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An injection well effluent transport study, Maui, Hawaii

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AN INJECTION WELL
EFFLUENT TRANSPORT STUDY,
MAUI, HAWAII

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ABSTRACT

AN INJECTION WELL
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by Kim A. Brown

A field study was designed to map submarine sources of plumes or seeps of primary treated sewage effluent discharged into a coastal injection well. The study was part of an effort to identify possible nutrient sources thought to be the cause of episodic algal blooms in the area. An artificial fluorescent dye, Rhodamine WT, was used as a tracer of the discharged effluent. The site of the study was off the west coast of Maui, Hawaii.

No submarine sources of effluent were positively identified. Elevated fluorescence, at values only slightly above background levels, was recorded in just a few areas. The maximum recorded concentrations were close to the seafloor and equivalent to minimum dilutions of 300. These dilutions increased to over 3,500 over horizontal distances of 100 - 200 m, suggesting effluent disposed via injection wells is effectively diluted in the groundwater or rapidly mixed upon entering the marine environment.

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CHAPTER 1

INTRODUCTION

Nutrients, Eutrophication, and Algal Blooms

The Earth is currently experiencing the greatest rate of environmental change in human history. The effects of these anthropogenic changes are most pervasive along the coastlines of the world, where human populations have always concentrated and where population growth is at its highest. Increasing population results in increasing demands on natural resources and greater impacts to coastal environments. These impacts include loss of natural habitats, invasion by exotic species, increased pollution in nearshore waters, and increased nutrient loadings in nearshore environments.

Wastewater discharges, fertilizer run-off, and atmospheric deposition have been identified as the most significant sources of nutrients (Boynton et al. 1992). Of these sources, increasing eutrophication of coastal waters due to the discharge of domestic and industrial sewage has been the most studied and regulated. Municipal sewage effluent contains high concentrations of nutrients. The input of anthropogenic nutrients can influence biogeochemical cycling and biotic structure of nearshore marine communities, leading to significant changes in the communities, degradation of water quality, and reduction in the beneficial uses of coastal waters (Boynton et al. 1992). One effect that has been linked to nutrient enrichment is the occurrence of algal blooms (Sewell 1982, Lapointe and O'Connell 1989). While coastal ecosystems can adapt to large amounts of nutrients, the increased loadings often have pushed the growth of algae far beyond normal limits, primarily in response to nitrogen or phosphorus additions. In several locations, including Bermuda (Lapointe and O'Connell 1989) and Australia (Sewell 1982), inputs of nutrients to groundwater from cesspools and septic systems have been implicated as a source of nutrients sufficient to result in benthic algal blooms in nearshore waters.

Consequences of increased algal growth include increased turbidity, loss of coral and seagrasses, and more frequent anoxia. In turn, these perturbations may lead to changes in the structure and function of ecological communities and detrimental economic and aesthetic impacts to human communities. Such changes can have serious impacts to the economy, as well as the environment, in areas such as Hawaii, where coastal recreation and tourism are the mainstays of the economy.

Algal Blooms off Maui

On the west coast of Maui (Figure 1), offshore from the Lahaina District, episodes of massive algal blooms of several species have been reported in local newspapers. Reported sightings of large masses of *Cladophora sericea* and *Hypnea musciformis* began in 1989. *Hypnea* forms floating mats, some of which wash ashore. The subsequent decomposition becomes odoriferous and obnoxious and interferes with the recreational enjoyment of heavily populated tourist beaches. *Cladophora* tends to accumulate near the bottom in large clumps. These algae have reportedly damaged corals, forced fish from their normal habitats, and interfered with recreational diving (Fitzgerald and Tenley 1993).

It is possible that the blooms are triggered and sustained by influxes of essential plant nutrients (nitrogen and phosphorus) emanating from anthropogenic activities. Local environmental activists and public opinion perceive the discharge of sewage effluent to be a potential contributor to the increasing frequency of algal blooms. Other possible nutrient

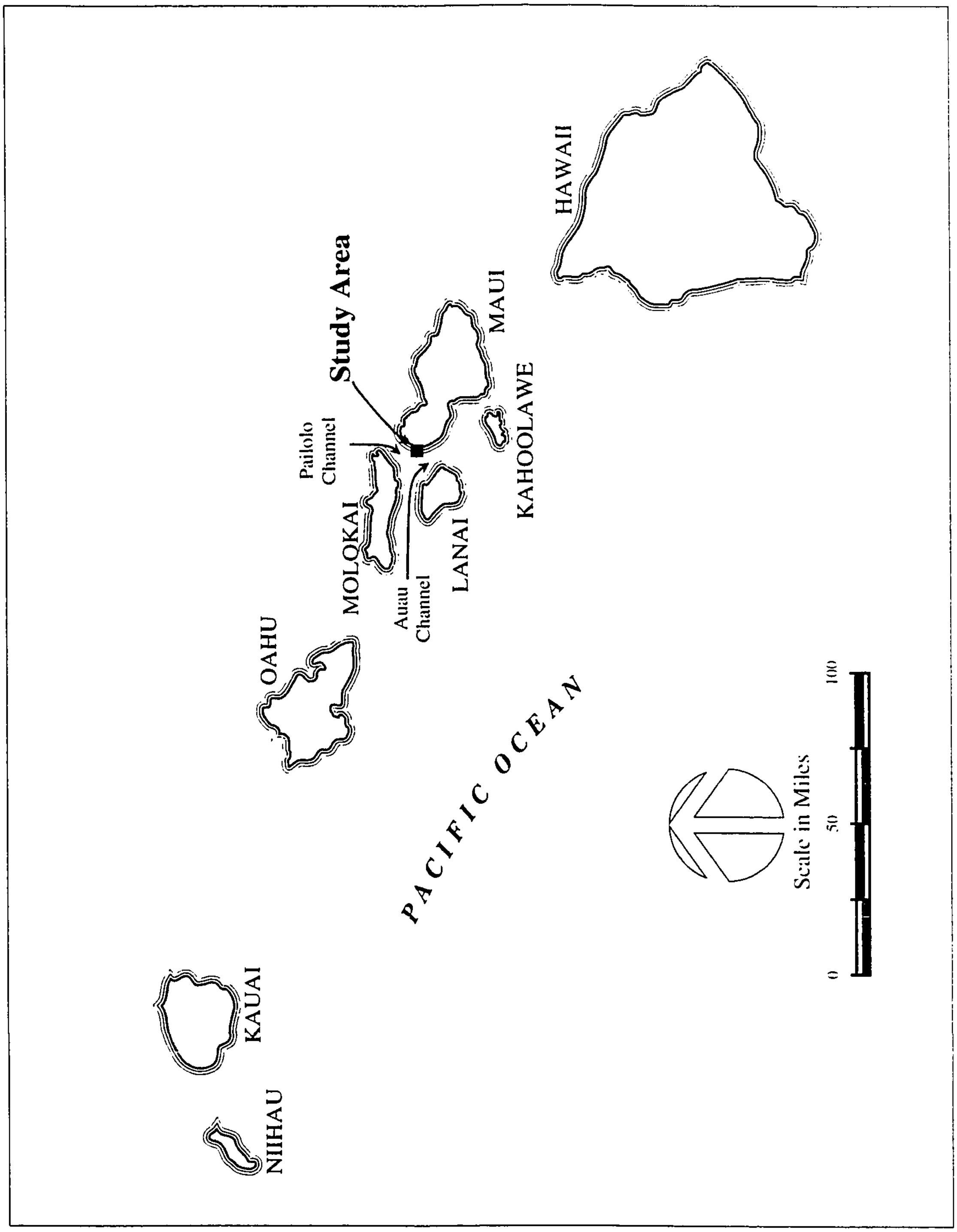


Figure 1. Location of the west coast of Maui, Hawaii.

sources include nonpoint inputs from agricultural, recreational, and urban runoff, and stream flow discharge (Tetra Tech 1993a).

Policy and Regulatory Background

Regulations promulgated by the USEPA in response to the Clean Water Act of 1977 (P.L. 92-500) and subsequent amendments have lead to the adoption of tertiary levels of wastewater treatment and monitoring of receiving waters for most municipal wastewater treatment plants that discharge via outfall pipes into coastal waters. These regulations have substantially decreased the input of solids, toxic chemicals, and nutrients into ocean waters. On the Island of Maui, nearly all of the municipal sewage is disposed of by pumping into nearshore injection wells (Dollar, Peterson, and Raleigh 1992). Under present regulations, if the treated sewage is not discharged directly into the ocean, sewage requires only secondary treatment and chlorination. This level of treatment removes solids but has little effect on the concentrations of nutrients in the effluent.

Local concerns over the occurrence of repeated algal blooms between 1989 and 1992 resulted in four congressional enquiries and the creation of the West Maui Algal Bloom Task Force. The Task Force, conceived by the Hawaiian Department of Health (HDOH), was composed of members from the local community and local government. USEPA also formed its own Maui Algae Team, staffed with specialists from several USEPA programs. The two organizations developed a preliminary watershed management strategy to recommend appropriate studies and subsequent nutrient source abatement actions.

Congress appropriated almost $1 million to fund investigations of the causes of and possible solutions to the algal blooms off Maui (Fitzgerald and Tenley 1993). One part of a research project funded with this appropriation involved an effluent tracer study in nearshore waters to determine if effluent and its associated nutrients from the Lahaina Wastewater Reclamation Facility (LWRF) injection wells were reaching the ocean (Figure 2).

The injected effluent tracer study, a part of a larger assessment of possible anthropogenic sources of nutrients within the Lahaina District of Maui, was funded by USEPA and contracted to Tetra Tech, Inc. Other environmental engineering firms and academic institutions were contracted to perform various other field studies. The Tetra Tech study was composed of several parts, including the determination of the submarine influx patterns of the injected effluent; an assessment of land-based nutrient contributions to coastal waters; a bathymetric survey of the study area; water column profile measurements of temperature, salinity, and density; and water sampling to measure nutrient levels. All efforts, except for the assessment of possible nutrient sources, were combined into an extensive field study. The focus of this thesis is one part of the overall study by Tetra Tech, namely, to investigate where the effluent enters the coastal waters.

The Objectives of the Study

The LWRF effluent pumped into injection wells was suspected of being a large source of nutrients to the near-coastal waters. This effluent may eventually be released to the marine environment, possibly elevating levels of nitrogen and phosphorus in the nearshore waters. These wells discharge approximately 19×10^6 liters/day (5 million gallons per day [mgd]) of secondary treated sewage effluent two hundred feet below ground surface. The effluent enters the saline groundwater below the basal groundwater lens at the site. It is most probable that injected wastewater ends up in the ocean, although

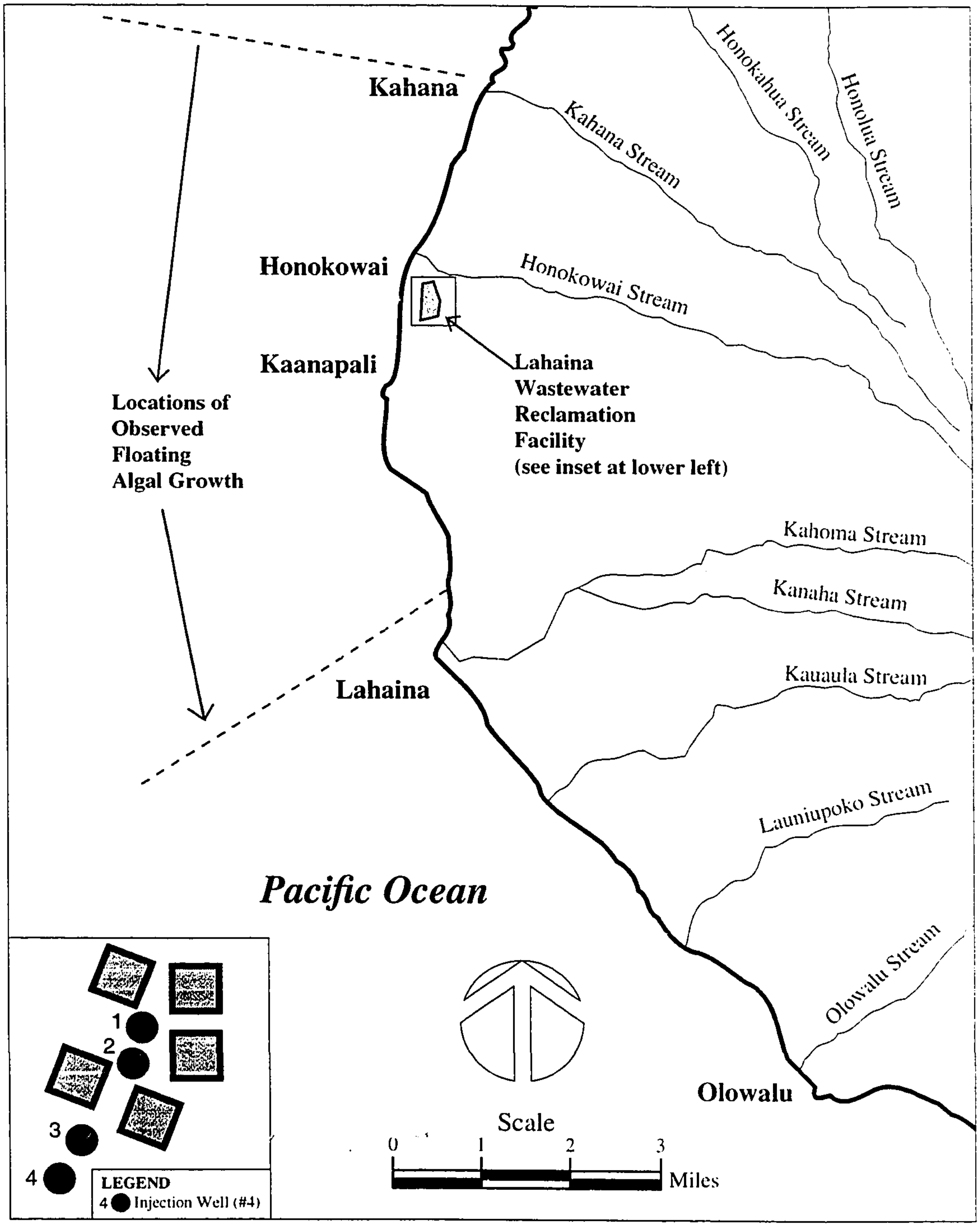


Figure 2. Location of area of reported algal bloom sightings, and Lahaina Wastewater Reclamation Facility (Tetra Tech 1993a).

its ultimate offshore transport and dispersal are not definitely known. In response to the lack of research into the uncertainties surrounding the transport and fate of the injected effluent, and to efficiently use the available research funds, a sequential set of objectives were developed for the overall project. They included water column measurements of salinity, temperature, and density, water sampling for nutrients, and detecting and mapping the submarine influx areas of the effluent.

The objectives of this thesis focus only on the study of the patterns of influx into coastal waters of sewage effluent from the LWRF, after injection into the groundwater and transportation beyond the shoreline. The specific objectives can be defined as follows:

1. Determine if effluent, discharged into injection wells at the LWRF, can be detected in the nearshore waters.

2. Locate and delineate the effluent seep(s) on the sea floor.

To accomplish these objectives requires the design of an appropriate field study using a fluorescent dye. The dye, added to the treated effluent, acts as a conservative tracer or marker of the treated effluent as it is transported by the groundwater and enters the marine environment.

The final product of this study was a scaled plot covering the study area delineating areas of elevated fluorescence, indicating the presence of fluorescent tracer and the submarine influx pattern of the associated effluent. The plot is computer generated from precision navigation data, seafloor bathymetry, and fluorometry.

As part of the overall project, of which this specific thesis study is a part, this map of elevated concentrations is an essential component. In conjunction with water column density data and nutrient analyses of water samples collected from the seafloor, the information this map contains was designed to be a cornerstone upon which potential regulatory actions will be based. An essential part of the requested study was to design the field study is such a way that the methodology could be carried over to any other injection well investigation, independent of location, flows, or effluent characteristics.

CHAPTER 2

RELATED RESEARCH

The primary focus of this study is the investigation of the submarine distribution pattern of treated wastewater discharged from a near-coastal municipal sewage treatment facility through injection wells. Many field studies, modeling efforts, and theoretical analyses concerning more conventional wastewater disposal via engineered ocean outfalls have been carried out for sites throughout the world. Few studies have investigated methods for mapping injection well discharges into coastal waters. The following literature review discusses some relevant studies pertaining to either subsurface injection, submarine groundwater discharges, or marine discharge of wastewater and their applicability to this proposed study. An emphasis of the literature search was to identify previous studies carried out in the Hawaiian Islands. The final section reviews the literature on effluent and groundwater tracer compounds to investigate the most appropriate types of tracers for this application. This area also appears to have limited study, especially for transport periods of more than a few hours to a few days.

Injection Wells

Oberdorfer and Peterson (1982) and Peterson and Oberdorfer (1985), investigating injection well use in Hawaii, report that more than 500 injection wells were constructed between 1965 and 1985. The majority are used for disposal of treated sewage effluent and are small, shallow, and privately owned. However, several municipal injection well facilities have been constructed on Oahu and Maui. These wells are generally deeper and have the capacity to accept several hundred thousand to a few million gallons of wastewater per day. They identify two distinctly different types of operational problems with this method of disposal of sewage effluent. The first, which corresponds to the focus of this study, is contamination of potable groundwater and shallow coastal waters. It is mentioned only briefly and is not considered a significant problem in the operation of the injection well. The second problem, which constitutes the major effort of their study, is clogging and the rapid reduction of injection capacity in the immediate vicinity of the wells. This decrease in efficiency of the wells will not be addressed in this study.

Modeling studies of varying complexity have been conducted for the Kahului Wastewater Reclamation Facility (Burnham et al. 1977, Larson et al. 1977, Mercer et al. 1980). This facility is located only 40 km (25 mi) north of the Lahaina Facility, on the north coast of Maui. The results of these modeling studies indicated that the Kahului effluent rises after entering the groundwater. This is due to the buoyant forces created by the lower density of the effluent relative to the saline groundwater. The effluent in the Kahului groundwater model spreads radially as it rises, forming a conical plume, the top of which is displaced toward the down-gradient direction (seaward). The final displacement of the effluent plume was strongly dependent on the assumed geology of the model. Groundwater modeling applied to measured conditions at Kahului predicted that the effluent was likely to rise along the freshwater/saltwater interface. However, because of the volume of effluent disposed, this interface, which would normally intersect the surface at the shoreline, was displaced 600 m (2,000 ft) offshore (Mercer et al. 1980).

Although the results of the Kahului modeling study have been presumed by previous investigators to apply to the LWRF effluent, the modeling study has never been validated with field observations or data. There is also much uncertainty in the migration pathway of the LWRF effluent, and it is unclear whether the Kahului modeling results can be extrapolated to the LWRF given the geologic and hydrogeologic differences at these sites. For example, the lava rock aquifer at the Kahului site has been described to be overlaid by residual soil, clay, coral reef deposits, and marine sand that form a low-permeability cap rock. This caprock is thought to restrict or confine the aquifer (Burnham et al. 1977). Another difference is that water depths at the Kahului site, within 1,500 m (5,000 ft) of shore, are quite shallow. For example, the 5.5-m (18-ft) depth is located approximately 900 m (3,000 ft) offshore. Conversely, at the LWRF the presence of a cap rock that may confine the aquifer is unknown; the coastal sediments are believed to be extremely permeable (Stearns and MacDonald 1942, Yamanaga and Huxel 1969) and 5.5–m (18-foot) water depths occur within 60 m (200 ft) of the coastline.

Two modeling studies of the transport of effluent injected into saline groundwater were identified. One used a physical, bench-scale, sand-packed hydraulic model (Peterson and Lau 1974), and the other was a numerical model using a simple two-phase system (Wheatcraft and Peterson 1979). The conclusion, in both studies, suggested that concentrated effluent would migrate vertically to the interface between the saline water and freshwater and then travel along the interface until it was discharged with the groundwater close to the shoreline. The numerical approach also indicated that some effluent, diluted by mixing with the saline groundwater, would travel horizontally, displaying little upward movement due to buoyant forces. The physical model was designed to investigate the mechanics of density stratification on the transport of effluent in an essentially homogeneous and permeable substrate. The numerical approach, although recognizing the anisotropic character of the aquifer, is limited by the same constraints; the effects of a complex geology could not be addressed. Preferential flow paths and impermeable or semi-permeable strata as may exist in the Lahaina area are expected to have significant effects on the effluent flow patterns within the aquifer.

Larson et al. (1977) used a simplified model that combined a numerical solution with an analytical solution. They found that, depending on the local geology, seaward flowing freshwater could extend up to 600 m (2,000 ft) offshore in the vicinity of the injection well.

Treweek (1985) presents a review of municipal subsurface wastewater disposal facilities, where disposal of the wastewater is achieved by either surface spreading or subsurface injection. Surface spreading operations dispose of primary, secondary, or tertiary treated wastewater, typically to recharge groundwater aquifers beneath the spreading basins. Wastewater treatment using only primary, or secondary treatment methods are often suitable for spreading operations due to in situ treatment that occurs in the subsurface soils. Most of the pollutant removal occurs in soil layers where biological activity is typically greater than in groundwater.

Subsurface injection operations typically dispose of tertiary treated wastewater. At Palo Alto, California (Roberts 1985); Los Angeles, California (Argo and Cline 1985); and Nassau County, New York (Oliva 1985); tertiary treated wastewater effluent is used to recharge groundwater aquifers. The Los Angeles injection facility was established to provide both a barrier to seawater intrusion and to replenish the groundwater aquifers used for potable water supply. The Palo Alto injection facility recharged shallow aquifers primarily to prevent salt water intrusion; the Nassau County facility recharges water supply aquifers to replenish the aquifers and mitigate a decline in the groundwater table caused by excessive pumping of groundwater.

Subsurface injection operations at West Palm Beach, Florida (Amy 1980) dispose of tertiary treated effluent 900 m (3,000 ft) below the surface, into a highly fractured dolomite formation. The West Palm Beach facility uses deep-well injection to dispose of effluent below confining layers and is not intended to recharge local aquifers or discharge into offshore coastal waters.

Monitoring has been conducted at the Palo Alto, Los Angeles, Nassau County, and West Palm Beach injection operations. Roberts (1985) used chloride as an effluent tracer to monitor effluent migration through the aquifer. Extensive monitoring networks were also established at Los Angeles and Nassau County facilities. These monitoring networks have not detected any significant degradation of groundwater in the aquifers used for disposal. Monitoring in aquifers, 700 m (2,300 ft) below ground surface, above the confining layers at the West Palm Beach Facility has not detected any change in water quality since the inception of the 0.9 m^3/sec (20-mgd) effluent disposal into the dolomite formation 900 m (3,000 ft) below ground surface.

Submarine Discharge Mapping

Canadian researchers have developed techniques for collecting information to detect where and at what rate groundwater infiltrates into rivers and shallow lakes and estuaries (Lee 1985, Lee and Welch 1989). These methods are complex, labor-intensive, and expensive. Sediment probes, in situ measurement of variations in electrical conductance and temperature measurements are required. Divers are needed to measure the in situ (i.e., on the lake bed) hydraulic heads and seepage rates. Swedish investigators have used similar seepage flux and electrical-conductance sediment probe methods to map freshwater discharge areas into coastal marine waters (Vanek and Lee 1991). However, these methods are only viable in shallow water. In the Swedish study water depths did not exceed 2 m (6 ft); in the Canadian studies, 8 m (26 ft) was the maximum depth.

Ocean Discharges

Historical observations of wastewaters discharged into open coastal waters are reviewed by Myers (1983). The impact of wastewater discharges varies, depending on the volume and characteristics of the waste and the receiving waters. Municipal wastewaters in coastal areas are typically treated and discharged via marine outfalls either into estuaries or on the oceanic continental shelf. Two open coastal water regions in the United States receiving significant quantities of municipal wastes have been extensively studied: the New York Bight and the Southern California Bight. The New York Bight is a wide continental shelf with typically shallow waters nearshore. In contrast, the Southern California Bight is a rugged, narrow continental shelf with numerous submarine canyons.

Monitoring studies in the Southern California Bight have been undertaken by many organizations to measure contaminant concentration in the water column, the sediments, and biota (Mearns and Young 1983; Southern California Coastal Water Research Project 1994). In addition, many monitoring programs have been established to assess the condition of the benthic environment, epibenthic fish and macroinvertebrates, and the pelagic environment in the Southern California Bight. Field studies documenting the dispersion of plumes from the Los Angeles County outfall at Palos Verdes indicate that the plumes are submerged between the 20-m and 40-m (65-ft and 130-ft) depths (Jones et al. 1991). The lateral width of the plumes varied with depth, and ranged from tens to hundreds of meters. Dilution of the plume increased at shallower depths. The monitoring program shows that benthic infaunal abundance and biomass increase near a discharge while the number of species and diversity generally decreases. It has been postulated that discharges stimulate the phytoplankton community by contributing nutrients, especially ammonia, into the mixed surface layer.

In October 1987 and June 1988, field tests were conducted on a 0.9-m^3/sec (20-mgd) ocean outfall in San Francisco, California (Roberts and Wilson 1990). Rhodamine WT dye was added to the effluent and dye concentrations were monitored continuously using flow-through fluorometers. Dye profiles indicated that the plume vertical thickness ranged from 10 to 15 m (30 to 50 ft), the height of rise of the plume varied from 5.5 to 23 m (18 to 75 ft), and the measurable dilutions ranged from 115:1 to greater than 1,000:1 (Roberts and Wilson 1990). Tidal current changes and turbulence in the ocean waters during the San Francisco field test resulted in a plume that changed orientation markedly during observation, to such an extent that data obtained at different times were difficult to correlate.

Hydrological Tracers

A hydrologic tracer is any substance whose physical, chemical, and biological properties are such that the tracer can be identified and observed to provide information about a hydrologic system. An ideal hydrologic tracer should be detectable at low concentrations, inexpensive, easily detectable, biologically and chemically conservative, non-volatile, and environmentally benign. This section reviews a number of possible hydrologic tracers, giving references to their use in hydrologic field studies, and brief reviews of favorable and unfavorable attributes.

A number of salts, such as sodium dichromate, sodium iodide, sodium chloride, lithium chloride, and potassium chloride are often used as tracers in hydrologic and hydrogeologic studies (Gaspar 1987). The anion chloride was used as a tracer of wastewater effluent in aquifers at Palo Alto, California (Roberts 1985), and in the Dan Region, Israel (Idelovitch and Michail 1985). Chloride is not a suitable tracer at the LWRF as the wastewater may be subject to interference by basal ground waters and surface waters that exhibit low chloride concentrations. Also, given the wide range of dilution possible in the ground water and coastal waters and the background concentrations of anions in coastal and fresh waters, anion salts are not likely to be useful in this study.

Coliform bacteria are common biological tracers used to monitor surface water quality. Pigmented bacteria have been used in surface water studies (Oetzel et al. 1991) and in karst ground water systems. However, microbial transport in porous media may not be conservative. Thus, microbes are more often used as qualitative rather than quantitative tracers (Gaspar 1987).

The environment contains a series of isotopes such as deuterium (^{2}H), tritium (^{3}H), carbon-14 (^{14}C), and oxygen-18 (^{18}O), which label water naturally, making it possible to trace water through the hydrologic cycle (Gaspar 1987). Several investigators have used these naturally occurring compounds in recent years to trace waters in the coastal environment and ground waters. Bowser and Ackerman (1992) used ^{2}H and ^{18}O data from inflowing ground water, atmospheric precipitation, and lake water to calculate water budgets for several lakes in Wisconsin. Scholl et al. (1992) used deuterium and ^{18}O data to trace ground water in the Kilauea Volcano Area, Hawaii. Karr et al. (1992) used deuterium and ^{18}O measurements and salinity/temperature data to study water masses and mixing in ocean waters on the Amazon Shelf. These isotopic tracer studies require the fingerprinting of all water sources and measurements of deuterium and ^{18}O to estimate mixing processes in the water body of interest. The isotopes are expensive and require technology-intensive procedures.

Organic chemical tracers, such as fecal sterols, chlorinated pesticides, and hydrocarbons have been used as indicators of municipal wastewater effluent in marine systems. The fecal sterol coprostanol has been used to trace effluent near a sewage outfall at Cocoa, Florida (Holm and Windsor 1986), and as an indicator of sewage contamination in the New York Bight (Hatcher and McGillivary 1979) and in Sarasota Bay, Florida (Pierce and Brown 1984). Alkane and aromatic hydrocarbons have been used as a sewage tracer in Chesapeake Bay (Brown and Wade 1984) and Cocoa, Florida (Holm and Windsor 1986). The success from using organic chemicals as sewage tracers depends on the concentration of these markers in the effluent, their background concentration in receiving waters and sediments, the duration of the sampling period, and the dimensions of the study area.

Coprostanol concentrations, for example, were well above background in dissolved and particulate fractions of samples taken from the water column at a municipal wastewater outfall near Cocoa, Florida. However, the dissolved and particulate fraction in the water column decreased rapidly away from the outfall such that samples only 9 feet from the outfall were below detection (Holm and Windsor 1986).

Fluorescent dye tracers are used extensively in hydrology and hydrogeology because of their unique characteristic of being able to be detected in situ at very low concentrations of between 10^{-10} and 10^{-12} g/mL or at ratios of 0.1 parts per billion to 1 part per trillion (Gaspar 1987). They are relatively cheap, nontoxic, not mutagenic, and are relatively conservative in freshwater and saltwater environments. Several dyes from this family have been used under different conditions.

Applications of dye tracing techniques in engineering are common. Infiltration measurements, studies for the calibration of flow control and measuring structures, and circulation studies have been reported by many investigators. Oil field tracing, time of travel measurements in surface waters, groundwaters and engineered structures, flow measurements in harsh environments, such as under ice cover or in steep rocky channels are other reported successful applications of fluorescent tracers (Smart and Laidlaw 1977).

Oberdorfer and Buddemeier (1986) used Fluorescein and Rhodamine B as tracers to investigate short-term vertical and horizontal permeability and flow patterns in a coral reef on Australia's Great Barrier Reef. Fluorescent tracers have been used in karst areas for many years to trace groundwater flow patterns and to gauge discharge rates in both aquifers and surface waters, such as rivers and streams. Brown et al. (1969) used Rhodamine WT to determine the water budget of a karst system in southern British Columbia. They chose Rhodamine WT because it is not easily adsorbed by clay and few interfering substances occur within the groundwater. The U. S. Environmental Protection Agency recommends the use of Rhodamine WT in dye studies to estimate concentrations of effluent in the receiving waters around ocean outfalls of municipal wastewater treatment plants. These effluent concentrations are then used together with laboratory tests to determine the toxicity of the effluent /receiving water mixtures at different locations around the outfall (U.S. Environmental Protection Agency 1989).

A qualitative assessment of eight different fluorescent dye was performed using laboratory and field experiments by Smart and Laidlaw (1977). Their purpose was to determine the utility of the dyes for qualitative tracer studies in different environments. The parameters considered included minimum detectability, reactivity in water, photochemical and biological decay rates, adsorption losses to sediments, toxicity to humans and aquatic organisms, and cost. Rhodamine WT, and lissamine FF, were the two tracers dyes recommended. Lissamine FF was recommended because it was found to be extremely stable and resistant to adsorption losses. However, it is over 9 times more expensive than Rhodamine WT, which was the second most cost effective dye studied. And, although it was not the most conservative dye, it was described as having no serious disadvantages compared to the other dyes studied.

Deaner (1973) studied the effects of chlorine on four different fluorescent dyes, including Rhodamine WT. Chlorine residuals of 2 to 9 mg/L are normally found in sewage effluent disinfected by the application of chlorine before discharge from the treatment facility. This study concluded that little effect was observed on Rhodamine WT at normal levels of chlorine residual, but at higher chlorine concentrations (greater than 13 μg/L) substantial quenching of fluorescence occurred.

Summary

Marine discharge via subsurface injection is not a common municipal wastewater effluent disposal alternative. It appears that this disposal method is somewhat unique to the Hawaiian islands and that it did not become a common practice until the late 1960s and early 1970s. The only discharge systems found elsewhere in the world that resemble this subsurface disposal method are small-scale extended aeration systems that are common in coastal tourist areas. These small-scale systems dispose of wastewater into the subsurface with the intent of ultimately discharging offshore. Septic system effluent is typically discharged into shallow wells in Hawaii (Oberdorfer and Peterson 1982) and into shallow trenches in islands and coastal areas of the Mediterranean (Christoulos and Andreadakis 1987). Except for the locations in Hawaii, no references were identified for studies of large municipal systems that dispose of effluent in the subsurface with the intent of discharging into nearshore coastal waters. Identified methods of detecting and mapping submarine groundwater influxes are limited to shallow waters and are effective only over areas in the order of a few thousand square meters. The use of fluorescent dyes as effluent tracers is common for engineered outfalls, where the study period is of the order of 24 hours or less. In groundwater studies, the duration of the study is generally even shorter. For surface water dilution and time-of-flow studies, durations are rarely longer than a few hours, and for engineering studies of flow control structures the study period can be as short as a few minutes.

CHAPTER 3

EXPERIMENTAL APPROACH

To accomplish the objectives of this study, a field study was designed using a fluorescent dye as a conservative tracer of the treated effluent as it transported through the groundwater and enters the marine environment. The tracer was added to the effluent immediately before discharge into Injection Well No. 2 of the LWRF (Figure 2). A survey vessel equipped with a submersible pump and fluorometer sampled the coastal waters offshore from the wastewater treatment facility in order to detect the tracer.

Specific design details were governed by three factors. The first factor was the lack of knowledge about the transport of the discharged effluent while underground. Possible ranges of values for residence time, transport pathways, dilution factors, permeability, and adsorption characteristics were allowed for in the design. The second factor considered was the combination of the environmental characteristics of the effluent, of the tracer, of the groundwater, and the ocean water through which the effluent and tracer traveled. A wide range of physical and chemical properties were considered for both the study design and the tracer attributes. The third factor was the external limitation of funding for the project and the subsequent constraints to the approach used in this thesis study. This factor influenced the possible combinations of sampling density, sampling frequency, the length of the field study, and the size of the study area.

Tracer Selection

The following factors strongly influenced the choice of a natural marker or tracer compound for the LWRF effluent:

1. Effluent, groundwater, and coastal water composition
2. Mixing, transport, and dilution within the aquifer
3. Unknown discharge location in the coastal waters
4. Environmental impact of the marker compound.

Inorganic chemical tracers could not be used because of possible interference between the effluent and other freshwater sources and the limited resolution resulting from dilution of the tracer in coastal waters. Radioactive tracers were not selected for use because of possible environmental concerns of release of a radioactive element in the west Maui coastal waters. Biological tracers were considered but not selected for use as a quantitative marker because of (1) the high mortality rate of microorganisms following chlorination of the effluent, (2) the possible filtering of biological tracer organisms during transport through the aquifer, (3) the nonconservative nature of biological markers in the coastal waters, and (4) the potential for interference between the LWRF effluent and septic systems. Naturally occurring alkane and aromatic hydrocarbons, and chlorinated hydrocarbons were not good choices for a tracer because these compounds were not present in significant quantities in the LWRF effluent.

The naturally occurring effluent markers that could potentially be used as tracers were the isotopes deuterium (^{2}H) and oxygen-18 (^{18}O). In order to use isotopes as a tracer, however, it would first be necessary to fingerprint all hydrologic sources in the vicinity of the LWRF. This would include the basal groundwater, surface waters, the coastal waters, and the LWRF effluent to verify a unique composition for the LWRF effluent. However, because the submarine effluent entry location was not known, a random sampling of a large area in the West Maui coastal waters would be required along with costly laboratory analyses of these samples to locate the plume. Thus, isotopes were not selected for use as LWRF effluent markers because these tracers could only be discretely sampled; the sampling and laboratory detection methods were costly, and the results would not be immediately available in the field.

The fecal sterol coprostanol was not chosen for a marker in this study because of the partitioning of this chemical to particulate phases and the small spatial scale over which this marker may be detected in the water column.

The fluorescent dye tracers were thought to be the best choice for use in this study. Fluorescent dye tracers have the following attributes:

1. are easily detected
2. are required in small quantities
3. are inexpensive to detect
4. will not modify hydraulic characteristics of system
5. will not volatilize or react (conservative)
6. are absent in all hydrologic sources in West Maui
7. are environmentally benign.

The fluorescent dyes are particularly useful for this study because real-time detection and measurement methods are available and can be used in a continuous sampling mode. The detection instrumentation can be configured to report essentially real-time analytical results that allow for immediate feedback into the choice and refinement of sampling locations. For example, if the fluorometer indicates an elevated concentration in one area, the field team can immediately respond to this data and conduct a more thorough study in the region for elevated fluorescence concentration.

In high concentrations (1 to 5 mg/L), fluorescent dyes are visible to the naked eye. Various methods are employed to measure samples below the visible range. A flow-through fluorometer with an internal data logger was selected for this study. This instrument is capable of detecting fluorescence at concentrations as low as 0.01 μg/L.

The only drawback to using a fluorescent dye as a tracer is the possible weak to moderate adsorption to soil particles as the dye moves through the aquifer. The adsorption can be minimized by choosing a dye that has the smallest sorption coefficient.

The following dyes were considered and rejected for the following reasons. Uranine (Fluorescein) is not used because it degrades more rapidly in sunlight and adsorbs more strongly to solids and sediments. Rhodamine B is not chosen because it adsorbs to sediments more strongly than Rhodamine WT. After consideration of the alternatives, the fluorescent dye Intracid Rhodamine WT (Crompton and Knowles, Reading, Penn.) was chosen as the tracer for the effluent. Rhodamine WT is chosen because it is the weakest adsorbing of the commonly used fluorescent tracers. This dye has been used extensively in both groundwater and coastal water applications, and it is commonly used to trace sewage in distribution lines, sewers, and treatment plants. Rhodamine WT is often referred to as the tracer of choice for groundwater studies (Gaspar 1987).

The Impact of Environmental Characteristics on the Field Sampling Design

No previous studies relating to the transport and fate of the effluent from the LWRF have been performed. Consequently, it was necessary to consider many uncertainties in the design of the field study. The study design was based on the addition of an artificial tracer to the effluent, prior to discharge into the injection wells, which could be detected in seawater at anticipated dilutions of up to 10,000 times the concentration in the effluent.

Several features of the coastal and groundwater systems in the Lahaina area may influence the transport of the LWRF effluent. The following groundwater and coastal water characteristics were considered in the design of the sampling plan.

Groundwater

Inland from the coast in the Lahaina District, the groundwater system is classified as an unconfined basal aquifer comprised of horizontally extensive lavas on the flank of the west Maui volcanic complex (Stearns and MacDonald 1942). Along the coastline, the aquifer is classified as an unconfined basal aquifer, consisting of sedimentary materials deposited in a narrow strip along the coastline. Souza (1981) describes the Lahaina District as:

> a low-head coastal area where little or no coastal sediment restricts the flow of fresh ground water to the ocean. Unlike continental alluvial aquifers, the water table near the coastline is not a subdued reflection of surface topography, and water levels increase only 8 ft above sea level 3 miles inland.

This is a hydraulic gradient of only 0.0005 m/m.

The regional coastal groundwater flow has been estimated at approximately 0.08 m^3/sec/km (3 mgd/mile) (M&E Pacific 1991). Thus, the 0.2 m^3/sec (4.5 mgd) flow of LWRF effluent was assumed to spread out laterally along the coastline a maximum of 2.5 km (1.5 miles or 4.5 mgd divided by 3 mgd/mile), assuming the effluent rises into the regional flow field in the basal lens (Figure 3).

However, the presence of confining layers, such as buried soil horizons or volcanic ash layers (Figure 4), or dikes, or preferential flow paths, along lava tubes or clinker beds,

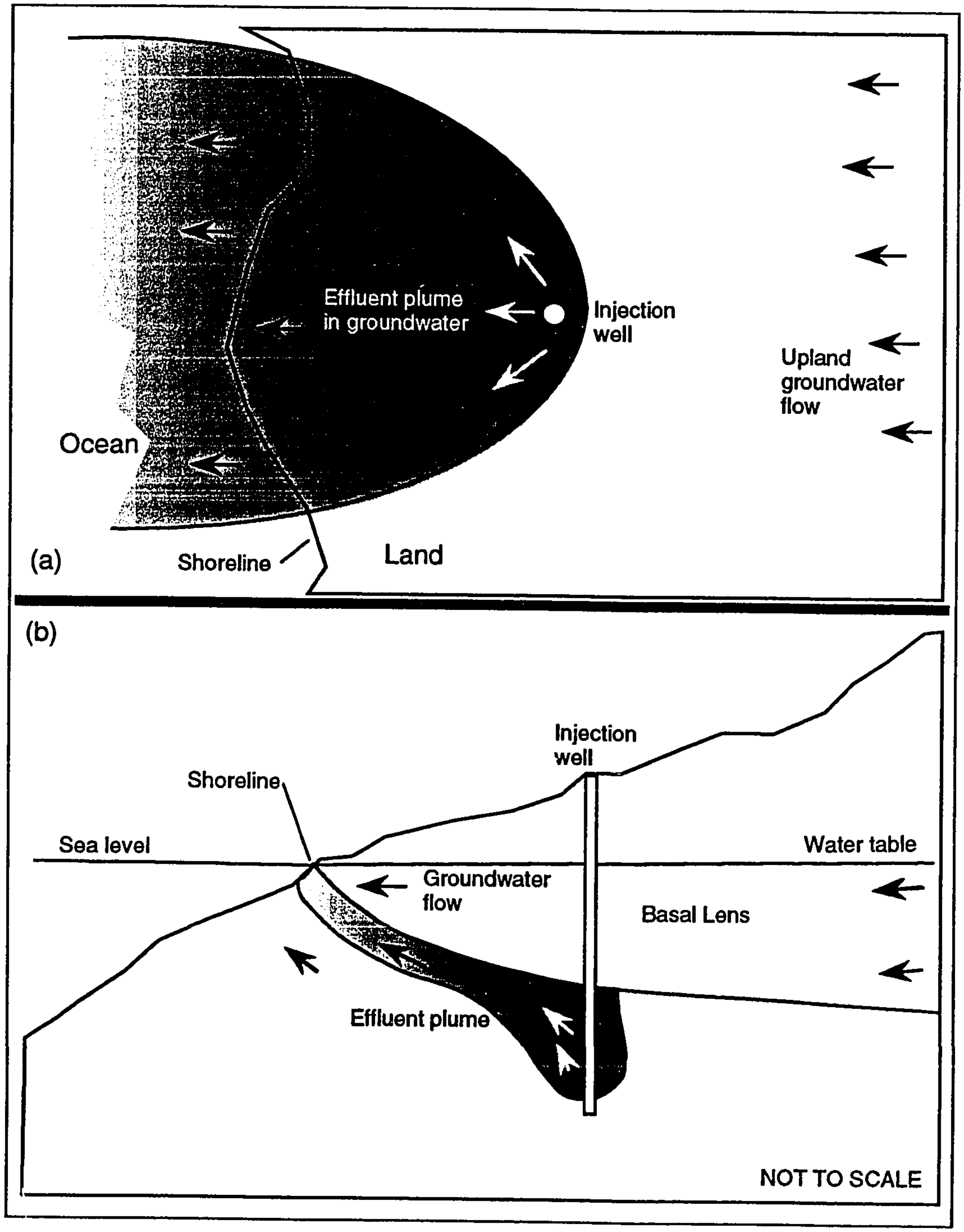


Figure 3. Conceptual model of a uniform and unconfined groundwater flow pattern. (a) Plan view. (b) Cross-sectional view.

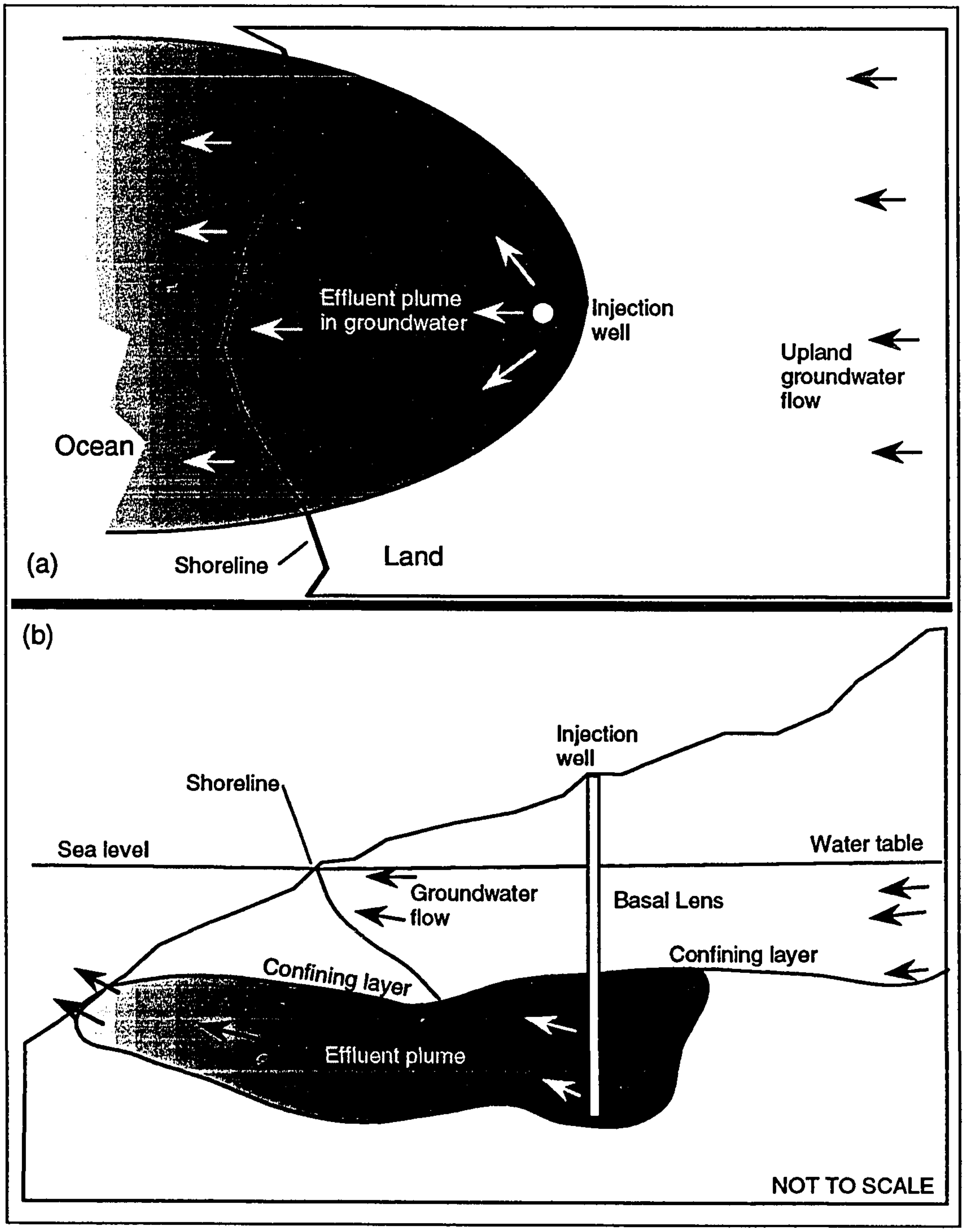


Figure 4. Conceptual model of a uniform and confined groundwater flow pattern. (a) Plan view. (b) Cross-sectional view.

offer the possibility of the effluent being constrained either horizontally or vertically (Figure 5).

The effluent is discharged into Well No. 2 of the LWRF and then into the saline aquifer below the basal aquifer at depths of between 30 - 61 m (95 - 200 ft) below the surface (Brown and Caldwell 1991). The depth of water at which the effluent discharges into the ocean is unknown. This will be influenced by factors such as the buoyancy of the wastewater, which may cause the wastewater to rise in the groundwater, the permeability of the strata, the possible occurrence of low permeability confining layers, and the dip and layering of the lava beds, which may cause the wastewater to remain in deeper strata.

Coastal Waters

The Auau Channel, between Maui and Lanai (see Figure 1), has an average maximum depth of about 90 m (300 ft). Offshore from the LWRF, water depths reach 30 m (100 ft) within 1,000 m (0.8 mi) off the shore and then increase more slowly to 90 m (300 ft) at 4,500 m (3 mi) offshore. Very little information on ocean current patterns has been reported. Nearshore currents in the Auau Channel have been reported to flow predominantly to the north at speeds of 12-25 cm/s (and the alongshore current is tidally reversing every 6 hr with an average flow of 13 cm/s (Grigg 1983). However, close to the shore, current flows are complicated by tidal, wind, and shoreline effects.

If the effluent emerges from a single point or small number of points on the seabed, the resulting plume is likely to elongate in a north-south direction due to the prevailing currents. The plume is also likely to extend a greater distance north of the influx location than it will to the south because of these currents. This effect was evident from the results of a dye study performed by Grigg (1983). Under these circumstances, the plume may rise

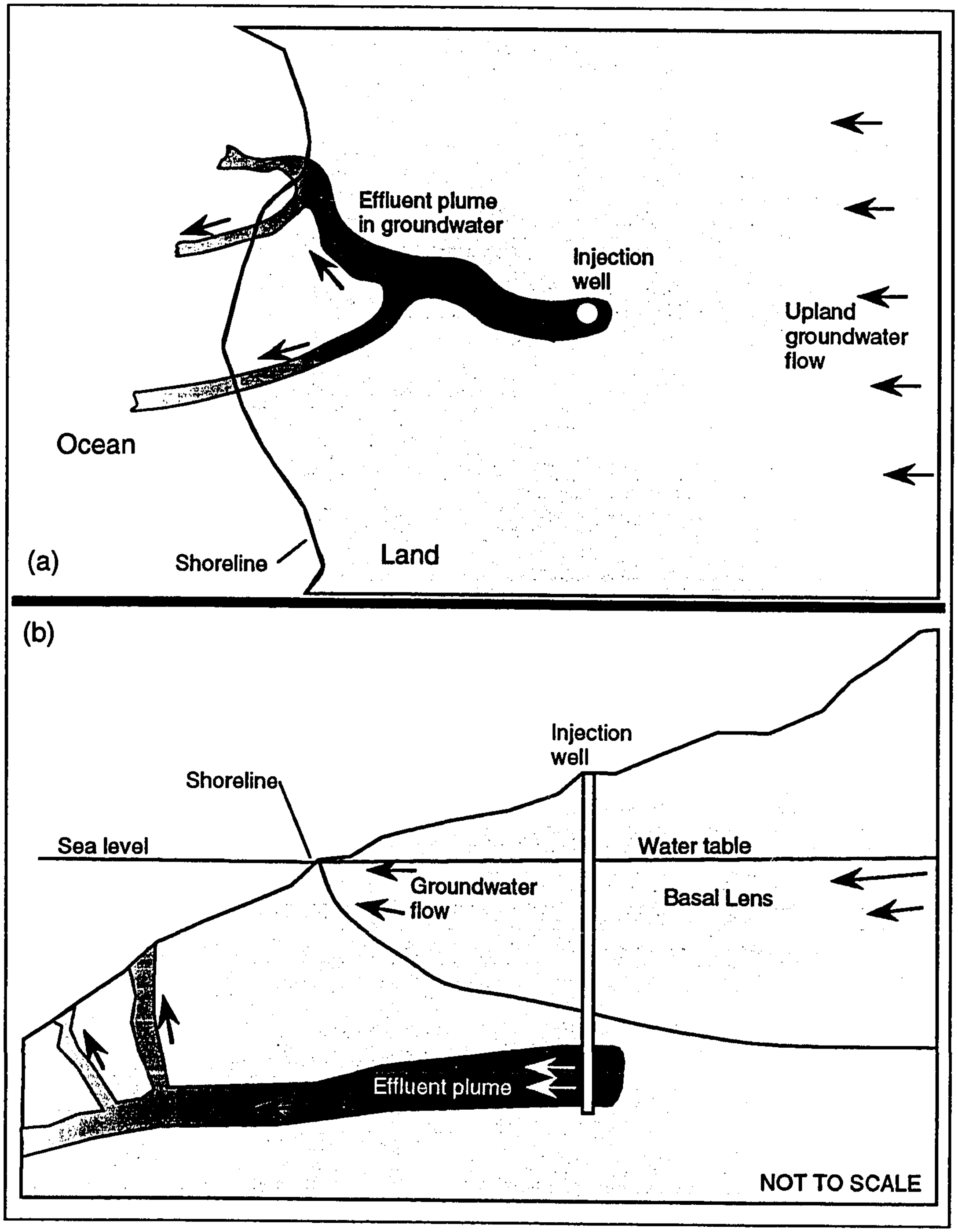


Figure 5. Conceptual model of groundwater flow through preferential channels. (a) Plan view. (b) Cross-sectional view.

to the bottom of the thermocline, if one has formed, or the plume may even surface if the discharge zone is a submarine spring. If the effluent enters the coastal waters from one or several diffuse influx zones, the plume(s) are more likely to remain submerged and thus detectable only within a few feet of the bottom. The horizontal turbulent shear flow zone close to the sea floor would result rapid mixing of the effluent to background levels before the relatively weaker buoyancy forces could exert appreciable upward movement.

Injection Wells

An average of 0.2 m^3/sec (4.5 mgd) of effluent is discharged from the LWRF by gravity flow into four injection wells at the facility. The four injection wells are located approximately 600 m (2,000 ft) from the coastline (Figure 2).The four wells are identified as Numbers 1, 2, 3, and 4, starting from the northernmost well. Each well has a diameter of 508 mm (20 inches) and is drilled to a depth of 61 m (200 ft). The surface elevation at the facility is approximately 9 m (30 ft) above mean sea level; the well bottom elevation is approximately 52 m (170 ft) below mean sea level.

Although the original rated capacity of the injection wells was 0.09 m^3/sec (2.0 mgd) for Well No. 1 and 0.33 m^3/sec (7.6 mgd) for Well Nos. 2, 3, and 4, the capacities have decreased significantly over the years (Brown and Caldwell 1991). Present capacities, as determined by LWRF staff, are estimated to be between 0.02 - 0.07 m^3/sec (0.5 - 1.5 mgd) for Well Nos. 1, 3, and 4. The estimated capacity of Well No. 2 is between 0.13 - 0.22 (3.0 - 5.0 mgd). This is greater than the combined capacity of all the other wells and two to ten times greater capacity than any one other well. The LWRF operations staff take advantage of this high capacity by discharging more than half the effluent to Well No. 2 on a routine basis.

This substantially higher capacity of Well No. 2 also suggests that the discharge section of the well intersects one or more regions of very high permeability. Because the other wells do not exhibit the same capacity, it is possible that the zone of high permeability is confined horizontally. A possible explanation for this that the well intersects a lava tube. If this was the case, a preferential flow path for the effluent through the aquifer could be possible. The considerable difference in capacity between the wells also indicate that the geological structure is not uniform and thus unconfined and uniform flow in the groundwater system is not certain.

Size of the Study Area

The maximum depth of the sampling area, 70 m (200 ft), was somewhat greater than the maximum depth of the wells, 52 m (170 ft). The possibility of a lava tube being present and acting as a preferential flow path required that the study area extend away from the shoreline, as shown in the conceptual model in Figure 5. Although it was possible that the seaward dip of the strata and the likelihood of impermeable layers within the strata may result in the effluent emerging at depths below the injection depths, the 70-m (200-ft) isobath was chosen as a practical seaward limit of the study area.

The maximum distance offshore of the 70-m (200-ft) isobath was approximately 2,800 m (9,200 ft). The sampling area, assuming a maximum effluent injection rate of 4.5 mgd, was estimated to extend along the shoreline no further than 2,800 m (9,200 ft) (Tetra Tech, 1993b). This results in a study area of 7.8 km^2 (3.0 mi^2), as shown in Figure 6.

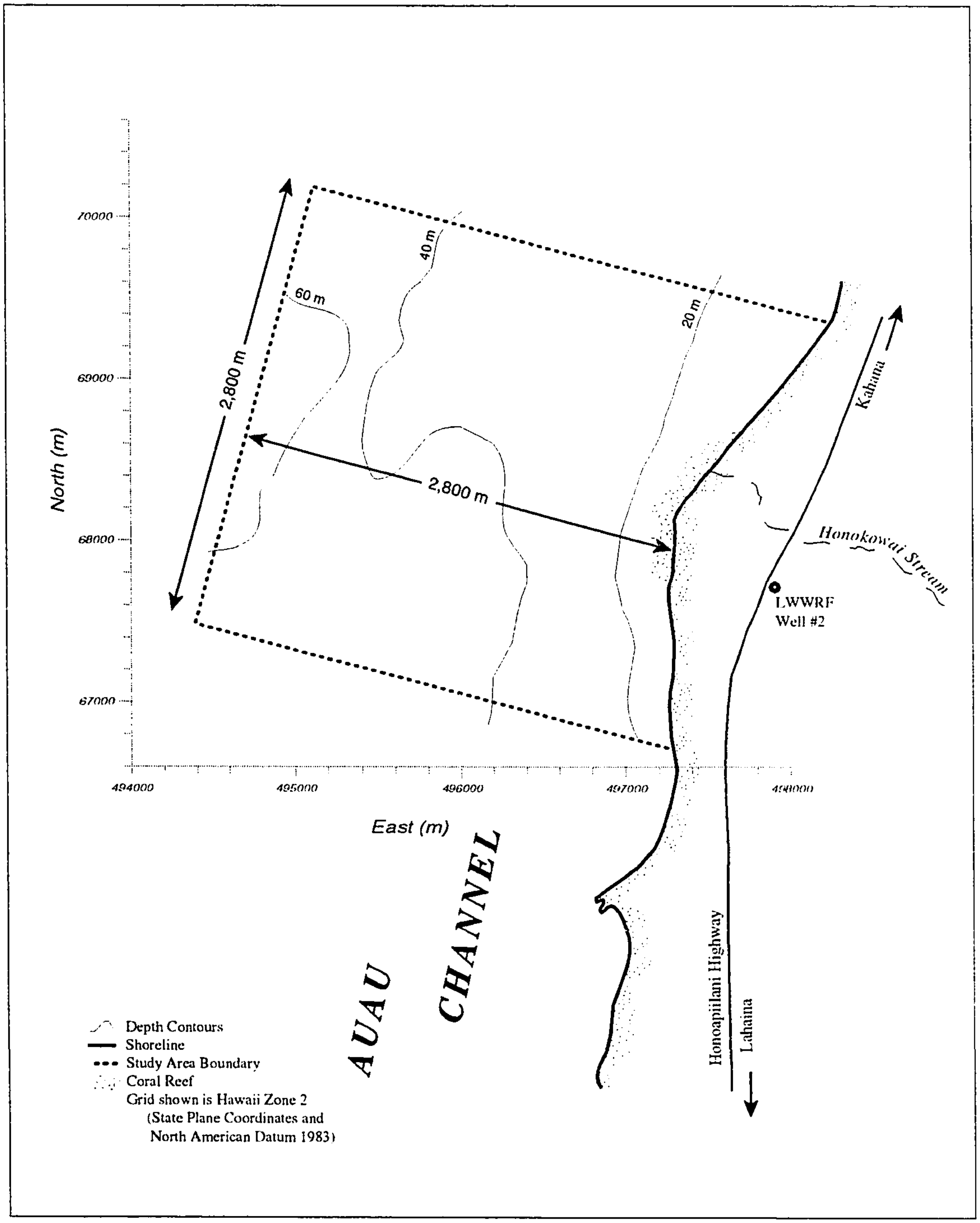


Figure 6. Dimensions and location of the study area.

Sampling Density

The determination of the appropriate sampling density, and size of the study area, depended not only on environmental parameters within the coastal and groundwater systems but also on operational constraints. To detect a wide area, low velocity seep, or a large number of discrete sources, the distance between adjacent sampling points was not a limiting factor. The focus of the sampling, however, had to be within 1 - 1.5 m (3 - 5 ft) of the bottom, close to where the effluent entered the ocean. In this case, the probability of detecting the tracer increased closer to the sea floor before additional dispersion occurred.

On the other hand, to detect plumes of effluent emerging from one or a small number of discrete points, the sampling point separation had to be less than the width of the plume. Because the plume disperses both laterally and vertically away from its source, sampling had to extend away from the bottom to increase the probability of detecting such a plume.

The sampling density was constrained by (1) the combination of the accuracy of survey vessel position fixing, estimated to be no better than ±3 m (±10 ft). This accuracy is a function of precision Global Positioning System satellite navigation accuracies (U.S. Environmental Protection Agency 1992) and the geometry of the receiving antenna relative to the sampling point; and (2) the position of the sampling apparatus near the sea floor relative to the vessel (estimated at a maximum of ±46 m [±150 ft] in 61 m [200 ft] of water). This implied that a sampling grid density greater than ± 50 m (165 ft) could not be achieved accurately throughout the sampling area.

Two extremes of possible effluent influx to the coastal waters were considered in the field study plan. For wide-area seeps or a large number of discrete vents, the probability of detection would be increased by sampling close to the bottom. For a small number of vents or point sources, the probability of detection would be increased as the density of the sampling grid increased. A sampling design was developed to address both possibilities. Semi-discrete, near-bottom sampling was incorporated to improve the detection of wide area seeps. A sampling grid of sufficient density was designed to maximize the detection of the extreme case of discrete sources: a single source and plume entering the coastal waters.

Based on the groundwater and coastal water characteristics, it was assumed that if the effluent enters the coastal water as a single plume, it would be elongated parallel to the prevailing current flow and the coastline. In the boundary condition of a single plume, the minimum detectable plume width in the direction perpendicular to the nominal current (and perpendicular to the shoreline) was estimated to be about 120 m (400 ft), based on dilution calculations (Tetra Tech, 1993b). This was calculated assuming the plume was vertically mixed over 50 ft of the water column, a 25 cm/sec current, 2 kg/day of dye being added to the LWRF effluent, and the ability to detect a variation of at least 0.05 µg/L of dye concentration. A three-to-one aspect ratio of length to width, based on the 25 cm/sec current (U. S. Environmental Protection Agency 1985) was used to estimate a plume length of 360 m (1,200 ft).

The probability of detecting the tracer in a wastewater plume of fixed dimensions and from a single source was determined using the method of geometric probabilities. Gilbert (1987) presents nomographs summarizing the probability of detection versus the ratio of the semi-major axis of the plume to the sampling grid space. The computed probability of plume detection for the selected study area size (2.8 km by 2.8 km), the estimated single plume size, 120 m by 360 m (400 ft by 1,200 ft), and the range of uniformly distributed sampling points (400-500) was 0.95 or higher (Table 1).

Table 1. Probability of detecting a single plume (after Gilbert 1987)

Study Area	Plume Dimensions (m)	Probability of detection			
		Grid Size (m)			
		300 x 150	240 x 120	200 x 100	185 x 90
2.8km x 2.8 km	24 x 73	0.04	0.05	0.07	0.08
	76 x 230	0.29	0.44	0.56	0.68
Note 1	122 x 366	0.65	0.85	0.95	0.99
	245 x 732	0.82	0.99	0.99	0.99
	762 x 2286	0.99	0.99	0.99	0.99
	1250 x 3750	0.99	0.99	0.99 Note 2	0.99

Note 1. Estimated dimensions for a plume if effluent discharges from a single point (400 ft x 1,200 ft)

Note 2. Sampling density used for fluorometric survey. Probability is found at the intersection of the Note 1 row and this column.

Sampling Rate

Sampling was designed to be semi-discrete due to time and equipment constraints and to satisfy both possible extremes of effluent discharge characteristics (wide-area seeps or plumes from one or a few point sources). For water to be pumped from close to the bottom to detect wide-area seeps, the survey vessel would have to stop at each sampling station to allow the sampling apparatus to reach the bottom. Then, as the vessel moved between stations, sampling continued as the sampling pump and hose rose through the water column. In this way, discrete bottom samples were taken, and continuous sampling throughout the water column was achieved. Using this approach, an average of 40 to 50 grid points were sampled each day.

A 100-m line spacing parallel to the shore was established (Figure 7) in response to the various environmental and operational constraints discussed above. Line coordinates were computed for each transect and then input into the navigation computer at the beginning of each field day. Near-bottom water samples were recorded every 200 m (650 ft) along each transect, effectively creating a 100-m by 200-m (330-ft by 650-ft) sampling grid. Approximately 450 near-bottom sampling stations, located at the intersections of this grid, were created using this design.

Timing of Field Studies

The time required for the discharged effluent to travel in the groundwater system to reach the coastal waters was also an important consideration in designing the study. The time required for the tracer to reach steady state within the groundwater was also important. Neither value was known, so estimates of the possible range of values were calculated by using numerical modeling.

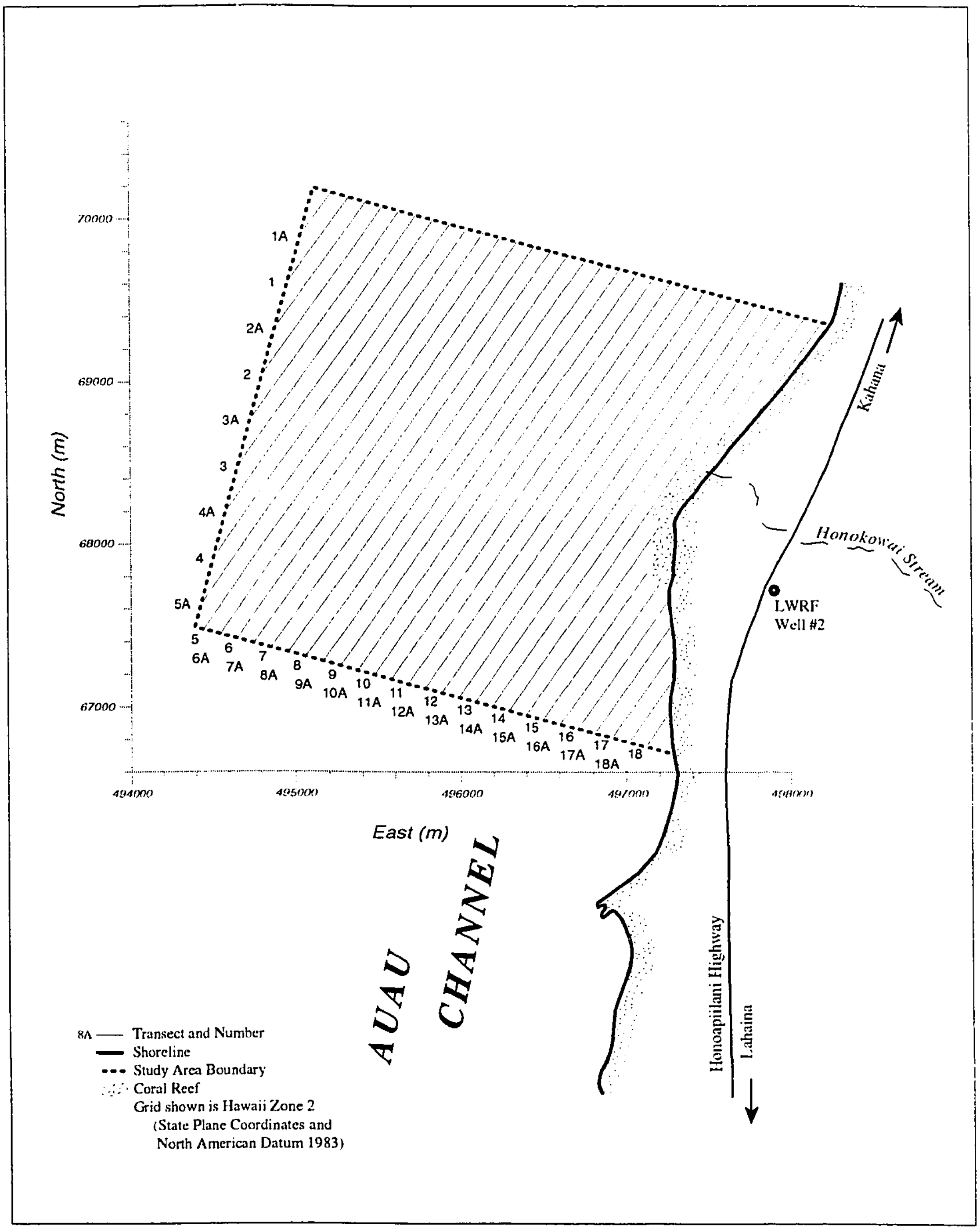


Figure 7. Position of 100-m transect lines within the study area.

A two-dimensional groundwater flow model (SWIFT; Reeves et al. 1986) was used to approximate the migration of the effluent once it has been discharged into the injection well and had entered the aquifer. This model approximation predicted that the plume would spread laterally 600 m (2,000 ft) from the injection point before reaching the shoreline. The effluent time of travel to the shoreline was predicted to be 25 days (Tetra Tech 1993b).

A groundwater solute transport model, also part of the SWIFT modeling package, was used to estimate the areal extent and dilution of the wastewater plume as it travels through the groundwater system. This modeling assumed a two-dimensional advection of the effluent in the basal aquifer. At the shoreline, 600 m from the source, the estimated dilution was only 2:1 along the plume centerline after 25 days of injection (Tetra Tech 1993b). As expected, the dilution increased with distance from the source and with horizontal distance from the groundwater plume centerline. Effluent dilution at a submarine influx point would be proportional to the distance from the injection well and the distance off the centerline of the effluent plume.

The same transport model was used to estimate the time required for the tracer to reach steady state within the groundwater system. For an assumed discharge point close to the shoreline and 600 m (2,000 ft) from the injection well, tracer added for periods of 1, 10, and 25 days resulted in peak tracer concentrations at the discharge point of less than one half of the steady state response. Tracer added for a period of 50 days reached 80 percent of the steady state response. Tracer added for a period of 100 days reaches a steady state where the predicted discharge concentration is the same as the source concentration.

However, it is important to recognize that these model estimates were unsubstantiated and based on limited hydrogeologic data. As explained earlier, three general and different potential pathways between the injection wells and the ocean have been hypothesized. Not knowing which is the most accurate representation, the model results could only provide estimates of the lateral extent of the effluent plume, transit time, and time required for the tracer to reach steady state (equilibrium) in the effluent plume.

These preliminary modeling results predicted that the optimal field sampling design would include:

1. adding tracer to the effluent for a minimum of 50 days, based on the steady state predictions
2. commencing the sampling at least 50 days after the start of the addition of tracer to the discharged effluent, based on the dilution and transit time predictions
3. continuing limited sampling after the completion of the main field survey to increase the probability of detecting tracer if the transit time of the effluent within the groundwater system was greater than the predicted period of 50 to 60 days.

Another possibility considered was that the transport of the effluent through the groundwater occurs rapidly. To monitor for this possibility, a preliminary survey effort involving the addition of 5 L slugs of tracer to the effluent every 8 hours for 3 days was designed. Within 6 hours of the addition of the first slug of tracer, nearshore monitoring would commence. Should the flow path from the injection well to the coast be on the order of a few hours to a few days, effluent dilution would be relatively low. In this case, high concentrations of tracer would be detectable in the coastal waters. If tracer dilutions were sufficiently small, the tracer may enter the coastal waters at visible concentrations.

If this were to occur, the travel time of the effluent would be known to within a few days. Tracer addition to the effluent would be stopped until a few days before the main survey commenced, higher concentrations of tracer could then be used, making detecting and locating the tracer easier. The cost of the tracer usage could be reduced and funds redirected to better delineate the effluent distribution patterns.

A flow chart of the proposed study design, outlining the sequence of activities for each phase is presented in Figure 8.

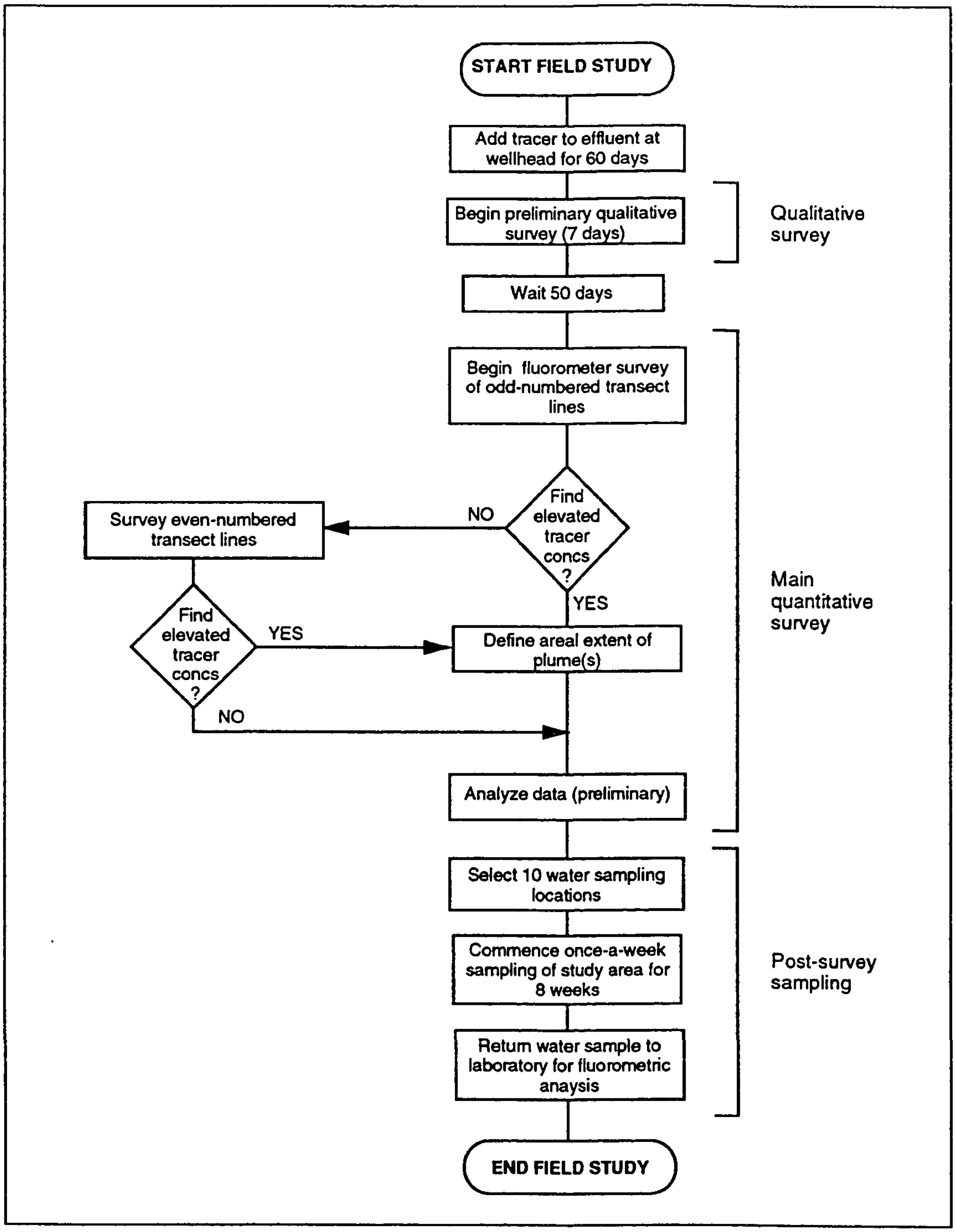


Figure 8. Flowchart of the field study activities

CHAPTER 4

FIELD AND LABORATORY METHODS

Data collection activities were divided into four separate phases:

1. A preliminary qualitative field monitoring effort
2. The main quantitative fluorometric survey
3. A subsequent postsurvey discrete sampling effort
4. A concurrent laboratory study of residual chlorine concentrations and loss of fluorescence.

Prior to beginning the field program, wastewater and ocean water samples from the study site were measured for naturally occurring fluorescence. The samples were also spiked with Rhodamine WT to verify the compatibility of the tracer with the effluent and ocean water. Estimations of the natural loss of fluorescence in both media over time were made. The detection limit of Rhodamine WT in the field samples was also determined.

The effluent fate field study commenced on July 1, 1993, with the addition of Rhodamine WT to the LWRF effluent injected into Well No. 2. Field work was performed over three periods, (1) a qualitative monitoring survey conducted between July 2 and July 12, 1993, (2) the main survey effort conducted between August 21 and August 31, 1993, and (3) postsurvey sampling, conducted approximately weekly between October 10 and December 12, 1993. A laboratory study of the possible effects of residual chlorine on the fluorescence of the tracer was conducted during this same period. Overall, the field and laboratory studies extended over a 6-month period (Figure 9).

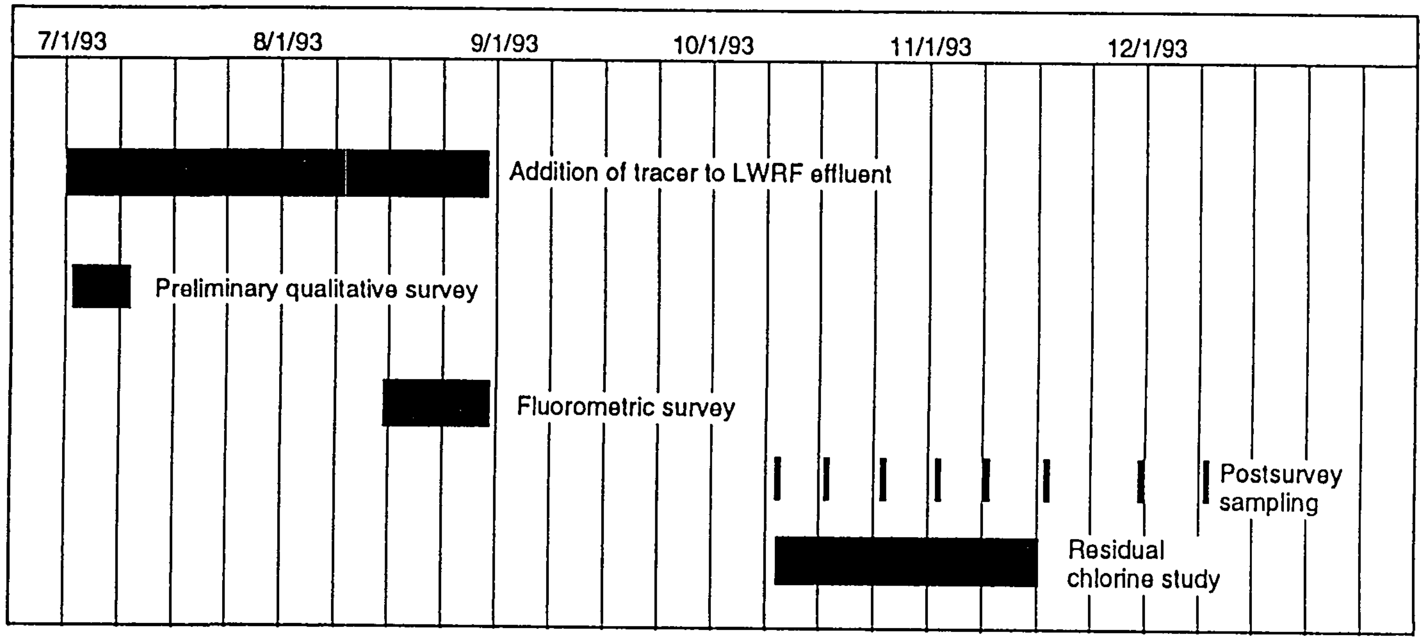


Figure 9. Timetable of field activities

Tracer Addition to LWRF Injection Well

For 3 days, commencing on July 1 1993, approximately 5 L of tracer (a 20 percent solution of Rhodamine WT) were added to the effluent every 8 hours. Starting on July 4 1993, a constant supply of tracer was added to the effluent, using a calibrated pump, at an average rate of 5.2 mL/min. This is equivalent to 7.5 L/day of dye solution or 1.5 L/day of active tracer, resulting in a concentration in the effluent of between 70 and 130 µg/L. The tracer was added continuously for a total of 58 days, until August 28, except for brief periods due to pump failure or a delay in the delivery of drums of Rhodamine WT to the LWRF. The lower concentration (70 µg/L) was calculated by using the total LWRF effluent volume discharged of 0.25 m^3/sec (5.6 mgd), assuming the effluent injected into all four wells was mixed completely in the groundwater system. The higher concentration (130 µg/L) was based on an average flow of effluent of 0.13 m^3/sec (3.0 mgd) injected into Well No. 2 and assumed no mixing of the effluent injected into different wells.

Navigation and Bathymetry

As part of the mobilization, a differential GPS reference station was established close to the survey area. The GPS equipment, a Trimble 4000 series system (Trimble Navigation, Sunnyvale, CA) and a MDL telemetry link (MicroTel, Birmingham, England) were used to establish survey control from a first-order U.S. Geological Survey station, *Laina*, located on a small rock outcrop east of the town of Lahaina and about 6 km (4 mi) from the study site. A temporary survey mark was established on the roof of an eleven-story hotel, located at the beach between the LWRF and the study area. Once the coordinates of the roof station were calculated, the GPS reference unit was moved to the roof location and the remote unit and telemetry link were mobilized on the survey vessel.

Transect lines were precomputed for the study area using State Plane Coordinates, Hawaii Zone 2, North American Datum 1983. Geographic conversions were calculated with CORPSCON, computer software published by the U.S. Army Topographic Engineering Center (Ft. Belvoir, Virginia). After consultation with the captain of the survey vessel, transects were calculated to run parallel to the prevailing wind direction to facilitate vessel handling during sample collection. The vessel could travel more slowly and be maneuvered more accurately when heading into the wind. To cover the study area, 36 transect lines, 100 m (328 ft) apart and running in the direction of 035° - 215° true (Figure 7), were computed, using Trimble HYDRO software (Auckland, New Zealand).

The same software package was used to interface and record real time navigation, depth, and fluorometry data. These data were recorded every 15 seconds or upon demand. The data were recorded directly to the hard disk of the portable computer (Toshiba 3200). The computer was also used to monitor all data during the survey and to control a remote monitor installed at the helmsman's position. This remote monitor provided continuous updates of the vessel's position as distance along the line and distance off the line for each transect.

The majority of sampling stations were occupied and sampled within 50 m (160 ft) of the precomputed coordinates. For all stations, the navigation system recorded vessel positions to an accuracy of better than ±5 m (15 ft). The position of the intake pump on the seafloor, relative to the vessel, was estimated to be less than ±35 m (115 ft) from the recorded position for most sampling stations. This difference is a function of the water depth. Large variations were most likely to occur only in depths greater than 40 m (130 ft) and during rough weather, when the wind and waves made it more difficult to hold the vessel steady and on location.

Bathymetric data were measured continuously and recorded every 15 seconds along transects. A MD100 digital fathometer (Meridata, Olhja, Finland), with a resolution of 0.1 m (0.3 ft) and accuracy of ±0.5 m (1.6 ft) was interfaced to the navigation computer. A small transducer was mounted on the stern of the vessel. No heave or tidal corrections were applied to the bathymetric data.

Preliminary Qualitative Field Monitoring

Because of uncertainties in the expected dilution of the effluent and concerns that the transit time through the groundwater system might be short (resulting in visible concentrations of tracer appearing in nearshore areas), preliminary monitoring by boat started the day after the first tracer additions. Seven half-day, near-surface sampling cruises were completed during a ten-day period. The vessel, with the aid of the echo sounder, followed approximately the bathymetric contours at 3 m (10 ft), 10 m (30 ft), 20 m (60 ft), and 50 m (160 ft), through the study area.

A 8-m (26-ft) survey vessel used for all field cruises. For the preliminary survey, a Model 10 Analog Field Fluorometer (Turner Designs, Sunnyvale, CA) was used as the monitoring instrument. A 12-volt DC pump was used to deliver water through a 1.6-cm (5/8-in) hose to the fluorometer. Between 23 - 61 m (75 - 200 ft) of hose was towed behind the vessel as it moved through the survey area. A steel depressor fin and lead weights were attached to the end of the hose, allowing water to be pumped from depths of 6 - 12 m (20 - 40 ft).

Problems were encountered with the hose collapsing and with maintaining the end of the hose at depth. It appeared that the hose was not sufficiently rigid to withstand the pressure difference created by the suction of the pump, and as the effective diameter of the hose was decreased, the flow rate decreased. The hose intake could not be maintained at depths greater than 6 to 12 m (20 - 40 ft) because of a number of factors. These included the buoyancy and drag of the hose and the minimum speed of the survey vessel, which was too high to allow the hose to sink further. The fluorometer calibration also appeared to drift markedly on two occasions. As a result, the hose, pump and fluorometer were changed for the main survey. Another style of hose with a smaller diameter and constructed of a less flexible material was used and a larger capacity submersible pump was chosen for the main survey.

<u>Quantitative Fluorometric Survey</u>

A recording, digital display fluorometer and higher capacity submersible pump were mobilized for the main survey. An integrated navigation system, capable of recording vessel position, water depth, and fluorometer data, was installed also.

Fluorometer

A Model 10-AU-005 Digital Field Fluorometer with temperature compensation (Turner Designs, Sunnyvale, CA) was used to analyze and record water column fluorescence. The fluorometer was set to display every 2 seconds. The internal data logger in the fluorometer was set to record time, water temperature, and the 3-second moving average of fluorescence. The navigation software recorded time, position, depth, and the instantaneous reading from the fluorometer every 15 seconds.

The limit of detection of the fluorometer was reported to be between 0.01 and 0.05 μg/L above background for Rhodamine WT in potable water and 0.1 μg/L in raw sewage (Turner Designs 1990). The seawater in the study area was exhibited very low turbidity. Because of this, the field detection limit was estimated to be 0.02 μg/L (S. Mokelke, Turner Designs, Inc., personal communication, January 17 1994). The temperature sensor accuracy is reported as ±0.2 °C (Turner Designs 1990). An exponential temperature coefficient of -2.6 percent was used for the automatic temperature compensation of the water flowing through the fluorometer cell. The instrument is designed to measure the relative difference in fluorescence between a sample and a calibrated standard concentration and the accuracy of the instrument is directly related to the accuracy of the calibration standard.

Three replicate standards of 1.00 μg/L and three replicate standards of 10.0 μg/L were prepared for the field calibration of the instrument. The variation in the readings for the 1.00 μg/L standard was consistently 0.02 to 0.03 μg/L. From these results, the accuracy (the measure of the difference between the real reading and the instrument reading) is estimated to be 0.03 μg/L. The fluorometer was calibrated before the beginning of the main survey, after every 2 days, and again at the end of the survey. A 1.00 μg/L standard solution of Rhodamine WT was used as a field standard and distilled water was used as a reference blank.

Submersible Pump

A 115-volt submersible pump, Little Giant Model 3E-12N (Tecumseh Products, Oklahoma City, Oklahoma), modified for use in 70 m of water, was attached to 76 m (250 ft) of 1.3-cm (1/2-in) diameter nylon hose. The pump was contained in a specially designed polyethylene housing (Figure 10). The pump was powered by a 650-watt portable gasoline powered generator via a three core, 14-gauge electrical cable taped to the hose. A 2.5-mm polypropylene mesh covered the pump intake. An additional 150-denier nylon mesh was

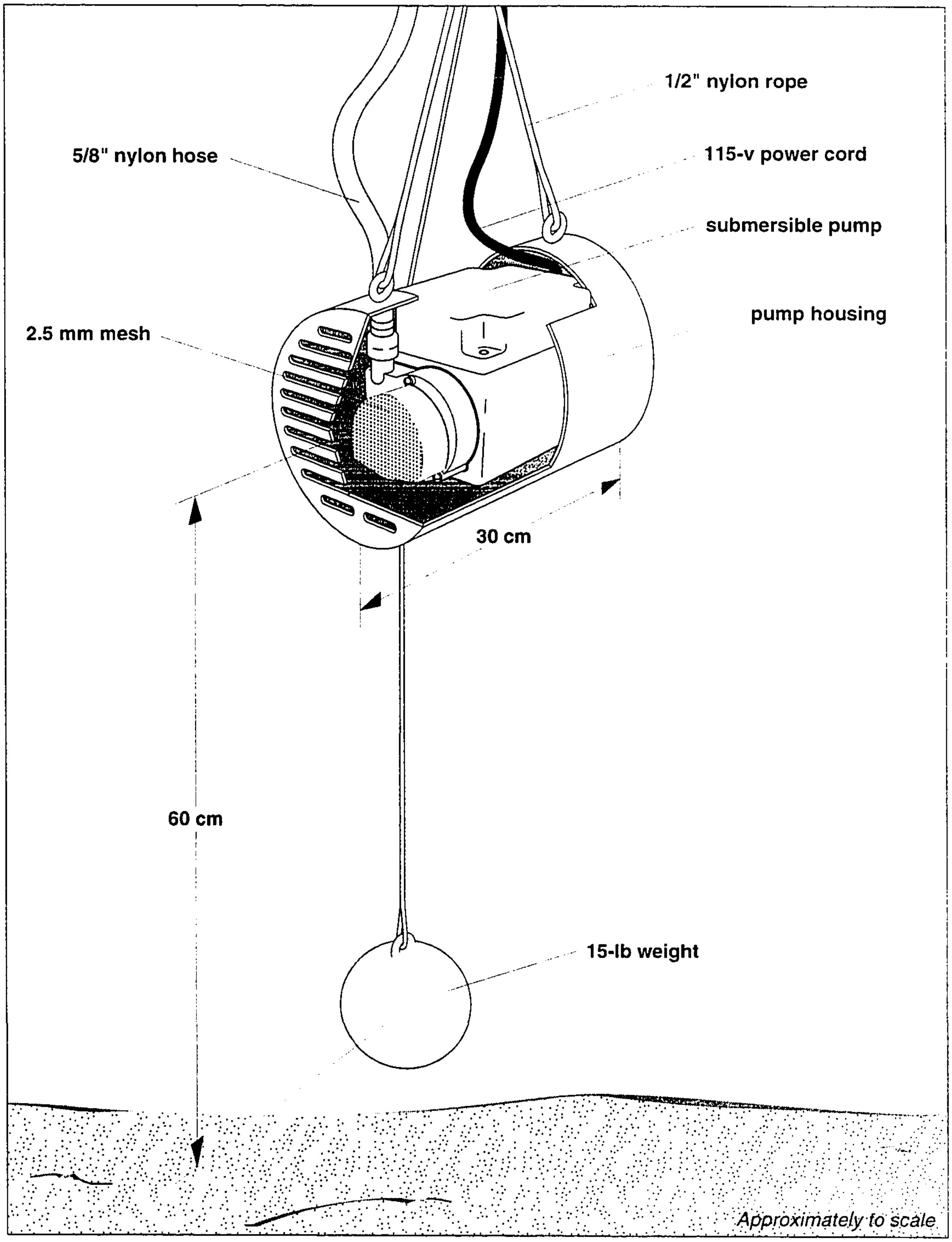


Figure 10. Diagram of submersible pump system.

installed after sand was detected being pumped through the fluorometer, and an additional inline 100-micron nylon filter (Pepco, Fresno, CA) was installed in the hose line immediately before the fluorometer.

Nylon rope (1.3 cm diameter) was taped to the hose to support the pump. A 7-kg (15-lb) lead weight was attached to the pump to speed its descent through the water column and to help detect when the pump was on the bottom. The weight was originally attached with a short cord, designed to sit on the bottom while the pump remained 60 cm from the bottom. However, in deep water, where a considerable length of hose was required to reach the bottom, it was difficult for the hose operator to discriminate between the weight and the pump hitting the bottom. It is assumed the pump was also on the bottom for most of sampling stations. The pump housing prevented the intake of the pump from being closer than 10 cm (0.3 ft) to the bottom.

Sampling Procedures

All transects, except one, were run in a south to north direction. For the first half of the survey every second transect, numbered from 1 to 18, was run. The alternate transects, numbered 1A through 18A, were run during the second half of the survey (Figure 7). The submersible pump was switched on before the beginning of each transect, and data were recorded continuously on both the navigation computer and the internal logger of the fluorometer for the time required to complete each transect.

The vessel stopped at 200-m (650 ft) intervals along the transect line, as close to the line as possible. The pump and hose would sink to the bottom, and as soon as the hose operator determined the pump and weight were on the bottom, time, position, and depth were recorded (in addition to the automatic 15-sec recordings). The vessel remained on station for a minimum of 1.5 min and then moved along the line another 200 m (650 ft) to the next station. Once the vessel started moving, the pump and hose rose through the water column to close to the surface and then sank again after the vessel had stopped. Typically, the pump was on the bottom for 1.5 min, rose through the water column for 30 sec, was near the surface for 1.5 min, and fell through the water column for 1.5 min. This created a sampling cycle of approximately 5 min (Figure 11).

Each transect was started at the precomputed position at the study area boundary. For those transects that ended in deep water, the final sampling point was outside the study area boundary, 200 m (650 ft) beyond the last point within the study area to ensure complete coverage of the area. For the precomputed transects that ended in shallow water or on the shore, the water depth determined the location of the last sampling point. The shallowest samples were collected in water depths of 2 - 3 m (6 - 10 ft). The distance of the shallowest sampling points from the shoreline varied. In the southeast section of the study area the bottom is sandy and the beach steep. The vessel was able to approach to within 10 m (30 ft) of the beach in this area. Further north, at Honokawai Point, a shallow coral reef extends away from the shore. The distance of the nearest sampling points to the shoreline here was approximately 10 - 30 m (30 - 100 ft). From the mouth of Honokawai Stream and north, shallow depths extended further offshore and the distance of the sampling stations also increased to approximately 300 m (1,000 ft).

The flow rate of the pump was measured at frequent intervals to determine the transit time of water in the hose from the pump to the fluorometer. This time was generally about 2 min. Data were recorded for an extra 3 min after the end of each transect, to allow for the transit time of the water sample traveling through the hose to the fluorometer. The flow rate of the pump was recorded at regular intervals by recording the length of time

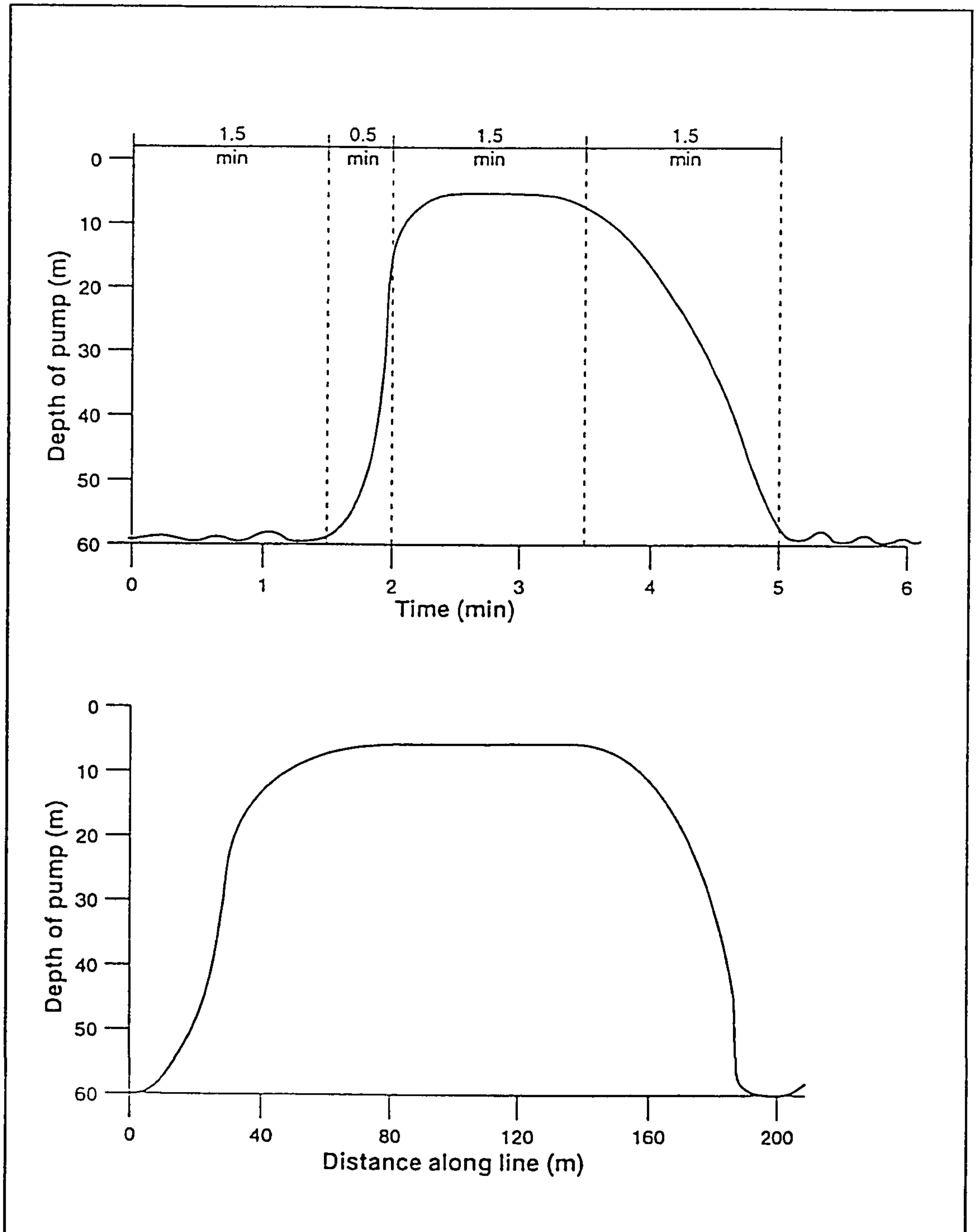


Figure 11. Graphs of pump depth versus time and pump depth versus distance between adjacent sampling stations

required to fill a graduated 2-gallon container. Several discrete water samples were collected from the fluorometer discharge hose at times when elevated concentrations were being recorded. These samples were inspected and analyzed at the end of the day. After the final sampling station was occupied for a transect, the vessel returned to the southern end of the study area to begin another transect. The pump was either streamed behind the vessel or brought on board for inspection and maintenance.

Data Recovery

At the end of each survey day, the data collected were downloaded from the navigation computer onto diskettes. The data were copied, and reviewed to verify that data collection had occurred correctly and to determine if spatial patterns of elevated fluorescence were detectable. Fluorometry data were recorded as a concentration in parts per billion, relative to the calibration standard. Near-bottom fluorometry values were correlated with respect to time with the depth and position data and then examined for spatial patterns.

Reference Stations

Six reference stations were chosen beyond the probable influence of the LWRF effluent influx to establish values and a measure of the variation of local background fluorescence. All stations were between 1,200 m and 5,500 m (0.75 - 3.5 mi) to the south of the study area, at locations approximately 1,200 m (0.75 mi) apart. Four stations were in 30 m (100 ft) of water. Two of the southernmost stations were at depths of 20 m (65 ft) and 10 m (33 ft) (Figure 12).

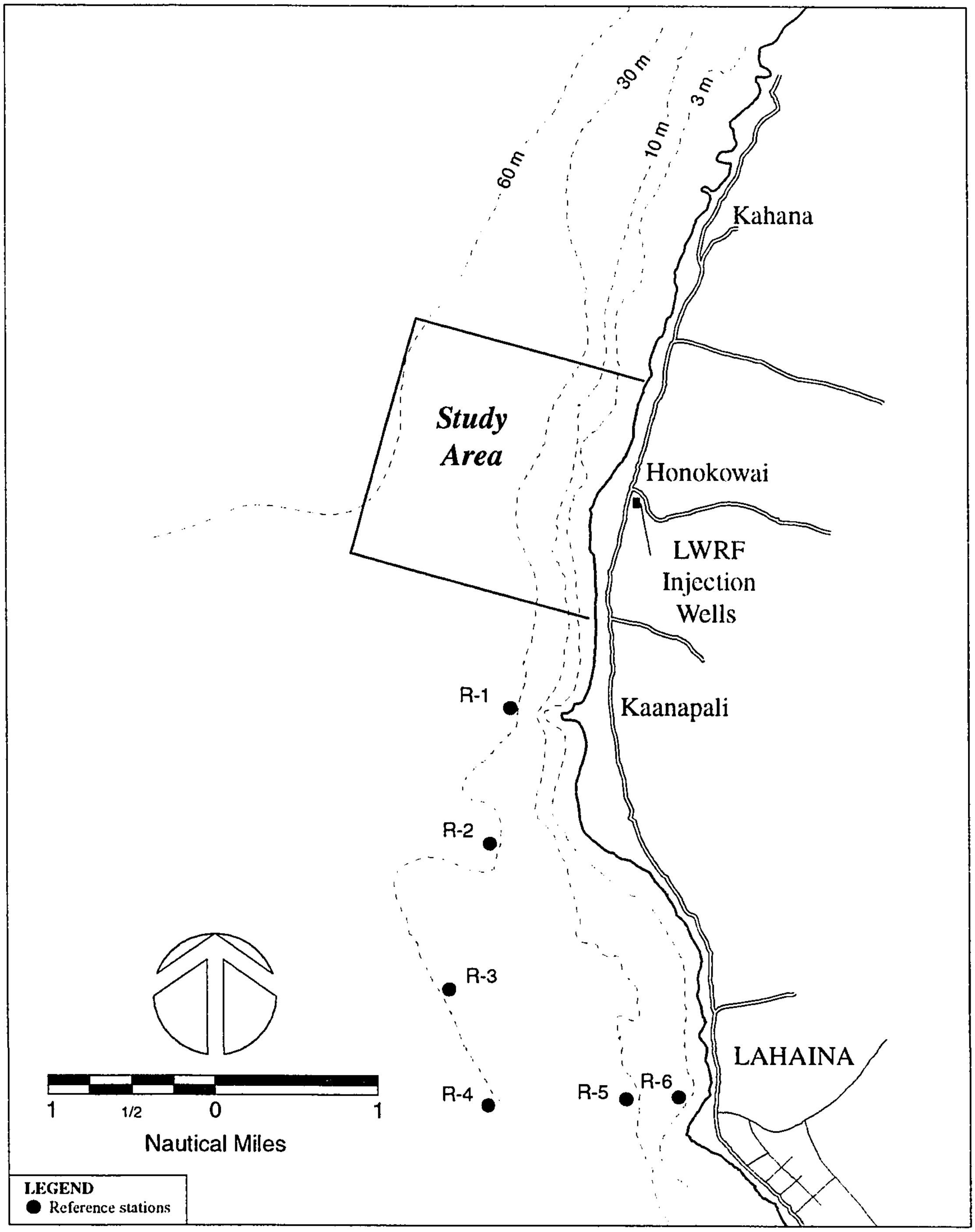


Figure 12. Location of fluorometry reference stations, relative to the study area.

Post-Survey Sampling

Approximately one month after the end of the fluorometric survey, near-bottom water samples were collected from ten locations within the study area (Figure 13). These locations were chosen after the initial analysis of the fluorometric survey data had been performed. The chosen sites represented areas of possible elevated readings (Stations S3, S4, S5, S6, S7, S8, S10) or other areas of interest, such as a nearshore area in which bubble seeps and freshwater influxes had been previously reported (Station S1). A station off the mouth of Honokawai Stream (Station S2), and a station at the deepest part of the study area (Station S9) were chosen also. This sampling occurred approximately once a week for 2 months, except when prevented by bad weather.

Samples were collected using a conventional water sampling bottle lowered by hand to the bottom. The sampling bottle was triggered with a messenger or weight dispatched along the line from the surface. At each site, a 250-mL polyethylene bottle was filled with seawater, labeled, and stored in the dark in a small cooler. The same vessel was used as for the main survey. A single GPS navigation unit (not differential GPS) was used to position the vessel at each of the 10 locations. The accuracy was estimated to be approximately ± 100 m at the deep sites and considerably better at the nearshore locations, where landmarks could be used to position the vessel more accurately.

The samples were shipped to San Francisco via overnight delivery. Upon arrival the samples were immediately analyzed for fluorescence using a Turner Model 112 Digital Filter Fluorometer. This laboratory fluorometer, although a different model, was of similar accuracy, sensitivity, and resolution as the field fluorometer. The fluorometer resolution is reported at 0.1 percent of full scale, which is equivalent to 0.002 μg/L using these standard solutions. The accuracy is reported to be 1 percent of full scale or 0.02 μg/L in this

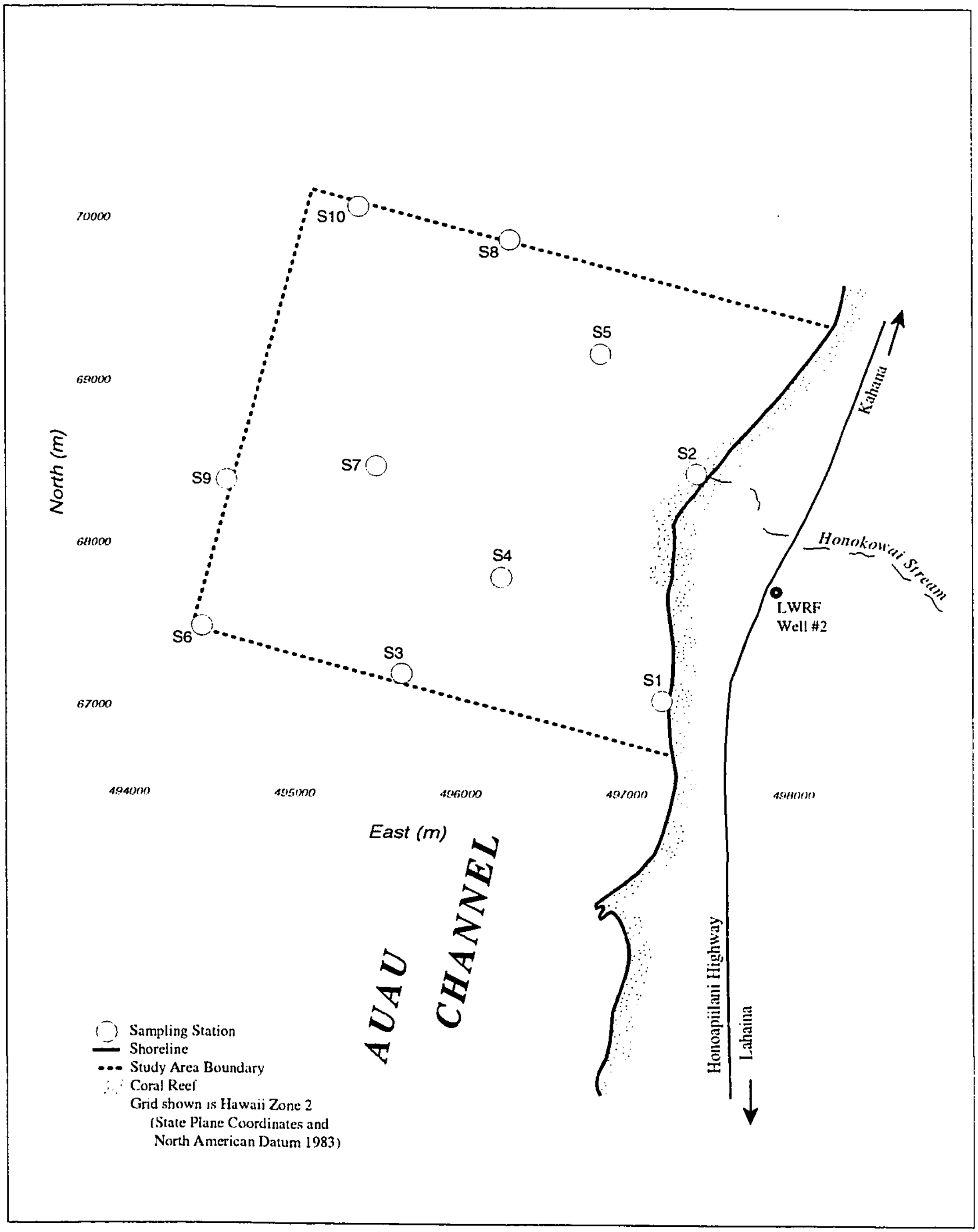


Figure 13. Location of postsurvey water sampling stations.

application (Sequoia-Turner Corporation 1982). The sensitivity (detection limit) varies with the type of fluorescent tracer analyzed. Under these test conditions with Rhodamine WT, the sensitivity was conservatively estimated to be 0.02 μg/L.

The fluorometer was calibrated before each analysis using four standard concentrations of Rhodamine WT dye, ranging from 0.4 - 2.0 μg/L. No temperature compensation device was available for the fluorometer, so all standards and blanks were allowed to come to room temperature (between 20 - 23 °C) before each analysis. Because the temperature of the sample increases when placed in the fluorometer, no sample was reanalyzed for a minimum of 1 hr after the initial reading.

Effect of Residual Chlorine on Fluorescence

A loss of fluorescence resulting from oxidation reactions between Rhodamine WT and the residual chlorine, present in the effluent after disinfection, appears to be the most likely source of loss of tracer in this study. This effect was investigated further. Tests performed on a single sample of chlorinated LWRF effluent sent to San Francisco before the beginning of the survey indicated that approximately a 5 percent loss of fluorescence occurred over a 48-day period. However, residual chlorine concentrations in the effluent may have declined in the 5 or 6 days between collection of the sample in Lahaina and the addition of the dye in San Francisco. Thus, the results may not be indicative of those expected on site because of the time delay and other general environmental conditions, such as aeration, mixing, temperature, and passage through the ground, which cannot be replicated in the laboratory.

Because of sparse literature on the oxidation of Rhodamine WT dye by residual chlorine (Deaner 1973), and the possibility of loss of fluorescence, a series of long-term tests was initiated by adding measured doses of chlorine and dye to locally available unchlorinated secondary-treated domestic wastewater effluent collected from the Central Contra Costa Sanitary District (CCCSD) treatment facility. A series of chlorine doses from 0 - 63.0 mg/L (0, 5.25, 10.5, 15.75, 21.0, 26.25, 31.5, 42.0, 52.5, and 63.0 mg/L) was added to the effluent within 2 hours of collection. A 50 μg/L concentration of Rhodamine WT was added to each of the test solutions 30 minutes after the addition of the chlorine dose to approximate the Rhodamine concentration and the time interval between the addition of chlorine and tracer to the effluent at the LWRF.

The reductions of fluorescence in the test solutions were measured over a 36-day period. Measurements were made frequently during the first 8 days (10 measurements) when the rate of change of fluorescence was greatest and less frequently for the remaining 28 days (3 measurements), once the rate of loss of fluorescence stabilized.

CHAPTER 5

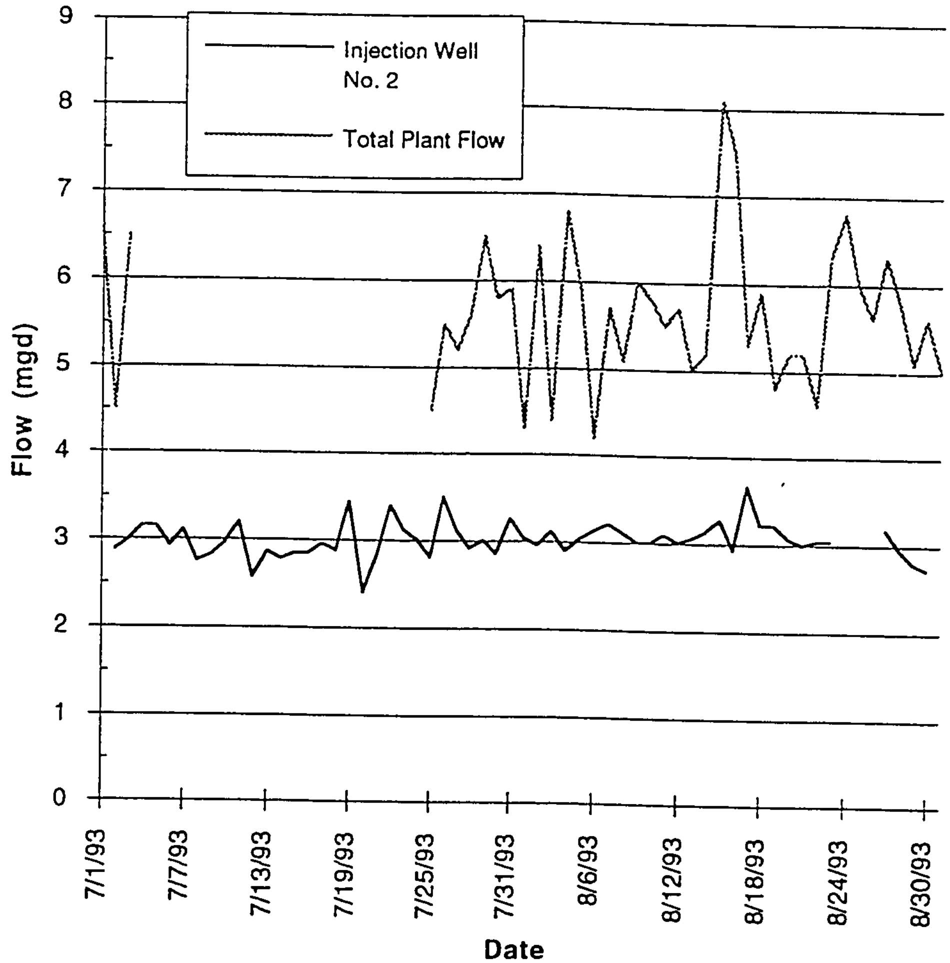
RESULTS OF FIELD AND LABORATORY INVESTIGATIONS

Lahaina Wastewater Reclamation Facility Operations

During July and August 1993, the Lahaina Wastewater Reclamation Facility (LWRF) treated and discharged an average of 21.2 x 10^6 L/day (5.6 mgd). Daily influent flow rates varied from a minimum of 16.3 x 10^6 L/day (4.3 mgd) to a maximum of 30.7 x 10^6 L/day (8.1 mgd).

The flow meter at the intake point of the facility was not operational. Influent flow data were estimated daily from a flow meter located at the chlorination plant, except for approximately three weeks in July, during which time this flow meter also was not operational. Other than this, no unusual operating conditions were reported by LWRF personnel during the study period. Effluent injection rates to Well No. 2 were recorded three times a day at a flow meter at the splitter box between Injection Wells No. 1 and No. 2. This is the same location at which the tracer was added to the effluent discharged to Well No. 2.

The tracer was added to the effluent injected into Well No. 2 at a daily average volume of 1.5 L of active dye. The average daily effluent flow into Injection Well No. 2 for the 2-month period was 11.4 x 10^6 L/day (3.0 mgd), with a maximum flow of 14.0 x 10^6 L/day (3.7 mgd) and a minimum of 9.1 x 10^6 L/day (2.4 mgd). A graph showing the total wastewater flow treated and effluent flow into Injection Well No. 2 is shown in Figure 14.



Note: Data gaps represent periods when the flow meter was not operational (influent) or when no data were recorded (effluent).

Figure 14. Graphs of total plant flow (mgd) and effluent flow into Injection Well No. 2

Bathymetry

Bathymetric contours were constructed from the depth data recorded during the main survey. Contours were computed after editing the depths recorded at each of the near-bottom sampling points (Figure 15). Heave and tidal corrections were not applied to the values and the depths were estimated to be accurate to ±0.5 m.

Preliminary Qualitative Monitoring Results

During the seven, half-day monitoring cruises, no concentrations of fluorescence above the background values of 0.4 to 0.6 µg/L were detected in the near-surface waters. No visible dye could be seen in the shallow and relatively calm water along the shore.

Quantitative Fluorometric Survey Results

The main survey was performed over two 4-day periods, with a break of one day between each period of the survey. Every second transect (Line 1 to Line 17) was run in the first half, and the remaining transects, Line 1A to Line 18A (see Figure 7), were surveyed in the second part. At times, adverse weather conditions prevented the lines from being run in numerical order. Background fluorescence was sampled at depth at the six reference locations and continuous near-surface sampling was performed as the survey vessel traveled between these reference sites.

The fluorometry data results were prepared in three steps:

(1) All the 3-sec averaged data were plotted as separate line graphs of fluorescence versus time and plots of temperature versus time (after instrument calibration corrections had been applied).

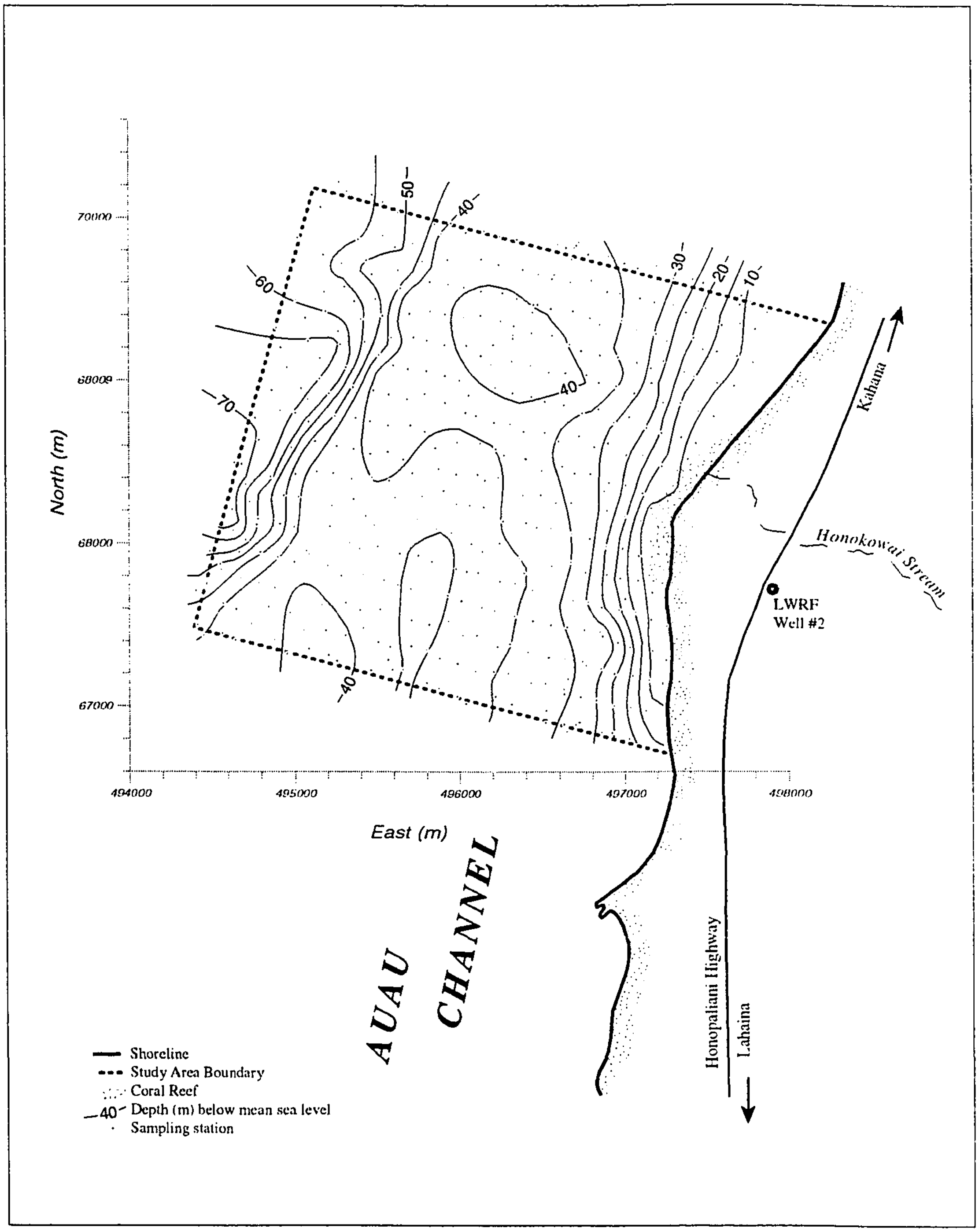


Figure 15. Measured bathymetry of the study area.

(2) A statistical analysis of the daily data collected was performed to distinguish signals from background and instrumental noise. Mean values and 95-percent confidence limits were calculated and scattergrams of concentrations were plotted for each of the eight survey days (Table 2).

(3) Contour charts for near-bottom fluorescence were compiled and plotted from the 3-sec averaged data collected from each half of the survey. These plots were used as the basis of the analysis of spatial patterns of the signals and to present the final interpretation of the data.

Elevated concentrations of fluorescence, when detected, were detected only near the bottom. Mid-depth and near-surface fluorescence readings remained at background levels of 0.4 - 0.6 µg/L throughout the survey.

Fluorescence and Temperature Plots

Instantaneous fluorometer readings were recorded every 15 sec onto the navigation computer. After an analysis using these 15-sec instantaneous data, the 3-sec averaged data from the internal data logger of the fluorometer were chosen for a more detailed analysis. These data were recorded more frequently and the data were more consistent than the instantaneous data. The 3-sec data were considered a more accurate representation of conditions during the study because, at a typical sampling location, the near-bottom water was pumped through the fluorometer for approximately 1.5 min and during this period only six 15-sec values were recorded compared to thirty 3-sec values.

Graphs of fluorometry data, reported as concentrations relative to a calibration standard of 1.00 µg/L of Rhodamine WT, and graphs of temperature (°C), were plotted for all transects. These plots are included in Appendix A. The plotted data were unedited 3-sec

Table 2. Statistical summary of fluorometric data

Date	Minimum	Maximum	Spread	Count	Mean	Std. Dev
Single inline filter. Transects 1 - 17. First half of survey.						
8/22/93	0.043	0.280	0.237	5517	0.053	0.0142
8/23/93	0.043	0.117	0.074	4776	0.051	0.0074
8/24/93	0.041	0.405	0.364	3107	0.060	0.0323
8/25/93	0.039	0.115	0.076	5554	0.052	0.0111
Three in-line filters. Transects 1A - 18A. Second half of survey.						
8/27/93	0.048	0.078	0.030	6132	0.057	0.0045
8/28/93	0.041	0.082	0.041	6866	0.054	0.0089
8/29/93	0.045	0.086	0.041	6358	0.050	0.0034
8/30/93	0.045	0.187	0.142	4052	0.052	0.0063

averaged values, as recorded by the internal data logger of the fluorometer. Corrections for the approximate 2-min travel time between the pump and the instrument have not been applied, nor are the data corrected for background fluorescence.

Fluorometric Contours

Contour plots of near-bottom fluorescence were prepared from the 3-sec averaged data and the navigation records of the vessel's position when the pump first reached the bottom. Fluorescence and position readings were corrected for the time delay of the water traveling through the intake hose. After these corrections were applied, the concentration values corresponded to the horizontal grid position at which they were recorded.

Where sections of transects had been resurveyed to verify areas of elevated concentrations, the highest readings were plotted initially and the subsequent resurvey data, which were lower in all cases, were not used in the contour plots. This was done to ensure that no valid elevated readings were discarded. However, because the elevated readings were not repeated, the validity of the initial readings is in doubt.

Initially, near-bottom concentration values were plotted relative to the grid locations (easting, northing) at which the values were recorded. This was done separately for each half of the data, unfiltered and filtered (Figures 16 and 17). Although these plots of discrete concentrations show the complete data sets, they are difficult to interpret. So, the data sets for both unfiltered and filtered samples were then computer-contoured to produce plots of lines of equal concentration (Figures 18 and 19). For each chart, the corresponding positions of the data points used for the contouring are shown as small dots. Each dot corresponds to the concentration value shown on the previous figures (Figures 16 and 17).

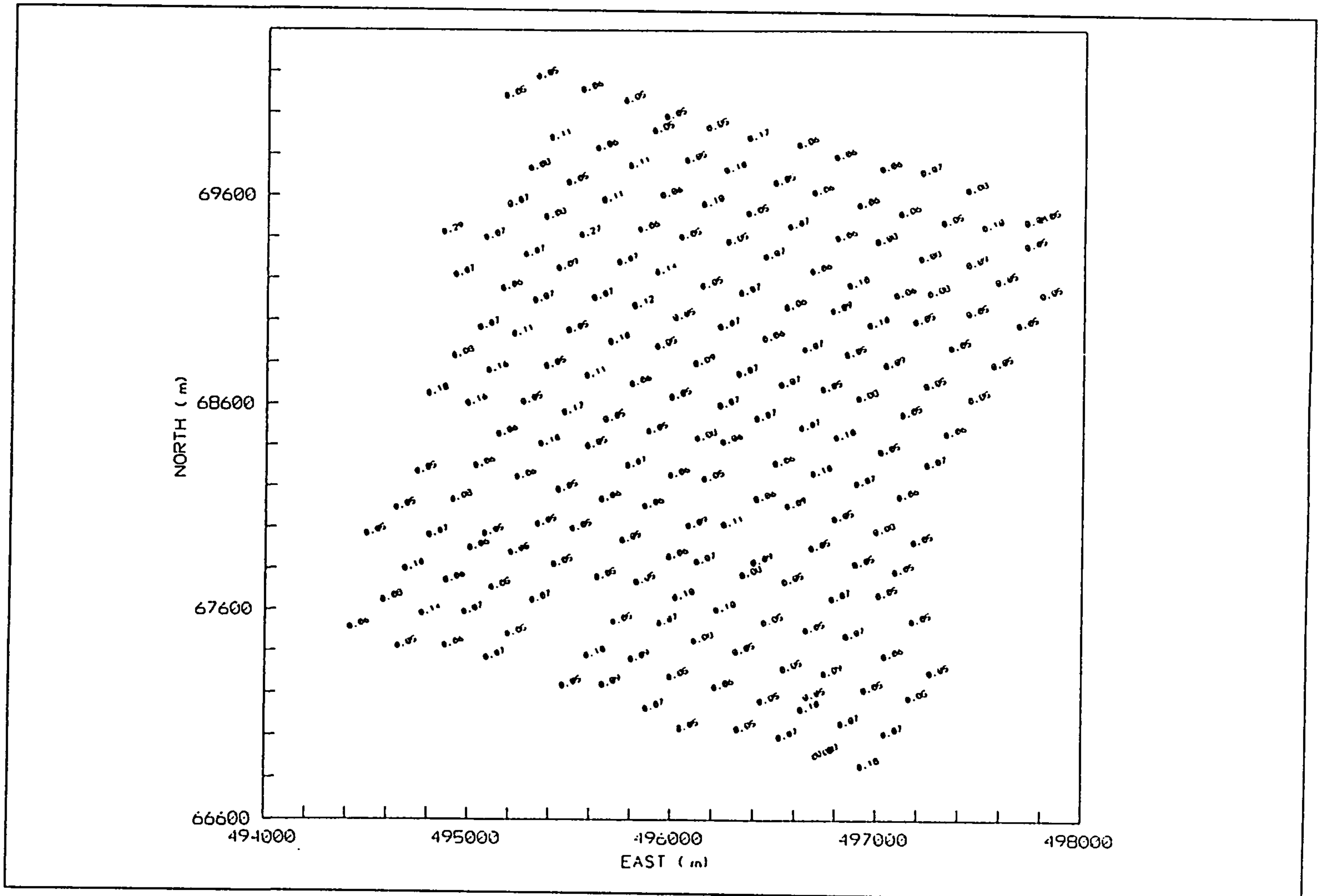


Figure 16. Fluorometric concentration (µg/L) versus location for unfiltered samples collected between August 22 - 25, 1993

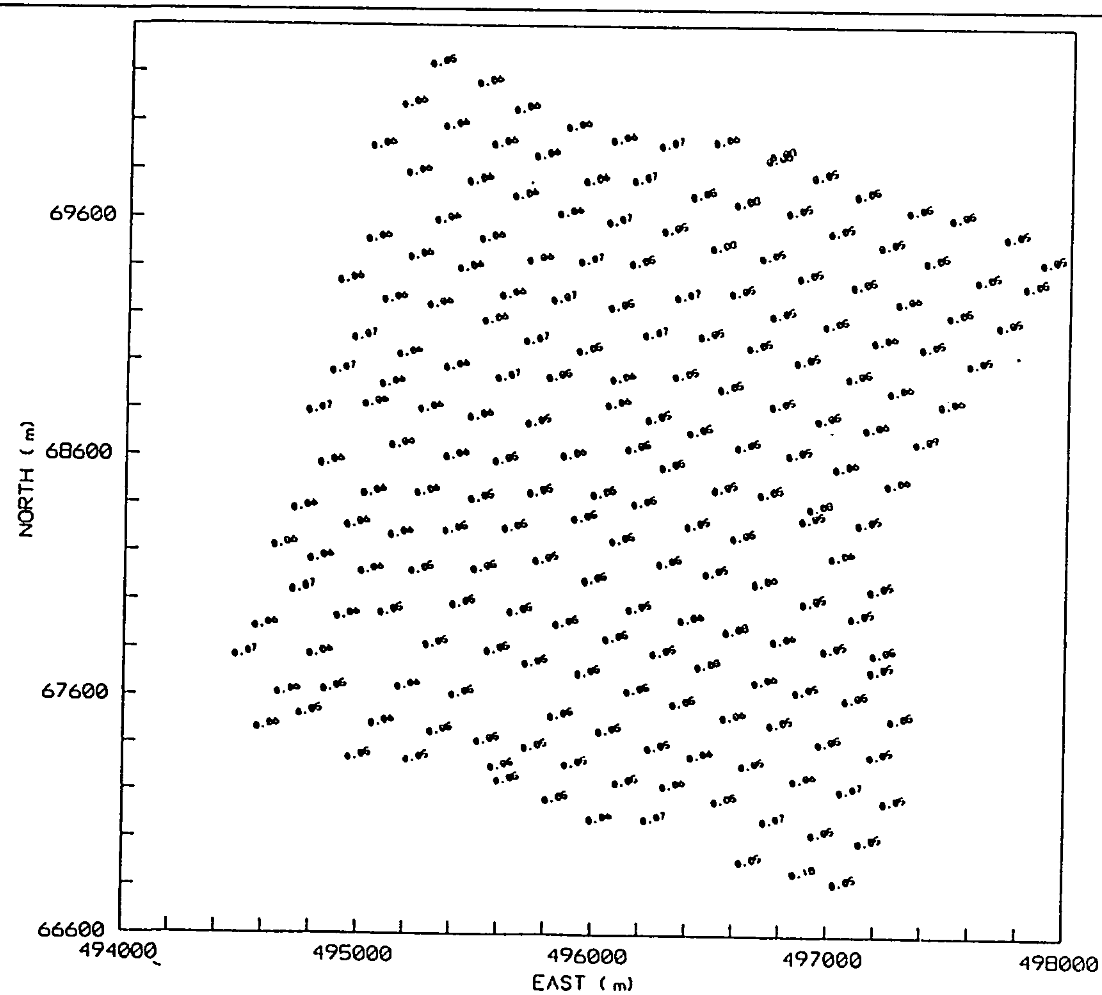


Figure 17. Fluorometric concentration (µg/L) versus location for filtered samples collected between August 27 - 30, 1993

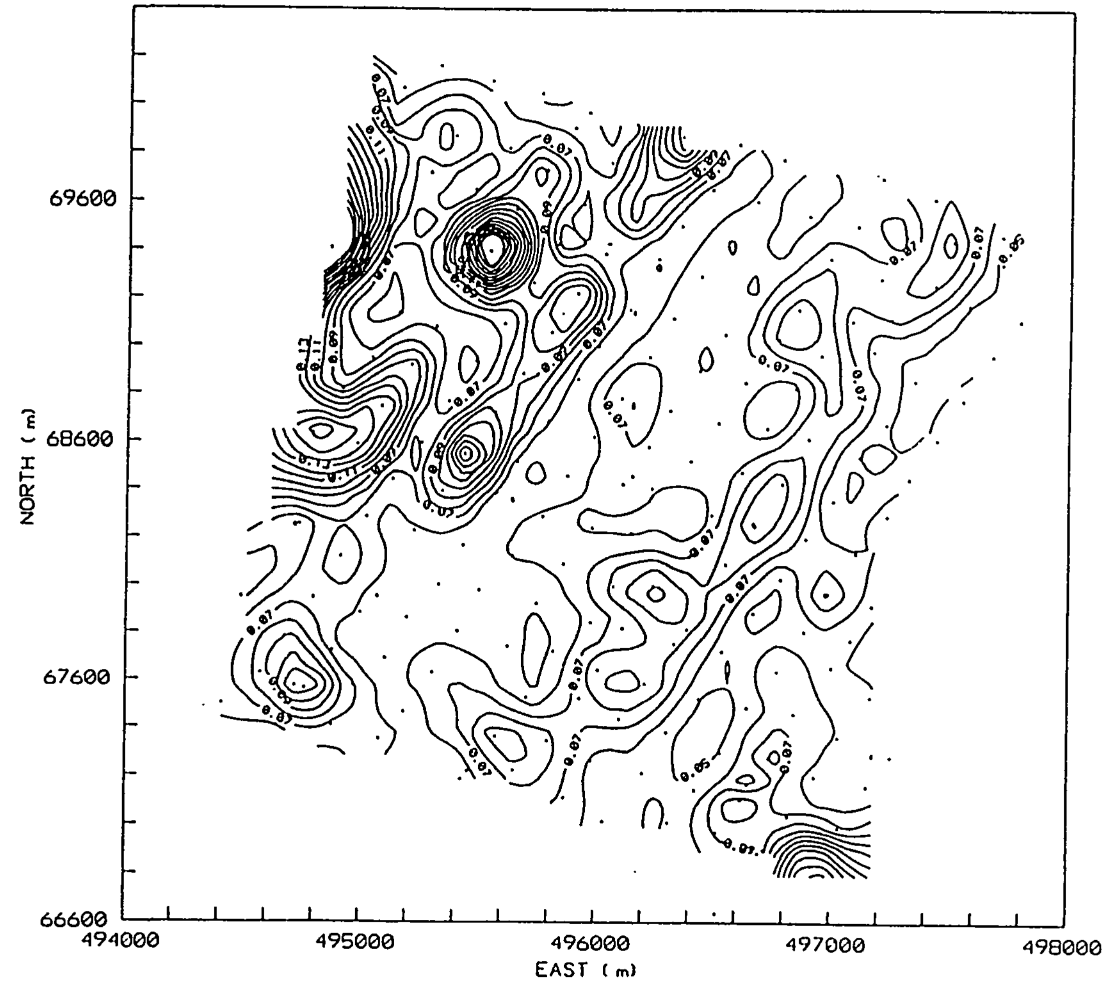


Figure 18. Contours of equal fluorescence (μg/L) for unfiltered samples collected between August 22 - 25, 1993

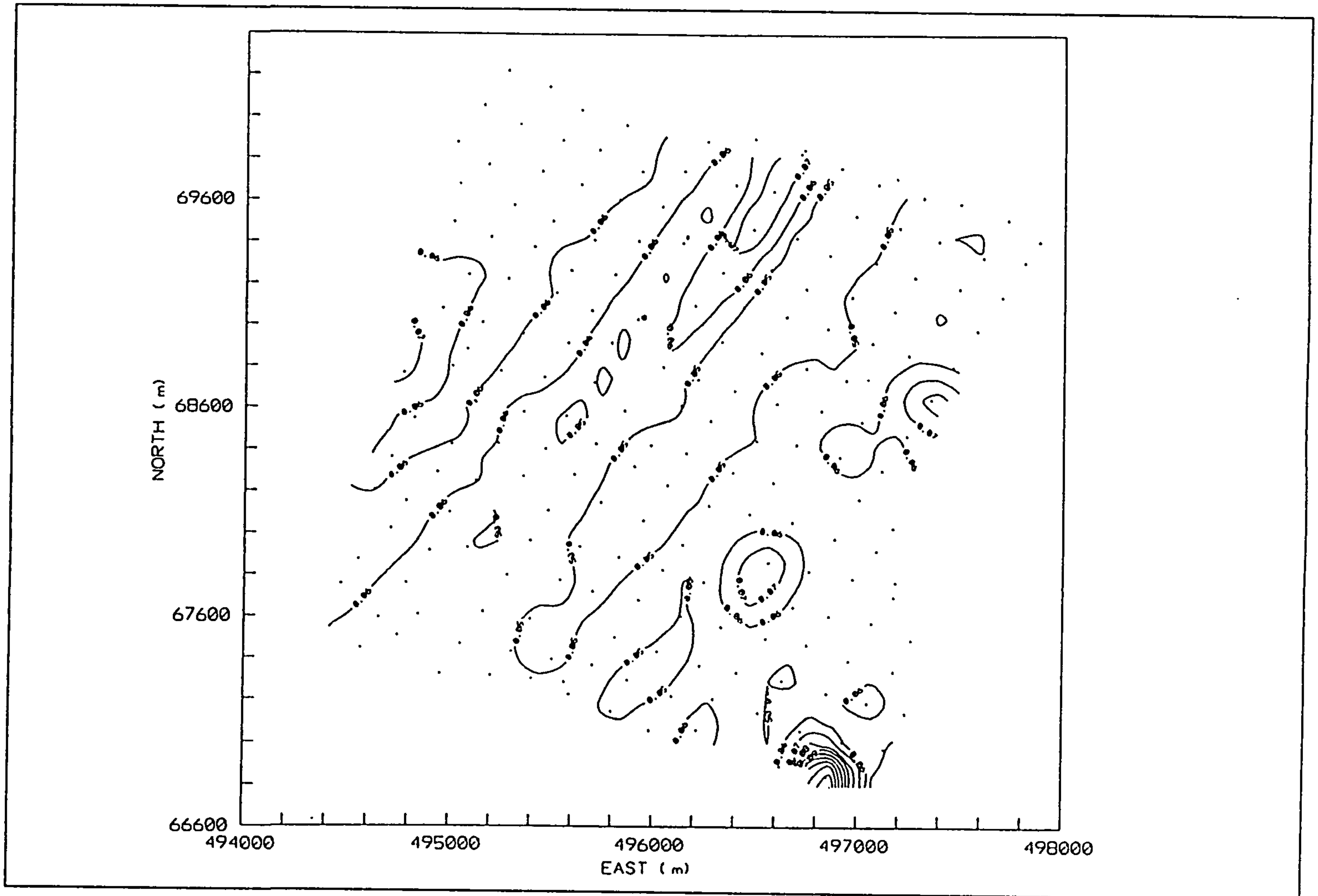


Figure 19. Contours of equal fluorescence (μg/L) for filtered samples collected between August 27 - 30, 1993

Postsurvey Sampling

Eighty discrete, near-bottom water samples were collected from ten locations over a 2-month period. Figure 13 shows the position of the sampling points within the study area. In addition to the samples collected in the study area, two background samples were collected at Reference Station R-1, 3,500 m (1.5 nm) south of the center of the study area (Figure 12). Elevated fluorescence was not detected in any of these samples (Table 3).

Effects of Residual Chlorine on Fluorescence

The results of the 36-day study on residual chlorine effects on fluorescence loss were similar to those reported by Deaner (1973), although his testing did not extend beyond two days. The majority of the quenching occurred within the first 48 hr in all cases. After this time, the loss of fluorescence decreases rapidly and the fluorescence remained constant through the remainder of the test (Figure 20). For applied chlorine doses of 5.25 and 10.5 mg/L, similar to the residual concentrations used by Deaner (1973), the observed loss of fluorescence over 2 days was less than 5 and 10 percent, respectively. For higher applied chlorine concentrations, the loss or quenching was higher, ranging from approximately 20 percent for 15.75 mg/L to 85 percent for 52.5 mg/L. At an applied chlorine dose of 63.0 mg/L, fluorescence was rapidly and completely quenched.

At the LWRF, the daily applied chlorine dose was 12 - 14 mg/L, for an average flow of 0.13 m^3/sec (3.0 mgd). The total residual chlorine concentration in the effluent was reported between 0.1 - 0.6 mg/L (County of Maui 1992), although concentrations of 2 mg/L or greater could have existed during periods of low flow (J. Oka, County of Maui, October 12, 1993, personal communication). These values are considerably lower than the residuals Deaner (1973) reports as found in practice (2 - 9 mg/L). The effluent used in the

Table 3. Measured fluorometric concentrations (μg/L) of postsurvey samples

Date	Sampling Station Number											
	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	Average	Std Dev.
10-Oct	0.002	0.000	0.000	0.002	-0.004	-0.012	0.008	0.008	0.012	0.000	0.002	0.007
15-Oct	-0.017	-0.015	-0.009	-0.015	-0.015	-0.011	-0.017	-0.013	-0.023	-0.025	-0.016	0.005
23-Oct	-0.019	-0.013	-0.011	-0.015	-0.015	-0.015	-0.009	-0.007	-0.015	-0.013	-0.013	0.003
30-Oct	0.006	0.004	0.004	0.016	0.022	0.010	0.022	0.010	0.012	0.020	0.013	0.007
07-Nov	-0.003	-0.011	-0.009	-0.011	-0.009	-0.009	-0.011	-0.013	-0.013	-0.009	-0.010	0.003
16-Nov	0.012	0.012	0.007	0.014	0.012	0.008	0.007	0.007	0.012	0.008	0.010	0.003
02-Dec	-0.018	-0.016	-0.014	-0.016	-0.016	-0.014	-0.012	-0.016	-0.014	-0.016	0.010	0.003
08-Dec	-0.001	0.005	0.007	-0.003	0.003	0.003	0.003	0.005	0.001	-0.003	-0.015	0.002
Maximum	0.012	0.012	0.007	0.016	0.022	0.010	0.022	0.010	0.012	0.020		
Average	-0.005	-0.006	-0.005	-0.004	-0.004	-0.006	-0.002	-0.003	-0.004	-0.005		
Std Deviation	0.013	0.011	0.008	0.014	0.015	0.011	0.014	0.011	0.015	0.015		

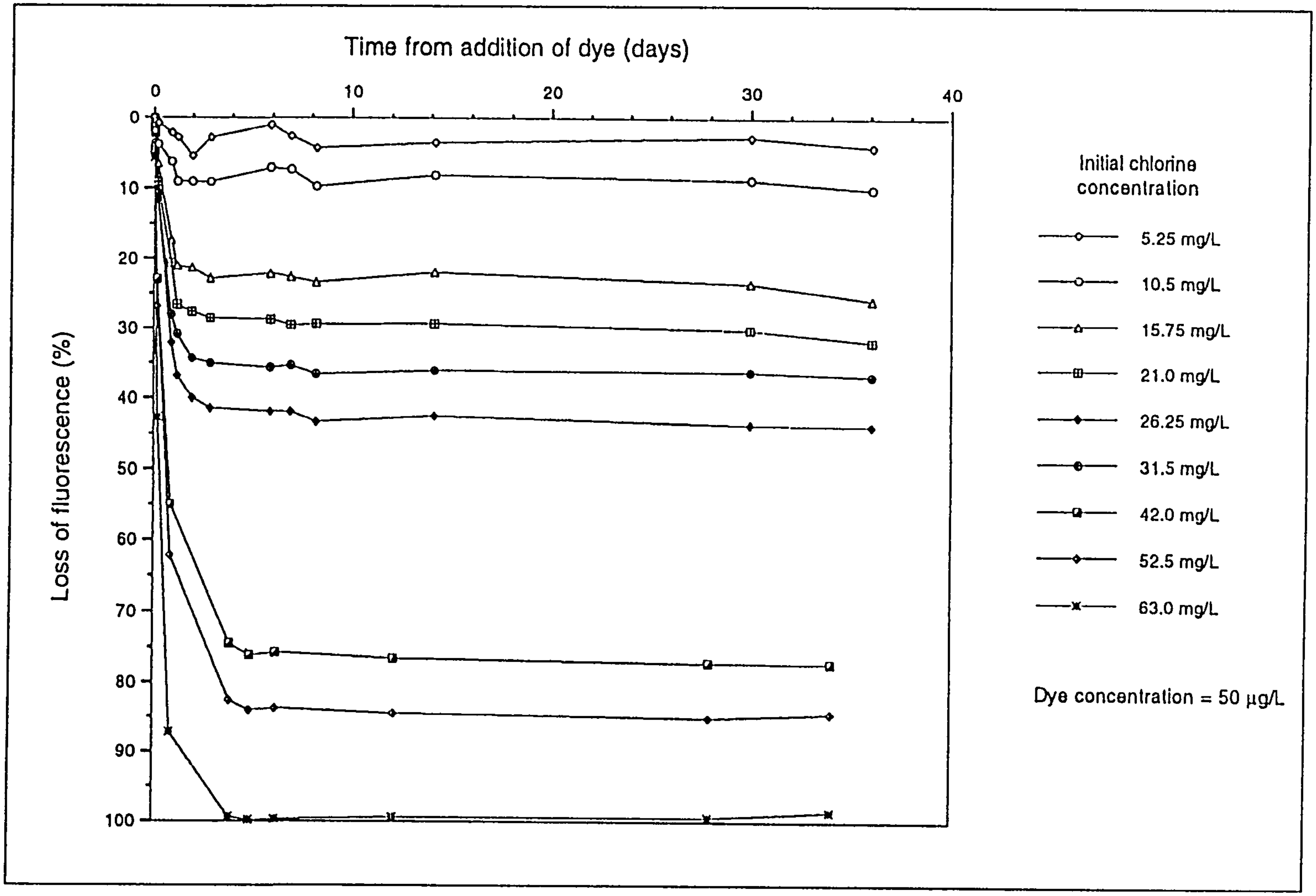


Figure 20. Loss of fluorescence versus time for different chlorine doses

laboratory tests was routinely dosed with approximately 27 mg/L of chlorine at the wastewater treatment facility. The resultant residual chlorine concentration measured at the facility was about 11 mg/L.

CHAPTER 6

DISCUSSION OF RESULTS

Bathymetry

Isobaths remain parallel to the shore throughout the study area except at the deep southwest corner (Figure 15). The major bathymetric feature of the study area is a broad ledge or bench, 35 m to 45 m in depth, between 1,200 m and 1,600 m wide and approximately 1,000 m offshore. Closer to shore, the water remains shallow across the coral on Honokawai Point and to the north past the mouth of Honokawai Stream. To the south, the inshore water deepens more quickly.

Away from the shore, the bottom drops off quickly from 10 m to 35 m in depth. This occurs closer to the shore in the southern portion of the study area. The depth increases more rapidly along the western edge of the study area, to a maximum of over 70 m. At the northwestern corner of the area, the depth increases, but more slowly, to less than 60 m. At the southern edge of the study area, a depression of 5 m or more is evident running south to north and extending about 700 m into the study area. However, from the data collected, it cannot be determined how far it extends toward the south. No unexpected bathymetric features were detected in this data. No submarine canyons or deep holes that might be associated with a possible point source of effluent were identified.

Preliminary Qualitative Monitoring

No fluorescence readings greater than the background levels of 0.04 to 0.06 µg/L were observed during this initial monitoring survey. This implies that either the tracer was not present in the surface waters or it was present at concentrations below the detection limit of the instrument.

In the shallow water along the shoreline, no visible traces of the dye were observed. This result is significant, indicating that no rapid flow pathway exists from the injection wells to the shoreline.

A further possibility supported by the nondetection of tracer is that the travel time of the effluent from the injection well to the ocean is greater than eight days and this preliminary survey was completed before any tracer had traveled the roughly 600 m to the shoreline.

Quantitative Fluorometric Survey

An analysis of the line plots of fluorometry and temperature data (Appendix A) identifies three major characteristics:

(1) Oscillations in temperature readings occurred consistently throughout both halves of the survey and related small variations of concentration were evident on some lines.

(2) Many distinct peaks of fluorescence at two to three times background values were recorded when the pump was near the bottom during the first half of the survey.

(3) Variations in fluorescence were much smaller and were close to the detection limit of the instrument during the second half of the survey when two extra filters were installed in the water intake line.

Background fluorescence, as sampled at the six reference stations, remained consistent for all stations. Recorded valves ranged from 0.046 to 0.050 μg/L, well below the instrumental precision. These values were also consistent throughout the water column at each reference station.

Effect of Temperature

Temperatures varied consistently by approximately 1 °C along each transect. The lower readings correspond to near-bottom water temperatures and the higher readings to the surface water temperatures. Corresponding small-scale oscillations, in the opposite direction, were discernible in the fluorometer readings for some transects. These oscillations were generally at or below the sensitivity limit of the instrument. They were thought to be the result of either backscattering by very fine particles near the seafloor or inaccuracies in the temperature compensation circuitry of the fluorometer or a combination of both. These effects were more pronounced because the background signal was very low and the fluorometer was operating at the low end of its sensitivity range (S. Mokelke, Turner Designs, January 17, 1994, personal communication). However, other than being an interesting qualitative phenomenon, the observed negative correlation between temperature and fluorescence is of no significance to the overall study.

Effect of Particulates

The majority of high fluorescence readings were recorded during the first four days of the survey. These elevated signals were associated with near-bottom water samples, and during the initial days of the survey, they were thought to be legitimate signals. However, sand particles were detected in the water at the fluorometer outlet during several periods of elevated readings. It was then realized that, as the weight and pump moved across the seafloor, sand and finer particles may have been disturbed and sucked into the intake hose. As the particles passed through the measuring cell of the fluorometer, light was refracted from the particles at different frequencies, creating false readings. To verify this possible source of interference, background near-bottom samples were measured at a shallow site remote from the study area. Similar elevated readings were recorded as the pump and weight were observed moving across the seafloor. To compensate for this interference for the second half of the survey, two extra filters were installed in the hose line to trap suspended particulates before the water entered the fluorometer measuring cell.

As further verification of interference, several discrete water samples were collected along different transects at the same time the fluorometer was recording high readings. These discrete samples, all of which contained visible particles, were analyzed later on the same day using the fluorometer set up for a discrete sample measuring mode. No elevated readings were recorded if the samples were not stirred before being poured into the measuring cuvette (Table 4). The same sample, if stirred briskly before pouring into the fluorometer cuvette, recorded a higher concentration than the unstirred sample, indicating that the higher reading was a result of backscattering of the light signal by the particles in suspension.

The readings for discrete samples, either stirred or unstirred, are substantially lower than the in situ flow through samples because of different sampling configuration of the fluorometer. For discrete sampling, the flow-through cell is replaced by a cuvette holder. The thickness of the cuvette walls is much less than the flow-through cell (because the cell is subject to some pressure from the pumping system). The temperature compensation apparatus cannot be used in the discrete sampling mode, and the ambient room temperature during the discrete sampling was several degrees higher than the temperature of the coastal waters. As a result, the fluorescence recorded from the discrete samples was consistently lower than the in situ readings recorded for the same samples. Because of the presence of the particles and the variations in readings, the elevated flow-through readings recorded in the first 4 days of the study were considered to be the result of light backscattering. If true

Table 4. Concentrations (µg/L) of unfiltered samples as measured in situ and as measured as unstirred discrete samples

			Concentration (µg/L)	
Transect No.	Date	Sampling Time	In Situ Flow-through Sample	Unstirred Discrete Sample
4	8/24/93	11:20:00	0.232	0.027
4	8/24/93	11:54:20	0.270	0.033
5	8/24/93	07:38:55	0.115	0.042
6	8/24/93	09:11:11	0.117	0.036
6	8/24/93	09:37:40	0.150	0.024
6	8/24/93	09:41:50	0.165	0.023
6	8/24/93	10:22:50	0.137	0.026
2	8/25/93	10:50:33	0.120	0.021
3	8/25/93	09:09:08	0.140	0.018
3	8/25/93	09:28:20	0.135	0.019
3	8/25/93	09:31:25	0.122	0.018
7	8/25/93	07:14:10	0.117	0.019
1A	8/27/93	11:46:10	0.056	0.018
5A	8/27/93	08:17:10	0.057	0.017
6A	8/27/93	11:11:25	0.058	0.016
7A	8/28/93	08:43:20	0.065	0.016
8A	8/28/93	09:25:00	0.080	0.017
16A	8/29/93	13:33:00	0.065	0.024
15R*	8/30/93	11:54:10	0.050	0.019

Notes
* R denotes the transect was resampled

signals were present, they would have been masked by this interference. Thus the near-bottom sampling data for this period were not considered accurate. However, this interference did not continue throughout the water column. Although the exact distance from the sea floor at which the interference from suspended particles ceased cannot be determined, near-surface and mid-depth data are considered accurate. These data show that no areas of elevated fluorescence, indicating the presence of an effluent plume, were detected.

Analysis of all the data sets supported the presence of interference in the samples collected during the first half of the survey. Background fluorescence concentrations varied between 0.04 and 0.06 µg/L within the study area and at the reference stations. This observed background concentration is a result of backscatter from and the natural fluorescence of naturally occurring dissolved and suspended particulates, either organic and inorganic (Hoge and Swift 1981). Throughout and beyond the study area, the background reading remained constant during the sampling periods.

In general, elevated readings were present only along every second transect, and did not occur along adjacent transect lines. An example of this can be seen from the following line graphs. Figure 21 is a line graph of fluorescence of the unfiltered water versus time recorded on transect Line 4. Elevated signals, due to suspended particulates in the sample, are obvious at nine near-bottom stations. Figure 22 shows the equivalent graphs for the adjacent Line 4A and Line 5A (100 m to the east and west of Line 4, respectively), in which the water passed through two additional filters before reaching the fluorometer. These plots show only a small variation of concentration of 0.05 - 0.07 µg/L. This alternating pattern of high readings and near background readings along adjacent transects was repeated along other transects, supporting the hypothesis that the elevated

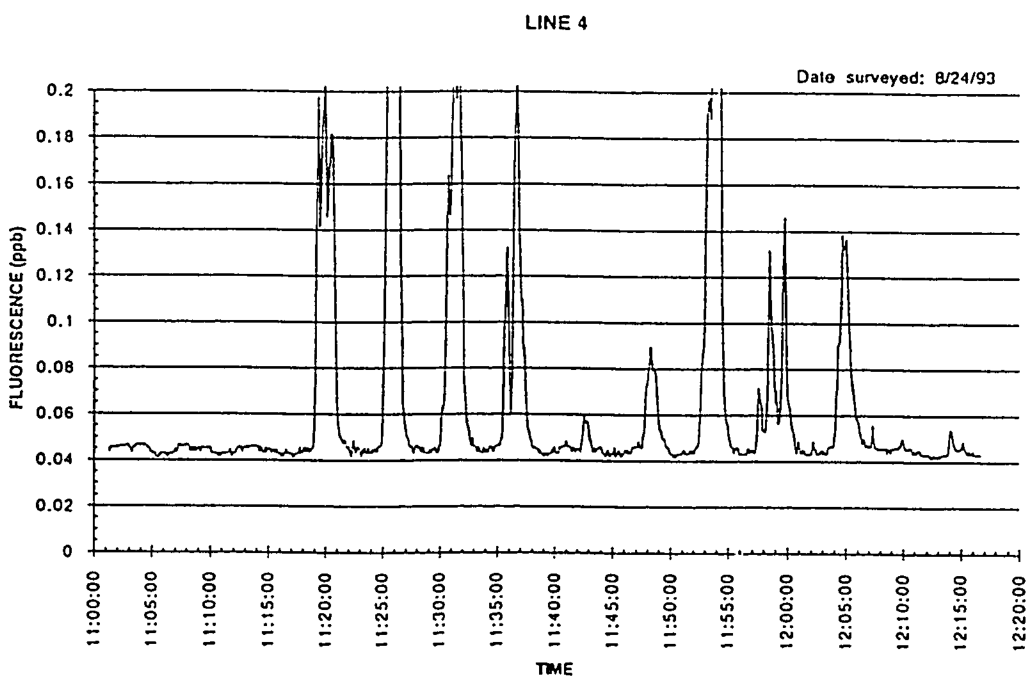


Figure 21. Unfiltered fluorescence data (µg/L) recorded on Transect 4

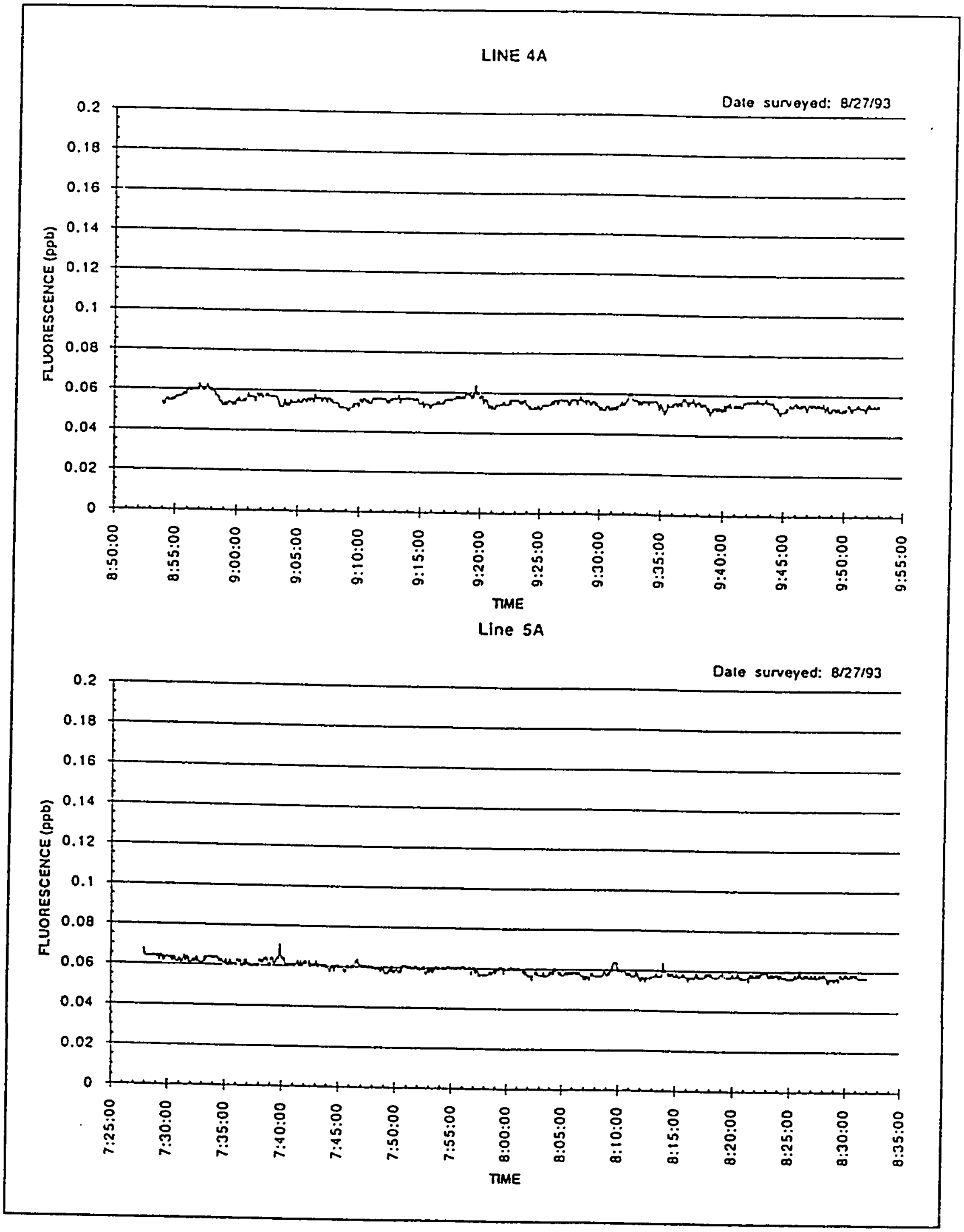


Figure 22. Filtered fluorescence data (μg/L) recorded on transects adjacent to Transect 4

readings were a result of an interference mechanism, and are not representative of a naturally occurring effluent discharge pattern.

Contour Patterns

Contour charts of near-bottom fluorescence were prepared from the edited 3-sec averaged fluorometry data and the navigation records of the vessel's position. The preliminary analysis of the data suggested that interference was the cause of high inconsistent readings during the first half (4 days) of the survey. To further investigate this possibility, the data collected during the two halves of the study were treated as separate data sets. This separation was a natural division of the data. The filtering system was changed from a single filter after the first 4 days to three separate inline filters for the second 4 days of the study. Also, every second transect was surveyed in each half of the study. So the study area was covered uniformly during each half.

The contour plot of the data collected during the first half of the survey (Figure 18) exhibits high concentrations in the western section of the study area, most corresponding to data collected along Line 4 and Line 6. A single high value is evident at the beginning of Line 2 along the western edge of the area, and another is evident in the southeast corner, at the beginning of Line 17. These characteristics and the overall structure of the pattern can be seen more clearly in Figure 23, in which only contours greater than the background value of 0.06 µg/L are plotted. The 0.01-µg/L contour plots show distinct concentration patterns. The apparent trend in the concentration contours from the southwest to northeast is thought to be an artifact of the data collection procedure because each transect was run in the same northeasterly direction.

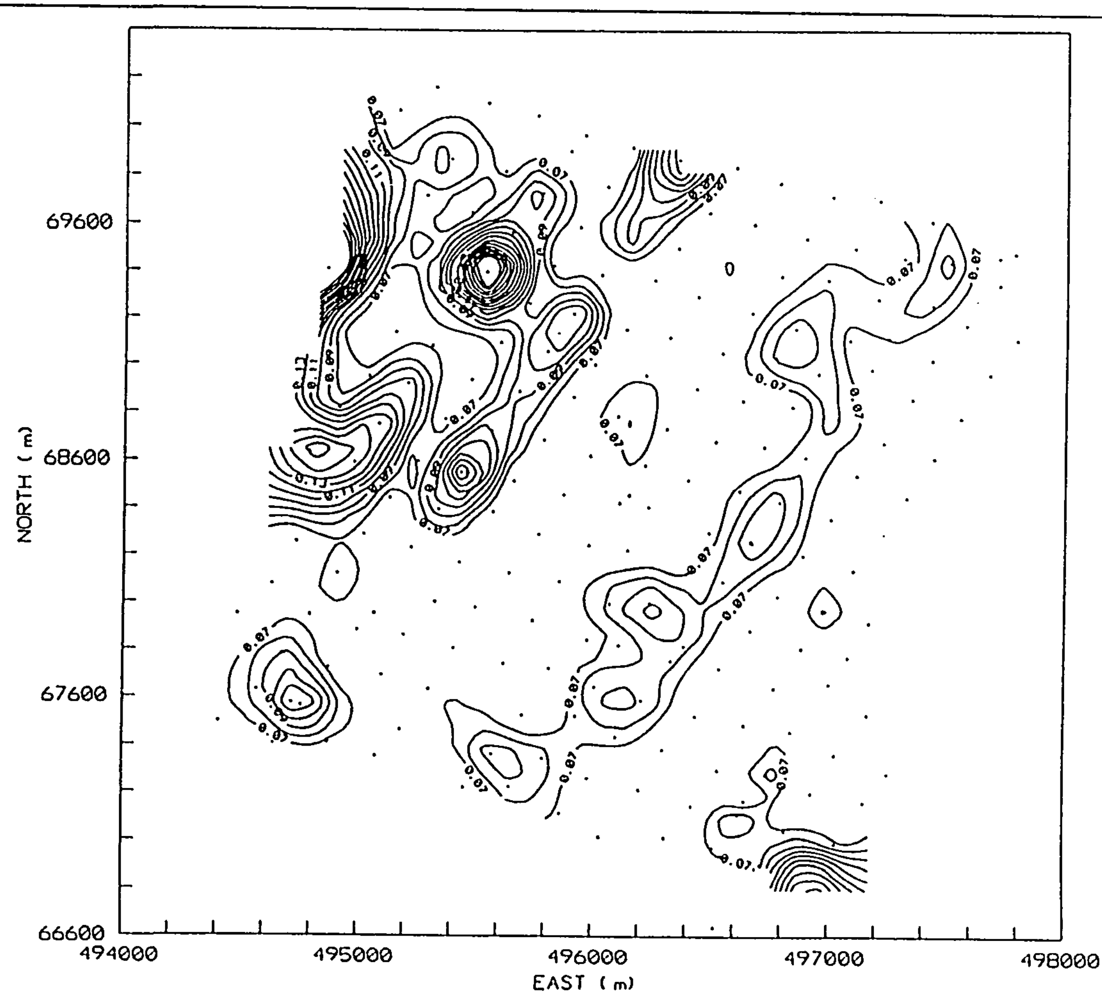


Figure 23. Fluorometric concentration contours greater than 0.06 μg/L for unfiltered samples collected between August 22 - 25, 1993

Figure 19 shows the contours generated from the data collected in the second half of the survey. This plot is very different from the unfiltered results presented in Figure 18. No elevated concentrations were present in the northwest section of the study area. Concentrations throughout the area were much lower than those recorded in the first half. The only exception was a single high value in the southeast corner. Again, these characteristics can be seen more clearly in Figure 24, in which only contours greater than the maximum background of value 0.06 μg/L are plotted. The considerable difference between the plots from each half of the survey are more obvious when Figures 23 and 24 are compared. Because of these obvious variations in the fluorometric contour patterns resulting from interference due to the passage of sand particles through the fluorometer, the near-bottom data collected during the first half of the survey were not considered for further analysis.

Overall, not considering signals resulting from interference from suspended particulates, the data are characterized by a small signal-to-noise ratio. That is, the variations in readings for the majority of the valid data are small, similar to background levels, and close to the limits of deductibility of the instrument. Statistical analyses of the data were performed to distinguish possible signals from the background variations. The mean and standard deviation were calculated for the 3-sec averaged data collected each day (Table 2).

A time-series analysis of the fluorometry data was performed to separate possible signals from the background variations. Data were plotted as scattergrams of observations versus concentration, and the mean value and 95-percent confidence limits, represented by 1.65 standard deviations were also plotted on the scattergrams. This information is presented in Appendix B.

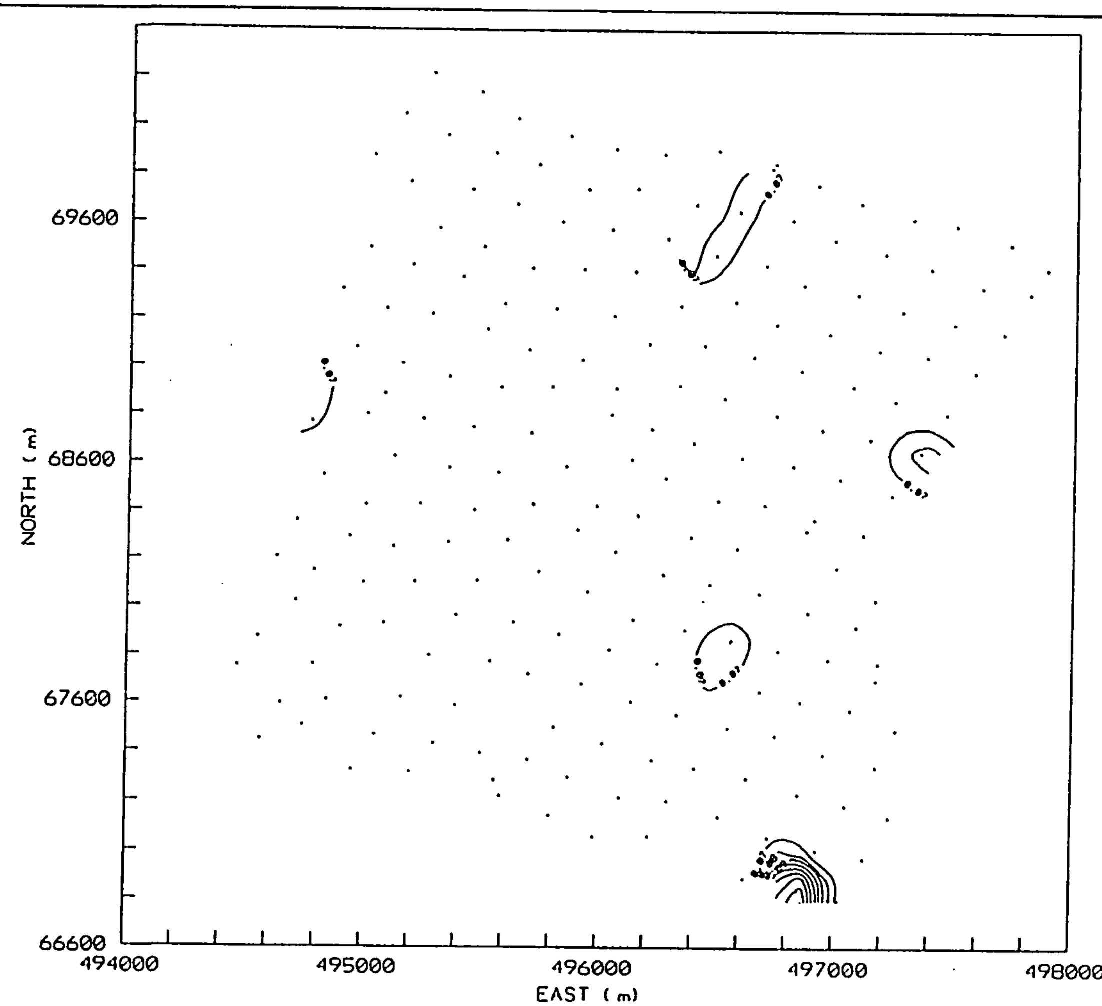


Figure 24. Fluorometric concentration contours greater than 0.06 μg/L for filtered samples collected between August 27 - 30, 1993

Data from the first half of the survey, when only one inline filter was used, were considerably more dispersed than data from the second half, when two additional filters were placed in the hose line. This large dispersion correlates to the high readings (shown in the corresponding line graphs in Appendix A), which were attributed to backscattering interference from suspended particles passing through the flow cell of the fluorometer.

Data from the second half of the survey were more consistent. Only short sections of a few lines had data exceeding the 95-percent upper confidence limit. These sections were at the beginning of Lines 5A, 3A, 16A, and 17A; the end of Lines 6A and 8A (which was surveyed in opposite direction to other lines); and in the center of Lines 17A and 13A (see Appendix A).

Points falling outside the upper confidence limit were identified for further inspection, after the data were contoured, to determine the extent of continuity of elevated values across adjacent sampling locations. If elevated concentrations were observed in any adjacent sample, it suggested that the elevated readings were more likely to be valid indications of higher concentrations of tracer. This was especially true if elevated readings were observed at adjacent points that were not sampled concurrently in time.

These statistical analyses and contouring of the data identified five possible areas of elevated concentrations. However, at three of the areas the differences (0.02 - 0.03 μg/L) between the elevated readings (0.07 μg/L) and background values (0.04 - 0.06 μg/L) were close to the sensitivity limit (0.02 μg/L) of the fluorometer. The fourth signal, although stronger (0.08 μg/L), was a single reading of short duration. The fifth reading, in the southeast corner of the study area (0.18 μg/L), was three times the background concentration. High concentrations were recorded here at adjacent locations on two different days in shallow water, close to where freshwater seeps and bubbles had been previously reported. The location is at the boundary of the study area and no data are available to the south.

Postsurvey Sampling

For each of the eight sets of analyses, the range of fluorescence concentrations was about 0.02 µg/L, similar to the sensitivity of the instrument, and also similar to the range between the two background samples. The average value for each set of ten samples varied by 0.028 µg/L, from -0.016 to 0.012 µg/L. The negative values are a result of the instrumental variation when measuring concentrations about zero and do not imply that negative concentrations were measured. Standard deviations for each of the eight sets of analyses are less than 0.007 µg/L and indicate a small spread in the measured concentrations (Table 4). Average concentrations for each of the ten samples, measured over the 2-month period, are less than 0.007 µg/L, which is also less than the sensitivity of the instrument. None of this postsurvey data supported the possible distribution patterns discussed in the previous sections.

Effects of Residual Chlorine on Fluorescence

The residual chlorine concentrations reported in the LWRF effluent, between 0.1 - 0.6 mg/L, and occasionally as high as 2.0 mg/L, are much lower than generally found in practice (Deaner 1973). The relationship between applied chlorine dose and the subsequent residual chlorine concentration depends upon the organic content and amount of ammonia present in the effluent (Deaner 1973). These characteristics vary widely between treatment facilities, being a function of wastewater influent characteristics. The effluent used in the laboratory testing appeared to have a lower chlorine demand (a dose of 27 mg/L resulted in a residual concentration of 11 mg/L) than the LWRF effluent (a dose of 12 - 14 mg/L resulted in a residual concentration of 0.1 - 0.6 mg/L). That is, the test effluent retained 41 percent of the added chlorine, whereas the LWRF effluent routinely consumed all but 1 - 4 percent of the added chlorine. Consequently, a much lower chlorine residual was available in the LWRF effluent to react with the fluorescent dye than in the laboratory tests, and it is concluded that the resultant fluorescence losses due to chlorine reactions were insignificant in the field study.

Maximum Measurable Effluent Dilutions

The observed background concentrations within and beyond the study area varied from 0.04 to 0.06 μg/L throughout the water column. The field resolution of the fluorometer was estimated to be 0.02 μg/L. Based on these values, for the tracer and effluent to be present in the study area at levels undetectable by the instrumentation (less than 0.06 to 0.08 μg/L), effluent dilutions of at least 3,500:1 would have occurred. This assumes the effluent from all four wells (0.25 m^3/sec [5.6 mgd]) is completely mixed before entering the ocean waters and the initial tracer concentration of tracer was 70 μg/L.

If the effluent from Well No. 2, injected at an average rate of 0.13 m^3/sec (3.0 mgd), did not mix with the effluent from the other wells after injection, effectively increasing the initial tracer concentration to 130 μg/L, then the overall dilution of the effluent, resulting from mixing with groundwater and subsequently from mixing with ocean waters before detection, would be at least 6,500:1.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Of the two objectives defined for this study, the first, determining if effluent can be detected in the coastal waters, was achieved. The second, delineating the effluent seeps on the sea floor was not fulfilled. Although elevated fluorescence was detected at the edge of the study area, the exact source was not identified. However, valuable information was gained from the study.

The field study was designed so that useful information about the characteristics of the submarine effluent dispersion patterns could be determined even though elevated concentrations of tracer were not conclusively detected.

1. The probability of tracer entering the coastal waters within the study area as a single plume is very low. No tracer was detected in the water column away from the seafloor or at the surface. It is more likely that if the tracer were present, it entered through a large number of discrete points or through one or more wide-area seeps at low flow rates and was diluted to undetectable concentrations rapidly and within short distances, horizontally and vertically, from the point(s) of influx.

2. Fluorescence readings of between 0.08 and 0.18 ppb were recorded at five small areas within the study area (Figure 22), but the data did not conclusively show the presence of the tracer at any of these locations. This was because the elevated readings were either close to the detection limit of the fluorometer, or because the elevated readings were

isolated events of short duration. No correlation was evident between the fluorometric survey results and the postsurvey fluorescence analyses, even through some locations of the postsurvey sampling sites were chosen to correspond with the locations of the five sites of elevated concentration. Further sampling would be required at each of the five locations to verify the presence of elevated effluent concentrations.

3. For the tracer to be present and undetectable in the water column, the tracer and the effluent must have undergone dilutions of between 3,500 and 6,500 times the injected concentrations. If the tracer were present in the near-bottom water, the tracer had been diluted to undetectable concentrations vertically within the first 10 to 30 cm of the bottom, or horizontally within 100 to 200 m of its seabed source, as it was not detected at the sampling points.

4. A final conclusion is that it is necessary to filter water samples though a 100-micron or finer screen prior to measuring fluorescence to eliminate interference. During the first half of the survey, frequent interferences were observed.These readings were attributed to a light backscattering effect, a result of sand and smaller particles passing through the fluorometer.

<u>Possible Explanations for Absence of Dye in the Study Area</u>

An alternative, but unprovable, conclusion is that the tracer was not present in the study area during times that sampling was being conducted. This suggests that any of the following may have occurred:

1. The tracer had not yet reached the study area 60 days after first being added to the effluent at the injection well.

2. The tracer reached the study area sometime between 60 and 163 days after initial injection but was not present at any of the ten postsurvey sampling locations.

3. The effluent disperses into the coastal waters at some location in deeper waters outside the study area after flowing through a lava tube or similar preferential pathway. Another alternative is that the effluent entry into the coastal waters occurs to the south or north of the study area, due to unknown subsurface hydrogeologic features.

4. The tracer and effluent emerged very close to the shoreline (within a few meters) in very shallow water (less than 2 m) and were not detected because the nearshore sampling locations were not sufficiently close to the shoreline. However, low current velocities, and shallow water would slow the dilution of the effluent compared to conditions in deeper waters. Thus the effluent at detectable concentrations would spread further from the influx points, both along the shore and offshore. Although this cannot be quantified, sampling close to the influx points would not be as critical in very shallow waters as it would be in deep waters, where the effluent experiences relatively more rapid dilution. So this possibility is considered unlikely.

5. The fluorescence of the tracer was attenuated by degradation from the residual chlorine in the effluent (unlikely at the low chlorine concentrations reported), adsorption in the groundwater, or other decay processes.

Recommendations for Continued Study

In support of the specific objectives of determining the dispersion patterns in the nearshore waters of effluent from the LWRF injection wells and of verifying if the slightly elevated levels of fluorescence were valid, the following recommendations are made.

The pump and sampling assembly could be improved only at significant increase in cost and complexity. A towable instrument platform, commonly called a "batfish," whose depth in the water can be controlled from the tow vessel would allow rapid and continuous sampling throughout the water column. However, a larger vessel, larger crew, and larger budget would also be required.

The fluorometer used was a state-of-the-art instrument. The resolution of this fluorometer was reported by the manufacturer to be at least 0.02 ppb for Rhodamine WT in seawater. This is approximately two to three times less than the background, naturally-occurring fluorescence (0.06 µg/L) and similar to the measured variation in background levels. The sensitivity is appropriate for this level of detection, and no advantages could be gained from using a more sensitive instrument.

Another possible approach, but not recommended, to better determine the presence of effluent in the study area would be to increase the concentration of the tracer added to the effluent before discharge into the injection well. This is not recommended because of the cost. An order of magnitude increase in initial tracer concentration, from about 100 ppb to 1,000 ppb, would result in equivalent increases in the concentrations at the study area. This would also require an equivalent order of magnitude increase in costs, from $20,000 to $200,000 for the dye.

Future study should be concentrated at the southeast corner of the present study area and further to the south. Of the five occurrences of elevated readings, those recorded in the southeast corner of the study area are the most likely to indicate the presence of the tracer. It is possible that unknown hydrogeologic features may constrain the effluent flow more toward the south than downslope, directly into the coastal waters.

A more efficient approach, perhaps, would be to focus the sampling effort in shallow waters along the shoreline. If indeed the aquifer is highly permeable and unconfined in the immediate area, the injected effluent would be expected to surface at the shoreline.

In-situ conductivity probes operated from the shoreline or from a small dinghy could be used to detect and delineate areas of freshwater discharges. Once the possible freshwater seeps had been identified, only these areas would be monitored for increased nutrient levels or with a fluorometer once the tracer had been added to the effluent. Using this approach all sampling equipment could be land-based. A motor vessel and navigation system would be unnecessary. A sampling hose or collection bottles could be carried by hand to waters at least 2 m deep, and by a scuba diver, or dinghy to waters 10 m deep or more.

Policy and Regulatory Implications

Determining the fate of injection well sewage effluent is important within west Maui and elsewhere from several perspectives. From a technical point of view, understanding the transport mechanics of injected effluent is necessary from an engineering and an urban planning perspective. Future wastewater treatment and disposal capacity are important considerations as population pressures continue in desirable coastal communities. Efficient engineering solutions must be balanced with appropriate, community-backed environmental safeguards. The experience and data gained from this study can add, albeit in a modest fashion, to the body of engineering knowledge.

From a regulatory standpoint, disposal of treated sewage effluent is under the jurisdiction of both state and federal agencies in Hawaii and nationally. The development of a method to determine the fate of effluent after injection will provide valuable data that can be used as a scientific datum for appropriate guidelines and regulations addressing the sometimes conflicting demands of community growth, economic prosperity, human health, and environmental protection.

USEPA and HDOH have developed a joint watershed strategy to investigate the causes of the algae blooms. As well as this effluent tracer study, an algae mapping effort, an assessment of terrestrial nutrient loadings entering the coastal waters, a survey of the biodiversity of the coral reefs to determine if algal predators are missing, and a study of algae physiology have been funded to create a sound scientific foundation for future government actions. In conjunction with possible regulatory changes, the agencies are pursuing a number of nutrient source control measures with the cooperation of the County of Maui, the U.S. Soil Conservation Service, and agribusiness partners. These activities include reclamation of treated sewage effluent and demonstration projects for reduction in fertilizer use by incorporating best management practices (Fitzgerald and Tenley 1993).

Although results from this effluent tracer effort did not conclusively attain the objectives of the study, the results do show that no strong sources of concentrated effluent were present in the study area. This information provides some justification to policy makers to concentrate on other potential sources of nutrients in the west Maui watershed and redirect research efforts toward these sources, toward the development of better management practices or toward research focused on the life cycles of the algae types and possible natural causes of the observed episodic algal blooms.

On a more general basis, the results of this study lend themselves to the implication that injection well disposal appears to dilute and disperse domestic wastewater at least as efficiently as conventional engineered ocean outfall systems. A substantial savings in construction costs could be realized in the construction or expansion of existing wastewater treatment plants if a choice was possible between drilling injection wells or building a seafloor outfall and diffuser structure.

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APPENDIX A

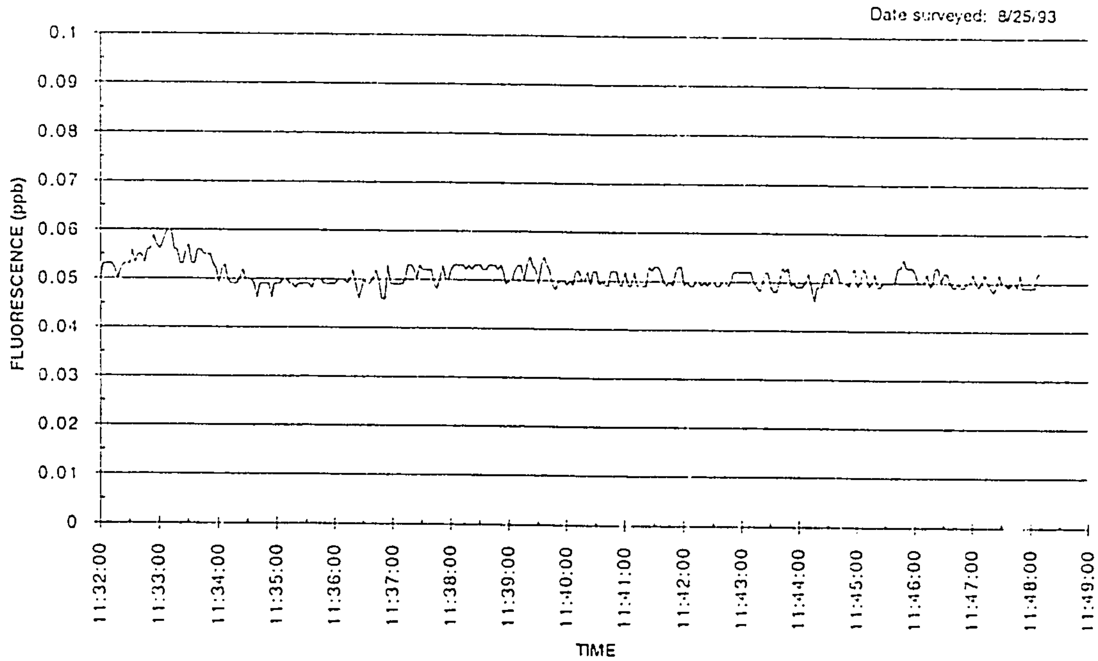
LINE PLOTS OF THE DATA

Line graphs of fluorescence versus time and water temperature versus time were plotted for each transect surveyed. Fluorescence is reported as concentrations (µg/L) relative to a calibration standard of 1.00 µg/L of Rhodamine WT in distilled water. The graphs are presented in numerical order. The date each line was surveyed is included on each graph. Transects with the suffix "-A" were surveyed during the second half of the study (August 27 - 30, 1993) and were located adjacent to and 100 m to the west of the lines numbered without the suffix (see Figure 7).

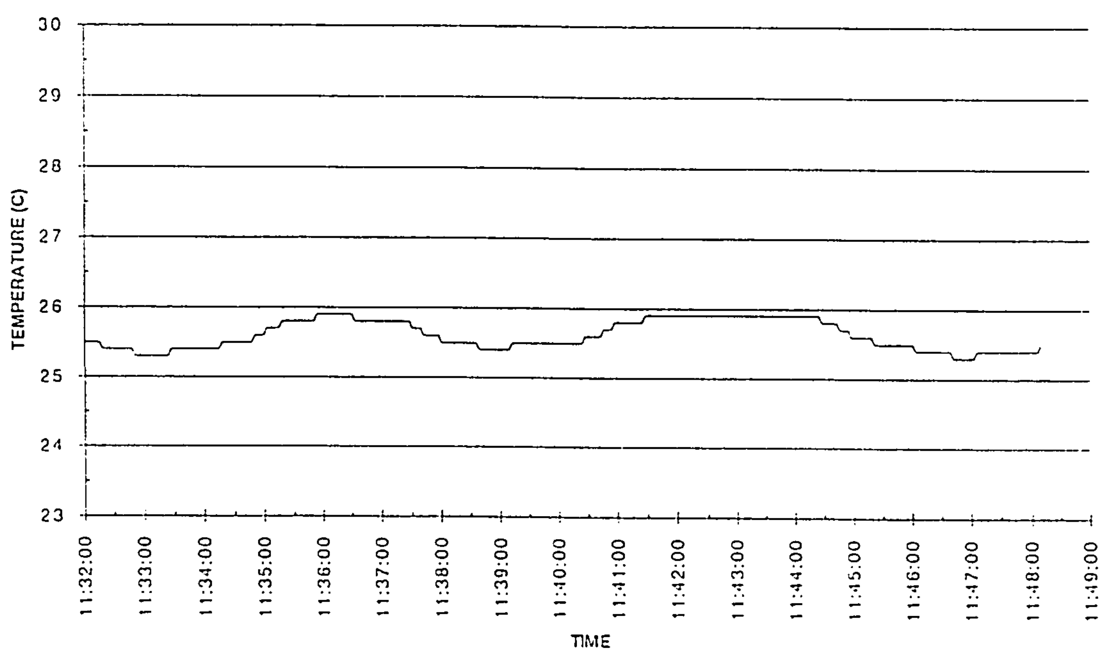
All line graphs of fluorescence are presented with a constant concentration range of 0 to 0.1 ppb (µg/L) to facilitate comparisons between graphs. For some transects, for which the peak recorded concentrations were greater than 0.1 ppb, a second graph, with a higher concentration range, is included.

Temperatures of the sample passing through the flow cell of the fluorometer were reported in °C. The instrument's internal temperature compensation device automatically adjusted the fluorometric reading for variations in temperature. A temperature coefficient of -2.6 percent per degree Celsius was used for Rhodamine WT (Turner Designs 1990). Recorded temperature variations were small throughout the survey, but were good indicators of the position of the pump within the water column. The lowest temperatures along any transect corresponded to near-bottom water; the highest temperatures were recorded from surface waters.

LINE 1

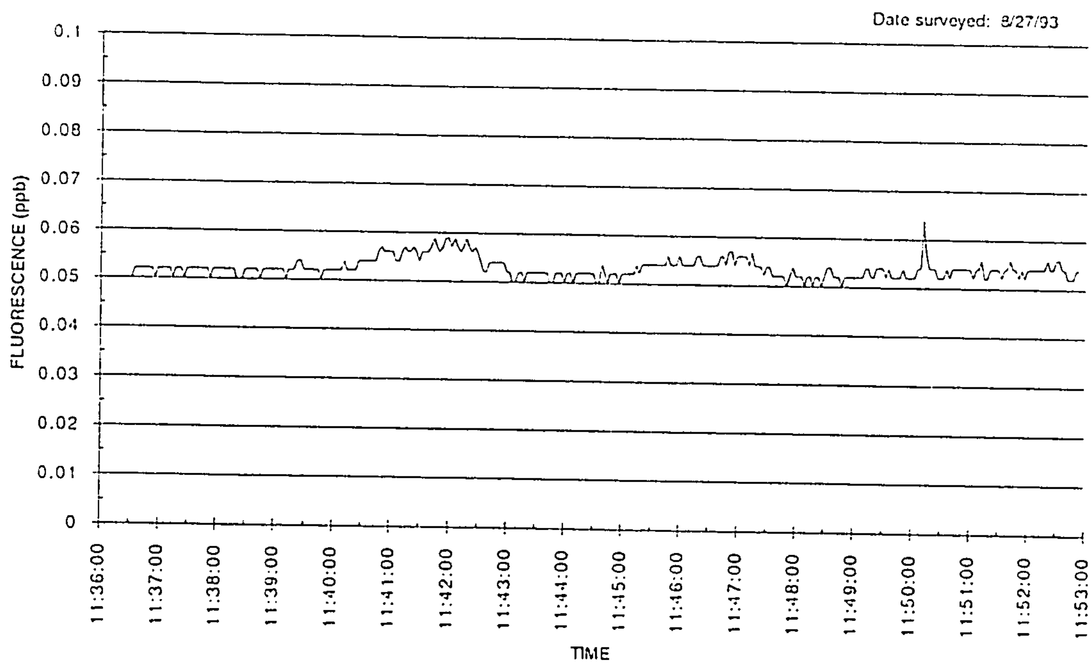


Date surveyed: 8/25/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
11:32:00
11:33:00
11:34:00
11:35:00
11:36:00
11:37:00
11:38:00
11:39:00
11:40:00
11:41:00
11:42:00
11:43:00
11:44:00
11:45:00
11:46:00
11:47:00
11:48:00
11:49:00
TIME

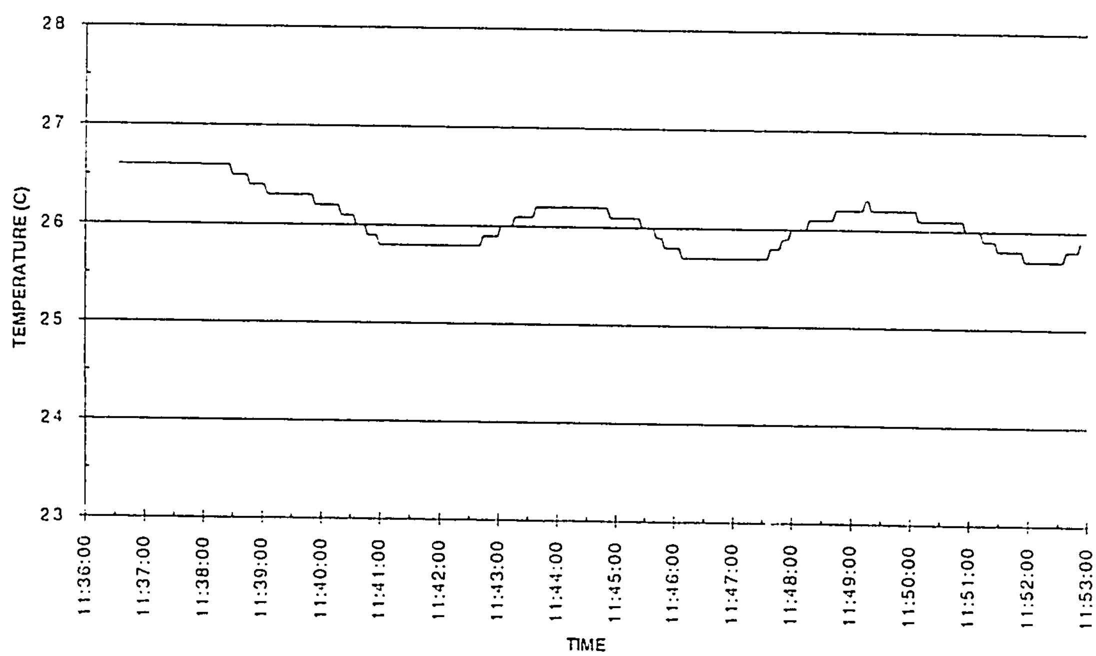


TEMPERATURE (C)
30
29
28
27
26
25
24
23
11:32:00
11:33:00
11:34:00
11:35:00
11:36:00
11:37:00
11:38:00
11:39:00
11:40:00
11:41:00
11:42:00
11:43:00
11:44:00
11:45:00
11:46:00
11:47:00
11:48:00
11:49:00
TIME

LINE 1A

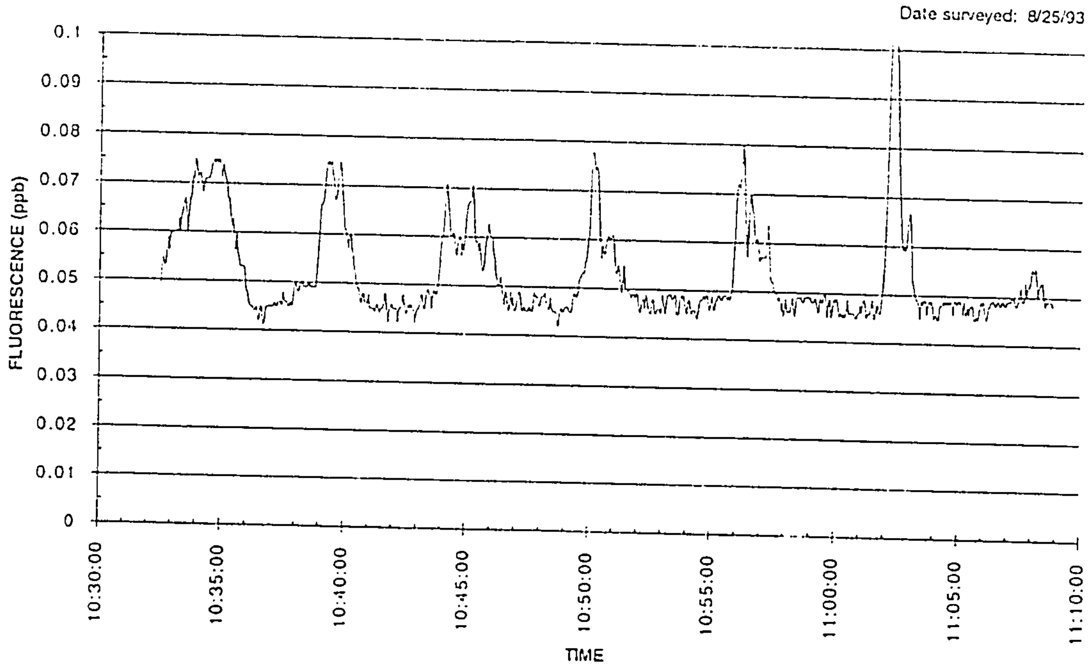


Date surveyed: 8/27/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
11:36:00
11:37:00
11:38:00
11:39:00
11:40:00
11:41:00
11:42:00
11:43:00
11:44:00
11:45:00
11:46:00
11:47:00
11:48:00
11:49:00
11:50:00
11:51:00
11:52:00
11:53:00
TIME

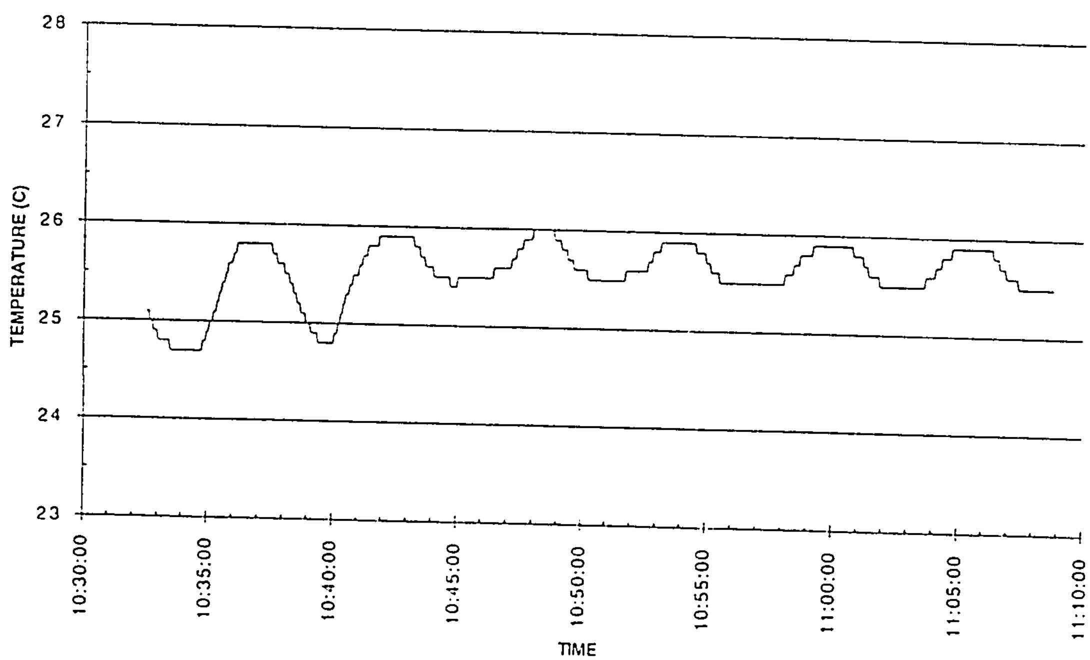


TEMPERATURE (C)
28
27
26
25
24
23
11:36:00
11:37:00
11:38:00
11:39:00
11:40:00
11:41:00
11:42:00
11:43:00
11:44:00
11:45:00
11:46:00
11:47:00
11:48:00
11:49:00
11:50:00
11:51:00
11:52:00
11:53:00
TIME

LINE 2

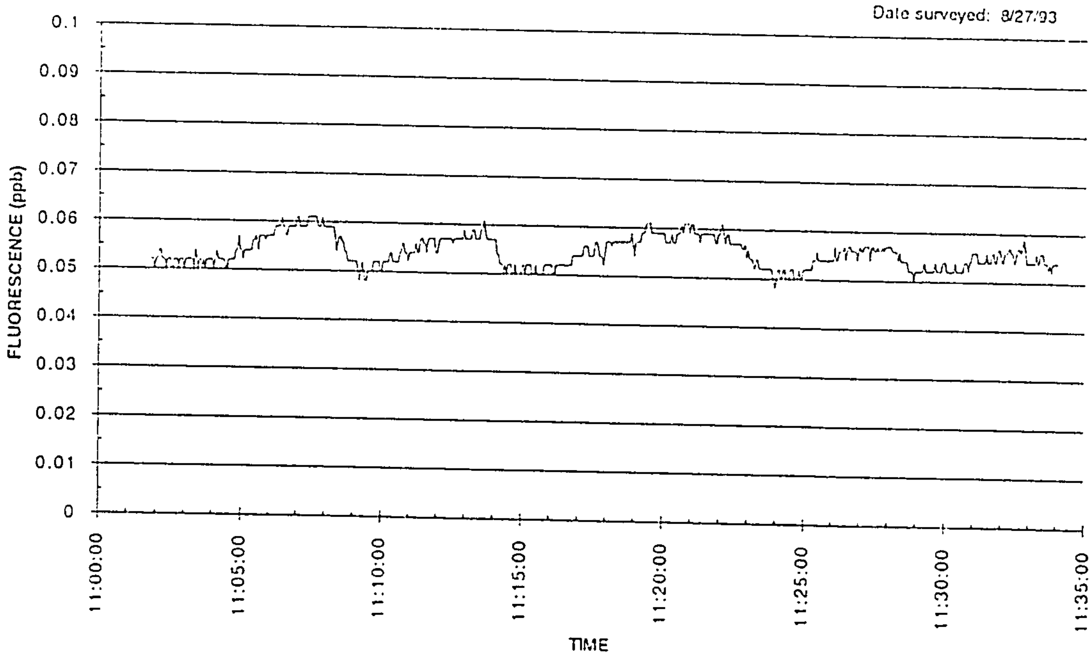


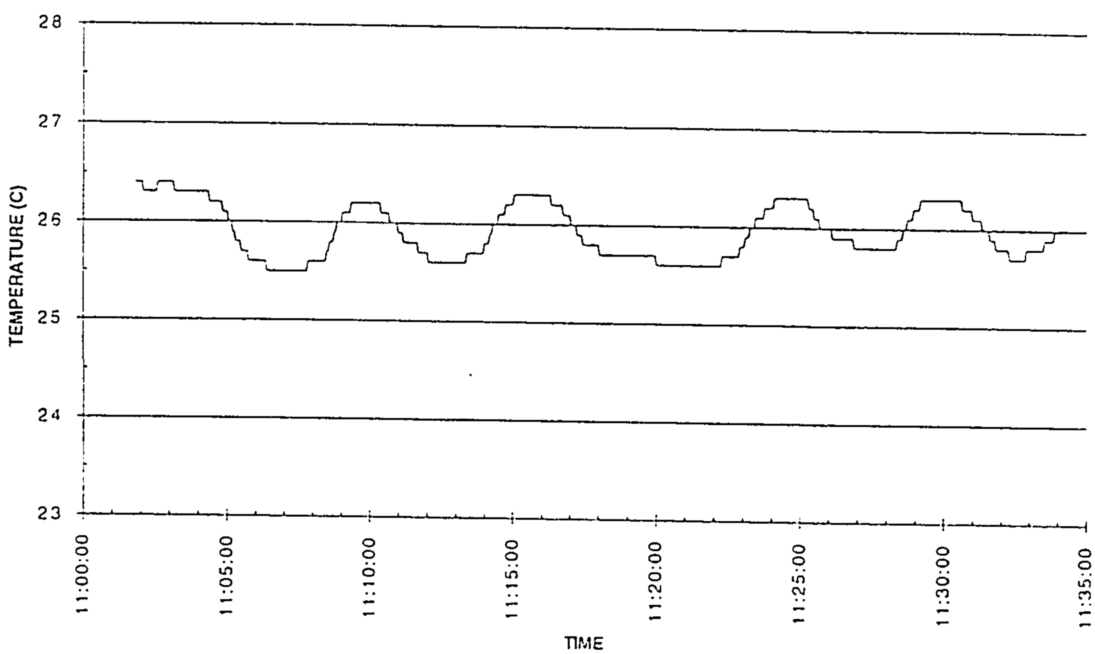
Date surveyed: 8/25/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
10:30:00
10:35:00
10:40:00
10:45:00
10:50:00
10:55:00
11:00:00
11:05:00
11:10:00
TIME



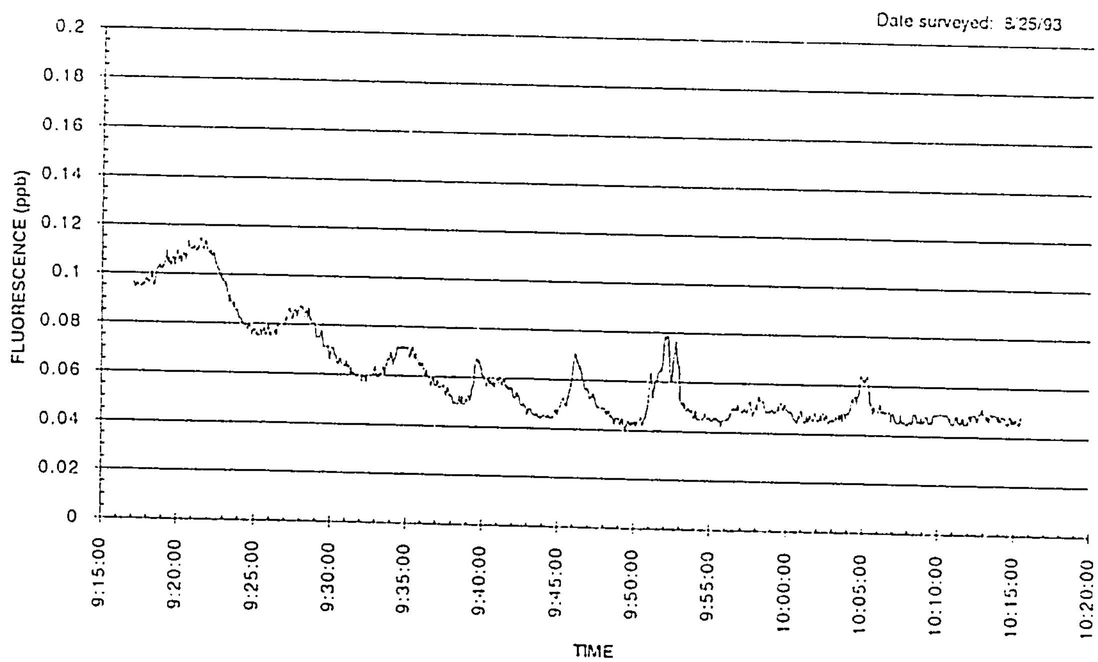
TEMPERATURE (C)
28
27
26
25
24
23
10:30:00
10:35:00
10:40:00
10:45:00
10:50:00
10:55:00
11:00:00
11:05:00
11:10:00
TIME

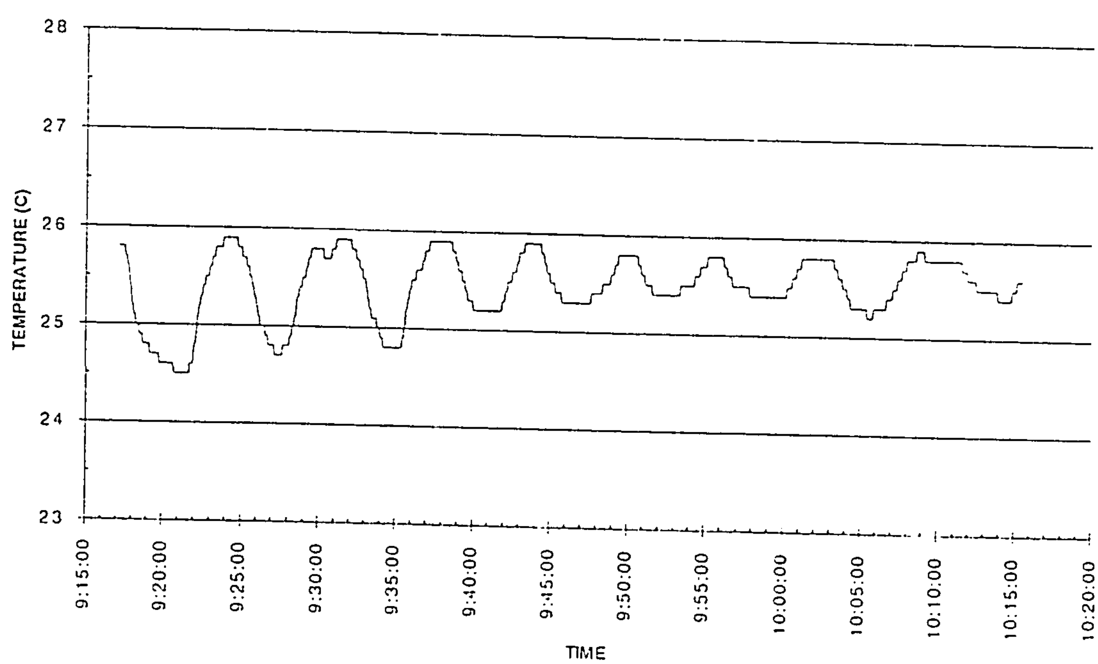
LINE 2A


Date surveyed: 8/27/93
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
FLUORESCENCE (ppb)
11:00:00
11:05:00
11:10:00
11:15:00
11:20:00
11:25:00
11:30:00
11:35:00
TIME

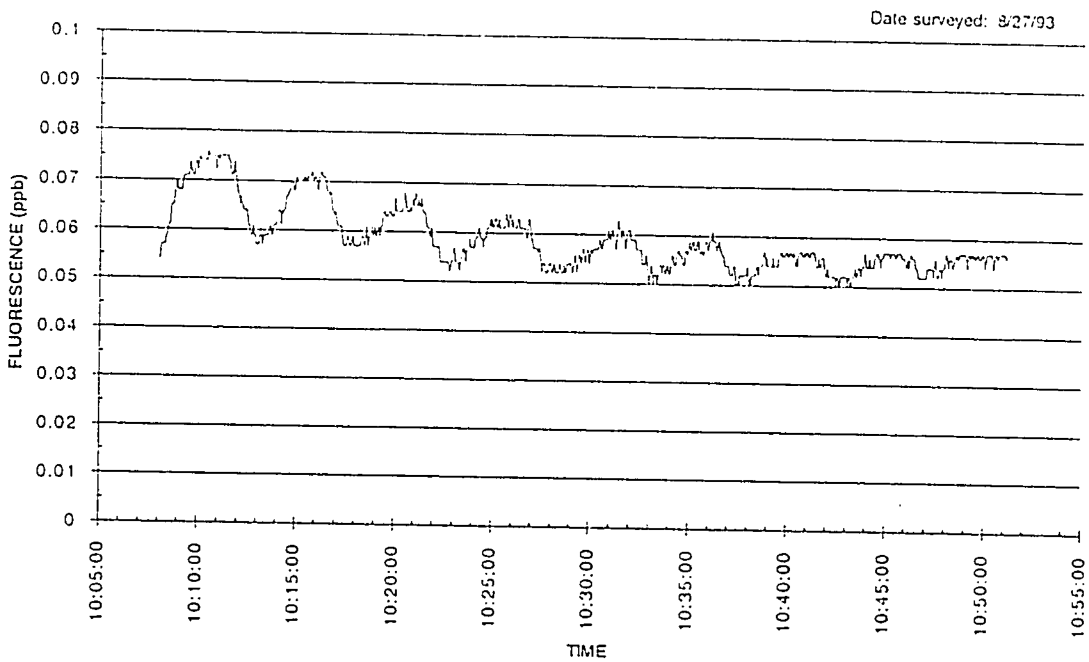

28
27
26
25
24
23
TEMPERATURE (C)
11:00:00
11:05:00
11:10:00
11:15:00
11:20:00
11:25:00
11:30:00
11:35:00
TIME

LINE 3

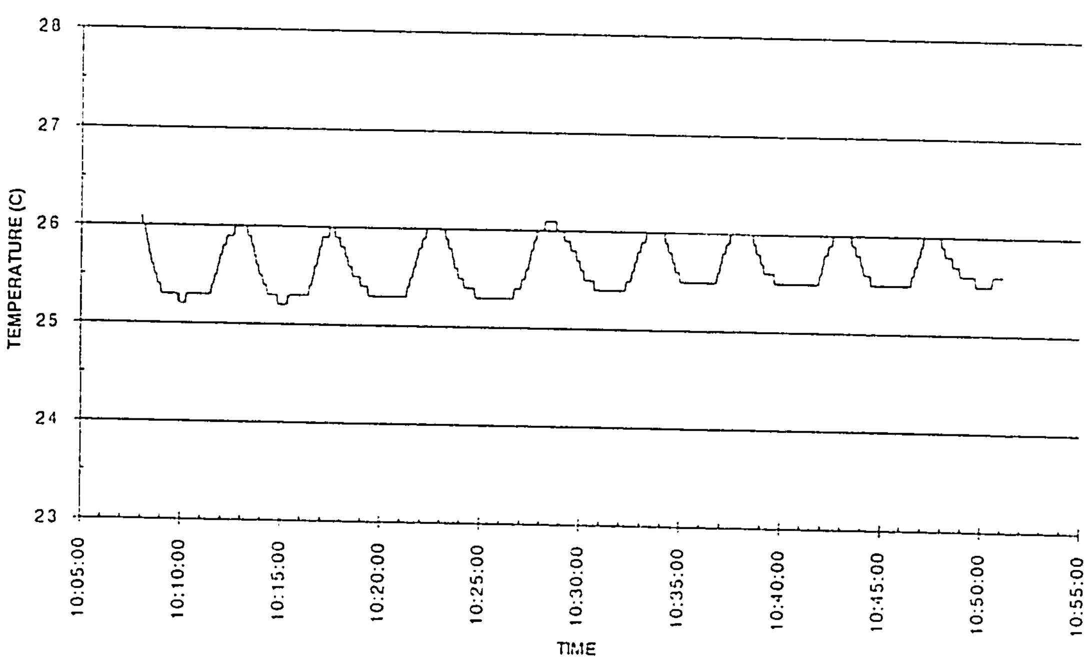

Date surveyed: 8/25/93
FLUORESCENCE (ppb)
0.2
0.18
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
10:00:00
10:05:00
10:10:00
10:15:00
10:20:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
10:00:00
10:05:00
10:10:00
10:15:00
10:20:00
TIME

LINE 3A

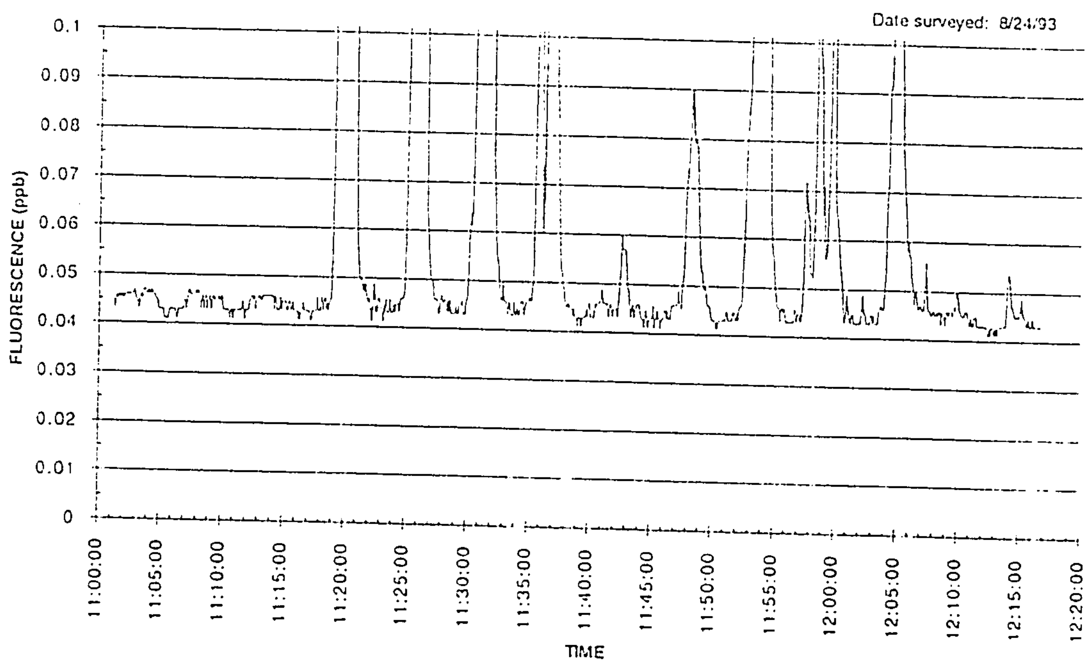


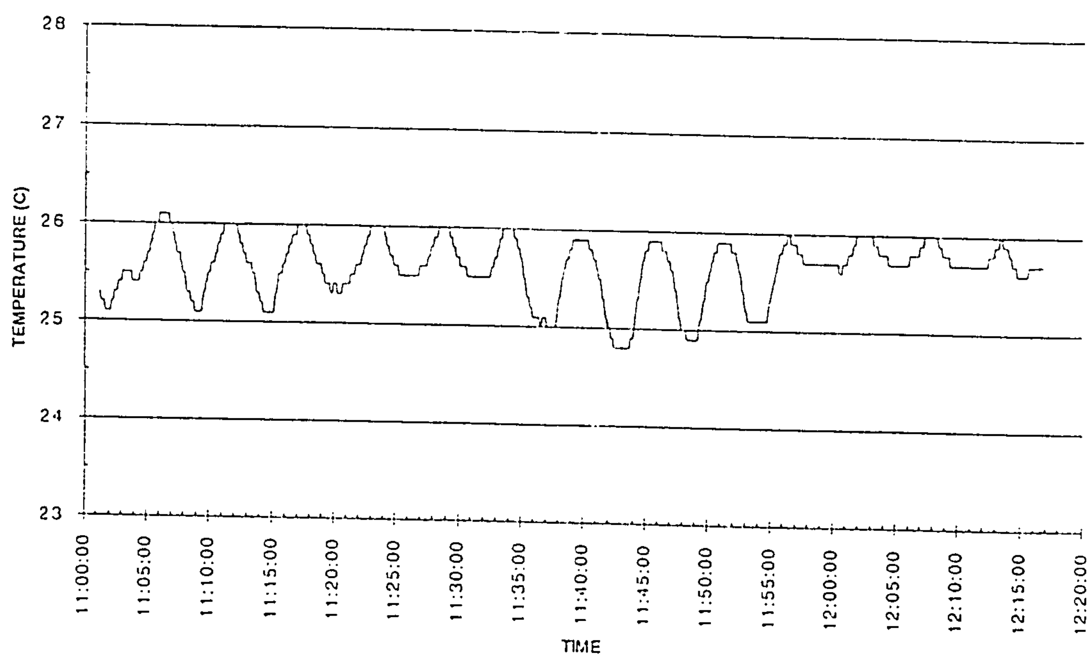
Date surveyed: 8/27/93
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
FLUORESCENCE (ppb)
10:05:00
10:10:00
10:15:00
10:20:00
10:25:00
10:30:00
10:35:00
10:40:00
10:45:00
10:50:00
10:55:00
TIME



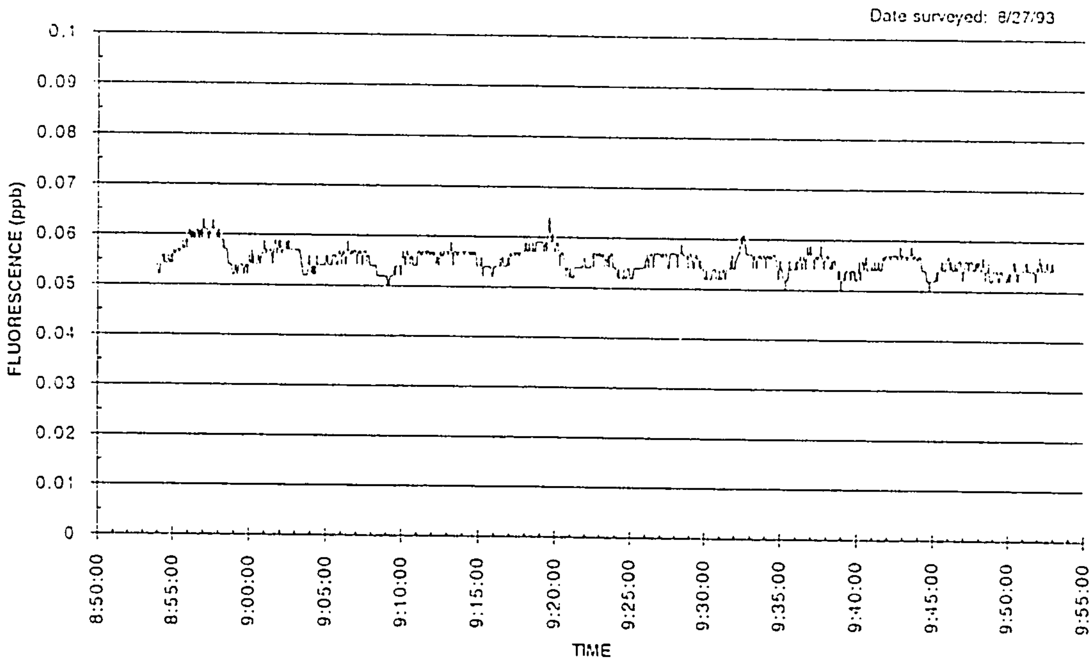
28
27
26
25
24
23
TEMPERATURE (C)
10:05:00
10:10:00
10:15:00
10:20:00
10:25:00
10:30:00
10:35:00
10:40:00
10:45:00
10:50:00
10:55:00
TIME

LINE 4

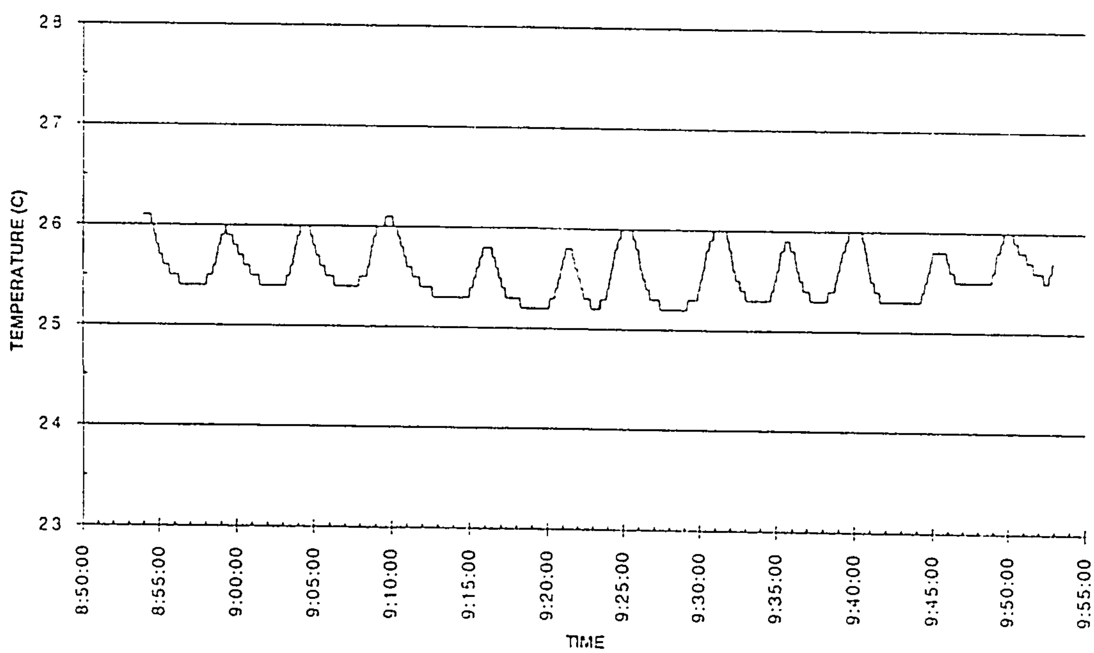

Date surveyed: 8/24/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
11:00:00
11:05:00
11:10:00
11:15:00
11:20:00
11:25:00
11:30:00
11:35:00
11:40:00
11:45:00
11:50:00
11:55:00
12:00:00
12:05:00
12:10:00
12:15:00
12:20:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
11:00:00
11:05:00
11:10:00
11:15:00
11:20:00
11:25:00
11:30:00
11:35:00
11:40:00
11:45:00
11:50:00
11:55:00
12:00:00
12:05:00
12:10:00
12:15:00
12:20:00
TIME

LINE 4A

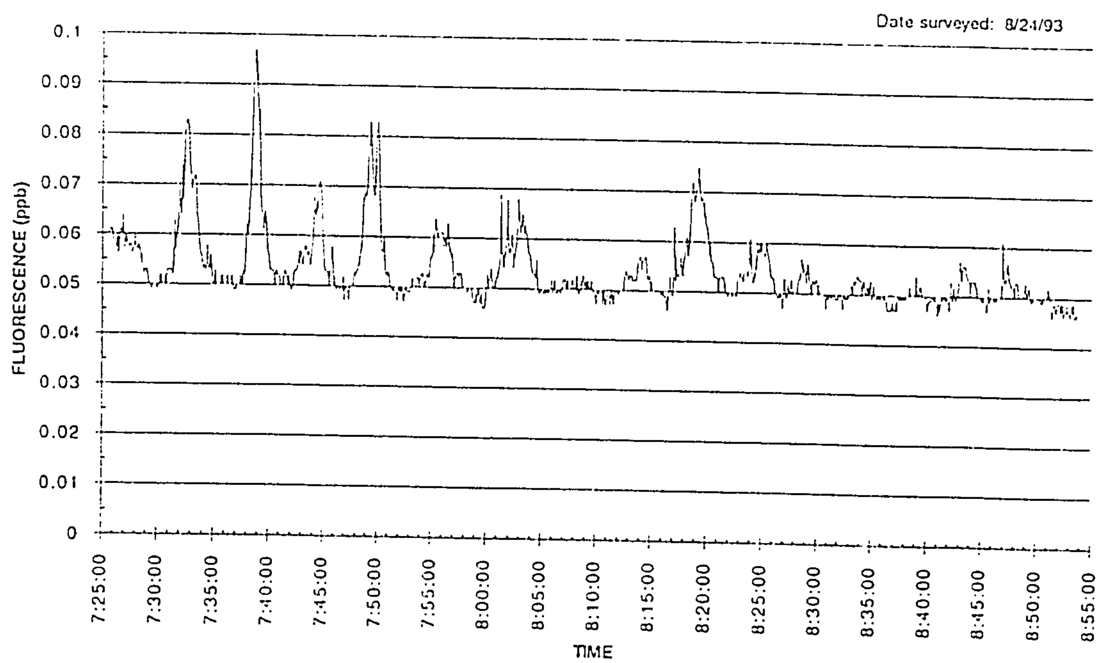


Date surveyed: 8/27/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
8:50:00
8:55:00
9:00:00
9:05:00
9:10:00
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
TIME

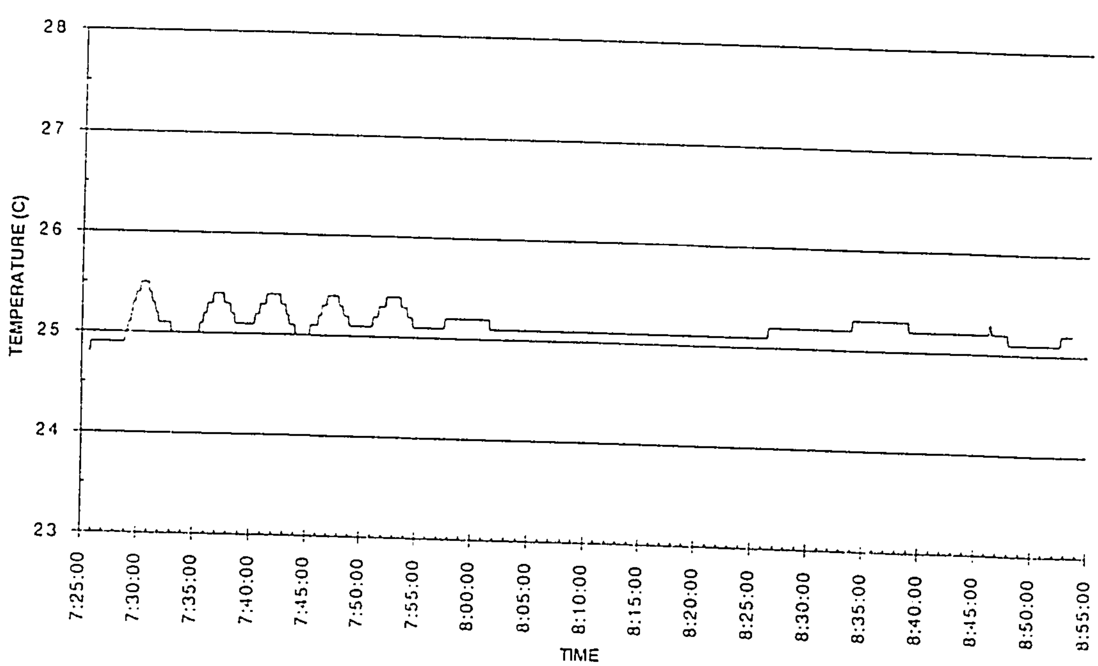


TEMPERATURE (C)
28
27
26
25
24
23
8:50:00
8:55:00
9:00:00
9:05:00
9:10:00
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
TIME

LINE 5

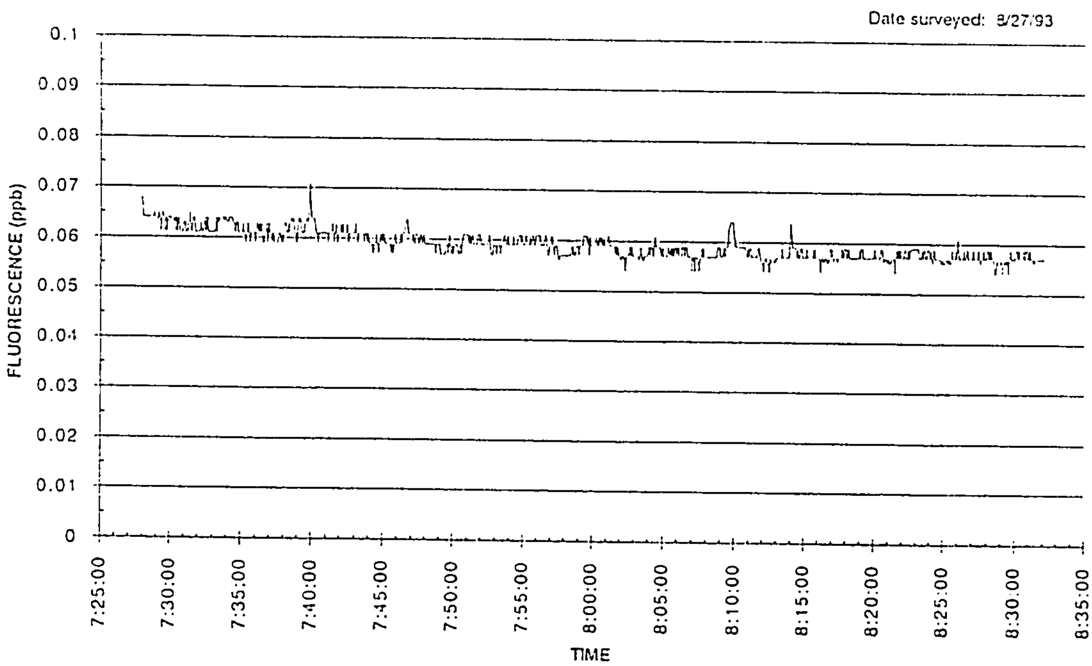


Date surveyed: 8/24/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
8:40:00
8:45:00
8:50:00
8:55:00
TIME

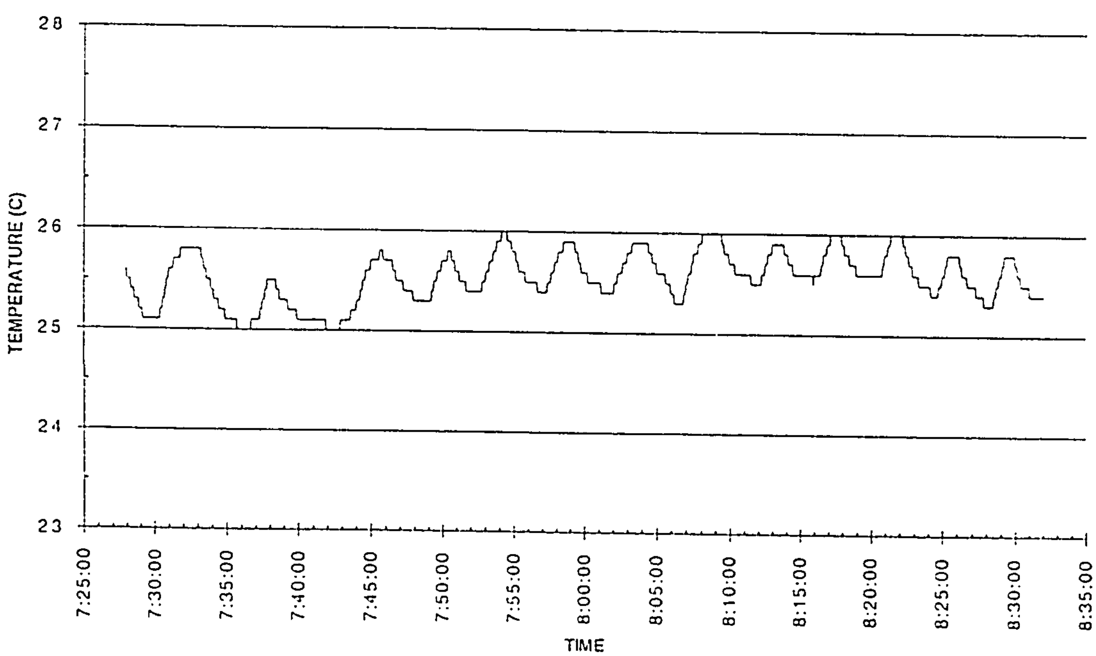


TEMPERATURE (C)
28
27
26
25
24
23
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
8:40:00
8:45:00
8:50:00
8:55:00
TIME

Line 5A

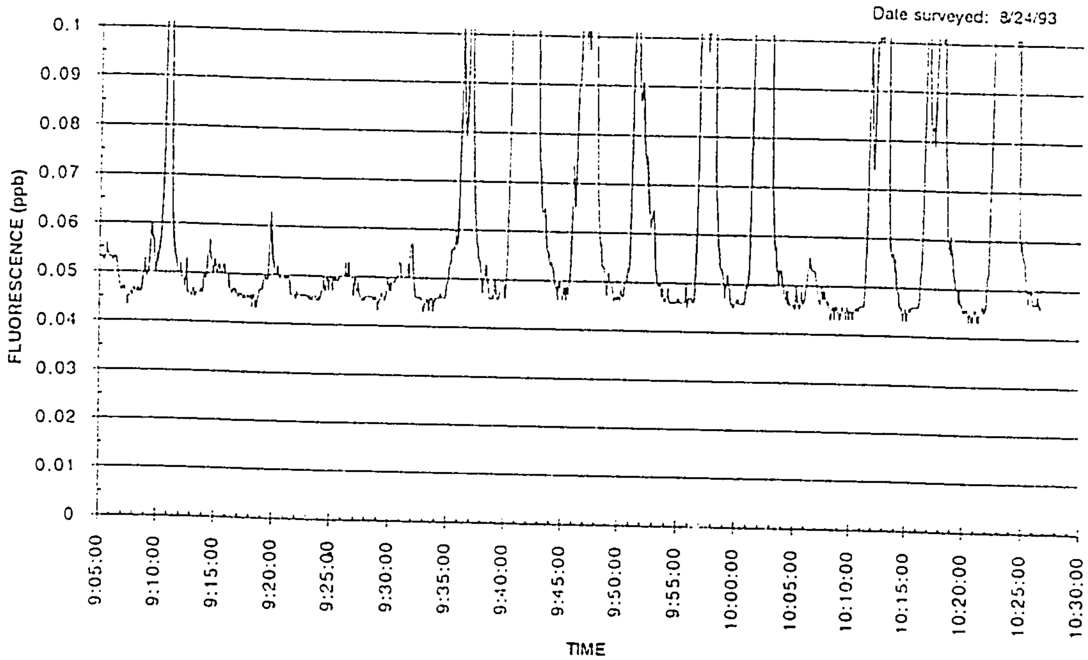


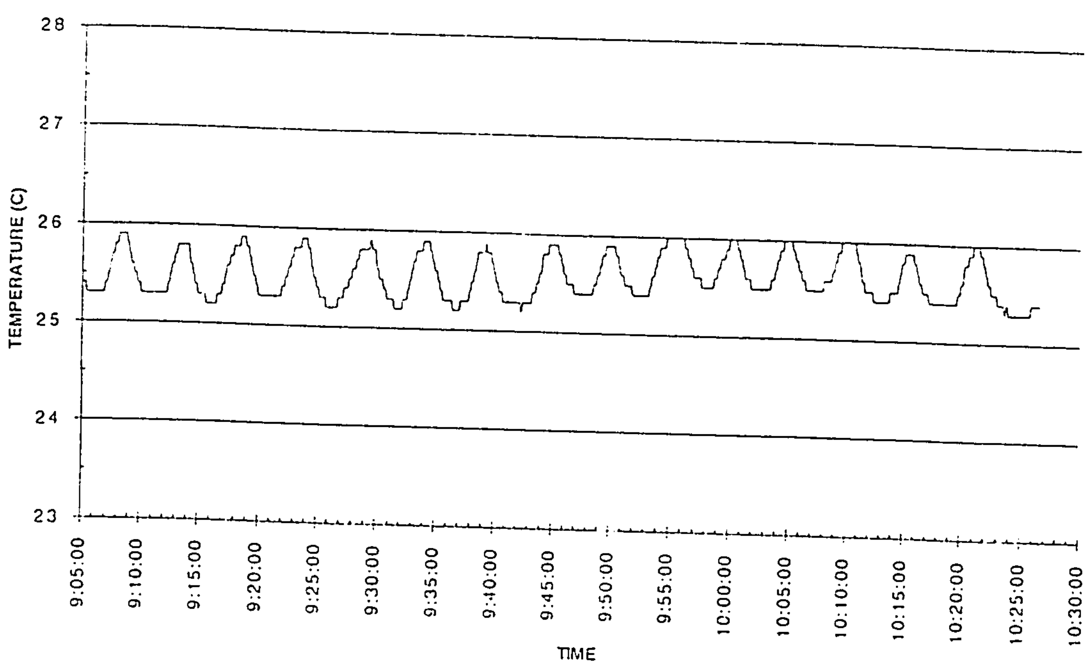
Date surveyed: 8/27/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
TIME



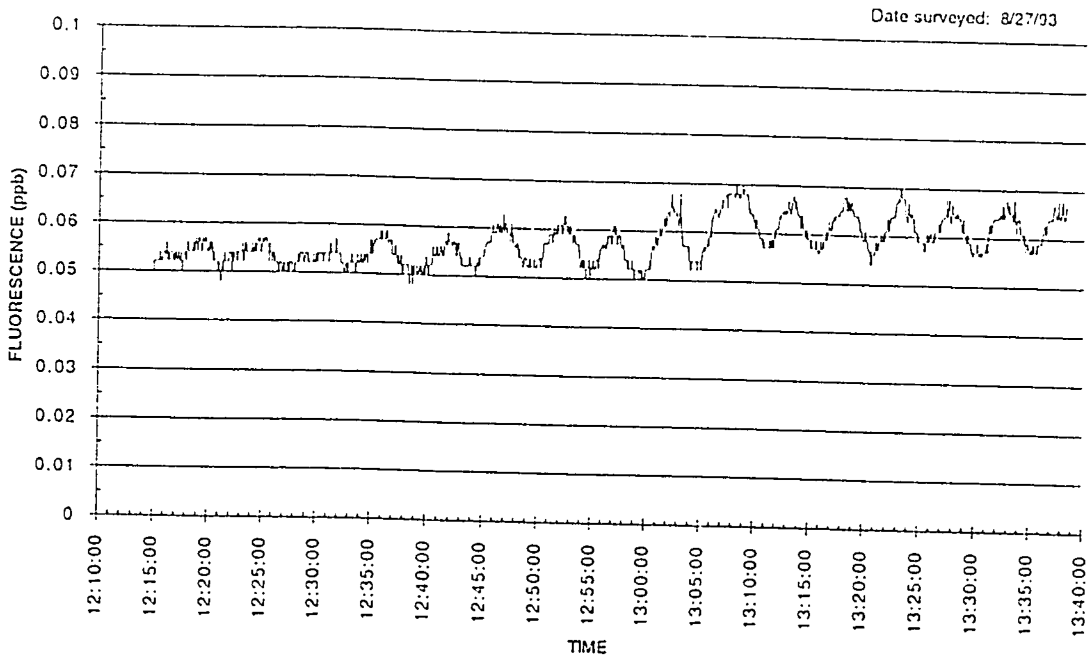
TEMPERATURE (C)
28
27
26
25
24
23
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
TIME

LINE 6

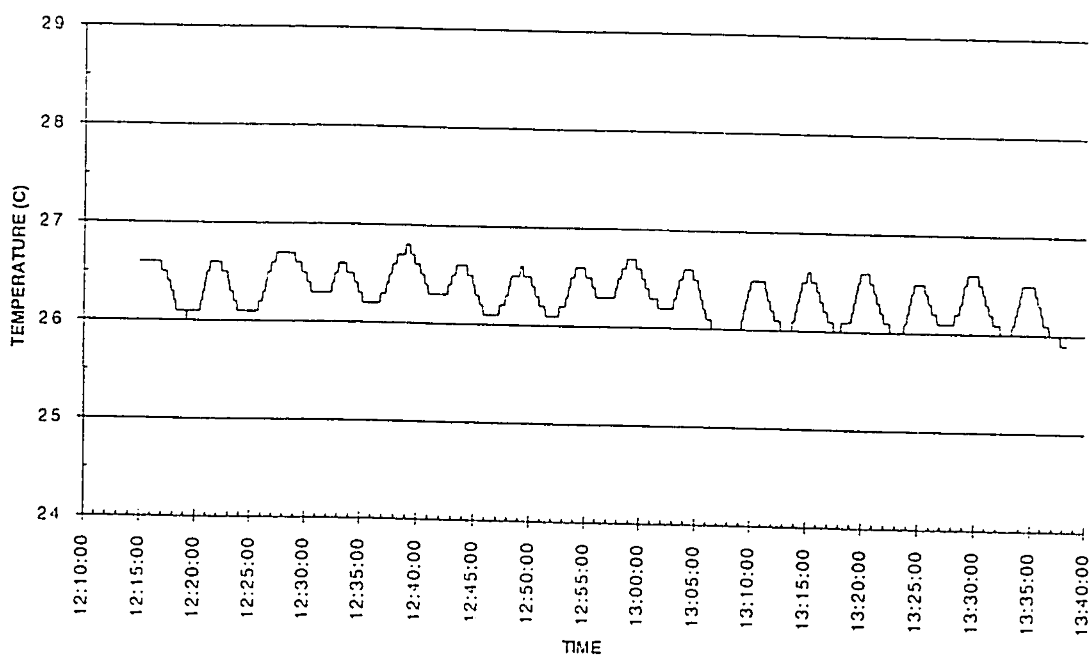

Date surveyed: 8/24/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
9:05:00
9:10:00
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
10:00:00
10:05:00
10:10:00
10:15:00
10:20:00
10:25:00
10:30:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
9:05:00
9:10:00
9:15:00
9:20:00
9:25:00
9:30:00
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
10:00:00
10:05:00
10:10:00
10:15:00
10:20:00
10:25:00
10:30:00
TIME

LINE 6A

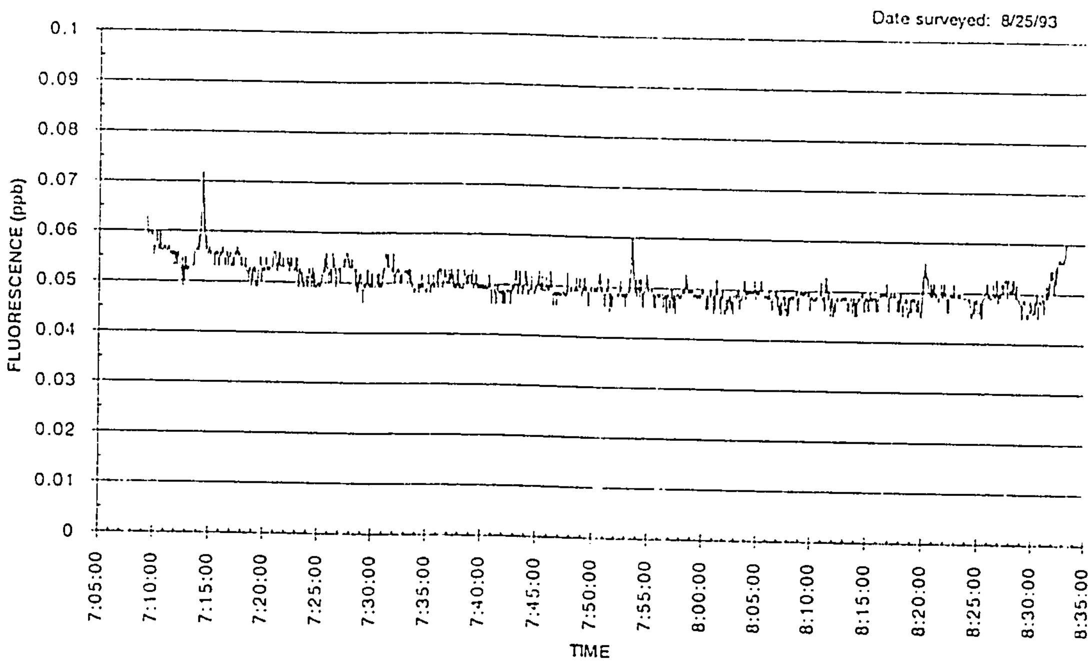


Date surveyed: 8/27/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
12:10:00
12:15:00
12:20:00
12:25:00
12:30:00
12:35:00
12:40:00
12:45:00
12:50:00
12:55:00
13:00:00
13:05:00
13:10:00
13:15:00
13:20:00
13:25:00
13:30:00
13:35:00
13:40:00
TIME

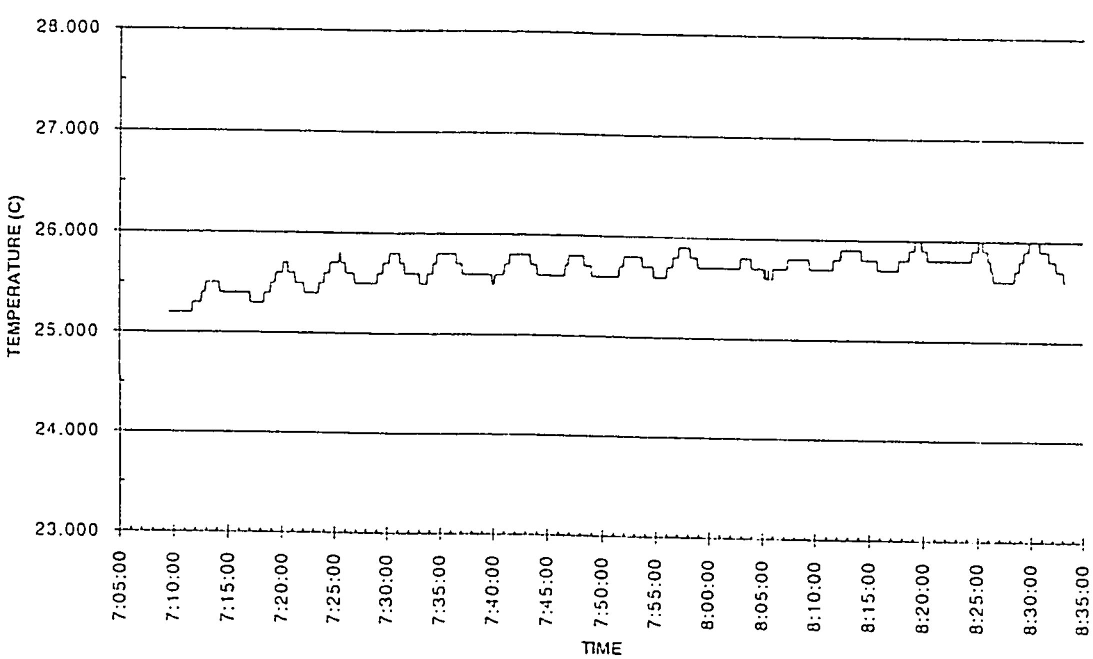


TEMPERATURE (C)
29
28
27
26
25
24
12:10:00
12:15:00
12:20:00
12:25:00
12:30:00
12:35:00
12:40:00
12:45:00
12:50:00
12:55:00
13:00:00
13:05:00
13:10:00
13:15:00
13:20:00
13:25:00
13:30:00
13:35:00
13:40:00
TIME

LINE 7

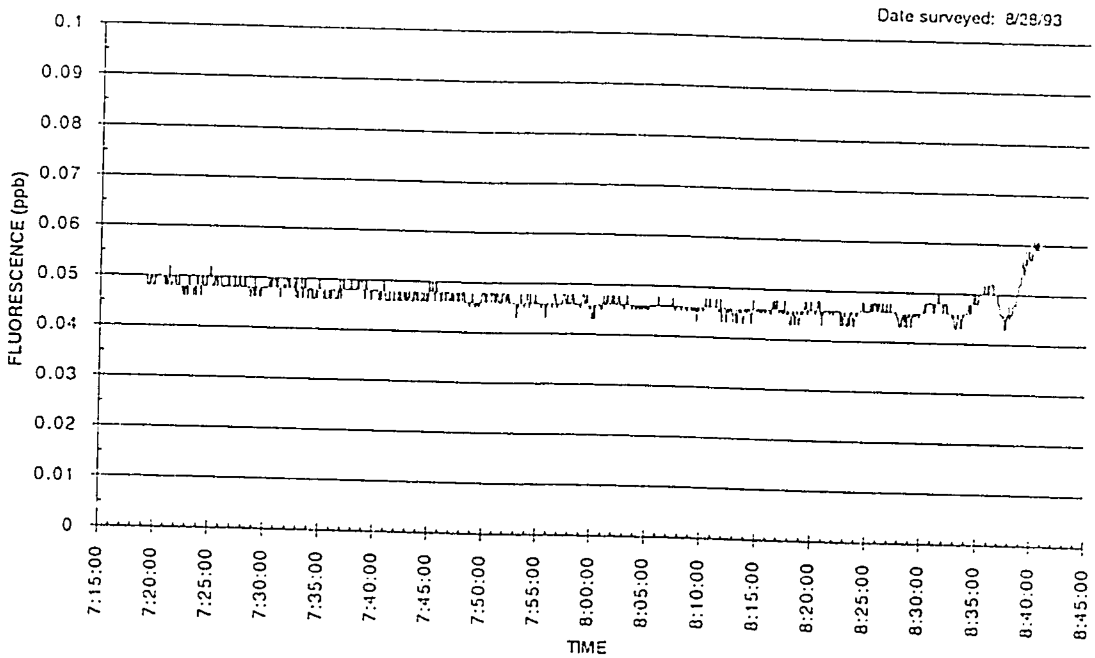


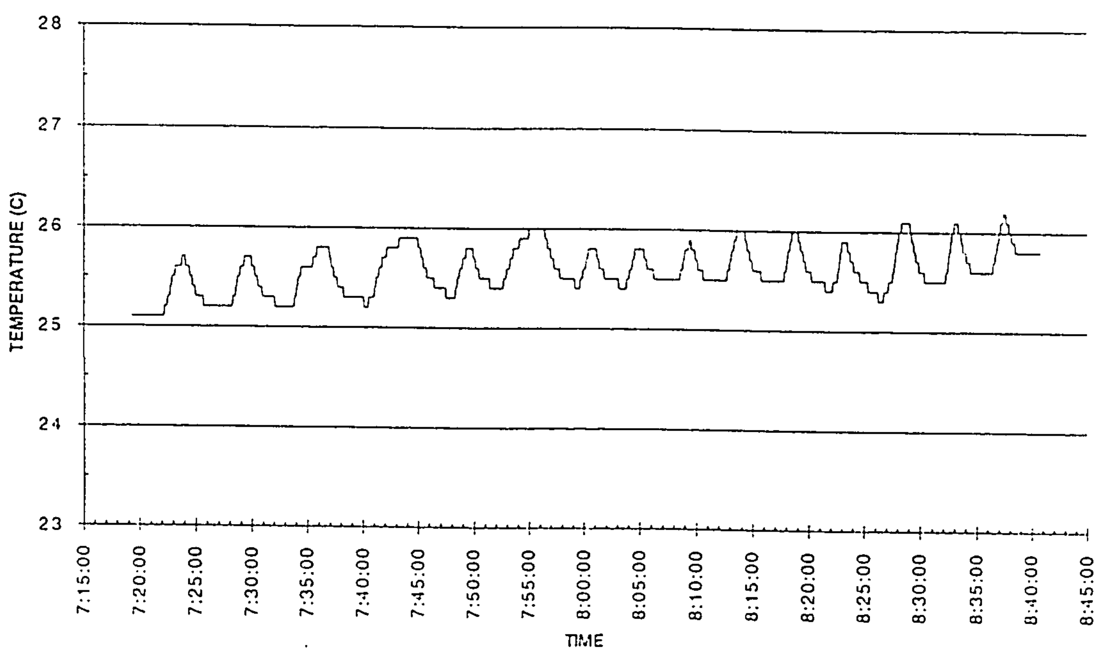
Date surveyed: 8/25/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
7:05:00
7:10:00
7:15:00
7:20:00
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
TIME



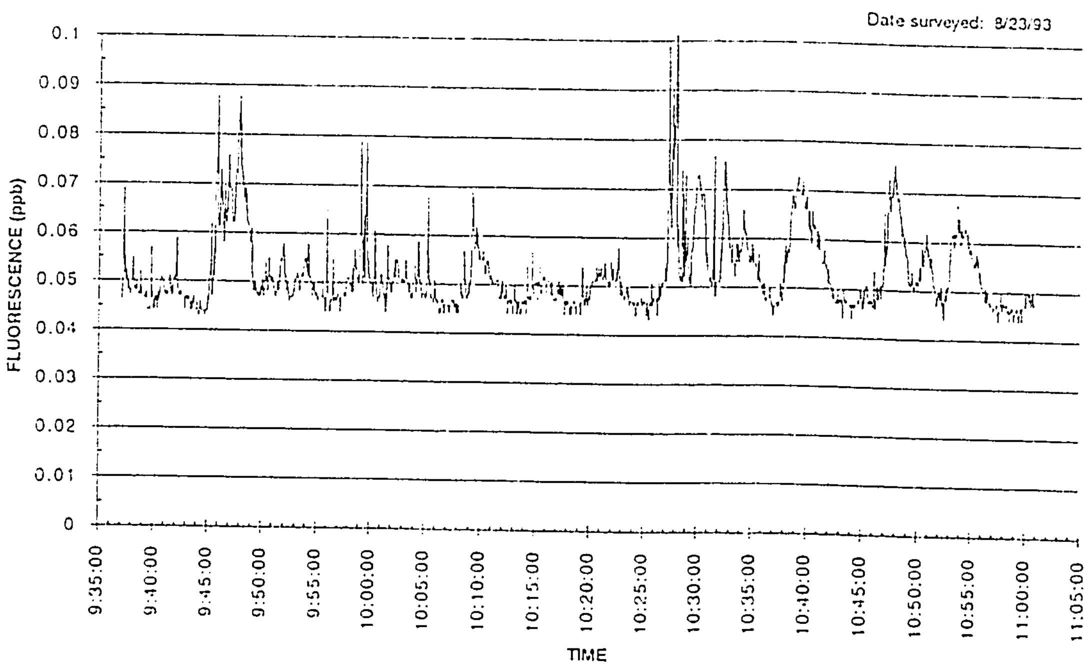
TEMPERATURE (C)
28.000
27.000
26.000
25.000
24.000
23.000
7:05:00
7:10:00
7:15:00
7:20:00
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
TIME

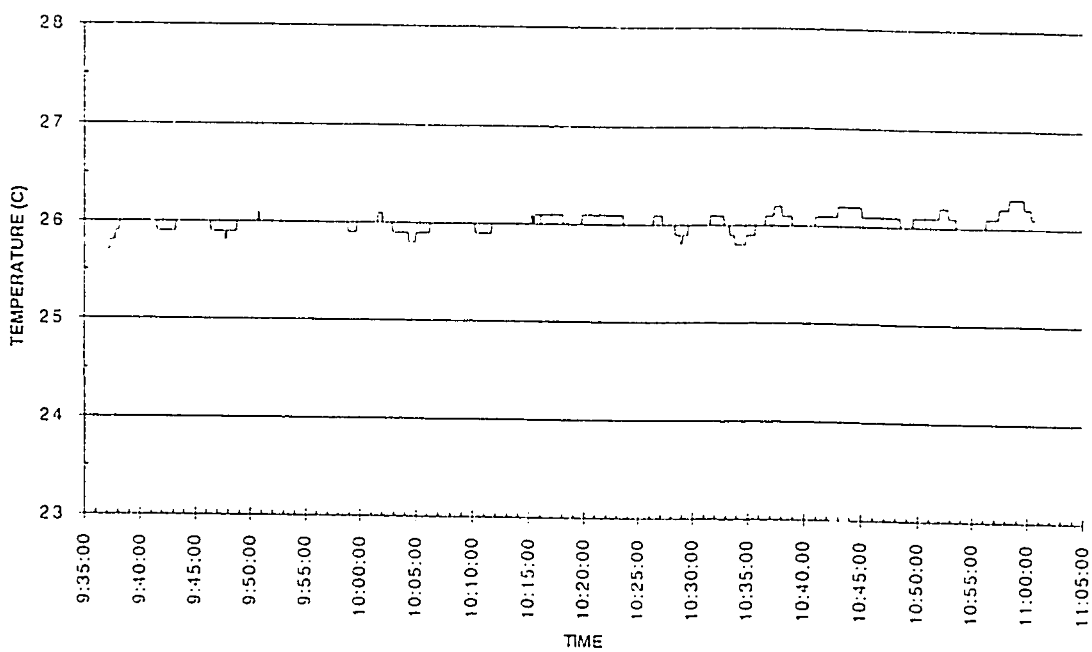
LINE 7A


Date surveyed: 8/28/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
7:15:00
7:20:00
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
8:40:00
8:45:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
7:15:00
7:20:00
7:25:00
7:30:00
7:35:00
7:40:00
7:45:00
7:50:00
7:55:00
8:00:00
8:05:00
8:10:00
8:15:00
8:20:00
8:25:00
8:30:00
8:35:00
8:40:00
8:45:00
TIME

LINE 8


Date surveyed: 8/23/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
9:35:00
9:40:00
9:45:00
9:50:00
9:55:00
10:00:00
10:05:00
10:10:00
10:15:00
10:20:00
10:25:00
10:30:00
10:35:00
10:40:00
10:45:00
10:50:00
10:55:00
11:00:00
11:05:00
TIME


TEMPERATURE (C)
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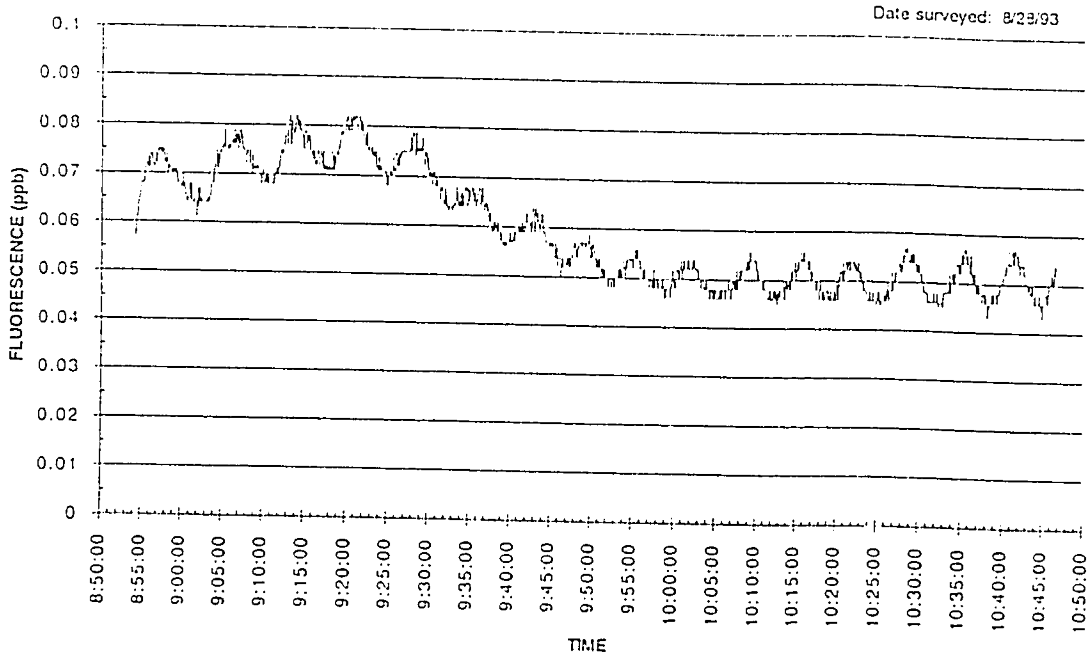
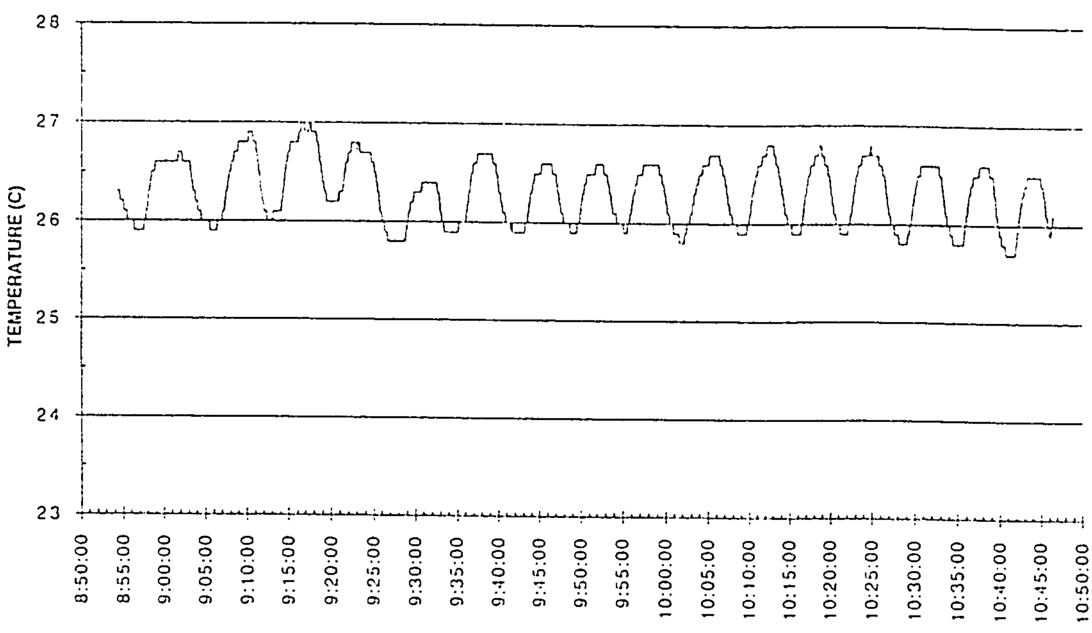
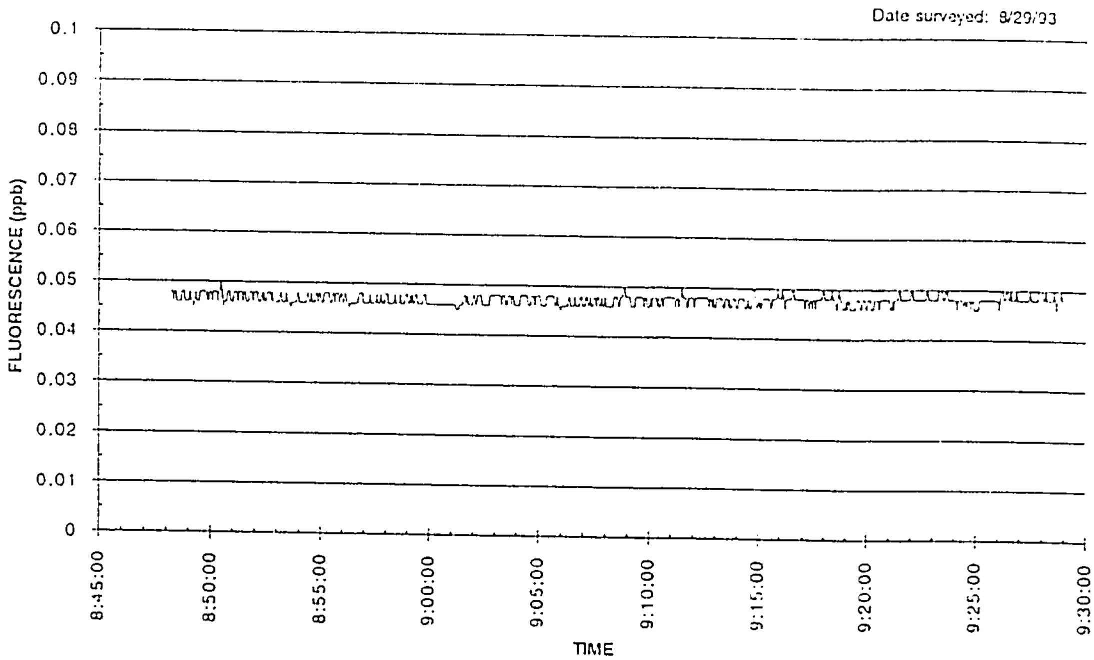
11:05:00
TIME

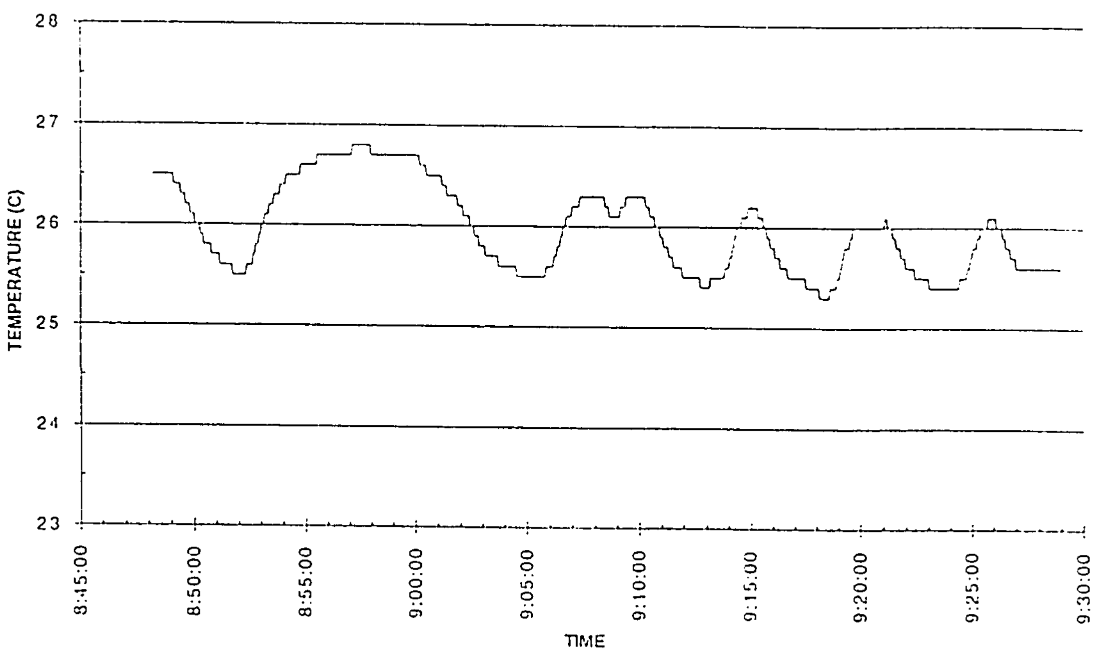
LINE 8A (REVERSE)



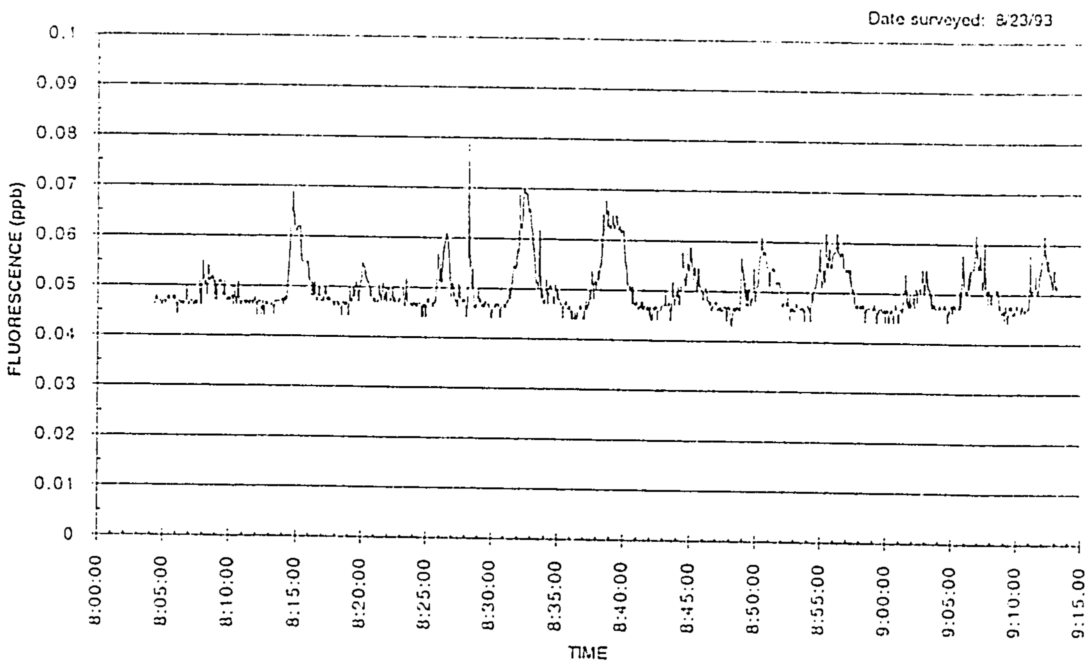


LINE 8A (REPEAT/REVERSE)





LINE 9



Date surveyed: 8/23/93
FLUORESCENCE (ppb)
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0.04
0.03
0.02
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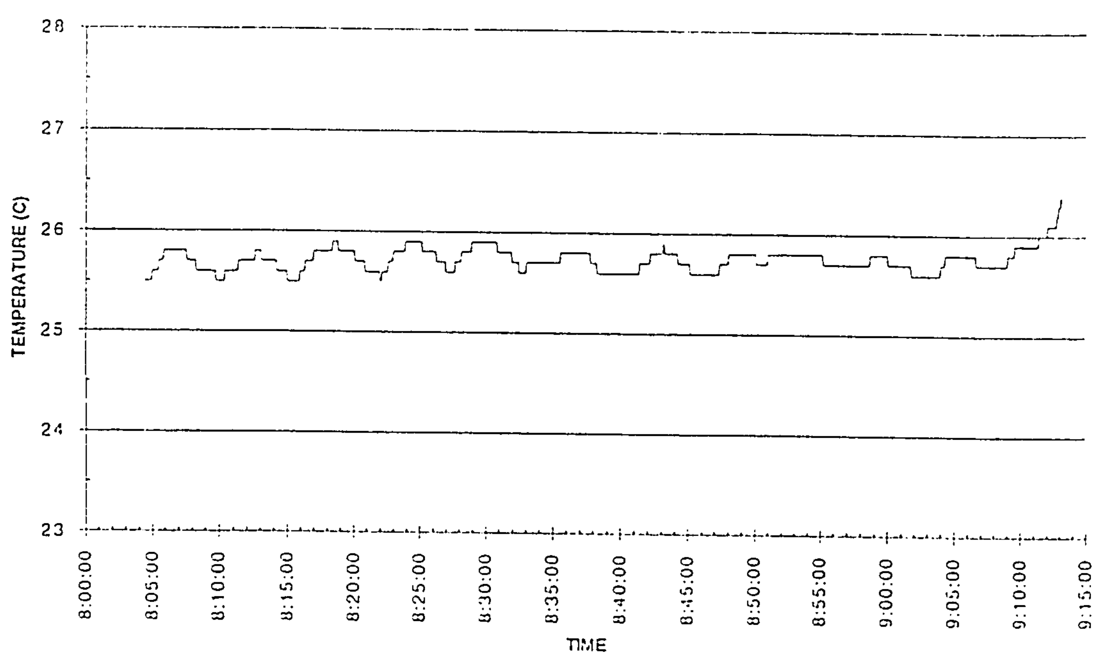
TIME



TEMPERATURE (C)
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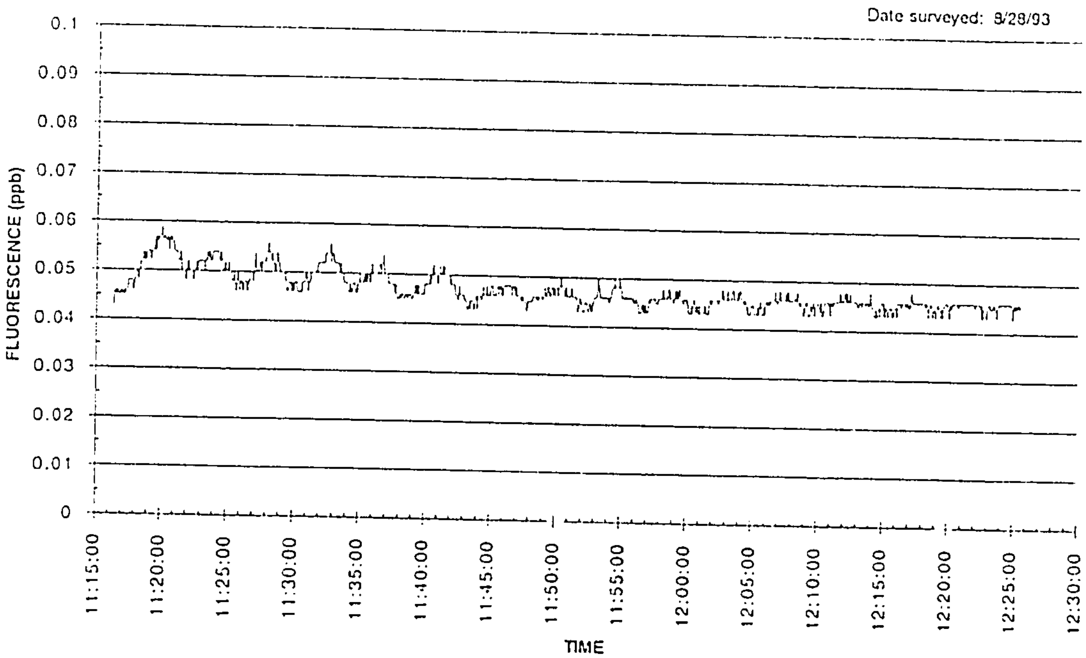
TIME

LINE 9A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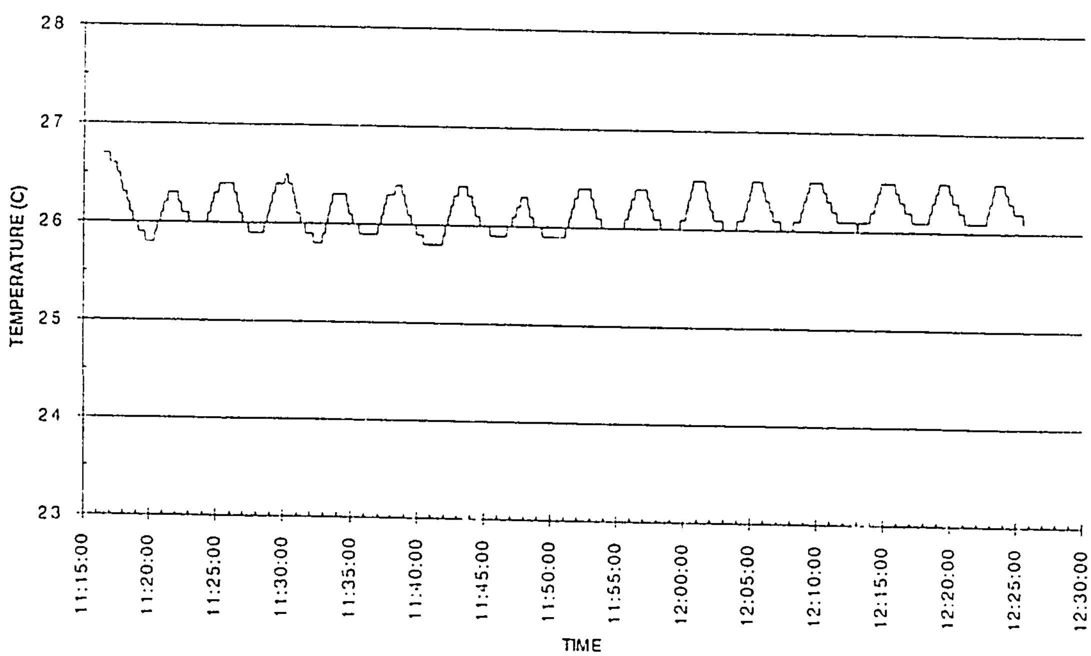
TIME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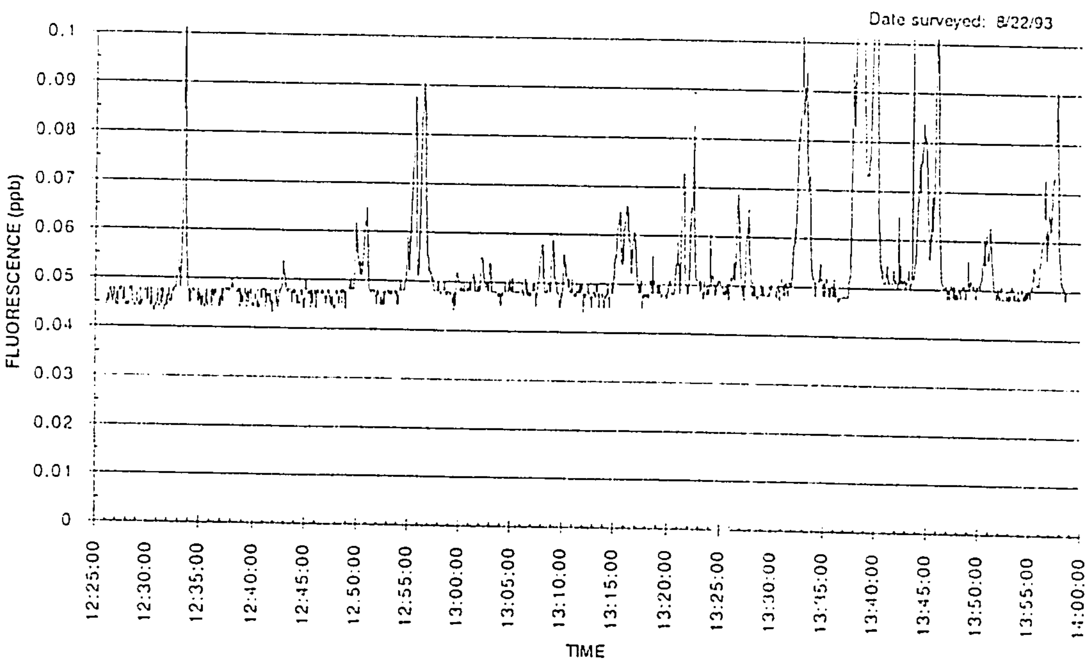
TIME

LINE 10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FLUORESCENCE (ppb)
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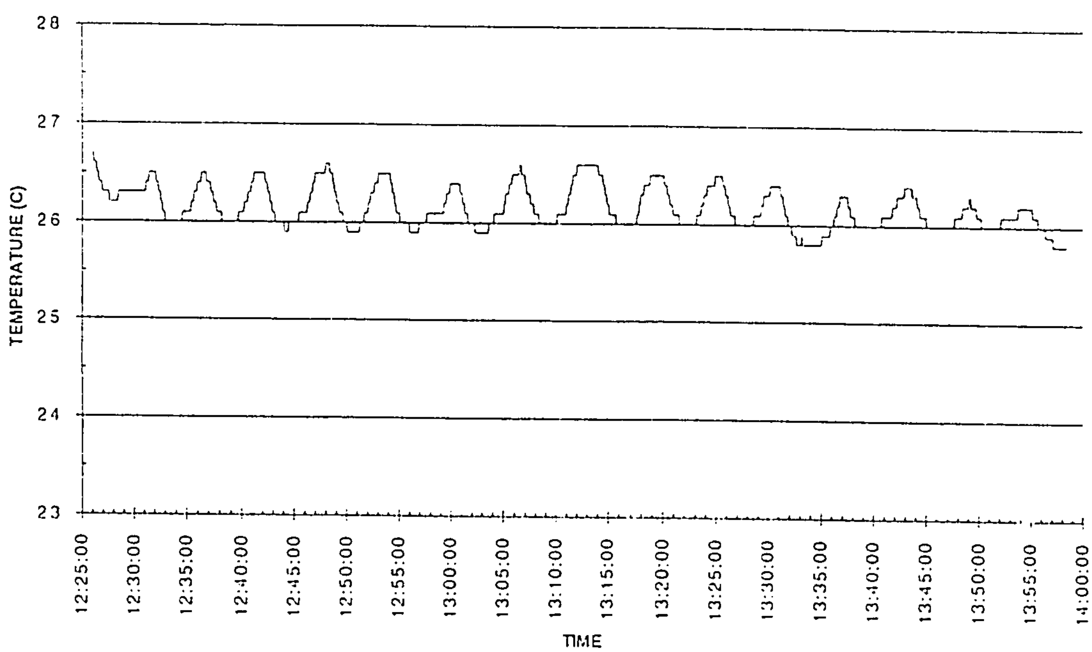
TIME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TEMPERATURE (C)
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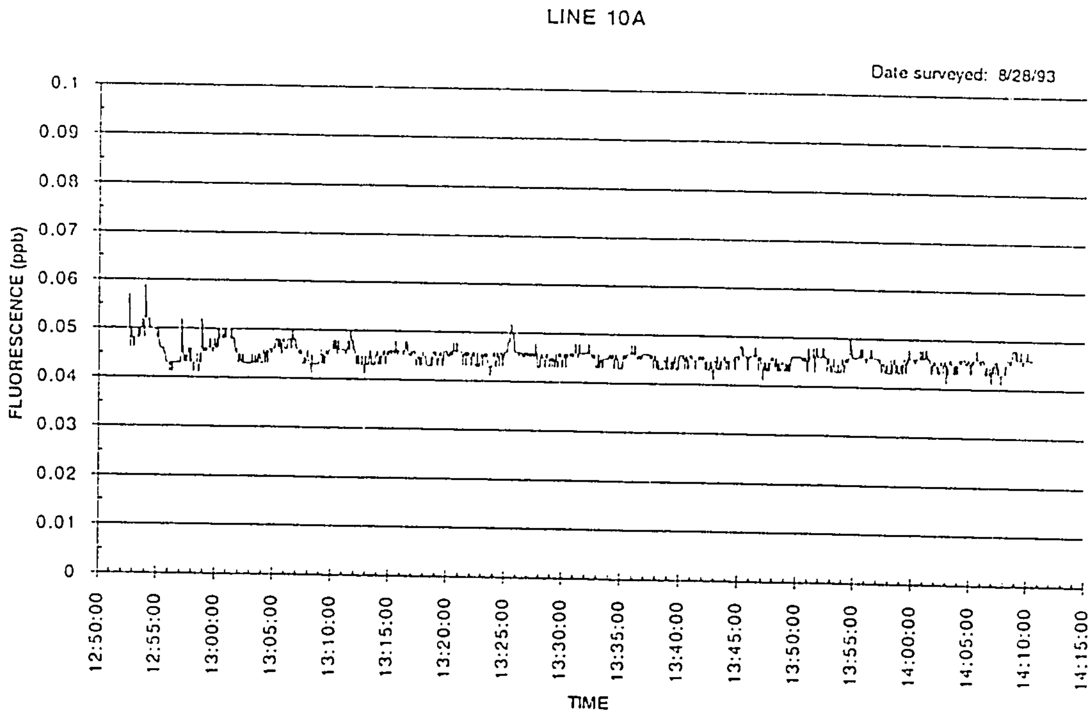
TIME


LINE 10A
Date surveyed: 8/28/93
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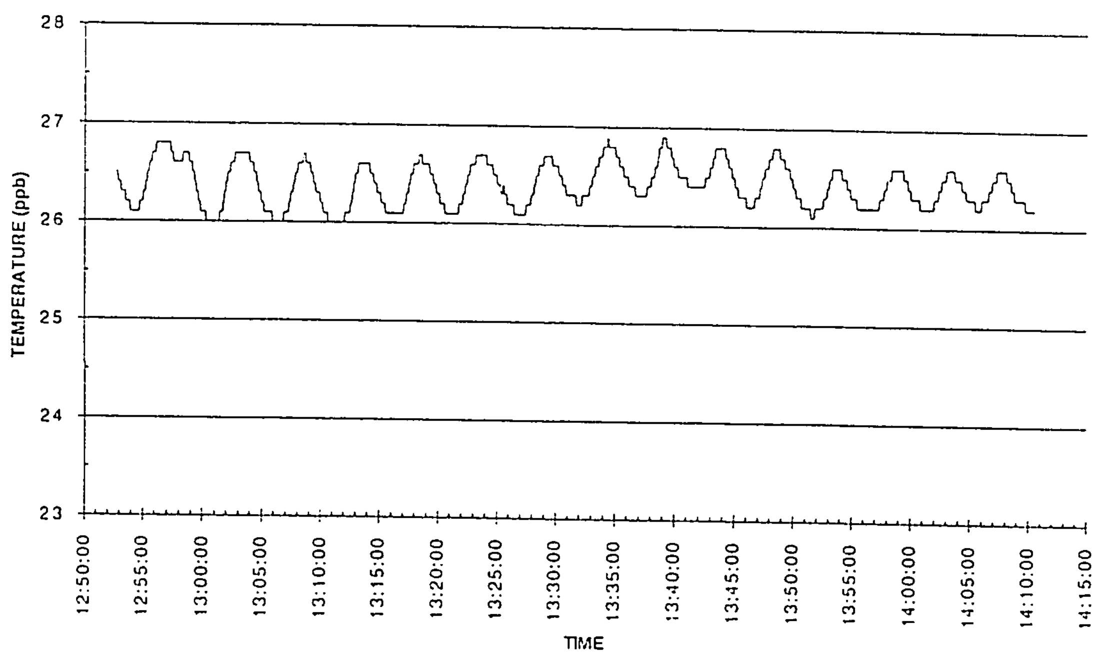
TIME


TEMPERATURE (ppb)
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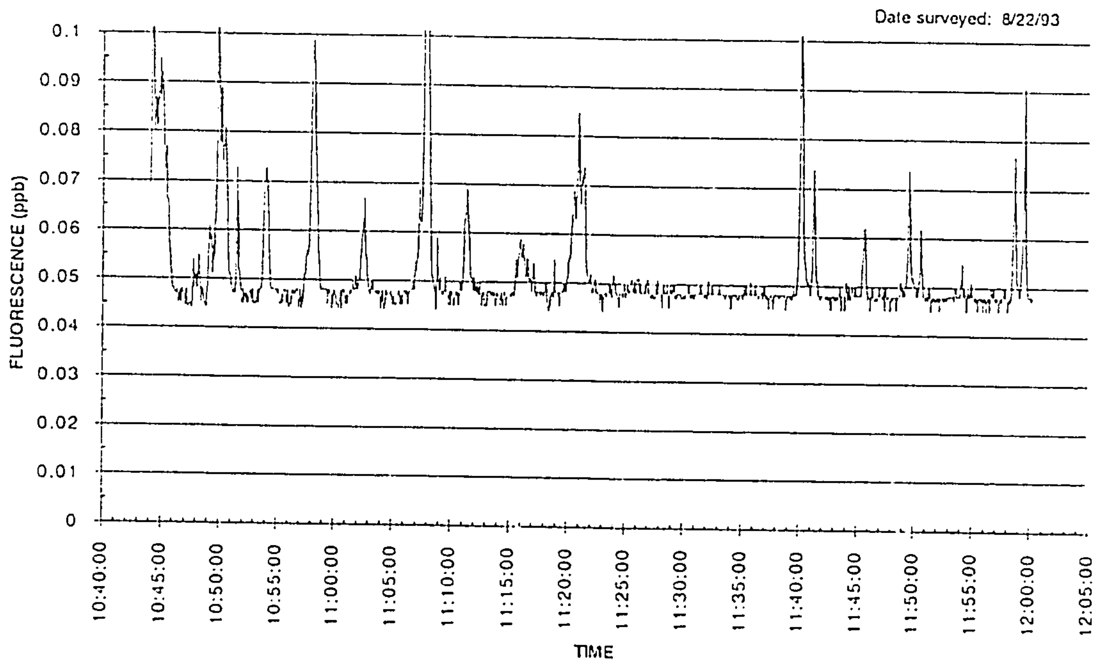
TIME

LINE 11


Date surveyed: 8/22/93
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FLUORESCENCE (ppb)
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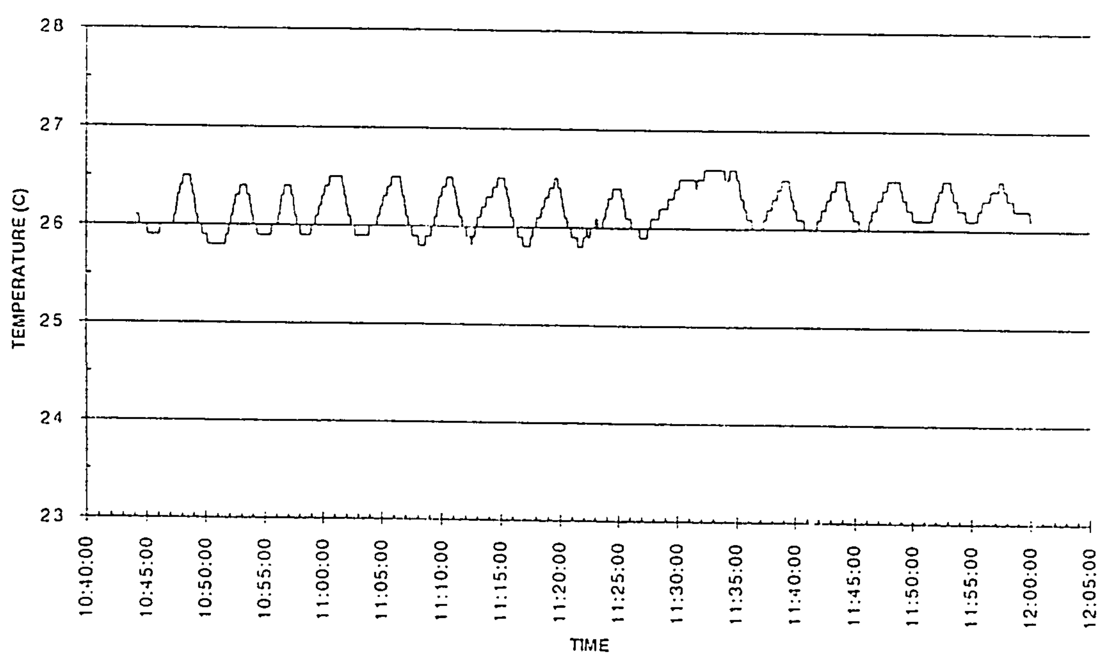
TIME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TEMPERATURE (C)
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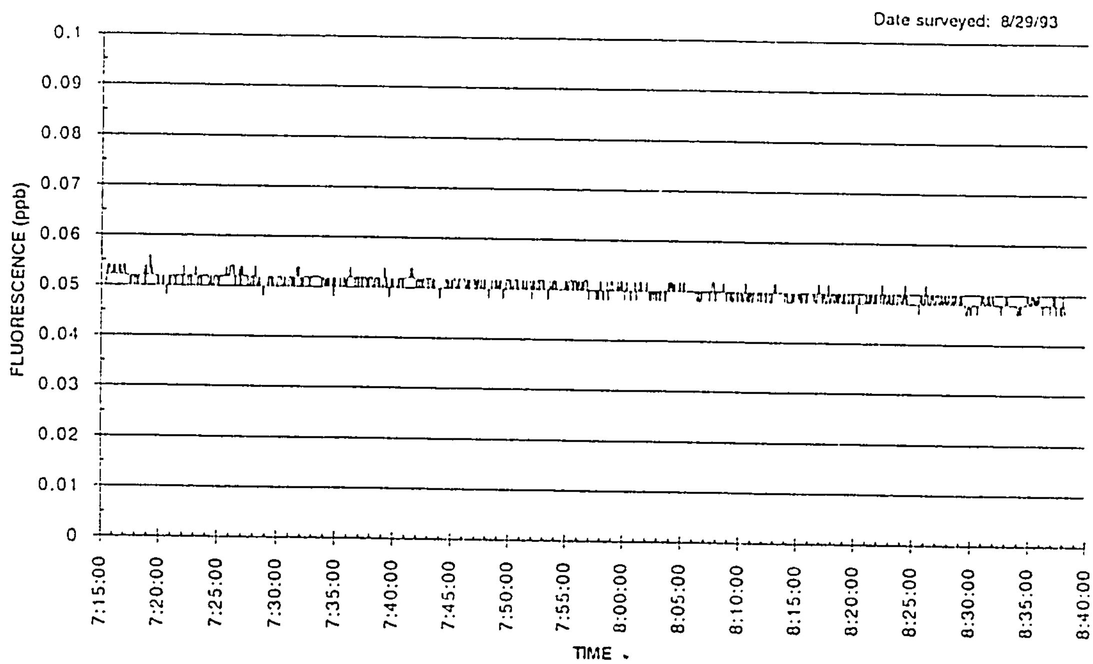
TIME

LINE 11A



Date surveyed: 8/29/93
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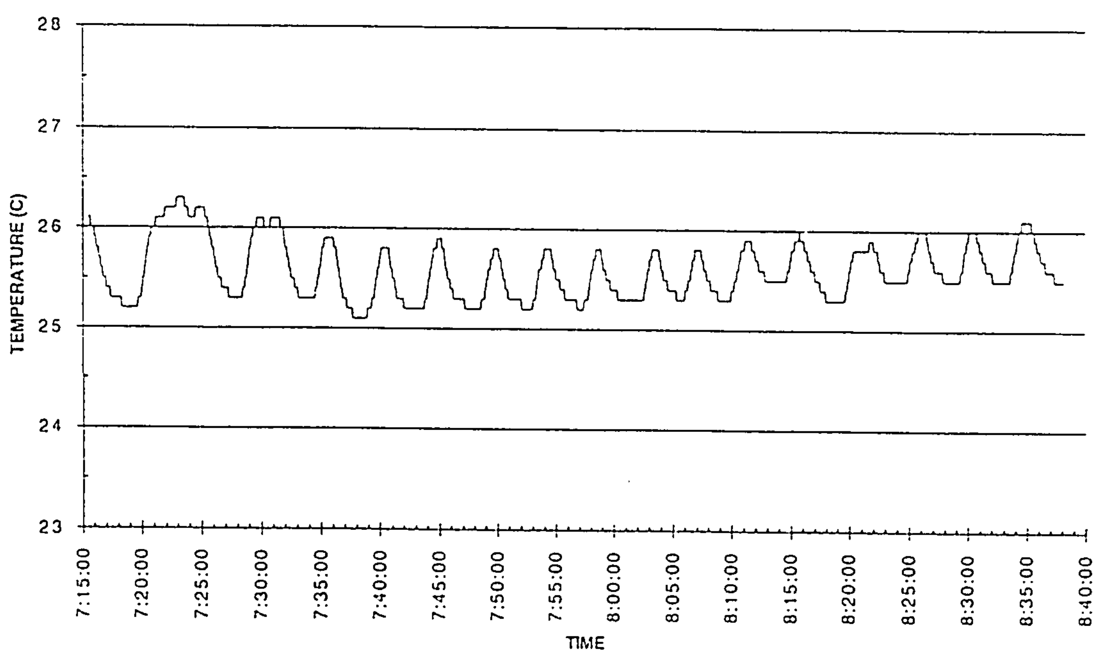

TEMPERATURE (C)
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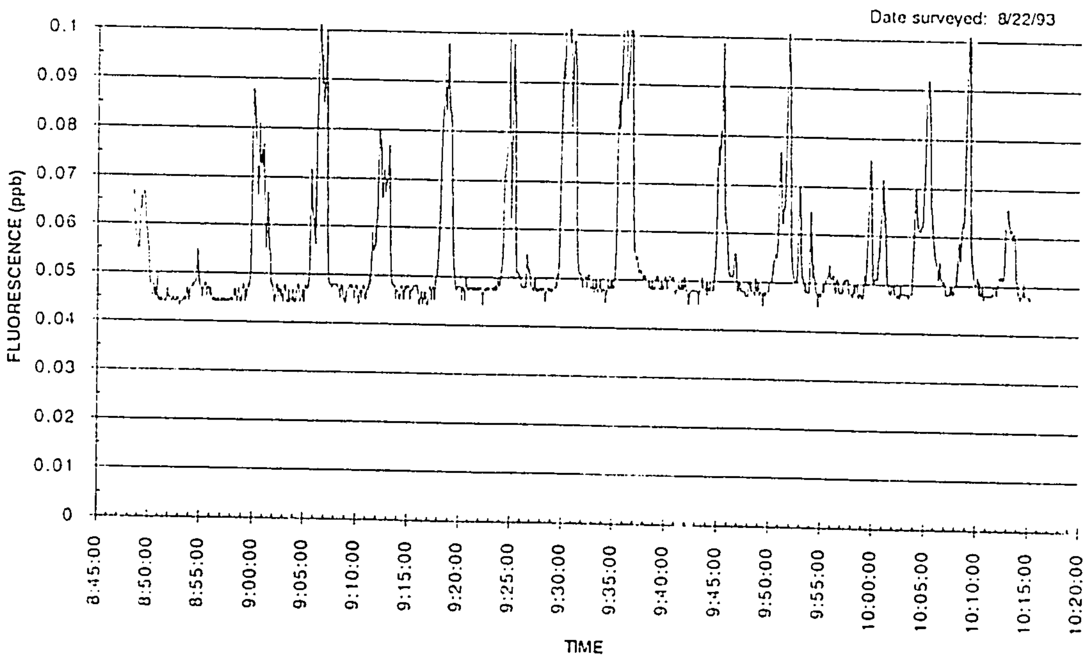
TIME

LINE 12



Date surveyed: 8/22/93
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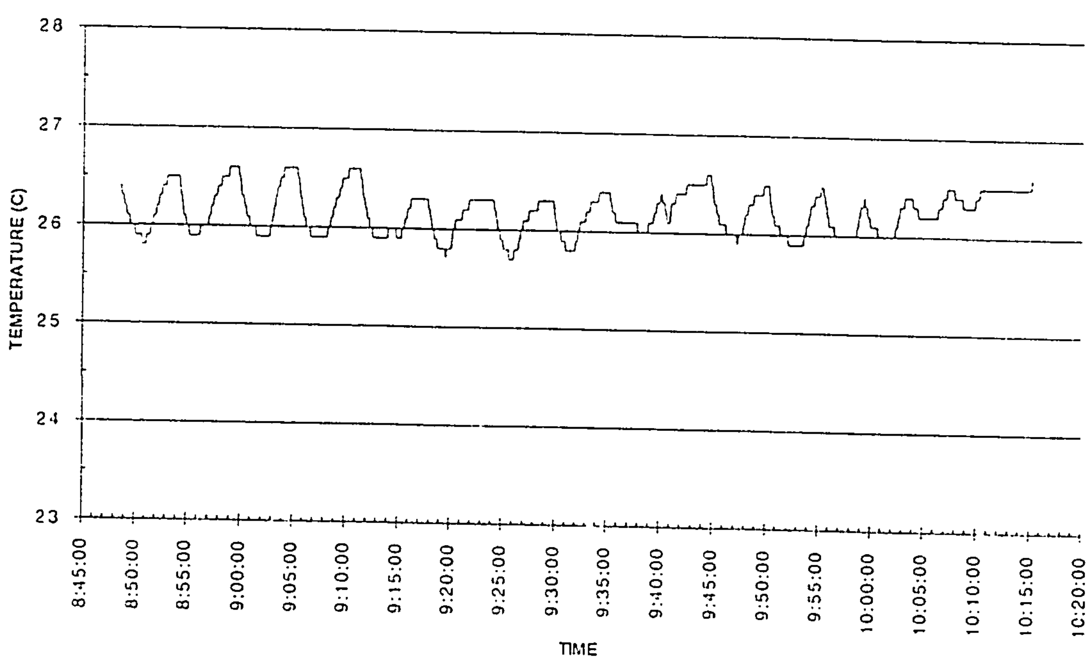

TEMPERATURE (C)
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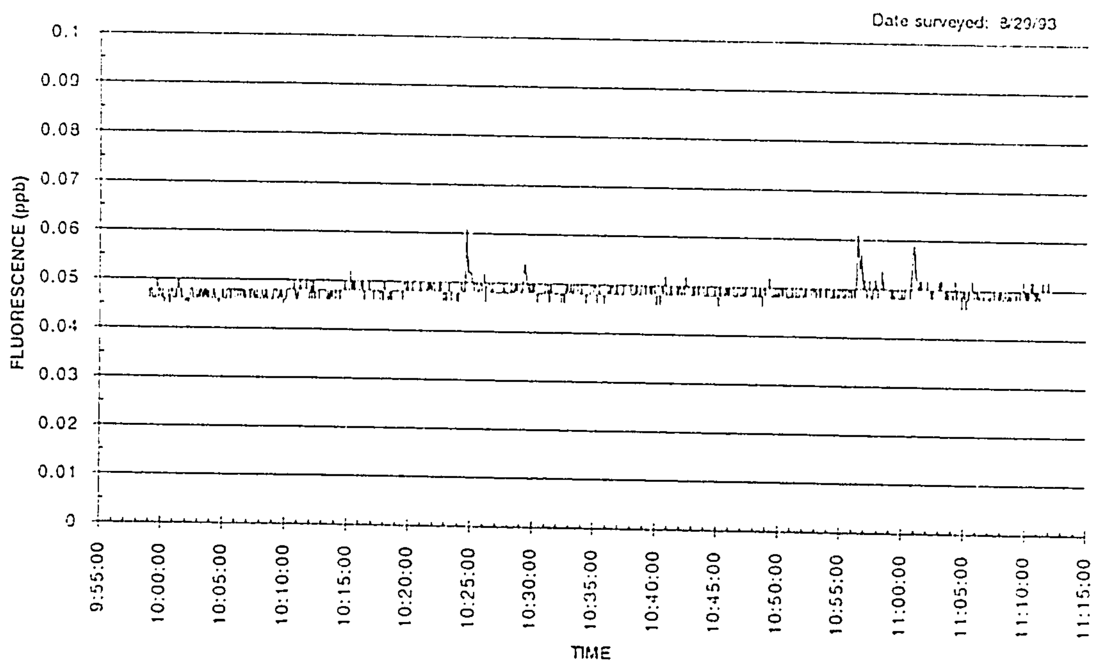
TIME

LINE 12A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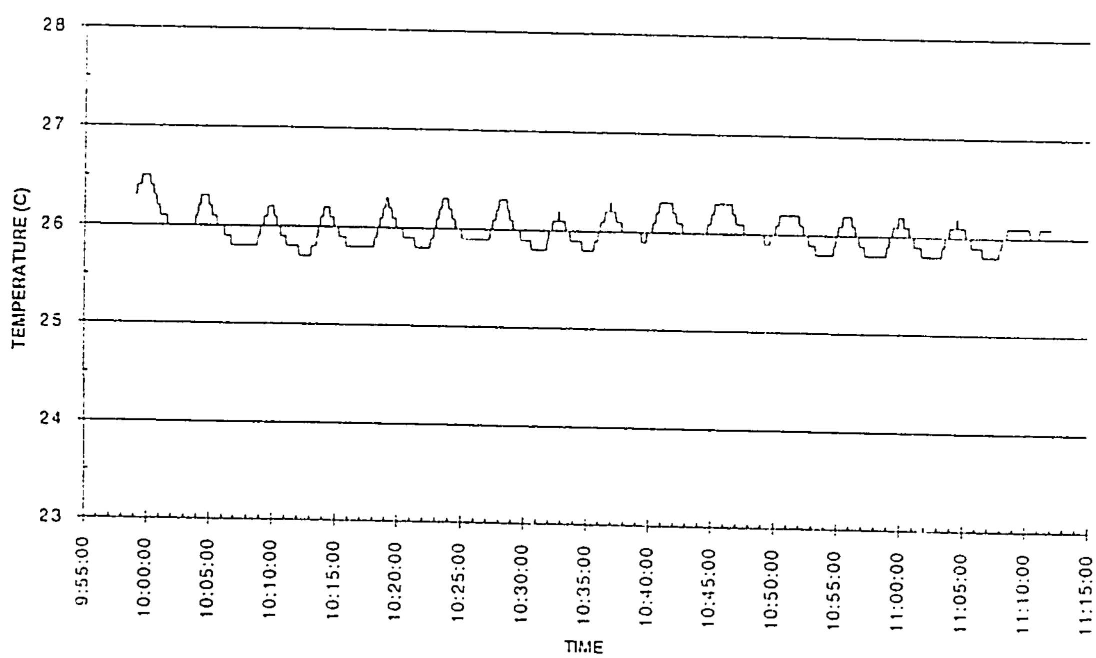
TEMPERATURE (C)
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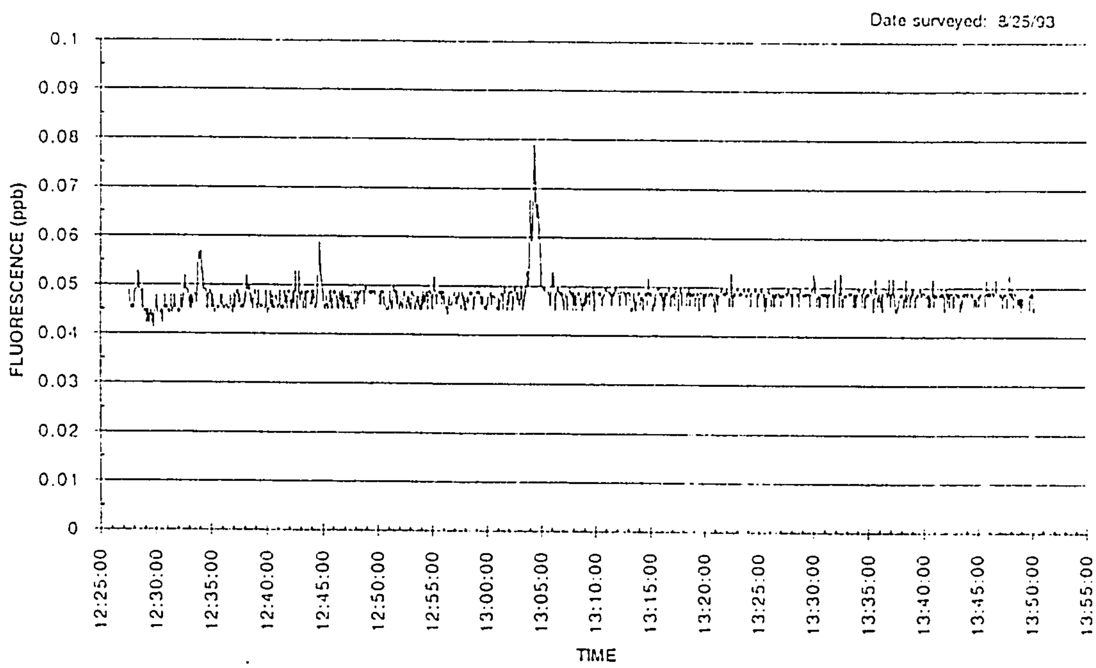
TIME

LINE 13



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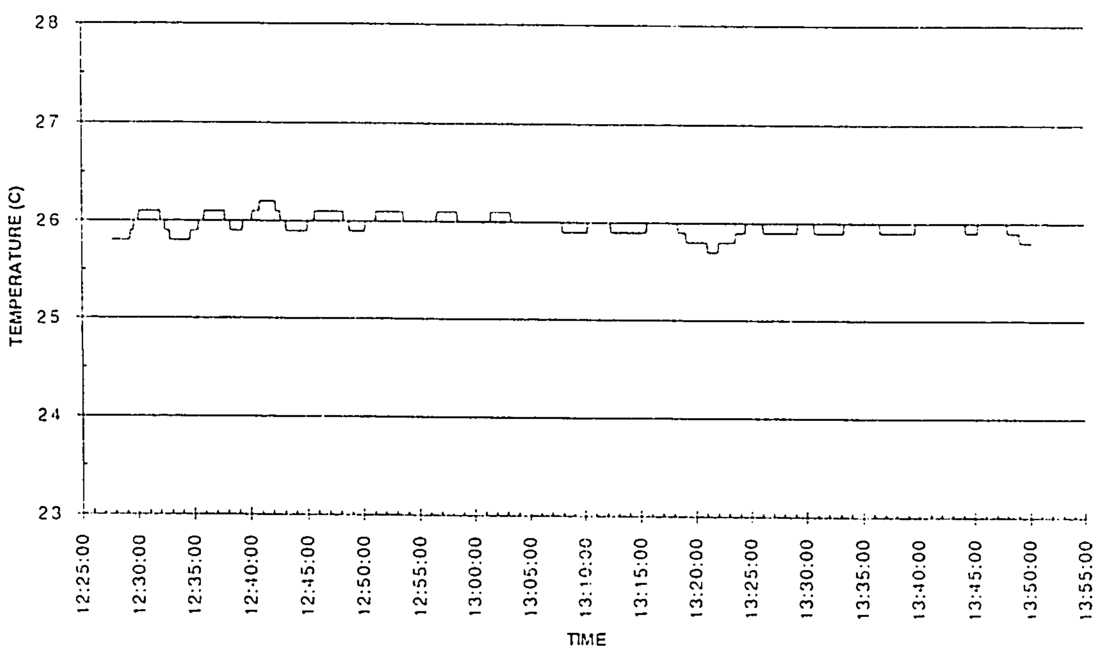
TIME



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TEMPERATURE (C)
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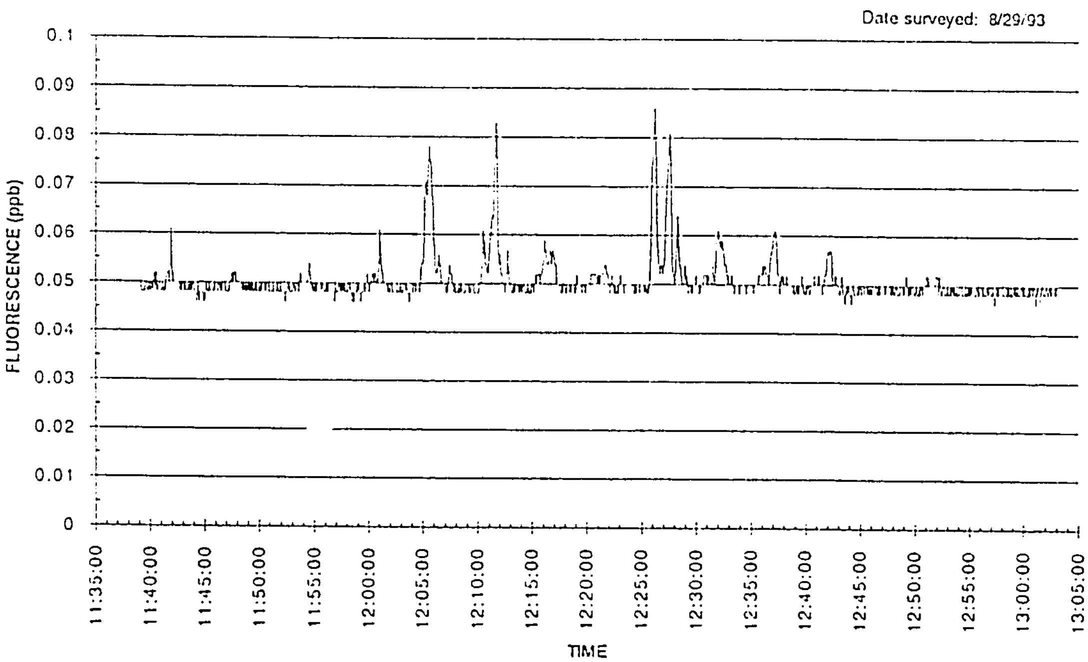
TIME

LINE 13A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FLUORESCENCE (ppb)
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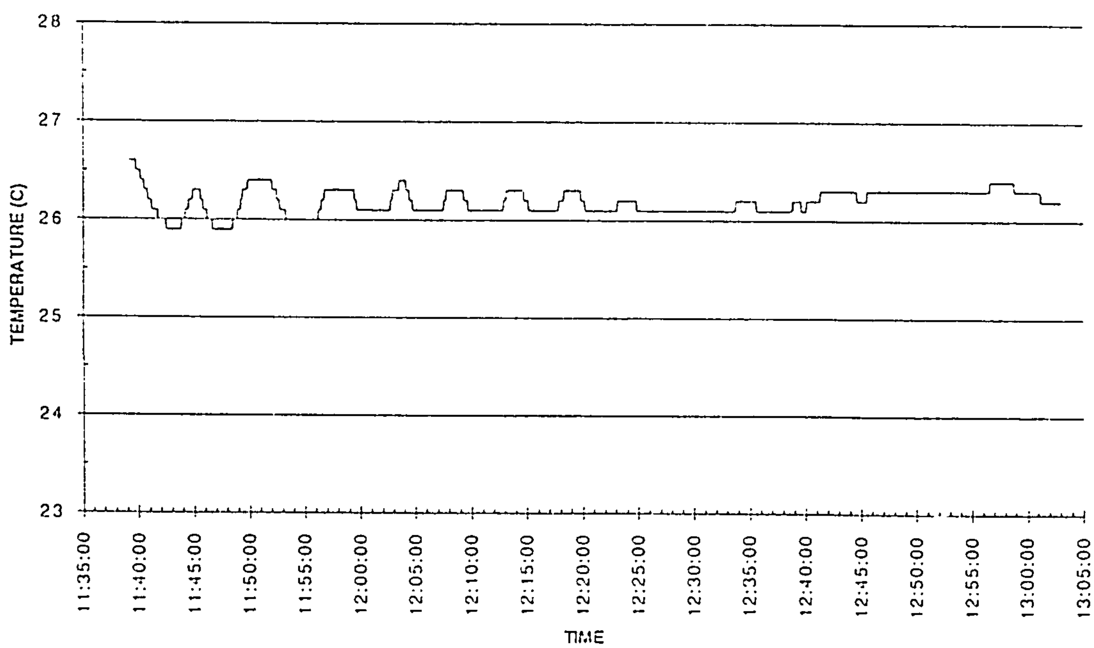
TIME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TEMPERATURE (C)
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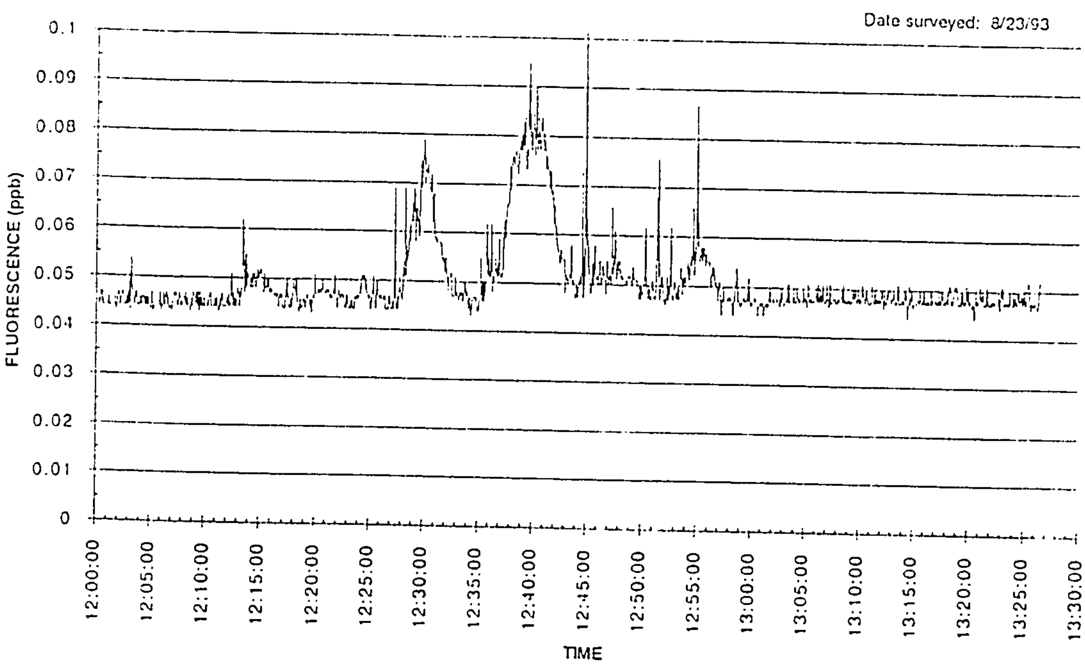
TIME

LINE 14


Date surveyed: 8/23/93
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0.02
0.01
0
FLUORESCENCE (ppb)
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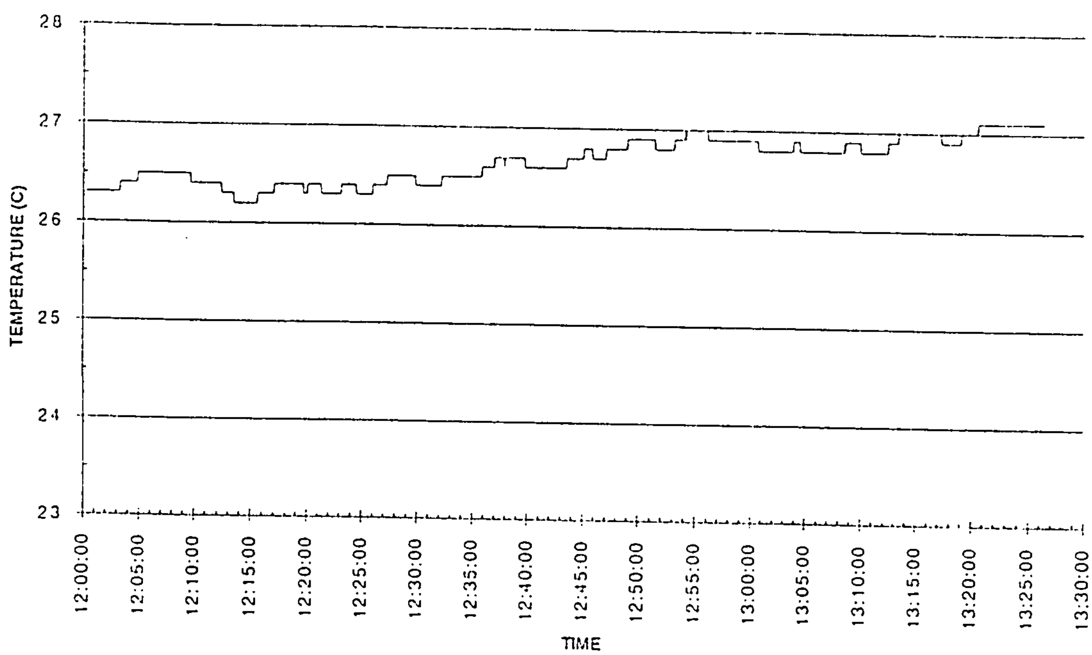
TIME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TEMPERATURE (C)
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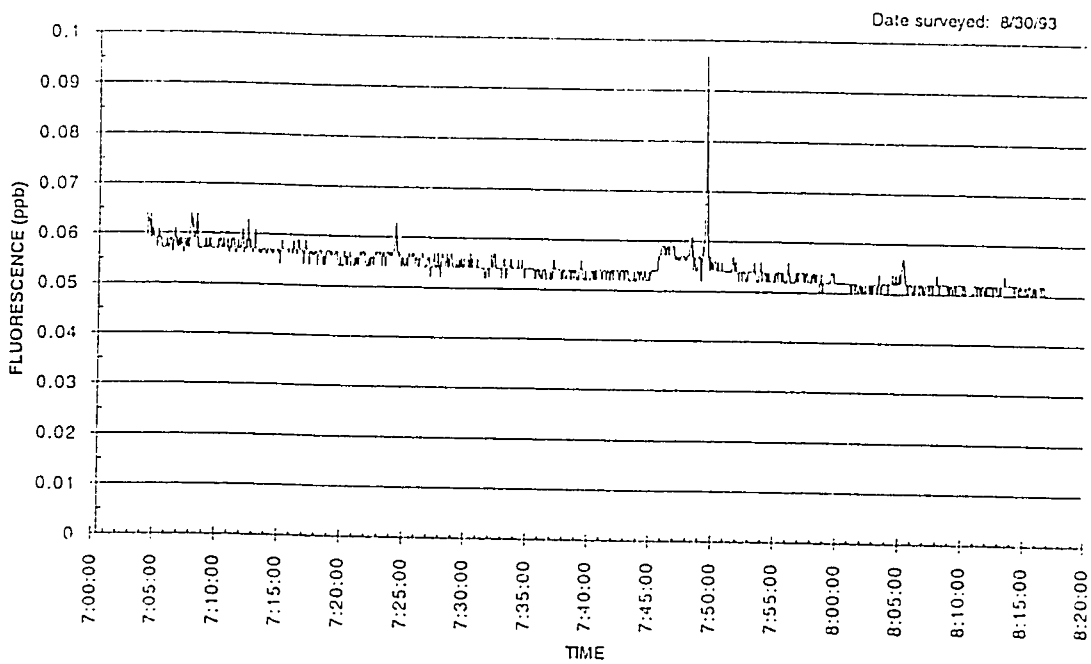
TIME

LINE 14A


Date surveyed: 8/30/93
FLUORESCENCE (ppb)
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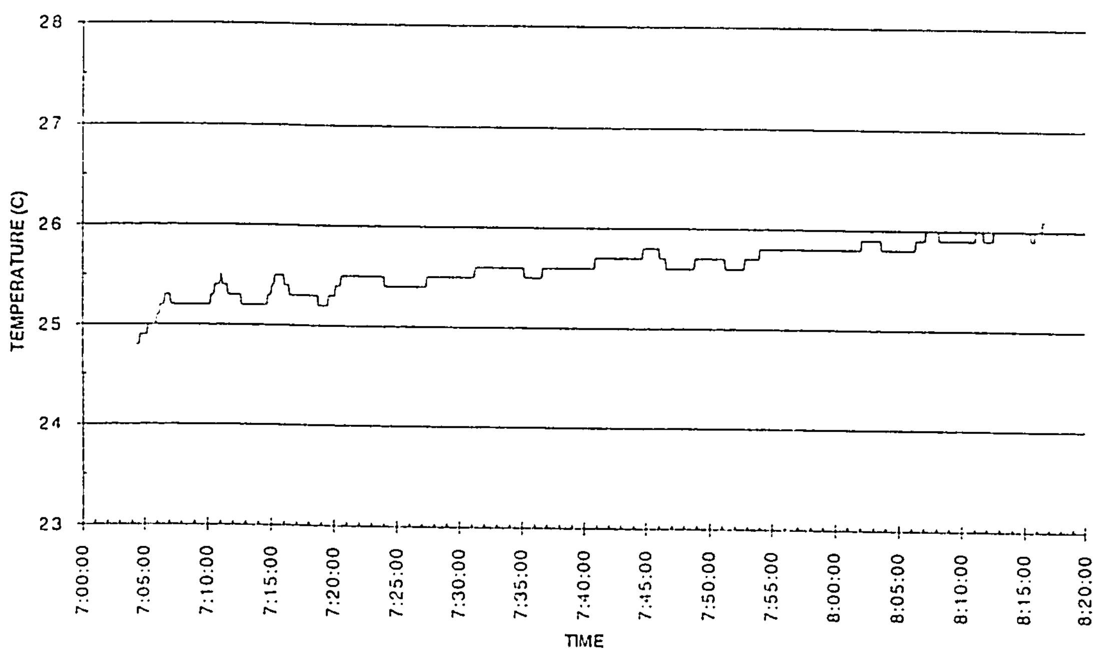
TIME


TEMPERATURE (C)
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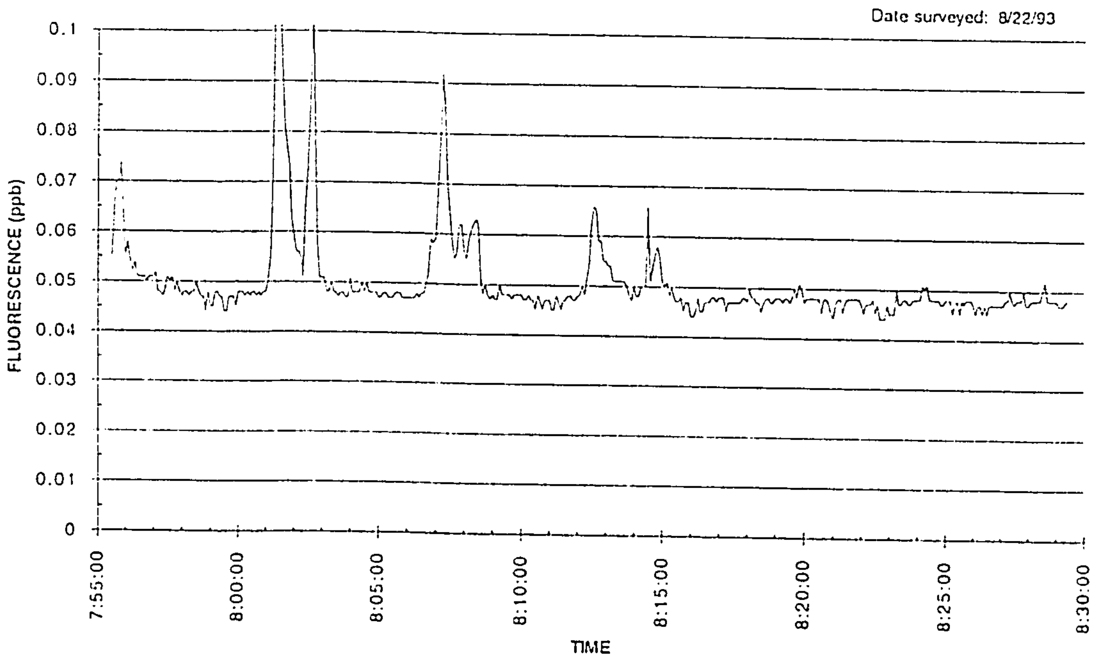
TIME

LINE 15


Date surveyed: 8/22/93
FLUORESCENCE (ppb)
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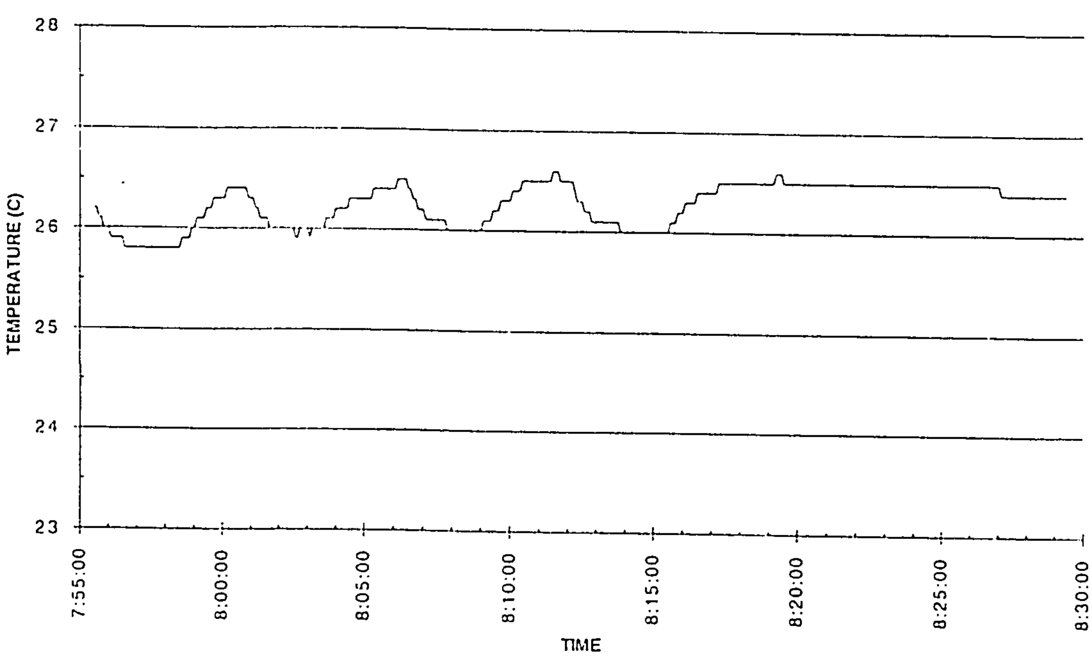
TIME


TEMPERATURE (C)
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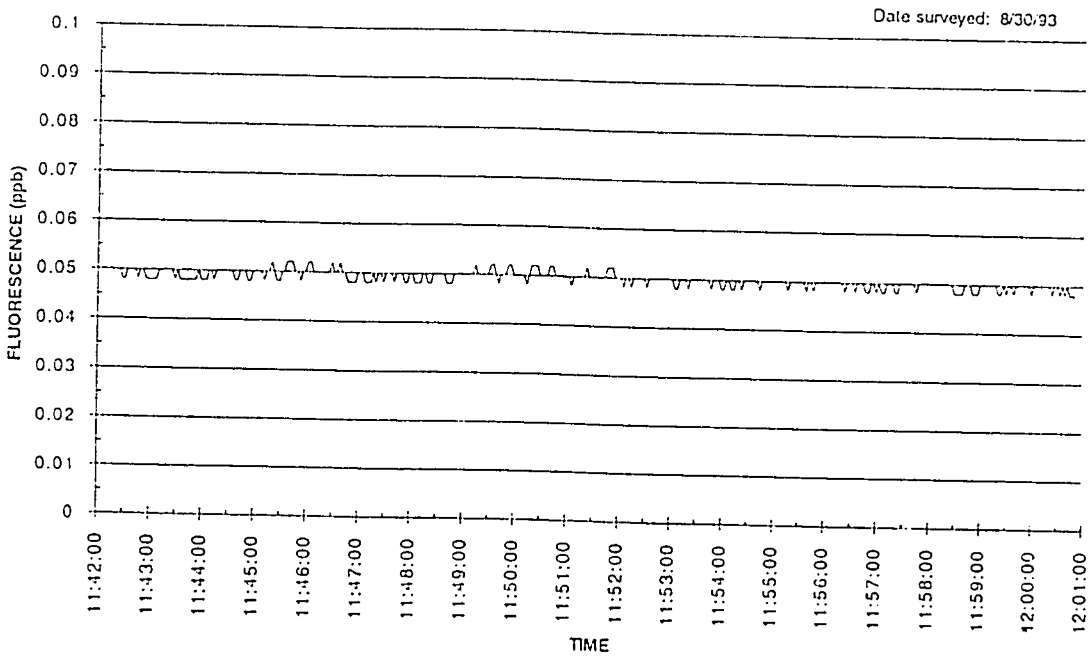
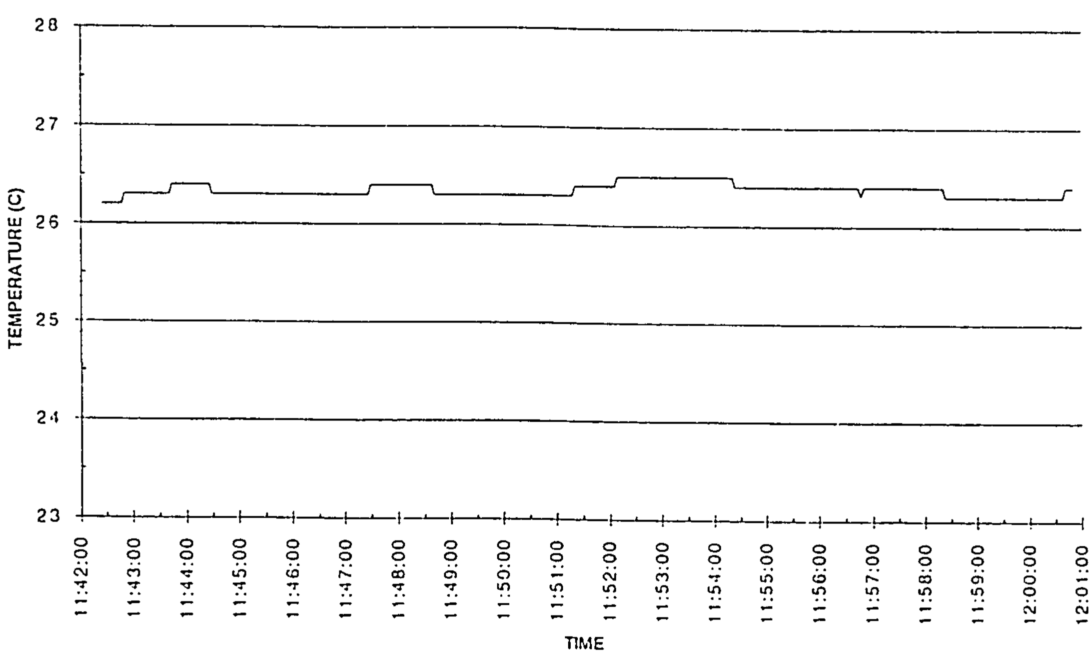
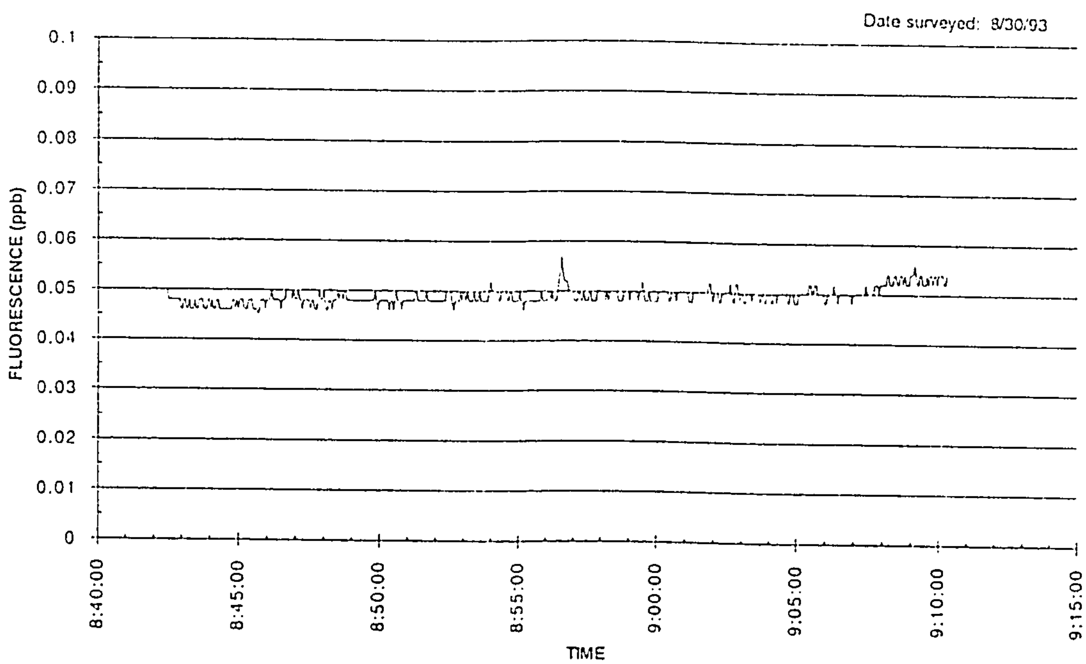
8:30:00
TIME

LINE 15/REPEAT

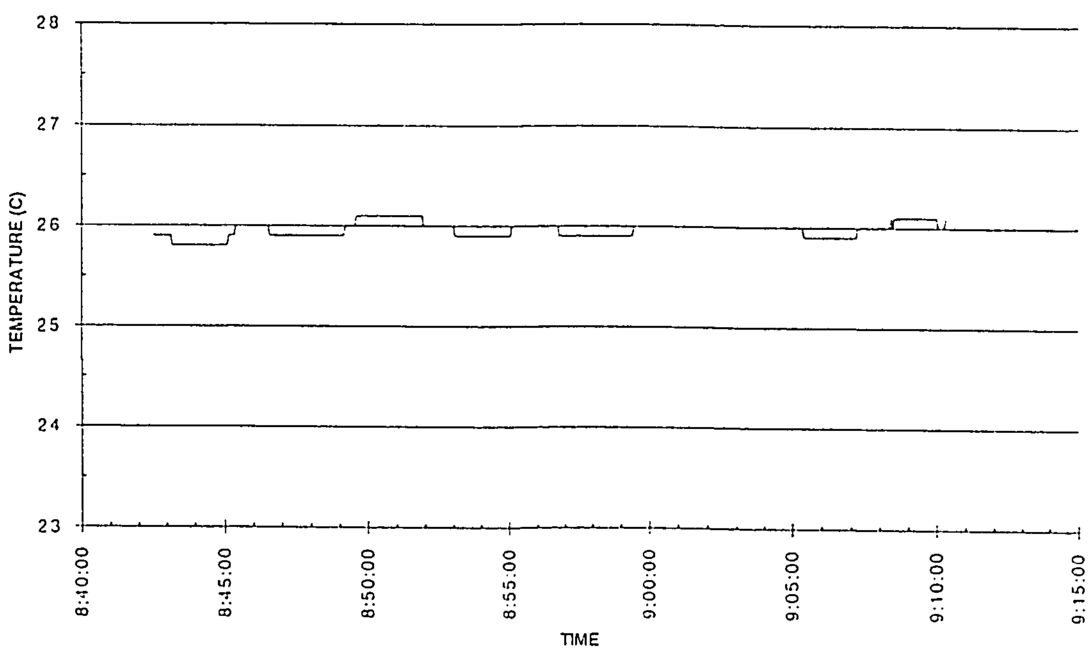




LINE 15A

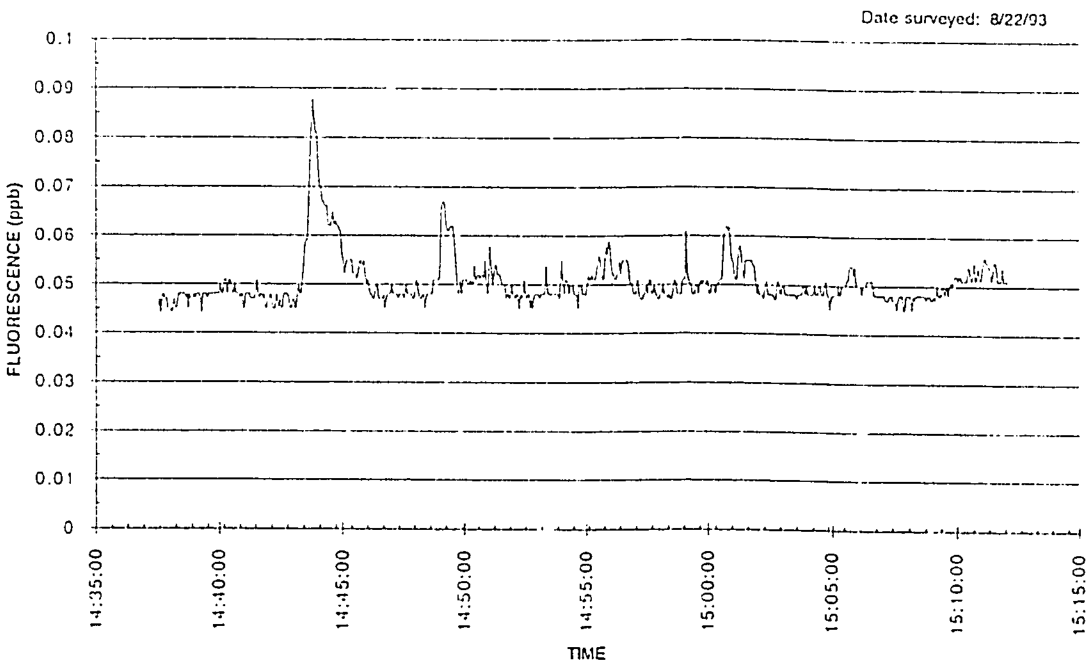


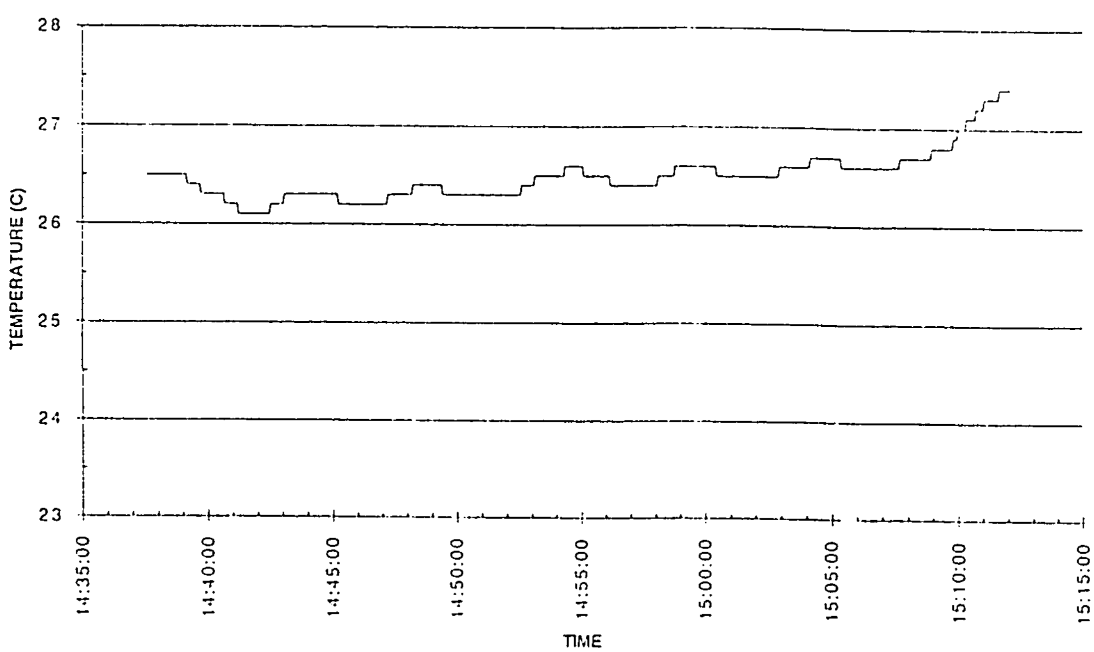
Date surveyed: 8/30/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
8:40:00
8:45:00
8:50:00
8:55:00
9:00:00
9:05:00
9:10:00
9:15:00
TIME



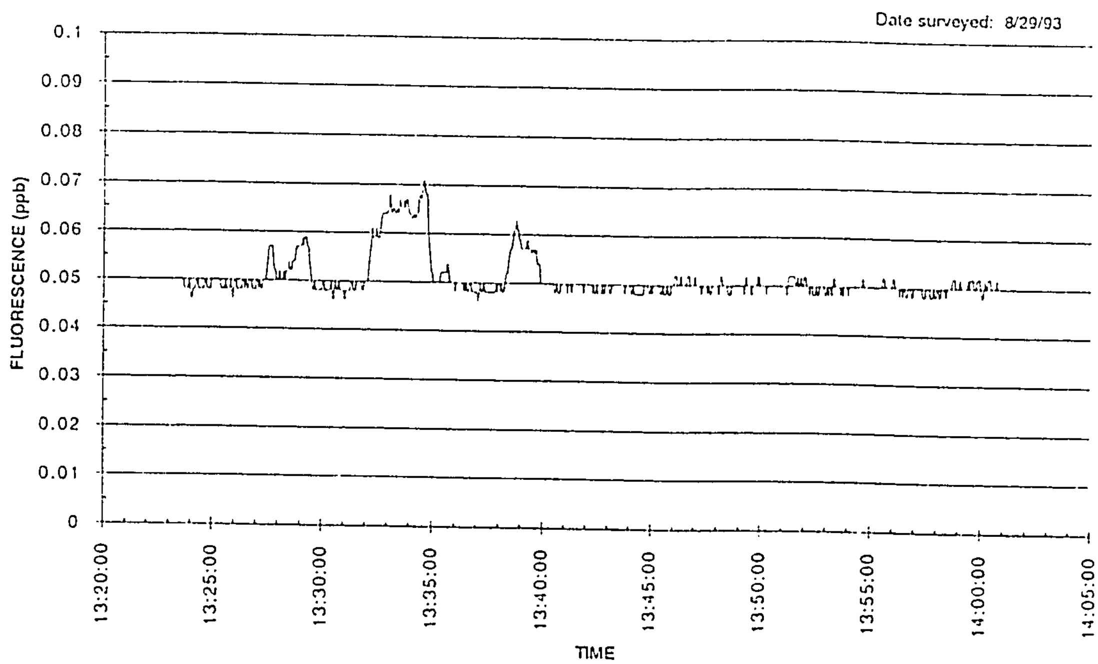
TEMPERATURE (C)
28
27
26
25
24
23
8:40:00
8:45:00
8:50:00
8:55:00
9:00:00
9:05:00
9:10:00
9:15:00
TIME

LINE 16

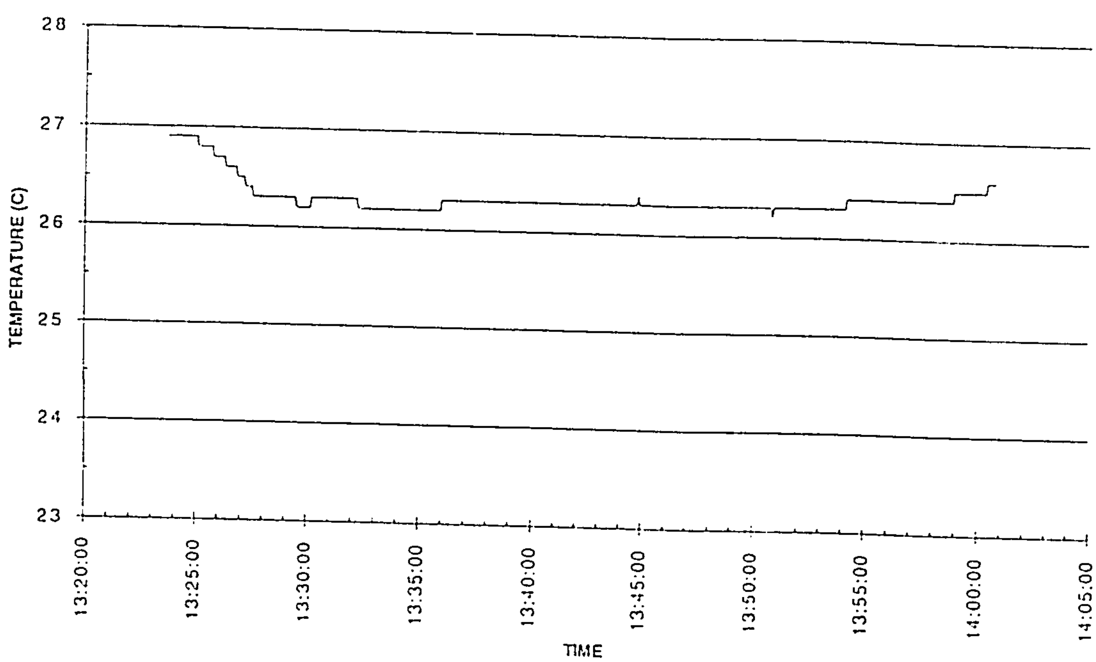

Date surveyed: 8/22/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
14:35:00
14:40:00
14:45:00
14:50:00
14:55:00
15:00:00
15:05:00
15:10:00
15:15:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
14:35:00
14:40:00
14:45:00
14:50:00
14:55:00
15:00:00
15:05:00
15:10:00
15:15:00
TIME

LINE 16A

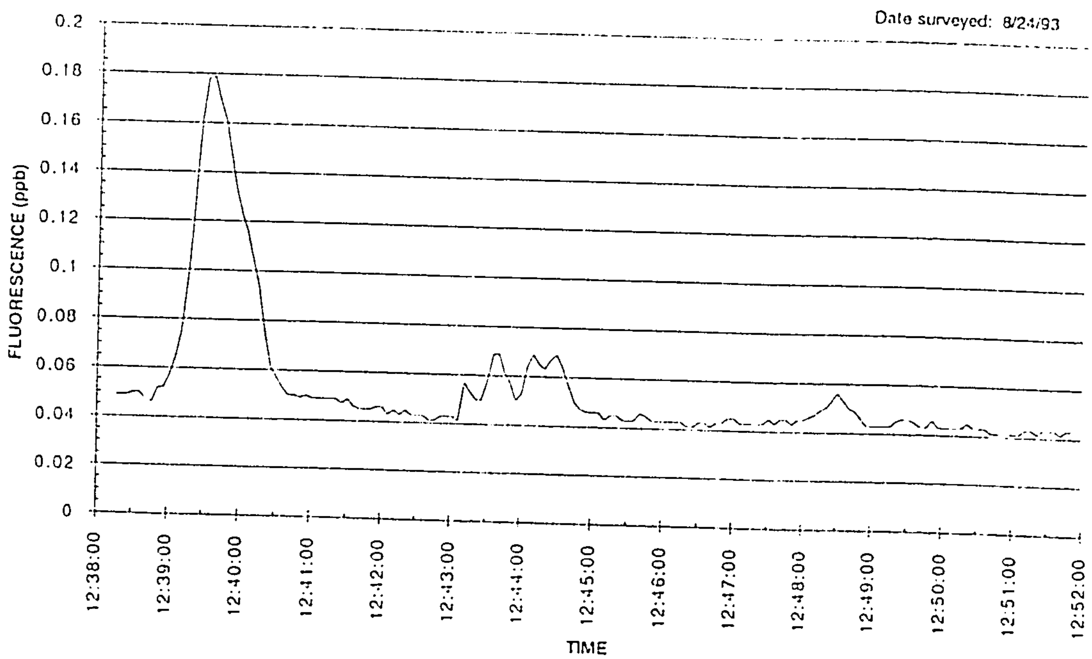


Date surveyed: 8/29/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
13:20:00
13:25:00
13:30:00
13:35:00
13:40:00
13:45:00
13:50:00
13:55:00
14:00:00
14:05:00
TIME

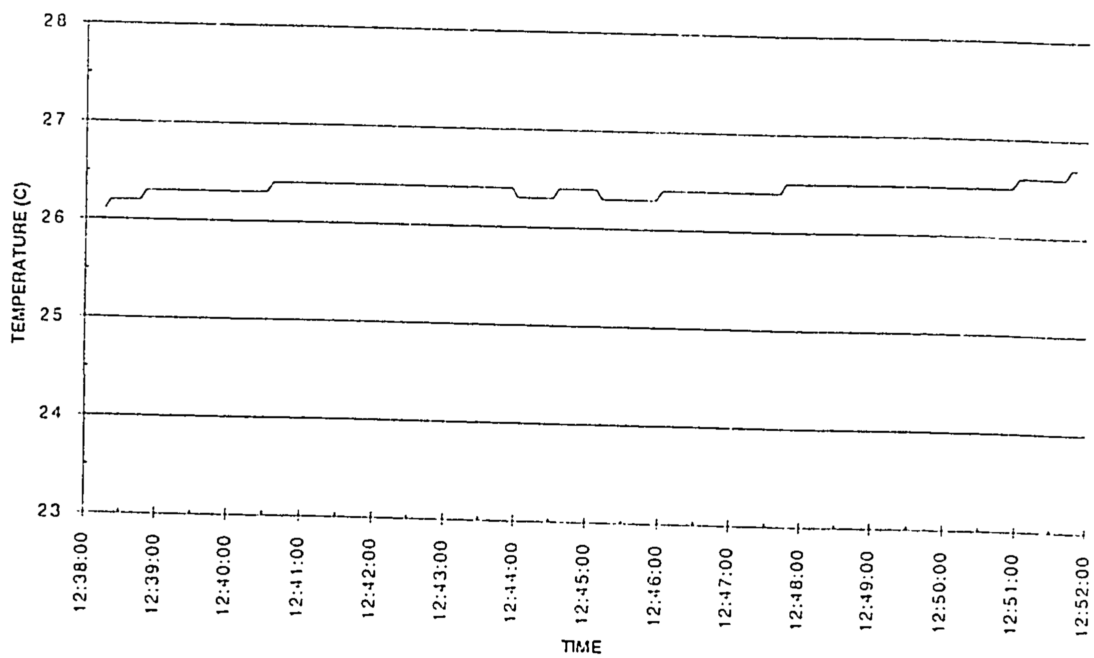


TEMPERATURE (C)
28
27
26
25
24
23
13:20:00
13:25:00
13:30:00
13:35:00
13:40:00
13:45:00
13:50:00
13:55:00
14:00:00
14:05:00
TIME

LINE 17

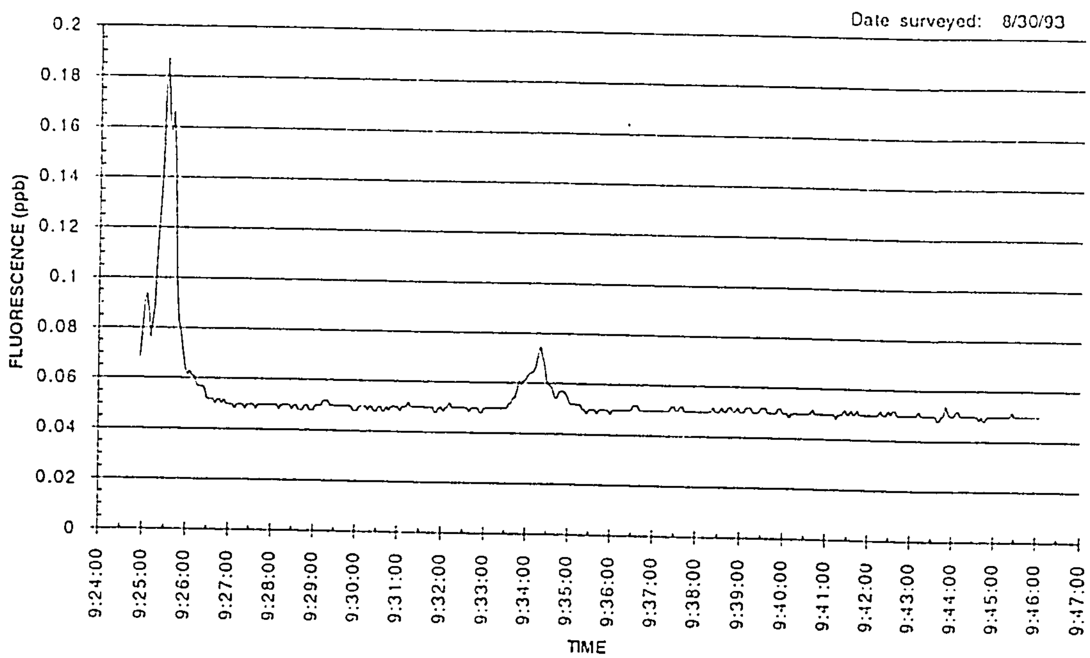


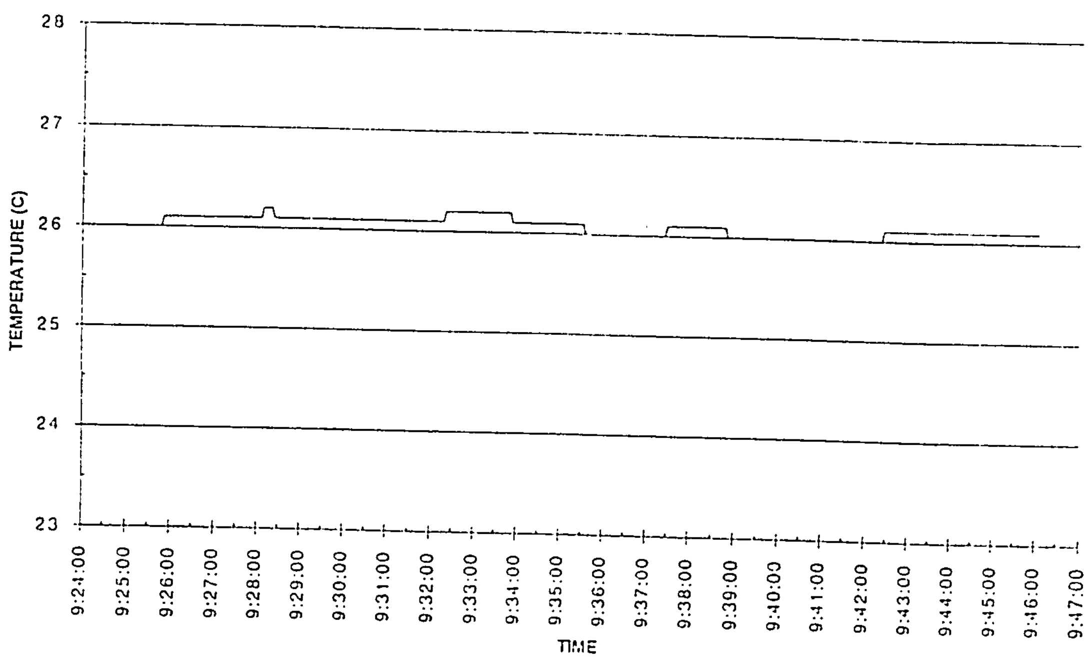
Date surveyed: 8/24/93
FLUORESCENCE (ppb)
0.2
0.18
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
12:38:00
12:39:00
12:40:00
12:41:00
12:42:00
12:43:00
12:44:00
12:45:00
12:46:00
12:47:00
12:48:00
12:49:00
12:50:00
12:51:00
12:52:00
TIME



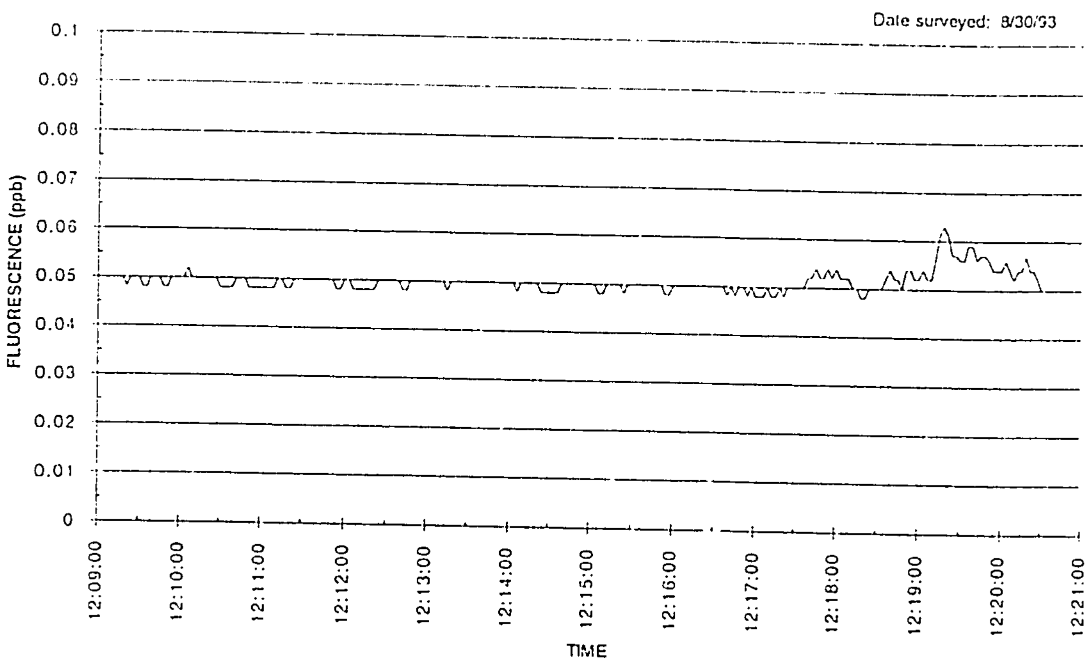
TEMPERATURE (C)
28
27
26
25
24
23
12:38:00
12:39:00
12:40:00
12:41:00
12:42:00
12:43:00
12:44:00
12:45:00
12:46:00
12:47:00
12:48:00
12:49:00
12:50:00
12:51:00
12:52:00
TIME

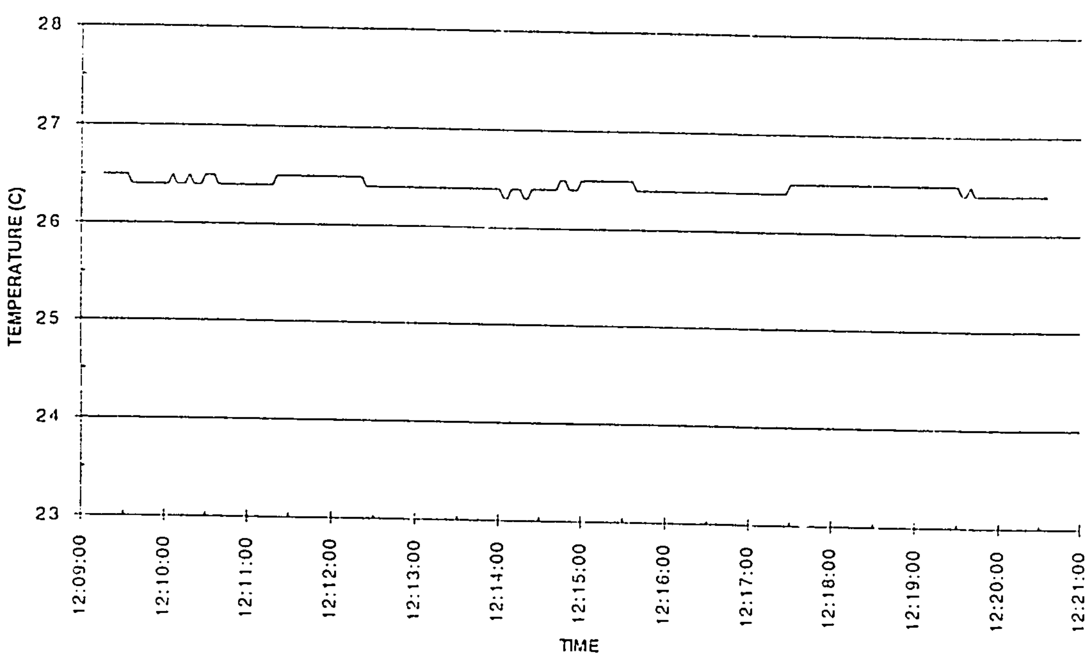
LINE 17A


Date surveyed: 8/30/93
FLUORESCENCE (ppb)
0.2
0.18
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
9:24:00
9:25:00
9:26:00
9:27:00
9:28:00
9:29:00
9:30:00
9:31:00
9:32:00
9:33:00
9:34:00
9:35:00
9:36:00
9:37:00
9:38:00
9:39:00
9:40:00
9:41:00
9:42:00
9:43:00
9:44:00
9:45:00
9:46:00
9:47:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
9:24:00
9:25:00
9:26:00
9:27:00
9:28:00
9:29:00
9:30:00
9:31:00
9:32:00
9:33:00
9:34:00
9:35:00
9:36:00
9:37:00
9:38:00
9:39:00
9:40:00
9:41:00
9:42:00
9:43:00
9:44:00
9:45:00
9:46:00
9:47:00
TIME

LINE 18A


Date surveyed: 8/30/93
FLUORESCENCE (ppb)
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
12:09:00
12:10:00
12:11:00
12:12:00
12:13:00
12:14:00
12:15:00
12:16:00
12:17:00
12:18:00
12:19:00
12:20:00
12:21:00
TIME


TEMPERATURE (C)
28
27
26
25
24
23
12:09:00
12:10:00
12:11:00
12:12:00
12:13:00
12:14:00
12:15:00
12:16:00
12:17:00
12:18:00
12:19:00
12:20:00
12:21:00
TIME

APPENDIX B

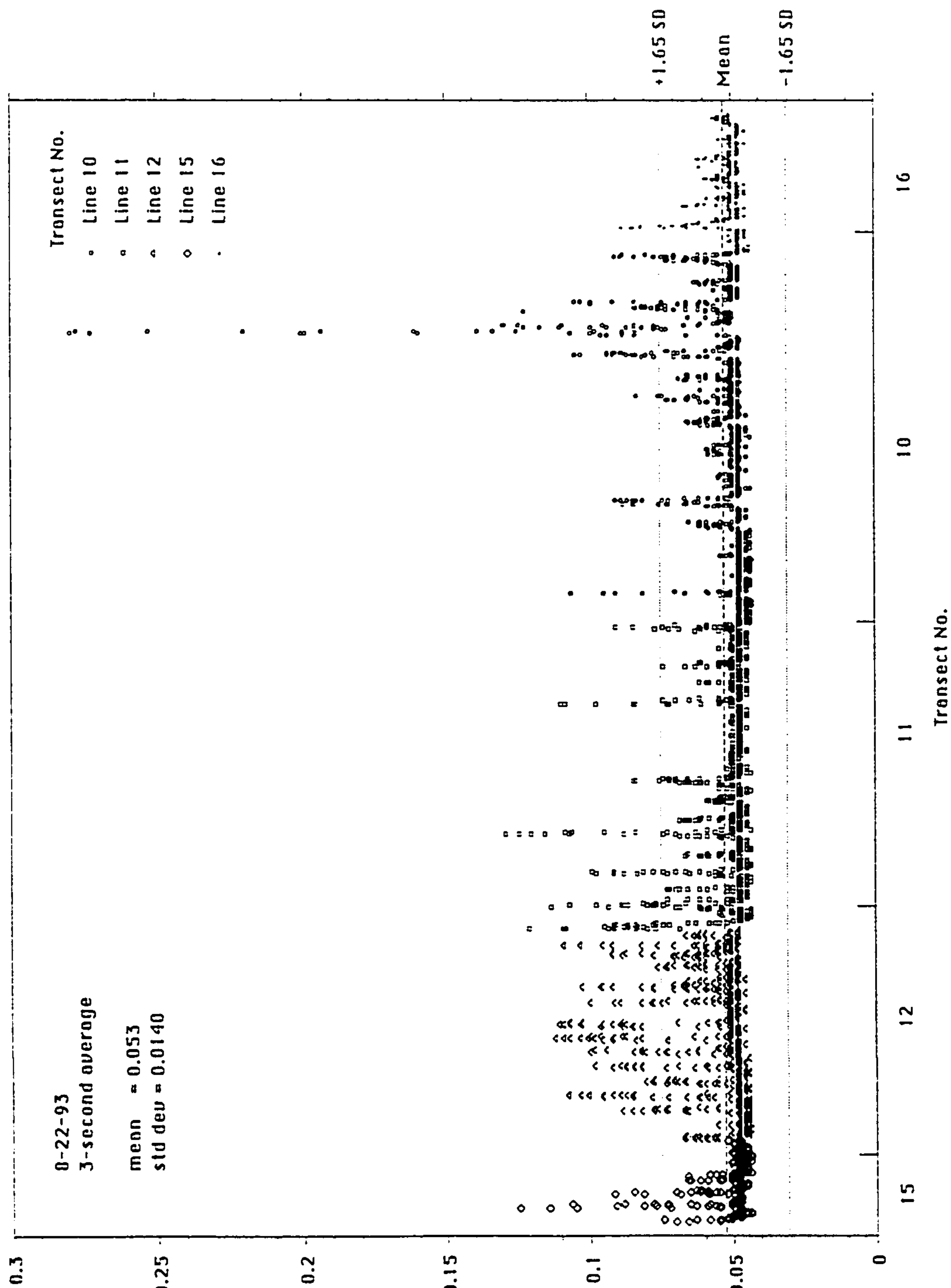
SCATTERGRAM PLOTS OF THE DATA

A statistical analysis of the fluorometry data was performed to separate possible signals from the background variations. The mean and standard deviation were calculated for the 3-sec averaged data collected each day (Table 2) and all data were plotted as scattergrams of daily observations versus concentration. The mean value and 95-percent confidence limits, represented by 1.65 standard deviations were also plotted on the scattergrams. This information is presented in the following eight plots.

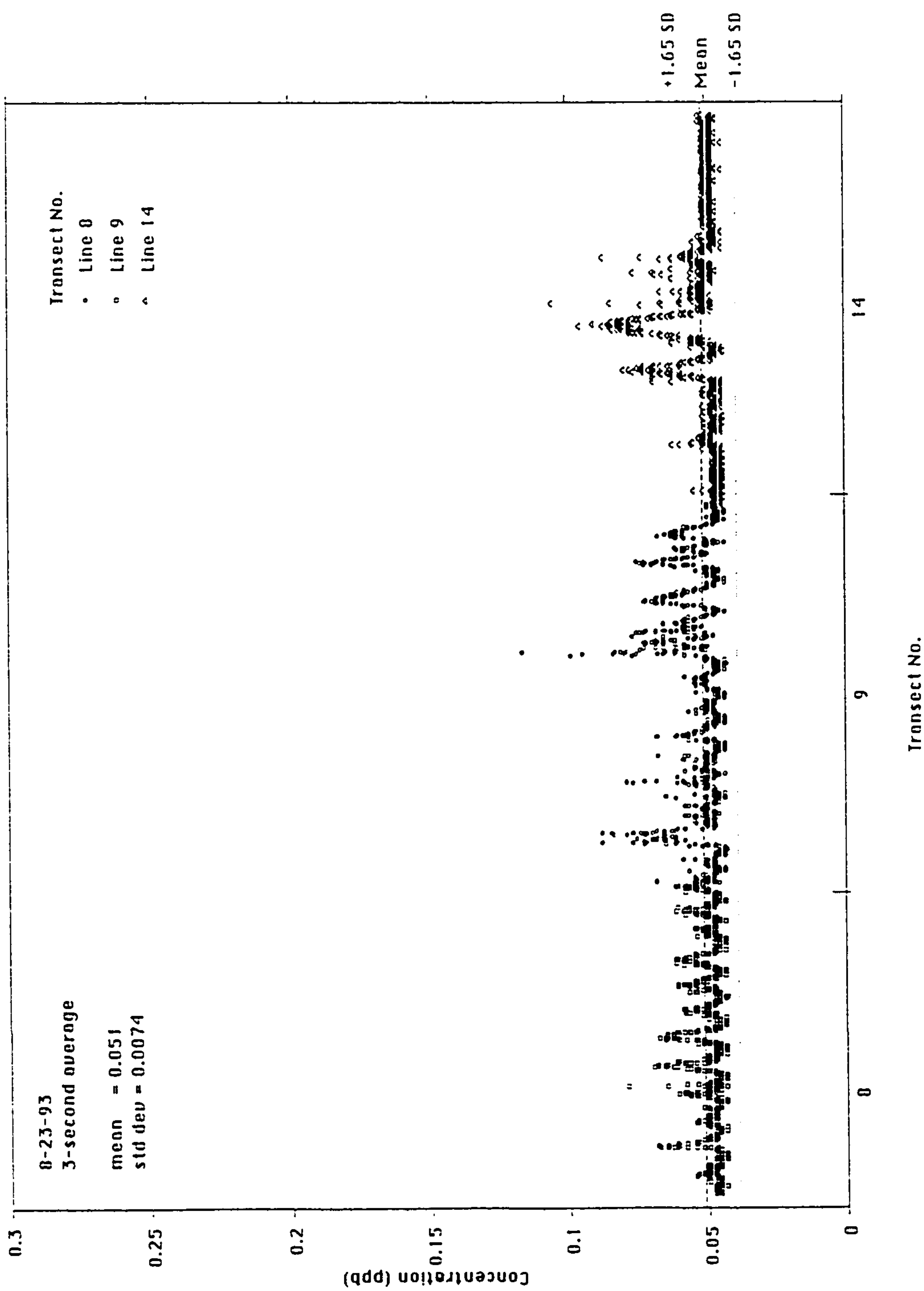
Data from the first half of the survey, August 22 - 25, 1993, were considerably more dispersed than data from the second half, August 27 - 30, 1993. This large dispersion correlates to the high readings (shown in the corresponding line plots in Appendix A), which were attributed to backscattering interference from near-bottom suspended particles passing through the flow cell of the fluorometer.

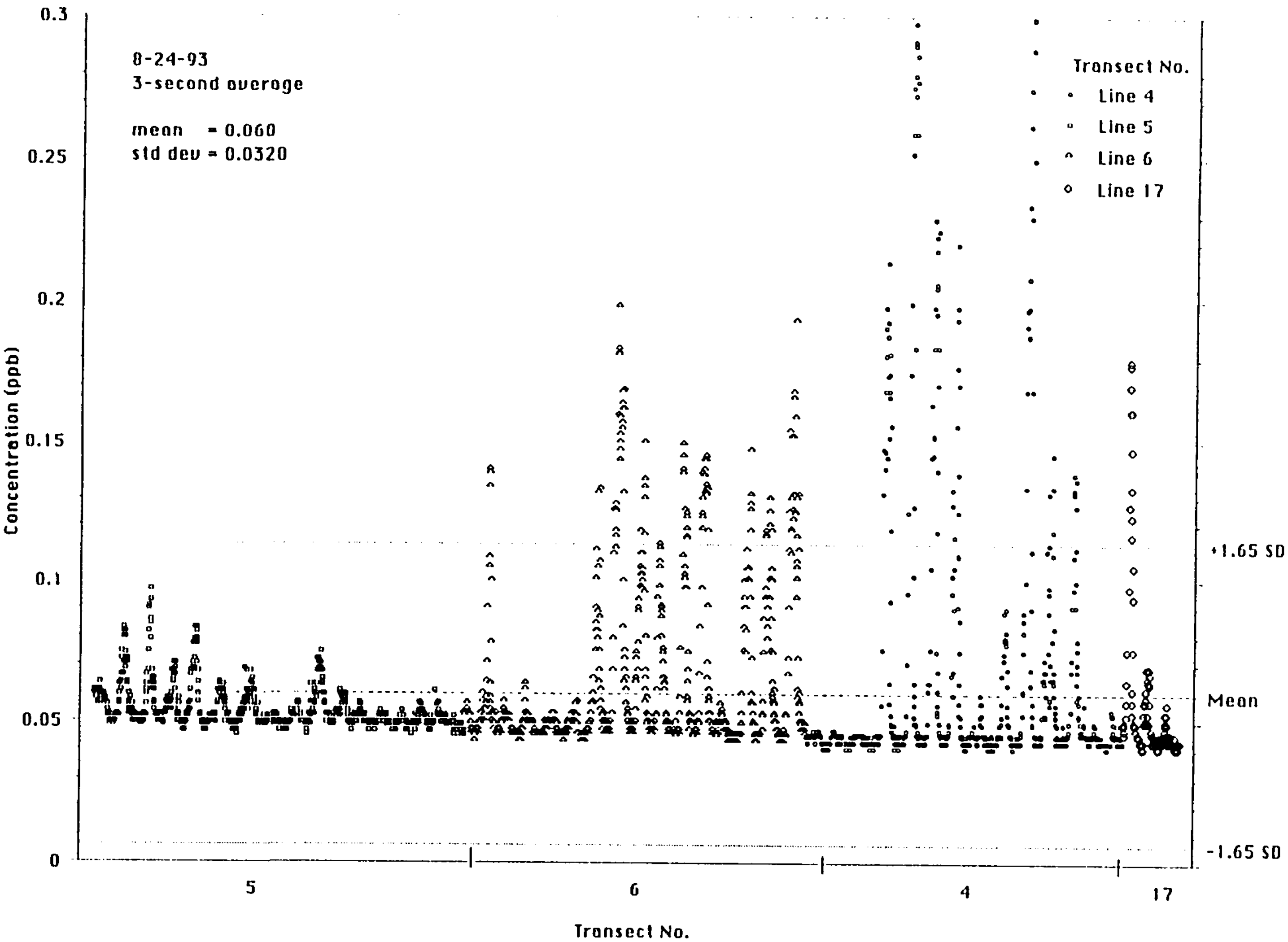
Data from the second half of the survey were more consistent. Only short sections of a few lines had data exceeding the 95-percent upper confidence limit. These sections were at the beginning of Lines 5A, 3A, 16A, and 17A; the end of Lines 6A and 8A (which was surveyed in opposite direction to other lines); and in the center of Lines 17A and 13A.

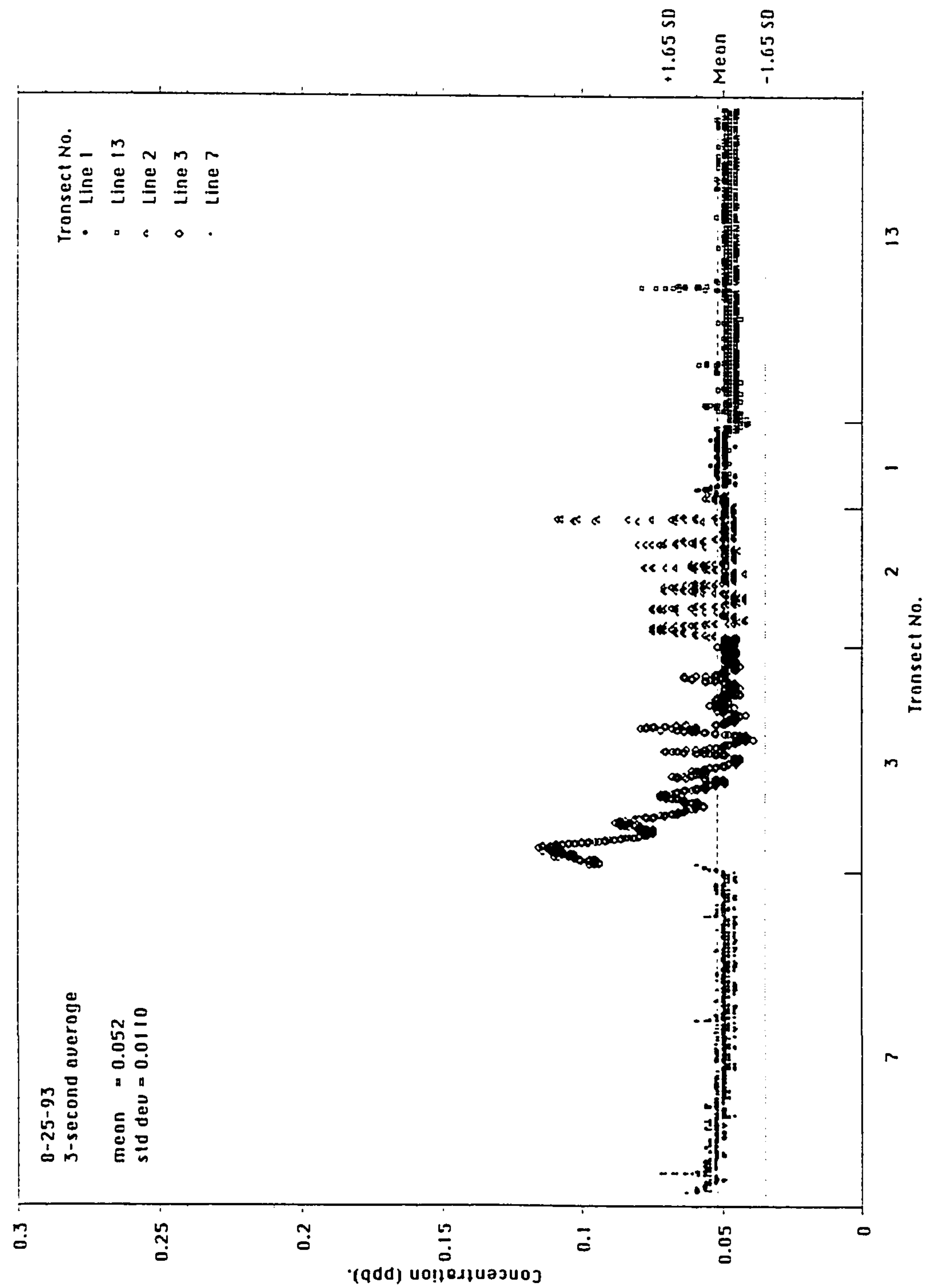
Each of these sections falling outside the upper confidence limit were identified on the contour plots to determine the extent of continuity of values across adjacent transects and to identify and verify any spatial patterns of elevated readings.

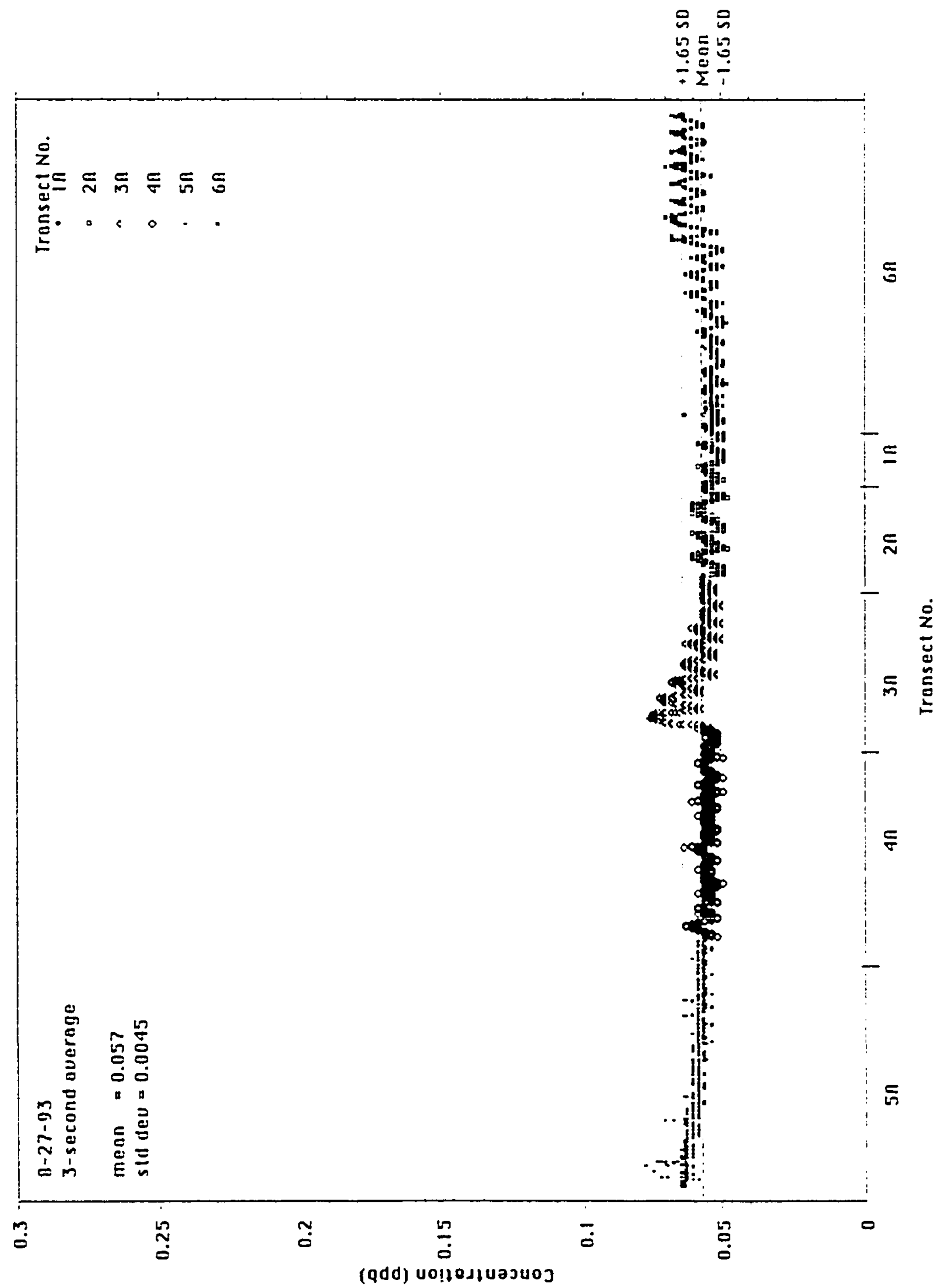


8-22-93
3-second average
mean = 0.053
std dev = 0.0140
Transect No.
Line 10
Line 11
Line 12
Line 15
Line 16
+1.65 SD
Mean
-1.65 SD
Concentration (ppb)
Transect No.
0.3
0.25
0.2
0.15
0.1
0.05
0
15
12
11
10
16

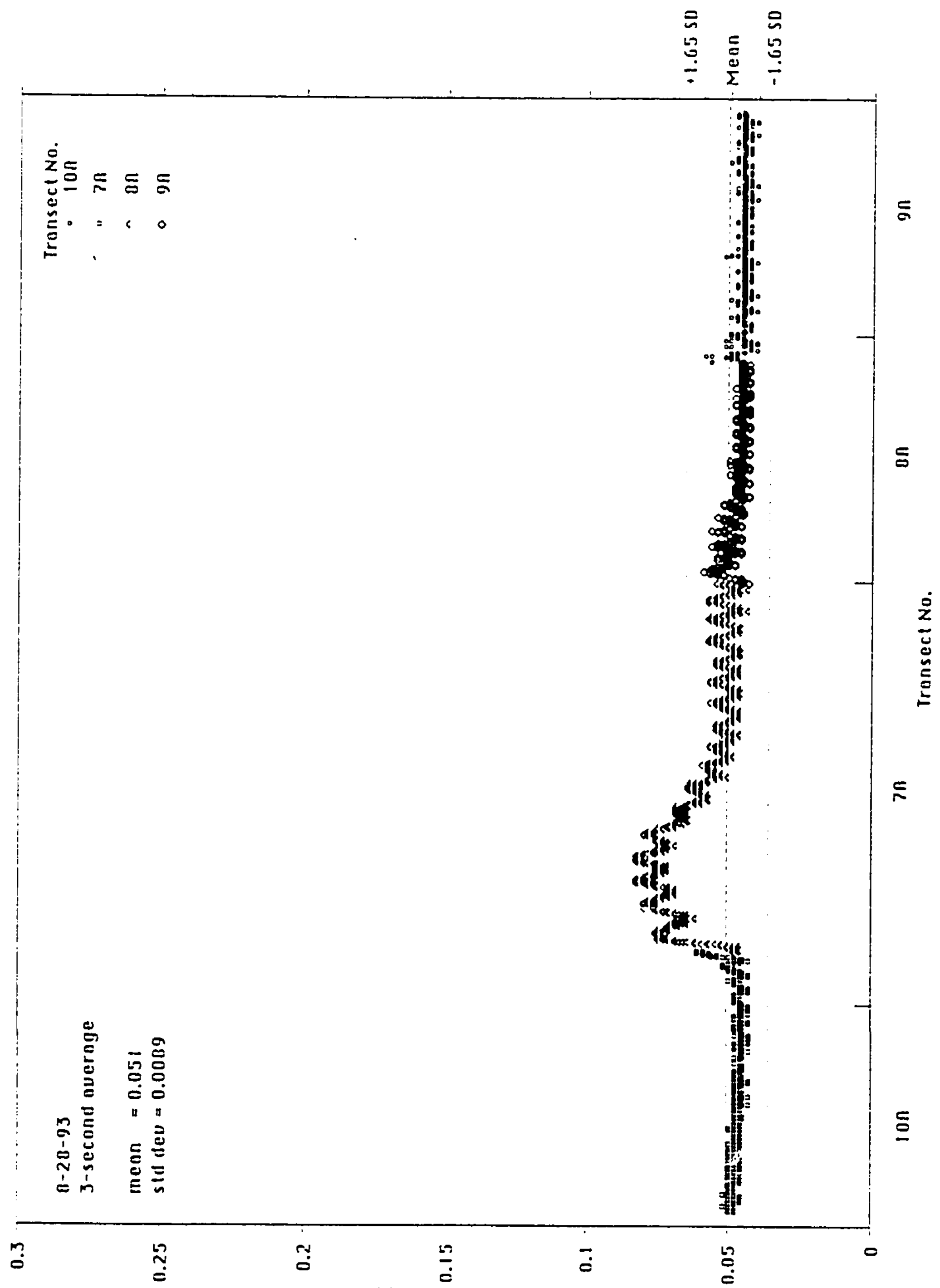

8-23-93
3-second average
mean = 0.051
std dev = 0.0074
Transect No.
Line 0
Line 9
Line 14
+1.65 SD
Mean
-1.65 SD
0
0.05
0.1
0.15
0.2
0.25
0.3
Concentration (ppb)
0
9
14
Transect No.

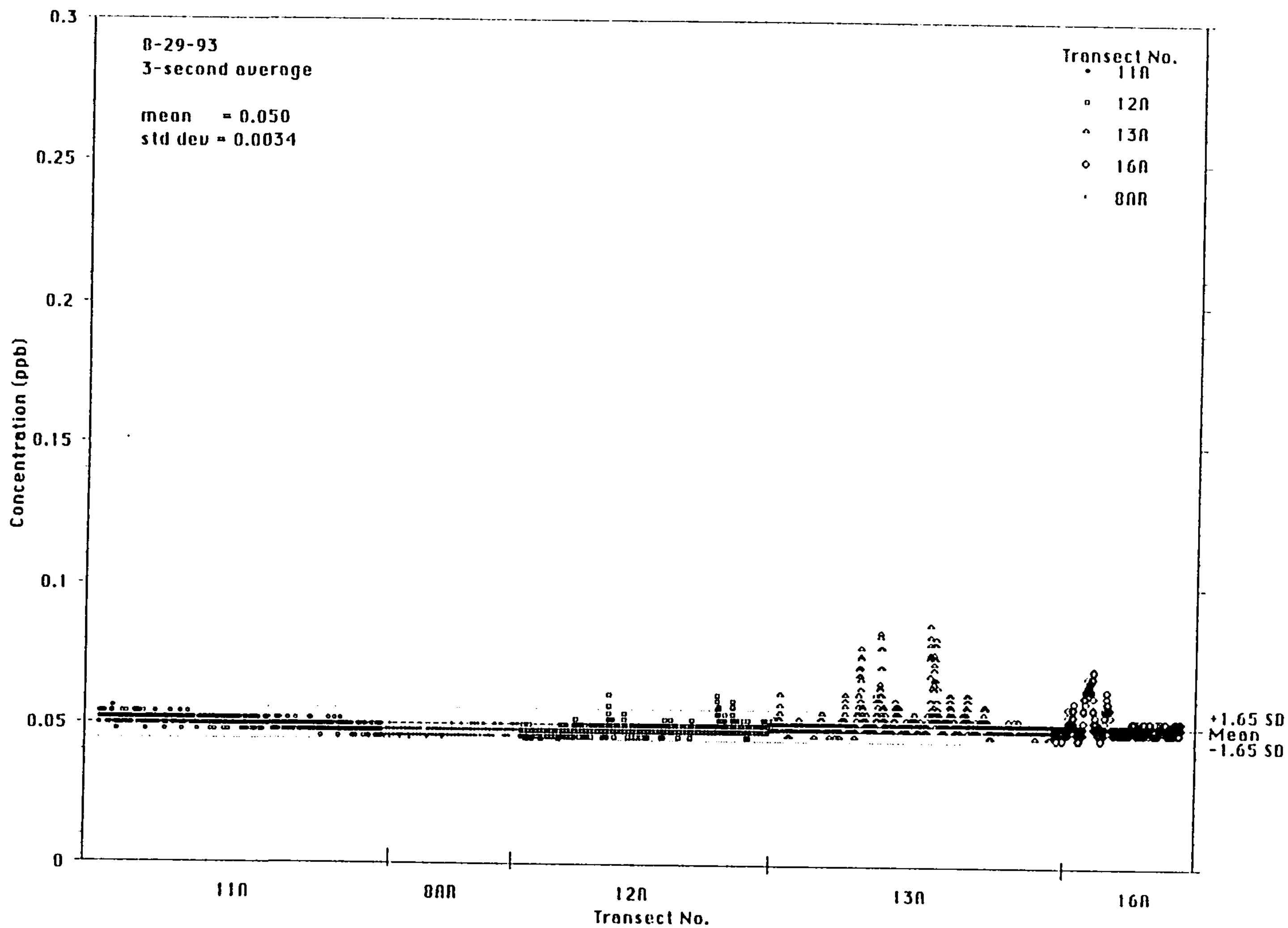

8-24-93
3-second average
mean = 0.060
std dev = 0.0320
Transect No.
Line 4
Line 5
Line 6
Line 17
Concentration (ppb)
0.3
0.25
0.2
0.15
0.1
0.05
0
+1.65 SD
Mean
-1.65 SD
5
6
4
17
Transect No.


0-25-93
3-second average
mean = 0.052
std dev = 0.0110
Transect No.
Line 1
Line 13
Line 2
Line 3
Line 7
+1.65 SD
Mean
-1.65 SD
Concentration (ppb)
0
0.05
0.1
0.15
0.2
0.25
0.3
Transect No.
7
3
2
1
13

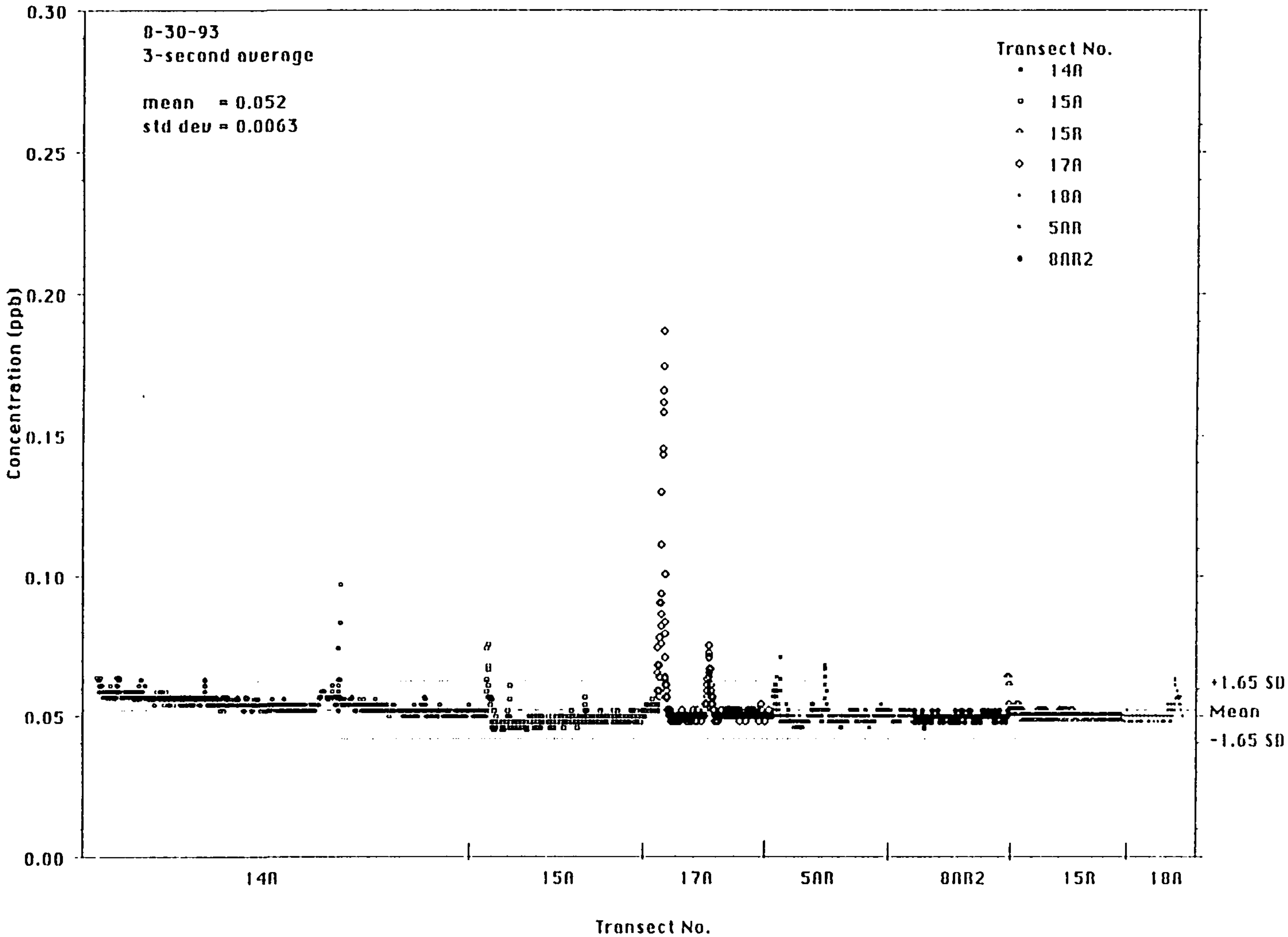


8-27-93
3-second average
mean = 0.057
std dev = 0.0045
Transect No.
1A
2A
3A
4A
5A
6A
+1.65 SD
Mean
-1.65 SD
Concentration (ppb)
0.3
0.25
0.2
0.15
0.1
0.05
0
Transect No.


8-28-93
3-second average
mean = 0.051
std dev = 0.0089
Transect No.
Transect No.
+1.65 SD
Mean
-1.65 SD
Concentration (ppb)
0
0.05
0.1
0.15
0.2
0.25
0.3



8-29-93
3-second average
mean = 0.050
std dev = 0.0034
Transect No.
11A
12A
13A
16A
8AA
0.3
0.25
0.2
0.15
0.1
0.05
0
Concentration (ppb)
+1.65 SD
Mean
-1.65 SD
11A
8AA
12A
13A
16A
Transect No.


8-30-93
3-second average
mean = 0.052
std dev = 0.0063
Transect No.
14A
15A
15B
17A
18A
5AA
8AA2
Concentration (ppb)
0.30
0.25
0.20
0.15
0.10
0.05
0.00
+1.65 SD
Mean
-1.65 SD
14A
15A
17A
5AA
8AA2
15B
18A
Transect No.