Ehlin 1981).

	Area, 1000 km2	Volume 1000 km3	Maximum depth, m	Volume below 100m, 1000 km3
Bothnian Bay	36	1.5	147	0.01
Bothnian Sea	79	4.9	301	0.5
Gulf of Finland	29	1.1	123	<0.001
Gulf of Riga	18	0.4	51	0
Baltic proper	210	13.0	459	1.2
Total Baltic Sea (inside Danish straits)	373	20.9	459	1.7

Table 2. Areas and volumes of the Baltic Sea. After Ehlin et al. (1974) and U. Ehlin and G. Zachrisson, pers comm.

In the Baltic proper, a primary halocline at about 70 m depth separates the low salinity surface layer (6-8 o/ooS), from a semi-stagnant deep water of higher salinity (8-11 o/ooS), in which secondary haloclines may further isolate the heaviest bottom water (11-13 o/ooS). The surface layer is mixed in the cold season, but develops a thermocline at 20-30 m depth in the summer period. The formation and breakdown of the thermocline is governed by the thermal balance and the wind mixing, since tidal mixing is negligible.

The bottom water of the deep basins is replaced intermittently, in connection with larger inflows of dense water through the Danish sounds. Such exchanges are infrequent, and stagnation periods as long as about a decade have been recorded in the bottom water of the Eastern Gotland Basin (Fig 1). The Gulf of Bothnia (= The Bothnian Bay + the Bothnian Sea) shows only weak salinity stratification, but the stratification of the Baltic proper extends into the Gulf of Finland.

Figure 1. The Baltic and its subareas. In text computations for 'Bothnian Sea' include the Åland and Archipelago Seas, and for 'Baltic Proper' the Arkona Sea, Bornholm Sea, and Eastern and Western Gothland Seas.

The Baltic Sea extends from 54°N to almost 66°N, nearly to the polar circle. In the far north, the Bothnian Bay is ice-covered for five months each year, whereas in the south open-sea ice is present at all only in harsh winters (Paluoso 1966).

The chemical and biological consequences

Respiratory processes during the long stagnation periods of the Baltic proper deep waters lead to low oxygen concentration, a feature noted already by the earliest investigators of the chemistry of the Baltic Sea. During stagnation periods, nutrients accumulate in the deep water, initially mainly as nitrate and phosphate, but as the deep water turns anoxic, and hydrogen sulphide starts to accumulate, nitrate is largely denitrified and inorganic nitrogen begins to accumulate as ammonium, while phosphate increases in concentration due to release from the now anoxic sediments.

The low but stable salinity of the Baltic Sea has a profound influence on its biological diversity. Only a few marine and freshwater organisms, and a few forms specially adapted to brackish water, are able to persist in those intermediate salinities, leading to an extremely species-poor biota (Remane 1934, 1940, Segerstråle 1957). And already early in this century, Ekman (1916) described how the oxygen-poor deep water in the southwestern Baltic was avoided by fish, with important consequences for the fishery.

The large, nutrient-poor freshwater input to the Gulf of Bothnia, and the much shorter productive season in the north (4-5 months v.s. 8-9 months in the south, Elmgren 1984) leads to a clear north-south gradient of increasing primary production, benthic secondary production (Elmgren 1978), zooplankton secondary production (Ackefors et al. 1978) and fish production and yield (Elmgren 1984).

The long-term trends

The long-term data sets available from the Baltic show relatively minor trends in the physical setting, such as a slight land subsidence in the entrance area (Striggow, 1983), and some changes in climatic forcing functions, such as precipitation and dominant wind speed and direction (summarized by Kullenberg 1986, Launiainen et al. 1987).

The chemical long-term data show many significant trends. These trends are often made up not of slow gradual changes, but a few rather sudden "events", separated by periods of relative stability. Single events, such as the exceptionally large inflow of high salinity water in 1951 (Wyrtki 1954), may influence the situation for decades, adding a stochastic aspect to the long-term trends.

The salinity of the Baltic surface water showed a statistically significant increase from the 1920's into the mid 1970's (Launiainen et al. 1987), but closer scrutiny shows that much of the increase took place in short periods, during World War II, and a few years after the 1951 salt water inflow. This salinity increase has been correlated to a tendency of decreased average precipitation in the Baltic drainage basin during the first half of our century, and a cause-effect relationship inferred (Launiainen et a1.1987). The mechanisms governing the salt water inflows have been modelled by Stigebrandt (1983).

The oxygen concentration in the deep waters of the Baltic Sea have deteriorated sharply in the last few decades. The current stagnation period in the eastern Gotland Basin is the longest on record (about 10 years), with hydrogen sulphide concentrations higher than ever measured there before (Launiainen et al. 1987). The length of the stagnation periods is not influenced by eutrophication, but the almost continuous lack of oxygen in the deep waters in recent decades can at present only be explained as the result of an increased input of organic matter to the deep waters, due to eutrophication (Shaffer 1979).

Alternative explanations of the deep water oxygen deficiency

When the deterioration of oxygen conditions in the Baltic first became an issue through the publications of Fonselius (e.g. 1969) it was impossible to say whether the primary cause was cultural eutrophication, or climatic fluctuations.

Fonselius (1969) showed that phosphate concentrations had increased in parts of the Baltic since the 1930's. He suggested that pollution of the Baltic with nutrients and oxygen consuming wastes could be at least partly responsible for the oxygen decrease. But he also suggested that much of this decrease, and of the increasing phosphate concentrations could be the end result of natural climatic fluctuations. He noted evidence that decreased precipitation in the Baltic drainage basin had led to decreased freshwater input to the Baltic. Less freshwater, he reasoned, could explain the raising of the level of the primary halocline by about 20 m he estimated from his data.

Fonselius also calculated an increased density gradient across the halocline, and suggested this meant less water exchange through the halocline and therefore longer mean stagnation periods in the deep water. A longer stagnation period should mean lower oxygen concentration in the deep water, even if no change occurred in the biological production. But as soon as the stagnation resulted in real oxygen deficiency in the bottom water, bound phosphate would be released from the sediments. When later, following a water exchange, this phosphate-rich water was mixed into the euphotic zone, increased production should follow. This could conceivably start a positive feedback process, with more production giving greater sedimentation of organic matter, giving higher oxygen consumption rates in the deep water, giving larger anoxic areas of sediment, giving further phosphate-releases from the sediment, and so on.

In this primarily climate-driven scenario the increased phosphate concentrations in the water could be explained as due to natural events. This scenario gained in credibility when Hallberg (1973) presented sedimentary evidence of earlier anoxic periods in the eastern Gotland Basin.

The alternative interpretation, that anthropogenic eutrophication was the major factor causing the Baltic anoxia was long considered doubtful. There was never any doubt that anthropogenic nutrient inputs to the Baltic area had increased, nor that this had resulted in severe local eutrophication, e.g. off the cities of Helsinki (Melvasalo and Viljamaa 1977) and Stockholm (Waern and Pekkari 1973). But the magnitude of the nutrient load was only vaguely known, and therefore its influence on the open sea areas of the Baltic was doubted.

Today, the climate-driven scenario has lost much of its credibility. The mean change in the halocline depth seems to be much less than he calculated, only 5-6 m (Matthäus 1980), and the stability across the halocline shows no consistent increase with time (e.g. Kullenberg 1986). Rydberg (1983) argued, based on the model of Stigebrandt (1983), that a decrease in precipitation and run-off in the Baltic area should increase, not decrease, the mixing across the halocline. Larsson et al.(1985) showed that the new estimates of the phosphorus load on the Baltic imply that the sediments must act primarily as net sinks, not sources, of phosphorus. Finally, it is *now* commonly accepted that nitrogen, not phosphorus, is generally the most limiting nutrient for Baltic primary productivity (see below). In the light of this new knowledge, the climate-driven scenario falls short in its physics, its geochemistry and its biology.

Nevertheless, the physically driven stagnation periods clearly decide the short-term dynamics of oxygen concentration in the deep waters of the Baltic, and of salinity and to some extent also nutrients in the surface water (Launiainen et al. 1987, Nehring et al. 1987). It is therefore not surprising that it took a decade to reach the conclusion that they cannot adequately explain the long-term eutrophication trend in the Baltic (Shaffer 1979), and almost another decade for this conclusion to become fairly generally accepted (Launiainen et al. 1987, Nehring et al. 1987). The increased nutrient load on the Baltic, no matter which of the recent estimates you find more convincing, seems to provide an explanation, if as yet only qualitative, of the long-term eutrophication trend.

LIMITING NUTRIENTS IN THE BALTIC SEA

The available evidence on limiting nutrients in the Baltic Sea has been reviewed by Larsson (1986), and further supporting evidence is presented by Granéli et al. (in press). The evidence derives from several types of measurements, and the conclusion is clear: In most of the Baltic Sea, nitrogen is normally the most limiting nutrient in the open sea area, and in coastal areas unaffected by local nutrient discharge. The Bothnian Bay differs sharply in having phosphorus as the most limiting nutrient, and the Bothnian Sea may be regarded as a transition area. There are reports suggesting that phosphorus may also be the most limiting nutrient in the Gulf of Riga (Nehring et al. 1987).

In coastal areas with large local nutrient loads, the situation may differ. Near the effluent release from large sewage treatment plants with efficient phosphorus removal, phosphorus is often the most limiting nutrient, e.g. today in the innermost Stockholm archipelago, where the runoff from Lake Mälaren brings in a large nitrogen surplus, in addition to that in the sewage from over a million inhabitants, from which most of the phosphorus has been removed (Brattberg 1986), and a similar situation is found in the nearby Himmerfjard area, at the outfall from a modern sewage treatment plant, treating the effluents from 240,000 people (Elmgren and Larsson 1987).

The limiting nutrient is not always the same for all algal species , and the relative availability of different nutrients may lead to changes in the species composition of the flora, not only to a limitation of the primary production (c.f. Tilman 1982, Tilman et al. 1986). Studies in the northern Baltic proper suggest, however, that the benthic macroalgal community as a whole tends to be limited by the same nutrient as the phytoplankton (I. Wallentinus, in press). An exception to the general nitrogen limitation in most of the Baltic Sea is the colonial heterocystous cyanobacteria, which are an important component of the Baltic phytoplankton, at least in warm summers. The ability of heterocystous cyanobacteria to fix atmospheric nitrogen normally makes phosphorus their most limiting nutrient (Brattberg 1986).

The Baltic is naturally rich in silicate, and silicon has been regarded as always present in surplus (Melvasalo et al. 1981, Nehring et al. 1987). However, silicate concentrations seem recently to have decreased in some Baltic areas (Kiel Bight: Babenerd and Zeitschel 1985; possibly the Tvärminne area: Hällfors et al. 1983), and may at times become almost depleted, especially in eutrophicated areas, such as the Himmerfjard

(Elmgren and Larsson 1987). A slow silicate depletion of the Baltic due to eutrophication, similar to that inferred for some of the Great Lakes of North America (Schelske and Stoermer 1971, Schelske et al. 1986), may be suspected. This may have large future consequences, if diatoms are replaced by other phytoplankton groups, that do not require silicon for their growth. As yet, silicon has, however, never been reported to be the most limiting nutrient, even for diatoms, in the Baltic Sea.

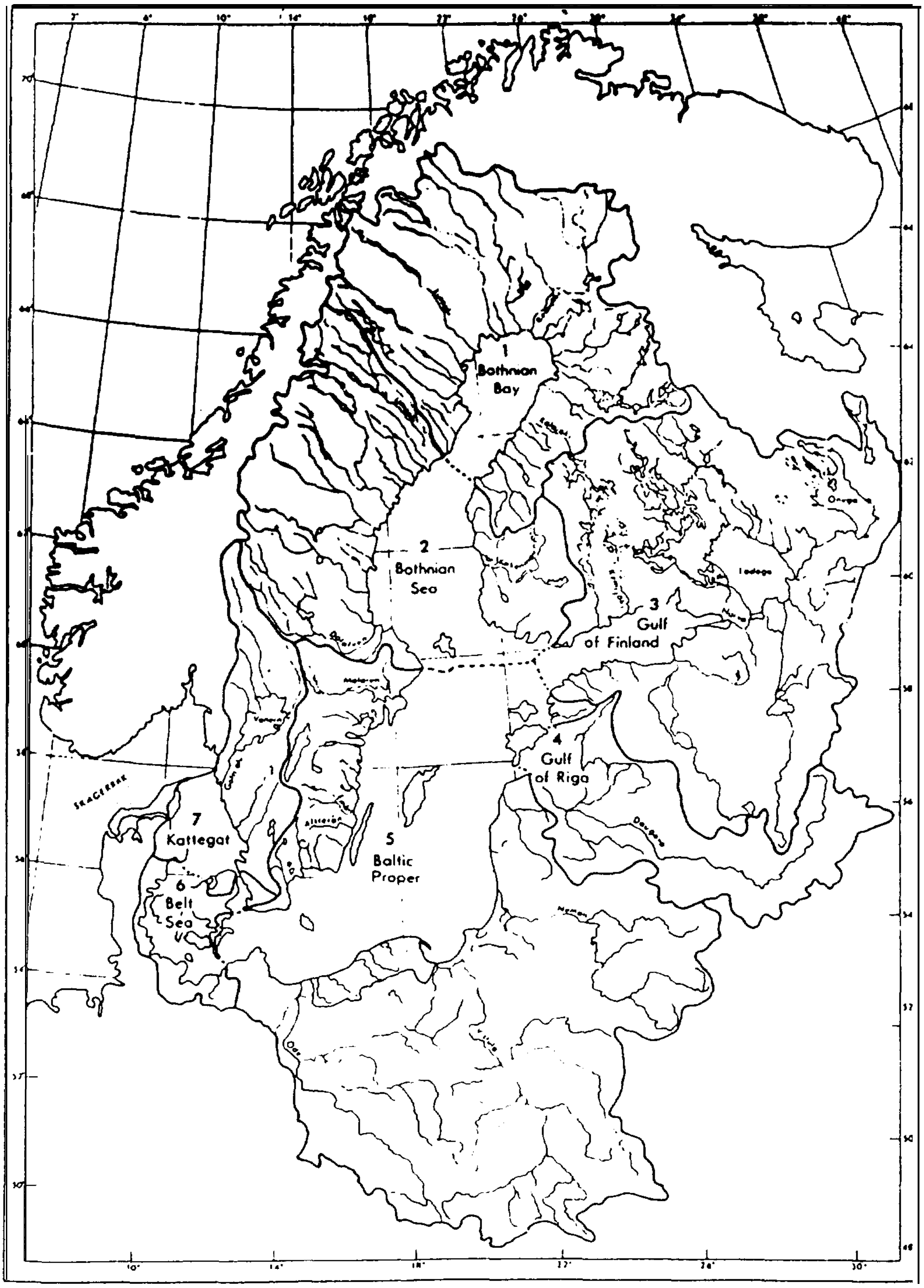

Fig. 2. Drainage basin and subregions of the Baltic Sea and its transition area (Falkenmark and Mikulski, 1974). Boundaries between Baltic Sea subregions (dashed lines). Boundaries between the corresponding drainage basins (thick lines).

NUTRIENT LOAD ON THE BALTIC SEA

Nutrient input

The drainage basin of the Baltic Sea (Fig 2) has an area about 3.5 times larger than the Baltic itself. Based primarily on documents submitted by the Baltic countries to a working group under the Helsinki Commission, Larsson et al. (1985) presented an estimate of the nutrient load on the Baltic Sea. The figures are given in Tables 3 and 4 together with the estimates later presented by the HELCOM Working Group itself (HELCOM 1987b). The HELCOM total values are considerably lower, especially due to the lower values for the Polish discharges. On the other hand Sweden, Finland and Denmark have revised their release data upwards since Larsson et al.(1985) made their calculations. A major difference between the two data sets is, that Larsson et al. tried to estimate missing data if they could find a reasonable basis for doing so, while in the HELCOM data set, it seems that a lack of knowledge often became mathematically equivalent to zero pollution load.

Trzosinska, in Nehring et al. (1987) quotes Polish multi-year mean estimates of the load from the river Vistula alone of $94\cdot10^3$ t/a of nitrogen and $6.4\cdot10^3$ t/a of phosphorus. These figures suggest that the original Polish report to HELCOM, used by Larsson et al. (1985), may have been too high, especially for phosphorus.They also indicate that the HELCOM (198713) estimate of the nitrogen load from Polish rivers is too low, since it is lower than the figure quoted above for the Vistula alone. Thus, it is possible that the nutrient loads from Poland may have been overestimated by Larsson et al. (1985), whereas both they and HELCOM have almost certainly greatly underestimated the nutrient load from the Soviet Union. Satellite images confirm the presence of large phytoplankton blooms, not only off the Polish coast, but also off the Soviet coastline in the southeastern Baltic proper (Horstmann et al. 1986), an area for which HELCOM (1987b) presents very low nutrient load estimates. That they are too low, is clear from the extremely low areal load (for nitrogen) and per capita load (for phosphorus) from the Soviet Union, calculated in the HELCOM (198713) compilation. Further support can be found in the much higher estimates for the Soviet Union published by a Soviet researcher (Maksimova 1982, see Table 4), who also estimates loads from Poland, intermediate between Larsson et al. (1985) and HELCOM (1987b).

		Larsson et al. 1985		HELCOM 1987b	
		N	P	N	P
Land discharge:	Rivers	641	50	403	27
	Coastal municipal	87	18	60	10
	Coastal industries	14	4	11	8
Atmospheric deposition		314	6	398 1)	6
External load subtotal		1056	78	872	51
Nitrogen fixation		134	-	134 2)	-
Total load		1190	78	1006	51

1) Estimated from Fig. page 41 in HELCOM 1987b.
2) Estimate from Larsson et al. 1985. HELCOM gives no estimate.

Table 3. Total nutrient load on the Baltic, including the Belt Sea and the Sound. Exchanges with the Kattegat are not taken into account. Figures give 10^3 t/a and are rounded to nearest integer.

Altogether this underscores the uncertainty of all load estimates for the Baltic Sea. I will use the Larsson et al. (1985) estimate as the basis for my discussion, since it is probably still the best available estimate of the total Baltic nitrogen and phosphorus loads. It may, in fact, still underestimate the real load on the Baltic Sea, at least for nitrogen. Larsson et al. (1985) also made rough estimates of how much the present nutrient load had increased, compared to a background load, representative of the end of the last century. They suggested that the phosphorus load had increased about eightfold and the nitrogen load fourfold. The increase must have been larger in the south and smaller in the north. Most of the increase must have taken place after the Second World War, probably accelerating after about 1955-1960.

	Maksimova 1982		Larsson et al. 1985		HELCOM 1987b		
	N	P	N	P	N	P	Population
Sweden	60	4	77	5	94	6	6
Finland	46	3	51	3	70	4	3
Soviet Union	236	19	149	9	130	6	20
Poland	166	19	386	43	110	19	33
GDR	no estimate		6	2	4	<1	2
FRG	no estimate		22	3	16	3	1
Denmark	no estimate		50	7	51	8	3
Total	(599)	(60)	741	72	475	45	70

Table 4. Land discharges of nutrients to the Baltic Sea, including, except for Maksimova's estimate, the Belt Sea and the Sound. Maksimova's figures for individual countries refer to river discharges only, but her total includes groundwater and direct pollutant discharges from the coast. Load figures in 10^3 t/a. Population is in millions and refers to the drainage basin of this part of the Baltic Sea area. All figures rounded to nearest integer.

Fate of the nutrients

It is possible to give only very rough estimates of where the nutrients added to the Baltic end up, since uncertainties in many terms tend to compound. The estimates given below are based on Larsson et al. (1985), Rönner (1985) and the review by Larsson (1986):

Phosphorus (P): ~ 10% is exported and removed in fish catch
~ 20% accumulates in the water column
~ 70% is deposited in the sediment
(calculated by difference)

Nitrogen (N): ~ 10% is exported and removed in fish catch
~ 10% accumulates in the water column
5-10% is sequestered in the sediments
70-75% is denitrified to molecular nitrogen,
(calculated by difference)

The two main conclusions are:
1. Most of the phosphorus load ends up in the sediment.
2. Most of the nitrogen load is denitrified.

These conclusions emphasize the importance of sediment-water interactions in water bodies with long hydraulic residence time, such as the Baltic.

The phosphorus deposited in Baltic sediments is not permanently removed from circulation, but may be mobilized into the water column again, should the water above the sediment turn anoxic (Fonselius 1969, Holm 1978), or if a layer of decomposing spring bloom algae settles onto the sediment, making its former surface anoxic (S. Blomqvist, pers comm.). It is not known how long the sediments will be able to absorb such a high proportion of the phosphorus inputs to the Baltic. Precipitating hydroxides of iron and manganese are thought to scavenge phosphate from the water column, but the stoichiometry of the process is not known, and predictions on the ultimate absorptive capacity of Baltic sediments cannot yet be made.

That denitrification, mostly in the sediments, is a major sink for nitrogen is supported by calculations by Shaffer and Rönner (1984), who estimated that 470 x 10^3 tons/year of combined nitrogen were returned to molecular nitrogen in Baltic proper below-halocline waters alone. A further 23 x 10^3 tons/year were supposedly lost as gaseous N20, also produced mostly during the nitrification-denitrification process. Denitrification thus acts as a "self-cleaning" mechanism in regard to the nitrogen load on the Baltic (Rönner 1985).

The assimilative capacity of the Baltic for nitrogen has, however, clearly been exceeded in recent years, as shown by the increased nitrogen concentrations in the water column. Furthermore, there are indications that the proportion of supplied combined nitrogen that is denitrified in a marine sediment may decline as the load increases (Seitzinger and Nixon 1985). This is probably due to to less efficient nitrification-denitrification pathways in sediments devoid of bioturbation and water-pumping macrofauna (Henriksen et al. 1980, Enoksson 1987), but competition for nitrate from dissimilative reduction of nitrate to ammonia may also be suspected in carbon-rich, oxygen-poor environments (Koike and Hattori 1978). Thus the relative assimilative capacity for nitrogen may actually decrease as the nitrogen load increases.

OUTLOOK FOR THE FUTURE

Given the uncertainties in the nutrient load, the fate of the nutrients and the mechanisms by which they are largely removed from the water column, it is hard to say whether the Baltic today is near a steady state in relation to its nutrient load, or whether the eutrophication process will continue, even if the nutrient load does not increase. We must suspect the latter, especially as regards phosphorus.

Whether the nutrient load on the Baltic can be decreased , or even prevented from increasing even further, depends on the political priority given this effort. It is uncertain whether the load is still increasing. Some data indicate an unchanged atmospheric deposition of nitrogen over Sweden since the mid-1970's (Rohde and Rood 1986, Granat 1986), and decisions already taken in western Europe concerning catalytic conversion of car exhausts give reason to believe the trend may even be reversed in the next decade. Further such decisions are needed also in both eastern and weatern Europe.

The other large source of nitrogen to the marine environment is agriculture, both through ammonia emissions to the atmosphere and through run-off from land. The nitrogen fertilizer use in agriculture stagnated in many western countries after the 1979 oil price increases, but are likely to increase again, due to recent oil price decreases, unless changes in agricultural policy intervene. In Sweden and Denmark, political decisions to reduce these emissions have been taken, and only the mere practical details of how really to bring this about remain mostly to be decided. Hopefully, within the next decade, these decisions will begin to take effect.

A look at Table 3 shows, however, that what happens in Poland and the USSR will largely decide whether the eutrophication of the open Baltic can be halted. Both these countries have major economic problems, making investment in environmental protection technology difficult. The low productivity of the agricultural sector is a major facet of the economic problems of both countries, and increased use of fertilizer has long been an important means whereby they have sought to increase agricultural productivity (Soviet Union:Medvedev 1987). Poland also has extremely urgent problems with heavy metal and other air pollution in large parts of the country, problems thought responsible for major acute health effects in the population (Kabala 1985, Kramer 1987). Measures against such problems may well be judged more urgent than protecting the Baltic from eutrophication.

Concerning phosphorus, Andersen (1986) estimated that if all Baltic countries removed 80% of the phosphorus from point sources, a lower efficiency than normal in Sweden today, the phosphorus load to the Baltic could be reduced by more than half. The phosphorus load from Sweden to the Baltic has decreased in recent years, due to improved municipal and industrial sewage treatment (Anon. 1987). Measures against agricultural (erosion) releases could decrease the load even further. If phosphorus concentrations in the water of the open Baltic in summer are allowed to increase further, the result may be even larger blooms of nitrogen-fixing blue-green algae than today. Such blooms, which at times are toxic (Edler et al. 1985), were a prominent feature in the eutrophicated inner archipelago of Stockholm, before phosphorus reduction was implemented in the local sewage works, but disappeared when phosphorus discharges were reduced (Brattberg 1986).

In partly enclosed coastal areas around the Baltic, local discharges are normally the main water quality determinants, and the fact that local discharges are only a small fraction of the total load on the Baltic is not a valid argument against discharge reductions. Such reductions will often, if carefully calculated to suit the local area, result in considerable, and rather rapid (a few years) local water quality improvement, as demonstrated in the Stockholm Archipelago (Brattberg 1986). They Will also be of some benefit for the open Baltic, and this should always be considered in the planning.

Only in a longer perspective (decades) are improvements possible also in the open Baltic Sea, given the necessary political will in all Baltic countries. An improved scientific understanding of marine eutrophication would be of great help in fostering such a development. Careful scientific investigations of the effects of the discharge reductions that are now being implemented will both demonstrate the effectiveness of reductions, and help us better predict the effects of further, future reductions. Improved basic understanding of the origin of algal blooms, including why some of them are toxic, and of the basic biogeochemical cycling of nutrient are needed, to provide a better basis for managing the Baltic Sea in the future.

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THE QUALITY OF URBAN STORMWATER : A STATE-OF-THE-ART REVIEW

Professor J Bryan Ellis*

ABSTRACT

A review of pollutant sources and loadings discharged from both separate (SWO) and combined (CSO) sewers in urban catchments is made and scouring/resuspension of contaminated in-pipe and in-stream sediments is identified as a significant factor in the poor quality of urban sewer discharges. Receiving water impacts are discussed in terms of both acute and chronic timescale effects and exceedance criteria advocated as a basis for toxic bioassay analysis. The significance of both exposure and recurrence time for the establishment of wet weather criteria is stressed as well as the speciation of the toxicant.

Keywords: Urban Runoff; Pollutant Sources and Loadings; receiving water impacts; wet weather criteria; bioassay toxicity.

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1. INTRODUCTION

During the last decade most European countries have achieved an increasing control over their point-source discharges from municipal treatment works and industrial plants. However, despite the very large sums of money that have been invested in new or upgraded treatment plant as well as in sewer renovation and replacement, many member states have not experienced as much corresponding receiving water quality benefits as they might have expected. In the UK, over 10% of all receiving waters (or 4210 km of river length) are currently in a state ranging from "polluted" to "grossly polluted". The proportion falling into these two classes is now showing an increase for the first time since national river quality surveys began some thirty years ago. A major reason for the persistence of these poor quality waters is the continued existence of uncontrolled or poorly controlled discharges from separate stormwater sewer outfalls (SWO's) and combined storm sewer overflows (CSO's).

Water quality problems resulting from urban runoff are now a top priority in most European countries. In the UK between 35 to 40% of CSO's are deemed to be unsatisfactory and causing serious short term, slug effects as well as longer term chronic impacts. The transient, randomly distributed SWO discharges from non-point urban sources also present a major control problem. Lack of hydraulic capacity and transporting efficiency in the existing urban drainage system can frequently lead to surcharging and local flooding as well as causing both in-pipe and in-stream pollution problems. The consequent flood and pollution alleviation works involve water authorities in major expenditure on sewerage systems which in the UK is estimated at being £100 M/year for flood avoidance and £40 M/year for pollution prevention. The annual costs for urban sewer management in Norway are placed at £30 M/year, whilst in Sweden sewer investment currently amounts to some £38 M/year with operating and maintenance costs estimated at about £40 M/year, which rises to nearly £55 M/year in W. Germany.

2. SEWER SYSTEMS AND LOADS

Most European countries have a very high percentage of their population connected up to a main sewer system, although as 'indicated in Table 1, the relative percentage of separate and combined sewers does vary quite considerably as does the age of the respective sewer systems.

TABLE 1 European sewer systems

Country	Percentage of Populatior Connected to Sewer	Percentage Combined	**Age**
UK	96	70	50% after 1945
Sweden	94	14	50% after 1964
Denmark	87	37	65% after 1955
Netherlands	86	90	
France	91	85	48% after 1950
W Germany	90	60	60% after 1963
Spain	52	02	50% after 1950
USA	77	25	

(After Fiddes [1])

Most new residential developments in urban areas are now designed on the separate principle, despite the proven poor quality of SWO discharges and the fact that only 2-5% of wrong connections will entirely eliminate the water quality benefits of such separation. Table 2 gives an indication of the relative pollutant concentrations and loadings associated with separate surface water and combined storm sewerage systems in W Europe.

TABLE 2 Mean Pollutant Concentrations and Unit Loads for Surface Water and Combined Sewer Discharges.

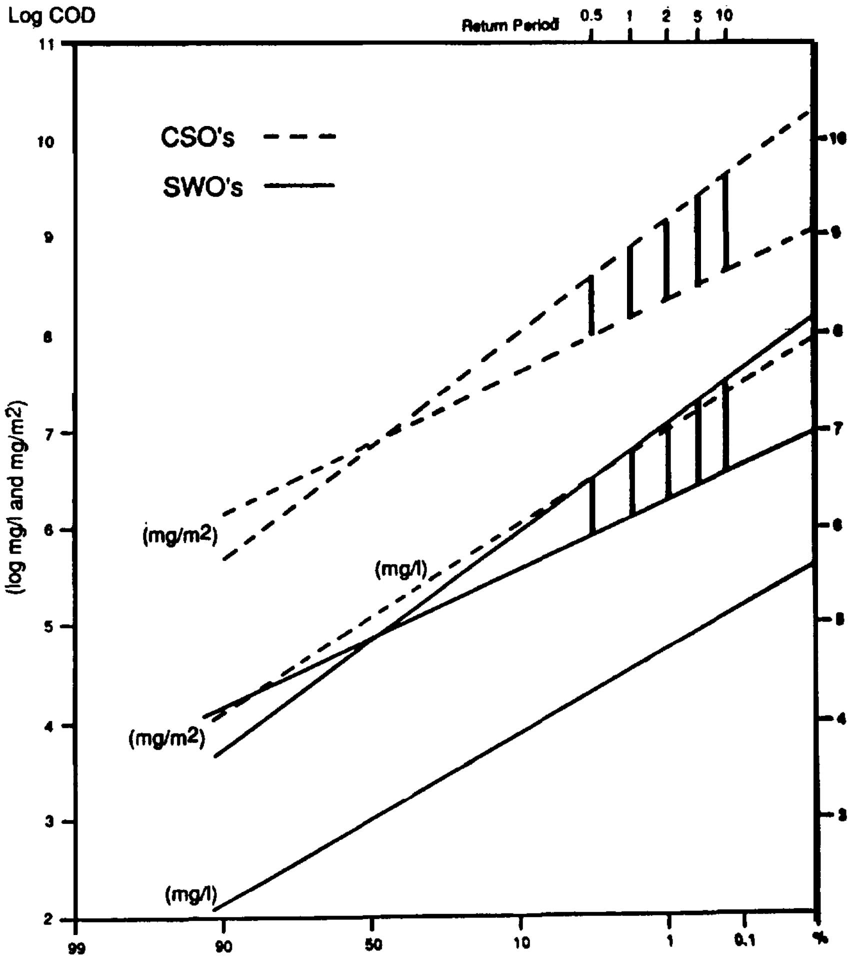
POLLUTANT PARAMETER	surf-ace Water Sewers (SWO's)		Combined Sewers (CSO's)	
	Load per Unit Area (kg/imp.ha/yr)	Event Mean Concentrations (mg/1)	Load per Unit Area (kg/imp.ha/yr)	Event Mean Concentrations (mg/1)
Suspended Solids	487 (347 - 2340)	190 (21 - 2582)	1260 (80 - 4900)	425 (176 - 647)
Biological Oxygen Demand	59 (35 - 172)	11 (7 - 22)	826 (500 - 1400)	90 (43 - 225)
Chemical Oxygen Demand	358 (22 - 703)	85 (20 - 365)	1600 (62 - 3800)	380 (250 - 530)
Ammoniacal Nitrogen	1.76 (1.2 - 25.1)	1.45 (0.2 - 4.6)	68.0 (15 - 85)	6.0 (3.1 - 8.0)
Total Nitrogen	9.0 (0.9 - 24.2)	3.2 (0.4 - 20.0)	20 (8 - 40)	8.3 (21 - 28.5)
Total Phosphorus	1.8 (0.5 - 4.9)	0.34 (0.02 - 4.30)	4.1 (2.2 - 8.8)	10.0 (6.5 - 14.01
Total Lead	0.83 (0.09 - 1.91)	0.21 (0.01 - 3.1)	1.25 (0.83 - 1.84)	0.25 (0.08 - 0.45)
Total Zinc	1.15 (0.21 - 2.68)	0.30 (0.01 - 3.68)	5.33 (4.06 - 6.61)	0.81 (0.10 - 1.07)
Oil		0.4 (0.09 - 2.8)		
Faecal Coliforms (E.Coli)	2.1 (0.9 - 3.8) (x10^9 counts/ha)	6430 (400 - 50,000) (MPN/100ml)		(10^5 - 10^8) (MPN/100ml)

(Figures in parentheses give range of observed mean values)

(After Ellis, 2)

With the exception of solids and metals, CSO pollutant concentrations exceed those of SWO's by about an order of magnitude although it should be noted that the COD:BOD ratio of stormwater runoff is much higher than that of combined sewerage discharges.

The event-mean concentration (EMC) values show considerable variability not only between events but also from one site to another. Coefficients of variation (Cv) for most pollutant parameters typically range between 0.5 and 1.0 increasing to 1.5 - 2.0 for solids and solid-associated parameters. Therefore whilst acute impacts are best evaluated on the basis of extreme values for the individual storm event, the accumulative, long term chronic effects are best considered in terms of the annual loadings. EMC values appear to be typically log normally distributed as can be seen from inspection of Fig. 1 where the statistical distribution of COD per storm event and unit catchment area is plotted for four Danish urban catchments. It should be noted however that a 3 parameter log normal distribution using a 'maximum likelihood' fitting procedure appears to give the best SWO data fit although an extreme value Frechet (EV2) distribution has also been shown to provide a good fit for organic demand parameters (Ellis, 2). Harremoes [3] has stressed the importance of taking into consideration the variability of the concentration in order to derive reliable predictions of pollutant discharges. The distance between the curves on Fig.1 can be used to determine the correction factors (which vary between 2 to 3) for different return periods. However the similarity, and indeed overlap, of the SWO pollutant distributions to those of CSO's would imply a widespread, even if limited, degree of cross-connection between these systems. Given that only a few percent of such cross-connections are necessary to negate any water quality



(After Harremoes, 3)

Fig. 1 - Statistical Distribution of SWO and CSO Pollutant Discharges.

benefits resulting from separation, it could justifiably be argued that the first and best management practise for urban runof quality control should be to ensure the elimination of all illegal connections to the surface water sewer.

Biomonitoring within urban catchments suggest that the magnitude of CSO/SWO concentrations and loadings is sufficient to provide the critical limiting factor for receiving stream ecology and would further imply a lower quality classification than that derived from chemical sampling. Therefore, even given full rehabilitation or renovation of CSO's, the remaining acute effects of SWO discharges combined with the longer term, delayed impacts due to overall pollutant sedimentation and adsorption, might well moderate and suppress many of the expected in-stream quality benefits.

The growth of an informed environmental lobby and the introduction of complex and stringent national and EC pollution control acts and directives, has meant that most European countries have instituted national programmes to review the scale and magnitude of the urban runoff problem and to identify cost-effective management solutions. Whilst none may have been as comprehensive as the US Environmental Protection Agency NURP or National Urban Runoff Program, the UK River Basin Management Programme (RBM, 1986-89), the French National Programme on Runoff Pollution (LHM, 1983-90) as well as the Swedish Environmental Protection Board, Stormwater Management guidelines (SNV, 1978-83) are all intended to develop appropriate technical procedures and tools for the control of urban sewer flow quality. More recently the newly constituted Portuguese Northern Region Water Authority (PGIRHN) has commissioned a national review of urban pollution abatement technology (PO-Rivers, 1988-92) and the Italian National Research Council is also conducting a review of the operational and legislative aspects of urban sewer quality (IRSA, 1988-90). The targets for sewer upgrading would thus appear to be primarily quality based although in the UK and France avoidance of surcharging and flooding is perhaps more frequently the major consideration. This is largely because of the age of their sewer systems (Table 1), where some 5/10% of all 50m sewer lengths between manholes are categorised as being unsound with higher percentages (25/30%) occurring in the older 1860/1920 brick-lined sewers of central urban districts.

3. POLLUTION SOURCES AND LEVELS

3.1 Atmospheric Deposits and Roof Runoff

It is widely recognised that aerial deposition can contribute on average some 40/50% of nutrients and heavy metals associated with the total mass discharged from urban catchments (Ellis, 4). In areas of acidic rainfall, wet deposition will be an important source of freely dissolved species which will be further enriched by leaching and solubilisation of the exchangeable fraction contained within impermeable surface dusts. Additionally, where copper and zinc are used in roofing and guttering as in Scandinavia, corrosion can elevate the average metal contributions to between 70-90% of the total mass discharged (Malmquist, 5; Hogland, 6). Their short times of concentration (3-5 minutes), also mean that roof runoff flows represent significant components of the early and peak discharges recorded in storm drains. For rainfall volumes between 0.25 - 0.6 mm it is probable that only roof surfaces contribute to sewer runoff (Pratt et al., 7). Whilst disconnection of roof downpipes can attenuate and reduce storm sewer peak flows by 10-30% and total runoff volume by up to 50%, the potential toxicity of roof discharges (Fig. 2) would suggest that on-site disposal may not be acceptable in every location. Infiltration controls for roof runoff need to be carefully installed and monitored, particularly where sensitive groundwaters may be affected.

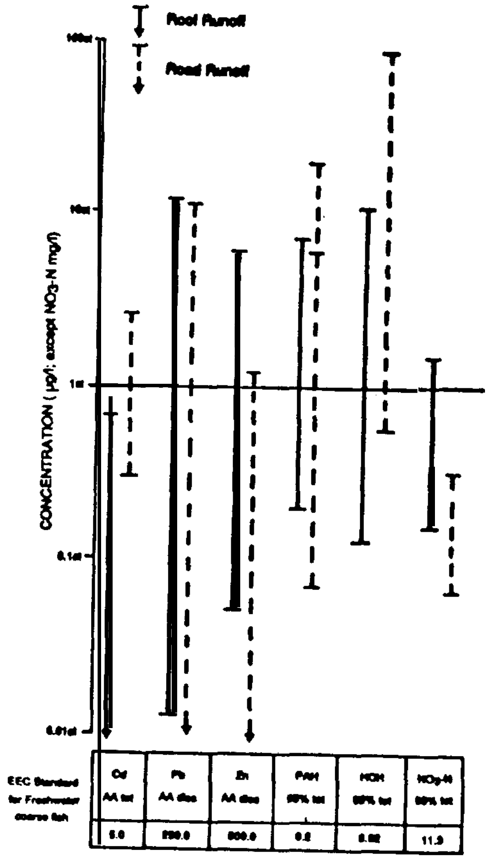
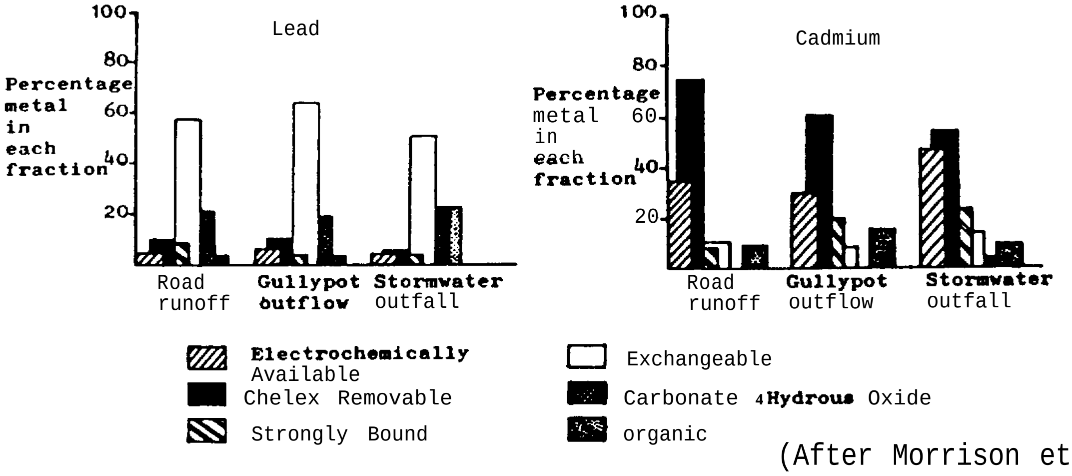


Fig. 2 - Runoff Exceedance of EC Quality Standards.

3.2 Highway Runoff

Runoff from impermeable highway surfaces can also be highly contaminated (Fig. 2) with accumulated mass rate of pollutant[3] being a function of aerial deposition rate, average daily traffic density and intensity-frequency of the rainfall event (Ellis, 4). Speciation studies show that substantial proportions of toxic metals in road runoff are in a potentially bioavailable form (Fig. 3) with as much as 50-60% of Pb being in the exchangeable fraction and 30-37% of Cd occuring as a freely dissolved species.



(After Morrison et al.,18)

Fig. 3 - Stormwater Runoff Metal Speciation.

Modelling studies would suggest that impermeable urban surfaces virtually provide an unlimited sediment store for washoff (Ellis et al., 8), such that surface accumulation can be adequately represented by a simple linear function:

P_M = $Po + K_1.dt.$

where P_M = pollutant mass per unit highway surface at time t (g/km)

$P0$ = residual on surface prior to time t (g/km)

K_1 = accumulation rate (g/km/day)

dt = time of accumulation (day)

The value of K_1, the accumulation rate, can be estimated from the average daily traffic density (ADT) as being equivalent to $2ADT^{0.89}$.

The high ionic strength and reduced pH of snowmelt runoff can liberate weakly associated species from the fine solids phase to elevate metal and dissolved organic fractions in the runoff by an order of magnitude. As much as half of the yearly pollutant mass transport can occur during intensive snow melting periods (Morrison et al., 9; Lygren and Gjessing, (10).

Although physically based simulation models based on sediment transport theory are being developed for the prediction of surface washoff rates (Moys, 11; Svensson, 12), they do require verification, calibration and sensitivity analysis to be reliably and widely applied as stormwater management tools.

Empirical first-order washoff equations still provide an appropriate basis for many operational modelling approaches and are of the general form:

$$dP/dt = P_M (1 - e^{-K_2 . v})$$

where dP/dt = pollutant mass discharged during runoff event (g/km)

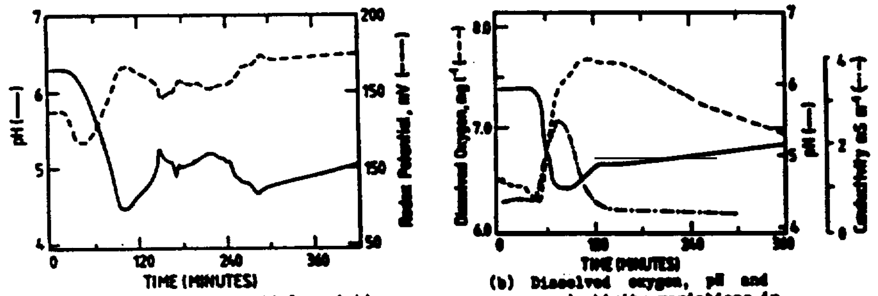
K_2 = washoff coefficient (hr/m), which varies between 5-12 for highly impermeable and semi permeable rural areas respectively.

v = average runoff rate (m/hr)

The above formulation and its many variants have been fully discussed by Huber (13) and the exponential function has been adopted by many European workers.

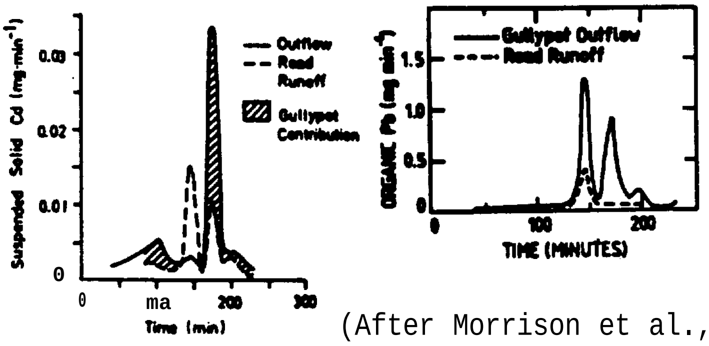
3.3 Roadside Gullypots and Inlets

Roadside gullypots (or catchpits) have been identified as a significant source of the poor quality of SWO discharges (Morrison et al., 14). Gullypot metal outflows for example, show substantial (and often early) contributions from the anoxic supernatant liquors and interstitial gullypot waters (Fig. 4), reflecting anaerobic degradation processes which act upon the trapped chamber sediments both during and between storm events. The disturbance and mobilisation of the interstitial sediment waters during road runoff are clearly indicated in Fig. 4b by the rapid changes that can be observed in the controlling parameters within the gullypot. Although these controlling factors can produce large changes in metal speciation during specific storm events, the overall effect on dissolved metals for complete storms does not appear to be of major significance (Fig. 3). Increases in dissolved organic carbon within the gullypot can lead to enhanced metal complexation as can be seen in the case of cadmium in Figure 3.



(c) pH and redox potential variations in gullypot liquor

(b) Dissolved oxygen, pH and conductivity variations in gullypot liquor.



(After Morrison et al., 14)

Fig. 4 - Variations in Cullypot Quality.

3.4 Sewer Deposits and Flushing

The greatest contributions to both total SWO and CSO pollutant loadings (50-60%) are undoubtedly derived from the scour and release of in-pipe slimes and sediments. An increasing number of European studies are showing that polluted sediments in the sewer pipe can be very considerable, amounting to some 20-44% of the in-line storage capacity. Not only do such sewer sediments restrict the hydraulic capacity and conveyance efficiency causing surcharging and premature SSO spillage but they also have a considerable acute, shock-loading effect upon the receiving water resulting from the 'first-foul' flush of contaminated solids and oxidisable/soluble organics.

Table 3 provides a pooled estimation of the equivalent annual loads derived from pipe deposits within CSO catchments in Scandinavia and Switzerland which collectively amount to 50/60% of the total mass loads discharged.

TABLE 3 In-pipe pollutant loads

Parameter	Load (kg/ha/yr)
BOD_5	2.2 - 8.8
COD	270 - 960
N_{total}	10.2 - 15.7
SS	340 - 1255

The BOD strength (g/kg O) of the cohesive, organic sludge layers found in combined sewers can reach 10-20 g/kg 0 with 4 hour BOD equivalents of about 10-15 g/kg per metre of pipe length and possessing critical shear strengths typically in the range of 10 - 100 KN/m^2 (Crabtree, 15).

The key criteria for the initiation of bed scour and mobilisation of the surficial sewer sediment layer will therefore depend on rheological characteristics such as sediment bulk density, cohesion and plasticity. Consideration must therefore be given to the critical flow and shear velocities for the various particle size ranges and sediment concentrations found in sewer deposits as well as their interstitial ionic strength and degree of cohesion. The work of Stotz and Krauth (16) as well as Lindholm and Aaby (17) has confirmed that whilst a critical shear stress of 1.8 KN/m^2 may be sufficient to washout recently deposited, unconsolidated sediment, residence times of 2-8 days will increase this value to 3 and 5 KN/m^2 respectively for the same flushing rates and volumes. These shear strengths would suggest a minimum sewer velocity of about 1 m/s is needed to maintain a self-cleansing capability in small diameter pipes.

In the UK, current recommendations are for self-cleansing velocities of 0.7-0.9 m/s at pipefull flows and assumed shear stress values of 6.2 KN/m^2. In Scandinavia and W Germany rather higher pipefull velocities of 1.5 m/s are recommended but both assume shear values of only some 2-4 KN/m^2 for this design. All quoted values fall well below the 10-100 KN/m^2 required shear stress threshold levels observed in the UK field studies referred to above and which have been independently confirmed by the French CERGRENE work being undertaken in Paris and Marseille. It would appear that for large pipe sizes above 1 m diameter, a critical minimum flow velocity of at least 1.5 m/s is required to maintain a reasonably satisfactory self-cleansing situation. Current sewer transport design is based on non-cohesive materials, with low sediment concentrations (< 100 mg/l) and for medium to coarse grain sizes (l-2 mm). Diameter-related velocity and shear design criteria need to be developed for the cohesive, fine grained sediments and high concentrations typically found in urban sewer pipes. Other parameters of design and operational interest would include shear stress values for DWF loads as well as frequency function curves (FDC's) for the distribution of shear stress along the sewer pipe length or perhaps for 10 and 50% of the length. Similar FDC's might also be plotted for slope as well as pipe diameter.

Both SWO and CSO discharges tend to display a characteristic first-flush which frequently coincides with the time of concentration for the drainage system and which primarily reflects the re-entrainment and transport of in-pipe contaminants. The first-flush phenomena is undoubtedly less marked in large catchments above 250/300ha in size and for storm events possessing low, uniform runoff rates. Delayed subsidiary concentration peaks can also be frequently detected, which have been taken to reflect the input of fresh surface sediments and pollutants that have been washed into the sewer system. It should be noted however that there is considerable variation in the temporal distribution and magnitude of pollutant concentrations depending upon catchment and sewer conditions, the antecedant dry period, the characteristics of the storm and flow event as well as upon the nature, age and incidence of in-pipe sediments. The nature of the chemical species involved and their solubility potentials also have a marked affect upon the magnitude and extent of the first-flush. Total lead and copper for example show a much more pronounced first-flush than their dissolved or exchangeable species. Total lead and copper, both of which have strong affinities for solid-associated phases (Fig. 3), also possess a greater tendency for first-flush than total zinc or cadmium (Morrison et al., 18). Cumulative mass loading curves indicate that on average only about 42% of solids, 35% of the organic demand, 24% of nutrients and 18% of metals are discharged during the initial 30-35% of the cumulative storm volume (Ellis, 4; Geiger, 19). There is therefore considerable uncertainty and difference of opinion as to the overall significance of first-flush as a sewer flow quality phenomena as well as a lack of knowledge about its relative importance in terms of receiving water impacts.

4. RECEIVING WATER IMPACTS

4.1 Time Scale Effects

Urban drainage has conventionally been considered as an engineering problem essentially concerned with hydraulic sewer design, but the pollution problems associated with sewer discharges have shifted the objectives of many European water agencies to optimise operational performance in terms of receiving water impact. Water quality responses to SWO and CSO discharges are measurable on both short (acute) and long term (chronic) timescales. The accumulation of sediments, containing elevated pollutant gradients, adjacent to and downstream of the sewer outfall can result in localised acute effects following their scour and resuspension under storm flow conditions. In addition they impose severe longer term delays on stream recovery rates and the chronic bioaccumulation of toxic materials induces permanent alterations in the aquatic biological community and ecosystem.

These delayed impacts are associated with sediment-accumulated nutrients, metals, hydrocarbons and bacteria as well as being the result of benthal sediment oxygen demands. Typical undisturbed SOD levels vary between 0.15 - 2.75 g/m^2/day, which can account for a permanent deficit of about 1.5 - 2.5 mg/l in the DO regime of the receiving water. Bed disturbance during stormflow conditions can elevate the SOD levels into the range of 240 - 1500 g/m^2/day and depress normally near-saturation, in-stream DO levels to 2 mg/l or less. This depressed DO balance is further exacerbated during storm event activity by the elevated in-stream COD and NH,-N concentrations which typically peak around

250-350 mg/l and 3-4 mg/l respectively following dilution and mixing; even minor runoff events can cause river NH,-N to rise to 0.7-1.0 mg/l. The effects of these and other dissolved pollutants may well be sufficient to cause highly visible shock impacts such as fish kills and odour problems in addition to the objectionable aesthetic problems caused by solids and gross floatable materials. However, very unusual conditions would be required for there to be urban runoff related DO and NH,-N depressions which are not controlled by the flow, transport, loadings, dispersion and reaeration coefficients after the end of a storm event.

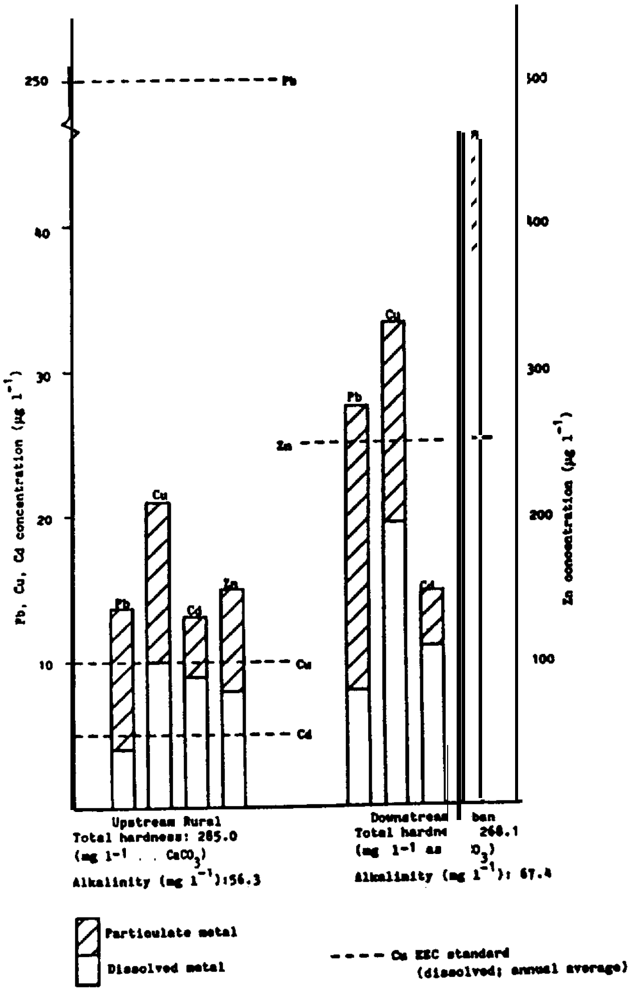
4.2 Exceedance Criteria and Bio-impacts

It is clear from inspection of Table 2 that the mean metal concentrations associated with urban discharges can be substantially in excess of existing water quality standards. Dilution ratios varying between 6:1 to 600:1 would be required to reduce the upper median values down to values that would conform with allowable annual average (AA) or maximum allowable concentrations (MAC) for freshwater life. Studies in the UK and Sweden (Morrison et al., 20) have shown that total Cd, Cu and Pb concentrations in undiluted SWO storm runoff regularly exceed US Environmental Protection Agency acute criteria (1 hour LC_{50} value) for about 75%, 70% and 20% respectively of the storm duration and exceed chronic criteria (96 hour LC_{50}) value for some 80%, 85% and 95% of the time. Violation of Zn standards is lower occuring at exceedance levels for 10% and 70% of the storm event for acute and chronic values respectively.

However, as can be seen from Fig. 3, a substantial proportion of the metal loading is labile being either in free ionic or readily exchangeable form. Comparison of the mean metal values in the dissolved and particulate phases, with the annual average EEC criteria (which are expressed in terms of the dissolved fraction), would imply that urban receiving waters are likely to permanently violate Zn, Cd and Cu standards (Fig. 5). The results clearly show that Cd and Cu present the most significant long term threat to receiving water standards, tending to consistently exceed both acute and chronic limits. Lead does not appear to pose much threat as most of the metal is strongly bound in complexation with inorganic colloids.

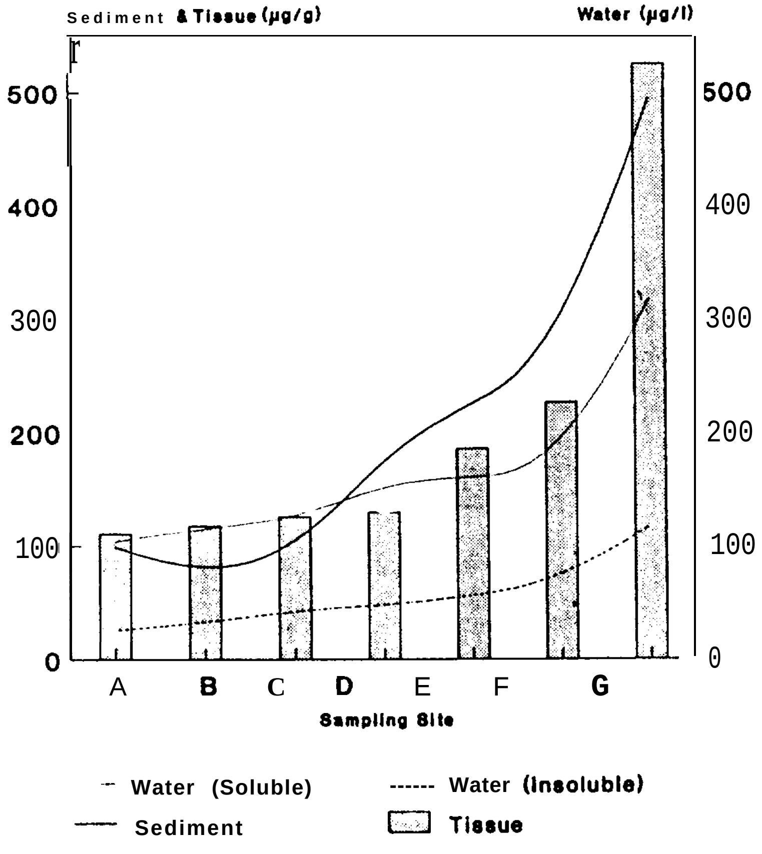
The toxic storm event exposure duration would normally be well within the acute 1 hour LC_{50} levels for the most sensitive macroinvertebrate and fish species. Not only are the discharged metal species directly bioavailable (Fig. 3), but through take-up in the sediment phase, they also become bioavailable on a much longer term timescale. Fig. 6 shows that both total water and sediment metal levels increase in receiving waters subject to urban discharges; locations E and G on the figure corresponding to site of SWO and CSO discharges respectively. Ambient water phase metals increase by some five times and sediment metal levels by up to ten times the upstream rural background levels. Benthal macroinvertebrate tissue metal levels reflect this sediment pattern, increasing dramatically in the vicinity of the CSO to reach four times the upstream levels.

The use of caged indicator species of Emphemera, Gammarus and Asellus suggest that metal uptake equilibria in the soft tissue is reached within 5-6 weeks and that Ephemeroptera and Gammarid species suffer mortality rates of about 60% and 40% respectively following storm events (Bascombe et al., 21). The species diversity indices of urban receiving waters are therefore substantially reduced (<50%) in comparison to surface waters not subject to intermittent, urban discharges. The more sensitive species such as Phyganea, Ryacophilia larva, Hydrospsyche and Polycelis disappear altogether from the biological community within the urban waters. Biomonitoring surveys of urban receiving waters reveals a very different quality situation to that implied by routine chemical



(After Bascombe et al., 21)

Fig. 5 - Metal Exceedance Criteria.



(After Bascombe et al., 21)

Fig. 6 - Variations in Metal Uptake for Water, Sediment and Tissue Phases..

sampling of the water phase. Biological index scores are frequently depressed and impoverished, being typical of polysaprobic conditions and indicating a 'poor' classification, rather than the apparently 'good' to 'fair' condition that would be assumed as a result of the chemical quality data. The use of colonisation samplers suggests that pollution sensitive species such as Gammarus and Ephemera can survive the impact of SWO and CSO discharges, if they are sealed from the uptake effects of contaminated, toxic benthal sediment.

Acute toxic criteria for these more sensitive species, as well as for salmonids and cyprinids, may perhaps be feasibly expressed by conventional 1 hour to 12 hour LC_{50} exposure (time-duration) curves, although LC_{25} or LC_{20} curves would provide an additional margin of safety. Such a safety factor incorporated into the criteria would ensure that a pollutant with a concentration equal to the standard would commonly be lethal to less than 20% or 25% of a population and would also offer some allowance for possible additive and synergistic effects. The safety margin might be stretched even further if the toxicity curves were derived from 'early life' and/or adult 'inter-moult' LC_{50} values. However, given the rather longer (<24 hour) generation and response times of many invertebrate organisms, it is probably more sensible and meaningful to relate toxicity criteria to recurrence intervals (or time between events) rather than to exposure time. This approach provides the basis for presumptive wet weather criteria being currently developed for metals by the US EPA as well as for DO by the Danish Water Pollution Control Committee. Organisms will often survive the short term, acute impact of SWO and CSO discharges, especially if the in-stream depressed DO levels are not prolonged, but will die later as a result of the combination of the acute stress and chronic exposure to toxic bed sediments.

There certainly appears to be a considerable ecological sensitivity to the varying concentrations associated with urban runoff discharges. At low concentrations or high dilutions, some workers have observed stimulatory effects and increased biomass particularly in the case of algae and zooplankton as well as with certain fish species (Lygren and Cjessing, 10). Adverse effects on growth and mortality rates, as well as the acceleration of eutrophic effects, are only felt once higher fractions of polluted runoff become mixed into the receiving water volume. The extent of accumulation and exchange reactions of nutrients across the sediment-water interface has a significant effect upon the trophic status of the receiving water. Bioassays suggest that available nitrogen, including ammonia and nitrate-nitrogen, should not exceed 60%-65%, although it must be realised that phosphorus is the limiting nutrient for algal growth in most urban receiving waters.

5. CONCLUSIONS AND RECOMMENDATIONS

Much greater attention must be given to both in-pipe and in-stream sediment criteria for evaluating receiving stream impacts of SWO and CSO discharges and water quality standards need to be specified for both acute and chronic timescales. It is necessary to generate urban runoff bioassay data for both events and sites as well as for specific indicator organisms, to develop a reliable 'whole effluent' toxicity methodology as a basis for establishing wet weather criteria. This working methodology should not only consider exposure durations of and time intervals between runoff events but should also consider the chemical form and bioavailability of the toxicant.

There is a need to develop an integrated biological and chemical monitoring protocol for long term water quality classification and cost-effective operational management of urban receiving waters. Best practical environmental options for the management of urban runoff need to seriously consider and actively realise the potential benefits that would accrue from enhanced source control as well as from more effective and efficient design of in-pipe hydraulic and storage capacity as well as overflow operation.

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QUALITY ASPECTS ON AND METHODS FOR REDUCTION OF COMBINED SEWER OVERFLOW DISCHARGE

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Abstract

Combined sewer overflow discharge is a pollution problem which has been largely neglected in most of the world. However, during the last five years extensive studies have been carried out in the Nordic countries in order to clarify CSO-problems. A Nordic cooperation project on the importance of sediment wash-out for the combined sewer overflow quality is currently under way. After a short summing up of combined sewer overflow pollution problems and a survey of the research in this field this paper presents some methods of pollutant reduction and discusses their effects.

Introduction

Combined sewer overflow (CSO) discharge means the flow from a combined sewer which exceeds the interceptor capacity and is discharged into a receiving water.

City areas in Sweden built before the 1950's are commonly drained by a combined sewer system. The rapid urbanization meant that new areas were connected to sewer nets with limited capacity leading to an increase in in-flow volumes and in the CSO pollution load to the receiving waters. This together with an extension and an improved efficiency of the wastewater treatment plants means that the relative effect of combined sewer overflow discharge has increased compared to other urban pollutant discharges.

The knowledge of CSO is usually scant which leads to large uncertainties when calculating both CSO volumes and mass transport. This lack of knowledge is serious when rehabilitation plans are to be established. Each municipality in Sweden is obliged to supply a rehabilitation plan for its entire sewer system. Uncertainties in the basis material of the rehabilitation plans implies that money allocated to rehabilitation is not used in the most economical way. Thus may e.g. improvements of the sewer systems be located at less effective sites, due to insufficient knowledge.

In 1982 studies in order to clarify the CSO situation were initiated at the Department of Water Resources Engineering, Lund. The project has been presented in several reports and conference papers, see for instance Hogland (1986), Hogland et al (1984), and Bemdtsson et al (1986).

Mass transportation via CSO

In order to calculate the CSO mass transport, it is necessary to know both the water volumes and the pollutant concentration during the CSO-event. There are five common ways of estimating the mass transportation from combined sewer overflow (see Hogland, 1986):

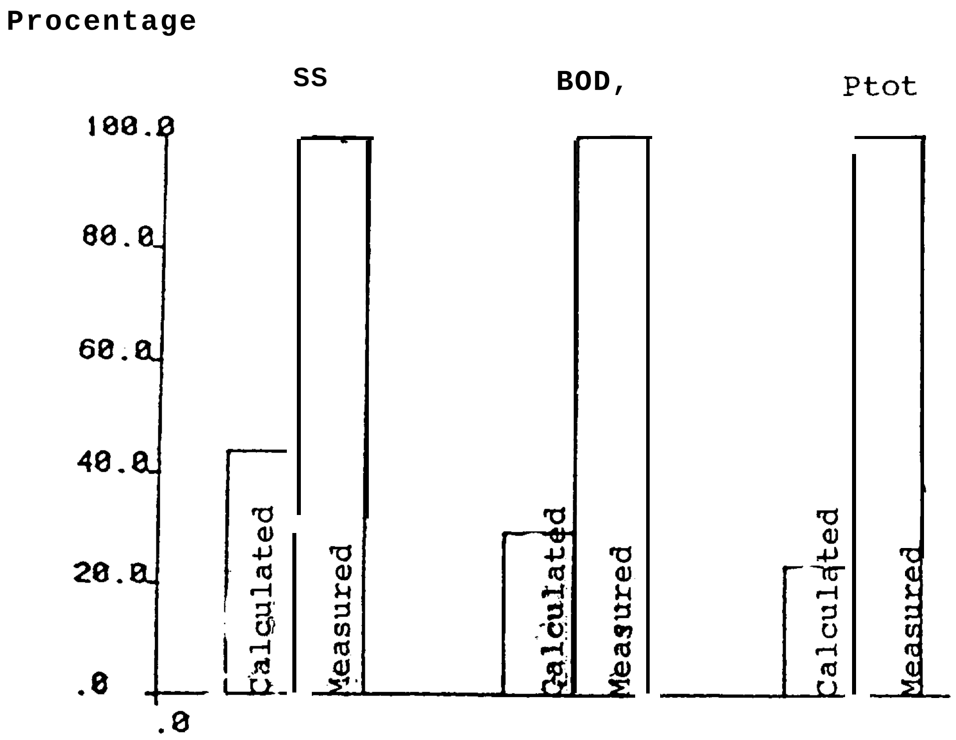
- the standard-value method
- the dilution method
- the CSO-wash-out model
- computer simulation
- measurements.

Usually the first four methods are combined with measurements in field.

The pollutants in the CSO discharge originate from three different sources during the event:

- the waste water
- the storm water
- accumulated sediments in the sewer network.

The amount of pollutants accumulated on surfaces and in the sewer network between CSO events seems to be the most important factor when it comes to determining the mass transport during short periods of time. Field studies show that the mass transport can be underestimated by a factor of 4-5; for single events when the sediment wash-out is not taken into account in the calculations (see Figure 1). The standard valve method gives reasonable results for longer periods of time. For standard values of CSO representative for Sweden see Figure 2.



Calculation method

$$C_B = \frac{C_W \cdot q_W + C_S \cdot q_S}{q_W + q_S}$$

q_W = wastewater flow (l/st)

q_S = storm water flow (l/st)

C_B = concentration of pollutant in CSO (mg/l)

C_S = " " " in storm water (mg/l)

C_W = " " " in wastewater (mg/l)

Figure 1 Pollutant transport according to the dilution calculation compared to actual measurements at a CSO-weir in Malmö.

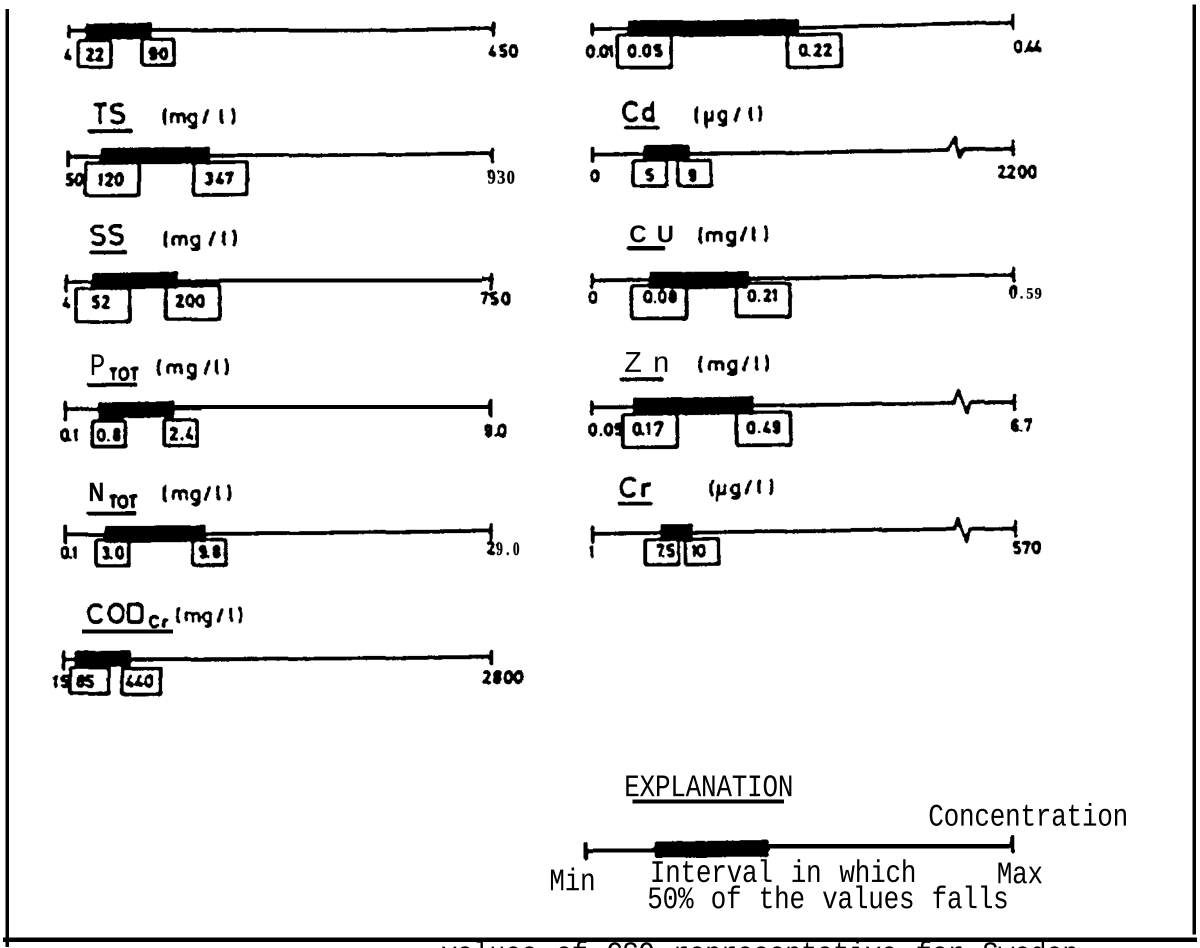


Figure 2 Standard values of CSO representative for Sweden (Hogland et al, 1984).

The pollutant concentration varies a lot from one CSO-event to another but also during one single event. Field studies show that the concentrations of the constituents decrease with time during the CSO-event.

Normally, a distribution function of the log-normal type can describe the average concentration (see Di Toro, 1983) during the CSO-event.

The differences in pollution concentration between and during CSO events can be explained by the characteristics of the rainfall, time elapsed since the preceding event, characteristics of the sewer network, and quality and quantity of the dry-weather flow.

Exceptional CSO-events with very high concentrations of pollutants can occur when accumulated sludge banks are flushed away and when the waste-water sewers are partly clogged during dry spells.

The rehabilitation plan

In order to establish a rehabilitation plan knowledge of the CSO-discharge and the function of the CSO weir is a necessity. Otherwise the assignment of relative priority to measures necessary to avoid inconveniences in connection with waste-water management cannot be worked out. A complete CSO study consists of the following phases:

1) Definition of CSO problems and the use of receiving waters.
2) Selection of type and extent of CSO-control.
3) Detailed planning of CSO-control facilities.

Effects of the CSO discharge on the receiving water must also be studied. Impacts of pollution loads have been presented by Driscoll (1981), Harremoës (1982), and Johansen et al (1984).

The present use of the receiving water as well as that planned for the future determine the amount of pollutant discharge allowed. The quality criteria for receiving waters of different size can be exemplified by the schedule presented in Table 1.

Reduction of CSO-discharge

In urban areas the stormwater runoff increases with the growth of impervious areas. This means that larger water volumes must be transported to the treatment plant and that the peak flow will be higher as well. The three main methods of reduction of CSO are:

Source control of the stormwater runoff
Down-stream control of the stormwater runoff
Treatment of CSO

In Sweden especially the first two methods have been practised. These are based on the idea of reducing the stormwater volumes and the peakflow. Until the beginning of 1970 the main goal of stormwater management in this country was to lead the stormwater out of the urban areas as fast as possible. The frequency of CSO-discharge has increased

Table 1. Quality criteria for different types of receiving waters (partly based on data from the Cothenburg Waterworks, 1985).

Pollutant load	Type of recipient								
	Sea	Big lake	Small lake	River	Stream	Brook	Fishing water	Bathing	Drinking water
Yearly load (P, N, Susp)	X	X			X	X	X	X	X
Total load during the summer period(6 months) (P, N, Susp)			X	X					
Total load during the driest summer month (P, N, Susp)					X				
Total load during the CSO event, dim. 2-year rainfall (BOD_7, heavy metal)					X	X			
Total load during the CSO event, dim. 2-year rainfall (BOD_7, heavy metal, coli bacterial)							X	X	X

very much during this period as have other urban water problems as for instance groundwater depletion. Hence, during the last decade alternative methods for stormwater management have been developed. An overview of these methods is given by Hogland (1984). Treatment of CSO in Sweden has not been very common until just recently. The experiences of CSO treatment presented in this paper are therefore based on a survey of the current literature.

Source control

The stormwater flow (i.e. the peakflow and the volumes transported in the combined sewer system) can be reduced through different kinds of source control. Source control means that the water volumes are reduced and/or the runoff delayed before the water enters the sewer system. Local deposition of stormwater and flow equalization through inflow control are the two main principles of source control.

Local deposition means that`the water is infiltrated into the ground close to the source. The most common methods are:

- Infiltration on pervious areas such as lawns
- Infiltration in open ditches
- Infiltration in pervious ponds
- Percolation directly into the soil
- Percolation storages for drainwater from roofs
- Percolation storages located in the road body.

A special interest has been directed towards a pervious road construction called "the unit superstructure" which infiltrates all the rainwater (see Hogland et al, 1986). The stormwater runoff can more ore less totally be reduced (see Figure 3). Flow equalization can also be achieved through inflow control. Reduction of the flow is then made on the impervious surfaces before the water enters the sewer system. The concentration time can be lengthened by for instance increasing the roughness of the impervious area, decreasing the slope of the surface, increasing the distance between the gully pots,infiltration

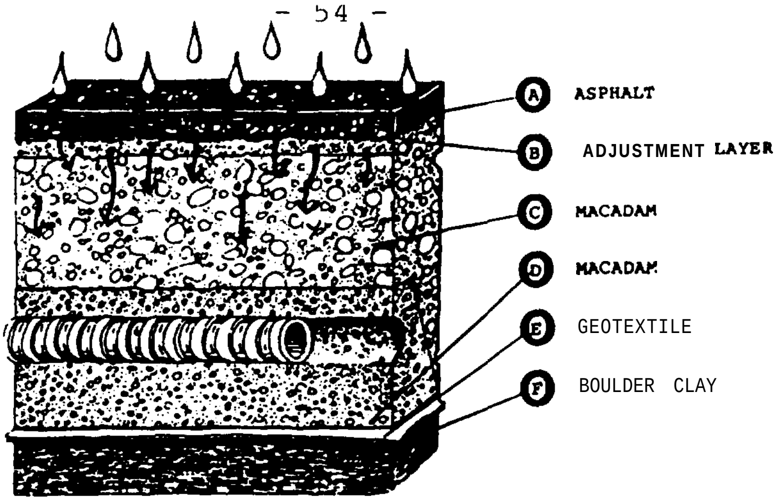


Figure 3 The pervious asphalt construction "the unit superstructure".

through pervious pavement constructions, storage on roof tops, restricted inlet holes to the gully pots, detention ponds and dams.

Downstream control

The downstream control means that the water flow is regulated in the sewer system.

Detention basins and concrete storages are commonly used close to the sewer system. These facilities can also have a treating effect on the water. Special regulators can also be installed in the sewer net, such as:

- flow regulators
- hydrobrakes
- wirbel drosser
- flow valve.

These regulators usually do not need any extra storage volume beyond the storage volume in the sewer system.

Detention of the flow can also be achieved at the wastewater treatment plants in special storages. In highly developed areas, closed reservoirs can be embedded in the receiving water. Most of the methods presented above are described in detail in Stahre (198).

The treatment of CSO

Advanced treatment of CSO-discharge has no tradition in Sweden. The reconstruction of combined sewer system into separated systems which was carried out mainly during the period 1950-1975 has eventually decreased the need for CSO-treatment.

Treatment of CSO-discharge can be an economically interesting alternative when the cost of separation of the sewer net, construction of retention basins etc is high.

CSO-treatment is usually only economically motivated where large amounts of water can be treated at the same time. This means that the CSO-discharge must be concentrated to a few closely located sites.

The most convenient location for CSO-treatment is close to the main waste water treatment plant. Treatment of only CSO-discharge means that the plant must be inactive for long periods of time. Badly functioning treatment methods and high costs due to a low utilization rate, characterise a non-continuous treatment. According to Innerfeld et al (1979) every method of treatment of CSO-discharge must include:

- Automatical activation at CSO-start
- Processes used must not be sensitive to quantitative and qualitative fluctuations in CSO-discharge
- Authenticity according to operation and treatment result.

The most important part of the treatment of CSO is to take care of the first most polluted discharge which is caused by washout of sediments accumulated in the pipes and on the surfaces during dry periods.

The pollutants in CSO discharge can be divided according to the processing methods as follows (Hultgren et al, 1974):

- Settleable suspended materials
- Non-settleable suspended materials
- Dissolved biologically degradable materials

- Dissolved biologically non-degradable materials and toxic materials
- Nitrogen compounds
- Microorganisms.

The treatment methods are traditionally divided into:

a) Mechanical Processing Methods
b) Biological Processing Methods
c) Chemical Processing Methods.

The optimal effect of treatment is often reached with a combination of the processes together with preventive measures such as street cleaning and rinsing of pipes. A total overview of the pollutant situation and function of the sewer net from the source to the receiving water is a necessity.

Mechanical treatment

The methods of mechanical treatment are usually simple constructions with high efficiency and high adaptability to fluctuations in the flow. No continuous operation of the plant is necessary. The plant can work effectively directly after a long dry period. Through chemical additives the efficiency of the mechanical treatment can be increased. Suspended materials but also a considerable amount of other constituents such as BOD_7 and phosphorus can be sufficiently removed by mechanical treatment. The mechanical treatment can be divided into the following methods:

- Grids, screens
- Sedimentation
- Filtering
- Flotation.

The separation rate of Grids and Screens varies a lot depending on the CSO quality and the size of the screen. Lager et al (1974) present removal rates for screens:

Separation rate	Constituent
25-90%	Suspended Solids (SS)
10-70%	Biochemical Oxygen Demand (BOD_7)

Screens can also beneficially be combined with other treatment processes such as filtration and flotation. The efficiency of treatment can be reduced by oil and lubricants in the water together with bacterial growth on the screens.

The sedimentation process works through gravitational separation of suspended materials. The method has also a flow equalization effect which reduces the momentaneous load on the receiving water. The efficiency of different types of sedimentation basins and storages in the sewer system is discussed by Koral and Saatci (1976).

A good reduction of suspended materials can be achieved but also a lesser reduction of BOD_7. Drehwing et al (1979) gives the reduction rate 30-60% for suspended solids and 15-30% for BOD_7 as normal values. The sedimentation basins can easily be combined with other treatment steps or with chemical dozing.

Different types of SWIRL-constructions (see Hogland et al, 1984) can be used for removal of sediment. Sedimentation is also done in open dams. In these dams the water is also biologically treated through organical destruction by bacteria and algae.

Suspended materials can be removed more efficiently by filtering than by the other methods mentioned. However, the incoming CSO must first be pre-treated by a screen.

Innerfelt et al (1979) present removal rates by filtering.

Separation rate	Constituent
60%	SS
30%	BOD_7
40%	COD

Murphy and Hrycyk (1979) state 40-60% reduction of suspended materials as normal values for filtering. Lager et al (1974) summarize the experiences of filtering CSO in the USA.

The flotation process separates the solids from the liquid phase when air bubbles are added to the CSO. The bubbles are fastened on the particles and float to the surface where they can be removed with a scraper.

The flotation removes suspended materials but a good reduction of BOD_7 can also be achieved. Small particles are removed more efficiently as compared to conventional sedimentation.

Lager et al (1974) summarize the experiences of flotation.

Biological treatment

Biological treatment methods are used for removal of non-sedimental and dissolved organic materials. The organic material is transformed biologically into cell substances, which can be removed by sedimentation. There are two methods of stabilisation of organic material:

- The aerobic method
- The anaerobic method.

The function of the biological system and its removal efficiency is dependent on the following factors (Lager et al, 1974):

- The inflow and the quality of the CSO (the availability of food)
- The growth speed of the organisms
- The total mass of organisms in the system
- The contact period between the organisms and the CSO
- Type of organisms
- The environmental condition (pH, temp. etc).

Graham et al (1979) give 70-95% as a reasonable removal rate for BOD_7 and SS.

One main disadvantage with biological treatment is the fact that the CSO-discharge has no continuous inflow. No dry weather flow exists. Therefore the organisms are kept alive between the CSO-events or developed again at the CSO-event. There are two possibilites of solving this problem. The plant can be located near the conventional waste water treatment plant and used for treatment of waste water during dry periods. Another way is to store the CSO-water and let the organisms grow so the wanted treatment can be achieved. The most common methods of biological treatment of CSO as well as of waste water are:

- Activated sludge
- Biological beds
- Rotating biodiscs
- Biodams.

Normally the activated sludge method is inconvenient for treatment of CSO as the BOD_7 content is not always very high. Lubricants and heavy metals in the CSO can disturb the biological process and the removal of the sludge. Under normal conditions the method works efficiently.

A biological bed consists of a bed of stone, carbon, plastic etc. through which the CSO-water passes under destructive influence by the micro organisms. The organisms adsorb the organic material.

Lager et al (1974) present advantages and disadvantages with CSO treatment by means of biological beds.

The rotating bio discs method is something in between the two biological treatment methods just mentioned. The construction consists of rotating discs sitting on an axis, where the organisms are growing, toghether with a low contact tank.

When treating CSO in biodams the organic substances are removed through sedimentation and through biological degradation. The degradation can be anaerobic or aerobic. The anaerobic method is not convenient for CSO as continuous inflow is necessary to keep the balance between the organisms. The surplus of algae and the number of other microorganisms are the most important factors for the removal rate. To keep aerobic conditions it can be necessary to instal special aerators. Problems can also occur if the CSO water has a high sludge content. Pre-sedimentation may then be necessary to avoid accumulation of sludge in the biodams.

Lager et al (1974) discuss the convenience of treatment of CSO in biodams.

Chemical treatment

CSO-water can be treated chemically when it consists of fine particles which cannot be removed in a mechanical way.

The following chemical methods are common:

- Chemical dozing
- Carbon adsorption
- Disinfection.

The efficiency of most of the methods presented earlier can be increased through chemical additives. Aluminium and iron salts and polyelectrolytes can be used. The removal rate is usually 65-75X for organic materials.

The carbon adsorption method is used to remove dissolved materials in CSO. The water passes up and down through activated carbon where watersoluble pollutants can be adsorbed. The method is costly but the carbon can be reused in big plants.

Graham et al (1979) state that 90% of the BOD_7 can be removed by carbon adsorption. The method can also be convenient for removal of chemical components such as heavy metals and pesticides.

Disinfection is used for reduction of microorganisms in CSO, mostly viruses and bacteria. The most common methods are:

- Mechanical methods
- Physical methods
- Chemical oxidation.

When the CSO is disinfected mechanically the microorganisms are removed by a mechanical treatment method. Disinfection of CSO demands a higher efficiency effect than when treating ordinary waste water due to higher loading during short periods of time. Physical disinfection is made by ultra-violet light or by heating methods. The method consumes much energy and the suspended materials must be removed from the CSO. The most common method of disinfection is chemical oxidation with chlorine or ozone. Because of the temperature fluctuations in the CSO, the dozing must be varied during the year.

Discussion

What method is selected among the existing control and treatment alternatives depends on **the usefulness,** at each special location, of the method concerned and the costs that a realization of the CSO control entails. The suggestion must be presented in relation to the main goal of the entire rehabilitation plan for the sewer network. Simple and sometimes less costly measures can also improve the CSO situation. Examples of such methods, which can be satisfactory as immediate rehabilitation work, are:

- Improvement and regular maintenance of the sewer system
- Repair and regulation of weir crests, screens and high-water gates
- Redistribution of the flow conditions in the sewer network
- Increasing the capacity of sewer under pressure, jets and pumping stations
- Disconnection of storm water pipes from combined sewer systems
- Sludge pumping of storage and sediment traps.

It must always be kept in mind that the effect of the CSO discharge also depends on the conditions in the receiving waters. A large discharge to a harbour dock usually gives less negative effects than a small discharge to a river. The CSO-discharge to the rivers in the south of

Sweden can during short periods in the summer be 8-10 times higher than the riverflow itself.

Further research

The current main research efforts in Sweden and in the other Nordic countries are directed towards an estimation of the importance of sediment build-up processes in combined sewers. A research co-operation between the Department of Water Resources Engineering and the municipalities of Stockholm, Gothenburg and Oslo is currently under way. The aim of this project is mainly to try to measure the pollution build-up in sewers of different types and flows. With the knowledge of the time-dependent pollution build-up processes, it will be possible to establish a CSO-model which can be used to calculate the pollution discharge on a single-event basis in a realistic way.

Acknowledgement

The authors would like to express their gratitude to the Swedish Council for Buidling Research (BFR), the National Swedish Environmental Protection Board (SNV), the Swedish Water and Sewage Works Association (VAV), and to the municipalities of Lund, Malmö, Helsingborg, Stockholm, Göteborg and Oslo which are funding the research project.

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Stahre P

Slide Rule Methods for Estimation of Combined Sewer Overflow-Volumes and Pollution Load.

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SLIDE RULE METHODS FOR ESTIMATION OF COMBINED SEWER OVERFLOW VOLUMES AND POLLUTION LOAD.

Introduction.

The relative effect of combined sewer overflow (CSO) discharge into receiving waters has increased during the last years due to improved efficiency at treatment works. New pipelines are connected to old sewerage systems with limited capacity and of not too good quality. In order to develop a responsible urban water quality management policy we must be able to describe overflow discharges in terms of quality and quantity. During the last 10-15 years big efforts has taken place in order to develop the necessary tools.

Earlier it was presumed that CSO-discharges represents from 10-75% of the total pollution produced over a year and that 25 to 40% of the pipevolumes could be filled with deposits, depending on parameter and field conditions.

I would like to start this presentation on slide rule methods for estimation of combined sewer overflow, by giving an overview of what kind of methods researchers have been using in predicting the pollution load and concentrations from overflow. So far no one has been able to give any good equation or method for calculating pollution load, and it would be useful to take a closer look into the nature of pollution deposits in combined sewers.

Methodology.

Oddvar Lindholm, Norwegian Institute of Water Research,(NIVA) in 1974-75 studied 9 catchment areas out of which 4 were combined sewers /1/.

He assumed that the concentration and load from the overflow system were high early in the rainfall in highly urbanised areas, at high rain-intensities and after long periods of drought. The load was also assumed to be closely connected to the slope of the pipes.

From September 1974 to july 1975, samples were taken manually from the time when the precipitation started. 5 to 20 samples were taken during each precipitation event. The samples where analysed for SS, COD and total Phosphorous (TP). He calculated the pollution discharge via overflows over a period of one year and for each precipitation event, but did not try to make any equations for deposits. He only stated some general rules for how the systems would act under different conditions, assuming that slope and drought periods would be the most important factors. His most important conclution was that the overflow discharges probably was greater than the discharges from secondary treatment works over a year.

William C. Pisano, Gerald L. Aronson, Celso S. Queiroz, Frederic C. Blanc and James C. O´Shaughnessy did a two year study on dry weather deposition and flushing for combined sewers./2/Their aim was to find thumb rules for artificial flushing of sewers to avoid deposited pollution to enter recipients during rain events. In doing so they tried to develope an equation for dry-weather depositions, using both clean water and sewage for flushing. The research took place from 1976 to 1978. The samples where taken manually from the front and in the tail of the wave. They made analysis on COD, BOD, TKN, NH_3, TP, TSS and VSS. Based upon constants and variables they made deterministic equations for deposits buildup and flushes. Geiger and Lager had also made such models in 1975 in two different studies.

In 1977 M.B. Sonnen, USA made a mathematical model connected to the SWMM which continually calculates the deposit in the pipes. This was based upon the works of Pisano et.al.

Wolfgang F. Geiger, Technische Universitat, Munchen, Federal Republic of Germany, over a period of 4 years, from 1977 to 1981, made a study of the characteristics of combined sewer runoff. His methodology was similar to Pisano et.al., but he was only using drinking water, when flushing the sewers in his catchment areas. Samples were taken automaticly every 90 minute under dry-weather conditions and the time lapse between every sample under precipitations events was 5 to 20 minutes. His conclusion was that: "In spite of other and previous assumptions pollution buildup seems to be of stochastic nature."/3/

William Hogland, Ronny Berndtsson and Magnus Larson made a study on combined sewers in Malmø, Sweden in 1982; it was especially focused on the time between the precipitation-events./4/ They stated that the time for build up in the pipes could be up to at least 12 days before a steady state occured, and that this fenomenon was of a greater importance, than the time since the last CSO event.

G.Stotz and Kh. Krauth, Institut fur Siedlungswasserbau, Universitat Stuttgart, made simular analasys as Geiger in Stuttgart in 1983. Their samples were taken on a manual basis, and the time between each sample was down to 15 seconds, even then they found that the time between samples taken were too long to measure pollution in the front of the wave. The time between each precipitation-event varied from 12 to 288 hours (12 days). Taking into consideration the dry weather period and the slopes, they tried to make equations for the quantity of washed out materials. For SS the equation had a relatively high correlation coefficient, but not for other parameters. /5/

From 1982 to 1986 the Norwegian Institute for Water Research made theoretical analysis and mathematical models for calculation of dry weather deposits in combined sewers, and the amount of pollution discharged to the recipients /6/. They tried to make equations for the discharges, based upon multiple correlations between parameters other authors had focused on. This work was to a great extent based upon the works of Pisano et.al. and M.B. Sonnen. In addition they took into account the deposit loads from surface areas during the rain-event. They calculated the theoretical pollution production from the area and the surface load. They then assumed that what was measured as mass load to a sampling point downstream the overflow, during the precipitation event, was the sum of produced pollution minus the overflow discharge. This provides that you have a good control over the production. In 1985/86 NIVA found that the assumed pollution production had been overestimated, regarding phosphorous. In other words, the pollution deposits in the pipes had been overestimated in these calculations.

In 1982-1984, the Norwegian Hydrotechnical Institute in Trondheim, did some research on selfcleaning pipes. This work concluded that the dry weather deposits only occured at very low flow or/and at very gentle slopes. In other words: deposits seldom takes place unless we have errors, like shallows or obstructions /11/.

Today research is taking place on a large scale all over the world. In the nordic countries a cooperative work is taking place and has by now been going on for 3 years. The project will measure the dry weather deposits after various drought periods. In addition they will do a study of the mecanisms behind sedimentation- and erosion-processes in pipes.

Some of their findings are being published theese days, and furter reports will be published during 1988. In a new, not published report from NIVA, they have studied a catchment area in Oslo for 5 years, an area with steep slopes and high population-density. The conclutions are that the overflow-masses were relatively small, compared with other and earlier findings, both in Norway and other countries. This is assumed to be caused by steep slopes and a high population density.

Water Research Center (WRC) has started a similar work in 4 catchment areas in England.

The university of Hannover has started a project to make a mathematical model for predicting deposits for the most commom pollution parameters. Also in other parts of the Federal Republic of Germany work are going on to optimize the management of the sewerage systems.

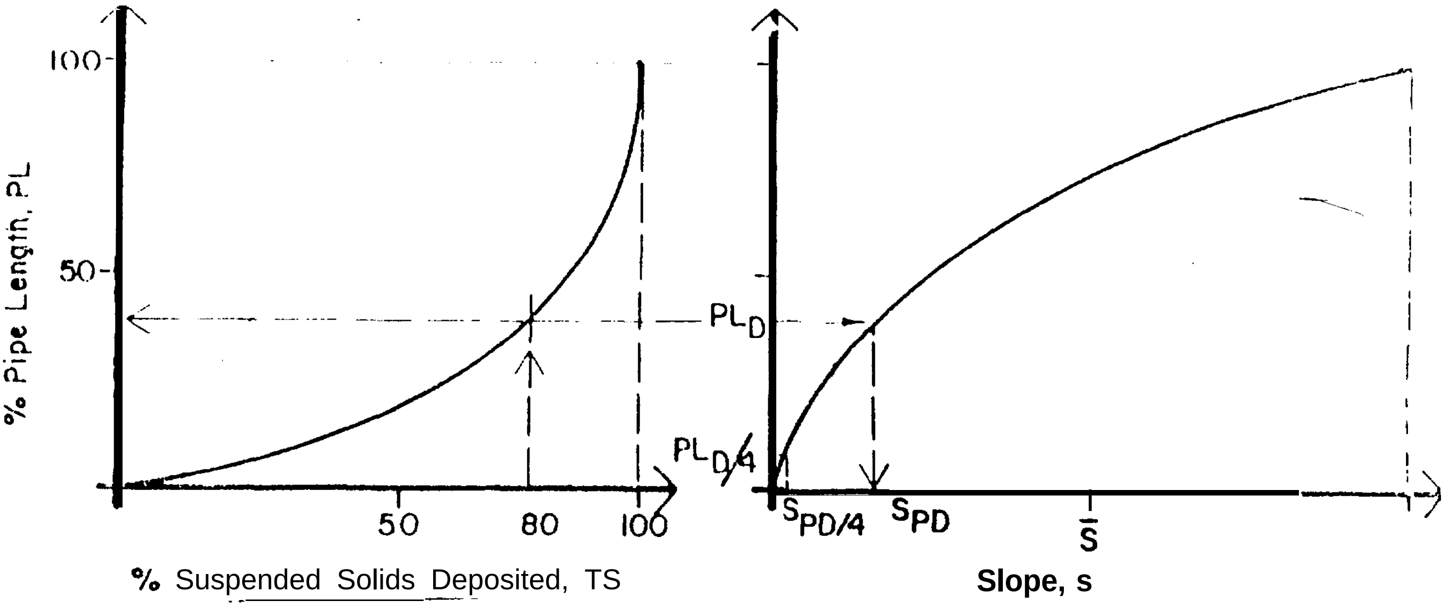
Slide rules.

Everyone who has tried to study combined sewer overflow discharges, has conluded that the variation in the discharges is great, it varies with factors up to 10. It is also difficult to predict these variations in models. It is now general agreement among researchers that the pipe deposits seems to be of a stochastic nature. In addition it is very hard to measure the discharge, because the first-flush wave is difficult to analyse correctly. The work of Stotz and Kraut shows that not even samples taken every 15 second was good enough to describe the variation in concentrations. What we can conclude from these works over a period of 10 years, is that we still not are able to predict the deposits in pipes and that measuring the discharges will be difficult, even for skilled researchers However, we are much further down the road in order to understand the nature of the deposits and will probably soon be able to give good slide rules, even if we probably never will be able to give accurate equations for calculating pollution discharges.

Pisano et.al. measured on 75 different catchment areas and did regression analasys on their data. They based their models on the following parameters;

- Total collection system pipe length (L') ft
- Service area of collection system (A) acres
- Average collection system pipe slope (S') ft/ft
- Average collection system pipe diameter (D') inches
- Length of pipe corresponding to 80% of the solids deposited in the system (Lpd) ft
- Slope corresponding to Lpd (Spd) ft/ft
- Slope corresponding to 1/4 of the percentage of pipe length (PLd) below which 80% of the solids deposit (Spd/4) ft
- Flow rate per capita, including allowance for infiltration - q gpcd.

From these paramterts they made 4 different equations with different complexity and different accuraty. The correlation-coefficient varied from 0.949 to 0.845.



Fiuure 1. Spd, Spd/4.PLd and PLd/4

$D = 0.0038*L^{08142} * Spd^{08187} * Spd/4^{01078} * q^{05098}$ (Eq. 1)

$D = 0.001303*L^{118} * A^{0178} * S^{\text{`}0418} * D^{\text{`}0604} * q^{051}$ (Eq.2)

$D = 0.00389*L^{12195} * A^{01866} * S^{\text{`}04343} * q^{051}$ (Eq.3)

$D = 0.0076* L^{1063} * S^{\text{`}04375} * q^{051}$ (Eq.4)

D = Deposits in the network

Later work has been focused on the dry weather flow (Lindholm) and the time of drought since the previous precipitation event (Hogland et.al. and Stotz/Kraus). Equations like these has not been confirmed and it is a general agreement among researchers that the nature of deposite and/or discharges cannot be described in deterministic equations.

I will now take a closer look at factors affecting the sedimentation and pollution dischage. The parameters which has been studied are;

- total pipe length
- slopecaracteristics
- dry weather flow and flow/p.e.
- rainfall intensity and volume per event
- type and use of the catchment area
- antecedent dry weather
- catchment area
  pipe diameters
- population density
- the quality of the construction work.

It is obvious that most of these parameters will play a part in the behaviour of the sedimentation and discharges, but it has been very hard to find good correlations because the sedimentation in the pipes is of stochastic nature.

Total pipelength.

Both Pisano et.al. and Lindholm has studied this parameter without finding good correlation between deposits and pipelength. This is probably due to the fact that many of the parameters which has been considered, interfere with each other. The pipelength is probably of a great importance because the probability of having areas with a negative slope is increasing with the pipelength, but this is taken care of by other parameters.

Slopecharacteristics.

Since the shearing stress we need to keep the sewers selfcleaning is rather low, it should not be a good relationship between deposits and slope. However, since the deposits largely takes place in basins caused by negative slopes, the volumes and the stability of these "stores" will depend on the slope (and the flow). Most findings concludes that we will have small deposits when the average slope in the catchment is steeper than 15%. Pipes that steep will in most cases keep the sewerage clean. Slopes between theoretical selfcleaning conditions and 10- 15% will not correlate directly with deposits.

A thumb rule will then be, that if you have sewerage with slopes below 15% you may have deposits, but you will then have to consider other parameters for predicting how great the problem will be.

Dry weather flow, flow/p.e. and p.e./m.

The flow pr. p.e. and meters per p.e., are parameters interfering with the total pipelength and dry weather flow. The flow pr. p.e. gives you a measure of the infiltration, and will obvious be of importance for the ability to keep the pipes clean, but the dry weather flow has so far been considered to be the best parameter for describing this ability. As earlier described, if the pipe lines is buildt with smooth slopes, there should not be a problem with deposits. But since this is not so, parts of the pipes will store sediments. The most important conteract against these stores seems to be a high dry weather flow (and steep slopes).

Rainfall intensity and volum per event.

Different conclutions has been drawn here from different authors, but a majority of reports tell that the volume is not well correlated to the discharge as the rainfall intensity is. Pisano et.al. conclude that small or medium rainfall intensity did not cause any discharges or could clean the pipes. This corresponds with the findings of Paul Lessard in Quebec, Canada and Hogland, Sweden. Geiger however found no correlation between the rainfall intensity and the concentrations in the discharges. The findings here will probably depend on the contribution from the surface.

Type and use of catchment area.

How important is the type and use of the catchment area? Full agreement has not been reached, but it seems obvious that erosion, street-sweeping practice will influence on the discharges from the surface. Lindholm and Reinertsen, Norway, assume that the load from the surface washed out during a rain event, is proportional to the impervious surface area and correlated to the intensity.

Considering nutrients and organic loads, there seems to a general agreement amoung researchers that the sewage plays the major part in the deposits, but not so for SS, metals and toxic compounds. Both Lindholm and Hogland has pointed out that street sweeping practice will be of great importance to the discharges during an rainfall event.

Antecedent dry weather period.

Pisano et.al. and Geiger did not take the period of drought between each overflow event into consideration in their works. Geiger concluded that this was not of any great importance.

Field, Hogland et.al. Lindholm and Stotz/Krauth however, found that this played an important role, because the sedimentation in the pipes could last for days and weeks before a steady state occured in the system. This is one of the most important factors the nordic project is studying in the work now going on. The length of the dry weather period will probably be of great importance calculating the pollution discharged to the recipient both for calculating the discharge over a year and for finding the maximum loads for each rain event.

Catchment area.

The total area will be of some importance for the total mass sedimenting in the pipes. Other parameters will however overshadow in most cases. Nobody has found a good corrolation between catchment-area and the the quantity of deposits.

Pipe diameters.

The pipe diameter is of course of great importanse when we consider the self cleaning ability for the sewerage. In addition a high flow in a small pipe will give better self cleaning conditions than the same flow in a bigger pipe - obviously. But since the diameter corresponds well with other parameters mentioned above, this factor is already taken care of. Parameters like population density and p.e. per meter pipe will tell more or less the same.

Population density.

The population density is of important because it should lead to a high dry weather flow compared to the area. We cannot use both the density and the dry weather flow, because they are expressions of the same physical factors - more or less.

The quality of the construction work.

As pointed out above, the sedimentation in the sewerage system is very much a consequence of negative slopes causing shallows. This should mean that the quality of the construction work and the geotechnical conditions are important. This has not yet been studied, but will be analysed in the nordic project. The project aim at measuring the volumes in the pipes caused by shallows and see if there is any relationship between the volume which can store sediments and the discharges. This may be difficult to document, both because of the general problem in measuring and because the surface of the local "stores" would be more directly linked to the pollution washed out from the pipes, than the volume. But anyhow, the quality of the construction and the geotechnical conditions must be of great importance, for deposits and overflow discharges.

Conclusions.

The nature of the sedimentation in pipes and the wash out, is complicated to describe and predict. Work has been going on for years and we still have a long way to go before we have a full understanding of the influencing factors. We do know a lot of about factors that plays a part here, but the problem is that we do not know how they interfere with each other and measuring during the discharge events is very difficult.

To predict the problem however it is possible to give some slide rules. I will point out 3 major factors which should be resonable to measure or determine:

- the quality of the construction of the pipes
- the slopecaracteristics
- the dry weather flow
- geotechnical conditions.

If the construction job is well done, like in tunnels, and the slope is steep enough to have self cleaning conditions, you should not have any problems at all with sedimentation in the sewers.

If the slope is bigger than 10-15% you should not have too much problems with sedimentation. With lower slopes you must consider other parameters like the dry weather flow, the population density and the meter of pipe per person.

If the dry weather flow is high, problems with deposits should not be too great. When it is lower you must look into other parameters like the slope and length of pipe and the length per person.

Litterature:

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SLIDE RULE METHODS FOR ESTIMATION OF POLLUTION LOAD DUE TO STORMWATER

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When making budget calculation for the pollution load due to urban areas it is often easy to estimate the load from sewage treatment plants. Much more difficult is the calculation of pollutants from separate storm sewers. In this case you could not in a realistic way measure all the stormwater outlets, they are usually to many and the technique of measuring continuously is difficult and expensive. In other words you have to do calculations of the pollutant load from information gathered at some few locations and during a limited period of time.

A method for crude hand calculation of pollutant load was presented at the Visby seminar 1986 (Falk, 1987). The description of the methodology is repeated here:

The stormwater runoff volume during the year considered may be calculated according to

$$Q_{year} = a \ . \ A \ . \ (Pr-b). \ 10^{-3} \ m^3$$

where a = constant showing the portion of impermeable surfaces that is drained via the stormsewer network a may be estimated according to

Land use	a
Single family houses	0.5-0.6
Multi family houses	0.6-0.7
Downtown areas	0.8-1.0

A = total area of impermeable surfaces in the catchment area (m^2). Varies from 10 to 100 % (single family houses sparsely distributed to downtown areas)

Pr = total amount of precipitation during the considered year (mm)

b = total loss of precipitation due to depression storage (mm)

b may for a year be estimated to 50 mm for steep areas and to 100 mm for flat areas.

The yearly outlet of pollutants may be derived by multiplying the volume calculated above with average pollution concentrations according to Table 1.

	COD mg/l	Oil mg/l	N_{tot} mg/l	P_{tot} mg/l	Pb µg/l	Zn µg/l	Cu µg/l
Single family houses	70	0.2	2.0	0.3	75	150	35
Multi family houses	100	0.4	2.0	0.3	150	200	50
Downtown areas	150	1.0	2.5	0.4	230	400	100
Traffic areas	200	2.0	2.0	0.3	230	250	50
Industrial areas	110	2.0	3.0	0.4	140	500	130

Table 1. Average concentration of pollutants in stormwater (Falk, 1987).

The method should give an accuracy of ± 30-50 % as regards volumes. For pollution load concentrations the accuracy of course is smaller. It is strongly advised to support the calculations by at least some measurements of quantity and quality on location.

Some warnings when using the method must be given:

- It should be stressed that the method is valid only for calculating stormwater pollution from a <u>separate stormwater network</u>.
- Only direct runoff is considered i.e. the runoff generated during a rainstorm. The baseflow from inflow/infiltration is not taken into account. Hogland et al, 1986 showed that the pollution load due to inflow/infiltration is far from insignificant. For certain catchment areas this contribution may be 50 % of the total load.
- The method is difficult to use during winter conditions.
- Again it must be strongly advised to support the calculations with measurements.

In spite of all the warnings the method has been used in several cases since 1985. The Swedish action plan for the eutrophication of the sea has used it when comparing different sources of pollution. Different national task groups for the protection of the Baltic Sea has used it as well. When the county of Östergötland made their water protection plan they also used the method. Several municipalities have used it in their water planning, among others Norrköping, Linköping, Mönsterås, Oskarshamn, Motala and Falkenberg. Recently the method was used when making budget calculations for the pollution load on lake Mälaren.

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Protection Commission,
Second Seminar on Wastewater
Treatment in Urban Areas,
Visby, Sweden, Sep 6-8, 1987

MEASURES AT OR CLOSE TO SOURCE TO PREVENT THE DETERIORIATION OF STORM WATER QUALITY

THE URBAN WATER CYCLE

Urban areas, that is areas where man made structures dominate over the natural structures, are from a hydrological point of view still a part of the hydrological cycle. Comparing a rural area to an urban area in terms of precipitation, evapotranspiration and runoff results in less evapotranspiration, more runoff and less ground water production in an urban area. The runoff process is much more rapid in urban areas, giving higher maximum flows and thus having a high potential of transporting particulate substances and substances bound to particles. The interaction between urban areas and the atmosphere is outlined in Fig. 1, which shows that there is not only a deposition of substances from the atmosphere but also an uptake of substances produced within the area.

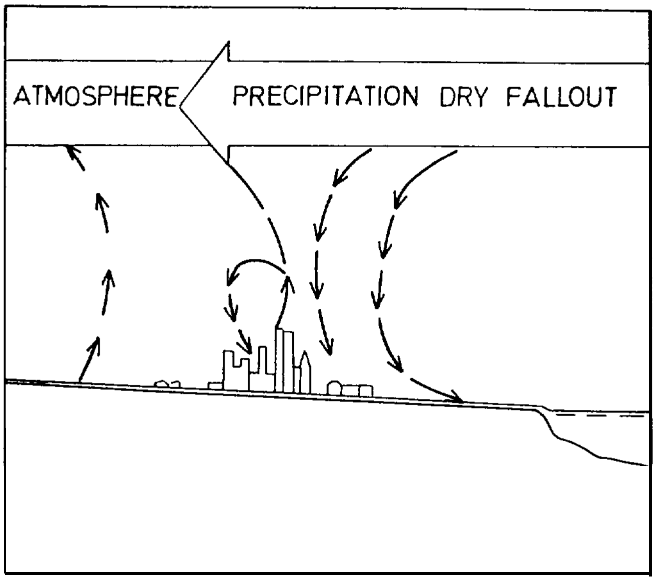

Figure 1. Interactions between the atmosphere and an urban area.

The local hydrological cycle is to a great extent disturbed by man made structures. In sewered areas the water balance is more or less controlled by man. In principle there are two water cycles in urban areas: One is concerned with the production and distribution of drinkikng water, which becomes sewage during the passage of the urban area with the later being discharged into a receiving water. The other is the rainfall-runoff process which is a part of the natural hydrological cycle. Both are outlined in Fig. 2, (Svensson (1987)), where some interactions between the two systems are also marked. For example the yearly storm water volume is of the same order of magnitude as the yearly infiltrated volume of ground water due to leaky sewers for many urban areas in Sweden.

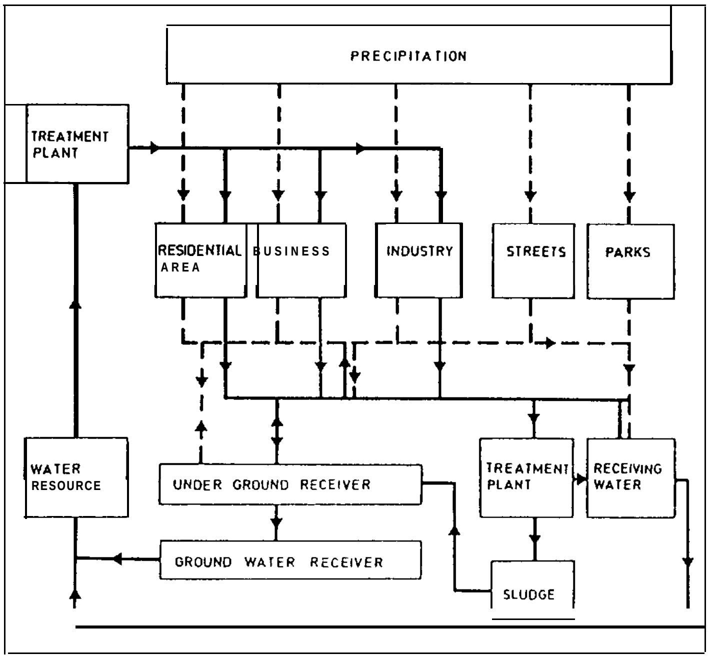

Figure 2. Water cycles of urban areas.

To get some impression of the magnitudes of the mass flows in the urban rainfall runoff process, an example from the City of Lund in the southern part of Sweden, (Hogland, 1986), is shown in Fig. 3.

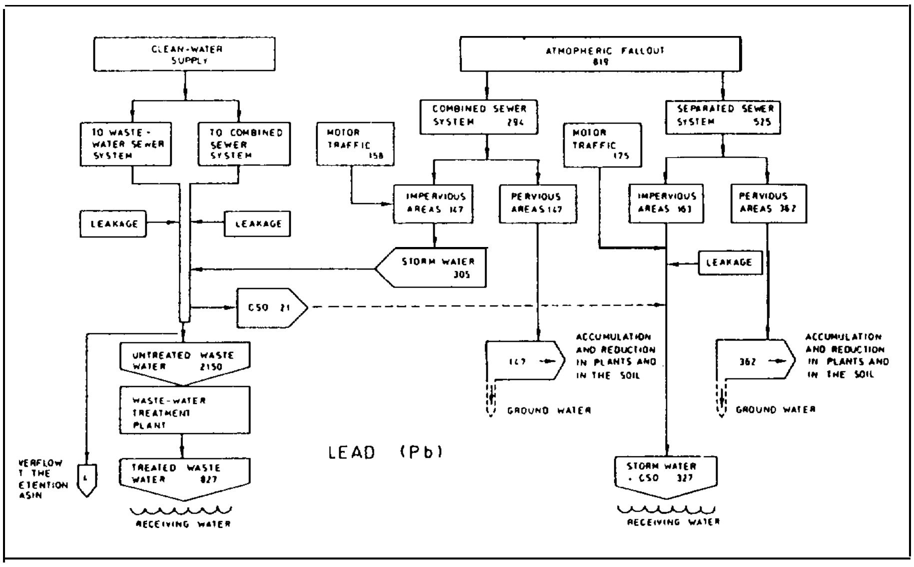

Figure 3. Yearly mass balance of Pb for the City of Lund, after **Hogland** (1986).

The major pollutant sources of storm water are:

- Atmospheric fallout
- Exhausts from vehicles
- Vehicle construction materials
- Building construction materials
- Road construction materials
- Droppings from animals

A conclusion, which can be drawn from mass balances made for different areas, (Malmqvist, 1983), is that there has to be parts of a catchment which act as sinks to the pollution. For example areas with vegetation, catch basins or parking areas.

THE RECOGNITION OF STORM WATER AS AN IMPORTANT TRANSPORT PATH FOR URBAN POLLUTION

The sources of the substances found in storm water can be divided into atmospheric sources and surface bound sources. Both can be further subdivided, as by Malmqvist (1983). A reproduction of this subdivision is made in Fig. 4.

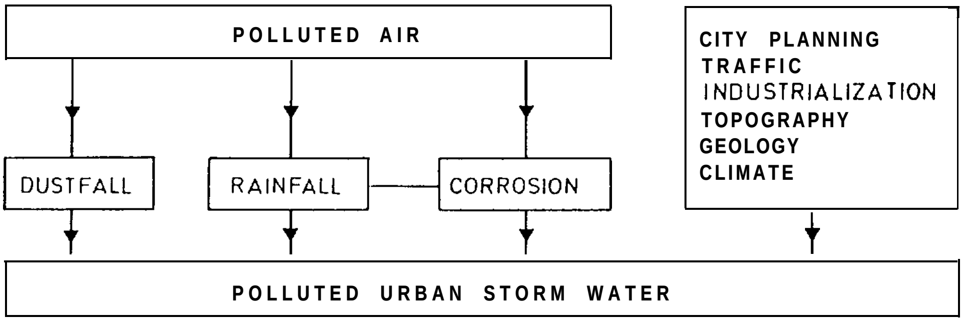

Figure 4. Factors influencing the quality of storm water, after Malmqvist (1983).

There are two main reasons why storm water is an important transport path of substances in the urban environment. The first is the fact that rainfall or snow passes through the atmosphere over an urban area and hits every part of the area at least some times every year. As water is a good solvent for many substances this causes a transport of soluble substances from the atmosphere and from urban surfaces by the runoff. The second reason is that raindrops, when hitting a surface, are a very good erosive agent, thus creating the possibility of transporting particulate substances. However, this cannot be done without the runoff having a sufficient transport capacity, which will vary with the runoff intensity, the topology of the area etc. Saying that the transport capacity varies within an area, implies that a substance can be transported from one part of an area to another, not necessarily reaching the receiving water during a single rainfall event.

An outline of the transport paths in a sewered urban catchment is shown in Fig. 5. All parts of the area accumulate substances between runoff events. During runoff, however, some parts will be washed, thus having less accumulated substances after the runoff event and some parts will act as a sink, thus having more accumulatated substances after the event. An example of a part acting as a sink is a catch basin, which not only accumulate substances, but also affects the composition of the storm water through processes going on in between runoff events, (Morrison (1985)).

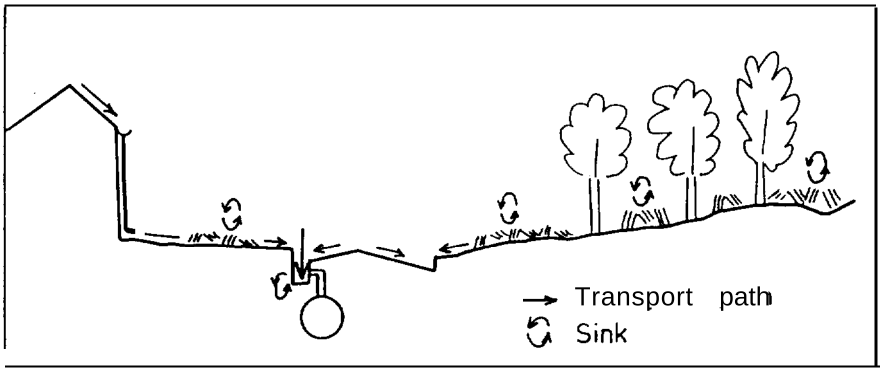

Figure 5. Transport paths and sinks for substances found in storm water.

Substances which are transported with storm water in relatively high concentrations are:

- Cadmium, Copper, Zink and Lead
- Phosphorous and Nitrogen
- Organics (COD)
- Solids
- Hydrocarbons

EFFECTS OF STORM WATER DISCHARGES

The physical, chemical and biological substances in storm water may be harmful to the aquatic life in the recieving waters. Looking at background concentrations of receiving waters, storm water most often increases these concentrations, thus storm water can be regarded as a polluter. However, wether the storm water is harmful or not depends on the concentration levels, and if they are critical for aquatic life or not.

When discussing concentration levels it is essential to distinguish between acute toxic effects and long term effects. The acute toxic effects are related to soluble or weakly bound substances, which are available for organisms and can be taken up rapidly. The long term effects are related to substances, which tends to accumulate in organisms and in plants.

The discussed effects of storm water on the aquatic life are the primary effects of storm water discharges. A secondary effect on the receiving water is the combined sewer overflow, where the flow increase during a storm event causes the overflow, and the mixture of sewage and storm water overflowed causes the environmental effect. Similary storm water discharges to sewage treatment plants often disturb the treatment process, thus effecting the quality of the outlet from the treatment plant.

MEASURES TO IMPROVE STORM WATER QUALITY OR TO REDUCE STORM WATER DISCHARGES

Source control will be effective to reduce pollutants in:

- Atmospheric fallout
- Exhausts from vehicles
- Construction materials

Reducing runoff volumes is effective in reducing the pollution impact on surface water bodies. However, the risk of polluting the ground water must be observed. Means of reducing runoff volumes are:

- Surface infiltration
- Percolation basins
- Permeable asphalt

Surface maintenance is another effective way of reducing the pullution of storm water. This can be done by:

- Street cleaning on a regularly basis
- Catch basin cleaning on a regularly basis
- Snow transport

At present reducing storm water volumes and diverting some runoff to the treatment plant will be the most realistic way of handling the storm water pollution.

Also, surface maintenance for industrial areas and restrictions for the handling of substances which could cause water pollution are effective tools.

In the long run, source control may improve air quality, thus improving storm water quality.

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CAUSES AND EFFECTS OF HAZARD FROM MUNICIPAL - INDUSTRIAL AGGLOMERATION TO THE WATERS IN THE CATCHMENT AREA OF THE UPPER VISTULA

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1. Introduction

The catchment area of the Upper Vistula of 34 thousand km^2 surface area constitutes 17 % of the total basin the Vistula River.

Participation of the flow from the Upper Vistula reaches about 30 % in relation to the total flow.

Five of the total number of 27 areas of ecological hazard in Poland are located in the basin of the Upper Vistula. (9)

2. Causes of ecological hazard

From among the mines, water of 7 Silesian coal mines are discharged into the Vistula River. (2)

According to investigations carried out by the Institute of the Technical University of Cracow the discharge from these mines of highly brackish water was very irregular in the course of a day and a week.

In consequence of activities of water management authorities from the Voievodship Katowice a temporary stabilization of this discharge took place.

The diagrams no. 4, 5 show the variability of mean concentrations of chlorides in the years **1934 to 1986 and** mean concentrations of dissolved matter in the years 1968 to 1982 /3/.

From the territory of Silesia and the adjacent territory polluted waste waters flow into the Vistula River from textile, cellulose-paper mills, metallurgical works and chemical industry.

These waste waters cause qualitative changes in the waters of the Vistula River in the region of Cracow. (Figure 6)

The list of concentrations corresponding to determined frequence occuring in the years in question shown in Table 1 gives evidence of the hazard to the waters of the Vistula River in the region of Cracow. /Table 1/. (8)

3. Effects of hazard to waters in the catchment area of the Upper Vistula

Pollution of surface waters in the territory of the Siliasian agglomeration made it necessary to look for pure water resources beyond this territory. /Figure 7/

This was carried out by building a water reservoir on the Small Vistula River at the locality Goczalkowice and by using the water stored in the chain of water reservoirs of the Sola River since local reservoirs on the Przemsza River and Brynica proved not to be sufficient.

At present already the Silesian agglomeration suffers from a serious water insufficiency and the conception of transporting water from the Carpathian tributaries of the Vistula River to Siliasia is discussed.

In consequence of water pollution in the Vistula River, Cracow, counting at present 800 thousand inhabitants, had to restrict water uptake from this river for its water works.

Water pollution in the Vistula River, which appears in the fifties and was caused mainly by phenol compounds from chemical and coke industry made it necessary to eliminate the water intake from the Vistula River, working since the beginning of this century by means of infiltration intakes.

Consequently it caused a necessity of building water intakes on small tributaries of the Vistula River in the vicinity of the city at the localities Rudawa and Dlubnia.

These sources of water supply assured water for the town in sufficient quantities only till the half of the sixties and since that time marked deficit of water in the town Cracow had been noticeable and made it inevitable for the municipal authorities to take water from the Vistula River at Bielany in spite of poor quality indices.

This water intake began to be supplied from the stream Sanka dewatering an agricultural catchment area.

A melioration of water supply conditions for the town took place by water intake from the Raba River a Carpathian tributary of the River Vistula where a storage reservoir of volume of about 130 mln m^3 was built.

Salination of the Vistula River is especially troublesome for industrial works which use water from the Vistula for production and cooling purposes.

The Metallurgical Works of annual steel production of about 5 mln ton, which take water from the Vistula River must cope with serious difficulties because of poor water quality.

This water used at present in the region of Cracow for cooling purposes is highly aggressive and does not satisfy the conditions which industrial water, used in ferrous metallurgy and heat energetics, should satisfy.

In dependence on the kind of water circulation in the metallurgical plant the rate of equipment corrosion is at present from 0,1 mm/year to 1,5 mm/year and leads in consequence to necessity of an exchange of pipe lines every 5-10 years.

In thermal power plants in the region of Cracow use of brackish water for cooling purposes causes a decreasing in work reliability of turbine condensers, steamboilers and cooling water pipelines.

Representants of the energy-industry find that maintainance or gravation of the existing water conditions in cooling circuits will cause in consequence a decrease in work reliability and life of the equipment.

It may also influence production of electric energy and heat supply for Cracow.

4. Fields of activities aiming at decreasing the hazard of industrial agglomeration pollution to the waters of the Vistula River

After recognition of the state of hazard of municipal industrial agglomeration pollutions in the catchment area in the upper course of the river, the basic activity consisted in defining a building plan of sewage treatment plants in the main points of hazard. In the years 1970-1980 a number of designs were elaborated the building of which was begun and partly completed.

These were built in the region of Upper Silesia among others at Tychy, Katowice, Czestochowa.

A sewage treatment plant for the Metallurgical Works in Cracow was built and finished in 1985, a sewage treatment plant in Nitrogen Works in Tarnów has also been built.

The general state of waste water treatment in the catchment area of the Upper Vistula is however, still unsatisfactory.

4.2 Limited pure water resources directed attention of a number of research centers in Poland to the possibilities of water recovery from waste waters and using it for industrial purposes.

First investigations on this possibility were carried out for the region of Upper Silesia, where a great water demand for industrial purposes exists and moreover this territory lies on the vidge between the Rivers Vistula and Odra, and does not dispose of sufficient resources of surface water.

The water drawn from coal mines are highly brackish and are not useable for industrial purposes.

The population get water from a number of Carpathian rivers on which, as it was mentioned, storage reservoirs were built.

Investigations indicate to the possibility of water reclamation from waste water with destination for industrial purposes. It was assumed that waste waters of hardly removable mineral pollutants not exceeding concentrations permissable for industrial water, admitted in the given area, can be designed for recovery. /4/

Water recovered from waste waters can be used for cooling purposes. Territories of the basin where waste waters are designed for recovery must be protected against inflow of substances complicating purification processes.

Waste waters, only after biological treatment in sewage treatment plants of output higher than 20 thousand m^3/day will be subject to the reclamation processes.

After biological treatment in which nitrogen compounds should be removed, waste waters designed for cooling circuits in metallurgical or energo-industry are subsequently subjected to further purification in coagulation processes by use lime or aluminium or iron sulphate. After the coagulation process, filtration and desinfection of the waste water in forseen.

A schematic presentation of recovered water utilization in the central part of Silesia is presented in figure 9. /4/

Similar studies on utilization of recovered waste waters as the source of industrial supply for Metallurgical Works in Cracow have been recently carried out in Poland, in the Technical University of Cracow, Institute of Sanitary Engineering and Environment Protection.

Municipal sewage from Cracow often shows lower salinity than the water in the Vistula River, since the inhabitants use mainly waters taken in from the Rivers Raba, Rudawa and Dlubnia which are not charged with mineral pollutants.

4.3 Increase in salinity in the waters of the Vistula River is caused by discharging brackish waters from coal mines into it; further prognoses concerning this increase have been an investigation and study problem since 1960. In a successive conception chemical treatment of brackish waters was forseen according to the proposition made by Chief Institute of Mining. This conception led to the establishment of an experimental plant at Debienska, where brackish waters were subject to chemical thermal treatment giving in consequence salt and water.

In this process, however, great amounts of wastes, especially gypsism, are obtained and moreover, it is a very energy-consuming process. According to the data of the Chief Institute of Mining from the year 1967 heat consumption in the heating period is 1047 kJ per 1 kg H_2O. /6, 7/

A successive conception concerned storing brackish waters in storage reservoirs and dosing them in proportion to the occuring flow so as to obtain in the waters of the Vistula River upper concentration values of 300 mg/dm^3, forseen by respective Polish regulations.

Increase in salinity of coal mine waters and their amounts caused by opening of new coal mines where exploatation reeched deeper and deeper layers caused the situation that the Vistula River above Cracow is no more able to receive chlorides. Simultaneously difficulties in building storage reservoirs for storing brackish waters occured. Another conception was, thus, forwarded it is to transport brackish waters below Cracow and subsequently below the estaury of the San River. /Figures 10, 11/

Studies in this line have been carried for many years. In 1985 a group of specialists from the Oil and Gas Company and a team of the Technical University of Cracow forwarded the suggestion of transporting brackish waters to the region where Cenomane sandstone occurs and pressing them into the sandstone strata. Preliminary investigation showed that the total capacity of the Cenomane reservoir, lying about 800 to 1000 m below the ground level, is about 18 mld m^3. The water quality in that reservoir is similar to waters from coal mines, but these waters have no contact with waters of higher levels.

The quantity of highly brackish waters of chloride concentration of 42 000 mg/l is about 1 cm^3/s, hence annually about 31 mln m^3/year. Annual amounts of brackish waters pressed into that reservoir would then to 1,6 % of the total capacity, of the Cenomane reservoir. Considering the life-time of the Silesian mines, which is 50 years, the amount to about 1,5 mld m^3 i.e. about 8 % of the total capacity of the Cenomane reservoir.

Investigations on the absorbtion ability of Cenomane reservoir have recently started in Poland. In case of positive results the waters of the Vistula River would be greatly lightened from pollution with brackish waters from Silesian coal mines.

4.4 Decrease in pollutions discharged is also the consequence of building sewage treatment plants.

On the basic of a number of examples of waste waters characteristics of the purification system of waste waters in the Metallurgical Works in Cracow is given. There are 3 sewage treatment plants in the Metallurgical Works for treating industrial wastes and rainfall waters.

In the waste water treatment plant the so called A /figure 12/ 420 thousand m^3/day of industrial waste waters and 6,4 m^3/s rainfall waters are to be purified. The purified waters from this plant supply the so called port basin from which water is taken for production purposes.

The quality of wastes should have achieved, according to the design, following values determined in the water-loyal permission COD 60 mg/dm^3, oxidability 20 mg/dm^3, total suspension 40 mg/dm^3, soluble matter 1000 mg/dm^3.

The purification plant consists of mechanical grates, longitudinal sedimentation tanks of retention time T = 3 hours earth sedimentation tanks for rainfall waters flowing into the purification plant during heavy rains. Control studies performed on the purification plant corroborate the values adopted in the design.

Reduction of suspended solid by 50 %, oil and fats by 20 %, iron by 70 % is achieved in the sewage treatment plant.

The second sewage treatment plant built in the Metallurgical Works, plant "B" has a daily flow capacity of about 140 thousand m^3/day for industrial wastes and about 1,6 m^3/s for rainfall wastes. This plant is of mechanical type and the reduction effect is 50 % as suspended solid is concerned and 20 % as oil is concerned.

The Metallurgical Works have also built a biological treatment plant for coke waste waters.

4.5 Waste waters from the region of Tarnów, discharged into the Dunajec River, a right tributary of the Vistula River, is a serious source of hazard for surface waters. In the territory of Tarnów there are Nitrogen Works, which have recently built their own sewage treatment plant. These Works came into operation on the turn of years 1920-1930 and have been greatly developed in the post war period.

A part of the Works built in the center-war period has a combined sewage system covering the area where production of unorganic products: ammonia, nitric acid, semi conductor silico is located.

From this part of the Works the wastes are discharged to the mechanical-chemical treatment plant into which also flow in wastes from production of acrylonitrile, where hydrogen cyanido is the main product.

With regard to their high toxicity these wastes are preliminary subjected to oxidation with active chlor so that cyanides are reduced to a level which is absorbable by active sludge in the local department purification plant. There from the waste waters are discharges to the mentioned mechanical-chemical treatment plant.

The above mentioned local purification plant is also designed for selective biological removal of waste waters from caprolaetan and formaldyhyde production. Activated sludge tanks are aerated with compressed air.

The plant has a developed system of retarding reservoirs what eliminates variations in pollutant concentration. This splant aims at preliminary wastes treatment before introducing them into the common sewage treatment plant for Nitrogen Works and the city Tarnów, which has been in operation since 1978. /1/

In this sewage treatment plant chemically contaminated waste waters with many toxic substances and easily decomposing sewage, carrying nutrients /phosphorus, nitrogen/ necessary for activated sludge undergo treatment and are discharged into the river.

In that plant a 58 % decrease of COD takes place, but no investigations on nitrogen removing are carried out.

Simultanously with building waste water treatment plant in Tarnów, technological production changes aiming at decreasing the amount of water uptake and waste load were introduced in the Nitrogen Works.

From the primary quantity 43 300 m^3 waste waters/day, in 1985 the amount of discharged wastes was 34 500 m^3 wastes/day and respectively decreased the following pollutant charges:

CDD from 16 497 kg/day to 7 225 kg/day
N_{NH_3} from 5 065 kg/day to 4 441 kg/day
and Nitrates N_{NO_3} from 199 kg/day to 11,7 kg/day.

4.6 Apart from investigations on water recovery from waste waters carried out in the Technical University of Cracow investigations connected with hydraulics of equipments for waste water purification are also performed. Those are carried out in the Institute of Engineering and Water Management Laboratory of Hydraulics.

Kinetics of flow and pollution in nitrification and denitrification chambers is investigated for one of the big sewage treatment plants to be built on the coast of the Baltic Sea.

Investigations are carried out on a model in the scale 1:17 aiming at checking the design version, determination on dead areas and inner circulation.

Establishing of optimal maxing conditions tends at decrease in energy consumption.

5. Summary

Hazard to the natural environment in Poland was caused by excessive development of national economy, and specially mining and heavy industry at simultanous underestimation of water and air protection.

This may be accounted for by a rapid increase in population which has grown in the recent 40 years by about 12-14 mln. This was connected with the necessity of creating new work possibilities to assure work for people from the overpopulated country regions.

Resources such as coal, zink and lead ore and sulfur occur in the South of Poland in the basin of the Upper Vistula where big municipal-industrial agglomerations developed. This caused difficulties in a planned protection of water against pollution at a lack of respective means.

At present this problem is not only noticed by professionalists but also by economical and political authorities. There force inhibition of further degradation of the natural environment and gradual, though a long term, a melioration of this state is to be expected.

Should locally pretreated industrial wastes be discharged to the municipal sewage system or to the stormwater system?

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1 Background

Waste water from industries in municipal areas are generally discharged to the municipal sewage system. The state or municipal authorities can require that the water shall be&pretreated in order to protect the sewerage system and the treatment plant.

In Sweden, industrial wastes from

- breweries
- dairies
- slaughterhouses and meat processing industries
- canneries
- tanneries and textile industries
- pharmaceutical industries
- metal/finishing industries
- workshops
- photographic laboratories

and some other industries in the city area are discharged to the municipal sewage system and treated together with the normal municipal waste. Guidelines with limit values have been prepared by the Swedish Water and Sewage Works Association, VAV. (VAV 1983, Rennerfelt 1986)

In principle a well pretreated waste water might also be discharged to the stormwater system. This was also discussed during the preparation of the first issue of the limit values.

In the following the advantages and disadvantages with a treatment with the municipal sewage on a disharge to the stormwater system will be discussed.

2 Composition of industrial wastes

Waste water from food industries contains carbohydrates, proteins and fats in suspended and soluble form.

Waste water from pharmaceutical industries contains complicated organic substances and organic solvents.

Waste water form workshops often contains heavy metals and oil (partly as emulsion) and water from metal finishing contains heavy metals, partly in complex form, cyanides etc.

Waste water from photographic laboratories contains toxic and carcinogenic substances.

Waste water from textile industries contains a variety of organic and inorganic compounds used in production and dyeing of textiles.

Waste water from laundries contains organic material and tensides and has a high pH.

Waste water from tanneries contain organic substances, sulphide and sometimes chromium.

3 Local treatment of industrial wastes

In order to protect the sewerage system and prevent damage to the treatment process, a local pretreatment is often done at the industry before the discharge to the municipal system.

Normal pretreatment techniques and results are summarized in table 1.

Table 1. Pretreatment of industrial waste waters

Industry/production	Quantity and Quality of untreated water’	Type of pretreatment	Quality of treated water
Breweries Beer and soft drinks	3-5 m3/m3 prod 4-7 kg BOD/m3 prod 2-3 kg SS/m3 prod BOD ~ 1 000 mg/l	Equalization in aerated tanks neutralization, sedimentation of kiselguhr biological treatment	BOD after biological treatment about 300 mg/l
Dairies Milk	1.2 m3/m3 prod, 1.5 kg BOD$_7$/m3	Neutralization, (equalization), biological treatment	
Cheese dairy Mixed production	1-2 m3/m3 prod, 2 kg BOD$_7$/m3 BOD 1 200 mg/l		BOD after biological treatment about 300-400 mg/l
Slaughterhouses Slaughtering	~7 m3/ton ~16 kg BOD/ton BOD ~ 1 200-3 000	Fat. grease and sludge removal chemical flocculation biological treatment	BOD after chemical treatm 600-1 500 mg/l BOD after biological treatment 300-800 mg/l
Meat processing	-15 m3/ton ~15 kg BOD/ton BOD ~ 500-1300 mg/l		
Canning Peas	18-35 m3/ton 17-25 kg BOD/ton	Screening, equalization, sludge removal, neutralization, biological treatment	BOD after biological treatment 300-600 mg/l
Potato	7-17 m3/ton 10-20 kg BOD/ton		
Textile mills	Varying concentration BOD 50-1 200 mg/l COD 230-Z 300 mg/l	Equalization, neutralization, screening	Contains salts, tensides nonbiodegradable substanc dyes etc
Laundries	10-15 l/kg textiles BOD 150-600 mg/l SS 50-150 mg/l	Equalization, pH-control screening	Contains consides, Risk f foaming
Tanneries	BOD 1 900 - 3 600 mg/l 30-120 kg BOD/ton SS 1500-3000 mg/l Sulphide 5 - 20 mg/l Chromium 10 - 30 mg/l	Ear screens, static screens, aerated lagoons, chemical precipitation. Catalytic sulphide oxidation	Contains organic substanc and Cr(III)
Workshops Metal finishing	See table 2		

• Good standard in modern industries

Of special interest are the possibilities to treat wastes containing heavy metals.

The most important way to decrease the concentrations is to change the production processes, e.g. the washing.

The external treatment of the wastes is facilitated and gives better results if waste water of different character are separated before the treatment.

The following types of water should be separated

- **cooling waters or other clean waters (which can be discharged directly to the recieving stream or to the storm water system)**
- **water containing cyanide**
- **water containing chromates**

The techniques of waste treatment and the expected results are summarized in table 2.

lab 2. Pretreatment of waste waters from workshops and **metal** finishing industries

Type of waste	Type of treatment	Quality of treated wastes
Metalfinishing waste with heavy metals	pH-adjustment, precipitation of metal hydroxides (pH 8,5-10) (flocculation and separation of flocs) complexing agents may disturb the formation of hydroxides.	Al 2 mg/l Fe 2 " Zn 2 " Pb 1 " cu 1 " Cr-tot 1 " Ni 1 "
	Destruction of the complexing agents, precipitation with H_2S etc may be used.	Ag 1 " Sn 1 " Cd 0,1 "
Cyanides	Oxidation with chlorine or sodium hypochlorite at pH > 11.	Free cyanide 0,1 mg/l Total cyanide 1 mg/l
	Two stages, mixing (>15 min) and after-reaction (>45 min)	
Chromates	Reduction of chromates with sodiumbisulfite at pH < 2,5 (15 min). Chromium hydroxide is precipitated with pH-adjustment.	Cr (VI) 0,1 mg/l
Oil containing wastes		
o with free oil and suspended solids	Flotation (gravitational)	
	Surface loading 1 m/h Detention time > 2 h	Oil < SO mg/l
	Flotation + filtration	Oil < S mg/l
	Filtration (if the oil concentration is low, < 5 mg/l)	Oil < 1 mg/l
o with emulsified oil		about 20 mg/l
	Breaking with acids, Hg-ions or chemical precipitation	(With Mg + separation 10-50 mg/l (With chem.precipitation 1-10 mg/l
	Ultra filtration	S mg/l
	Adsorption	20 mg/l
	Filtration **with activated** carbon	5 mg/l

4 Advantages and disadvantages with discharge to a municipal treatment plant

4.1 Advantages

Even after a local pretreatment, waste waters from food industries, textile mills and tanneries contain a considerable amount of organic material that may consume oxygen in the recieving water. In a municipal treatment plant (especially with biological treatment) a larger or smaller part of the biodegradable material is reduced. Nonbiodegradable organic material such as PAH and PCB may to certain extent be adsorbed to the primary sludge and is thus removed from the water. EPA:s investigations showed however big differences in the reductions between different plants and different groups of organic substances.

4.2 Disadvantages

Pretreated water from workshops and metal finishing plants contains small amounts of heavy metals and other toxic substances. If there are disturbances in the operation of the pretreatment plant, the concentration of mineral oil, heavy metals and other toxic material increases. This may upset the operation of the municipal plant and cause a loss both of the BOD and the toxic substances to the recieving water.

If water from workshops, metal finishing plants, printers etc are discharged to a municipal treatment plant, the heavy metal concentration in the sludge increases. The removal of Cu, Zn, Cr, Pb and Cd was on an average 40 % in the primary sedimentation, 25 % in the biological stage and 15 % in the chemical postprecipitation according to the SWEP research (SNV PM 1942). Ni was hardly removed at all in the primary sedimentation and chemical precipitation but to some extent in the biological step.

The increase of metals in the incoming municipal sewage due to the industrial wastes has been estimated to 40 - 80 % in the SWEP studies.

The main disadvantage with an industrial discharge to the municipal system is the transport of heavy metals and nonbiodegradable organics to the sludge. If the sludge is used in agriculture there may be a further transport of these substances to the environment. Even if the sludge is taken to a garbage dump, there may be a leakage to the ground water.

5 Advantages and disadvantages with a discharge to the stormwater system

5.1 Advantages

The advantage with a discharge of pretreated water to the stormwater system is above all that there is no risk to upset the operation of the municipal treatment plant with toxic material. The concentration of heavy metals and nonbiodegradable organics in the sludge can be decreased, which facilitates the sludge disposal.

Another advantage is that the hydraulic load of the municipal treatment plant is decreased.

A pretreated industrial waste water with a considerable organic concentration must however in most cases be treated in the municipal treatment plant.

5.2 Disadvantages

The main disadvantage with discharge to the stormwater system is that the recieving waters may be polluted in case of a breakdown of the industrial pretreatment system. The aquatic life may thus be damaged. On the other hand, an accidental discharge to a municipal treatment plant may decrease the treatment efficiency and cause more damage during a longer period to the environment.

A practical disadvantage is the necessity of changing the connections from the stormwater system to the sewage system if the character of the industrial production is changed.

6 RECOMMENDATIONS

Stormwater, cooling waters and well pretreated industrial wastes without biodegradable organic substances should be discharged to the stormwater system.

Industrial wastes containing biodegradable organic substances should be discharged to the sewage system and treated together with the municipal sewage.

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NUTRIENT REMOVALAT LOW TEMPERATURES
DANISH EXPERIENCE

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INTRODUCTION

Within five years nitrogen removal is required at all Danish treatment plants above **15,000** PE, and phosphorus removal is required at all plants above **5,000** PE.

These requirements will bring about construction or extension of about 250 treatment plants of a size from 5,000 PE to 2 million PE.

Today **phosphorus** removal takes place at about one tenth of the plants above 5,000 PE and nitrogen removal takes place at the 35 plants shown in figure 1.

It is seen that plants ranging from 800 to 200,000 PE are in operation.

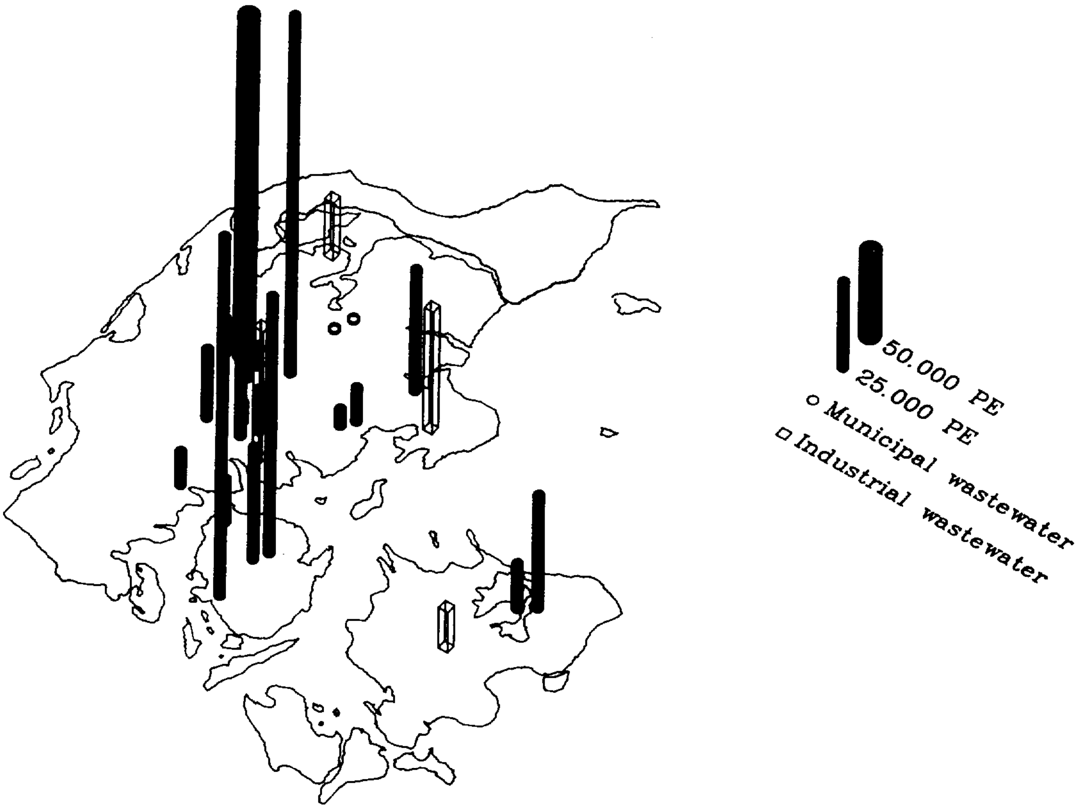

Figure 1. Treatment plants performing nitrogen removal in Denmark.

Long term experience with nitrogen removal exists in Denmark. The first treatment plant has been in operation more than ten years. An example of the operational results from a former trickling filter extended with activated sludge to perform nitrogen removal is shown in figure 2. The plant is designed for 85,000 PE. The frequency distributions of the operational results for nitrogen compounds are given. The first axis gives the effluent concentration and the second gives the frequency. It is seen that the ammonia concentration is fairly low, and **that the** mean effluent concentration of total nitrogen is about 5 mg/l which is well below 8 mg/l which will be the typical standard. Further full scale results from Danish treatment plants are presented in /1/.

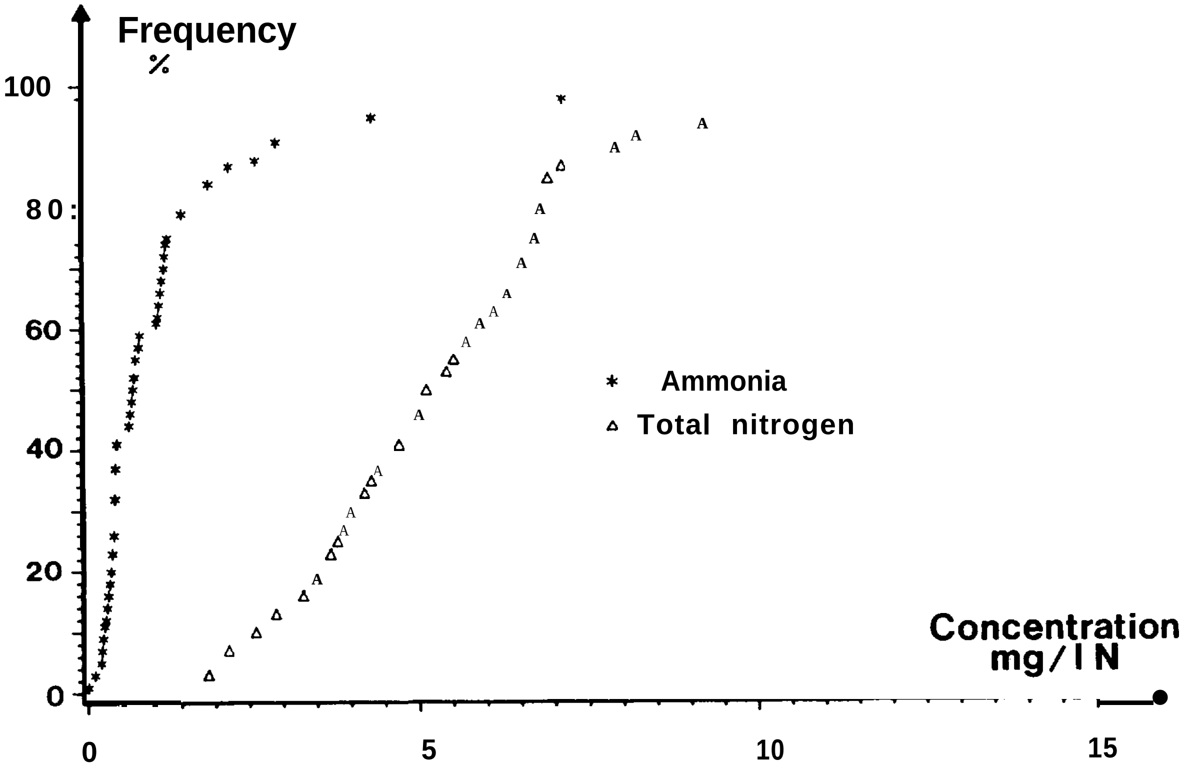

Figure 2. Frequency distribution of effluent results for nitrogen compounds from a trickling filter, upgraded with activated sludge treatment for nitrogen removal.

The nitrogen requirement makes it necessary for the plants to maintain nitrification to a considerable extent throughout the entire winter, and since nitrification is expected to be highly influenced by temperature drop during winter operations, special attention is necessary.

The principal factor determining nitrification is the sludge age, i.e. the average "life time" of the bacteria in the treatment plant and the temperature. Therefore, design of plants in Denmark basically takes place according to a correlation between the necessary sludge age and the temperature as shown in figure 3.

Even at treatment plants designed on the safe side of the guideline in figure 3, problems with nitrification may occur.

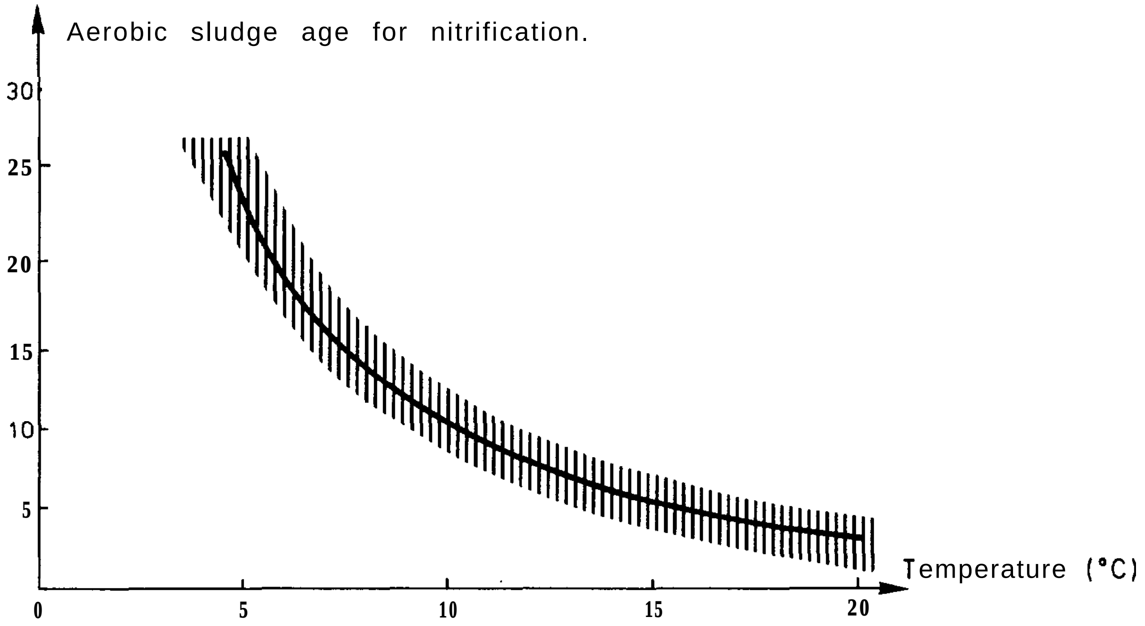

Figure 3. Design curve for nitrification.

The course is that the process depends on a number of things of a very different nature whose influence may be difficult to separate from each other.

<u>The natural conditions</u> of central importance are of course primarily the special growth conditions of the nitrifying bacteria and the temperature conditions, but also precipitation conditions including the snow cover during thaw may have a decisive influence on the course of the process.

<u>The planning conditions</u> in connection with sewerage and treatment plants play an important part in the influence of the natural conditions on the process. By ensuring a reasonable design of the plant in relation to the amount of pollution in the catchment area and by preventing restrictive substances in the wastewater, basic conditions can be established for a satisfactory growth of nitrifying bacteria.

A suitable planning of the sewerage may prevent unfavourable temperature conditions from acquiring a dominant influence, and by forethought when choosing the constructive design of the treatment plant, the importance of unfavourable natural conditions may be limited. (Covering, shelter planting, choice of aeration device, etc.)

<u>The operational conditions</u> decide whether the possibilities for nitrification are used or not. These conditions constitute a complex interaction between wastewater load, means of operation and adjustment, and the knowledge of the operator and his possibilities of taking the necessary corrective measures.

The central conditions have been collocated in figure 4.

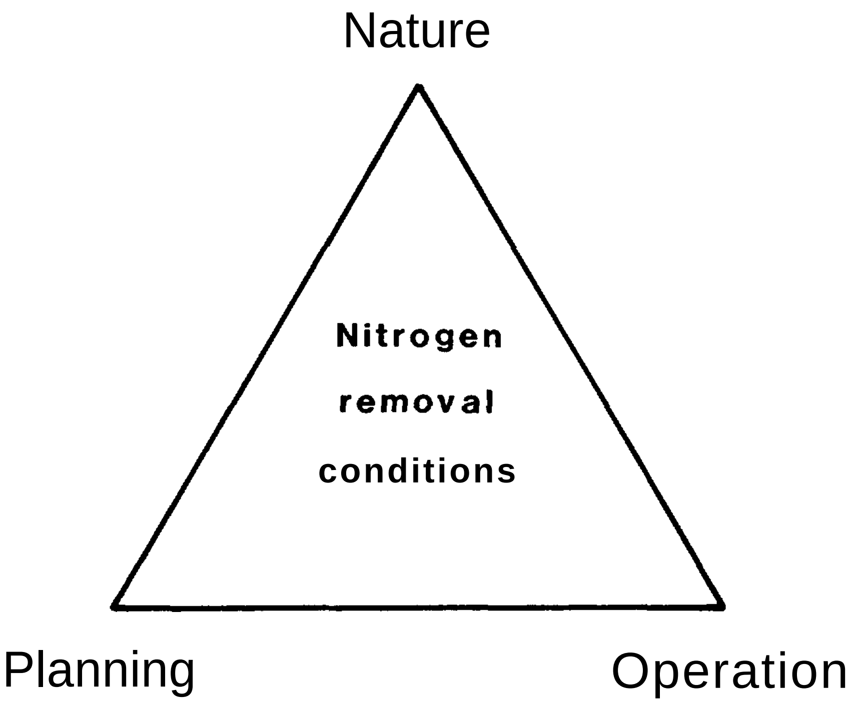

Figure 4. Conditions of importance for nitrification (nitrogen removal).

From a theoretical point of view the principal problems can be regarded as having been solved, but the transition to a judgment of the actual conditions is difficult.

The problems are illustrated below with data from 3 plants where data concerning load and. outlet quality are compared with the information deduced from the operational journal of the plants.

DANISH WINTER CONDITIONS

Temperature

Figure 5 shows the winter temperatures (average of the daily maximum and minimum temperatures) during the months from December to March in the winters from 1980-1981 to 1985-1986, measured at Søholt Treatment Plant.

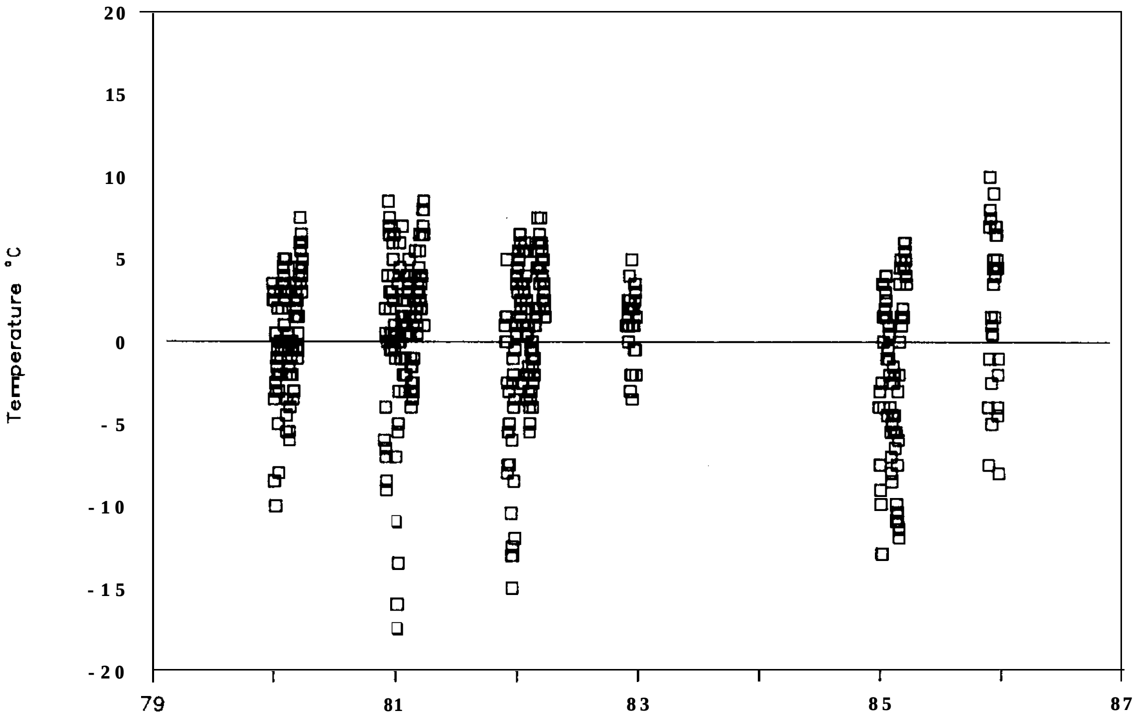

Figure 5. Daily average temperatures measured at Søholt Treatment Plant during a couple of winters.

The figure shows that the temperature is often about 0°C over long periods and often with longlasting periods of frost in between.

Wastewater Temperature

The temperature of the Danish underground, where the sewers are situated, is normally around 8°C. This means that the wastewater temperature at the plant inlet is above this value, since the wastewater normally has a higher temperature. However, low temperatures do occur during snow melting and rain during the winter period.

Figure 6 shows the variation of the temperature at the inlet to Søholt Treatament Plant. The temperature is approximately 7 to 10°C for long periods.

Furthermore, the construction of each individual plant will contribute to determining the temperature at which the biological processes are to proceed.

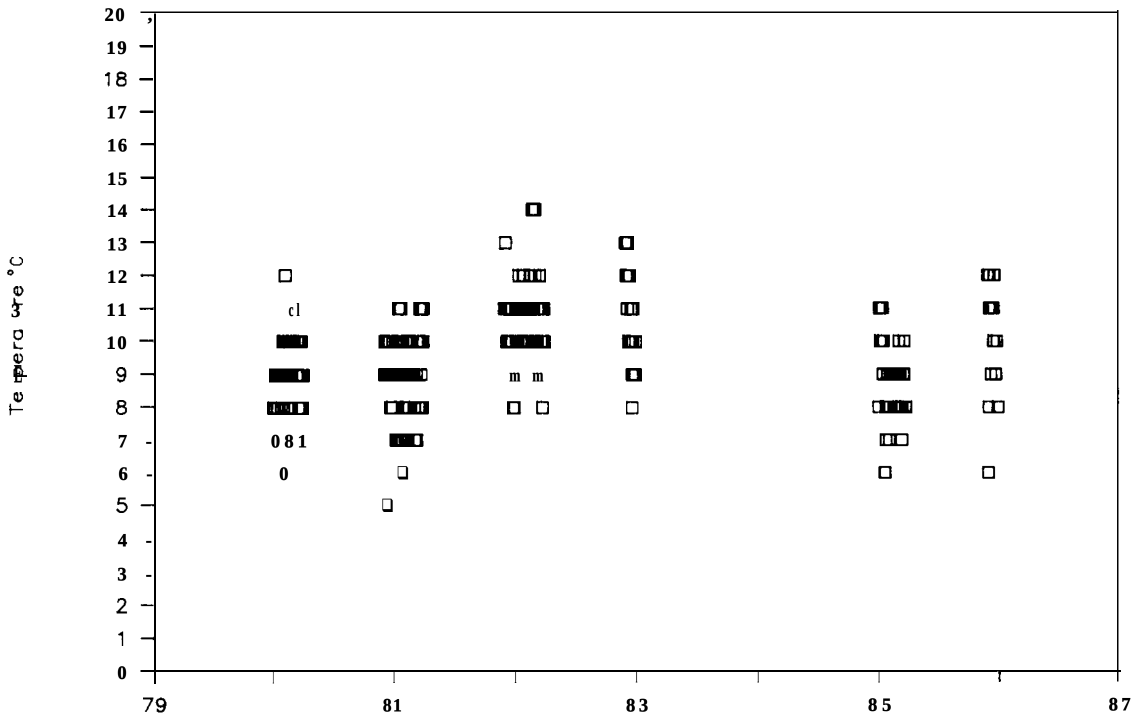

Figure 6. Inlet temperatures to Ssholt Treatment Plant during a couple of winters.

<u>Soholt Treatment Plant.</u> (105,000 PE)
Figure 7 shows the winter temperatures measured at Søholt Treatment Plant during a number of years.

Figure 8 shows the ammonia concentration in the discharge from 1985 to 1986 measured in 24-hour composit samples taken at an interval of 2 weeks. It can be seen that nitrification fails in part. Generally, the possibilities of the plant to nitrify may be judged on the basis of the design curve in figure 3, though the data basis for judging the sludge age is2 flimsy and inaccurate. In figure 9 the sludge age is shown such as it can be calculated in the winter months on the basis of the operational journal. It can be seen that the sludge age of the plant is near/below the critical limit, so that a securing of the nitrification will require favourable wastewater conditions and optimum operation.

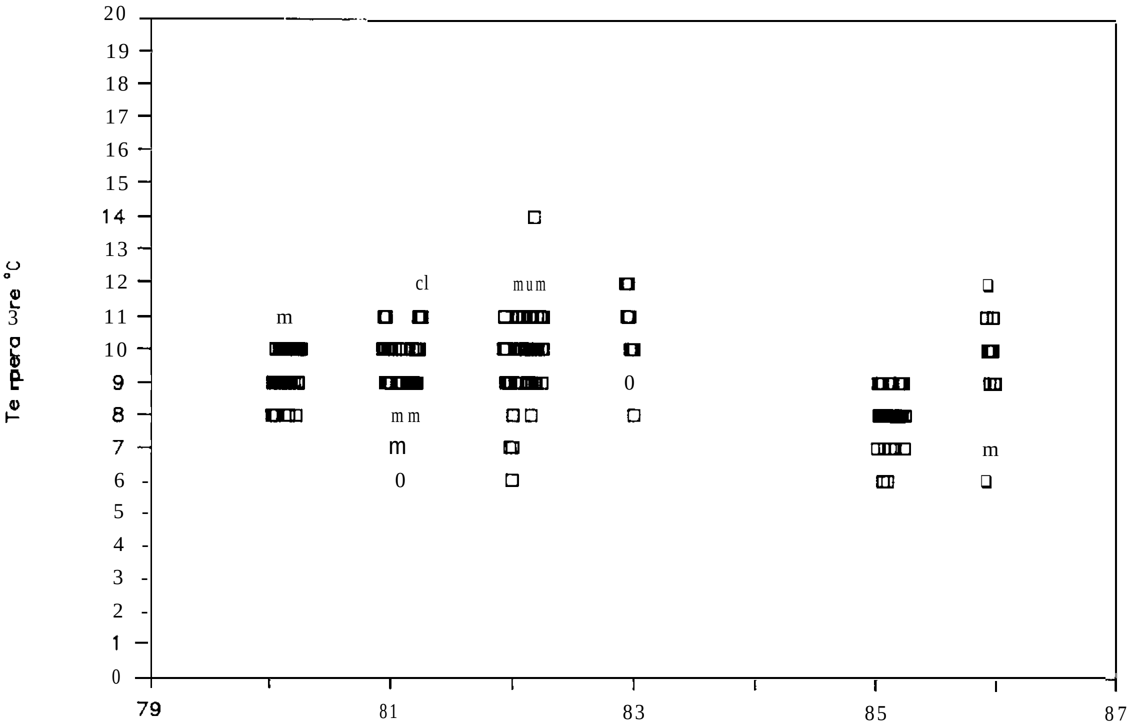

Figure 7. Plant temperature at Soholt Treatment Plant during a couple of winters.

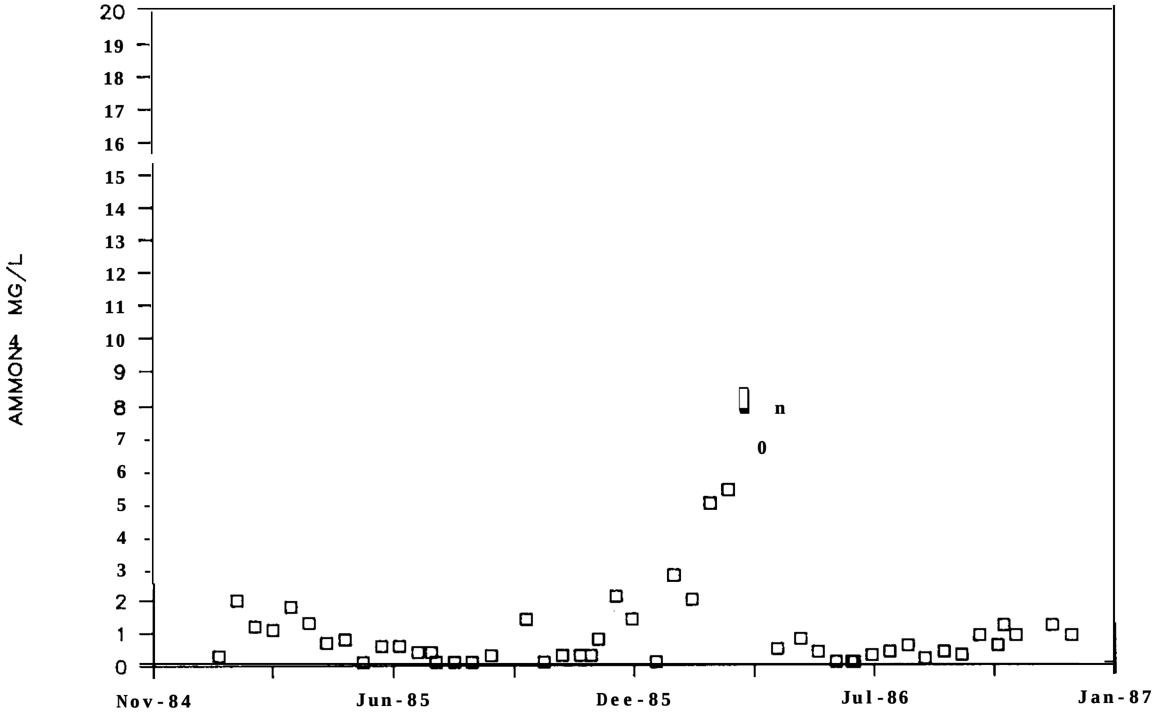

Figure 8. Discharge concentrations of ammonia at Søholt Treatment Plant from 1985 to 1986.

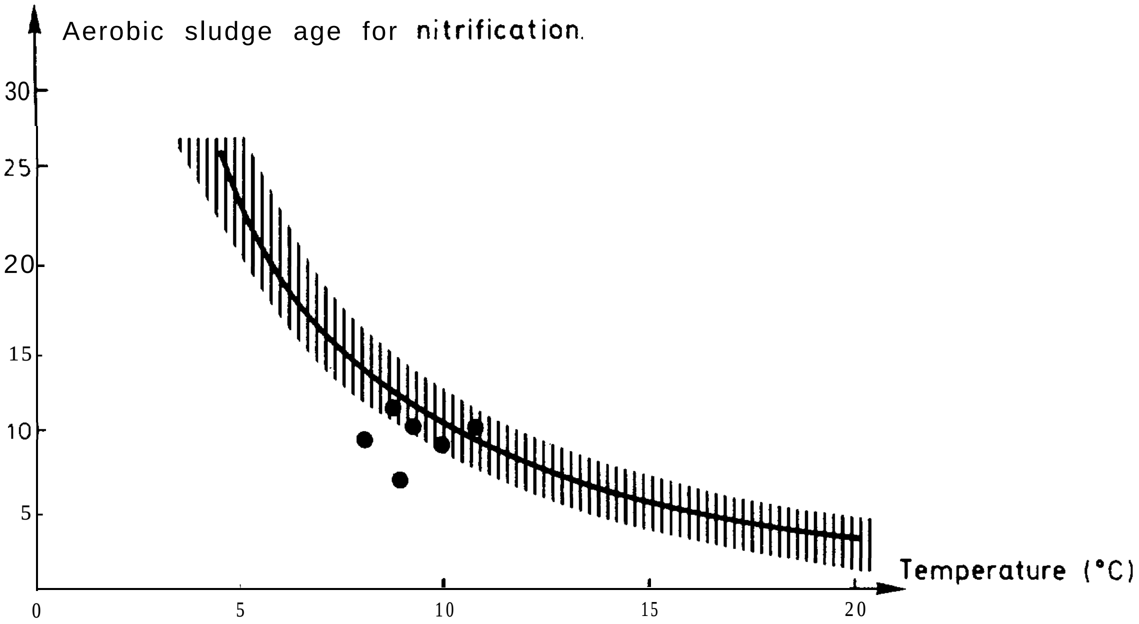

Figure 9. Estimate of the sludge age and nitrifying conditions at Søholt Treatment Plant during the winter from 1985 to 1986.

<u>Odense N.W. Plant.</u> (85,000 PE)

Figure 10 shows the discharge quality from Odense N.W. Treatment Plant of total inorganic nitrogen and ammmonia in 1986 till now (average 1.3 mg/l NH_3, 5 mg/l inorganic N). On an average the plant has been loaded with 70% of the design load. The plant can be seen to nitrify excellently during both winters, but at no time has the temperature been lower than 12°C. The one high measurement in January 1987 was due to a heavy hydraulic and organic overloading, the organic substance loading being 30% above the design basis. Further to the discharge of ammonia reduced discharge quality was recorded for BOD and suspended matter.

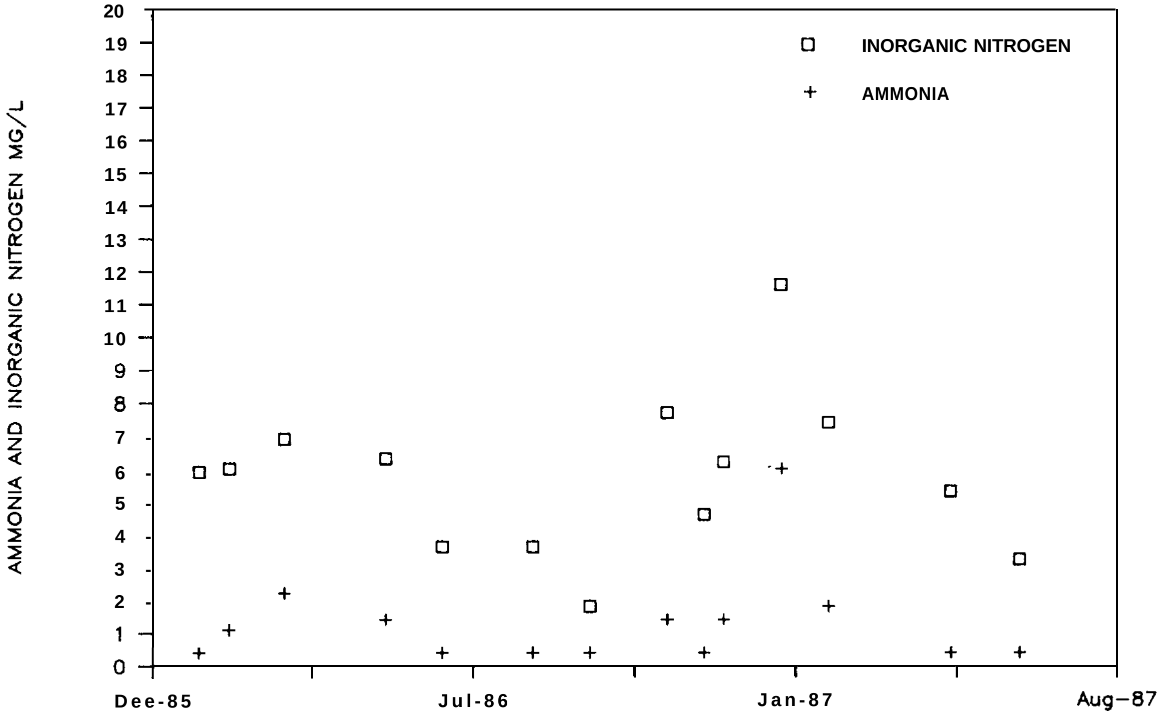

Figure 10. Discharge quality for ammmonia and inorganic nitrogen for Odense N.W. Treatment Plant.

Small Treatment Plant (8,500 PE)
Figure 11 shows the water temperature of a small treatment plant and the corresponding discharge results for ammonia. It can be seen that for long periods the nitrification runs at a temperature well below 5°C. Reduced discharge quality can be demonstrated when the temperature drops to 0°C due to snow melt. However, the reduction is not predominant and only lasts a short time. The plant is estimated to have a load of 60% of the design load.

In figure 12 the calculated sludge age is compared with the temperature and the typical curve for nitrification. It can be seen that the plant nitrifies at considerably lower temperatures than expected!

Therefore, figure 3 can only be used as a guideline.

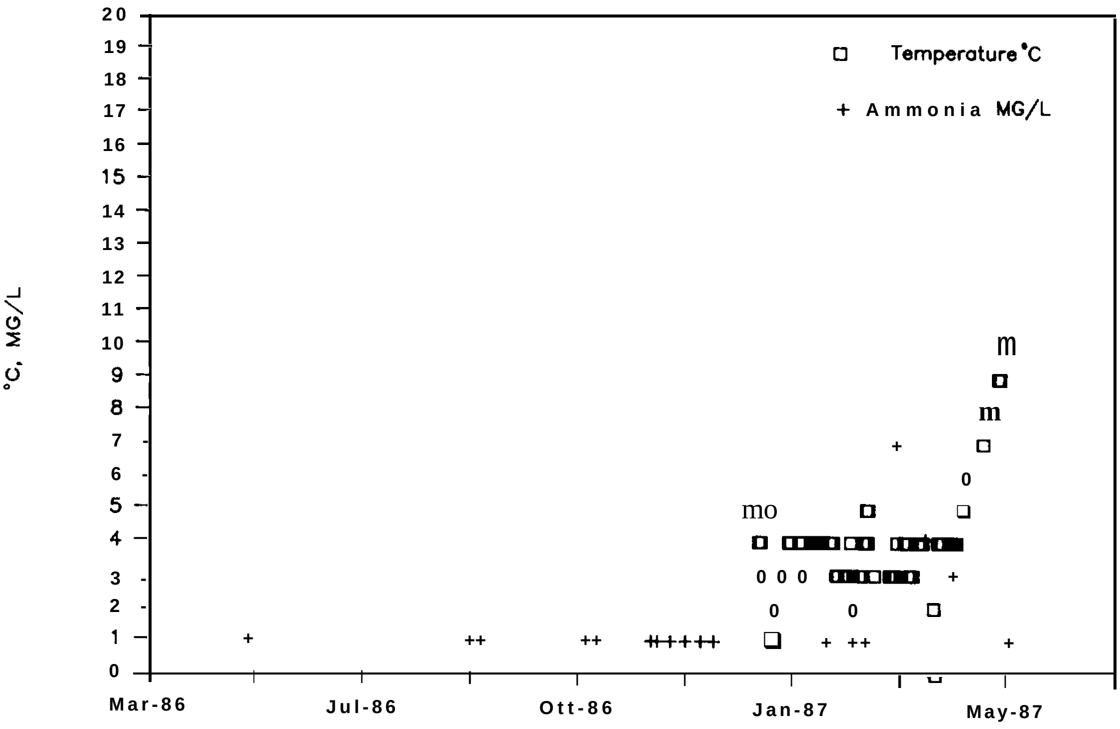

Figure 11. Water temperature and discharge of ammonia at a small treatment plant during the winter 1986 to 1987.

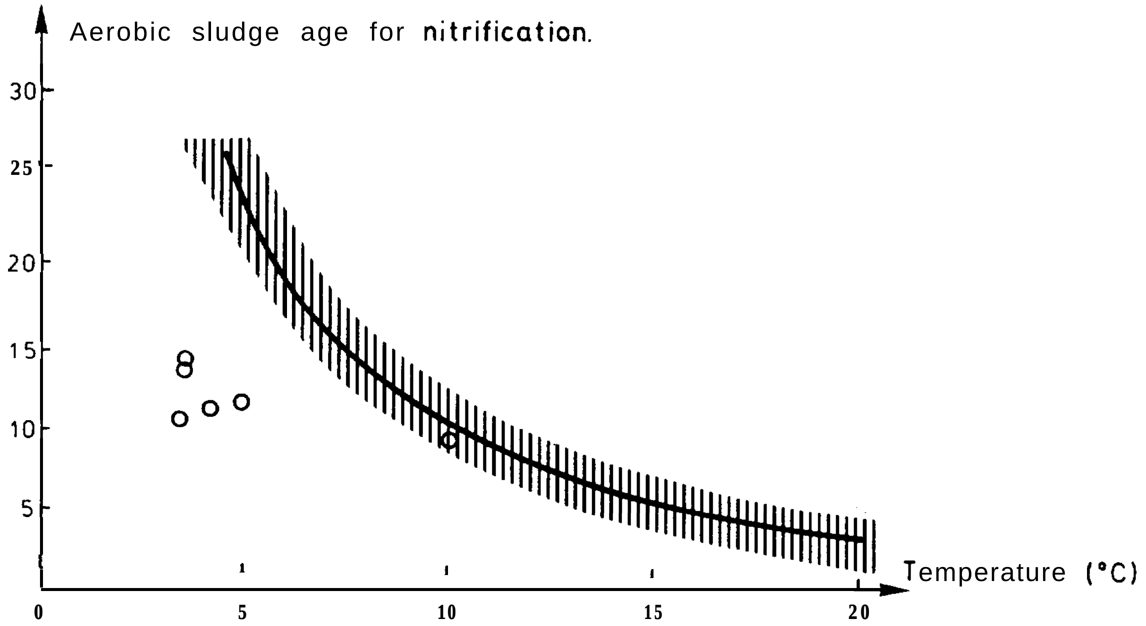

Figure 12. Estimate of the sludge age and nitrifying conditions at a little treatment plant during winter months in 1986 and 1987.

CONCLUSION

It is possible to operate treatment plants with nitrogen removal all the year round under typical and extreme Danish temperature conditions so that requirements for an average nitrogen content of 8 mg/l can be met.

Nitrification, which is the critical process, may be carried out effectively at the treatment plants even at temperatures below 5°C during several months.

Plants performing nitrogen removal require careful operation, if the process is to be secured under unfavourable temperature conditions and unfavourable loading conditions.

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CONTROL OF PHOSPHORUS AND NITROGEN DISCHARGES - PRESENT SITUATION AND TRENDS IN FINLAND

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National Board of Waters and Environment
Finland

1. INTRODUCTION

In order to prevent eutrophication phosphorus removal is required without exception in municipal wastewater treatment. It was taken into practice during the 1970s, and at present, practically all municipal treatment plants have chemical phosphorus precipitation.

In the 1980s, nitrification has been required in several cases, mainly to prevent oxygen depletion in the recipient.

Nitrogen has been considered an important factor limiting growth in some coastal and inland waters and nitrogen removal is now of current interest. As yet, however, no requirements concerning the total nitrogen have been set.

2. TREATMENT METHODS

At the beginning of 1986, 582 municipal treatment plants were in operation, treating the wastewaters of 3.5 million people. Wastewater treatment covered 99 per cent of the population served by sewer systems and 72 per cent of the total population of Finland (Table 1).

Table 1 Municipal wastewater treatment in sewage works designed to serve more than 200 inhabitants 1. Jan. 1986.

Method	Number of plants	People served	
		1000 inh.	%
No treatment		27	0.8
Chemical	61	471	13.2
Biological-chemical	487	3041	85.3
Other methods	34	26	0.7
Total	582	3565	100.0

The share of simultaneous precipitation and the combinations of simultaneous and pre or post precipitation is about 80 per cent of the total plant capacity. Simultaneous precipitation has been introduced in activated sludge plants without significant costructional, operational or economic disadvantages. Ferrous sulphate ($FeSO_4 \cdot 7 H_2O$) is the most commonly used precipitant. The usual dosage corresponds with the Fe to P molar ratio of about 2. Many plants are operated with partial pre precipitation, with addition of ferrous sulphate at two points.

In most cases, nitrification has been carried out in old simultaneous precipitation plants which originally have been too heavily loaded for nitrification. Pre precipitation is often used for prolonging the sludge retention time. The design values of sludge load in combined carbon oxidation-nitrification processes have been 0.06 - 0.08 kg BOD_7/kg MLSS·d.

The alkalinity of waters in Finland is low and a lime dosage of 50 - 100 g $Ca(OH)_2/m^3$ or denitrification is needed in connection with nitrification. Denitrification, which has been used for reducing the operating costs of new nitrification plants, has resulted in a 50 - 60 per cent removal of total nitrogen as a yearly average.

3. TREATMENT REQUIREMENTS

Treatment requirements are expressed as maximum allowable concentration values and minimum percentage reduction values of BOD_7 and phosphorus (Table 2). The figures refer to average results during a specified time period (3, 6 or 12 months), including possible by-passes and process disturbances.

Table 2. Common requirements for municipal wastewater treatment.

Method	BOD_{7ATU}		Phosphorus	
	mg/l	red. %	mg/l	red. %
Direct precipitation	60-70	60-70	1.0	85-90
Simultaneous prec.	15-20	85-90	0.5-1.5	80-90
Post precipitation	15	90	0.5	90

When nitrification has been required, the maximum permitted concentration of ammonia nitrogen has been 4 mg N/l and the minimum reduction 80 - 90 per cent. The reduction calculation is based on influent total nitrogen and effluent ammonia nitrogen.

4. TREATMENT RESULTS

In 1985, the phosphorus load discharged to the receiving waters from municipal sewer systems was 520 tons, which corresponds to 88 per cent removal and effluent concentration of 0.95 mg/l.

For nitrogen the figures were, respectively, 14 400 tons, 31 per cent and 27 mg/l (Table 3).

Table 3. Municipal wastewater load and treatment results in 1985.

	Load t/a	Reduction %	Effluent conc. mg/l
BOD_7	16 500	a5	31
BOD_{7ATU} (est.)	a 500	92	16
Phosphorus	520	88	0.95
Nitrogen	14 400	31	27

5. SUMMARY

The present situation in Finland regarding phosphorus removal is satisfactory, even if the operation and management of the treatment plants can still be improved.

The share of nitrification will be increased. For economical reasons many nitrification plants will utilize denitrification, which improves the removal of total nitrogen.

The question of need for requirements concerning nitrogen removal is still open.

Quality of Stormwater

- state of the art in Schleswig-Holstein -

by Günther Leymann, Kiel

Federal Republik of Germany

In the German part of the Baltic Catchment area, in Schleswig-Holstein, seperate systems are mostly prefered. Only in elder parts of some towns you will find few combined sewer pipes.

Investigations showes, that even the rain-water is polluted with the same substances like domestic sewage, it is only a question of concentration.

Ph	4,5
NH_4-N	2 mg/l
NO_3	1 mg/l
P tot.	0,03
Pb	0,01
Cd	0,002 mg/l

(mean concentration 1984/85 by Landesamt für Wasserhaushalt und Küsten Schleswig-Holstein)

These figures showes also, that measures aiming to the reduction of the air-pollution take also affect to the waterbody.

The pavement run off often is additional polluted by oil, rubber and other carbon compounds.

The mean pollution load of storm water is for some parameters general the tenth of the rain water load.

BOD	10 mg/l
COD	50 mg/l
N tot.	3,4
P tot.	0,4
Pb	0,1
Cd	0,001

(mean Concentration of pavement run off, Brunner 1977 and Krauth 1978).

Normaly there is no treatment of storm-water in connection with seperate systems, but beside of oil seperators at petrol stations we run a lot of storm-water tanks to equalize the discharge.

These tanks, sometimes lagoons, additional fittet out with an oil barrier in case of an accident,.work like an sedementation tank. Beside these funktions they are often very nice biotops.

In some cases e.g. at Auto wrack places, uncovered industrial areas or quays where fertilizer is handled we run overflow tanks with the facility to discharge all sewage to a central treatment plant, or in case of the quay only the first highly polluted flush.

Beside these measures in my opinion roofs are the best method to avoid highly polluted storm-water.

Experiences and remarks to the advanced sewage treatment in Schleswig-Holstein

by Günther Leymann, Kiel
Federal Republik of Germany

1. Introduction

Schleswig-Holstein, situated in the north, is one of the eleven federal states in Germany. It borders in the west to the North Sea in the north to Danmark and in the east to the Baltic. Schleswig-Holstein has about 2.6 millions inhabitants, and about 1.0 million of those live in the catchment area of the Baltic.

The specialyties for the water management are about 300 lakes, slow floating short waters and more than 1.100 km coastline.

In 1971 the district government of Schleswig-Holstein defined and published the project “Sewage treatment an protection of water, general line". One of the aims of these plan was a reduction of phosphorus and BOD in order to stop the increasing eutrophication in the lakes due to the nutrients, especially phosphorus.

The development of sewage treatment in Schleswig-Holstein can be seen at fig. 1. In the German catchment area of the Baltic today nearly 90% of all wastewater (urban areas and industry) is being collected and treated at central sewage treatment plants at least mechanical and biological (including sludge treatment).

In order of this fixed plan today we run about 53 treatment plants with additional phosphorus precipitation, further 5 of those plants are under construction, 15 are designed and 10 are projected. Most of these plants with advanced treatment are in operation in the East of Schleswig-Holstein (see fig. 2), that means in the catchment area of the Baltic. In order to the plan I mentioned, in the recent 15 years the investments for protection of waters amount up to 1.9 billion German marks (DM).

2. The German water act

Before I try to explain the German point of view concerning the advanced wastewater treatment especially the elemination of nutrients I must explaine some definitions of our water act.

2.1 Wastewater and permission for discharging

Every wastewater discharging into a surface-water must be promitted by the local water authority.

"Wastewater" means: "Water, which is polluted by human usage or changed otherwise in his characteristic. Pavement run off is wastewater too." All wastewater discharges must fulfil special minimum requirements concerning the concentration of the parameter BOD, COD, SS and after 1. 1. 1992 additional P and NH_4-N.

The water authority is able, due to the quality of the receiving body, to require a higher discharging standard (fig. 3). To fulfil these requirements the use of "Advanced Wastewater Treatment" in our definition is necessary. The permission for discharge contents max. concentration values for some parameters, which are not allowed to ecxeed. These values are fulfilled if the arithmetic mean of the last five 2-h-samples doesn't exceed these values. Each exceeding of these values result in a higher sewage tax and can be punished as a crime. Dilution isn't allowed and no solution for pollution!

Due to these regulations it must be possible to fulfil these requirements and to avoid a crime by mean of available waste water technic,

2.2 Stages of technic

There are three or four stages of technical perfection:

1. General employed technic,
2. state of technic (best available economic acceptable technic),
3. state of technical science,
4. state of technical scientific discussion.

Due to the remarks I made, the highest stage, which can be required by the authority is the stage "Best available technic". That means, it must be able to buy these technic, the producer must be able to guarantee the promised result - and the user must be able to believe him!

3. Consequence to the “advanced wastewater treatment”.

3.1 Phosphorus

Today in Schleswig-Holstein about 53 treatment plants with phoshorus precipitation are in operation. Due to the requirements of the recipient and due to the joy for experiments of some engineers we run all sorts of precipitation:

pre-simultan-combined-after-precipitation etc. (fig. 4).

3.1.1 Example 1: Simultan precipitation

The simultan-precipitation is the most applied method. Fig. 5 shows the results of P-precipitation at the treatment plant Kiel.

The outlet concentration P tot. during the last year:

P tot. 1.87 mg/l
(mean value January 1985 to May 1987 unfiltred sample).

This treatment plant is the largest treatment plant in Schleswig-Holstein. It is designed for about 500,000 population equivalents and at present a filtration stage is under construction.

3.1.2 Example 2: After-precipitation

The treatment plant of the city Raisdorf is equipped with two biological treatment stages and an after precipitation as a combined flotation and filtration. The outlet concentration P tot. during the last years:

0.37 mg/l (unfiltrated sample, mean value).

Figures 6 and 7 gives an overview about the efficiency of these plant.

3.1.3 Consequence concerning "phosphorus"

The technic for phosphorus precipitation, without doubt, is an available economic acceptable technic, which can be required by the water authorities.

3.2 Nitrogen and nitrogen compounds

At present phosphorus but especially nitrogen are in the focus of interest, because by human activities during the past decades increasing amount of these two nutrients have been released to the sea. These elements, which we call nutrients are available only in limited quantities. They control plant growth according to Liebig´s law of the minimum factor, when other enviromental factors are optimal. What are the effects of these increasing inputs of nutrients7 Since some years this question is discussed among marine Scientists. It seemed to be, that nitrogen is the nutrient, that generally limits growth in the Baltic Proper, the Sound, the Kattegat and the Skagerrak. But there is also another question discussed among the Scientists: The right method of nitrogen-analysis. For example, when the Kjeldahl wet-oxydation method is applied, more organic nitrogen compounds are included, which are not easyly available to plants. Applying pyrolytic methods the results are also problematic. I don't intend (and I'm not able) to complete this theme, but we should be carefull with nitrogen budgets. Before going on the discussion about the advantages and possibility of a requirement of nutrient removal, especially nitrogen oxydation (nitrification) and nitrogen removal (denitrification), some remarks to the nutrient-sources.

3.2.1 Nutrient sources

Concerning phosphorus, wastewater from urban areas is the main source of these nutrient. More than 55 % of the total load receiving our waters result from domestic sewage. The calculated specifications for nitrogen coming from domestic sewage differ from 10 % up to 22 %. Last calculations in Schleswig-Holstein resulted in 16 %. Fig. 8 gives some details for nitrogen-sources, but at the moment only such sources are of interest, which we can influence by wastewater engeneering. In case of nitrogen we talk about 16 %.

3.2.2 Example 3: Nitrification/Denitrification

Now I intend to present you the brandnew new wastewater treatment plant of Husum, as an example for a modern plant with a controled nitrification and since some month with denitrification.

Husum is a small village at the North Sea coast, but that doesn't mind in this example, with about 25 000 inhabitants, important tourism during summer and some industry. The main content of the pollution is caused by a slaughter house, so that the plant had to be designed to a seize of 110.000 populant equivalents (PE). The requirements for discharge are shown in fig. 9.

Because the purified wastewater is disoharged into the harbour, the discharge requirements specify not only the reduction of carbon compounds, but also a reduction of nitrogen compounds to an ammonium (NH_4-N) concentration of 5 mg/l at a water temperature of 15° c.

In accordance with the inlet conditions - the nitrogen oxydation must be secured at any time - the arithmetical BOD sludge load had been laid out to 0.12 kg/kg x d at max. load. (This correlates to a sludge age of about 6 days). The plant has been designed as an oxydation ditch. (Design parameters fig. 10).Tankform and aerator arrangement (fig. 11) create best conditions for a simultan nitrification/denitrification. The return sludge and mechanicaly pretreated wastewater are intermixed in a contact tank and are discharged into the anoxic area of the aeration basin. The input of oxygen is done by the rotors 1 and 6 as continues runners. The NH_4-N concentration is continually measured at the tank effluent and with exeeding a certain concentration the necessary additional oxygen input is done by the step-wise shutting in of the other rotors. These control and steering equipment is additional controled by an oxygen regulation.

3.2.2.1 Discussion of results

Figure 12 shows the results of the first test december 1986 until january 1987. The reached purification effect was much better than requested, but the plant works only with a capacity of 60 % compared with the design data, Even under extrem conditions, 3.5° C water temperature, the required ammonium nitrogen concentration of 5 mg/l wasn't exeed at any time (fig. 13). But the nitrat-nitrogen value was higher than expected. A control of the oxygen concentration along the slope results, that there was no point with an oxygen concentration below 0,2 mg O_2/l (picture 14). After changing the rotors for the basic load (instead of the rotors 1 and 6 now the rotors 3 and 4) for the basic load the results of the plant change in a significant way (fig. 15).

3.2.3 Consequence concerning "nitrogen"

Well, these very simple treatment plant, beside of controling equipment, could be a good argument for the general require of nitrogen reduction. But please have a look to fig. 12. The corresponding mean sludge load was about 0.075 kg/kg x d, and there is no possibility for calculate the denitrffication effiency, because there is no value for total nitrogen (TKN).

I think Husum is one of the necessary experiences and it shows, that nitrification is feasible without operation problems. But it is also an example for wasting energy and last not least money. Investments in wastewater treatment must not be in any case economical, but it's necessity must be explainable.

Today the decision to require and define a max. outlet concentrations for nitrogen would halve our treatment capacity and result in an enormous investment. As a subvention to the building trade, – o.k. But as an argument resulting from

- the water situation,
- the expected efficiency,
- the state of science about the minimum factor,
- the consideration of the sources and
- the lack of available wastewater-technic

it should be very difficulty to get the needed money.

3.2.3.1 Available technic for nitrogen removel?

The biological proceedings for the nitrogen removal are well known and often practised by operation reasons. In this case denitrification as an method for nitrogen removal is of interest. Precondition for denitrification is the total nitrification. The only practical design parameter for the nitrification is the BOD-sludge load (B_{TS} 0.15 kg/kg $_x$ d) believing, that at a certain sludge age the conditions for the Nitroso and other bacterial specialists are good enough for life. There are a lot of experiences, where this assumption seemed to fit not in any case.

Figure 16 gives a little overview about the different technical ways for nitrogen removal. But there is none fixed rule for designing and it is recommended to make tests in a large as possible scale, for getting the best fitting method. Methods for denitrification today are object of technical scientific discussion, but they are not ready for the general application1 Special in Germany, due to our water act, we would criminalize all treatment plant operators, if we would require a limitation for nitrogen total, because there is no “available technic” in the mean of our water act. This is, beside economical questions, today our main argument against the limitation of nitrogen.

4. Final remarks

Beside this, please allow following remarks. The removal of nitrogen needs the "high-tech" of wastewater engineering. It’s possible, but what is not possible? In my opinion in the whole catchment area of the Baltic we first need thousands kilometer of sewer-pipes and a lot of simple treatment plants with mechanical and biological purification. After reaching that aim, we should look further. I’m a little bit afraid, that while discussing high-tech of wastewater engineering, we don’t recognize that we lost view to the reality. It‘s a fact that there are differences in the econimical sources and possibilities within the Baltic-states. I think, we should wait a little and reduce the speed in order to go the way together.

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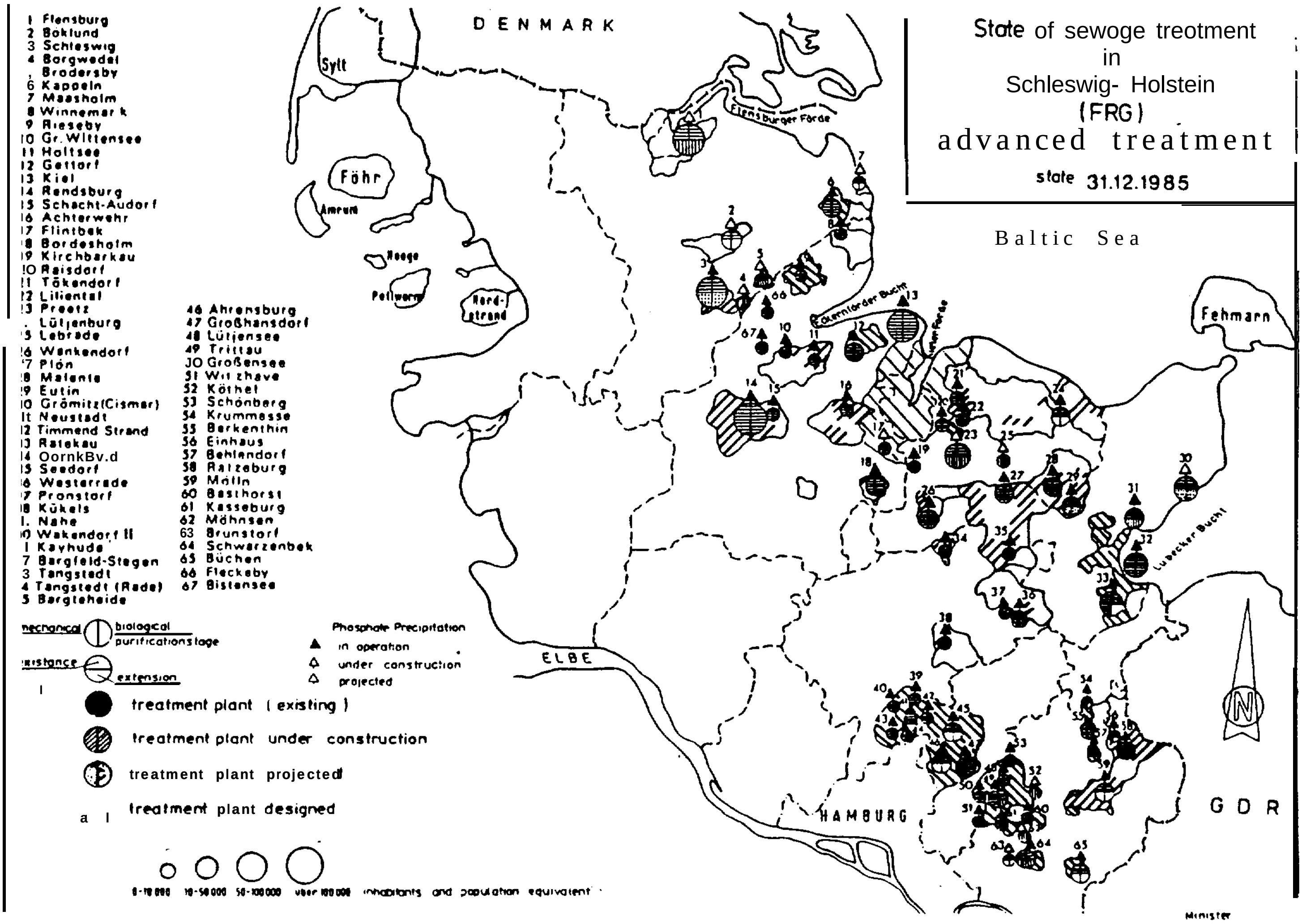
Development of the sewage treatment **in Schleswig-Holstein (FRG)**

year	**number** or **inhabitants in 1000 inh.**	central sewageplant * **joined on in** 1000 **inh.**	**in %**	**not joined** on in **1000** Inh.	**in %**	**sewerpipe** total length **in km**
1971	2.555	1.400	54.8	1.155	45.2	**5.900**
1976	2.582	1.748	57.7	034	32.3	**9.300**
1979	2.599	1.918	73.8	681	26.2	11.370
1984	2.616	2.120	81 .o	496	**19.0**	13.400

Fig. 1 *) central **sewageplants =** at least **mechanical** and biological **stage** **and** sludge treatment(')



Fig

on ground of clearness only plants with advanced treatment are figured

Requirements:"advanced uasteuater treatment"

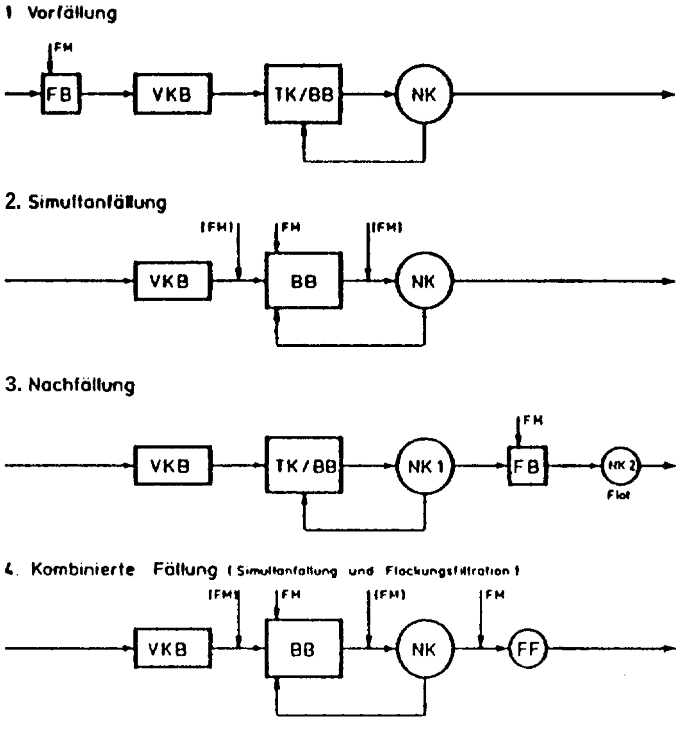
Parameter	Ablaufkonzentrationen (mg/l)	
	Uberwachungswerte *	mittl. Ablaufwerte
1	2	3
Chemischer Sauerstoffbedarf CSB (COD)	60,0	37,5
Biochem. Sauerstoffbedarf BSB_5 (BOD_5)	16,0	10,0
Gesamtohosohor P (P_{tot})	1.6	1,0

- Die in Soalte 2 angegebenen Werte gelten für Einleitungen ≥75 m^3/d. Es sind Uberwachungswerte an 2-Stunden-Mischproben, die auch dann als eingehalten gelten, wenn der arithmettsche Mittelwert aus den letzten 5 im Rahmen der staatlichen Gewässeraufsicht durchgeführten Untersuchungen unter dem angegebenen Wert liegt. Dies ist aus statistisch-mathematischen Gründen im allgemeinen nur zu erreichen. wenn im praktischen Klärwerksbetrieb die in Soalte 3 angegebenen mittleren Ablaufwerte eingehalten werden.

Fig. 3 (3)

Methods for precipitation

↓ FH= addition of precipitant



VKB : Vorklärbecken
N K : Nachkllrbeckon
BB : Belebungsbecken
FM: Fällmitteldoolerung
TK : Tropfkörper
FF: Flockungsfilter
FB : Fällungsbecken

Fig. 4

Treatment plant

Kiel 500 000 population equivalent

P_{tot} (mean concentrations)

1985: 1,4; 1,5; 1,2; 2,6; 1,3; 1,1; 1,7; 6,0; 1,6; 2,4; 2,0 und 4,5 mg/l
1986: 1,9; 1,7; 1,9; 1,4; 1,0; 1,2; 0,72; 1,7; 1,1; 1,0; 1,9 und 2,3 mg/l
1987: 3,1; 1,5; 1,3; 1,9 und 1,27 mg/l
i.M. = 1,87 mg/l

Fig. 5

Treatment plant Raisdorf - Phosphorusreduction

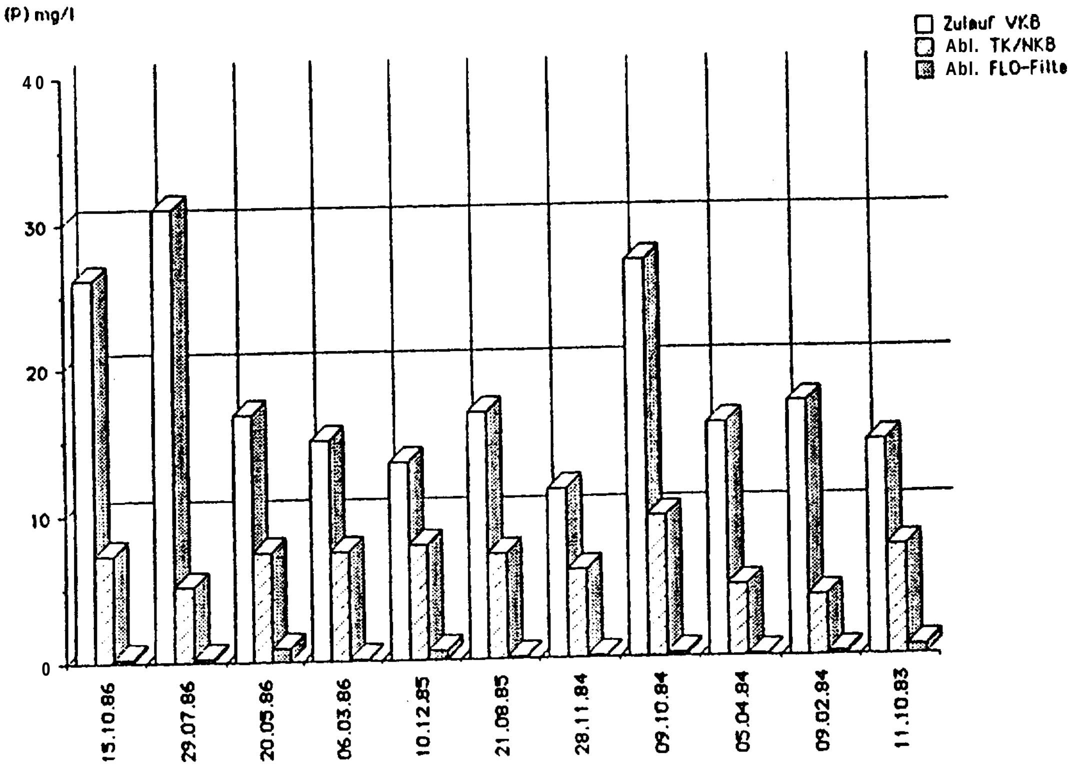


Fig. 6

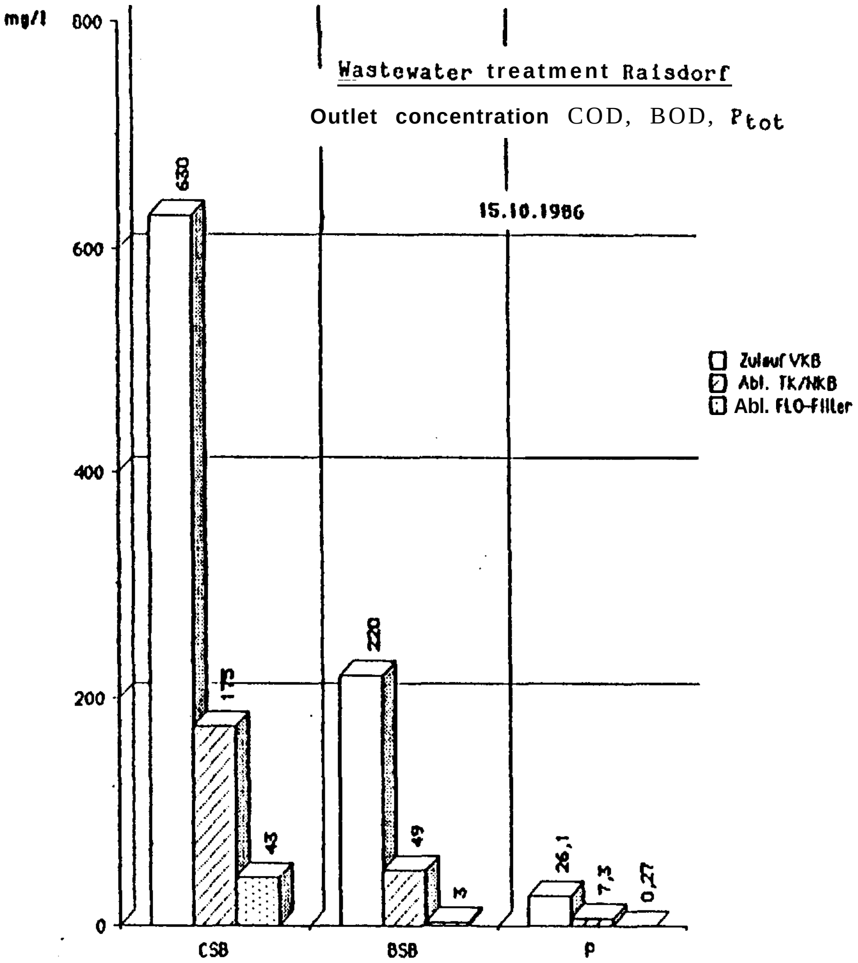


Fig. 7

Sources and amount of nitrogen in the waters of the FRG

Herkunft		being discharged into the waters		
	t/a	%	tN/a	%
1	2	3	4	5
1. domestic sewage	240.000	50	120.000	22
2 manure eta.	650.000	8.	52.000	10
3. fertilizer	1.100.000	5	55.000	10
4. natural mineralisation aore			151,000	28
greenfields			80.000	15
forest			59.000	12
			290.000	55
5 precipitation	200.000	4	8.000	3
6 N-Fixing	255.000	4	10.000	
total	2.455.000		535.000	100

The nitrogen load originates from

○ agriculture	61%	○ sewage treatment	10
○ forestry	26%	○ rural areas	2
○ industry	1%		

National Swedish
Enviromeot Protection ,
Board (6)

Fig. 8 (3)

Load parameter for design

sewage inlet	=	12,750	m^3/d
stormwater inlet	=	1,300	m^3/h
BOD_5 charge	=	6,600	kg/d
BOD_5 concentration	=	520	mg/l
nitrogen charge	=	1,320	kg/d
nitrogen concentration	=	104	mg/l

Requirements for discharge

BOD_5 concentration	=	20	mg/l
COD concentration	=	130	mg/l
SS	=	0.3	ml/l
NH_4-N concentration (at 15° C)	=	5	mg/l

Fig. 9 (1)

Oxydation ditch (design parameters) (1)

max. BOD_5 space load	=	0.6	kg/m' x d
SS	=	5.0	kg/m^3
max. BOD_5 sludge load	=	0.12	kg/kg x d
max. nitrogen sludge load	=	0.24	kg/kg x d
final clarifier volume 2 x 3,300	=	6,600	m^3
final clarifier surface 2 x 700	=	1,400	m^2
mean water depth - final clarifier	=	4.50	m
developed length per tank	=	220	m
water depth	=	3.20	m
channel width	=	7.00	m
eff. volume = 2 x 5,500	=	11,000	m^3
installed aeration capacity per tank 6 mammoth rotors dia	=	1,000	mm
length	=	1.50	m
O^2 input	=	54	kg/h
O^2 input - total plant 12 x 54	=	648	kg/h

Fig. 10

Treatment plant Husum (1)

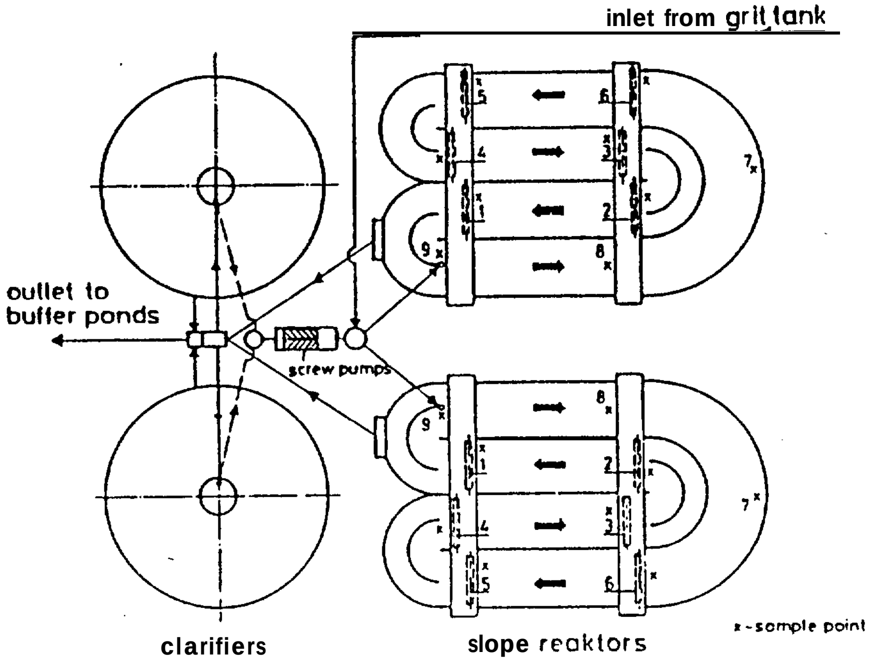


Fig. 11

		december 86	january 87	april 87
		start up		control unit modif ied
inlet values				
sewage inlet	m³/d	8,130	5,545	0,720
BOD concentration	mg/l	464	515	494
BOD charge	kg/d	3,772	2,056	4,312
COD concentration	mg/l	587	792	675
TKN concentration	mg/l	61	87	93
TKN charge	kg/d	496	402	811
tank values				
SS	kg/m³	4.4	5,2	5.4
BOD sludge load	kg/(kg x d)	0,080	0,052	0.075
N sludge load	kg/(kg x d)	0,011	0,009	0,014
temperature	°C	9.8	3,5	9.5
outlet values				
BOD concentration	mg/l	2,9	7,6	6,4
COD concentration	mg/l	38,0	47.1	44.5
NH_4-N concentration	mg/l	0,2	2,1	1,8
NO, -N concentration	mg/l	13.3	33,5	2.3

Fig. 12 Summarized results of investigation [illegible]

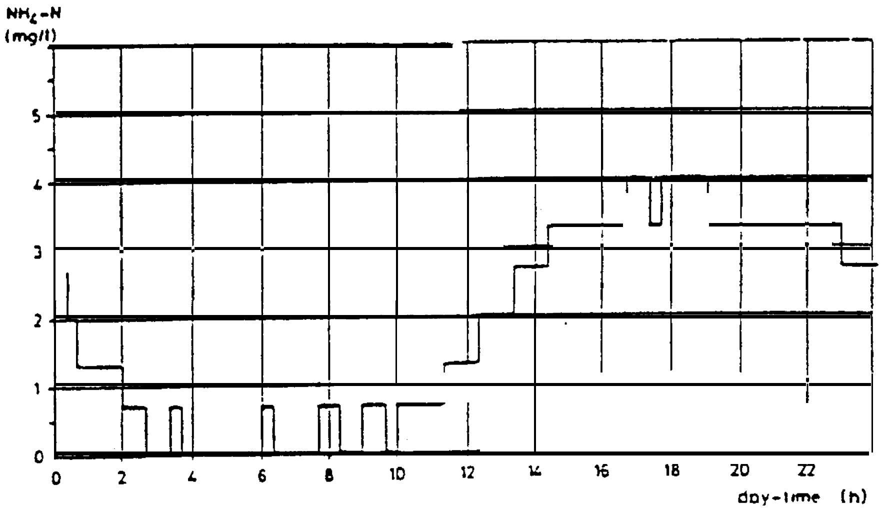


Fig. 13 Course of NH_4-N concentration in the activation tank at a temperature of 3.5 °C. (1)

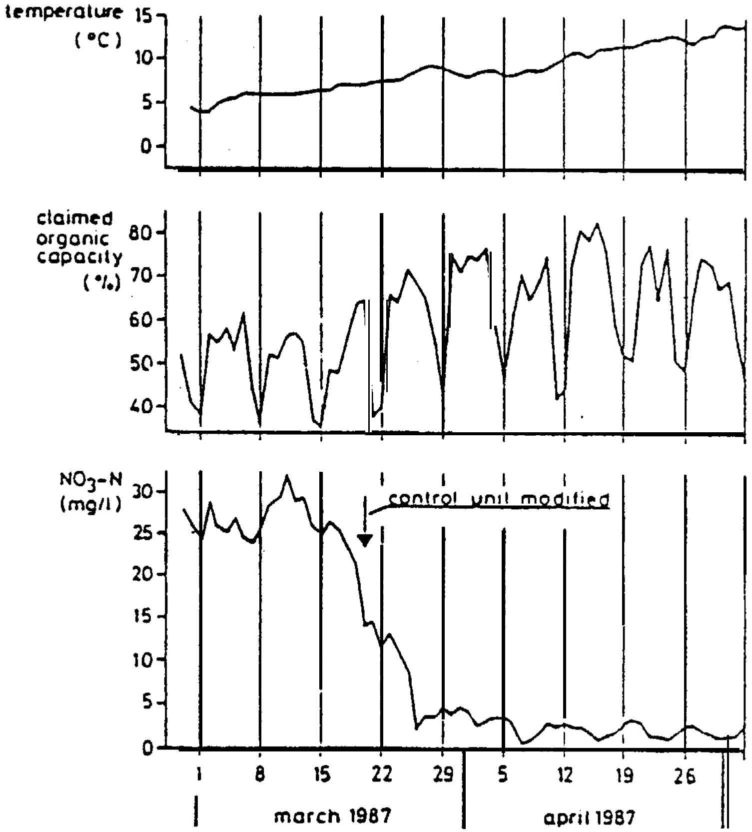


Fig. 15 Course of NO_3-N, organic capacity and temperature in the activation tank (NH_4-N never exceeded 4 mg/l). (1)

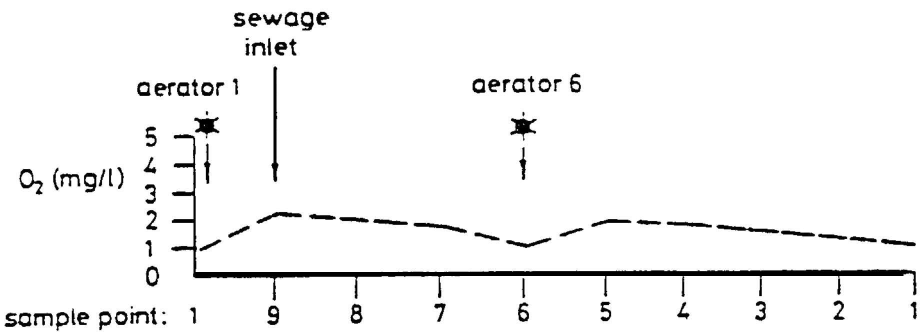
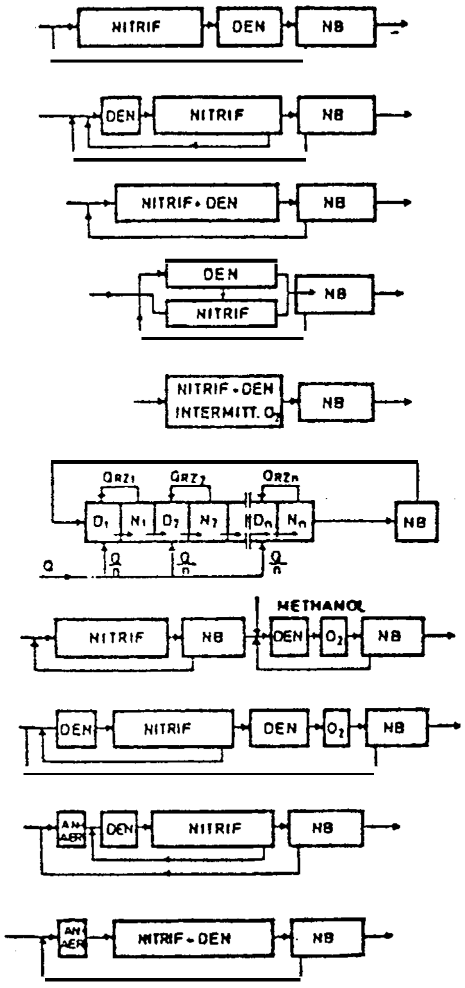


Fig. 14 Oxygen profile along the slope. (1)



1) Nachgeschaltete
Denitrifikation

2) Vorgeschaltete
Denitrifikation

3) Simultane
Denitrifikation

4) Alternierende
Denitrifikation
Bio Denitro

5) Intermittierende
Denitrifikation

6) Kaskaden-
Denitrifikation

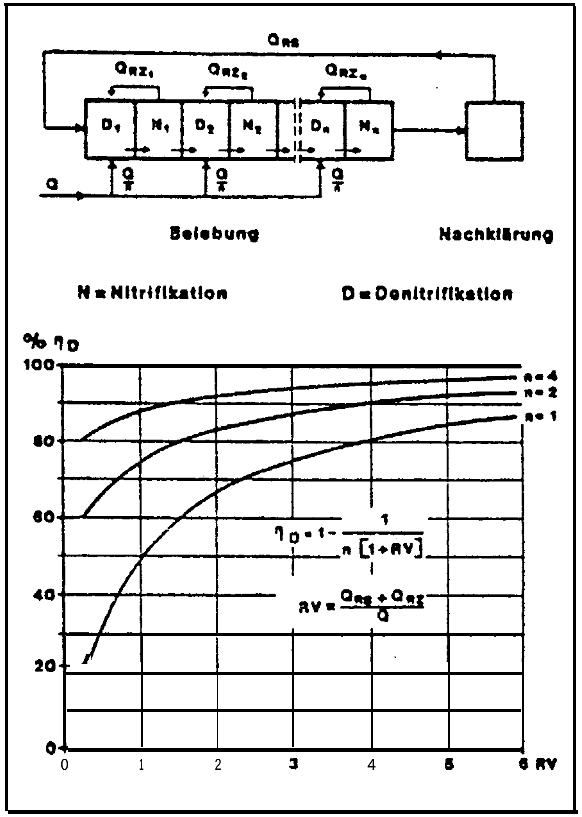
7) Getrennte Denitrifikation
+ Methanolzugabe

8 , Kombinierte
Denitrifikation

9) Vorgeschaltete Anaerob-
u. Denitrifikationsstufe

10) Simultane Denitrifikat.
mit vorgeschalteter
Anaerobstufe

Fig. 16(5)



η_D = Deni· Efficiency
n = quantity of Den-stages
RV recirculation ratio

Q_{RS} = Rücklaufmenge aus Nachklärung
Q_{RZ} = Rezirkulationsmenge innerhalb einer aus D- und N-Stufe bestehenden Kaskade
Q = Zulaufmenge

Fig. 17 Max. denitrification efficiency depending of the recirculation ratio and the number of stages.

RESEARCH AND DEVELOPMENT IN PHOSPHORUS AND NITROGEN REMOVAL FROM MUNICIPAL WASTEWATER IN POLAND

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INTRODUCTION

Nitrogen and phosphorus compounds are one of the most important constituents of nater contamination. Their presence in wastewaters discharged into the surface waters creates a danger of recontamination by promoting aquatic growth and eutrophication.

At the sane time, both these substances are the main nutrients controlling biological growth in the aquatic environment. Phosphorus has been considered the limiting nutrient in Poland.

Protection of receiving waters, lakes and impoundments is only one reason for developing advanced wastewater treatment technologies presence of an excess nutrients results also in poor water quality making water unsuitable for the prospective municipal and industrial users. The latter becomes very significant in the highly. industrialized region of South Poland where the deficit of water resources is severe.

Water that has been recovered from municipal sewage at nater reclamation plants in Siicsia and Cracow regions is planned to be reused in the cooling systems of metal and electric power industry. Nitrogen is a limiting nutrient for this application. Maximum concentration of ammonia ions allowed in the cooling systems due to a corrosion hazard ranges from 2 to 5 g N/m^3. The actual, permissible concentration depends upon the type of the cooling cycle and water condensation as well as water alkalinity.

Considering all above it seems natural that the problem of nutrients and their presence in wastewater have. received considerabl attention in Poland for many years. It has been reflected in both research and legislative actions concerned with environmental protection issues. Unfortunately, any practical applications and implementation of nutrients removal technology in municipal wastewater treatment plants are still in a very preliminary phase and more delays in their commissioning schedule are expected.

The Water Law, amended in 1974, sets very low stream concentration standards for both nitrogen and phosphorus in surface inland waters. The concentrations are specified for each of 3 purity classes of surface waters existing in Poland, and established according to their planned usage.[7].

They are:

- class I - waters supplying the population with drinking water, supplying food processing industry and salmony fish raising;
    - phosphates- less than 0.2 g PO_4/m^3
    - ammonia nitrogen - less than 1.0 g N/m^3
    - nitrates - less than 1.5 g N/m^3
    - organic nitrogen - less than 1.0 g N/m^3
- class II - water for animal farming, water contact recreation;
    - phosphates - less than 0.5 PO_4/m^3
    - ammonia nitrogen - less than 3.0 g N/m^3
    - nitrates - less than 7.0 g N/m^3
    - organic nitrogen - less than 2.0 g N/m^3

- class III - water supplying industry and agricultural irrigation
    - phosphates - 1.0 g PO_4/m^3
    - ammonia nitrogen - 6.0 g N/m^3
    - nitrates - 15.0 g N/m^3
    - organic nitrogen - 10 g N/m^3

Despite existing water quality standards, local water protection authorities tended to overlook their execution with respect of nutrients removal in the 1970-ties even when it would be advisable due to an insufficient dilution in receiving waters. Such policy was a result of high costs and technical problems associated with new technologies of nutrients removal as well as the difficulties with an accurate transfer of water standards into maximum wastewater concentrations. Municipal wastewater treatment plant design criteria were mainly based on organic loading expressed as BOD_3.

Existing stream water quality standards for surface waters were difficult to enforce in case of lakes and impoundment reservoirs and that is why the Institute of Environmental Protection in Warsaw has recently developed a set of new regulations. They are based on the wastewater effluent limits and may be used in those lake drainage basins which are especially eutrophication sensitive. Proposed new standards include also phosphorus considered as a limiting nutrient. A legislative action on this issue is still pending.

Lake standards apply to the same f-class scale of water purity:

Class I - direct wastewater discharge to the lake is not allowed; any discharge to the rivers or streams located within the lake drainage basin cannot exceed the total phosphorus level in wastewater effluent of 1 g P/m^3

Class II and III - direct wastewater discharge to the lake is allowed; maximum concentration of total phosphorus in wastewater discharges either to the lake or to rivers and streams in its drainage basin cannot exceed 1 g P/m^3.

These standards comply with the similar ones existing in most of the countries.

CHEMICAL PRECIPITATION OF PHOSPHORUS

Concentration of total phosphorus found in Polish municipal wastewaters ranges from 3 to 15 g P/m^3 with the average concentration not exceeding 10 g P/m^3. As an example, wastewater originated in Warsaw and Cracow had approximately 6 g P/m^3; in Wrocław the average concentration in 1985 was 10 g P/m^3.

The conventional biological treatment plants operated in our country achieves from 30% to 50% removal of total phosphorus /average 40%/. Only some Of them, like the City of Żywiec treatment plant, which receives a significant volume of wastewater from paper industry, arc able to obtain 60% to 80% removal.

A limited efficiency of biological treatment is caused by the fact that based on an average conposition of cell tissue of $C_{15}H_{105}O_{30}N_{15}P$ about 1.7 percent by -weight of phosphorus is required and nay. be taken up for incorporation into biomass. Thus, removal of phosphorus in a conventional biological system depends on phosphorus to organic carbon /TOC/ ratio.

Conventional biological treatuent, as it was, turned out to be inadequate to meet high quality standards set for I class stream and lake waters and in the late 1970s Poland had to introduce, developed in other countries, process of chemical precipitation of phosphorus. The principal chemicals used for this purpose are lime, alum and ferric chloride or sulphate. As it was found in a comprehensive study $FeSO_4$ added to the activated sludge aeration tank appeared to be optimal; it resulted in the effective phosphorus removal, though pH adjustment with lime was necessary in some cases. Polish research - design office CTK, together with VEB Projektierung Wasserwirtschaft Halle /GDR/ has developed a set of standard installations for chemical preparation and dosing.

Also, the Technical University at Poznań developed a detailed design criteria for chemical precipitation of phosphorus in an activated sludge type package plant "Bioblok" working in an extended aeration mode. The applied dosages of precipitant vary from 40 to 150 g $FeSO_4 \cdot 7H_2O/m^3$ depending on the required degree of phosphorus removal. The dosage has been expressed as a molar ratio Fe : P that yields to the ratio of orthophosphates to total phosphorus P_{ort}/P_{tot}. The latter becomes a convenient indicator of process effectiveness; for example, a dosage Fe : P = 2 :1 results in an effluent orthophosphates concentration 0.2 g P_{ort}/m^3, which amounts to 1 g P_{tot}/m^3, providing that $P_{ort}/P_{tot} = 0.2$.[5].

If tertiary treatment process include also filtration, additional 30% phosphorus is removed. It has been found, that the chemicals introduced into the aeration tank do not inhibit the nitrification, and enhance sedimentation and dewatering of mixed primary and activated sludge.

Phosphorus removal in an activated sludge aeration tank with $FeSO_4$, as a main chemical, has been recently implemented in many wastewater treatment plants in Poland. They will cover a wide range of capacities and some of them are already in their final stage of construction. Some of these plants are located along the coast with a direct discharge into Baltic Sea. They are:

1/ Gdańsk-Wschód treatment plant /180 000 m^3/d/; mechanical treatment plant will be upgraded to an activated-sludge treatment plant with nitrififaction, denitrification and chemical precipitation of phosphorus in an seration tank. Maximum dosage of chemical /150 g $FeSO_4 \cdot 7H_2O/m^3$/ resulted in 94% removal of phosphorus and final effluent concentration 0.6 g P/m^3. The plant is scheduled to begin its operation in 1992.

2/ Gdynia-Dębogóra treatment plant /120 000 m^3/d/; activated sludge treatment plant /system A-B/ with chemical precipitation of phosphorus. The plant will come into operation in 1990.

3/ Swarzewo treatment plant /12 000 m^3/d/ receiving wastewater from Puck and Władysławowo . The construction of biological reactors "Biooksyblok" with chemical precipitation of phosphorus is almost completed.

4/Kartuzy treatment plant /1200 m^3/d/; expending of the existing biological treatment plant with chemical precipitation of phosphorus using $FeSO_4$.

The other areas where there is a need for construction of wastewater treatment plants combined with phosphorus removal are impoundment drainage areas located on the Upper Vistula tributaries. They are a main source of I class drinking water for Cracow and Upper Silesia regions.

A very interesting, comprehensive study has been done at a pilot-scale treatment plant in Wrocław. Chemical precipitation of phosphorus with aluminium and iron salts has been performed on a mixture of municipal and industrial wastewaters, with an average phosphorus concentration 4.8 g P_{tot}/m^3. The chemicals were added at a variety of different points in the treatment process[.6].

Conventional biological treatment process itself /sludge organic loading 0.2 - 0.38 g BOD_5/g MLVSS·d/ was able to achieve severage 49.3% removal of total phosphorus and 40% removal of phosphates.

The best effects were observed for alum, though, for economic reasons, the precipitation with ferric sulphate was chosen for further application.

Phosphorus removals with $FeSO_4$ varied with the point of chemical addition, and were: in primary sedimentation - 94%, in biological treatment - 96.5%, in tertiary treatment - 99.8%.

PHOSPHORUS AND NITROGEN REMOVAL IN BIOLOGICAL WASTEWATER TREATMENT

In recent years there has been a growing interest in nutrients removal both through an extensive processes carried out in natural conditions and through advanced intensive methods carried out in a sophisticated man-made installations.

1. Nutrients removal in stabilization ponds

Accumulation of nutrients in algae and solids settled on the bottom was estimated in two kinds of stabilization ponds. The research was done in full-scale at Lowicz treatment plant /15 000 m^3/d/. Percent of total annual nutrient loads accumulated in solids were:

- Facultative pond. total nitrogen 8%
  total phosphorus 2.3%
- Aerobic pond : total nitrogen 31.5%
  total phosphorus 5.9%

A significant amount of nutrients was also accumulated in the algae , which should be harvested on a regular basis. Annual combined removal of total phosphorus observed in both ponds was 55%; total nitrogen 51% with a summer high value 70%. Nutrients removal was accompanied by COD5 reduction up to 90%. [2].

2. Biological nitrification and denitrification

Biological removal of nitrogen, as the only nutrient, is recommended in case of industrial wastewaters coming from fertilizer industry; municipal wastewaters treatment combines usually both nitrogen removal and chemical phosphorus precipitation, as it is designed at Gdańsk-Wschód treatment plant.

Construction of the plant followed a laboratory-scale research conducted at the Institute of Environmental Protection in Gdańsk. The system tested comprised of an one-sludge activated sludge system with preliminary denitrification. The process parameters were: biomass concentration = 3000 g/m^3, recirculation rate - 300%, temperature 17°C /summer/ and 10^oC /winter/, hydraulic detention time: 6 hours /denitrification/, 8.2 hours /nitrification/, sludge organic loading = 0.12 g BOD_5/g·sm·d, total nitrogen concentration in the influent 49 g N/m^3.

Efficiency of nitrogen removal, varied with seasonal changes of temperatures, and was: in summer - above 90%, in winter - 57.0% lowering to 53.8% when chemical precipitation of phosphorus with $FeSO_4$ was practiced.
Nitrogen removal was accompanied by 95% removal of BOD_5 during summer time. [9].

Data gathered during the testing period will be analyzed and veryfied during phase I of the construction to include the scaling-up factor. Adjusted process parameters will be used to operate the ultimate phase of the wastewater treatment plant.

8. Integrated removal of carbon, nitrogen and phosphorus in the activated sludge process

The very first research in Poland on the combined removal of nutrients from the municipal wastewaters was done at the Institute of Environmental Protection in Warsaw by J. Bernacka, in 1980-1982. The project combined denitrification and phosphorus removal in multifunctional reactors with activated sludge and was conducted on a laboratory scale. The model was able to simulate anaerobic/aerobic conditions similar to Bardenpho system in different configurations. The optimum process parameters, at the temperature 10-20°C, were[1] :

- anaerobic tank - hydraulic detention tine - 2 h
  - sludge recirculation rate - 200%
  - dissolved oxygen - 0 g O_2/m^3
- aerobic tank - hydraulic detention time - 6-7 h
  - sludge organic loading - 0.1 - 0.2 g BOD_5/g MLVSS·d
  - dissolved oxygen - 2 g O_2/m^3

During the experiments 90% removal of BOD_5, total nitrogen and ammonia nitrogen was observed; phosphorus removal ranged from 60 to 80%.

Phosphorus removal was limited by BOD_5: P ratio /optimum 100:1.3/ and a low sludge production, caused by low sludge organic loading and high sludge age 16 - 31 daye.

It was confirmed, that lowering the temperature down to 10-12°C resulted in a significant drop in nitrification and denitrification efficiencies; removal rates were reduced by 50% and 80%, respectively.

On the basis of operating experience, a multifunctional package treatment plant PS 150 for extensive integrated biological nutrient removal has been design by POWOGAZ in Poznań. The first installation has been located in Buk near Poznań and is in a start-up phase.

4, Removal of nitrogen and phosphorus in wastewater reclamation system

As it has been stated previously, nutrients removal appears to be the most urgent in the wastewater reclamation systems supplying water to the industrial users. The schematic flow diagrams of water reclamation plants /WRP/ planned to go into construction in the nearest future in South Poland are shown in Figs 1 and 2. They are: WRP in Żory supplying water to the coal mine /Fig. 1/ and WRP in Cracow /Fig. 2/ supplying water to these industries in the Cracow area, which cannot draw any more water from the Vistula River due to high salinity.[3,4].

As shown in the diagrams, the nutrients removal process bears its own individual characteristic, with nitrogen and phosphorus removal achieved in separate processes. Nitrification and denitrification are conducted as a part of biological treatment followed by coagulation and chemical precipitation of phosphorus during tertiary treatment. Coagulation is required in system providing water for cooling cycles, because it removes also other undesired refractory substances like COD, alkalinity, coloids.

The water reclamation system, recommended for Cracow differs from the others, and phosphorus is going to be removed in a multi-functional biological reactor. It will allow for a lower coagulant dosage in the phosphorus precipitation process.
Chemical precipitation of phosphorus will serve as a buffer in case an extensive biological phosphorus removal in the reactor is temporarily inhibited.

At present, the wastewater reclamation process is being tested at Cracow in a small scale model.It comprises all processes of water reclamation together with a multifunctional biological reactor The reactor works in a closed cycle with the preliminary active process which returns 8 supernatant from a preliminary sludge thickener. The experiment has begun in July 1987 and it is still too early to draw any conclusions, though the data are promising.

5. Denitrification of nitrified wastewater using biological fluidized bed reactors

A biological fluidized-bed combines the advantages of both: activated sludge and high-loaded trickling biofilters.

Most significant advantages of the reactor of this' type are:

- very large biomass surface area in a reactor unit volume,about 3300 m^2/m^3 which is approximately 20 times higher than in trickling biofilters,
- high concentration of biomass in fluidized reactors - 8 to 53 kg of dry mass/m^3 - that is in average 10 times higher than in activated sludge process.

The model experimental unit was set up at a trickling filter plant treating municipal sewage from Kraków suburb Kliny. [8].

The pilot scale fluidized-bed unit consisted of two denitrification bed columns, 3.5 m high each. The diameter of the reactor was 70 and 120 mm.

The reactors were filled with sand and having particle size d_{60} equal 1.0 and 0.74 mm and with carbon - particle size d_{60} = 0.5 mm.

The average value of mass indicator of denitrification rate was 0.4, the maximum summer value went up to 0.67 g $N-NO_3$ rem./g V_{ss} · d.

These values are higher than the ones quoted by Jeris who found 0.18 g NNO_3/g VSS·dandmuch higher than the ones found at the activated sludge process /0.06 - 0.12 g NNO_3/g VSS·d/.
The volumetric indicator of denitrification rate, characterising the degree of the reactor utilization ranged from 1.5 /with no methanol added/ to 9.26 g $N\text{-}NO_3$ rem/m^3·d for summer time. The latter value is 10 times higher than the one found in conventional denitrification processes.

The experiments showed that the process proceeded with a high kinetic removal rate - 0.4 g $N\text{-}NO_3$ removal per g VSS·d.

- A high concentration of biomass in a fluidized-bed reatcr /20-30 kg VSS/m^3/, ten times higher than in activated sludge, results in a large value of the volumetric rate of denitrification - up to 9 kg $N\text{-}NO_3$ removal/m^3·d.
- Nitrates removal efficiency was 85-95%, at hydraulic detention time 3-6 min. and temperature 15 - 20°C.

On the basis of experimental data the complex mathematical model of denitrification kinetics was developed and verified.

SUMMARY

1. A great need to implement the optimum technologies of nutrient removal from wastewater in Poland is associated with two factors
   - water quality standards for the Polish rivers, and the Baltic Sea which ultimately receives 95% of the wastewaters generated in our country ,
   - reclaimed wastewater quality standards for the wastewaters that are to be used again in the industry as a source of cooling waters.

2. Phosphorus has been found to be a limiting nutrient, affecting eutrophication in the aquatic system and therefore the technology of its removal is receiving the priority. An extensive pilot-scale study is being conducted that concentrates on the development of the optimum nitrification and denitrification process parameters.

Nitrogen removal is considered to be an important unit process in the water reclamation technology planned for Upper Silesia and Cracow. It is also proposed as a tertiary treatment for the newly designed wastewater treatment plants at the Baltic Coast.

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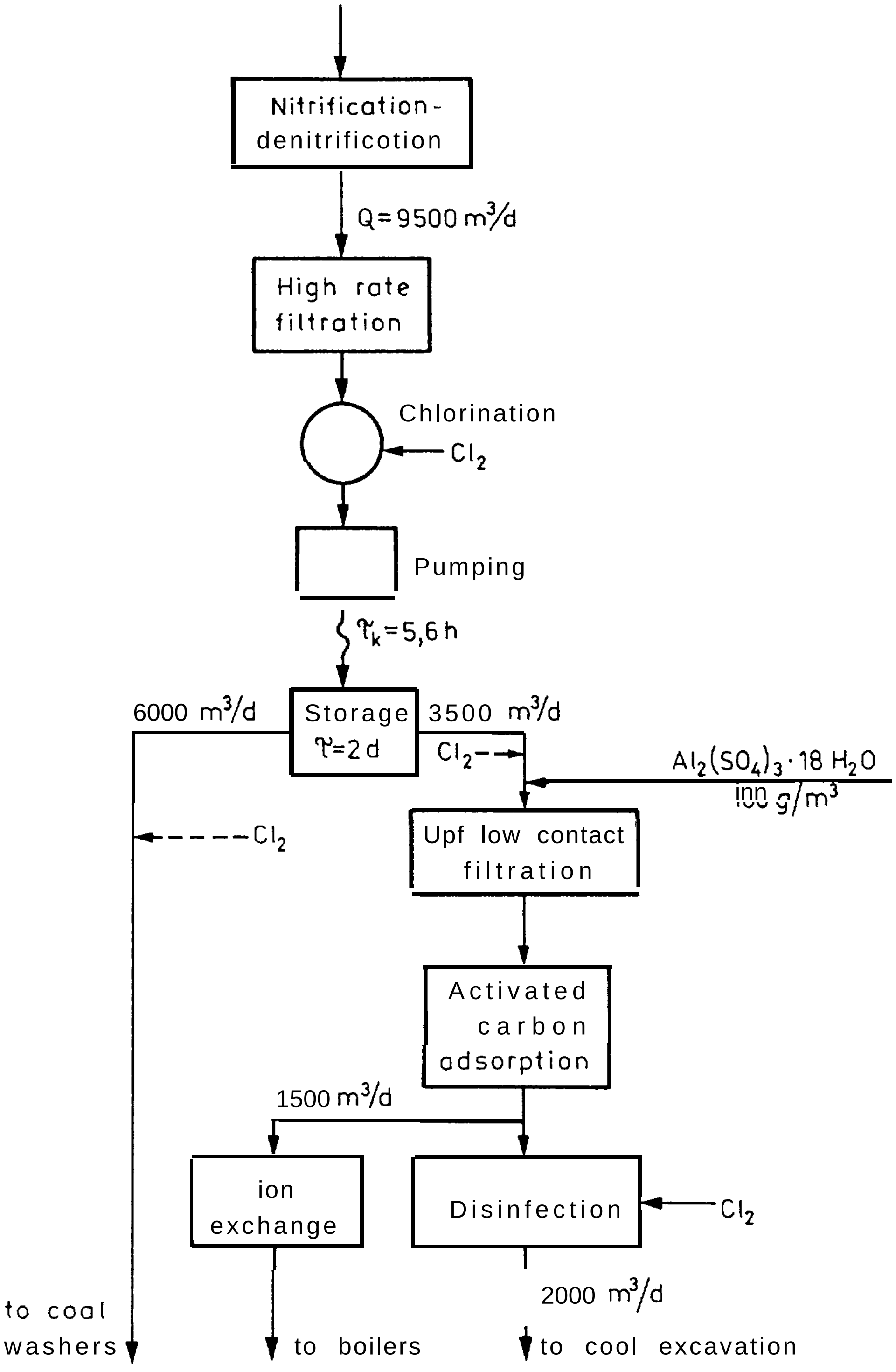
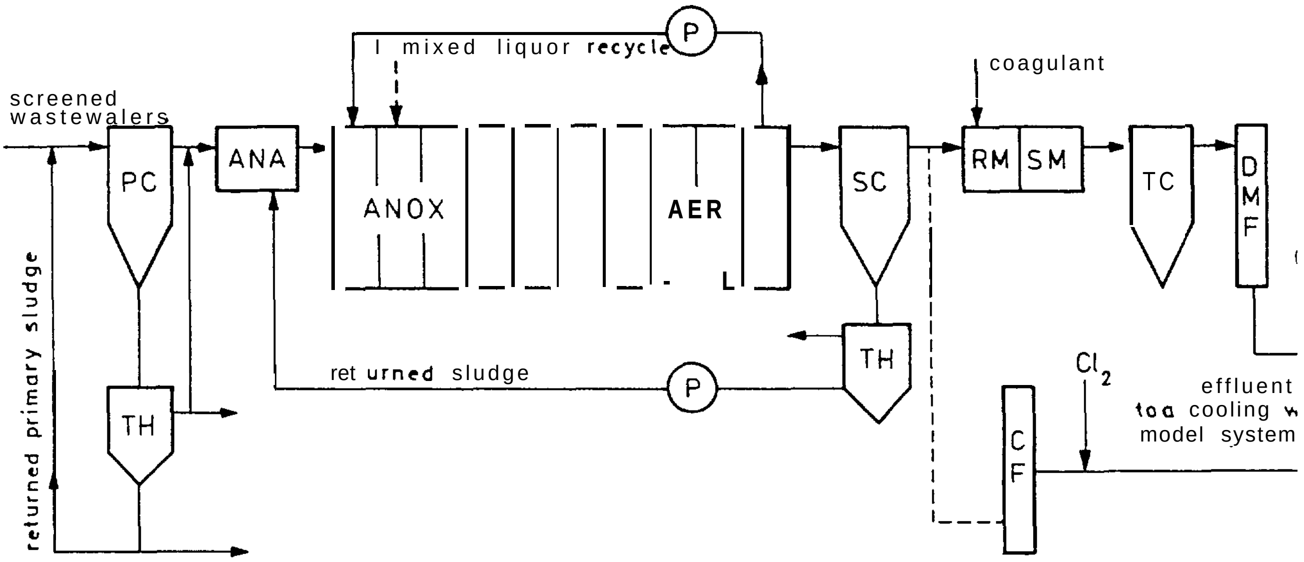


Fig.1. Design conception for wastewater reclamation plant at Żory (9500 m^3/d).



PC	primary clarifier
TH	thickener
ANA	anaerobic zone
ANOX	anoxic zone
AER	aerobic zone
SC	secondary clarifier
SM	slow mixing tank
RM	rapid mixing tank
TC	tertiary clarifier
DMF	dual media filter
CF	contact filtration
P	pump

Fig. 2 . **Scheme of the experimental pilot units for Cracow wostewater reclamation .**

INTRODUCTION INTO SEQUENCING BATCH REACTOR TECHNOLOGY

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DESCRIPTION OF THE PROCESS

The Sequencing Batch Reactor (SBR) process is commonly understood to be a modification of the continuous flow activated sludge process, although the SBR principle can be and has been applied to biofilm systems as well.

In general, the SBR process is distinguished by three major characteristics:

(1) periodic repetition of a sequence of well defined process phases;
(2) planned duration of each process phase in accordance with the treatment result to be met;
(3) progress of the various biological and physical reactions in time rather than in space.

Applied to the activated sludge process, the typical sequence of process phases is sketched in Figure 1 and 2.

At the beginning of each cycle, the SBR contains a certain volume of water, and activated sludge settled at the bottom of the reactor. The cycle starts with a fill phase of distinct duration. The fill phase may be short or long depending on the effects which are desired to be achieved.

With the beginning of the fill phase, or some time later, the aerator is turned on. The aeration phase which now begins may last until the biodegradable portion of the organic wastewater constituents has been degraded, and/or nitrification has been completed.

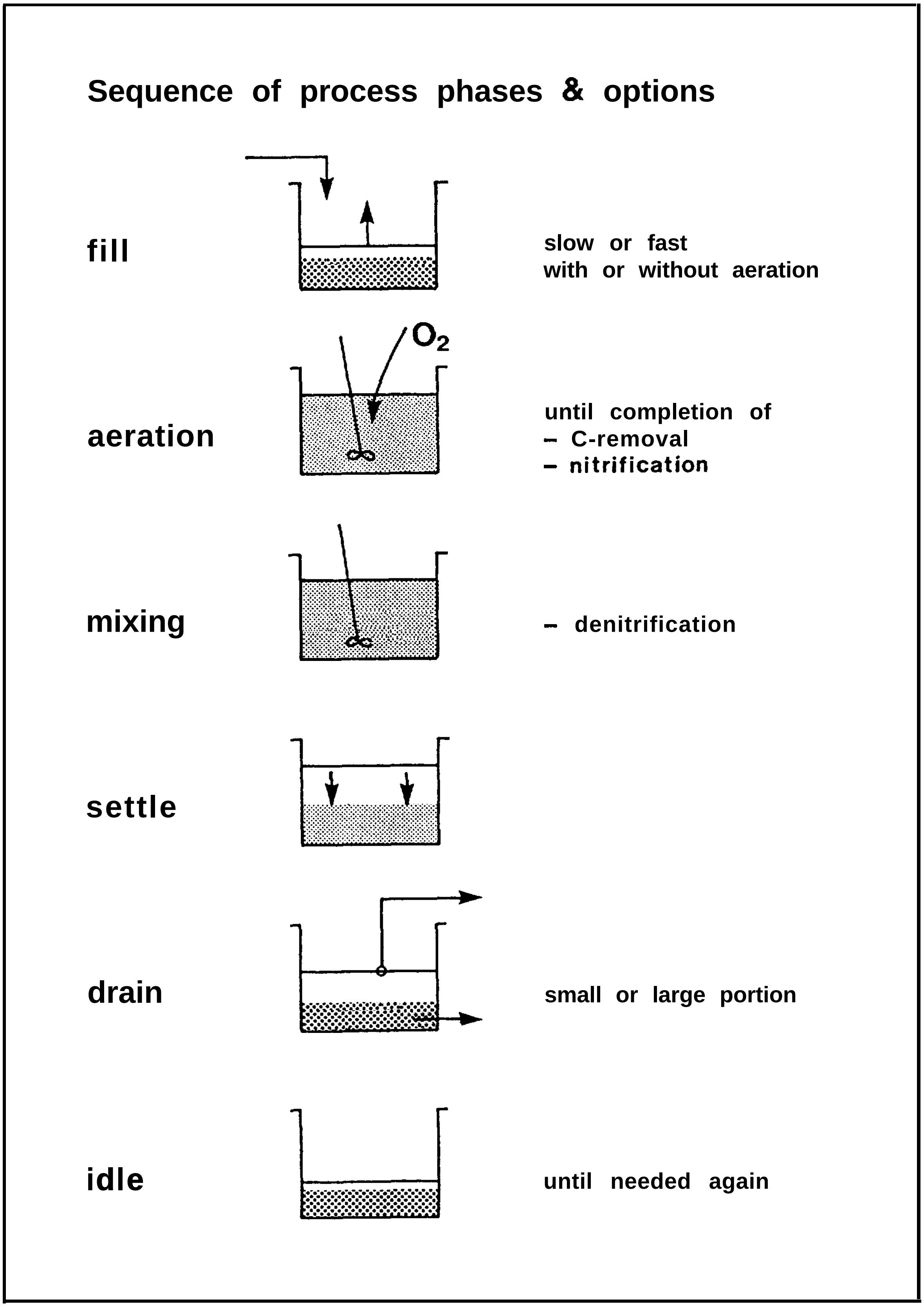

Figure 1 Typical sequence of SBR process phases

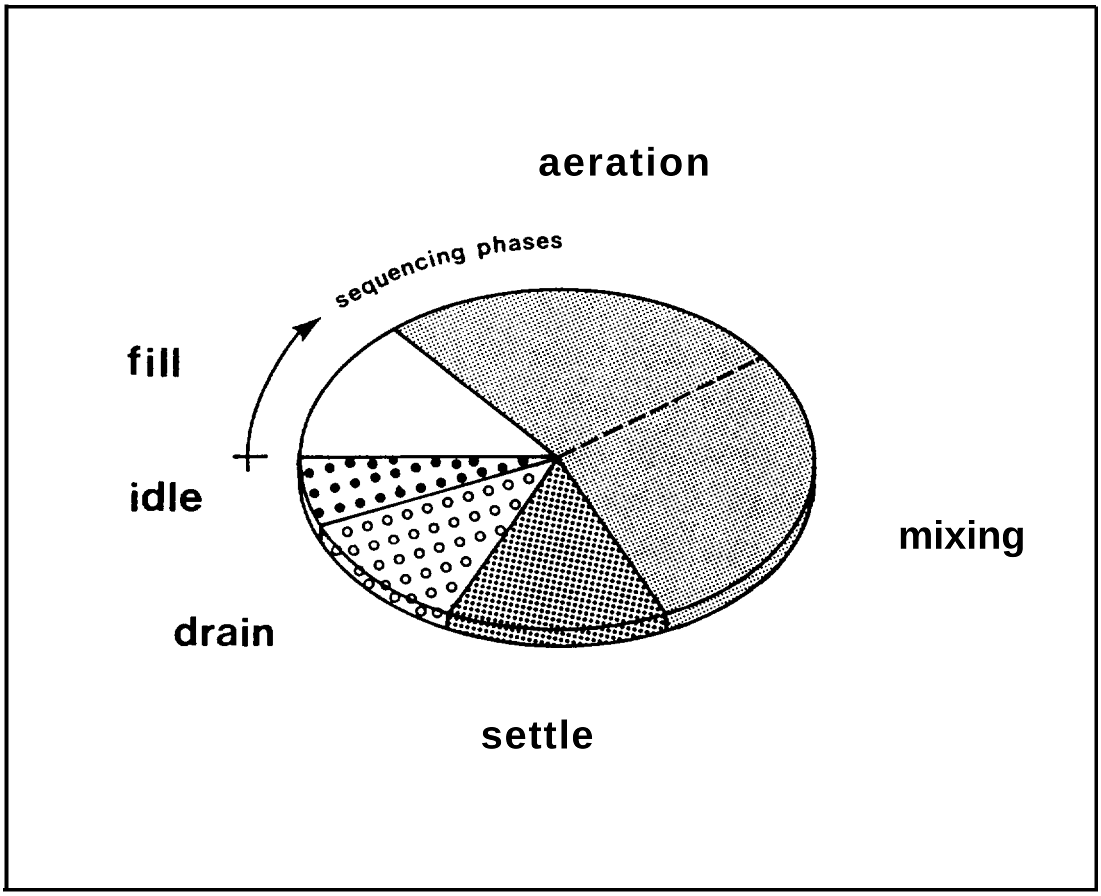

Figure 2 Representation of the SBR process phases as they are periodically repeted.

To achieve denitrification it is necessary to add an anoxic phase during which the reactor content is smoothly mixed.

Eventually, mixer and aerator are turned off. The sludge is allowed to settle under entirely quiescent conditions. A clear water zone (supernatant) appears which can be progressively withdrawn as the sludge blanket moves downwards. Once the water level in the reactor has reached the lower setpoint the mixer is turned on for a short period of time to distribute the sludge evenly, and to allow controlled withdrawal of the surplus sludge. From now on the SBR idles until a new batch of wastewater is available to be filled into the reactor.

ADVANTAGES OF SBR OPERATION

Periodic repetition of process phases is not a unique feature of the SBR process but of the activated sludge process in general. The periodic nature of continuous flow activated sludge systems becomes obvious, when the activated sludge is observed while recirculating through a cascade reactor system (Figure 3).

In such a system the microorganisms may experience very different environmental conditions as they flow from one to the other compartment of the cascade. Some of the environmental conditions can be actively influenced by the engineer. Among these are the mean concentration of the organic wastewater constituents in the influent compartment, the occurance of starvation conditions for the heterotrophic bacteria in the downstream compartments, the oxygen concentration, and the turbulence.

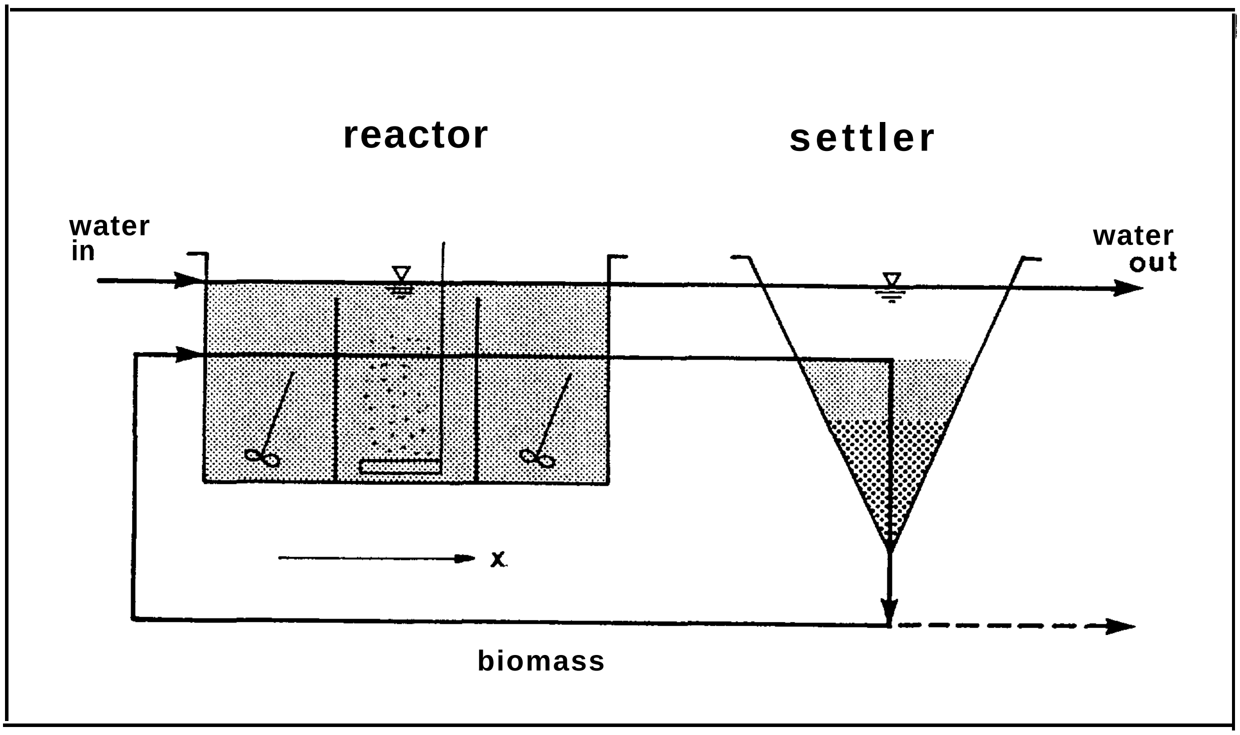

Figure 3 Representation of a continuous flow activated sludge treatment system. The micoorganisms circulating in the system experience periodically changing environmental factors as in the SBR

Periodic change of environmental conditions is a factor which has a significant effect on the composition of the activated sludge and on it's overall metabolic potential. For instance, nitrifiers and denitrifiers can only be forced to live together and form a biocommunity, when the activated sludge is periodically exposed to aerobic and anoxic conditions. Chudoba (1985), van den Eynde (1983), Chiesa (1982) and many others demonstrated experimentally the effects of periodic feast/famine conditions on sludge settleability. It is assumed that as long as the substrate concentration is high, floc forming organisms become capable of outgrowing their filamentous competitors; starvation conditions, on the other hand, favor synthesis of the slime material (extracellular polymers) the flocs are made of.

Of importance, however, is not only the existence of periodic changes of environmental conditions but also the duration during which the microorganisms are exposed to specific situations. The duration of the various process phases controlls enrichment of particular groups of microorganisms. And of course, the duration of the process phases determines which degree of treatment can be obtained.

The great advantage of the SBR process is that the duration of the various process phases can be chosen and maintained independently of any variations of the treatment plant influent. Once the reactor has been filled the process is executed according to it's own rules. The resulting effluent values are low in variability. The process is controlled by timer settings and, if required, by the result of analytical parameters which can easily and continuously measured, as for instance: dissoved oxygen concentration, pH, water level in the reactor and turbidity. The SBR, inherently, is an automatically controlled process (Wilderer et al., 1987), and requires only very limited operator assistance.

DESIGN OF SBR SYSTEMS

Since wastewater is normally delivered to treatment plants on a regular basis but is filled into the SBR only during distinct periods of time an SBR plant must consist of several reactors to be operated in parallel, and must provide holding capacity. Figure 4 contains the typical flow schematic of an SBR treatment plant.

If the actual fill regime is not of concern, and the wastewater is allowed to fill the available SBR by gravity (Irvine et al., 1985), the demand for holding capacity shrinks to a minimum. A by far higher demand of holding tank capacity exists, when the fill regime is strictly regulated (Görg and Wilderer, 1987), and the inflow to the treatment plant is highly variable. A storm water standby tank is necessary, when the wastewater is received from a combined sewer system.

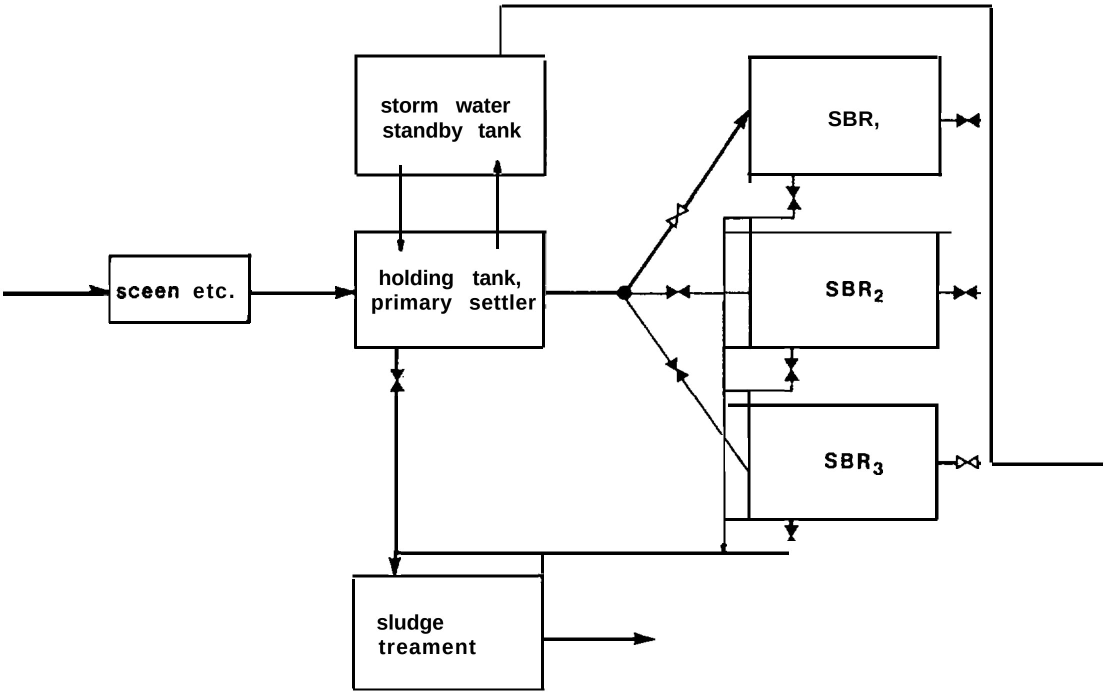

Figure 4 Flow schematic of a Sequencing Batch Reactor wastewater treatment plant

NITRIFICATION/DENITRIFICATION

As in continuous flow activated sludge systems, nitrification can be achieved, when the SBR is operated at a sludge age of above 5 days. The possibilities to enrich for nitrifiers, and to force nitifiers to express their metabolic capabilities is especially favorable in SBRs. That is because organic substrates are converted at the very beginning of the aeration phase so that heterotrophic bacteria switch soon to endogenous respiration, and nitrifiers have free access to oxygen, then.

Termination of the nitrification process is characterized by a significant drop of the overall respiration rate (Figure 5), and can be easily and automatically detected by examination of the profile for the concentration of dissolved oxygen.

Denitrification can theoretically be accomplished during a mixing phase which is added to the aeration phase but the time required would in this case be extremely long. That is because organic substances required as electrone donors were already converted during the intial phase of the cycle, and only stored products are left to drive the denitrification process.

Various methods have been proposed to speed the denitrification process up (Wilderer and Schroeder, 1986). Most favorable results could be achieved by application of quasi-simultaneous operation strategies (Wilderer et al., 1986). The reactor used for the respective pilot studies is sketched in Figure 6. It consists of two interconnected cylindrical compartments, the inner cylinder for denitrification and the outer tubular compartment for nitrification. An adjustable pump allows control of the retention time of the mixed liquor within the two compartment.

The results obtained from a comparison study are summarized in Table 1. To be treated was septic tank supernatant. The efficiency of the SBR process was comparable with the efficiency of a full scale continuous flow activated sludge plant. However, only half of the reactor tank volume was required.

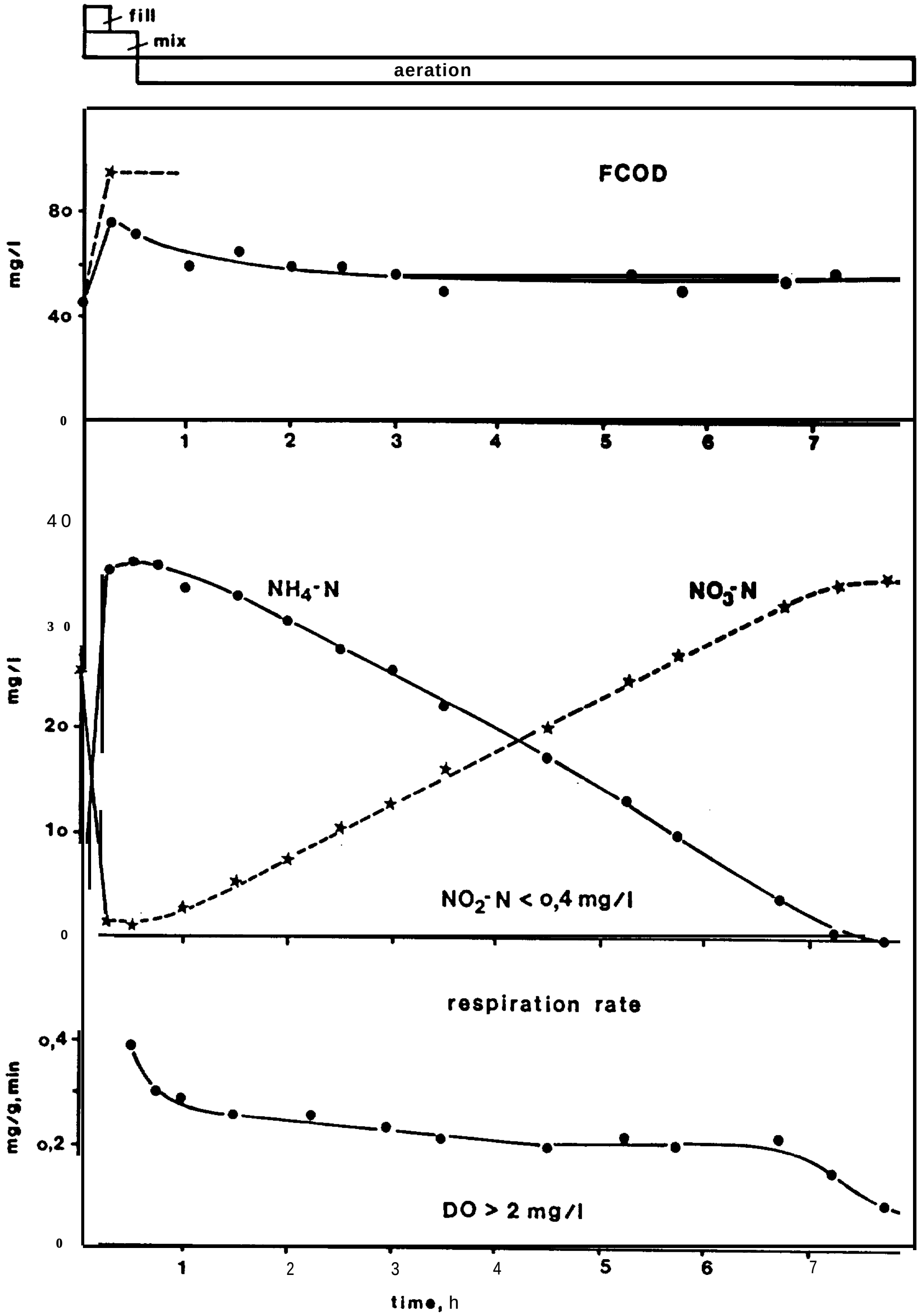

Figure 5 Development of the concentration of filterable organics (FCOD), ammonia and nitrate nitrogen, and the development of the respiration rate during the react phase of an SBR cycle.

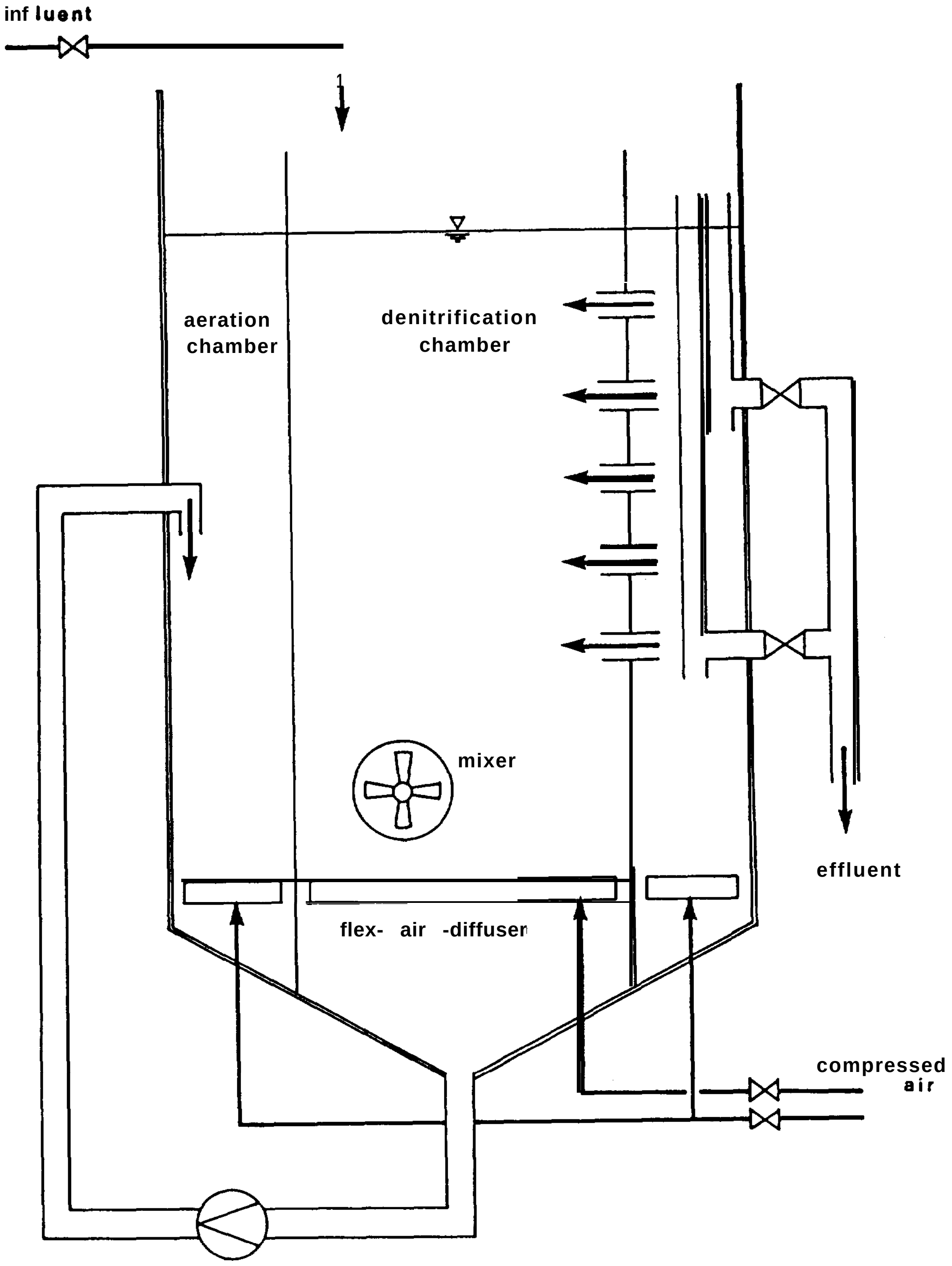

Figure 6 Cross section of an SBR reactor designed to achieve nitrification and denitrification quasi-simultaneously .

Table 1 Results of pilot studies at a treatment plant in Schleswig-Holstein
Comparison between an existing continuous flow activated sludge system and an SBR.

	COD mg/l	TKN mg/l	NO_3-N mg/l
influent i.e. septic tank supernatant	2 040	148	< 1
effluent of the continuous flow system (mean value); hydraulic retentions time: 5 days	58	2.3	2.2
effluent of the SBR (mean value); hydraulic retention time: 2.5 days	68	< 1	1.2

ASSESSMENT

Basically, it can be assumed that SBR systems can be applied to treat the same types of wastewater as continuous flow activated sludge systems. The same treatment results can be expected, provided the SBR system is properly designed and operated.

The SBR provides great advantages over continuous flow activated sludge systems, when the wastewater to be treated varies heavily in flow and concentration. Since the duration of the various process phases can any time be adjusted to the actual needs, consistently high effluent quality is achieved.

Since the duration of the process phases, in particular the duration of the aeration phase is adjustable to the actual needs, SBR systems are impressively energy efficient.

The SBR can and must be operated automatically. Simple timers are the minimum requirement to operate an SBR system. Higher sophistication of process control and management can be achieved without the need for employment of complicated analytical devices. As a result, SBR systems appear to be especially well suitable at locations where only little or no operator skill is available.

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EFFECTS OF LOW TEMPERATURE ON NITROGEN REMOVAL PROCESSES

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ABSTRACT

Temperature must be understood to be an environmental factor which affects not only the reaction rate but also the composition of the multi-species biocommunities employed in biological wastewater treatment plants. Both effects must be taken into account, when impacts of low temperature on the efficiency of biological treatment systems are to be assessed.

Based on the results of pilot scale studies biofilm systems appear to be superior over suspended growth systems. Nitrification could be maintained even at a temperature of only 1 ^{o}C. Trickling filters employed for nitrification proved to be reliable even at low temperatures but, nevertheless, limited in capacity.

BACKGROUND

Temperature affects the microbial systems in wastewater treatment plants in two different but interrelated ways. Short term and long term effects have to be distinguished.

The immediate response of any temperature change is change of the rate of metabolic reactions such as substrate uptake, respiration and growth. The dependency of the reaction rates on the temperature can be modelled according to the Arrhenius law:

$$\ln r_{max} = \ln k_o - Ea\, R^{-1}\, T^{-1}$$

where r_{max} = maximum reaction rate
k_o = frequency factor
Ea = activation energy
R = kinetic gas constant
T = absolute temperature

Different groups of microorganisms may respond to temperature changes with different sensivity, however. The factors k_o and Ea of the Arrhenius equation are system parameters, and are specific for the various microorganisms and substrates of concern.

Long term effects are to be considered, when the temperature after having been shifted to a lower or higher level stays at that level for an extended period of time. On the long run, the organisms may be able to adjust to the new environment. Enzymatic and/or biocoenotic adaption may take place.

Enzymatic adaption is, in this context, the result of shifts of the enzymatic outfit of the microorganisms. When the composition of the microbial community shifts, biocoenotic adaption occurs. Some of the microbial species present in the biocommunity expand their populations size on the expense of others. The dislodged species, however, may find their niche some place else, for instance at a higher sludge age in an activated sludge system or in the deeper regions of a plug flow trickling filter (Figure 1).

During the process of adaption and dislocation the observed reaction rate may change gradually towards a new steady state level. As demonstrated in Figure 2, the steady state reaction rate differs from the rate which is observed right after the temperature change. Again, a linear relationship may be applicable to describe the dependency between the steady state rate ($\ln r_{max}'$) and the water temperatur (T^{-1}), but this model has no direct thermodynamical meaning, of course.

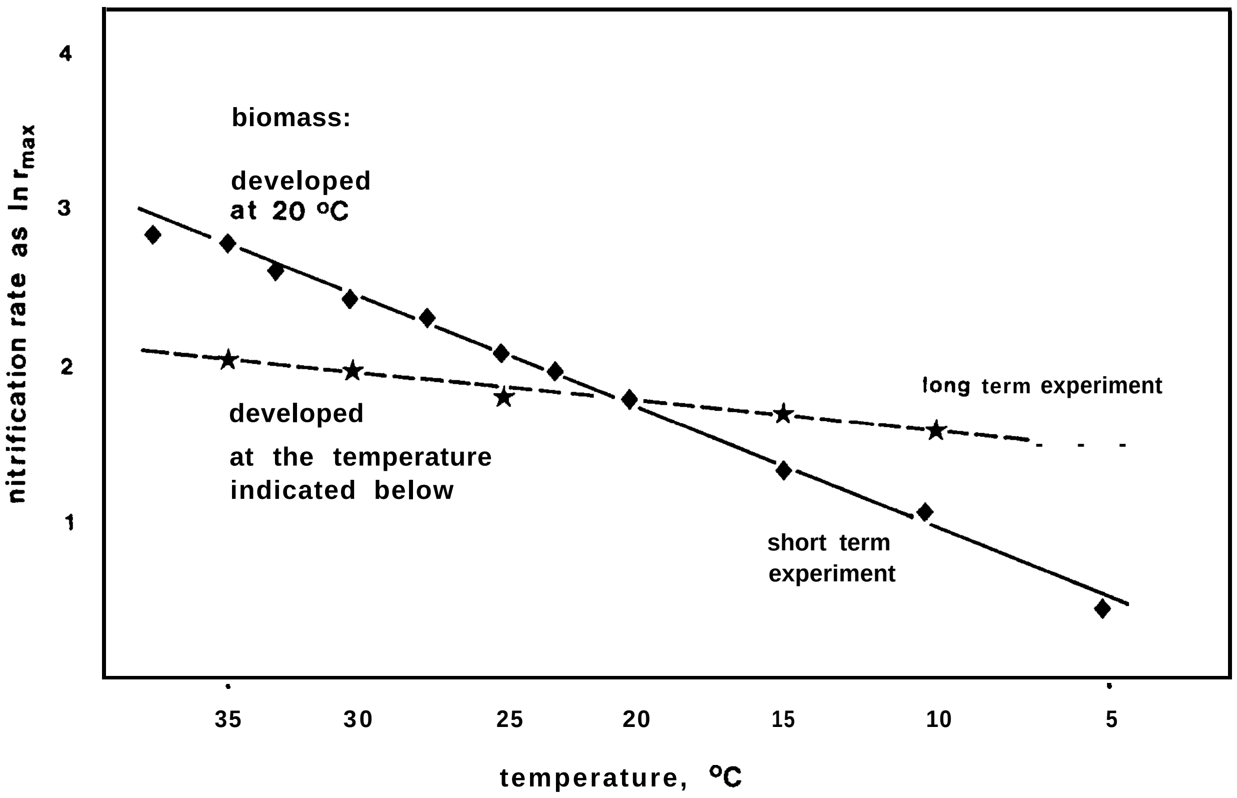

Figure 1 Long term and short term effects of temperature on the nitrification rate expressed by a heterogenuously composed biofilm community

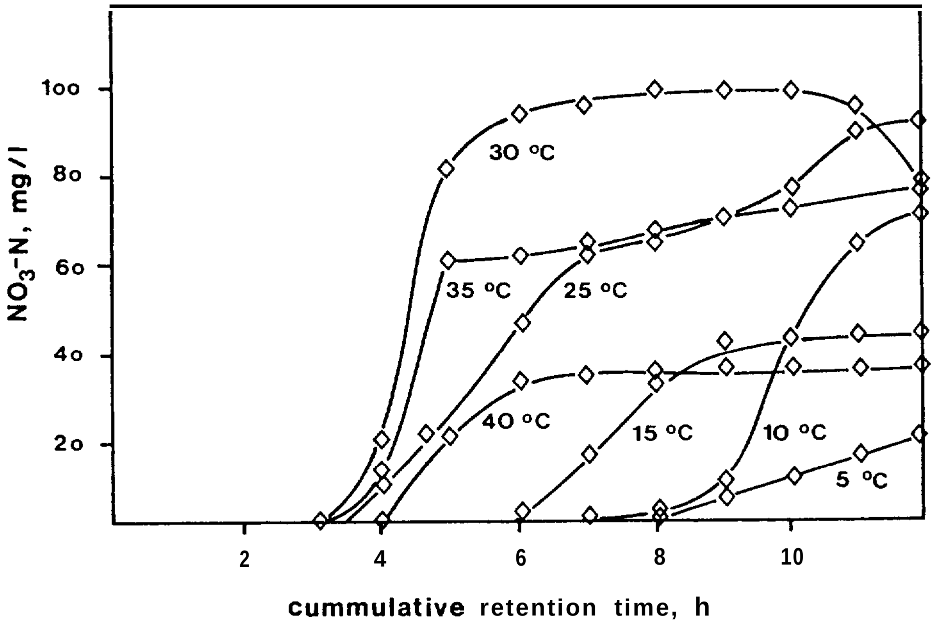

Figure 2 Evolution of nitrate downstream a plug flow fixed bed biofilm system after the process of biocoenotic adaption to the actual water temperatur had been accomplished (exception: the succession at 5 °C was still in progress at the time of observation)

As a conclusion, it appears to be necessary to take the duration of the situations of high and low temperature into account in order to assess the effects of temperature on microbial communities.

With reagard to biological wastewater treatment technologies in general it appears to be necessary, although hardly practicable, to modify sludge age and trickling filter depth respectively as the seasonal mean temperature of the wastewater changes. Reduction of the temperature related fluctuations of the effluent results would be achievable.

With respect to nitrification, biofilm reactor systems appear to be superior over suspended growth systems. The loss of nitrifiers through sludge wasting and washout of suspended cells over the weir of the clarifier is of particular concern, when activated sludge systems are applied. Nitrifiers, on the other hand, are retained in the background of biofilms, even when the organisms are not able to grow for a long period of time.

PRACTICAL EXPERIENCES

Pilot scale experiments have been conducted to develop a method to treat the run-off water from a dredged material landfill site at Hamburg, West Germany (Sekoulov et al., 1985). A fixed bed biofilm reactor was applied to achieve nitrification.

The observed nitrification rates over a one year period are shown in Figure 3, together with the temperature graph. During the summer months, the water temperature varied between 12 and 18 ^{o}C. The observed nitrification rate varied roughly between 14 and 20 mg ammonia nitrogen per square meter of support media surface and per hour. During the winter months the water temperature dropped considerably, for several weeks even below the freezing point. Actually, the entire reactor content froze two times during the observation period. But, once the reactor content had been thawed nitrification continued, although at a relatively low rate of about 5 to 8 mg/m^{2}, h.

The nitrifications rates which were observed during a one year study period are plotted in Figure 4 against the corresponding temperatur of the water. No

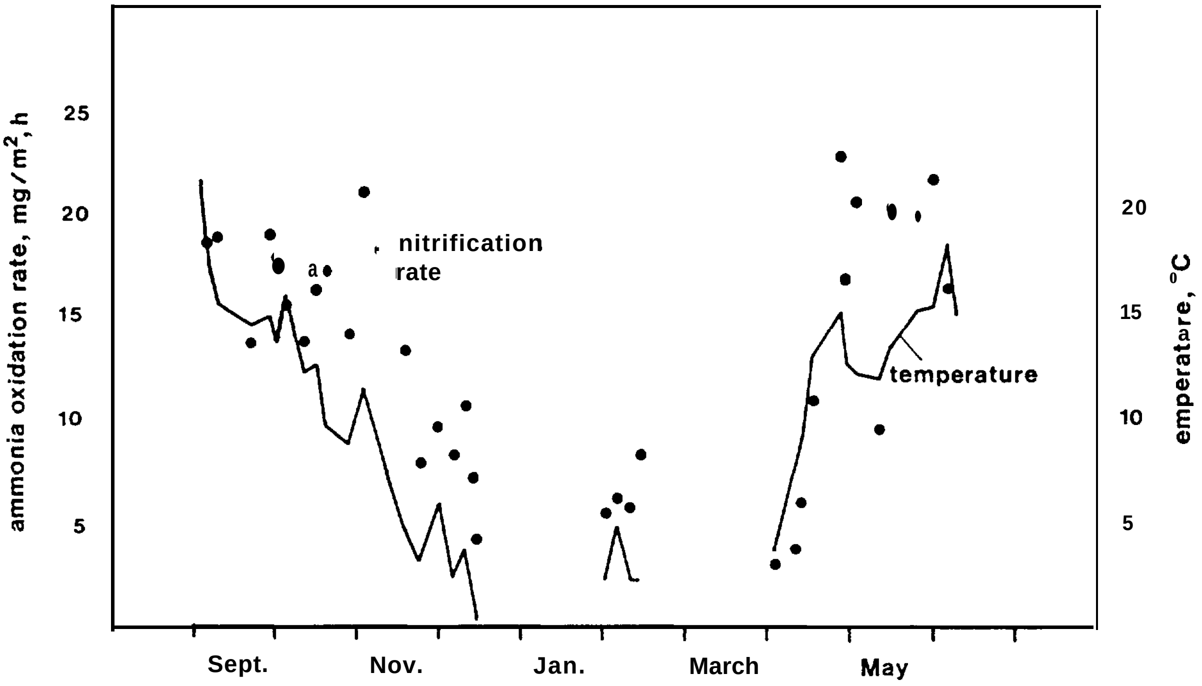

Figure 3 Observed nitrification rate and water temperature. Results from a fixed bed biofilm reactor for nitrification of dredged material runoff

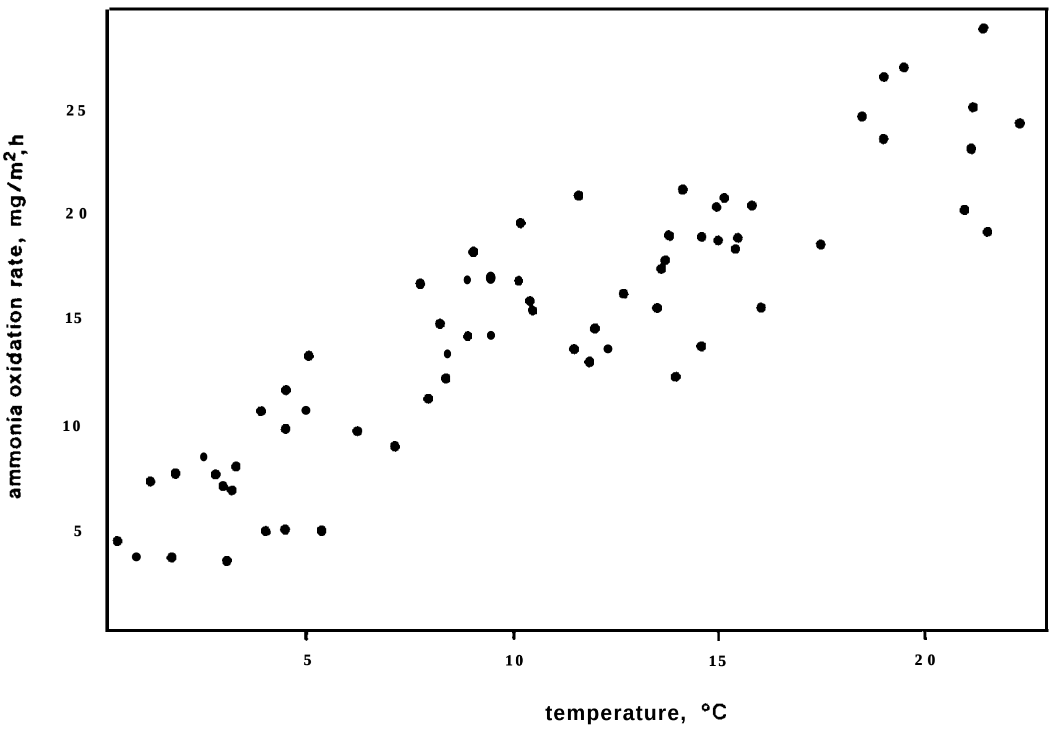

Figure 4 Effect of temperature on the nitrification rate. Results of pilot scale experiments to nitrify dredged material runoff

distinction can be made between data referring to the temperature adapted and non-adapted state of the biofilm community. Anyway, the results support the conclusion that biofilm reactors provide significant advantages under low temperature conditions. The nitrifiers being attached to the suport medium were able to perform nitrification regardless the growth rate and the yield which the nitrifiers could realize.

Severe sludge losses over the weir due to uncontrolled denitrification in the clarifier were experienced at a pilot plant operated in Schleswig, West Germany (Sekoulov and Wilderer, 1985; Wilderer, 1987). The pilot study was conducted to collect information about the capacity of a single sludge activated sludge system in comparison with a two stage activated sludge-trickling filter system. The goal was to achieve advanced carbon removal and nitrification. Denitrification was not directly required but recommended.

The flow schematic of the pilot plant is sketched in Figure 5. Since a full scale plastic media trickling filter was already in operation and could not be easily be bypassed, a plastic media filter had to be incorporated into the flow schematic of the pilot plant as a first biological stage.

The partly unusual results which are summarized in Figure 6 can be explaind by the fact that the first stage trickling filter removed, on the average, about 50 per cent of the soluble organic carbon. As a result, it was not possible to establish controlled denitrification in the activated sludge tratment plant. Denitrification occured, nevertheless, but uncontrollable in the clarifier. Although the surface of the clarifier was regularly cleared from floating sludge, it could not be avoided that large amounts of rising sludge got lost over the weir. Because of this, and because of sludge wasting the nitrifier population decreased steadily as the time went by, and was completely lost during cold weather conditions.

In contrast, nitrification in the rock media trickling filter following the intermediate activated sludge system improved steadily, and was not at all affected by temperature drops down to as low as 6 °C during the winter season. However, the mean concentration of ammonia nitrogen in the effluent of the trickling filter could not be reduced to a level below 5 to 10 mg/l.

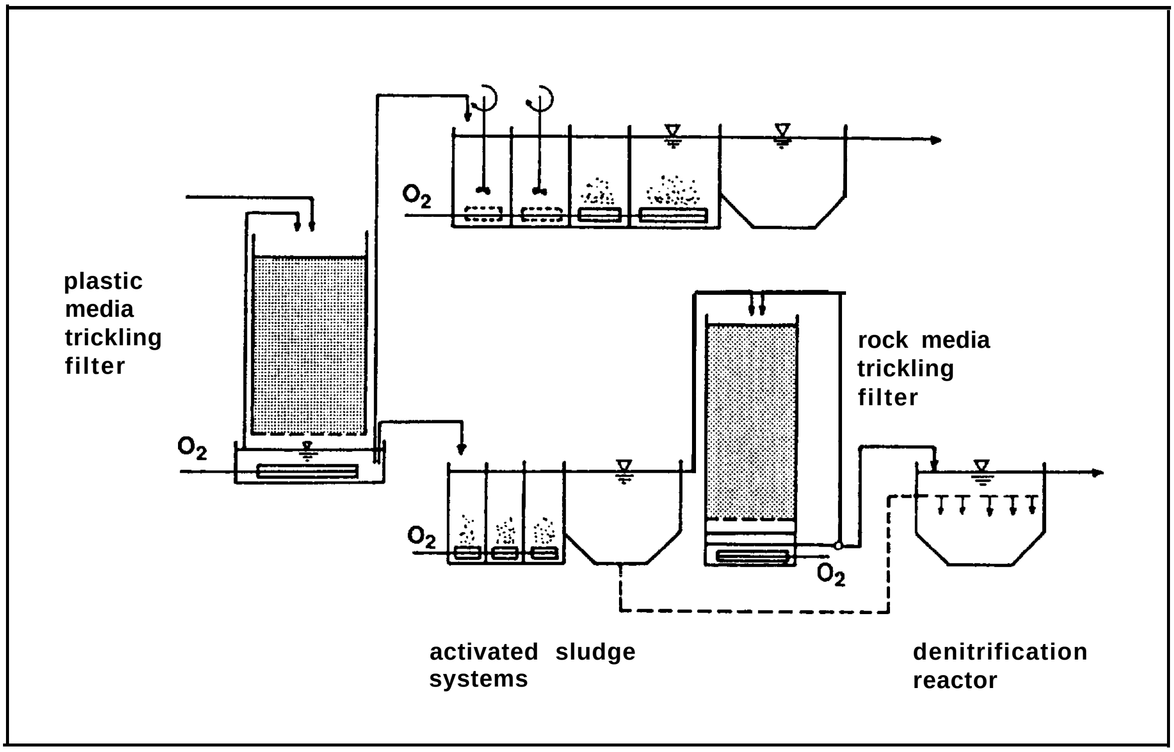

Figure 5 Flow schematic of the pilot plant operated in Schleswig

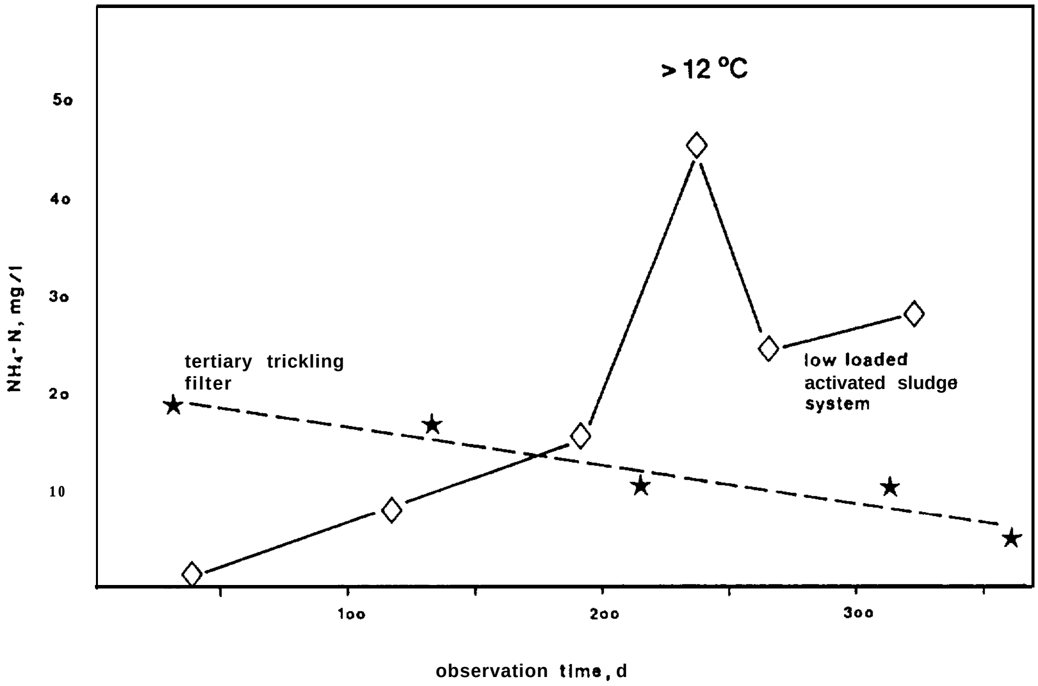

Figure 6 Effluent ammonia concentration of the activated sludge system and of the trickling filter. Mean values for observation periods of several weeks each.

Relatively high effluent concentrations were observed, although the mean removal capacity of the trickling filter was remarkably high.

The quantity of ammonia nitrogen which was converted in the trickling filter is plotted in Figure 7 against the frequency of occurance. Represented are the results of a one year observation period. A linear relationship indicates Gauss normal distribution of the observed removal values.

The slope of the regression line reflecting the variance of the observed data appears to be high. Further investigation of that phenomenon revealed that the variability of the effluent results was mostly due to the variations of the influent ammonia concentration. The influent concentration was occasionally as low as 22 m/l N. The mean influent value, however, was 45 mg/l. The highest values were observed during the cold weather season with concentrations between 60 and 72 mg/l. At the same time, the highest ammonia removal rates were achieved so that the effluent ammonia concentration remained, on the average, close to 7 mg/l even at a water temperatur of 10 ^{O}C and less.

Nevertheless, the capacity of the trickling filter to convert ammonia was limited. On the average, 41 mg/l of ammonia nitrogen could be transformed into nitrate. With reference to the influent concentrations this number is impressive, but not really satisfying.

Advanced denitrification could be achievd by pumping surplus sludge from the activated sludge system to the final clarifier, and distribute the sludge evenly in that tank. The effluent nitrate nitrogen concentration varied between 1 and 7 mg/l (Figure 8), and the concentration of suspended solids between 10 and 20 mg/l. These results are remarkable. Despite the opinion of many experts, denitrification can be accomplished in two stage, two sludge treatment systems without the need to add external carbon sources.

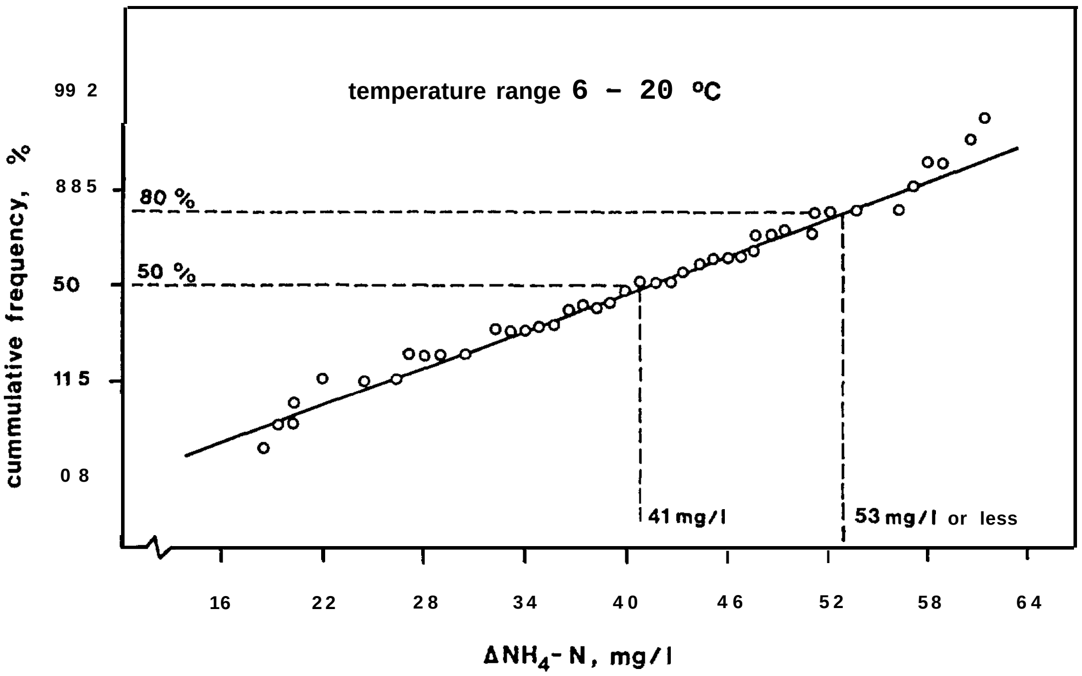

Figure 7 Ammonia nitrogen removed in the trickling filter. Collected data of a one year observation period. Temperature range: 6 to 20 °C.

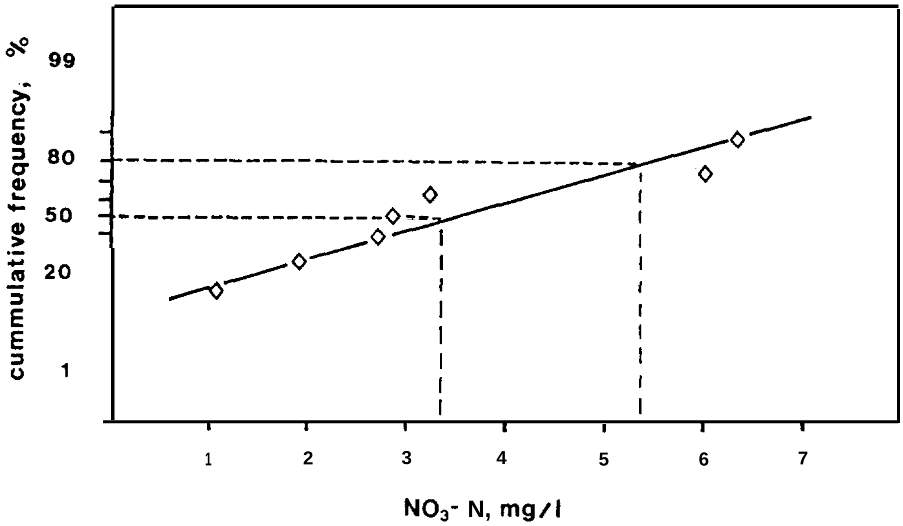

Figure 8 Nitrate concentration in the effluent of the clarifier serving the trickling filter, after conversion to a combined sedimentation-denitrification reactor.

CONCLUSIONS

The effects of low temperature on the nitrification rate should not be overestimated. Biocoenotic adaption must be considered to take place, when the temperature starts to decline, so that the decrease of the reaction rate is eventually leved off to a considerable degree.

In general, biofilm processes provide under low temperature conditions a significantly higher process stability as compared to suspended growth systems. Trickling filters belong to the group of biofilm reactors, but cannot be recommended without reservation. Trickling filters appear to be limited in efficiency for a number of reasons, and they are susceptible to significant variations of the effluent concentration. The latter is not primarily caused by temperature but by the limited hydraulic buffer capacity the trickling filter provides. For that very reason it is recommended to pay more attention to other biofilm reactor systems such as rotating biological contactors and fixed bed biofilm reactors.

Activated sludge treatment systems are susceptible to significant losses of slow growing organisms such as nitrifiers through sludge wasting and hydraulic washout. Special attention must be paid to perturbation of the flow regime, and to biological reactions in the clarifier. Advanced denitrification must be achieved in the biological reactor in order to minimize un-controlled denitrification in the sedimentation tank, and subsequently washout during cold weather conditions.

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MODELLING AND IMPROVEMENTS OF SINGLE SLUDGE PROCESSES FOR BIOLOGICAL NITROGEN REMOVAL

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INTRODUCTION

Hultman and co-workers (1987) have reviewed possibilities to implement nitrogen removal at Swedish municipal wastewater treatment plants. The interest has thereby mainly been focused on single sludge systems for biological nitrogen removal. In this paper will be discussed:

- Modelling of single sludge systems
- Strategies for improving single sludge processes based on the model

MODELLING SINGLE SLUDGE SYSTEMS

Parameter for determining of nitrogen removal

Nitrogen removal may be accomplished by two mechanisms:

- Assimilative nitrogen removal, which depends on withdrawal of sludge containing nitrogen. The removal may be described by the formula:

$$N_a = (s_o - s_e)Y_{obs}b_1 \qquad (1)$$

in which N_a = removed concentration of nitrogen due to assimilation of nitrogen
s_0 = influent concentration of organic material
s_e = effluent concentration of organic material
Y_{obs} = observed yield coefficient
b_1 = nitrogen content in sludge

0 Denitrification, which depends on reduction of nitrate to nitrogen gas according to the chemical formula:

$$5C \text{ (organic carbon) } + 2H_2O \text{ t } 4NO_3^- \longrightarrow 2N_2 \text{ t } 4OH^- \text{ t } 5CO_2$$

The denitrification process may be described by the formula:

$$N_d = (s_o - s_e)a(1 - b_2)b_3/b_4 + b_5 \qquad (2)$$

in which N_d = removed concentration of nitrogen due to denitrification
a = fraction of anoxic zone
b_2 = fraction of organic material that is transferred to sludge, i.e. $(1 - b_2)$ is the oxidised fraction of the organic material
b_3 = oxidation velocity of organic material in the presence of nitrate/oxidation of organic material in the presence of oxygen
b_4 = stoichiometric coefficient between oxidation of organic material and reduction of nitrate
b_5 = coefficient describing initial rapid removal of nitrate

The total concentration of nitrogen that is removed may be calculated as according to formulae 1 and 2:

$$N_r = (s_o - s_e)(ab_3(1 - b_2)/b_4 + Y_{obs}b_1 + b_5) \qquad (3)$$

in which $N_r = N_a \text{ t } N_d$ = total removed concentration of nitrogen

Two parameters, K_1 and K_2, will be defined, that are helpful in evaluation of nitrogen processes. The parameters may be written, respectively:

$$K_1 = (1 - a)\frac{N_r}{s_0 - s_e} \qquad (4)$$

in which $K_1 = (1 - a)(ab_3(1 - b_2)/b_4 + Y_{obs}b_1 + b_5)$

$$K_2 = (1 - a)\frac{N_d}{S_0 - S_e} \qquad (5)$$

in which $K_2 = (1 - a)(ab_3(1 - b_2)/b_4 + b_5)$

Values of parameters K_1 and K_2

Based on an introductory literature study, typical values are of different coefficients describing K_1 and K_2 (see for instance Matsché, 1986, and Hultman and co-workers, 1987):

- $Y_{obs} = 0.3$ g VSS/g COD
- $b_1 = 0.08$ g N/g VSS
- $b_2 = 0.4$ $(1 - b_2 = 0.6)$
- $b_3 = 0.7$ (pre- and simultaneous denitrification)
  $b_3 = 0.4$ (post-denitrification)
- $b_4 = 2.86$ g COD/g N
- $b_5 = 0.03$ g N/g COD (pre-denitrification)
  $b_5 = 0$ (simultaneous and post-denitrification)

For pre-,. simultaneous, and post-denitrification, respectively, K_1 and K_2 may be written:

Pre-denitrification:

$$K_1 = (1 - a)(0.147a + 0{,}054) \qquad (6)$$

$$K_2 = (1 - a)(0.147a \text{ t } 0{,}03) \qquad (7)$$

Simultaneous denitrification:

$$K_1 = (1 - a)(0.147a \text{ t } 0{,}024) \qquad (8)$$

$$K_2 = 0.147a(1 - a) \qquad (9)$$

Post-denitrification:

$$K_1 = (1 - a)(0.084a \text{ t } 0{,}024) \qquad (10)$$

$$K_2 = 0.084a(1 - a) \qquad (11)$$

The different relationships according to formulae 6-11 are shown in Table 1 for K_1 and K_2, respectively.

Table 1. Values for K_1 and K_2, respectively, for different values of the fraction of the anoxic zone (a)

a	(1 - a)	a(1 - a)	Values of K_1 and K_2, g N/g COD					
			Preidenitrification		Simultaneous denitrification		Post-denitrification	
			K_1	K_2	K_1	K_2	K_1	K_2
0.1	0,9	0,09	0.0618	0.0402	0.0348	0.0132	0.0292	0.0076
0.2	0.8	0.16	0.0667	0.0475	0.0427	0.0235	0.0326	0.0134
0.25	0.75	0.1875	0.0681	0.0501	0.0456	0.0276	0.0338	0.0158
0.3	0.7	0.21	0.0687	0.0519	0.0477	0.0309	0.0344	0.0176
0.5	0.5	0.24	0.0677	0.0533	0.0497	0.0353	0.0346	0.0202
0.6	0.4	0.25 0.24	0.0569 0.0638	0.0518	0.0488	0.0368	0.0330	0.0210
[illegible]	[illegible]			0.0473	0.0449	0.0353	0.0298	0.0202
0.7	0.3	0.21	0.0471	0.0399	0.0381	0.0309	0.0248	0.0176
0.8	0.2	0.16	0.0343	0.0295	0.0283	0.0235	0.0182	0.0134
0.9	0.1	0.09	0.0186	0.0162	0.0156	0.0132	0.0108	0.0076
Average value for a between 0.25 and 0.6:		0.23	0.065	0.051	0.047	0.033	0.033	0.019

It may be seen from Table 1 that the values of K_1 and K_2 are approximately constant for values of a between 0.25 and 0.6.

Residence time needed for nitrogen removal

In order to obtain nitrification (and thus also denitrification) the critical sludge age must be exceeded for nitrification bacteria. The critical sludge age may be written:

$$\theta_{d,crit} = k_1 \cdot k_2^{(15 - t)} \qquad (12)$$

in which $\theta_{d,crit}$ = critical sludge age for nitrification bacteria

k_1 = constant

k_2 = temperature coefficient

Typical values of k_1 and k_2 are:

$k_1 = 2.13$ d
$k_2 = 1.103$

In order to obtain a stable nitrification process a safety factor (SF) is needed:

$$SF = \theta_d/\theta_{d,crit} \tag{13}$$

in which θ_d = actual sludge age for nitrification bacteria

Nitrification bacteria only grow under aerobic conditions. Thus, the actual sludge age for nitrification bacteria may be written:

$$\theta_d = \frac{(1 - a)\theta_{tot}VSS}{Y_{obs}(s_o - s_e)} \tag{14}$$

in which θ_{tot} = total hydraulic residence time
VSS = sludge concentration in aeration basin

By substitution of $(1 - a)/(s_o - s_e) = K_1/N_r$ according to formula 4 into formula 14 it is obtained:

$$\theta_{tot} = \frac{N_r Y_{obs}\theta_d}{K_1 VSS} \tag{15}$$

Combination of formulae 13 and 14 gives:

$$\theta_{tot}/N_r = \frac{Y_{obs}SF \cdot \theta_{d,crit}}{K_1 VSS} \tag{16}$$

Suppose that the following coefficient values are valid:

$Y_{obs} = 0.3$ g VSS/g COD
SF = 3
$k_1 = 2.13$ d
$k_2 = 1.103$
$K_1 = 0.065$ g N/g COD (pre-denitrification)
VSS = 2,000 g/m^3

If the temperature is 10 ^{0}C formula 16 gives:

$$\theta_{tot}/N_r = \frac{24\cdot 0.3\cdot 3\cdot 2.13\cdot 1.103^{(15 - t)}}{.065.2000} = 0.58 \text{ h/g N/m}^3$$

According to the calculation example, a residence time of about 0.6 h is needed for every g N/m^3 that is removed. If it is necessary to remove 15 g N/m^3 in the pre-denitrification step at 10 ^{0}C, a residence time of about 9 h is needed at the sludge concentration 2000 g VSS/m^3.

Fraction of anoxic zone

By use of formula 4 the fraction of the anoxic zone may be written:

$$a = 1 - \frac{K_1(s_o - s_e)}{N_r} \tag{17}$$

If, for example, $s_o - s_e$ = 200 g COD/m^3, N_r = 15 g N/m^3, and K_1 = 0.05 g N/g COD the value of a may be calculated to 0.33. For a fixed value of a = 0.25, $s_o - s_e$ = 200 g COD/m^3, and K_1 = 0.05 g N/g COD the value of N_r may be calculated to 13.3 g N/m^3.

Formula 4 may also be written:

$$N_r = \frac{K_1 (s_o - s_e)}{1 - a} \tag{18}$$

It is seen from the formula that N_r is increased by a decreased value of (1 - a) for constant values of K_1 and $s_o - s_e$. However, (1 - a) must not be lower than a certain value for a stable nitrification process. By use of instrumentation (for instance measurement of the effluent ammonium concentration) and possibilities to control the air supply, it may be possible to optimize the operation with respect to the fraction of the anoxic zone (a).

Parameters K_1, and K_2 (calculation example

Hultman and co-workers (1987) reported data for pre-denitrification plants (mainly full-scale plants in the Nordic countries) and for post-denitrification plants (mainly from literature) and found the relationships:

Pre-denitrification:

$$N_r = s_{o,BOD}(a/3.7 + 0.044) \tag{19}$$

Post-denitrification:

$$N_r = s_{o,BOD}(a/5.6 + 0.044) \tag{20}$$

in which N_r = removed concentration of total nitrogen, g N/m^3
$s_{o,BOD}$ = influent total BOD concentration, g/m^3

If $s_o - s_e = 1.85 s_{o,BOD}$ formulae 19 and 20 may be written, respectively:

Pre-denitrification:

$$N_r = (s_o - s_e)(a/6.85 + 0.024) \tag{21}$$

Post-denitrification:

$$N_r = (s_o - s_e)(a/10.4 + 0.024) \tag{22}$$

If formula 4 is compared with formulae 21 and 22, respectively, K_1 may be written:

Pre-denitrification:

$$K_1 = (1 - a)(a/6.85 + 0.024) \tag{23}$$

Post-denitrification:

$$K_1 = (1 - a)(a/10.4 + 0.024) \tag{24}$$

By use of similar calculations as in Table 1, K_1 may be determined to about 0.047 g N/g COD and K_2 to about 0.036 g N/g COD for values of a between 0.25 and 0.6.

If $b_S = 0$, a comparison of formula 5 with formulae 21 and 22, respectively, shows that K_2 may be written:

Pre-denitrification:

$$K_2 = a(1 - a)/6.85 \tag{25}$$

Post-denitrification:

$$K_2 = a(1 - a)/10.4 \tag{26}$$

From calculations similar as in Table 1, K_2 is approximately 0.034 g N/g COD for pre-denitrification and approximately 0.022 for post-denitrification for values of a between 0.25 and 0.6.

The calculated values of K_1 and K_2 for post-denitrification is in agreement with the values of K_1 and K_2 in Table 1. The calculated values for pre-denitrification of K_1 and K_2 are approximately the same as for simultaneous denitrification in Table 1. This may depend on that large recirculation makes the pre-denitrification process similar as the simultaneous denitrification process.

Design procedure

Based on the presented model, a design procedure may be for the removal of nitrogen in single sludge systems:

- Calculation of the necessary total residence. time, θ_{tot}, by use of formula 16 and with values on:

  the concentration of total nitrogen, N_r, that is needed to remove in the activated sludge process

  the critical sludge age, $\theta_{sl\ crit}$, for nitrification (dependent on temperature) the safety factor (SF)

the observed yield coefficient (Y_{obs})

the parameter for nitrogen removal (K_1)

the sludge concentration in the aerated/anoxic zone (VSS)

o Calculation of the fraction of the anoxic zone by use of formula 17 and with values on:

the concentration of total nitrogen, N_r, that is needed to remove in the activated sludge process

the parameter for nitrogen removal (K_1)

the removed concentration of organic material in the activated sludge process ($s_0 - s_e$)

(the model is valid for a approximately between 0.25 and 0.6)

Possibilities for improved nitrogen removal in an existing plant

In order to estimate the possible nitrogen removal in an existing plant, the following calculations may be used:

o Calculation of the necessary residence time, θ_n, needed for nitrification:

$$\theta_n = \frac{SF \cdot \theta_{d,crit} \cdot Y_{obs}(s_0 - s_e)}{VSS} \qquad (27)$$

0 Calculation of the fraction of the anoxic zone:

$$a = \frac{\theta_e - \theta_n}{\theta_e} \quad (\text{for } \theta_e > \theta_n) \qquad (28)$$

in which θ_e = existing total residence time

o Calculation of the removed total concentration of nitrogen, N_r, in the activated sludge process by use of formula 18 (for a between about 0.25 and 0.6).

Experimental determination of parameters

Basic parameters

Formula 16 may be considered as the basic formula describing the necessary residence time for the removal of a certain concentration of nitrogen. The formula contains the following parameters:

- o Observed yield coefficient, Y_{obs}
- o Safety factor, SF
- o Critical sludge age for nitrification, $\theta_{d,crit}$
- o Parameter for nitrogen removal, K_1
- o Sludge concentration, VSS

The observed yield coefficient may be determined as:

$$Y_{obs} = \frac{\text{produced volatile suspended solids}}{\text{removed organic material } (s_o - s_e)} \qquad (29)$$

Formulae 12 and 13 may be combined to:

$$\frac{\theta_d}{k_2^{(15 - t)}} = SF \cdot k_1 \qquad (30)$$

The constant k_2 describes the temperature dependance for the growth of nitrification bacteria and may be determined by use of:

- o Literature data
- o Batch experiments in which the ammonium oxidation rate is determined as a function of the temperature for a sludge sample

For a known value of k_2 it is possible to plot $\theta_d/k_2^{(15 - t)}$ as a function of the effluent ammonium concentration. From this plot a value of $\theta_d/k_2^{(15 - t)} = SF \cdot k_1$ may be chosen which gives a stable nitrification process. The sludge age is described by formula 14. With a chosen value of $SF \cdot k_1$ and a known value of k_2 the product of $\theta_{d,crit} SF$ may be written according to formula 12:

$$\theta_{d,crit} SF = SF \cdot k_1 k_2^{(15 - t)} \qquad (31)$$

The parameter K_1 is determined by use of formula 4. From formula 16 it is seen that the basic processes influencing nitrogen removal are:

- Factors influencing sludge production and sludge properties described by Y_{obs}/VSS.
- Factors influencing the nitrification process described by $\theta_{d,crit}^{SF}$ (Cf formula 31).
- Factors influencing the nitrogen removal process described by K, (position and size of anoxic zone, type of organic material etc).

In order to determine basic parameters influencing the nitrogen removal processes measurements should be made for the biological step:

- Influent flow
- Excess sludge flow
- Sludge concentrations in aerobic and anoxic zones
- Sludge concentration in excess sludge
- Sludge concentration in the effluent
- Volatile fraction of sludge
- Influent organic concentration
- Effluent organic concentration
- Influent total nitrogen concentration
- Effluent total organic concentration
- Effluent ammonium concentration
- Temperature
- Temperature dependance for nitrification bacteria
- Fraction of anoxic zone
- Sludge volume after 30 min sedimentation

Assimilative nitrogen removal

The assimilative nitrogen removal is described by formula 1. In addition to the measurements of the influent and effluent organic concentration and the observed sludge yield, it is necessary to measure the nitrogen content in the sludge (b_1) in order to determine N_a.

Denitrification

By use of formulae 4 and 5, the parameter K_2 may be written:

$$K_2 = K_1 - (1 - a)Y_{obs}b_1 \tag{32}$$

With a known value of K_2 it is possible to calculate the removed nitrogen concentration by denitrification, N_d, by use of formula 5. The denitrification rate, r_{N_d}, in the anoxic zone may be calculated as:

$$r_{N_d} = \frac{N_d}{\theta_{tot} a\,VSS} \tag{33}$$

In the parameter K_2, important coefficients are b_2, b_3, b_4 and b_5. b_2 may be determined as $b_6 Y_{obs}$ where b_6 is g organic material per g vss. The value of b_4 is 2.86 g COD/g N. Possible methods to determine b_3 ans b_5 are:

0 According to the IAWPRC model of the activated sludge process (Henze and co-workers, 1987) b_3 is influenced by two factors. The first is a correction factor which adjusts for either the the change in maximum growth rate associated with anoxic conditions, or for the fact that only a portion of the biomass can denitrify. The second is a correction factor is a correction factor which adjusts for the observation that hydrolysis of slowly biodegradable organic matter occurs more slowly under anoxic conditions than under aerobic conditions.

In the IAWPRC model is shown how these two factors may be determined. The tests imply the measurements at the same time of the oxygen and nitrate consumption rates in two batch reactors which are equivalent in every respect except for the terminal electron acceptor oxygen in one (aerobic) and nitrate in the other (anoxic).

- b_5 may be determined by use of formula 2, which may be written:

$$\frac{Nd}{s_0 - s_e} = a(1 - b_2)b_3/b_4 + b_5 \tag{34}$$

If $N_d/(s_0 - s_e)$ is plotted against a the slope is $(1 - b_2)b_3/b_4$ and the intercept is b_5. In simultaneous or post-denitrification the value of b_5 seem to be approximately zero.

Nitrification

The oxidised nitrogen, N_{ox}, may be calculated as:

$$N_{ox} = N_{Kj,o} - Y_{obs}b_1 - N_{Kj,e} \tag{35}$$

in which $N_{Kj,o}$ = influent Kjeldahl nitrogen concentration
$N_{Kj,e}$ = effluent Kjeldahl nitrogen concentration

With a known value of N_{ox} it is possible to calculate the nitrification rate, $r_{N_{ox}}$, as:

$$r_{N_{ox}} = \frac{N_{ox}}{\theta_{tot}(1 - a)VSS} \tag{36}$$

Additional measurements

In addition to measurements necessary for determination of basic parameters the following parameters are of interest according to the presented model:

- Nitrogen content of VSS
- Organic content (for instance as COD) of VSS
- Oxygen and nitrate consumption rate by use of batch reactors
- $N_d/(s_0 - s_e) = f(a)$
- Influent Kjeldahl nitrogen concentration
- Effluent Kjeldahl nitrogen concentration

STRATEGIES FOR IMPROVING SINGLE SLUDGE PROCESSES FOR BIOLOGICAL NITROGEN REMOVAL

Possibilities to improve the single sludge biological nitrogen process will be discussed from formula 16, where important parameters are:

- Observed yield coefficient (Y_{obs})
- Safety factor for nitrification (SF)
- Critical sludge age for nitrification ($\theta_{d,crit}$)
- Parameter for nitrogen removal (K_1)
- Sludge concentration (VSS)

Suppose that Y_{obs} and SF could be diminished by 10 % and 20%, respectively, and that K_1 and VSS could be increased by 20% and 25%, respectively. In order to remove a certain concentration of nitrogen the volume need would then according to formula 16 be diminished by a factor of $1.2 \cdot 1.25 / 0.9 \cdot 0.8 = 2.1$. This example shows that the imorovement of different process factors may give a combined effect to significantly diminish the volume need for biological nitrogen removal.

Observed yield coefficient

A possible way to diminish the observed yield coefficient, except for increasing the sludge age, is to substitute part of the organic material in the influent by a soluble organic carbon source. In that case the inert VSS fraction will be diminished.

Safety factor for nitrification

In order to reach below a certain value of the ammonium concentration in the effluent a certain value is needed for the safety factor. By use of instrumentation and possibilities to control the oxygen supply, it is possible to use a lower value of the necessary safety factor.

Critical sludge age for nitrification

Two ways may be helpful in decreasing the critical sludge age:

- The critical sludge age for nitrification is highly dependent on temperature. Thus, partial nitrification of side-streams from dewatering of digested sludge may be a suitable technique.
- The critical sludge age may be diminished if external nitrification bacteria are seeded into the activated sludge process

Parameter for nitrogen removal

The parameter, K_1, for nitrogen removal seems to be positively influenced by:

- Use of pre-denitrification
- Use of a soluble carbon source
- Use of plug flow conditions in the anoxic zone in pre-denitrification

Sludge concentration

Several methods may be used to increase the sludge concentration:

- Improvement of sludge settleability by addition of weighting agents
- Improvement of sludge settleability by use of concentration gradients (plug flow etc)
- Use of fixed bed material in the aeration basin
- Nitrification in the return sludge (use of contact stabilisation)
- Denitrification in the return sludge (addition of a carbon source to the return sludge and use of a separate denitrification step for the return sludge)

ACKNOWLEDGEMENT

This study is part of a research project on nitrogen removal with support from the Swedish Board for Technical Development (STU), the Swedish Water and Waste Water Works Association (VAV) and the National Environmental Protection Board (SNV).

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The State of Advanced Biological Wastewater Treatment in the Federal Republic of Germanv

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1. Introduction

Late in the 60s the need of new wastewater purification technology for the protection of the environment became evident.

The start of a special research program SFB-82* (in 1972) supported by the DFG** can be considered as the beginning of an intensive research in the field of advanced wastewater treatment in the FRG.

The main research projects of this program had the object to enlarge the basic knowledge on:

- Nitrification (activated sludge systems and fixed bed reactors)
- Denitrification (activated sludge systems without extended H-donator)
- Suspended solids removal (microstraining, filtration, particle size distribution in biological treatment plants effluents)
- Phosphorus removal (chemical precipitation)
- Refractory CSB (identification of the single substances, activated carbon, the use of ozone)
- Trace elements in the effluents
- Treatment methods with algae

SFB-82* - Sonderforschungsbereich-82, at the University of Stuttgart
DFG** - Deutsche Forschungsgemeinschaft, Bonn

The importance of the advanced uastewater treatment was rapidly recognized. The received results were influencing the requirements on the effluent quality and the planning of treatment plants.

Parallel to the achievement of better results by the advanced wastewater treatment and the rising consciousness of the people concerning the environmental protection, new more stringent laws and regulations were issued. This process is still going on and cannot be assumed as finished.

This has to be considered by the planning of new treatment plants. That means that we have to design treatment plants today, which must fulfill the expected effluent requirements of tomorrow.

That is why, the engineers have to develop first of all a concept for a full possible wastewater treatment, independent of lower current requirements. This will allow a stepuise building of the treatment plants according to the coming requirements in the future. Thus, unexpected difficulties by expanding of an existing purification system can be easily avoided.

2. The State of the Biological Treatment

Because of the lacking of practical experience with advanced wastewater treatment plants, their application was at the beginning very careful.

The first requirement to stop or prevent eutrophication was regulating only the phosphorus concentration in the effluent of the treated wastewaters. This was followed later by limiting also the ammonium-nitrogen ($N-NH_4^+$) concentration. After introducing the nitrification, the operation of these plants was modified for carrying out a denitrification. Additional to some benefits by protecting the recipient of eutrophication, the use of NO_3^- as homolog to O_2 helps to save some energy costs. The denitrification is more or less optional, but it finds a good acceptance and is very often planned together with the nitrification.

The elimination of the eutrophication factors (P, NH_4^+, NO_3^-) were obligatory only for the protection of special surface waters. This did not belong yet to the general requirements, but it is expected that this can occur in the next few years.

The 5th novel of the law regulating the water resources in the FRG (WHG) has its validity from January 1987. The new bylaws regulating the effluent quality (AbwAG and VwV) are in discussion and will be introduced in 1989. The application of advanced wastewater treatment to be required generally. That means that besides the full BOO-removal, the treatment plants have to be planned today for full nitrification, denitrification, phosphorus- and suspended solids removal.

2.1 Preferred Treatment Systems

The investment- and operational costs for the upgrading of a conventional treatment plant with stages of advanced wasteuater treatment are usually high. To avoid erratic planning and disappointments the ATV*-organization tried to elaborate recommendations for the planning of treatment plants with advanced wastewater stages.

The design methods are based on the existing experience with technical plants. Basic research, theoretical models, results of Laboratory or pilot scale only are not usually considered. The worked out recommendation from the specialized groups (ATV-FA** 2.6 and 2.8) are published in the "Korrespondenz Abwasser" 1987 (1).

In the FRG good practical experience is available for the following advanced wastewater treatment processes:

- chemical removal of phosphorus
- nitrification
- denitrification and
- suspended solids removal

ATV* - Abwassertechnischer Verband

FA** - Fachausschuß

The proposed and accepted technical solutions for the above mentioned processes depend to some extent on subjective decisions. It is not seldom that the personal opinion of a responsible government officer, the special experience of the engineers and the research interests of the professors involved in projects influence the decision of a treatment plant system.

That means that the wanted effluent quality can be achieved with different techniques and systems.

A reliable comparison of costs, energy, efficiency, process stability etc. of different systems is not available. The existing results (data) are not sufficient for a critical review of the known plants.

2.1.1 Activated Sludge Plants

The best results for nitrification and denitrification up till now are achieved with activated sludge plants.

In Fig. 1 prospective systems for full nitrification and denitrification are summarized.

The interest of combining these systems (Fig. 1) with a biological phosphorus removal is very great. Research work is carried out now on such plants. To reach high stability of a phosphorus removal is still considered as a problem. The results are promising and in spite of not sufficient safety such treatment plants are now in planning.

The efficiency of these treatment systems is also depending on the sewer system. Combined sewerage causes difficulties to the denitrification and biological phosphorus removal. The role of equaliting basins and primary sedimentation tanks on the wanted biological purification processes has to be considered in each case very carefully.

There have been published several methods to design plants for nitrification and denitrification. They will not be discussed in this paper. However,

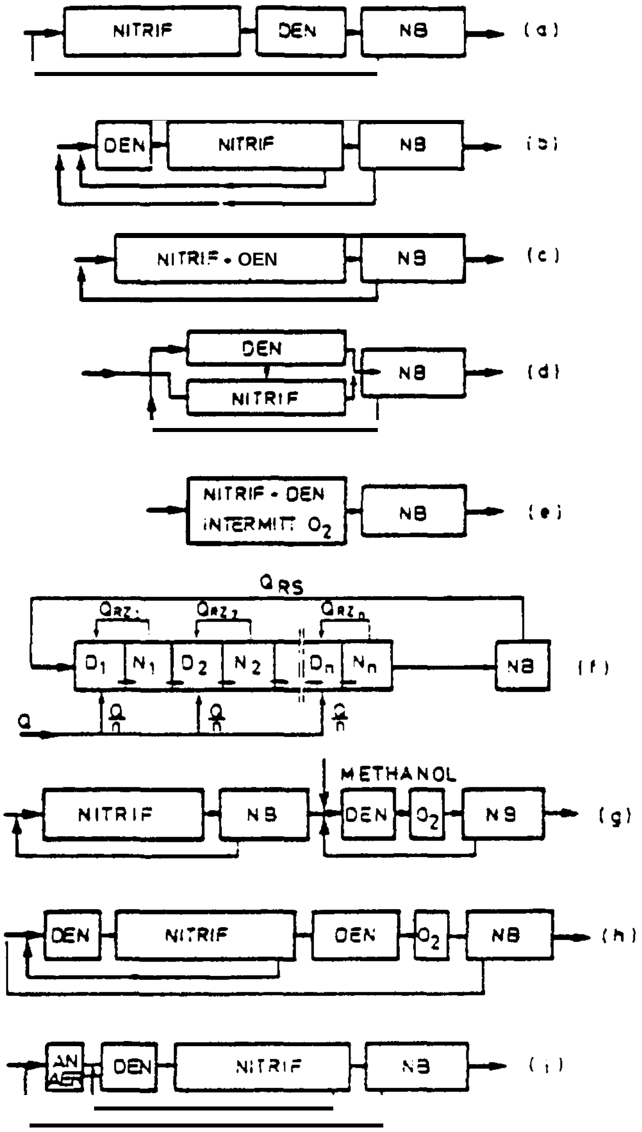

Fig. 1: Prospective treatment plants for nitrogen removal (nitrification and denitrification) (1).

it must be mentioned that in most of the cases the design parameters and the systems are proven with pilot plants studies carried out in situ.

For receiving of full nitrification generally a daily average organic load of

$$B_{TS} = 0.1 \ ./. \ 0.2 \ (\text{kg BOD/kg SS .d})$$

is recommended.

2.1.2 Biofilm Reactors

Biofilm reactors have some advantage for full nitrification. Once the biofilm has grown on the support material it can reach a very high sludge age. This is of great benefit for growing of the nitrifying bacteria.

Good results in the practice are achieved with two-stage treatment plants in which the first stage is an activated sludge system and the second stage a biofilm reactor e.g. a trickling filter unit.

On Fig. 2 can be seen the dependance between the organic load (BOD) and the reached nitrification efficiency of a trickling filter (2).

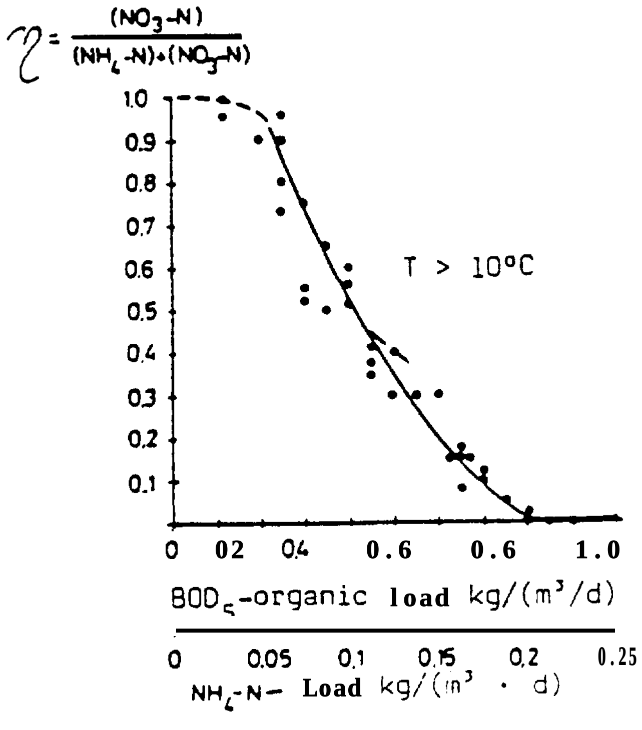

Fig. 2: Dependance of the nitrification efficiency from the organic load by trickling filters (2).

The profiles. of COD, NH_4^+, and NO_3^- measured in trickling filters by different organic loads are given in Fig. 3 (3).

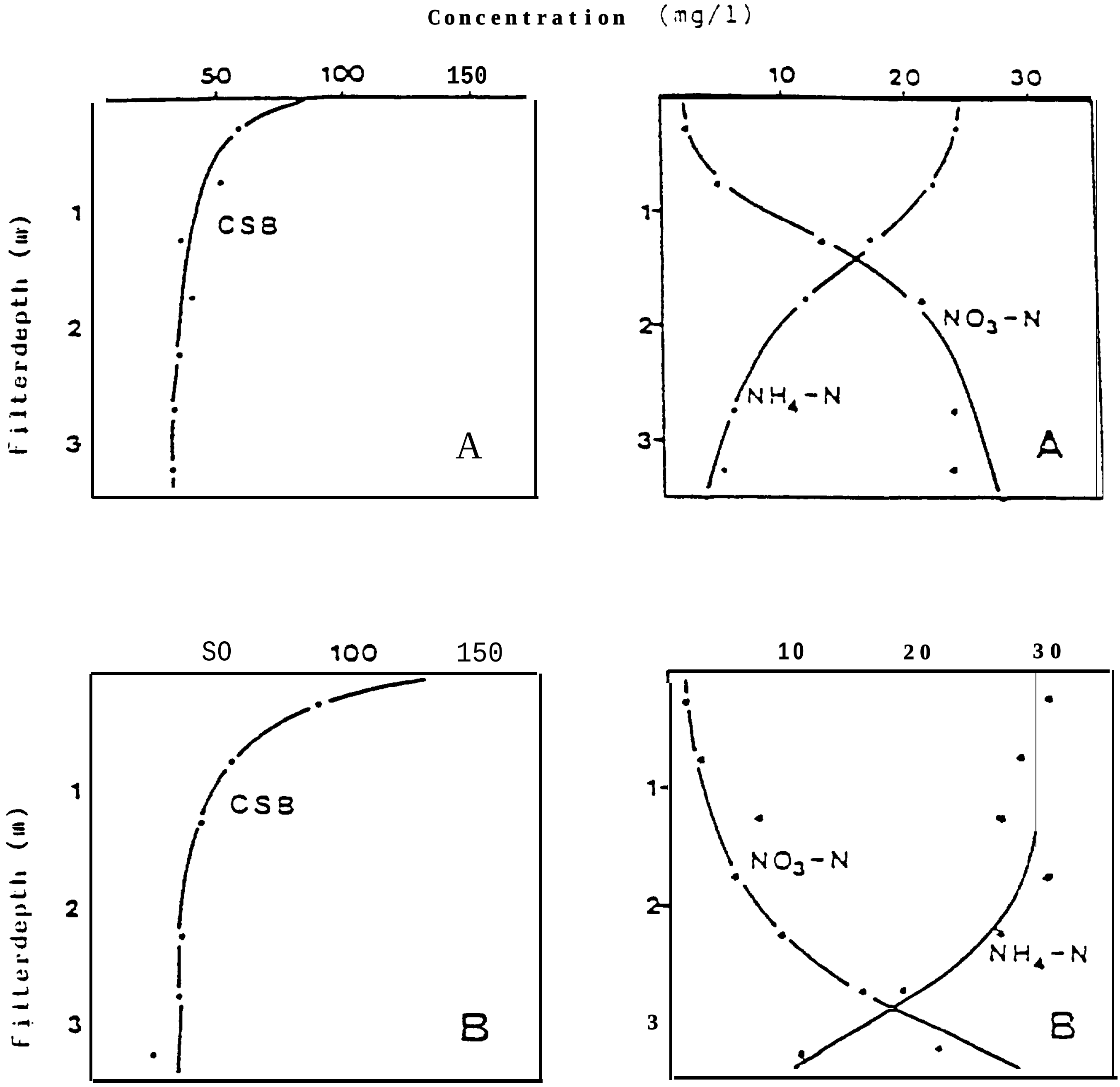

Fig. 3: Profiles of COD, NH_4^+, and NO_3^- measured in a trickling filter (3). Hydraulic load $q = 0{,}75\ (m^3/m^2$ ● $h)$.

For low ammonia concentrations in the effluent ($N_e \leq 5$ mg/l) by $t \geq 10$ °C a specific load of 15 ./. 20 $(mg/m^2 \cdot h)$ $N\text{-}NH_4^+$ can be recommended. The hydraulic load of the trickling filter has a big influence on the nitrification rates.

Good results are received with a hydraulic load

$$q_h \geq 1.5 \ (m^3/m^2 \cdot h) \ .$$

The recirculation rate can be chosen 100%.

To receive good results from a two-stage treatment plant it is necessary to optimite the first stage-activated sludge together with the second-stage trickling filter.

To use the equalizing efficiency of the activated sludge system it is reasonable to design this stage for partial nitrification

$$B_{TS} \ 6 \ 0.2 \ ./. \ 0.3 \ (kg \ BOD/kg \ SS \ . d)$$

The second stage trickling filter is nitrifying with good success, inlet concentration of ammonia (NH_4^+) not exceeding 20 ./. 30 mg/l ($N\text{-}NH_4^+$) (without calculating the dilution due to recirculation).

The detention time in the trickling filter is too short for equaliting peak $N\text{-}NH_4^+$ concentrations (Fig. 4).

For the enlargement of the treatment plant of town Flensburg (ca. 300000 inhabitants) are carried on pilot studies in situ.

A two-stage system consisting of first stage activated sludge for BOD removal and partial nitrification, second stage trickling filter for full nittification and denitrification in the final clarifiers was chosen.

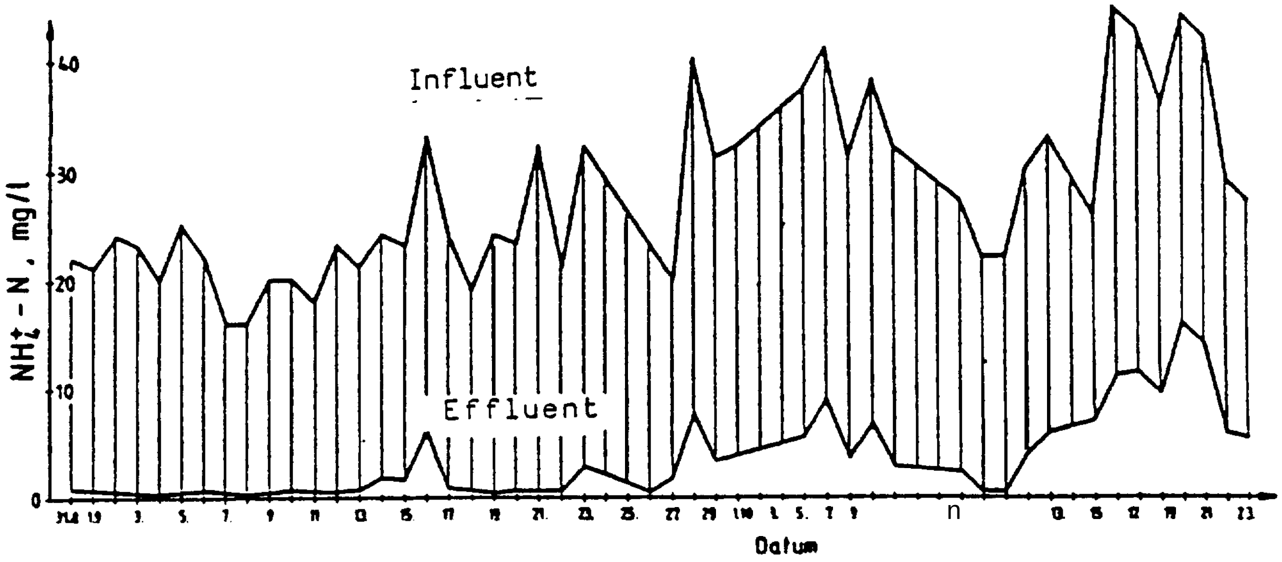

Fig. 4: Variation of the effluent concentration of the ammonia (NH_4^+) depending on the variation of the influent concentration.

On Fig. S can be seen the flow scheme. A biological. phosphorus removal is integrated to the first stage.

The research work will be finished at the end of May and the final report will be available in July.

3. Conclusions

- The requirements on the effluent quality became more stringent. The treatment plant today must be designed for the expected requirements of tomorrow.

- There are reliable techniques for achieving full nitrification and denitrification.

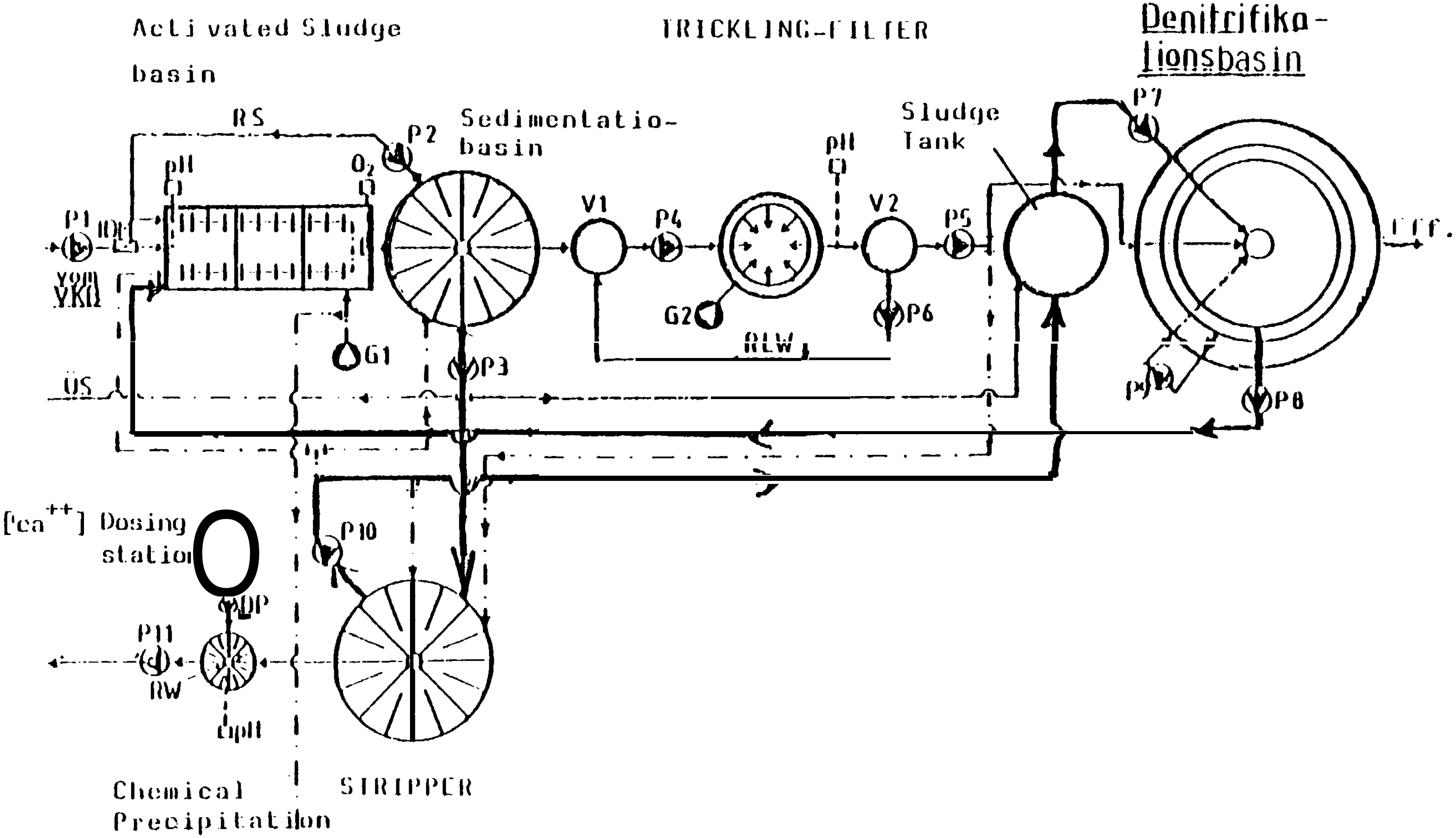

Fig.5: Flow scheme of the pilot plant at the treatment Flensburg for achieving BOD-removal and partial nitrification (activated sludge process), nitrification (trickling filter), denitrification and biological phosphor elimination.

- The activated sludge systems are the most reliable systems *to* reach high requirements today. The integration of a biological phosphorus removal to these systems can be considered as reasonable. The recommended organic load is $B_{TS} \leq 0.1$ ./. 0.2 (kg BSB/kg SS . d)

- The nitrification process is strongly temperature dependent. That is why to the limits of ammonia in the effluent the temperature range must be given also.

- The biofilm reactors e.g. trickling filters are suitable for nitrification. The biofilm grown on the support material has usually high sludge age benefitial for the growth of the nitrifyers.

- The two-stage treatment plants with first stage activated sludge system and second stage nitrifying trickling filter can have some advantages over the activated sludge systems.

- The nitrification rate of the trickling filter can be chosen $v_N = 15$ ./. 20 ($mg/m^2 \cdot h$) by $t \geq 12$ °C. The first stage activated sludge process must be designed in this case with an organic load

  B_{TS} 4 0.2 ./. 0.3 (kg BOD/kg SS .d)

  With the nitrifying filters normally it can be achieved a difference of ammonia between the influent and effluent of ca. $\Delta = 20$ ./. 30 mg/l ($N\text{-}NH_4^+$)

For optimizing this system further research work is necessary.

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SECOND SEMINAR ON WASTEWATER TREATMENT IN URBAN AREAS
VISBY, SWEDEN 6-8 SEPTEMBER 1987

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