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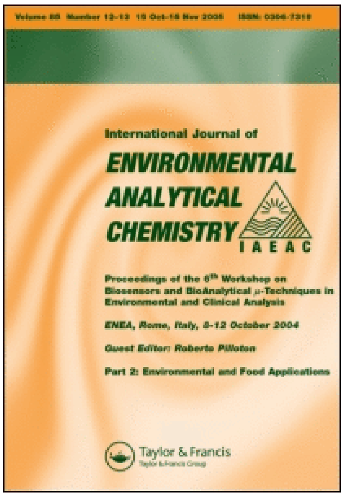

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CHARACTERIZATION OF SEDIMENTS IN AERATED LAGOONS AND WASTE STABILIZATION PONDS

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Sediments affect the performance of aerated lagoons and waste stabilization ponds in many ways. This paper presents the results of a three years study conducted on real-size facilities and implementing numerous analytical procedures. Sediment accumulation rates and physico-chemical characteristics are described as well as their activity in terms of oxygen consumption and exchange rates with overlying waters.

The deposits had a mean accumulation rate of 4.7 cm per year, their main characteristics being low viscosity and high organic content (> 30%). They also accumulated numerous organic and mineral compounds, such as nutrients and heavy metals. Vertical concentration profiles, measured in the sediment and interstitial liquid phases, are presented and discussed. These results emphasize the importance of surface activity.

Sediment oxygen demand, which can be divided into biological and chemical components, ranged from 1 to 3 $gO_2/m^2 \cdot d$. Finally, the exchange rates of COD, nitrates, ammonia and orthophosphates existing at sediment-water interface were quantified under several redox conditions and substrate additions.

Keywords: Aerated lagoon; stabilization pond; sediment; accumulation rate; oxygen demand; sediment-water interactions

INTRODUCTION

Stabilization ponds and aerated lagoons are extensive means of wastewater treatment that differ from other biological processes by the fact that biological sludges and others suspended particles settle directly inside their basins. The gradual accumulation of these sediments at ponds bottom obviously affect their performances by reducing ponds volume and shortening their hydraulic resi-

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dence time.[1] However, the role played by these deposits is not restricted to such a physical effect. Far from being inert, sediments represent an important oxygen sink which must be taken into account when designing aerator power and oxygen supply. Moreover, an effective model of ponds purification kinetics should include an appropriate description of all the interactions, feedbacks and exchanges existing at the sediment-water interface.[2]

As a major source or sink of pollutants in aquatic environments, deposits definitely contribute to the removal yields of aerated lagoons and waste stabilization ponds, especially for nitrogen and phosphorus compounds. The mechanisms involved in these adsorption and remobilization processes, which are dependent on a subtle web of environmental conditions (pH, redox potential, temperature, bacterial activity,...) can also be the cause of numerous types of dysfunction, such as the release of foul odors or resuspension of deposits.

Even when they are removed from ponds bottom, sediments still pose environmental problems of sludge treatment, disposal and reuse. Deposits from aerated lagoons and stabilization ponds are likely to contain pathogenic microorganisms as well as organic and inorganic contaminants which may be hazardous or toxic to humans and could therefore limit their suitability for agricultural applications. For these reasons, the proper handling of these sludges is one of the most important components of the design and operation of these two wastewater treatment processes.[3]

Our paper intends to characterize these sediments as completely as possible by measuring their accumulation rates, analyzing their physical, chemical and bacteriological properties, and estimating their oxygen consumption and rates of carbon and nutrient exchanges with overlying water. In order to understand sediment-water interactions better and to study the evolution of sludges composition, vertical concentration profiles were measured in the solid and liquid phases of deposits. Diffusive and overall benthic fluxes at the sediment-water interface were also estimated.

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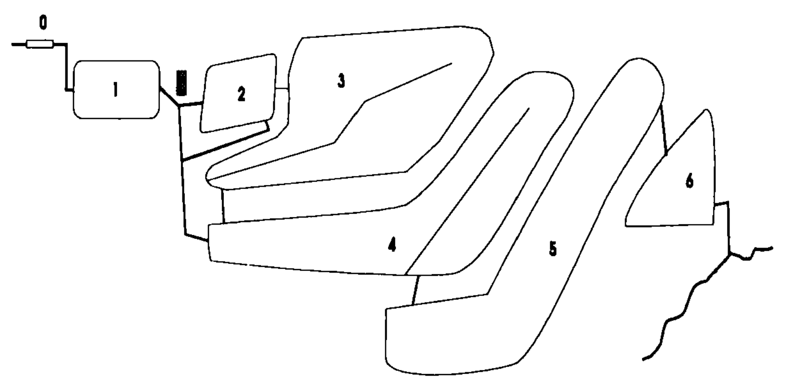
EXPERIMENTAL

Description of Bertrix Wastewater Plant

The data presented in this study were collected at the Bertrix wastewater plant. This facility was designed and built for experimental purposes, especially for comparison of aerated lagoons and stabilization ponds, as well as surface and diffused air aeration. Operational since the early 90's, this plant was designed for a nominal capacity of 7,500 inhab.eq. It treats mostly domestic sewage from



0 Preliminary treatment
1 Aerated lagoon (surface aerators)
2 Aerated lagoon (air diffusion)
3 Stabilization pond with deflector
4 Stabilization pond with deflector
5 Stabilization pond
6 Maturation pond

	Volume (m^3)	Surface (m^2)	Depth (m)	Length X width
Pond 1	3404	1710	2.7	70 X 26.5m
Pond 2	3518	1905	2.7	58 X 34m
Pond 3	3100	6200	0.5	255 X 23m
Pond 4	3700	7400	0.5	368 X 18.5m
Pond 5	3900	5000	1.5-0.5	199 X 25m
Pond 6	2000	1500	2	42.5 X 55m

FIGURE 1 Schematic representation of Bertrix wastewater plant.

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the rural community of Bertrix but also handles the industrial wastes from a small slaughterhouse. Those effluents are extremely diluted since the combined sewer system which serves the plant carries huge amounts of rainwater, storm water and parasite water.

Primary treatment consists of a mechanically cleaned bar screen, followed by grit removal and a rotary drum screen. The secondary treatment is carried out by a series of two aerated lagoons and four facultative stabilization ponds, combining microphytes and macrophytes. Their hydraulic circuit can be modified at will by means of numerous floodgates distributed over the plant's 4 ha site (Figure 1).

To limit the risk of seepage, which is particularly severe in this type of slate-and-sandstone bedrock, the bottoms of the two aerated lagoons have been covered with a waterproof membrane.

The average residence time in the entire plant is about one week, with influent wastewater flowrates varying from 70 to 300 m^3/hr, depending on weather conditions. Belgian standards for such facilities impose a maximum BOD at plant's outlet of 15 mg/l.

Materials and Methods

A three years study was conducted at Bertrix wastewater plant in order to characterize the sediments accumulated at the bottom of the ponds. All the different aspects and roles played by these deposits were considered. This entailed setting up and implementing a host of analytical procedures.

- *Accumulation rate*—Deposit thicknesses encountered after 6 years of operation were estimated by depth sounding in the two aerated lagoons and in the first stabilization pond; these three basins showing the highest accumulation rates. Measurements were conducted every 2 m, over the entire surface area of the ponds in order to reconstitute an accurate 3-dimensional view of their bottoms and locate the areas where suspended matters mainly settle. At the same time, about ten sediment traps were placed on the pond floors and collected after a 1 year exposition period. These traps, which consist of 30 cm diameter PVC plates, were also used to assess sludges physico-chemical characteristics.
- *Physico-chemical and bacteriological characteristics*—The main physico-chemical analysis were conducted on the material collected with sediment traps, regardless of their thickness. Sludges density, percentage of dry solids, granulometry, organic content, humic acids, organic carbon, total nitrogen and total phosphorus were determined on fresh sediments. Their concentrations in Fe, Zn, Mn, Pb, Cu, Ni, Cr, Hg and Cd were also monitored, as were the hydrolysable fractions of nitrates, ammonia and orthophosphates. All these characteristics were determined by procedures taken from *Standard Methods*.[4]

  Indicator organisms of faecal contamination present in the sludges were isolated by culturing one-gram samples of suspended sediments on various solid media. Aerobic bacteria were determined on Plate Count Agar (30°C, 48h). Total and faecal coliforms were isolated in lactose broth with bromocresol purple, incubated during 48h at 30°C and 44°C, respectively. Faecal streptococci were estimated according to Slanetz and Bartley method (37°C, 48h). Sulfate reducing clostridia were isolated by incorporation in sulfite agar (44°C, 24h).

  Vertical concentration profiles of organic carbon, total nitrogen and total phosphorus were determined by means of sediment cores, frozen and cut in 2 cm slices. ATP concentrations were measured as an indicator of sludge activity at different depths using a bioluminescence technique. Finally, the oxygen gradient at the sediment-water interface was estimated using an oxygen microsensor, described in the literature,[5] coupled with a micrometer screw.

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- *Composition of interstitial waters*—2 cm slices of the sediment cores were centrifuged at 5000 rpm for 10 mn to collect the liquid phase of the deposit. COD, nitrates, ammonia and dissolved orthophosphates were determined in each fraction.
- *Sediment oxygen demand*—To compare the various analytical procedures described in the literature, sediment oxygen demand was measured by means of respirometric tests conducted on disrupted and undisrupted sludges samples. «Non-disruptive» measurements were carried out either *in situ* using a benthic chamber or in the laboratory using sediment cores. The laboratory measurements were performed with and without substrate addition, *i.e.*, by replacing overlying water by oxygen-saturated dilution water with or without a mixture of glucose and glutamic acid, equivalent to a BOD of 100 mg/l. When the tests were conducted in the presence of substrate, an acclimation period of 12 hours proved necessary before any measurements could be performed. «Disruptive» measurements consisted in determining the rates of oxygen consumption of suspended sediments maintained under continuous stirring. To quantify the chemical component of these respirations, respirometric tests were also performed after the addition of $HgCl_2$, which, by acting as a metabolic inhibitor, stops all biological activity.
- *Exchange rates at sediment-water interface*—Exchanges of organic matter, nitrogen and phosphorus compounds at the sediment-water interface were studied by means of sediment cores taken from the different lagoons and placed in the laboratory in the presence of a synthetic wastewater of known composition. To prevent gradient formation, the overlying waters were kept under continuous stirring while avoiding sludge resuspension. COD, nitrates, ammonia and dissolved orthophosphates in the liquid phase were monitored daily or weekly, depending on the experiment's duration, *i.e.*, one week or one month.

Our study will present the results of more than 200 sediment cores taken from aerated lagoons and waste stabilization ponds. Almost fifty respirometric tests were conducted on disrupted sediments while ten SOD determinations were carried out *in situ* with a benthic chamber. Sediment-water interactions were quantified by means of a dozen analysis campaigns. At the same time, sludges physico-chemical characteristics were measured on eight sediment traps and ten vertical concentration profiles.

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RESULTS AND DISCUSSION

Accumulation Rate

Sludge accumulation rates, estimated by means of sediment traps, varied between 1.8 and 7.5 cm per year, with an average of 4.7 cm per year for the entire plant. Although this value is relatively high, it remains within the ranges usually found in the literature.[6,7] More recently, slower accumulation rates of less than 1.5–2 cm per year were reported for waste stabilization ponds preceded by a sedimentation tank.[8] This demonstrates the superiority of this primary treatment system over the rotary drum screen set up at Bertrix wastewater plant.

Nevertheless, the accumulation rates estimated with sediment traps are certainly maximal since some traps are likely to have sunk into pre-existing sediments because of their great thickness and low density.

Sludge thickness after 6 years of operation frequently exceeded 30 cm. Depth sounding revealed that the sediments were not spread uniformly across the whole pond surface. As already noticed by several authors,[3,9] deposits naturally tended to build up in the least mixed areas, *i.e.*, away from aeration devices, near embankments, and especially at ponds inlet and outlet where they formed sedimentation cones.

Of course, the first basins, which receive raw sewage, are always characterized by the most important silting up. For example, at Bertrix wastewater plant, the thickest sediments were most frequently encountered in the two first aerated lagoons but also in the first waste stabilization pond, which, despite its shallowness, acts as a real settling tank for the sludges that are resuspended during aerator operation in the two previous lagoons.

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Physical, Chemical and Bacteriological Characteristics

Table I presents the physico-chemical characteristics assessed by means of sediment traps. The deposits, which had a mean density of less than 1.07 kg/dm^3 and very low levels of dry solids (varying from 8 to 25%), were characterized mainly by their extremely liquid nature. This is a common property of most sediments, whether they come from rivers, lakes[10,11] or operating ponds and aerated lagoons.[3,8] Its main consequence is to facilitate sludge removal from pond bottom since this operation can be conducted simply by sludge pumping.

Fifty to eighty percent of the dry solids consisted of minerals amongst which coarse particles (over 20 μm) predominated. However, fine silts and clays, the adsorption capacities of which play an important role in sediment-water interactions, were also represented (15 to 40% of the total mineral content). The

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TABLE I Physico-chemical characteristics of sediment (sediment traps).

	Pond 1 Aug-95	*Pond 1 Aug-95*	*Pond 2 Feb-95*	*Pond 2 Feb-95*	*Pond 2 Feb-95*	*Pond 4 Feb-95*	*Pond 5 Jan-95*	*Pond 5 Jan-95*
Exposition period (month)	14	14	8	8	8	8	8	8
Accumulation rate (cm/year)	4.3	4.3	7.5	7.5	3.0	7.0	1.8	2.6
% clay	13.2	14.4	5.1	5.1	13.3	11.4	15.8	14.8
% fine silt	13.5	7.5	27.5	11.0	13.1	35.3	44.5	20.5
% coarse silt	27.5	25.4	4.6	24.7	19.9	31.6	30.2	41.2
% fine sand	20.9	12.8	20.6	15.1	20.9	9.9	5.0	14.6
% coarse sand	24.9	39.9	42.2	44.1	32.8	11.8	4.5	8.9
% $CaCO_3$	5.1	5.3	7.1	6.9	6.5	5.4	4.6	2.5
Percentage of dry solids (%)	17.3	12.5	10.1	9.6	8.9	23.9	20.3	13.8
Heat loss (%)	32.5	46.4	37.7	40.9	38.6	12.1	14.3	28.5
Humic acids (mg/kg D.W)	81.6	92.5	32.4	38.6	65.4	21.0	47.1	75.0
Organic carbon (g/100g D.W)	10.0	18.4	8.1	13.2	9.7	6.4	9.3	17.5
Total nitrogen (g/100g D.W)	4.9	4.4	2.0	1.8	2.1	1.2	1.3	2.1
Total phosphorus (g/100g D.W)	1.0	1.3	1.7	1.9	1.4	0.5	0.4	0.6
Hydrolysable fraction								
N-NO_3 (mgN/kg D.W)	12.1	9.2	<1.0	7.0	1.0	<1.0	1.7	13.6
N-NH_4 (mgN/kg D.W)	534.2	1661.5	952.2	1587.1	745.7	49.3	57.6	29.5
P-PO_4 (mgP/kg D.W)	16.9	48.8	16.6	47.2	56.4	6.6	75.0	104.5
SOD ($gO_2/m^2 \cdot d$)	1.61	0.48	0.87	1.19	0.67	1.38	4.14	1.01

TABLE II Heavy metals content of sediment.

mg/kg D.W	*Pond 1*	*Pond 2*	*Pond 4*	*Pond 5*
Fe	17566	2815	12325	15932
Zn	1154.2	2375.0	1775.0	445.1
Mn	436.8	435.0	612.5	392.4
Pb	151.3	180.0	193.8	33.5
Cu	137.3	250.0	312.5	50.0
Ni	37.4	47.6	52.5	33.6
Cr	28.1	41.3	32.5	29.7
Hg	10.9	7.1	9.4	6.5
Cd	<0.5	3.3	2.5	1.1

sediments were further characterized by their high organic content, exceeding 30 percent of the dry solid content on average. Without reaching the extremely high levels of typical secondary sludges, these values were similar to the ones reported in the literature for several aerated lagoons and waste stabilization ponds.[3] Despite years of residence at pond bottom, sediments seem to be poorly stabilized. Their extreme instability is confirmed by their low humic acid concentrations and C/N ratios, which were consistently below 10.

The nitrogen and phosphorus compounds contained in the sludges from Bertrix wastewater plant averaged 2.5 and 1.1% of the dry solids, respectively. On a mass basis, nitrogen and phosphorus concentrations in the sediments were 200 and 1000 times higher than in the overlying water. This indicates their ability to act as a major sink for pollutants in aquatic environments. However, most nutrient accumulation processes seemed quite irreversible since just a tiny proportion of the sorbed quantities were directly mobilizable. Less than 3% of the nitrogen and 0.2% of the phosphorus contained in the deposits were water-soluble and took part in the exchanges occurring at sediment-water interface without any prior solubilization processes.

Tables II and III summarize the heavy metal contents and bacteriological characteristics of the sediments collected at Bertrix wastewater plant. These sediments were characterized by high levels of Fe, Zn, Mn, Pb and Cu that approached, even exceeded, the values given by the EPA as being typical of heavily polluted sediments.[12,13] Heavy metals appear to be 1000 to 7000 times more concentrated in the sludges than in the overlying water.

These high concentrations of heavy metals result from the same mechanisms involved in the accumulation of nutrients such as nitrogen and phosphorus compounds. Sedimentation, chemical precipitation, adsorption at clay surface and the formation of organic complexes are considered the dominant processes responsible for the immobilization of contaminants in sediments.[14] Obviously, some of those mechanisms are still at least partly reversible, since sediments

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TABLE III Bacteriological characteristics of sediment and overlying water.

LogMPN/l	*Pond 1*		*Pond 2*		*Pond 4*		*Pond 5*	
	sediment	overlying water	sediment	overlying water	sediment	overlying water	sediment	overlying water
Aerobic bacteria	8.5	8.6	8.4	7.7	8.3	6.7	8.3	7.4
Total coliforms	8.0	8.0	7.8	6.6	7.3	6.5	8.1	6.6
Faecal coliforms	6.1	7.5	6.0	6.0	4.9	5.4	5.2	4.9
Faecal streptococci	5.5	5.5	5.0	4.3	4.9	4.0	5.0	3.4
Sulfate-reducing clostridia	3.3	2.5	4.0	1.9	4.2	1.6	3.9	1.5

always have the ability to release micropollutants if pH and redox conditions change.[15]

Nevertheless, heavy metal concentrations in pond sediment stay far below the concentrations in other by-products used in agriculture as organic fertilizers. The French standards, setting the maximum allowed levels for land application, are systematically respected (Table IV). Logically, trace elements present in sludges should not limit their use in agriculture. However, one will notice that copper, lead and zinc exceed the concentrations measured in sludges from other aerated lagoons and waste stabilization ponds.[3]

The bacteriological characteristics of the sediments at Bertrix wastewater plant were not only similar to the rare values reported in the literature[3,8,16] but also extremely close to those of the overlying water. Only sulfate-reducing clostridia appear to be 30 to 40 times more abundant in sediment, where they find ideal anaerobic conditions to growth. The constant densities of faecal indicator organisms from the inlet to the outlet of the plant indicate the relatively low inactivation of these bacteria in sediment over time.

Compared with the other by-products used in agriculture, pond sediments do not present any particular health risk. They even contain fewer faecal coliforms than manure and others biological sludges frequently spread on agricultural land.

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Vertical Concentration Profiles

The mean vertical concentration profiles measured in ten sediment cores are presented on Figure 2. They give a rough idea of sludges composition and illustrate the vertical stratification of their liquid and solid phases. Actually, the calculated coefficients of inter-sample variation frequently exceeded 50%. This reflects the extreme variability of sludges physico-chemical characteristics. However, the shape of their mean profiles remains significant since all the samples we studied have the same overall pattern.

The steady reduction in sediment water content with increasing depth reflects the sludges gradually packing, which is accompagnied by an increase in density and decrease in porosity. As a result, sediment-water exchanges occurring in the deeper layers of deposits are certainly slowed down since lower porosity leads automatically to lower diffusive fluxes.

One will see in this simple phenomenon, already noticed by several authors,[16,17] an initial, at least physical, explanation of why only the top sediment layers are likely to be active: not only are the surface layers in direct contact with the overlying water but they also develop a larger exchange surface.

The decreasing organic carbon and total nitrogen profiles measured in the solid phase of sediment and rapidly or gradually rising dissolved COD, phos-

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TABLE IV Sanitary properties of the main by-products used in agriculture.[3]

	Faecal coliforms (logMPN/l)	*Cu (mg/kgD.W)*	*Zn (mg/kgD.W)*	*Cr (mg/kgD.W)*	*Pb (mg/kgD.W)*
Manure	7	280–2150	1430–4220	125–137	275–287
Municipal secondary sludges	7–8	800	3000	250.00	700.00
pond sediments	5–6	35–120	100–530	17–46	25–61
Standard (France)	–	1000	3000	1000.00	800.00

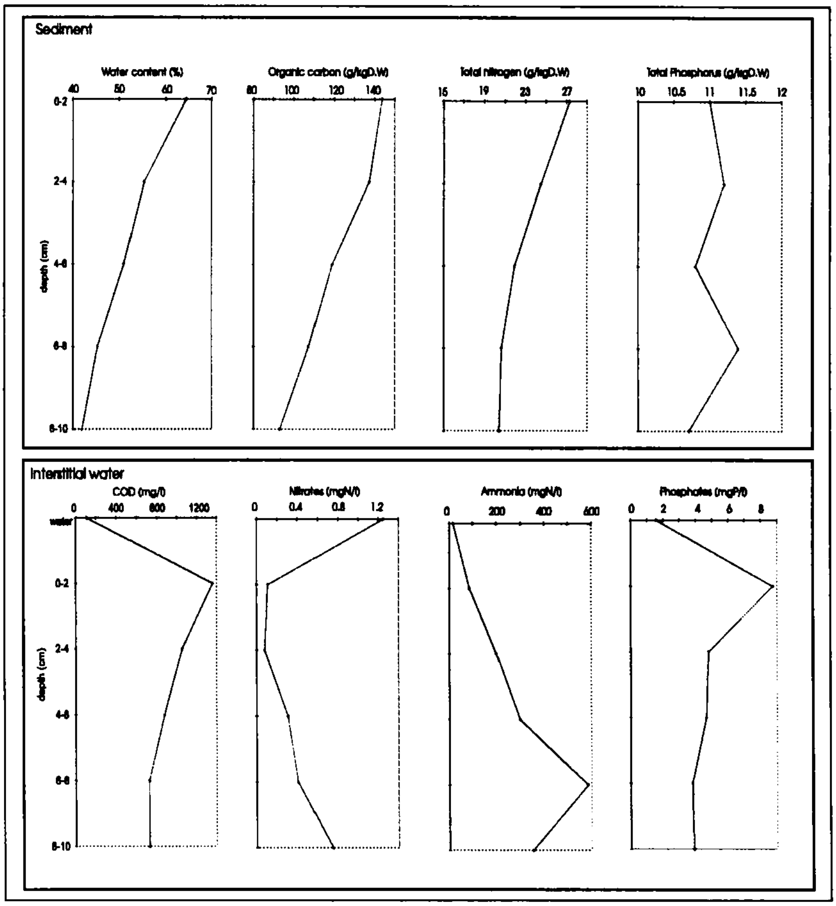

FIGURE 2 Vertical concentration profiles in sediment.

phates and ammonia concentrations in interstitial water reflect the mineralization and solubilization of sediment organic matters, especially in superficial layers where these two phenomenons are particularly intense. Unlike nitrogen, the higher dissolved COD and phosphorus concentrations observed in the upper levels show that their release mainly occurs in the young sludge; while their stabilization, even decrease, in deep sediment corresponds to slower biodegradation kinetics, removal into the overlying water, or «readsorption» by the sediment solid phase.

Several mechanisms, such as precipitation of phosphates with calcium carbonates or phosphorus adsorption on iron and aluminum hydroxides, that regulate the dissolved phosphate pool in interstitial water, support this last

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hypothesis.[16–19] The relative constancy of phosphorus vertical profiles in the sediment solid phase also pleads in favor of such a regeneration of the initial P pool.

The nitrate profiles reflect a completely different logic, since their concentrations in the sediment were systematically lower than in the overlying water. Denitrification is certainly the key-factor explaining why the first two layers (0–4 cm) of deposits contain less than 0.1 mg $N\text{-}NO_3^-/l$. Redox conditions are indeed so rapidly reducing that nitric nitrogen serves as the electron acceptor instead of oxygen. The increase in the nitrate concentrations below 4 cm is much more difficult to explain. Whether this is just an artifact or a result of the progressive reduction in total bacterial activity, *i.e.*, aerobic *and* anaerobic, beyond a certain depth, is not known.

Nevertheless, examination of these vertical concentration profiles shows that, if sediment-water exchanges were exclusively governed by molecular diffusion, deposits would always act as a source of pollutants, releasing important quantities of COD, ammonia and dissolved orthophosphates. In fact, the real processes are infinitely more complex. Other biological and physical mechanisms, such as bioturbation, advection or sludge scouring, affect the rates of exchange between sediment and overlying water.

Dissolved oxygen penetration into the sediments is restricted to a very thin surface layer of less than the first two millimeters of deposits (Figure 3). Except for this aerobic biofilm, all the materials contained in sediment are exposed to markedly more reducing conditions. This, of course, has important consequences for biodegradation kinetics and redox-controlled remobilization of various inorganic elements such as phosphorus and heavy metals. Furthermore, the dissolved oxygen profiles seem to vary with the seasons, oxygen being all the more rapidly consumed when temperature rises. While the oxygen gradients stretched over more than 1500 μm in January 96, they shrank to less than 200 μm, three months later, in March 96, after a temperature rise of only 3 to 4°C. The same phenomenon had previously been observed in a recent study, which also revealed the role played by sludge particle size on dissolved oxygen penetration.[20]

Sediment ATP profiles, shown in Figure 4, emphasize the importance of surface activity. The ATP concentration, which has frequently been used to characterize microbial populations, was systematically highest in the two first centimeters of sediment. Below this surface layer, the ATP concentration decrease with increasing depth to stabilize around 100 picomols per gram of fresh weight. Unfortunately, the only basis for comparison we could find in the literature was the observation of an increase in the total ATP content of river sediment just below a domestic sewage discharge pipe.[21]

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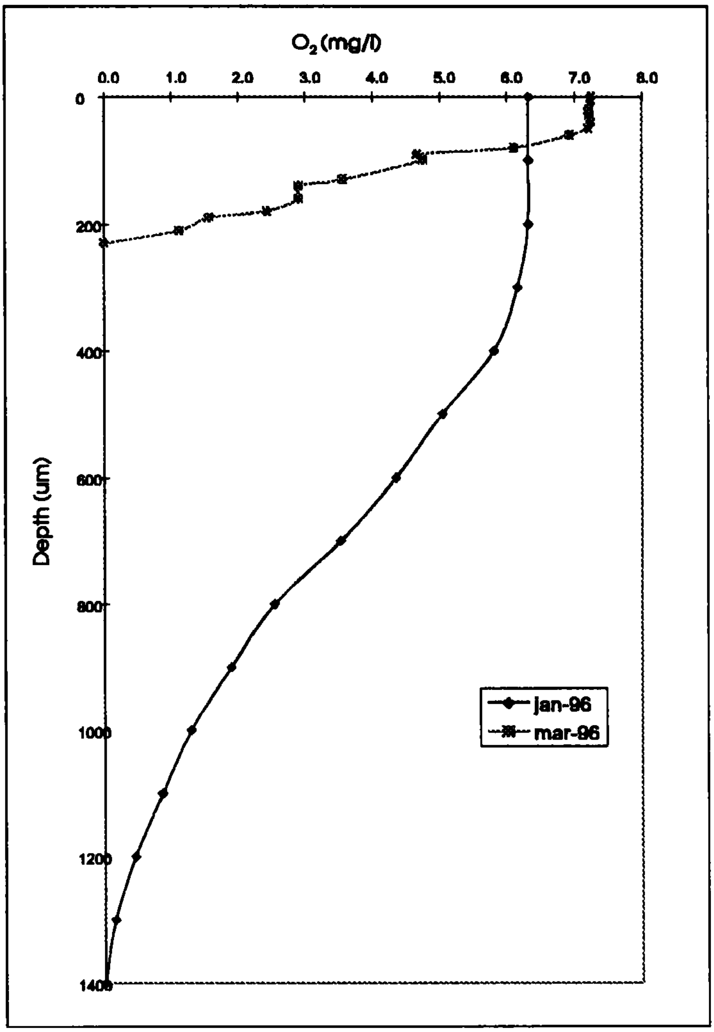

FIGURE 3 Oxygen profiles at sediment-water interface.

Sediment Oxygen Demand

The most direct and important influence sediments exert on overlying waters is their oxygen uptake rate which may account for more than 50% of the total oxygen consumption in river and lakes.[20,22] Sediment oxygen demand (SOD) may also affect the purification performances of waste stabilization ponds and aerated lagoons and must imperatively be taken into account when designing their oxygen supply, at least to avoid the development of anaerobic conditions.

SOD is usually divided into two components:

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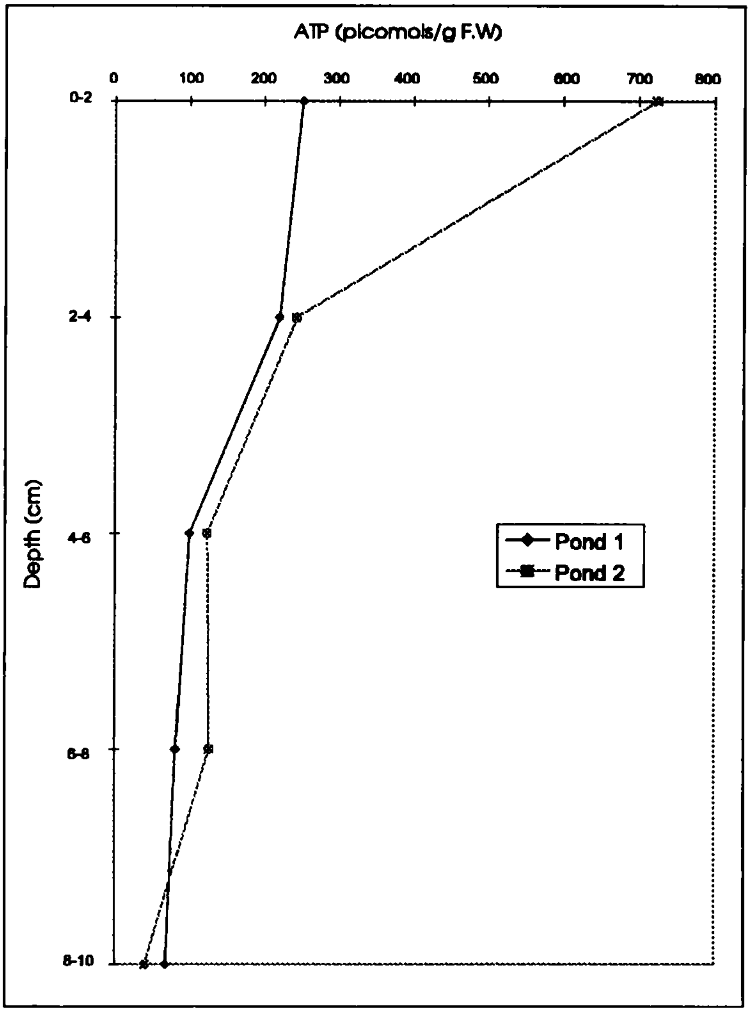

FIGURE 4 ATP profiles in sediment.

- a biological term, which encompasses respiration of all living micro- and macro-organisms dwelling in the sediment; regardless of their activity (whether the result of internal or external biodegradation); and
- a chemical term, corresponding to the vertical migration and oxidation in the overlying water of reduced, oxygen-demanding substances such as bivalent iron, manganese or sulfide.

Our objective was to quantify this benthic oxygen consumption and to identify some of the mechanisms involved in this phenomenon. Since many different

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TABLE V Sediment oxygen demand (sediment cores).

SOD ($gO_2/m^2 \cdot d$)	*Pond 1*	*Pond 2*	*Pond 4*	*Pond 5*
mean	2.53	1.46	2.43	1.97
standard deviation	1.22	0.87	1.43	1.57
number of values	74	51	44	46

analytical procedures have been used to estimate SOD, we also tried to compare them to determine whether they yielded consistent results.

The most widely used method for measuring sediment oxygen uptake consists in transferring undisturbed sediment cores to the laboratory and measuring the oxygen depletion in their overlying water. More than two hundred such respirometric tests were conducted at Bertrix wastewater plant. Their principal results are summarized in Table V.

Most values were between 1 and 3 $gO_2/m^2 \cdot d$, with an average, calculated over the entire plant, of 2.14 $gO_2/m^2 \cdot d$. Surprisingly, very few differences were observed between basins or even between seasons. However, we must keep in mind that all these measurements were conducted at the same temperature (20°C). On the other hand, the few authors who noticed seasonal changes in benthic oxygen demand were never able to conclude if these changes reflect seasonal variations in temperature, nutrients supply (resulting from algal decay), or populations of micro- and macro-organisms living in the sludges.[23,24]

The role played by benthic macrofauna, especially tubificids and chironomids, seems particularly determining, since they maintain better mixing and oxygen availability in upper layers. The irrigational effects of these macro-organisms, better known as «bioturbation», have frequently been observed in aquatic ecosystems such as lakes and rivers.[23,25,26] The same phenomenon certainly occurs in sediments of waste stabilization ponds and aerated lagoons since they contain comparable macro-invertebrate densities of from 5000 to 20,000 larvae/m^2.

We must emphasize that this first series of respirometric tests was conducted without substrate addition, in a situation close to endogenous respiration. While sediments often contain sufficient internal organic reserves to support an important oxygen demand,[20] their oxygen consumption nevertheless appears to be extremely dependent on the BOD of the overlying water.

This could explain why several authors measured higher respiration rates *in situ* than in the laboratory.[24] Respirometric tests, conducted on pond bottom by means of a benthic chamber, enabled us to do the same observation. According to this second analytical procedure, the average SOD calculated for the entire plant would be 3.25 $gO_2/m^2 \cdot d$, a value fifty percent higher than that yielded by sediment cores. During these experiments, the BOD in the overlying water was about 35 mg/l. However, we must admit that only twelve such field measure-

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TABLE VI Sediment oxygen demand after sustrate addition (BOD = 100 mg/l. sediment cores).

SOD ($gO_2/m^2 \cdot d$)	*Pond 1*	*Pond 2*	*Pond 4*	*Pond 5*
mean	4.80	5.81	4.05	9.04
standard deviation	1.50	2.53	1.62	2.39
Multiplication factor	2.4	4.6	2.3	9.7
number of values	13	17	10	18

ments were conducted at Bertrix wastewater plant, for they rapidly proved tedious and much more restricting.

Despite the differences existing between *in situ* and laboratory measurements, all the data collected at Bertrix wastewater plant are similar to these related in the literature.[20,24,27,28] Unfortunately, most of these results concern lakes or rivers and very little attention has been paid to artificial ecosystems such as aerated lagoons or waste stabilization ponds. Only one study, conducted on waste stabilization ponds treating papermill and domestic sewage, indicates SOD varying from 4 to 11 $gO_2/m^2 \cdot d$.[2]

The influence of the overlying BOD on sediment oxygen uptake rate was confirmed by a third series of respirometric tests conducted on sediment cores, successively with and without substrate addition. Added substrate consisted of a mixture of glucose and glutamic acid, with an equivalent BOD of about 100 mg/l.

With an average SOD of 6.28 $gO_2/m^2 \cdot d$, substrate addition induced a fivefold increase in benthic respirations (Table VI). Again, very few differences were observed between basins, apart from the high values obtained in pond 5. The principal interest of these results, which seem to have no equivalent in the literature, is to demonstrate the importance of the biological component of SOD. Moreover, the fact that these sediments are able to oxidize and thus to consume organic matter in the overlying water attests to their direct participation in pond purification performances.

While the applied BOD per unit of surface area has a major part to play, sediment oxygen consumption appears to be independent of its internal organic content. The nature and biodegradability of these organic substances rather than their quantity would have a decisive effect on the extent of benthic oxygen uptakes.[23]

Several authors[11,20,29] prefer measurements on suspended sediments to all these respirometric tests conducted both *in situ* and on non-disruptive samples in the laboratory. Given the absence of comparison of these two completely different analytical approaches in the literature, we decided to run a comparative series of analyses on about forty samples. The results of this comparison are reported in Table VII.

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TABLE VII Oxygen consumption of suspended sediments (without $HgCl_2$).

SOD ($mgO_2/gFW \cdot d$)	*Pond 1*	*Pond 4*	*Pond 5*
mean	3.92	6.56	5.55
Multiplication factor	1083	863	416
number of values	23	12	8

Obviously, the respiration rates calculated for suspended sediments are several hundred times higher than those determined for undisturbed samples. On average, suspended sediments consume as much as 4.96 mgO_2 per day and per gram of fresh weight, that is to say, nine hundred times more than undisturbed sediments.

Such large differences are definitely due to better oxygen availability, higher surfaces of exchange and the sudden release of reduced inorganic compounds that normally diffuse much more slowly towards sediment-water interface. To quantify the proportion of this chemical SOD, which is governed exclusively by oxydo-reduction processes, respirometric tests were conducted on suspended sediments treated with $HgCl_2$. Mercury chloride acts as a metabolic inhibitor, which stops any kind of biological activity. According to the results presented in Table VIII, chemical SOD would account for over one-third of overall respiration. This proportion is similar to those reported in the literature.[22,26]

In many ways, respiration rates calculated on suspended sediments appear to correspond to a potential oxygen demand. Usually restricted to particular situations such as ponds desludging, they also coincide with periods of huge oxygen consumption. In fact, such «resuspension» or «scouring» events are much more frequently encountered in aerated lagoons, since each time their aerators are switched on, about 3 tons of fresh sediment are resuspended. The direct result is a surge in overall respirations measured over the two aerated lagoons.

Sediment-water Exchanges

The exchange of dissolved substances across the sediment-water interface is an important process affecting the purification performances of waste stabilization ponds and aerated lagoons. Accumulation and remobilization rates in or from sediments depend on a host of extremely complex physical, chemical and biological processes. Settling of suspended particles, precipitation/dissolution, ad-

TABLE VIII Oxygen consumption of suspended sediments (with $HgCl_2$).

SOD ($mgO_2/gFW \cdot d$)	*Pond 1*	*Pond 4*	*Pond 5*
mean	1.08	1.97	1.31
% of total respiration	33	36	35
number of values	11	10	1

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TABLE IX Diffusive fluxes calculated at sediment-water interface.

	ϕ (%)	D_0 (m^2/h)	dC/dZ ($mg/m^3 \cdot m$)	Fd ($g/m^2 \cdot d$)
Ammonia	90	$4.87 * 10^{-6}$	$3.56 * 10^6$	-0.303
Orthophosphates	90	$2.52 * 10^{-6}$	$3.59 * 10^6$	-0.016

sorption/desorption, ligand exchange and enzymatic hydrolysis are certainly amongst the main factors that determine dynamic equilibria between sediment and overlying water.

Dissolved pollutants return to the water column via several transport mechanisms (induced turbulence, gas convection, bioturbation,...) but molecular diffusion has always been considered the most important one. This physical process, which is based on the existence of a concentration gradient at the sediment-water interface, can be described by means of Fick's first law:

$$Fd = -\phi \cdot Ds \cdot dC/dZ$$

With
- Fd — diffusive flux ($g/m^2 \cdot j$)
- Ds — diffusion coefficient in sediment (m^2/h)
- dC/dZ — concentration gradient
- ϕ — sediment porosity (%)

(1)

Ds can be estimated from the diffusion coefficient in an infinitely diluted solution D_0, corrected for sediment characteristics:[30]

$$Ds = \frac{D_0}{\phi \cdot f}$$

With
- D_0 — diffusion coefficient in an infinitely diluted solution (m^2/h)
- f — sediment formation factor ($1/\phi^3$)

(2)

Several diffusion coefficients for $N\text{-}NH_4^+$ and $P\text{-}PO_4^{3-}$ are found in the literature.[17,22] Concentration gradients at sediment-water interface dC/dZ were estimated from the vertical profiles measured in interstitial water. Assuming that only superficial layers were involved in these exchanges, we restricted our calculations to the two first centimeters of sediment. Results are given in Table IX. Of course, those calculations suppose the existence of an equilibrium between deposit and interstitial water, a steady state which is probably achieved in deposits after several years.

Negative diffusive fluxes show that nutrients should be released from sediment to overlying water, *i.e.*, from higher to lower concentrations. Surprisingly, those

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TABLE X Daily exchange rates at sediment-water interface.

flux ($g/m^2 \cdot d$)	*COD*	*N-NO$_3$*	*N-NH$_4$*	*P-PO$_4$*
with oxygen				
with substrate addition	29.9	0.58	−0.59	0.18
without substrate addition	41.1	0.69	−0.64	0.51
without oxygen				
without substrate addition	2.7	0.74	−0.48	0.02

Nb: -average fluxes estimated in laboratory over a three days period
-positive values correspond to an adsorption from overlying water to sediment
-negative values correspond to a release from sediment to overlying water

theoretical fluxes, which are based on a well-known, established logic, were systematically lower than the daily exchange rates measured on more than 12 sediment cores in the laboratory (Table X). Even greater discrepancies were observed for orthophosphates since sediments continue to adsorb large quantities of $P\text{-}PO_4^{3-}$ despite the important pool of P they already contain.

The contribution of molecular diffusion to sediment-water exchanges must therefore be relativized while other transport processes, mainly of biological nature, must be considered. The fact that adding substrate increases not only the respiration rates but also all the exchange rates observed at the sediment-water interface proves the importance of bacterial activity.

Moreover, sediment-water interactions seem to be extremely dependent on redox potential, since the exchange rates of COD and phosphates, measured under anaerobic conditions, were depleted by as much as 90%. The high adsorption rates of nitrates observed under the same reducing conditions certainly result in an intense denitrification.

However, we must be extremely cautious in interpreting these results, first, because of their large coefficients of variation and, second, because they can only reflect an overall situation whose internal mechanisms are still largely unknown.

Nevertheless, those fluxes demonstrate how sediments influence pond purification performances. Their contribution to water quality seems even more important than in lakes and rivers since all the exchange rates we measured at Bertrix wastewater plant were higher than these reported for natural ecosystems. For example, the maximum release of ammonia measured in a relatively polluted river at sediment-water interface was about 250 $mgN/m^2 \cdot d$.[27] Similar results were obtained for orthophosphates in the river Seine[17] while maximum uptake rates of nitrates, ranging from 50 to 170 $mgN/m^2 \cdot d$, were observed on Hamilton Harbour sediments.[31] No corresponding data about ponds and aerated lagoons could be found in the literature. As far as we know, virtually no study has ever

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focused on sediment-water exchanges in such heavily polluted ecosystems.

When applied to the 2.5 ha surface area covered by the six basins of Bertrix wastewater plant, exchange rates calculated in Table X show that sediments would be able to remove as much as 90% of the organic load and 80% of the phosphorus load daily received by this facility. Of course, this is only an extremely rough calculation and certainly an overestimation of what is really happening at pond bottom, as our estimated fluxes were obtained under controlled conditions (sediment cores—20°C—synthetic wastewater), relatively far from field situation.

CONCLUSION

The main conclusion of our study is the recognition of the major role played by sediments in aerated lagoons and waste stabilization ponds. Far from being inert, they participate actively in their purification performances. All attempts to model these two wastewater treatment processes require an appropriate description of the sediment compartment, whether in terms of accumulation rates, oxygen consumption or exchange rates with overlying water.

The ambitious program of analyses we conducted at Bertrix wastewater plant enabled us to understand better, even to quantify, some of the aspects and mechanisms that contribute to the importance of this compartment, as follows:

- With a mean accumulation rate of 4.7 cm per year, deposits have a physical effect on ponds performances, since they reduce their volume and shorten their hydraulic residence time.
- As far as their physico-chemical properties are concerned, sediments are mainly characterized by their low percentage of dry solids and high organic content. These highly unstable organic materials explain the sediment intense internal activity and high rates of endogenous respiration. Sediments also have the ability to concentrate important quantities of pollutants and mineral compounds, such as heavy metals and nutrients, in the form of ammonia and dissolved orthophosphates.
- Pond deposits support an important bacterial population of more than 10^8 MPN/l but appear to be heavily contaminated with faecal indicator organisms. However, their microbiological content should not limit land disposal, since it stays far below the values observed in other organic by-products frequently used in agriculture.
- Sediment oxygen demand, which is particularly important when designing the oxygen supply of aerated lagoons, varied from 1 to 3 $gO_2/m^2 \cdot d$. Such

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oxygen consumption is the direct consequence of extremely high surface activity in the first millimeters of the deposit. Evidence of the thinness of this «active layer» was obtained from ATP and dissolved oxygen vertical profiles. However, nearly one-third of the benthic respiration appeared to be linked to purely chemical oxydo-reduction processes.

The fact that these respiration rates increase with applied BOD load per m^2 demonstrates not only their biological component but also that sediments participate actively in pond removal yields, at least in term of organic mass loading. Scouring the sludges surface while aerators are working or achieving the same effect by gas bubbling would enhance benthic respirations several hundred times, as sediment oxygen demand becomes the largest entry in the oxygen consumption budget of the entire basin.

- Finally, the different fluxes of COD, nitrates, ammonia and phosphates, estimated at the sediment-water interface, revealed the extreme complexity of the mechanisms involved in these exchanges. Molecular diffusion is not the only governing process. Biological mechanisms, such as the aerobic-anaerobic bacterial activity or bioturbation caused by the benthic macrofauna, also contribute significantly to the phenomenon.

Sediment-water interactions certainly involve the greatest number of unknowns. Release and uptake rates at sediment-water interface in aerated lagoons and waste stabilization ponds had never even been quantified before. Further investigations should focus on the identification of the various physical, chemical and biological factors affecting their exchanges with overlying water.

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NOTATION

C	concentration (mg/l)
D0	diffusion coefficient in an infinitely diluted solution (m^2/h)
Ds	diffusion coefficient in sediment (m^2/h)
f	sediment formation factor ($1/\phi^3$)
Fd	diffusive flux ($g/m^2 \cdot d$)
SOD	sediment oxygen demand
Z	depth (m)
ϕ	sediment porosity (%)

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