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Kinetic and Empirical Design Criteria for Constructed Wetlands

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Abstract - Design criteria were derived for estimated the size of constructed wetlands needed to treat oil sands mining wastewater using a field-scale wetland experimental facility in northeastern Alberta, Canada. The objective of this research was to demonstrate the capabilities of such wetlands as long-term, self-sustaining "natural" systems for the treatment of large quantities of wastewater anticipated to be released from tailings ponds after mine abandonment. Using empirical data (i.e., hydrocarbon loading rates versus effluent quality), the optimal range of hydrocarbon loading was 5 to *25* gTEH/m^2/month. Using more conservative kinetic data (i.e., microbial mineralization rates), the range of optimal treatment effectiveness was 9.6 to 13.2 gTEH/m^2/month. Since these two design criteria methodologies were calculated using two independent analytical methods, the similarity in results represents a substantial validation of their accuracy. Further research is being undertaken to confirm these findings, to improve treatment performance, and to assess the ecological characteristics of these wetlands.

Key Words - constructed wetland, design criteria, hydrocarbons, mineralization

INTRODUCTION

Suncor Inc., Oil Sands Group, is an oil sands mining and extraction company in northeastern Alberta, Canada. It produces approximately 85 x 10^6 m^3 of tailings slurry per year from which about 13 x 10^6 m^3 of water will eventually be drained into the receiving environment. A principal concept for the reclamation and/or disposal of this tailings water is the use of treatment wetlands as reviewed by Hamilton et al. (1993). It is anticipated that these wetlands would treat any dyke drainage or seepage water from tailings held in untreated or treated (i.e., reclaimed) tailings ponds and/or leachate from other dry land disposal areas. A major consideration in the design of such wetlands would also be their capability to act as "self-sustaining" treatment systems; that is, a system which would not require any manipulations (i.e, artificial structures such as weirs) or long-term maintenance.

In addition to their function as treatment systems and since the potential volume of treatable water would be large, these wetlands would also create additional "natural" aquatic systems which would be part of the local environment compared with conventional confined engineered treatment systems which would be essentially isolated from the environment. As a consequence, any proposal to develop such treatment wetlands has greater environmental implications, both with respect to likely ecological benefits (e.g., increased habitat for waterfowl) or potential adverse impacts (e.g., the bioaccumulation of hydrocarbons into plants); however, this aspect of our research is not included in this paper.

Released water will contain some amount of organic (e.g., naphthenic acids, hydrocarbons) and inorganic (e.g., ammonia) compounds and therefore require treatment since these contaminants can have an acute or sublethal toxic impact on either aquatic organisms or terrestrial plants and animals in the receiving environment (Nix and Martin, 1992). No acute toxicity to plants has been identified but sublethal effects are possible. Dyke Drainage water has historically been much less toxic than tailings pond water by virtue of the loss of contaminants as it is filtered through tailings ponds dykes.

This project utilized two representative types of wastewater: Dyke Drainage (i.e., seepage water from tailings pond dykes); and, Pond 1A recycle water (i.e., water from the surface of a tailings pond). Dyke Drainage water would likely be similar to seepage water from a reclaimed tailings pond/lake. Pond 1A water contains higher levels of contaminants compared with Dyke Drainage and hence represents a worst-case scenario for any process water or surface runoff water from dry land reclamation operations.

Constructed wetlands were built in 1991 on the Suncor lease and planted with aquatic plants in preparation for the start of a multi-year research program in 1992 (Gulley and Klym, 1992). The experimental design included three replicate trenches for each of two treatment systems (Pond 1A, Dyke Drainage). The three replicate wetlands using surface runoff water were intended as a control for comparison with the two wastewater treatment wetlands. The objective of this aspect of Suncor's research was to develop preliminary design criteria for these wetlands; that is, to assess the optimum contaminant loading rates which would result in an acceptable quality of the effluent water (i.e., treatment effectiveness).

MATERIALS AND METHODS

Nine constructed wetlands (Fig. 1) were tested for treatment effectiveness using three types of inflow water: Pond 1A, Dyke Drainage and Control (i.e., surface runoff). Water input flow rates were 2 L/min in 1992 and 6 L/min in 1993 to provide a range of contaminant loading rates (respective hydraulic retention times were 18 d and 6 d). In parallel with increased flow rates, there was a concurrent and unanticipated increase in TEH (total extractable hydrocarbons) concentrations in Pond 1A inflow water in 1993; therefore, there was a ten fold increase in contaminant loading rates compared with 1992. Changes in both flow rates and TEH concentrations resulted in a range of contaminant (i.e., TEH) loading rates varying between 0 to 100 g/m^2/month.

Weekly surveys consisted of water quality, microtox toxicity, ammonia nitrogen, and nitrite-nitrate nitrogen analysis. Monthly surveys included a suite of chemical parameters (e.g., nutrients, chlorophyll *a,* metals and Hg, TEH, ammonia-ammonium, phenols, chemical oxygen demand (COD), total organic carbon (TOC), cyanide, sulphide) and used the aquatic invertebrate, *Daphnia magna,* as the principal bioassay test organism to determine both lethal toxicity (i.e., mortality) and sublethal toxicity (i.e., number of progeny as a percentage of concurrent laboratory controls). The daphnia test were conducted following the methods of ASTM (1989) except that only five replicate neonates were used. TEH (total extractable hydrocarbons) was analyzed in accordance with U.S. EPA method 3510/8015 (EPA, 1984) using methylene chloride to extract the hydrocarbons and gas/chromatography for analysis using diesel fuel as a standard. Chemical and Toxicological data sets were analyzed for both the inflow and outflow water. Inflow samples were composites of the three replicates for each treatment. Outflow samples were collected from each wetland at the outflow (Station D).

A comprehensive survey was undertaken in August to determine bacterial mineralization rates. The methodology for the measurement of carbon dioride production was a modification of a method used to measure $C0_2$ production rates from soils (Anderson, 1982). In the field, 10 L plastic buckets were inverted over the surface of the water and held in place using wooden stakes. A small styrofoam pad or "boat" was floated inside the enclosed space to support a glass beaker containing 30 mL of IN potassium hydroxide (KOH). After a period of 24 to 48 h, the inverted bucket was removed and the KOH placed in small glass vials with rubber-sealed screw caps, which were then transported to the laboratory for titration. The use of inverted chambers to measure mineralization rates is common in soil studies. The technique is inexpensive, simple to use and had the advantage of integrating flux over time (Rochette et al., 1992).

Carbon dioxide production rates were calculated as follows (Anderson, 1982):

Milligrams of $C0_2$ = (B - V - C) NE

where B = mL of acid needed for titration of the reagent blank
V = mL of acid needed for titration of test samples
C = mL of acid needed for titration of blank control samples (i.e., empty 10 L bucket)
N = normality of acid
E = equivalent weight; expressed in terms of CO_2 (E = 22)

In terms of mass balance calculations, both output (carbon dioxide) and input (diesel fuel TEH) were converted to units of grams of carbon (C). Carbon dioxide output was multiplied by a factor of 0.273 (atomic weight C/atomic weight of CO_2) to convert to milligrams of carbon (C). These data were then multiplied by a factor to convert to mg/m^2 of surface area and divided by the exposure time (hours) to transform the data to $mgC/m^2/h$. On the input side (i.e., loading rates of hydrocarbons), TEH concentrations in the inflow water were multiplied by a factor of 0.85 to obtain loading rates as mgC/h or $mgC/m^2/h$ month since it can be assumed that 85% of total hydrocarbons is carbon (Bartha, 1986).

RESULTS AND DISCUSSION

Kinetic Design Criteria

The ability of indigenous species of bacteria and/or other microbes (e.g., fungi) to degrade hydrocarbons as been well documented (e.g., Catello, 1993). This study focussed on the production of carbon dioxide as an indicator of hydrocarbon-degrading activity since the enumeration of bacteria using plate counts or other techniques is often in error by one or more orders of magnitude (Roszak and Colwell, 1987) and since even observed microbes may be dormant and hence their enumeration may not indicate the extent of activity or degradation (e.g., Pickup, 1991). Our purpose was also to demonstrate that organic contaminants in oil sands wastewater can be mineralized in field-scale wetlands treatment systems; that is, to show that these contaminants can be completely removed during treatment rather than being transferred from the water into various environmental compartments (e.g., sediments, plant or insect tissue).

The conversion of organic contaminants to carbon dioxide by microbial processes (i.e., mineralization) is a standard test procedure to assess the biodegradability of organic contaminants in soils. A soil study by Sharabi and Bartha (1993) supported two major underlying assumptions of soil mineralization test procedures: a) that the conversion percentage to CO_2 is relatively independent of chemical structure; and b) that CO_2 production is relatively proportional to the amount of added test compound.

It is generally considered that the anaerobic degradation of petroleum hydrocarbons is negligible compared with aerobic processes (Leahy and Colwell, 1990) although some form of anaerobic metabolism of aromatics has been demonstrated. Also, it is probable that a certain amount of methane (an end product of anaerobic metabolism) would be oxidized in the water column (to form CO_2 and cellular biomass) before "escaping". In peat wetlands in Canada, methane emissions represent a relatively insignificant proportion of total carbon loss (< 1%) during decomposition (Hogg et al., 1992). Therefore, it is likely that the measurement of CO_2 would account for most of the mineralized carbon. The overall biochemical process of mineralization may therefore be viewed as:

$$\text{organic material} + O_2 \longrightarrow CO_2 + H_2O + \text{new organisms} + \text{heat}$$

In general terms, about 50% of completely metabolized organic compounds (i.e., aerobic systems) are converted into carbon dioxide, while the remaining 50% is utilized and converted into cellular material (Dold, 1989). It is also likely that some CO_2 is converted into algae and/or plant biomass through the process of photosynthesis. On the other hand, it cannot be assumed that background mineralization rates are unaffected by the addition of the test compound; for example, more than half of the CO_2 produced in soils may represent the effect of stimulated mineralization of biomass and other organic matter by the test compound (Sharabi and Bartha, 1993). Since these two processes work in opposite directions, it was assumed that they cancelled each other out and therefore the carbon dioxide produced in this study was multiplied by a factor of 1.0 (Le., unchanged) to calculate the total amount of hydrocarbon (i.e., TEH) depleted. This calculation is likely a conservative estimate of the amount of organic carbon utilized by microbial processes since: 1) mineralization rates may be underestimated due to the formation of carbonates in the water column from the release *CQ&* 2) bacterial processes may not completely mineralize hydrocarbons to carbon dioxide but rather produce other byproducts (Hinchee and Ong, 1992); and, 3) a certain percent of organic material was likely metabolized anaerobically.

In field studies, nine empty and sealed buckets were used as controls. The mean value for the total amount of carbon dioxide in the three buckets exposed over 72 h was 4 mg CO_2 or 1.1 mgC. This value was close to the theoretical value for the amount of carbon dioxide contained in 10 L of air (1.6 mgC), assuming a concentration of 0.03% which is the global average in the atmosphere (Barry and Chorley, 1976). Since the value for CO_2 concentration in the atmosphere can vary substantially, the derived and theoretical values for carbon dioxide levels in the blanks were comparable and therefore the mineralization field protocol was considered valid.

For the Control series of treatment wetlands, the rate of mineralization was comparable among inflow, mid-trench and outflow sites, reflecting the low amount of hydrocarbon loading into the system and the relatively uniform characteristic of the water within each wetlands (Fig. 2). Mineralization rates for the Dyke Drainage treatment wetlands were substantially elevated compared with Control trenches, almost certainly a consequence of the input of TEH/TOC compounds. Unlike the Control trenches, mineralization rates also showed a marked decreasing trend from inflow to outflow, suggesting that biodegradable compounds were being utilized and that biological treatment of contaminants in the water was ongoing. For the Pond 1A treatment wetlands, mineralization rates were comparable with the Dyke Drainage trenches; however, there was no trend of declining values from inflow to outflow, suggesting that the biological degradation of contaminants had not been completed. In addition, standard deviations tended to be higher than was the case for Dyke Drainage trenches, likely reflecting a more heterogeneous mixture of TEH compounds in both the water and sediment. Visual observations during the sampling surveys confirmed the presence of heterogeneous "lumps" of bitumen-like hydrocarbons within both the water column and sediment of the Pond 1A trenches.

Mineralization rates for the constructed wetlands ranged between 9 to 27 mgC/m^2/hour which converts to 79 to 236 gC/m^2/year. Notwithstanding inputs of organic carbon, this range is low compared with values obtained from the literature. For example, Werttzel (1992) has cited values ranging from 804 to 254 gC/m^2/year for total carbon dioxide release rates from a salt marsh and lake sediment respectively. In a study of constructed wetlands, Gale et al. (1993) quoted mineralization rates of between 361 and 733 gC/m^2/year. However, comparison between mineralization rates in the Suncor treatment wetlands and other data is tenuous since physical, chemical and biological characteristics of these wetlands would not likely be similar to these referenced wetlands, since published data are very limited, and since Suncor's constructed wetlands are relatively immature (i.e., only two years old) compared to most natural or constructed wetlands.

To determine the effect of hydrocarbons and/or other organic compounds on mineralization rates in these wetlands, data for carbon dioxide production from the Control trenches were subtracted from the two series of treatment trenches (Dyke Drainage and Pond 1A) at each site. These corrected data suggest that the mineralization rates for TEH/TOC compounds were comparable between the two treatment wetlands (Fig. 3). The range for Dyke Drainage water was 6.3 to 16.2 mgC/m^2/hour (mean = 11.4 mgC/m^2/hour or 9.6 gTEH/m^2/month) and for Pond 1A was 11.8 to 18.7 mgC/m^2/hour (mean = 15.3 mgC/m^2/hour or 13.2 gTEH/m^2/month).

From a regression analysis of the data at Station D, moderate to high coefficients of determination (r^2) were demonstrated for several organic compounds as shown in Table 1. Phenols (i.e., phenolic compounds) had the highest correlation with mineralization rates, indicating that these compounds may be good indicators for the presence of biodegradable contaminants (concentrations of phenolic compounds in themselves were too low to have resulted in a major portion of the produced carbon dioxide). However, both TEH and TOC also have relatively high r^2 values compared with BOD_5. There was a lower r^2 value for TEH (0.74) compared with TOC (0.82), an indication that the TOC component contained a subcomponent of non-TEH organic compounds which were more readily biodegradable and thereby produced a closer relationship for TOC between concentrations and mineralization rates.

The relatively low correlation with mineralization for BOD_5 is likely due to the slow rate of metabolism for most contaminants present in these wastewater streams; that is, requiring longer than 5 days for their biodegradation/mineralization. This conclusion is supported by the data which show that most BOD_S values were less than 5 mg/L notwithstanding much higher TEH and TOC values of 20 to 80 mg/L. This observation also suggests that the HRT (hydraulic retention time) for these treatment wetlands should be longer than 5 days (i.e., time of a BOD test) to assure an effective treatment process.

Finally, these data indicate that toxicity was not a factor in terms of any adverse effect of these compounds on microbial degradation since all correlations are positive; that is, the mineralization rate increased with increasing concentrations.

Empirical Design Criteria

An extensive comparison of hydraulic loading rates for 31 surface and 13 sub-surface flow wetlands treating municipal effluents in North America has been provided by Watson et al. (1987). Advanced treatment, based on the survey, requires 1.9 to 3.1 cm/day, (190 to 310 m^3/m^2/day) for a surface flow system, and 3.1 to 4.7 cm/day (310 to 470 m^3/m^2/day) for a sub-surface system. In the Suncor surface flow wetlands where each trench has an area of 175 m^2, this would translate to a range of flows from 2.3 to 3.8 L/min.

Initial analysis of the data sets did not indicate clear trends between TEH loading rates for the various chemical and lexicological parameters. For example, there was only a weak correlation (r^2 = 0.29) between sublethal toxicity (i.e., percent progeny of *D. magna)* in the outflow and TEH loading at the inflow for all treatments (including the Control wetlands) and for all months (Fig. 4). Therefore, the data were reanalyzed with the following exclusions: 1) all Control wetlands data (since TEH values in surface runoff water likely did not represent the same type of hydrocarbons that originate from Dyke Drainage and/or Pond water); 2) all September data (since temperatures during this month were substantially decreased and hence treatment effectiveness was reduced); 3) all data from trench 5 (since the inflow rate in this wetland were altered in both August and September and treatment processes may not have re-established an equilibrium during this relatively short period).

Following these guidelines, Dyke Drainage and Pond water data that are subsequently presented became more illustrative of relationships in the field in some instances. For example, the unclear relationship between sublethal toxicity and TEH loading rates (Fig. 4) become comprehensible (Fig. 5) with an r^2 value of 0.71 compared with 0.29 for the unselected data. Using these selected data, a threshold loading of between 10 to 25 gTEH/m^2/month was found when using sublethal toxicity (i.e., *D. magna* progeny) as a lexicological criterion. This value is about 300 to 600% higher than the estimated potential removal rate using kinetic analysis of bacterial mineralization data. At all loading rates above 42 gTEH/m^2/month, no progeny were produced. In most cases, statistical treatment of these data have not been undertaken since the number of data points are limited and, more importantly, since the presence of zero values (i.e, no progeny for the *D. magna* test) would have distorted any analysis such as regressions and associated correlation coefficients.

Using TEH concentrations at the outflow as a criterion of treatment effectiveness, a weak relationship (r^2 = 0.61) between TEH loading rates and effluent quality was apparent (Fig. 6). The data suggest that a loading rate between 5 to 15 gTEH/m^2/month would be required to obtain an effluent TEH level of less than 5 mg/L (a permit level used by regulatory agencies).

SUMMARY

Design criteria for the treatment of TEH contaminants in oil sands wastewater are: Empirical, 5 to 25 gTEH/mVmonth; Kinetic, 9.6 to 13.2 gTEH/m^2/month. Kinetic analysis was expected to be more conservative (i.e., lower TEH loading rates) since optimum rates were based on a single criterion; that is, the complete removal of TEH contaminants from the water column. In the case of the empirical analyses, calculations of optimum loading rates were based on two less stringent criteria; that is, the removal of sublethal toxicity (i.e., some toxicants may still be present but at levels below those which would cause any adverse impacts) and/or the partial reduction of TEH concentrations (i.e., such that levels in the effluent would be less than a specified maximum, in this case, 5 to 15 mg/L.

These criteria provide a method to estimate the size of a wetland required to treat oil sands wastewater with a known TEH concentration and at a specific flow rate (i.e., loading rate). However, further research is required since these design criteria have been derived from a limited data set and since the characteristics of treatment processes within these wetlands are not completely understood. Furthermore, since wastewater flow rates may be very large (i.e., millions of cubic meters per year), the required land area will be large to achieve a satisfactory level of treatment (i.e., hundreds of hectares). Therefore, the initial concept of self-sustaining wetlands may need to be revised to include engineered designs for supplementation with nutrients and aeration.

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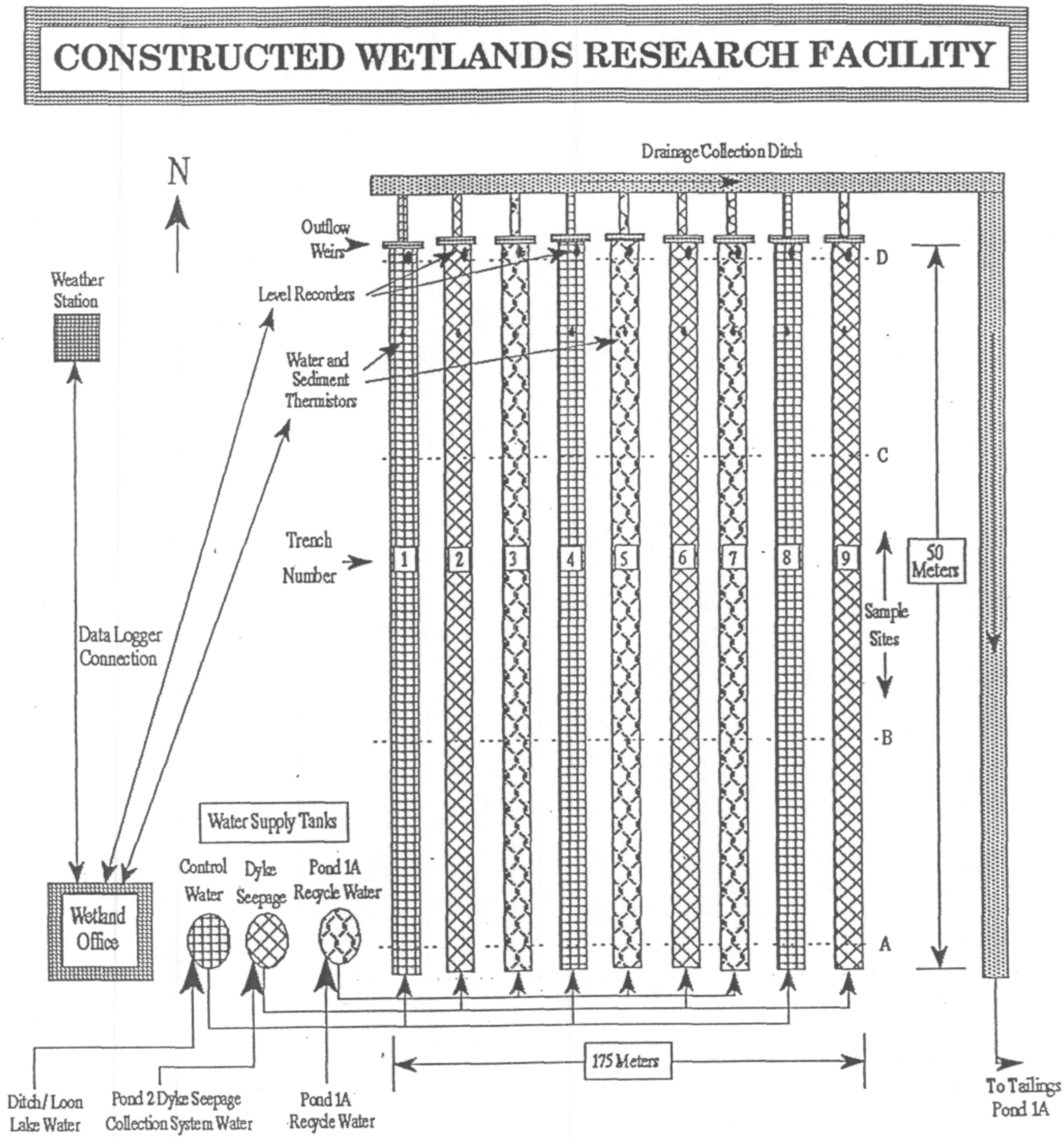

Figure 1. A schematic of Suncor's wetland research facility

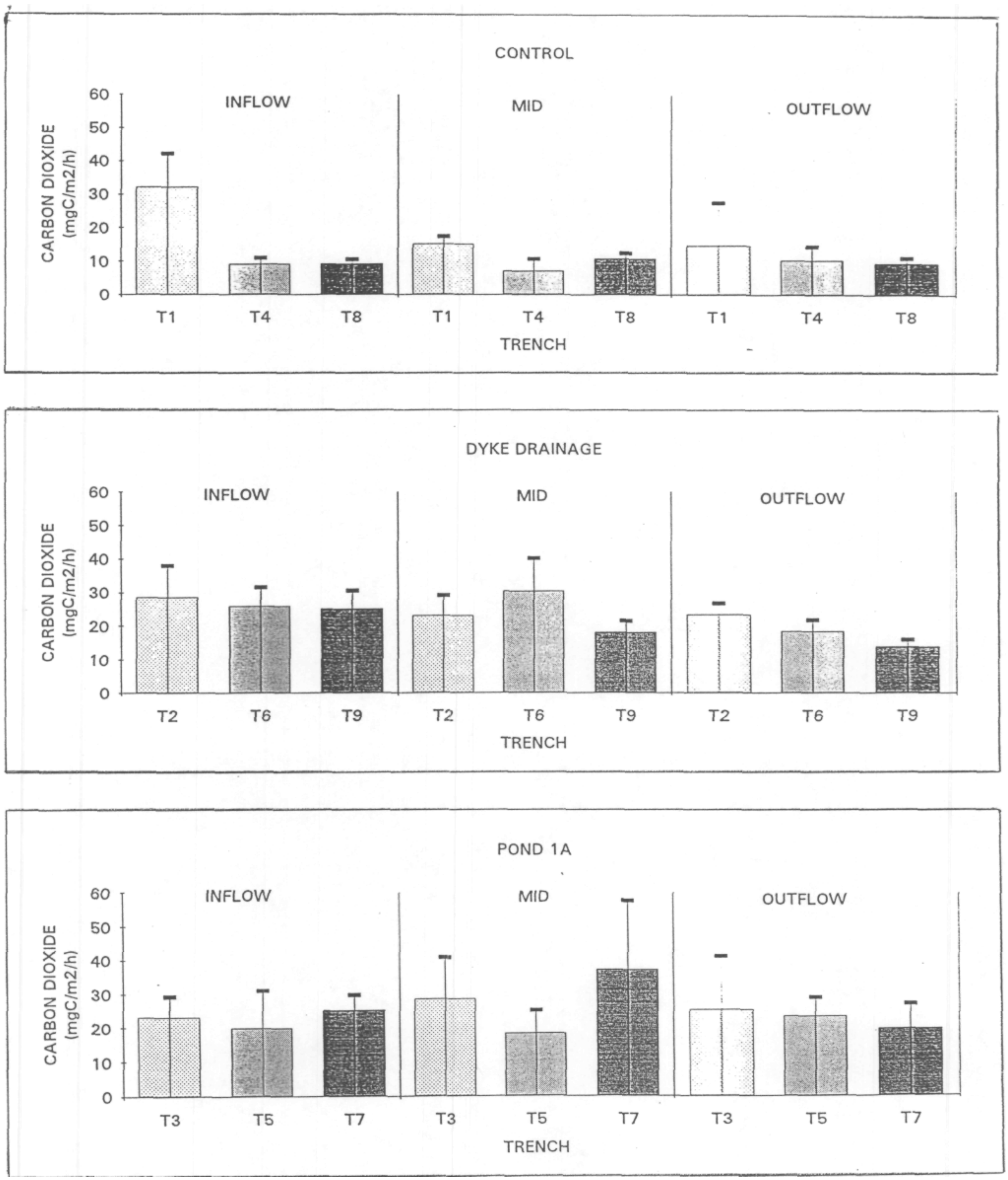

Figure 2. Mineralization rates in the constructed wetlands, August 24-26, 1993 (means ± St. Dev.).

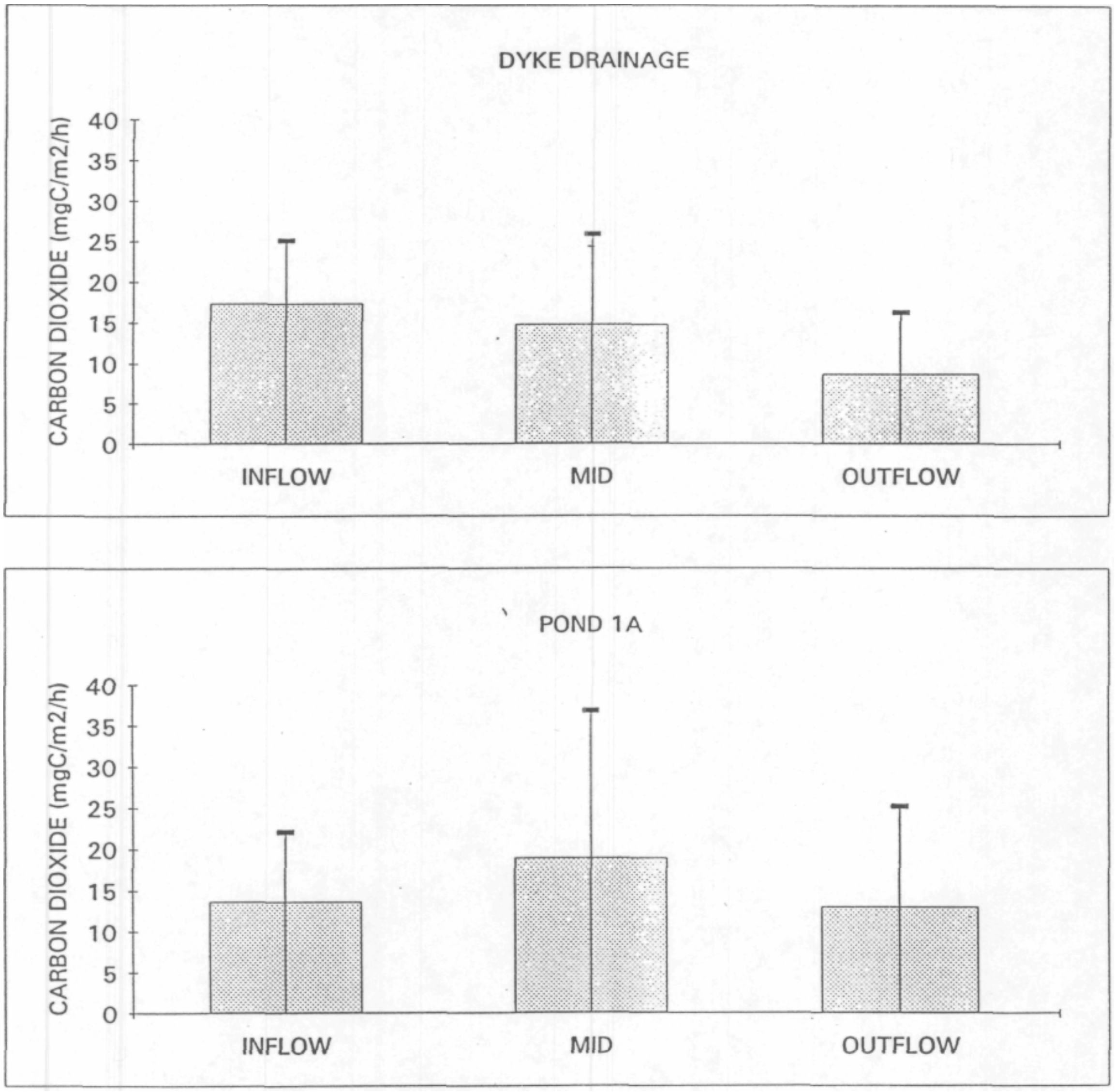

Figure 3. Corrected mineralization rates in the two treatment wetland systems resulting from the input of contaminants, August 24-26, 1993 (means ± St. Dev.).

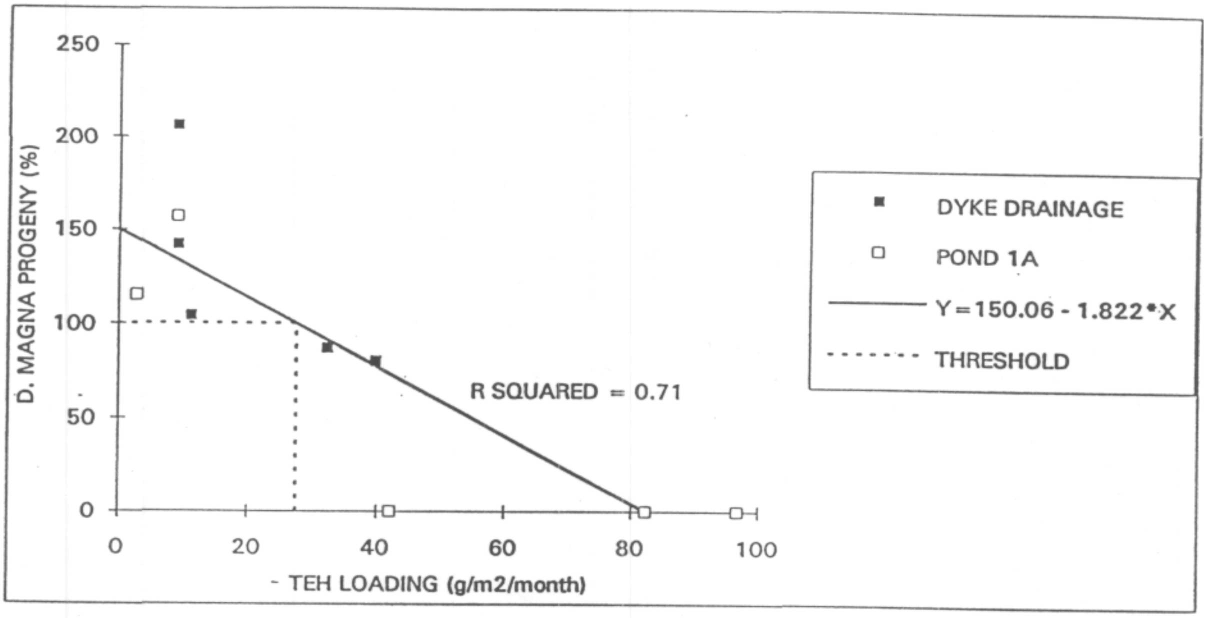

Figure 4. The impact of TEH loading rates on sublethal toxicity in wetland outflows using selected data (i.e., omitting T5, September and Control wetland data).

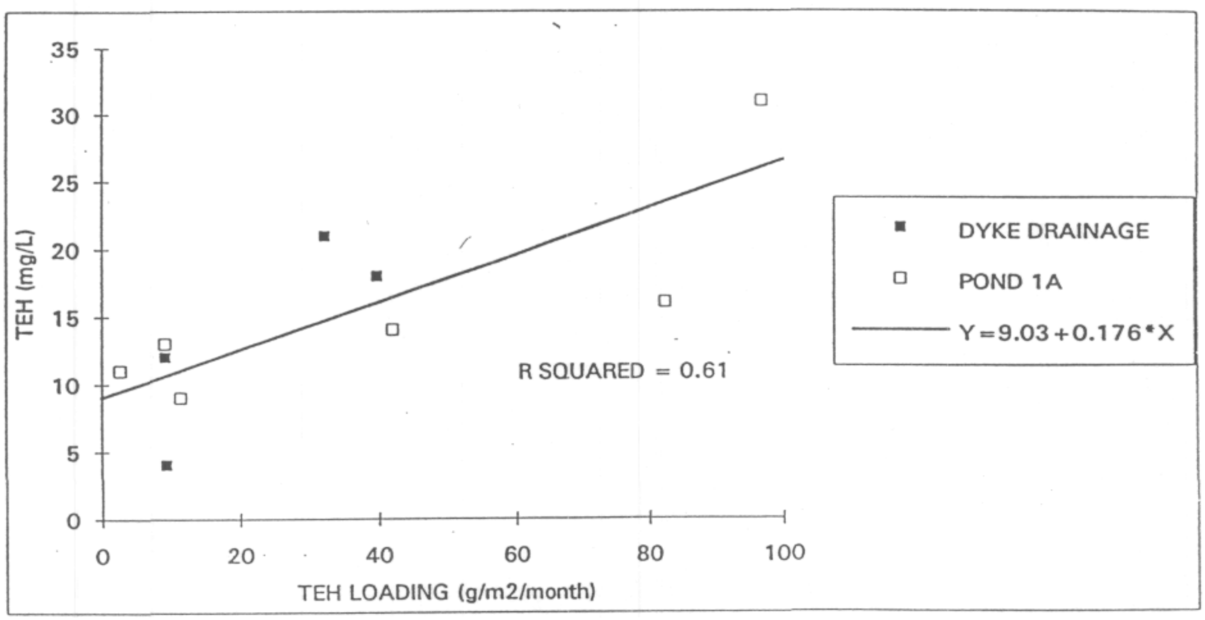

Figure 5. TEH loading rates compared with hydrocarbon levels in wetland outflows using selected data (i.e, omitting T5, September and Control wetland data).