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Evaluation of Ecological Consequences of Seawater Scrubber Effluent from the Fläkt-Hydro Sulfur Dioxide Removal Process when Applied to a 1200 MWe Coal Fired Power Plant

Norwegian Institute for Water Research NIVA

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Abstract

Based on dilution and dispersal calculations for effluents from a proposed 1200 MWe coal fired power plant in the Oslofjord, S. Norway, it is concluded that without treatment of waste water there is a risk for serious deterioration of the marine environment due to lowering of pH and oxygen concentrations. The effects will be less pronounced or insignificant with discharge to open waters. Aeration and lime treatment of the effluent will usually eliminate or reduce these problems considerably. Several uncertainties make the evaluation of metal pollution speculative. Presently available information indicate that unwanted effects should be moderate or small. The same applies to polycyclic aromatic hydrocarbons. Possibly mercury will be the most critical toxic substance. These conclusions are temporary as there is need for more data on effluent composition as well as results from long-term bioassay.

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Director: *Kjell Baalsrud*

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PREFACE

This work has been conducted for Norsk Viftefabrikk A/S by contract of October 11th 1979. The report is based on two previous reports to the Norwegian Water and Electricity Board (Bjerkeng and Knutzen 1978, 1979), with supplementary comments respecting two alternative treatment procedures (aeration with or without lime admixture) to reduce the potentially deleterious effects caused by the acidifying and oxygen demanding character of the untreated effluent.

After appearance of the above reports, Norsk Viftefabrikk A/S has performed studies with a pilot scrubber at the coal fired power plant of I/S Vestkraft, Esbjerg, Denmark. As a result of these studies the estimated discharge of metals has been reduced 75% compared to previous data about effluent composition. Consequently, the conclusions with regard to possible metal pollution have been somewhat modified.

Fil.kand. Jan Magnusson and cand.real. Lars Kirkerud have contributed to this work, respectively on the hydrography of the Oslofjord and concequences of pollution by metals and oxygen consuming material.

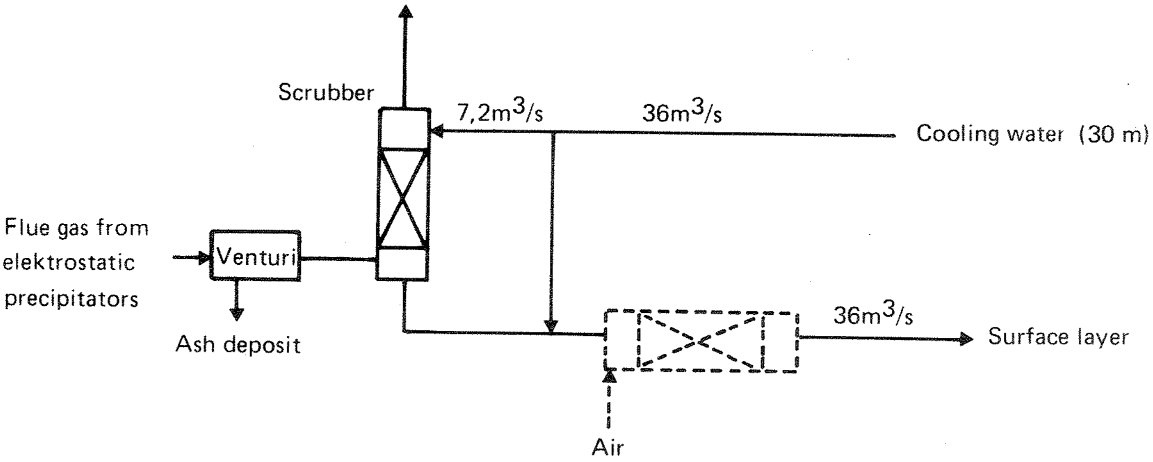
Oslo, April 10th 1980

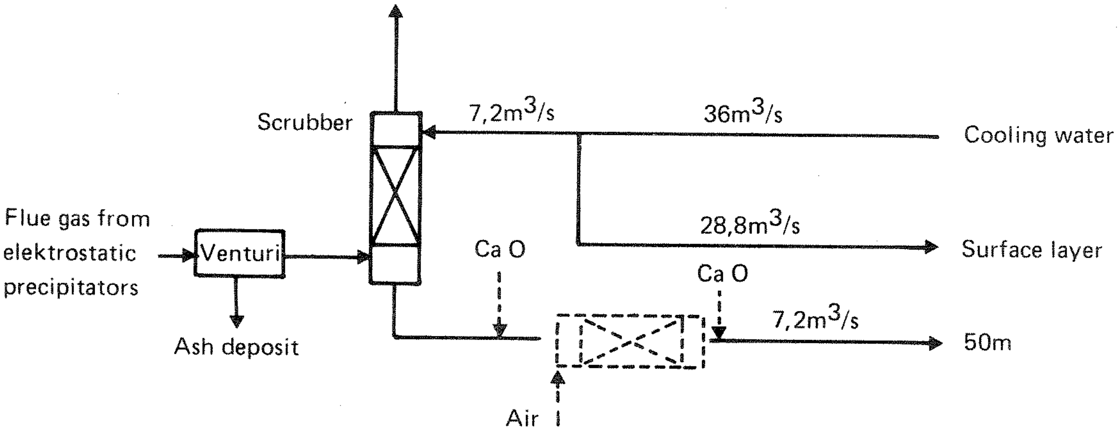
Jon Knutzen

Jon Knutzen

SUMMARY AND CONCLUSIONS

I The possible ecological concequences of the seawater scrubber effluent from the Fläkt-Hydro Sulfur Dioxide Removal Process applied to a 1200 MWe coal fired power plant have been evaluated. Information on effluent characteristics (metal concentration, pH, chemical oxygen demand) are based on metal content of the coals, percentage retention of different metals in electrostatic precipitators and studies with a pilot scrubber. Further, the evaluation is based on dilution and dispersal calculations primarily relevant for the proposed power plant sites in the Oslofjord area (fig. 5). Two alternative discharge arrangements are considered, respectively without and with effluent treatment (See fig. below, with broken lines marking treatment units and processes). The alternative effluents are considered with respect to acidity, oxygen demand and content of metals and polycyclic aromatic hydrocarbons.





II Scrubber effluent may be discharged to the surface layer together with the cooling water, or from a separate outlet at about 50 m depth.

Initial turbulent jet mixing results in diluted effluent being trapped by the density stratification in a layer of limited thickness, spreading horizontally outwards from the outfall area by gravitational forces. Dilution and entrapment depth are calculated by an established theory for turbulent jet mixing, but the limitations in gravitational transport capacity for the outfall as a whole is also considered.

Outside the close vicinity of the outfall, the effluent is thought to follow shifting recipient currents, spreading normally to current direction by gravitational forces and diffusion. A simplified model of such dispersion is used to estimate the extention of water volumes with effluent concentrations above given limits.

Concentrations for the whole recipient area are calculated from rough estimates of water renewal, based on current measurements and hydrographic data.

The result of the calculations are summarized in the following tables:

EFFECTS ON WATER QUALITY IN RECIPIENT FOR MIXED OUTFALL TO SURFACE LAYER. Alternative 1A: No Treatment, 1B: With treatment.

Dilution limit S	Water masses affected by effluent in dilution $\leq$ S			Water quality at dilution S			
				pH		O_2-contents (mg/l)	
	Max. distance (km)	Horizontal area (km^2)	Volume ($10^6 m^3$)	1a	1b	1a	1b
1	-	-	-	5.3-6.1	6.0	-8	2
2-3	2	2	2	6.3-6.7	7.0	∿1	5-6
5	5	5	5	6.8-7.0	7.5	4.5	6.7
10	15	20	30	7.1-7.4	7.8	6.5	7.4
20	40	150	350	7.3-(7.6)	7.9	7.0	7.7
30-50	-	∿300*	∿3000*	∿7.7	∿8.0	∿7.5	∿7.8
Background values				8.0	8.0	8.0	8.0

* Whole surface layer of outer Oslofjord.

EFFECTS ON WATER QUALITY IN RECIPIENT FOR SEPARATE OUTFALL OF SCRUBBER EFFLUENT AT 50 m DEPTH. Alternative 2A: No Treatment Alternative 2B: With treatment.

Dilution limit S	Water masses affected by effluent in dilution ≤ S			Water quality at dilution S			
				pH		O_2-contents (mg/l)	
	Max. distance (km)	Horizontal area (km^2)	Volume ($10^6 m^3$)	2a	2b	2a	2b
1	-	-	-	3.0	6.0	-70	-10
≤ 10	0-0.4	0-0.6	0-0.8	6.3-6.5	7.7	- 1	4-5.8
≤ 15	0-0.8	0-1.5	0-2.5	6.6-6.7	7.85	0.4-3.4	4.5-7.3
≤ 25	0-2	0-5	0-10	6.8-7.1	7.9	2.4-5.4	4.9-7.8
≤ 40	0.2-5	0.1-12	0.2-34	7.0-7.3	~ 8.0	3.6-6.6	5.1-8.1
≤ 60	0.7-12	0.5-40	1.5-140	7.2-7.5		4.2-7.2	5.3-8.2
≤ 100	>2	1.5-150*	8 -6000*	7.4-(7.7)		4.7-7.2	5.35-8.35
≤ 250	>8	10 -150*	120-6000*	7.7-(7.9)		5.2-8.2	5.45-8.4
Background values				8.0	8.0	5.5-8.5	5.5-8.5

* Total area and volume for outer Oslofjord between 20m and 60m depth.

III With reservation for lack of data from adequate, long-term and life cycle studies of representative marine organisms, there is a risk that mixed discharge of cooling water and untreated scrubber waste water to the surface layer will harm the marine communities.
In the Oslofjord, the untreated effluent will cause a mean pH reduction close to 0.5 units in the upper 10 m of an area corresponding to the outer part of the fjord (table above, fig. 5). The resulting conditions would be particularly hazardous for sessile, benthic organisms, but may also affect phytoplantkton primary production. Aeration of the above effluent will reduce the seriously affected area and volume of surface water masses with at least 90%.

By separate, deep water discharge of untreated scrubber effluent lowering of pH would be restricted to intermediate layers, and to a relatively minor part of the total pelagic environment. Respecting pH effects this discharge arrangement would be more acceptable, as primary production and shallow water biota would not be exposed. Lime treatment and aerations followed by deep water discharge would make the pH effects negligible.

IV In spite of high chemical oxygen demand the untreated, separate discharge of scrubber effluent to 50 m depth will not cause serious oxygen deficit in the primary recipient area with the exception of water masses close to the outfall (cf. above table). However, under the circumstances in the Oslofjord, the oxygen consuming pollutional load must none the less be deemed unacceptable for some of the outfall sites. This is because of the deteriorating effects on the oxygen environment of the inner Oslofjord, which is already critical for bottom fauna and demersal fish.

Treatment with lime and aeration will reduce the oxygen demand of the effluent about 80%. Generally, this will make the effluent acceptable for many recipients. However, the COD load from the treated effluent corresponds to the organic content of untreated municipal sewage from about 70 000 persons, which may be significant in land-locked and/or polluted waters.

Significantly harmful effects due to oxygen consumption is not to be expected from surface layer discharge of mixed scrubber and cooling water. Even without treatment the water volume with oxygen content below 4.5 mg O_2/l would be small.

V Possible metal pollution is difficult to evaluate due to considerable variation in the metal content of coal (table 1). Further, there is need for more information about the collection efficiency of the electrostatic precipitatiors with regard to the individual elements and also their distribution between the air and water phase. It is assumed here that a mean of 25% of the amount passing the filters follow the waste water. For mercury in particular this may be an overestimation.

Assuming "normal" metal concentrations in coal (table 1) only beryllium, mercury and lead are discharged in amounts giving overconcentrations several times higher than the background level in oceanic water. These overconcentrations will be restricted to to relatively small water wolumes in the vicinity of the outfall Even with high estimates of metal content in coal, the risk of toxic effects is moderate. (Cf. table 10 with tables 4, 7). Reservations must be taken, however, due to lack of data from long-term tests and insufficient knowledge about possible synergistic effects.

The risk of accumulation to unwanted metal levels in seafood organisms probably is small and should not give rise to serious concern, provided that the assumed metal levels in the effluent holds true.

Based on present information mercury seems to be the critical metal both with regard to poisonous effects and bioaccumulation.

With regard to the effects of metals, deep water discharge probably would be preferable in most (but not all) cases. The reason for this is avoidance of stress in the zone of primary production and on shore and shallow water organisms.

VI The amounts of polycyclic aromatic hydrocarbons (PAH) and other organic micro pollutants in the scrubber effluent are not known. There is some indications from flue gas analysis that the quantities of PAH from modern boilers are rather small. If, on the other hand, high estimates should appear to be correct, at least some categories of recipient areas would experience accumulation problems warranting serious concern.

VII Particularly for the evaluation of the effects from large amounts of scrubber water, there is a need for long-term biotest data on the effects of moderate pH reduction on different life stages of representative marine species. In connection with such tests one should be aware of the influence of pH-reduction on the equilibrium between simple ions and other dissolved forms of those metals which may act toxic in concentrations close to natural levels.

1. INTRODUCTION

Flue gas from fossil fueled power plants contain considerable amounts of several pollutants, among them particles, oxides of nitrogen and sulfur, polynuclear aromatics and metals. Emittance of these substances may have serious consequences for the atmospheric environment, freshwater and soil resources.

Common protection measures include removal of most particulate matter and the utilization of fuel with low sulfur content. Various flue gas desulfurization processes have also been put into use. Among the most promising of these are sea water scrubbing.

The transfer of sulfur oxides from air to water, however, results in an acid effluent in spite of the high buffer capacity of sea water. The considerable quantities of water required in the scrubbers of large power plants may give rise to unwanted effects in the receiving waters. Besides lowered pH the scrubber effluent is characterized by a high chemical oxygen demand (COD). It also contains some metals and organic micropollutants (polycyclic aromatic hydrocarbons, related heterocyclics and others); mostly associated with particulate matter. The concentrations of metals and organics are largely unknown since there ar no coal fired power plants with sea water scrubbers installed.

The discharge of cooling water may represent an additional stress on marine biota in terms of increased temperature and residual chlorine. These stress factors have been the subject of thorough reviews elsewhere, and are not considered here.

2. OUTLINE OF ALTERNATIVE TREATMENT PROCESSES AND DISCHARGE ARRANGEMENTS

From the boilers the flue gas passes electrostatic precipitators and a closed liquid loop venturi scrubber before it enters the "once through" sea water scrubbers. The electrostatic precipitators remove approximately 99.5% of the flue gas content of particulate matter. The removal efficiency varies somewhat according to the characteristics of the coal.

About 1/5 of the cooling water is utilized in the scrubbers. Scrubber effluent and cooling water may be discharged together (alternative 1) or separately (alternative 2). In the first case the effluent may be discharged without further treatment (1a) or aerated to reduce the acidity and COD (1b). By separate outfalls the scrubber water may be discharged without treatment (2a) or treated with lime and thereafter with air and lime (2b).

Effluent treatment, intake and discharge depth must be considered in relation to recipient conditions, economy, etc. In the following, the above mentioned alternatives (figs. 1-4) are evaluated with localities in the Oslofjord area as examples (fig. 5). The flow of scrubber and cooling water denoted in figs. 1-4 concerns a 1200 MWe coal fired power plant.

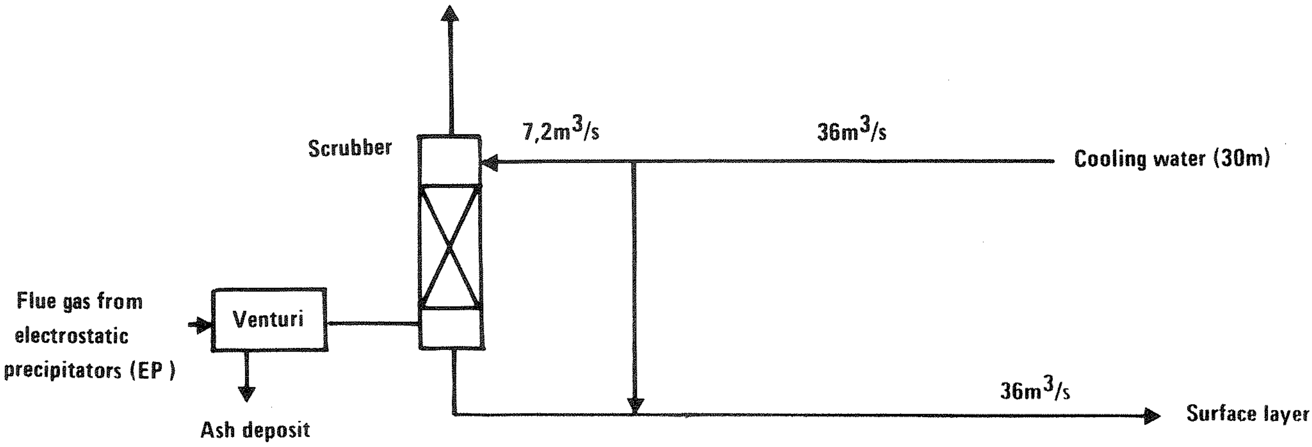


Fig. 1. Alternative 1a. Intake depth 30 m. Combined discharge of scrubber and cooling water to surface layer. No treatment of waste water.

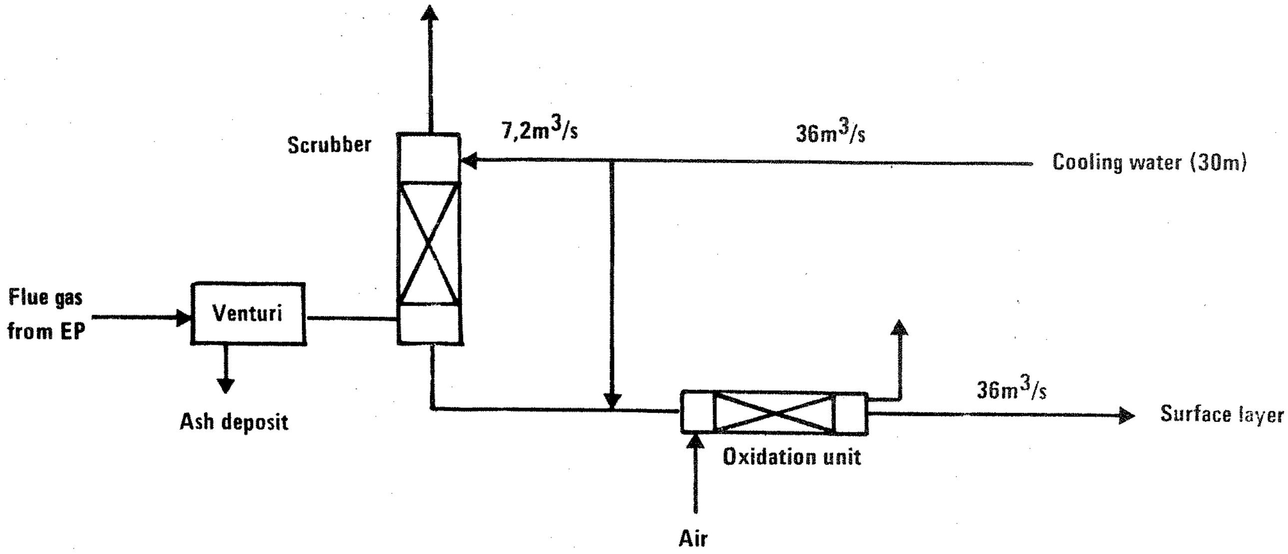


Fig. 2. Alternative 1b. As 1a, but with aeration of the effluent.

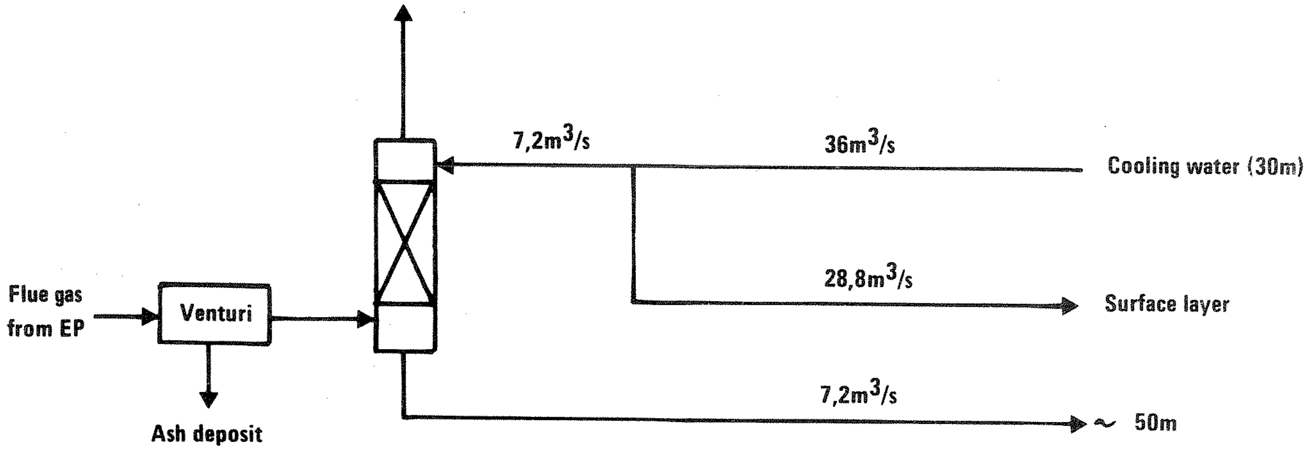


Fig. 3. Alternative 2a. Intake depth 30 m. Separate discharge of scrubber effluent (to about 50 m depth) and cooling water (to surface layer).

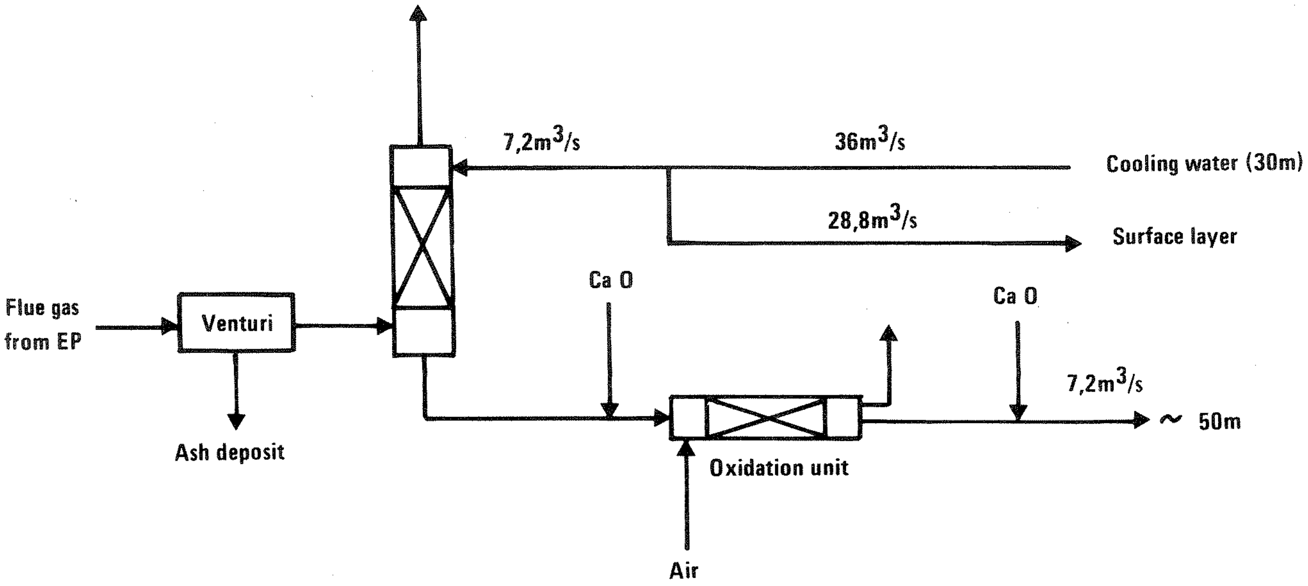


Fig. 4. Alternative 2b. As 2a, but treatment of scrubber water with lime and aeration.

Total lime consumption is about 5.5 t/h (0.8 mole Ca per 1 mole SO_2 removed). About 5% residue of particulate matter (mainly MgO, SiO_2, $CaCO_3$) will go to the recipient. Lime may be replaced by lime stone and hydrated lime.

3. POWER PLANT AND EFFLUENT DATA

Size: 2 units of 600 MWe, 7000 hours per year.

Coal consumption: 2.8×10^6 tons per year, i.e. ∿400 t/h.

Sulfur content of coal: 1%. This is presumed to be a reasonable estimate of the mean sulfur concentration. Mostly, the sulfur content of coal deposits will be less than 1% (Swedish Department of Industry/Energy Commission 1978, Løvblad, 1977). In some cases, however, it may be considerably higher.

Ash content of coal: 15%

Cooling water: 36 m^3/s.

Scrubber water: 7.2 m^3/s.

Temperature increase of effluents: About 12°C in cooling water and mixed effluent and about 18°C in separate scrubber effluent.

Suspended solids: About 300 kg/h, i.e. about 12 mg/l (in lime treated scrubber effluent of 7.2 m^3/s).

Sulfur dioxide: About 8 t SO_2/h. By evaluation of the consequences for the water recipient it is assumed that 5% of the sulfur is removed with the ash, Thus 95% goes to water.

COD: The chemical oxygen demand will vary with the alternatives shown in figs. 1-4. Even if the COD is higher than the natural oxygen content of the sea, the oxidation of sulfite to sulfate takes some time, also depending on pH (Gramme & al., 1974). By relatively high pH (above 5.5-6.0) the complete consumption of the natural oxygen content of sea water will take place within a few minutes or even less than a minute. This applies to alternative 1a, but not alternative 2a, in which pH is considerably lower. However, in order to facilitate dilution calculations and comparison between the alternatives, COD is given as theoretical gross values for both these alternatives. For the alternatives with aeration (1b, 2b), values are given as indicated by the percentage reduction of sulfite as measured in pilot plant tests by A/S Norsk Viftefabrikk (unpubl.).

Alt. 1a (fig. 1): ∿ 16 mg O_2/l

Alt. 1b (fig. 2): ∿ 4.8 mg O_2/l (70% reduction of original COD. At the same time 80% oxygen saturation in the effluent.)

Alt. 2a (fig. 3): ∿ 77 mg O_2/l

Alt. 2b (fig. 4): ∿ 15 mg O_2/l (∿ 80% reduction of original COD and about 80% oxygen saturation in effluent water).

The untreated effluent represent a load of approximately 50 t COD per day, corresponding to the content of organic material in domestic sewage from about 350 000 persons (Bjerkeng and Knutzen 1978). It should be stressed that the content of plant nutrients in municipal sewage (phosphorus and nitrogen compounds) may cause primary production representing several times more organic matter than the organics directly discharged.

pH: Varies with the alternatives presented in figs. 1-4.

Alternative 1a (fig. 1): ∿ 5.7 (somewhat depending on the speed of oxidation of sulfite to sulfate, see fig. 6).

Alternative 1b (fig. 2): ∿ 6.0. (Increase in pH due to stripping of CO_2 by aeration).

Alternative 2a (fig. 3): ≤ 3

Alternative 2b (fig. 4): ∿ 6.0

Metals: The metal content of coal shows considerable variation between different deposits. By the combustion and the subsequent flue gas treatment the metals are distributed between boiler bottom ash, particulate matter collected by the electrostatic precipitators and the venturi scrubber, the stack gas and the water effluent. The proportions ending up on site, in air or water, will depend on the metal in question, coal characteristics,combustion conditions, and other details of the treatment equipment, operation and maintenance. Consequently, neither input nor removal efficiency of each treatment stage can be stated in exact terms. Table 1 contains estimates based on available data. Figures for element concentrations in coal and percentage passing the electrostatic precipitators are in accordance with data given in a report from the Swedish Industry Department/ Energy Commission (1978). Values in parenthesis refer to extreme concentrations quoted in the same report or Lövblad (1977). For mercury there are indications that the efficiency of the electrostatic filter may be considerably higher than it appears from the table (cf. ref. in Norsk Viftefabrikk A/S, 1979, unpubl. report).

Table 1. Estimates of metal discharge in the water effluents from a 1200 MWe coal fired power plant. High estimates in parenthesis. Increase in ambient sea water concentrations given for mixed effluent (alternative 1, figs. 1-2) and for separate discharge of scrubber water (alternative 2, figs. 3-4). For further comments, see text.

Element	Conc. in coal, g/t	% passing electrostatic precitator	Discharge per year kg	Increase in metal conc. µg/l Alt. 1	Increase in metal conc. µg/l Alt. 2
As	5.0 (15)	1.8	63 (190)	0.07 (0.21)	0.35 (1.1)
Be	1.5 (<5)	2.0	21 (<60)	0.02 (<0.06)	0.10 (<0.30)
Cd	0.2 (2.5)	2.9	4 (50)	<0.01 (0.05)	0.02 (0.28)
Co	6.0 (18)	0.67	28 (85)	0.03 (0.09)	0.15 (0.45)
Cr	12.0 (30)	1.2	100 (250)	0.11 (0.28)	0.55 (1.4)
Cu	15.0 (24)	0.47	50 (80)	0.06 (0.09)	0.28 (0.45)
Hg	0.1 (0.5)	90.0	63 (315)	0.07 (0.35)	0.35 (1.75)
Mn	40.0 (225)	0.47	130 (730)	0.14 (0.80)	0.71 (4.0)
Ni	15.0 (45)	0.43	45 (135)	0.05 (0.15)	0.25 (0.75)
Pb	10.0 (60)	3.9	275 (1650)	0.32 (1.9)	1.6 (9.5)
Se	2.0 (3.7)	1.8	25 (45)	0.03 (0.06)	0.15 (0.28)
V	25.0 (80)	1.0	175 (560)	0.19 (0.60)	0.95 (3.0)
Zn	45.0 (270)	1.9	600 (3060)	0.65 (3.3)	3.2 (16.5)

Amounts discharged to water have been calculated on the basis that about 1/4 of the quantity passing the electrostatic filters will end up in the water. According to introductory pilot plant investigations (Norsk Viftefabrikk A/S, 1979 unpubl. report), this may be taken as a reasonable mean. In practice one must expect considerable variations between metals, and further experimental data are needed.

Organic compounds: The formation of polycyclic aromatic hydrocarbons (PAH) and heterocyclic compounds is known to take place, but the amounts of cancerogenic and other substances are much disputed. Estimates of PAH and benzo(a)pyrene (B(a)P) range from 0.1-10 and 0.01-1 mg/kg coal, respectively (Swedish Department of Industry/Energy Commission, 1978). For the power plant in question this gives 0.3-30 t PAH and 0.03-3 + B(a)P per year. In large and modern boilers it is believed that the lowest figures are most probable (Alfheim & al., 1978, Alfheim and Gether, 1978). Again, as for metals, one lacks data for the distribution between the air and water phases.

Chlorinated compounds: In the mixed scrubber and cooling water effluent (alternative 1) residual free chlorine and chlorinated compounds probably will be largely neutralised by the reducing power of the scrubber water. In the other cases (alternative 2), the concentrations of hypochlorite, chlorinated amines, etc. will as usual depend on the chlorination procedure.

4. DILUTION AND DISPERSAL OF EFFLUENT

4.1 General description of recipient

The effects of the different effluent alternatives have been studied theoretically for locations around the Oslofjord in Norway. The area is shown in fig. 5, with location alternatives marked.

The Oslofjord consists of two distinct parts. The inner fjord north of Drøbak has a surface area of about 200 km^2 and maximum depth around 150 m. The deep layers of the inner fjord are closed off by a sill of depth 20 m at Drøbak. The rest of the fjord has a maximum depth of 350 m, and is open towards Skagerrak down to 100 m depth.

The fjord is usually divided into a central part between Drøbak and Horten-Moss, and an outer part extending south to Færder. In this report we take the fjord between Drøbak and Tønsberg as being the main recipient for the effluent. This part of the fjord has a surface area of $\sim$ 300 km^2, and a maximum depth 300 m. About half of the bottom area has a depth of more than 50 m. The locations are not studied individually, though some differences are pointed out. The locations at Larvik are not studied specifically. The near zone concentration are assumed to be about the same for these locations as for the others, but the overall recipient concentrations calculated does not apply.

The Oslofjord has two major freshwater sources. The river Dramselva discharges about 300 m^3/s into the Dramsfjord connected to the northwestern part of the central Oslofjord, and the river Glomma about 700 m^3/s into the southernmost part of the fjord, at Fredrikstad.

As a result the Oslofjord has a brackish surface layer of thickness 10-20 m and with salinities in the range 10-25 ^{o}/oo. Below the surface layer, and separated from it by a layer of rapidly changing salinity and density, are intermediate layers, where salinity increases more slowly with depth. The water may be quite homogeneous below 20 m depth, but usually there is a significant gradient down to 50-100 m depth.

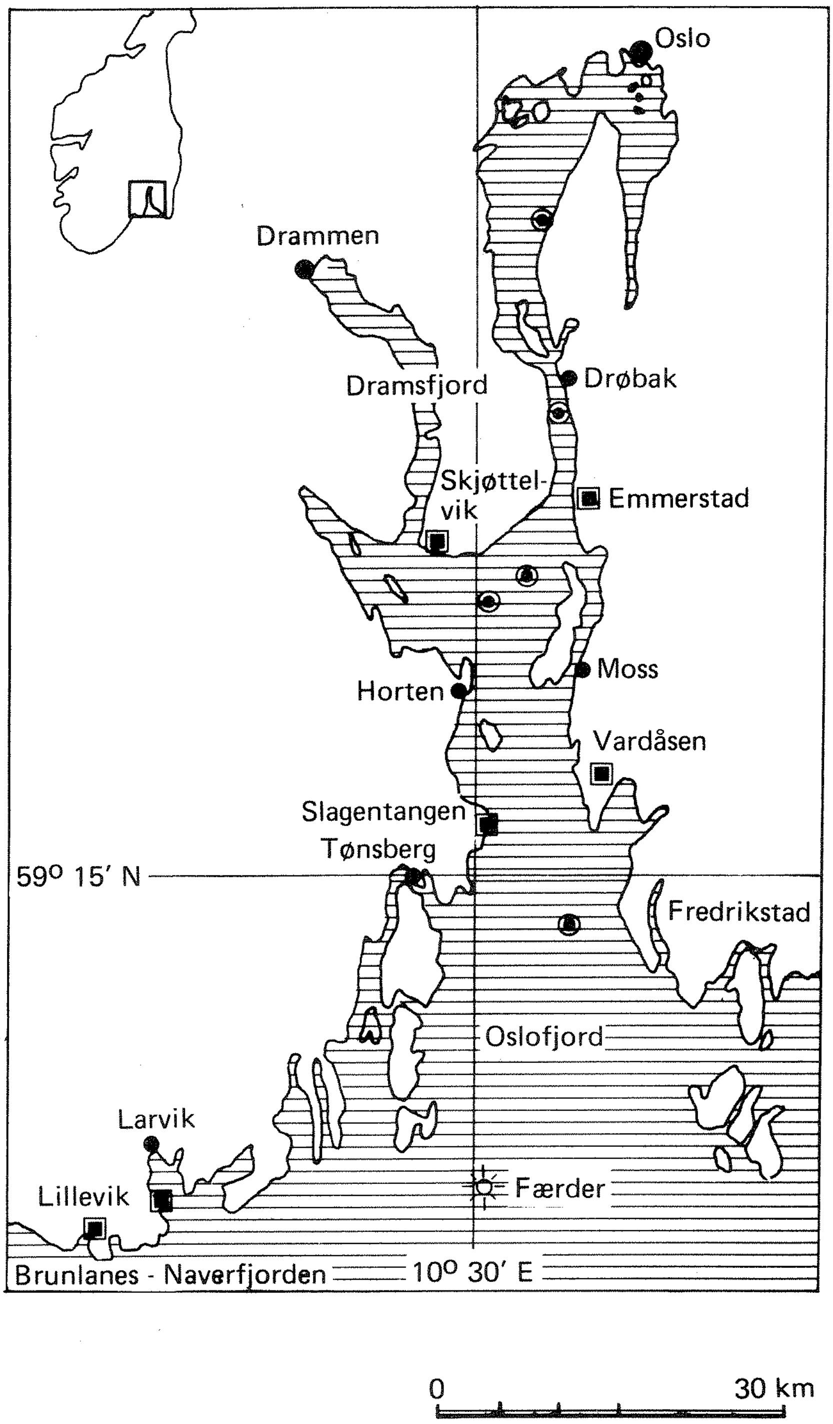


Fig. 5 . Alternative power plant sites in the Oslofjord area ▣ , with stations for hydrographic observations used in dilution and dispersal calculations ◉.

Hydrographic data and current measurements show that the central part of the fjord is not dominated by stationary flow patterns. Water in all depths are renewed by large horizontal movements into and out of the fjord, with corresponding vertical displacements, due to outside hydrographic changes and variations in meteorological conditions and fresh-water discharge.

4.2 General description of dispersal processes

The effluent is assumed to be discharged in submerged, turbulent jets, directed horizontally into the recipient.

The cooling water outfall nearsurface will consist of relatively few, rather large jets, with initial diameters 1-3 m. The scrubber effluent will be mixed into the cooling water (alternatives 1a, 1b) or discharged at about 50 m depth through a diffusor arrangement (alternatives 2a, 2b), consisting of one or more long pipe segments with a large number of holes along the sides. In the latter case the jet diameters are 20-40 cm.

Jet velocities are assumed to be in the range 2-4 m/s.

The dispersal of the effluent will in principle take place in the same way for both outlet arrangements, with quantitative differences due to outflow magnitude and discharge depth.

In the first phase after discharge the jets move along defined trajectories, mixing violently with recipient water because of the created turbulence, and thus increasing in volume flux and changing in density.

The cooling water, taken from 30 m depth and discharged at the surface, will normally be heavier than the surface water in the recipient, even with a temperature increase of 10°C. The jet trajectories will be deviated downwards, and the mixed effluent is trapped in a submerged layer around the depth where the density of diluted waste water, decreasing from mixing, just equals the ambient density, increasing with depth.

For separate discharge of scrubber water at 50 m depth the jet density will be less than ambient density at discharge depth due to density difference between 30 and 50 m depth in the recipient, and this is increased by the temperature increase of 18°C. The jets will rise towards the surface, and be trapped in a layer around the depth where the increasing jet density equals the ambient density.

If there is no practical limits to the available amount of dilution water, and the jets do not interfere with each other, primary dilution and resulting depth are determined by the dynamics of each individual jet. A theory presented by Fan and Brooks (1969) was solved numerically by a computer program (Bjerkeng and Lesjø, 1973) to give those values for varying outlet conditions and observed density profiles.

The estimates given by this theory will be achieved if the natural currents in the recipient transports the necessary amounts of dilution water through the outfall area.

In situations with more stagnant recipient conditions, the dilution may be further limited by the capacity of the recipient to transport dilution water towards the outlet area, and diluted effluent out from the area.

When a plume of relatively homogeneous density is trapped in a stagnant recipient where density is varying with depth, gravitation will create horisontal pressure gradients, spreading the plume out in a layer of limited thickness. In the close vicinity of the outfall the gravitational spreading will dominate in all directions. Similar pressure gradients will draw dilution water towards the mixing area.

For both types of flow there is an upper limit to the amount of water which can be transported through a given cross-section.

This is the critical flow, which for a single flowing layer in an otherwise stagnant recipient is given by the equation

$$h_c = \gamma \left(\frac{Q}{N \cdot b} \right)^{\frac{1}{2}} \tag{1}$$

where h_c = flowing layer thickness at the critical cross-section.
Q = total volume flux
N = measure of stability (Brunt - Väisäla - frequency) defined by

$$N^2 = \frac{g}{\rho} \frac{d\rho}{dz}$$

b = horisontal width of critical cross-section

g = gravitation constant ($\simeq$ 9.81 m/s^2)

ρ = water density

z = depth

γ = dimensionless coefficient, with empirical values
2 for surface or bottom flow,
2.8 for intermediate flow

The equation may be deduced by simple application of Bernoullis equation on outward front spreading of an intermediate layer or for flow towards a sink ("selective withdrawal") and represents the situation with lowest possible total energy for a given volume flux and a given density gradient.

The mean velocity is given by

$$u = \alpha N h, \ \alpha = \frac{1}{\gamma^2} \qquad (2)$$

An incoming flow of dilution water, drawn radially towards the outlet area, will be fetched from a stagnant layer of thickness

$$h_{oc} = \frac{3}{2} h_c \qquad (3)$$

The critical width b is determined either by bottom topography, or by the extention of the outlet arrangement or the mixing area.

For an outflowing layer, towards increasing width, an energyconservative flow at given b would become super-critical for larger b-values. Such a flow is unstable, and may be expected to dissipate its excess of kinetic energy, so that equation (1) is approximately valid for increasing distance (and width), with decreasing u and h_c (maybe somewhat increasing Q due to mixing).

The equations assume that the waters above and below the flowing layer are stagnant, and are therefore strictly speaking not valid for a two-flow situation, where there is an incoming flow just below or above the outgoing flow. The restrictions may therefore in reality be somewhat stronger in terms of necessary thickness for given flows than calculated.

Estimates of gravitational spreading from eq. (1) compared to observed currents in the recipient indicate that the gravitational forces will dominate in all directions merely within a few hundred meters from the outlet.

Further away the effluent is supposed to be transported as a plume following the currents of the recipient, and spreading out horizontally and vertically in the cross-section normal to the flow. With recipient currents changing in velocity and direction over time this is expected to create "patches" of varying size and concentrations above the overall values for the affected layer in the recipient.

A theory for such plume spreading is outlined in an appendix. Spreading is calculated by this theory under varying conditions for a stationary plume, and the results are assumed to describe statistically the extent of given concentrations also for varying recipient currents.

4.3 Effect of SO_2 on pH and oxygen content as a function of dilution

Abdulsattar et al. (1977) have calculated pH in seawater theoretically as a function of SO_2-content, in a closed system with no free oxygen present. Calculations are made for the equilibrium of the following set of chemical reactions

$$SO_2 + H_2O \rightleftharpoons H^+ + HSO_3^-$$

$$HSO_3^- \rightleftharpoons H^+ + SO_3^{--}$$

$$CO_2 + H_2O \rightleftharpoons H_2CO_3 \rightleftharpoons H^+ + HCO_3^-$$

$$HCO_3^- \rightleftharpoons H^+ + CO_3^{--}$$

$$HSO_4^- \rightleftharpoons H^+ + SO_4^{--}$$

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

with mass-balances for CO_2, SO_2, SO_4 and electro-neutrality condition.

According to Bromly (1972) and Gramme et al. (1974) SO_3^{--} will oxidize by the reaction

$$SO_3^{--} + \frac{1}{2} O_2 \longrightarrow SO_4^{--}$$

in about 1 hr for pH ∿ 4, in a few minutes for pH ∿ 7. The reaction will deplete oxygen content of the water, and for pH-values below ∿ 6.0 the acidity will increase compared with no oxidation (Bromley, 1972, see also fig. 6). This effect can be meutralized by aeration, leading to an increase in pH due to stripping of carbon dioxide from the liquid.

Maximum 10% of the amount of SO_2 is assumed to be oxidized by oxygen from the flue gas. Oxygen will mainly be taken from the effluent and the recipient water.

In table 2 oxygen data are given for the four alternatives outlined in chapter 2. The values for oxygen content are estimated from hydrographic data for the recipient, and from tables of oxygen saturation values.

Table 2. Oxygen values in effluent and recipient

Outlet alternatives	Oxygen demand in effluent COD (mg/1)	Initial oxygen reserve in effluent R (mg/1)	Oxygen content in dilution water C_o (mg/1)
Alt 1a:	16	8	8
Alt 1b:	4.8	∿ 6.5	8
Alt 2a:	77	∿ 7	∿ 7
Alt 2b:	15	∿ 5.5	∿ 7

The effluent will have an oxygen content of R-COD if oxidation is complete. For a dilution factor S, the oxygen content will be:

$$c = \frac{C_o (S-1) + R - COD}{S}$$

moving towards C_o as S increases.

The expected effects on pH by alternative effluents and dilutions are summarized in fig. 6.

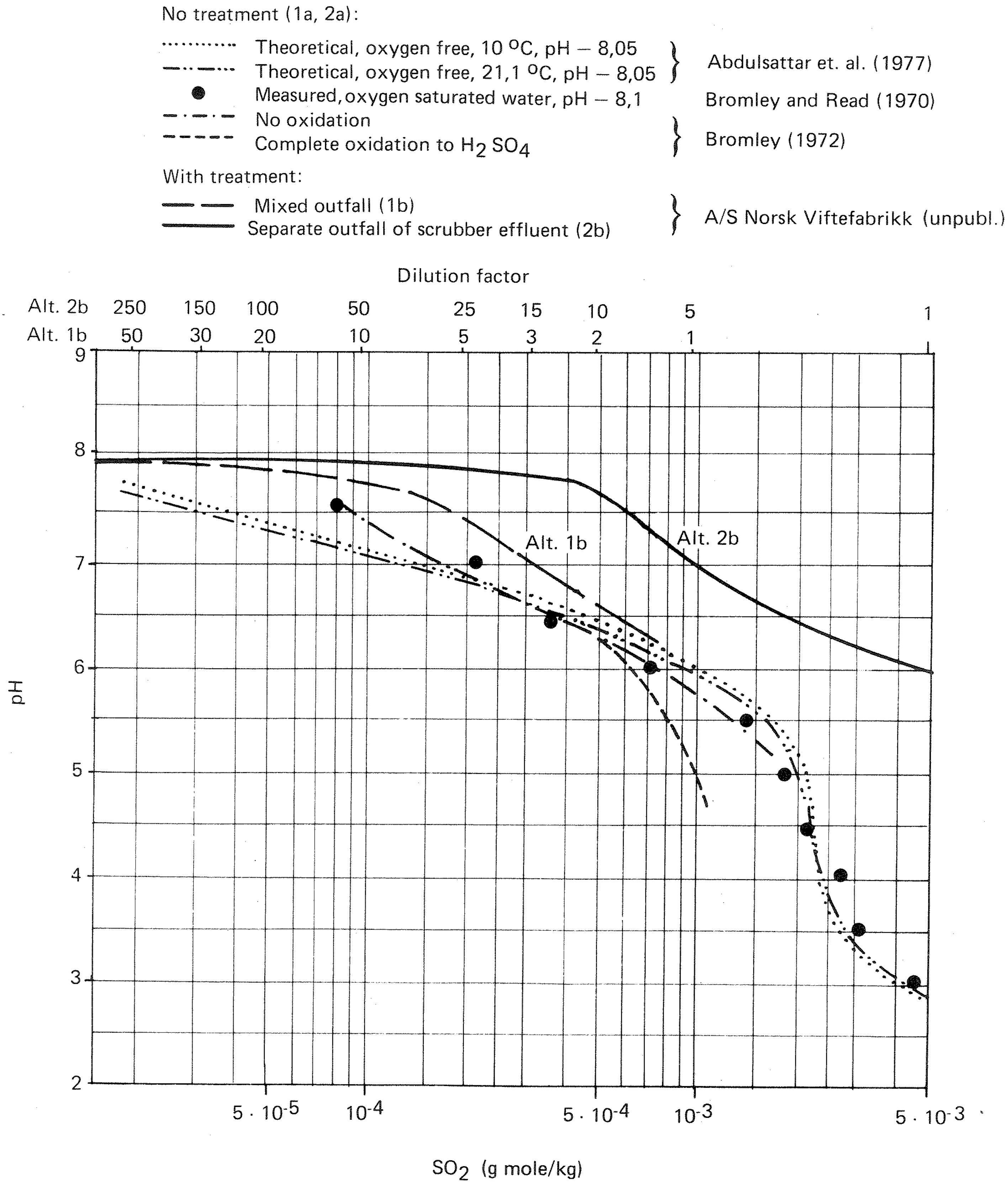


Fig. 6. pH in seawater as function of sulfur contents.

4.4 Estimated effects of effluent on water quality of outer Oslofjord

4.4.1 Mixed outfall of cooling water and scrubber effluent to the surface layer

Calculations were performed for three alternative outlet arrangements, shown in table 3.

Table 3. Assumed alternatives, mixed outfall. Velocity 2 m/s, volume flux 36 m^3/s.

Alternative	Nr. of jets	Jet diameter (m)	Depth (m)
I	2	3.4	1.7
II	8	1.7	0.8
III	32	0.8	0.4

Calculations are based on two density profiles for each of the five locations, observed 1.-2. June 1978 and 10. August 1978. The profiles were observed specially for this purpose, and gave a more detailed description of the surface layer than usual.

Primary dilution and depth of the effluent plume were first calculated without regard to the limited transport capacity in the recipient. These calculations indicated dilution factors 3-4 for alt. I, 5-9 for alt. III, with entrapment depth 5-7 m and 1-6 m respectively, varying with location and date.

When compared to the required thickness of the incoming flow of dilution water at the surface according to eq (1), the results were seen to be clearly unrealistic, because the diluted effluent would be found inside the incoming layer.

A new calculation was therefore made, with mixing coefficient reduced to 2/3, to indicate the distortion of the density profile near the jets according to eq. (3).

The conclusion is that for two large jets near surface the mixed effluent will normally be trapped in a layer at 5-10 m depth with a mean dilution factor of about 3, about 2 in the center of the layer. Even for the largest number of jets considered dilution factors below 5 must be expected.

If the effluent is discharged some distance below surface (∿ 5 m) dilution may be increased 25-50%. Horizontal distance between each jet should be 80, 45 and 30 m respectively, for the three alternatives.

The low dilution factors are due partly to the strong stratification in the surface layer, which means the effluent moves a small vertical distance before being trapped in a horizontal layer, and partly to the restricted transport capacity of the recipient.

For the surface layers of the recipient as a whole the water reneval is roughly estimated by Rye and Thendrup (1978) to be in the range 1000-1800 m^3/s, based on current measurements. This gives overall concentrations of effluent in the range 1:30 - 1:50, and the larger parts of the surface layers of the recipient are assumed to be affected by these concentrations.

The water volumes affected by larger concentrations are calculated by the plume spreading theory and parameters given in the appendix.

The results are summarized together with values for pH, oxygen and metal concentrations in table 4.

For dilutions 20-30 the affected water volumes may be very much smaller if the overall dilution is higher than assumed in the calculations.

TABLE 4. EFFECTS ON WATER QUALITY IN RECIPIENT FOR MIXED OUTFALL TO SURFACE LAYER. ALTERNATIVE 1A, 1B.

Dilution limit S	Water masses affected by effluent in dilution $\leq$ S			Water quality at dilution S										
				pH		O_2-contents (mg/l)		Metal overconcentrations (High estimates in parenthesis) µg/l						
	Max. distance (km)	Horizontal area (km^2)	Volume ($10^6 m^3$)	1a	1b	1a	1b	Be	Cd	Co	Cr	Hg	Pb	Zn
1	-	-	-	5.3-6.1	6.0	-8	2	0.0200 (<0.0600)	<0.0100 (0.0600)	0.0300 (0.0900)	0.1100 (0.2800)	0.0700 (0.3500)	0.3200 (1.9000)	0.6500 (3.3000)
2-3	2	2	2	6.3-6.7	7.0	∿1	5-6	0.0080 (<0.0240)	<0.0040 (0.0240)	0.0120 (0.0360)	0.0440 (0.1120)	0.0280 (0.1400)	0.1280 (0.7600)	0.2600 (1.3200)
5	5	5	5	6.8-7.0	7.5	4.5	6.7	0.0040 (<0.0120)	<0.0020 (0.0120)	0.0060 (0.0180)	0.0220 (0.0560)	0.0140 (0.0700)	0.0640 (0.3800)	0.1300 (0.6600)
10	15	20	30	7.1-7.4	7.8	6.5	7.4	0.0020 (<0.0060)	<0.0010 (0.0060)	0.0030 (0.0090)	0.0110 (0.0280)	0.0070 (0.0350)	0.0320 (0.1900)	0.0650 (0.3300)
20	40	150	350	7.3-(7.6)	7.9	7.0	7.7	0.0010 (<0.0030)	<0.0005 (0.0030)	0.0015 (0.0045)	0.0055 (0.0140)	0.0035 (0.0175)	0.0160 (0.0950)	0.0325 (0.1650)
30-50	-	∿300*	∿3000*	∿7.7	∿8.0	∿7.5	∿7.8	0.0005 (<0.0015)	<0.0003 (0.0015)	0.0007 (0.0023)	0.0027 (0.0070)	0.0018 (0.0087)	0.0080 (0.0475)	0.0162 (0.0825)
Background values **				8	8	8.0	8.0	0.005	0.03	0.05	0.3	0.02	0.05	3

* Whole surface layer of outer Oslofjord.

** Metal levels in oceanic water; oxygen levels from the Oslofjord.

4.4.2 Separate outfall of scrubber effluent at ∿ 50 m depth

An upper limit for primary dilution and depth of the effluent layer were first calculated by the theory of Fan and Brooks (1969) with no regard to transport capacity of the recipient, as for a mixed outfall.

The calculations were made for a number of different outlet geometries, and for the six density profiles in fig. 7. The profiles constitute representative examples from 77 profiles studied, covering the periods 1963-1970 (Norwegian Institute for Water Research) and 1973-1974 (River and Harbour Laboratory).

Outlet alternatives are shown together with the results in table 5.

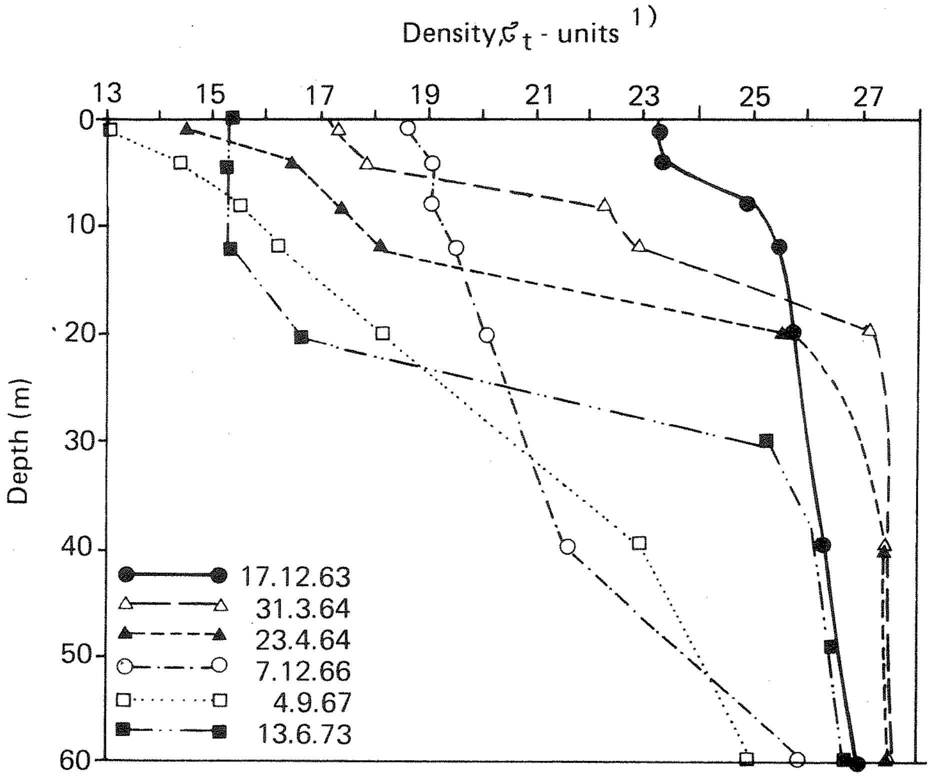


Fig. 7. Representative examples of density profiles in the outer part of the Oslofjord. Based on data from 1963-1970 (Norwegian Institute for Water Research) and 1973-1974 (River and Harbour Laboratory).

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[1] σ_t: (density -1) · 1000

Table 5. Upper limit for primary dilution with alternative jet geometries and outlet depths. First values correspond to strong density stratification (low dilution, entrapment close to outlet depth), the second value to weak stratification. Dilution figures in parenthesis are median values.

Alternative nr.	Outlet depth (m)	Jet data			Mean dilution factor S	Depth of effluent layer (m)	Width of jet at entrapment depth (m)
		Diameter (m)	Velocity (m/s)	Approx number			
1	40	1.5	4	1	9-19 (13)	35-20	10-18
2		0.4	2	30	12-40 (20)	37-23	3-8
3		0.2	2	120	17-55 (30)	38-28	2.5-5
4		0.2	4	60	22-60 (35)	39-30	3.5-7.5
5	50	1.5	4	1	9-22 (15)	45-20	10-20
6		0.4	2	30	12-45 (25)	47-30	3.5-8
7		0.2	2	120	17-75 (30)	48-34	2.5-6
8		0.2	4	66	25-80 (45)	48-35	3.5-8
9		0.1	2	450	25-125 (55)	48-37	2-5
10	60	1.5	4	1	9-25 (15)	54-24	10-20
11		0.4	2	30	15-60 (30)	56-34	3.5-9
12		0.2	2	120	20-110 (40)	57-37	2.5-7
13		0.2	4	60	25-110 (50)	58-37	3.5-10

The results are assumed to be realistic if the necessary thickness of the outflowing layer, given by eq. (1) do not exceed 1/3 of the vertical distance between outlet depth and depth of the effluent layer.

The results are not realistic for a stagnant recipient, due to limitation in transport capacity. Only a few of the outlet alternatives (2, 6, 7, 11, 12) may achieve the values in table 5 at common current velocities in the recipient (∿ 10 cm/s) if the length of the diffusor arrangement is 500-1000 m normal to the direction of the current.

For a stagnant situation we require that the sum of thicknesses of the outflowing and inflowing layer do not exceed the vertical distance between outlet depth and entrapment depth for the jet center, where dilution is slowest and this distance is therefore largest. When assuming

$$S_{center} = S_{mean} / 1.7$$

(normal distribution) in two dimensions and applying eq. (1) to each layer, iterating until the condition is just met, we get the results given in table 6, where transport width b is now twice the effective circumference of the diffusor arrangement.

Table 6. Lower limit of primary dilution, stagnant recipient. First values apply to strong stratification, second values to weak stratification.

Outlet depth (m)	Transport width b (m)	Dilution factor	Sum of thickness of layers (m)
40	200	5-8	9-22
	500	6-12	6-20
	1000	7-19	5-20
	2000	9-23	4-16
50	200	4-9	9-30
	500	6-18	7-30
	1000	7-23	6-26
	2000	9-30	5-22
60	200	5-12	11-40
	500	7-20	8-37
	1000	9-25	6-33
	2000	11-35	5-28

The results show that even a diffusor length of 1000 m (b = 2000 m) will not give primary dilution exceeding 10-35 in a stagnant recipient. The effluent layer may reach 20-30 m depth for all alternatives, but more seldom and in lower concentrations for increasing outlet depth and increasing diffusor length.

Hydrographic data from the Oslofjord show that large-scale water movements take place quite frequently. Fig. 8, giving depth variations of water at fixed salinity values for a representative station, shows vertical movements of 10-20 meter in periods of 14 days, and the volume of water between given salinity limits also varies greatly. A comparison with other stations indicates that the changes are largely parallell in all main parts of the basin.

The variations indicate that primary dilution do not take place in the same water layers over a longer period of time. The effluent will be distributed over a layer of 20 - 40 m thickness.

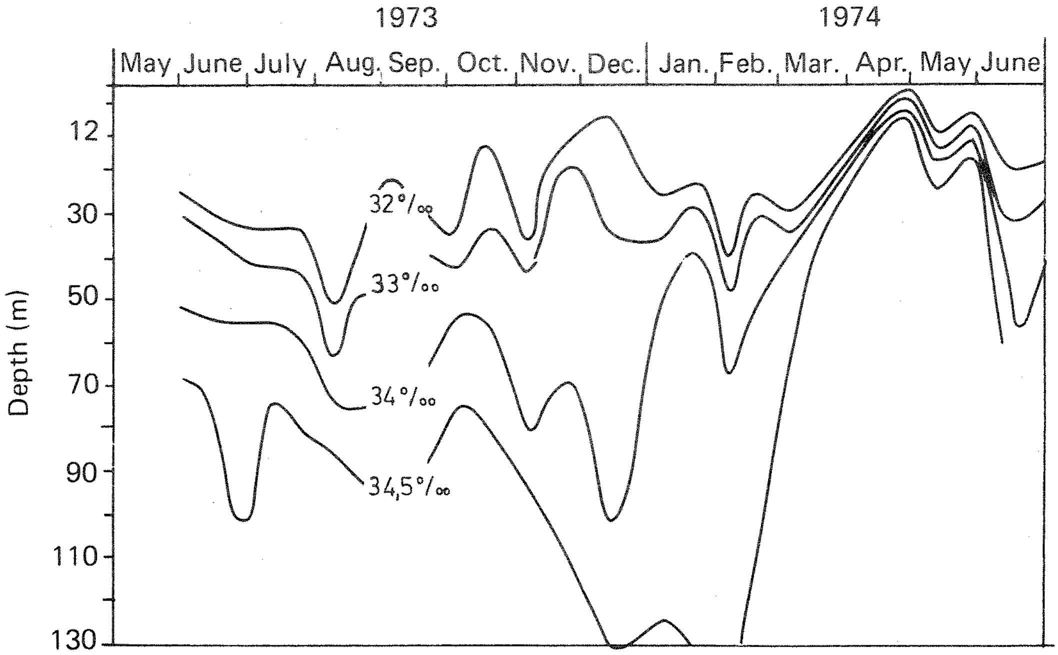

Fig. 8. Salinity as function of depth and time, representative station in the central part of the Oslofjord, 1973-1974 (Data from River and Harbour Laboratory).

The vertical movements are also connected to large-scale horizontal exchanges of water, dispersing the effluent horizontally, and giving water renewal from Skagerrak. From the volume changes shown in fig. 8 the exchange of water per 10 m depth is estimated to vary from 250 to 1500 m^3/s for the whole outer part of the Oslofjord. This agrees well with what may be deduced from a rough comparison with the surface layer estimate, based on current measurements. This gives a residence time varying from 1/2 to 3 months. The effluent will then be distributed over a layer of ∿ 30 m thickness, with a total water exchange from 750 to 5.000 m^3/s.

The overall concentration of effluent in the affected layer will vary between 1:100 and 1:700. This layer will be found mainly between 20 and 60 m depth, but closer to surface in special hydrographic situations.

For the inner-most alternatives there will be somewhat higher concentrations in the inner parts of the outer fjord.

In the region around the outfall, plumes of higher concentrations are expected. Affected water volumes for given dilution factors are calculated as for a mixed outfall under the conditions given in appendix.

The results are summarized in table 7 together with the effects on pH, oxygen and metal concentrations.

TABLE 7. EFFECTS ON WATER QUALITY IN RECIPIENT FOR SEPARATE OUTFALL OF SCRUBBER EFFLUENT AT 50 M DEPTH. ALTERNATIVE 2A, 2B.

Dilution limit S	Water masses affected by effluent in dilution ≤ S					Water quality at dilution S								
	Max. distance (km)	Horizontal area (km^2)	Volume ($10^6 m^3$)	pH 2a	pH 2b	O_2-contents (mg/l) 2a	O_2-contents (mg/l) 2b	Metal overconcentrations µg/l Be	Cd	Co	Cr	Hg	Pb	Zn
1	-	-	-	3	6	÷70	÷10	0.1000 (0.3000)	0.0300 (0.3000)	0.1500 (0.4500)	0.5500 (1.4000)	0.3500 (1.7500)	1.6000 (9.5000)	3.2000 (16.5000)
≤ 10	0-0.4	0-0.6	0-0.8	6.3-6.5	7.7	÷ 1	4-5.8	0.0100 (0.0300)	0.0030 (0.0300)	0.0150 (0.0450)	0.0550 (0.1400)	0.0350 (0.1750)	0.1600 (0.9500)	0.3200 (1.6500)
≤ 15	0-0.8	0-1.5	0-2.5	6.6-6.7	7.85	0.4-3.4	4.5-7.3	0.0067 (0.0200)	0.0020 (0.0200)	0.0100 (0.0300)	0.0367 (0.0933)	0.0233 (0.1167)	0.1067 (0.6333)	0.2133 (1.1000)
≤ 25	0-2	0-5	0-10	6.8-7.1	7.9	2.4-5.4	4.9-7.8	0.0040 (0.0120)	0.0012 (0.0120)	0.0060 (0.0180)	0.0220 (0.0560)	0.0140 (0.0700)	0.0640 (0.3800)	0.1280 (0.6600)
≤ 40	0.2-5	0.1-12	0.2-34	7.0-7.3	∿ 8.0	3.6-6.6	5.1-8.1	0.0025 (0.0075)	0.0007 (0.0075)	0.0038 (0.0113)	0.0138 (0.0350)	0.0087 (0.0438)	0.0400 (0.2375)	0.0800 (0.4125)
≤ 60	0.7-12	0.5-40	1.5-140	7.2-7.5		4.2-7.2	5.3-8.2	0.0017 (0.0050)	0.0005 (0.0050)	0.0025 (0.0075)	0.0092 (0.0233)	0.0058 (0.0292)	0.0267 (0.1583)	0.0533 (0.2750)
≤ 100	>2	1.5-150*	8 -6000*	7.4-(7.7)		4.7-7.2	5.35-8.35	0.0010 (0.0030)	0.0003 (0.0030)	0.0015 (0.0045)	0.0055 (0.0140)	0.0035 (0.0175)	0.0160 (0.0950)	0.0320 (0.1650)
≤ 250	>8	10 -150*	120-6000*	7.7-(7.9)		5.2-8.2	5.45-8.4	0.0004 (0.0012)	0.0001 (0.0012)	0.0006 (0.0018)	0.0022 (0.0056)	0.0014 (0.0070)	0.0064 (0.0380)	0.0128 (0.0660)
Background values **				8.0	8.0	5.5-8.5	5.5-8.5	0.005	0.03	0.05	0.3	0.02	0.05	3

* Total area and volume for outer Oslofjord between 20m and 60m depth.

** Metal levels for oceanic water; oxygen levels from the Oslofjord.

A depth outfall will also affect the renewal of oxygen contents of the deep layers of the inner fjord, taking place each year by inflows of heavy water over the sill at Drøbak.
The inflows draw water from 5m depth and down to perhaps 30 m depth, and take place especially in situations where water of high salinity is brought near the surface outside the sill. We assume therefore that layers influenced by scrubber effluent may dominate the inflow.

For the outer locations we expect the inflowing water to have normal overall concentrations of effluent, perhaps somewhat lower for large inflows. For a location at Emmerstad, where mixed effluent may enter directly into the inflowing layer, the concentrations may be substantially higher, from 1:200 up to 1:20. The estimates are based on volume and duration of inflows in the years 1962-1965 and 1973-1978, given by Gade (1967) and Magnusson & al. (1979) respectively.

A similar effect may be expected from a location at Skjøttelvik regarding the Dramsfjord, but we have no basis for quantifying this effect.

5. POSSIBLE EFFECTS

The potential effects of the scrubber effluent from a coal or oil fired power plant is connected with :

- acidification
- large chemical oxygen demand
- metal toxicity and accumulation in organisms
- possible discharge of polycyclic aromatic hydrocarbons (PAH) and other micropollutants

In addition to this come the impact of cooling water (thermal pollution, chlorination). Problems connected with chlorination and discharge of heated water are not treated in this report, as they are not specific for the Fläkt-Hydro process.

5.1 Acidification

The normal variation of pH in sea water of 35^{o}/oo salinity is 7.8-8.2. Intensive photosynthesis may rise pH considerably. Correspondingly, respiration results in lowering of pH. Under aerobic conditions, however, this decrease is less pronounced than the increase effected by photosynthesis. In open waters pH very rarely should sink below 7.6. It should be stressed that in all probability moderate pH decrease is more frequent in the immediate surroundings of plants and animals (on their body surface) due to respiratory activity. This would be most common in places with little water movement.

Seawater pH is mainly governed by the strong buffering capacity of the carbon dioxide system. In figures 9-10 the quantitative relations between pH and the individual constituents of the equilibrium are depicted:

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

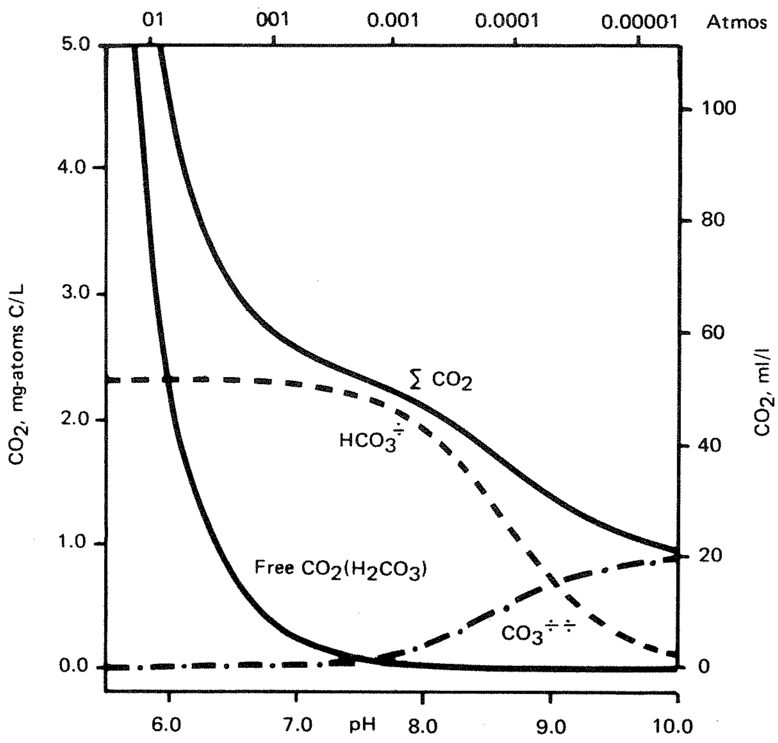

Fig. 9. Variation in total CO_2, free CO_2, $H_2CO_3^-$ og CO_3^{--} in seawater (Cl = 19 °/oo, 20°C) as function of pH and partial pressure of carbon dioxide. Modified after Sverdrup & al. (1942.

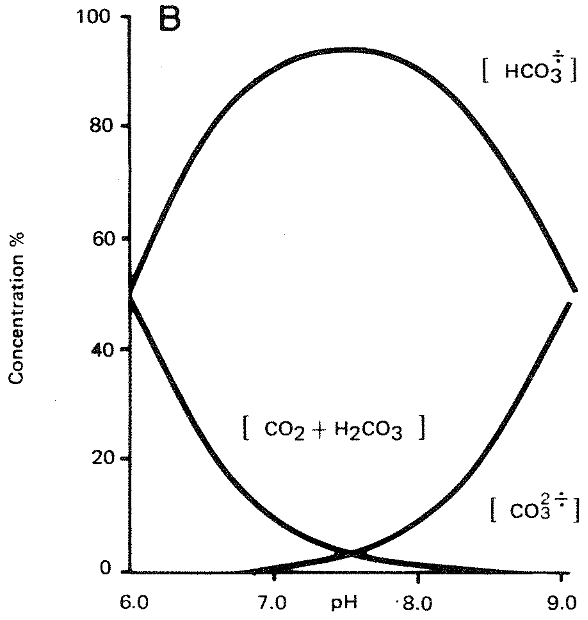

Fig 10. Relative distribution between CO_2 ($+H_2CO_3$), HCO_3^- and CO_3^{--} in seawater (Cl = 19 °/oo, 25°C) as function of pH (From Borowitzka and Larkum, 1976).

It appears from the figures that bicarbonate is predominant at normal pH values, whereas lowering of pH effects a shift in the equilibium towards no carbonate and increasing concentrations of free carbon dioxide. (Molecular carbonic acid is of little significance).

In addition to this effect, reduction of pH will influence other aspects of sea water chemistry:

- Chemical form of metals (particularly the equilibrium between different complex ions and the simple metal ions)
- Charge and other characteristics of amphoteric substances
- Solubility of various compounds.

Quantitatively the details of such reactions are largely unknown, but in principle they are important in relation to both toxicity and bioaccumulation.

The experimental evidence respecting effects of reduced pH on marine organisms is summarized in table 8. Considering the various modes of life of the large number of marine organisms, data are rather sparse. Thus there is great need of more experimental work, in particular chronic tests. Reservation must also be taken for the large difference in experimental design between the investigations referred to. Tests of pH-effects are difficult to perform because the organisms tend to modify the incipient conditions by their metabolic activity. In some cases this has resulted in pH variation (mostly rising pH) during the time of the experiment. The quoted figures refer to the start of the experiments.

TABLE 8. EFFECTS OF REDUCED PH ON MARINE ORGANISMS.

ORGANISMS	pH	EFFECTS/COMMENTS	REFERENCES
Algae			
Nitzschia sp, *Navicula* sp (Benthic diatoms)	< 6.4/< 7.0	No, respectively reduced growth. (Variable pH during tests)	Bachrach and Luccicardi, 1932
Prorocentrum micans, *Peridinium* sp. (Dinoflagellates)	< 6.0/< 7.0	No growth with start pH 6.0, possibly not with pH 7.0 (pH increased during tests)	Barker, 1935
Ectocarpus sp. (Phaeophyceae)	7.0	Reduced growth (pH increased from 7.0 to 8.0 during test).	Boalch, 1961
Halimeda tuna (Chlorophyceae)	6.0 - 9.0	Approx. constant photosynthesis at pH 9.0 to 7.0, increasing in the interval 7.0 to 6.0	Borowitzka and Larkum, 1976
Halimeda tuna	6.0 / 6.5	No, respectively low calcification	Borowitzka and Larkum, 1976
Species of Chlorophyceae, Cryptophyceae and Chrysophyceae	<7.0	"Toxic"	Droop, 1955
5 brine chlorophyceans	7.2	Good growth, two species with optimal growth at 7.2-7.5	Gibor, 1956
Ulva lactuca (sea lettuce, Chlorphyceae)	< 6.0—>6.5	Died within 5 days. Increased uptake of Yttrium 90 at low pH, no change for Strontium 90.	Hampson, 1967
Several plankton algae (diatoms, chlorophyceans, dinoflagellates)	5.3 - 10.0	Max. photosynthesis usually in the interval 7.0 - 8.0. Some species with maximum at pH ∿ 6.0, others with reduced photosynthesis below pH 7.0.	Humphrey, 1975
Species of benthic algae (Rhodophyceae, Phaeophyceae). Thallus parts and spores	3.6 - 10.0	Some difference in sensitivity. Increased cell death rate below 6.6/6.8, strongly harmful effects below 6.0. Seemingly normal development above 6.8/7.0 (2-3 days observation)	Kylin, 1927
Prymnesium parvum (Haptophyceae)	6.0 - 8.0	Similar growth rate, but seemingly increased need of trace metals at low pH	Mc Laughlin, 1958
Red, green and brown benthic algae (*Porphyra*, *Ulva*, *Petalonia*)	< 7.0 (7.3)	Reduced photosynthesis, but still measurable at pH 5.9.	Ogata, 1966
Coccolithus (Emiliana) huxleyi (Coccolithophorid)	< 6.7 (7.0)	Reduced photosynthesis, but measurable at 5.7.	Paasche, 1964
Coccolithus (Emiliana) huxleyi	< 6.0 (6.3)	No calcification	Paasche, 1964
Palmaria palmata (Rhodophyceae)	< 6.5	Reduced photosynthesis (Larger in the interval 7.5-6.5 than at pH 8.1).	Robbins, 1977
Chondrus crispus (Rhodophyceae)	6.6	Somewhat reduced growth. Optimal growth at pH 7.0/7.5/8.0, respectively with pH regulation with CO_2, HCl and H_2SO_4.	Simpson & al., 1978.
Bossiella orbigniana (Rhodophyceae)	< ∿ 7.0	Reduced calcification (at increased CO_2 concentration).	Smith and Roth, 1979
Thalassiosira pseudonana (diatom)	7.7	Increased copper toxicity (indirect effect).	Sunda and Guillard,1976.
Cricosphaera elongata (Coccolithophorid)	5.8/6.4/7.4	Respectively no cell division (but viable cells), clumping of cells, and reduced growth rate.	Swift and Taylor, 1966
Dunaliella tertiolecta (Chlorophyceae)	5.0/6.2	Optimum for photosynthesis and cell division, respectively.	Wegman and Metzner, 1971

(cont. next page)

Table 8 cont....

ORGANISMS	pH	EFFECTS/COMMENTS	REFERENCES
Bacteria			
Methane bacteria	5.0	50% reduced growth, optimum at pH 6.0 - 7.0	Hutton and Zobell, 1949
Aerobic heterotrophs	6.5	50% reduced growth. Optimum pH 7.5-7.8. Some growth at 5.5.	Zobell, 1941
Animals			
Mercenaria mercenaria and *Crassostrea virginica* (Mussels)	< 6.7 / 7.0 < 6.3 / 6.5 > 6.8	Reduced egg development. Reduced survival of larvae Normal growth of larvae	Calabrese and Davis, 1966
Ostrea edulis (Oyster)	7.0	Normal development of larvae	Gaarder, 1932
Pinctada fuscala (Japanese pearl oyster)	7.4 / 7.7	Dead, respectively reduced growth after about 20 days.	Kuwatani and Nishii, 1969
Ostrea virginica (Oyster)	< 6.5/6.8-7.0	Reduced pumping frequency and shorter opening time, respectively increased pumping frequency.	Loosanoff and Tommers, 1947
Acartia tonsa (Crustacea)	< 6.7	Increased death rate.	Ross & al., 1977
Mytilus edulis (Common mussel)	< 7.0	Increased heart beat frequency (Same effect by adding CO_2)	Schlieper, 1955

It is seen from the table that pH-lowering to 7.5 mainly appears to be tolerable, with two noteworthy exceptions. Kuwatani and Nishii (1969) found increased death rate and reduced growth in the Japanese pearl oyster at pH 7.4 and 7.7, respectively. They also found indications of shell dissolution at pH 7.6. Sunda and Guillard (1976) reported the very important effect that reduction to pH 7.7 resulted in higher copper toxicity to a marine phytoplankton species at constant total concentration. This is explained by a shift in chemical equilibrium towards more of the metal in the form of cupric ion.

There are also rather few cases of deleterious effects in the interval 7.5 - 7.0. Below this, the potentially harmful concequences from acidification become increasingly evident. It should be noted, though, that even below pH 6.0 there are examples of highly tolerant algae.

Little light has been thrown upon the mechanisms of toxicity and the difference in tolerance between species. With respect to photosynthesis, one should perhaps expect a stimulating effect of lower pH due to the increase in free carbon dioxide relative to bicarbonate. (All algae can use CO_2, whereas the utilization of HCO_3^- is disputed both with regard to effectiveness and how widespread it is). The results of Borowitzka and Larkum (1976) clearly point to such a stimulation. It has also been demonstrated by several authors that the photosynthesis saturation level of carbon dioxide is higher than its normal concentration in seawater (Paasche, 1964; Ogata and Matsui, 1965; Simpson & al., 1978). It remains unexplained, however, why then several species show reduced photosynthesis or growth already at pH values as high as about 7.0.

To study the isolated effects of changes in pH, it is necessary with other strong buffers than the bicarbonate system. It is somewhat questionable whether the widely used Tris (hydroxymethyl) aminomethane can meet this need. High Tris concentrations may be toxic to some species (Mc Lachlan, 1963). On the other hand, Ogata (1966) found a stimulating effect due to the absorbance by Tris of carbon dioxide into the growth medium.

Many marine animals and algae have calcified external skeleton, carbonate bodies covering the cells, or carbonate encrusted cell walls and surfaces. In animals all calcification of importance take place outside the cells (Istin, 1975). In algae widely differing mechanisms are utilized

(Borowitzka, 1977). Calcification may occur within the cells (coccolithophorids), in the intercellular space (many green and red algae), within the cell wall (Corallinaceae) or on the surface (Charophyceae).

The carbonate of calcified tissue mainly occurs in the two crystal forms calcite and aragonite, of which the latter is slightly more soluble and goes into solution at pH 6.3 and below (Borowitzka and Larkum, 1976). It is seen from table 5.1 that stop in algal calcification has been demonstrated below pH 6.0 - 6.3 (Paasche, 1964; Borowitzka and Larkum, 1976), and reduced rate of calcification below pH 6.5 - 7.0 (Paasche, 1964; Borowitzka and Larkum, 1976; Smith and Roth, 1979).

The pH dependance of calcification in marine animals has not attracted much interest, but Kuwatani and Nishii (1969) found indication of shell dissolution in the Japansese pearl oyster at pH as high as 7.6.

Information with respect to reduced pH and animal calcification is perhaps most badly wanted for hermatypic (reef building) corals, due to their ecological importance in warm seas. In theory, it is possible that their skeleton, although usually pure aragonite , is better protected against acidification than other carbonate skeletons and shells. The outer layer of living polyps protects against calcium leakage, and the associated zooxanthellae (unicellular plants) make the microenvironment more alcaline through their photosynthesis (Cf. Goreau, 1961; Woodhead and Weber, 1973).

Reduction in pH can reduce or increase the toxicity and availability for uptake of many substances, in particular weak acids, bases and metals. Thus, hydrogen cyanide will increase in toxicity by lowering of pH, until all is in the molecular (undissociated) state (Doudoroff & al., 1966). Other examples of such dependences are to be found in papers by Marking (1975), Mc Leay & al. (1979), Sano (1976) and others. Through consideration of theoretical models, Zirino and Yamamoto (1972) pointed out that uncomplexed metal ions might increase considerably in relative abundance if pH was reduced from 8.0 to 7.0. The theoretical relative increase of cupric ion from about 1 to 30% in this pH interval may be quite significant; ionic copper acting toxic towards some algae at concentrations close to the background level in seawater. (Steemann-Nielsen and Wium Andersen, 1970). As mentioned above, Sunda and Guillard (1976) have demonstrated this effect with even smaller pH decrease (but with higher copper concentrations).

From the above discussion one may on the one hand conclude that it is difficult to substantiate through experimental evidence the criterion from the U.S. Environmental Protection Agency, that pH in open seawater should not deviate more than ± 0.2 units from the normal variation interval (EPA, 1976). On the other hand, the results of Kuwatani and Nishii (1969) call for caution, and so does the serious lack of knowledge into the matter.

5.2 Effects of reduced oxygen concentrations

Mainly with the exception of certain bacteria and protozoa, the presence of free oxygen is a prerequisite for marine life. Within the limits of natural fluctuation in oxygen concentration, various animals and plants differ widely in their tolerance of reduced levels.

Saturation concentrations at full salinity and 0-30°C varies between about 4.4-8.0 ml/l (6-11 mg/l), decreasing with increasing temperature. In a report from FAO (1969), mostly based on experimental work, it is stated that most marine organisms cannot live for any length of time at lower concentrations than 0.8 ml/l and that in fish stress symptoms occur below 2 ml/l. 3.5-5 ml/l is regarded as satisfactory; approximately coinciding with the lower limits recommended by Perkins (1976).

Nevertheless, there are many examples of demersal fish and bottom dwelling animals having sustained low oxygen concentrations for quite long time in nature (see review by Kinne, 1972). In nature and for short periods, animals adapted to pelagic life may tolerate levels which result in stress symptoms under experimental conditions.

Oxygen tolerance limits are difficult to establish, or even to define exactly. The reason is that the tolerance and preference of any one species may vary with life stage and also depend on other variables, particularly temperature and salinity, but also the presence of toxic substances, competition or other stress factors (for instance in experimental environments).

Excepting extreme cases of tolerance and/or reproductive adaptability, present knowledge does not allow for statements with regard to oxygen requirement of various species in exact terms. Thus one has to base evaluation of possible risks of deleterious effects on the general assumption that oxygen depression below 4-5 ml/l may result in poorer communities and reduction of biological resources.

5.3 Metals - toxicity and accumulation

The following considerations will be based on

- Available information about background levels of metals in oceanic water.
- Assumed overconcentrations in the alternative effluents compared to background levels
- Information on lower limits of toxicity for the metals in question.
- Factors affecting toxicity and bioconcentration (uptake and excretion).
- Recommended upper limits for human intake of metals in food (Hg, Cd, Pb).

5.3.1 Background levels of metals in seawater

With reference to Riley and Chester (1971), Brewer (1975), Bryan (1976) and Sugawara (1978), approximate mean background concentrations (µg/l) of metals (particulate + dissolved) in oceanic waters can be given as below. Some Oslofjord data (Bjerkeng and Knutzen, 1978) are included for comparison:

	As	Be	Cd	Co	Cr	Cu	Hg	Mn	Ni	Pb	Se	V	Zn
Oceanic water	3	0.005	0.03	0.05	0.3	1	0.02	1	1	0.05	0.2	2	3
Oslofjord	-	-	0.1	-	<0.5	1	0.1	-	1	∿0.5	-	-	3

Seawater analysis for metals is difficult, and comparable and reliable data still are relatively scarce. Often the above figures should be replaced by intervals close to one order of magnitude. In other cases there are just a few records. Differing methods of sampling and analysis are the main causes of doubtful comparability. Further, lack of separation between the particulate and the dissolved state is common. Thus the data is of somewhat restricted relevance for scientific as well as management purposes.

5.3.2 Overconcentration of metals in effluent

As stated above and in chapter 3, information about background levels and effluent composition both are very approximate. To evaluate the possible risks of toxic effects and for accumulation in edible and other organisms, it may nevertheless be useful to state the effluent concentrations in terms of background or ambient levels. This is done in table 9 for the two alternativ effluents (Cf ch. 2 and table 1).

Table 9. Approximate number of times increase in metal concentrations of alternative effluents from a 1200 MWe coal fired power plant (effluent concentrations divided by assumed levels in oceanic and Oslofjord waters). The interval limits correspond to estimates of normal, respectively high metal content in coal.

	Alt.	As	Be	Cd	Co	Cr	Cu	Hg	Mn	Ni	Pb	Se	V	Zn
Oceanic water	Alt. 1	<0.1	4-<12	<0.3-2	0.6-2	0.4-1	<0.1	4-18	<1	∿0.1	6-40	<0.3	<0.3	0.2-1
	Alt. 2	<0.5	20-<60	0.7-9	3-10	2-5	<0.5	∿18-90	0.7-4	<1	32-190	<1.5	<1.5	1-5
Oslofj. water	Alt. 1	-	-	<0.5	-	<o.5	<0.1	<1÷4	-	∿0.1	<1÷4	-	-	0.2÷1
	Alt. 2	-	-	0.2-3	-	1-3	<0.5	4-18	-	< 1	3-19	-	-	1-5

It appears that the relative overconcentrations are most striking for beryllium, mercury, and particularly lead; to a smaller degree for cadmium, cobalt, chromium and zinc. Discharge of the rest of the metals appears to be insignificant in comparison with the recipient content of these elements. Some reserve must be taken, however, for the state and chemical form of the main component of each metal in the effluent as well as in the receiving waters.

Overconcentrations of Be, Cd, Co, Cr, Hg, Pb and Zn at different dilutions are shown in tables 4 and 7.

5.3.3 Toxicity data and water quality criteria

Table 10 gives a summary of information on the toxic levels recorded for the metals in question. Here too, data are fragmentary for most elements, in the sense that relatively small numbers of organisms have been tested. Often the experimental procedures have been inadequate with respect to mode and duration of exposure.

TABLE 10. PRELIMINARY QUALITY CRITERIA FOR SEAWATER AND EXAMPLES OF METAL CONCENTRATIONS (µg/l) WITH HARMFUL EFFECTS. DUBIOUS OR EXTREME VALUES IN PARENTHESIS.

As	Be	Cd	Co	Cr	Cu	Hg	Mn	Ni	Pb	Se	Zn	REFERENCES	COMMENTS
	1100$^{a)}$	5$^{b)}$	100$^{a)}$	100$^{a)}$	X$^{c)}$	0.1	100$^{d)}$	Y$^{c)}$	–	Y$^{c)}$	Y$^{a,c)}$	EPA (1976)	Quality criteria
		60 – <10^3		10^3– >10^4	20 – >100	∿ 1		625 – 10^3	100– >10^3			Ref. in EPA (1976)	
7.5 x 10^3		300 – 10^3	>10^5	100– 10^3	50 – 10^3	(3) 10– 100	16·10^3 – >10^5	10^3– >10^5	2.5·10^3 – 10^5	>10^5	(100)– 10^4	Ref. in Bryan (1976)	Increased lethality in tests for acute toxicity
		300			(1) 10– 100$^{e)}$	(2) 5– 100			∿ 100		40 – 10^3	Ref. in Bryan (1976)	Sublethal effects (growth, reproduction metabolism, etc.)
		100 – 3·$10^{3}$$^{f)}$	g)	(<12)– $10^3$$^{h)}$	(1) 10– 10^3	(0.5)$^{i)}$ –100		j)	10^3		50 – 10^3	Ref. in Beijer & al. (1977)	
				(12) 100 –200								Mearns and Young, 1977	Cr (VI)
100	0.1$^{k)}$	15	10$^{l)}$	(0.1)$^{m)}$	(1)	5	16·10^3	(0.2)$^{n)}$	150	(10)$^{o)}$	50	Davey and Phelps, 1977	
		(200) 10^3–10^4				(1) 10– 10^3						Ref. in Taylor (1977 a,b)	Lethal effects
		(1) 10– 10^3				(0.1)$^{p)}$ 1–100						Ref. in Taylor (1977 a,b)	Sublethal effects
						1–5						Grice and Menzel, 1978	Effects on bacteria fytoplankton and zooplankton

a) Adapted from freshwater.

b) Partly due to risk of accumulation in mussels.

c) X, Y: Respectively 0.1 and 0.01 x 96 hours LC_{50} for least tolerant species in the recipient area.

d) Due to risk of accumulation in mussels.

e) 1 µg/l may be toxic to some species of fytoplankton.

f) Down to 0.02 µg/l in tests with some freshwater organisms.

g) Down to 5 µg/l in tests with freshwater organisms.

h) Cr (VI)

i) Reduced photosynthesis in fytoplankton.

j) Down to 30 µg/l in tests with freshwater organisms.

k) Depressed enzymatic activity

l) 72 hours LC_{50}, copepods.

m) Reduced yield in algal cultures.

n) Depressed growth in algae.

o) 96 hours LC_{50}, copepods.

p) Down to 0.04 µg/l in test with freshwater daphnia.

Most of the results quoted in table 10 are based on registrations of increased lethality over a short time period (LC_{50} type of tests for 48 or 96 hours). Experiments aiming at the demonstration of possible chronic, sublethal effects, are still quite few. Even such effects may in the long run cause eradication of whole populations. To use LC_{50} values as basis for water quality criteria is somewhat precarious, even when it is applied a safety factor of 10 or 1%. The ratio between LC_{50} and safe concentrations will differ with substance as well as species and factors in the natural environment. The trend of the latest years has been a steady lowering of concentrations which can be regarded as safe without reserve.

On the other hand, some of the concentrations reported to have had poisonous effects must be judged as extremes, in the sense that they are close to or even below naturally occurring concentrations. The typical example is copper, but levels naturally occuring in coastal waters overlap with toxicity thresholds also for mercury (and zinc and cadmium, if one include tests performed with freshwater organisms).

It is seen from the table that Davey and Phelps (1977) claim to have found that nickel acted poisonous towards algae in a concentration which is about 1/5 of the natural background. As the test procedure has not been described to any detail, this result must for the moment be taken with strong reservations. This also applies to the chromium toxicity claimed by the same authors.

Overlapping between toxic and natural concentrations points to the fundamental importance of the chemical form of metals in the aquatic environment, and also to the possible role of humus and other chelators as detoxifying agents. These are difficult subjects, which hitherto have not attracted sufficient attention.

5.3.4 Factors affecting toxicity and bioconcentration

Both toxic effects and accumulation in plants and animals will depend on the way in which the organisms are exposed. In the main, exposure is mediated through water or food.

Metal in water may be present in one or (usually) several of the following states:

- Incorporated in particles (inorganic, organic, live or dead).
- Adsorbed to, or precipitated upon surfaces of particles
- Colloidal
- Dissolved, and then either as simple ions or complexed with organic or inorganic chelators.

Within the group of dissolved species, the relative abundance of theoretically possible components often are insufficiently known. As a rule, simple metal ions are regarded as most toxic. The obvious reason for this is that uncomplexed ions are the most diffusible and adsorbable form. (Methyl mercury is an important exception).

In addition to the metal's state, thresholds of toxic effects depend on many environmental factors (temperature, salinity, oxygen saturation, presence of other toxicants etc.). Life stage, physiological conditions of the exposed specimen and length of exposure may be decisive. Various species differ widely in tolerance.

Although the distribution between different states are not governed by simple equilibrium reactions, it is worthy of notice that most metals may be dissolved in much higher concentrations than their ambient levels in seawater (Bryan, 1976).

Small organisms, with a high surface to volume ratio, are most inclined to take up metals in the dissolved state. After adsorption to the surface and diffusion through membranes, the metals are often bound to very stable complexes within the plant or animal. Benthic algae are perhaps the best known example of practically not regulated and irreversible uptake of metals. Animals, particularly the larger species, have more or less effective mechanisms for excretion and regulation of uptake.

Mostly among larger animals, uptake through the food often play a dominant role, but diffusion through the gills and other surface membranes may still be important, especially in mussels, but also in fish. Often only a minor fraction of the metal ingested is absorbed in the intestine.

Biomagnification of metals (i.e. increasing concentrations with higher levels in food chains) appears to be of limited occurence, and does not take place to the same degree as for chlorinated hydrocarbons and other persistant organic compounds. Usually it requires a combination of large intake through the food and inadequate excretion mechanisms. Length of life is also important. According to Bryan (1976) excretion ability seems to be least for metals with no known physiological function (mercury, lead, cadmium).

5.3.5 Recommended maximum human intake of metals (Hg, Cd, Pb)

With exception of the Minamata disease there are few indications that metal contaminated seafood has caused damage to man (Bryan, 1976).

From the considerations of expert committees within FAO/WHO (1972) and WHO (1973) it is evident that non essential metals are regarded as potentially most dangerous with respect to occurrence in food. Preliminary recommendations for maximum weekly intake per person (60 kg) of mercury, cadmium and lead are as follows:

Mercury:	0.3 mg (of which maximum 0.2 mg methyl mercury)
Cadmium:	0.4-0.5 mg
Lead :	3 mg

In fish 90% of the mercury content usually occur in the methylated form (Bryan, 1976). Bryce-Smith and Waldron (1974) maintain that the maximum lead intake by children should be set considerably lower per kg body weight than for adults, owing to higher absorption of lead in the intestine of children and their higher sensivity.

All three metals may damage kidney tissue and have the ability to pass from the mother into the fetus. Small doses of mercury and lead affect the central nervous system.

Transformation of the above recommendations into acceptable levels in seafood is very complicated, and requires detailed analysis of other sources (water and food consumption, air pollution). Particularly exposed and/or vulnerable population groups must also be taken into account. At present, some countries have put into force an upper limit of 0.5 or 1 mg mercury per kg wet weight.

5.4 Polycyclic aromatic hydrocarbons (PAH)

PAH occur naturally as the result of forest fire, volcanic activity, other geochemical processes and, probably, biosynthesis. In many areas, however, human activities are the predominant sources. PAH and related substances attract attention because metabolites of some of the compounds have been demonstrated to be cancerogenic. Benzo(a)pyrene (B(a)P) perhaps is the best known example. With regard to physical, chemical and physiological characteristics, most important sources, occurrence in the aquatic environment etc., it is referred to reviews by Andelman and Suess (1970), Andelman and Snodgrass (1974), Knutzen (1976), Landner (1977) and Neff (1979).

For the moment, one does not know much about the effects of PAH in the marine environment, beyond the fact that high concentrations have been recorded in the vicinity of large point sources. In Saudafjorden, W. Norway very high levels were found in mussels and sediments (Bjørseth & al., 1979) together with a poor fauna, but there are other examples of relatively rich communities in heavily polluted environments (Palmork, 1974). Table 11 gives some examples of harmful effects on marine organisms, mostly observed in the laboratory, but also in the field. For other examples it is referred to the review by Neff (1979).

Table 11. Observation of harmful effects of PAH on marine organisms

Substances	Conc. (ppb)	Organisms	Effect	References
Various PAH compounds	10-300	Rhodophyceans	Reduced or stimulated growth, cell size	Boney and Corner (1962) Boney (1974)
Dimethylbenzoanthracene (DMBA)	5000	Sea urchin	Egg development	de Angelis and Giordano (1974)
DMBA B(a)P Benzanthrone	10 5 0.1	Dinoflagellate	Growth (Cell and nucleus division)	Ishio & al. (1977)
B(a)P	20-80	Bacteria	Reduced growth	Calder and Lader (1976)
Benzanthrone	Field observation	*Porphyra* sp. (Rhodophyceae)	Cancer	Ishio & al. (1971)
Creosote	Field observation	Moss animal (Bryozoa)	Cancer	Powell & al. (1970)

PAH is biodegradable or can be excreted in unchanged form in a relatively rapid way. 90-99% reduction has been observed within 5-100 days in contaminated animals after transfer to pure seawater, somewhat depending on species, compound, and way and time of exposure (Anderson and Neff, 1975; Lee & al, 1972a; Cox & al., 1975; Lee & al., 1972b, Neff & al., 1976, Lee, 1975, Neff, 1979).

Generally, the excretion seems to be faster in species with the mixed function oxidase enzym system (MFO). Mussels and snails are the most important groups in which the MFO system appears to be absent or very weekly developed (Neff, 1979).

On the above grounds it may be concluded that the risk of biomagnification is considerably smaller than with persistant organics like PCB, etc. High PAH levels in organisms will usually not occur unless there is relatively large and continuous contamination of the environment.

6. DISCUSSION

pH

According to EPA (1976) maximum allowable deviation from normal pH variation in open marine waters should be 0.2 units. Depending to a certain extent on the recipient conditions this would mean a minimum acceptable pH of 7.6 - 7.8. In table 12 it is shown for the four innermost sites of the Oslofjord area what volumes and areas that risk these reductions by the four effluent alternatives (cf. figs 1-4).

Table 12. Approximate volumes (10^6 m^3) and areas (10^6 m^2) potentially affected with pH $\leq$ 7.6 or $\leq$ 7.8 due to alternative effluents from 1200 MWe coal fired power plant in the Oslofjord area (Assumed background pH:8.0)

Effluent alternative	$\leq$ 7.6		$\leq$ 7.8	
	Volume	Area	Volume	Area
1 a	∿ 350	∿ 150	∿ 3000	∿ 300
1 b	∿ 10	< 10	30	20
2 a	∿6000*	*	∿ 6000*	*
2 b	< 1	< 1	< 2.5	< 1.5

* Subject to great variation depending on hydrographic conditions and water renewal in intermediate depth, cf. table 7.

It is evident from the table that untreated effluent may represent a hazard to large water masses in the recipient. Treatment with lime and/or air reduces the risk considerably.

Discharge of untreated effluent to the surface layer (alt. 1 a) is judged to be the most hazardous alternative. The affected volumes would be about the same if untreated waste water is discharged at about 50 m (alt. 2 a), but the consequences would be more serious for the upper ∿ 10 m for the following reasons:

- Lower pH over the whole surface area between Emmerstad - Skøttelvik and Vardåsen - Slagentangen (fig. 5).
- Potential damage to the zone of primary production
- Habitats of shallow water benthic organisms risk continuous exposure to unfavourable pH conditions, possibly resulting in the decimation of less tolerant species, particularly among the sessile life forms. Mobile species might tend to prefer somewhat deeper water than before the change in ambient pH. Any harmful effect on benthic communities would also affect the recruitment of those fish species which as younglings depend on healthy shore and shallow water communities. (It may also be speculated that littoral benthos in general would be more sensitive than deep water species. Being adapted to less water movement, the latter group should experience microclimatic pH reduction due to respiration more often than surf-zone organisms). With the exception of fishermen, the prime interest of most people is connected with the amenities and recreational quality of the surface layer.

Discharge of untreated waste to deep water (alt. 2a) should only affect a minor part of the total subsurface fjord environment. The communities exposed would presumably be less sensitive because of relatively easier avoidance of stress and the fact that the main parts of pelagic populations would be unexposed at any one time.

Aerated mixed effluent to the surface layer (alt. 1 a) might be locally important, but compared to the untreated effluent the area and volume affected would be less than 10 and 1%, respectively. Moreover, with the relatively low critical dilution of 10 (cf. table 4), an appropriate discharge arrangement would make it possible to avoid stress on shore communities and benthic organisms.

Separate deep water discharge of neutralised and aerated scrubber effluent (alt. 2 b) should not cause any damage due to low pH outside the primary dilution zone. Effects would hardly be detectable unless bottom fauna is directly exposed to low waste dilution.

With regard to acidification, deep water outfalls should in general be preferable to surface layer discharge.

A weak point in the above reasoning is that there are few facts (proven harmful effects) which support the criterion of maximum 0.2 units reduction. On the other hand, there are several indications that a safety margin is warranted:

- General lack of long-term or life cycle studies
- Results of Kuwatani and Nishii (1969) and Sunda and Guillard (1972)
- Probable relative increase of simple ions of some metals (Zirino and Yamamoto (1972)
- Enhancement of the effects of some pollutants (possible reduction in the effects of others)

The safety margin is particularly justified for surface waters of inlets, bays and fjords to which there are attached large recreational and/or economic interests.

Oxygen

Even without aeration, surface layer discharge of the mixed scrubber and cooling water effluent (alt. 1 a) will not usually result in potentially harmful depression of the oxygen concentration beyond the immediate vicinity of the outfall (cf. table 4 and ch. 5.2). The oxygen level will be satisfactory above dilution 10, and the affected water mass restricted to about $30x10^6$ m^3. A small volume of about 2×10^6 m^3 near the outlet will be directly toxic (anaerobic).

In contrast, a separate discharge to deeper waters of the untreated scrubber waste stream (alt. 2 a) may have quite serious consequences, somewhat depending on recipient conditions. In land-locked waters the subsurface layers often experience long periods with marked oxygen deficit or even anoxic conditions. Under such circumstances the aquatic environment may be drastically changed if exposed to an additional stress. The same applies to recipients with reduced oxygen concentrations due to high loading of degradable organics.

The Oslofjord area is a good example in both these respects. Bottom waters inside the sill in the Drøbak Sound (fig. 5) often have critically low oxygen content (sometimes complete exhaustion in some parts). Large load of organic matter and plant nutrients from domestic sewage, has resulted in a negative trend with regard to oxygen content of the bottom water during the last decades (Magnusson, 1979) and deterioration of the soft bottom fauna (Beyer, 1968). Old bottom water is periodically renewed by sill overflow of water from the 5-30 m layer of the middle and outer parts of the fjord.

The calculations in ch. 4 show that the diluted waste water from a discharge at 50 m depth periodically may rise to 20-30 m; and that even with maximum attainable dilution the renewal water will have considerably reduced oxygen content compared with the ambient level today. For the innermost localisation at Emmerstad (fig. 5) the mean reduction may be about 1-2 mg/1 (max. ∿ 4.0 mg/1); for the alternative Vardåsen (fig. 5) up to 0.8 mg/1 at the worst. Under such circumstances the deep water discharge of untreated scrubber effluent obviously would be unacceptable or highly questionable.

Lime treatment and aeration (alt. 2 b) reduce the chemical oxygen demand of the scrubber waste water with about 80% (table 2), at the same time ensuring about 80% oxygen saturation in the effluent (ch. 3). Consequently, the effluent will be far less objectionable with respect to oxygen conditions. Even so, the remaining chemical oxygen demand may still represent a hazard to sensitive recipients.

Consideration of potential oxygen problems alone leads to the conclusion that surface water discharge would be preferable, other conditions being equal.

Metals

Evaluation of the possible impact of effluent metals on marine life, recreational and economic interests, has to be mainly speculative for the time being. Firstly, the metal content of coal shows considerable variations (table 1). Secondly, the only information on metal concentration in the waste water is from introductory pilot plant studies conducted by Norsk Viftebarikk A/S (unpubl. report, 1979). Thirdly, the state of the metals in the effluent are not exactly known. Probably the main part is incorporated in, or at least associated with small ash particles.

Lastly, our knowledge of metal effects on marine organisms (bioconcentration, bioaccumulation, toxicity) is far from satisfactory.

From the tables 4, 7 and 9 it appears that considerable overconentration in the effluent probably are restricted to beryllium, mercury and lead. Owing to its high toxicity, cadmium might be included in this group of primary concern. The other metals are not taken into account because there is little evidence that they are toxic even in concentrations found in the undiluted waste water (cf. table 10 and ch. 5.3.3. The reasons for excluding chromium and nickel are also given there). Finally, it should be stressed that the highest metal levels mentioned for the effluent refer to the less probable estimate of high metal content in coal. Thus, for the majority of the metals, arguments against discharge are restricted to general concern for protection of the marine environment.

Data on the effect of beryllium on marine organisms are very scarce, and does not allow evaluation of the possible consequences in the recipient. So far, this metal has not been regarded as particularly dangerous in the aquatic environment, but the result of Davey and Phelps (1977, cf. table 10) ought to stimulate further research.

With the most probable cadmium content in coal (table 1), the pollutional load from this metal should not give rise to particular concern. The matter is possibly somewhat different if the cadmium content should be 10 times higher. Even if cadmium should turn out to be poisonous in concentrations close to the background level, the volume of toxic water would still be rather small (tables 4 and 7).

The hygienic risk from higher levels of cadmium in edible organisms would be of more interest. Some commom seafood species, particularly among crustaceans and molluscs, may have a high natural content of cadmium (Bjerkeng and Knutzen, 1978, Bokn & al., 1978). Together with the previously mentioned low limit of acceptable weekly intake, this indicate that care should be taken with any discharge of cadmium into the sea. All uncertainties considered, it is rather improbable, though, that the cadmium discharge should cause any serious harm.

Neither should the effluent content of lead give rise to concern. This metal apparently has relatively low acute toxicity towards marine organisms, and the effluent concentrations would be 1-2 orders of magnitude below the expected threshold of toxic effects. (Reserve must be taken for the general lack of results from long-term bioassays). Compared to other sources, such as polluted precipitation and urban runoff, the contribution of lead from a power plant scrubber usually would be of minor importance. This is the case in the Oslofjord area, in which other lead sources dominate at present (Bjerkeng and Knutzen, 1978). The few results of water analysis from the fjord indicate a considerable increase of lead concentration compared to the background in oceanic water (about 5-10 times), but this has not resulted in a corresponding increase of the lead level in organisms (Bjerkeng and Knutzen, op.sit.)

Even when assuming high lead levels in coal, the relative increase of lead concentrations would be moderate and of limited extent in the Oslofjord. In unpolluted waters the matter would be different; with high percentage increase in large water masses (tables 4, 7). The possible consequences of this difference in basic conditions in the two types of recipient waters are difficult to evaluate. If the toxic and accumulative activity of lead forms in scrubber effluent is the same as for lead already present in recipient waters, then the discharge should be more doubtful in waters of oceanic quality than in coastal waters.

Possibly, mercury would be the most critical element in the scrubber waste. Although the concentrations are quite low, there is a risk that the effluents are acutely toxic (table 10). The main concern, though, is with respect to accumulation in edible organisms. With the high estimate of mercury content in coal even a dilution of about 100 (based on scrubber water alone) could result in a near doubling of mercury concentration compared to the normal level in oceanic water (tables 4, 7). Thus it is a possibility that considerable water masses would be affected. This is judged to be more serious for mercury than for lead, particularly due to the considerably lower limit of acceptable levels of mercury intake (FAO/WHO, 1972). As in the case with cadmium, any discharge of mercury to the sea is in principle unwanted.

Bearing in mind the inadequacy of present knowledge, it may be concluded as follows with respect to metals:

- Acute toxic effects are not probable beyond the immediate surroundings of the outfall. (The possibility of synergistic effects represent a uncertainty).
- With "normal" metal levels in coal there is a moderate risk of unwanted accumulation of cadmium, mercury and lead in organisms close to the discharge site. With high metal concentrations the affected area and volume might interfere with other uses of the receiving waters.
- Generally, discharge to deep, open waters would be preferable, thus reducing or avoiding the contamination of the productive zone and benthic organisms of the shore and shallow waters.

PAH

As various estimates of PAH quantities in flue gas differ with two orders of magnitude or more (ch. 3), and there is no data for transmission to the water phase, the evaluation of possible consequences must be speculative.

If it is correct that modern boilers have modest dischare of PAH (Alfheim & al., 1978), e.g. with less than 0.1 mg PAH and 0.01 mg benzo(a)pyrene per kg coal, there would be a relatively small risk of either toxic effects or accumulation in biota.

At the other extreme, discharge of 10 and 1 mg/kg of PAH and B(a)P, respectively, should be deemed serious for recipient conditions. Concentrations in the 36 m^3/s effluent would then be of the order ∿ 30 µg PAH/l and ∿3 µg B(a)P/l.
level in scrubber effluents from Norwegian aluminium smelters with Søderberg electrodes (Knutzen, 1978), but the amounts of waste water from a 1200 MWe power plant are considerably larger. Particularly with discharge into the surface layer one must expect high levels of PAH in benthic organisms, and the possibility of interference with other interests in the recipient area (harvesting of mussels, aquaculture. The effluent

concentration of $\sim$ 3 µg B(a)P/l is close to the range in which toxic effects have been observed (table 11). But the risk of harm to marine communities is judged to be moderate or small.

From the above discussion it is obvious that conclusions in one or the other directions are heavily dependent on data from analysis of representative effluents. The need for such information is most acute in cases where large amounts of waste water is discharged to sensitive recipients.

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APPENDIX

THEORY AND PARAMETERS FOR CALCULATION OF PLUME SPREADING

1. THEORY

For a plume following a current in the x-direction, spreading by diffusion in y-(horizontal) and z-(vertical) direction, the concentration of a substance is described by the equation.

$$c(x,y,z) = \frac{Q_o \cdot C_o}{2\pi\sigma_y \cdot \sigma_z \cdot U_x} \cdot \exp\left(-\frac{y^2}{2\sigma_y^2} - \frac{z^2}{2\sigma_z^2}\right) \qquad \text{A 1}$$

where spreading follows the equations

$$\frac{d\sigma_y^2}{dx} = \frac{2\,k_y}{U_x}, \quad k_y = \varepsilon\, B^{4/3} \qquad \text{A 2}$$

$$\frac{d\sigma_z^2}{dx} = \frac{2\,k_z}{U_x} \qquad \text{A 3}$$

σ_y, σ_z is the standard deviation of a normal distribution in y,z-direction. The other parameters are:

Q_o = volume flux of effluent
C_o = concentration in effluent
U_x = current velocity
k_y, k_z = diffusion coefficient

For a physical distribution, total thickness h and width B of the plume are expressed as:

$$h = f_z \cdot \sigma_z, \quad B = f_y \cdot \sigma_y$$

where $f_z \simeq 4$ and $f_y \simeq 4$ for two-sided plume, $\simeq 2$ for one-sided plume along shore-line.

Equations (4), (5) may then be rewritten as differential equations for diffusive spreading:

$$\left.\frac{dB}{dx}\right._{\text{diff}} = f_y^{\,2} \cdot \frac{\varepsilon \cdot B^{1/3}}{U_x} \tag{A4}$$

$$\left.\frac{dh}{dx}\right._{\text{diff}} = f_z^{\,2} \cdot \frac{k_z}{U_x \cdot h} \tag{A5}$$

Expressions for purely gravitational spreading are found by using equation (1) in chapter 4, the condition $dB/dt = u \cdot f_y/2$ and the continuity relation for mass, $h \cdot B$ = constant. The resulting equations are:

$$\left.\frac{dB}{dx}\right._{\text{grav}} = \frac{f_y \cdot \alpha\, Nh}{2\, U_x}\,, \qquad \alpha = \frac{1}{\gamma 2} \simeq 0.128 \tag{A6}$$

$$\left.\frac{dh}{dx}\right._{\text{grav}} = -\frac{h}{B} \left.\frac{dB}{dx}\right._{\text{grav}} \tag{A7}$$

The mechanisms of diffusion and gravitational spreading are now combined into one set of equations. Horizontal spreading will be a non-linear combination of the processes in eq. A4 and A6 If we assume that spreading energy is combined linearly, it is natural to sqare-sum velocities, i.e:

$$\frac{dB1}{dx} = \left(\left.\frac{dB_1}{dx}\right._{\text{diff}}^{2} + \left.\frac{dB}{dx}\right._{\text{grav}}^{2} \right)^{\frac{1}{2}} \tag{A8}$$

for the increase of total width B_1.

Inside this width there is a central part where gravitation dominates, and where concentrations are constant horizontally across the plume. The width of the central part B_2 equals B_1 initially, decreasing to zero as horizontal diffusion takes over. If the central part is "eroded" with the same velocity as the plume spreads outwards from the gravitation front, we have

$$\frac{dB_2}{dx} = \begin{cases} \frac{dB}{dx}_{grav} - \left(\frac{dB_1}{dx} - \frac{dB}{dx}_{grav} \right) & \text{if } B_2 > 0 \\ 0 & \text{hvis } B_2 = 0 \end{cases} \qquad \text{(A 9)}$$

The vertical mixing according to eq.(A5) and the flattening according to eq.(A7) are combined linearly, with a mean B-value applied in eq.(A7):

$$\frac{dh}{dx} = \frac{dh}{dx}_{diff} - \frac{2h}{(B_1 + B_2)} \frac{dB}{dx}_{grav} \qquad \text{(A10)}$$

Diffusion is expected to smooth out the differences in density gradients inside and outside the plume, this is expressed by a decreasing factor α:

$$\alpha(x) = \alpha_o \cdot ({}^{So}/S_{min})^{\frac{1}{2}} \quad , \quad \alpha_o = 0.128 \qquad \text{(A11)}$$

where

S_o = primary dilution

S_{min} (x) = center dilution of plume at distance x from the outlet area.

S_{min} is calculated as

$$S_{min} = \frac{0.6 \cdot h \cdot (B_2 + (B_1 - B_2) \cdot 0.6) \cdot U_x}{Q_o} \qquad \text{(A12)}$$

with 0.6 being the ratio between lowest and mean dilution in a one-dimensional normal distribution, or as

$$S_{min} = 0.75 \cdot S. \qquad \text{(A13)}$$

given by NIVA-Samfunnsteknikk (1978) for the outfall area. The lowest of the two values are used.

Eq. (A8) through (A10) may now be solved numerically as 1.order differential equations in x, with expressions (A4 - A7), (A11 - A13) inserted.

To accumulate water volumes and horizontal area affected by dilution below given limits, the normal distributions are approximated by cosine functions with Π corresponding to 4σ, going to zero at $\pm 2\sigma$ and thus giving a more realistic description of the fringes of the plume.

To account for the over-all concentrations of effluent in the recipient a corrected dilution factor

$$\frac{1}{S'} = \frac{1}{S} - \frac{1}{S_{background}}$$

is applied in the equations above.

2. PARAMETERS FOR CALCULATION OF PLUME SPREADING. MIXED OUTFALL.

Standard set of parameter values:

$f_y = 4$

$k_z = 10^{-5}\ m^2/s$

$N = 0.085\ s^{-1}$

$\alpha_o = 0.128$

$\varepsilon = 4.55\cdot 10^{-5}\ m^{2/3}\ s^{-1}$

$Q_o = 36\ m^3/s$

$S_o = 3$

$S_{lim} = 30$

$U_{res} = 0.1\ {}^m/s$

$h_o = 5\ m$

Calculation no.	Perturbation of parameter set	Calculation no.	Perturbation of parameter set
1	none	8	$h_o = 2$ m
2	$U_{res} = 0.03\ {}^m/s$	9	$h_o = 10$ m
3	$U_{res} = 0.3\ {}^m/s$	10	$f_y = 2$ (along shore)
4	$S_o = 2$	11	$k_z = 0.5\cdot 10^{-5}\ m^2/s$
5	$S_o = 5$	12	$k_z = 5.10^{-5}\ m^2/s$
6	$S_{lim} = 50$	13	$N = 0,\ \alpha = 0$ (only diffusion)
7	$S_{lim} = 1000$	14	$\varepsilon = 0$ (Only gravitation in horizontal direction)

The standard value of k_z may be somewhat low, but the order of magnitude is in accordance with estimates by Anderson & al., (1974), Gade (1970) and Kullenberg (1969). The stability N is the mean value for the observed density profiles. The value of ε is taken from Rye and Thendrup (1978) and is in accordance with Kullenberg (1972).

3. CALCULATION OF PARAMETER VALUES FOR PLUME SPREADING. SEPARATE OUTFALL OF SCRUBBER EFFLUENT.

Calculation nr	Stability N (s^{-1})	Vertical diffusion coeffisient kz (m^3/s)	Current velocity (m/s)	Primary dilution S_o	Initial thickness of plume h_o (n)	Background dilution S max
1	0.0031	$6 \cdot 10^{-4}$	0.02	20	15	100
2	"	"	"	"	"	700
3	"	"	0.10	50	6	100
4	"	"	"	"	"	700
5	0.016	$9 \cdot 10^{-5}$	0.02	12	5	100
6	"	"	"	"	"	700
7	"	"	0.1	20	3	100
8	"	"	"	"	"	700
9	0.044	$2 \cdot 10^{-5}$	0.02	8	2	100
10	"	"	"	"	"	700
11	"	"	0.1	12	1	100
12	"	"	"	"	"	700

Fixed parameters:
$f_y = 4$
$\alpha_o = 0.128$
$\varepsilon = 4.55 \cdot 10^{-5} \ m^{2/3}/s$
$Q_o = 7.2 \ m^3/s.$