GEOTECHNICAL RESEARCH CENTRE

GRC STUDIES
on
ENVIRONMENTAL GEOTECHNICS

GEOENVIRONMENTAL SERIES No. 91-6

August 1993

McGill University
Montreal, Que Canada

ISSN 1183-9503

FOREWORD

The following selection of papers on **Environmental Geotechnics** has been published in the Proceedings of the First Canadian Conference on Environmental Geotechnics held in Montreal, Quebec - May 1991.

1. Yong, R.N.
   Landfilling Waste Management Compliance with Regulatory Policy and Guidelines. Keynote Address. pp.3-8.

2. Elektorowicz, M., Hadjinicolaou, J. and Yong, R.N.
   Selection des sites alternatifs pour l'elimination des dechets de grandes municipalites. pp.143-148.

3. Yong, R.N., Mohamed, A.M.O. and Elmonayeri, D.S.
   A Multiphase Flow Model for Remediation of Subsurface Contamination by Nonaqueous Phase Liquid. pp.293-298.

4. Yong, R.N., Cabral, A. and Weber, L.M.
   Evaluation of Clay Compatibility to Heavy Metals Transport and Containment; Permeability and Retention. pp.314-321.

Proceedings, First Canadian Conference on Environmental Geotechnics
Eds. R.P. Chapuis and M. Aubertin. Montreal, Quebec.
May 1991

KEYNOTE ADDRESS

LANDFILLING WASTE MANAGEMENT COMPLIANCE WITH REGULATORY POLICY AND GUIDELINES

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ABSTRACT

The management and "safe" disposal of hazardous solid and liquid wastes pose formidable problems and difficult issues which need to be resolved if "*protection of human health and the environment*" are to be obtained. This keynote address focusses on the principal issues that appear to be at the centre of the sets of problems: (a) the need to obtain proper specification of "*safe protection requirements*" relative to the kind of waste being managed, (b) concerns regarding development and utilization of the "*best available*" disposal technology, and (c) perception of governmental policy, regulations, and conformance (and acceptance?) requirements.

INTRODUCTION, AND PROBLEM DEVELOPMENT

The management of hazardous solid and liquid wastes (HW)[1], - i.e. generation, storage, transport, and disposal - offers several challenging issues if the requirements for ***protection of human health and the environment*** are to be met. The need for protection of human health and the environment has been the subject of numerous discussions and debate, with resulting effects articulated in terms of minization of threats to human health and the environment. As will be seen later, in the discussion of various aspects of the problem, articulation of the <u>need</u> is indeed much easier than establishment of a "*concrete" set of rules and standards.* For the purpose of this keynote presentation, the aspects dealing with "safe" disposal in the ground will be addressed.

Three basic sets of issues can be identified: (a) What is meant by "safe" protection requirements? - i.e. what are the rules, standards and criteria governing *SAFE* protection? - relative to the nature of the HW being managed? (b) What is "*best available*" disposal technology, and how is this to be obtained (decided upon) and evaluated? (c) Perception of governmental policy, regulations, and conformance requirements.

It is difficult to mount separate discussions addressing each of these issues without transgressing from one (issue) to the other. For example, in determining the parameters that need attention in the development of "*What is safe protection*", it will be immediately evident that the associate question should be "*Who is responsible for articulating the SAFE requirements and standards?*" If the response identifies the regulatory agencies, it then needs to be asked whether the various sets of criteria used will indeed satisfy the requirements for protection of human health and the environment. The record shows that there is considerable reluctance in providing actual concrete answers, hence one "talks" about *minimization* of threats to human health and the environment. One therefore immediately asks "*What is meant (quantitatively) by minimization?*"

Whereas <u>*disposal*</u> of waste, in the general context used in North America[2] refers to the discharge of waste forms into the atmosphere, receiving waters and land, the subject of concern to the geotechnical engineering community is land disposal of waste, with particular attention to HW. The immediate and particular concerns regarding landfilling technology relate to the availability, or lack thereof, of proper specifications and technical information regarding disposal standards and criteria, - critical in the establishment of the basic land disposal/containment design standards and technological requirements. Whilst direct questions (to control bodies) concerning the details and specifications constituting **safe protection requirements** do not, as yet, have specific or definitive answers, assurances are nevertheless continually being sought by various control bodies or agencies for "safe and secure" HW containment and final disposal. The conundrum that is perceived as "*safe and secure*" landfilling, requires resolution to several very difficult interactive issues - i.e. issues which require interaction between, for example, epidemiologists, toxicologists, environmental scientists and engineers, biologists, legal experts, geotechnical engineers and technologists, etc. Some specific issues or questions that need to be addressed include:

- What is *safe and secure?* Policies, rules, regulations, controls, expectations, measurement, ...
- What are the "controls" or targets for design and disposal strategies?
- Assessment criteria for *damage to human health and the*

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[1] The abbreviation "HW" (hazardous waste) is used to cover *hazardous solid and liquid wastes.*

[2] Directive 78/319, Article 1c, of the European Community allows for transformational operations necessary for recovery, reuse and recycle in its definition of "disposal".

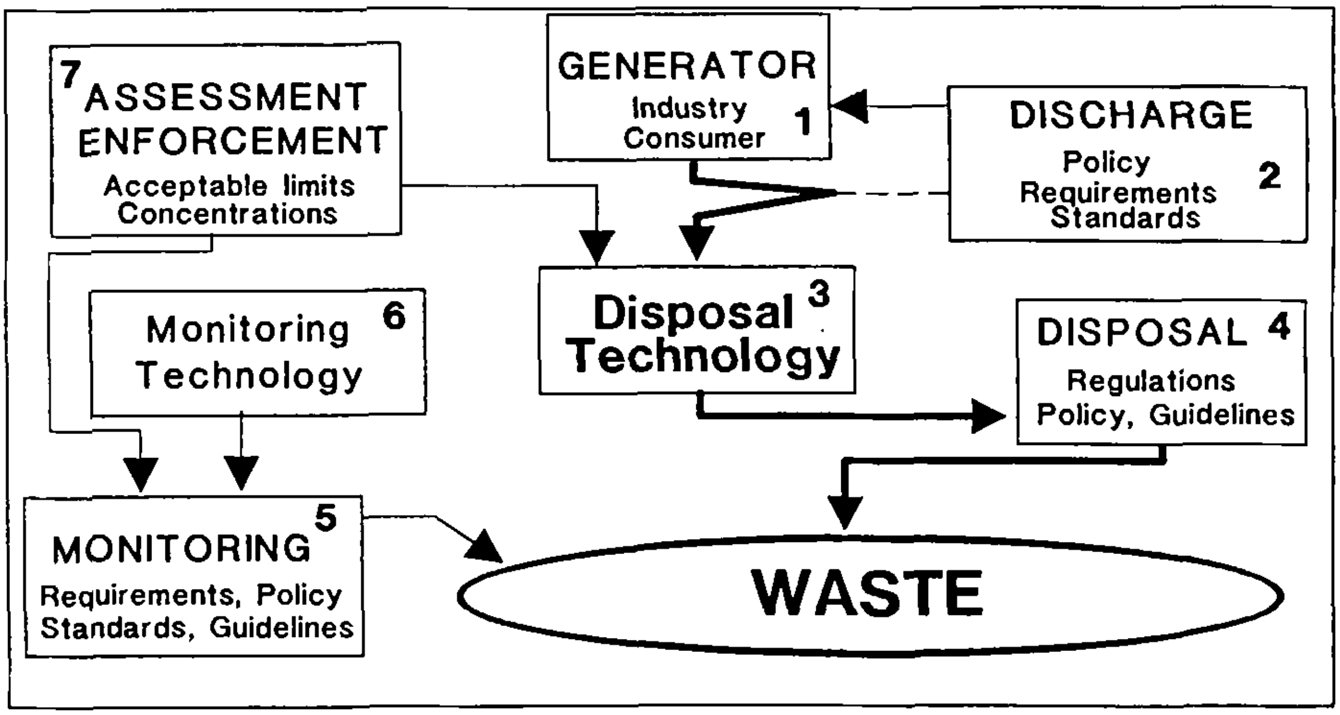

Fig. 1 - INTERACTIVE FACTORS IN WASTE MANAGEMENT

environment in relation to *"safe and secure"?*

- If disposal strategies and procotol are implemented in accord with the present policies and regulations, are we assured that *"safe and secure"* landfilling to meet protection of human health and the environment are met?
- What are the technological requirements to meet *"safe and secure"*.....and protection of human health and the environment?
- Accountability? Who sets the rules? Culpability? Responsibility?
- FAILURE? Measure of failure? Onset of irreversible damage to human health and the environment?

The list of issues and questions can obviously be considerably increased. However, for the geotechnical engineer, the various interacting items that SHOULD pose direct concern to the practice of safe geotechnical disposal technology can be directly deduced from the summary diagram presented in Fig. 1. The various interactions occurring across the different interest-groups is evident from the Figure.

"SAFE" DISPOSAL INTERACTIVE FACTORS

As noted from Fig. 1, landfilling waste disposal strategy depends on several factors not directly within the control of the geotechnical engineer. The implementation of the "constraints" shown in the Figure generally take the form of procurement of construction and operational permits. This presumes that the necessary requirements, conditions and guidelines for *siting* have been satisfied, and that permission to site the disposal facility has been received. "Permits" for implementation of the disposal scheme issued to the geotechnical consultant by the "Permitting Agency (PA)" occurs when (presumably) the PA is assured that the scheme and technology will satisfy the requirements for *safe and secure disposal*. This operational procedure implicitly requires that the many aspects of HW management - from HW health threatening issues to technological solutions and risk assessment - have been fully considered, evaluated, and thoughtfully analyzed. For this to occur, a proper understanding of the details of interactions and consequences concerning waste type and chemistry, handling procedures, permissible limits and threats to human health and the environment, the soil buffering system and transport processes, toxicology and ecotoxicology, etc. need to be obtained.

Conventional wisdom expects that the PA members will have the required expert knowledge or expertise in the various individual disciplines - either personally or via consultative advice. The preceding notwithstanding, experience shows that the transdisciplinary interaction required to produce effective scrutiny of the disposal scheme is rarely achieved. Some of the major factors that impact directly on the interactions between the disciplinary groups in the consideration of disposal strategy include:

1. **Industry:** Composition and toxicity of waste discharged, quantities, storage and handling.
2. **Standards and criteria for waste discharge:** Acceptable (permissible?) concentrations or limits, lethal doses and/or concentrations, exposure indices, etc.
3. **Disposal requirements and technology:** Short-term and long-term performance expectations and requirements, risk, etc.
4. **Monitoring:** Strategy and technology.
5. **Assessment:** Criteria, pathways, fate, exposure.

The above factors are subordinate to one single over-riding set of controls: i.e. governmental legislative policies regulating total HW management - generation, re-use and recycle, treatment, storage, transportation and disposal. The ideal set of circumstances presumes that regulations and guidelines available from the regulatory agencies are properly developed and "in-place". Accordingly, the *"DISCHARGE policy, requirements, standards"* (Box 2) shown in the Figure, not only controls (directly) the quality and nature of the waste discharged by the *"Generator"* (Box 1), but also indirectly "dictates" the disposal strategy.

By and large, *"DISPOSAL regulations, policy and guidelines"* (Box 4), which should deal directly with the *DISCHARGE POLICY* will "indicate" the technology requirements that would "safely" contain the waste - i.e. the *Disposal Technology* (Box 3). We should note that short and longterm performance requirements for the completed (and operational) landfill, generally set in terms of detection of control limits of concentrations of in the liner and substrate at set time intervals and distances, are in actual fact the sets of controls for the design of the landfill. Interpretation of these performance requirements is key to the successful compliance of the landfill capability and performance (to regulatory expectations). Since longterm performance is to be "predicted" as part of the compliance procedure, the importance of *"MONITORING requirements, policy, standards and guidelines"* (Box 5), is self evident.

As noted from Fig. 1, the sets of interactions needed for proper and coherent HW management to meet the requirements for protection of human health and the environment can only occur if regulatory policies can reflect the "real" conditions and constraints needed to provide for "safe" longterm disposal. In short, disposal requirements and conditions, which are directly associated with the nature (and quantities) of the waste. (composition, chemistry, properties, characteristics, etc.), provide the first step in "safe" disposal strategy. The recommended levels or concentrations of pollutants allowed for "safe" discharge are needed, failing which, the requirement for safe disposal containment must ensure that no "unsafe" discharge to the environment occurs. The lack of strict regulations or conformance requirements, improper interpretation or "prediction" performance expectations of the completed landfill will lead to severe consequences.

The preceding notwithstanding, it is apparent that even if all published standards and criteria are met, and if all regulatory requirements have been strictly adhered to, there is no guarantee that waste generation and disposal will still meet the conditions designed to protect human health and the environment. The fundamental need to establish categorically *what constitutes a threat to human health and the environment, and what constitutes "safe" disposal design/technology,* is paramount.

HW MANAGEMENT AND CONTROL

The U.S. *Hazardous and Solid Waste Amendments, 1984 (HSWA 1984[3])* which amended the U.S. *Resource Conservation and Recovery Act, 1976 (RCRA 1976[4])* provides a good example of "policy and general guidelines" which impact directly on the practice of landfilling waste disposal. The "National Policy" articulated in Sec. 1003 of RCRA 1976 states: *"The Congress hereby declares it to be a national policy of the United States that, wherever feasible, the generation of hazardous waste is to be reduced or eliminated as expeditiously as possible. Waste that is nevertheless generated should be treated, stored, or disposed of, so as to minimize the present and future threat to human health and the environment".*

In addition to the preceding, Section 224(a) of HSWA 1984 requires waste generators to certify on the manifest accompanying every waste shipment that:

"(1) *the generator of the hazardous waste has a program in place to reduce the volume or quantity and toxicity of such waste to the degree determined by the generator to be economically practicable; and*

(2) *the proposed method of treatment, storage, or disposal is that practicable method currently available to the generator which minimizes the present and future threat to human health and the environment."*

The RCRA 1976 policy represents a *market policy*[5] approach which has been advocated by many other countries (e.g. EEC) in regard to HW management and control. With this type of market policy, the burden of responsibility for management and control of HW lies with the generator and the geotechnical engineer (consultant), within a set of guidelines articulated by "Permitting" bodies and regulatory agencies. With enforcement of regulations and use of penalties, it is presumed that the market will satisfy compliance requirements. However, the record shows that gross mismanagement and illegal practices[6] abound. Accordingly, it becomes difficult to provide assurances that disposal technology practices required to satisfy "safe disposal[7]" requirements can be met.

PROBLEMS FROM HW CLASSIFICATION

Handling and disposal requirements are generally different between HW containing toxic substances and HW without toxics. The need for regulatory control of HW is especially important in transboundary shipments of HW, where control and management of the manner in which HW is transported from Generator to disposal facility are critical. For example, the Committee on Transport of Dangerous Goods (CTDG) within the Economic and Social Council (ECOSOC) of the UN classifies HW (as Dangerous Goods) into nine hazard groups, requiring labelling for each group, and conditions and criteria for packaging in multi-modal transport[8]. Proper and continuous tracking of HW ensure accountability for safe handling and disposal[9] - provided that the technology available for safe handling and disposal is applied (or exists?). Since there exists considerable economic advantage for the Generator to have a waste declared a non-HW - because of the less-than-stringent handling and disposal requirements governing disposal of non-HW (e.g. little or no tracking requirements, etc.) - the guidelines, criteria or standards used to define or characterize a HW or toxic substance become very important.

The (Canadian) Federal Department of the Environment, formed in June 1971 conferred upon the Minister the duties of *protection and enhancement of the natural environment, including water, air and soil quality.* Since the "Government Organization Act, 1970" limits the Minister's responsibility to those areas *not by law assigned to any other department, branch or agency of the Government of Canada,* the role of the Provinces becomes important vis-a-vis waste regulation and control. This

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[3] Pub.L. 98-616, Nov. 9, 1984.

[4] 42 U.S.C.A. 6901 et seq., as amended by Pub. L. 96-482, Pub. L. 96-510

[5] In some other countries, a *State policy* approach has been adopted. HW disposal facilities are owned and managed by the State, and waste generators are required to transport their waste to the State-owned or State-subsidized facilities.

[6] U.S. Congress. Office of Technological Assessment. (1984). *OTA Review of Experience of the "Superfund" Program*, Washington, D.C.; UK Hazardous Waste Inspectorate. (1985). *Hazardous Waste Management: An Overview*, London: HMSO. First Report. DOE.

[7] We presume that *eventually*, a "set of rules" or "whatever" will be produced that would inform one as to what "safe disposal" requirements are; or better yet, a general Code of Practice will be produced which would ensure that "safe disposal" occurs.

[8] The comparable Canadian requirements in the Transportation of Dangerous Goods Act (TDGA), bear considerable similarity to the UN CTDG recommendations.

[9] As for example in the *cradle to grave* approach set forward in RCRA 1976.

situation provides the scenario that now exists, i.e. varying criteria, standards, conformance requirements, and even definitions or classification schemes for HW between the different Provinces and the Federal government. The evolvement of CCREM, the Canadian Council of Resources and Environment Ministers, and the recent development of the Canadian Environmental Protection Act (CEPA) have addressed some of the preceding shortcomings. However, a considerable effort and commitment are needed to develop the fundamentals of HW management and control, not the least of which is a common approach to waste management.

The differences in HW classification criteria between the Provinces and the Federal government, together with differences in requirements for "safe disposal" as perceived by each Province or the Federal government, make for possible varying "standards" for disposal of the *same hazardous waste or toxic substance.* What is classified as a HW in one region may not be classified as such in another region, thus providing the opportunity for more or less stringent conformance requirements for disposal. Whereas some control exists through the Transportation of Dangerous Goods Act (TDGA), the TDG regulations (TDGR) which identify a host of "dangerous materials" in nine classes, are not specifically designed to address permissible levels and concentrations of contaminants or toxic substances. These (levels or concentrations) are key elements in design and protection requirements for the geotechnical engineer in formulating disposal technology, and for the generator in the discharge of HW. The priority substance list of CEPA which deals with toxic substances considers only a small number of the more than 70,000 chemicals presently listed. Thus, with no published permissible levels, the problem of "safe dosage" is left to standards established by "Drinking Water Standards" or some such equivalent.

Various countries, and even various regions/cities in individual countries have different definitions or criteria characterizing or differentiating between hazardous[10] and non-hazardous waste, and even municipal solid waste. The *Directive on Toxic and Dangerous Waste*[11] of the European Economic Community establishes the Community Law for all the twelve member states. However, only three member states have adopted the Directive list of materials[12] without modification. Other member states have either modified the List or published their own lists. In the U.S., the lists of hazardous wastes are found in 40 CFR parts 261.31 through 261.33. The four lists include "hazardous wastes from non-specific sources" (261.31), "hazardous wastes from specific sources" (261.32), "acute hazardous" (261.33(e)), and "toxic" (261.33(f)). It should also be noted that as in the case of the TDGR, the Directive List and the U.S. lists are not specifically designed to establish target levels (i.e. permissible concentrations or levels).

In some Provinces (in Canada), no exact definition of what constitutes a HW exists (except as a "special waste" or perhaps through special legislation by the Provincial Minister of the Environment). Reliance on the TDGR for tracking and overall management of hazardous goods constitutes a major part of the HW handling procedures. Disposal requirements and standards do not fall within the jurisdiction of the TDGR; these are left to the individual Provincial and the Federal governments for their respective jurisdictions. The CCREM 3-year Action Plan for the Management of HW[13] was initiated in 1987 in recognition of the fact that HW was being managed in Canada *"in a manner that could pose a significant danger to public health and the environment"*, – a conclusion which was reached because of the realization of the inconsistencies in regulations and inadequacies of treatment and disposal facilities.

Paramount in the problem of waste management is the manner in which HW is finally disposed (grave). Because of the dangers posed by disposal of potentially hazardous materials (waste) in the ground, initiatives taken by many countries – via legislation or recommended guidelines/standards – presently require that some proper means for neutralization and detoxification of the HW be implemented before discharge (e.g. HSWA 1984 and RCRA 1976). The lack of absolute procedures for neutralizing or detoxifying ("treating") all of the elements in HW require that the technology used for "safe disposal" (grave) be most secure. Frequently this means using procedures which provide for a form of encapsulation of the "treated" HW.

SAFE DISPOSAL TECHNOLOGY

The Perceived Role of Regulations, Guidelines and "Permitting" Requirements

As seen from the perspective of the geotechnical engineer responsible for creating a safe "grave", the role of regulations, guidelines and/or "permitting" requirements is to provide the conformance requirements and criteria – "targets" – which if met, would ensure that the HW has been safely "managed". Regulation 309 under the Ontario Environmental Protection Act[14] provides comprehensive lists which describe HW and general waste management disposal practice covering such items as standards and siting for waste disposal sites, carrier requirements, access roads, and supervision. The technical details concerning siting, design and operation of a landfill which are embodied in the *Reasonable Use Policy*[15], advocates the use of the *natural attenuation* concept as the most reasonable concept for landfilling. By doing so, it is admitted that some contaminant discharge (leachates) from the landfills will occur. In the Québec document *Guide d'implantation et de gestion de lieux d'enfouissement de sols contaminés*[16], the landfill disposal procedures and "rules" are given without necessarily acceding to the natural attenuation concept. The evident "fuzziness" in many of the "signposts or rules" in both types of documents has often been cited as being deliberate – to permit "discussions" to occur between the "Consultants" and the Permitting Agency (PA)[17].

The concerns of the geotechnical engineer in regard to responsibilities attendant with safe handling and disposal of HW arise from conformance requirements and guidelines. It is not unusual for published guidelines[18] for disposal to be inadequate, and misleading – based on outdated knowledge and information or even faulty knowledge. If "failure" occurs because of (or in spite of) adherence to conformance requirements, regulations or published standards, what is the responsibility of the "Consultant (geotechnical engineer)" ? The Province of Ontario Bill 220[19], sometimes referred to as Ontario's answer

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[10] e.g. the U.S. definition of hazardous wastes in RCRA 1976

[11] Directive 78/319/EEC

[12] The list of materials attached to the Directive establishes the identified wastes as candidates for control according to the provisions of the Directive.

[13] The Action Plan was approved in principle in October, 1986.

[14] Amended by O. Reg. 175/83; O. Reg. 574/84; O. Reg. 464/85; O. reg. 460/88; and O. Reg. 750/88.

[15] Ontario Ministry of the Environment, Policy No. 15-08, *The Incorporation of the Reasonable Use Concept into the Environment.* This policy became a formal Ministry policy in 1986.

[16] Ministère de l'Environnement du Québec, Direction des substances dangereuses, janvier, 1988.

[17] Subtitle C of RCRA 1976 provides the equivalent format for HW management in the U.S. In the UK, DOE (1986) Waste Management Paper No. 26, *Landfilling Wastes*, is generally used as a code of practice for the "Consultants" and PA.

[18] Or even "permitting" requirements, and/or consultations with the PA.

[19] Passed on June 28, 1990. Contains technical amendments

to the Superfund law[20] raises several very interesting, albeit disturbing, issues to the geotechnical engineering community – not the least of which is the responsibility to "properly" detect and assess environmentally unacceptable subsurface contamination. Technology, criteria, permissible levels, tests, toxicity and exposure assessments vis-a-vis "damage" are some of the items that need very serious study and understanding by the geotechnical engineer participating in monitoring, environmental impact assesment (EIA), landfill disposal siting and implementation.

Permissible Levels, Concentrations – "Targets"

Assessment standards and criteria are central to the whole issue of determination of acceptance (or denial) of a safe HW landfill facility. Without the publication or issuance of permissible levels or concentrations, not only will handling and disposal technology be affected, but also monitoring and assessment. Since design "standards" exist only in the form of performance (short and longterm) requirements, and since these are generally articulated in the form of permissible levels of pollutants detected at some point distant from the landfill site over a particular period, the importance of the "limiting concentrations" specified cannot be overstated. Not only must these limits cover the spectrum of toxic and health-threatening pollutants emanating from the waste as leachates, these (limits) must also encompass the likelihood of other toxic compounds formed within the substrate. The attendant requirement for design performance predictions to properly model the interactions and "predict" transport and attenuation of the pollutants in the substrate is critical. Contaminant (pollutant) transport modelling constitutes one of the most challenging issues – if total account of the various complex interactions is to occur – especially since acceptance of the capability and reliability of the performance of the completed disposal facility (landfill?) revolves, almost exclusively, around the "accuracy" of the predictions made by the model.

Monitoring of effectiveness of treatment prior to disposal, together with monitoring longterm effects of disposal technology, requires quantitative assessments and target limits. While permissible concentrations and limits (endpoint "targets") for various types of waste substances[21] in water, groundwater and leachates are available, the lists by and large do not cover the complete spectrum of contaminants found in HW leachates. The endpoint "targets" refer only to concentrations in the fluid phase (water or groundwater). In groundwater assessment, extractable pore water from the soil substrate[22] is used for chemical (contaminant) examination. Since the "rules" and regulations do not pay any attention to pore water not "easily" extractable from the soil, monitoring and assessment procedures do not therefore account for the contaminants remaining in the non-extractable pore water. Since the processes of contaminant accumulation in soils generally result in greater concentrations (per unit pore water volume) of contaminants in the non-extractable pore water, awareness of the real concentration of contaminants at any one point, using present accepted procedures, is denied. Adsorbed contaminant ions in the inner Helmholtz layer are not easily "removed". It is also doubtful that all the adsorbed contaminant ions in the outer Helmholtz layer are easily extractable using the techniques presently practised in geotechnical engineering[23].

DISPOSAL OPTIONS AND STRATEGY

For the geotechnical engineer (consultant?), the *"siting"* of the disposal facility, and the design of technology for waste containment in the ground (grave) must not only satisfy criteria developed from the "Regulations" or through a set of guidelines, but also the scrutiny by the PA (Permitting Agency). The combination of these are (in theory) designed to obtain the "most compatible site and the best technology" which would protect both human health and the environment. The general specification of synthetic membranes, required as liners underlying the waste in landfills, to provide impermeable barriers to leachate transport into the ground substrate, constitutes the primary and single most important design consideration in landfill containment of HW[24]. The various design ramifications encountered deal with site specific issues, and do not detract from the fundamental requirement – the need to ensure that *no leachate escapes from the contained waste material.* The life-span of the synthetic membrane liners is an issue that has yet to be satisfactorily resolved. Since the need to provide proper entombment of HW over the period required for it to become *"neutralized"* or *"detoxified"* can be over a hundred years (depending on the nature of the waste and the environmental factors), the integrity of the membrane liners must be assured throughout this period. To date, manufacturer guarantees and assurances for survival times for such long time periods are not available. Test procedures for simulating both time and chemical assault, on an accelerated scale, have not provided the kinds of information which would permit one to confidently assume survivability of the membrane liners.

Partly in recognition of the above, and partly because of "differences" in containment philosophy, the use of "engineered clay barriers" for landfills designed to contain HW is also permitted in many countries. Under such circumstances, the regulations expect that some contamination of the ground will occur[25]. The belief is that when the contaminants (in the leachate) reach the groundwater, the attenuating processes in leachate contaminant-clay barrier interaction, have reduced the concentration levels of the pollutants to sufficiently low levels, and will not pose health threatening risks. This is sometimes referred to as *DILUTION IS THE SOLUTION.* However, two particular disturbing circumstances exist:

• The capability of the geotechnical engineer to understand and design the buffer system to provide full attenuation of the contaminants in transport through the engineered clay barrier is generally less than adequate. Insufficient background work in this difficult subject prevents one from dealing with all the various types of contaminants and their interaction with the ground.

•• Published permissible limits (targets) generally do not

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to Ontario's Environmental Protection Act and the Water Resources Act.

[20] U.S. Comprehensive Environmental Response, Compensation, and Liability Act (Superfund), 42 U.S.C. §9601 et seq., enacted by the U.S. Congress in 1980, has often been referred to as CERCLA or the "Superfund Law". Significant revisions to CERCLA were enacted by the U.S. Congress on October 17, 1986 through the Superfund Amendments and Reauthorization Act (SARA), Pub. L. No. 99-499, which essentially made CERCLA bigger and more complex. The combination of CERCLA (Superfund) and SARA should strictly be referred to as CERCLA.

[21] e.g. Schedule 4, Regulation 309, Environmental Protection Act, Ontario; Water quality and groundwater quality standards for the various provinces in Canada and states in the U.S.

[22] Since there are no standards governing extraction procedures, the amount of water extracted from soil varies between different organizations, groups, etc., dependent to a large extent on the technical capability of the "consultant".

[23] Determination of contaminant species and concentrations in the soil solid phase provides a partial resolution to the problem. However, this does not necessarily relate to the published endpoint "targets".

[24] In one Province, the exclusive use of the double-membrane liner is prohibited (i.e. reliance only on the liner without necessary added substrate protection via engineered clay barriers.

[25] As in the Reasonable Use Policy of the Ontario Ministry of the Environment *(footnote No. 15).*

cover the spectrum of pollutants being transported in the ground, and generally refer only to the contaminants in the extractable pore water (or groundwater).

A necessary function of the ground in containment of waste material, and in protection of the quality of groundwater, is to provide the capability to buffer the waste leachates such that the contaminating constituents (substances) are *held* within the soil matrix For proper design considerations (or analysis), it is necessary to study and understand the many kinds of interactions established between the various contaminants and soil constituents. To do so there is a need to understand what constitutes the waste material being disposed, *via landfilling or some other means of land containment*, and in turn understand what the soil constituents are since interactions established are directly related to the characteristics and properties of the interacting partners. Attenuation of contaminants (hopefully to the permissible targets) in transport through the substrate, generally determined from laboratory tests and analyses, do not as a rule consider longterm chemical, biological and physical changes in the various interactions between the contaminants and the substrate. In consequence, it is not clear whether predictions made in regard to contaminant concentrations in the substrate, at various periods, (and distances away from the waste site), are correct. Verification procedures are inadequate because of the problem of time simulation procedures

CONCLUDING REMARKS

Much work remains in the study of contaminant buffering and transport problems in the landfill (grave) aspect of HW management Until proper information becomes available, published guidelines concerning disposal standards cannot provide the necessary assurances for HW containment to satisfy the health and environment threat problems Design procedures for implementation of HW disposal technology cannot be considered as "standard", not only because the nature of the waste material to be landfilled cannot be fully categorized - not insofar as the contaminant load is concerned - but because of the various items which deal directly with regulatory policies, site specificities, and the very large unknown "package" of *safe disposal requirements and minimization of threat to human health and the environment*. It is nevertheless clear that the "safe" disposal of HW cannot await the outcome of all the detailed sets of studies and deliberations that need to occur between researchers, practitioners and policy makers. For the geotechnical engineering profession, procedures for "safe" design of disposal facilities require "best" knowledge capability in understanding the many sets of interactions occurring between the various pollutants in the waste leachate and the engineered clay barrier and soil substrate - particularly since regulatory policies appear to rely on the "attenuating" properties of the barrier and substrate to neutralize pollutant concentrations to the point where they pose *no* threat to human health and the environment. Modelling procedures, vital to the success of predicted performance capability and reliability of the constructed disposal facility, need to be properly aware of the longterm interaction effects. The burden of responsibility for "safe" design containment in disposal technology needs to be shared between the policy makers/implementors and the practitioners Accordingly, the sets of interactions that need to be developed between them should occur throughout the process of disposal technology development and implementation.

Comptes-Rendus
Premiere conference canadienne de geotechnique environnementale
Editeurs: R.P. Chapuis et M. Aubertin. Montreal, Quebec. mai 1991

SÉLECTION DES SITES ALTERNATIFS POUR L'ÉLIMINATION DES DÉCHETS DE GRANDES MUNICIPALITÉS

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RÉSUMÉ

Au Québec (selon le MENVIQ), 40% de sites qui présentent un potentiel de risque élevé pour l'environnement et pour la santé publique, appartiennent aux anciens dépotoire des déchets. A la lumière des études géoenvironnementales récentes, il semble nécessaire de changer les exigences courantes contre l'ensemble des critères bien déterminés. Ceux-ci sont indispensables pour sélectionner des lieux alternatifs pour le traitement des déchets municipaux, en minimisant l'impact considérable sur l'environnement. L'article démontre la sélection des critères convenables pour différents scénarios de traitement des déchets.

INTRODUCTION

Présentement, les normes environnementales deviennent de plus en plus exigeantes pour contrer la pollution. Cependant, selon l'étude du ministère de l'Environnement du Québec, la plupart des anciens sites d'enfouissement présentent actuellement un potentiel de risque élevé pour la santé publique et/ou pour l'environnement (GERLED, 1986). Environ 40% de tous les lieux contaminés, détectés au Québec, sont des sites d'enfouissement. Quelles sont donc les raisons de cette situation? Sorte de déchets enfouis? Localisation des sites? Probablement les deux.

Les grandes métropoles cumulent plusieurs millions de mètres cubes de déchets municipaux chaque année. La province du Québec génère annuellement 5,6 millions de tonnes de déchets solides dont 2,1 millions sont les déchets municipaux composés des déchets résidentiels, urbains et ceux de petits commerces.

La composition des déchets varie en fonction du temps, du développement de la ville et des habitudes de sa population. Par exemple, au Québec, la composition des déchets se situe entre celle des États-Unis et celle des pays de l'Europe occidentale, et elle contient: 30,4% de matière putrescible, 28,5% de papier 10,9% de plastique, 10.9% de verre, 6,8% de métaux et 13,4% d'autres (MENVIQ, 1987).

Il est évident qu'une partie importante des déchets est dissimulée sous le nom "autres". Des études récentes approfondies, menées par le GRAIGE (1989), ont permis de diviser les déchets municipaux en 47 catégories. Cette information semble être primordiale au cours de l'évaluation du degré d'influence de la composition des déchets sur les processus physico-chimiques et biologiques ayant lieu (à long et à court terme) parmi les déchets diversifiés et entre les déchets décomposés et l'environnement.

La réglementation concernant les déchets acceptables à enfouir sur un site a évolué, et est inclus dans les normes canadiennes sous la forme actuelle. Pourtant, elle diverge toujours de celle développée dans les autres pays (ex.: Allemagne ou France, reliées aux normes de la Communauté européenne). Cependant, prenant en considération la sorte des déchets présentement acceptés, il faut établir une nouvelle échelle de critères, servant au choix des site pour les installations à traiter les déchets.

La politique vis-à-vis l'élimination des déchets évolue constamment. Le site, qui est prévue pour vingt ans de l'enfouissement, ne cumule pas nécessairement la même sorte de déchets au cours de toute la période de son fonctionnement. Par exemple, la politique de la revalorisation des déchets (surtout de la matière organique) peut changer de façon diamétrale les processus physico-chimiques et biologiques ayant lieu dans le site d'enfouissement de même que la composition des déchets brûlés dans l'incinérateur ou compactés à la station de transbordement.

La mauvaise expérience des années passées a démontré la nécessité d'établir une politique plus consciencieuse quant à la gestion des différentes sortes de déchets. L'environnement est beaucoup plus sensible que l'être humain ne puisse le croire.

Donc, les sites de traitement des déchets, sujets au principe de la protection de l'environnement, doivent être localisés et préparés soigneusement. La lixiviation à partir des sites d'enfouissement, l'émission des gaz à partir de l'incinérateur, la déposition des cendres volantes, le recueil des déchets encombrants, la déposition des déchets secs provenant de démolition ainsi que plusieurs autres problèmes complexes exigent une attention particulière.

Dans ces conditions, la vérification des plusieurs paramètres de localisation et des paramètres géologiques, hydrologiques, et sociologiques est indispensable pour choisir un site en accord avec les exigences des normes contemporaines. Ces exigences se réduisent à la minimisation de l'impact environnemental, à la minimisation du volume des déchets déposés aux sites d'enfouissement, à l'optimisation de la revalorisation des déchets et à la minimisation des coûts d'opération.

Dans ce contexte, il y a présentement plusieurs systèmes de traitement des déchets et dans chaque système plusieurs technologies de plus en plus sophistiquées. Est-il nécessaire d'utiliser toujours la même méthode pour choisir le site convenable pour chaque système spécifique de traitement? Quels sont les critères adéquats pour effectuer ce choix?

Jusqu'à présent, les critères précis, pour le choix des sites, ne sont pas disponibles. Par conséquent, dans cet article, on présente l'ensemble des critères classifiés, nécessaires à vérifier avant l'implantation d'une installation quelconque. La sélection de ces critères est basée sur les diverses réglementations (canadiennes, américaines et européennes) ainsi que sur les résultats d'études scientifiques récentes dans ce domaine. Les études ont été effectuées pour six types de traitement des déchets, incluant les installations le plus souvent implantées dans les grandes agglomérations urbaines. Les types de traitement pris en considération sont les suivants: centre de tri et de récupération, déchetterie, usine de compostage, incinérateur, centre de transbordement et site d'enfouissement.

SYSTÈME DE TRAITEMENT DES DÉCHETS MUNICIPAUX

Centre de tri et de récupération

La récupération des déchets à être recyclés peut être réalisée par le système de tri à la source ou par le tri effectué dans une usine spécialisée. Les matériaux sélectionnés peuvent être recyclés pour servir de matière première dans la fabrication de nouveaux produits. Les techniques de tri se développent constamment, mais ils peuvent être réparties en deux groupes: 1) tri avec broyage préliminaire réalisé en milieu humide, en milieu semi-humide et en milieu sec; 2) tri en milieu sec sans broyage préliminaire.

Déchetterie

Il existe toujours certaines quantités de déchets qui ne peuvent pas être ramassés de façon conventionnelle; ce sont les encombrants, les déchets secs, les produits dangereux domestiques, les produits verts, etc. Ces déchets pourraient être gérés par les résidants, s'il trouvaient dans leur voisinage un endroit spécialement prévu à cette fin. Ces installations sécuritaires, où chaque type de déchets ou matière est stocké en un lieu distinct s'appelle une déchetterie.

Usine de compostage

Certaines quantités importantes de déchets organiques putrescibles, ramassés et enfouis, peuvent être recyclés. L'usine de compostage peut convertir ces déchets en amendement stable au moyen d'une activité bactérienne aérobie. Il existe actuellement une trentaine de procédés de compostage qui se basent soit sur la fermentation lente soit sur la fermentation accélérée.

Incinérateur

Une autre méthode appliquée à l'élimination des déchets est leur combustion. Plusieurs types d'incinérateurs sont construits à des volumes qui varient d'une municipalité à l'autre. La tendance actuelle va vers la construction d'incinérateurs de large capacité donc le coût de système d'épuration est moins élevé que celui des petites constructions. Les types d'incinérateurs des déchets le plus couramment installés sont les suivants: incinérateur à système modulaire, incinérateur de déchets non conditionnés et incinérateur à combustible dérivé des déchets (CDDM). Maintenant, l'unité à combustion possède un système qui permet de récupérer l'énergie dégagée.

Site d'enfouissement

Entre autres, il existe toujours certaines quantités de déchets qui doivent être enfouis avec sécurité. Les terrains recherchés pour le site d'enfouissement devraient être de taille à inclure des déchets provenant d'une grande agglomération, sur une période de 20 ans.

On considère quatre procédés de décharge sur le site: 1) décharge contrôlée d'ordures brutes (en vrac); 2) décharge contrôlés d'ordures déchiquetées; 3) décharge utilisant la technologie de pré-fermentation (charge supplémentaire de matière organique facilement dégradable, ex.: boues des stations d'épuration); 4) décharge des déchets compactés provenant soit des stations de transbordement soit des installations implantées sur les sites (production des ballots).

Centre de transbordement

Dans le cas d'implantation d'un site d'enfouissement éloigné par rapport à la source productrice des déchets, il est nécessaire de construire une ou plusieurs centres de transbordement de déchets dans les milieux urbains. Dans ces endroits, les déchets sont transférés des véhicules soit au lieu de reprise, soit aux containers prévus pour le transport à longue distance. Dans plusieurs centres, un compactage supplémentaire est effectué avant le dépôt dans les containers.

CRITERES DE CHOIX DES SITES ADÉQUATS

Une gestion inadéquate des installations susmentionnées apporte un risque potentiel pour la santé des résidants avoisinants et pour l'environnement. Donc, il est nécessaire vérifier les sites selon des critères adéquats pour éliminer ce risque potentiel.

Le choix du site adéquat dépend des critères qui sont pris en considération. Si les critères sont incomplets, le site considéré ne serait pas optimal. Par conséquent, l'évaluation des critères pour chaque installation est un élément primordial et doit être développée soigneusement. On propose la méthodologie qui suit le chemin critique (critical path) démontré aux figures 1 et 2. Il est évident que le chemin critique doit être utilisé pour toutes les installations. Cependant, les extensions des critères spécifiques peuvent différer d'une installation à l'autre. Le chemin critique prend en considération les groupes de critères de localisation, les critères géologiques, hydrologiques, sociologiques et autres.

Pour optimiser ces critères, les auteurs ont tenu compte des normes provinciales, des normes et des lignes directrices fédérales ainsi que des normes et des guides tant techniques qu'environnementaux utilisés dans d'autres pays, en plus, des plans directeurs des grandes agglomérations ainsi que des publications et des travaux récents pertinents.

CRITERES DE PREMIERE GENERATION
↓
PREMIERE SELECTION
↓
CRITERES DE DEUXIEME GENERATION
↓
DEUXIEME SELECTION
↓
CRITERES DE TROISIEME GENERATION
↓
TROISIEME SELECTION
(voie critique)
↓
première étape

deuxième étape

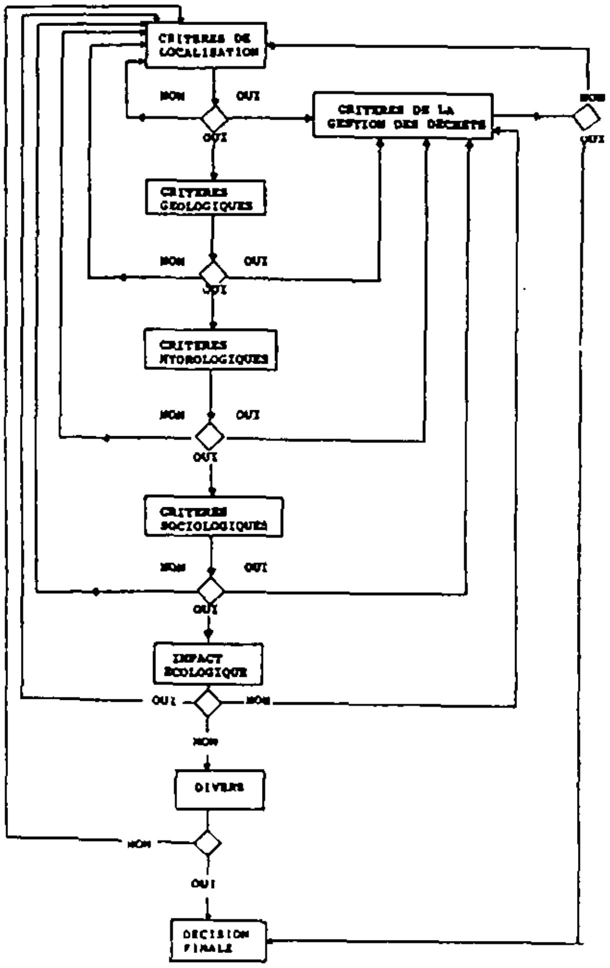
troisième étape

quatrième étape

cinquième étape

Fig. 1. Système de sélection des sites identifiés

Fig.2. Méthologie de choix des sites adéquats à l'implantation des installations pour traiter les déchets municipaux



Critères de la première génération

Les critères établis pour la première sélection concernent la superficie approximative du terrain nécessaire à l'implantation des installations ainsi que la distance de la source productrice.

Pour les six groupes d'installations pris en considération (voir chapitre précèdent), il est évident que chacune de ces installations représente des besoins d'aires différentes. Parmi ces groupes, la superficie de la déchetterie est la plus petite. Elle doit quand même assurer l'espace nécessaire pour déposer les déchets sélectionnés, donc les surfaces inférieures à 50 m x 80 m doivent être rejetés. Tous les autres terrains identifiés peuvent être divisés en cinq catégories: I - superficie de moins de 5000 m^2, II - entre 5000 et 10000 m^2, III - entre 10000 et 100000 m^2, IV - entre 100000 et 500000 m^2 et V - superficie supérieure à 500000 m^2. Leur affectation est examinée en fonction des besoins des installations particulières.

La superficie approximative du site d'enfouissement peut être évaluée à l'aide d'une formule empirique ou à l'aide des données particulières disponibles.

La superficie de l'usine de compostage peut être évaluée selon les hypothèses suivantes: - la fermentation lente exige des aires où les andains à section triangle (6 m x 100 m x 3 m) correspondent généralement au traitement sur une semaine; - une programmation des retournements nécessite au moins quatre semaines de travail; - la fermentation accélérée exige des aires suffisantes pour la construction d'installations utilisant les procédés les plus sophistiqués; - tant la fermentation lente que la fermentation accélérée exigent des aires pour les déchets transportés, pour la maturation et pour le stockage, (ex.: pour une unité de 60 t/j de déchets, la superficie totale nécessaire est de 8000 m^2).

Une première sélection des sites potentiels et leur affectation est réalisée en se basant sur la superficie du terrain et leur distance de la source productrice.

Critères de deuxième génération

La deuxième sélection doit être réalisée selon les exigences spéciales soulevées par les services municipaux concernés et impliqué dans la gestion des déchets. Les éléments se rapportent à la politique municipale, à la vocation de certains terrains ainsi qu'aux priorités des installations.

Les exigences des agglomérations étendues sont beaucoup plus grandes que celles des plus petites ou des villages. Les terrains sont plus chers, les vocations sont plus diversifiés, la politique municipale est plus sensible, la circulation est plus fréquente, l'habitation plus dense, la monopolisation de la gestion des déchets est problématique, le danger de l'échec est plus élevé, les rebuts d'industries secondaires plus diversifiés etc.

La politique d'une ville joue un rôle important dans la sélection de sites convenables à l'implantation d'installations et une sélection basée sur les critères de deuxième génération doit être effectuée avant la vérification des critères plus précis.

Critères de la troisième génération

Au cours de la troisième sélection, des critères plus précis doivent être pris en considération. L'ensemble des critères potentiels contient quatre grandes groupes, tels les critères de localisation, les critères géologiques, les critères hydrologiques et les critères sociologiques. Ces groupes ont été divisés en catégories plus précises. Ils sont établis en fonction de six traitements divers et en fonction de la nature des déchets.

Critères de localisation

Les éléments qui doivent être étudiés dans le groupe des critères de localisation sont les suivante: 1) distance minimale devant être respectée par rapport aux différents cours d'eau; 2) accessibilité au site; 3) utilisation des sols, prenant en considération la distance qui les sépare des terrains résidentiels, des parcs, des terrains de récréation, des terrains agricoles, des zones protégées (ex.: zone d'inondation, réserve faunique, etc), lignes électriques de haute tension; 4) espace supplémentaire (ex.: parc mécanique, station de lavage des véhicules, équipement anti-incendie); 5) superficie du terrain nécessaire à implanter chaque installation selon le procédé particulier ainsi que la hauteur des constructions avoisinantes (ex.: études sur la dynamique de la dispersion des gaz).

Critères géologiques

Les critères géologiques comprennent les éléments suivants: 1) stratigraphie du terrain nécessaire pour étudier la superposition des couches géologiques (barrières naturelles); 2) disponibilité des matériaux de recouvrement (particulièrement importants lors de l'enfouissement des déchets); 3) minéralogie du terrain, pour donner la chance de prévoir des processus interactifs entre les eaux de lixiviation et les particules des sols; 4) tectonique du terrain afin de détecter la fissuration des rocs et les fractures naturelles, les failles, les terrains séismiques, etc; 5) géomorphologie du terrain (inclinaison du terrain, pente, talus, affleurement des rocs, glissements de terrains, etc); 6) comportement des sols à long terme, vis-à-vis de la qualité et de la concentration du lixiviat; 7) critères hydrogéologiques comprenant: les paramètres géotechniques, les paramètres physico-chimiques, la possibilité de former des barrières géologiques, la possibilité du contrôle continu de l'impact, les résultats des tests de compatibilité des sols et du lixiviat nécessaires afin de prévoir les processus complexes biochimiques ainsi que la transmissibilité du lixiviat, la diffusion, la absorption, la désorption, la chélation, etc.

Critères hydrologiques

L'ensemble des critères hydrologiques, pris en considération est déterminé par : 1) la variation de la nappe d'eau, permettant ainsi d'estimer la dynamique des eaux souterraines (influençant la solution de la construction d'ingénierie, la possibilité d'inondation, la migration du lixiviat, etc); 2) la distance d'un cours d'eau (continuels ou temporaires), permettant d'évaluer le ruissellement direct vers les cours d'eau, la variation du gradient des eaux souterraines (qui doit être vérifié en aval et en amont du site), l'inondation, le drainage, etc); 3) la facilité d'infiltration d'eau exprimée par la conductivité hydraulique du terrain qui est testé in situ et au laboratoire; 4) la situation du site en rapport avec les prises d'eau existantes et potentielles qui est importante pour les raisons suivantes: - possibilité d'influencer la qualité d'eau de surface captée; - possibilité de diminuer la qualité de l'eau dans les puits avoisinants; - croissance du gradient d'eau souterraine vers les puits et/ou bassins captants, démontrant la vitesse réelle de la filtration; 5) la facilité de capter des eaux de lixiviation qui est en rapport avec l'étude sur le drainage du lixiviat, des eaux d'infiltration et des eaux de ruissellement sur le terrain en question et qui influence la faisabilité des installations particulières; 6) la qualité des eaux souterraines qui doit être évaluée non seulement au cours de l'exploitation des installations, mais aussi avant la décision de les implanter (la chimie de l'eau peut exclure la possibilité d'implantation immédiate des installations et exiger de nettoyer premièrement le terrain); 7) les paramètres climatiques tels les variations de température, la force et la direction des vents, le niveau de congélation des sols, etc, (influençant le choix du procédé, la localisation des installations, l'extension de travaux d'ingénierie, etc.)

Critères sociologiques

Les critères sociologiques sont divisés en plusieurs groupes. Cependant, il est évident qu'ils ne couvrent pas toutes éventualités qui peuvent apparaître selon le caractère spécifique de l'agglomération et les habitudes des résidants. De manière générale, les critères sociologiques sont groupés comme suit: 1) la *santé et la sécurité*, exigeant d'attirer l'attention sur la vermine, le transfert pathogène, la prolifération des espèces, l'accès possible des animaux et des enfants, les dangers de la circulation: 2) l'environnement, prenant en considération la localisation du site vis-à-vis des voisins, l'émission d'odeurs, l'émission des gaz, l'émission de poussière, la contamination des eaux souterraines et de surface, la contamination des rues par les pneus des véhicules ou par les rebuts répandus, le bruit; 3) l'esthétique, tenant compte de l'impact visuel, de l'impact commercial et de la possibilité de rebuts répandus; 4) la communauté qui est assujettie au décroissement de la valeur des propriétés, au changement de la topographie du terrain, au changement de l'image historique du quartier, au départ des compagnies, à la circulation intensive, au soin continuel des espaces fermés, à la création des nouveaux emplois, au changement de la composition des habitants, aux frais d'accessibilité et aux taxes imposés, à la récupération de l'énergie (gaz et vapeur), l'usage du compost, l'impact sur les réserves indiennes; 5) l'impact sur les espèces sauvages ou uniques; 6) la dégradation des voiries; 7) l'exigences de maintenance et de sécurité; 8) l'impact de l'extension éventuelle du site; 9) les exigences de la construction qui influencent les critères économiques.

La *sélection des sites selon* la troisième génération doit être effectuée après avoir vérifié les critères susmentionnés. La troisième sélection exige d'être effectuée en quelques étapes successives.

La première étape est un collecte des données, et par conséquent, la première sélection des sites est faite à la base des données déjà disponibles dans les rapports et publications ultérieurement effectués, des visites descriptives sur place, des photographies, et des tests macroscopiques des types de sols, de la topographie ainsi que d'études sociologiques générales.

La deuxième étape consiste en études géologiques, tectoniques, geomorphologiques des terrains potentiels ainsi que en études hydrogéologiques macroscopiques.

La troisième étape de sélection étudie les critères sociologiques qui doivent être vérifiés à plusieurs niveaux: la direction de la ville, gestionnaires du quartier, les habitants du quartier, les associations impliquées et les principales compagnies installées dans le quartier. L'étude doit être effectuée sous forme d'enquête (mettant en cause un nombre suffisant de répondants) et sous forme d'études spéciales: écologiques, sociologiques et techniques.

L'étape suivante est une évaluation plus précise de tous les critères hydrogéologiques susmentionnés.

Cinquième étape est une estimation des coûts probables pour le nombre limité d'installations alternatives.

CONCLUSION

L'implantation d'installations servant à la gestion optimale des déchets exige des études approfondies. Les réglementations actuelles ne s'appliquent qu'à certaines installations. Par conséquent, elles ne couvrent pas toutes les éventualités de la gestion des déchets municipaux (incluant la revalorisation des matériaux et la récupération de l'énergie). Dans ce contexte - la nature des déchets gérés dans chaque cas n'est pas ressemble; les processus physico-chimiques sont différents, et par conséquent, les critères d'évaluation *des sites convenables divergent*.

Le réglement existant se rapporte à certaines méthodes d'élimination des déchets; elle ne contiennent pas tous les critères nécessaires à la construction du site optimale et sûr.

Selon les études menées, plusieurs paramètres ne sont pas pris en considération, en surplus, il y a divergence entre les normes des différents pays. En raison que la science géoenvironnementale est présentement capable de répondre à plusieurs questions fondamentales sur lesquelles, devraient être basées les normes et les guides techniques actuelle au Canada.

Le choix d'un site convenable à l'implantation des installations susmentionnées pourrait donc être fait à partir des critères présentés dans cet article.

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A MULTIPHASE FLOW MODEL FOR REMEDIATION OF SUBSURFACE CONTAMINATION BY NONAQUEOUS PHASE LIQUID

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ABSTRACT

Groundwater contamination by nonaqueous phase liquids (NAPL) such as organic solvents and petroleum hydrocarbons frequently occurs as a result of surface spills, tank leaks, and improper disposal practices. In this paper a multiphase flow model for NAPL through porous media is derived. The model takes into account the effect of volume change due to the surfactant injection in NAPL/soil mixtures. The model predicts the experimental data satisfactory. In addition, a comparison with some of the existing models is discussed.

INTRODUCTION

In recent years groundwater contamination by organic solvents and other petroleum products has reached significant proportions. This contamination has occurred both at low level contamination from sanitary landfills and as concentrated liquid spills due to leakage from underground tanks, accidental surface spills, or disposal sites. This paper addresses the problem of a nonaqueous phase liquid (NAPL) flow in porous media during surfactant injection as a means of remedial action on the waste disposal site.

The earlier literature on groundwater pollution by petroleum hydrocarbons has generally recognized that when a NAPL is spilled, the NAPL migrates through pore spaces and some liquid is retained within the pores by capillary forces. The review of multiphase flow theory by Van Dam (1967) is very perceptive of dominant mechanisms and is very clear about uncertainties which arise in real filed situations and which, of the most part, re still unresolved 20 years latter.

Furthermore, during surfactant injection through porous media, as a method for cleaning up, the mobile materials (NAPL and suspended solids) are brought into contact with the fluid stream. The rate of flow of the mobile materials is dependent on the surfactant flow rate. Two approaches have been proposed for prediction of surfactant flow rate. In the first approach, investigators have used many methods to solve the problem of the flow of suspension. In one method, the work by Argon (1952) with Sternerdin's suggestions (1962), leads to a simplified energy balance equation which has been used with further refinements by Leung and Wiles (1976) to calculate the velocity of the fluid containing suspended solids. Another method has been followed by Mertes and Rhodes (1955) and Lapidus and Elgin (1957), based on the slip velocity of the particles. It leads to another formula for the velocity of the flowing suspensious.

In the second approach, the fluids have been considered as a diffused material through the porous medium. The concentration profiles can be determined by solving the classical diffusion equation (Parlange, 1972).

$$\frac{\partial\theta}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial\theta}{\partial x}\right) \qquad (1)$$

where; θ is the fluid volumetric content; t, time; x, coordinate axis; and D, the soil fluid diffusivity.

However, Eq. 1 does not represent the physical phenomenon in the porous medium during surfactant injection. This is due to the dispersion of both NAPL and suspended solids and hence the flow of both materials in the flow direction. It is also believed that during the flow there will be a time lag between the surfactant front and the suspension solution front. This study is intended to include the above mentioned effects in the diffusion equation and hence predicts the surfactant flow through porous media during the cleaning up. Furthermore, a comparison between the proposed model and some existing work is discussed.

FORMULATION

By considering the mass balance equation for the injected surfactant, the continuity equation can be written as follows:

$$\frac{\partial\theta_e}{\partial t} + div(\bar{q}_e) = 0 \qquad (2)$$

where: $\bar{q}_e$ = velocity vector of injected surfactant; θ_e = surfactant volumetric content; and t = time. In order to evaluate the soil-surfactant flux, $\bar{q}_e$, relative to a stationary co-ordiante system, it is assumed that $\bar{q}_e$ is equal to the sum of two vectors, the first, $\bar{q}_s$ is related to the movement of slug (NAPL and/or suspended solids), carrying with it its surfactant content θ_e and the second, $\bar{q}_{es}$, is related to the movement of surfactant relative to the moving particle; thus

$$\bar{q}_e = \bar{q}_{es} + \theta_e\bar{q}_s \qquad (3)$$

combining Eqs 2 and 3 gives

$$-\frac{\partial\theta_e}{\partial t} = \text{div}\bar{q}_{es} + \theta_e\text{div}\bar{q}_s + \bar{q}_s\text{grad}\theta_e \qquad (4)$$

In order to evaluate $\bar{q}_s$, it is assumed that it consisted of two vectors; the first, q_r related to the suspended solid flux, and the second, $\bar{q}_{sr}$ related to the movement of NAPL relative to the suspended solid flux. Thus,

$$\bar{q}_s = \bar{q}_{oc} + \theta_o \bar{q}_c \qquad (5)$$

where θ_o is the NAPL volumetric content or:

$$\mathrm{div}\bar{q}_s = \mathrm{div}\bar{q}_{oc} + \theta_o \mathrm{div}\bar{q}_c + \bar{q}_c \mathrm{grad}\theta_o. \qquad (6)$$

Substituting Eq. 6 into Eq. 4 yields

$$-\frac{\partial\theta_\epsilon}{\theta t} = \mathrm{div}\bar{q}_{es} + \theta_\epsilon(div\bar{q}_{oc} + \theta_o div\bar{q}_c + \bar{q}_c \mathrm{grad}\theta_o) + (\bar{q}_{oc} + \theta_o\bar{q}_c)\mathrm{grad}\theta_\epsilon \qquad (7)$$

Simplifying Eq. 7 by taking one-dimensional flow in the horizontal direction only yields

$$-\frac{\partial\theta_\epsilon}{\partial t} = \frac{\partial\cdot\bar{q}_{es}}{\partial x} + \theta_\epsilon\frac{\partial\bar{q}_{oc}}{\partial x} + \theta_o\theta_\epsilon\frac{\partial\bar{q}_c}{\partial x} + \bar{q}_c\frac{\partial\theta_o}{\partial x} + \hat{q}_{oc}\frac{\partial\theta_\epsilon}{\partial x} + \theta_o\bar{q}_c\frac{\partial\theta_\epsilon}{\partial x} \qquad (8)$$

The solution of the above equation is extremely difficult even numerically. Thus, another simplification may be introduced by ignoring the second order terms, e.g., $\bar{q}_{oc}$ grad θ_ϵ; $\theta_o\bar{q}_\epsilon$; and $\bar{q}_c$ grad θ_o. Thus, Eq. 8 becomes:

$$-\frac{\partial\theta_\epsilon}{\partial t} = \frac{\partial\bar{q}_{es}}{\partial x} + \theta_\epsilon\frac{\partial\bar{q}_{oc}}{\partial x} + \theta_o\theta_\epsilon\frac{\partial\bar{q}_c}{\partial x} \qquad (9)$$

In order to evaluate the NAPL flux, $\bar{q}_{oc}$, and the suspended solid particles flux, $\bar{q}_c$, it is assumed that these fluxes are in direct proportion to the pore water pressure gradient as discussed by Yong et al. (1990). By taking into account the effect of volume change in the continuity equation and using Darcy's law, the suspended flux, $\bar{q}_c$, and the NAPL flux, $\bar{q}_{oc}$, can be written as:

$$\frac{\partial\bar{q}_c}{\partial x} = -D_c\frac{\partial^2 V_c^*}{\partial x^2} \qquad (10)$$

$$\frac{\partial\hat{q}_{oc}}{\partial x} = -D_{oc}\frac{\partial^2 V_{oc}^*}{\partial x^2} \qquad (11)$$

where: D_c is the suspended solid particles diffusivity; V_c^*, change in volume of suspended solid particles for a detailed deerivation of Eqs. 10 and 22, the reader should refer to Yong et al. (1990) ; D_{oc}, NAPL diffusivity; V_{oc}^*, the change in NAPL volumetric content.

Combining Eqs. 9, 10, 11 yield the following:

$$\frac{\partial\theta_\epsilon}{\partial t} = -\frac{\partial\bar{q}_{es}}{\partial x} - \theta_\epsilon\frac{\partial V_o^*}{\partial t} - \theta_o\theta_\epsilon\frac{\partial V_c^*}{\partial t}$$

$$= \frac{\partial}{\partial x}D(\theta_\epsilon)\frac{\partial\theta_\epsilon}{\partial x} - \theta_\epsilon\frac{\partial V_o^*}{\partial t} - \theta_o\theta_\epsilon\frac{\partial V_c^*}{\partial t} \qquad (12)$$

Eq. 12 represents the classical diffusion equation with two additional terms which accounts for the volume change due to NAPL and suspended solid movements. It should be noted here, that the additional terms in Eq. 12 imply that the case of flow described by any negative value θ_ϵ is a physical impossibility. If in the course of computation, θ_ϵ becomes negative at some value of the injection length, this should be taken as an indication that the injection process should be stopped at, or before, this time. In addition, the calculations should be made at small time intervals, since any error involved in determining the surfactant profile will be magnified at larger time values.

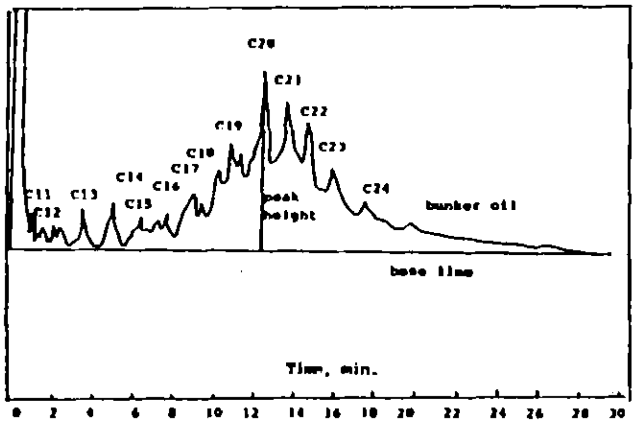

Fig. 1a Gas chromatograms of unweathered bunker fuel oil

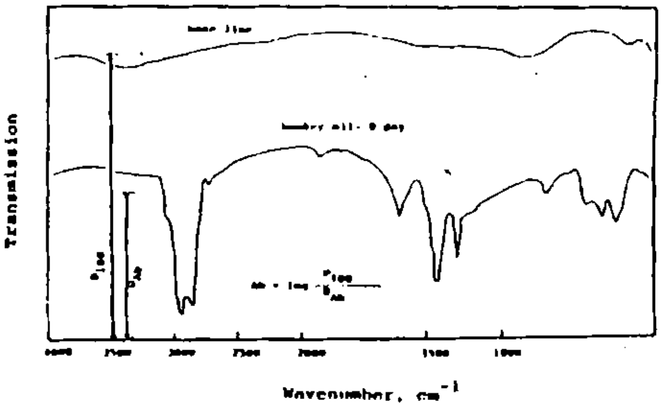

Fig. 1b Infrared spectra of unweathered bunker fuel oil

MATERIAL AND METHODS

Materials

1.0 Bunker Fuel Oil

The samples of bunker fuel oil (6C) was supplied by Gulf Oil Canada Limited. The gas chromatograms and Infrared spectra are shown in Figs. 1a and 1b respectively. The bunker fuel oil has the following properties : (1) flush (point) = 88°C; (2) sulphur weight = 2.6%.

2.0 Surfactant

A ratio of nine between emulgin 05/emulgin 010 by weight is taken after El Monayeri, 1983.

3.0 Solvent

A triple mixture, consisting of carbon disulfide, acetone and methanol in the ratio of 70:15:15 by volume, was used to extract the oil from the substrate. This mixture has been known to remove four to five times more oil than petroleum ether (Yong and Sethi; 1975).

4.0 Soils

The soils used in this study were montmorillonite and sand, and their x-ray diffraction patterns are shown in Fig. 2. The chemical analysis of the montmorillonite is given in Table 1.

Sample Preparation

To a known weight of bunker oil, a dry mixture of sand and clay (montmorillonite) was added and mixed thoroughly with a spatula. The bunker oil was preheated to obtain better mixing conditions. The ratios of clay to sand (W_c/W_s) were 0,0.05, 0.15, and 0.25, while the ratio of the oil to sand (W_o/W_s) was fixed at 0.15.

Test Procedures

The oil/soil mixture was placed in a 100.0 mm long and 50.0 mm inside diameter lucite cylinder. Equal amounts of the mixture were compacted to a length of $\approx$ 30.0 mm. The porosity of the mixture was controlled by the clay/sand ratio. The cylinder was then placed in a hot-water path adjusted to the desired temperature. The samples were ready for testing when temperature equilibrium was attained. The surfactant solution was heated to the same temperature as that of the soil sample. It was then injected through a constant head device at one end of the cylinder. The surfactant supply system consisted of two separate units: the surfactant supply tank, and the constant temperature tank. This arrangement allowed for a continuous supply of heated surfactant.

Surfactant Content Measurements

After injecting the surfactant through the oil/soil mixture for a specified period of time, samples of the oil/soil mixture were taken from the inlet and outlet positions. The samples were oven dried, and the surfactant content was determined as the ratio of the difference between the sample weight before and after the drying process, to the dry weight of solids.

Oil Content Measurements

The oven dried soil samples were crushed and then placed in thimbles, where they were washed with a triple mixture solvent (carbon disulfide, acetone and methanol in the ratio of 70:15:15 by volume). This mixture has been known to remove four to five times more oil than petroleum ether (Yong and Sethi, 1975). The soil samples were washed with the solvent until no further color developed in the solvent. The oil content, then, was determined as the ratio of the difference between the weight of the sample before and after washing to the weight of the sand.

Clay Content Measurements

After washing out the soil, the samples were air dried. They were then mixed in a sodium bicarbonate solution to disperse the montmorillonite. The supernatant liquid was removed, and the procedure was repeated until, practically, no more clay was observed in the supernatant. The ratio of the clay to sand by weight was then determined.

It should be mentioned here that these procedures sometimes yielded erroneous clay contents due to the fact that some of the clay particles could be lost during the oil washing process. These errors could be observed from the fact that $\frac{V_s}{V_T} + \frac{V_a}{V_T} + \frac{V_c}{V_T} + \frac{V_c}{V_T}$ was not equal to unity as it should be. This error could not be avoided, and it did contribute to the observed deviation between the experimental results ands the proposed analytical solution.

CALCULATION OF DIFFUSIVITY COEFFICIENTS

The present methods for calculating the surfactant, NAPL and suspended solid particle diffusivity are the same as those given by Yong and Wong (1973). There are basically two methods: the first one calculates surfactant diffusivity, and the second calculates NAPL or suspended solid particles diffusivity. The procedures are illustrated below.

Table 1 Chemical Analyses of Montmorillonite

	%			%
SiO_2	49.4	CaO		1.80
Al_2O_3	20.10	Na_2O		0.43
TiO_2	0.4	K_2O		1.12
Fe_2O_3	3.7	H_2O	moisture	5.00
FeO	n.d	H_2O	loss at 900°C	15.15
MgO	2.55			

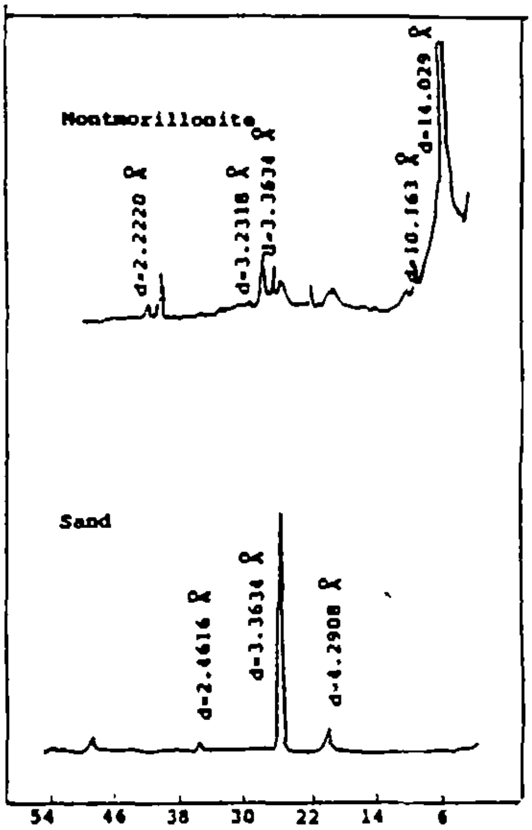

Fig. 2 Typical X-ray diffraction pattern of montmroillonite and sand (oriented particles using CuKα radiation)

Calculation of Surfactant Diffusivity

The method of calculation of surfactant diffusivity is the same as that proposed by Yong and Wong (1973), which in turn is a simplified version of that proposed by Bruce and Klute (1956). The principle of this method is to ignore the additional terms in the diffusion Eq. 12, i.e., those representing the effect of oil and suspended solid movement. This means that the usual diffusion equation is used. The calculation procedures, then, are as follows.

1. the best fitted surfactant saturation ratio $\frac{\theta_s}{\theta_{ss}}$, or s, distance (x) curve at a certain given time should be constructed. In the present study a linear relationship was assumed between two points due to the fact the measurements have been carried out at these points;
2. the interval $(S_o - S_i)$ is divided in equal intervals n, thus obtaining the values;

$$\Delta S = \frac{S_o - S_i}{n}$$

and

$$S_r = (n - r)\Delta S.$$

3. the values of n-r in column 1 of Table 2, and the values of S_r in column 3, are calculated according to step 2;
4. the values of $X(S_r)$ can be read from the experimental data and are entered in column 3;
5. column 4 in Table 2 is the value of $X(S_{r+1}) - X(S_r)$;
6. column 5 is column 4 divided by the value of ΔS;
7. column 6 can then be obtained by multiplying ΔS by the corresponding value in column 3;
8. The first row in column 7 is equal to the value of $X(S_i)(S_{n-\frac{1}{2}} - S_i)$. The second row is equal to the sum of the first row in column 6 and the first row in column 7; the third row is equal to the sum of its second row and the second row in column 6, and
9. the values of $D(S_{r+\frac{1}{2}})$ in column 8 are calculated by multiplying column 7 and column 5 and dividing the product by 2 t.

An example of calculation is given in Table 2 for 5% clay content samples at time intervals of 40 min.

TABLE 2 Example Calculation of Surfactant Diffusivity

1	2	3	4	5	6	7	8
$n - r$	S_r	$X(S_r)$	$X(S_{r+1})$ $-X(S_r)$	$X(S_{r+1})$ $-X(S_r)$	$X(S_r)\Delta S$	$\sum_{S_r}^{S_r+\frac{1}{2}} X ds$	$D(S_r + \frac{1}{2})$
0	0.977	10	1	434.78	0.23	0.0115	0.0625
1	0.9793	9	1	434.78	0.0207	0.0345	0.1875
2	0.9816	8	1	434.78	0.184	0.0552	0.2999
3	0.9839	7	1	434.78	0.0161	0.0736	0.400
4	0.9862	6	1	434.78	0.138	0.0897	0.4875
5	0.9885	5	1	434.78	0.0115	0.1035	0.5625
6	0.9908	4	1	434.78	0.0092	0.115	0.625
7	0.9931	3	1	434.78	0.0069	0.1242	0.675
8	0.9954	2	1	434.78	0.0046	0.1311	0.7124
9	0.9977	1	1	434.78	0.0023	0.1357	0.7375
10	1.0	0			0	0.1380	0.7500

Calculation of Oil (or Suspended Solid Particle) Diffusivity

The NAPL diffusivity, D_{oc}, (or suspended solid diffusivity, D_c) Calculation procedures are as follows:

1. As explained before, the sample is divided into n divisions. *These values of V_{oc}^* (or V_c^*) are calculated from the experimentally* obtained values of the oil (or suspended solid) volumetric contents $\theta_o (or \theta_c)$;
2. For each value of V_j^*, there will be a corresponding value of V_{j+1}^*;
3. The value of D_{oc} (or D_c) can, then, be obtained by solving Eq. 11 (or 10) respectively, and
4. the entire procedure is then repeated for the next iteration.

An example of calculation is given in Table 3.

TABLE 3 Sample Calculation of Oil (or Clay) Diffusion Coefficients

n	V_o^j	V_o^{j+1}	$D \times 10^{-3}$ cm^2/sec
1	0.0071	0.0143	
2	0.0068	0.0138	0.85
3	0.0066	0.01333	0.852
4	0.0063	0.0129	0.889
5	0.00612	0.0124	0.854
6	0.00588	0.0119	0.871
7	0.0054	0.01094	0.836
8	0.00517	0.01046	0.856
9	0.0049	0.00998	0.858
10	0.0047	0.0095	

$D_{oc} = 0.858 \times 10^{-3} cm^2/sec$

PREDICTION AND COMPARISON WITH EXISTING WORK

In this section, the analytical model used in this study will be compared with the existing theories. Three sets of measurements will be used as the basis for the comparison. These sets refer to three samples containing different amounts of clay and sand. The clay/sand rations were 5, 15 and 25% (W_c/W_s). The injection pressure head in all cases were kept constant during the injection period.

The study of multiphase flow can be approached by two methods which will be referred to as the first approach and the second approach.

In the first approach the flow quantities are defined by the following three conditions:

(1) Darcy's Law

$$\bar{q}_i = -\frac{K_i}{u_i} K(grad p_i - \bar{g}\rho'_i) \qquad (13)$$

(2) Continuity equation

$$-n\frac{\partial \rho'_i S_i}{\partial t} = div(\rho'_i \bar{q}_i) \qquad (14)$$

(3) Relationship between pressure p and density of fluid ρ'

$$\rho'_i = \rho'_i(p_i) \qquad (15)$$

where $i(= 1, 2)$ refers to fluid 1 and fluid 2, e.g., water and oil, respectively. In addition, it is assumed that the following conditions are satisfied:

$$S_1 + S_2 = 1 \qquad (16)$$

$$\rho_2 - \rho_1 = \rho_c(S_1) \qquad (17)$$

where: p_c = capillary pressure; $\bar{q}$ = fluid velocity; K = total intrinsic permeability; and K_i = relative permeability (a fraction of the total intrinsic permeability).

In the second approach, the problem of the surfactant-oil-fine mixture is treated as a fluid-solid slurry passing through a packed bed. A summary of the equations proposed in the field of slurry flow is given in Table 4. Most of these equations yield the velocity of the flowing fluid. Thus, in order to carry out the comparison between the proposed analysis and the existing theories, the velocity of the carrier fluid must be calculated. A schematic representation for the calculation methods, for both the existing and the proposed models, is given in Table 5.

In order to decide which approach is more suitable for the present work, the following points were taken into consideration.

1. in the first approach., it is required that the saturations be time-independent, in order to obtain meaningful values of relative permeability. Also, the velocity vectors of the continuous immiscible flowing fluids are required to be collinear. Furthermore, the momentum transfer at the fluid interface (Rose, 1969) is ignored. In other words, the first approach is based on the assumption that injected fluid pushed out and replaces the stationary fluid in the pore skeleton. This imposes certain limitations which cannot be easily met in the conditions chosen for the present experiments.
2. on the other hand, the flow of a surfactant-oil-clay mixture can be, practically speaking, described by the flow of a slurry through a porous medium. This problem has been thoroughly investigated in the second approach.

Accordingly, the present study adopts the second approach as a basis for the comparison with the proposed analytical model.

The results obtained from the various models, as well as those obtained from the proposed model, are compared to the experimental results in Figs. 3 to 5. It can be seen from these figures that:

1. Model No. 1 deviates considerably in terms of both the rate of change of θ_s and its values (Figs. 4 and 5), i.e., this method is not capable of predicting the rate of surfactant volumetric content change under the present conditions.
2. Model No. 3 is strongly dependent on curve accuracy, which is the first step in this approach (see Table 5). Thus the initial value of the volumetric surfactant content, and hence the subsequent (calculated) value, are dependent on the accuracy. In this case, a lower value of surfactant volumetric content than the experimental initial values was calculated. This resulted in generally lower values of the surfactant volumetric content, at all tested time intervals, especially in the first set of experiments (Figs. 3a and 3b). The discrepancy was less considerable for the second set of results (Figs. 4a and 4b). However, in the third set of results, this model could not predict the surfactant content, due to the lower accuracy of the curve fitting procedure. It was concluded from these results that model No.3 was not sufficiently reliable as a predictive tool since some of the assumptions on which it is based cannot always be fulfilled in practice.
3. Model No. 2 yields results that are in fair agreement with the present theoretical model, as well as with most experimental results (Figs. 3 to 5).

TABLE 4 A SUMMARY TABLE FOR MODELS #1, 2 and 3

Model

1. Yoon & Kunii (1970)

$$\Delta p = \left(\frac{150}{R_N} + 1.25\right) \cdot \frac{L\rho}{D_p\epsilon^3} \cdot (V_\epsilon - \frac{V_s}{1-\epsilon}\epsilon) \cdot (1-\epsilon)$$

2. Stemerding (1962)

$$\Delta p = \frac{\rho_s V_s}{1-\epsilon} + 4\tau L/D_p$$

+

Leung & Wiles (1976)

$$\frac{4\tau L}{D_p} = \frac{fLV_\epsilon^2}{2g_c D_p} + 2\frac{f_s L\rho_s V_s^2 D_p}{1-\epsilon}$$

$$f_s = \frac{0.005(1-\epsilon)}{V_s}$$

3. Ergun (1952)

$$t = \frac{u\alpha w}{g_c A^2(-\Delta p)}\left(\frac{V^2}{2} + BV\right)$$

where:

Δp = pressure drop; R_N = Reynolds no.,; L = sample length; ρ = fluid density; D_p = pore diameter; ϵ = porosity; V_ϵ = fluid average velocity; V_s = solid particle average velocity; ρ_s = solid particles density; f, f_s = fluid and solid friction factors; τ = shear stress at the wall; $V_{p,s} = V_{p,\epsilon}(1-\epsilon)$; $V_{p,\epsilon}$ = particle velocity in a fluid-solid system of porosity; V_{sl} = slip velocity of particles in a fluid-solid system; u = fluid viscosity; V = permeate volume; t = time.

TABLE 5 Calculation Procedures for Models # 1, 2 and 3

The proposed model	Models # 1 and 2	Model # 3
1. Calculate the volumetric Surfactant content $/\theta_s$ at different times.	1. Calculate V_s as $\frac{(\Delta\theta_s)\bullet Total\ volume}{\epsilon\bullet Area\bullet\Delta t}$	1. Plot reciprocal filter rate $\frac{dt}{dV}$ vs. volume of filtrate collected (V).
2. the porosity, $\epsilon, = \theta_s$ $\frac{volume\ of\ surfactant}{Total\ volume}$	2. Calculate the surfactant fluid velocity, V_s from models 1, 2 or 3.	2. The slope yields the value $\frac{\mu\alpha w}{g_c A_c^2 \Delta p}$
	3. Calculate the surfactant volumetric content as $(\theta_s)_t = (\theta_o)_{t_o} + (V_s\epsilon)\Delta t$ where $\Delta t = t - t_o$ $V_s = V_s$ Calculated from molds 1, 2 or 3.	3. The intercept = slope x B, thus calculate the value B.
	4. Compare with the surtant volumetric content calculated from the proposed model	4. At any time, t, calculate the total volume of filtrate (= the volume of surfactant, oil and clay)
		5. Calculate the volume of surfactant by substracting the oil and clay volumes from step #4.
		6. Compare with the volume of the surfactant as calculated from the present model.

CONCLUSIONS

A study of the existing theories in related fields shows that the various methods of solution of the problem of multiphase flow through packed beds needs refinement. In spite of the difficulties encountered in solving this problem, the proposed analytical model, as well as the methods presented by Stermerding (1962) and modified by Leung and Wiles (1976), are in fair agreement with the actual test results. On the other hand, the results from the method developed by Yoon and Kunii (1970) deviated considerably from the present data. In addition, the first approach is difficult to apply in such flow problem. Its is due to the fact that their model required a knowledge of the permeability values of the involved fluids: this is difficult, if not impossible, task to achieve at such low oil loads and experimental conditions.

ACKNOWLEDGEMENTS

This study was supported by a Grant-in-aid of Research by the National Science and Research Council of Canada (NSERC Grant No. A-882).

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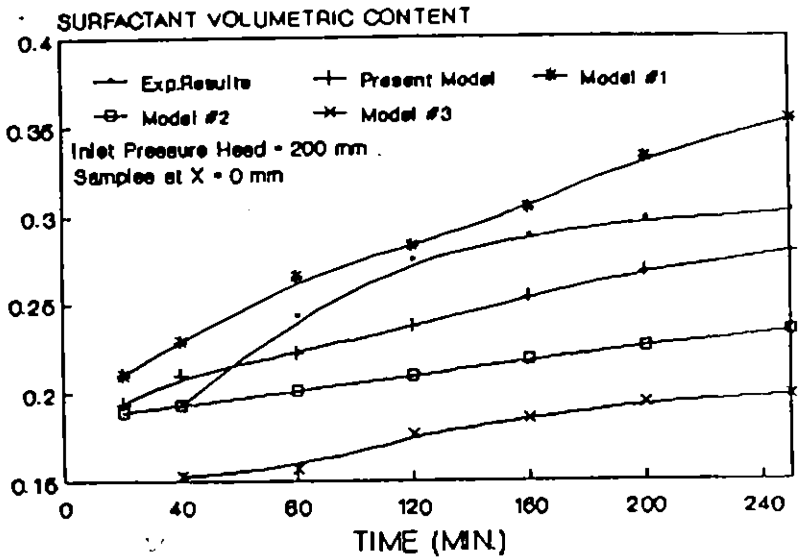

Fig. 3a Comparison of actual and predicted surfactant volumetric content with time (montmorillonite .05 wc/ws)

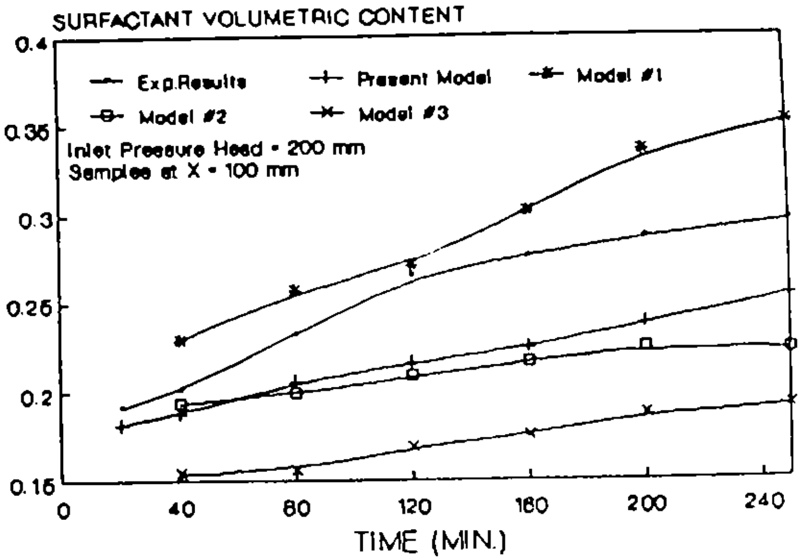

Fig. 3b Comparison of actual and predicted surfactant volumetric content with time (montmorillonite 0.05 wc/w

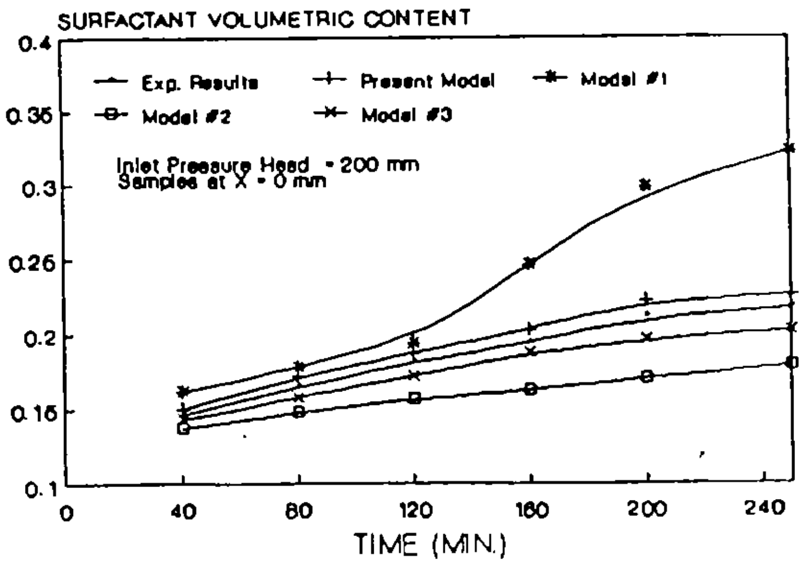

Fig. 4a Comparison of actual and predicted surfactant volumetric content with time (.15 · montmorillonite sample)

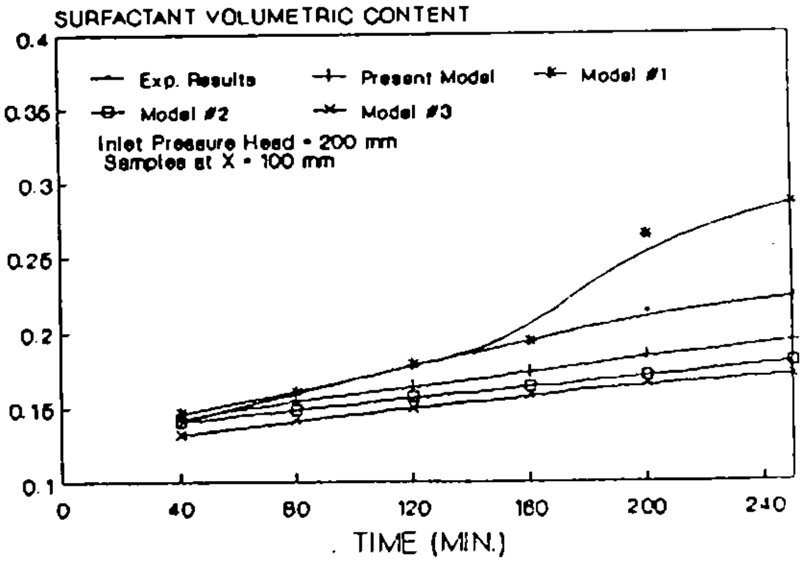

Fig. 4b Comparison of actual and predicted surfactant volumetric content with time (montmorillonite .15 wc/w

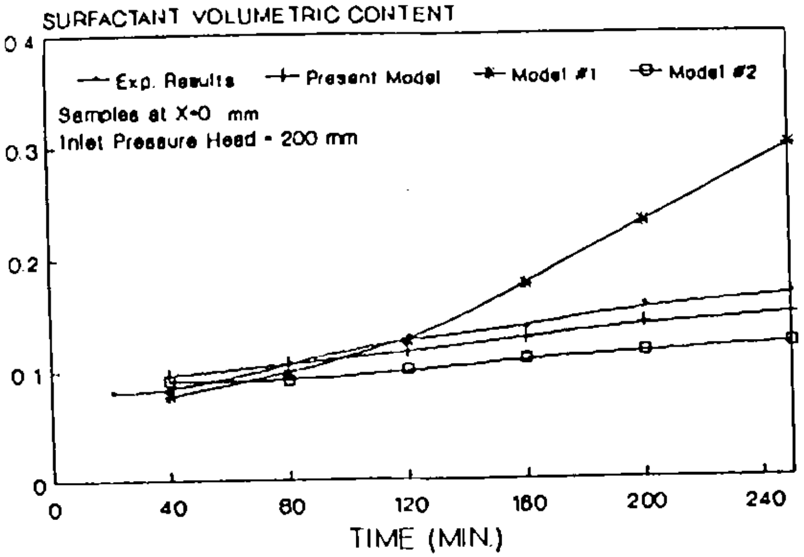

Fig. 5a Comparison of actual and predicted surfactant volumetric content with time (montmorillonite 0.25 wc/ws)

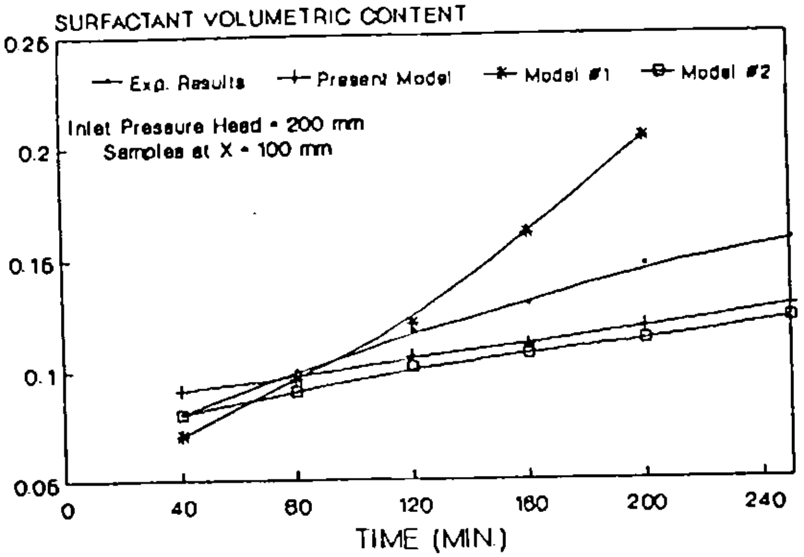

Fig. 5b Comparison of actual and predicted surfactant volumetric content with time (montmorillonite 0.25 wc/ws)

Proceedings, First Canadian Conference on Environmental Geotechnics
Eds. R.P. Chapuis and M. Aubertin. Montreal, Quebec.
May 1991.

EVALUATION OF CLAY COMPATIBILITY TO HEAVY METALS TRANSPORT AND CONTAINMENT: PERMEABILITY AND RETENTION

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ABSTRACT

The study confirms previous experience with kaolinite in regard to the relative insensitivity of the clay to pore-water chemistry changes. This can be explained in terms of the buffering capability of the material. The laboratory tests for permeability, using flexible and rigid wall test devices, indicate that little difference in k values exist between tests determined with either device in the case of tests with kaolinite. This is not the case for tests with a sand-bentonite (S/B) mixture. The amounts of Pb adsorbed/g of soil for both the kaolinite and S/B samples were greater as a result of permeability tests with the triaxial cell - in contrast to permeability tests with the consolidation cell. Partitioning modelling must respect the difference between the compact soil system and the soil suspension. The non-linear Freundlich model was used with success for both soil suspensions and compacted kaolinite samples. For the S/B test series, this model needs to be further investigated.

INTRODUCTION

The relatively low hydraulic conductivity and high adsorption capacity of clay soils make them prime candidates for constructed or engineered clay barriers in waste disposal landfills - as a means for control of migration of waste leachate to the groundwater. By and large, the compatibility of clay barriers with leachates is generally evaluated only in respect to changes in the coefficient of hydraulic permeability (k) of the clay barrier material, on the assumption that this reflects changes in the index properties and/or in the internal structure of the soil (Eklund 1985; Bowders et al. 1986). If little or no change is observed during permeability testing of the clay liner material, the liner may be considered satisfactory.

Common regulatory requirements generally establish an upper limiting value of k for engineered clay liners, with little or no attention for the many mechanisms of contaminant-clay interaction which produce attenuation of contaminants (ions) - particularly in regard to how the k value will respond to the interaction and accumulation processes. One can only conclude that if longterm considerations insofar as survivability and effectiveness of the engineered liner are important, then the parameter k must indeed accurately reflect the result of the various transport and accumulation processes occuring in the liner barrier.

Evidence from many studies show that hydraulic conductivity is one of the most difficult soil properties to measure accurately because of the numerous variables influencing it (e.g. *Lambe 1958; Mitchell et al. 1965; Dunn and Mitchell, 1984*). Its measurement in the laboratory can be very misleading if proper care is not taken. A review of the many experimental methods available can be found in Olson and Daniel (1981). Because of the need to obtain more precise and representative values for parameter k, greater emphasis on the type (system and method) of permeability testing is required - not only for the immediate assessment of k, but also for determination of the longterm leachate exposure effects on survivability and integrity of the test samples. Chemical assault of test samples can create conditions in the samples which would not be representative of actual field conditions - e.g. cracking, shrinkage, etc. Accordingly, the ability of the test system to anticipate and avoid such occurrences would constitute a prime requirement in the choice of system to be used.

This study presents the results of compatibility tests performed with two soils : a kaolinite clay and a mixture of sand and bentonite (90% sand/10% Na-bentonite). Two different types of permeameters - rigid wall and flexible wall - were used, with initial test conditions (except for sample size) maintained as similar as possible between the two types of test apparatus. Following leaching (permeation) of contaminant solutions under several contaminant concentrations, and under a range of gradients, each sample was analyzed to determine contaminant retention. The quantities of contaminants adsorbed by the compacted sample were compared to the results obtained from adsorption tests conducted on soil suspensions of the same clay soil material.

MATERIALS AND TEST METHODS

Soils

A kaolinitic clay, Hydrite PX, (Georgia Kaolin Co.) and a mixture of 90% fine to medium sand (Silica 40) and 10% Na-bentonite were tested. Typical sand/bentonite liners in Québec have bentonite contents in the 5-15% range (Chapuis, 1990). The geotechnical properties of both soils are presented in Table I and their chemical properties are shown in Table II. X-ray diffraction results indicated very little amorphous materials and no quartz, smectites or organics in the kaolinitic soil. Trace amounts of feldspar and mica were present in the bentonite obtained from Avonlea Minerals Ind. Ltd. of Saskatchewan.

Though kaolinite is rarely used in environmental geotechnology practice as a barrier material, it has been studied and reported on extensively, thereby permitting one to compare results obtained from this study with other reported results.

The cation exchange capacity (CEC) for both materials [illegible]rmined according to Black (1965; in Chhabra et al., 1975) [illegible] well within the range of reported values (Grim, 1968; Yong [illegible]al.,1986). Exchangeable H^+ was determined *following the* [illegible]thod described by Jackson (1967). The CEC of kaolinite is [illegible] dependent; in the acidic range the amounts of exchangeable [illegible] and *Al* are extremely important (Grim, 1968; Boland et [illegible] 1980; Duquette and Hendershot, 1987), as can be seen in [illegible]ble II. The concentration of the various cations was obtained [illegible] Atomic Absorption Spectrometry (AAS).

[illegible]rmeants

Two permeants were used: distilled water and *Pb* solutions at *different* concentrations. The distilled water was used to saturate samples prior to contaminant leaching. The contaminant solution, obtained by mixing distilled water and $Pb(NO_3)_2$, maintained its solution pH of 3.6 by the addition of nitric acid – thereby avoiding the occurrence of significant precipitation. The adsorption isotherms, (Figs. 9 and 11) were obtained using batch equilibrium tests on soil suspensions of the two materials, (i.e. equilibrium tests using different concentrations of lead in the suspensions). In the isotherm plots, the mass of lead adsorbed per gram of soil (q) at each equilibrium concentration (C), is obtained as an average of three values (triplicates).

The three concentrations of contaminant solution used are presented in the experimental scheme in Figure 1. The values of C1, C2 and C3, which depend on the nature of the material, were chosen to cover a wide range of concentrations.

Permeameters

The rigid wall permeameter used for the laboratory program was a modified consolidation cell, as shown in Fig. 2. The outer ring was made of stainless steel, with a Delrin inner ring and a Teflon top cap. All fittings were stainless steel. The flexible wall permeameter (Fig. 3), used a standard triaxial cell with a Teflon base and top cap and stainless steel fittings. The two membranes surrounding the sample were tested in a highly concentrated lead solution (5000 ppm), and found to be non-reactive.

The initial testing conditions were as similar as possible for both permeameters. These included: (1) type of soil; (2) type of contaminant; (3) initial water content; (4) dry density, and (5) the method and energy of compaction.

Experimental Procedures

To obtain better dispersivity of the soil particles, samples were statically compacted on the wet side of the Proctor curve, i.e. optimum moisture content (OMC) + 2 to 3% for kaolinite, and OMC + 2% for the sand-bentonite (S/B). The maximum dry densities and OMC of the materials are shown in Table I. Samples tested in the rigid-wall permeameter were compacted directly in the cell, whereas an 80 cm^3 compaction mold was used to prepare samples for the triaxial cell. The kaolinite samples were trimmed to a final height of 7 cm, whereas S/B samples were trimmed to 4 cm in order to decrease the long time test requirement.

Samples tested in the consolidation cell were saturated by flushing with 2 pore volumes of distilled water. Measurements taken during this procedure provided a reference permeability value of the soil to water. Kaolinite samples tested in the triaxial cell were saturated by the same procedure, except for one which was saturated using the back pressure technique. Sand-bentonite (S/B) samples tested in the triaxial cell were saturated by application of back pressure, except for one test sample which was used for comparative purposes.

After saturation, permeation from the bottom of the sample began under a constant pressure head (see test scheme in Figure 1). The hydraulic gradients used were : 25 , 50 and 100. The two burettes "in" and "out" were monitored to ensure that the inflow and outflow rates were equal. Swelling of the S/B samples in the consolidation cell was prevented through the use of an applied vertical pressure of 13.6 KPa. Preliminary experiments showed that at lower applied pressures, some swelling occurred. The same pressure was applied (as a chamber pressure) to the triaxial test samples.

After completion of the permeability test, each test sample was sliced into sections for chemical analysis. Triplicate samples were obtained from each slice. Each sample was washed two times with distilled water in order to recover the lead in the pore solution. The cations adsorbed on the clay surface (Pb, Na, Ca, Mg, K, and Al) were recovered by mixing the soil with a concentrated solution of ammonium acetate, and by exchange, these cations were released into solution and their concentrations were measured by the AAS.

RESULTS AND DISCUSSIONS

Permeability Tests

From the results of the kaolinite permeability tests shown in Table III, it is seen that the range of k values for all tests is between 1.3 x 10^{-7} and 6.5 x 10^{-7} cm/s, i.e. the k value of the material was not substantially affected by the introduction of the lead permeant. The same can also be said for virtually all the variations used – e.g. sample compaction energy, samples prepared with 1700 ppm lead solution instead of distilled water, backpressure and sample preparation by slurry consolidation. In addition, there appears to be no appreciable difference between the k values measured in the triaxial cell and the consolidation cell. This (lack of difference) is in agreement with observations made by other authors (e.g. Daniel et al. 1985, Bowders and Daniel, 1986, Peirce et al, 1986). Any variation between the actual values obtained in this study and those reported by other researchers may be attributed to several factors, including: type of kaolinite, the use of 0.01$NCaSO_4$ or tap water, instead of distilled water, as permeant; type of compaction; sample water content and density; and applied hydraulic gradient.

In the permeability test results for the S/B triaxial cell samples, shown in Figure 4, similar test conditions were used for for all four tests, i.e. the samples were saturated by application of back pressure in steps of 34 KPa (except TSB33NB which was saturated by flushing 2 pore volumes of distilled water). The concentration of lead in the permeant was approximately 2500 ppm, and the density of the samples varied between 19.2 - 19.5 KN/m^3. The effective confining pressure used for all tests was 13.6 KPa. Note that the position of the hydraulic gradient arrows shown in the Figure indicates the stage at which the gradient was applied. The initial gradient for the top two curves was 25, whereas the initial gradient for the bottom two curves was 100.

Sample TSB13(3), which was compacted in 3 lifts rather than the standard 5, as is the case for sample TSB13(5), produced an equilibrium permeability at i = 100, five to six times greater than one compacted with 5 lifts. Since the initial water contents of both samples are almost the same (17.3% and 18.1%), a seductive argument would be that the better oriented fabric obtained with the 5 compaction lifts was responsible for the lower k value. Whereas this reasoning might be sustained in a complete clay system, the mixture of sand and clay however presents complications in clay distribution that would need further fabric study before final definitive arguments can really be offered.

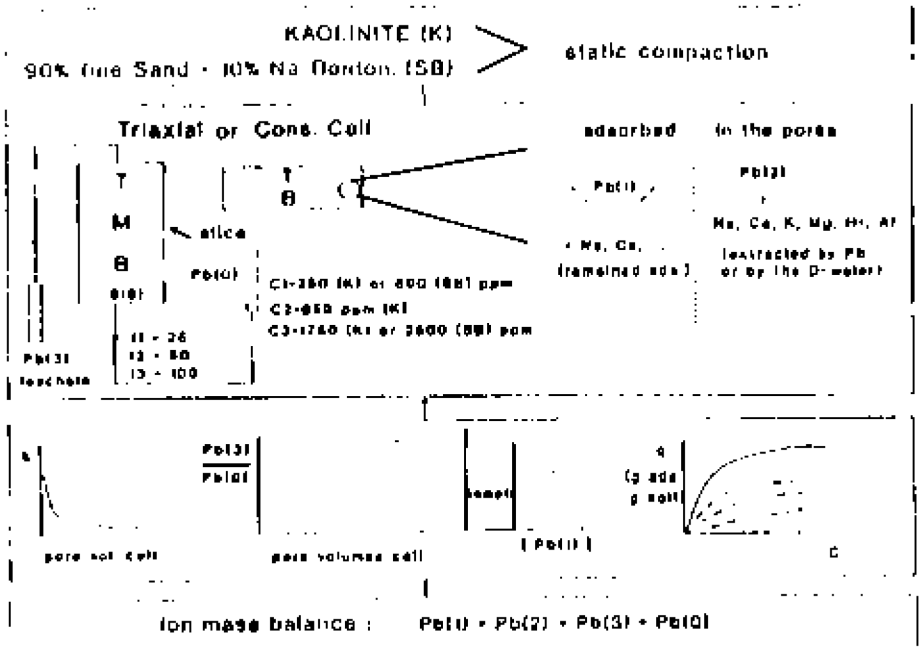


Figure 1 - Scheme of Experimental Procedure & Data Reduction

Table I - Geotechnical and Physical Properties of Kaolinite and a Sand/Na-Bentonite Mixture

PROPERTY	KAOLINITE	S/B
Liquid Limit (LL)	89.0 %	365% (bentonite)
LL (w/ 200 ppm Pb-soil)	70.5 %	326% (500 ppm)
LL (w/ 1600 ppm Pb-soil)	70.2 %	-
Plastic Limit (PL)	30.6 %	65% (bentonite)
Specific Gravity	2.6	2.68
Maximum dry density (kN/m3)(a)	13.1	17.2
Optimum moisture content (a)	32.0 %	16.0%
% clay size material	86.0 %	10.0%
Specific surface area (m2/g)	24.0	800 (Grim,1968)
Diam. of sand grains (mm)	-	0.06 - 0.8

(a) Standard Proctor

Table II - Chemical Properties of Kaolinite and a Sand/Na-Bentonite Mixture

PROPERTY	KAOLINITE	S/B
soil pH (a)	4.3 - 4.7	8.6 - 8.8
Cation Exchange Capac. (meq/100g)	7.00	109.0
Index exchang. cations (meq/100g):		
Na	0.31	63.7
K	0.46	1.9
Ca	0.16	35.7
Mg	0.02	7.6
exchangeable H+	2.86	-
exchangeable Al	3.23	-
% by mass of:		
Al_2O_3	0.48	N.A.
Fe_2O_3	0.00	
SiO_2	0.70	
total amorphous content	1.18	N.A.
organic content	0.00	0.00

(a) pH units, range for three tests

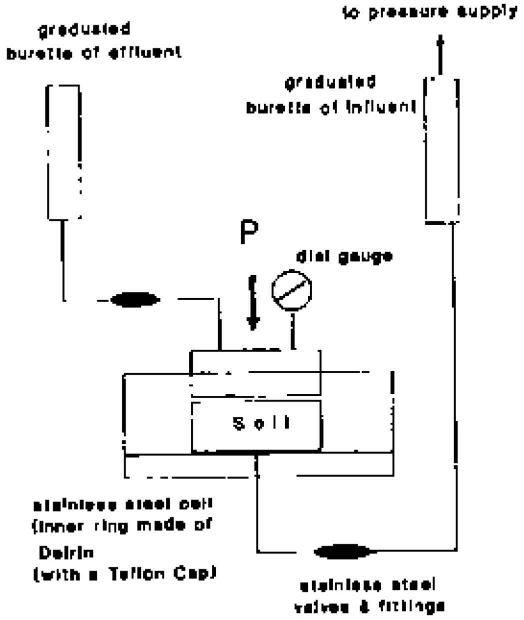


Figure 2 - Schematic Presentation of the Rigid-wall Permeameter (Consolidation Cell)

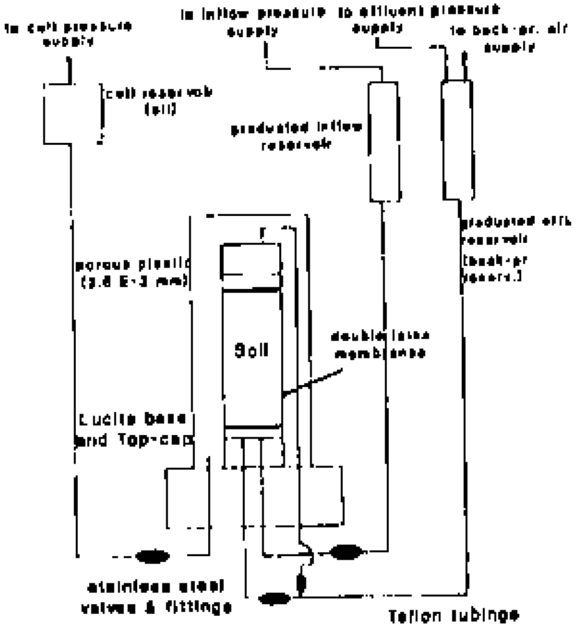


Figure 3 - Schematic Presentation of the Flexible-wall Permeameter (Triaxial Cell)

Table III - Hydraulic Conductivity Results for Kaolinite

Sample	γ (kN/m³)	ω (%) (OMC=32)	i	C (ppm)	k to H_2O (x 10^{-7}cm/s)	k to Pb (x 10^{-7}cm/s)
TK21	16.8	31.3	50	250	3.8	3.2
TK31	17.5	32.0	100	250	2.5	2.7
TK32	17.2	35.6	100	800	3.2	3.2
TK13	17.3	34.3	25	1700	4.2	2.0
TK23	17.5	31.0	50	1700	3.0	3.4
TK33b	17.2	35.2	100	1700	2.2	2.9
TGAMA	17.8	33.2	100	1700	6.5	5.0
TGAMB	16.2	33.0	100	1700	2.2	2.3
TK* (prepared with Pb)	17.3	35.5	100	1700	-	4.0
TSLURR (slurry consol.)	16.8	33.7	50 100	-	4.3 3.7	-
TKBP (back-pressure)	17.3	32.6	100	1700	2.4	2.4
CK21	17.3	**	50	250	3.3	3.7
CK31	17.2	**	100	250	1.3	1.7
CK22	17.5	**	50	800	2.7	2.9
CK32	17.3	**	100	800	1.8	2.7
CK13	17.4	33.3	25	1700	2.1	3.2
CK23	17.3	**	50	1700	3.5	3.2
CK33	17.4	**	100	1700	1.9	2.4
CK* (prepared with Pb)	17.4	**	100	1700	-	3.6

** No measurement was taken. Assumed to be 32 %

Fig. 4 shows that the equilibrium permeabilities of the other two samples (TSB33BP and TSB33NB), which had initial gradients four times higher ($i = 100$) than the other two samples described heretofore, were two orders of magnitude lower (than the other two samples). Two possible reasons exist for this significant reduction in permeability: (1) partial pore blockage (pore constriction) due to seepage forces that displaces particles and/or group of particles, and (2) precipitation of lead as oxides due to the high pH of the mixture (pH between 8.6 and 8.8). The results indicate that the drop in k values occurred very rapidly over the first pore volume of Pb-solution, with a less rapid but albeit fast drop in k values over the next 3 pore-volume permeation. This contrasts quite dramatically with the results demonstrated by the two tests with initial gradients of 25. From these initial observations, one can be persuaded that detachment of particles could occur under the initially "high" gradient of 100, leading thereby to pore constriction (partial blockage). Observations of effluent discharge showed that the effluent from sample TSB33BP (lowest curve in Fig.4) became cloudy as the permeability dropped from 4.0 x 10^{-7} to the equilibrium value of about 1 x 10^{-9} cm/s. It is useful to note that an increase in the hydraulic gradient once the coefficient of hydraulic permeability had stabilized did not lead to a significant change in the k values of test samples TSB13(3) and TSB13(5) *(two upper curves)*.

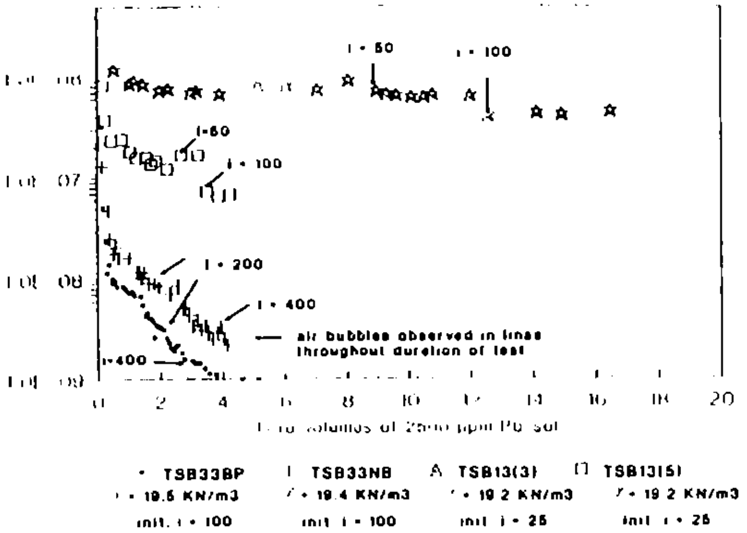
The effect of possible detachment of particles under initial high gradients raises some very interesting questions concerning the application of high hydraulic gradients to accelerate required-time procedures in permeability tests (with clay soils). The implications arising from the results shown in Fig.4 are very severe. Judgements based solely on the coefficient of hydraulic permeability k obtained with samples TSB33BP or TSB33NB where high initial gradients were used (bottom two curves, k = approx. 1 x 10^{-9} cm/s), would indicate that the S/B mixture tested easily qualifies as an engineered barrier material. However, if the results obtained with samples TSB13(3) and TSB13(5) where low initial gradients were used (top two curves in Fig.4, k = 1 to 6 x 10^{-7} cm/s) are considered, the judgement would be highly tempered, and would perhaps lead to rejection of the material for use as an engineered barrier material.

Limited tests on the influence of a lower concentration of Pb (500 ppm Pb in the permeant instead of 2500ppm) on the coefficient of hydraulic permeability k, using the low hydraulic gradient of 25, produced k values in the order of 10^{-8} cm/s, - at least one order or magnitude lower than the k values obtained in the test series reported in Fig. 4 for the same initial gradient of 25. The lower k value obtained with the lower Pb concentration is not unexpected in light of the diffuse double-layer (DDL) model. The model calculations (Yong and Warkentin, 1975; Greenland and Hayes, 1978) show that the DDL thickness is inversely proportional to the valence and to the square root of the concentration (of the participating ions). The schematic diagram in Fig. 5 shows the reduction in the DDL due to the replacement of Na by Pb. Interpenetration of the DDL, shown in the lower diagram will produce greater or lesser midplane (electric) potentials, depending on the species of dissolved solute (Pb, in this case) and concentration of the dissolved solutes. The magnitude of the midplane potential is indicative of the resistance to flow through the pore space established between the two interacting particles (and DDL). A description of the transport process through this type of system can be found in Yong and Samani, (1988).

Fig. 6 presents the results of permeability tests performed with the consolidation cell. All samples were compacted to the "same" density (between 19.65 and 19.85 KN/m^3) and saturated by flushing with a *minimum of 2 pore volumes of distilled* water. The effect of Pb concentration can be evaluated by examining the results obtained for samples CSB33 and CSB31 where the Pb concentrations were 2500 ppm and 500 ppm respectively for the two samples - with same initial hydraulic gradient of 100. Whereas the sample with the lower Pb concentration (CSB31) started with an initial k value about one order of magnitude higher than the sample with the higher Pb concentration, it is noted that both samples virtually developed the same k value after permeation with 3 pore volumes of their own respective Pb permeant.

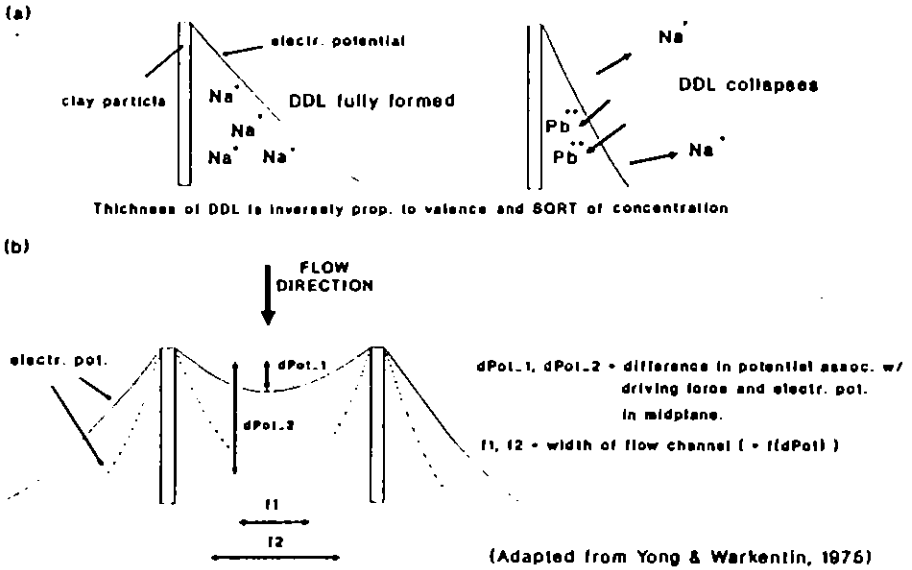
To further explore the "same k value" phenomenon demonstrated by the two samples, the gradient for the sample with the higher Pb concentration (CSB33) was increased from 100 to 400 after permeation with 5 pore volumes of its Pb permeant, and (the gradient) subsequently decreased from 400 to 200 after about 6.5 pore-volume permeation. In concert with this, the sample with the lower Pb concentraton (CSB31) stayed with its common initial gradient of 100 until after 6.5 pore-volume permeation, at which time, the gradient was also raised to 200, to match the gradient of the higher Pb concentration sample test. The end result of this somewhat tortuous gradient change procedure is to determine whether hydraulic gradient path dependency is a sufficiently important factor in development of the k value. As can be seen from the graphical results shown in Fig. 6, the two curves remain virtually coincident - except for the higher Pb concentration sample after 7 pore volumes of permeation when an apparent leak occured in the soil system.

To provide more information, a sample with the same high Pb concentration of 2500 ppm was tested in the consolidation cell under a lower hydraulic gradient (sample CSB13, $i = 25$). As seen in Fig. 6, the initial k value of the sample lay inbetween the previous two samples. However, as was the case of the previous two samples, the k value of the low gradient sample dropped to a k value slightly lower than (or nearly equal to) the equilibrium k value obtained by the other two samples after permeation by 3 pore volumes of its own Pb permeant. To further complicate the problem of test results evaluation, the hydraulic gradient of this sample was increased from 25 to 100 after the 3 pore-volume permeation.



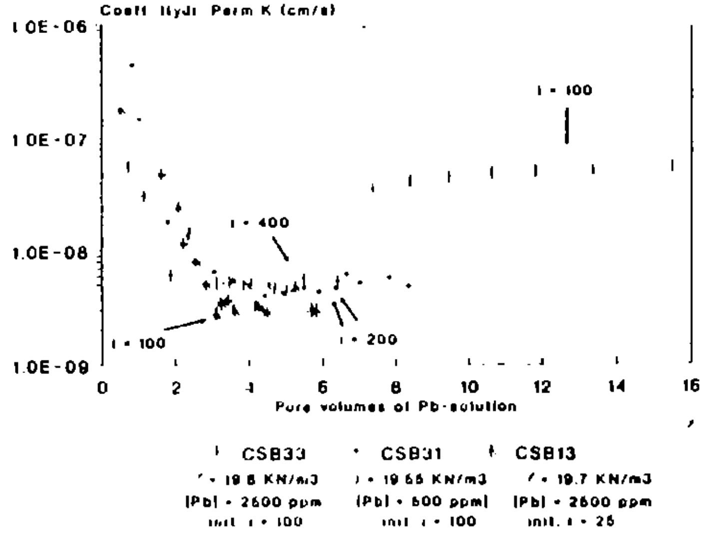
Obs. Back Press. applied in steps of 34 KPa (except TSB33NB which was saturated by flushing 2 pore vol. of distilled water)

Figure 4 - Variation of the Coeff. of Hydraulic Permeability of S/B Samples Tested in the Triaxial Cell



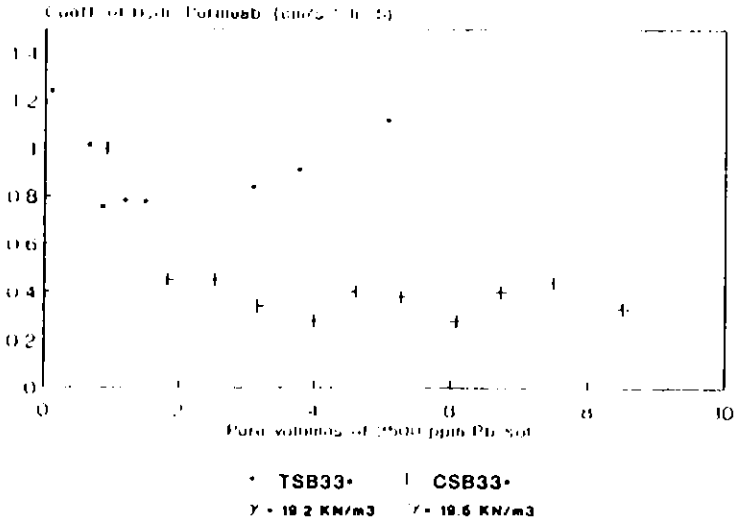
(Adapted from Yong & Warkentin, 1975)

Figure 5 - Schematic presentation of : (a) DDL collapse due to cation exchange; (b) the influence of changes in surface potential due to DDL collapse in flow characteristics.



Obs.: - saturation ended after 2 p.v. of distilled water had been flushed
- initial w% = OMC + 2%

Figure 6 - Variation of the Coeff. of Hydraulic Permeability of S/B Samples Tested in the Consolidation Cell



Obs.: - Sample TSB33• saturated by application of back pressure
- hydraulic gradient i = 100 throughout tests

Figure 7 - Variation of the Coeff. of Hydraulic Permeability of Precontaminated S/B Samples (Triaxial and Consolidation Cells)

It is not possible to provide a conclusive set of arguments concerning the performance of the sand-bentonite mixture in the consolidation cell – for determination of the "permeability" of the sample. Since the tests were designed as "exploratory" tests as opposed to property-determination tests, replicate and systematic parametric variation testing was not conducted. All that can be said about the test results at this stage is that the equilibrium value for the sand-bentonite mixture tested appear to asymptote to an equilibrium k value of between 2 x 10^{-9} and 6 x 10^{-9} cm/s – within the range found for the tests with the flexible wall test system for the high hydraulic gradient tests.

The effect of precontamination was also evaluated and the results of two tests are presented in Figure 7. Rather than mixing dry material with distilled water, a Pb solution (2500 ppm) was utilized. Samples were compacted to virtually the same density and initial water content as the "non-precontaminated" samples, and a hydraulic gradient of 100 was applied to the Pb concentrated permeant (Pb concentration of 2500 ppm). The results (Fig. 7) show that the k vlaues of the precontaminated samples may be several orders of magnitude greater than all other tests conducted on "non-precontaminated" sand bentonite mixtures. The apparent difference of less than one order of magnitude in k values between the triaxial and consolidation test results after 2 pore-vlume permeation for the precontaminanted samples shown in Fig., 7 is not considered significant – considering the size and sample preparation (Pb contamination) and saturation procedure differences between the two types of test system.

Chemical Analyses

The adsorption isotherm curves for the kaolinite, together with the Pb adsorption results for both the triaxial and consolidation permeability tests, shown in Figs. 8 and 9 respectively, can be "fitted" by power curve fitting using the Freundlich model (Greenland and Hayes, 1978). As noted on the diagrams, the Pb adsorption results for the triaxial permeability tests are obtained from the respective slices of the samples as designated (top, middle and bottom). In the case of the consolidation (permeability) tests, only the top and bottom slices are obtained – in view of the initial small sample thickness. The relatively high correlation coefficients (R^2) obtained indicate the capability of the Freundlich model to model the adsorption characteristics of the kaolinite used in this study.

As is obvious from Figs. 8 and 9, the quantities of Pb adsorbed by the compacted kaolinite samples at various Pb input permeant concentrations are much less than quantities which characterize the adsorption isotherm (Pb adsorbed by the corresponding kaolinite soil suspensions), even in the case where almost total breakthrough ($C/Co \simeq 0.9$) was reached after several pore volumes had leached through a sample. The differences in Pb adsorption which are more significant at lower concentrations are due primarily to availability of exposed clay particle surfaces. This reasoning argues that with soil suspensions, where all dispersed clay particles can contact the dissolved contaminant, accumulation processes are at their optimum. In the case of compacted clays however, aggregate and cluster formation will considerably decrease effective specific surface area and will also severely hamper cation exchange, because the permeant fluid will (preferably) flow around rather than through these clay structures (Olsen, 1962). The greater flow length, represented by the triaxial samples, appears to be responsible for the higher degree of Pb adsorbed by the samples tested in the flexible wall permeameter (in g_{ads}/g_{soil}) than those tested in the rigid wall (consolidation) cell.

Whereas Figs. 8 and 9 show that the amounts adsorbed by each section do not vary appreciably between samples, in spite of differences in effluent concentration at termination of test, hydraulic gradient, density and other parameters investigated, it is noted that the differences between Pb adsorbed in the top and bottom sections are significant, – reflective of the accumulation processes occurring within the sample and the dependency on flow path. Whilst the buffering capacity of kaolinite is relatively low (see Yong et al, 1991), the present test results show that Pb retention by the kaolinite samples does occur – as witness the observations showing that the metal was identified in the effluent only after 2 to 3 pore volumes had been collected during tests using the highest concentration permeant. From the chemical analyses, a simple set of calculations indicate that the retardation factor for the samples ranged from 4 to 5.

The ion mass balances calculated after each test for the kaolinite samples (amount of Pb introduced in the system compared to the sum of amounts found in the adsorbed form, in the pore fluid, and in the leachate collected) show at least a 90% accountability (of the ions). The differences in output/input can be attributed to experimental errors.

The Freundlich model was also found to fit the adsorption isotherm for the sand-bentonite (S/B) mixture – (Fig. 10). With the limited results available to-date, it is seen from the Figure that at low input Pb concentrations, very little Pb adsorption occurs. This contrasts dramatically with Pb adsorbed at the high input concentrations. The similar sets of reasoning can be postulated for S/B samples (as in the case of the kaolinite) for the differences between the quantities of Pb adsorbed in the samples and the soil suspension. Further investigation would be necessary to establish a correlation between the amounts of lead adsorbed by compacted samples and soil suspensions using the "effective" exposed surface area as input.

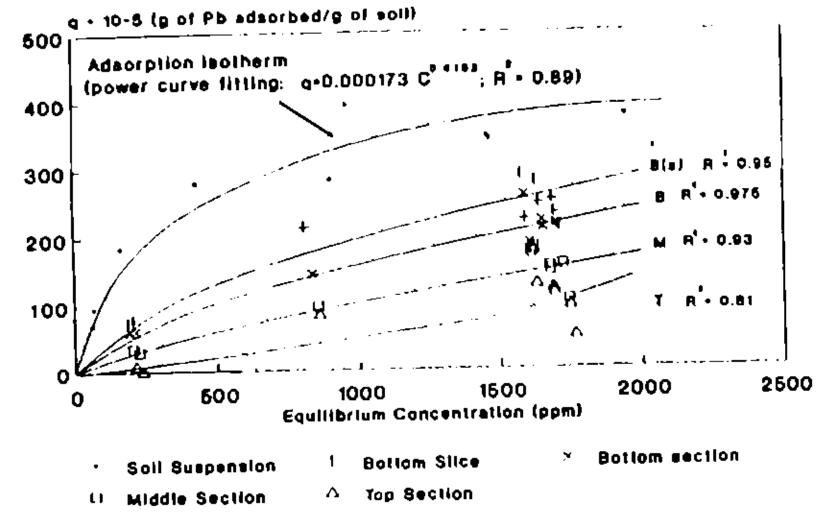
Since, in general, adsorption by compact clays will probably never equal the total adsorption by the same soil suspensions, the specification of an appropriate equation to represent the partitioning of (the amounts of) contaminant associated with the solid and liquid phases becomes particularly important. The uninformed use of the adsorption isotherm in all cases may lead to an improper evaluation of the adsorption capability of the compact soil, particularly when only the linear portion sometimes observed in the lower concentration range is considered. The appropriate model to describe the adsorption curve is also of considerable concern. In this study, the nonlinear Freundlich model was used with success for both soil suspensions and compacted kaolinite samples. However, the indications in regard to the compact sand-bentonite mixtures suggest that considerably more testing is needed to support the use of this model.

The breakthrough results shown in Fig. 11 indicate the high retention capability, e.g. detection of Pb in the effluent occurred only after 4 to 5 pore volumes had been collected. The breakthrough point ($C/C_0 = 0.5$) occurred at 7 to 8 pore volumes.

CONCLUDING REMARKS

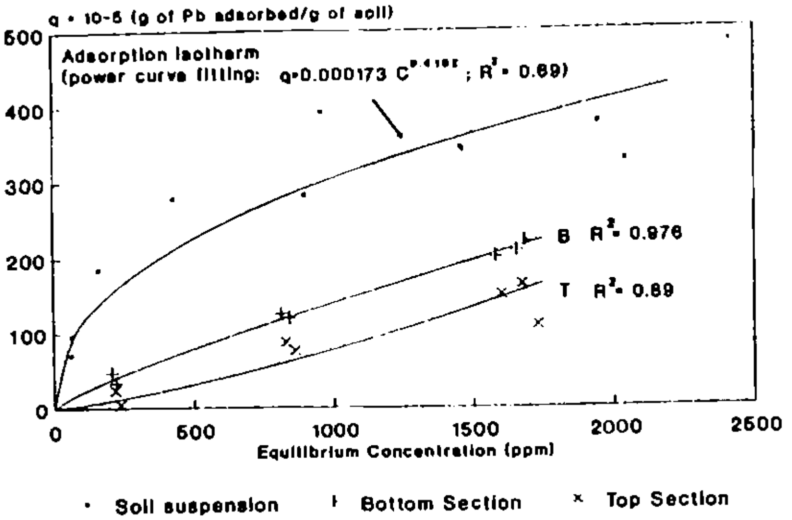
The results from the study agree with previous experience with kaolinite in regard to the relative insensitivity of the clay to pore-water chemistry changes. This is not totally unexpected, and can be explained in terms of the buffering capability of the material (Yong et al, 1991). As seen in the laboratory tests for permeability, using flexible and rigid wall test devices, little differences exist between the k values determined with either device. This has been confirmed by other researchers, and testifies to the relatively "inactive" nature of the material – i.e. "cracking" is not as severe a problem as in other clays with more pronounced surface active forces.

In Pb retention analyses, it is seen that the amounts of Pb adsorbed/g of soil for both the kaolinite and S/B samples were greater as a result of tests with the triaxial cell – in contrast to permeability tests with the consolidation cell. It would appear that path length and exposure time would have some decided effect on adsorption efficiency. This would need to be confirmed



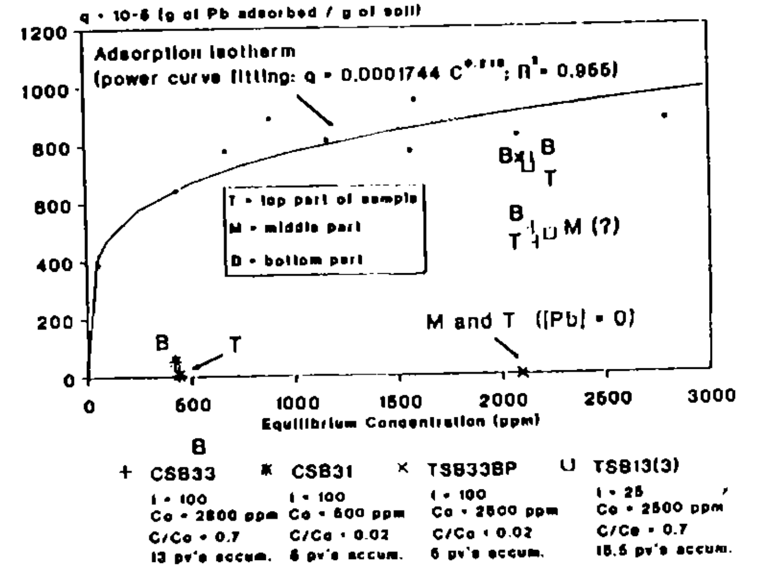
Obs.: - the Freundlich Model was used for all curves
- each point represents the average of three values

Figure 8 - Adsorption of Lead by Kaolinite. Tests Executed in the Triaxial Cell



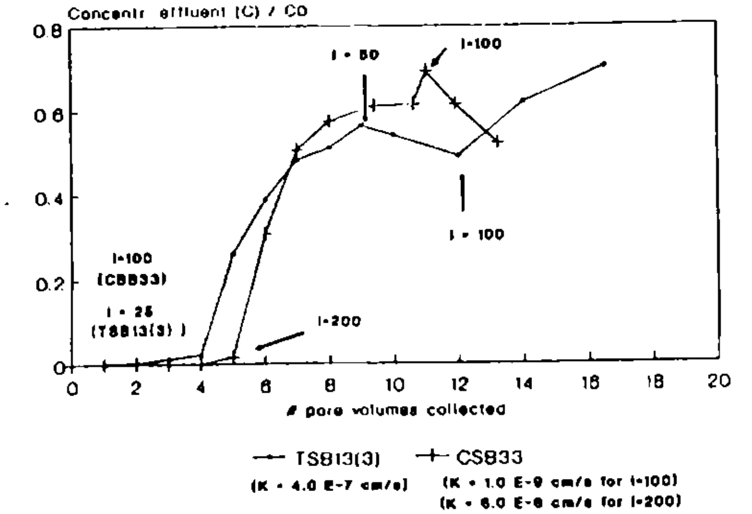
Obs.: - the Freundlich model was used for all curves
- each point represents the average of three values

Figure 9 - Adsorption of Lead by Kaolinite. Tests Executed in the Consolidation Cell



Obs.: - the Freundlich model was used for the Ads. Isot.
- each point represents the avg of 3 values (except M of sple TSB13(3))
- Co is the initial concentration of Pb in the permeant

Figure 10 - Adsorption of Lead by S/B



Obs.: - Co is the initial concentration of Pb in the permeant (2500 ppm)
- Sample TSB13(3) saturated by application of back-pressure
- Sample CSB33 saturated by flushing 2 pv of distilled water (values not plotted)

Figure 11 - Breakthrough Curves for Two Tests with Two S/B Samples

with supporting calculations and measurements of the effective exposed surface areas between the triaxial and consolidation samples.

The selection of the relationship "needed" to represent the partitioning of the amounts of contaminants associated with the solid and liquid phases at various contaminant concentrations appears to be particularly important. Partitioning modelling must respect the difference between the compact soil system and the soil suspension. The unenlightened use of "textbook" values or models for all cases may lead to an improper evaluation of the adsorption capability of the compact soil, particularly when only the linear portion, sometimes observed in the lower concentration range, is considered. In addition, the choice of a proper model to describe the adsorption curve is also of considerable concern. In this study, the non-linear Freundlich model was used with success for both soil suspensions and compacted kaolinite samples. In regard to the compact sand-bentonite mixtures, more results are needed to test use of this model.

ACKNOWLEDGEMENTS

This study was conducted under a Grant in aid of research from the Natural Sciences and Engineering Council (NSERC), Grant No. A-882.

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