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Modeling of the Fate of Lead due to the Removal of Corroded Sub-marine Cables in Aquatic Environments

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A Thesis

in

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ABSTRACT

Modeling of the Fate of Lead due to the Removal of Corroded Sub-marine Cables in Aquatic Environments

Alaa Al-Hawari

Removal of contaminated sediments from waterways to maintain desired depths or to remediate sites affected by contaminated sediments, has led to increased concerns related to the release of contaminants into the water column during dredging activities. Concerns were extended to other activities such as the removal of objects buried in sediments (e.g. corroded sub-marine cables), which also cause the release and dispersion of contaminants into the water column.

The processes by which lead is released from corroded cables and dredged material are complex, but have to be taken into consideration when assessing the potential risk of dredging and removal activities. In this research, the effect of various physical factors on the release and dispersion of lead from corroded sub-marine cables and bottom sediments was studied. Among these factors are the hydrological and physical properties of the water body (e.g. flow rates, resuspension and settling), situation and position of cables (e.g. on the surface or at different depths), physico-chemical characteristics (e.g. adsorption, desorption and diffusion rate), transport of toxicant (e.g. advective and dispersive mixing) and dredging activities (e.g. mechanical and hydraulic). In order to show the effect of different factors, a conceptual model called the Model for Fate of Corroded Lead (*MFCL*) was developed. The objective was to predict lead concentration in the water column in dissolved and particulate forms. The *MFCL* model is a powerful tool for predicting the changes in risk resulting from the implementation of various remedial actions, including the "no action" alternative.

The *MFCL* model indicated that the water quality decreases proportionally with the increase of corrosion rate. It was also found that the buried cables compared to cables situated on the surface would decrease the amount of released lead into the water column. However, the depth of cable would have minimal impact on the concentration of lead released into the water column. With respect to dredging schedule, the model demonstrated that intermittent dredging would decrease the effect on the water quality.

Rivers with high flow rates would cause immediate dispersion of particulate and dissolved contaminants thereby provoking increased dissolution of lead at the cable position. The highest impact on aquatic species was observed to be in the interface area between sediments and water column. Unfortunately, this area is recognized to be the spawning area for fishes having habitats in the St. Lawrence River in Montreal.

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To My Beloved Parents

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CHAPTER 1

INTRODUCTION

Removal of contaminated sediments from waterways to maintain desired depths or to remediate sites affected by contaminated sediments has led to increased concerns related to the release of contaminants into the water column during dredging activities (Vale *et al.*, 1998). The concerns are also associated with the removal of objects containing metals (e.g. sub-marine cables), which can also cause the release and dispersion of contaminants into the water column during this activity.

Many types of modern sub-marine communication or electrical power cables have been in use for underwater distribution and transmission. These cables consist of several metallic sheaths (e.g. lead sheaths) that can be damaged due to corrosion. Corrosion of these metallic sheaths cause the release of lead (having a very high toxicity) and other heavy metals into the aquatic environment (Schubert, 1985).

The processes by which lead is released from corroded cables and dredged material are complex, but must be taken into account when assessing the potential risk of dredging and removal activities. It is necessary to estimate and study the effect of various factors on the release and dispersion of lead from corroded sub-marine cables and bottom sediments. Therefore, a computer simulation is considered to be a useful tool for studying the interactions between the different factors affecting the release and fate of lead in the aquatic environment.

Particular scenarios with and without cables removal have to be considered. If the decision of removal is taken, it is also necessary to know which type of dredging activities must be chosen, and the timing of this work.

At present, the impacts of these activities have not been studied. Consequently, an investigation of potential risk of above mentioned activities on biota has to be performed. Classification of factors and their associated impacts on water quality and biota have to be also considered.

1.1 OBJECTIVE OF THE THESIS

The principal objective of this thesis is to investigate the impact of corroded submarine cables on water quality during corrosion and removal processes.

The secondary objective is to identify and evaluate the most important factors that would affect the release of lead from dredged bottom sediments into the water column.

CHAPTER 2

LITERATURE RIEVIEW - STATEMENT OF THE PROBLEM

The literature review has shown that there are no previous studies dealing with the fate of corrosion products due to the removal of corroded sub-marine cables from bottom sediments. Consequently, the review of literature concerning various topics associated with the removal of corroded cables was performed in order to accumulate information on the factors affecting the fate of metals released due to corrosion and dredging.

The first part of the literature review describes materials used for communication and electrical sub-marine cables and their corrosion process which could cause the release of heavy metals (e.g. lead) into the water column. The second part describes the behavior of heavy metals in aquatic environments and during dredging processes. The final section of the literature review deals with the description of different dredging techniques, which could be used to remove corroded sub-marine cables, and their impact on water quality.

2.1 Corrosion of sub – marine cables

Components and corrosion of sub-marine cables will be discussed in this section, from the perspective of the corroded components, particularly lead, released into the aquatic environment.

2.1.1 Components of Sub – Marine Cables

"Many types of modern communication and electric power cables have been in use for underwater distribution and transmission of power. They vary greatly in their design and constructional techniques. However, all power cables consist of three essential components" (King and Halfter, 1982):

1) The metallic core of the cable which provides the electrical conducting path.
2) An external protective layer, preventing the cable from any mechanical or chemical damage.
3) The protective sheath of the cable, which covers the core of the cable and prevents damage, especially the corrosive effect of water (Schubert, 1985).

Figure 2.1 shows a cross-section of a medium tension electric cable (Pintado and Montero, 1992).

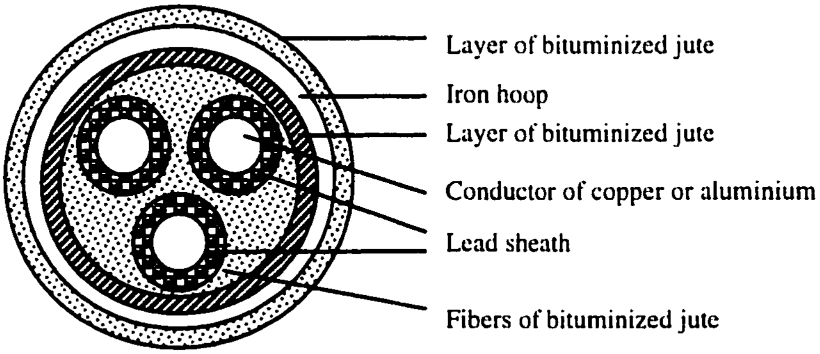


Figure 2.1: Cross-section of a medium tension electric cable structure by Pintado and Montero (1992).

2.1.1.1 Lead Sheath

Since lead has high flexibility at normal temperature and because the slight heating (about 170 °C) could extrude it around the cores without causing any damage to the paper insulation inside the cable, it is used as a primary protective material (Schubert, 1985).

Lead used in cable sheaths is manufactured either with or without the addition of copper. Antimony or tin could also be added to lead in order to increase the hardness of lead sheath. This would allow it to withstand the shaking associated with its transport and installation (Schubert, 1985).

Due to a high corrosion resistance, antimony-alloyed lead sheaths are commonly employed in cable sheaths. Typically, the average antimony required is 0.8%. In the case of buried cables, an alloy of tellurium is more advantageous to use because of its inter-crystalline stability (Schubert, 1985).

The thickness of the lead sheath depends on the application for which the cable is used, the design of the cable and the thickness of the core beneath the sheathing (Schubert, 1985). According to Schubert (1985) the wall thickness of the lead sheath is between 1.3 and 3.2 mm.

2.1.1.2 Steel Sheath

The thickness of the steel sheath depends on the core thickness beneath the sheathing. According to Schubert (1985) steel sheaths have a wall thickness between 0.3 and 0.6 mm. Steel sheaths must have an external covering applied directly to the metal sheath, and must be carefully immersed in a waterproof liquid compound before application to the cable, because when the steel is subjected to moisture, it will rust without forming an anti-corrosion layer. Consequently, gas pressure should be used when employing cables with a steel sheath (Schubert, 1985). As stated by Schubert

(1985): "If the sheath of the cable becomes damaged, the excess internal pressure of the dry gas will normally prevent the penetration of moisture into the cable core".

2.1.1.3 Asphalt Sheath

Asphalt sheaths are used to protect the metal sheathing and any other sheaths from corrosion. Usually cables are provided with an extra anti-corrosive covering (e.g. bitumen) which is then covered with a protective layer of PE or PVC (King and Halfter, 1982, Pansini and Smalling, 1993).

Cables usually have an inner and outer protective asphalt covering. The internal and external protective coverings are usually made from a combination of substances such as, rubber foils, plastic foils, fibers and plastics (Schubert, 1985). The external protective cover usually includes jute, and non-adhesive chalk, or plastics.

2.1.2 Corrosion of Cables

When metals are exposed to reactive environments in the surrounding medium corrosion will occur. Corrosion reactions cause metals to change to their original ores via different oxidation reactions (Wranglen, 1985). According to Montgomery (1985), the mechanism by which metals are oxidized involves three steps: "(1) transport of dissolved reactants to the metal surface; (2) electron transfer at the surface; and (3) transport of dissolved products from the reaction site." When transport steps are the slowest, the corrosion reaction is said to be under "transport control". When transfer process of electrons to the metal surface is slower, corrosion is said to be under "activation control".

Literature review shows that the process by which cables placed in soil become corroded is complicated, since many factors will govern this process. However, it is necessary to generate the most important factors affecting the corrosion process in sediments and the fate of corrosion products.

Analysis of lead sheath of communication cables after corrosion (Pintado and Montero, 1992; Schubert, 1985), revealed that the main corrosion components were lead carbonate ($PbCO_3$), and lead hydroxide ($Pb(OH)_2$). According to Schubert (1985) "the corrosion is uniform and usually is only superficial. The holes produced in lead are not usually tunnel-like, but adopt a somewhat deep, saucer-like shape". The average yearly corrosion rate of lead was observed to be approximately 0.3 mm for cables buried in soil (Schubert, 1985).

According to Pintado and Montero (1992), areas of localized attack were due to the presence of bacteria and fungi in both lead and asphalt sheaths. The biological degradation products such as short chain fatty acids, succinic acid, acetic acid, α-ketoglutaric acid and glyoxylic acid were found around the degraded asphalt, which are all products of biological degradation. Due to the combination of CO_2 from fungal metabolism and metallic lead, lead carbonate was the main product around the lead sheath. Table 2.1 summarizes the corrosion products, which were found in the study by Pintado and Montero (1992).

Table 2.1: Corrosion products of cable materials (after Pintado and Montero, 1992).

Corrosion Products	Asphalt Degradation Products
Hydrocerusite ($Pb_3(CO_3)_2(OH)_2$)	Asphalt paraffinic compounds Acetic acid α-ketoglutaric acid
Lead Tetroxide (Pb_3O_4)	Citric acid Succinic acid Glyoxylic acid

It was found that corrosion products of lead formed an adherent protective film around the metal surface, isolating the underlying metal. The rate of lead corrosion would increase if this protective film is damaged by different mechanical or chemical processes (Al-Kharafi and Badawy, 1998 and Palraj *et al.*, 1995).

Corrosion of metals in water is strongly affected by pH. At a pH value greater than 9.5, iron and steel would not corrode while lead would be affected by strong alkaline solutions since it reacts with bases (Schubert, 1985).

Wranglen (1985) discussed the effect of the water content in soil on the corrosion rate. With an increase in the soil water content, the corrosion rate tends to increase upon a saturation value (as the case in sediments), then would be followed by a decrease in the corrosion rate due to the lack of oxygen.

A comparison between the corrosion of lead in oxygen-free and oxygen saturated acidic solutions, and the effect of chloride was studied by Al-Kharafi and Badawy (1998). The results of this study showed that under acidic conditions, the presence of oxygen in the solution increases the lead corrosion resistance, and the presence of chloride ions increases the rate of corrosion.

Removal of corroded sub-marine cables from bottom sediments could provoke the release of the corrosion compounds and other metals already existing in sediments, into the water column. Therefore, the following section describes the behavior of different metals in aquatic environments.

2.2 Behavior of Heavy Metals in the Aquatic Environment

In aquatic environments, metals exist in different forms and concentrations. Heavy matals in the sediment can be released into the water body, thereby having a significant effect on river water quality (Xianghua and Herbert, 1999). Physicochemical characteristics of the surrounding environment (e.g. pH and redox potential) and sediment properties (e.g. clay content, organic matter content, amounts of reactive iron and manganese), govern the mobilization and immobilization processes of metals in the sediment (Gambrell *et al.*, 1980).

According to Yong *et al.* (1995), the increase of the pH value would increase the accumulation of heavy metals in the sediment. At pH = 7, metals will be mainly in hydroxide, sulfate, and chlorate forms. At lower pH values, heavy metals could be mobilized. At the same time, sorption onto sediment particles would be lower due to competition from H^+ ions at the exchange sites. The amount of heavy metals adsorbed depends on the sediment composition (Yong *et al,* 1995).

A review of heavy metal behavior in aquatic environments could be summarized as follows:

- **Lead**

Lead , in aquatic environments, exists in Pb^{+2} and Pb^{+4} oxidation states, with Pb^{+2} being more stable than Pb^{+4} (Jaagumagi, 1992). Lead has a complicated speciation which depends on several different factors such as pH, dissolved oxygen and the concentrations of other organic and inorganic compounds (Galvez-Cloutier, 1995). According to Bennett (1995), the solubility of lead, as a function of pH, depends upon the alkalinity of water. When pH and alkalinity are low, soluble lead is the dominant species. In low alkalinity waters ($\geq$ 25 mg/l as $CaCO_3$), lead is highly sensitive to pH and lead solubility would decrease with increasing pH. In highly alkaline waters ($\geq$ 100 mg/l as $CaCO_3$), the solubility of lead is insignificant in the pH range between 6.5 and 8.0. In these conditions, the most stable species of lead is $PbCO_3$ (cerussite).

Gambrell *et al.* (1991) observed that soluble Pb is slightly affected by redox potential except at intermediate potentials (approximately 100 mV). Salinity did not have a significant effect on soluble Pb levels in the water column. Furthermore, Xianghua and Herbert (1999) found no release of Pb in their long-term experiment. These findings were similar to results of other studies performed on the availability of Pb in water (Calmano *et al.*, 1993, Gambrell *et al.*, 1980 and Galvez-Cloutier, 1995).

Lead in the presence of inorganic and organic ligands, is able to form various complexes. According to Galvez-Cloutier (1995), lead has the ability to form a wide variety of complexed ions. Most complex ions are electrically charged and consist of a central atom or ion covalently bonded to a number of other atoms, ions or molecules. Since lead has a high charge density, it tends to form stable complexes. The inorganic ligands of interest in the sediment environment are H_2O, NH_3, Cl^-, OH^-, $(CO_3)^{-2}$ and $(SO_4)^{-2}$. Lead also has the ability to form strong complexes with organic ligands containing the donor atoms S, N and O. The most signficant of these organometallic lead (IV) compounds are tetraethylead $Pb(C_2H_5)_4$ and tetramethylead $Pb(CH_3)_4$ (Dojlido and Best, 1993).

A significant portion of lead in sediment is associated with iron oxides (Chang *et. al.*, 1998). Lead is strongly bound to hydroxides and organic matter in oxidized sediment. In reducing sediment, lead can be released into the water column in two different forms, either as free ions in the dissolved phase, or as sulfides, as iron and manganese hydroxides dissolve (Calmano *et al.*, 1994). Table 2.2 shows that the largest fraction of iron species in sediments is sulfide-bound iron.

Table 2.2: Iron species in sediments (Calmano *et al.*, 1994)

Iron Species	Percent of total iron
Vivianite ($Fe_3(PO_4)_2$)	20
Siderite ($FeCO_3$)	8
Fe(II) in Ca-Mg carbonate	4
Fe(II) in humic matter	1
Fe(II) in clay minerals	18
Fe(III) and sulfide bound Fe(II)	49

According to Calmano *et al.* (1993), re-suspension of the anoxic layer in bottom sediments would occur during dredging activities. The overlying water, which contains high concentration of dissolved oxygen, is in direct exposure to these sediments. Therefore, re-suspended anoxic sediments are oxidized by oxygen dissolved in the water.

Calmano *et al.* (1993) reported that the amount of Pb released from the sediment to the water column in the re-suspension process was 0.68% of the total Pb content of the sediment. However, after re-suspension, 85.5% of the released lead was scavenged again. Therefore, the net amount of Pb released into the water column was only 0.1% of the total amount of Pb in the sediment.

Scavenging of lead is mainly attributed to re-adsorption and co-precipitation of the released metal with Fe and Mn hydroxides. Calmano *et al.* (1993) also reported the order of scavenging percentage for metals as below:

$$Pb > Cu > Zn > Cd$$

This order is the reverse of the mobilization percentage for metals. These findings were similar to the results obtained by Yong *et al.* (1995), Galvez-Cloutier (1995) and Xianghua and Herbert (1999).

Vale *et al.* (1998) performed both field and 24-hour laboratory experiments on sediment re-suspension and release. They measured total metal concentration in suspended sediments created around a dredging vessel and a nearby estuarine area. Results showed that the mean concentration of zinc, lead, and cadmium were lower near the dredger than a nearby estuarine area.

The fate of lead compounds could be affected by the presence of other metals in aquatic environments. Consequently, a review of the behavior of other metals present in aquatic environments is required. Zinc, cadmium, and copper were considered due to the fact that they can also be present in the cable material.

- Zinc

Zinc, in aquatic environments, occurs mainly as Zn^{+2}, which is capable of combining with both acids and bases (Jaagumagi, 1992). Solubility of zinc is affected strongly by pH and redox potential (Gambrell *et al.*, 1980). At neutral pH, zinc is adsorbed into the sediment by different compounds such as iron and manganese hydroxides, clay minerals and organic matter. The structure of the sediment and concentration of organic matter affect the adsorption properties of these compounds. In coarse sediment with low organic matter, zinc is mainly adsorbed by iron and manganese hydroxides. In fine-grained sediment, sorption to organic matter is the most significant process. Concentration of zinc in pore water is controlled by the solubility of iron and manganese hydroxides in the oxidized layer and by sulphides (FeS and MnS) in the reduced layer (Jaagumagi, 1992).

- Cadmium

In water, cadmium can exist as free ions (Cd^{+2}) or complexed to various ligands such as humic acids, organic particles and various oxides. The solubility of iron and manganese hydroxides in the oxidized layer and sulphides in the reduced layer, control the concentration of cadium in sediments (Jaagumagi, 1992).

According to Gambrell *et al.* (1991), "Soluble Cd increases by a factor of more than 10 under oxidizing conditions compared to strongly reducing conditions at all pH levels". Also, there is a high effect of salinity on Cd, where an increase of salinity causes an increase of Cd in solution (Gambrell *et al.*, 1991).

- Copper

Cu^{+} and Cu^{+2} , are the most common states of copper in aquatic environments. Copper can form complexes with inorganic or organic ligands (Jaagumagi, 1992). According to Gambrell *et al.* (1991), "Copper in sediment has a high affinity for iron and manganese hydroxides, clays, carbonate materials and organic matter, though formation of these complexes is pH and redox dependent". As the redox potential increases, soluble copper in the water column has a tendency to increase. Copper is mainly present in the form of organic complexes and cupric carbonate complexes. At neutral pH, copper could co-precipitate with the presence of inorganic carbon and iron and manganese hydroxides. Copper appears primarily in the form of sulphide complexes in a reducing sediment, which cause its immobilization in the sediment. Copper is adsorbed to organic matter or to iron and manganese hydroxides in the oxidized zone (Jaagumagi, 1992).

Dredging is required for the removal of corroded sub-marine cables. Re-suspension of the bottom sediments containing metals would occur during dredging activities causing the release of those metals into the water column.

2.3 Dredging Technologies

A wide variety of dredging equipment exists on the market. The equipment can be categorized into two main groups, namely mechanical dredges and hydraulic dredges (St. Lawrence Center, 1993a). Regarding the removal of the sub-marine cables from bottom sediments, hydraulic and mechanical dredging methods may be used. As a result, different kinds of dredging technologies will be discussed in this chapter. The following section describes different dredging techniques and there impacts on the water quality.

2.3.1 Mechanical Dredges

Mechanical dredges remove sediment by direct application of mechanical force on the sediment surface (St. Lawrence Center, 1993a). Their main advantage is that they remove sediments at the near in-situ density since little amounts of water are retained in the bucket. This would decrease the amount of sediments that must be transported and treated afterwards (St. Lawrence Center, 1993a and Herbich *et. al,* 1995). Mechanical dredges can be operated in small and confined areas and are effective in locations with obstructions (St. Lawrence Center, 1993a). Mechanical dredging techniques can be used for the removal of objects (e.g. cables) from bottom sediments.

The production rate of mechanical dredges is low compared to hydraulic dredging. It can be around 500 m^3/h and 7600 m^3/h for mechanical and hydraulic dredges respectively (St. Lawrence Center, 1993a). There are several kinds of mechanical dredges, but the ones most commonly used in the St. Lawrence River will be discussed in the following sections.

2.3.1.1 Clamshell Dredge

Clamshell dredges consist of a bucket or clamshell operated by a crane (Barnard, 1978). "The bucket descends to the bottom in the open position and penetrates the material to be dredged by the force of its weight and the action of the closing mechanism" (St. Lawrence Center, 1993a). Releasing the cable that closes the bucket would unload the dredged material.

According to the St. Lawrence Center (1993a), the accuracy of dredging would decrease with increasing water body depth; the current would interfere with bucket control. The capacity of the clamshell bucket ranges from 0.75 m^3 to 6.0 m^3 and the operation rate is from 20 to 30 cycles/hour, depending on the water depth and sediment characteristics.

A majority of sediment re-suspension caused by clamshell dredging is from impact, penetration and removal of the bucket from the bottom (Hayes *et al*, 1988, Barnard, 1978 and St. Lawrence Center, 1993a). Implementing operational controls and changing the bucket design can reduce the re-suspension of sediments during clamshell dredging operations. Operational controls can be applied by monitoring the speed of the bucket through the water column. The speed of the bucket should be kept below 0.6 m/s, in order to keep the sediment inside the bucket. Equipment design modifications include the use of enclosed clamshell buckets (McLellan *et al.*, 1989).

2.3.1.2 Dipper Dredge

The dipper dredge is a mechanical dredge that uses a shovel mounted on a pontoon. Dipper dredges are mainly used for dredging soft, broken rock and dense sediments. They are also used to remove old structures, dikes, and layers of rock (St. Lawrence Center, 1993a). The working depth of dipper dredges is generally not greater than 12 m. The bucket capacity varies and the operation rate is between 30 and 60 cycles/hour (St. Lawrence Center, 1993a).

2.3.1.3 Backhoe Dredge

The backhoe dredge removes sediments by pulling the bucket towards the dredge. Dredged material is collected afterwards on shore or in scows (St. Lawrence Center, 1993a). "The backhoe dredge can usually operate to a depth of about 12 m in a wide range of sediment type: gravel, coarse sand, cohesive sand and compact clay. Bucket capacity is generally 1 m^3 to 3 m^3. Losses of dredged material can be significant with this dredge, and for this reason it is rarely used to excavate fine sediment" (St. Lawrence Center, 1993a).

2.3.2 Hydraulic Dredges

"Hydraulic dredges remove and transport sediment in the form of a slurry. The slurries usually contain 10 to 20% solids by weight" (St. Lawrence Center, 1993a). The dredged material is transported over long distances from the place of dredging by loading the dredged sediments into barges or scows.

Hydraulic dredges are faster than mechanical dredges and they also cause less re-suspension at the dredging site. However, they are difficult to operate in rough waters, and debris can clog up suction lines and anchoring systems could interfere with navigation (St. Lawrence Center, 1993a). There are several kinds of hydraulic dredges, but the ones most commonly used in the St. Lawrence River will be discussed in the following sections.

2.3.2.1 Cutterhead Dredge

The cutterhead dredge is equipped with a rotating cutter at the intake end of the suction pipe (McLellan *et al.*, 1989). For different types of sediments different types of cutterheads can be used (St. Lawrence Center, 1993a).

Concentration of suspended sediments generated from a cutterhead dredging operation would range from 10 to 300 mg/l near the cutterhead to a few milligrams per liter at the distance of 300 to 600 m away from location of dredging (Barnard, 1978; Raymond, 1984; Hayes *et al.*, 1984). The amount of suspended solids created due to a cutterhead depends on different factors such as the thickness of cut, and the cutter rotation rate (Barnard, 1978). According to Hayes *et al.* (1984), a proper balance between the mechanical action of the cutter and the pickup ability of the pump could reduce sediment re-suspension.

2.3.2.2 Hopper Dredge

These dredges differ from other hydraulic dredges due to the fact that they are equipped with sediment containers (hoppers). Therefore, they would transport dredged sediment instead of dumping it into barges (St. Lawrence Center, 1993a and McLellan *et* *al.*, 1989). According to McLellan *et al.* (1989). Hopper dredge can dredge in depths from 3 m to over 30 m. However, the minimum dredging depth is restricted by the vessel draught (St. Lawrence Center, 1993a).

Hopper overflow is a practice that increases the capacity of the hopper by allowing excess water with low concentrations of suspended solids to overflow the hoppers and back into the water body while retaining settled solids in the hopper (St. Lawrence Center, 1993a). This practice is most effective when solids separate quickly, such as granular sediments (Hayes *et al.*, 1984). Re-suspension during hopper dredge operation is mainly caused by agitation of bottom sediment by the drag-arm, the action of the vessel's propeller and hopper overflow (St. Lawrence Center, 1993a). According to Barnard (1978), reducing the flow rate of the slurry being pumped into the hopper during the hopper-filling operation, would reduce the solids concentration in the plume.

2.3.2.3 Mudcat Dredge

A Mudcat dredge is a relatively small hydraulic machine that is mounted on a pontoon (St. Lawrence Center, 1993a). "Instead of the conventional cutter, the Mudcat has a horizontal cutterhead equipped with cutter knives and a spiral auger that cuts the material and moves it laterally towards the center of the auger where it is picked up by suction" (Herbich and Brahme, 1991).

The main advantage of the Mudcat dredge is its small size, which means it can be easily transported and can operate in inaccessible areas. The Mudcat can operate in shallow water up to 0.5 m, but cannot operate at depths greater than 6.1 m and its production rate is approximately 90 m^3/h. (St. Lawrence Center, 1993a).

Table 2.3 (Nakai, 1978) summarizes and compares the re-suspension of sediment due to some dredging operations for different types of sediment. The field studies show that re-suspension rates are generally higher with mechanical dredges than with hydraulic dredges. Type and fraction of dredged material and hydrological characteristics of the water body have important impacts on the amount of suspended solids released into the water column during the dredging process and their subsequent settling. As shown in Table 2.3 the suspension of clayey sediments is approximately 40 times more than sandy sediments. The adsorption/desorption process is strongly controlled and affected by a category of sediment characteristics, which in turn determines the number of available sites and interface contact area. Due to their high specific surface area and high cation exchange capacity, clays tend to adsorb high concentration of metals (Yong *et al.* 1992). The adsorbed metals could change phase to dissolved form and diffuse into the water column (Thomann and Mueller, 1987).

Table 2.3: Approximate suspended solids levels generated by different dredges (adapted from Nakai, 1978)

Type of Dredge	Installed power or bucket volume	Dredged material	Suspension (kg/m^3)*
Pump	4000 hp	Silty clay	5.3
		Sandy loam	1.4
		Clay	45.2
	2500 hp	Silty clay	9.9
		Sand	0.2
	2000 hp	Sand	0.1
Trailing suction	2400 hp	Silty clay loam	7.1
	1800 hp	Silt	25.2
Grab	8 m^3	Silty clay	89
	4 m^3	Clay	84.2
	3 m^3	Silty loam	11.9
Bucket		Sandy loam	55.8

*kg of suspended sediment per m^3 material dredged

Advantages and disadvantages of some dredging techniques described previously are summarized in Table 2.4; it shows that the main advantage of hydraulic dredges is associated with less amount of re-suspended material during dredging than mechanical dredges. However, sediments removed by mechanical dredges, keep density of material dredged close to their in-situ density. This situation would decrease the amount of sediments that must be transported and treated afterwards.

Table 2.4: Advantages and disadvantages of mechanical and hydraulic dredges (adapted from St. Lawrence Center, 1993a).

OPTIONS	ADVANTAGES	DISADVANTAGES
Mechanical dredges		
CLAMSHELL DREDGE DIPPER DREDGE BACKHOE DREDGE	* Maximum preservation of in-situ solids content of excavated material. * Minimum facilities required for transport treatment and disposal of dredged material * Almost unlimited dredging depth for clamshell dredges. * Can work safely close to wharves and other fixed structures. * Efficient for removal of contaminated sediment at shorelines or in flood plain * Precision work in shallow water	* Relatively low production rate decreases with depth (30 to 500 m^3/s) * Relatively high rate of sediment re-suspension into water column, especially during work in fine, noncohesive material. * Effectiveness low or nil in liquid sediment. * May hinder navigation. * Rehandling required when open water disposal is impossible
Hydraulic dredges		
CUTTERHEAD DREDGE	* High production rate * Lower rate of sediment re-suspension into water column than mechanical dredges. * Not limited by speed of currents. * Pipeline transport of contaminated sediment minimizes risks for workers and public. * Unit cost of removing large volumes of material is typically lower than with mechanical dredges.	* High water content of material removed (80% - 90%) * Hydraulic dredges are very sensitive to swells and waves. * Dredge and pipeline may hinder navigation. * Open-water disposal from the pipeline generates more turbidity than does dumping from barges loaded by mechanical dredges. * Gas in sediment can affect pump operation

Table 2.4: Advantages and disadvantages of mechanical and hydraulic dredges (adapted from St. Lawrence Center, 1993a) (con't).

OPTIONS	ADVANTAGES	DISADVANTAGES
Hydraulic dredges (cont)		
Hopper Dredge	* No interference with navigation. * Overflow allows for maximal use of hoppers * Not limited by speed of currents.	* High water content of material removed (80% - 90%) * More sediment re-suspension in water column at dredging site than with mechanical dredges * Less precise than other dredges * Limited in shallow waters.
Mudcat	* Small size and easy to transport * Leaves an even bottom	* Modest production rate * Maximum water depth of 6.1 m

2.4 Conclusion

The processes by which lead is released from bottom sediments, corroded sub-marine cables and dredged material are complex and governed by a substantial amount of factors. A literature review demonstrated that under undisturbed, near neutral pH and strongly reduced conditions in sediments, the sediments would immobilize the metal contaminants effectively.

Regarding the removal of sub-marine cables, both hydraulic and mechanical methods have to be considered where the hydraulic method itself is not sufficient. The manipulation of cables during the removal process can provoke re-suspension and/or loss of materials similar to mechanical dredging. Dredging activities would cause oxidation and acidification of bottom sediments, thereby causing the release of lead in both particulate and dissolved forms, into the upper water column. Re-suspension rates are generally higher with mechanical dredges than with hydraulic dredges. However, both hydraulic and mechanical dredging techniques could have an impact on the biota.

The literature review did not demonstrate the direct relationship between the existence and behavior of corrosion compounds (e.g. lead) and the removal of corroded sub-marine cables from bottom sediments using different dredging techniques. In addition, the impact of cable removal on water quality and biota was not demonstrated. It is therefore necessary to identify and classify the main factors affecting the release and fate of corrosion compounds (e.g. lead), and assess their impact on water quality and biota.

CHAPTER 3

MODELING OF FACTORS AFFECTING THE FATE OF CORRODED LEAD

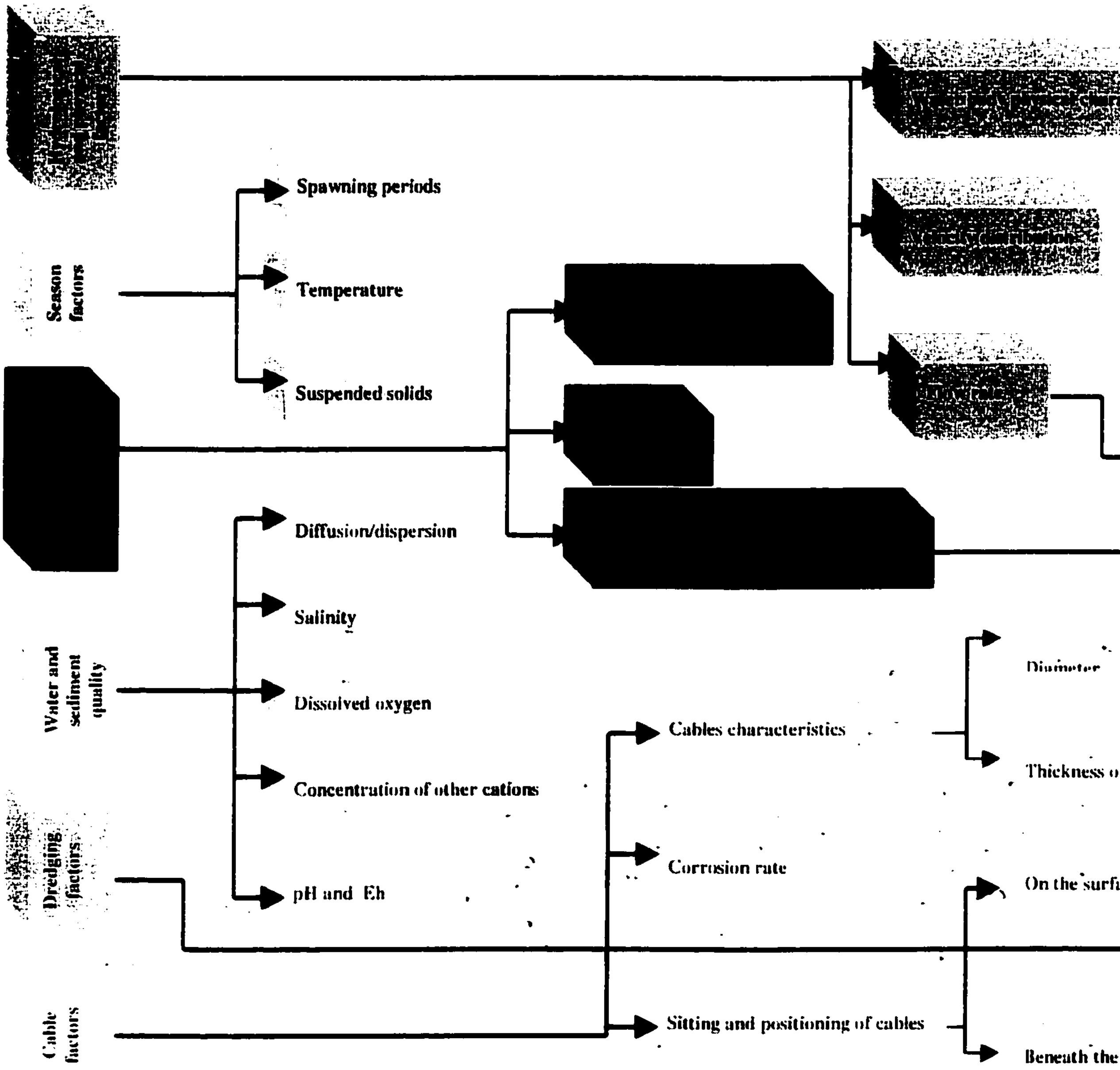
The fate of released corrosion products into water bodies is governed by an important number of factors. Classification of the most important factors that would affect the fate of corrosion products due to release, dredging of sediments and removal of cables was not performed. Therefore, this chapter includes identification, classification and description of factors affecting the fate of lead.

Among the most important, the following factors have to be considered in evaluation of the fate: hydrological and physical properties of the water body, sitting and position of cables, physico-chemical characteristics of the water column and sediments, and the impact of different dredging technologies.

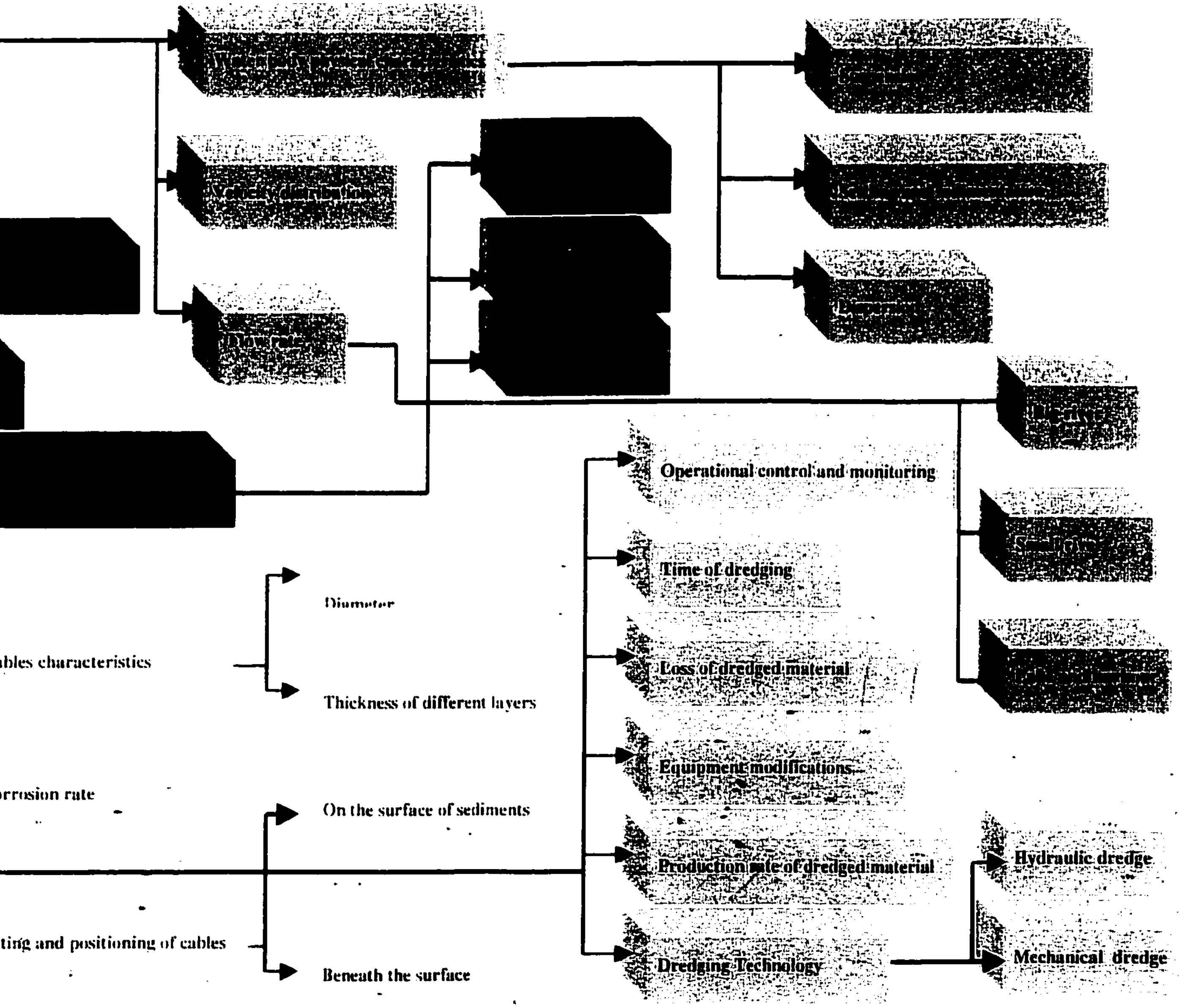
3.1 Classification of Factors Affecting the Fate of Corroded Lead

Based on a literature review and the knowledge of physico-chemical processes in the aquatic environment, the following groups of factors are identified as having the impact on the fate of lead. The most important factors may include: factors associated with the types of cables, factors associated with hydrological conditions of water bodies, physical characteristics of sediments, factors associated with seasonal changes, factors related to the chemistry of the water column and pore water in sediments and factors arising from the use of a particular type of dredging technology. These factors are grouped and presented in Figure 3.1.

Figure 3.1: Most important factors affecting the release and fate of corroded lead



nd fate of corroded lead



3.1.1 Factors Associated with Cables

Among the most important cable factors are:

- Cable characteristics;
    a) Components of sheaths (Chapter 2);
    b) Thickness of different sheaths (Chapter 2);
    c) Diameter and length of cables;
- Siting and positioning of cables;
    a) Positioning of cables on the surface of bottom sediments;
    b) Positioning of cables beneath the surface of sediments;
- Corrosion rate of cable components;

Cable factors would mainly affect the concentration of corrosion products due to their release into the aquatic environment.

3.1.2 Hydrological Factors

Physical and hydrodynamic characteristics of the water body have a large impact on dispersion, persistence and mixing of released components from corroded materials. The following factors should be included among the most important physical and hydrological factors:

- Flow rate (which varies with type of rivers, lakes, harbours and bays etc.);
- Spatial velocity distribution (which varies within the depth, and within different kinds of water bodies such as the bay area);
- Sediment re-suspension (due to the advective effect of the flow rate);
- Settling of suspended colloidal particles (Thomann and Mueller, 1987);

- Diffusion phenomena (which will occur across the sediment-water interface and between the different layers of the water column itself, due to the different concentration gradient) (Thomann and Mueller, 1987);
- Adsorption/desorption phenomena (Yong *et al.*, 1992);

3.1.3 Sediment Characteristics

Type and fraction of particles, type of mineral material, and fraction of organic matter in sediments have an important impact on the release and transport of corrosion products. The adsorption/desorption process is strongly controlled and affected by a category of sediment characteristics, which in turn determines the number of available sites and interface contact area.

Sediment characteristics, generally considered for the prediction of dispersed fate of corrosion products, include the following:

- Type of mineral and organic fractions which have an influence on:
  - Sorption of lead into the particulate suspension (Thomann and Mueller, 1987);
  - Desorption of lead from the particulate suspension into the water (Thomann and Mueller, 1987);
  - Ion exchange between lead and particulate suspension (Yong *et al.*, 1992);
- Porosity (which would differ according to the kind of sediments) (St. Lawrence Center, 1996);
- Cohesion and compaction (which would differ according to the kind of sediments) (St. Lawrence Center, 1996);

3.1.4 Water and Sediment Quality

The physico-chemical characteristics of water and sediment would affect the release speciation and toxicity of corrosion products in the water column. Among the main water and sediment quality factors are:

- pH;
- Oxidation-reduction potential (Eh);
- Concentration and mobility of other cations in sediments and water column (e.g. Fe. Mn, and heavy metals) (Chapter 2);
- Dissolved oxygen concentrations (Thomann and Mueller, 1987);
- Diffusion associated with concentration gradient (Thomann and Mueller, 1987);
- Presence of various anions (e.g. salinity of water);

3.1.5 Seasonal Factors

The following seasonal variations would affect the release and fate of contaminant by affecting other factors:

- Temperature (mostly variable in lakes and bays);
- Density of water (changes observed in lakes, bays, and harbors) (Thomann and Mueller, 1987);
- Amount of suspended solids in the water body (usually higher in spring season) (St. Lawrence Center, 1993b);
- Flow rate (higher in spring season) (St. Lawrence Center, 1993b);
- Spawning periods of aquatic species (Chapter 5);

3.1.6 Dredging Factors

Removal of cables from bottom sediments would probably require the use of different dredging methods. In the case of buried sub marine cables, hydraulic dredging could be used first to remove sediments at the level of the cables. Thereafter, some mechanical dredges would be used to pick up and remove this cable. The application of different dredge activities would enhance re-suspension, release and dispersion of corrosion products into the water column.

Among the main dredging factors that would affect water quality are:

- Type of dredges used:
    a) Mechanical dredging;
    b) Hydraulic dredging;
- Dredge capacity or production rate (including hydraulic dredges which have a higher production rate);
- Equipment modifications in order to reduce resuspension;
- Sediment loss during operations;
- Operation, monitoring and control of dredging;

Study demonstrated that more than 300 factors could directly or indirectly affect the water quality during dredging and cable removal activities.

3.2 Description of Model for Fate of Corroded Lead (*MFCL*)

In order to show inter-relationships between the factors classified in Chapter 3.1, computer modeling was required. The review of available simulation software shown in Table 3.1, demonstrated that no one software can adequately simulate the fate of compounds released due to the corrosion of sub-marine cables and their removal. Consequently, the creation of a new model to achieve these objectives was necessary.

The modeling has to include the most important factors shown in Chapter 3.1. To perform this modeling, the Model Maker package (Walker and Crout, 1997) which is a tool for developing numerical models was used. Whether the area of interest lies in chemistry, physiology, sociology, epidemiology, pharmacokinetics, ecology, mathematics or environmental kinetics the Model Maker package visualizes and tests different hypotheses, with functions such as optimization, sensitivity analyses, events, lookup tables, lookup files, sub models, delays, arrays, conditional and unconditional events. In this research the Model Maker package has been used to develop a new Model for Fate of Corroded Lead (*MFCL*), which can reflect a variety of aquatic scenarios of sub-marine cable corrosion and removal. This model could provide environmental engineers and toxicologists with an estimated view of contaminant distribution within a water body.

A particular objective of the modeling was to create a tool for the prediction of the fate of dissolved and particulate lead released from corroded sub-marine cables in scenarios that include the major factors identified in Chapter 3.1. The principal physico-chemical processes associated with the previously described factors and included in the *MFCL* model are:

1- Sorption and desorption of dissolved and particulate forms of lead in the water column and sediment.

2- Settling and resuspension mechanisms of particulates at the interface of the water column and sediment, and settling in the water column itself.

Table 3.1: Description of some simulation softwares for aquatic environment

N	Simulation models	Application	Interface to other software	Environmental parameters				Speciation of contaminant	Application to metallic contamination	Impact of hydrological parameters	Abiotic transformation	Biotic transformation
				pH	T	O_2	CO_2					
1	OTIS, One-Dimensional Transport with Inflow and Storage	Quantifies hydrologic transportation process (advection, dispersion, transient storage), simulate the transport of solutesin which one-dimensional transport may be assumed hydrologic and hydraulic characteristics of the stream	MINTAQ and OTEQ	N	N/D	N	N	N	N/D	Y	N/D	N
2	OTEQ, equilibrium-based transport model	Hydrology, water quality. Combines OTIS and a chemical equilibrium submodel	MINTAQ, OTIS	Y	N/D	N/D	N/D	N/D, sorbtion desorbtion	Y	Y	Y	N
3	BRANCH, Branch-Network Dynamic Flow model	Simulates steady or unsteadyflow in a single open-channel reach (branch) or throuhout a system of branches, time varying water levels, flow discharges, computed velocities and volumes can be calculated at any location within the the open-channel network	MODBRNCH (USGS, MODFLOW)	N	N	N	N	N	N	Y	N	N
4	CE-QUAL-W2, hydrodynamic and water quality model for rivers, estuaries, lakes and reservoirs	Hydrodynamic, water quality	N/D	N/D	N/D	Y	N	N	N/D	Y	N	N

Legend: N-No, Y-Yes, N/D-No Data

Table 3.1: Description of some simulation softwares for aquatic environment (con't)

N	Simulation models	Application	Interface to other software	Environmental parameters				Speciation of contaminant	Application to metallic contamination	Impact of hydrological parameters	Abiotic transformation	Biotic transformation
				pH	T	O_2	CO_2					
5	CE-QUAL-W2, application to Rhodhiss Lake	Simulates movement and mixing of spills or releases in the reservoir and reservoir nutrient (NH3, PO4), algal and DO concentrations in response to possible changes in external loadings and flows	N/D	N	Y	Y	N	N	Y (Fe only)	Y	N	N
6	CE-QUAL-W2, application to Thalatin river, Oregon	Hydrodynamic, water quality	N/D	N/D	N/D	Y	N	N	N	Y	N	Y
7	MINTAQA2, geochemical equilibrium speciation model for dilute aqueous systems	Water quality	WATEQ, PRODEFA	Y	N/D	Y	Y	Y	Y	N	Y	N
8	QUAL 2E, enhanced stream water quality model	Waste load allocations, discharge permit determinations and other pollutant evaluations in the Unated States	Y	N/D	Y	Y	N	N	N/D	Y	N	N
9	WASP, water quality analysis simulation program	Modeling contaminant fate and transport in surface waters	N/D	N/D	N/D	Y	N	N	Y	Y	N/D	N/D
10	AGNPS, Agricultutral Nonpoint Source Solution model	Analysis of large agricultural watersheds, sediment yields	GIS	N	N/D	N/D	N/D	N	N/D	N/	Y(N, P, for others - N/D	N

Legend: N-No, Y-Yes, N/D-No Data

3- Diffusive exchange between the sediment and the water column, and between the layers of the water column.

4- Movement of lead due to advective transport and dispersive mixing.

3.2.1 Processes in the Aquatic Environment Simulated by the *MFCL* Model

The mass balance of the total lead concentration in the water column, c_{T1}, is given by Thomann and Mueller (1987):

$$V_1 \, dc_{T1}/dt = K_f A \, (f_{d2} \, c_{T2}/\phi_2 - f_{d1} \, c_{T1}) - v_s \, A f_{p1} \, c_{T1} + v_u \, A f_{p2} \, c_{T2} \quad \text{..........} \; (1)$$

- $V_1 \, dc_{T1}/dt$: Time rate of change of lead concentration in the water column.
  - $K_f A \, (f_{d2} \, c_{T2}/\phi_2 - f_{d1} \, c_{T1})$: Diffusion exchange of dissolved lead between the sediment and the water column, where:
    - K_f : Diffusion coefficient (m/day).
    - A : Interfacial area between the sediment and the water column (m^2).
    - f_{d2} : Dissolved fraction of lead in sediments.
    - f_{d1} : Dissolved fraction of lead in the water column.
    - ϕ_2 : Porosity of sediments.
    - c_{T1}, c_{T2} : Total concentration of lead in the water column and sediments respectively (mg/L).
  - $v_s \, A f_{p1} \, c_{T1}$: Settling of the particulate lead from the water column to the sediment at velocity v_s.
    - v_s : Settling velocity (m/day).

- $v_u A f_{p2} c_{T2}$: Resuspension into the water column of the particulate lead from the sediment at velocity v_u.
    - v_u : Resuspension velocity (m/day).

For the stationary sediment underlying the moving water column, the mass balance equation is (Thomann and Mueller, 1987):

$$V_2\, dc_{T2} / dt = - K_f A\, (f_{d2}\, c_{T2} / \phi_2 - f_{d1}\, c_{T1}) + v_s A f_{p1}\, c_{T1} - v_u A f_{p2}\, c_{T2} - v_d A f_{p2}\, c_{T2} \quad \text{..........} (2)$$

- $v_d\ A\ f_{p2}\ c_{T2}$: The loss of toxicant from the sediment due to net sedimentation (which was assumed to be zero, in order to consider the worst case where most of the lead is released into the water column).
    - v_d : Sedimentation velocity (m/day).

The mechanism of sorption and desorption of lead is represented by the following equations (Thomann and Mueller, 1987):

$$dc_p/dt = K_u\, m\, c'_d - K\, c_p \quad \text{..........} (3)$$

$$\phi\, dc'_d/dt = - K_u\, m\, c'_d + K c_p \quad \text{..........} (4)$$

- c_p : Particulate concentration of lead $[\, M_T / L_{w+s}^3]$
- c'_d : Dissolved concentration of lead $[\, M_T / L_w^3]$

$$c_T = c_p + \phi\, c'_d \text{..........} (5)$$

$$c_T = c_p + c_d \text{..........} (6)$$

- Where c_d $[M_T / L_{w+s}^3]$ is the dissolved lead concentration corrected for porosity.

The fraction of the total concentration, f_d is as follows:

$$f_d = 1/(1+\P\,\mathrm{m}) \text{ .... } (7)$$

- m: Total mass of suspended solids.

Particulate lead concentration as a fraction of total concentration, f_p as follows:

$$f_p = \P\,\mathrm{m} / (1+\P\,\mathrm{m}) \text{ .... } (8)$$

$$f_p + f_d = 1 \text{ .... } (9)$$

3.2.2 Algorithm of the MFCL Model

The algorithm of the *MFCL* model is shown in Figures 3.2, 3.3 and 3.4. Figure 3.2 shows that the water body is divided into three compartments (a) the upstream compartment, presented as "up_condition", (b) the cable position compartment presented as "Cable_Position", and (c) the downstream compartment presented as "Sub1". These compartments contain further sub-compartments. Figure 3.3 shows that the downstream compartment "Sub1" itself, is divided into four different sub-models. Each sub-model represents a different location in the downstream, shown as "Sub1", "Sub2", "Sub3", and "Sub4". Figure 3.4 shows that at each location of the water body, the water column was divided into three different layers with different depths, and each layer was divided into particulate and dissolved concentrations of lead. Each layer of the water column is presented as a different compartment and the transfer of lead between the compartments is shown as arrows. These arrows represent the effect of different phenomena on each compartment. Other compartments such as "up_condition" and "Cable_Position" include sub-models similar to the one shown in Figure 3.4. Variables presented in the model (Figure 3.2 and 3.3) are considered either as global or universal components, where global components can be referenced by any other component in the same sub-model. The use of global components avoids having lots of influences.

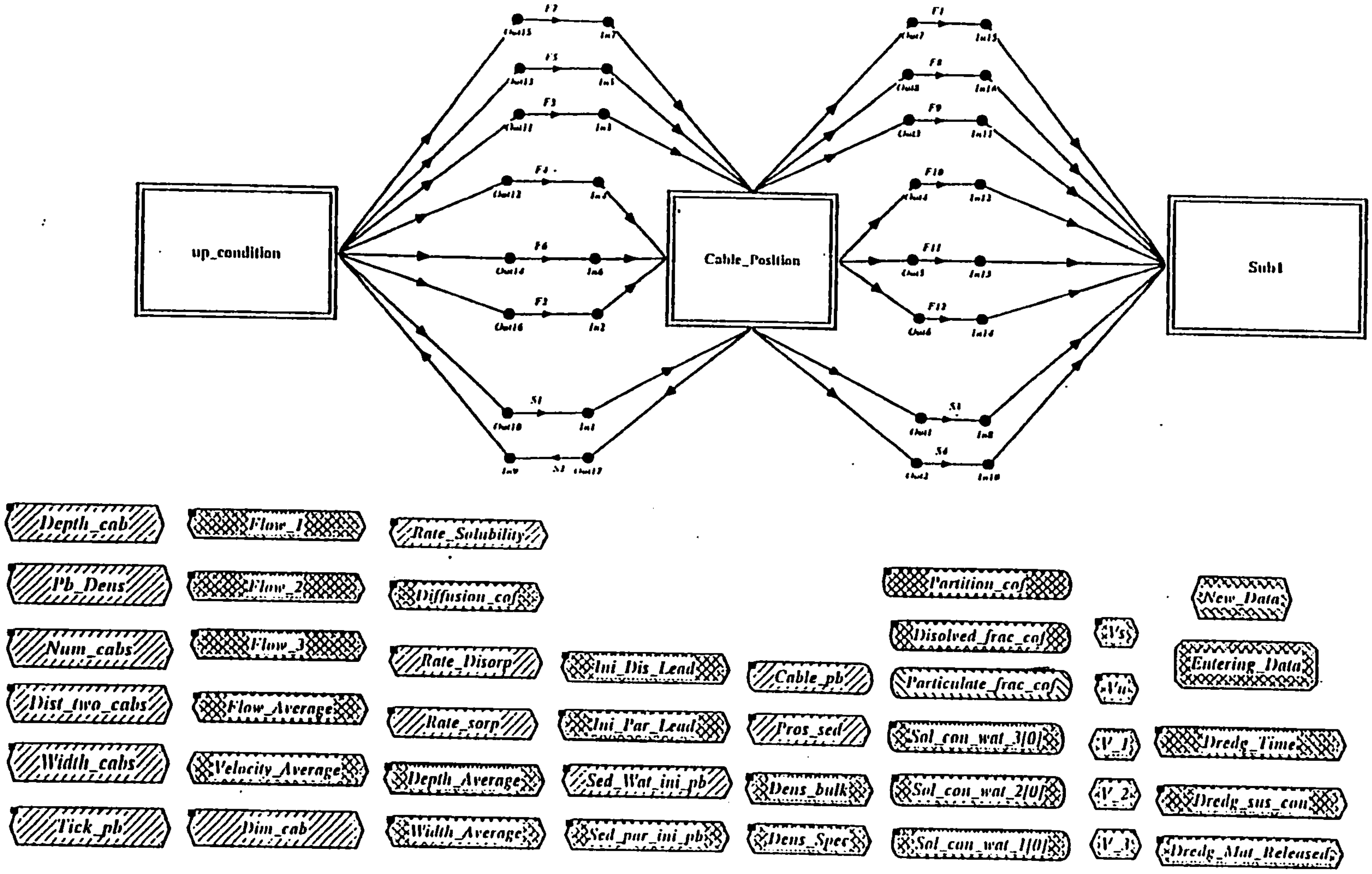


Figure 3.2: Algorithm of the *MFCL* model (upstream, cable position, and downstream) including the different global and universal variables.

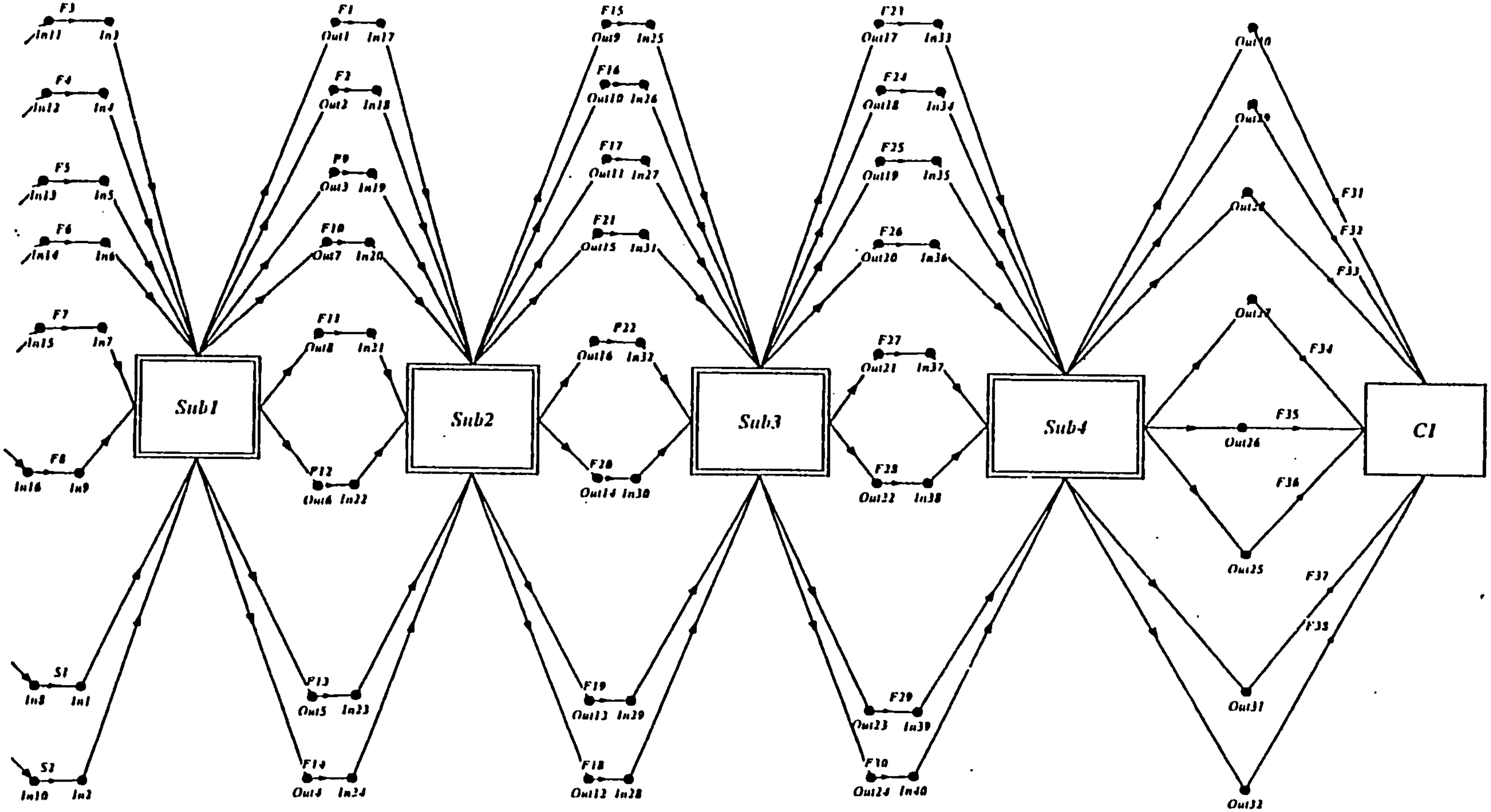


Figure 3.3: Algorithm of the *MFCL* model, downstream sub-models.

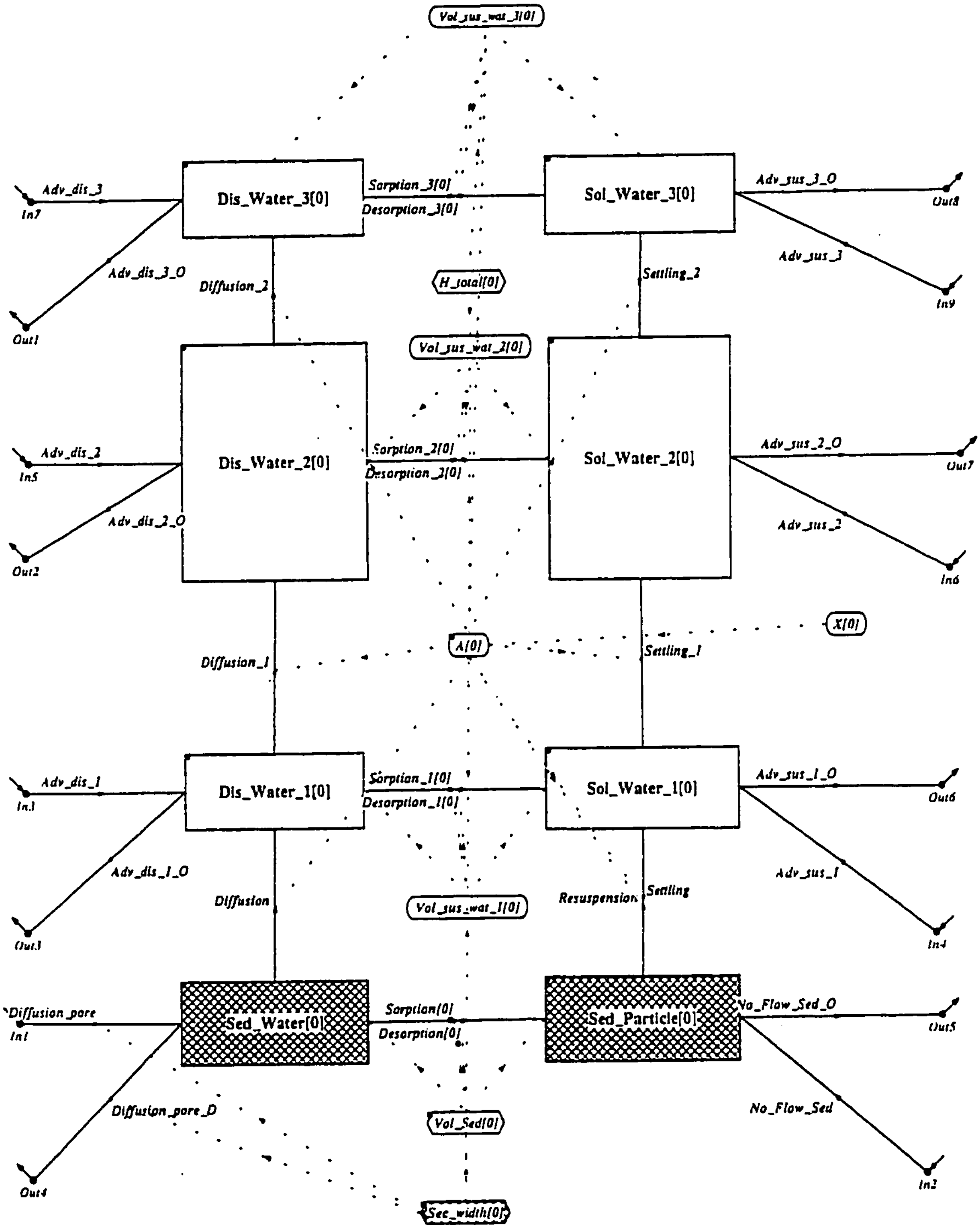


Figure 3.4: Algorithm of the MFCL model, stratification of water column in cable position and different downstream locations.

Universal components can be referenced by any other component in the entire model. Making the component universal avoids having to create many sub-model influences.

Different variables included in the *MFCL* model are shown in Figure 3.2. For example (a) cable variables: depth of cables "*Depth_cab*", number of cables "*Num_cabs*", distance between cables "*Dist_two_cabs*", length of cables "*Width_cabs*", diameter of cables "*Dim_cab*", and thickness of lead sheath "*Thick_pb*"; (b) hydrological and physical variables: average flow rate "*Flow_Average*", flow rate in the bottom layer "*Flow_1*", flow in the middle layer "*Flow_2*", flow in the upper layer "*Flow_3*", average velocity "*Velocity_Average*", depth of water body "*Depth_Average*", and width of water body "*Width_Average*"; (c) lead characteristics: lead density "*Pb_Dens*", rate of desorption "*Rate_Disorp*", rate of sorption "*Rate_sorp*", initial particulate lead concentration in the water column "*Ini_Par_Lead*", initial dissolved lead concentration in the water column "*Ini_Dis_Lead*", initial dissolved concentration of lead in sediments "*Sed_Wat_ini_pb*", initial particulate concentration of lead in sediments "*Sed_par_ini_pb*", partition coefficient of lead "*Partition_cof*", dissolved fraction coefficient "*Dissolved_frac_cof*", and particulate fraction coefficient "*Particulate_frac_cof*"; (d) sediment characteristics: porosity of sediments "*Pros_sed*", bulk density "*Dens_bulk*", and specific density "*Dens_spec*"; (e) dredging factors: dredging time " *Dredge_Time*", and solids release due to dredging "*Dredg_sus_con*" in addition to those variables such as settling velocity "*Vs*", suspension velocity "*Vu*", and volume of water body "*V*" were included.

Every compartment and arrow presented in each model and sub-model (Figures 3.2, 3.3, and 3.4) includes one of the equations introduced in Chapter 3.2.1.

These equations were introduced into the model in both ways: (a) as equations incorporated into each compartment; (b) as inter compartment equations introduced into the arrows between the compartments. For example Figure 3.5 shows the compartment "Sol_Water_1[0]" which represents the concentration of particulate lead in the bottom layer of the water column. The inter compartment equations introduced into the arrows in this case can be summarized as follows:

Settling_1 ↓ $= v_s \, A \, f_{p1} \, c_{T1}$

v_s: Settling velocity (m/day)

A: Interfacial area between the sediment and the water column (m^2)

f_{p1}: Particulate fraction of lead in water column.

c_{T1}: Total concentration of lead in the above compartment "Sol_Water_2[0]" (mg/L).

Settling ↓ $= v_s \, A \, f_{p1} \, c_{T1}$

c_{T1}: Total concentration of lead in the same compartment "Sol_Water_1[0]" (mg/L).

Resuspension ↑ $= v_u \, A \, f_{p2} \, c_{T2}$

v_u : Resuspension velocity (m/day)

A: Interfacial area between the sediment and the water column (m^2)

f_{p2}: Particulate fraction of lead in sediments.

c_{T2}: Total concentration of lead in sediments (mg/L).

Sorption_1[0] → $= K_u \, m \, c'_d$

K_u: Adsorption coefficient (μg/g).

m: Concentration of solids (mg/L).

c'_d: Dissolved concentration of lead (mg/L).

Desorption_1[0]← = K c_p

K: Rate of desorption (1/day).

c_p: Particulate concentration of lead (mg/L).

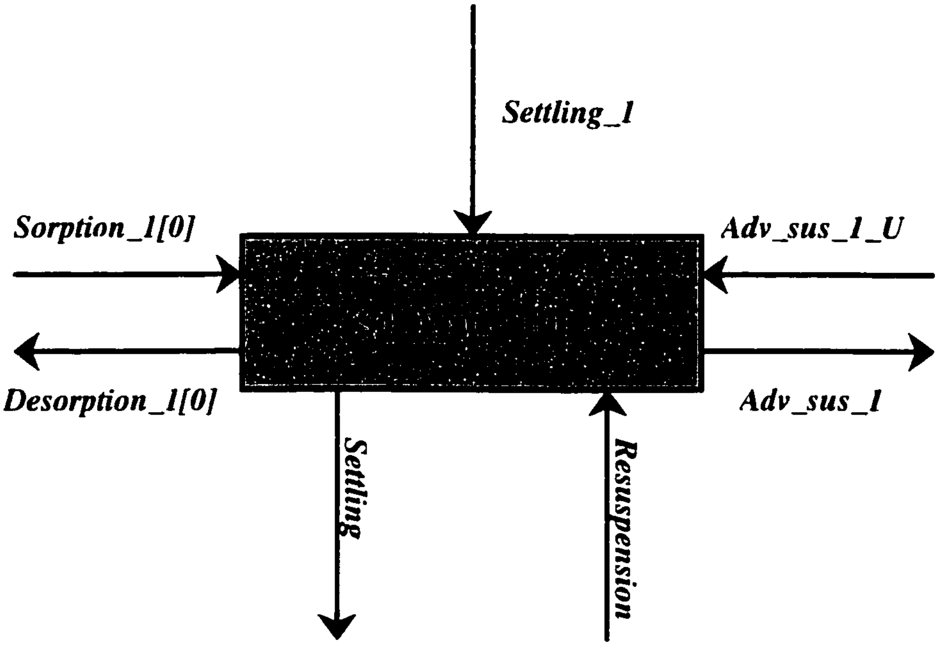


Figure 3.5: Micro-compartment of the *MFCL* model

Adv_sus_1_U← = Q c_p

Q: Flow rate in the bottom layer (m^3/day).

c_p: Particulate concentration of lead in the up stream (mg/L).

Adv_sus_1→ = Q* c_p

Q: Flow rate in the bottom layer (m^3/day).

c_p: Particulate concentration of lead in the same compartment (mg/L).

Consequently, the equation incorporated into the compartment will be as follows:

$$\textbf{Sol\_Water\_1[0]} = \textit{\textbf{Settling\_1}} - \textit{\textbf{Settling}} + \textit{\textbf{Resuspension}} + \textit{\textbf{Sorption\_1[0]}} - \textit{\textbf{Desorption\_1[0]}} + \textit{\textbf{Adv\_sus\_1\_U}} - \textit{\textbf{Adv\_sus\_1}}$$

Other equations were incorporated into the other compartments and arrows in the same procedure.

3.3 Input to the *MFCL* Model

Hydrological, environmental and ecological conditions of the St. Lawrence River around the Montreal area were introduced into the *MFCL* model. This area, due to the presence of industry and a high population distributed on both banks of the river, has to have a potential localization of sub-marine cables.

Data entered into the model was collected from various available publications. Unfortunately, no one publication included all parameters related to the area of interest. The list of parameters required as input to the model are summarized as follows:

1- Information about cables
  - Number of communication cables (assumed to be 10 cables).
  - Diameter of communication cable (assumed to be 40 cm).
  - Thickness of lead sheath between 1.3 and 3.2 mm (According to Schubert (1985)).
  - Lead density (13000 kg/m^3).
  - Depth of cables (assumed to be 1 and 5 m).
  - Length of cables (assumed to be the same as river width).
  - Distance between each two cables (assumed to be 50 cm).

2- Information about water column

- 0.000034 mg/L, initial concentration of dissolved lead (after St. Lawrence Center, 1996).
- 0.00054 mg/L, initial concentration of particulate lead (after St. Lawrence Center, 1996).
- Settling velocity (v_s), 5.0 m/day (assumed after Di Toro and Nusser, 1976).
- Re-suspension velocity calculated from the following equation (Thomann and Mueller, 1987):

$$v_u = v_s\,(m_1/m_2) - v_d \quad \text{............} \ (12)$$

Where:

v_u: Re-suspension velocity (m/day).

v_s: Settling velocity (m/day).

v_d: Sedimentation velocity (m/day) (assumed to be zero, in order to consider the worst case, when most of the lead is released into the water column).

m_1: solid concentration in the water column. The average solid concentration around the Montreal area is equal to 9.0 mg/L (after St. Lawrence Center, 1996).

3- Information about sediments

- Sediments in the Montreal area are mostly gravel, sand and silt material since this area is considered as a transport and erosion zone (Loiselle *et al.*, 1997).

- 0.000034 mg/L initial concentration of dissolved lead in pore water sediments (after St. Lawrence Center, 1996).
- 14.4 mg/kg initial concentration of particulate lead (after St. Lawrence Center, 1996).
- m_2: solid concentration in sediments, 1060000 g/m^3. (after Di Toro and Nusser, 1976).
- Sediment-water diffusion transfer coefficient is between 0.1 and 1.0 cm/day (assumed after Thomann and Mueller, 1987).
- Partition coefficient ($\P$) is equal to 10^5 L/kg (assumed after Thomann and Mueller, 1987).
- Adsorption coefficient, $K_u = 0.04$ µg/g sediment (assumed after Thomann and Mueller, 1987).
- Desorption rate (K), Calculated from the following equation (Thomann and Mueller, 1987):

  $$K = K_u/\P^{'} \quad .... (13)$$

  Where:

  $\P^{'}$: Corrected partition or distribution coefficient of lead at equilibrium between the particulate and the dissolved form, which is equal to:

  $$\P^{'} = \P * \phi \, ... (14)$$

  K = 5E-5 1/day.

  ϕ : Porosity of sediments (ϕ), 0.7- 0.8 (according to Thomann and Mueller, 1987).
- Bulk density (ρ_b) (1.06 g/ cm^3).

- Specific density (ρ_s) (2.6 – 2.75 g/ cm^3).

4- Physical and hydrological characteristics
  - Flow rate; average flow rate of the St. Lawrence around the Montreal area is 9000 m^3/s (St. Lawrence Center, 1996). This flow rate was divided into three different values, following the vertical velocity distribution within the depth:
    - Bottom layer: flow rate was assumed to be 5% of the total flow rate.
    - Middle layer: flow rate was assumed to be 65% of the total flow rate.
    - Upper layer: flow rate was assumed to be 30% of the total flow rate.
  - Width of the river, 300 m close to the old port (according to St. Lawrence Center, 1996).
  - Depth of the water column, 9.0 m (after St. Lawrence Center, 1996). The water column was divided into three parts following the velocity distribution in the water body:
    - Layer-1: bottom layer, which is 1 m high.
    - Layer-2: middle layer, 6 m high.
    - Layer-3: upper layer, 2 m high.
  - Suspended solids concentration. The average concentration around the Montreal area is 9 mg/L (according to St. Lawrence Center, 1996).
  - Flow velocity in the Montreal area generally ranges from 0.15 - 0.30 m/s (St. Lawrence Center, 1996).

5- Dredging parameters
  - Kind of dredging activity (hydraulic or mechanical).

- Amount of suspension during dredging:

  a) Mechanical dredging: 89 kg of sediments suspended for each m^3 of dredged sediments (assumed after Nakai, 1978).

  b) Hydraulic dredging: 23 kg of sediments suspended for each m^3 of dredged sediments (assumed after Nakai, 1978).

- Time of dredging. Assumed to be 7 hours per one shift or dredging for two consecutive shifts.

Parameters introduced into the *MFCL* model were divided into two categories, defined values and variables. The key characteristic of defined values is that they are calculated only at the start of the model run, or in response to the actions of an event, making them useful for holding values, which do not change very often. Table 3.2 summarizes the list of defined values that were introduced into the *MFCL* model. Variables are values, which are entered by the user at the beginning of each run to simulate different cases. Table 3.3 summarizes the list of variables introduced by the user of the *MFCL* model.

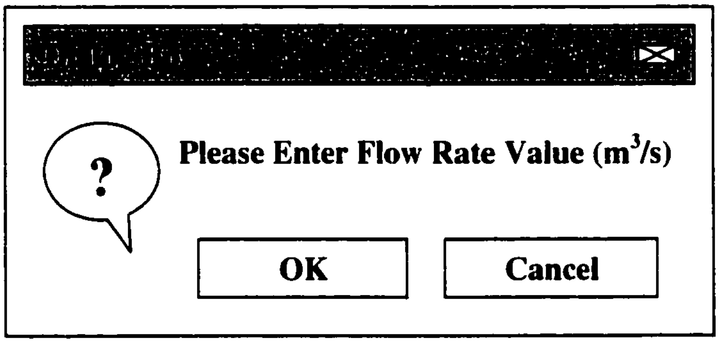
Table 3.2: Summary of defined values entered into the *MFCL* model

Defiened Values Entered to the *MFCL* Model	**Value**	**Reference**
Adsorption coefficient (Ku)	0.04 µg/g	Thomann and Mueller (1987)
Bulk density (ρ_b)	1.06 g/cm^3	Assumed
Desorption coefficient (K)	5.0E-5 1/day	Thomann and Mueller (1987)
Initial concentration of lead in pore water sediment	3.4E-5 mg/L	St. Lawrence Center (1996)
Initial concentration of particulate lead in sediments	14.4 mg/kg	St. Lawrence Center (1996)
Initial dissolved lead concentration	3.4E-5 mg/L	St. Lawrence Center (1996)
Initial particulate lead concentration	5.4E-4 mg/L	St. Lawrence Center (1996)
Lead density	13000 kg/m^3	Assumed
Partition coefficient (¶)	10E5 L/kg	Thomann and Mueller (1987)
Porosity of sediments (ϕ)	0.8	Thomann and Mueller (1987)
Re-suspension velocity (Vu)	4.25E-5 m/day	Thomann and Mueller (1987)
Settling velocity (Vs)	5.0 m/day	Di Toro and Nusser (1976)
Suspended solids concentration in water column	9 mg/L	St. Lawrence Center (1996)
Solids concentration in sediments	10.6E5 g/m^3	Di Toro and Nusser (1976)
Sediment-water diffusion transfer coefficient (K_f)	0.55 cm/day	Thomann and Mueller (1987)
Specific density (ρs)	2.65 g/cm^3	Assumed
Suspension due to clamshell dredge (mechanical dredge)	89 kg/m^3	Nakai (1978)
Suspension due to cutterhead dredge (hydraulic dredge)	23 kg/m^3	Nakai (1978)

Table 3.3: Summary of variables entered into the *MFCL* model.

Variables Entered to the *MFCL* Model	**Value**	**Reference**
Number of cables	10	Assumed
Diameter of cables	40 cm	Assumed
Thickness of lead sheath	3.2 mm	Schubert (1985)
Distance between cables	50 cm	Assumed
Flow rate		
High	9000 m^3/s	St. Lawrence Center (1996)
Medium	30 m^3/s	St. Lawrence Center (1996)
Low	1.5 m^3/s	Assumed
Depth of water column	9 m	St. Lawrence Center (1996)
River width	300 m	St. Lawrence Center (1996)
Duration of one shift of dredging	7 hours	Assumed

Data mentioned in Table 3.3 can be introduced by the user using a message box containing the specified formatted text shown in Figure 3.6. If the entered value satisfies the given validation condition, it is assigned to the defined component.



Please Enter Flow Rate Value (m^3/s)

Figure 3.6: Message Box in the *MFCL* model

3.4 Running the *MFCL* Model

Once the model is completed, the *MFCL* model is ready to run. Running the model is the process of solving the incorporated equations through the desired range of time. Independent variable of the *MFCL* model is time where all equations are solved for various introduced values with respect to time. Package of the Model Maker manual (Walker and Crout, 1997) can be always helpful in the run of simulated models such as *MFCL* model. The run option dialog box permits to choose start value and desired output steps. A brief description of the running process is described as follows: The run calculations would start a two-stage process:

(1) The model is compiled. The equations of the model are checked to make sure that they are defined correctly, and that *MFCL* model can perform the necessary calculations. If a problem is found, an appropriate error message is displayed.

(2) The calculations are carried out by the model taking into consideration both defined values and variables entered by the user. When calculations are complete, a message appears stating how long the calculation has taken (for the *MFCL* model usually each run took around 7 hours to be completed).

After the calculations are performed by the model, the dialog box permit to choose the form of the outputs either graphs or tables. Customizing both graphs and tables is accessible and easy to change

3.5 Study Scenarios

Above described model served to perform a series of runs to find out the most important relations between identified factors in Chapter 3.1. Cables are crossing large and small rivers, lakes, bays, and channels. In all of these cases, it is expected that corrosion products would have a different fate and effect on aquatic biota. Subsequently, the *MFCL* model has been used for a spectrum of water bodies with small and larger urban agglomerations.

For the positioning of cables as associated with the dynamics of sediments, where some cables are found up to the depth of 5 meters, an investigation has to be done for several positions of the cables. As was mentioned in Chapter 2 the concentration of the corroded lead around the cables is a function of time and redox conditions; hence, an investigation has to be done for different corrosion rates.

The situation of cables, dredging activities, formation of sediment suspension, and fate of dissolved and particulate compounds strictly depend on the hydrological and physical characteristics of water bodies. These characteristics also influence the type and metabolism of aquatic biota. Consequently, the specific hydrological factors can play a crucial role in the fate of contaminants associated with dredging activities and their impact on biota. Therefore, an investigation has to be done for different flow rates.

Different cases were introduced to the model to see the effect of particular factors. The inter-relationship of factors such as flow rate, cable positioning, corrosion rate and dredging (removal) operations has been estimated.

The following case studies were introduced to the *MFCL* model:

1) Three case studies considered the impact of flow rate variation (Figure 3.7):
    a) High flow rate, simulation of large rivers such as the St. Lawrence River (Case Study-1).
    b) Medium flow rate, simulation of small rivers such as the Chateauguay River (Case Study-2).

    c) Low flow rate, simulation of bays and harbors (Case Study-3).

2) Three case studies considered the impact of cable position (Figure 3.8):
    a) Cables on the surface of sediments (Case Study-4).
    b) Cables at 1 meter depth (Case Study-5).
    c) Cables barred at 5 meter depth of sediments (Case Study-6).
3) Other case studies were chosen to show the impact of corrosion rate. Three case studies were investigated (Figure 3.9) assuming that:
    a) The total amount of lead was released into the water column in the case of full corrosion of cables (Case Study-7).
    b) 50% of total lead was released into the water column in the case of partial corrosion of cables (Case Study-8).
    c) No corrosion occurred (Case Study-9).
4) Other case studies were applied to compare different dredging systems (Figure 3.10):
    a) Mechanical dredging (Case studies 10 and 12).
    b) Hydraulic dredging (Case Study-11).
    c) Two consecutive days of dredging (Case Study-13).

Above mentioned cases were introduced to the *MFCL* model to obtain dissolved and particulate lead distribution in different layers of the water column and at different cable positions versus time. The model was also used to calculate the time needed to reach the pre-disturbed state. Subsequently, the *MFCL* model allowed for governing the factors influencing the fate of corrosion products.

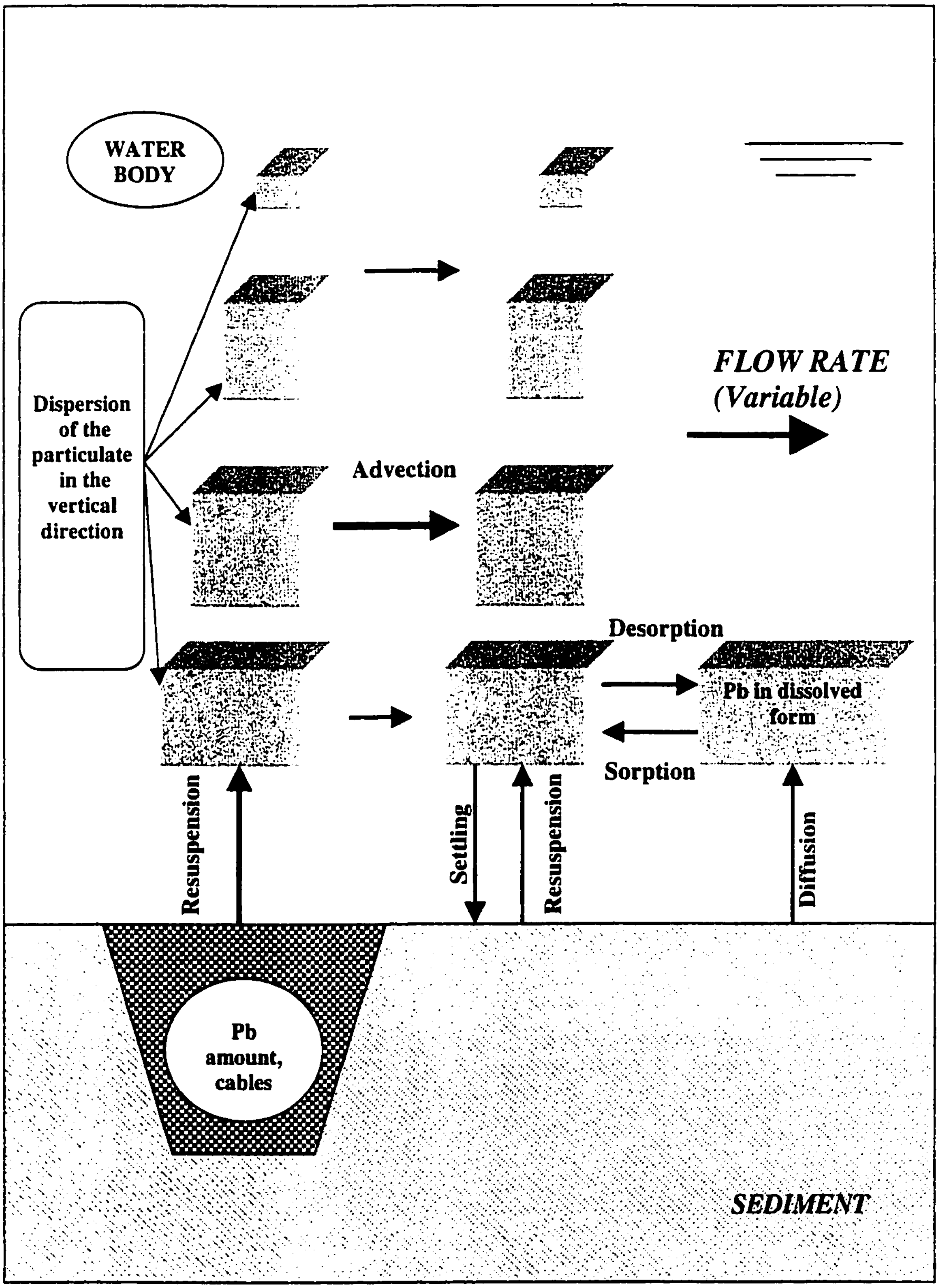


Figure 3.7: Simulation of the impact of flow rate on Pb fate in water column by means of MFCL model (Case Studies: C1, C2 and C3).

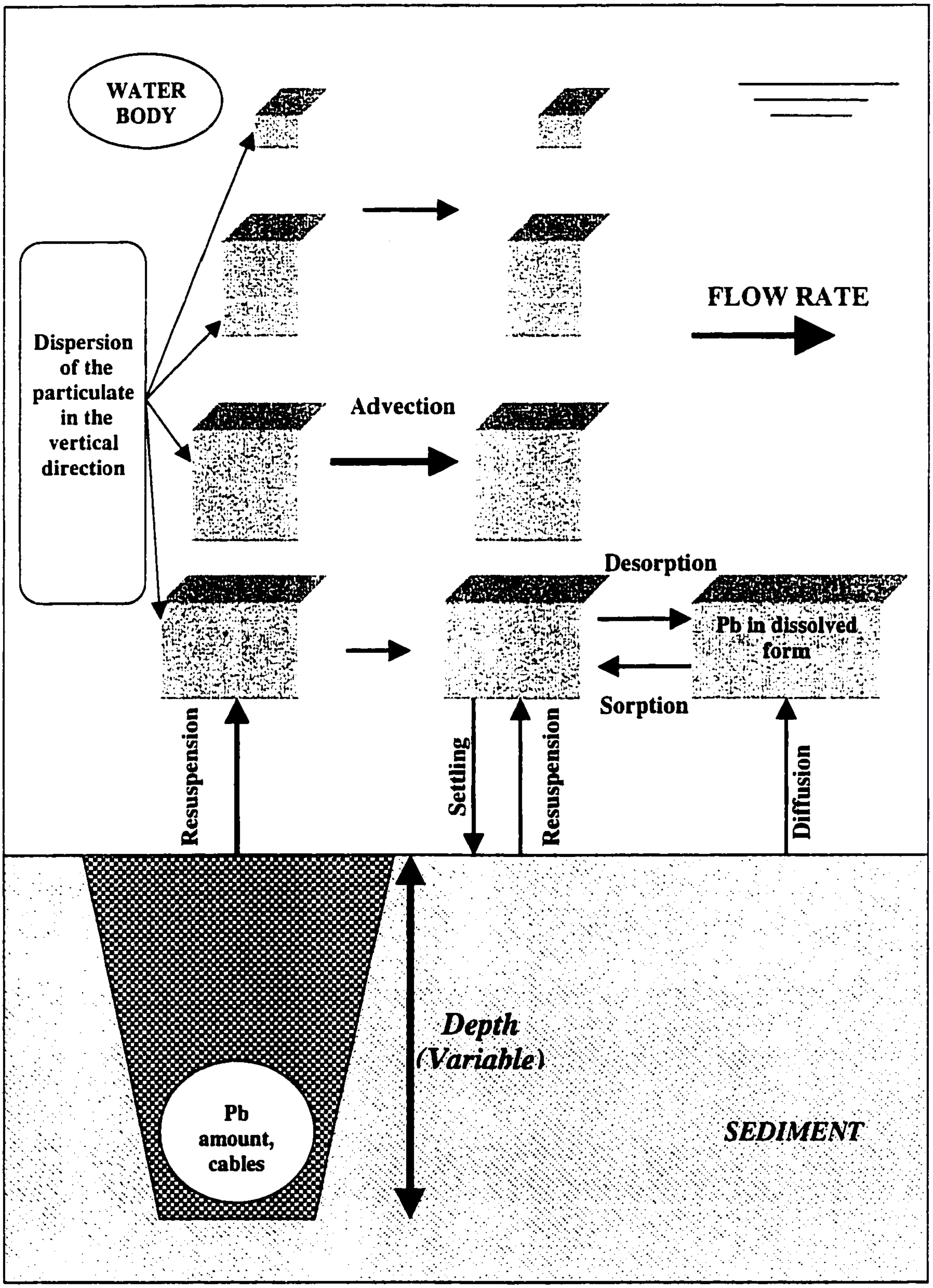


Figure 3.8: Simulation of the impact of cable position on Pb fate in water column by means of MFCL model (Case Studies: C4, C5 and C6).

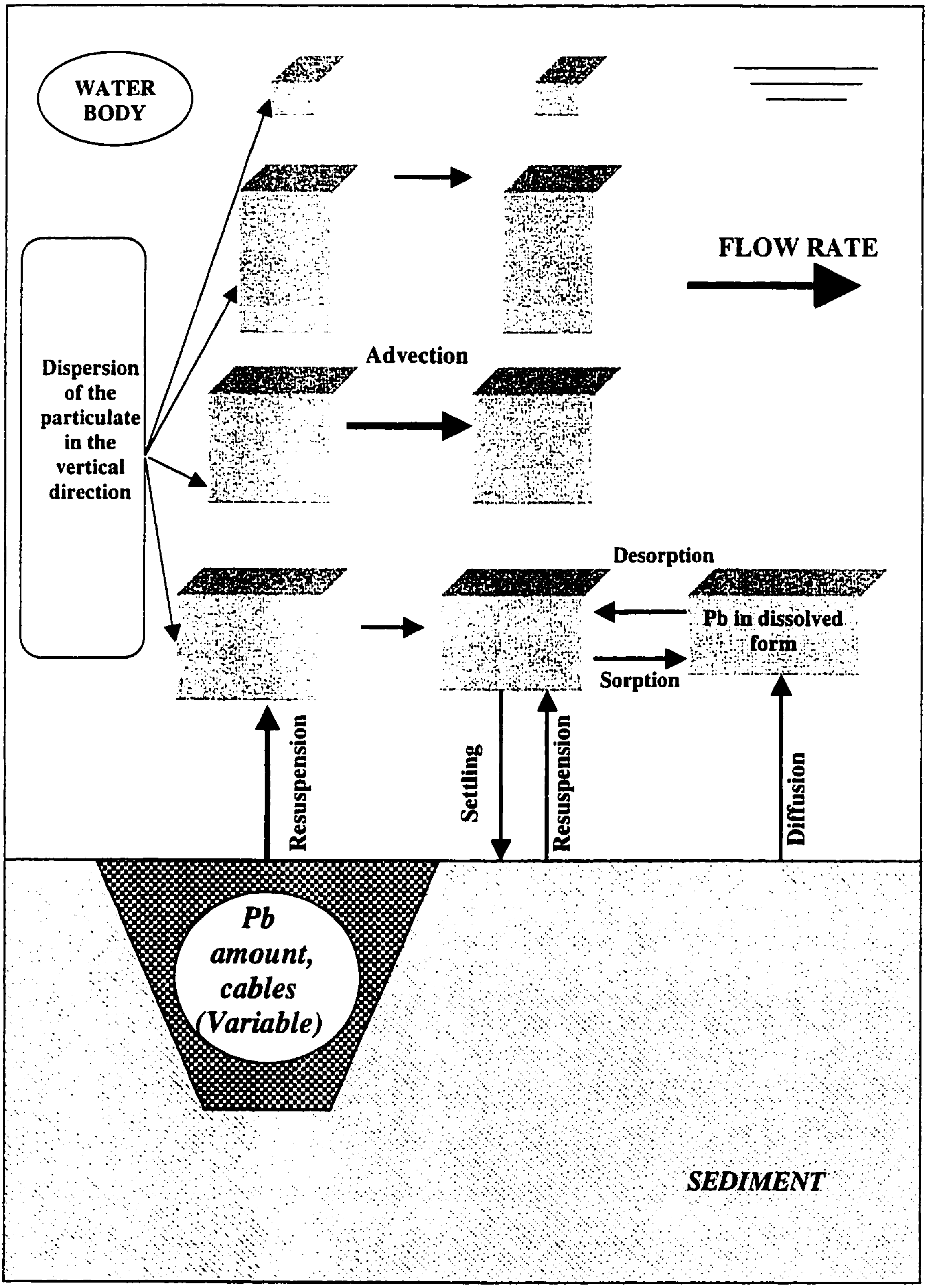


Figure 3.9: Simulation of the impact of corrosion rate on Pb fate in water column by means of MFCL model (Case Studies: C7, C8 and C9).

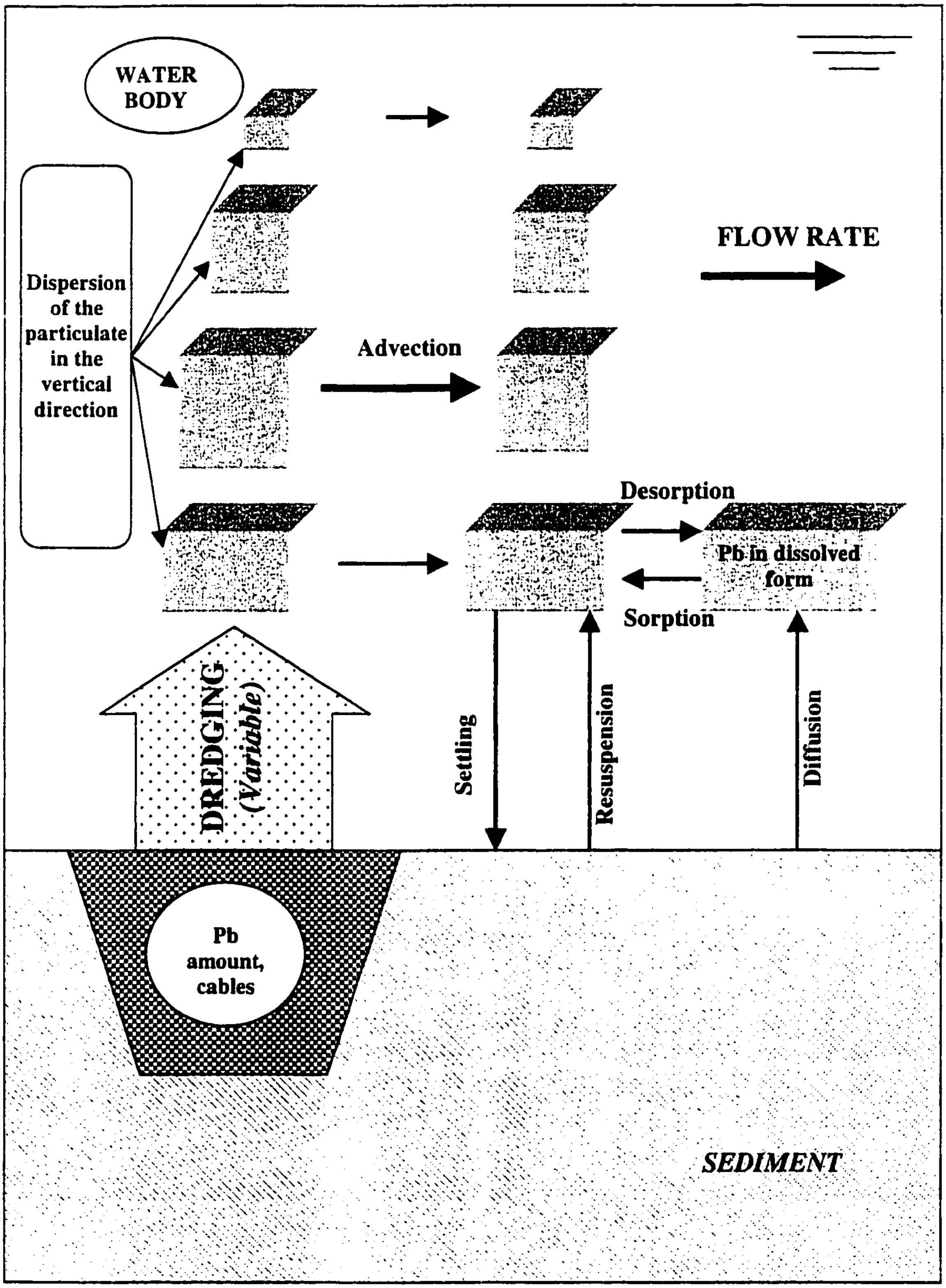


Figure 3.10: Simulation of the impact of dredging operation on Pb fate in water column by means of MFCL model (Case Studies: C10, C11, C12 and C13).

The model evaluates the impact of the engineering work schedule, looking at the best scenarios for dredging activities within continuous or intermittent dredging. Also, the model calculates dissolved and particulate lead concentrations at different downstream locations due to dredging up to 800 m.

3.5 Conclusion

The fate of corrosion products released into water bodies is governed by an important number of factors. The removal of corroded cables increases the complexity of processes and introduces new factors that interact with the previously mentioned ones (Chapter 2).

The development of the *MFCL* model allowed for management of the main factors which would influence the fate of corrosion products.

This model permits the prediction of the environmental impact of dredging activities in all types of river bodies (large and small rivers, lakes and bays). The model is powerful for use in dredging activities, preparation monitoring, and time shifts of engineering works with minimal disturbance to the environment.

The *MFCL* model can allow for a free choice of variable and constant factors and performs different scenarios in order to study the effect of each factor.

CHAPTER 4

RESULTS OF MODELING: DISSCUSION OF THE IMPACT OF FACTORS AFFECTING THE FATE OF CORRODED LEAD

To provide necessary information for Environmental Impact Assessment (EIA), the *MFCL* model was used to simulate some ecological scenarios. Different cases were introduced to the model to see the effect of particular factors.

By providing a choice of variable and defined data, the *MFCL* Model manages the main factors, which influence the fate of particulate and dissolved lead in different layers of the water column and also at different downstream locations.

4.1 Impact of Flow Rate

Results of modeling showed that the flow rate is the most important factor that affect the fate of particulate and dissolved lead released from corroded cables. To study the impact of this factor, the model was run with different flow rates: high, medium and low. During runs for the three flow rates, the cables were assumed to be laying on the surface of sediments. Lead was released into the water column due to full corrosion of cables.

4.1.1 Case Study-1 – High Flow Rate

In case study-1, the average flow rate of the St. Lawrence River in the Montreal area was taken into account, which is approximately 9000 m^3/s.

Results showed that particulate concentration of lead increased from the initial value of 5.4E-4 mg/L to a value of 1.345E-3 mg/L (Table 4.1) at the cable position, which is almost 25 times more than the initial particulate concentration. For dissolved lead the concentration almost remained constant with little changes, where it increased from the initial value of 3.4E-2 μg/L to a value of 3.615E-2 μg/L (Table 4.2).

As shown in Tables 4.1 and 4.2, concentrations of particulate and dissolved lead are almost the same in the three different layers of the water body at the cable position, which indicates a "completely mixed" situation. It is also shown that after reaching the highest concentration, particulate and dissolved lead starts to decrease at a very small rate.

Figures 4.1 and 4.2 show that it would take around 3 months for both particulate and dissolved lead concentrations to decrease to the background concentration.

Table 4.1: Vertical particulate lead distribution in different layers of the water column (Flow rate = 9000 m^3/s).

Time (hour)	Layer-1 (mg/L)	Layer-2 (mg/L)	Layer-3 (mg/L)
0	0.00054	0.00054	0.00054
0.5	0.013461	0.013458	0.013455
1	0.01346	0.013457	0.013454
1.5	0.013459	0.013456	0.013453
2	0.013458	0.013455	0.013452

Layer-1: Bottom layer, Layer-2: Middle Layer, Layer-3: Upper layer

Table 4.2: Vertical dissolved lead distribution in different layers of the water column (Flow rate = 9000 m^3/s).

Time (hour)	Layer-1 (μg/L)	Layer-2 (μg/L)	Layer-3 (μg/L)
0	3.40E-2	3.40E-2	3.40E-2
0.5	3.606E-2	3.616E-2	3.625E-2
1	3.606E-2	3.616E-2	3.625E-2
1.5	3.606E-2	3.616E-2	3.625E-2
2	3.606E-2	3.616E-2	3.625E-2

Layer-1: Bottom layer, Layer-2: Middle Layer, Layer-3: Upper layer

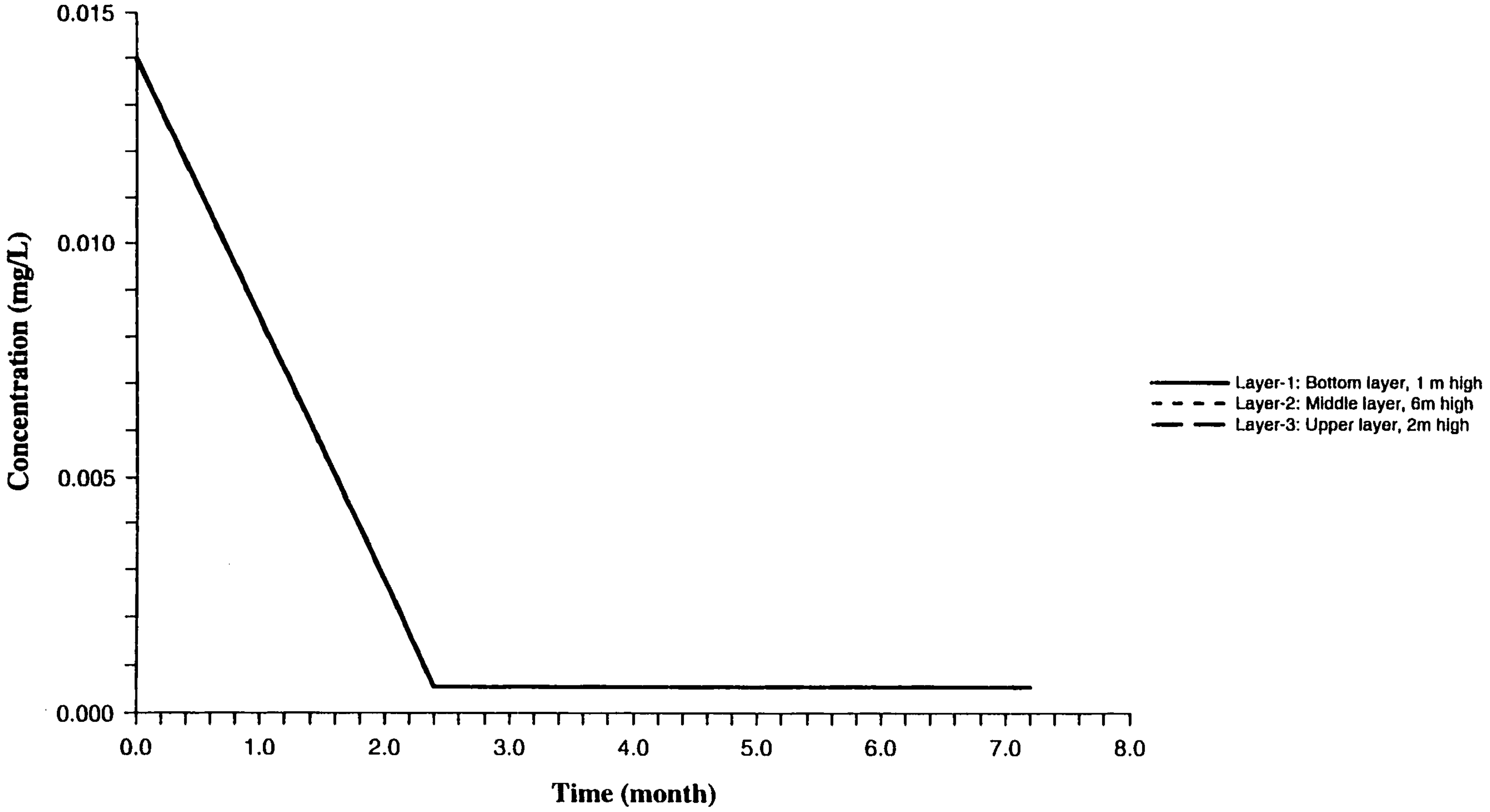


Figure 4.1: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-1: High flow rate (9000 m^3/s), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

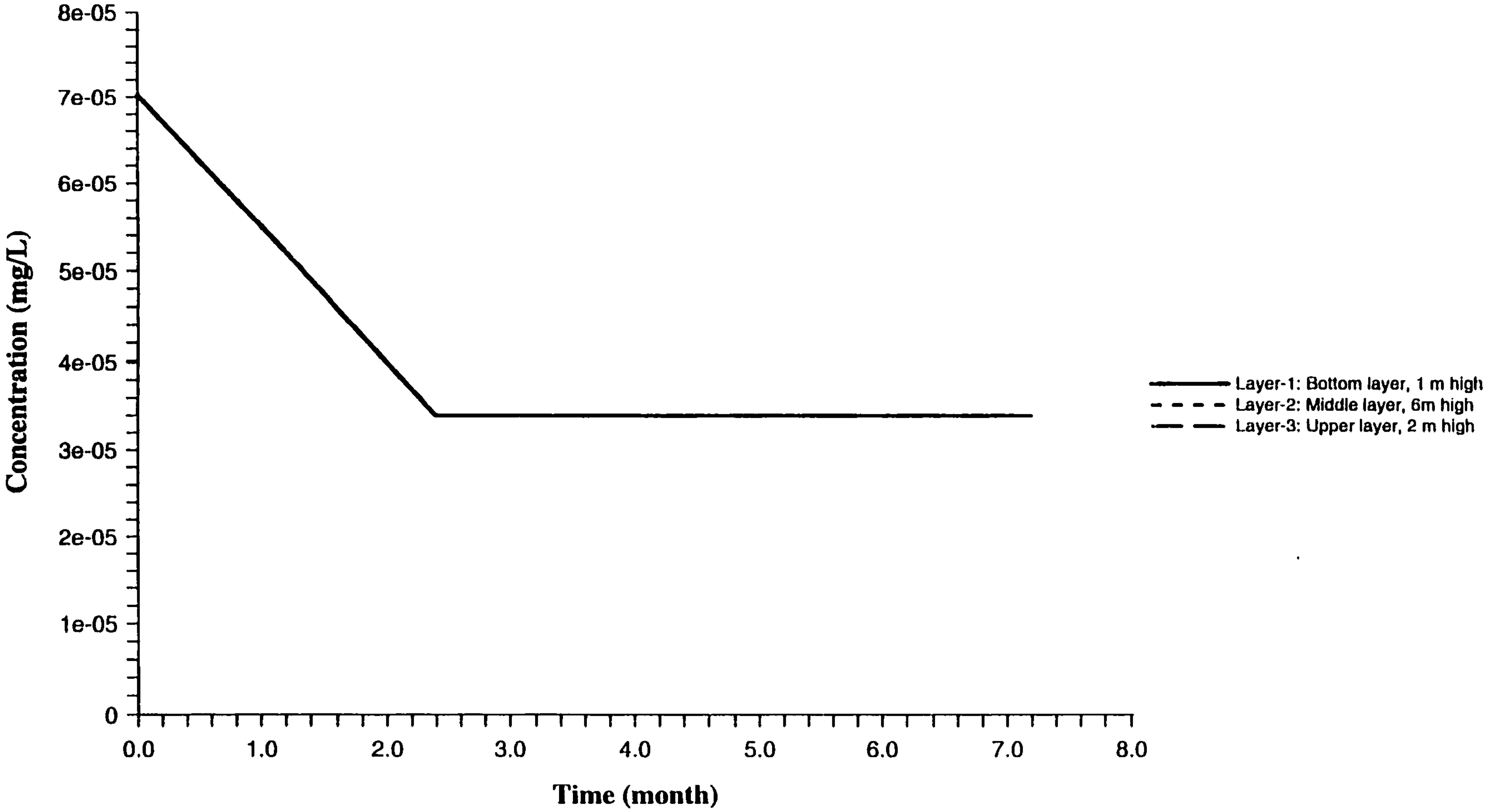


Figure 4.2: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study 1: High flow rate (9000 m3/s), cables on the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

The obtained data result from the very high flow rate, which will cause the rapid and high dispersion and mixing of released lead in the water column.

4.1.2 Case Study-2 – Medium Flow Rate

This case study represents a small size river with a flow rate of 30 m^3/s, which is representative of the Chateauguay River which has a flow rate of 33 m^3/s (St. Lawrence Center, 1996). This flow rate is almost 300 times less than the flow rate of the St. Lawerence River in the Montreal area.

In this case study, the entire corrosion of cables, situated on the surface of bottom sediments, was assumed. The vertical distributions of concentration were obtained for both particulate and dissolved lead (Figures 4.3, 4.4). Stratification of the water body above the sediment surface at the cable position was observed. For layer-1 which is 1 m above the sediment surface particulate lead concentration, increased from the initial concentration of 5.4E-4 mg/L up to 18.1 mg/L (Figure 4.3). This was almost 1350 times more than the case of the St. Lawerence River in the Montreal area. The dissolved lead concentration in the same layer increased from an initial value of 3.4E-2 µg/L up to 46.3 µg/L (Figure 4.4), which was almost 1280 times more than in Case Study-1. In the middle layer, the particulate lead concentration increased to a value of 8.7 mg/L (Figure 4.3) which was almost 650 times more than Case Study-1, the dissolved lead increased to a value of 35.4 µg/L (Figure 4.4) which was almost 980 times more than Case Study-1. In the upper layer, the particulate lead concentration increased to a value of 4.18 mg/L (Figure 4.3) which was almost 300 times more than Case Study-1. The dissolved lead

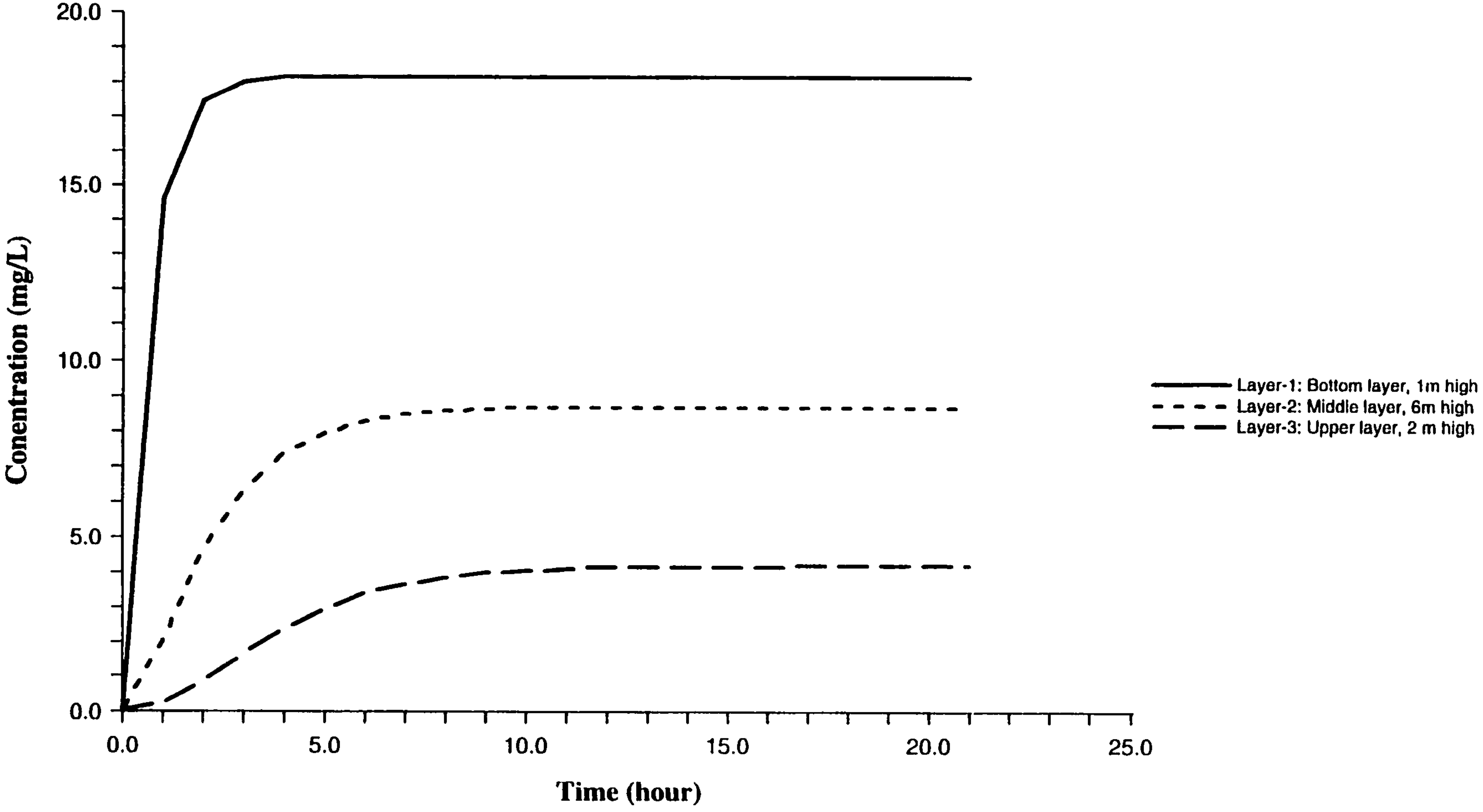


Figure 4.3: Particulate Pb concentration at cable position in different layers of the water column first day after release (Case Study-2: Medium flow rate (30 m^3/s), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

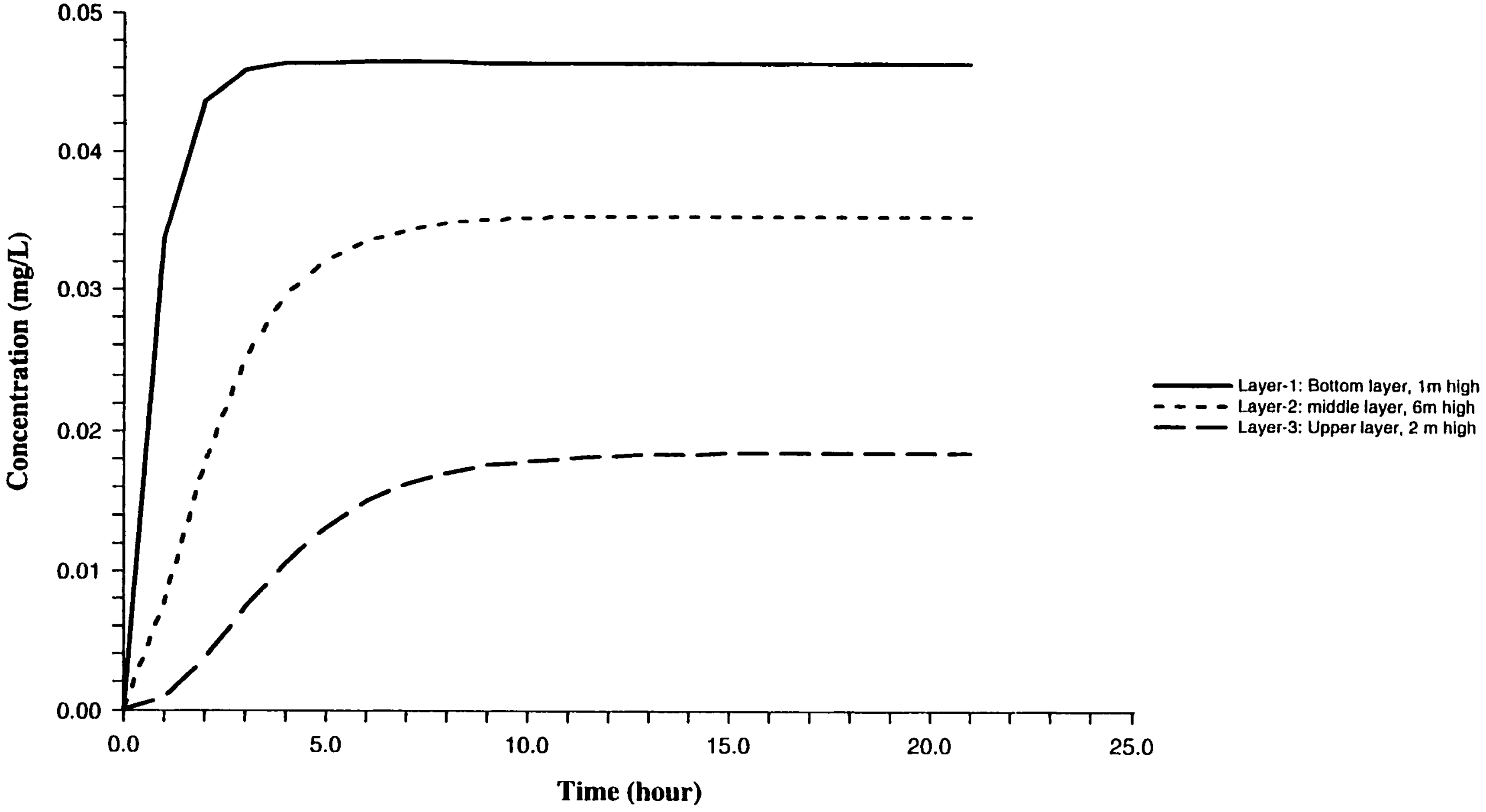


Figure 4.4: Dissolved Pb concentration at cable position in different layers of the water column first day after release (Case Study-2: Medium flow rate (30 m^3/s), cables on the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

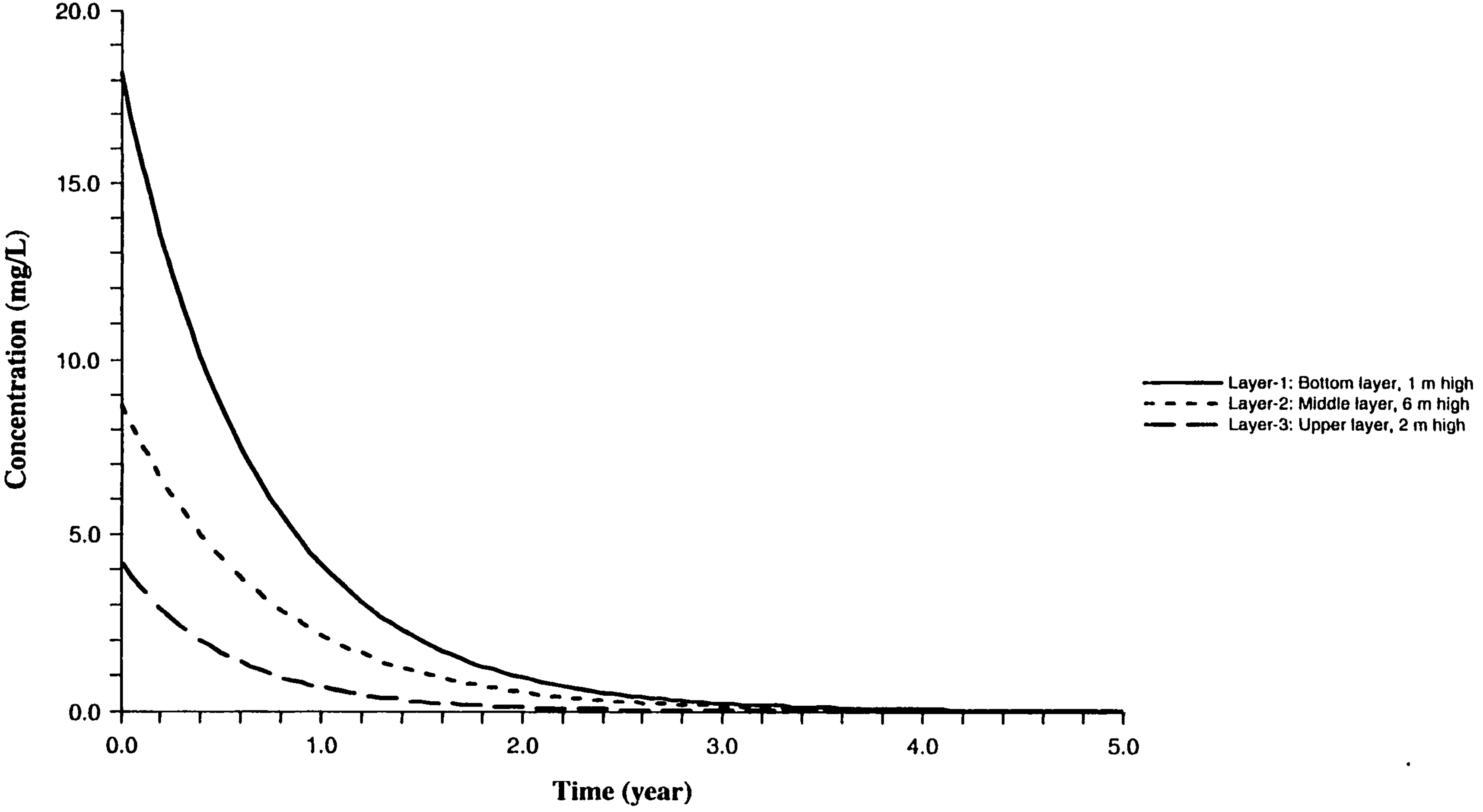


Figure 4.5: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-2: Medium flow rate ($30m^3/s$), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

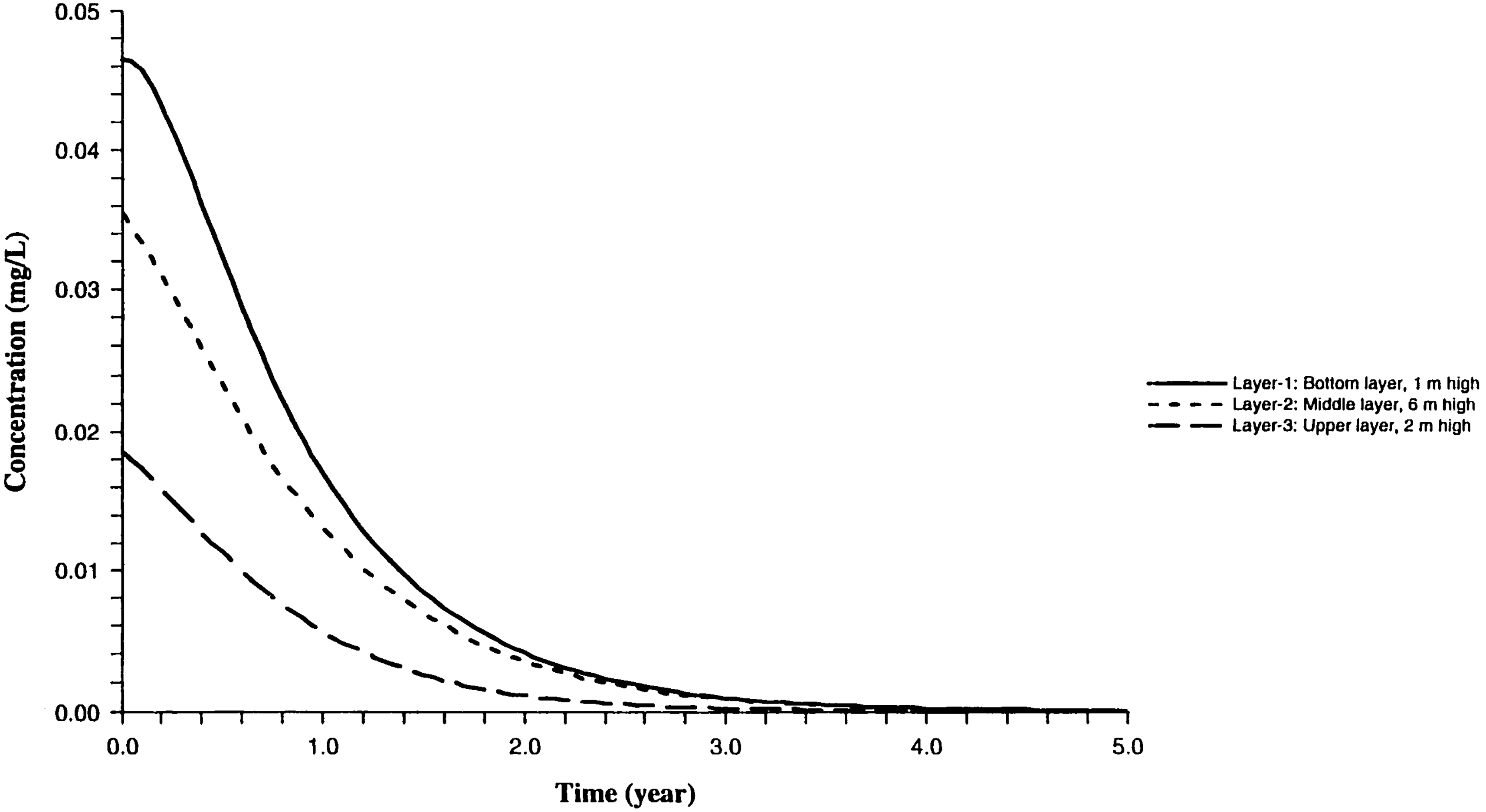


Figure 4.6: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study-2: Medium flow rate (30m^3/s), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 3.4E-5 mg/L increased to a value of 18.5 µg/L (Figure 4.4) which was almost 500 times more than Case Study-1.

Figure 4.5 shows that it took around 3.5 years for the particulate lead to decrease to its initial concentration before the discharge, which was 14 times more than Case Study-1. Figure 4.6 shows that for the dissolved lead concentration, it took approximately 4.2 years to decrease to the background concentration which was almost 17 times more than Case Study-1.

4.1.3 Case Study-3 – Low Flow Rate

This case study was performed with a flow rate of 1.5 m^3/s, is representative of lakes bays and harbours. This flow is almost 6000 times less than the flow rate of the St. Lawrence in the Montreal area and 20 times less than that in Case Study-2.

In this case study, a range of concentrations was obtained for both particulate and dissolved lead. Stratification of the water body above the sediment surface at the cable position was observed. Higher concentrations were found in layer-1, which was 1 m directly above the sediment surface. In this layer, the particulate lead concentration increased from an initial concentration of 5.4E-4 mg/L up to 52.0 mg/L (Figure 4.7) which was almost 3 times more than Case Study-2, and 3900 times more than in the St. Lawrence River (Case Study-1). In the same layer, the dissolved lead concentration increased from an initial value of 3.4E-2 µg/L up to 1850 µg/L (Figure 4.8), which was almost 4 times more than Case Study-2 and 5100 times more than Case Study-1.

In the middle layer, the particulate lead concentration increased to a value of 8.1 mg/L (Figure 4.7) which was almost 600 times more than Case Study-1 and almost the same as Case Study-2. The dissolved lead concentration increased to a value of 35.4

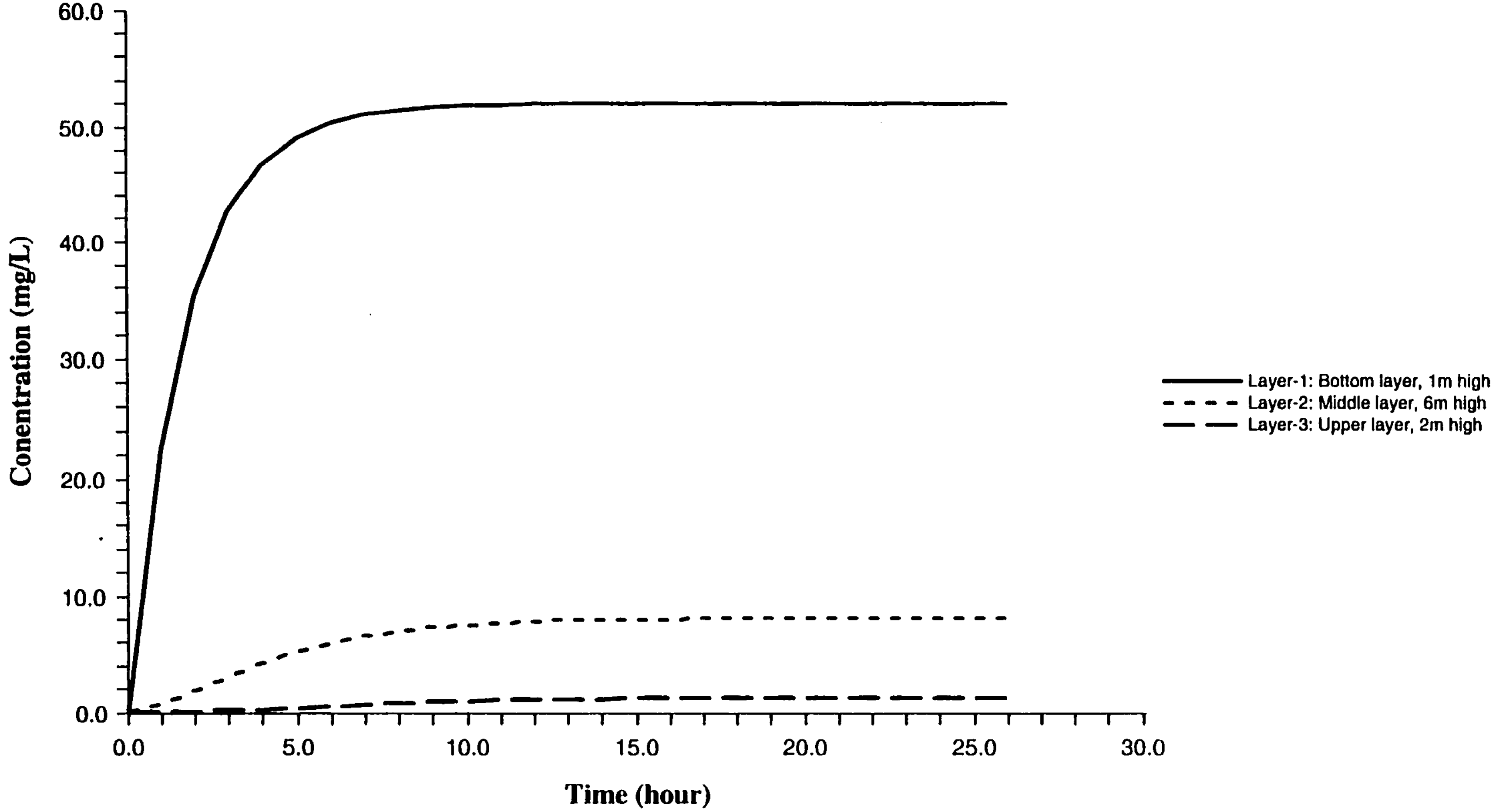


Figure 4.7: Particulate Pb concentration at cable position in different layers of the water column during the first two days (Case Study-3, 4 & 9: Low flow rate (1.5 m^3/s), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

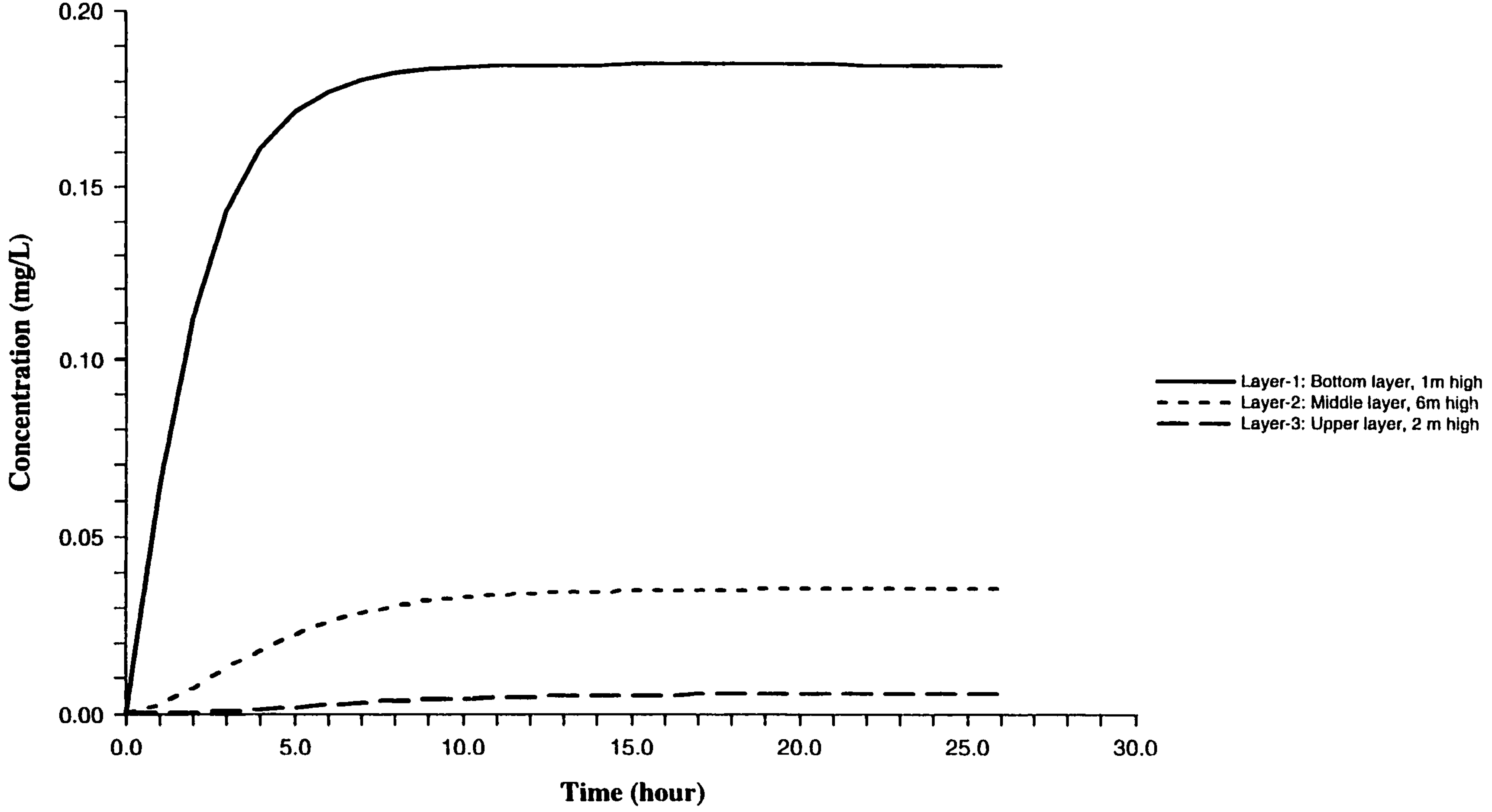


Figure 4.8: Dissolved Pb concentration at cable position in different layers of the water column during the first two days (Case Study-3, 4 & 9: Low flow rate (1.5 m^3/s), cables on the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

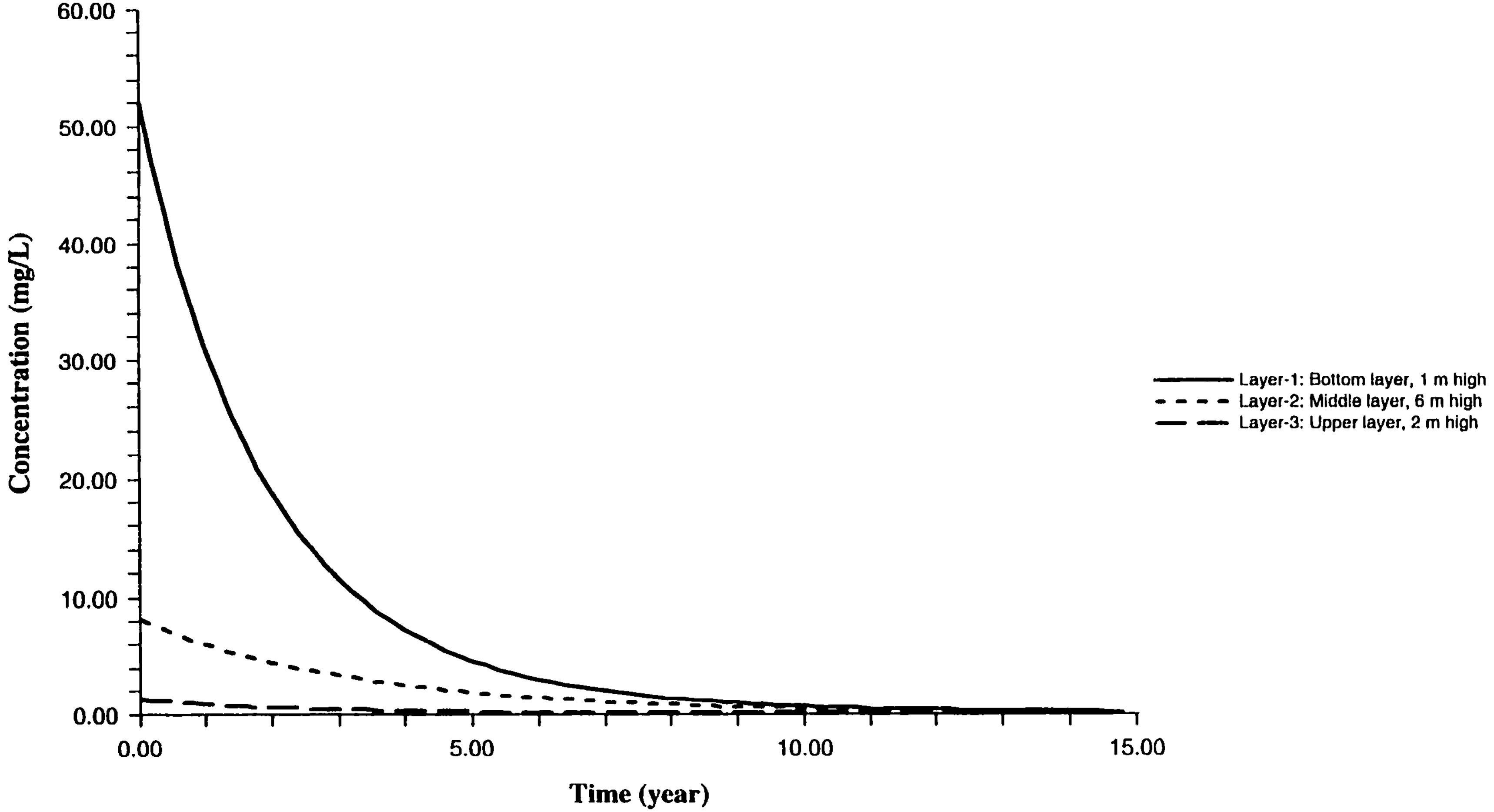


Figure 4.9: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-3, 4 & 9: low flow rate (1.5 m^3/s), cables on the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

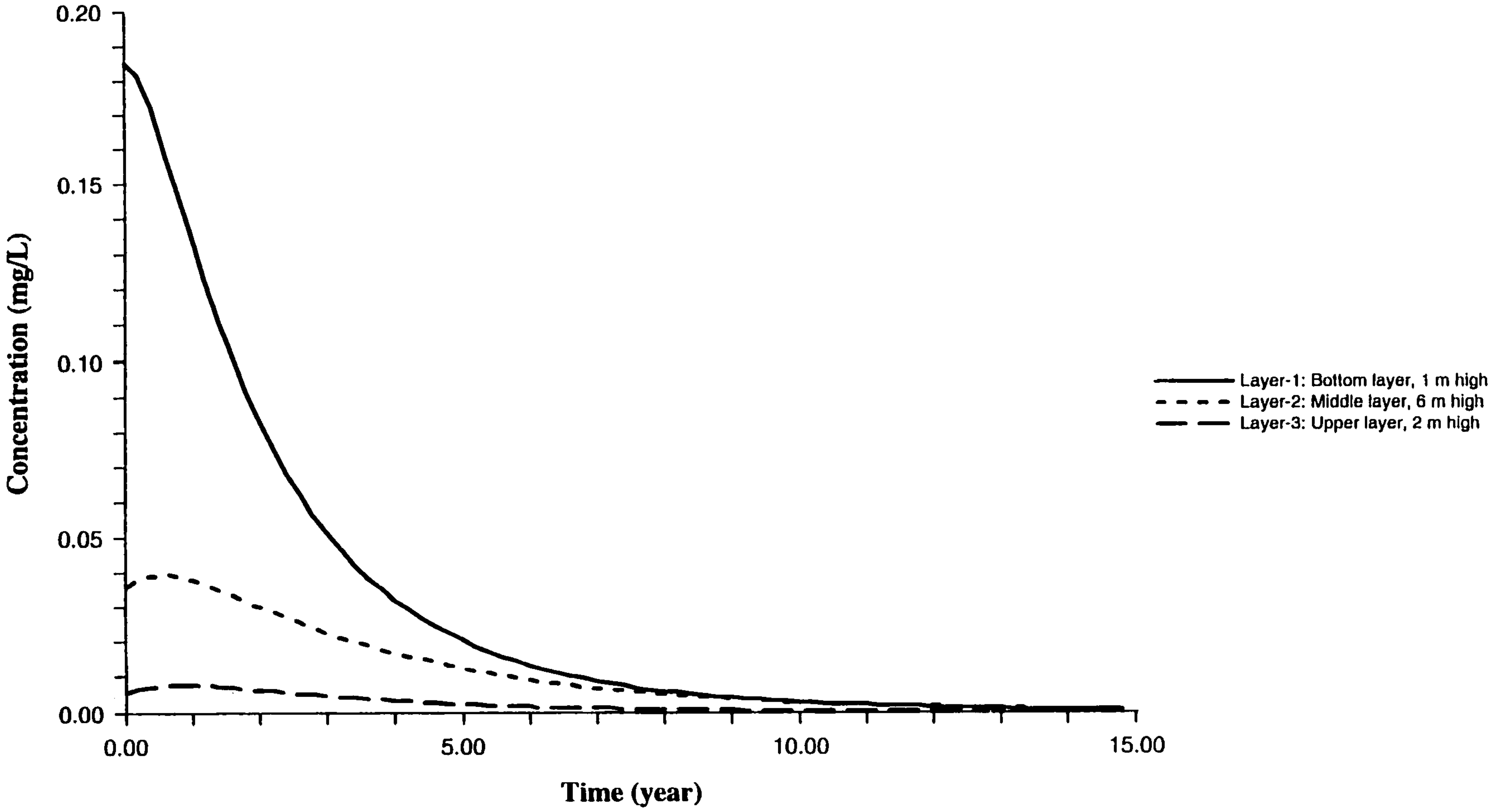


Figure 4.10: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study-3, 4 & 9: low flow rate (1.5 m^3/s), cables on the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

μg/L (Figure 4.8) which was almost 980 times more than Case Study-1 and the same as Case Study-2.

In the upper layer, the particulate lead concentration increased to a value of 1.27 mg/L (Figure 4.7) which was almost 95 times more than Case Study-1 and almost 3 times less than Case Study-2. The dissolved lead increased to a value of 5.72 μg/L (Figure 4.8) which was almost 160 times more than Case Study-1 and almost 3 times less than Case Study-2.

Figure 4.9 shows that it took approximately 13 years for the particulate lead to decrease to its initial concentration before the discharge which was almost 3.5 times more than Case Study-2 and 52 times more than Case Study-1. Figure 4.10 shows that for the dissolved lead concentration, it took around 14 years to decrease to the background concentration which was almost 3 times more than Case Study-2 and 56 times more than Case Study-1.

4.2 Impact of Cable Position

Situation and positioning of sub-marine cables can also affect the concentrations of particulate and dissolved lead released into the water column. In order to study the effect of this factor, different case studies were simulated, while maintaining the same flow rate, and the same amount of initial total lead amount (59000 kg) but changing the depth of cables from 1 m to 5 m in each case study. Previous studies showed that a small flow rate provided the largest concentration and longest period of time to reach initial concentrations. Therefore, this condition was adapted in further scenarios.

4.2.1 Case Study-4 – Cables on Sediments Surface

This case study is the same as Case Study-3 (Figures 4.7 and 4.8) with cables situated on the surface, and a flow rate of 1.5 m^3/s. Entire corrosion of cables was assumed.

4.2.2 Case Study-5 – Cables 1m Beneath Sediments

It was assumed that the flow rate was 1.5 m^3/s and the entire amount of lead contained in the cables (59000 kg) was released into the water column. In this case, cables were laid 1 m beneath the surface of the sediments.

The particulate concentration of lead in the bottom layer increased from the initial value of 5.4E-4 mg/L to 5.21 mg/L (Figure 4.11) at the cable position, which was almost 10 times less than the final concentration when the same cables were laid on the surface. In the middle layer, the particulate concentration of lead increased up to 1.137 mg/L (Figure 4.11), which was almost 7 times less than the final concentration in Case Study-4. In the upper layer, the particulate concentration of lead increased up to 0.253 mg/L (Figure 4.11), which was almost 5 times less than the final concentration in Case Study-4.

In the bottom layer, the dissolved concentration of lead increased up to a value of 18.5 μg/L (Figure 4.12), which was 10 times less than the final dissolved concentration in Case Study-4. In the middle layer, it increased up to a value of 4.95 μg/L (Figure 4.12), which was almost 7 times less than the final concentration in Case Study-1. For the upper layer, the dissolved concentration of lead increased up to 1.13 μg/L (Figure 4.12), which is almost 5 times less than the final concentration in the previous case study.

Figure 4.13 shows that it took around 13 years for the particulate lead to decrease to its initial concentration before the discharge, which was the same as Case Study-4.

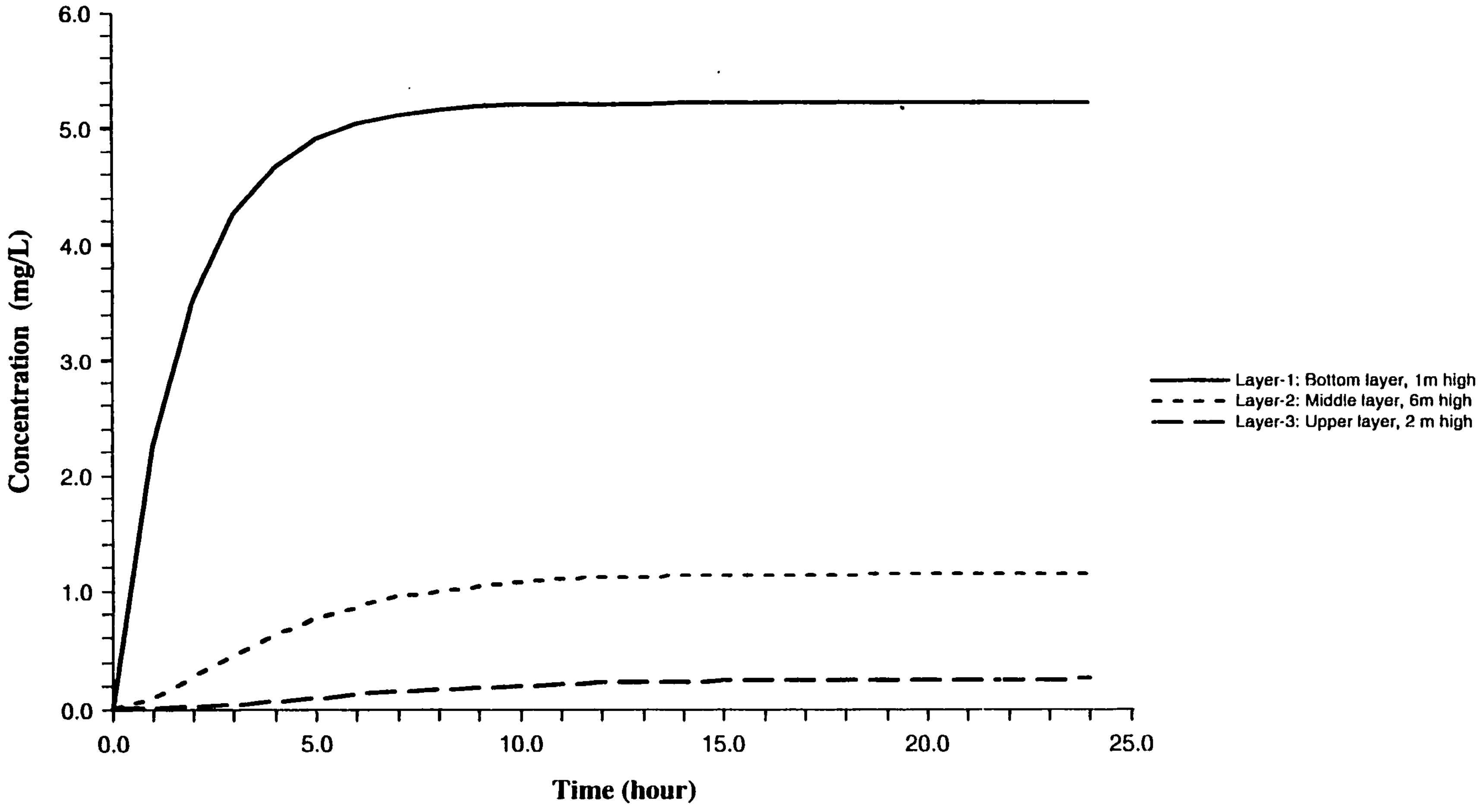


Figure 4.11: Particulate Pb concentration at cable position in different layers of the water column first day after release (Case Study-5: low flow rate (1.5 m^3/s), cables 1m beneath the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

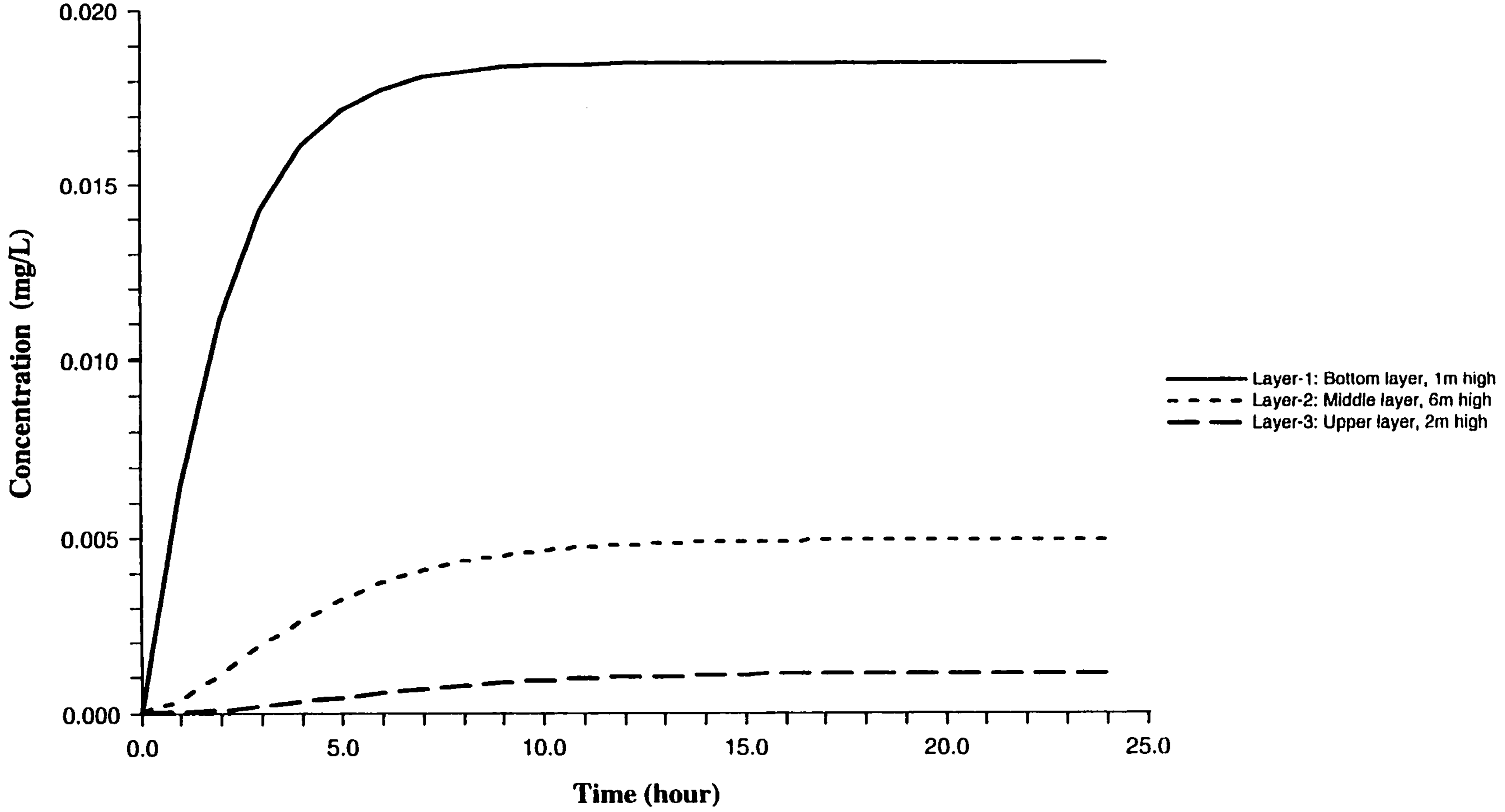


Figure 4.12: Dissolved Pb concentration at cable position in different layers of the water column first day after release (Case Study-5: low flow rate(1.5 m^3/s), cables 1m beneath the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

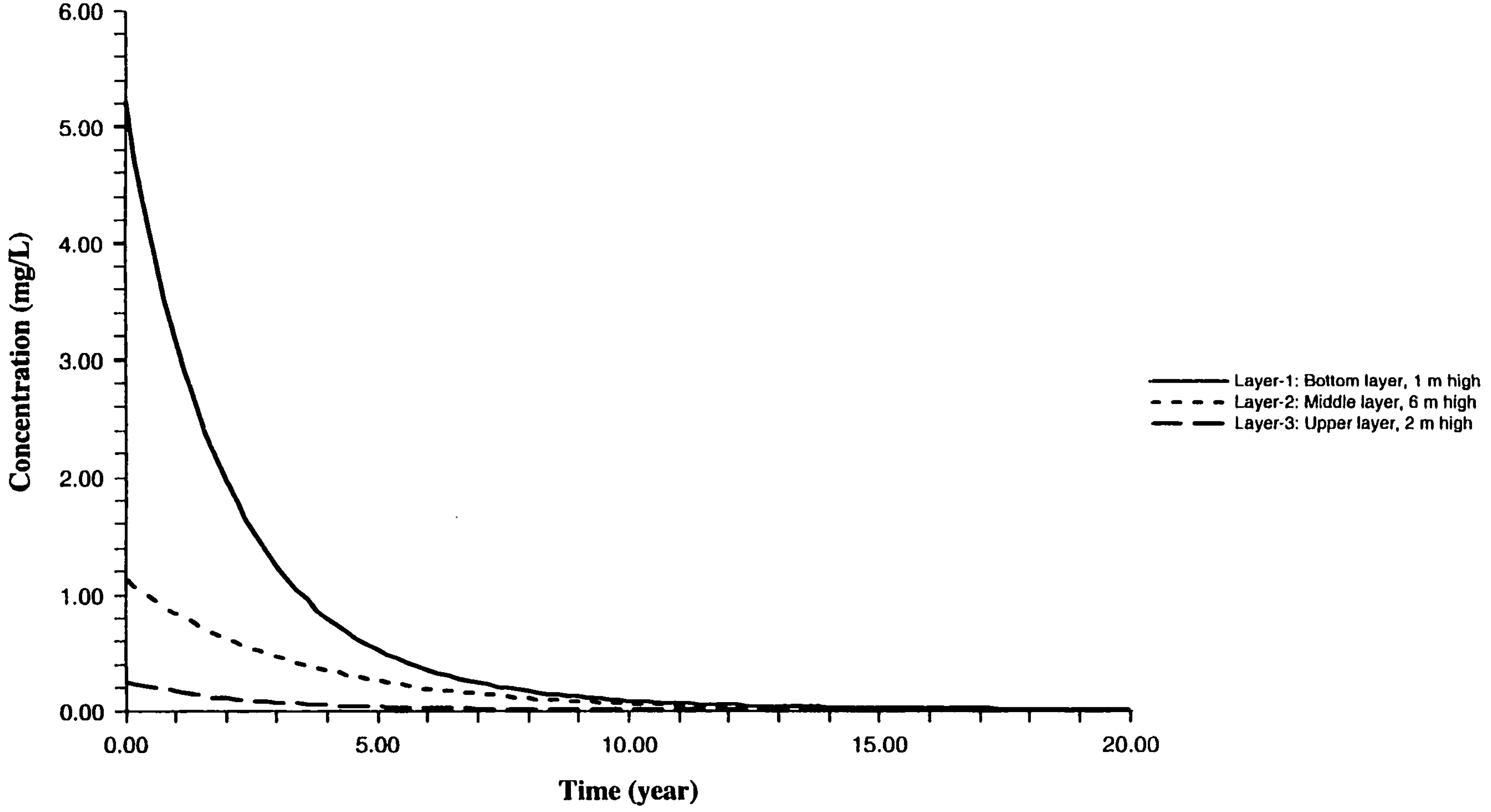


Figure 4.13: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-5: low flow rate (1.5 m^3/s), cables 1m beneath the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

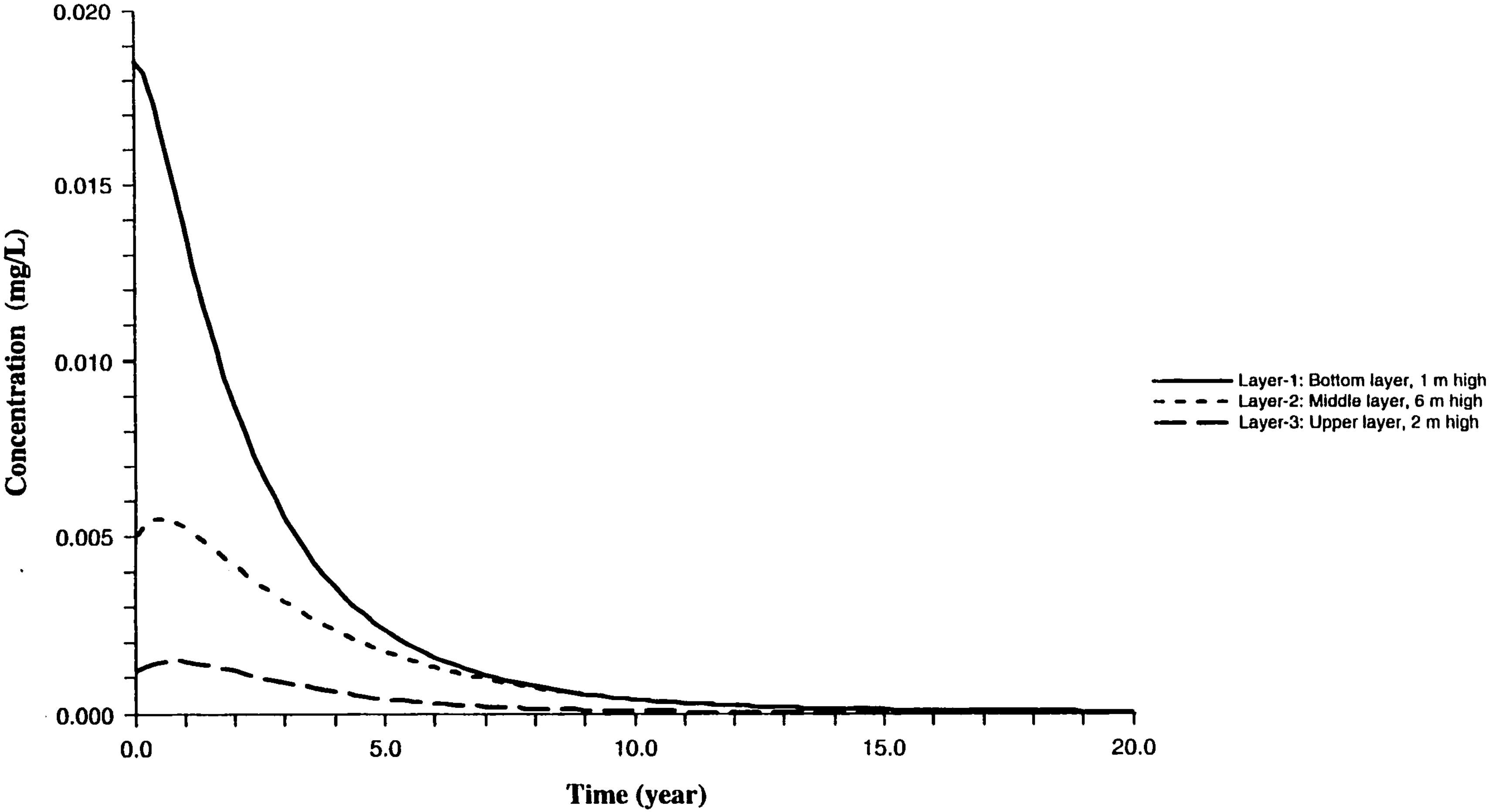


Figure 4.14: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study-5: low flow rate (1.5 m^3/s), cables 1m beneath the surface, entire corrosion of cables).

Initial particulate lead concentration = 3.4E-5 mg/L

Figure 4.14 shows that for the dissolved lead concentration, it took approximately 16 years to decrease to the background concentration which was two years more than Case Study-4.

4.2.3 Case Study-6 - Cables 5m Beneath Sediments

In this case, cables were assumed to be laid 5 m beneath the surface of the sediments but other conditions of Case Study-4 and 5 were kept constant.

The particulate concentration of lead in the bottom layer increased from the initial value of 5.4E-4 mg/L to 1.42 mg/L (Figure 4.15) at the cable position, which was almost 35 times less than the final concentration when the cables were laid on the surface, and 3.5 times lower when the cables were 1 m beneath the sediment surface. In the middle layer, the particulate concentration of lead increased up to 0.314 mg/L (Figure 4.15), which was almost 26 times less than the final concentration in Case Study-4 and almost 17 times less than Case Study –5 when the cables were at a depth of 1 m. In the upper layer, the particulate concentration of lead increased up to 0.075 mg/L (Figure 4.15), which was almost 18 times less than the final concentration in Case Study-4 and almost 3 times less than Case Study-5.

In the bottom layer, the dissolved concentration of lead increased up to a value of 5.04 µg/L (Figure 4.16), which was also 35 times less than the final dissolved concentration in Case Study-4 and 3.5 times less when the cables were 1 m beneath the sediment surface. In the middle layer, it increased up to a value of 1.37 µg/L (Figure 4.16), which was almost 26 times less than the final concentration in Case Study-4 and almost 4 times less than Case Study-5. In the upper layer, the dissolved concentration

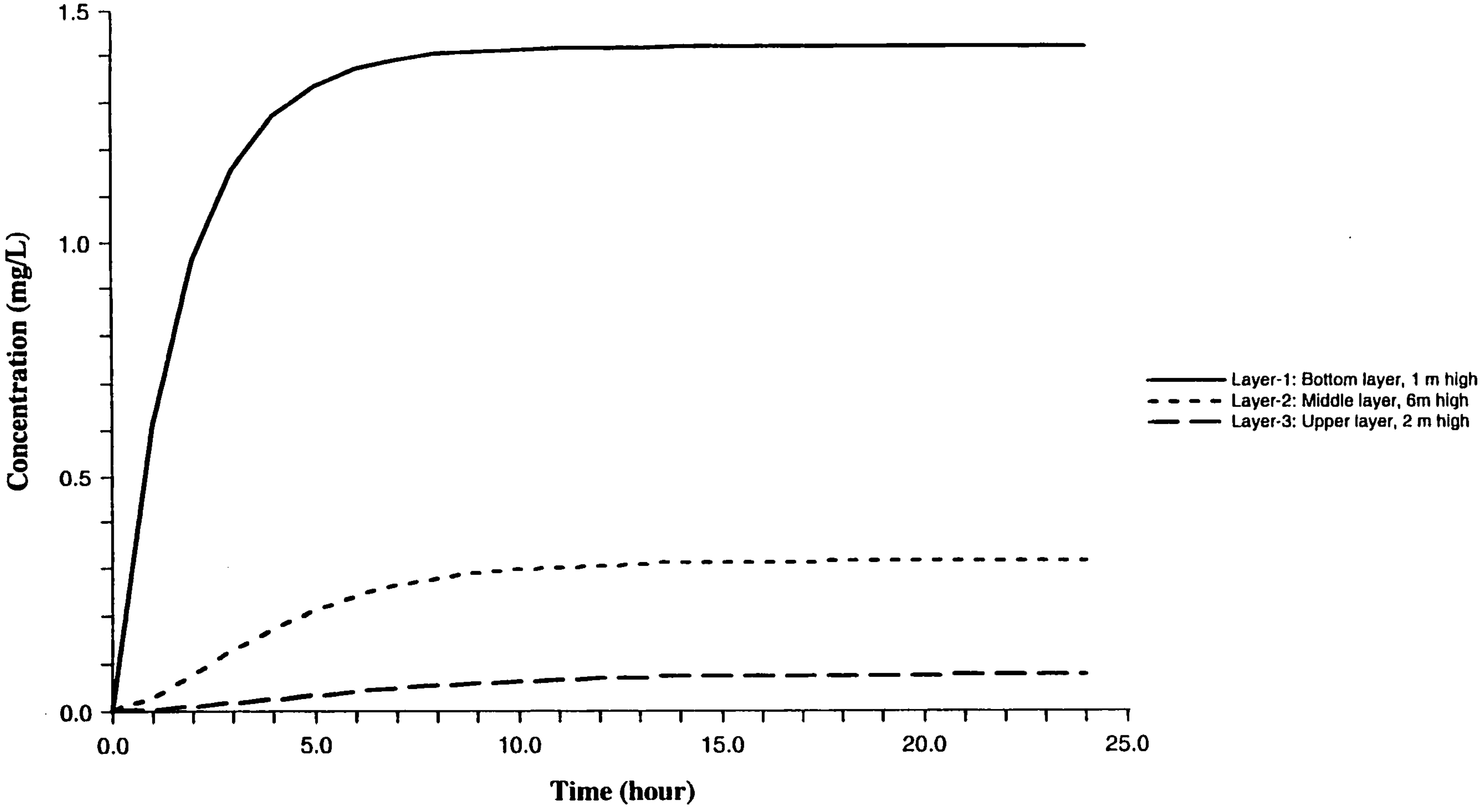


Figure 4.15: Particulate Pb concentration at cable position in different layers of the water column first day after release (Case Study-6: low flow rate (1.5 m^3/s), cables 5m beneath the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

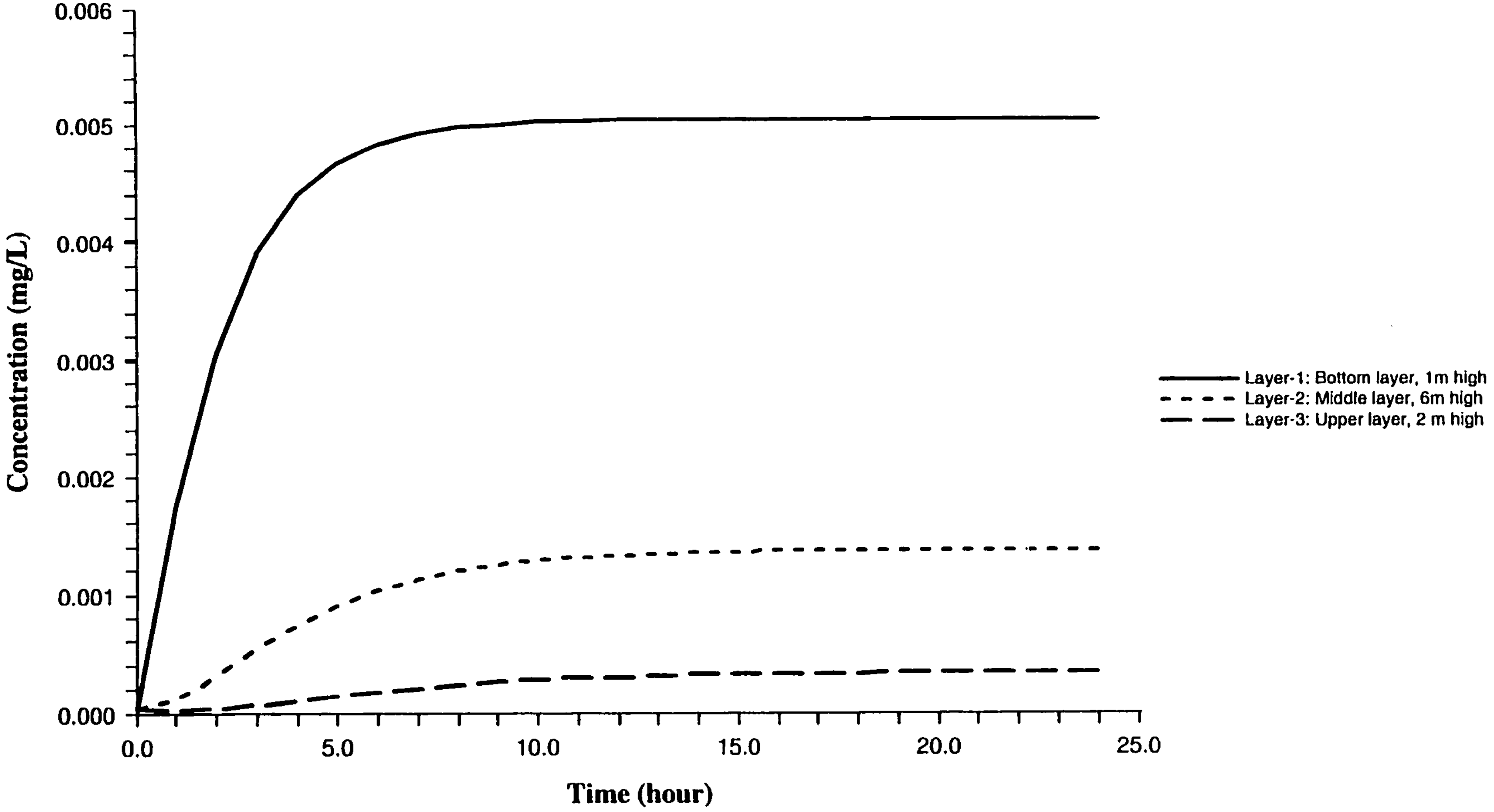


Figure 4.16: Dissolved Pb concentration at cable position in different layers of the water column first day after release (Case Study-6: low flow rate (1.5 m^3/s), cables 5m beneath the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

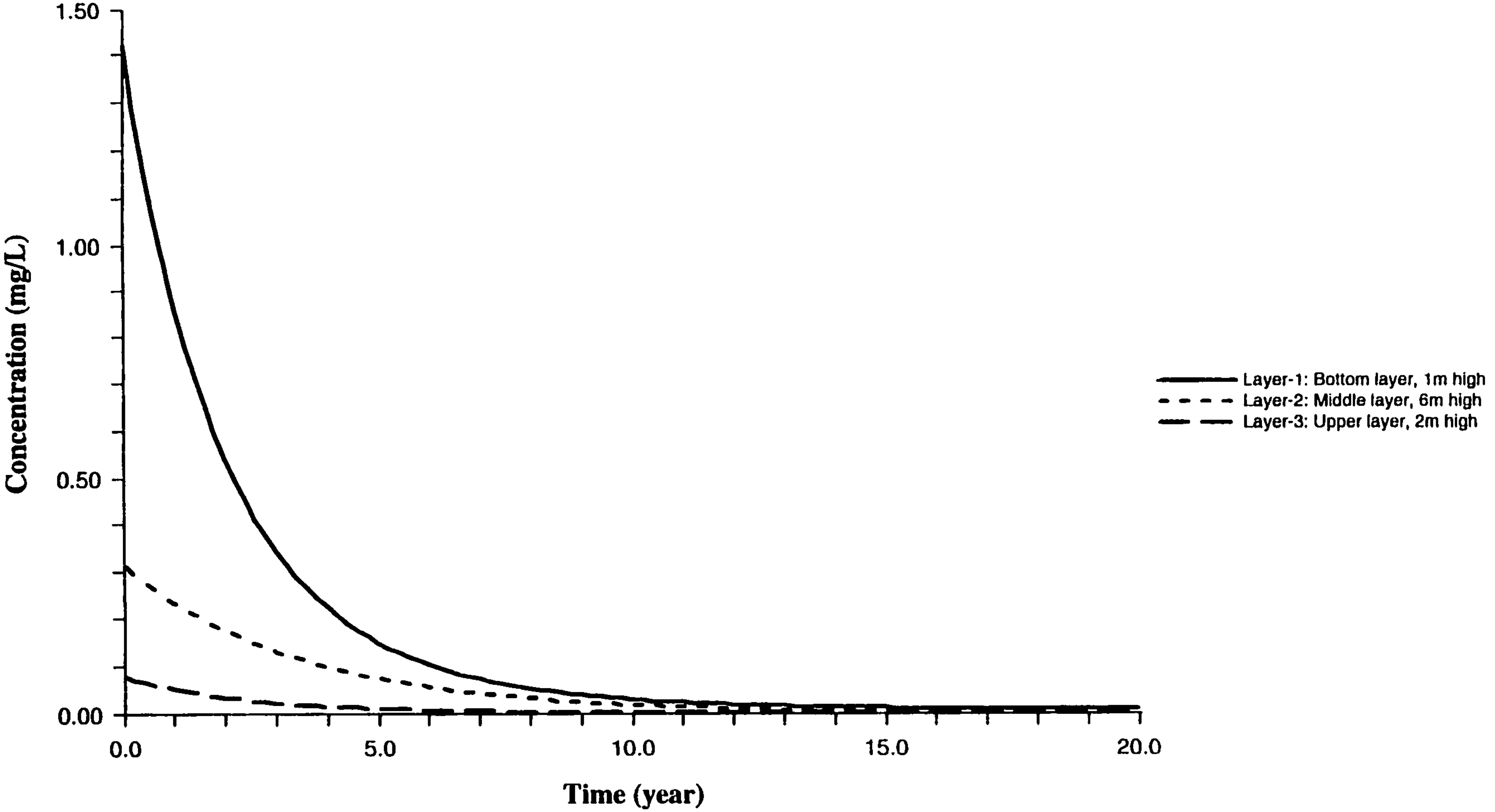


Figure 4.17: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-6: low flow rate (1.5 m^3/s), cables 5m beneath the surface, entire corrosion of cables).

Initial particulate lead concentration = 0.00054 mg/L

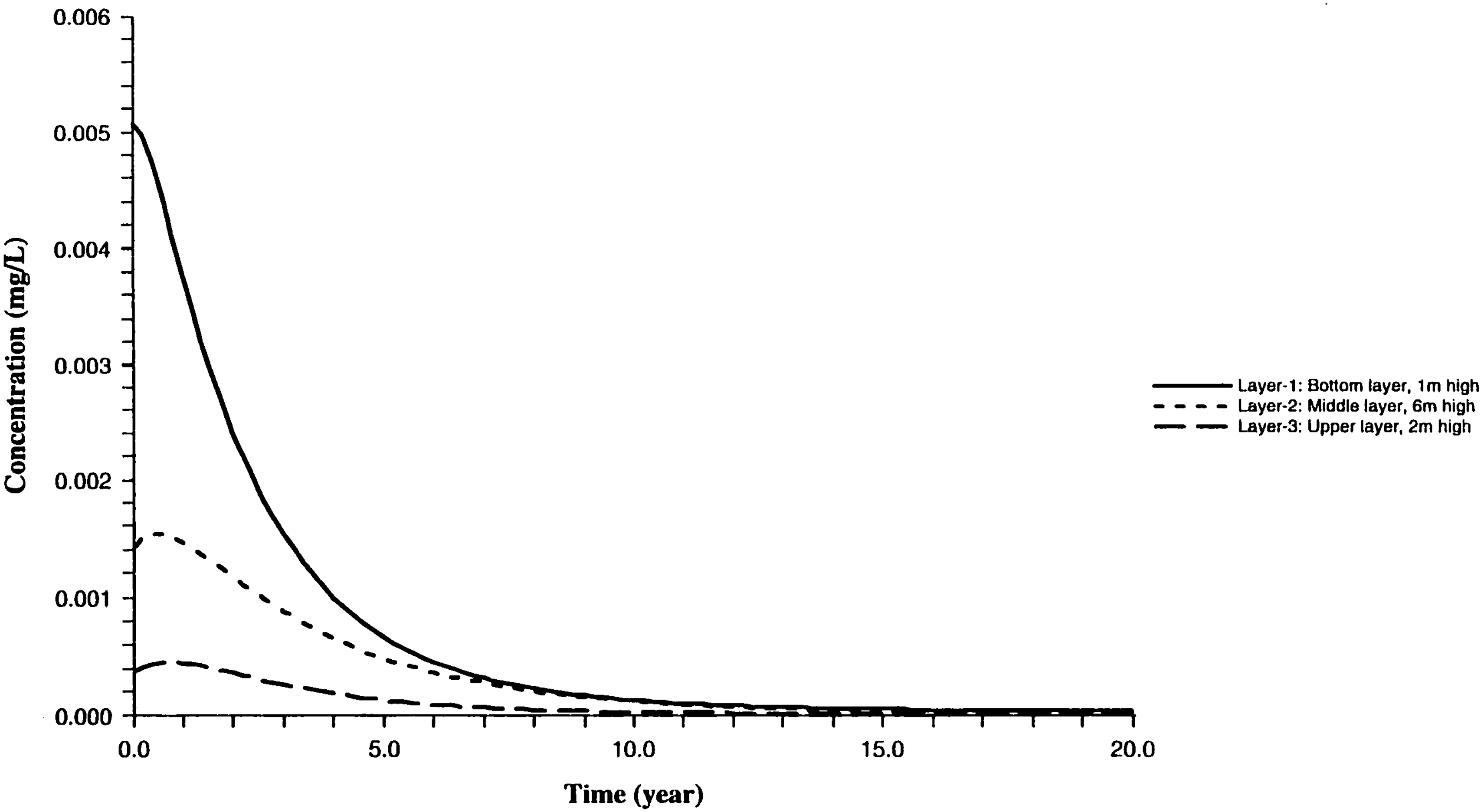


Figure 4.18: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study-6: low flow rate (1.5 m^3/s), cables 5m beneath the surface, entire corrosion of cables).

Initial dissolved lead concentration = 3.4E-5 mg/L

of lead increased up to 0.333 µg/L (Figure 4.16), which was almost 17 times less than the final concentration in Case Study-4 and almost 3 times less than Case Study-5.

Figure 4.17 shows that it took around 18 years for the particulate lead to decrease to its initial concentration before discharge, which was four years more than Case Study-4 and Case Study-5. Figure 4.18 shows that for the dissolved lead concentration, it took approximately 18 years to decrease to the background concentration which was two years more than Case Study-4 and four years more than Case Study-5.

4.3 Impact of Corrosion Rate

To show the effect of the corrosion rate, three case studies were investigated assuming that: 1) The total amount of lead was released into the water column due to full corrosion, 2) 50% of total lead was released into the water column and 3) No corrosion occurred.

In all cases, a flow rate of 1.5 m^3/s was assumed and cables were always laid on the surface of sediments.

4.3.1 Case Study-7 – Lack of Corrosion

In this case, no release of lead into the water column occurred, the flow rate was equal to 1.5 m^3/s and the cables were laid on the surface. Concentrations of background particulate and dissolved lead in the water column were 0.54 µg/L and 0.034 µg/L respectively.

4.3.2 Case Study-8 – Partial Corrosion of Cables

In this case, cable corrosion provoked the release of 50% of the total lead. The particulate concentration of lead in the bottom layer increased from the initial value of

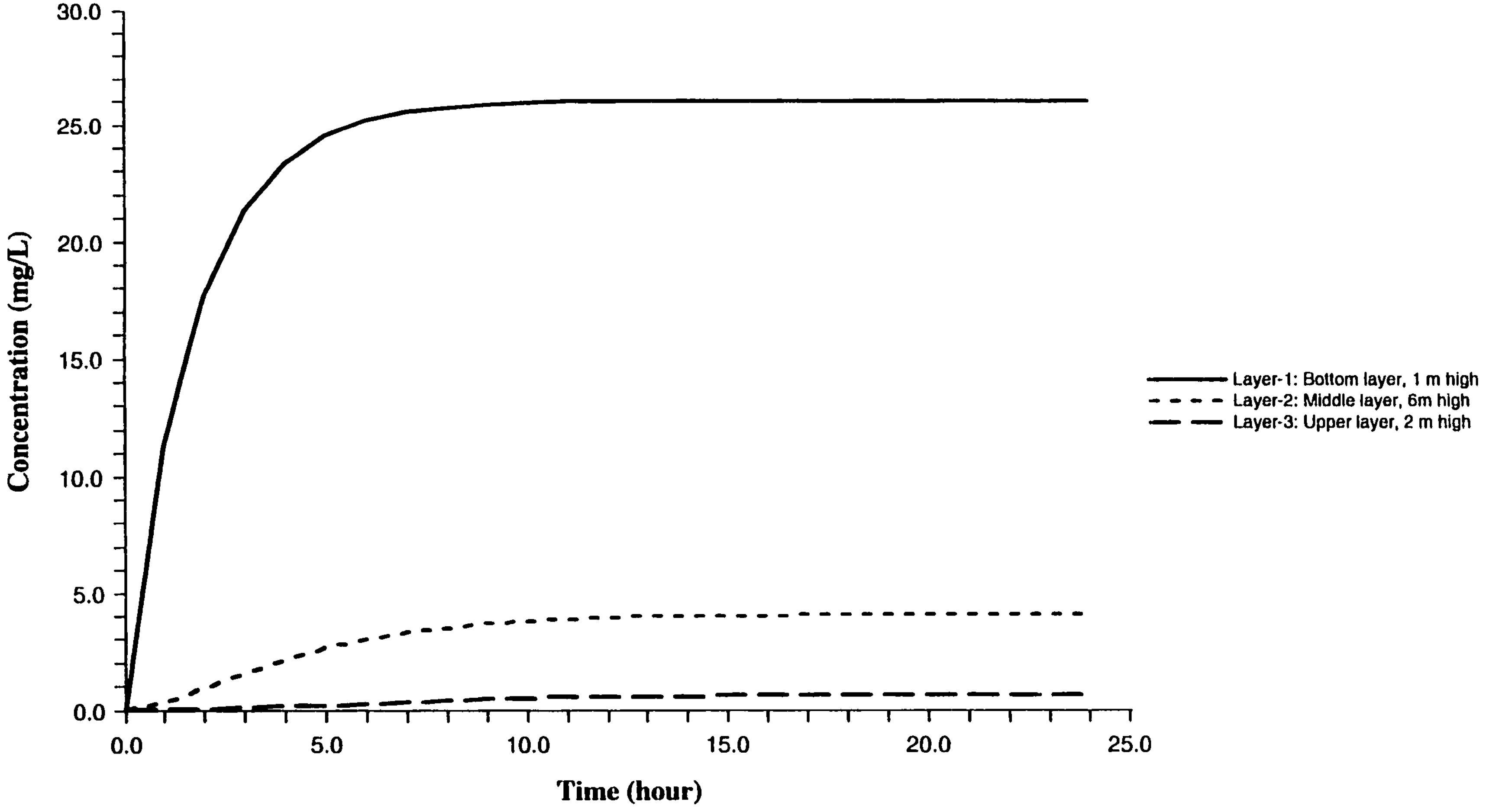


Figure 4.19: Particulate Pb concentration at cable position in different layers of the water column first day after release (Case Study-8: low flow rate (1.5 m^3/s), cables on the surface, and 50% of entire lead concentration).

Initial particulate lead concentration = 0.00054 mg/L

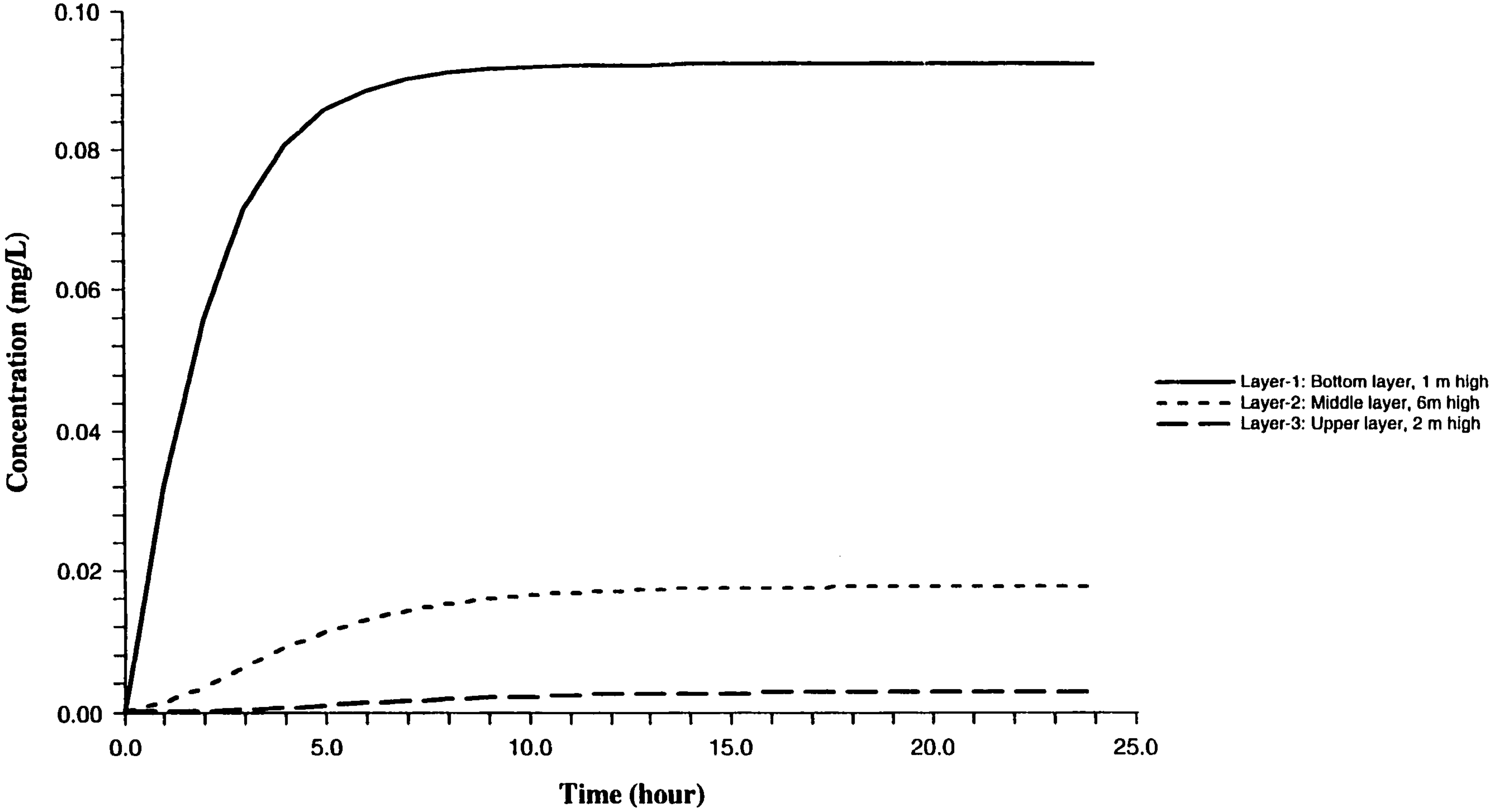


Figure 4.20: Dissolved Pb concentration at cable position in different layers of the water column first day after release (Case Study-8: low flow rate (1.5 m^3/s), cables on the surface, and 50% of entire lead concentration).

Initial dissolved lead concentration = 3.4E-5 mg/L

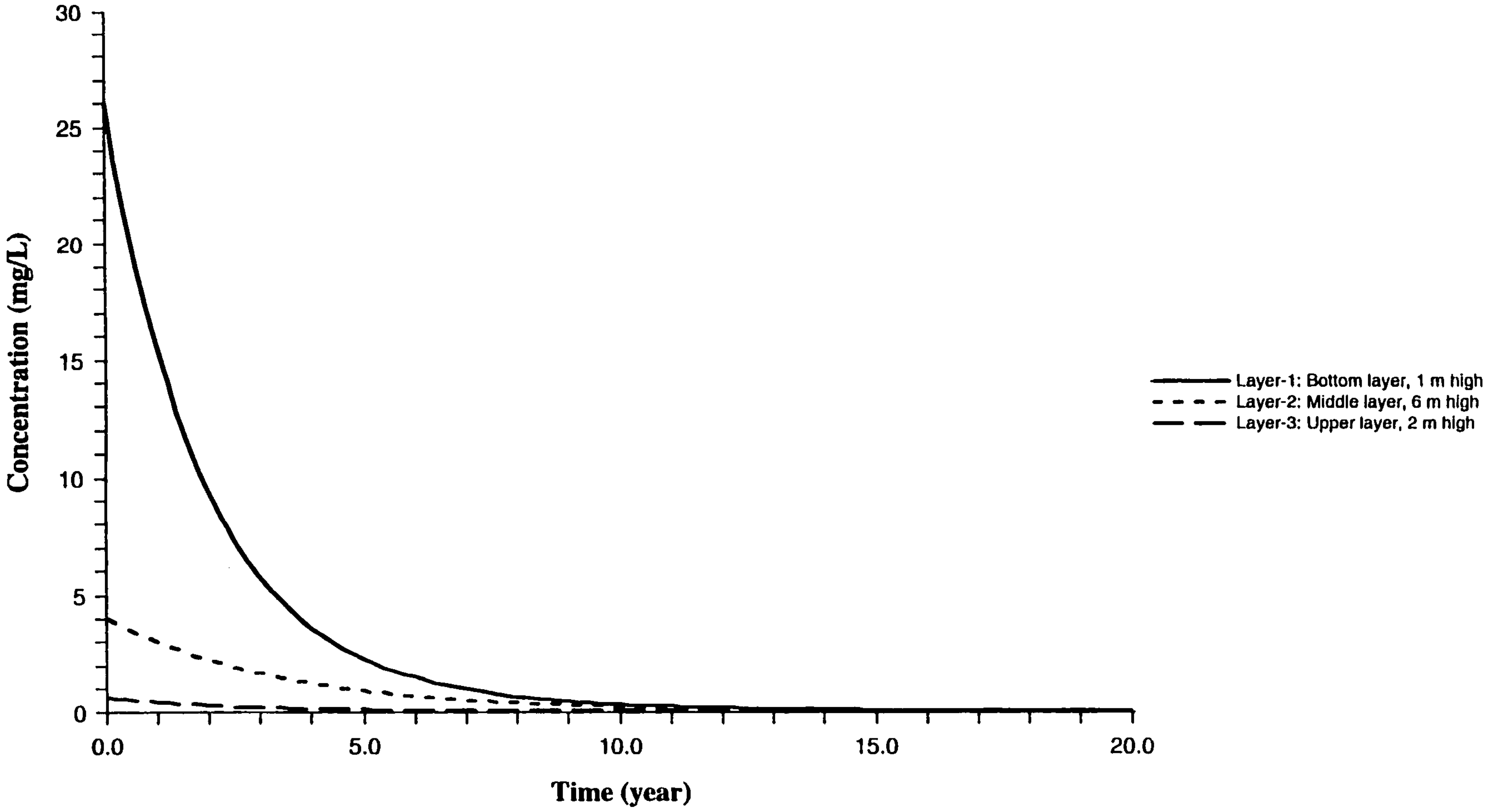


Figure 4.21: Particulate Pb concentration at cable position in different layers of the water column with time (Case Study-8: low flow rate (1.5 m^3/s), cables on the surface, and 50% of entire lead concentration).

Initial particulate lead concentration = 0.00054 mg/L

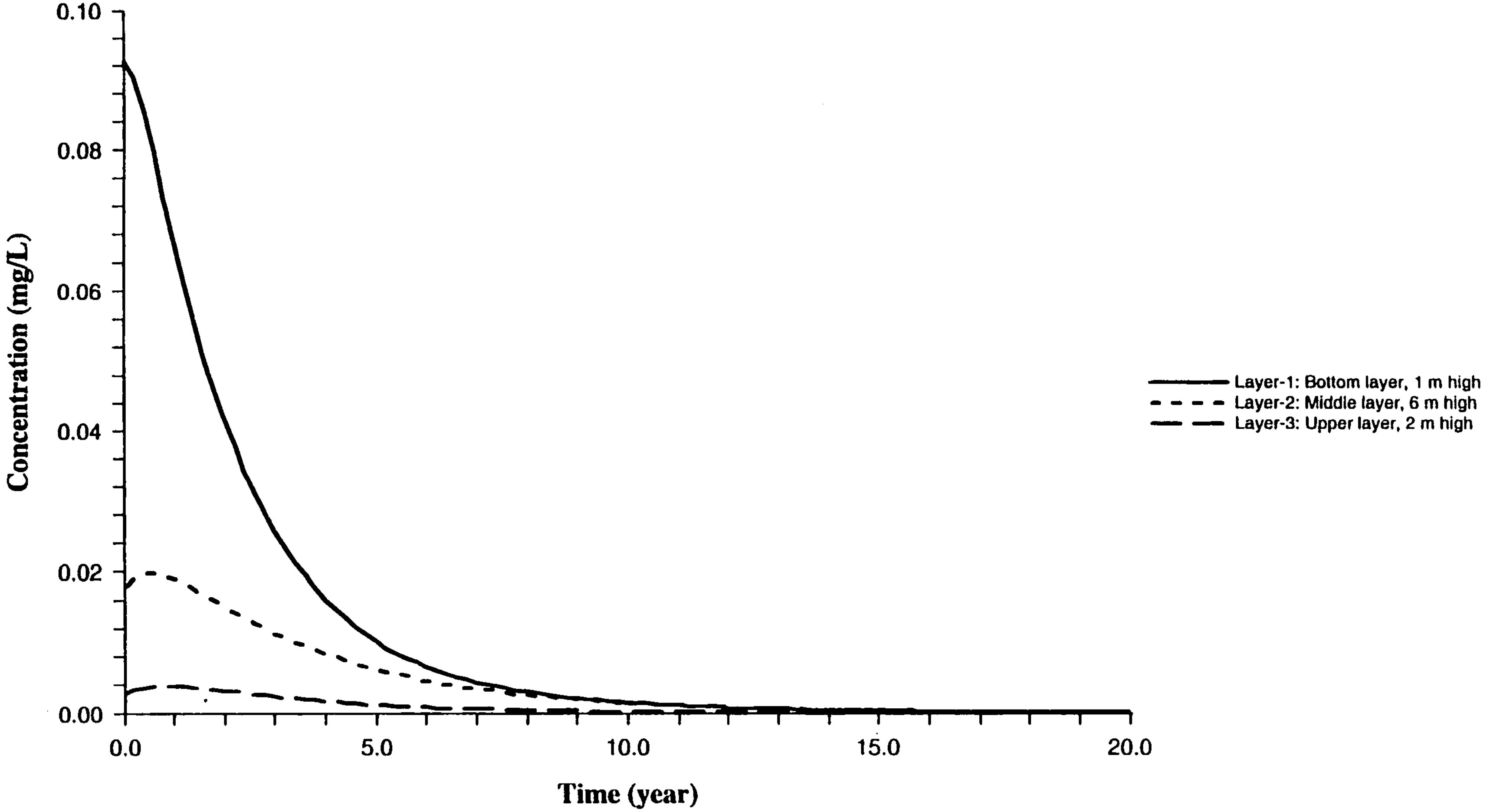


Figure 4.22: Dissolved Pb concentration at cable position in different layers of the water column with time (Case Study-8: low flow rate (1.5 m^3/s), cables on the surface, and 50% of entire lead concentration).

Initial dissolved lead concentration = 3.4E-5 mg/L

5.4E-4 mg/L to 26.0 mg/L (Figure 4.19) at the cable position, which was almost 5000 times more than the concentration of particulate lead in Case Study-7. In the middle layer the particulate lead concentration increased up to 4.07 mg/L (Figure 4.19), which was almost 7500 times more than Case Study-7. For the upper layer, the particulate lead concentration increased up to 0.642 mg/L (Figure 4.19), which was almost 1200 times more than the concentration in the Case Study-7.

In the bottom layer, the dissolved lead concentration increased up to a value of 92.3 µg/L (Figure 4.20), which was almost 3000 times more than Case Study-7. In the middle layer, it increased up to a value of 17.7 µg/L (Figure 4.20), which was almost 520 times more than Case Study-7. In the upper layer, the dissolved lead concentration increased up to 2.9 µg/L (Figure 4.20), which was almost 85 times more than Case Study-7.

Figure 4.21 shows that it took around 12 years for the particulate lead to decrease to its initial concentration before discharge. Figure 4.22 shows that for the dissolved lead concentration it took approximately 14 years to decrease to the background concentration.

4.3.3 Case Study-9 – Full Corrosion of Cables

In this case study, full corrosion of the cables was assumed and all of the lead was released into the water column. Conditions of this case study were the same as in Case Study-3, where the particulate lead concentration increased from an initial concentration of 5.4E-4 mg/L up to 52.0 mg/L (Figure 4.7), which was almost 2 times more than Case Study-8 and 96000 times more than Case Study-7 (which indicates no corrosion). In the same layer, the dissolved lead concentration increased from an initial value of 3.4E-2 μg/L up to 185 μg/L (Figure 4.8), which was almost 2 times more than Case Study-8 and 5500 times more than Case Study-7.

In the middle layer, the particulate lead concentration increased to 8.1 mg/L (Figure 4.7) which was almost 2 times more than Case Study-7 and 15000 times more than Case Study-8. The dissolved lead concentration increased to a value of 35.4 μg/L (Figure 4.8), which was almost 1000 times more than Case Study-7 and 2 times more than Case Study-8.

In the upper layer, the particulate lead concentration increased to 1.27 mg/L (Figure 4.7), which was almost 2300 times more than Case Study-7 and 2 times more than Case Study-8. The dissolved lead increased to a value of 5.72 μg/L (Figure 4.8) which was almost 170 times more than Case Study-7 and 2 times more than Case Study-8.

The results of this investigation show that, with the corrosion rate increasing the water quality decreases in a proportional way.

4.4 Impact of Dredging System

Two different dredging systems were compared, hydraulic and mechanical dredging. Case studies were performed keeping the same flow rate (1.5 m^3/s), the same depth of cable (5 m), and the same amount of initial lead concentration contained in the cables. However, in each case, the kind of dredging technology was changed. A dredging time of 7 hours was kept constant in both cases. Another case study was done to study the effect of continuous and intermittent dredging activities.

4.4.1 Case Study-10 – Hydraulic Dredging

In this case study, the hydraulic dredging was used to dredge the sediments. The particulate lead concentration in the bottom layer increased from the initial value of 5.4E -4 mg/L to 27.0 mg/L (Figure 4.23) at the cable position, which was almost 20 times more than Case Study-6, (same cable situation but without dredging). In the middle layer. the particulate lead concentration increased to 16 mg/L (Figure 4.23). which was almost 51 times more than the concentration obtained in Case Study-6 without dredging. In the upper layer, the particulate lead concentration increased to 3.5 mg/L (Figure 4.23), which was almost 47 times more than Case Study-6.

The dissolved lead concentration increased to the same value of 1200 µg/L in all water column layers (Figure 4.24), which was almost 240 times more than the concentration of lead dissolved in the bottom layer in Case Study-6, and almost 880 times more than the concentration of lead dissolved in the middle layer observed in Case Study-6, and almost 3600 times more than dissolved lead concentration in the upper layer observed in Case Study-6.

Figure 4.25 shows that the plume of particulate lead fumed due to hydraulic dredging, extended to a distance of approximately 800 m away from location of dredging, and it took approximately 35 hours for the plume to reach equilibrium after 7 hours of dredging.

Figure 4.26 shows that the plume of dissolved lead, due to hydraulic dredging, was at the place of dredging and did not extend downstream. It took around 5 hours for the plume to reach equilibrium after 7 hours of dredging.

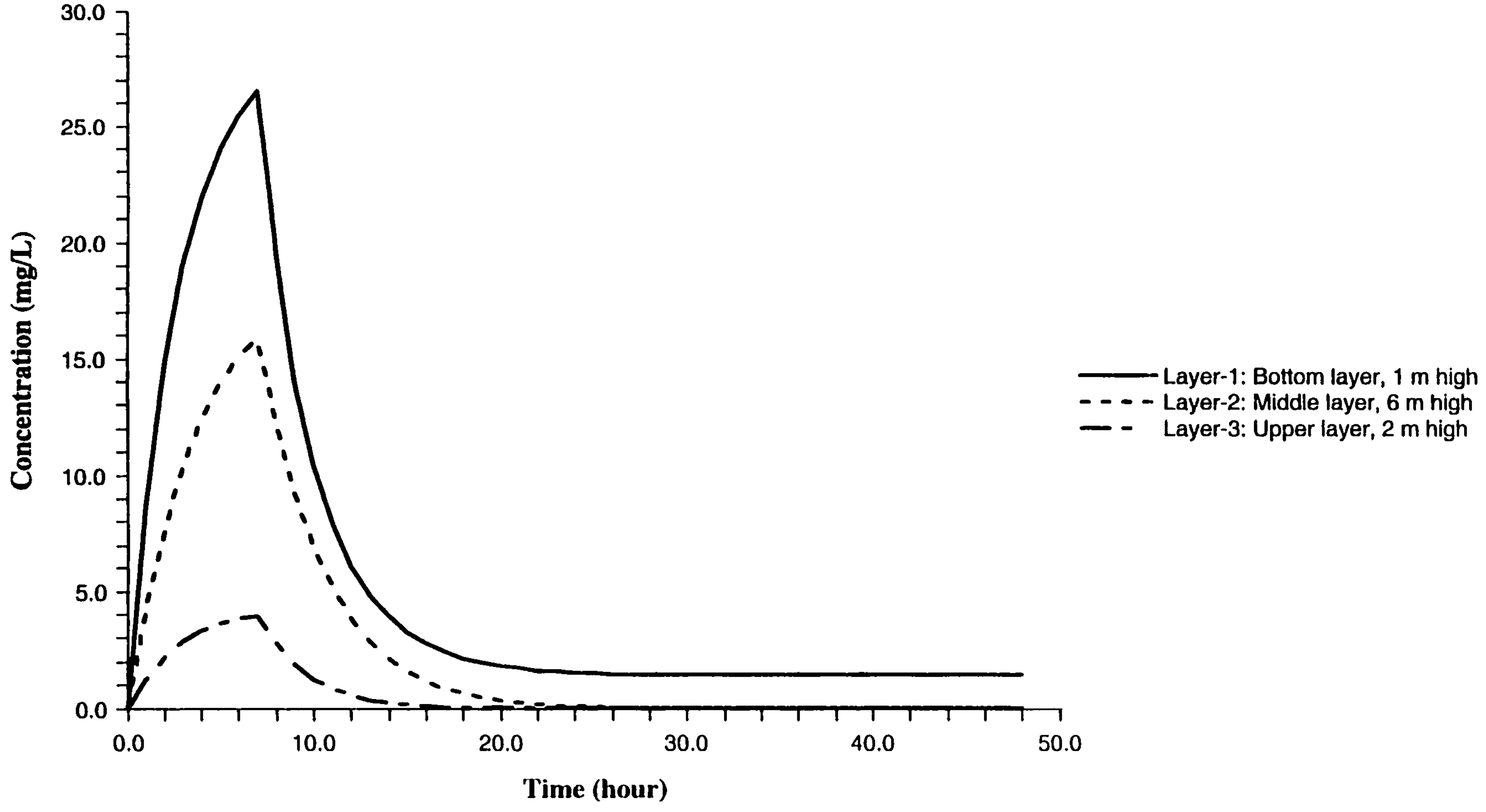


Figure 4.23: Particulate Pb concentration at cable position in different layers of the water column with time due to hydraulic dredging (Case Study-10).

Initial particulate lead concentration = 0.00054 mg/L

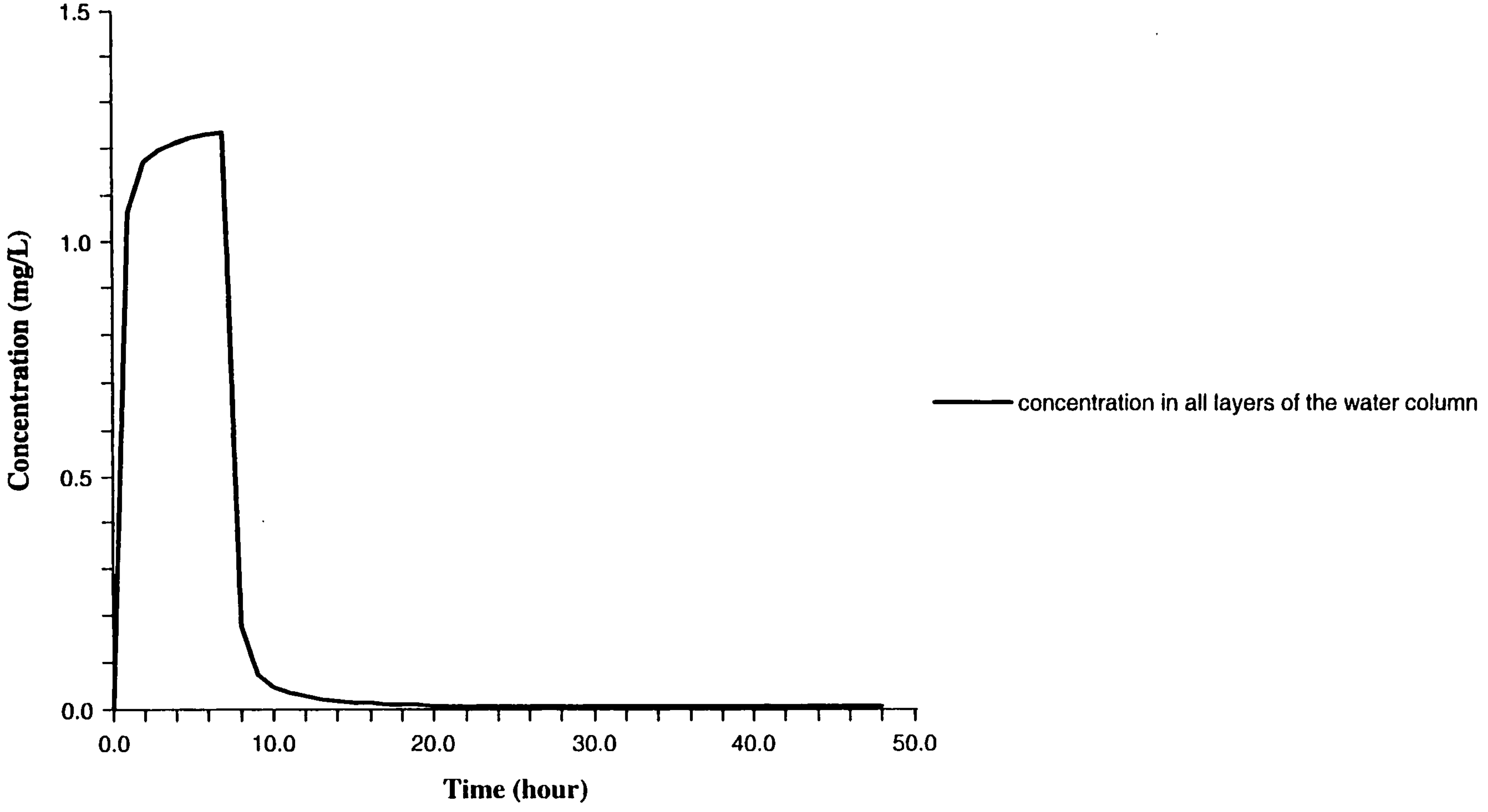


Figure 4.24: Dissolved Pb concentration at cable position in all layers of the water column with time due to hydraulic dredging (Case Study-10).

Initial dissolved lead concentration = 3.4E-5 mg/L

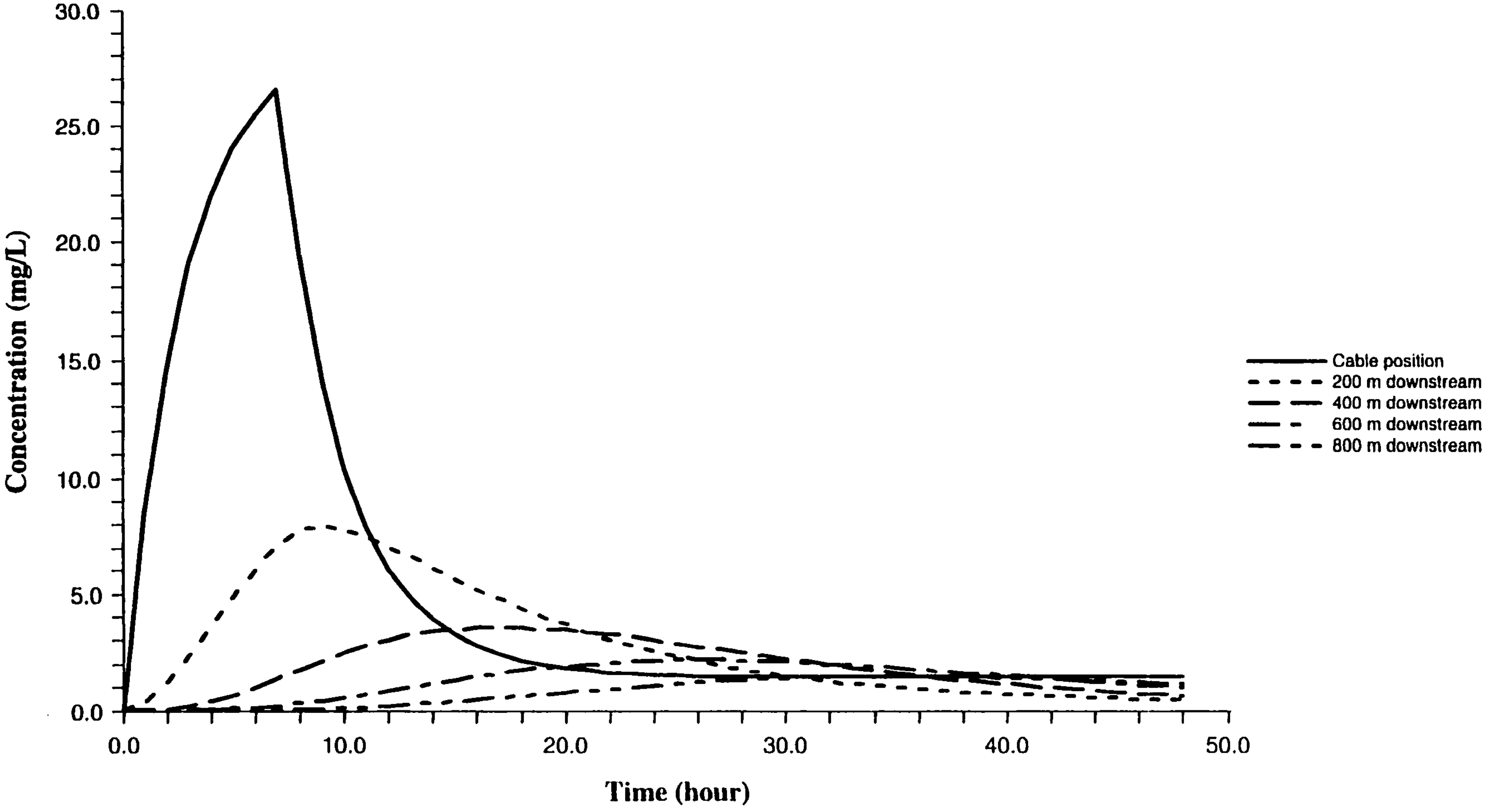


Figure 4.25: Particulate Pb concentration versus time at different downstream locations due to hydraulic dredging (Case Study-10).

Initial particulate lead concentration = 0.00054 mg/L

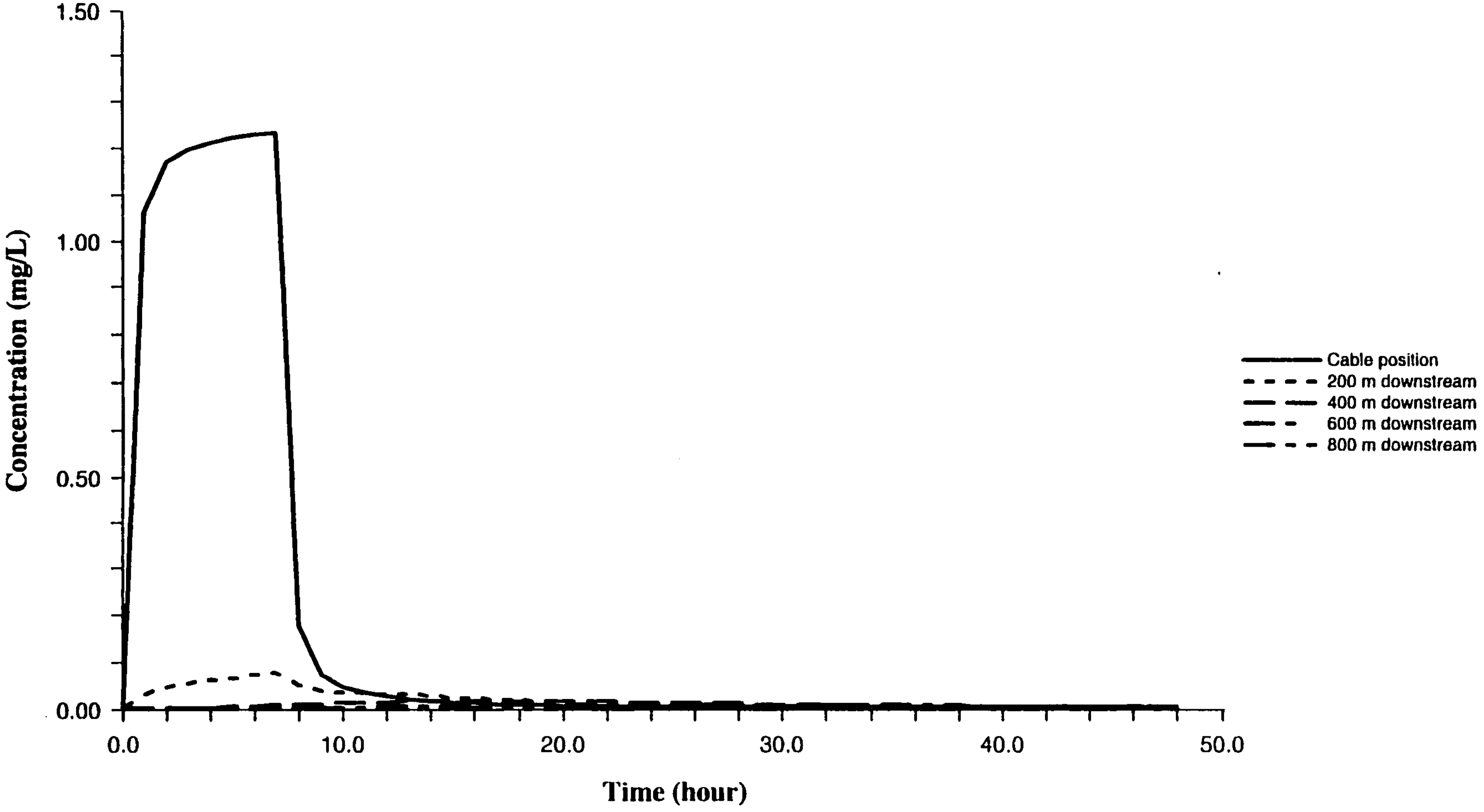


Figure 4.26: Dissolved Pb concentration versus time at different downstream locations due to hydraulic dredging (Case Study-10).

Initial dissolved lead concentration = 3.4E-5 mg/L

4.4.2 Case Study-11 – Mechanical Dredging

In this case study, mechanical dredging was used to dredge the sediments and remove the cables. The particulate concentration of lead in the bottom layer increased from the initial value of 5.4E-4 mg/L to 100 mg/L (Figure 4.27) at the cable position, which was almost 70 times more than Case Study-6 (with the same cable location but without dredging) and was almost 4 times more than Case Study-10 (with hydraulic dredging). In the middle layer, particulate lead concentration increased to 60 mg/L (Figure 4.27), which was almost 190 times more than the concentration obtained in Case Study-6 and almost 4 times more than Case Study-10. In the upper layer, the particulate lead concentration increased to 14.0 mg/L (Figure 4.27), which was almost 185 times more than Case Study-6 (without dredging) and 4 times more than Case Study-10.

The dissolved lead concentration increased to the same value of 4800 µg/L in all water column layers (Figure 4.28), which was almost 790 times more than the concentration of dissolved lead in the bottom layer in Case Study-6, and 3 times more than Case Study-10. In the middle layer, it was almost 2900 times more than the concentration of dissolved lead in Case Study-6, and 3 times more than Case Study-10. In the upper layer, it was almost 12000 times more than the dissolved concentration of lead in the upper layer of Case Study-6, and 3 times more than Case Study-10.

Figure 4.29 shows that the plume of particulate lead concentration due to the hydraulic dredging extended to a distance of more than 800 m away from the place of dredging. It took around two days (50 hours) for the plume to reach equilibrium after 7 hours of dredging which is more than in Case Study-10. However, at the same period of

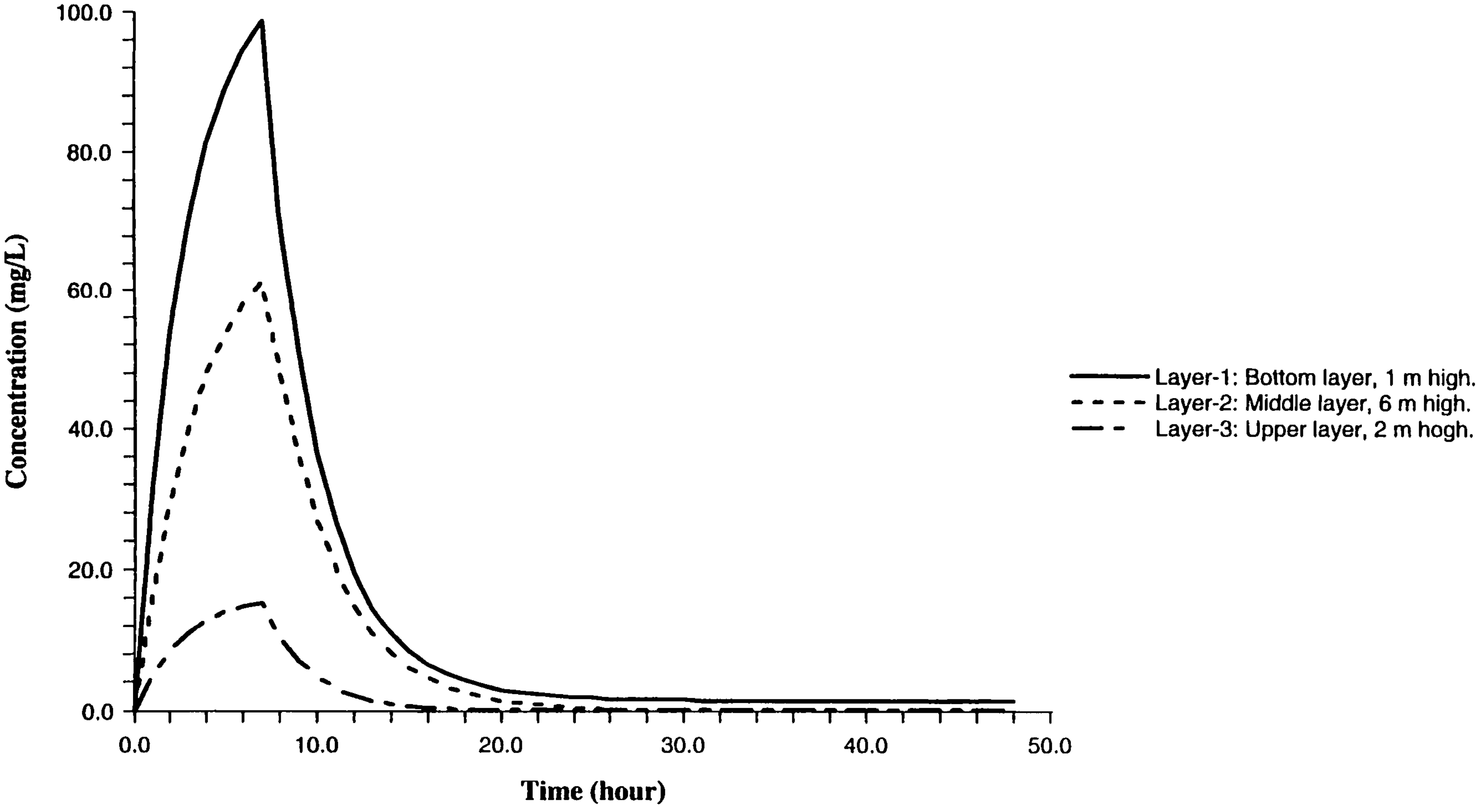


Figure 4.27: Particulate Pb concentration at cable position in different layers of the water column with time due to mechanical dredging (Case Study-11).

Initial particulate lead concentration = 0.00054 mg/L

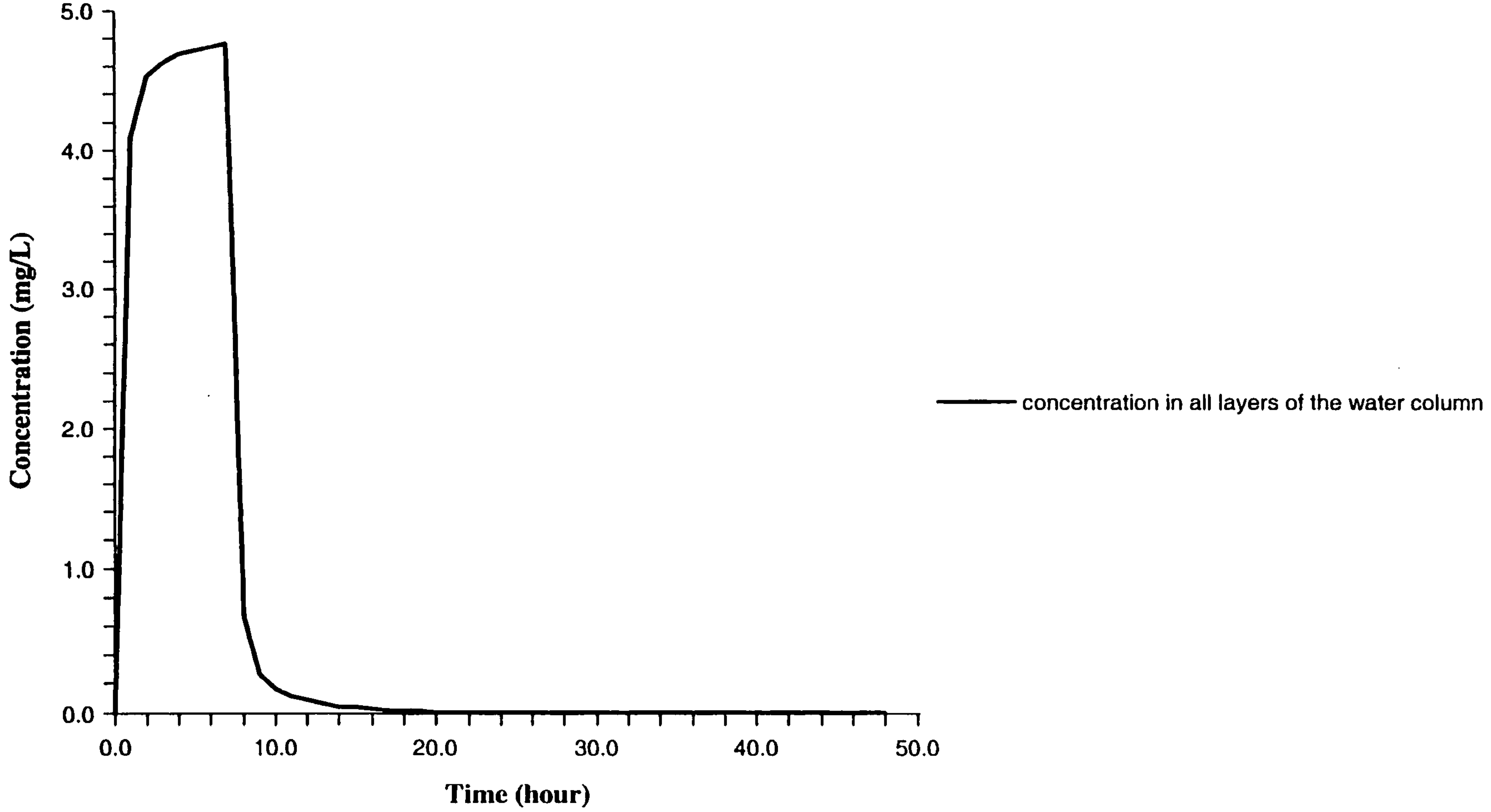


Figure 4.28: Dissolved Pb concentration at cable position in all layers of the water column with time due to mechanical dredging (Case Study-11).

Initial dissolved lead concentration = 3.4E-5 mg/L

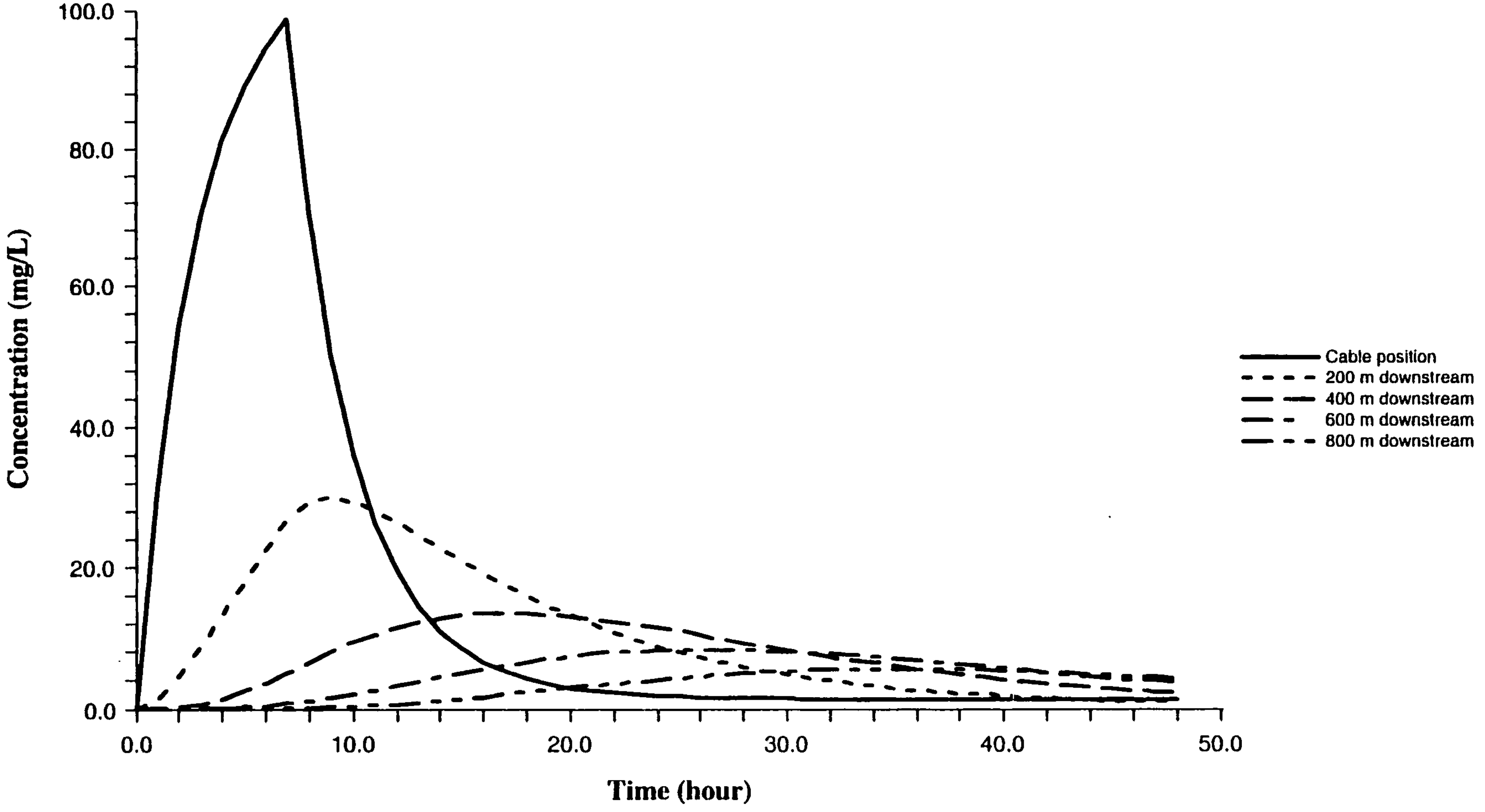


Figure 4.29: Particulate Pb concentration versus time at different downstream locations due to mechanical dredging (Case Study-11).

Initial particulate lead concentration = 0.00054 mg/L

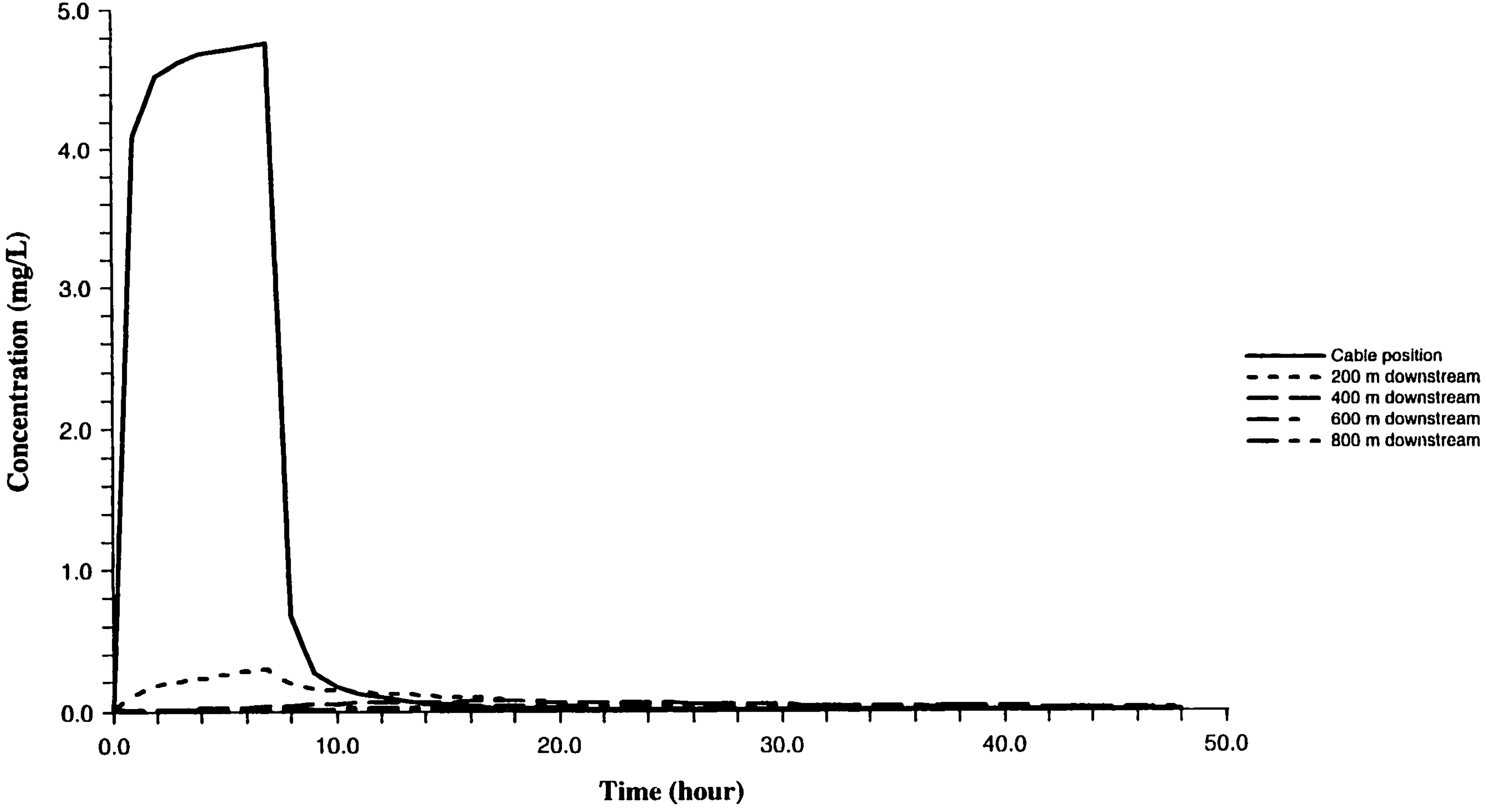


Figure 4.30: Dissolved Pb concentration versus time at different downstream locations due to mechanical dredging (Case Study-11).

Initial dissolved lead concentration = 3.4E-5 mg/L

time the concentration of particulate lead in the downstream was still higher than the initial concentration.

Figure 4.30 shows that the plume of dissolved lead concentration due to hydraulic dredging, was situated close to the place of dredging, and it took only 5 hours for the plume to reach equilibrium after 7 hours of dredging, which was the same as Case Study-10.

4.4.3 Case Study-12 – Mechanical Dredging & High Flow Rate

This case study was done to show the effect of high flow rate, such as 9000 m^3/s, during the dredging process. As in case studies 10 and 11, the same depth of cables at 5 m, the same amount of initial total lead concentration, and the same dredging time of 7 hours were used. In this case study, mechanical dredging was used to introduce the worst case, since it would produce more suspension than hydraulic dredging.

As shown in Tables 4.3 and 4.4, the concentration of particulate and dissolved lead remained constant during the dredging activity. The particulate concentration increased from an initial value of 5.4E-4 mg/L to 0.0198 mg/L in the bottom layer at the cable position, which was almost 5000 times less than Case Study-11 (which had the same cable situation but with a low flow rate of 1.5 m^3/s). In the middle layer, the particulate concentration of lead increased up to 0.00957 mg/L and in the upper layer, it increased up to 0.00289 mg/L.

The dissolved lead concentration in all layers of the water column increased from the initial value of 5.4E-4 mg/L to 0.0061 mg/L (Table 4.4) at the cable position during dredging, which was almost 800 times less than Case Study-11. After one hour of stopping the dredging process, both particulate and dissolved lead concentrations decreased to their initial values as shown in Table 4.3 and 4.4.

Table 4.3: Vertical particulate lead distribution due to mechanical dredging (Case Study-12: Flow rate 9000 m^3/s).

Time (hour)	Layer-1 (mg/L)	Layer-2 (mg/L)	Layer-3 (mg/L)
0	0.00054	0.00054	0.00054
1	0.0198	0.0096	0.0029
2	0.0198	0.0096	0.0029
3	0.0198	0.0096	0.0029
4	0.0198	0.0096	0.0029
5	0.0198	0.0096	0.0029
6	0.0198	0.0096	0.0029
7	0.0198	0.0096	0.0029
8	0.00101	0.00054	0.00054
9	0.00101	0.00054	0.00054
10	0.00101	0.00054	0.00054

Table 4.4: Vertical dissolved lead distribution in different layers of the water column due to mechanical dredging (Case Study-12: Flow rate 9000 m^3/s)

Time (hour)	Layer-1 (mg/L)	Layer-2 (mg/L)	Layer-3 (mg/L)
0	3.40E-05	3.40E-05	3.40E-05
1	0.0061	0.0061	0.0061
2	0.0061	0.0061	0.0061
3	0.0061	0.0061	0.0061
4	0.0061	0.0061	0.0061
5	0.0061	0.0061	0.0061
6	0.0061	0.0061	0.0061
7	0.0061	0.0061	0.0061
8	3.40E-05	3.40E-05	3.40E-05
9	3.40E-05	3.40E-05	3.40E-05
10	3.40E-05	3.40E-05	3.40E-05

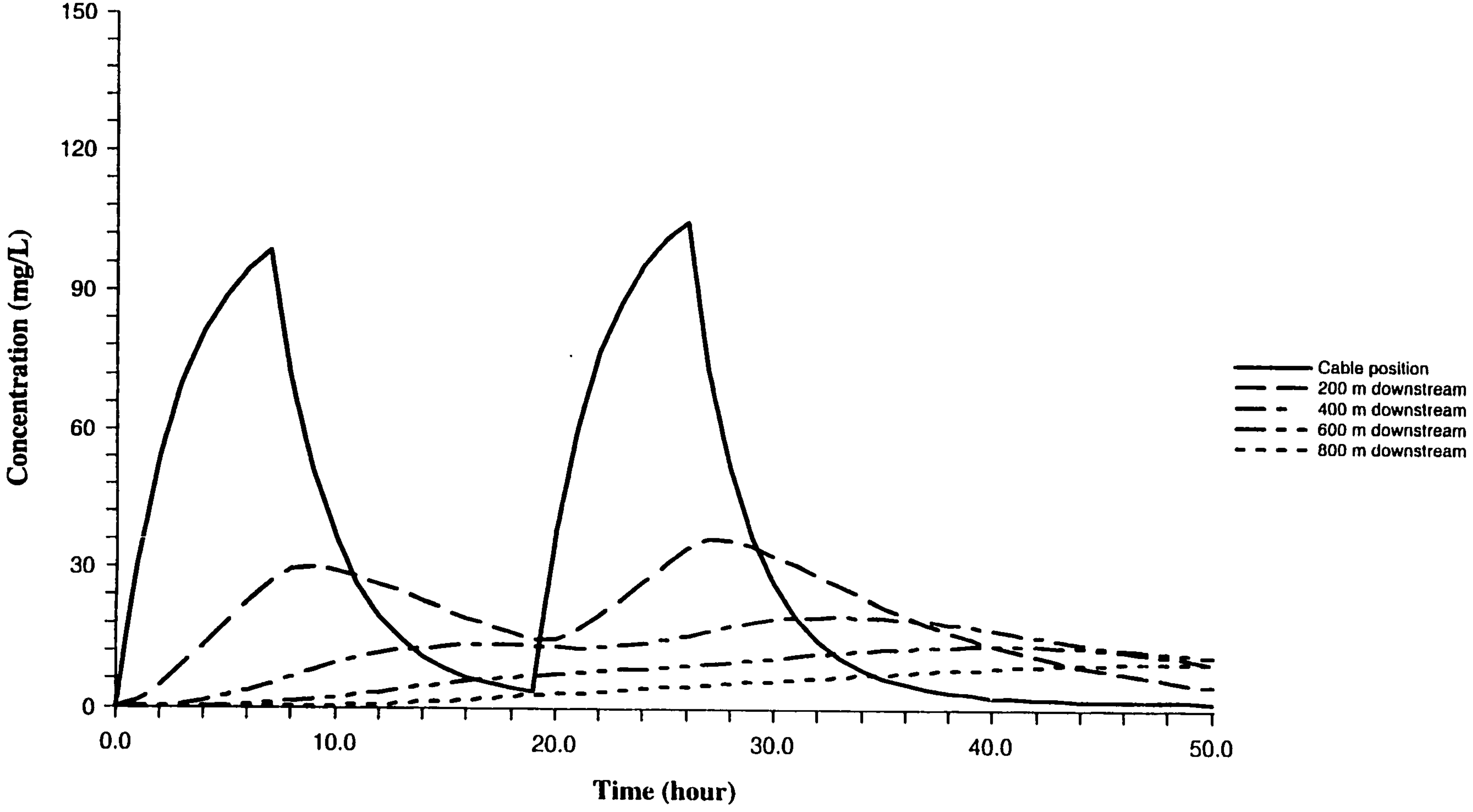


Figure 4.31: Particulate Pb concentration due to mechanical dredging for two consecutive days at different downstream locations (Case Study-13).

Initial particulate lead concentration = 0.00054 mg/L

4.4.4 Case Study-13 – Dredging for Two Consecutive Days

As shown in case studies 10 and 11, after dredging for one shift (7 hours), it took approximately two days for the particulate lead concentration to decrease and reach equilibrium. Case Study-13 was performed to show the effect of dredging for two consecutive days. As shown in Figure 4.31, if the dredging process starts again immediately on the next day, the particulate lead concentration will increase again before reaching equilibrium. By the end of the dredging process, the concentration will reach a value of 105 mg/L at the cable position, in the bottom layer. Downstream, after starting dredging on the next day, the particulate lead concentration will also start to increase before decreasing and reaching equilibrium. In this case, dredging for two consecutive shifts, it will take three or more days for the lead concentration to reach initial values. Higher lead concentrations will be reached and longer exposure periods for biota will occur if dredging is done in continuous shifts.

4.5 Conclusion

Changes in the flow rate had a large impact on the concentration of released lead into the water column. A 300-fold decrease in flow rate provoked an increase in lead concentration by 1000 times. The highest effect was observed in the layer close to bottom sediments. In addition, it was concluded that the measurements, which were not performed in an entire water column, could give a wrong estimation concerning benthic conditions, usually influencing the areas of fish spawning.

Changes in flow rate had the largest effect on the time required for the lead concentration to decrease to its initial value. For the case of a high flow rate, the time required for the lead concentration to decrease to its initial concentration was 17 times less than the case of a medium flow rate and almost 100 times less than the case of a low flow rate.

The cables buried at a level of 1 m decreased the release of lead to the water column by 5 to 10 times less than the case when cables were laid on the surface. By increasing the depth of cables from 1m to 5 m, the release of lead species to the water column can decrease from 4 to 17 times less than the case when cables were laid on the surface. Consequently, it was concluded that the depth of cable would have little impact on the concentration of lead released into the water column.

The water quality decreases proportionally with the increase of corrosion. It was also found that the use of hydraulic dredging techniques could decrease the concentration of particulate and dissolved lead almost 4 times when compared to mechanical technologies. In addition, the plume resulting from hydraulic dredging would extend to a shorter distance of 200 m away from the place of dredging. It was also found that during dredging, the dissolved lead concentration increased only in the place of dredging. A small increase in the dissolved concentration was found in the downstream.

Higher lead concentrations will be released to the water column and longer exposure periods for biota will occur if dredging is done in continuous shifts. If mechanical dredging is done in two consecutive shifts, in rivers with a low flow rate, three days or more will be needed for the lead concentration to reach initial values at the place of dredging. At the same period of time the concentration of particulate lead was still approximately 18000 times more than the initial concentration at 600 m in the downstream. It is recommended that two consecutive dredging projects have a time gap of at least two days, in order that the auto purification process could take place.

CHAPTER 5

VULNERABILITY OF AQUATIC SPECIES TO LEAD RELEASED FROM CORRODED CABLES

Adverse effects caused by the exposure of aquatic organisms to chemical substances are determined through the use of applied toxicological methods. Because the fundamental mechanisms that cause toxic responses are highly complex and are not fully understood, toxicological findings are largely based on experimental observations.

Adverse effects are described in relation to the time of exposure, frequency, and administered dose. The exposure period over which the dose is administered is divided into short and long-term toxicity tests.

Short-term toxicity tests, often referred to as acute toxicity tests, are used to evaluate severe effects due to the exposures for short periods of time. The LC_{50} is used to measure lethality or exposure in an aqueous solution. The standard test for measuring acute lethality is designed to produce a 50% lethal response after a 96-hour exposure duration. Responses for shorter exposure durations are often cited, but the 96-hr LC_{50} is recognized as the standard (LaGrega *et al.*, 1994).

Long-term toxicity tests are commonly referred to as chronic toxicity tests. These tests measure effects on development, growth, and reproductive potential. Since chemical concentrations associated with long-term effects are typically lower than those that produce acute responses, long-term toxicity usually provides a more sensitive measurement of toxicity (LaGrega *et al.*, 1994).

5.1 Toxicological Properties of Lead

Lead is rapidly transported and accumulated in bones of living organisms, where it tends to bioaccumulate in bones throughout life. The liver and kidney, of the soft tissues, tend to have somewhat elevated lead levels (Manahan, 1989).

"The most common biochemical effect of lead is the inhibition of the synthesis of heme, a complex of a substituted porphyrin and Fe^{+2} in hemoglobin and cytochromes. Lead interferes with the conversion of delta-aminolevulinic acid to porphobilinogen with a resulting accumulation of metabolic products. Lead inhibits enzymes that have sulfhydryl groups. However, the affinity of lead for the -SH group is not as great as that of cadmium or mercury" (Manahan, 1989).

Psychological symptoms due to the effect of lead on the central nervous include restlessness, dullness, irritability, and memory loss. In cases of extreme exposure, convulsions followed by coma and death may occur (Manahan, 1989).

Another important group that has an adverse effect on the health is represented by organolead species. Toxicological research has concentrated on some organolead compounds among them: tetraethyllead, dimethyldiethylead, trimethylead chloride, and diethyllead dichloride. The toxicological effect of tetraethylead is different from that of inorganic lead. "Symptoms of tetraethylead poisoning reflect effects upon the central nervous system. Among these symptoms are fatigue, weakness, restlessness, ataxia, psychosis, and convulsions. Death can occur as soon as one or two days after exposure" (Manahan, 1989). The organolead toxic action is due to its metabolic conversion to triethyl form (Manahan, 1989).

5.2 Effect of Lead on Aquatic Species

The most commonly used test endpoint for fish is the 96 h LC_{50} for adult or juvenile (post-larval) individuals. A problem with LC_{50} is that "in most cases only the response at 96 h is reported. Many assessments involve transient events, in which case the time to mortality is more important than the percent mortality" (Suter II *et al.*, 1993).

The lethal dose (LC_{50}) for lead differs between salmonid and non-salmonid fish. LC_{50} of lead for non-salmonid fish occurs at 900 µg/L for 40 days exposure and 550 µg/L for 90 days exposure (Crompton, 1977). The LC_{50} of lead for salmonid fish occurs at 1000 µg/L for 48 h exposure and 500 µg/L for 96 h exposure (LaGrega *et al.*, 1994). According to Crompton (1977) long-term (365 days) exposure of lead leading to mortality of non-salmonid fish was 6 µg/L.

Crompton (1997) showed that Annelids, Rotifers, Insects, Crustaceans and Gastropods are less sensitive to lead than non-salmonid fish and Gastropods. Short-term (4-14 days) exposure leading to mortality is associated with the following lead concentrations in water:

- Annelids C> 10 000 µg/L
- Rotifers C> 10 000 µg/L
- Insects C> 10 000 µg/L
- Crustaceans C> 10 µg/L
- Gastropods C> 10 µg/L

5.3 Vulnerability of Aquatic Species in the Montreal Area Based on Modeling

According to St. Lawrence Center (1993b) the most common species that appear in the St. Lawrence River in the Montreal area are: Rock bass, Pumpkinseed, Northern pike, Black crappie, Longnose sucker, White sucker, American shad, Lake sturgeon, Walleye, and Channel catfish.

Table 5.1 presents a general assessment of aquatic species sensitivity to lead. This assessment has been done for each case simulated with *MFCL* model (Chapter 4). The assessment considered: acute toxicity for 40 and 90 days for non-salmonid fishes, acute toxicity for 48 and 96 hours for salmonid fishes and short term exposure (4-14 days) for aquatic creatures.

Case Studies 1 and 7 introduced in Chapter 4 reflect the condition of the St. Lawrence River in the Montreal area. The concentrations of 0.01345 mg/L and 3.61E-2 μg/L were found for particulate and dissolved lead respectively. It can be concluded that all aquatic species exposed during the period from 48 to 96 h are not affected by these administrated lead concentrations.

Results from Case Studies 2 and 3 (Chapter 4) represent conditions of small rivers, harbors or bays. The concentrations (18 and 52 mg/L) in deep waters of these cases can have an adverse effect on all species.

According to Case Studies 5 and 6 (Chapter 4) where the impact of cables laid at 1 m and 5 m depth was investigated, a minimal effect on aquatic species would occur. However, a predicted impact on aquatic creatures such as Crustaceans and Gastropods has been found.

Table 5.1: Assessment of aquatic species sensitivity to lead released in different conditions described in Chapter 4.

Case Study (Chapter4)	Non-salmonids (Chapter 5.2)		Salmonid (Chapter 5.2)		Aquatic Creatures (Chapter 5.2) Short-term (4 - 14 days) exposure				
	LC50 40 days	LC50 90 days	LC50 48h	LC50 96h	Annelids	Rotifers	Insects	Crustaceans	Gastropods
Case Study-1	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*
Case Study-2	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*
Case Study-3	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*
Case Study-4	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*
Case Study-5	*AL*	*AL*	*AL*	*AL*	*BL*	*BL*	*BL*	*AL*	*AL*
Case Study-6	*AL*	*AL*	*AL*	*AL*	*BL*	*BL*	*BL*	*AL*	*AL*
Case Study-7	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*
Case Study-8	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*
Case Study-9	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*	*AL*
Case Study-10	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*
Case Study-11	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*
Case Study-12	*BL*	*BL*	*AL*	*BL*	*BL*	*BL*	*BL*	*BL*	*BL*

Legend: *BL*: Below Limit *AL*: Above Limit

According to Case Studies 10 and 11 which reflects dredging for an 8 h shift only, calculated lead concentrations were less than the LC_{50} value. However, timing of dredging activities, i.e. intermittent dredging for more than one day, had impact at least on salmonid fishes for 48 h exposure time. Other, ecological risk assessments have to be performed for long term effects. "Toxicity tests for fish that are termed chronic include

(1) full life cycle tests that go from egg to egg and may require two years, (2) partial life cycle tests that go from adults to juveniles, (3) early life stage tests that go from eggs to juveniles, and (4) five to eight day "short-term chronic" tests involving eggs and larvae or just larvae" (Suter II *et al.*, 1993)

Fish species common in the Montreal area have specified spawning, egg laying and hatching times. Spawning periods, temperature required for spawning, acceptable current, and depth at which the spawning occurs, are characterized in Table 5.2 (St. Lawrence Center, 1993b)

Table 5.2: Characterization of spawning habitats (St. Lawrence Center, 1993b).

Species	Current (cm/s)	Depth (m)	*Substrate	Temperature (°C)	Spawning Period
Rock bass	0 - 40	0 - 2	CSaGR	15 - 21	mid. May - mid June
White sucker	0 - 110	0.3 - 2.3	SaGR	9 -18	early May - early June
Pumpkinseed	0 - 20	0.1 - 0.3	CSaGR	20 - 28	mid. April - mid July
Northern pike	0 - 20	0.1 - 1.2	SC	4 - 15	April - early May
Black crappie	0 - 20	0.2 - 2.5	SSaG	18 - 20	mid April - mid July
Longnose sucker	0 - 45	0.1 - 0.3	SaGSi	5 - 18	mid March - mid May
Lake sturgeon	≥10	0.6 - 4	SaGR	8 - 18	mid April - mid June
American shad	≥20	0.5 - 2	SaGSi	12 - 18	mid April - mid June
Walleye	0 - 200	0.1 - 1.5	SaGR	3 - 11	mid March - mid June
Channel catfish	0 - 60	0.6 - 1.8	SSaG	20 - 30	mid May - mid July

*SUBSTRATE
C: Clay, S: Silt, Sa: Sand, G: Gravel, Si: Shingle, R: Rock

It can be seen from Table 5.2 that spawning periods of these fishes are generally between April and July. The acceptable temperature varies from 0 to 30°C and spawning areas are characterized by low current velocity. At the same time *MFCL* model (Chapter 6) shows that the highest impacts are associated with low flow velocities. Table 5.2 shows that spawning areas are close to the bottom sediments which are recognized in the *MFCL* model to be the most affected area. To assess the entire impact of dredging activities tests on early life stages (that goes from eggs to juveniles) have to be performed; these tests can evaluate long-term effects on populations. For time being it is recommended to avoid the spawning period of dredging activities.

CHAPTER 6

GENERAL CONCLUSIONS AND RECOMMENDATIONS

The literature review demonstrated that an environmental impact associated with the removal of sub-marine cables did not get adequate attention from researchers.

Publications provide information about dredging activities but no information is given on monitoring the removal of these objects from freshwaters.

A few dredging activities, with environmental monitoring, were reported; however they did not provide all the necessary data to allow for an accurate assessment of environmental impacts.

In order to obtain an accurate relationship among the factors affecting the release of corroded lead and cable removal a simulation model known as Model for Fate of Corroded Lead (*MFCL*) was constructed. The *MFCL* was capable of investigating a number of factors affecting water quality and biota. Among these factors are the hydrological and physical properties of the water body (e.g. flow rates, resuspension and settling), situation and position of cables (e.g. on surface or in different depths), physico-chemical characteristics (e.g. adsorption, desorption and diffusion rate), seasonal factors, transport of toxicant (e.g. advective and dispersive mixing) and dredging activities (e.g. mechanical and hydraulic).

The *MFCL* model is a tool that manages the factors influencing the fate of lead and other heavy metals. This model is powerful and it can provide a free choice of variable and defined values for calculating dissolved and particulate metal distribution in different layers of the water column for any cable position. In addition, the model provides a sensitivity analysis concerning the effect of different scenarios on the biota.

The model evaluated the impact of the engineering work schedule by looking for the best scenarios for dredging activities within a continuous or intermittent time frame. Higher concentrations will be reached and longer exposure periods for biota will occur if dredging is done in continuous shifts. For two consecutive dredging projects, it is recommended to have a time gap of at least two days, in order that the auto purification process could take place.

Also the model was able to calculate dissolved and particulate lead concentrations at different downstream locations due to dredging up to 800 m. It was concluded that dredging in harbors (the most contaminated areas) and bays (most populated by biota) are the worst scenarios.

Results from the *MFCL* model provided important information about ranking of particular factors. Among them, the size of the river and hydrological conditions play a leading role. The time required for the lead concentration to decrease to its initial concentration was 17 times less than the case of a medium flow rate and almost 100 times less than the case of a low flow rate. The rate of corrosion had a proportional relation with the water quality. For both cases, with entire and partial corrosion, it took approximately 15 years for lead to reach its initial value.

In rivers with medium flow rates (e.g. Chateauguay River) and in bays or harbours, for all cable positions (on sediment surface or below), and with partial or full corrosion, lead concentrations released into the water column were higher than the LC_{50} value for salmonid, non-salmonid, Crustaceans and Gastropods. Results showed that concentrations of lead released due to both mechanical and hydraulic dredging (for one day 7 hours shift), were less than the LC_{50} value for non-salmonid and salmonid fish. However, the concentration of lead in the water column due to dredging for two consecutive days (7 hours shift each day) was higher than the LC_{50} value for salmonid fish. Consequently, dredging techniques, protection barriers and monitoring have to be carefully designed since sensitivity of aquatic biota seems to be important factor. In addition, recognition of the spawning area and season should be considered. In general, dredging activities within the Montreal area must be avoided in spring.

A deep positioning of partially corroded cables seems to have an important impact on water quality. Corrosion of cables in aerobic conditions can provoke an adverse effect on benthic organisms. The highest effect was observed in the layer close to bottom sediments. This information is important for toxicologists evaluating and studying the effect on population, and also for engineers to prepare accurate protocol for sampling and measurement.

It is recommended:

- To proceed with an investigation based on well-designed field studies, which provide actual input data to the *MFCL*. The *MFCL* has to be integrated with software performing the speciation of toxic compounds (e.g. FACT), and 3D graphical distribution (e.g. GIS).

- To carry laboratory tests useful for collection of data regarding principal physico-chemical processes, sediment interface processes, and corrosion rate, etc.

- To develop new tools for cable position detection and corrosion rate estimation in the field.
- To run a series of bioassay tests for biota from a particular area in an ecotoxicological lab.
- The application of new sonar equipment would avoid several problems. Removal of sub-marine cables is associated with dredging activities, consequently, mass of dredged material depends on cable positioning within the water body. The precise detection of cable position will permit an accurate estimation of corrosion conditions, type of sediments and their porosity. It also contributes to the decrease of the volume of sediments required for removal.
- The *MFCL* model can be useful for the evaluation of other metals fate in aquatic environments.

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