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Guðjón Atli Auðunsson

Summary and evaluation of environmental impact studies on the recipient of sewage from the STP at Ánanaust, Reykjavík

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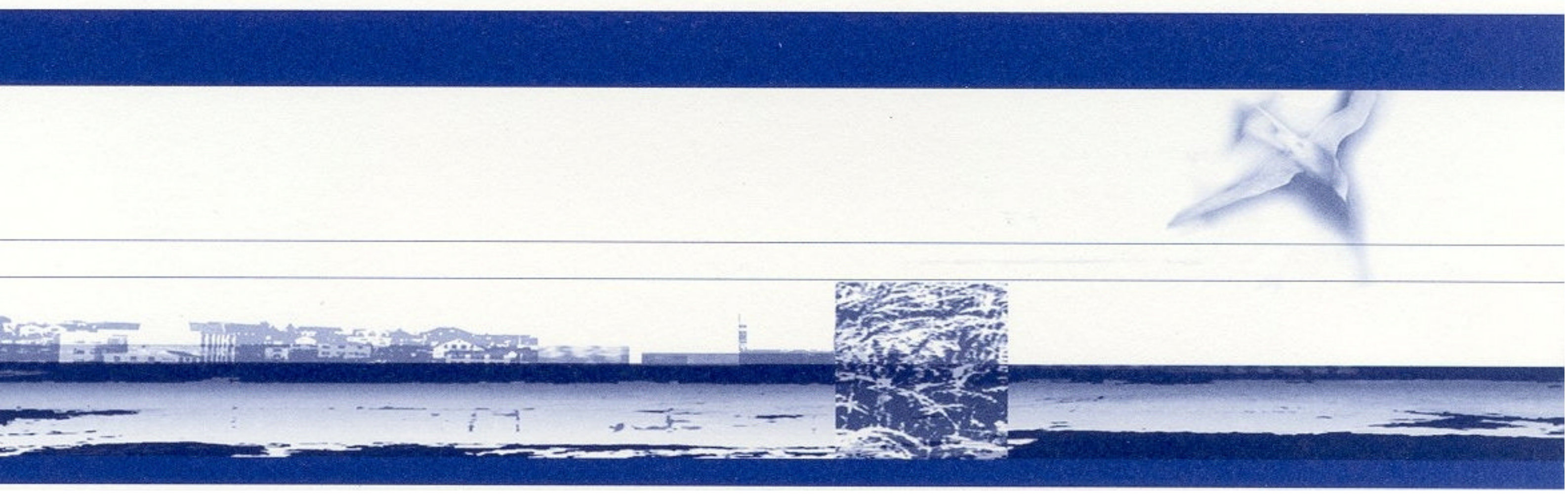

efnagreiningar KELDNAHOLTI

Orkuveita Reykjavíkur

IceTec TECHNOLOGICAL INSTITUTE OF ICELAND

SUMMARY AND EVALUATION OF ENVIRONMENTAL ASSESSMENT STUDIES ON THE RECIPIENT OF SEWAGE FROM THE STP AT ÁNANAUST, REYKJAVÍK

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1. TOPOGRAPHIC AND PHYSICAL OCEANOGRAPHIC CONDITIONS

1.1 Disposal sites, location

The area of sewage disposal from Reykjavik and neighbouring communities (Reykjavík, Seltjarnanes, Kópavogur and Garðabær) is in Kollafjörður, SE-Faxaflói, see maps 1 and 2. Kollafjörður may be defined as the area bounded by the line between Grótta and Kjalarnes with an area of about 83 km^2. The whole of Faxaflói, *i.e.* within Garðskagi and Malarrif, Snæfellsnes, is about 90 km wide and about 50 km long with an area of about 4940 km^2.

Map 1. Faxaflói is the bay within Garðskagi and Malarrif.

Map 2. Kollafjörður is the bay within Grótta and Kjalarnes. Small red dots indicate pump stations while the two larger dots show the STPs at Ánanaust and Klettagarðar.

All sewage from the above communities is collected and treated by only two sewage treatment plants (STP) located on the northern shore of Reykjavik, one is the STP at Ánanaust, Skolpa, and the second is at Klettagarðar, see map 2 and figure 1. The disposal of sewage from the STP at Ánanaust is along a 500 m diffuser 3.6 to 4.1 km NW off the station but the disposal site of sewage from the STP at Klettagarðar is along a 1000 m diffuser 4.45 to 5.5 km NW off Klettagarðar (3.3 km northeast of the disposal site from the STP at Ánanaust). Only these two outfalls are used for all sewage from Reykjavik and neighbouring communities but discharge from the former was started in the beginning of year 1998 while the second was put into operation in the middle of 2002 and in full operation in the autumn of 2005 when all sewage to be treated had been collected and directed to the station. Future disposal is estimated to amount to 400,000 person equivalents (pe), 150,000 p.e. from Ánanaust-STP and 250,000 p.e. from the Klettagarðar-STP. At present, the disposal is much less, however, being approximately hundred thousand p.e. from each STP. One person equivalent is defined as 60 g of BOD_5 per day (C.E.C. 1991). Because of the short distance between these two disposal sites in addition to their environmental similarities as described below, both sites should be considered as one integral recipient of the sewage.

In addition to data collected since the operation of the STP at Ánanaust started in 1998, the following report is based on studies prior to its operation, which were summarised earlier as a basis for the classification of the two recipients (City of Reykjavík's Environmental and Technical Sector 1997 and 2000). The environmental impact studies made after the operation started in the STP of Ánanaust and discussed below have appeared in the following reports (all in Icelandic):

Vatnaskil Consulting Engineers 2000. Dreifing mengunar frá útrás við Ánanaust. Unnið fyrir Gatnamálastjórann í Reykjavík. April 2000. Skýrsla 00.06. (*Dispersal of pollutants from the Ánanaust outfall*).

Jón Ólafsson og Sólveig R. Ólafsdóttir 2001. Ástand sjávar á losunarsvæði skolps undan Ánanaustum í febrúar 2000. Unnið fyrir Gatnamálastjórann í Rreykjavík. Nóvember 2001. Hafrannsóknastofnunin. Fjölrit nr 81. (*Marine conditions in the recipient of sewage off Ánanaust in February 2000*).

Guðjón Atli Auðunsson 2002. Hegðun og samsetning fráveituvatns í Skolpu 2000-2001. Unnið fyrir Orkuveitu Reykjavíkur. Mars 2002. Verkefnaskýrsla Rf 06-02. (*Behaviour and composition of wastewater in the sewage treatment plant at Ánanaust 2000-2001*).

Jörundur Svavarsson 2002. Lífríki botns við skólpútrásarstað undan Ánanaustum,-staða eftir opnun útrásar. Skýrsla til Gatnamálastjórans í Reykjavík. Líffræði-stofnun Háskólans.(*Benthic biota in the recipient of sewage from the outfall of Ánanaust,-status after the operation started*).

Kjartan Thors 2003. Ánanaustræsi: Setflutningur við hafsbotn kannaður með ljósmyndun. Jarðfræðistofa Kjartans Thors hf. Unnið fyrir Gatnamálastofu Reykjavíkur. Júní 2003. (*The Ánanaust outfall: Sediment transport examined by bottom photography*).

Guðjón Atli Auðunsson 2005. Setgildrurannsóknir út af Ánanaustum '00-'01: hafræn meðferð skolps. Unnið fyrir Orkuveitu Reykjavíkur. Desember 2005. Skýrsla ITÍ0605/EGK01. (*Sediment trap studies off Ánanaust '00-'01: Marine treatment of sewage*).

Guðjón Atli Auðunsson 2006. Kræklingsrannsóknir: Ánanaust 2000. Unnið fyrir Orkuveitu Reykjavíkur. Júní 2006. Skýrsla ITÍ0606/EGK02. (*Monitoring contaminants with blue mussels: Ánanaust 2000*).

The following Figure 1 shows the routes of the ocean outfalls in relation to the nearby coastal areas.

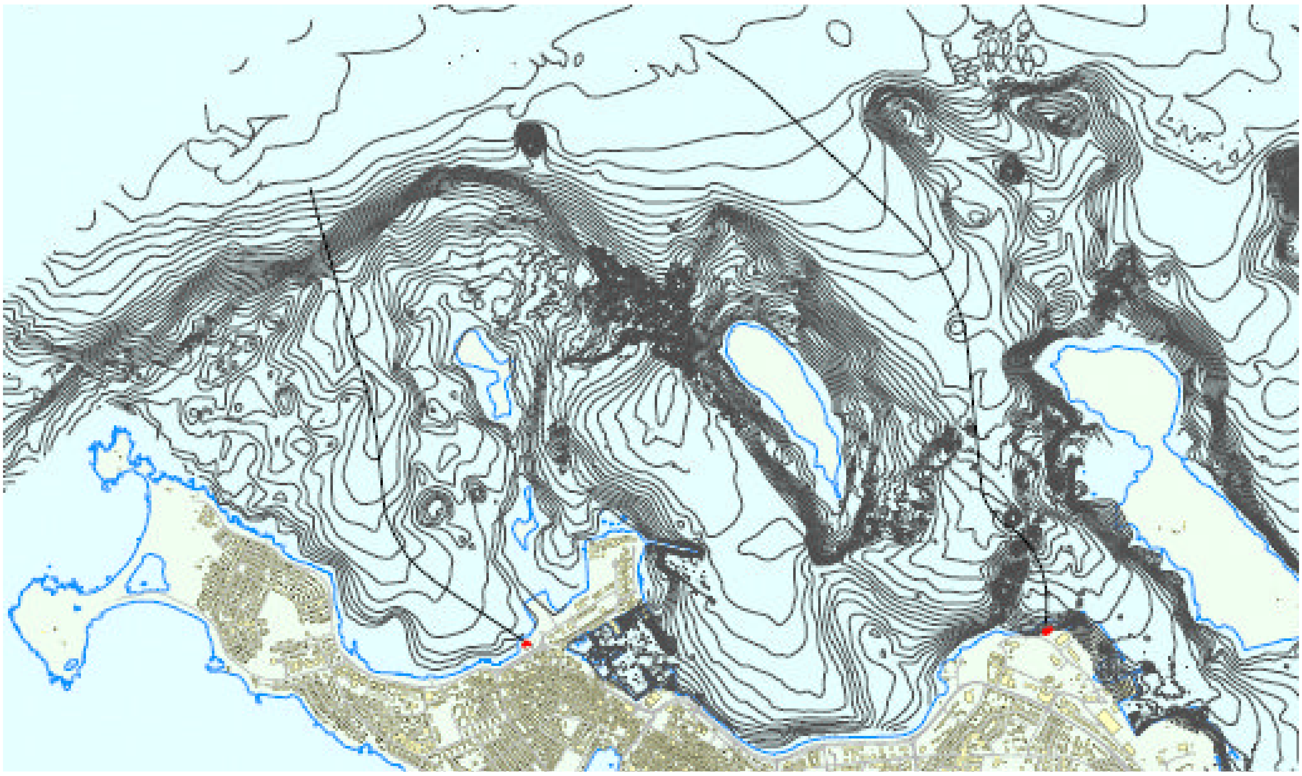


Figure 1 Route of the ocean outfall pipelines.

1.2 Seafloor topography

The volume of the 83 km^2 area of the Kollafjörður at spring tide ebb is about 1.4 km^3. Therefore, the bay is shallow with an average depth of 17 m and the area within the 10 m isobath is about 44% of the total area of Kollafjörður. The area beween the 10 and 20 m isobaths is 26% and the rest, 30%, is within the 20 m and 50 m isobaths. No sills are found in the Kollafjörður area that might prevent mixing with the water outside.

The inner part of Faxaflói is relatively shallow and the area within the 50 m depth contour is 60% of the total area (37% within the 20 and 50 m isobaths) but the area within the contours between 50 and 100 m depths is about 30%. Out from this deep between 50 and 100m is a channel, Kambsleira, towards southeast of Faxaflói reaching towards Hvalfjörður and Kollafjörður (Svend-Aage Malmberg, 1968). Near the mouth of Faxaflói there is a small region, 10-11% of the area, with depths greater than 100 m. This is the innermost part of the Jökuldjúp, cutting into the continental shelf from southwest (Unnsteinn Stefánsson and Guðmundur Guðmundsson 1978). The total volume of Faxaflói at ebb on spring tide is 245 km^3 while the volume of Kollafjörður is 1.4 km^3. No sills are between the disposal sites and the open ocean that might prevent mixing of waters in Faxaflói with ocean water.

In the summer of 1993 and again in May and June 1994 the Marine Research Institute carried out depth measurements by echosounding (with a resolution of ±1 m) along the planned route of the outfall from Ánanaust. The nature of the bottom with regard to rocks and other possible obstructions was described by a diver (Kjartan Thors *et al.* 1994). Similar studies on depths and of the bottom for the Klettagarðar-pipeline were carried out on November 26 1998 and in February and March 1999 (Kjartan Thors 1999). The depth profiles of the two pipelines are illustrated in Fig. 2.

The diffuser of the Ánanaust site starts at 3600 m distance at a depth of 19.5 m while the depth is 32 m at the diffuser's end at 4100 m.

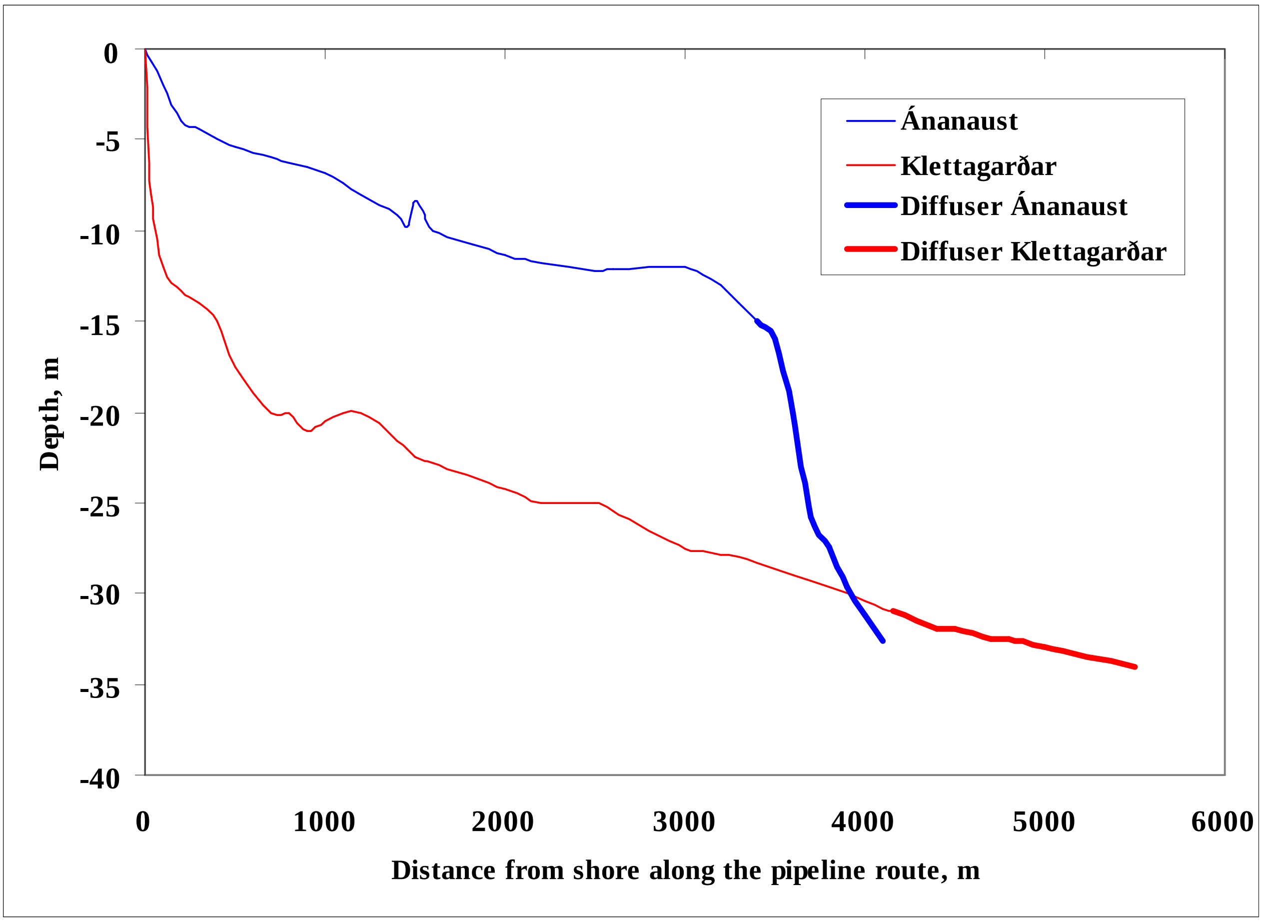


Figure 2 Depths along the route of the pipelines.

The pipe from Klettagarðar crosses a line of rocks which lies north of Laugarnes towards Viðey after which the depth increases fast down to about 20 m at a 1000 m distance along the pipeline. Thereafter the depth increases more slowly and the depth is about 32 m at the start of the diffuser and about 34 m at its end.

1.3 Nature of the seafloor at the disposal sites.

On May 20 1994, the bottom along the path of the Ánanaust pipeline was photographed with deepwater camera (Kjartan Thors 1994). The outermost 500 meters or so (from 4200 m to 3700 m) are characterised by coarse sediments (gravel and coarse sand) with frequent ripples of 10-20 cm wave heights and 20-50 cm wavelengths. Sampling by two types of bottom grabs (Shipek and Van Veen) during biological surveys of the benthos between and around 4000-3700 m turned out to be impossible as the thin carpet of unconsolidated sediments gave rise to very small samples (Jörundur Svavarsson 1996). At 3700 m or 24 m depth, a hard rocky bottom appears and at about 3540 m the lower limit of the brown kelp *Laminaria hyperborea* is reached but the plants of *L. hyperborea* in this area are small and scattered. The kelp-forest grows denser in direction to shore and the plants become taller. Still closer to shore the plants become scattered again with zones of no plants and at 3200 m the kelps disappear. Within 3200 m off shore the bottom is sandy.
Figure 3 is a photograph of a typical area of the bottom at the disposal site of the Ánanaust STP.

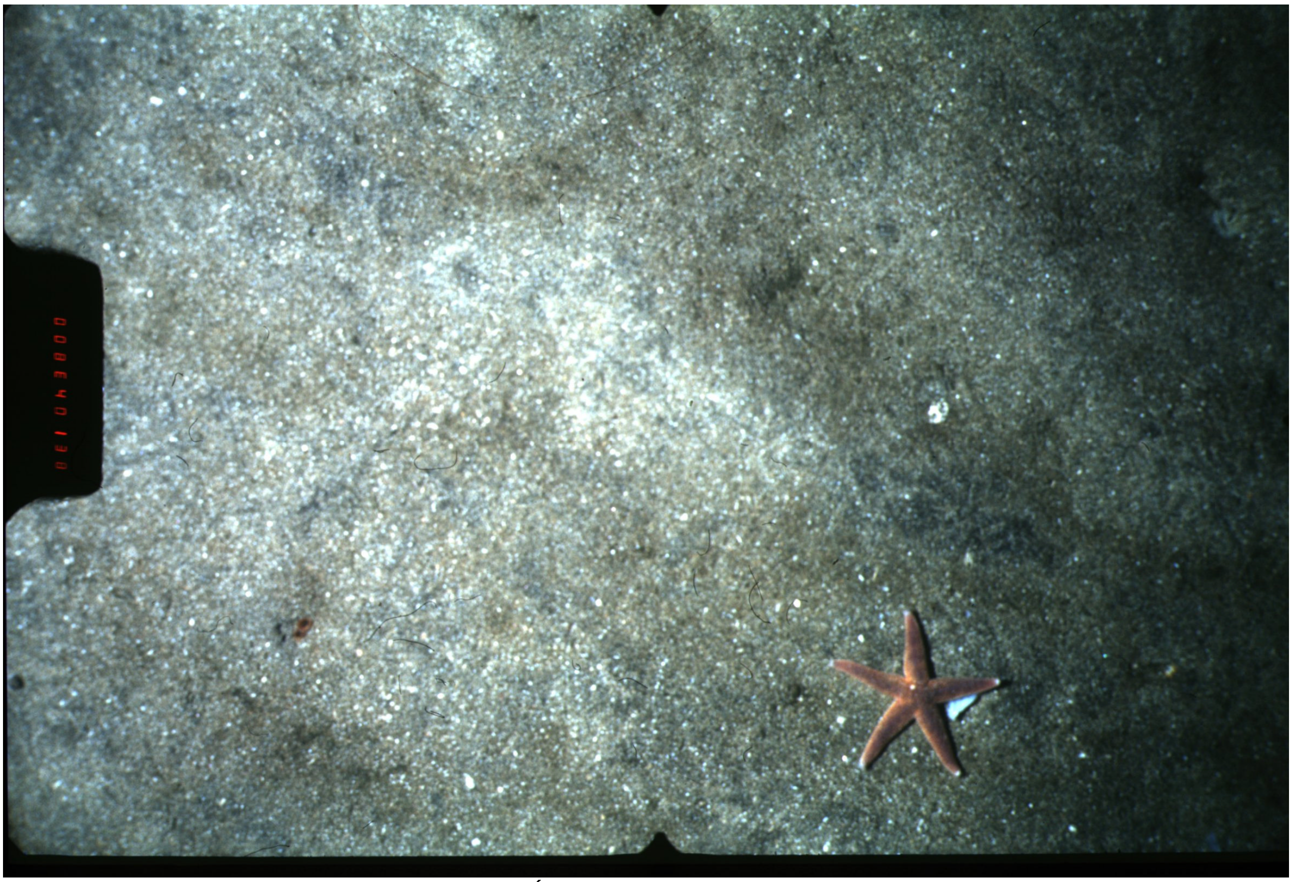


Figure 3 Picture of the bottom at the disposal site of Ánanaust site, May 1997. Coarse sand with large ripples. Ripple crests from upper left to lower right

Corresponding photographing and description of the bottom floor by a diver took place in February and March 1999 along the planned route of the pipeline from the STP at Klettagarðar and simultaneously 10 samples of sediments were taken for determination of particle size distribution (Kjartan Thors, 1999). Next to shore and out to about 2300 m the seafloor is muddy, after which it becomes more sandy with gravel zones out to 5500 m. The disposal site itself is very flat and smooth. During a biological survey at the bottom at the planned disposal site, six samples were also collected and their particle size distribution determined (Jörundur Svavarsson, 2000). Around the New Year 1997-1998 unconsolidated sediments were investigated in the area and their depth to rock was examined, both with seismic profiling and coring, and their particle size distributions determined (Jón Skúlason 1998). These investigations showed that a thin layer of sediment lies upon a hard layer along the path of the planned pipeline from the STP at Klettagarðar to the line of rocks to the west of the STP, see figure 1. In the area from the line of rocks to a gravel pit at the eastern end of Engey, there is a 7 m thick layer of sandy silt with shells but only few meters down to rock in the gravel pit itself. North of the gravel pit, the thickness of the sediment layer is only 1 to 2 meters in a 100 m wide region (sandy silt with shells) after which the sediment layer becomes thicker towards north. This thick sediment layer covers the bottom to the end of the pipeline.

The net sedimentation rate the last 8-10 thousand years decreases with distance from shore as is generally the case with increased depth worldwide. It may be assumed that the net sedimentation at the Ánanaust disposal site the last 8-10 thousand years is somewhere between 30 and 90 cm thick (Kjartan Thors, pers.comm.), *i.e.* 3-4 cm per 1000 years or 0.03-0.1 mm per year. This is very slow rate so close to shore but it is not known whether this rate is equally distributed over these 10 thousand years and it could well be slower still at present. The reasons for this slow sedimentation rate are the strong currents in the area, storm waves during winter, eroding the seabed and keeping the sediment particles in suspension whereupon the currents transport them away from the area. In addition, a lack of sediment supply (silt, clay, and fine grained sand) may also be a reason as well as bioturbation, which keeps the top layers of the sediments loose (non-cohesive) and amenable to transport. Therefore, the likelihood of fine grained particles or the type of particles originating from the sewers settling at the diposal site is very small, cf. further chapter 5 below. For comparison, the sedimentation rate in the deep oceans where the rate is slowest is of the order of 1 cm/ka (Degens and Mopper 1976), usually below 3 cm/ka, *i.e.* of the same order of magnitude as at the disposal sites (ka: kiloannum = thousand years). In valleys and basins on the continental shelves and continental slopes, the rate is 10-100 cm/ka while it can reach much higher rates as for example in the continental shelf off Eastern USA, where it can reach 1000-3000 cm/ka. In enclosed or semi-enclosed seas and in fjords it is 30-500 cm/ka while in some estuaries and deltas where tidal water exerts an influence, *e.g.* Fraser River delta, British Columbia, the rate can be 700,000 cm/ka.

1.4 Water exchange and the freshwater regime in Faxaflói

Faxaflói contains mostly relatively warm and saline Atlantic water but impacts of freshwater occur with seasonal variation. The amount of freshwater in Faxaflói was estimated to be about 3000×10^6 m^3 on average in the years 1966 and 1967 but high seasonal variation occurs (Unnsteinn Stefánsson and Guðmundur Guðmundsson 1978). In these years, the total flow of rivers into Faxaflói, mainly from its eastern part, was about 29×10^6 m^3/day on average or 336 m^3/s, the river Hvítá in Borgarfjörður (glacier water) contributing most. Elliðaár, the river closest to the disposal sites, has an average flow of 4.6 m^3/s (average of 30 years (National Energy Authority, Hydraulic Laboratory 2006)) but the freshwater effects of Elliðaár in the area have been evaluated (Vatnaskil Consulting Engineers, 1984). There is also considerable inflow of freshwater into the southwestern part of Faxaflói, the more so when the winds are from the south or 20-24.5$\times10^6$ m^3 d^{-1} for each knot of southerly winds, *i.e.* about 0.5 m s^{-1} which is unusually low for the area, even for southerly winds, cf chapter 1.5 below. This water originates from rivers in Southern Iceland (mostly glacier water) that enters the coastal current which carries this water around Reykjanes and into Faxaflói. From direct flow of rivers into Faxaflói, the mean residence or flushing time of freshwater in the whole of the bay is about 100 days, but if freshwater entering the bay from S-Iceland is also included, assumed to be equal to direct river flow, the residence time of freshwater is probably 50 days or shorter (Unnsteinn Stefánsson and Guðmundur Guðmundsson 1978). In other words, inflow of freshwater into Faxaflói from S-Iceland is of a similar magnitude or greater than from all the rivers flowing into the bay.

Because of unhindered mixing of the bay-water with ocean water, freshwater effects are mainly seen within the 100 m depth contour of Faxaflói (Unnsteinn Stefánsson and Guðmundur Guðmundsson 1978; Unnsteinn Stefánsson and Jón Ólafsson 1991).

The data presented by Unnsteinn Stefánsson and Guðmundur Guðmundsson (1978) together with total volume of Faxaflói may be used to give a very rough estimate of the flow into Faxaflói if the average inflow of freshwater from the south is assumed to be equal to direct river inflow and that water within the bay (fresh water and seawater) is assumed to be well mixed. Under these conditions, conservation of volume and salinity gives a total flow into the bay of about 5-6$\times10^4$ m^3/s and a flushing time of 7-8 weeks for Faxaflói, which, of course is the same as for fresh water. It has to be borne in mind that these are rough estimates and may fluctuate greatly during the year and from one

month to another. For example, if water inflow from south is nil as above, the flushing time is 100 days but if it is half of the direct inflow or double, the residence times are 70 days and 35 days, respectively.
The total anticyclonic transport around Iceland has been estimated to be about 10^6 m^3/s (as cited in Unnsteinn Stefánsson and Jón Ólafsson 1991), which means that less than 10% of the anticyclonic transport enters Faxaflói. If half of the anticyclonic transport takes place within the 200 m isobath as has been roughly estimated (Unnsteinn Stefánsson and Jón Ólafsson 1991), one may assume that very roughly 1/8 of the transport takes place within the 100m isobath. This means that 40-50% of the flow within the 100 m isobath enters Faxaflói. To give a further example, this flow into Faxaflói would equal the flow of a current of velocity 5 cm s^{-1} at a water depth 50 m (about the average depth north of Garðskagi) and a width of 22 km (12 nautical miles).

These flushing or residence times are of considerable importance since they suggest that most of the discharge of nutrients during October to February will be flushed out before phytoplankton might start to utilise them. The growth season usually starts in mid-March and lasts until mid-October (Þórunn Þórðardóttir 1986). Additionally, not less importantly, these short residence times, especially during summer, mean that the nutrients released cannot sustain phytoplankton or macrofaunal biomass as evidenced by a number of studies in estuaries and bays of the North Atlantic Ocean (Nixon *et al.* 1996; Josefson and Rasmussen 2000). Short residence times result in net export of the nutrients from the system and thereby decreased risk of eutrophication. Using the empirical model of Nixon *et al.* (1996), this residence or flushing time of Faxaflói means that 60-80% of the total nitrogen load and 70-95% of the total phosphorus load will be exported from Faxaflói. Discussion on flushing times and water exchange of Kollafjörður is given in chapter 2.2 below.

1.5 Currents in Faxaflói and in the coastal area north of Reykjavík, general remarks

Clockwise coastal currents predominate in most parts of Faxaflói (Svend-Aage Malmberg 1967 and 1968). The main **residual currents** in Faxaflói are depicted in figure 4 below. The figure shows surface currents during summer. The bottom currents have similar paths and are about half a nautical mile per day (1 cm/s) while surface currents are about 5 nautical miles per day (11 cm/s) (Svend-Aage Malmberg 1968).

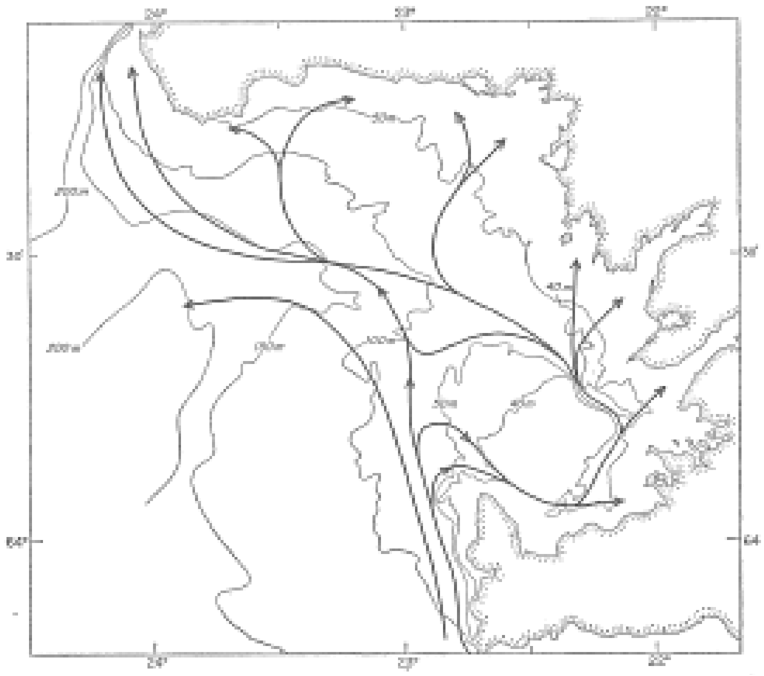


Figure 4 An outline of surface currents in Faxaflói during summer (reprinted from Svend-Aage Malmerg 1968 with permission).

Closer to shore at Reykjavik the residual currents have been measured to be 1-5 cm/s at the disposal sites. The net current follows the coastline with net currents towards northeast at the Ánanaust site but southeast at the Klettagarðar site.

The current system close to Reykjavik is however dominated by **tidal currents**. The tidal range in Reykjavík at spring tide is 4m at flood (0.2 m height above zero level on ebb) on average while it is 3.0 m on average at neap tides (1.3 m height above zero level on ebb) (Vatnaskil 1994). Average tidal range is about 2.1 m. The tidal currents are therefore much larger than the residual current or between 5 and 25 cm/s. The maximum currents occur at high and low tides (actually 1-2 hours after slack water (Svend-Aage Malmberg 1968)), lowest at neap tides and highest at spring tides. The direction of the tidal currents at the Ánanaust site are towards northeast on flood, thus reinforcing the residual current and southwest on ebb. However, the vectoral currents during the tidal cycle of 12 h and 25 minutes form an ellipse (hodogram) with significant currents in all directions although the currents at high and low tides take up most of the time. At the Klettagarðar area the tidal directions are towards southeast on flood reinforcing the net current and northwest on ebb. For the sake of simplicity, one cycle time (12 h and 25 m) at the Ánanaust-area may be divided into four intervals:

Slack water, low water	From 0.5 h prior to and 0.75h after slack water, 1.25 hours in total; turbulent mixing
Flood	0.75 h after inflowing slack water until 0.5 h after high tide, 5 hours in total; currents dominantly towards northeast, about 10 cm/s on average (lunar month).
Slack water, high water	From 0.5 h prior to slack water until 0.75 h, 1.25 hours in total; turbulent mixing
Ebb	From 0.5 h after the outflowing slack water until 0.5 h after low tide, 5 hours in total; currents dominantly towards southwest, 10-15 cm/s on average (lunar month)

Therefore, the tidal currents only partially flush the sewage in each cycle since outgoing water on ebb returns on flood and *vice versa*, the returning water has been estimated to be about 70% in an area closer to shore (Isotopcentralen, 1971). However, in addition to affect net transport from the area, the turbulence of the water during the tidal cycles causes enormous mixing of the seawater and thereby seawater with sewage water, *i.e.* contributes significantly to the initial dilution. The tidal currents are approximately at right angles to the diffuser of the Ánanaust pipe which have the openings on the sides of the pipe directing the jets horizontally into the seawater (or more exactly at 30° from vertical, see below).

Wind induced currents and accompanying **wave motion** are also of considerable importance. Wind induced currents are due to the friction the wind imparts to the surface layer by wind stress (proportional to the wind speed squared). This motion is transferred to deeper layers involving drag forces beneath the sea surface and at the seabed and thereby turbulence in the water masses in between these boundaries by vertical transport of small masses of water (eddies) and their inherent momentum. Turbulence is caused by all currents but the vertical turbulent diffusion is usually at maximum at mid-depths for wind induced currents (Quetin and De Rouville 1986). Most of the energy transferred from the wind to the sea surface, however, is manifested by waves that cause circular motion of water particles at the surface. The orbital velocity of this circular motion (proportional to wave height) at the surface is transferred vertically down the water column but decreases fast and exponentially with depth where the orbits become smaller and gradually elliptical. At any particular place, the waves may have formed locally or have arrived from elsewhere, even originating quite far from the observed waves. The vertical transfer of this motion caused by winds may be hampered by density-stratification, which, however, is neither of long duration nor very steep in the areas north of Reykjavík, see below.

Both of these types of motion caused by winds are of considerable importance down to the seabed as seen by the flux of sediments to the overlying water above the seabed (Guðjón Atli Auðunsson 2005). It is known by experiments that the most important processes for sediment being transported are the so called burst-sweep cycles where downward sweeps of high velocity water penetrate lower velocity layers of water near the seabed exerting instantaneously high shear stress at the bed and displacing both sediment and the immediately overlying lower velocity water as bursts upward into the main flow, which carries the suspension away. Additionally, wave motion is particularly effective in moving sediment particles from the seabed by its circular or stirring to and fro motion at the seabed carrying sediments to the main flow.

Figure 5 shows an example of wind speeds at Reykjavík harbour (Miðbakki) grouped on directional sectors. The figure shows that the highest wind speeds may be expected from the north, northeast and east. The average wind speed for this particular year was 4.7 m/s (median 3.9 m/s). Figure 6 shows the distribution of wind directions in Reykjavik harbour indicating similar behaviour from one year to another and dominance of wind from east and southeast, *i.e.* parallel to the coast of Reykjavík. Winds parallel to the coast impart about 1.5-2 times higher current velocities than directions perpendicular to the coastline, a fact that was observed experimentally during sediment trap studies at both the Ánanaust site and Klettagarðar site (Guðjón Atli Auðunsson 2005). Finally, Figure 7 shows monthly average of wind speeds in the years 1961-2005 where lowest wind speeds may be expected in July (median of all years being 4.6 m/s) and highest in January and February (median of all years being 6.5 m/s). Surface currents are proportional to the wind speed and directed about 45° to the right of the wind direction. Therefore, one might expect that most of the time the wind-driven current gives surface currents from shore towards north. These currents may be expected to be of the order of more than 13-24 cm/s more than 90% of the time on a monthly basis using a semi-empirical relationships for coastal areas (Quetin and De Rouville 1986). However, these currents decrease sharply with depth at the same time as they turn more to the right and at a certain depth they are in opposite direction to that on the surface. Already at 2-3 m depth at the Ánanaust-discharge area, these currents are about half the surface current when wind is parallel to the coast while they are only 15% of the surface current when wind directions are perpendicular to the coast. At 1 m from the bottom at the Ánanaust site, the wind-induced currents are 25% of the surface current if winds are parallel to the coast while they are only 10% of and in opposite direction to the surface current if winds are perpendicular to the shore. An investigation of the sediment fluxes show that wind speeds greater than 3 m/s from southeast do set the bottom fines (clay and silt) into suspension (Guðjón Atli Auðunsson 2001b and 2005) showing that the wind-generated waves and currents reach down to the bottom. According to the Beufort scale the significant wave heights produced are greater than about 0.5 m if these winds prevail more than 2 hours. Significant wave heights measured at Garðskagi, about 40 km WSW of Reykjavík across open sea, correlated well with sediment fluxes at the Ánanaust discharge area when greater than 0.9 m (periods always less than 15s) (Guðjón Atli Auðunsson 2001b and 2005). These wave heights have empirically been shown to move particles of 0.2-0.3 mm diameter (fine and medium sized sand) at a depth of 30 m. Winds have thus a very substantial influence on the currents and mixing in the area.

The system into which the sewage is discharged is apparently very well mixed most of the time. During the summer months, notably July and August, the system is calmest due to comparatively low wind velocities. Only during this time, small density gradients are formed, which decrease somewhat the buoyancy of the sewage plume as well as inhibiting vertical transfer of wind-induced motion from surface down. This is therefore the period of the year in which most of the studies below have been concentrated.

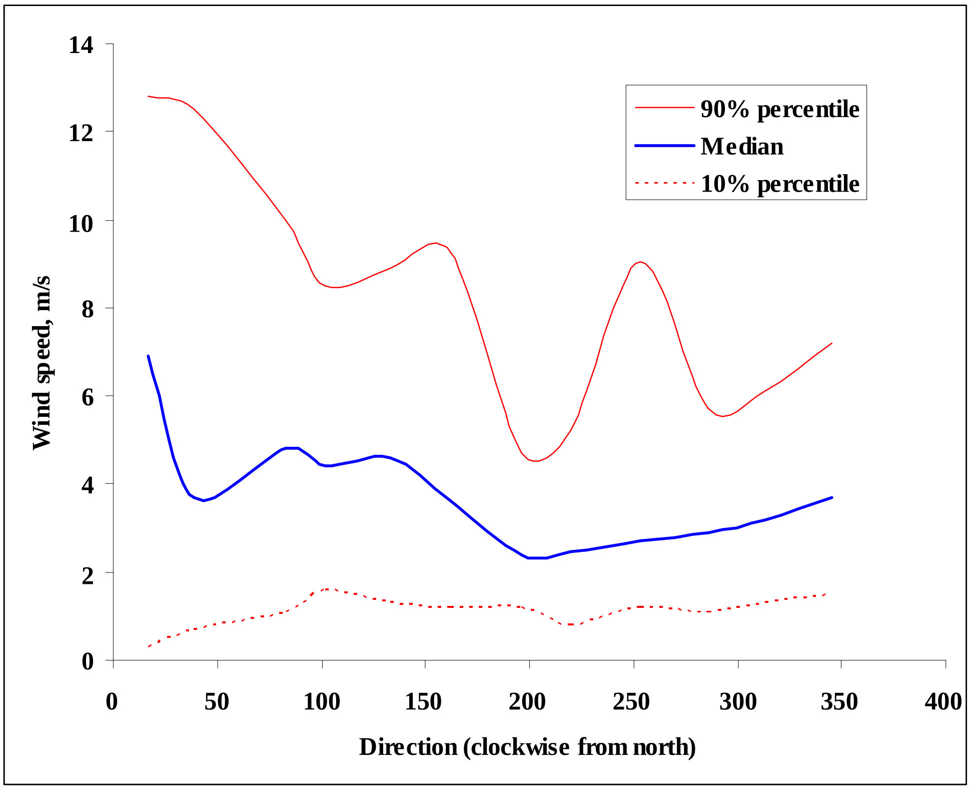


Figure 5 Median wind speeds in Reykjavík harbour in the period 01/05/200-30/04/2001, i.e. a whole year. Measurementscarried out every hour.

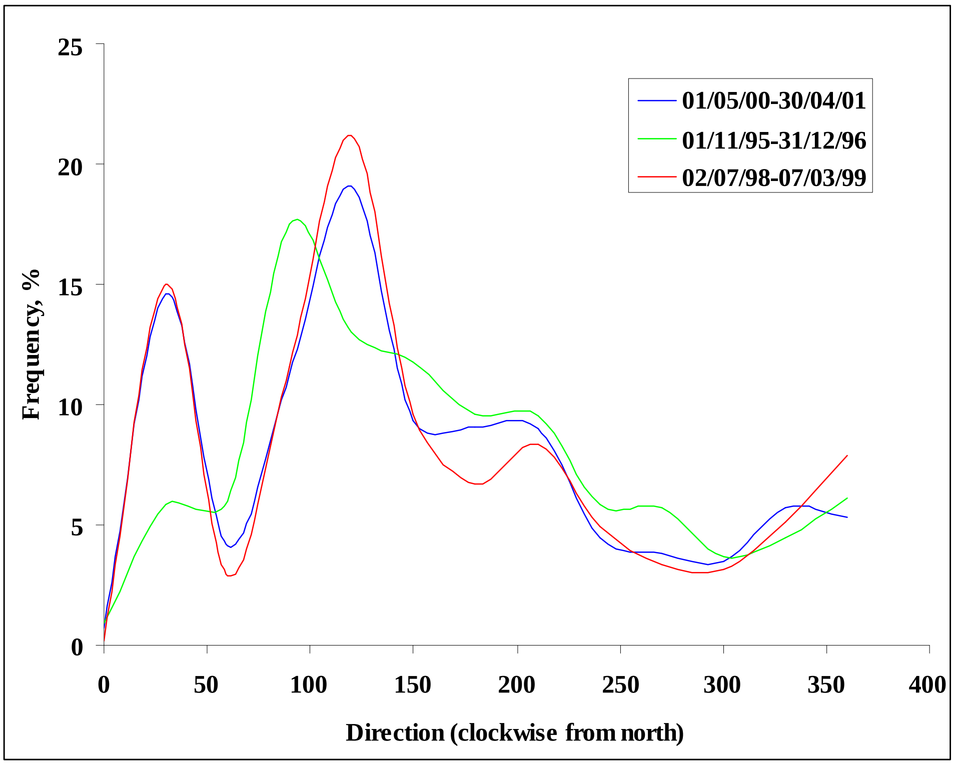


Figure 6 Distribution of wind direction in Reykjavík harbour in the years indicated, whole years except '98-'99.

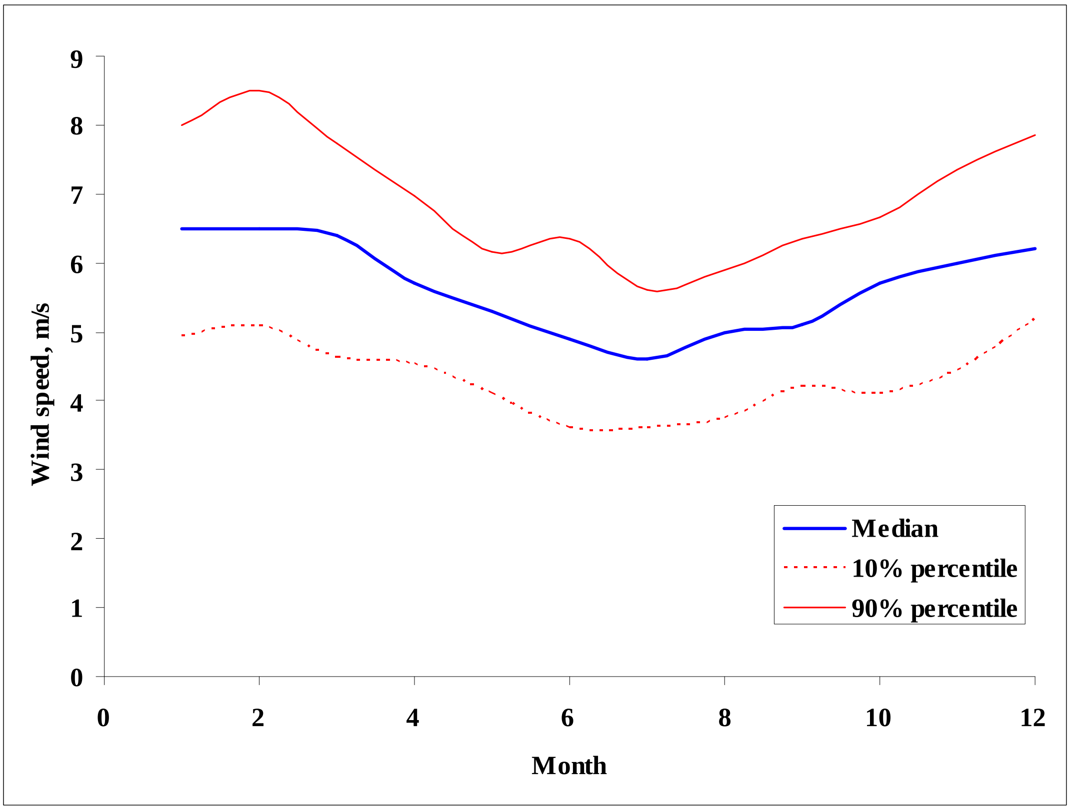


Figure 7 Median monthly wind speeds in Reykjavík in the years 1961-2005.

2 Dilution of sewage at the disposal sites. General description and simple empirical models

The general picture of the currents at the Ánanaust and Klettagarðar disposal sites may be used to estimate the dilution and flushing of water in the areas. The issue of sewage dilution has been the subject of innumerable worldwide studies in the laboratory, in the field, and with mathematical models. Simple calculations based on the general currents presented above may give an idea of how the system works but results of more elaborate mathematical modelling are presented in chapter 3. In the discussion below, a general picture of the dilution and dispersion processes is given, after which simple calculations are presented with comparisons with actual measurements.

2.1 General description of the behaviour and fate of effluent constituents

The diffuser at the Ánanaust site is 500 m long with circular 0.075 m diameter orifices on each side at about 60° from horizontal line through the centrum of the pipe. The 60° angle for the nozzles has prevailed in outfall design ever since Zeitoun *et al.* (1972, cited in *e.g.* Roberts and Toms 1987) found this angle to give the longest trajectory and therefore the greatest dilution. The orifices are about 14 m apart on each side, *i.e.* 7 m between holes alternating between sides. Therefore, the plumes from two adjacent diffuser holes will not overlap on their way up to the surface (the diameter on the surface containing 90% of the liquid discharge will be 10-12 m in stagnant homogenous media). The jet discharge velocity from each orifice will be about 3 m/s at an average effluent flow from the Ánanaust STP of 1 m^3/s. This is the first stage of dilution since this orifice velocity will impact the surrounding media with turbulence caused by the transfer of high momentum flow. Empirically, it has been found that the dilution in a stagnant medium is roughly inversely proportional to the orifice diameter (and directly proportional to the water depth) (Fischer *et al.* 1979) while a more detailed experimental and theoretical analysis for the jet in cross flow results in buoyancy and orifice velocity entering the equation (Chu and Goldberg 1974; Roberts and Toms 1988).

Dilution may be considered to occur in four phases, where the first is the dilution immediately after release of sewage from the diffuser by the jet from the orifices. The second stage is the subsequent immediate mixing that is effected by the buoyancy of the discharge. Thses two first stages are collectively the initial dilution in the near field area. The third stage is the dispersion of the sewage on the surface, the far field dilution, and the fourth is the long run effects on a larger scale.

Initial dilution or near field

Since wastewater has a density close to fresh water, it is lighter than the surrounding seawater (which is colder and more saline) and the resulting buoyancy deflects the jets from each hole of the diffuser upward, forming a plume which is swept downstream by the current. The plumes entrain seawater as they rise causing dilution and increased density and finally, if the surrounding seawater is not stratified, will reach the surface. However, if the seawater has a large enough density gradient, the plume may become trapped under the surface. This inital dilution (jet momentum and buoyancy) occurs with vigourous mixing and causes rapid dilution well within 10 minutes after the release. This is the so called “near field” or “initial mixing region”. Within the initial mixing region, the fluctuations in concentrations may be large compared to their average value. At the end of the inital mixing region the fluctuations are small, however. When the plume hits the surface, the average dilution at the surface is about two times the minimum dilution (maximum concentration) at the centerline of the plume (Roberts 1991). With the dominating tidal currents at both the disposal sites, Ánanaust and Klettagarðar, for which the current vectors form an elliptical path, the plume at the Ánanaust site for example will appear as an ellipse on the surface with the long axis perpendicular to the diffuser. This elliptical form is also due to the nature of the dispersion when the sewage plume enters the surface, see below. Subsequently, the established wastefield drifts with the surface currents to a region called the “far field” with further dilution by dispersion (diffusion and convection by turbulence). This farfield dilution is the third stage of dilution and is acting with totally different mechanisms from those in the nearfield dilution.

If there is any residual density difference left when the plume hits the surface, *i.e.* still some upward or positive buoyancy, the plume may spread or float like an oil slick on the denser natural seawater and will be pushed by the continous flow without any vertical mixing with the underlying water. For this to happen, however, the surface has to be relatively smooth (*e.g.* wind velocities less than 3 m/s) and the density difference must amount to an inital dilution of less than about 125 (Quetin and De Rouville 1986). Neither of these conditions are likely to ever exist simultaneously in the coastal waters north of Reykjavík.

Another aspect of this initial dilution applies if the current speeds are large or in the range 8-80 cm/s for the Ánanaust site, when a so called forced entrainment regime may take place, where the discharge is rapidly carried out to sea. In this case the plume cannot entrain all of the oncoming flow and the rise height and thickness decreases with increasing current speed (Roberts 1991). This situation may occur in the area of discharge, especially during winter.

The initial mixing zone ends when there is no further dilution due to turbulent mixing caused by the jet momentum flux, buoyancy and accompanying friction. During the first two phases, the thickness of the wastefield increases. In the end of the initial dilution region, the turbulence collapses, *i.e.* when the inital kinetic energy of the jet and the initial potential energy due density difference (gravitational forces) have been dissipated.

For the Ánanaust site at 10 cm s^{-1} average current, this will take place within 50 meters from the diffuser or within 10 minutes. Further downstream, the wastefield width increases and its thickness decreases due to gravitational collapse. In the transition between inital dilution and far field, the plume flows at a rate of the discharge flow times the inital dilution factor which is also approximately equivalent to a plume moving at the ambient current speed with a width roughly equal to the diffuser length (Medcalf&Eddy 1991). With these assumptions and using a depth average current of 10 cm/s for the Ánanaust site, an average dilution of 1:1000, see below, and a discharge flow of 1 m^3/s (Guðjón Atli Auðunsson 2000 and 2002), and assuming homogenous medium like in winter, the thickness of the plume would take up about 80% of the water column. Alternatively, a depth average current of 5-8 cm s^{-1} would make the transition plume extend over the whole water column at Ánanaust under the same assumptions. Therefore, it seems likely that in the transition between near field and far field, the plume probably extends over most of the water column most of the time.

Far field

In the far field, the mixing is caused by ambient oceanic turbulence and proceeds much slower than in the near field. Generally, an additional dilution of 5-20 is expected within a zone of 1 km (Quetin and De Rouville 1986). The plume broadens on its way downstream by lateral spreading and with increasing rate since larger and larger turbulent eddies participate in the dispersion (*i.e.* the turbulent diffusion coefficients increase with time and/or distance from the source, see *e.g.* Stacey *et al.* 2000, Okubo 1971, and Okubo 1974). In a tidal environment, the reduction in contaminant on the plume centerline may be relatively small for the first hours of plume travel. At moderate distances the levels are intermittent, consisting of long periods of no detectable concentrations or of concentrations close to the low background levels intermixed with much higher concentrations when the plume is present. These higher concentrations would occur in a region about the size of the tidal excursions, *i.e.* within about 5 km east and west of the diffuser at the Ánanaust site. The turbulent diffusion is mainly caused by shear stress at the seabed and wind stress at the surface and are far from being equal in all directions, the horizontal dispersion coefficient in direction of the current being highest although of little importance in relation to the current speed with which the plume moves while the one perpendicular to the current increases with time (distance) usually following the 4/3-law, *i.e.* proportional to the length of plume (diffuser length at the start) to the power of 4/3 (Stacey *et al.* 2000). Therfore, the first hours it may be 0.1-1 m^2/s, then dispersion that is due to a combination of turbulent diffusion and shear stress becomes increasingly important resulting in a dispersion coefficient around 25 m^2/s. For longer times, horizontal shear dominates, and the dispersion may grow up to 100-1000 m^2/s (Roberts 1999a; Zimmerman 1986). This large ultimate dispersion coefficient combined with flushing, see below, results in very low background contaminant concentrations. The vertical dispersion coefficient is usually about one third of the transverse coefficient (Quetin and De Rouville 1986) but it decreases with increasing stratification (inversely proportional to the density gradient) and is affected by wave characteristics, *i.e.* increases with wave height (proportionally) and shorter wave periods (inversely proportional to the wave period squared) (Koh and Brooks 1975). The dispersion coefficients may be approximated fairly well but measurements of these at the discharge site by for example release of dyes or radiaoctive tracers are usually necessary since the coefficients may vary in space and time with the flow characteristics, *i.e.* they are not physical constants. If the coefficients have been approximated or found by measurements, they can be used to estimate downstream dilution with simple calculations (Metcalf & Eddy 1991; Quetin and De Rouville 1986) where the simple solution of the diffusion equation by Brooks is most commonly applied (Brooks 1960). The Brooks solution assumes a steady current and negligible vertical and longitudinal dispersion. These calculations may give quite accurate avereage values for simple systems, *i.e.* for uniform currents as opposed to varying current speeds and direction across the water column and time and when density gradients are linear. This may also give good insights into the processes, *e.g.* at the worst conditions. Otherwise, the use of more elaborate mathematical models is required (Roberts 1991; Medcalf&Eddy 1991; Roberts 1999a), see chapter 3, but the same general principles as described above are, however, always used.

The dilution factor in the far field multiplied by the initial dilution factor gives the total dilution at any distance from the plume center within the mixing zone.

It is important to keep in mind that the descriptions above relate to time-averaged situations since, during short travel times after release from the diffuser, the plume meanders around and breaks up into successive "puffs" like a smoke from a chimney (Quetin and De Rouville 1986; Roberts 1999a). This causes episodic, or intermittent, "fumigation" of a given fixed point by the contaminants it carries, *i.e.* the local contaminant concentrations there are negligible most of the time but is relatively high during fumigation episodes. For this reason, the picture of both the near field and far field are patchy at any given time which many studies have revealed (*e.g.* Carvalho *et al.* 2002; Petrenko *et al.* 1998). For far field modelling the horizontal trajectories of these patches have been examined with progressive vector diagrams and resulting visitation frequency diagrams (Robets 1999; Washburn *et al.* 1999) and with particle tracking, computationally not practical, however, for very long simulations extending over many months (Roberts 1999a). In short, the deterministic prediction of transport in far field may not be possible because the trajectory of the plume in an unsteady tidal environment is a chaos phenomenon (Zimmerman 1986), resulting in individual plume trajectories that are unpredictable, random, and not repeatable. However, the best present day models describe the gross features of the plumes reasonably well but there remains aspects which cannot be modelled yet, particularly the patchy nature of the wastefield. Further complications are for example due to decay processes, *e.g.* the decay rate of bacteria is fastest during daylight hours and slowest at night, and decreases with increased depth below the water surface. This variable decay rate of bacteria therefore also results in varying die-off times with season in Iceland, see chapter 3.

Long run effects

Finally, exchange rates on a larger scale, even at continental shelf scales, and biological and chemical decay processes determine the possible build-up of contaminants in the long run. This is the fourth stage of dilution. This phase is often described by simple tidal prism models (*e.g.* Isotopcentralen 1971a; Strain, Wildish and Yeats 1995) which work well if sills are not present or/and if there are no strong/stable stratifications (Strain and Yeats 1999).

In the UK, the different mixing zones have been the subject of comprehensive studies and modelling. The European Union Urban Waste Water Treatment Directive (CEC 1991) sets out the general conditions for which primary treatment, rather than secondary, may be sufficient for a marine discharge. The Urban Waste Water Treatment Directive (UWWTD) sets secondary treatment of urban wastewater as standard (article 4), which may though be simplified to primary treatment only if discharging from agglomerations of between 10,000 and 150 ,000 p.e. to coastal waters defined as "less sensitive" recipients and if comprehensive studies indicate that such discharges will not adversely affect the environment (article 6). In exceptional cases, primary treatment is also sufficient for agglomerations over 150,000 p.e. when "it can be demonstrated that more advanced treatment will not produce any environmental benefits" (article 8). Enhanced or tertiary treatment is required if discharging into "more sensitive" waters (article 5). The presence or absence of eutrophic conditions is one of the indicators of "sensitive" or "less sensitive" waters. Article 6 requires "comprehensive studies" and so the in the UK the "Comprehensive Studies Task Team" (CSTT) was set up to provide guidelines on the implementation of the directive. These guidelines cover four main aspects of the problem: initial dilution; oxygen demand; nutrient loadings, particularly in respect of eutrophication; and suspended solids (all these parameters will be addressed numerically in 2.2 below except for suspended solids which will be dealt with in chapter 5). CSTT introduced a distinction between "zones" or spatial scales (CSTT 1997; Sherwin 2000):

- Zone A: that around and close to the end of a pipeline or a fish farm; concentrations will be similar to the initial dilution in the boil.
- Zone B: that of a sea-loch basin or other water area in which the residence time of pollutants is a few days; this is the near field region according to CSTT, denoted to be at least the size of the tidal excursion and where phytoplankton growth might take place in favourable circumstances.
- Zone C: a larger region, *e.g.* an offshore water providing the "boundary conditions" for zone B. This is the far field region where the residence time is of the order of weeks to months.

These guidelines have been used in the UK and are likely to be applied in other European countries and the latest version of the CSTT-model appears in Tett *et al.* (2003) where it was applied to a large variety of systems (Kongsfjorden, Spitzbergen; Gullmaren, W-Sweden; Himmerfjärden, E-Sweden; Firth of Clyde, Scotland; Golfe de Fos, S-France; Ria Formosa, S-Portugal).

The UWWTD defines "eutrophication" as the enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned. The definition of OSPAR is essentially that of UWWTD but with an emphasis on "undesirable changes ("Eutrophication" means the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned, and therefore refers to the undesirable effects resulting from anthropogenic enrichment by nutrients as described in the Common Procedure") (OSPAR 2001). Therefore, eutrophication is a process of accelerated growth or a shift from lower to higher trophic status, see also Nixon (1995). The team developed the CSTT model (CSTT 1997), and proposed the "Water Quality Standards" (now Environmental QS) in which winter values of dissolved available inorganic nitrogen (DAIN) should not exceed 12 µM and summer chlorophyll *a* should not exceed 10 mg m^{-3} (DAIN is the sum of nitrite, nitrate and ammonia). The value of cholorophyll a was chosen 10 mg m^{-3} since at that level the phytoplankton will be seen by the bare eye. However, these are not applicable everywhere, since the trophic status may differ from one location to another and naturally eutrophic areas will exceed these values. According to Nixon (1995) marine waters are classified according to the following scheme:

Status, trophic state	Primary productivity, g C m^{-2} y^{-1}
Oligotrophic	<100
Mesotrophic	100-300
Eutrophic	300-500
Hypertrophic	>500

For example, the trophic status of the coastal waters north of Reykjavík are naturally eutrophic (Þórunn Þórðardóttir 1994) since the annual productivity is more than 300 g C m^{-2}. Thus, using the relationship between primary productivity and chlorophyll *a* (Chl *a*) in Icelandic waters (Kristinn Guðmundsson og Kristín J. Valsdóttir 2004) and an average euphotic zone down to a depth of 20m in the recipient (light attenuation evaluated by Secchi depth (k_D=1.4/Ds) and relationship between Secchi depth, Ds, and Chl *a* (Kristinn Guðmundsson, Þórunn Þórðardóttir og Gunnar Pétursson 2004)), the average Chl *a* in the waters north of Reykjavík during the growing season will be more than 15 mg m^{-3}, a result also obtained if a yield of 1.1mg Chl *a* per mmol nitrate is used (Tett *et al.* 2003) and assuming most of the winter nitrate in Faxaflói, 13.7±0.3 µM (n=25 on a transect west of Akranes out to Jökuldjúp) (Sólveig R. Ólafsdóttir 2006), to be used for primary production. Since this is an average value of Chl *a*, considerably higher values of Chl *a* concentration may occur during autumn but especially spring blooms. The nitrate quality standard value of 12 µM in the CSTT model does therefore not apply for Icelandic waters since natural winter values of nitrate are higher. Therefore, the approach proposed by OSPAR (2001) as a measure of significant shift from oligotrophic to eutrophic without providing an absolute standard for oligotrophic or eutrophic, is more appropriate in general and involves the following criteria:

- a 50% increase in winter DAIN (dissolved available inorganic nitrogen) and DIP (dissolved inorganic phosphate) concentrations, and in maximum and mean growth season phytoplankton chlorophyll concentrations;
- increase is relative to “historical region-specific background concentrations” or “spatial (offshore) background concentrations”;
- in waters of varying salinity, concentrations should be corrected to standard salinity.

The eutrophication proceeds as follows:

- nutrient enrichment;
- increased phytoplankton growth, biomass and primary production;
- *undesirable* disturbances to water quality and “balance of organisms”

Finally, the undesirable disturbance used in the definition of eutrophication of both the UWWTD and OSPAR may be described by the following:

1. shifts in relative abundance of diatoms and dinoflagellates or flagellates (in favour of the latter);
2. increases in species considered to indicate eutrophic conditions and which themselves might be nuisance or harmful species;
3. increase in harmful algal blooms (HABs) and/or “red tides”;
4. decrease in water transparency with adverse effects especially on benthic sea-grasses;
5. increased sedimentation of organic matter, resulting in increased rates of oxygen consumption in near-bottom water or seabed, and hence to potential deoxygenation in these regions;
6. (increased) kills of plankton, benthos or fish as a result of above effects;
7. changes in food webs as a result of above effects and also because change in phytoplankton might cause change in grazing zooplankton.

The simple CSTT model deals with the eutrophication potential by the following criteria (Tett 2000) after a zone has been defined and water exchange in a single and well mixed box has been evaluated from data on currents:

1. How much will the receiving water concentration of nutrients increase as a result of the discharge?

2. If the extra nutrients were used by phytoplankton, would the resulting concentration of Chl *a* exceed the environmental criteria, 10 mg m^{-3} in CSTT but 50% increase by OSPAR, assuming total conversion of nutrients into Chl *a*?

3. Can phytoplankton in fact use these nutrients, *i.e.* is the receiving water too turbid to allow plant growth, or diluted too fast to sustain an increase in the abundance of plankton, provided information on photosynthetically available radiation (PAR) is monitored?

4. Would the answers to the questions above be any different if the discharge received primary rather than secondary or more stringent treatment?

Both the OSPAR and the CSTT models use chlorophyll *a* as a measure of microplankton biomass.

2.2 Forecasting dilutions and effects by simple empirical and semi-empirical models

In the simplest of cases where no density gradients are present, maximum dilution in the vicinity of the diffuser may be found by comparing the outfall discharge with the probable seawater flow through a sea cross-section. Assuming an average current speed over the whole water column of 10 cm/s perpendicular to diffuser length, a conservative estimate for the Ánanaust area, the maximum initial dilution at the average depth of 25 m at the Ánanaust site (neap tide level), is between 1000 and 1500. Here, the average sewage flow of 1 m^3/s from the Ánanaust STP is used (Guðjón Atli Auðunsson 2000 and 2002).

Initial dilution or near field

This is equivalent to zone A in the CSTT-model (Sherwin 2000; CSTT 1997). Oceanographic parameters were monitored close to the discharge area of Ánanaust STP in the years 1966 and 1967, and showed stratification during June, July and August 1966 with a maximum aproximately linear density gradient of 1.2 g/L over 30 m water column for a short while at the end of June and the beginning of July (Unnsteinn Stefánsson *et al*. 1987). The pycnoclines are, however, often disrupted by winds in this area (Þórunn Þórðardóttir 1986) although occurring when wind speeds are expected to be lowest as discussed above. A density gradient, nonlinear however, was also observed much closer to shore in the summer of 1977, *i.e.* a maximum of 1.3 g/L difference across 30 m water column between Viðey and Engey 24/07/1970 (Isotopcentralen 1971a). Therefore, this gradient seems to reflect the most adverse conditions as regards dilution rate in the discharge area. Taking this into account and using a simple but well established semi-empirical model for the initial dilution from a diffuser (Roberts 1991; Roberts, Snyder and Baumgartner 1989a, 1989b, 1989c), a minimum dilution of 300 is found or an average initial dilution of 600 (assuming an average current of 10 cm/s and a discharge rate of 1 m^3/s). However, apart from a short period in mid-summer, the water is not stratified in this area (Unnsteinn Stefánsson *et al*. 1987), and simple empirical relationships give a minimum dilution of 750 and average dilution of 1400 (25 m average depth to diffuser, 10 cm/s current perpendicular to diffuser, and a discharge rate of 1 m^3/s). This empirical relationship is actually 60% of the simple box-model value above. These calculations show that an average initial dilution of about 1000 may be assumed most of the time. Additionally, under this stratification conditions and a depth-average current of 10 cm/s, the plume will probably not reach the surface.

These simple calculations of inital dilution are in line with experimental results for several microbial parameters in caged mussels showing lowest dilution factor of 1000 during July and August 2000, this minimum dilution occurring at the start of the diffuser (Guðjón Atli Auðunsson 2006). A somewhat higher median dilution factor was obtained for faecal coliforms when results for surface water samples collected on 02/02/2000 (*i.e.* long die-off times, sampling close to spring tide during ebb, and the wind speed was 2.6 to 3.6 m s^{-1} from SSW) in an area along and close to the diffuser were compared with estimated levels in the discharge (using experimental *per capita* value of $1.3x10^{10}$ MPN/p.e./d, see chapter 4). However, about two and a half months later or on 13/04/00 (also close to spring tide where winds were 1.5-3.1 m s^{-1} from the north), hardly any sample gave results of faecal coliforms above 2 MPN/100mL (Vatnaskil 2000), one value being high, the station at the beginning of the diffuser, the same station giving highest response in the mussel samples. Investigations on the level of faecal enterococci in 60 beaches in S-California showed highest values during ebb at spring tides (Boehm and Weisberg 2005) suggesting that these sampling dates, especially on 02/02/2000, may have been at the most adverse conditions. Additionally, a minimum dilution of 600 was observed in one sample on 02/02/2000 above the middle of the diffuser when salinity-silicate relationships obtained prior to and after discharge from the STP at Ánanaust were applied (Jón Ólafsson and Sólveig R. Ólafsdóttir 2000). However, the average dilution in 24 surface samples taken in a small area close to the diffuser showed an average dilution of 1100 or between 600 and 1600 (Jón Ólafsson and Sólveig Ólafsdóttir 2000). It is more difficult to use either nitrogen or phosphate to evaluate dilution due to their natural presence in seawater. However, application of the *per capita* values below together with discharge volumes that particular day, and using the station of highest salinity in the study of Jón Ólafsson and Sólveig Ólafsdóttir (2000) as the background or natural level of the nutrients, a total average dilution factor can be estimated. Ammonia (ammonia being 33% of total nitrogen in the effluent (Guðjón Atli Auðunsson 1992)), total nitrogen, and total phosphorus in the close vicinity of and above the diffuser show dilution factors that are log-normally distributed as found in other studies (Roberts 1999b). The dilution factors for these three nutrients did not differ significantly and are on average 1000 as found above but with 10 and 90% percentiles ranging from 400 to 2000.

Therefore, all results above taken together, the average initial dilution is about 1000 both inferred from experiments and calculations by semi-empirical models, where the average dilution is 1.5-3 times the minimum dilution.

Applying this dilution to nutrients, the relative increase in nitrogen and phosphorus from 150 thousand person equivalents in the surface water above the diffuser is less than 15% and 10% on average of the winter values of nitrate and phosphate in Faxaflói in 2002 (13.7 µM and 0.92 µM, respectively (Sólveig R. Ólafsdóttir 2006)). 150,000 person equivalents are the maximum amount expected but the present value is only about 100,000. During dry and warm summer months when the dilution of the sewage itself is minimal or only twofold (Guðjón Atli Auðunsson 2000 and 2002), the increase in concentrations at the surface above the diffuser (in an area extending 20-30 m on either side of the diffuser) amounts to less than 30% and 18% for nitrogen and phosphorus, respectively, from 150 thousand person equivalents (using the measured *per capita* volume of 270 L/p.e./d, see chapter 4 (Guðjón Atli Auðunsson 2000 and 2002). Therefore, even in the near field right above the diffuser the average elevation of these nutrients is well below the limit set by OSPAR (2001) of 50% increase as a measure of significant shift up a trophic level. Applying this average dilution to chemical oxygen demand, the near field depletion of oxygen would be less than 2.5% resulting from a discharge of 150,000 p.e., a decrease that would require very extensive sampling and analysis to reveal. The sampling on 02/02/2000 from 90,000 p.e. did not reveal any significant decrease in oxygen, *i.e.* the decrease was significantly less than 0.25% (95% CI).

Extended mixing and source region: far field

This is the region defined as zone B in the CSTT-model. The Brooks-solution (Brooks 1960) to the diffusion equation with estimated transverse turbulent diffusion by semi-empirical means (*i.e.* the "4/3-law") given in *e.g.* Roberts (1999a), Stacey *et al.* (2000), and Koh and Brooks (1975) may be used to define this area. The solution of Brooks assumes negligible vertical diffusion or longitudinal dispersion from a finite line source in a steady current and is therefore a conservative estimate. Using a time period of one tidal cycle to define the far field region in an average current of 10 cm s^{-1} gives an ellipse with a short axis of 1.7 km and a long axis of 4.5 km, *i.e.* an area of about 24 km^2. At these distances, the Brooks equation predicts an additional dilution of a factor of about five at the edges of the ellipse and thereby the nutrients nitrogen and phosphorus are on average at levels less than 3 and 2%, respectively, of the winter values from 150,000 p.e. or less than 6 and 4% during dry and warmest summer months.

Another way to look at the far field is to assume the water and sewage underneath the ellipse to be well mixed as is done on the CSTT-model, and assuming further the residual current to be 3 cm s^{-1} (neglecting turbulent mixing and water exchange by winds, wave motion and tidal currents to have a very conservative estimate), then the water exchange is 0.012 h^{-1} and the average additional or build-up concentrations of nitrogen and phosphorus in this volume is about 1 µM and 0.04 µM, respectively, from 150 thousand person equivalents or less than 10 and 5% of the winter values, well below the criteria of OSPAR (2001). Under these assumptions, the average decrease in oxygen in this volume from 150,000 p.e. is at maximum 0.12 mg/L or less than about 1.5% of the equilibrium level of oxygen neglecting decrease by nitrification and increase by uptake of oxygen from the atmosphere and production during photosynthesis. Thus, even this conservative estimate of oxygen decrease would be very difficult to detect experimentally.

The question remains to be answered to which degree the nutrients in the far field may be converted to phytoplankton biomass during photosynthesis. During the winter months (middle of October to middle of March), however, there is hardly any primary production in Icelandic waters (Þórunn Þórðardóttir and Unnsteinn Stefánsson 1977). Macoralgae may utilise nutrients for growth during winter, however, where the growth is controlled by length of day, *i.e.* starts to grow when daylight duration is less than a certain limit, approximately 12 hours in late winter, and ceases to grow in the spring when daylight duration is more than 12 hours. The kelp tangle (*Laminaria saccharina*) starts to grow already in January while other species start to grow in March (Sjötun and Gunnarsson 1995; Karl Gunnarsson and Agnar Ingólfsson 1995). The maximum growth rate is usually in May. During the summer months, when nutrients are at low concentrations, the kelp uses photosynthesis to produce polysaccharides, mainly mannitol and laminarin, *i.e.* compounds deficient in nitrogen and phosphorus. Thus, the kelp closest to the diffuser might utilise possible additional supply of nutrients during winter but this addition is very limited as discussed above.

Assuming total conversion and yield factors from Tett *et al.* (2003), *i.e.* 1.1 mg Chl *a* per mmol nitrogen and 30 mg Chl *a* per mmol phosphorus, the additional chlorophyll *a* in the far field is at most 1-1.5 mg Chl a/m^3. This amounts to less than 10-15% increase in the average Cla *a* in the far field and much less during the spring blooms, far below the

50% increase proposed by OSPAR (2001). Although these yield factors have been used across Europe (Tett *et al.* 2003), they have not been validated for Icelandic waters.

For answering the question whether the phytoplankton is able to utilise the nutrients and give maximum yield in Chl *a*, an estimate of the effective photosynthetic efficiency is needed as well as information on radiation. Figure 8 shows the measured radiation in Faxaflói (station at Húsafell) showing June to have the maximum PAR (photosynthetically available radiation) of about 21 µE m^{-2} s^{-1} on average. This maximum radiation is reflected by 6% by the water surface (*albedo*) and further attenuated by Chl *a* and turbidity in the water column, resulting in a total of 80% average decrease in PAR at the surface and the euphotic zone of 20 m (using a radiation attenuation factor by k=1.7/Ds, Ds being Secchi depth, and data on relationship between Secchi depths and Chl *a* in Icelandic waters from Kristinn Guðmundsson and Kristín J. Valsdóttir (2004) and Kristinn Guðmundsson *et al.* (2004). Further attenuation occurs as the light penetrates the surface, somtimes called the hyperexponential decrease, due to different penetration efficiency of some wavelengths. This subsurface decrease is commonly 35-40% (Tett *et al.* 2003) but is neglected here for a conservative estimate. Using the effective photosynthetic efficiency in Tett *et al.* (2003) of 0.006 d^{-1} (µE m^{-2} s^{-1})$^{-1}$, the photosynthetic growth rate is 0.001 h^{-1}, far less than the exchange rate in the area by the residual current alone, 0.012 h^{-1}. Therefore, the phytoplankton is only able to use a small fraction of the additional nutrients discharged by the STP at Ánanaust to build up biomass, provided the present form of the CSTT-model is valid and that the effective photosynthetic rate holds true for Icelandic waters in which it has not yet been validated.

From these results it may be concluded that secondary treatment of the sewage, decreasing nitrogen levels by 10% at best, provides little extra benefit and the present primary treatment alone will not result in a measurable increase in primary production nor a decrease in oxygen saturation in the far field. Therefore, the present primary treatment is suitable for the STP at Ánanaust under Article 6 of the UWWTD (C.E.C. 1991).

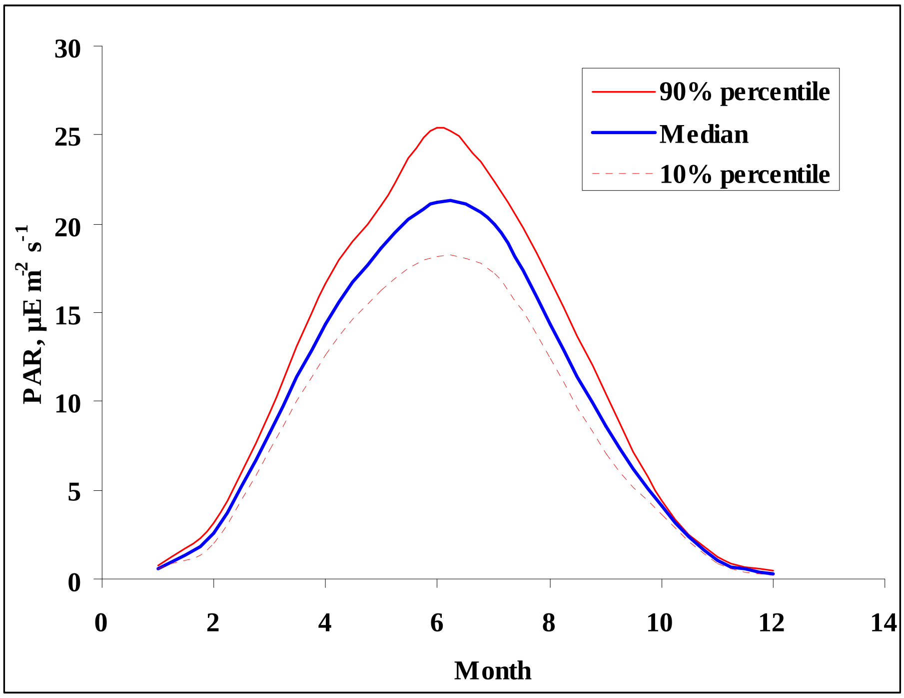


Figure 8. Median photosynthetically available radiation (PAR) in Faxaflói during 1990-2002 with 10 and 90% percentiles for each month. PAR is calculated from data on total radiation measured by the Meteorological Office of Iceland using a conversion factor of 0.455 for the latitude of Faxaflói (http://www.atmos.umd.edu/~srb/par/04status.htm) and 1W m^{-2} PAR = 4.15 µE m^{-2} s^{-1} (valid for 500 nm).

The distribution of faecal coliforms is discussed with in chapter 3 and suspended solids are described in chapter 5.

Long range effects

One may use a simple tidal prism model to evaluate the dilution and flushing times (or turnover and half life) (*e.g.* Luketina 1998) in a larger area, here the Kollafjörður. Kollafjörður was defined above as the area inside the line from Grótta to Kjalarnes with an area of about 83 km^2, volume at spring tide ebb about 1.4 km^3 and an average depth of 17 m.

Total freshwater inflow into Kollafjörður is about 10 m^3/s, mainly from Elliðaár (4.6 m^3/s, average of 48 years), Leirvogsá (about 3.3 m^3/s) and Úlfarsá (1.5 m^3/s) (National Energy Authority, Hydraulic Laboratory 2006). Assuming the concentrations of nutrients in all these rivers to be the same as in Elliðaár (winter time) (Sigurður Reynir Gíslason *et al.* 1998), freshwater contributes 20 kg/d and 6 kg/d of dissolved nitrate-N and dissolved phosphate-P, respectively, to Kollafjörður. The total amounts of nitrogen and phosphorus are somewhat higher but these discharges are, however, small compared with the discharges of the STPs at Ánanaust and Klettagarðar. In 2004, the discharge from both STPs was from about 200,000 p.e. amounting to 3,400 kg/d of nitrogen and 320 kg/d of phosphorus. In the future, the total discharge from these STPs is expected to be twice as much or 400,000 p.e. in total.

Tidal prism models are very simplified picture of an estuary (here the sewage effluent is the "river") where the main simplification is the assumption of very well mixed waters vertically and horizontally during the whole tidal cycle. The models provide a first estimate of possible effects, and as such have proven a useful management tool. They are frequently used when there is a lack of comprehensive data on currents and oceanographic parameters and due to their simplicity which enables the user to get an understanding of the major physical processes at work. If a return factor of 0.5 is assumed as proposed by USEPA in absence of any data (USEPA 1985 as cited in Luketina 1998), the flushing time of Kollafjörður is about 9-10 days on average. Additional flushing by residual currents is estimated to be roughly of the same magnitude and the total flushing time of Kollafjörður therefore about 5 days. Under these conditions, the increase in nitrogen in the Kollafjörður would be about 0.9 µM and phosphorus about 0.04 µM from 200 thousand p.e. but 1.7 and 0.075 µM, respectively, from the future 400 thousand person equivalents. These latter values represent 10-15% increase in winter nitrate nitrogen and less than 10% increase in winter phosphorus values of Faxaflói, both well below the 50% increase proposed by OSPAR (2001) to warrant caution for increase in trophic status of the area. Nitrogen is the limiting nutrient in Faxaflói and other Icelandic waters (Sólveig R. Ólafsdóttir 2006; Unnsteinn Stefánsson and Jón Ólafsson 1991) and therefore the sewage nutrient that might affect the trophic status of the waters. The maximum increase in Chl *a* is about 1 mg/m^3 if phytoplankton could utilise this increase in nitrogen level. However, the estimated water exchange rate in Kollafjörður, about 0.01 h^{-1}, is greater than the growth rate of phytoplankton under the CSTT-assumptions above, about 0.001 h^{-1}, and therefore the phytoplankton is only able to utilise a fraction of these nutrients for growth within Kollafjörður.

With the estimated exchange rate of Kollafjörður, and neglecting nitrification as well as oxygen uptake from air and oxygen production during photosynthesis, the oxygen decrease will only be about 0.2 mg/L on average when sewage from 400 thousand person equivalent are discharged into Kollafjörður Bay or roughly 2% of the natural oxygen levels in the area. This is well below the criterium set by CSTT (1997), 0.5 mg/L with oxygen above 7 mg/L.

As regards nutrients and oxygen levels, secondary treatment of sewage where the nitrogen decrease is less than 10% will not result in any environmental benefits greater than those already offered by the present primary treatment.

Source	TN* (tons/day)	TP* (tons/day)	COD_{Cr} (tons/day)
All rivers into Faxaflói	1.6	0.12	78
Hvítá in Borgarfjörður	0.9	0.07	52
Elliðaár, Reykjavík	0.02	0.006	0.8
400,000 p.e.**	6.8	0.64	54

*In case of rivers these values pertain to nutrients in dissolved state which is ≤ total amount used for sewage. **Untreated sewage.

Finally, the whole of Faxaflói Bay may be considered. The following table gives estimates of the total discharge into the bay assuming the number of person equivalents from the Ánanaust STP will be 150,000 p.e. and 250,000 p.e. from the Klettagarðar STP or about 400,000 p.e. in total. The contribution from sewage is estimated from the *per capita* values given in chapter 4 while contribution from all rivers is estimated from the composition of Hvítá in Borgarfjörður in the years 1973-1974 (Sigurjón Rist 1986) and total direct flow of freshwater into Faxaflói in the years 1966 to 1967 from Unnsteinn Stefánsson and Guðmundur Guðmundsson (1978). The contribution from Elliðaár in TN and TP are estimated from its winter values (Sigurður Reynir Gíslason *et al.* 1998). COD in Hvítá is derived from the permanganate number by asuming $COD_{Mn} \approx 0.4 \times COD_{Cr}$ but obviously the organic matter of rivers and sewage is of different nature.

As discussed above (1.5), the inflow of freshwater by rivers of Southern Iceland into Faxaflói is of the same order as direct inflow by rivers into the bay, which thereby roughly doubles the contribution of freshwater in the input of nutrients by rivers into Faxaflói. From the table above, it is seen that the input of the nutrients nitrogen and phosphorus released by the expected 400,000 person equivalents from Ánanaust and Klettagarðar STPs, is about 4 to 5 times the discharge from all rivers that direcly enter the Faxaflói. However, the seawater entering the bay contains nutrients which account for its high primary productivity. Using the estimated flow of seawater into the bay in 1.4 and the winter values of nutrients from Sólveig R. Ólafsdóttir (2006), the amount of nitrate-nitrogen inflow is 800-900 tons/d while the input of phosphate by seawater is 120-150 tons/d, exceeding by far the direct load by rivers and sewage. This input of nitrate is also predicted by the empirical relationship between primary productivity and nitrate input found by Nixon *et al.* (1996) where average primary productivity of Faxaflói is about 300 g C m^{-2} y^{-1} (Þórunn Þórðardóttir 1994). The effects of the input rates of the nutrients nitrate, phosphate and silicate from rivers on their levels in coastal waters of Iceland within the 200 m depth contour have been assessed (Unnsteinn Stefánsson and Jón Ólafsson 1991). According to their assessment, the contribution from rivers is negligible in the case of nitrate and phosphate but the effects of dissolved silicate from rivers are considered to be substantial.

The longest estimate of residence time of freshwater in Faxaflói is about 100 days (Unnsteinn Stefánsson and Guðmundur Guðmundsson 1978). As the volume of the Faxaflói is 245 km^3, the increase in nitrogen and phosphorus caused by sewage discharge from 400,000 p.e. is estimated to be 0.2 µM and 0.08 µM on average, respectively. However, the increase will be smaller since the flushing time is shorter than 100 days most of the time. This increase in nutrients, about 1%, will be impossible to detect experimentally when seawater is analysed. However, the phytoplankton may utilise this addition resulting in increased biomass since the residence time of water in Faxaflói is longer than the growth rate of phytoplankton under the conditions of the CSTT-model (Tett *et al.* 2003). The relative increase in biomass will be extremely small, however.

The residence time of sewage from 50,000 p.e. that was released 500 m from the shore at the STP at Ánanaust on 28/11/1995, was estimated to be 15-22 hours and this release of sewage had no effect on the oxygen-saturation of the seawater (Jón Ólafsson *et al.* 1996).

Additionally, on a transect along the piplene's position from Ánanaust 8/11/1996, when the release of sewage was about one fourth of that on 28/11/1995, the undersaturation of oxygen next to shore was only about 3% as compared with a station at 6000 m NW of the STP at Ánanaust (Jón Ólafsson *et al.* 1995). This investigation showed that the affected area or distribution of sewage was best estimated by salinity and silicate but organic matter, either as BOD_5 or COD, was not a good tracer due to fast mixing times and low sensitivity of the methods for BOD_5 and COD (Guðjón Atli Auðunsson 1996).

Finally, it may be noted that in February of both 1991 and 1992, nutrients were analysed on a transect west of Akranes (Jón Ólafsson *et al.* 1994). In short, the results showed that the concentration of nitrate was lowest at small depths and at similar level as seawater at the same salinity south of the Reykjanes peninsula. It was therefore concluded that the anthropogenic influence was very small on this transect or that dilution due to mixing of sewage made that source undetectable.

From the discussion above, it is clear that even in the near field, the risk of eutrophication (increase in microplankton growth) or reduced oxygen levels (due to total organic matter, BOD or COD) is not at hand in the recipient of the present primary treated sewage from Reykjavík and neighbouring communities. This holds true for the future discharge of up to 400,000 person equivalents. Secondary treatment reduces nitrogen only to a small degree, the limiting nutrient in the receiving waters. Therefore, secondary treatment will only result in very limited, if any, environmental benefit as regards eutrophication potential in the recipient.

The fate of suspended solids will be described in chapter 5 while the possible effects of contaminants in the sewage on sediments and biota are discussed in chapters 5, 6 and 7.

3. HYDRODYNAMIC RESEARCH AND DISPERSAL OF FAECAL BACTERIA.

3.1 Introduction.

According to the water quality standards in Iceland, the following two conditions need to be fulfilled regarding faecal coliforms at sewage disposal sites.

1. The number of faecal coliforms or enterococci outside the dilution area shall during at least 90% of the time be under 1000 per 100mL for a minimum of 10 samples.

2. Where recreational areas are located at the coastline or food processing plants near the shore, the number of faecal coliforms or enterococci, shall during 90% of the time be under 100 per 100mL ouside the dilution areas for a minimum of 10 samples.

These requirements are additional to the ones found in the Council directive 91/271/EEC (C.E.C. 1991) concerning urban waste-water treatment which is also in force in Iceland with some minor amendments.

In order to calculate the concentration of faecal coliforms and other pollutants from sewage outlets in the sea north of Reykjavík, Iceland, a numerical model of surface water flow and transport was constructed. The optimal length of the sewage outlets was determined so that calculated concentrations of pollutants fulfilled water quality standards. Calculated results were compared with measured data in order to calibrate the model. The calibration involved determining the following parameters:

1. Current speed and direction in a large area surrounding the proposed outlets.
2. Dispersion coefficients of pollutants in the sea.
3. Sewage flow from the outlets.
4. Concentration of faecal coliforms in the sewage.

5. Concentration of other pollutants in the sewage.
6. Lifetime of faecal coliforms in the sea.

In order to determine the above-mentioned factors, extensive measurements were required, and are discussed in detail in the following sections.

3.2 Flow model

The flow model covers all of Faxaflói and takes into account tidal and wind-induced currents in the sea. The coastline from Sandgerði on the Reykjanes peninsula to Malarrif on the Snæfellsnes peninsula delineates the inner model boundary. The outer boundary is defined by a straight line across Faxaflói from Malarrif to Sandgerði. The boundary conditions at the outer boundary are estimated by extrapolating measured tidal elevation in Reykjavík harbour. The tidal amplitude and phase were adjusted so that measured and calculated tidal elevation were in agreement. The results of these adjustments were that the amplitude on the boundary was 93% of the measured amplitude in Reykjavík harbour and the phase shift across Faxaflóí was approximately 15 minutes.

The flow model was calibrated against current measurements taken in Faxaflói between 1967 and 2000. The first current measurements were made in Fossvogur (creek between Reykjavík and Kópavogur, see map 2) and Skerjafjörður by the National Energy Authority's Hydraulic Laboratory for the City of Reykjavík's Environmental and Technical Sector (National Energy Authority 1967). In 1970, the Danish company Isotopcentralen compiled and carried out extensive measurements for the City of Reykjavík's Environmental and Technical Sector in Skerjafjörður and in the sea north of Reykjavík. Among these were current measurements at five locations made by the Marine Research Institute (Isotopcentralen, 1971). Current measurements were made at two locations in 1989 for the City of Reykjavík's Environmental and Technical Sector (Vatnaskil Consulting Engineers 1989). In addition, current measurements were performed at high tide under Gullinbrú bridge (across Grafarvogur) as well as off the isthmus of Geldinganes. The most extensive current measurements in the area were supervised by Vatnaskil Consulting Engineers for the City of Reykjavík's Environmental and Technical Sector. The actual measurements were performed by the Marine Research Institute and the National Energy Authority during the summer of 1994 (Vatnaskil Consulting Engineers 1994) and included:

1. Measurements of tidal elevation in Reykjavík harbour.
2. Continuous current measurements at the end of the proposed Ánanaust sewage outlet, 4 kilometers from the coast. Currents were recorded for approximately two months.
3. Current measurements at different depths along a line from Bygggarðsboði to Gunnunes, made up of segments from Bygggarðsboði to Akurey, Akurey to Engey, Engey to Viðey, Viðey to Geldinganes and finally Geldinganes to Gunnunes. Measurements were recorded over a period of 12 hours at low and high tides.
4. Current measurements at different depths along the length of the proposed Ánanaust outlet pipe. Measurements were recorded over a period of 12 hours at low and high tides.
5. Current measurements at different depths at the end of the proposed Ánanaust sewage outlet. These measurements were taken for comparison with the continuous measurements described in number 2 above. Measurements were taken over a period of six hours.
6. Wind speed and direction measurements taken from a two-meter high mast in Akurey during the same time period as the continuous current measurements.

In 1998 during the design of the Klettagarðar sewage outlet, it was decided to perform additional current measurements in the sea north of Reykjavík (Vatnaskil Consulting

Engineers 1999). Current flow was measured along three cross sections (Table 3.1) and current speed was measured at different depths at the end of the proposed outlet pipe.

Tavle 3.1 Current flow measurements

Cross section	Time period	Tides
Skarfaklettur-Viðey	28-29 Oct '98	Low
Viðey-Engey	28-29 Oct '98	Low
Engey-Laugarnes	28-29 Oct '98	Low
Skarfaklettur-Viðey	3-4 Dec '98	High
Viðey-Engey	3-4 Dec '98	High
Engey-Laugarnes	3-4 Dec '98	High

Similar current measurements have been conducted in Hafnarfjörður in association with the design of sewage outlets. The Marine Research Institute performed current measurements at four locations in Hafnarfjörður: off Garðar on Álftanes, Langeyri, Hvaleyrarhöfði and near Helgasker (west of Hvaleyrarhöfði) (Vatnaskil Consulting Engineers 1990). In 1994 the National Energy Authority performed current measurements for the city of Hafnarfjörður. Current measurements were taken at different depths along a line from Hvaleyrarhöfði to Helgasker and then back to the mainland at Álftanes. Measurements were made over a 12-hour period during low and high tides (Vatnaskil Consulting Engineers 1995). Finally, current measurements were made by the Marine Research Institute at four locations near Keflavík and Njarðvík in connection with sewage disposal studies in the area (Vatnaskil Consulting Engineers 1993).

All of the above-mentioned current measurements have in one way or another been used to calibrate the flow model. In order to calibrate current speed and direction, both the bottom friction coefficient (C) and the wind shear stress coefficient (C_D) were adjusted. The following values gave the best results:

$C = 20D^{1/6}$ where D is the average depth

$C_D = 8 \cdot 10^{-4}$

Both values are within published limits. As can be seen, the flow model has been calibrated against extensive current measurements in the sea north of Reykjavík.

3.3 Transport model.

As mentioned previously, in order to calibrate the transport model it was necessary to determine the dispersion coefficients of pollutants in the sea, volume of sewage flow, concentration of faecal coliforms and other pollutants in the sewage and lifetime of faecal coliforms in the sea.

Dispersion coefficients of pollutants were determined by fitting measured and calculated pollutant concentrations in the sea. The first measurements of pollutant concentrations were made by the National Energy Authority's Hydraulic Laboratory in 1967. A colored dye was released into the sea in Skerjafjörður and its dispersion measured. In 1970 Isotopcentralen conducted similar experiments (Isotopcentralen 1971) using radioactive tracer. In 1995 the Marine Research Institute conducted extensive measurements of nutrients in the sea near Ánanaust (Jón Ólafsson *et al.* 1996). These measurements were primarily used for the calibration of the dispersion coefficients. Measured and calculated concentrations fit best with dispersion coefficients of 10 m^2/s in the current direction and 5 m^2/s perpendicular to the current direction.

In order to determine the amount of sewage flow expected from the outlet, the City of Reykjavík's Environmental and Technical Sector performed flow measurements at the following four stations:

1. Pumping station at Laugalækur
2. Pumping station at Ingólfsgarður
3. Pumping station at Ánanaust
4. Manhole at Neðra-Breiðholt

The results of the measurements showed that the sewage flow, not including rainwater or heating water, was approximately 275 liters/person/day (Vatnaskil Consulting Engineers 1991, 1992, and 1996).

The concentration of faecal coliforms in the sewage was measured at the four above-mentioned stations during the same time as the flow measurements. The average daily concentration was approximately nine million faecal coliforms per 100 mL sample. Concentrations of other pollutants were also measured at the four locations.

The lifetime of faecal coliforms in the sea was determined by fitting measured and calculated concentrations. The first measurements of faecal coliforms were performed by Isotopcentralen (Isotopcentralen 1971). Measurements were later performed by the Icelandic Fisheries Laboratory (1991) in the sea north of Reykjavík. Several measurements were taken in Grafarvogur in 1984 (Vatnaskil Consulting Engineers 1984). The most extensive measurements of faecal coliforms concentration were taken in the sea near Laugarnes, Ingólfsgarður and Ánanaust in conjunction with the above-mentioned measurements of flow at the pumping stations (Vatnaskil Consulting Engineers 1991, 1992, and 1996). The results showed that the lifetime of faecal coliforms is longer during winter months when there is less sunlight than during summer months. The lifetime of faecal coliforms, T_{90}, is the time it takes for 90% of the faecal coliforms population to die. Table 3.2 shows the measured T_{90} values on a monthly basis.

Table 3.2 T_{90} values

Month	**T_{90}, hours**
January	9
February	8*
March	5
April	4*
May	3*
June	1*
July	2
August	3*
September	5*
October	8
November	9
December	10

*Estimated

3.4 Model validation

In 1999, Vatnaskil Consulting Engineers published a report for the City of Reykjavík's Environmental and Technical Sector describing the above-mentioned flow and transport model. The report included calculations of dispersion of pollutants from the current and proposed outlets in Ánanaust and Klettagarðar, respectively. The concentration of faecal coliforms, biochemical oxygen demand (BOD), chemical oxygen demand (COD), nitrogen, phosphorous and suspended particles have been calculated. Figure 9 shows the calculated faecal coliforms concentration from 130,000 p.e. from each outlet.

A monitoring programme was developed for the Ánanaust outlet where concentrations of faecal coliforms and other pollutants would be measured at determined intervals in order to compare with concentrations predicted by the transport model, and therefore test its validity. The first set of measurements were taken in February 2000 at 31 locations around the end of the Ánanaust outlet. A similar monitoring schedule was developed for the Klettagarðar outlet after its construction.

Additional measurements of concentration of faecal coliforms were performed on April 13, 2000 at 35 locations covering a more extensive area around the end of the Ánanaust outlet (Vatnaskil Consulting Engineers 2000).

The results show that the calculated concentrations of faecal coliforms compare well with measured concentrations, taking into account uncertainties in both the measurements and calculations.

3.5 Summary and conclusions

The flow and transport models have been calibrated against extensive measurements spanning a 30-year time period. The results show that the Ánanaust and Klettagarðar outlets shown in Figure 9 fulfill all of the demands of the water quality standards. In summary, the measurements and calculations show the following:

1. The measured net current is 1-5 cm/sec and flows in an easterly direction at the proposed outlet sites. The direction changes farther out into Faxaflói as depth increases.

2. Calculated current speed and direction compare well with measurements. The current at the proposed outlet sites is roughly 5 to 30 cm/sec in an east-west direction.

3. The proposed outlets satisfy the demands of the water quality standards for concentration of faecal coliforms.

4. With respect to nutrients and organic matter, the dilution zone is defined as an area confined by 50 meters distance in all directions around the diffuser. Within this dilution zone, the excess concentration resulting from the discharge has reached a small percentage increase in the background concentration in the sea. Considerable water exchange takes place at the location of the outlets.

5. The main conclusion of these studies is that the recipient north of Reykjavík can be classified as less sensitive as defined by the water quality standards.

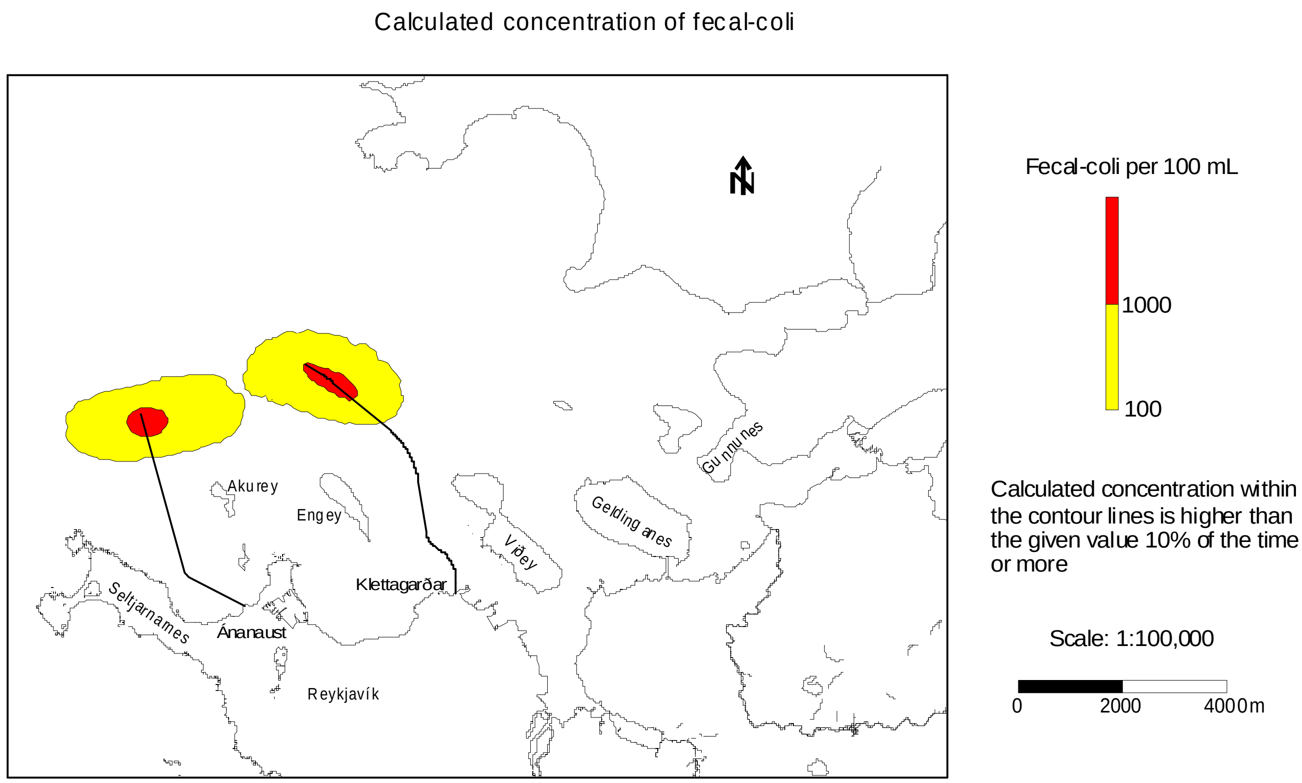


Figure 9 Calculated dispersal of faecal coliforms at the disposal sites for 130,000 p.e. from each outlet.

4. CHEMICAL COMPOSITION OF THE SEWAGE IN REYKJAVIK

The composition and behaviour of sewage in Reykjavik have been studied extensively whereupon a good baseline has been established for later monitoring as well as a base for decisison-making regarding sewage treatment and for the impact assessment of the sewage disposal into the recipient (Guðjón Atli Auðunsson 1992, 1994a, 1996, 2000, and 2002).

4.1 Main characteristics of the sewage in Reykjavík

The main characteristic of sewage in the pumping stations of Reykjavik is its great dilution. This dilution is both diurnal due to daily activities of humans and strongly seasonal which mainly relates to hydrothermal water used for space heating. Most of this water ends up in the sewage, either directly in composite systems or after its heat has been used up in dual systems (Guðjón Atli Auðunsson 2000). At the Ánanaust STP, the dilution is about two times in summer while it is up to five times in winter (Guðjón Atli Auðunsson 2000 and 2002). This dilution results in relatively low concentrations of the various chemical constituents but the composition of the diluting water may also affect the composition of especially ash content and the content of trace elements in the sewage. The dilution also results in short residence times in the sewage system.

Other peculiarities of the sewage is that the input of hydrothermal water adds to the silicate content and results in relatively high pH and temperature.

Studies on the treatment of sewage in the catchment area of the Ánanaust STP and the plant itself, showed that the cleanup of organic matter (COD_{Cr}) was 20% and 15% with respect to suspended matter (Guðjón Atli Auðunsson 2000). The same treatment is applied in the Klettagarðar STP.

4.2 Main chemical constituents of the sewage

Per capita values of the main nutrients of untreated sewage are shown in the table below.

Constituent	*Per capita* value g/p.e./ d	Source of information
Biological oxygen demand, BOD_5	60	Definition of p.e. (used to calculate *per capita* values)
Chemical oxygen demand, COD_{Cr}	135	Evaluated from the definition of p.e. and direct experiments in Skolpa (2.25xBOD_5)
Total nitrogen, TN	17	Faxaskjól, Nov. 1995, 50,000pe. [March, July and Oct. 1991: Breiðholt (3.8 thous. pe): 35; Laugarnes (62 thous. pe): 25; Ingólfsgarður (10.9 thous. pe): 40. Gelgjutangi Sept. 1993 (8.5 thous. pe): 97 g/pe d]
Total phosphorus, TP	1.6	Faxaskjól Nov. 1995, 50 thous. pe. [Breiðholt (3.8 thous. pe) : 2.8; Laugarnes (62 thous. pe) : 1.4; Ingólfsgarður (10.9 thous. pe) : 2.2. Gelgjutangi Sept. 1993 (8.5 thous. pe): 1.9 g/p.e./d]
Total volatile nitrogen, TVN	6.3	Faxaskjól Nov. 1995: 50,000 p.e. [Breiðholt March, July and Oct. 1991 (3.8 thous. p.e.) and Gelgjutangi Sept. 1993 (8.5 thous. p.e.): 13g/p.e./d]
Organic matter (TS-ash)	125	Breiðholt March, July and Oct. 1991, Gelgjutangi Sept 1993.
Suspended solids, SS (<1.6µm)	65	Breiðholt March, July and Oct. 1991, Gelgjutangi Sept 1993.

The *per capita* volume in the STP at Ánanaust has been found to be 270 L/p.e./d, a value in agreement with estimated usage of cold (150 L/p.e./d) and warm water (120 L/p.e./d) for household purposes in Reykjavík, *i.e.* excluding warm water for space heating (Guðjón Atli Auðunsson 2000 and 2002).

Extensive analyses have been carried out for faecal coliforms in sewage (Vatnaskil 1991 and 1996). In most of these samples COD and/or BOD were analysed at the same time (Guðjón Atli Auðunsson 1992 and 1996). From these measurements, altogether 119 pairs, a good *per capita* value of faecal coliforms could be estimated and found to be 1.3×10^{10} MPN/p.e./d with 10 and 90% percentiles being 0.5×10^{10} and 3.8×10^{10} MPN/p.e./d, respectively. This value agrees very well with a *per capita* value calculated from data for the STP of Hveragerði 2004 (averages of monthly sampling) (Hveragerðisbær og Háskólasetrið í Hveragerði 2004).

4.3 Trace elements

The total concentrations of trace elements were determined in the sewage of the pump stations of Ingólfsgarður and Laugarnes in addition to a manhole in Breiðholt in March, July and October 1991 (at two-hour intervals for 24 hours each time), but organic micropollutants were also analysed in the samples from Laugarnes (Guðjón Atli Auðunsson 1992). Analyses of the total trace element concentrations were also made on 24 hour composite samples of sewage from the pump station at Gelgjutangi in September 1993 (Guðjón Atli Auðunsson 1994a) and a few samples from the STP of Ánanaust 2000 (Guðjón Atli Auðunsson 2005 and 2006).

Per capita values

All the measurements of trace elements in the sewage of Reykjavík are summarised as *per capita* estimates in the table below.

It was noteworthy that in the 1991 investigation that the *per capita* values of the trace elements (copper, lead, cadmium, zinc, arsenic, and mercury) in all three stations were proportional to the *per capita* values of water volumes. This, in addition to the fact that there was not a significant difference in concentrations in these three stations at the three different seasons, renders the hypothesis reasonable that the source of the trace elements is not solely from households but rather the water itself after transport by the distribution system, weathering of soils and drainage from roofs and streets. In this context, it may be mentioned that hydrothermal water may be quite corrosive for metal pipes.

Trace element	*Per capita* value, mg/p.e./d
Iron (Fe)	2200
Selenium (Se)	0.2
Chromium (Cr)	5
Silver (Ag)	1.3
Arsenic (As)	<2
Zinc (Zn)	85
Nickel (Ni)	1-1.5
Copper (Cu)	12
Lead (Pb)	8
Mercury (Hg)	0.15
Manganese (Mn)	60
Cadmium	0.2

In the examination of sewage water in the pump station at Gelgjutangi in 1993 (copper, lead, cadmium, zinc, arsenic, mercury, chromium, nickel, silver, iron, and manganese) all *per capita* values were found to be lower than the lowest in the 1991 investigation (Breiðholt). Only copper was found in comparable concentrations. Noticeable were much lower *per capita* values of cadmium and especially lead.
The *per capita* values for inorganic trace elements in Reykjavik are high when compared with sewage water in Europe, particularly in the case of zinc. Even though the *per capita* value of zinc was lower in the examination at Gelgjutangi 1991, it is nonetheless higher in Reykjavik than in most other places. The reason for this high *per capita* value for zinc may be due to extensive usage of galvanised and corrugated roof and gutter materials in Reykjavik in addition to galvanised pipes for water, lamp posts, car parts *etc*. The high flow of water and the potential relationship between *per capita* volumes and trace elements is a possible explanation of these relatively high *per capita* values.

Concentrations

The levels of trace elements in the sewage water are very low and one has to compare them to maximum permissible levels in drinking water to find similar concentrations. The lead level in sewage water is only about 5% of the permissble level in drinking water while arsenic, chromium cadmium, silver, and copper are about 10% of their maximum levels or guidance concentrations in drinking water. Mercury, however, is at 20% of its maximum level in drinking water and zinc at 50% of its guidance value for drinking water. Only iron and manganese are above their maximum levels in drinking water, ten and three times higher, respectively. The reason for these high levels in sewage water is probably dissolution of the pipes in the water distribution system since for example cast iron contains manganese. No maximum limits or guidance values do exist for these two metals in sewage water.

A comparison with undiluted sewage water in Stockholm 1989 (Stockholm Vatten AB, 1990), where the same trace elements were analysed as at Gelgjutangi (except for arsenic, iron and manganese), reveals lower or equal levels in Reykjavik. Levels are similar (Cd), lower (Cr, Ag, Zn, Ni, Pb, and Hg) or much lower (Cu) than in other Nordic countries (Nordisk Ministerråd 1993). Recent investigations of the sewage in Ánanaust STP and drainage water in Reykjavik (Guðjón Atli Auðunsson, 2005 and 2006) confirm the low levels found in the sewage water in Gelgjutangi. It is worth noting that the level of cadmium in the Gelgjutangi examination is equal to the level of cadmium found in tap water in central Reykjavik in 1986 (Jón Ólafsson 1987) and the level of zinc in the same tap water sample was only one fifth of the zinc concentration found in the sewage water in the pump station at Gelgjutangi. These levels in the tap water stem most probably from the dissolution of pipelines in the water distribution systems. At their sources (both cold and warm water), the levels are, however, 10-1000 times lower than in the sewage water except for mercury which may be at the same level to ten times lower in warm water than in the sewage water. In seawater the levels of these trace elements are 1000 times lower than in sewage water although copper is only at 10-500 times lower level and cadmium 100 times lower.

Finally, the initial 1000-fold dilution of sewage in the near field from the STP at Ánanust, takes all trace elements except silver to lower or much lower levels than found in North-Atlantic seawater.

4.4 Organic micropollutants

PCBs, AOX (Adsorbable organic bound halogens), EOX (Extractable organic bound halogens), total phenols, anionic surfactants (methylene blue active substances, CTAS), and nonionic surfactants (cobaltothiocyanate active substances) were analysed in sewage water from the pump station at Gelgjutangi in 1993 (Guðjón Atli Auðunsson 1994a).

PCBs, AOX, anionic surfactants, and nonionic surfactants were analysed in the sewage water of the pump station at Laugarnes in March, July, and October 1991 (Guðjón Atli Auðunsson 1992).

PCBs All results for PCBs in sewage water of the Gelgjutangi pump station 1993 were below the detection limits or less than 0.05µg/L, which is half the permissible level in drinking water. This means that the *per capita* value is less than 0.06 mg/p.e./d to be compared with 2.1 mg/p.e./d in the pump station at Laugarnes 1991 and about 1.2 mg/p.e./d in Stockholm 1989 (Stockholm Vatten AB, 1990). In the sewage of the Laugarnes pump station, the levels of PCBs were found to be from being less than 0.05 µg/L up to 6.6 µg/L. From these results it seems that the sources of PCBs were related to the older districts of Reykjavík rather than industrial areas of Reykjavik.

Research on the possible sources of PCBs was carried out in 2000. This showed very low levels in the sewage water of the Ánanaust STP on a dry summer day (below or a slightly above the detection limit, which now was ten times lower than in 1991-1993 or 0.005 µg/L). The level of PCBs had fallen dramatically in the sewage water of the pump station at Laugarnes (Guðjón Atli Auðunsson 2001a, 2005 and 2006). The *per capita* value of PCB153, the congener usually highest of the PCBs in nature, was less than 5 µg/p.e./d (Guðjón Atli Auðunsson 2006).

AOX and EOX AOX-substances in the pump station at Gelgjutangi 1993 were on average at a level of 51±23 µg/L (from 18 to 90 µg/L). This is a similar concentration as may be found in ground water around Stockholm (Stockholm Vatten AB, 1990), but twice the level found in sewage water from the pump station at Laugarnes in 1991, which was in the interval from <10 µg/L to 59 µg/L. However, the *per capita* value of AOX at Gelgjutangi is little less than 60 mg/p.e./d which is twice the *per capita* value in Laugarnes 1991, 27 mg/p.e./d, and Stockholm 1989, 25 mg/p.e./d. The level of AOX in sewage water in Stockholm is twice the ground water level in that area and its source accounted for by bleaching of clothes with chlorine-solutions.

The EOX-analysis gave a weekly average of 22±15 µg/L (from <10 µg/L to 47 µg/L) at Gelgjutangi, about half of the AOX, which is a relatively high ratio since EOX is usually in the range of 2-20% of AOX. The twofold levels of *per capita* values at Gelgjutangi compared with Laugarnes indicate that the source of these substances at Gelgjutangi may be for example organic solvents from industry. No further studies have been carried out to elucidate the nature of the compounds included in the EOX but there is reason to believe that EOX-compounds may indeed accumulate in organisms. However, the use of AOX in environmental science has been questioned in a recent review on this summary parameter (Müller 2003).

Anionic surfactants Anionic surfactants in the pump station at Gelgjutangi ranged from 0.5 to 1.0 mg/L, which is ten times the levels found in Laugarnes in 1991, 0.09-0.4 mg/L. The *per capita* value, 800 mg/p.e./d, is furthermore nearly ten times the *per capita* value in Laugarnes, 75-100 mg/p.e./d, indicating industrial sources. The *per capita* value in Stockholm 1989 was, however, higher than the *per capita* value at Gelgjutangi or 1200-2300 mg/p.e./d. EU (80/778/EC) has a maximum level of 0.2 mg/L in drinking water.

Nonionic surfactants Nonionic surfactants in the sewage water of the pump station at Gelgjutangi ranged from 0.2 to 0.8 mg/L, resulting in a *per capita* value of 0.5 g/p.e./d. Both levels and *per capita* values are about six times higher than those detected in sewage water in the pump station at Laugarnes in 1991. Similar levels for nonionic and anionic surfactants in the sewage waters are in line with their production in Iceland and because of their use in car washing for example it comes as no surprise that their levels are higher in Gelgjutangi than Laugarnes.

Phenols The total amount of phenols in the sewage of the Gelgjutangi pump station was in the range <5 µg/L to 24 µg/L, which is 25 times the maximum EU level for drinking water. Individual chlorophenols in domestic sewage in Stockholm 1989 were in the range 0.01 to 0.45 µg/L and therefore much higher total concentrations. The EU has much higher permissible maximum levels for some specific industries (Commission Directive 86/280/EC of 12 July 1986). Thus. the monthly average of pentachlorophenol may be up to 1000 µg/L in discharge water from the production of sodium pentachlorophenol and a daily average of 2000 µg/L. For pentachlorophenol, the EU has a quality target according to this Directive, which for surface waters, estuarine waters, and coastal waters is less than 2 µg/L. A high ratio of EOX to AOX calls attention to the nature of the phenols that comprise the total phenol content here analysed, where it is especially important to examine whether pentachlorophenols are present.

5. SUSPENDED SOLIDS AND SEDIMENT TRANSPORT: BEDLOAD AND SUSPENSION LOAD

5.1 Introduction

Suspended matter behaves differently in seawater than dissolved constituents of the sewage. However, models for predicting their fate are less well established than those for microbes and dissolved compunds although models for suspended solids are obtaining more attention lately, especially due to the emerging problems associated with the heavy load of organic solids from aquaculture (Cromey *et al.* 1998 and 2002). Predictions of the fate of suspended solids involve all the problems of dissolved pollutant modelling but with the added problems of variable particle sizes, density of particles, colloidal and flocculant properties, and response of bottom sediment and benthic infauna to settling particles (nutrients and pollutants). Additionally, the fate of the suspended matter is also dependent on the shear stress on the seabed by currents slightly above the bottom (0.5-1m), the bed form and type, the extent of consolidation with natural sediments and the ability of water currents to resuspend and erode the sediments at high tidal velocities and storm events. Modelling of the fate of suspended solids from ocean outfalls have been approached by widely different techniques (Mearns and Word 1982; Koh 1982; Farley 1990; Neves *et al*. 1995 and 2002; CSTT 1997; Cromey *et al.* 1998) but it may be concluded that they are still only in their infancy and modelling has to be accompanied by monitoring and verifying studies.

Chemical nature of suspended solids

Studies on the suspended solids at Ánanaust STP showed that they are about 90% organic matter by weight and their share in the total organic matter in the sewage is about 40% (Guðjón Atli Auðunsson 2000). According to Australian data, the composition of suspended solids is 35% organic carbon, 5% nitrogen, 1% phosphorus, and 5% silicate (Bickford 1996). Assuming the same composition of suspended solids in Reykjavík and using the *per capita* values in 4.2, approximately 20% of the total nitrogen and 40% of the total phosphorus are found in the suspended solids. Thus, primary treatment, which should reduce suspended solids by at least 50% and BOD_5 by at least 20% according to UWWTD (CEC 1991), will reduce nitrogen by only about 10% and phosphorus by about 20%. Thus, sedimentation by gravitation in primary treated sewage will hardly satisfy both these criteria simultaneously since a 20% decrease in COD is accompanied with only about 25-30% reduction in suspended solids (Christoulas *et al.* 1998). However, reduction in suspended solids by 90% in secondary treatment is optional according to the UWWTD (while 70-90% reduction in BOD_5 and/or 75% reduction in COD_{Cr} is required), which would result in 15-20% and 35-40% reduction in nitrogen and phosphorus, respectively. Since nitrogen is the limiting nutrient for algal utilisation as discussed above, its reduction is of concern in the recipient. Finally, it may be mentioned that pollutants in sewage (microbes as well as organic and inorganic micropollutants) are mainly associated with particulate matter in the sewage water.

Effects of suspended solids in the marine environment

As shown in chapter 2, where total amount of organic matter was discussed in relation to possible oxygen depletion, the sewage solids will hardly have any detectable effects in the recipient if they are assumed to be solely in suspension (oxygen depletion was based on total organic matter by way of COD). Even in the near field of the initial dilution, the maximum concentration of COD is well below the amount of 1 mg/L, *i.e.* a level at which experiments show no effects of COD in seawater (Quetin and De Rouville 1986) and a decrease in oxygen will not be detectable. If the solids settle down onto the sediments, however, they may affect both the benthic biota and sediment chemistry. The following consequences may occur, see for example CSTT (1997), most likely interacting but their relative importance will vary according to the nature of discharge and the nature of the receiving environment:

i) Physical: Particles in sufficient quantity may induce changes in species composition or relative abundance as a result of physical interference with *e.g.* feeding or tube building activities.

ii) Physico-chemical: At least in stable areas, a buildup of organic matter within sediments may, through microbial actions, result in increased oxygen demand. Chemical changes associated with anaerobic metabolism may increase the toxicity of sediments, resulting in favouring of more tolerant or adaptable taxa over others.

iii) Chemical: Accumulation within sediments of certain particle-bound contaminants, if present in sufficiently high concentrations in sewage discharge, may induce a toxic response in “sensitive” species.

iv) Organic: Inputs of organic matter may provide a direct food source for certain benthic macrofaunal taxa, which may then be favoured over others.

These effects are often seen by examining the benthic community (CSTT 1997; Mearns and Word 1982) where the guidelines of Pearson and Rosenberg (1978) are used where increased biomass, decreased biodiversity and a shift from the more sensitive suspension-feeding to the more tolerant deposit-feeding benthic communities are among the responses observed with decreased distance from the source of organic matter. These guidelines of Pearson and Rosenberg are best suited for areas of low turbulent energy, whereas areas of high energy like the disposal areas of sewage from Reykjavík, are less amenable to predictions as they include gravelly seabeds and disperse the effluent constituents more effectively. In this regard CSTT (1997) defined adverse effect when primary treatment induces change in community structure at a distance greater than 100 metres from the outlet, effects which would not have occurred had secondary treatment been installed. As a working tool in these investigations, the Infaunal Trophic Index (ITI) is frequently applied. This index is based on four groups of feeders, *i.e.* suspension feeders (group I), surface detritus feeders (group II), surface deposit feeders (group III), and subsurface deposit feeders including the characteristic *Capitella capitata* at high organic loads (group IV) (Mearns and Word 1982). The index takes on values from zero (domination by subsurface deposit-feeding invertebrates) to 100 (domination by suspension-feeders) at unaffected areas. The index has been shown to be inversely proportional to the concentration of organic matter in the sediments (Mearns and Word 1982) and correlate well with the better known Shannon-Wiener index (Cromey *et al.* 1998). Thus, ITI between 80 and 100 indicates unaffected area (suspension-feeders), ITI between 60 and 80 occur near outfalls and non-outfall sites and is still numerically dominated by detrital-feeders while ITIs between 30 and 60 designate moderately affected or "changed" areas (surface deposit-feeding infauna), and below 30 (dominated bu subsurface deposit-feeders) the ITI reflects "degraded" area. Empirical relationships were found between the total emission of suspended matter from ocean outfalls and both the elliptical area affected as well as the total excess standing crop of organisms resulting from the organic load in soft-bottom Californian sites (Mearns and Word 1982). Thus, like all models proposed hitherto, this model applies only to the more sensitive soft-bottom areas in relatively low turbulent areas. Additionally, this very simple empirical model does not take any currents int consideration, and thus must be oversimplistic in spite of astonishingly good correlation for eight outfall sites in southern California. These relationships were advised by the initial CSTT in 1994 (CSTT 1994) to be used for UK-outfalls. However, in its revised form in 1997 it was stated that these relationships had "overpredicted the benthic effects" but that these relationships could still be used to predict the worst case scenario (CSTT 1997). It is worth while to apply these relationships to the Ánanaust site but emission rates of the present primary treatment is similar to the lowest rate found in southern California for which the model was founded. The total area occupied by total excess biomass, areas with ITIs lower than 60 and lower than 30 together with extensions of the areas on both sides of the diffuser, *i.e.* the long axis of the ellipses, are given in the following table.

The table shows that even in this worst case scenario, the model does predict the affected area to extend to less than 100 metres on each side of the diffuser. Therefore, according to UK guidelines, secondary treatment of sewage would not impart any environmental improvement farther away than 100 m from the diffuser and, therefore, primary treatment of sewage is deemed sufficient even at this stage of evidence. Secondly, the area of possible degradation according to this model, *i.e.* ITI<30, would not be observed even in the closest vicinity of the diffuser. Also evident from this model prediction is a much smaller affected area than could be foreseen for dissolved constituents of the sewage, chapter 2.1.

As shown below, even these low values are not at all likely to occur in the recipient of Ánanaust and Klettagarðar.

Person equivalents p.e.	**Total emission of SS tons/year**	**Total excess biomass tons wet weight**	**Total area km^2**	**Biomass density g/m^2**	**Area ITI<60 km^2**	**Long axis ITI<60 m**	**Area ITI<30 km^2**	**Long axis ITI<30 m**
100,000	2373	12.3	0.98	12.6	0.031	31	0.0005	0.5
150,000	3559	25.0	1.44	17.4	0.074	74	0.0014	1.4

Physical behaviour of suspended particles

The physical behaviour of the suspended solids is of considerable importance. Several investigations all over the world have shown that between 10 and 20% of the suspended solids of sewage may settle out of suspension in the water column (Berelson *et al.* 2002; Baker *et al.* 1995; Mearns and Word 1982) while about 80% remain in suspension and about 3% of SS of primary treated sewage may rise to the surface (Baker *et al.* 1995). Therefore, at least about 80% of the suspended solids are transported from the recipient in a similar fashion as dissolved constituents albeit at a somewhat slower rate. However, particulates decay in the water column and CSTT (1997) assumes a decay rate of 0.3 d^{-1} while Farley (1990) assumes 0.05 d^{-1} in cold water and 0.1 d^{-1} in surface water off southern California. The particles that settle are heterogenous in composition and irregularly shaped. Additionally, and more importantly, Baker *et al.* (1995) showed by experiments that 90% of the sinking particles have a diameter less than 80 µm and Lavelle *et al.* (1988) showed experimentally that 80-85% of the particles in sewage have diameters below 63 µm and in UK, more than 90% of particles in primary treated sewage are less than 100µm (Cromey *et al.* 1998). This is of great importance since particles smaller than about 100µm are not transported by bedload if deposited but taken directly into suspension and, when in suspension as pertains to all sewage particles from the jets of the diffuser, they are transported very long distances with very small currents before they are deposited. Current velocities of 1 cm/s or higher are sufficient to keep mineral sediment particles less than 100 µm in suspension (see textbooks on physical sedimentology, *e.g.* Anon. 1999) and to retain them in suspension as long as the current is above 1 cm/s. This holds true for mineral particles which have considerably higher density and thereby greater settling velocities than the suspended solids normally found in sewage. As an example, a mineral particle of 100 µm has a settling velocity of about 10 mm/s (e.g. Anon. 1999) while the highest velocity recorded for sewage particle from untreated and primary treated STP is 2.4-3 mm/s (Lavelle *et al.* 1988; Baker *et al.* 1995). Furthermore, inorganic sediment particles less than 63µm (>80% of sewage particle sizes) only need a current of 0.7 cm/s to stay in suspension and a current of 0.1 cm/s keeps sediment particles less than 20 µm in flowing suspension (Anon. 1999). Therefore, the predictions below assuming all the settlable sewage particles to settle at velocities less than 1 cm/s based on behaviour of mineral sediment particles, is a very conservative approach.

Also of concern regarding sewage particles is their possible flocculation into larger particles with greater sinking rates. Flocculation will not happen, however, if the concentration of suspended solids is below about 7 mg/L (Lavelle *et al.* 1988), a condition occurring instantaneously when the sewage leaves the diffuser from Ánanaust resulting in levels below 1 mg/L. Additionally, the experiments done by Baker *et al.* (1995) were carried out with indiluted sewage and thus giving maximum sinking rates and therefore maximum amounts of solids that might settle to the bottom.

Possible settling of suspended solids at the Ánanaust site

Continous current measurements in the calm period of July and August 1994 showed that speeds lower than 1 cm/s at 1m above the seabed occurred for less than 22% of the time (Guðjón Atli Auðunsson 2005), see figure 10 (current measurements were performed at 15 m above the seabed but calculated for 1 m above the seabed by using the common van Karman function assuming the lowest measureable speed of 1.1 cm/s to be zero at 1 m above the seafloor). This is also manifested by the fact that the seabed is of sand and gravel in a relatively thin sheet above a hard bottom (Kjartan Thors 1994 and 2003) as a result of frequent currents fast enough to remove particles greater than 100 µm from the bottom, *i.e.* currents greater than 10-20 cm/s. This is also borne out by the slow burial rate of organic matter in the area or less than 1 g C m^{-2} y^{-1} as elucidated in

Guðjón Atli Auðunsson (2001b) with a resulting flux of organic matter to the sediments of less than 10 g C m^{-2} y^{-1} in spite of the large primary production in the area or more than 300 g C m^{-2} y^{-1}. Thus, less than 4% of the primary production seems to reach the sediments in the area, the rest is decomposed in the water column and/or advected away which further reveals the high energy of the waters north of Reykjavik.

Settling rates of phytoplankton are in the range of 0.006-0.03mm/s (Heip *et al.* 1995; Wassmann 1991), which is similar to the settling rates of the fine fraction (<63µm) of sewage particles, which is 0.007-0.03 mm s^{-1} but this fraction makes up about 80% of the sewage solids (Lavelle *et al.* 1988). The coarse fraction (>63µm) had a settling velocity in the range 0.6-3.0 mm s^{-1} (Lavelle *et al* 1988; Baker *et al.* 1995) or two orders of magnitude higher than detritus from phytoplankton. The lowest rate found by Baker *et al.* (1995), 0.02 mm/s, is most likely due to aggregation of particles in the undiluted sewage in their experiments.

The data shown in figure 10 can be used to estimate how much of the settling solids (maximum 22% of the total solids (Baker *et al.* 1995)) will have the chance of reaching the seafloor at the Ánanaust site. Only currents are used for these predictions here and not the effects of wave motion. As shown below in chapter 5.3, the latter are of great importance in keeping particles in suspension and especially eroding them from the seabed if they had the chance to settle. Thus, the worst case scenario is once again used for the prediction of the fate of sewage particles.

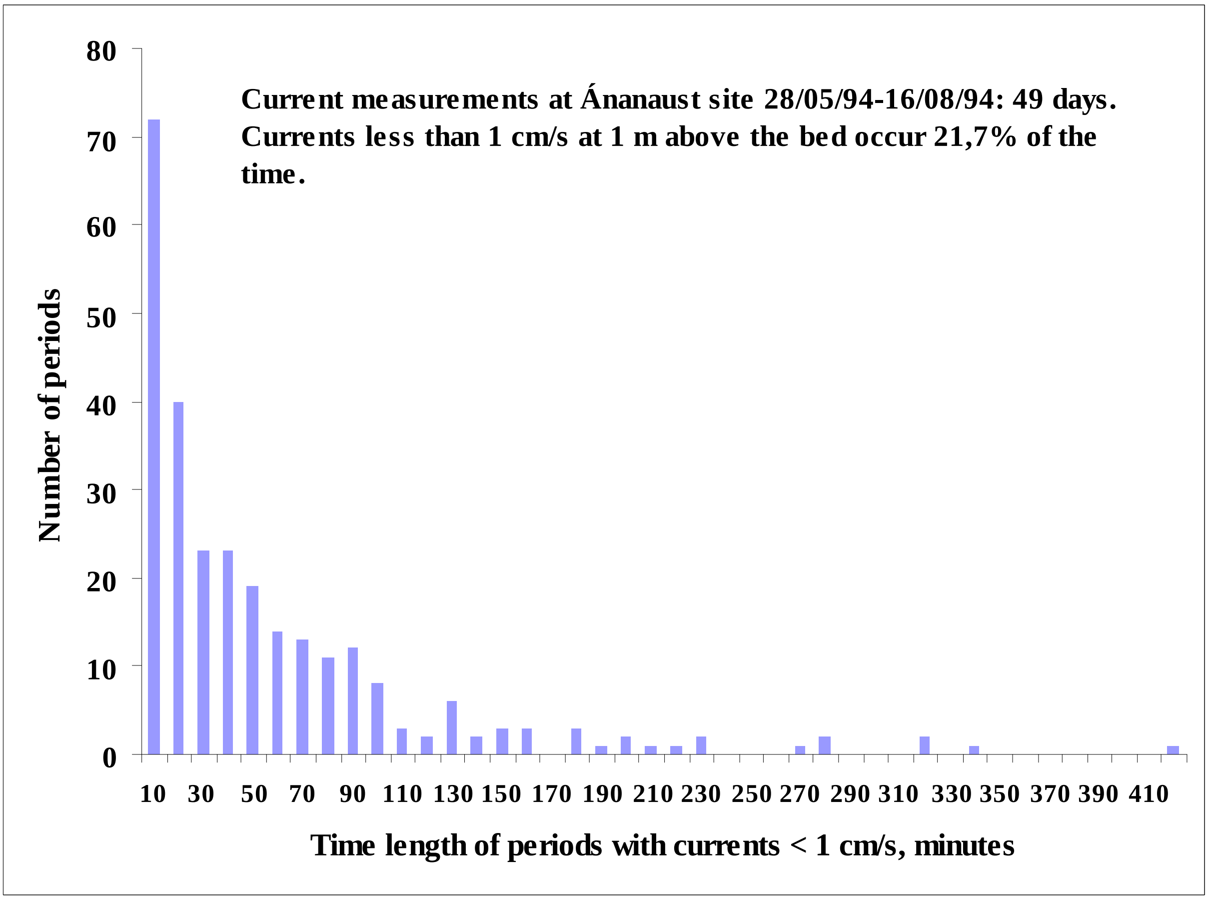


Figure 10 Frequency of time periods with currents less than 1 cm/s at 1 m above bottom at the Ánanaust-disposal site.

Firstly, the height(s) at which solids will start to fall has to be estimated. The density of sewage solids has been estimated to be in the range 1.04-1.5 g/cm^3 (Lavelle *et al.* 1988) and thus both buoyancy and jet momentum flux from the diffuser orifices takes them to heights well above the diffuser (at least to about 8 m for the lighter particles at the worst conditions during stratification in mid-summer (Koh 1982; Neves *et al.* 1995 and 2002)). To be very conservative, a height of 1.5 m above the seabed or 0.3 m above the top surface of the diffuser pipe is assumed. Secondly, the empirical relation between mass fraction of settling sewage solids of primary treated sewage as a function of settling rate in Baker *et al.* (1995) is assumed to be valid in the area. This way one can calculate the maximum mass of solids reaching the seabed. The highest settling rates of sewage solids are between 2.2 and 2.4 mm/s (this applies to about 0.3% of the total solids). Solids can therefore settle out on the occasions when currents less than 1 cm/s prevail for more than 10 minutes. Similarly, the particles of lowest settling rates of sewage solids (0.024-0.028 mm/s) constitute 0.77% of the total solids of the sewage. These would settle to the bottom if currents (much) less than 1 cm/s prevail for more than 890 minutes (which will never occur according to figure 10). Summing up, a maximum of 2.4% of the total solids will have any chance of reaching the bottom even at this worst case scenario at the disposal site of Ánanaust STP. Assuming further an average current speed of 10 cm/s, the absolutely highest carbon flux of 0.1-0.2 g C m^{-2} d^{-1} will occur in a narrow band between 50 and 100 m from the diffuser (once again a worst case scenario). Applying the empirical relationships for on one hand the oxygen demand resulting from this maximum carbon flux to the sediments and on the other hand the diffusion of oxygen into the sediment boundary layer (dependent on currents at 1 m above the bed) as described in Findlay and Watling (1997) for Maine, USA, and further verified in New Zealand (Morrisey *et al.* 2000), it is evident that the current will supply the sediments with much higher oxygen fluxes than required by the settling solids from the sewage at all times. The Findlay-Watling model relates carbon flux to the sediment to both sediment chemistry and effects on benthic biota where the key criterion is that the carbon flux should not exceed the oxygen supply for more than 2 hours since the most sensitive organisms were assumed to be harmed permanently if exposed to reduced oxygen for longer periods. Therefore, according to the Findlay-Watling model elaborated to predict effects from carbon flux into sediments below salmon net-pens, the sewage will not give rise to any organic enrichment in the sediments as it will be decomposed aerobically and further, no effects will be observed in the benthic biota. In other words, the layer of seawater closest to the sediment will not be depleted of oxygen to a degree affecting the benthic fauna.

Another approach is to distribute the maximum amount of solids that could settle to the bottom, *i.e.* 2.4% of the total solids, onto the total area predicted to be affected by the Mearns-Word model above giving about 0.06 g C m^{-2} d^{-1} on average. This is well below the amount of organic matter shown by mesocosm-experiments to give little or no effects on the macrobenthic communities or 0.1 g C m^{-2} d^{-1} while 1 g C m^{-2} d^{-1} enriched the sediment community, and loadings over 1.5 g C m^{-2} d^{-1} produced degraded conditions (Kelly and Nixon 1981; Oviatt *et al.* 1987; Maughan and Oviatt 1993; Cromey *et al.* 1998). These figures are in line with an empirical relationship found by Findlay and Watling (1997) between oxygen demand of the sediments caused by flux of organic matter and carbon flux, where no effects on oxygen saturation above the seabed occurs for carbon fluxes below 0.37 g C m^{-2} d^{-1}.

Erosion or resuspension of settled sewage particles at the Ánanaust site

Finally, as regards predictions of the fate of sewage solids at the Ánanaust site, the periods of currents below 1 cm/s are most of the time followed by maximum currents within two hours, *i.e.* periods of low currents occur at slack water followed by maximum currents at 1-2 hours after after slack water. Figure 11 shows that even if solids might settle, they will most of the time be resuspended or eroded from the sediments again within two hours and transported away. This occurs since the critical resuspension speed for sewage particles has been measured to be 5.0 cm/s (Burt and Turner (1983) as cited in Cromey *et al.* 1998) which will most often follow the periods of minimum current speeds. This resuspension rate for sewage particles is less than half that for corresponding sizes of sediment particles, see discussion on the Hjulström-curve in *e.g.* Anon. (1999). A closer look at the periods of currents less than 1 cm/s shows that the time between them is in 47% of cases less than 50 minutes, and in 42% of cases half a tidal cycle as expected. The period from the onset of currents less than 1 cm/s until the current becomes 5 cm/s at 1 m above the bottom may be considered as the residence time of sewage particles on the seafloor. Figure 12 shows that 97% of the periods are less than 10 hours, 82% are less than 5 hours, and that 50% of the periods are less than two hours. In one instance was there a period of 52 hours estimated residence time ocurring from 02/08/94 (15:00) to 04/08/94 (17:50). Wind speed in this period was on average 2.4 m/s (1.0-3.6 m/s) to be compared with the average of 4.2 m/s for the two months of current measurements in July and August 1994. During this long period, an estimated flux to the sediments is about 0.2 g C m^{-2} d^{-1} from 150 thousand person equivalents assuming the conservative area of 1.4 km^2 from Mearns and Word (1982) to be affected. The calculated average current 1 m above the bottom was 1.34 cm/s during this period. This carbon flux is well below the threshold that might result in any decrease in oxygen levels above the sediments according to the empirical model of Findlay and Watling (1997). Actually, the oxygen supply to the sediments during this longest period is about fifty times larger than the carbon flux requires for aerobic decomposition.

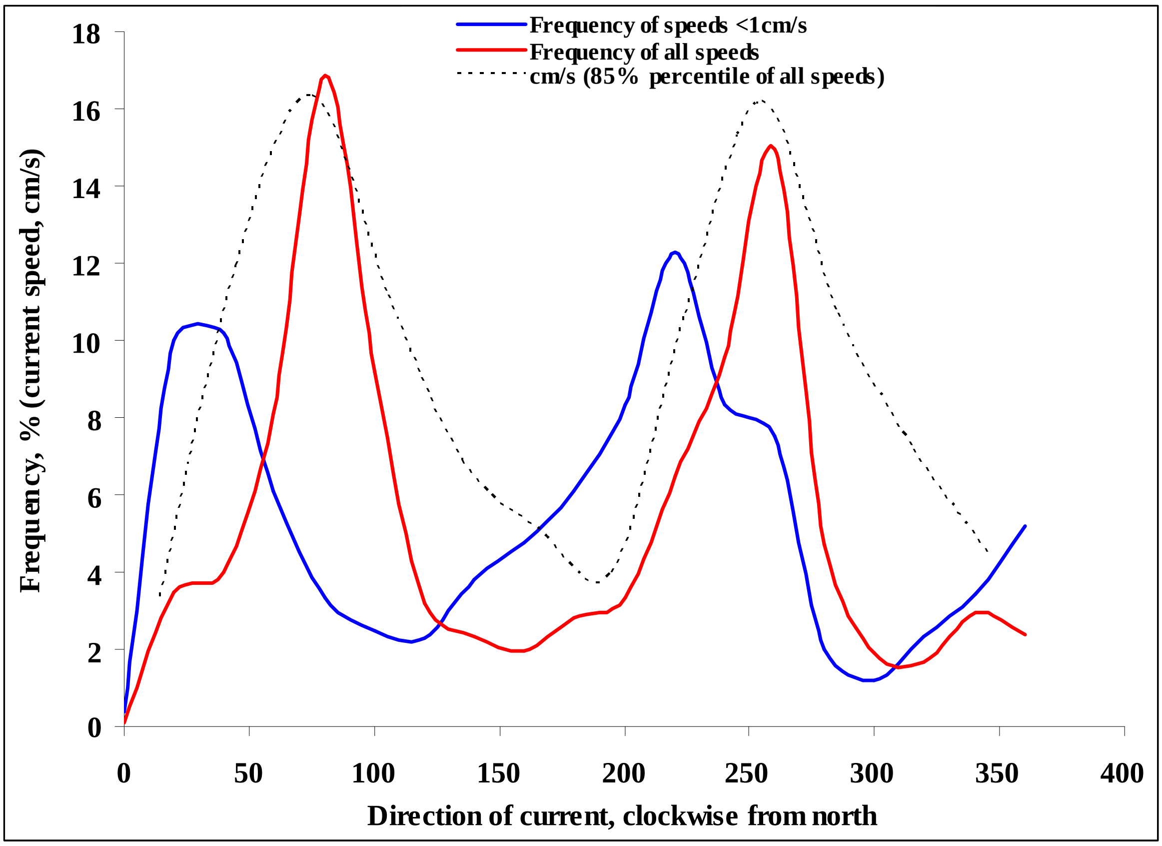


Figure 11 Frequency of currents less than 1 cm/s and all currents at the Ánanaust-disposal site at different current directions (intervals of 20°) at 1 m above the bottom. Also shown is the 85% percentile current velocity.

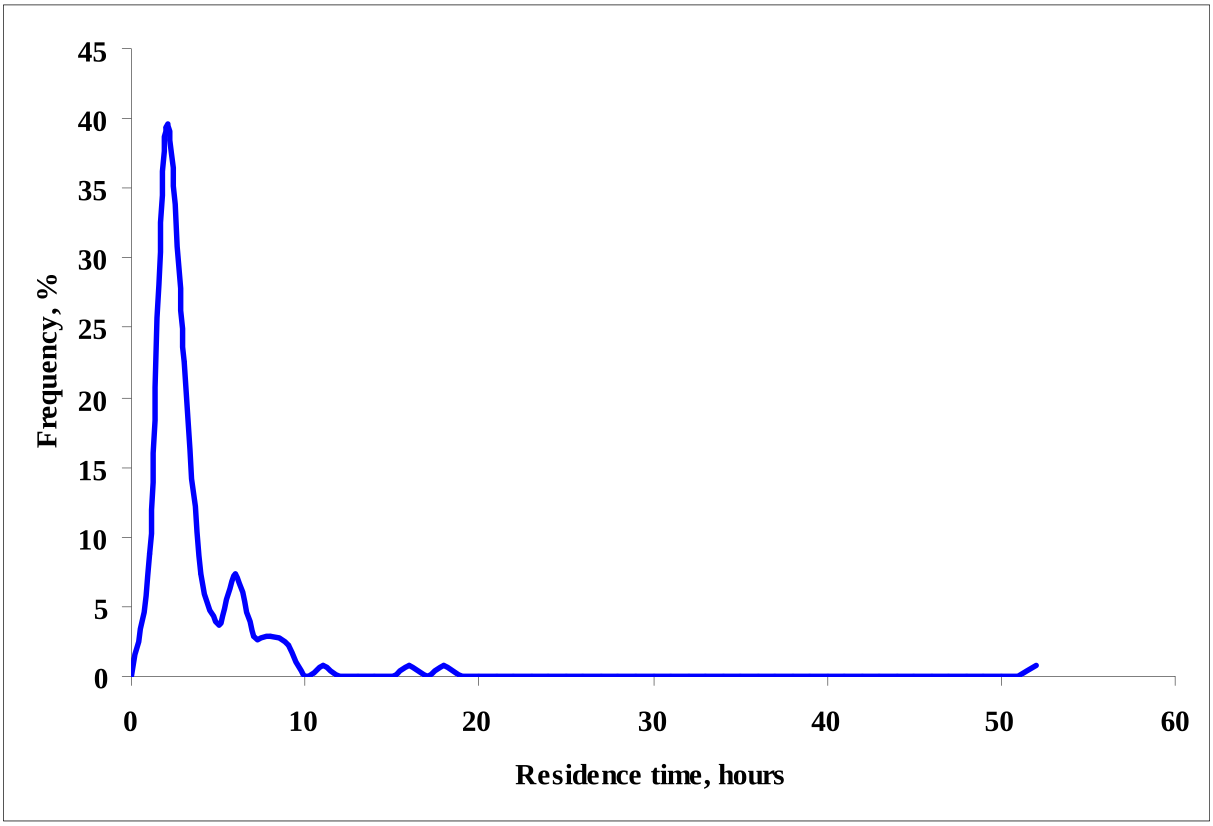


Figure 12 Frequency of periods from the onset of currents less than 1 cm/s until the current has reached 5 cm/s or more in the period 28/06/94-16/08/94.

In addition to the assumptions of equalling the behaviour of sewage particles with mineral particles as regards their stay or residence in suspension as well as not accounting for wave motion in keeping the particles in solution, there is also considerable turbulent mixing during slack water when currents change direction rapidly. In spite of the predictions above representing absolutely worst case scenarios, no effects are foreseen on the sediments by emissions from150 thousand person equivalents at the Ánanaust site by the present treatment of sewage, *i.e.* screening followed by sand sedimentation and fat floatation. Thus, the adaptation of UWWTD (CEC 1991) by Iceland, *i.e.* that screening is equivalent to primary treatment when sewage is discharged to less sensitive areas (article 20.2 of regulation 798/1999), is therefore fully justified. Furthermore, it is not to be expected to find any effects on the benthic community nor sediment chemistry as a result of the ocean outfall off Ánanaust at current emission rates nor future discharge from 150,000 p.e.. Finally, further reduction in suspended matter than the present day treatment offers will not result in any environmental improvement in the recipient since no effects are foreseen nor detected experimentally, see 5.3, 5.4 and 6.

Using the minimum exchange rate estimated in 2.1, one may calculate the average concentration of suspended solids in the near field from future 150,000 p.e. as 120 µg/L. This level corresponds to about 50 µg particulate carbon (POC) per litre. Here it is assumed that a negligible part of the suspended solids of the sewage will settle onto the seafloor. This may be compared with the natural content of offshore water (33.82‰≤S≤35.15‰) around Iceland or 29-420 µg POC/L (Unnsteinn Stefánsson 1981), where POC increases with primary production or Chl *a* as expected. Regression of POC with primary production or Chl *a* in May-June 1976 gives a value of about 100 µg POC/L at zero primary production (Unnsteinn

Stefánsson 1981), which is an estimate of winter values. Higher naturally occurring POC-values may be expected closer to shore due to both resuspension from bottom and from fluvial and aeolian inputs from land.

In the far-field estimated in 2.1, the concentration is only about 20 µg POC/L. Particulates decay in the water column and CSTT (1997) assumes a decay rate of 0.3 d^{-1} while Farley (1990) assumed a rate of 0.05 d^{-1} in cold water and 0.1 d^{-1} in surface water off southern California. Therefore, an average excess POC due to the sewage from 150,000 p.e. in the far-field is 10-20 µg/L resulting in 10-20% increase compared with off-shore winter values but most probably the relative increase at the Ánanaust site will be considerably lower. The suspended matter derived from sewage, however, is of a different nature to the naturally occurring suspended matter especially during summer months when primary production is effective. Finally, the POC excess due to effluents from the future 400,000 p.e. in the Kollafjörður, will on average be about 0.3-0.7 µg/L, which would be extremely difficult to detect experimentally.

5.2 Preexaminations and experimental layout

In February 2000, sampling of surface water above and around the present diffuser of sewage from the Ánanaust STP took place for the analysis of nutrients and microorganisms with additional examination of vertical profiles for turbidity, temperature and salinity (Jón Ólafsson and Sólveig R. Ólafsdóttir 2000; Vatnaskil 2000). Turbidity (measured as loss of transmission of light at 670 nm) was usually greatest at the bottom as expected but never high in surface water. Sewage particles may have contributed to this bottom turbidity but it stems also from incoming seawater into the area. The most important source of the turbidity, however, is erosion of the seafloor. The so-called benthic nepheloid layer is created by the naturally increasing turbidity of seawater with proximity to the seafloor. Transmittance, *i.e.* the proportion of light at the wavelength of 670 nm that penetrates seawater, is a measure of suspended particles in the water. Measurements of transmittance require calibration against direct analysis of suspended particles at each site and each time if the data are to be converted to levels of suspended particles. Such a calibration was not performed on this occasion. At stations where the highest dissolved nutrient concentrations were found, the turbidity was variable down the water column but not high. The study showed, not unexpectedly, that suspended particles behaved differently from dissolved nutrients (Jón Ólafsson and Sólveig R. Ólafsdóttir, 2000). During the February 2000 survey the discharge from the Ánanust STP was about 95,000 person equivalents (Guðjón Atli Auðunsson 2000).

A preexamination of the Ánnaust-disposal site was carried out by videophotography from a ROV on 20. May 1994 (Kjartan Thors 1994) and a similar study was undertaken at the Klettagarðar disposal site in February and March 1999 (Kjartan Thors 1999). The sediment at the Ánanaust disposal site was rich in shell fragments and at some places the coarse sediment was overlain by finer materials. As mentioned above, the sediment has ripples with wave heights of 10-20 cm and wavelengths of 20-50 cm. The rippled surface is seemingly burrowed by organisms.

The examination in 1994 did not result in an estimate of the residence time of the sediments. Such an estimate is however of considerable importance since a prerequisite for possible oxygen depletion and effects on the benthic biota caused by sewage disposal is the temporary deposition of organic matter on the seafloor as discussed above. Such a situation was deemed very improbable as elucidated above. Nevertheless, it was considered important to account for the behaviour of the sediment particles on the seafloor and to estimate the vertical flux and composition of the particles at the disposal sites as a baseline for later monitoring of the areas. In principle, the flux of the bedload may be estimated from the height of the ripples and their rate of movement. Similar studies have been carried out in for example Boston-Harbour-Masssachusetts Bay (Anon. 1998).

In November 1995 an examination of fluxes by way of sediment traps was started at the planned disposal site of the Ánanaust STP and these studies continued until December 1996. For the estimation of sediment residence times and bedload transport, a programme of time-lapse photography of the seafloor was started in May 1996, and these studies continued until June 1997. In the period July 1998 to April 1999 parallel studies were carried out on the by then planned disposal site from the Klettagarðar STP. These studies provided information on a period of least activity during the summer and a period of greatest activity during the winter. Evaluation of the data from the photograpy is found in Kjartan Thors (2000) while the data for the sediment trap studies are found in Guðjón Atli Auðunsson (2001b) where also a comprehensive discussion is given on the fate of organic matter at the disposal sites of sewage from Reykjavik and comparisons made with coastal areas in different parts in the world. Corresponding studies were conducted at the disposal site of the Ánanaust STP after the discharge of sewage started (June 2000 - April 2001) (Kjartan Thors 2003; Guðjón Atli Auðunsson 2005). Below is a short summary of the results of these studies.

5.3 Flux of suspended load studied with sediment traps

Fluxes and erosion

In the first study at the Ánanust disposal site, 5 sediment traps were deployed, one at about 3700 m from shore (middle of the planned disposal site) and four at 500m distance from the first one, *i.e.* to the west, north, east and south of the middle of the disposal site. In a later study at this site, only three traps were deployed. These were located at the east, west and north positions. On the by then planned disposal site of Klettagarðar STP, four traps were deployed around the middle of the planned diffuser at 5500 m from shore, *i.e.* at a distance 500 m west, north, east and south of the middle station. The traps were visited monthly and the total amount of material, particle size distribution and nutrients (LOI, organic carbon, organic nitrogen, and phosphorus) were analysed while trace elements (Cu, Cd, Zn, Hg, Cr, Pb and Ni) were analysed in selected samples.

The sediment trap studies at the Ánanaust disposal site show large amounts of particles in suspension, see figure 13, and a higher flux during winter than during summer.

The fraction of fine sediment particles, i.e. silt with sizes less than <63µm, depends on the weather conditions each time and ranged from as little as 4% up to 95%. The materials in the traps are mostly due to erosion from the seafloor as shown by the good correlation between flux and wave height (at Garðskagi) to a power of about three where more than 80% of the variation is explained by wave height (Guðjón Atli Auðunsson 2001b and 2005). This is a well known semi-empirical relationship for bedload transport, see for example Anon. (1999).

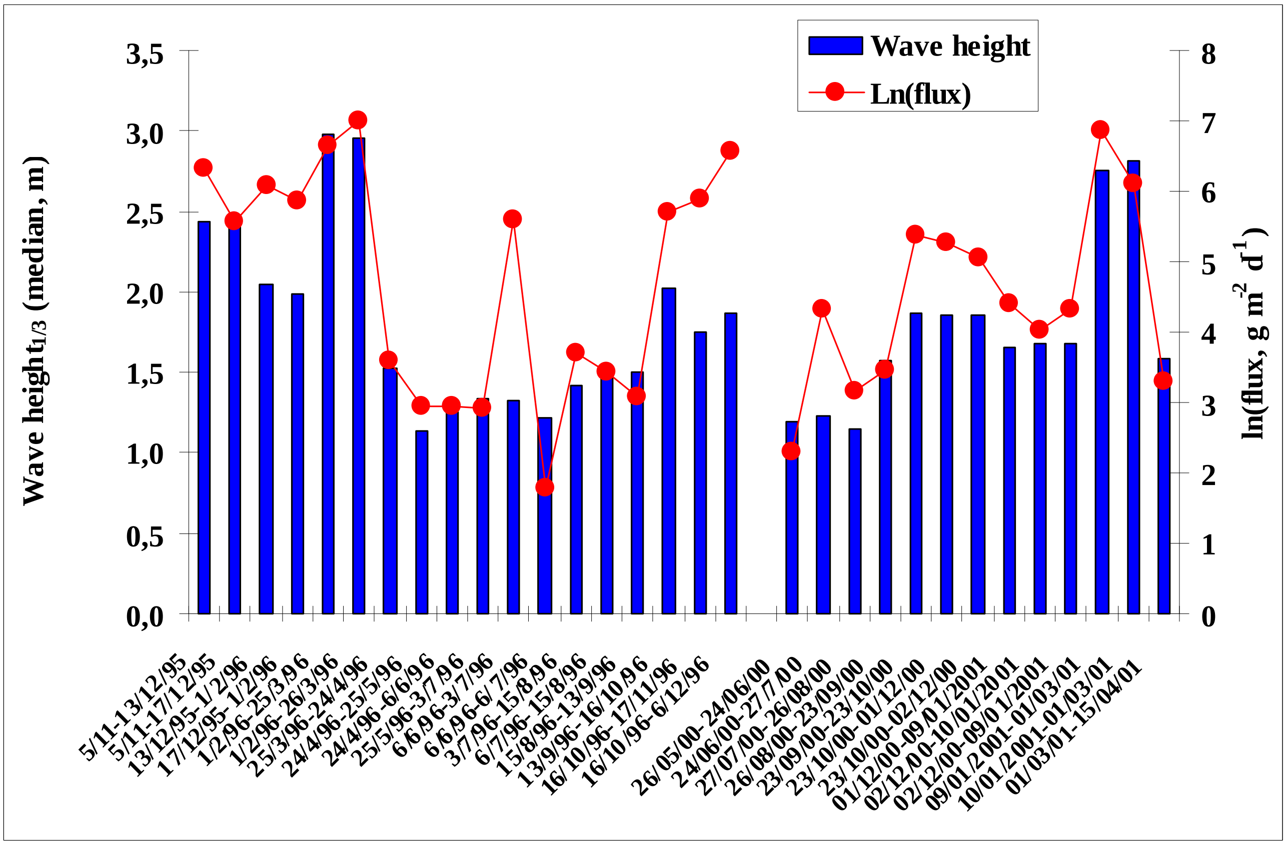


Figure 13 Comparison of significant wave height at Garðskagi and total flux of suspended material in the traps at the Ánanaust-recipient before ('95-'96) and after ('00-'01) the discharge started.

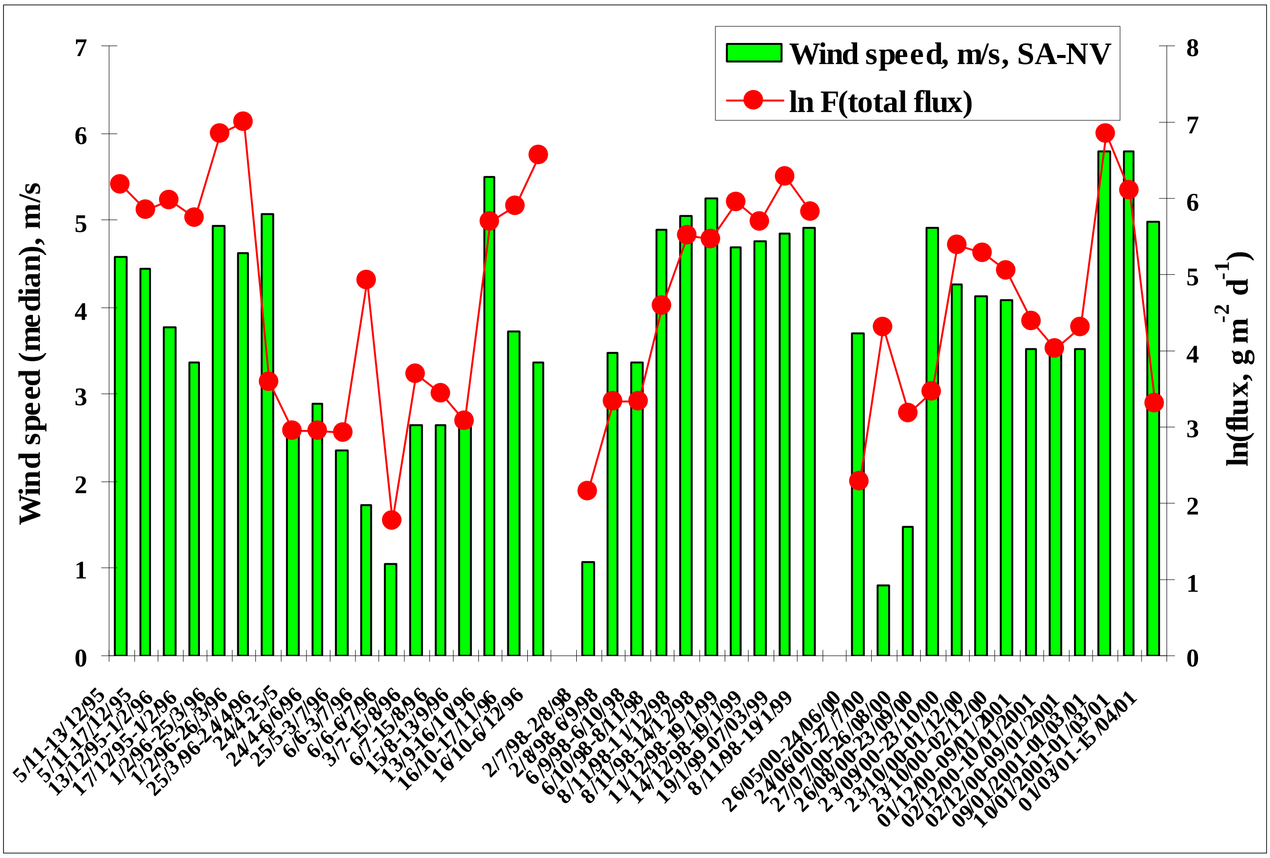


Figure 14 Comparison of wind speed from SE and NW and flux of suspended material in the traps at the Ánanasut-recipient before ('95-'96) and after discharge was started ('00-'01) together with data for the Klettagarðar-recipient before the discharge started ('98-'99).

In addition, materials transported to the areas from other marine waters as well as from land may contribute. The total flux and flux of silt particles behave very similarily at both the Ánanaust disposal site and the Klettagarðar disposal site. The wave height at Garðskagi also correlates well with wind speed at Reykjavik harbour. Figure 14 shows that speeds of winds from SE and/or NW, *i.e.* parallel to the north coast of Reykjavík, also correlate well with the total flux. Thus, synergic effects are at work between winds and currents where particles are eroded from the bottom material by the wave action while tidal and residual currents transport it away.

Even gravel and pebbles are transported into the water column to heights of at least of 45 cm above the bottom, *i.e.* height of the traps. When the behaviour of silt fraction is compared to total flux it may be seen that at a flux of 200 g m^{-2} d^{-1}, the fine fraction starts to decrease, *i.e.* sand and gravel start to move upward from the bottom. This flux corresponds to a wave height of 2.2 m at Garðskagi but wave heights are 35% of the time higher than 2.2 m in the three periods investigated. Similarily, this flux corresponds to wind speed of 5 m/s in Reykjavík but wind speeds higher than 5 m/s occurred 40-50% of the time during the sediment trap studies at Ánanaust. Monthly averages of wind speeds in Reykjavík 1961-2005 are 70% of the time higher than 5 m/s where lower wind speeds occur predominatly in June and July, see figure 7.

At SE-wind speeds >3m/s the flux increases fast and so does wave height from 1.2-1.5 m. Wave heights <1.5 m increase the fluxes significantly but not wind speeds <3 m/s. The monthly average wind speed in the years 1961-2005 is 3 m/s or lower only 0.3% of the time. At the lowest recorded wave heights at Garðskagi, *i.e.* 0.4-0.5 m (≤0.4m 0.1% of the time and ≤0.5 m 0.5% of the time) there is still a flux of silts of about 0.5 g m^{-2} d^{-1} that may occur for 1-3 days (this happened only once during these three study periods, *i.e.* at the end of June 2000). As organic carbon is a constant fraction of the silt or about 8.5%, there is a flux from the bottom of about 0.04 g C m^{-2} d^{-1} on these rare occasions. This flux is little less than the maximum settling rate of organic carbon from the sewage during the longest estimated residence time of sewage particles on the seafloor or 52 hours under the very conservative approach for settling rate described in 5.1.

More than 90% of the time, the wave height is greater than 0.9 m and corresponds to a flux of more than 4.5 g m^{-2} d^{-1}. Thus, more than 90% of the time, the erosion rate of carbon from the bottom sediments is greater than about 0.5 g C m^{-2} d^{-1}. This rate is much greater than the maximum settling rate under the worst case scenario described in 5.1 and therefore it is highly improbable that possible additional organic matter could be observed in the sediment traps.

These experiments verify that there is considerable movement of seawater above the sediments caused by wave motion in addition to the movement accounted for by currents. This in turn manifests that the settling rates and residence times in 5.1 are greatly overestimated and represent truly a worst case scenario. These observations explain the very low burial rate of organic carbon in the sediments in this area and therefore low flux of carbon to the sediments (Guðjón Atli Auðunsson 2001b) in spite of high primary productivity. Organic matter is mostly oxidised in the water column due to long residence times caused by the great turbulent mixing which in turn supplies ample amounts of oxygen for their aerobic decomposition. If the organic matter settles on the seafloor, it is shortly thereafter eroded from the bottom into the water.

From these observations it may be concluded that there is always considerable flux of silt particles at the seafloor except, very rarely, for very short periods, 1-3 days as an absolute maximum when a minimum flux is observed. During these periods, the oxygen supply to the sediments is always well above the oxygen demand and thus reduced oxygen above the sediments due to sewage particles alone will not occur.

As described above, the area in SE-Faxaflói, where the disposal sites are located, is one of the most productive marine areas around Iceland with primary production of more than 300g-C m^{-2} y^{-1} to be compared with about 218 g C m^{-2} y^{-1} on average within the the 200 m depth contour (Þórunn Þórðardóttir 1994). In spite of this high primary productivity, oxygen depletion has not been observed in the area due to high currents and fast water exchange. Episodically, some oxygen undersaturation may be observed in this area as could be seen in 1966 when some undersaturation took place close to the bottom due to fast decomposition of organic detritus from phytoplankton and related materials, starting at the end of June, peaking in middle of August, and ending in late October (Unnsteinn Stefánsson, Þórunn Þórðardóttir & Jón Ólafsson 1987). The minimum oxygen saturation was about 80%. It may also be mentioned that the rate of decomposition is faster the higher the primary productivity.

Nutrients in the trapped materials

Organic carbon, organically bound nitrogen, and phosphorus of the organic matter indicate that almost solely organic matter of marine origin is present at the disposal sites prior to sewage release as well as after discharges started from the Ánanaust STP (local Redfield-relationships). The natural ratios of these nutrients in living marine biota may, however, vary to a degree that encompasses composition of sewage particles (Guðjón Atli Auðunsson 2005). Furthermore, seasonal differences may be observed as can variation from one year to another. Thus, evaluation of these data may be a complicated task. When comparing the composition of the organic matter in the sediment traps before and after sewage discharge started it turned out that no changes could be observed in the total amount of organic matter. Furthermore, both nitrogen and phosphorus were significantly lower with respect to carbon after the release than before. The linear relationship between nitrogen and carbon was lower by a constant but the rate of change of nitrogen with carbon was the same, *i.e.* the same rate of change prior to and after release of sewage and the same as in typical organic matter of marine origin. When compared with the Ánanaust study 1995-1996, lower nitrogen by a constant relative to carbon was also observed at the Klettagarðar-disposal site prior to any disposal, *i.e.* 1998-1999. Phosphorus also correlates with nitrogen in the same manner before and after discharges started and in the same manner as these nutrients relate in the ocean south of Reykjanes (Selvogsgrunn). These observations bring out the variations between years in the age of organic matter in the area but phosphorus and nitrogen are ususally released faster from the particles during their decomposition, *i.e.* the ratio of carbon to either nitrogen or phosphorus increases with age.

In summary, no changes in supply nor in compostion of organic matter caused by discharge of sewage from 100,000 person equivalents could be detected in the area. This is in line with predictions on the behaviour and fate of sewage particles in the area discussed above.

Trace elements in the trapped materials

Some trace elements (Hg, Cr, Pb, Cd, Cu, Zn, Ni) were analysed in selected trap samples before and after release of sewage and the results show that their concentrations were not significantly different from their levels in surface sediments from uncontaminated areas around Iceland (Guðjón Atli Auðunsson 2005). The levels of nickel are, however, generally somewhat higher in the sediments north of Reykjavik than in most sediments around Iceland. The reason for this is at present not known but this nickel does not stem from the current discharge of sewage (Guðjón Atli Auðunsson 2005). Lead and mercury increase with organic carbon as do zinc and copper although their rate of increase was slower. This behaviour indicates that these metals originate in organisms in the area. Nickel and chromium, however, decrease in concentration with organic carbon, which indicates that their behaviour may be explained by the geochemistry of the area. Since organic matter on the seafloor is made up of the marine biota of the area as well as faecal pellets, these results show that neither particulate nor dissolved trace elements of the sewage are taken up by the local biota. This is in line with the predicted rate of dilution of these metals but already in the near-field the levels of all these metals are well below their natural concentrations in offshore ocean water (Guðjón Atli Auðunsson 2006). These results also indicate that the sediments are well oxygenated as predicted above because suboxic or anoxic sediments accumulate metals.

When the results for trace elements in sediments from unaffected areas around Iceland are compared with sediment quality guidelines from all over the world it becomes clear that quality guidelines need to be established on a regional basis. Such guidelines for some elements, *e.g.* copper and zinc, do not appear to be suitable for prevalent conditions around Iceland.

5.4 Benthic photography

The time-lapse photographs were examined with respect to signs of bedload transport. These signs are all types of changes on the seafloor, transport of sand and particular grains, especially shells and fragments of shells, and formation of ripples in the seafloor material and erosion of the seafloor. Additionally, fine materials are seen as resuspended solids or turbidity. Information on both wind speeds and wind directions in Reykjavík were used to assess their possible effects on the transport of bottom sediments in the area. It has not been possible to find any confident relationship between winds and sediment transport nor between tidal currents or wave height (at Garðskagi) and movement of the sediment.

For the Ánanaust disposal site, both before and after discharges started, it could be seen that transport of the sand and gravel bottom of the investigated area was frequent in the period of the research, especially during winter. The nature of the sediments, especially their coarse grain size, suggests high-energy conditions not likely to allow sedimentation of fines

From September to the end of March, frequent periods of sediment transport occurred, often several times each month, scenarios that may be divided into 1) formation of ripples, 2) erosion of ripples, and 3) sedimentation. The last event was characterised by sand that covered the whole area of the photograph and masked older formations, see figure 15. Quite frequently a cloud of sediment particles in suspension was formed above the bottom and often so dense that the bottom could not be seen (1 meter from the lens of the camera). These results demonstrate considerable movement of sediments during the winter time and verify that fines (fine sand, silt, and clay) do not reside in the area but are transported away by currents (Kjartan Thors 2000 and 2003).

The situation was much calmer in the areas during the summer 1996, spring 1997, spring 2000 and summer 2001. For example in the period between end of May to the end of August 1996, hardly any ripples were formed in the sediment except for one time, *i.e.* at the end of July and beginning of August. The formation of ripples is an indication of forceful transport of bottom sediments caused by currents and this was seen only this time. The same is true for the period from the middle of April to 24th of June 1997, when sediment transport of the aforementioned type was only observed once, *i.e.* end of May and beginning of June. This does not mean that no sediment transport takes place during the summer outside these periods. An examination of the photographs reveals that changes of the bottom sediments are frequent although they are not as decisive as they are in winter time, see figure 16.

a

b

c

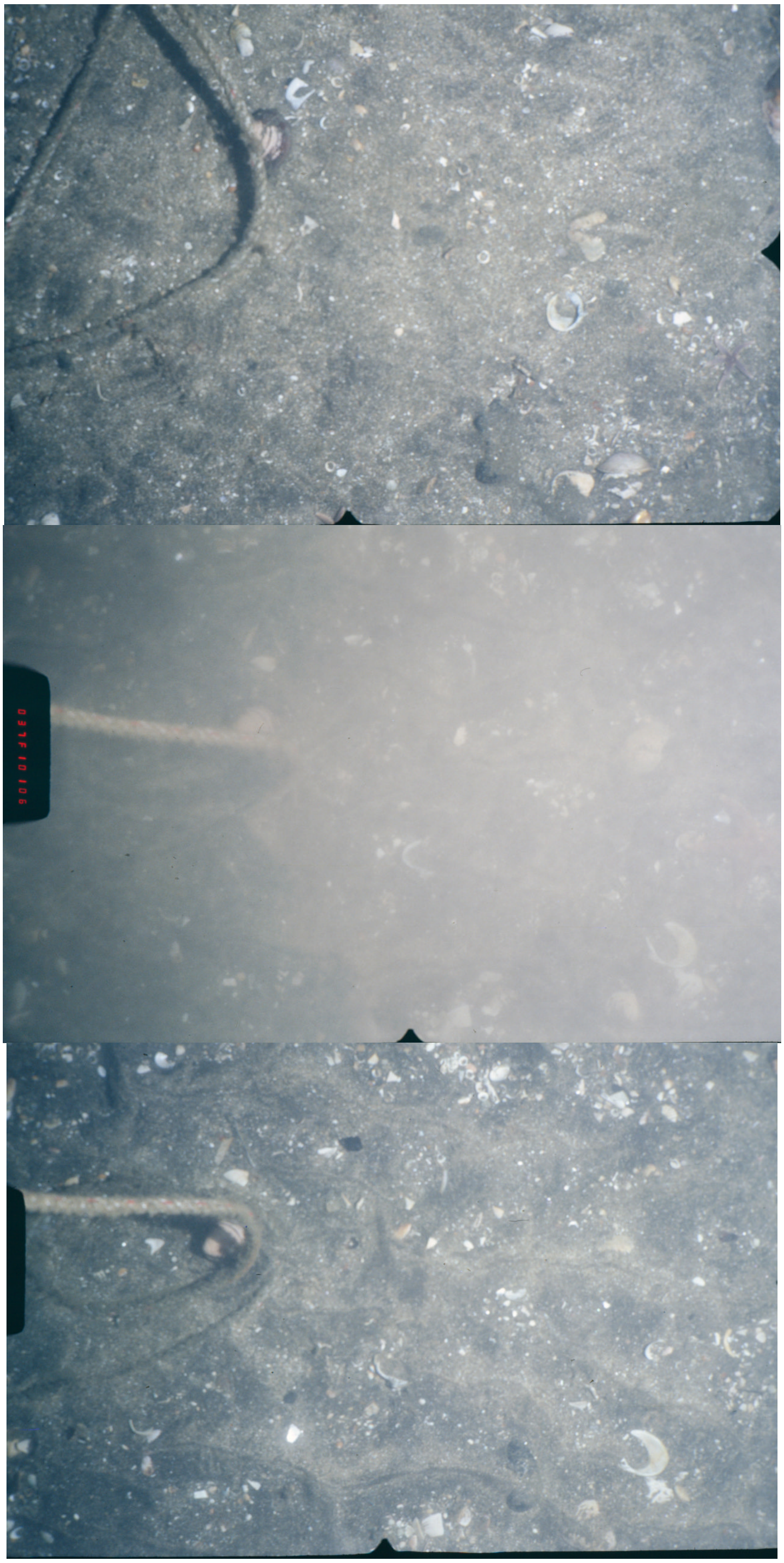


Figure 15 Photographs from December 2000.
a) December 11 at 13:06. Bottom with sand and gravel with shells and shell- fragments. Some ripples may be seen in the sediments.
b) December 16 at 13:06. Some turbidity above the bottom. The rope has been moved. New ripples may be noticed on the bottom.
c) December 18 at 13:06. The turbidity has disappeared but the new sandripples are clearly seen.

a

b

c

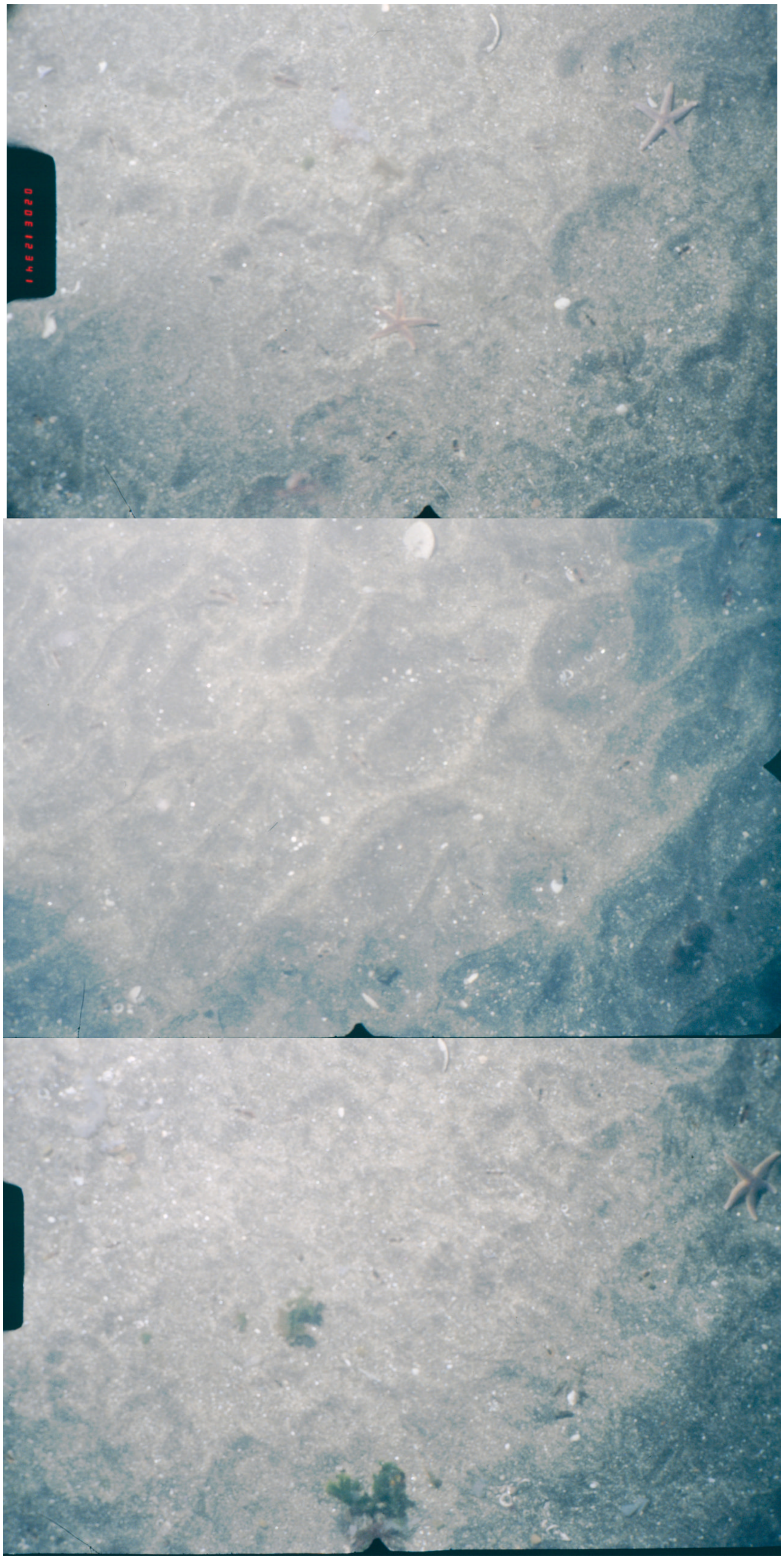


Figure 16. Photographs from June 2000.
a. 4 June. Ripples in the sand rupturing.
b. 6 June. Newly formed ripples seen in the sand.
c. 7 June. Ripples disappear again.

A series of photographs from 25th of June to 3rd of July 1996 contains 77 photos. In 33 of the pictures, changes from the preceding picture are observed. In some of these cases, the changes are relatively small but in others the changes are more marked. The changes indicate either erosion or sedimentation, but clearly indicate transport of sediments. A series of pictures from 6th of July to 15th of August the same year contain 78 pictures of which 51 show evidence of sediment transport.

It is therefore evident that sediment transport occurs frequently in the spring and during summer months although not as vigorous as seen during winter months, often starting already in September. Not all transport is caused by waves and currents. In some of the cases where changes in the bottom sediments are observed, these are probably caused by benthic organisms, bioturbation (Graf and Rosenberg 1997). In many of the pictures, starfish, gastropods, crabs and fish including flatfish, are seen which undoubtedly affect the bottom sediments and move sediment grains. Although the effects of these organisms are not very important in transporting sediments, they dislodge sediment grains which are subsequently more prone to transport by the flowing seawater. When the changes of the bottom sediments are considerable, these can only be explained by motion of the seawater.

One event is worth mentioning regarding sedimentation and movement of sediments although no explanation is provided at this time. In pictures from the springs of 1996 and 1997 a brownish hue is seen on the bottom in every second picture but less conspicuous or absent in other pictures. The surface spread is seen during the day but has disappeared the following night. The nature of this features has not been confirmed by direct experiments but there is strong evidence that epipelagic diatoms are the reason for this effect since they attach themselves to the surface of sediment grains and move to the top during the day but to the underside of the grains at night.

The sedimentation/erosion of bottom sediments at the planned disposal site from the Klettagarðar STP was studied by the same method. Pictures were taken of the bottom twice every 24 hours in the period July 2nd 1998 to February 1999.

The pictures show small changes of the seafloor in summer and autumn 1998. Occasionally, some turbidity was seen in the seawater but it disappeared after a short while without visible settling on the floor. The bottom at this site is sandy with shell fragments and shells and does not seem to change much as mentioned above. There is considerable activity of organisms at this site, fish are frequently seen but the camera frame might have affected that. Starfish are frequently seen but stay only for a short time in the path of the camera. The alterations on the bottom are slight enough to be caused solely by the passing organisms.

During winter, *i.e.* from middle of November 1998, the frequency and time length of turbid waters increases and the waters become richer in suspended particles. As was seen during the summer months, the suspended matter does not seem to settle on the floor and all of it seems to disappear from the site. The motion of the seawater starts to relocate the coarser materials on the bottom and in the middle of December, ripples on the seafloor start to form but these are signs of sediment transport. New ripples are also seen in the end of January and beginning of February 1999.

The main conclusion is that silty materials, *i.e.* materials finer than sand, do not accumulate at this site, the planned disposal site from the Klettagarðar STP. Turbidity that forms occasionally as a result of the interaction between winds and currents, is flushed away. During windy periods, especially during winter, the motion of the seawater is strong enough to relocate the bottom sediments.

From the discussion above, it may be concluded that net sedimentation rate at the Klettagarðar disposal site is extremely slow or none at all. This is supported by the fact that modern sediment at this site is very thin, only few tens of centimeters. Modern time encompasses approximately the last ten thousand years and therefore, in the time frame of discharge from the Klettagarðar STP, the sedimentation rate is not detectable.

Summary of the sediment bedload studies

The bottom photography, which is mainly directed at the coarser sediments on the seafloor, shows that sand and gravel is moved several times each year, especially during the winter time, with the sand more frequently than the gravel. Fine materials settle temporarily and disappear long before the currents start to move the sand. The net effect of this process is transport of coarse material through the area.

The photographs only gives indirect information on the fate of the sediments that emerge from the sewage diffusers. The silty material which is often seen in suspension at the bottom, seems to settle only for a very short time on the seafloor. It is reasonable to conclude, therefore, that the sediments emerging from the sewage diffuser will not accumulate at the disposal site.

This conclusion is in agreement with the predictions of 5.1 that sewage particles will not settle on the seafloor at the disposal site of the Ánanaust site and on the rare occasions they might do so, their residence times would always be less than 10 hours and in a flux onto the sediments that would hardly be observable by the camera.

6. BIOLOGICAL SURVEYS OF THE BENTHOS

In the summer of 1995, the benthos of the planned disposal site of the Ánanaust STP was investigated for the purpose of determining its potential value for conservation. The study was also meant to serve as a baseline for monitoring of the area after sewage disposal started (Jörundur Svavarsson 1996). The post-examination of the disposal site of Ánanaust STP took place in the summer of 2000, two and a half years after disposal started (Jörundur Svavarsson 2002). Parallel investigations were performed at the planned disposal site from the Klettagarðar STP in 1999 (Jörundur Svavarsson 2000).

The sandy bottom further out than 3700 m at the ***disposal site of Ánanaust STP*** is unsuitable for monitoring since uncertainty associated with the sampling limits its usefulness. The problems of sampling was described in 1.3. Therefore, only the area within 3700 m was investigated which means that only the first 100 m of the diffuser area was investigated since the diffuser starts at 3600 m. Sandy bottom beyond 3700 m offshore is generally limited in the number of species and, as stated above, reflects a rather harsh environment. Most benthic studies around Reykjavík have been carried out on muddy sediments which have greater species richness; and studies on benthic communities in relation to sewage outfalls discussed in 5.1 are generally on sediments rich in mud. For this reason, investigations at the Ánanaust site were carried out by photographing the bottom and by studies on kelp-holdfast wherever they could be found.

The outer limit of the brown kelp *Laminaria hyperborea* is reached at 3540 m. At this distance the plants are small and scattered. The biota on the rocky bottom from 3540 to 3700 m offshore is without kelp and therefore the diverse biota associated with their holdfasts is not available. The number of species and species density was in 1995 rather low in this area. The kelp forest gets more dense in direction to land and plants become larger while it again turns more scattered still closer to shore with barren zones. In an area between 3200 to 3540 m from the shoreline, considerable biodiversity exists on the rocky and sandy bottom where horse mussels (*Modiolus modiolus*) dominate, inhabiting pockets in the sand and are used by kelp as substratum. In spite of considerable species diversity in the kelp forest, the area is not considered unique and similar in biodiversity to other areas of similar sediments around Iceland. Within 3200 m from the shoreline, the bottom becomes more sandy.

In 2000 no changes in the benthic community could be detected as a result of the discharge of sewage effluents. The number of species and number of individuals on kelp-holdfasts in 2000 after discharge started, was in agreement with what had been found prior to release of sewage in the area, *i.e.* in 1995. Furthermore, there was not a significant difference in biodiversity or biodensity between the disposal site and the reference sites west and east of the study area.

Natural variability in the benthos is, however, considerable and environmental stress not easily detected above this natural variability even on soft bottoms (Livingston 1982). In 2000 the number of horse mussels had decreased significantly from 1995 but most probably due to a large number of the star fish *Asterias rubens* in 1995 feeding on horse mussels. This natural event is further demonstrated by a similar decrease in horse mussels at the reference sites west and east of the disposal area.

The absence of significant change in the benthic communities is in good accordance with predictions on the fate of sewage particles, *i.e.* they do not settle in the area and are thereby not available to the organisms.

The benthos at the ***disposal site from the Klettagarðar STP*** is homogenous but with considerable diversity. The investigation at the Klettagarðar site was carried out by grab sampling although difficult to carry out on the hard bottom with a thin layer of sand. The biodiversity is not among the highest found on sandy bottoms in shallow waters in SW-Iceland. Probably the diversity of the area reflect young individuals of species that will presumably not thrive there as adults. Most of the species found in the area have been found elsewhere in shallow waters around Iceland. No species were found unique enough to warrant conservation of the area off Klettagarðar.

Many investigations have been carried out around and in the vicinity of Reykjavik as regards the possible effects of the past disposal of sewage on organisms on the beach and benthos off shore and baseline data collected (Sólmundur Einarsson 1973; Einar Jónsson 1976; Arnþór Garðarsson and Kristín Aðalsteinsdóttir 1977; Agnar Ingólfsson 1977; Arnþór Garðarsson, Jónbjörn Pálsson and Agnar Ingólfsson 1974; Guðmundur Víðir Helgason and Jörundur Svavarsson 1991; Karl Gunnarsson and Konráð Þórisson 1976; Guðmundur V. Helgason and Arnþór Garðarson 1992; Karl Gunnarsson 1993; Guðmundur V. Helgasaon and Arnþór Garðarson 1995). Of the investigations that have taken place, only those related to macroalgae on the beach closest to the outlet of sewage in Skerjafjörður, showed effects that could be associated with sewage disposal (Karl Gunnarsson and Konráð Þórisson 1976) but the benthos in Viðeyjarsund and Eiðsvík had possibly been affected by pollution (Guðmundur V. Helgason and Arnþór Garðarson 1992). The present disposal of sewage has relieved the pressure on the areas that were affected by sewage disposal through temporary outlets close to or onto beaches and thereby of limited initial dilution. All sewage is now released through diffusers at the disposal sites of the Ánanaust and Klettagarðar STPs, where mixing with seawater is fast and efficient and accumulation of organic matter on the sediments is extremely limited, nil for more than 97% of the time. As discussed under 5.1, the accumulation of sewage particles in or onto the sediments is a necessary prerequisite for them to exert any effects on the benthos. Therefore, the net effect on the biosphere has been scaled down extensively and the sites of present disposal are evidently under no detectable pressure from the effluents of the outfall. This was to be expected from the predictions of initial dilution in chapter 2, verified by distribution of coliforms and nutrients, and fate of particulates from the sewage in 5.1 verified by the results from the sediment trap studies in 5.3, and benthic photography in 5.4.

7. MONITORING OF CONTAMINANTS WITH BLUE MUSSELS

Monitoring with mussels

Environmental effects of sewage disposal are generally reflected in changes in the species living at the disposal sites. In order to facilitate the study of these effects and not least to make the evaluation of the data less demanding, organisms may be deployed at different distances from source as well as in a reference site unaffected by sewage. Various biological, physiological and biochemical changes in the organisms may be monitored for this purpose but most commonly the accumulation of certain pollutants is looked for since that is often the direct cause of most of the detrimental changes in the biological, physiological and biochemical parameters. Information on accumulation also reveals whether the chemicals found in the organisms are bioavailable to them, often the prerequisite for their possible harm to organisms. The nature of the effects may often be related to the nature of and/or level of the substances found in the organisms since much is known through experiments about detrimental substances found in sewage.

The most commonly used organism for this purpose is the blue mussel (*Mytilus edulis)* but other species have also been used. Blue mussels at SW-Iceland have been used for trace metal studies and regional monitoring procedures developed (Jón Ólafsson 1983 and 1985). By placing the mussels at different distances from the sewage source, distribution of sewage is revealed. Additionally, the health of the organisms is evaluated both generally and in relation to the type and levels of chemicals found in their tissues. If these chemicals correlate with sewage exposure, *e.g.* faecal coliforms, their source in the sewer system may be located and their release terminated. Studies with blue mussels are commonly used in Iceland for monitoring industrial emissions and discharges from various activities. Additionally, blue mussels in the wild are widely used for long term monitoring. By this way a good Icelandic data bank is available, which, together with international data banks, greatly facilitate evaluation of the results.
Since the accumulation of pollutants in mussels is rapid and the analytical methods very sensitive, relatively short period (about two months) will suffice to find out whether and to what extent the organisms in the area are affected by the sewage or other discharges. If further sensitivity than that obtained in two months is required, the time period of deployment may easily be increased. Furthermore, for increasing the certainty in the assessment, the cages may be placed at stations where effects of sewage are most probably found irrespective of whether appropriate species for monitoring are naturally present at the stations or not. It will therefore take a relatively short time to find out whether sewage treatment is effective or not and to reveal possible effects of sewage disposal on the organisms in the recipient and, if not satisfactory results are obtained, one may react rapidly to improve the situation (*e.g.* by readjustment of the sewage treatment process, readjustment of the route of the pipeline or by preventing disposal of hazardous substances).

Comprehensive studies have been carried out as regards past disposal of sewage from Reykjavik for both classification of the planned disposal sites from Ánanaust STP and Klettagarðar STP as sensitive or less sensitive as well as creating baseline data for later monitoring at these sites (Guðjón Atli Auðunsson 1994b and 2001; Guðjón Atli Auðunsson and Hannes Magnússon 1995a and 1995b). Similar studies have also been carried out in relation to sewage in Hafnarfjörður (Guðjón Atli Auðunsson 1996). In the summer of 2000, examination by means of deployed blue mussels along the pipeline from the Ánanaust STP took place where organochlorines, PAHs, trace elements and microbiological indicators were analysed in the whole soft tissue of the blue mussels (Guðjón Atli Auðunsson 2006). Together with growth rates, death rates, and morphometry, proximate chemical composition was also determined.

A short review of the main conclusions drawn from the monitoring of mussels is as follows.

Bacterial indicator organisms

The situation prior to disposal of sewage by the ocean outfalls may be summarised as follows (Guðjón Atli Auðunsson 1994b; Guðjón Atli Auðunsson and Hannes Magnússon 1995a and 1995b)

a) The distribution of bacterial indicator organisms off Klettagarðar and Ánanaust was greater than expected from their distribution in surface water at the same season. This is probably due to the fact that mussels accumulate these bacterial indicator organisms whereupon their concentrations become about ten times (coliforms) and 100 times (enterococci) that of the surface water they live in.

b) The level of bacterial indicator organisms decreased continuously with distance from shore both off Klettagarðar and Ábnanaust.

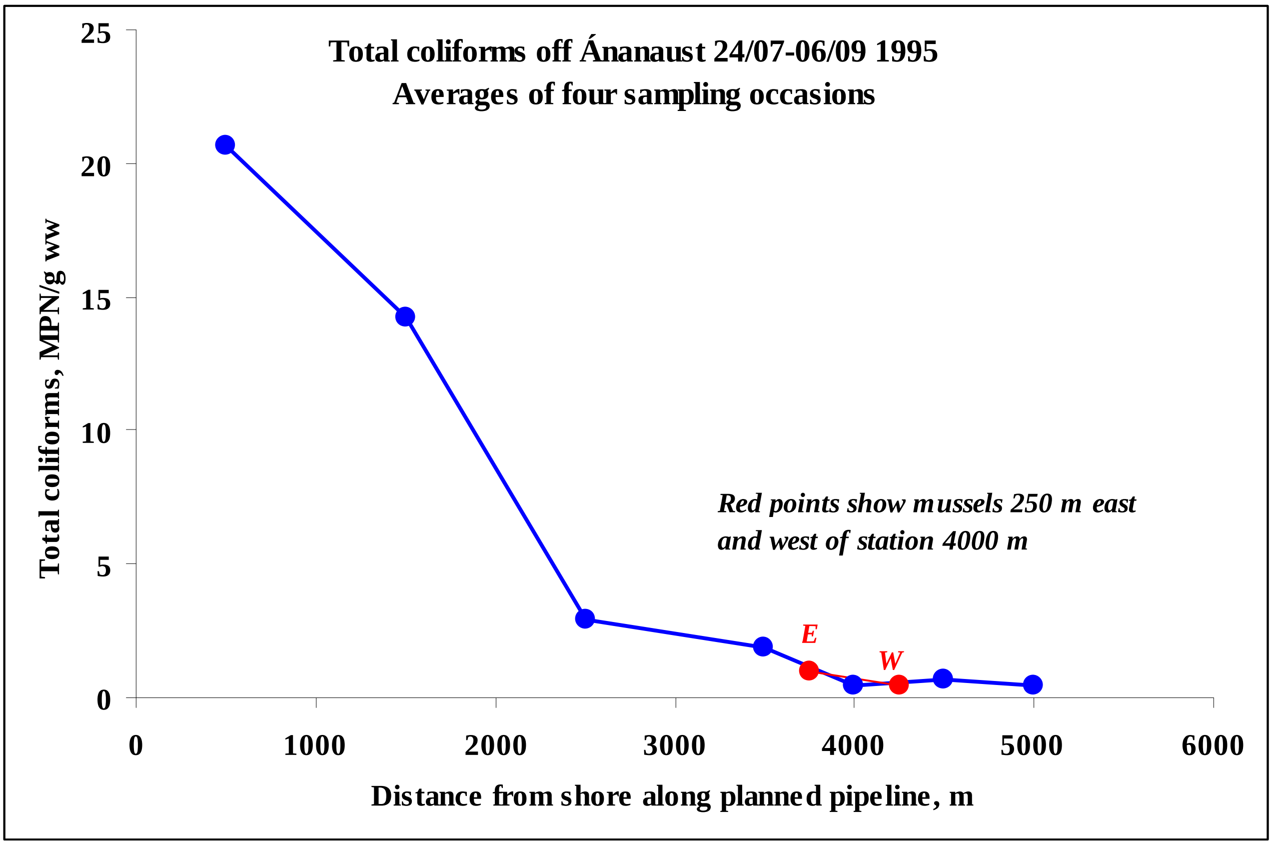
c) Total coliforms in blue mussels within 4000 to 5000m from shore were in all cases higher than Icelandic guidelines for bivalves stipulate (1 MPN/g).

Figure 17 shows the distribution of faecal coliforms in blue mussels off Ánanaust in 1995. The release of sewage to the coast at that time was about 15 thousand person equivalents distributed over a long distance along the shore (Guðjón Atli Auðunsson 1994).

The lower diagram shows the distribution of total coliforms in blue mussels in the period July to September 2000 after release of sewage started at the beginning of 1998 and during this sampling period it amounted to 100,000 p.e. (Guðjón Atli Auðunsson 2000 and 2002). The diffuser is at 3600 to 4100 m and the maximum concentration of total coliforms was found above the landward end of the diffuser at 3600 m (Guðjón Atli Auðunsson 2006). The concentration is low to the east, *i.e.* in the direction of the residual current. Only one sample could be obtained to the west of the diffuser (a cage was lost) but its value was fairly high. A single sample is only an indication since the variability in measurements of bacteria is very high. For this reason, this single measurement is not shown in the diagram. There is considerable variation in these measurements but good correlations were found between total coliforms, faecal coliforms, enterococci and *E.coli* indicating a common human source (Guðjón Atli Auðunsson 2006) where the following was always found:

total coliforms ≥ faecal coliforms ≥ enterococci ≥ *E.coli*

As seen in figure 17, the maximum at 500 m from the coast from 15 kpe is about one fourth of the maximum from 100 kpe. However, a sample taken 50 m from a preliminary outlet of 50-60 kpe from Laugarnes in 1994 gave total coliforms on average 60 MPN/g in the mussels (Guðjón Atli Auðunsson, 1995a), indicating further the greater dilution efficiency of the present discharges. These results on bacteria in mussels together with the measured *per capita* values of the bacteria confirm the the average dilution of about 1000 in the near field (Guðjón Atli Auðunsson 2006).



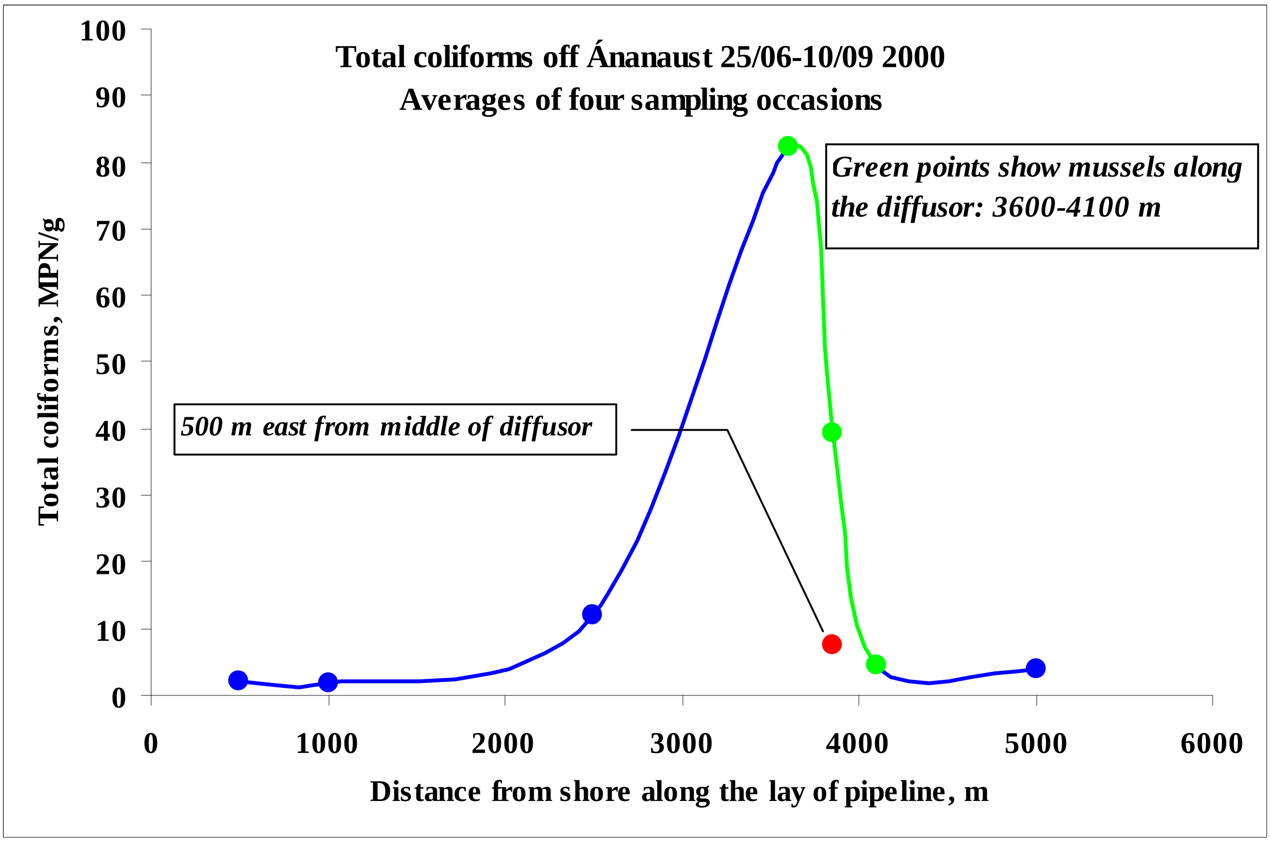


Figure 17 Distribution of total coliforms along the pipeline off Ánanaust. In the upper diagram, a release of an estimated 15,000 p.e. occurred at various places along the shoreline. In the lower diagram, a discharge of about 100,000 p.e. took place along the 500 m diffuser.

Trace elements

A comprehensive review on trace elements in all the investigations that have been carried out using blue mussels in the recipient (Guðjón Atli Auðunsson 2006) shows that except for the metal silver, the past disposal of sewage could not be found to have had any effect on their levels in blue mussels in relation to unpolluted areas around Iceland and when compared with international data banks on trace elements in mussels. A slight increase in lead was observed closest to shore (and sewage outlets) prior to the transfer of discharge to the present disposal sites. That slight increase in lead remained after the preliminary outlets were removed indicating either lead in past time sewage or, more probably, various human activities close to shore (Guðjón Atli Auðunsson 2006). The calculations on dilution in chapter 2 together with results of the levels of trace elements in the sewage itself, predict that all the trace elements are diluted to levels much lower than their current levels in ocean water except for silver. Thus, the data on currents in the area predict the outcome. This holds true even though the trace elements are bound to particles which distribute differently to dissolved species. This is due to the facta that particles are always suspended in the recipient and that many trace elements are released from the particles when entering seawater (Sólveig R. Ólafsdóttir and Jón Ólafsson 1999). As shown in figure 18, the present discharge of sewage has relieved and improved the situation for silver quite markedly, where mussels closest to shore off Ánanaust are as low as the reference site while the levels at the disposal site are the same as prior to discharge of sewage in that area. For reasons not yet elucidated, there was an elevation of copper, nickel, mercury and cobalt in the middle of the disposal site from the Klettagarðar STP prior to any sewage release at the site (Guðjón Atli Auðunsson 2001a).

The levels of trace elements in blue mussels are in all cases lower or much lower than their stipulated maximum limits for seafood for human consumption (maximum limits exist for lead, cadmium and mercury in mussels (Commission regulation (EC) No 466/2001 of 8 March 2001)).

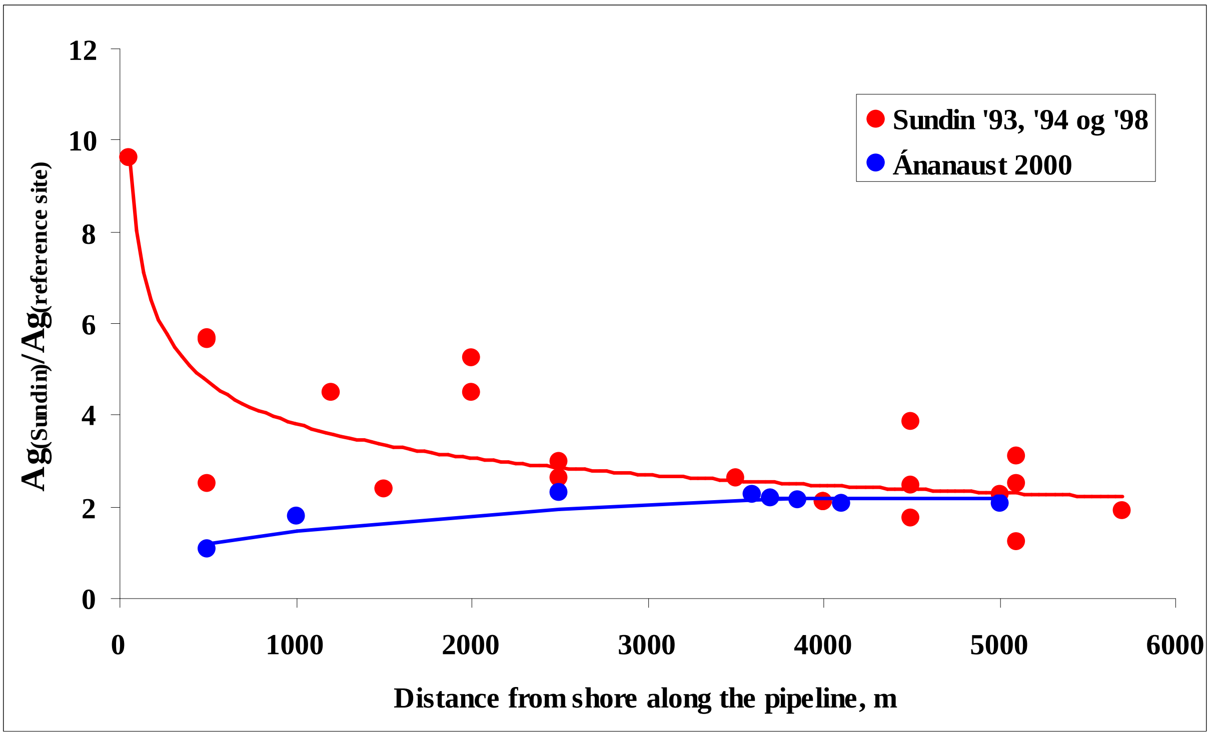


Figure 18 Concentration of silver in blue mussels as a raio of silver in the mussels along the pipelines (Sundin: both off Ánanaust and Klettagarðar) to the concentration of silver in mussels at the reference site (Hvalfjörður).

Since the levels of trace elements off the north coast of Reykjavik and in the near field of the Ánanaust site are the same as in non-affected areas around Iceland, except for silver, they are also low in comparison with, for example, Norwegian quality guidelines, rendering the area fit for aquaculture and recreational purposes (Molvær *et al.* 1997), *i.e.* the criteria for highest quality. This holds also true for silver at the Ánanaust site.

In general, it may be stated that in spite of the relatively large release of some of the inorganic trace elements by sewage (apart from silver which is at extremely low levels in the sewage of Reykjavik), they do not accumulate in the blue mussels except for silver. This is due to two reasons: their low concentrations in sewage caused by dilution of sewage prior to release and relatively fast and efficient dilution of the sewage when it enters the recipient. Nevertheless, silver accumulates to a very high degree in mussels.

Organochlorine micropollutants

Several classes of organochlorine compounds have been analysed in the blue mussels.

A comprehensive review on organic micropollutants in all the investigations that have been carried out on blue mussels in the recipient (Guðjón Atli Auðunsson 2006) may be summarized as follows.

As regards PCBs, DDTs, and HCB, their levels fall off at similar rates with distance from shore, both off Klettagarðar and Ánanaust, and the levels of the reference site in Hvalfjörður is obtained within approximately 4.5-5 km distance from the coast. Other compounds like chlordanes, toxaphenes, HCHs, and trans-nonachlor also decrease with distance from shore *albeit* at different rates. This holds true both before and after the release of sewage started. Apart from the results for the study off Laugarnes in 1993 (west of the planned STP at Klettagarðar where a preliminary outlet was situated prior to the installation of the Klettagarðar STP in spring 2002) neither increase nor decrease has been discerned with time in these three classes of compounds in the period 1993-2000. These compounds were found to increase towards shore. The present sewage water is not the source of the compounds since no change is observed in the gradient off Ánanaust in 2000, two years after the STP there started its operation. The source may be sewage from earlier times that may have contained these substances with resulting accumulation in the sediments. It may also be various activities on land or at shore that may have emitted these pollutants. Whether the situation will be the same for the gradient off Klettagarðar awaits to bee seen when caged mussels will be deployed along the pipeline and at the disposal site. These results were also predictable from the levels found in the sewage itself and the dilution calculated in 1.6 since that dilution takes these compounds below levels found in ocean water around Iceland.

The results show that the levels closest to shore range from being two times (HCB) to five times (DDE) higher than in wild mussels at the reference sites but the PCBs are up to four times the levels of the reference sites. No increase is detected above the diffuser at the Ánanaust site.

According to Norwegian guidelines for organochlorines in blue mussels (Molvær *et al.* 1997), the areas north of Reykjavik, the recipient, fulfill the highest standards and may be rated as suitable for aquaculture and recreational fishing.
The levels of the organochlorines in blue mussels off the north coast of Reykjavik, even closest to the shore, is below (PCBs) or far below (DDTs, HCB, and HCHs) the stipulated maximum limits for these compounds in seafood for human consumption in Iceland.

Polyaromatic hydrocarbons (PAHs)

PAHs were analysed in mussels along the ocean outfall from Klettagarðar in the summer of 1998 (Guðjón Atli Auðunsson 2001) and off Ánanaust in 2000 (Guðjón Atli

Auðunsson 2006). For both areas, their behaviour was similar to for example the PCBs and at a distance 4000-5000m from the shoreline, these substances had reached the levels found at the reference site (Hvalfjörður). No change in the gradient was observed above the diffuser at the disposal site off Ánanaust. Thus, the present sewage water is not the source of these compounds closest to shore but various human activities close to the shoreline. Samples closer to shore than about 4000m had more than four times the levels found in the reference site. The outermost station off Klettagarðar, farthest away from sewage disposal, showed an elevation in PAH-level which most probably is due to release of oils or oil-related materials.

According to Norwegian guidelines for PAHs in mussels (Molvær *et al.* 1997), the area off Ánanaust, even closest to shore, fulfilled the highest quality criteria and thus considered suitable for aquaculture and recreational fishing. The levels off Klettagarðar were higher, however. Only after discharge of sewage has started in the planned disposal site from the Klettagarðar STP will it be conclusive if and to what degree the sewage itself is the source of these substances closest to coast. For several decades comprehensive handling of petroleum products took place in the area around and off Klettagarðar. The disturbance caused by removal of the large oil depot in the area during the investigation for Klettagarðar may have affected the results obtained there. However, comparisons with mussels from other areas around N-Atlantic, including mussels for human consumption, show that even the highest levels found in mussels off the north coast of Reykjavík, *i.e.* off Klettagarðar, are very low (Guðjón Atli Auðunsson 2006).

Maximum limits have recently been established for benzo(a)pyrene in mussels as food commodity (Commission regulation (EC) No 208/2005 of 4 February 2005). The mussels off both Ánanaust and Klettagarðar contain benzo(a)pyrene less than one-tenth of these limits, even the samples containing the highest levels off Klettagarðar.

8. CONCLUSIONS

Comprehensive studies have been carried out on sewage and the recipient of sewage from Reykjavik and neighbouring communities, the area north of Reykjavík, Sundin. This report focuses on the recipient of sewage from the sewage treatment plant at Ánanaust, Skolpa, although results from the Klettagarðar site are also presented.

The recipient is freely open to ocean water and is characterised as a high energy area due to strong tidal currents and wave motion. The exchange rates are therefore fast and result in a dilution factor of about 1000 on average above the diffuser.

The bottom of the recipient area reflects a high energy where sand and gravel dominate. The bottom sediment is moved several times each year, especially during the winter time.

This study shows that the discharge of nutrients (nitrogen and phosphorus) will not cause increased rate of algal growth in the recipient with resulting adverse effects. This is shown by empirical and semi-empirical models of worst case scenarios. This indicates that the currently applied treatment of sewage up to the planned maximum rate of sewage discharge equivalent to 150,000 person equivalents is more than sufficient to fulfill international criteria. Further treatment of sewage than applied at present (screening followed by sand sedimentation and fat floatation) will not result in any detectable environmental improvement as regards algal growth.

Modelling dispersion of faecal bacteria outlines the largest area of sewage effects in seawater and thereby the dilution area. The accumulation of bacteria in deployed mussels during summer time, during which shortest half-lives of faecal bacteria occur, shows that a much smaller area is affected than predicted on an annual basis. The present disposal of sewage has resulted in a more confined dispersion area of faecal bacteria than past near shore disposal. More stringent treatment of sewage than applied at present, *i.e.* secondary treatment, will only marginally decrease the dipersion zone.

Studies and modelling of currents and wave motion indicate that the settling of sewage particles onto the sediments is highly improbable. This settling of particles is a prerequisite for any detrimental effects on the benthic community of organisms and sediment chemistry. Therefore, reduced oxygen above the sediments due to sewage particles alone will not occur. A worst case scenario might result in the settling of sewage particles to the bottom but only for a very short time (few hours during summer) and at a rate experimentally known to cause no effects on organisms on or in the sediments nor result in but very slight oxygen reduction above the sediments for such occasional short periods (hours). Sediment traps show unambigously that neither is there increase in organic matter at the disposal site nor a change in the composition (nutrients, trace elements). A survey of the benthic communities at the disposal site did not indicate any effects caused by the present discharge. Therefore, further treatment of sewage than applied at present will not result in any environmental improvement up to the planned maximum rate of sewage discharge equivalent to 150,000 person equivalents.

Studies of the accumulation of trace elements, organohalogen compounds, and polyaromatic hydrocarbons in blue mussels show that the present discharge of sewage renders the recipient to fit the highest environmental quality criteria in Norway and well below maximum limits stipulated for seafood. Of all the chemicals studied, only silver seems to be affected by past discharge of sewage, a situation that has also been observed internationally. The present disposal of sewage has improved the situation markedly as compared with past disposal through provisional outlets. More stringent treatment of sewage or secondary treatment is not expected to reduce these concentrations of silver.

The comprehensive studies on the waters north of Reykjavík receiving sewage demonstrate that information on currents and preferably also wave motion at planned disposal sites is necessary for model-based predictions of the fate of sewage-derived constituents provided the composition and behaviour of the sewage itself is known. These predictions are also sufficient if evaluations of worst case scenarios give unambigous results as regards acceptable effects with respect to eutrophication potential, dispersion of faecal bacteria, change in macrobenthic communities, accumulation of sewage particles onto the sediments, and accumulation of contaminants in sediments and biota. Only if one or more of these effects are not within acceptable limits, *e.g.* those set by experiments in internationally accepted research and/or set by international organisations, should there be further studies directed specifically at these possible effects.

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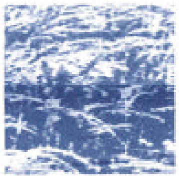

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Keldnaholti 112 Reykjavík phone 570 7100 fax 570 7111 www.iti.is