Bergvesenet

Postboks 3021, N-7441 Trondheim

Rapportarkivet

Bergvesenet rapport nr	Intern Journal nr	Internt arkiv nr	Rapport lokalisering	Gradering
7264				
Kommer fra ..arkiv	**Ekstern rapport nr**	**Oversendt fra**	**Fortrolig pga**	**Fortrolig fra dato:**
		Nordlandske		

Tittel

SFT-prosjekt "Utprøving av tildekning av gruveavfall" - delrapport 1

Forfatter	Dato	År	Bedrift (Oppdragsgiver og/eller oppdragstaker)
Håbjørg Atle, Robertsen Andrew	04.06	1988	SFT

Kommune	Fylke	Bergdistrikt	1: 50 000 kartblad	1: 250 000 kartblad

Fagområde	Dokument type	Forekomster (forekomst, gruvefelt, undersøkelsesfelt)
Miljø		
Råstoffgruppe	**Råstofftype**	
Malm/metall	Zn, Cu, Pb, Py	

Sammendrag, innholdsfortegnelse eller innholdsbeskrivelse

SFT har gitt en rådgivningsgruppe oppdrag å utprøve ulike tildekningsmåter av gruveavfall for å begrense utlekking av tungmetaller.

Rapport nr. 1 gir en sammenstilling av erfaringer i Norge med revegetering av gruveområder og andre vegetasjonsraserte områder. Rapporten gir i tillegg oppsummering av tekniske tildekningsløsninger benyttet i forskjellige land. Temaene er beskreet i 2 artikler.

- Revegetering av gruveområder (Bjørgo)
- Cover technology for acid mine drainage abatement, literature survey (Robertsen)

BV 7264

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Deres ref. | Vår ref. (bes oppgitt ved svar) | Dato

88/07006-1 KaK/AL 27 JULI 1988
612. mp. IV

DELRAPPORT I FOR SFT-PROSJEKTET
"UTPRØVING AV TILDEKKING AV GRUVEAVFALL".

Herved oversendes delrapport nr. 1 for prosjektet "utprøving av tildekking av gruveavfall" til Deres orientering.

Rapporten er utarbeidet av A/S Miljøplan i samarbeid med Steffen, Robertson and Kirsten, SEFO og NOTEBY.

Oppdragsgiver for prosjektet er Statens forurensningstilsyn.

Med hilsen

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STATENS FORURENSNINGSTILSYN

UTPRØVING AV TILDEKKING AV GRUVEAVFALL

RAPPORT NR. 1

REVEGETERING AV GRUVEOMRÅDER

COVER TECHNOLOGY FOR
ACID MINE DRAINAGE ABATEMENT

JUNI 1988

STEFFEN ROBERTSON & KIRSTEN
Geotechnical, Mining & Environmental Engineers

SEFO
SENTER FOR FORSKNINGSOPPDRAG

NOTEBY
NORSK TEKNISK BYGGEKONTROLL A/S

A/S MILJØPLAN
RÅDGIVENDE INGENIØRER M.R.I.F.

FORORD

Statens forurensningstilsyn har gitt en rådgivningsgruppe i oppdrag å utprøve ulike tildekkingsmåter av gruveavfall for å begrense utlekking av tungmetaller til nedenforliggende vassdrag.

Rådgivningsgruppen består av:

- STEFFEN, ROBERTSON & KIRSTEN, SRK, Vancouver, Canada
- Senter for oppdragsforskning, SEFO, Norges landbruksvitenskapelig forskningsråd, Ås
- Norsk Teknisk Byggekontroll, NOTEBY, Oslo
- MILJØPLAN, Sandvika

Miljøplan koordinerer rådgivningsgruppens arbeid.

Rapport nr. 1 gir en sammenstilling av erfaringer i Norge med revegetering av gruveområder og andre vegetasjonsraserte områder. Dertil gir rapporten en oppsummering av tekniske tildekkingsløsninger som er benyttet i de senere år i forskjellige land. Fordeler og ulemper ved ulike løsninger diskuteres med hensyn til materialvalg, praktisk utførelse, effektivitet i å redusere avrenning, holdbarhet over tid m.m.

Paul Liseth

Paul Liseth
Prosjektleder

REVEGETERING AV GRUVEOMRADER

av

PROFESSOR DR. ATLE HÅBJØRG

Institutt for hagebruk, NLH

Revegetering av gruveområder

Innledning.

Globale og lokale forurensninger og avfall fra ulike kilder kan føre til større eller mindre skader på vegetasjon, jordsmonn og vatnflora/-fauna, enten i primær form eller som reaksjonsprodukter. Spesielt utsatt er områder nær industrisentra, gruveområder, fyllplasser og sterkt trafikkerte veger. På disse områder er det ofte en total ødeleggelse av økosystemene, dvs. flora og fauna på land og i vatn. Videre vil det ved store tekniske inngrep i landskapet - vasskraftutbygging, oljeinstallasjoner, vegbygging osv. - oppstå store vegetasjonsraserte sår i landskapet med omfattende erosjonsproblemer.

Institutt for hagebruk har gjennom 20 år hatt forsøksvirksomhet på området revegetering og landskapspleie i forbindelse med tekniske inngrep i landskapet. Innsatsen har spent over et vidt problemspekter.

- revegetering av vegetasjonsraserte områder i forbindelse med oljevirksomhet, vasskraft og vegbygging m.m.
- revegetering av gruveområder og sterkt forurensete industriområder - med oppbygging av sperresjikt, dyrkingsmedium og vegetasjonsetablering. A/S Sydvaranger, Rana Gruver, Løkken Verk, Kjøli Gruver, Knaben Gruver, Titania Gruver, A/S Borregaard, Falconbridge Nikkelverk, DN Zinkkompani, Norsk Hydro Aluminium mf.
- naturlig vegetasjonsinnvandring på tungmetallforurensete områder og utvikling av tungmetallresistente økotyper i planter.

Prosjektene har dels vært et kortvarig samarbeid med de enkelte bedrifter og dels et mer langsiktig samarbeid med BVLI. Resultatene er publisert i vitenskapelige journaler og i 6-8 diplomoppgaver ved NLH.

Problemene med revegetering av tungmetallforurensete steder i Norge er gjerne knyttet til de klimatiske og edafiske forhold på stedet.

Klimaforhold

Norges beliggenhet, som det nordligste i Europa (58-71° N) med 1/3 av landarealet og 1/10 av befolkningen bosatt nord for den arktiske sirkel, spenner over et enormt klimaspekter (Figur 1).

Lengst nord er vekstsesongen maksimalt 2 måneder og det er praktisk talt kontinuerlig sollys hele perioden. I sørvest er vekstsesongen gjerne 7-8 måneder og daglengden er maksimalt 18 timer midtsommers. Videre har vi et maritimt klima i vest hvor maksimal temperatur-differanse mellom sommer og vinter er ca 35° C (+ 25- ÷ 10°) og med årsnedbør opp imot 3000 mm. Tilsvarende temperaturdifferanse i de mest kontinentale strøk i fjellområder i øst er bortimot 85° C (+ 33 til ÷ 50) og hvor årsnedbør kan gå ned mot 300 mm.

Den arvelige variasjon innen plantearter er meget stor og gjennom generasjoner vil plantene tilpasse seg forholdene på stedet. Det vil bl.annet bety at lokale økotyper praktisk talt alltid vil ha større evne til å overleve enn "innførte" planteslag. Graden av vellykkethet i et revegeteringsprosjekt vil derfor alltid være betinget av hardføre økotyper.

Tabell 1. Vinterherdighet (% overlevende) hos ulike økotyper av bjørk (Betula pubescens).

	Dyrkingssted		
Økotype	Oslo	Trondheim	Alta
Dansk	85	45	0
Trondheim	95	100	35
Alta	55	90	100

Edafiske forhold.

Bergverkdrift er blant de eldste industripregete virksomheter vi kjenner, og går helt tilbake til det 12. århundre. I en oversikt fra 1925 er det notert 1414 gruver i Norge. 51 av disse er pr. idag antatt å medføre betydelige forurensningsproblemer (NIVA 1984).

De fleste av disse bygger på forekomster av kismineraler, dvs. svovelsalter med Fe, Cu, Zn, Pb og Ni i varierende forhold.

Jordanalyser fra avgangsmasser gir eksempler på følgende kjemisk innhold (mg pr. l) (HÅBJØRG 1988)

	pH	P	K	Ca	Fe	Cu	Zn	Al
Kjøli	2,8	0,5	5	50	5090	106	13	95
Fredrik IV.(s)	7,4	0,5	46	2170	1370	4260	108	108
Kvikne	2,8	0,6	0,1	22	1650	35	14	16
Vigsnes	3,5	0,7	9	104	3160	34	16	77
Gravdal	3,2	1,9	7	63	1862	24	8	27

Giftvirkningen i forbindelse med eldre gruveanlegg er registrert på alle disse steder. Videre er det registrert store skader på vegetasjonsdekket bl. annet på Svalbard der avgangsmasser er brukt som fyllmasse ved utbygginger. Dessuten er det store skader på vegetasjonen i og ved vassdragene nedenfor veltene grunnet sigevatn fra disse.

Eksempler på innhold i sigevatn fra kisgruver. (HÅBJØRG 1988)

	pH	Fe	Cu	Zn	Kond. MS/M
Kjøli	3,0	35	3,5	0,15	63,6
Kvikne	3,5	7,4	1,2		19,8
Gravdal	3,6	2,8	0,2	0,3	20,1
Killingdal	2,5	692	62	273	396

Partikkelstørrelsen i veltene er vanligvis fin nok for kapillær ledningsevne. Dette sammen med relativt liten fordampning av vatn fra massene gjør at vatntilgangen for eventuell vegetasjon på veltene stort sett er tilfredsstillende.

Vegetasjonsetablering.

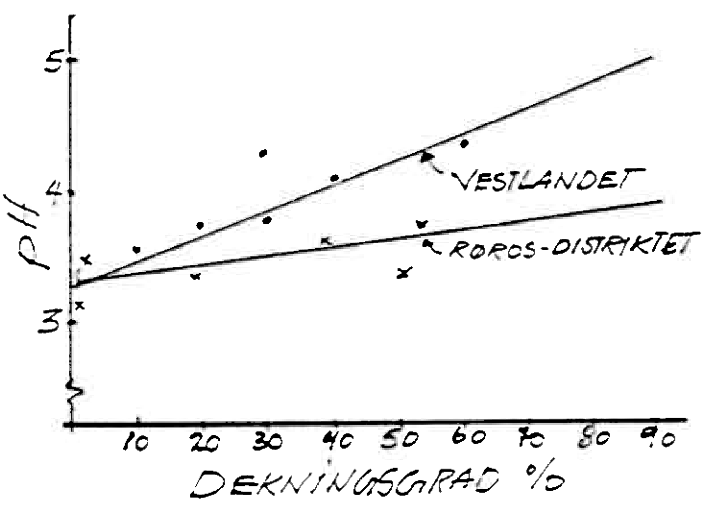
Feltforsøk og forsøk under kontrollerte forhold har vist at plantevekst er praktisk talt umulig i masser med pH under 3,5 (Figur 1). I våre forsøk er det gjort få detaljstudier for å klarlegge den egentlige årsak til vekstproblemene på slik jord, NRIAGU (1978) antyder at det kan ha følgende årsaker

- økt løselighet av AI- og Mn-ioner med fare for giftvirkning på plantene

- redusert tilgjengelighet av essensielle plantenæringsstoffer særlig P, K, samt Ca og Mg.

Fra jordanalysene som er oppgitt tidligere, går det imidlertid fram at sammenhengen mellom plantevekst og forurensning nok er betydelig mer nyansert. Blant annet har det vist at enkelte plantearter og økotyper kan vokse tilfredsstillende på visse gruvemasser, mens de går ut om de dyrkes på andre med tilsynelatende minst like ekstreme forhold. For andre arter/økotyper kan situasjonen være omvendt. Toleranseegenskapene hos arter/økotyper er altså med få unntak spesifikke. Multitoleranse kan utvikles i de tilfeller plantene utsettes for et allsidig seleksjonspress (Bradshaw 1975, Håbjørg 1979). Tillaging av slike typer er imidlertid en såpass omstendig og arbeidssom prosess at det neppe er gjennomførbart for norske forhold hvor variable klimaforhold ytterligere kompliserer situasjonen

Figur 1. Virkning av pH på vegetasjonsdekket på velter i Rørosdistriktet og på Vestlandet.



Revegetering av sterkt forurensete områder kan i praksis skje ved hjelp av

1 - sperresjikt og/eller dekningsmasser
2 - kalking/gjødsling direkte eller i kombinasjon med sperresjikt/dekningsmasser
3 - stimulere naturlig vegetasjonsvandring og/eller såing/planting i kombinasjon med pkt. 1 og/eller 2.

Oppbygging av sperresjikt /dekningsmasser

Norske velter er relativt små med en kupert og dels bratt overflate. Ofte ligger også veltene i en dal- eller fjellside med betydelig smeltevatnsmengder i vårløsningen. Det er derfor ikke problemfritt med et glatt og helt tett sperresjikt av tjukk, glatt plast. Dekningsmasser på toppen av et slikt sperresjikt vil lett bli spylt bort i sterkt regnvær eller i snøsmeltinga om våren.
Mangel på løsavsetninger for dekningsmasse er et annet hovedproblem. Ofte er også veltene lokalisert på temmelig utilgjengelige steder slik at massetransporten kan bli relativt problematisk.
Igangværende forsøk tar hensyn til disse rammebetingelser.
Følgende faktorer er variert
Sperresjikt

Slisseplast
Armert, noe vatngjennomtrengelig plast
Blank, tett plast

Det ideelle sperresjikt bør etter min mening kunne sprøytes på (skumplast e.l.) og helst være noe gjennomtrengelig for vatn.

Dekningsmasse

Mangel på løsavleiring har gjort at det på Kjøli bare er brukt bark. To barktykkelser - 4 og 8 cm - er brukt i kombinasjon med ovennevnte 3 sperresjikt.
På Eitrheimsneset (DNZ) er det forsøkt med 3 cm jord (uten sperresjikt) som dels er frest ned i eksisterende toppjord og dels lagt på toppen av opprinnelig jord (opptil 2% Zn, 1100 ppm Cu og 1500 ppm Pb og 100 ppm Cd i jorda).
Tilføring av torv i kombinasjon med kalking og fosforgjødsling førte til en betydelig reduksjon i lettløselige tungmetaller.

En kombinasjon av finpartikla morenemasser og organisk jord (torv eller bark) vil gi god nærings- og vatnkapasitet i massene og dermed et godt buffret dyrkingsmedium. Minimumstykkelsen på de respektive sjikt bør være 4-5 cm og 3-4 cm.

På sperresjikt/dekningsmasse-området kreves relativt omfattende forsøksvirksomhet

Kalking/gjødsling.

Kalking/gjødsling må tilpasses jordbunnsforholdene. Dekningsmassene trenger særlig tilførsel av N P og K. Til en viss grad kan det også være en positiv virkning av kalking. Derimot vil det sjelden være nødvendig å tilføre mikronæringsstoffer. Gjødslingsmengde og frekvens må tilpasses dekningsmassens tykkelse. Tynt jordlag krever små gjødselmengder men hyppig gjødsling - eks. 15 kg NPK-gjødsel 3 ganger i vekstsesongen. På dette felt er det mange forskningsresultater å støtte seg til. Ytterligere forsøk er derfor nødvendig i begrenset grad. Områder (uten sperresjikt) med tynt eller uten tilført toppsjikt av organisk materiale, må kalkes og tilføres store mengder fosfor. Dette vil bidra til å heve pH og redusere konsentrasjonen av lettløselige tungmetaller i toppsjiktet. I forsøk (Løkken) på jordarter med pH på ca 3,6 er det ved kalking med 1000 kg kalksteinsmjøl og 100 kg fosforgjødsel/da ved planting oppnådd god etablering og tilvekst på lokale treslag.

<u>Kalking og fosforgjødsling bidrar såpass sterkt til bedring av dyrkingsforholdene i sur (pH min. 3,5) og forurenset jord at vegetasjonsetablering vanligvis er mulig uten sperresjikt og dekningsmasser. Lokale tilpasninger krever relativt omfattende forsøksvirksomhet</u>.

Valg av planteslag.

Analyse av vegetasjon på velter i kyststrøk og i fjellstrøk viser at det er en viss forskjell i artssammensetning i de to klimaområder, men at denne ikke er så stor at det har noen praktisk betydning for revegetasjonsarbeidet. Håbjørg (1988) viser at treslagene bjørk (Betula pubescens), furu (Pinus silvestris) og vier (Salix spp) er de mest vanlige planteslag. Videre er grasslagene smyle (Deschampsia flexuosa) og engkvein (Agrostis tenuis) og dels rødsvingel (Festuca rubra) temmelig vanlige.

Gundersen (1988) viser dessuten at de genotyper (individer) som vokser på veltene, alle grunnet sterkt seleksjonspress, har spesifikk toleranse mot forholdene i jorda på velta. Videre er det vist i tabell 1 at adaptasjon til viktige klimafaktorer vanskeliggjør flytting av økotyper fra ett klimaområde til et annet grunnet overvintringsproblemer.

Konklusjonen på dette blir at det må satses på treslagene bjørk, furu, og vier og på grasslagene engkvein, fåresvingel og rødsvingel. Smyle er ikke aktuell i revegeteringsarbeidet grunnet dårlig frøproduksjon. Sortsmateriale av engkvein, rødsvingel og fåresvingel som er på markedet vil ha store problemer med overvintring i høgfjellet og nordover. Men fordi det er viktig å etablere et erosjonshindrende grasdekke, må det allikevel i hovedsak satses på disse. I den grad det er mulig bør det i tillegg høstes frø av tungmetallresistente og hardføre genotyper på de respektive områder. Dette frøet blandes med det kommersielle frø og sås ut. På dette vis vil man spredt over hele revegeteringsområdet få endel tungmetallresistente, hardføre og frøproduserende genotyper som med tiden kan bidra til full vegetasjonsdekning på området.

Det bør også samles frø av lokale treslag for innplanting på veltene - bjørk, furu og vier. Forsøk har ellers vist at bjørk og vier har så stor reproduksjonsevne og spredningskapasitet at det også er mulig å satse på naturlig vegetasjonsinnvandring. Naturlig vegetasjonsinnvandring av trær vil imidlertid ta noe lengre tid enn planting. Planting av trær muliggjør ellers valg av arter som lever i symbiose med nitrogenfikserende organismer og som dermed er sjølforsynt med nitrogen. Dermed kan vedlikeholdsgjødslingen reduseres til et minimum.

Vegetasjonsetableringen må primært ta sikte på å etablere et markdekkende grassjikt som kan hindre erosjon der det er påkjørt dekningsmasser. Det bør brukes kommersielle sorter av engkvein, fåresvingel og rødsvingel blandet med frø av lokale, helst tungmetallresistente genotyper. Tre- og buskvegetasjon bør fortrinnsvis søkes etablert ved å aksellere den naturlige vegetasjonsinnvandring. Ved planting av trær bør det brukes lokale økotyper av nitrogenfikserende planteslag - gråor.

Ved revegetering av massetak i forbindelse med uttak av dekningsmasser, transportveger o.l. nyttes samme teknikk som nevnt over.

Vegetasjonsetableringen må følges opp med et vedlikeholdsprogram som strekker seg over 3 år, og som i hovedsak består av gjødsling og kalking.

Ås, 4. juni 1988

Atle Håbjørg

Atle Håbjørg

COVER TECHNOLOGY FOR ACID MINE DRAINAGE ABATEMENT

LITERATURE SURVEY

by

DR. ANDREW ROBERTSEN

STEFFEN, ROBERTSEN & KIRSTEN, Vancouver, Canada

REPORT NO. 64702/1

COVER TECHNOLOGY FOR ACID MINE DRAINAGE ABATEMENT: LITERATURE SURVEY

FOR

NORWEGIAN STATE POLLUTION CONTROL AUTHORITY

June 1988

REPORT NO. 64702/1
COVER TECHNOLOGY FOR ACID MINE DRAINAGE ABATEMENT: LITERATURE SURVEY

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REPORT NO. 64702/1
COVER TECHNOLOGY FOR ACID MINE DRAINAGE ABATEMENT: LITERATURE SURVEY

1.0 INTRODUCTION

1.1 OVERVIEW OF ACID MINE DRAINAGE IN NORWAY

Norway has a history of intensive pyritic ore mining and processing for over 300 years. NIVA, 1987, lists some 39 substantial pyrite deposits as shown in Figure 1, the majority of which have been mined. Most of these mines are now abandoned.

During mining and milling, pyrite rich waste rock and mill tailings are deposited on the earth's surface. Mine workings are also opened up exposing ore and waste rocks to the entry of oxygen and water. Oxidation of the exposed pyrite, by both chemical and biological processes, results in the generation of sulphuric acid. The resulting acidic wastes leach heavy metals, notably copper and zinc, and other contaminants from the mine wastes and carry them away from their source of generation and into the environment.

Impacts on the environment of the acid mine drainage (AMD) can be severe. The Norwegian Institute for Water Research (NIVA) has conducted investigations and evaluations of the extent and impact of AMD from abandoned mines in Norway and the results are reported in NIVA, 1987. Table 1 provides a summary of the pollution and environmental impact assessments for the various mining areas in Norway. Severe impacts are experienced or threatened in the Orkla, Glama, Gaula and Sulitjelma water courses.

1.2 OBJECT OF THIS STUDY

The Norwegian State Pollution Control Authority has initiated a program for the abatement of the environmental impacts of AMD in Norway. A preliminary evaluation of alternative abatement measures at a select number of AMD sites was undertaken in 1987 (Steffen, Robertson and Kirsten, 1987). This study was requested by the Norwegian State Pollution Control Authority, in a letter dated 26th January, 1988, with the following objectives:

i) To evaluate technical solutions to covers for mine waste and tailings deposits. The testing was to include needs and possibilities of, as well as effects of, a capillary breaking layer, soil cover, and finally the establishment of vegetation over the waste pile.

ii) The study was to include a full-scale test in the field which should be part of the solution to the AMD problem at an existing waste deposit where abatement measures are urgently needed.

iii) The study for 1988 was to be considered as the first part of a program and would form the basis for further work. The program would, over a period of several years, test alternative abatement options to determine their effects on the waste deposit and the on the quality of the drainage.

A study program was developed by the Study Team (Steffen, Robertson and Kirsten; SEFO - Senter For Forskningsoppdrag; NOTEBY - Norsk Teknisk Byggekontroll; MILJOPLAN) and described in a proposal dated 24th February, 1988. The 1988 study program comprises three tasks:

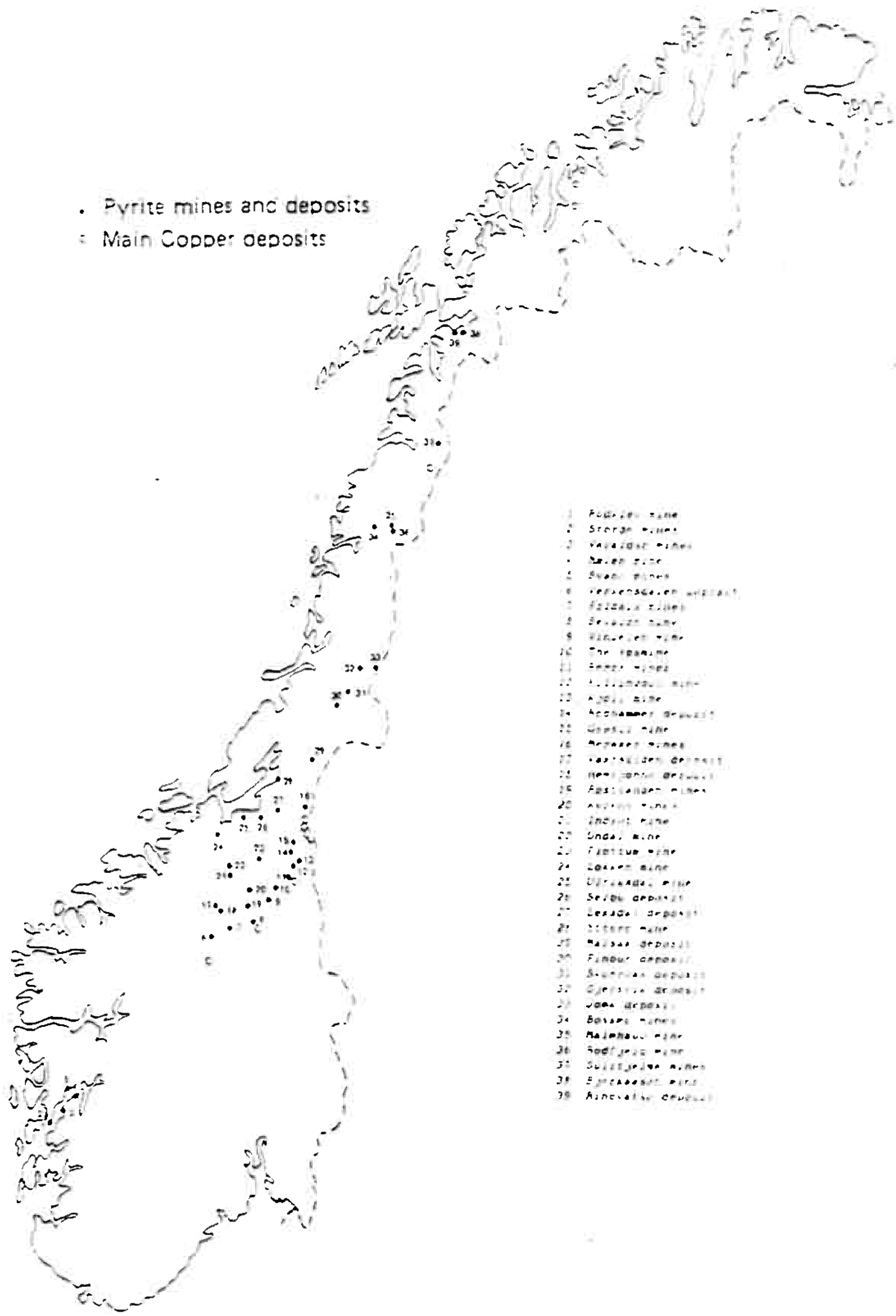

FIG. I PYRITE DEPOSITS IN NORWAY AS KNOWN IN 1926
(Foslie 1926)

TABLE 1 SUMMARY OF POLLUTION AND ENVIRONMENTAL IMPACT ASSESSMENTS FOR NORWEIGIAN METAL MINES (NIVA, 1987)

In the table O and X denote the following:

Effects		Inputs	
O	Unknown, negligible	O	Unknown, negligible
(O)	Unknown (possible effects)	X	Moderate
X	Minor effects	XX	Substantial
XX	Significant effects	XXX	Large
XXX	Severe effects		

Sect.	Name	Municipolitz	Draining to	Inputs	Effects	Further measures investigations
2.1.1	Vigsnes Copper Works	Karmøy	Lake Vigsnesvatn	XX	X	Inspection - controll programme
2.2.1	Evje Nickel Works	Evje and Hornnes	Otra	O	O	Inspection
2.2.3	Ertelien Nickel Works	Modum	Henoa (Tyrifj.)	XX	X	Inspection + controll investigation
2.2.4	Langdalen	Romerike/ Modum	Skjærdalswater-course (Tyrifj.)	X	X	Lower stretch of Skjærdalswatercourse should be investgated
2.2.5	Eiker Copper Works	Øvre Eiker	Drammenswater-course	X	X	
2.2.6	Modum Cobalt Works	Modum	Snarumselva-Simoa Drammensw.course	X	O	Inspection
2.2.7	Konnerud	Drammen	Drammenselva-Sandew.course	X	O	Affects ground water in the area
2.2.8	Grua	Gran	Viggaw.course	X	O	
2.2.9	Espedalen	Sør-Fron/ Gausdal	Gausa	O	O	
2.2.10	Fossgruva	Tolga-Os	Vangrøfta-Glåma	X	X	
2.2.11	Oscar II	Tolga-Os	Vangrøfta-Glåma	X	X	
2.2.12	Folldal	Folldal Alvdal	Folla-Glåma	XXX	XX	Detailed investiga-tined ... carried out 1984/85. Measures are under concerderation
2.2.14	Røstvangen etc.	Tynset	Gløta-Tunna Glåma	X	X	Effect of measures is being monitored
2.2.15	Kvikne Copper Works	Tynset	Ya-Orkla	XX	X	To be monitored of after the regulations of the Orkla w.course

TABLE 1 (cont'd.)

In the table O and X denote the following:

Effects		Inputs	
O	Unknown, negligible	O	Unknown, negligible
(O)	Unknown (possible effects)	X	Moderate
X	Minor effects	XX	Substantial
XX	Significant effects	XXX	Large
XXX	Severe effects		

Sect. investigations	Name	Municipality	Draining to	Inputs	Effects	Further measures,
2.3.1A	Røros Copper W. Stortvartz	Røros	Glåma	XX	X	Measures in Lake Djupsjøen
2.3.1B	Røros Copper W. Kongens	Røros	Glåma	XXX	XX	Lake Orvsjøen is has been monitored
2.3.1C	Røros Copper W. Muggruva	Røros	Gaula	X	O	
2.3.1D	Røros Copper W. Hessdalen	Holtålen	Hesja-Gaula	X	X	
2.3.2.	Kjøli	Holtålen	Gaula	XX	XXX	Furhter liming etc. in 1985?
2.3.3	Killingdal	Holtålen	Gaula	XXX	XXX	Measures in draft
2.3.5	Undal Works	Rennebu	Skauma-Orkla	X	X	
2.3.6	Løkken Works	Meldal	Raubekken-Orkla	XXX	XXX	Further measures should be drafted. The situation is being closely monitored
2.3.7	Dragset Works	Meldal	Vorma-Orkla	XX	X	The situation should be monitored by means of a con-troll programme
2.3.8	Høydalsgruva	Meldal	Svorka	X	O	
2.3.10	Lillefjell	Meråker	Stjørdals-watercourse	X	(X)	The Meråker area should be inspected
2.3.11/12	Gaulstad-Mokk			O	O	
2.4.1	Bossmo	Mo i Rana	Ranafjorden	X	(X)	
2.4.3	Sulitjelma	Fauske	Sulitjelma watercourse	XXX	XX	
2.4.4	Bjørkåsen	Balangen		X	O	
2.4.5	Tårstad	Evenes	Lavangseidet	O	O	
2.4.7	Bidjovagge	Kautokeino	Altawatercourse	O	O	

Task 1 - Literature survey of technology of covers design.

Task 2 - Evaluation of alternative test sites and preparation of a test program.

Task 3 - Description of technical measures with cost estimates for the test program.

This report outlines the objectives of the study and documents the results of Task 1 - the literature survey.

It briefly reviews the process of acid generation and AMD with the view of identifying the objectives, design criteria, operating conditions and long term effectiveness of covers. It reviews published information on the design of covers, evaluates alternative design methodology and discusses the results of tests or remedial actions conducted elsewhere.

1.3 BASIS FOR THIS STUDY

For the evaluation of the effectiveness of alternative cover types it is important to understand the mechanisms of acid generation and drainage into the environment. The review therefore starts with a brief non-technical summary of these processes and the factors which control them. This is followed by a brief description of the alternative cover types and how they serve to control AMD. Finally the control technology is reviewed and design methods discussed. Since AMD is a long term process, the effectiveness of the control technology in the long term is reviewed and technology for the achievement of long term stability is described.

2.0 FACTORS SIGNIFICANT TO COVER DESIGN

2.1 ACID MINE DRAINAGE

The process of acid generation in mine wastes has been described by numerous workers including Knapp (1987); Paine (1987); Errington and Ferguson (1987a). It is a time-dependent process controlled primarily by:

- the presence and nature of reactive sulphides
- the exposure of the reactive sulphides
- bacterial action
- availability of water
- availability of oxygen (by convection and diffusion)
- temperature
- pH or presence of alkaline reactants

Kleinmann et al (1981) identified three stages of acid generation. These are illustrated in Figure 2. During the first stage, chemical and/or biological oxidation of pyrite and other sulphide minerals slowly produce acid. This acid may initially be neutralized by basic minerals in the rock. Once the base source is exhausted the second stage initiates as the pH drops in the microenvironment around the sulphide minerals and acid generating bacteria such as Thiobacillus ferrooxidans begin to multiply. These bacteria cause a further decrease in the pH and the third stage is initiated. When the pH decreases below 3, ferric iron remains in solution, ferrous iron is converted by bacteria to ferric iron which in turn oxidizes the sulphur minerals. The rate of acid formation is rapid in the last stage and is limited by the concentration of ferric iron.

Robertson, 1988, has proposed an analogy which is useful in understanding the processes of acid generation and drainage. It is illustrated for a waste dump in Figure 3. A description follows.

Factors Controlling Acid Generation

Acid generation occurs in a **sulphide reactor**. This reactor contains a finite load of sulphide. The rate at which the reaction proceeds is dependant on:

i) The nature of the reactive sulphides; with some oxidizing much more rapidly than others, EPA, 1977. The form of the sulphide is also important with disseminated framboidal pyrite oxidizing more rapidly than large cubical crystal forms.

ii) The rate at which the other fuels:
   - oxygen and
   - water,

   are introduced into the reactor.

iii) The initiation of bacterial oxidation may increase the rates of oxidation from 50 to 1 million times, Lundgren, 1971. Both the chemical and biological oxidation rates are substantially dependant on the pH in the reactor as illustrated in Figure 4 (Knapp, 1987). Typically the reactor starts up slowly with local slow chemical oxidation, and increases rapidly as biological oxidation starts after the pH has dropped below 5. On a single lump of waste rock, as illustrated in the inset in Figure 3, individual crystals of pyrite may develop a surface coating of low pH adhered water, providing the conditions for rapid bacteriological oxidation long before similar conditions develop on other surfaces of the otherwise naturally alkaline host rock.

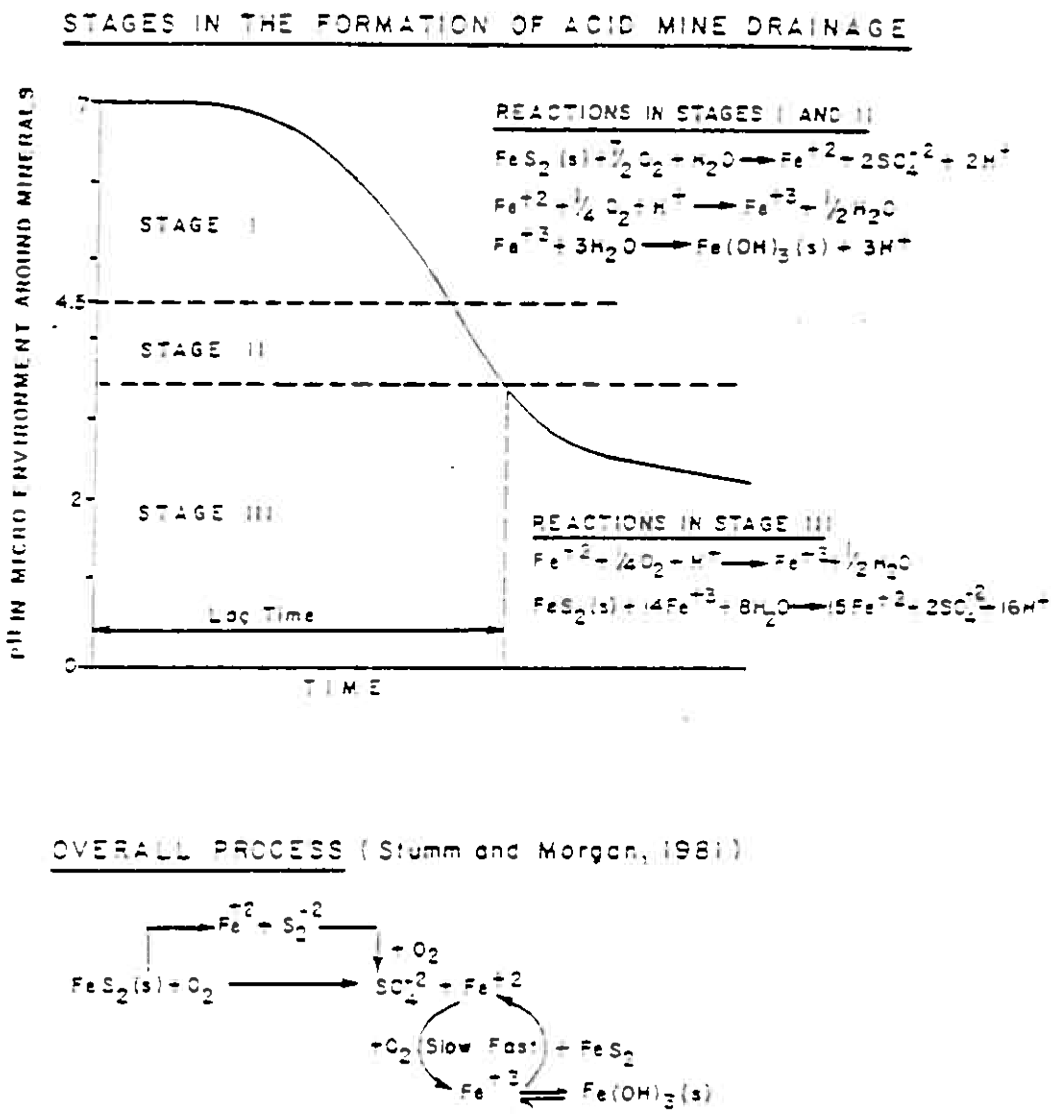

FIG. 2 FORMATION OF ACID MINE DRAINAGE FROM PYRITE OXIDATION

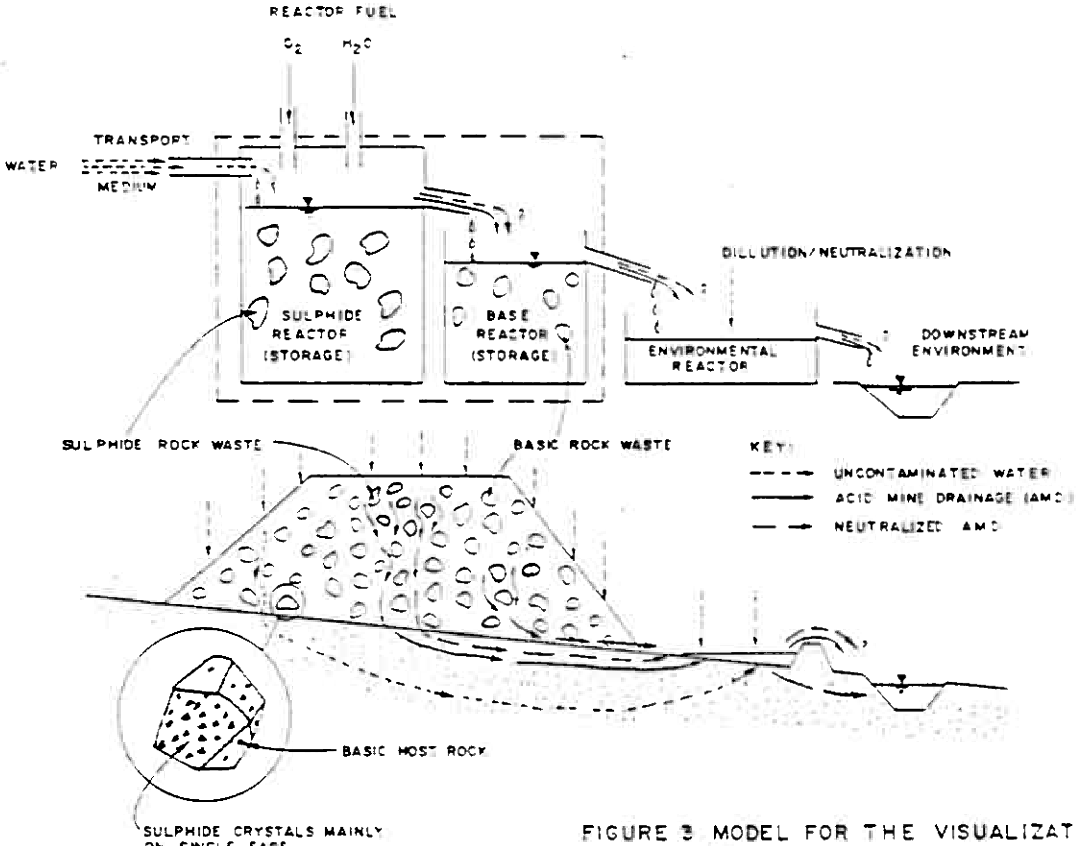

FIGURE 3 MODEL FOR THE VISUALIZATION OF LONG TERM AMD

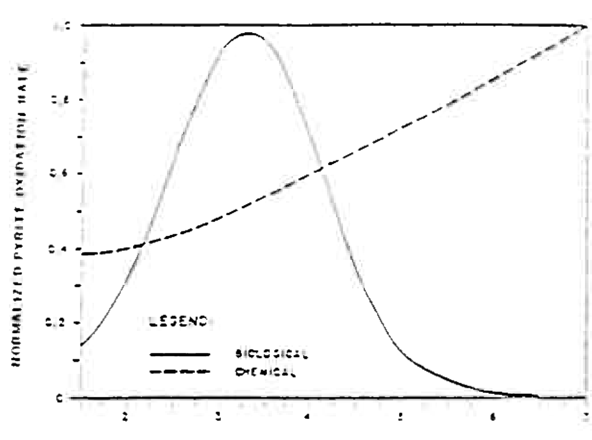

FIGURE 4 : EFFECT OF pH ON BIOLOGICAL AND CHEMICAL OXIDATION RATES (AFTER KNAPP, 1987)

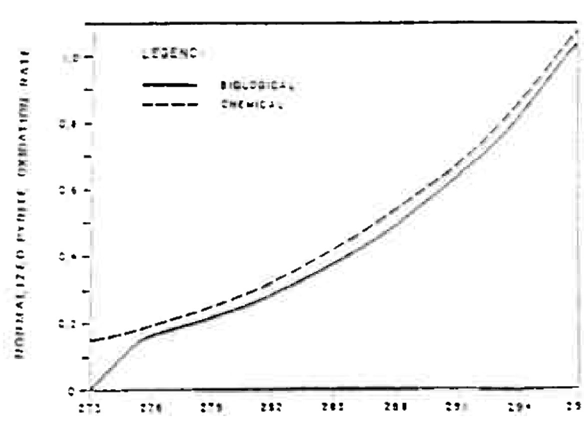

FIGURE 5 : EFFECT OF TEMPERATURE ON BIOLOGICAL AND CHEMICAL OXIDATION RATES (AFTER KNAPP, 1987)

iv) Oxidation rates increase as the temperature increases as illustrated in Figure 5 (Knapp, 1987).

Acid is produced as a result of the oxidation reactions and stored in the vicinity of the site of generation unless transported away from the site by diffusion or advective water flow. In the absence of flowing water the movement rate is extremely slow, and the acid products accumulate (are stored) in the vicinity of the sulphides and inhibit further oxidation. Movement of acid from the sulphide reactor site is therefore dependant on the presence of water as a **transport medium**. Removal of the acid products allows the oxidation reactions to continue. Metals and other products are dissolved in the acidic water. Flow or discharge from the sulphide reactor is therefore an acidic water flow containing dissolved metals commonly referred to as Acid Mine Drainage or simply AMD.

Factors Controlling Natural Neutralization

AMD is discharged into the **base reactor** in which it is neutralized. If the reaction rate in the base reactor is slower than the rate at which AMD is delivered to it then only partial neutralization occurs. There may still be acidic drainage while a net base potential remains. In practice this may occur on a lump of rock waste where the sulphide is on the exposed "joint" surfaces and oxidizing rapidly and the base material is contained in the interior of the waste lump from which it is released more slowly.

The base reactor has a finite quantity of base material and after this is consumed no further neutralization occurs. Depending on the amount of base material contained in the base reactor, and the rate at which AMD is introduced to it, the effective period of neutralization will vary. Thus if AMD generation or transport from the sulphide reactor is slow (as may occur in a dry climate or if a low permeability cover is installed) then the neutralization may be effective for a very long time, but the end result of acidic drainage may be the same. Some of the dissolved metals in AMD, including zinc, are not adequately precipitated on neutralization and such drainage may still be detrimental to the receiving aquatic environment.

The acid/base accounting method of AMD potential testing measures the total quantities of acid and base generating potential but does not take into account the rates at which the two reactors work. Hence the need for kinematic tests, particularly when the acid and base potential are nearly equal. Both static and kinematic test methods for acid generation potential determination are reviewed and summarized by Ferguson and Erickson, 1987.

Factors Controlling AMD and Environmental Impact

AMD draining from the waste deposit passes through or over soils which have a neutralization capacity. AMD is also mixed with surface and groundwater streams which both neutralize and dilute the AMD. The **environmental reactor** results in some improvement of the AMD. As the store of basic materials close to the deposit is consumed the AMD plume migrates further from the deposit, resulting in an ever increasing impact zone. At some point the volume of dilution and neutralization streams with which it merges is sufficient that the effect of AMD is abated.

Some environments have an extremely high load of basic materials in the soils, surface and ground waters. This is the case in many of the old deeply weathered landscapes with well developed old soils. In Canada and Norway, where recent glaciation has stripped off all old soils, only relative fresh rocks and new, unweathered soils are exposed. These conditions generally yield soils, ground and surface waters which are low in neutralizing capacity. Thus the potential for extensive impact on the surface and groundwaters downstream from the deposit are large.

For all waste deposits there is a level of AMD release which can be sustained by the environment without significant damage. It is the objective of AMD abatement measures to reduce AMD releases to below this level.

2.2 ROCK WASTE DUMPS AND TAILINGS DEPOSITS AS REACTORS

Waste Rock Dumps

A waste dump can be visualized as a series of sulphide and base reactors on both a micro and macro scale. On each piece of rock waste there may be zones which are either acid generating (and acidic) or base yielding (and basic). The most reactive lump produces acid which allows the pH to depress, initiating rapid biological oxidation. Drainage from this acid lump onto lumps below it rapidly consumes excess base allowing a zone of high reaction rates to spread along the drainage path.

Zones of predominantly acid rock waste will yield AMD and zones having a net base potential will neutralize AMD flowing into them. The pH of the seepage from such a dump is dependant on the flow path which the transporting water has followed and may be either basic or acidic as illustrated in Figure 3. It is common to have both basic and acidic seeps from the same dump during the early stages of AMD development.

Neutralized AMD will contain some of the dissolved metals taken up in the acid flow zones and not deposited on neutralization. Zinc is often a persistent metal in this respect. Thus the evolution of AMD from the dump is cronologically as follows:

i) All seepage characteristic of unoxidized leaching of the waste rock.

ii) Some seeps develop characteristics of neutralized AMD (Lundgren et al, 1972). This stage may be evident within weeks or months of waste placement but may take years.

iii) Some seeps begin to exhibit depressed pH and characteristics of AMD. The stage may develop within months of placement but a number of Western Canadian examples exist of it taking 10 to 20 years for the <u>first</u> such seeps to develop.

iv) pH depression develops in all seeps from AMD generating zones of the dump. For many mines this may be the entire waste dump. Documentary evidence exists of highly reactive dumps that have developed to the maximum rate of AMD within a few years of construction. Our understanding is not complete for the time rate of increase for the much slower developing reactors. Based on the observations for (iii) above it may take tens to hundreds of years.

v) The quality of seepage improves as the sulphides are consumed and that in the faster of the sulphide reactors is exhausted. Slow reactors (such as sulphides in the interior of large hard rock lumps) can last for a very long time. This phase is observed to last for hundreds of years for large waste dumps.

Seepage flows through waste dumps are usually partially saturated, and flow tends to follow preferred flow channels. Some of the waste lumps are regularly flushed by infiltrating water while the majority are seldom wetted by flowing water. Acid products accumulate on the seldom flushed lumps with the result that acid products accumulate. During periods of high flushing (spring run-off or high precipitation periods) some of these products are flushed out resulting in high flows with very high contaminant concentrations. Because hydraulic conductivities are usually high in the dump, the period between infiltration and seepage is short, resulting in rapid flushing and seepage responses to precipitation.

The air conductivity of dumps are also extremely high. The tendency for coarse waste with large air voids to accumulate at the base of the dumps usually provides an easy passage for air into the base of the dump. Thermal gradients develop due to natural temperature ranges in the ambient air as well as due to the exothermic reactions of oxidation in the waste dump. Such thermal gradients result in a 'chimney effect' in the dump with the result that oxygen is drawn into the dump and is readily available to fuel sulphide reactions. Changes in the barometric pressure causes the dumps to 'breath'. This 'lung effect' results in air flowing in along the more conductive channels deep into the dump. The entire dump therefore serves as a reactor with an ample supply of oxygen and water for sulphide oxidation.

In hard rock dumps which are resistant to weathering the amount of sulphides exposed and available for oxidation is limited, limiting the rate of reaction. The mass of sulphides exposed to oxidation conditions are almost directly proportional to the surface area of dump material which increases approximately as the inverse of the cube of the average lump or particle mass in the dump. Where the rocks slake (break down with time) rapidly, increasing the surface area of exposed sulphides, the rate of sulphide oxidation may increase rapidly. Fortunately the slaking effect, if it reduces the rock to soil size particles, may also reduce the dump conductivity to both air and water sufficiently to result in reduced reactions.

Tailings Impoundments

The factors controlling acid production and AMD from tailings impoundments differs significantly from those operative in waste dumps. These differences are summarized in Table 2.

The sulphide content in tailings is often much greater than in the mine wastes from the same ore deposit, resulting in a potential for much larger acid production.

Because of the fine grind, a high percentage of the sulphides are exposed and available for oxidation. If only these two conditions controlled reaction rates, then tailings would be expected to be much greater acid producers than rock wastes.

The fine grind and mixing that occurs in the milling process results in a much more even distribution of the sulphides and base materials in the deposit. The tailings are also generally mainly in water or in a high moisture state, particularly during the operating life of the impoundment. The combination of intimate mixing of sulphides and base materials and the ample water to transfer acid and base ions over the small distances separating grains results in a hugely reduced potential for the development of 'trigger' or 'hot spots', and hence the initiation of fast reactions due to biological leaching. Conditions suitable for acid generation usually only develop after tailings placement is discontinued. Thus the initiation of acid generation in a tailings dam usually starts long after the same initiation in waste dumps at the same mine.

In contrast to dumps, infiltration and flow of air and water is restricted by the relatively low conductivity of the tailings. This flow restriction results in the development of zones of oxidation (sulphide reactor zones), neutralization (neutralization reactor zones) and contaminant migration as illustrated in Figure 6. These zones have been described and discussed by Robertson, 1987.

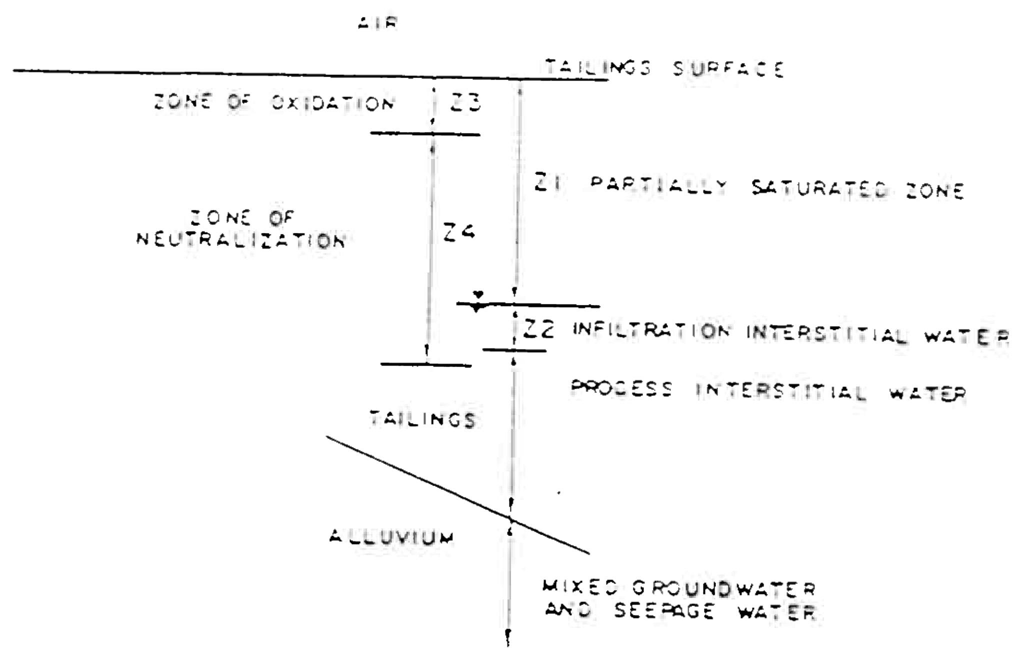

FIGURE 6 (a) WATER QUALITY MODEL FOR TAILINGS COLUMN

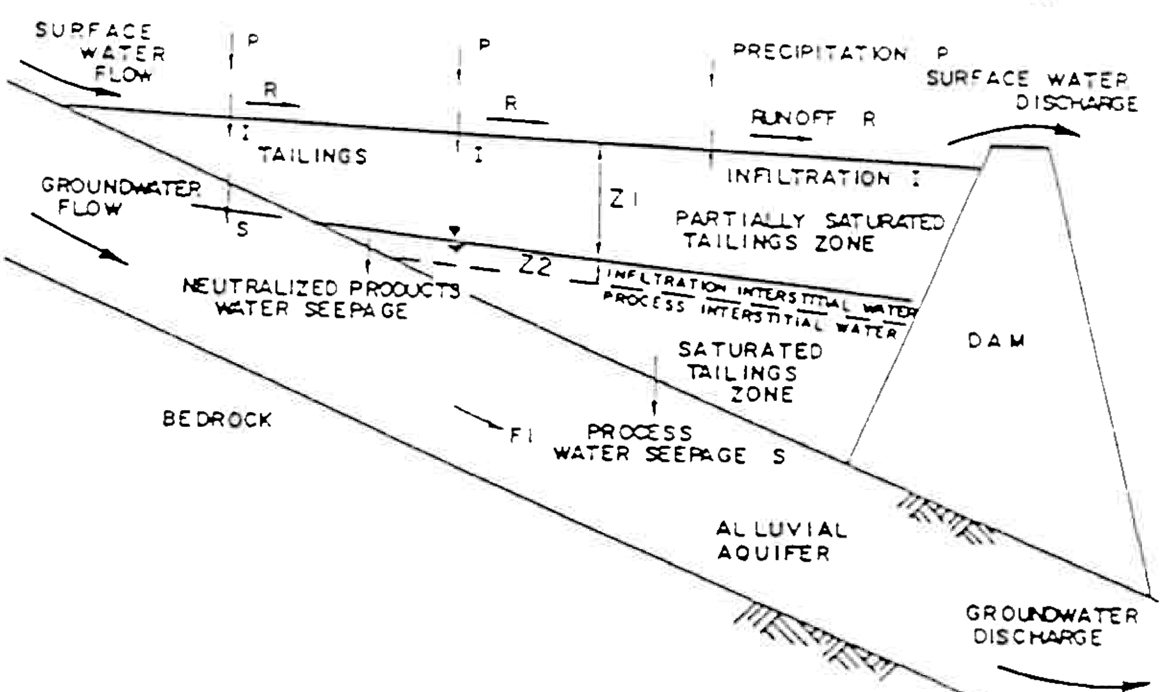

FIGURE 6 (b) INFILTRATION CONTAMINANT
SEEPAGE MODEL

TABLE 2

COMPARISON OF ACID MINE DRAINAGE FACTORS IN WASTE DUMPS AND TAILINGS IMPOUNDMENTS

A. ACID GENERATION	WASTE DUMPS	TAILINGS IMPOUNDMENTS
i) Sulphide source	- Variable in concentration and location - Conditions may vary from sulphide rich to basic over short distances	- Conditions uniform often with very high sulphide content.
ii) pH variation	- Highly variable conditions over short distances	- Fairly uniform conditions with a few major horizontal zones
iii) Initiation of Rapid Oxidation	- Usually starts immediately after first wastes are placed (in "trigger" spots)	- Usually starts after tailings placement ceases
iv) Oxygen Entry	- Enters freely along highly conductive flow paths at base of dump and large open void spaces. 'Chimney' and 'lung' effects.	- Restricted by water in the tailings void spaces and the lower conductivity of the partially saturated void spaces
v) Seepage	- Seepage rapid along preferential flow paths	- Seepage slow and uniform
vi) AMD Releases	- Large infiltration resulting in large seepage from toe and to groundwater - Rapid release following generation, sometimes with both neutralized and acid AMD seeps.	- Large early surface AMD run-off - Lower infiltration - Gradual transition in seeps from process water, to neutralized AMD, to AMD.

After tailings placement is discontinued, oxidation usually initiates fairly quickly at the surface of the tailings impoundment where tailings are exposed above water and where oxygen has free access to the tailings. Surface drainage from the tailings impoundment turns acid fairly quickly. The downward movement of the AMD is extremely slow because of the generally low permeability of the tailings and foundation soils. Flow is relatively uniform and there is not a route for rapid preferential seepage as in dumps. The period for the immergence of the first AMD contaminated groundwater may be extremely long (tens, possibly hundreds of years).

After impoundment closure, there is often a slow lowering of the water table in the impoundment as drainage takes place. The partially saturated zone, in which oxidation can occur, increases. Figure 7 illustrates the rapid increase in the gaseous difusivity of tailings or soil as the moisture content reduces. Drying of the upper surface of the tailings impoundment also results in dessication cracking of the fine tailings. This increases the conductivity to both air and water. If sandy zones are located in the deposit the cracks connect with the pore spaces in the sandy zones and the lung effect results in oxygen transmission to these zones. Thus we often observe yellow, oxidized sand zones interlayered with gray unoxidized slimes layers near the surface of the tailings impoundment.

2.3 ALTERNATIVE AMD CONTROL TECHNOLOGIES

AMD control technology is conveniently divided into three broad classifications.

- Prevention of acid generation
- Prevention of AMD migration
- AMD collection and treatment

Covers can serve to both control acid generation and AMD migration. Since covers may be used in combination with one or more of these other technologies, they are briefly reviewed.

Prevention Of Acid Generation

Prevention of acid generation in the first place is usually the most desirable of the control technologies. If generation does not occur, then there is no risk of its transportation into the environment.

This may be achieved by:

i) Removal of the pyrite source

Methods of pyrite removal have been reviewed by Hester and Associates, 1984. Removal of a portion of the sulphides from tailings, by flotation for example, may make the difference between net acid production and consumption. Oxidation of the sulphides, as part of the mineral extraction process, is being achieved at a number of gold mining properties (refer paper by Robertson et al, this conference). If applicable, the results are effective in the long term and no additional abatement measures or maintenance is required. Costs are high and the technique is not applicable to wastes.

ii) Rendering acid generation minerals inactive by the development of surface coatings

Hester and Associates, 1984, demonstrate that while these approaches hold promise they do not as yet represent applicable technology.

iii) Exclusion of water

Exclusion of water to the extent that acid generation could not occur is not considered practical (Robertson, 1987).

iv) Control of biological oxidation

Bacterial action control by the application of bactericides have been discussed by Sobek, 1987. Periods of effective control for admixed bactericides do not exceed 5 years and surface spraying has been found to be not effective. This measure provides no control of chemical oxidation.

v) Temperature control

If freezing conditions can be achieved, acid generation can be prevented. This control has application in regions of permafrost, with adequate precautions being taken to prevent seasonal surface thawing. Elsewhere surface covers have the effect of reducing the maximum surface temperatures, and therefore oxidation rates, but do not provide adequate control for abatement.

vi) Exclusion of oxygen

The exclusion of oxygen to the extent that acid generation is reduced to acceptably low levels requires the placement of a cover with an acceptably low level of oxygen diffusion.

vii) Addition of Base

Base addition can take a number of different forms.

- Blending of net acid producing and net acid consuming wastes to achieve a net acid consuming mixture. This is what is effectively practiced in the coal strip mines of the eastern USA (Skousen et al. 1987). Its effectiveness for hard rock mines has not been demonstrated. The flat horizontal layering of the sedimentary overburden over the coal seams and the strip mining method are well suited to achieving consistent and intimate blending of the overburden. The distribution of waste rock types and the mining sequence in hard rock mines is considerably more variable and consistent blending more difficult and costly to achieve. Where poor blending has been done, acid generation may still continue in localized zones resulting in neutralized AMD seepage of a quality which is still of concern to the downstream environment.

- Base addition and blending can be practiced to allow consistent control throughout the waste. The base material must be sufficiently fine and uniformly distributed that the rate of base neutralization is sufficient to prevent AMD development. Large isolated lumps of limestone would be inadequate even though the acid/base account may indicate a reserve of base. Coarsely ground limestone or calcareous rock is desirable. The rate of solution (hence depletion) and pH resulting from the addition of large quantities of slaked lime may be unsuitable for both short and long term AMD control.

- Surface applications are not considered effective for long term control.

Prevention Of AMD Migration

Where acid generation is not prevented, it is necessary to resort to prevention of acid and acid product migration. Since water is the mode of transport, the control technology relies on the prevention of water <u>entry</u> to the waste pile. Control of water <u>exit</u> from the pile is of little value since in the long term all water entering the pile must exit, long term storage being neglectible. The control requirements are as follows:

- diversion of all surface water flowing towards the pile
- interception or isolation of groundwater flow towards the pile
- prevention of infiltration of precipitation into the pile

Diversion facilities usually consist of ditches. Diversion of surface flows, while easily implemented are often difficult to maintain in the long term as discussed in the next section. The best long term solution to such surface flows is to select a disposal site which minimizes the need for diversion.

If the pile is located over a groundwater discharge area, interception and isolation of the groundwater is very difficult to achieve and maintain in the long term. While measures such as underdrains and sealing layers may be employed, their performance in the long term is questionable. The most effective solution is to select a site which is not located on a groundwater discharge area.

The secure prevention of infiltration, over the long term, is the most difficult to achieve. Covers of different types may be considered.

Reductions in infiltration do not necessarily mean equivalent reductions in contaminant loadings in AMD, since the concentrations may increase in the reduced flows. A 75% percent reduction in infiltration may only result in a marginal reduction in contaminant loading. On the other hand the placement of a cover also has other abatement effects due to reductions in air entry and thermal variation control. Reliance on covers for infiltration control requires their continued effective operation, in the long term, as discussed in the next section.

AMD Collection And Treatment

Collection and treatment provides the last defence in AMD abatement. It involves technology which is well established, and is working effectively at a large number of mines. Its main drawback is that collection and treatment must be continued for a very long period in time, while AMD continues.

The risk of failure of any form of treatment system is large. Extreme events such as flooding and fires can prevent treatment operations. Mechanical failure, labour disputes, power loss, reagent supply failure, all represent large risks to continued treatment.

Alternative chemical treatment methods are reviewed by Skousen et al, 1987. The primary treatment method currently involves treatment with lime to produce a sludge. The mass of sludge produced exceeds by many times the mass of the sulphides responsible for acid generation. In the long term the volume of sludges may exceed the initial volume of the wastes producing the acid. Disposal of these sludges are an increasing problem. Long term dissolution of these sludges may represent an environmental hazard as great as the initial acid producing waste. Chemical treatment, while it offers a secure short term method of achieving environmental protection, may not offer long term affordable solutions.

Treatment of AMD by wetlands has been demonstrated by a number of workers to be effective (McHerron, 1986, Huntsman, 1985 & 1986, Pesavento and Stark, 1986). For northern climates, concerns exist regarding both the short term continuous effectiveness and the long term maintenance requirements. Biological activity reduces in the winter and water flow in winter is canalized by ice formation. Unless adequate treatment can be maintained all year round it will be necessary to store winter flows for treatment in the summer. The long term fate of metals accumulated in the organic deposits have not been determined. Where feasible, wetlands should be considered as the final polishing step in the treatment of residual AMD, after the implementation of other primary abatement measures.

3.0 ALTERNATIVE COVER TYPES

3.1 WATER COVERS

Water covers can be achieved by placing mine wastes or tailings under water in the sea (as is practiced at sites such Island Copper mine, Kitsault mine and the now closed Britannia mine in British Columbia), in lakes or in man made impoundments. Ample demonstration exists of the effectiveness of acid generation control with under water placement of potentially acid generating tailings in the numerous tailings impoundments that have been investigated.

Both the cover and the water in the interstices help to prevent the oxygen transfer to the sulphides. It may be necessary to place a fine grained cover layer over submerged coarse rock waste to prevent oxygen transfer by convection of the pore water. Care must be exercised in the placement of old wastes below water to ensure that the solution of contained acid products is allowed for.

3.2 SYNTHETIC COVERS

Synthetic membrane covers such as polyvinyl chloride (PVC), high density polyethelene (HDPE) etc. have been applied with some success (Caruccio and Geidel, 1983, and 1986,). The extremely low conductivity of such membranes to air offers the potential for oxygen exclusion. Main causes of failure would be holes due to imperfection in manufacturing and installation and formed after installation due to dump deformations and puncturing. The long term degradation and loss of plasticity and ultimate cracking limits the long term effectiveness of such thin membranes. In combination with low conductivity bedding materials or cover layers, the effects of such cracking can be considerably reduced. To entirely prevent oxygen entry it may be necessary to entirely encapsulate the waste in such a membrane.

Other synthetic membranes such as geopolymers, asphalts, and cements suffer from cracking and disruption. The potential for cracking increases with:

- reduced tensile strengths
- reduced plasticity and ductility
- increased dimensional instability due to humidity or temperature changes
- degradation with aging.

Because of their dimensional instability, bentonite covers have displayed a tendency to crack on dessication (Geidel and Caruccio, 1985). A cover material currently being evaluated which shows promise is high volume polypropylene fibre reinforced sulphate resistant gunite. The corrosion resistant fibres provide tensile strength and flexibility and spread out any cracking which occurs. It has the considerable advantage that it can be applied to steeply sloping surfaces and areas inaccessible to large mechanical equipment. Cracking generally would result in sufficient oxygen entry for acid generation to continue, though at a reduced rate. The costs of such synthetic covers are often prohibitively high.

3.3 SOIL COVERS

Gaseous diffusion through soil covers differs little from that for tailings (Halbert et al, 1983, Silker and Kalkwarf, 1983). The effectiveness of the soil cover reduces rapidly as the moisture content of the cover reduces (Figure 7). Long term disruptions of the soil cover may occur as a result of erosion, cracking, frost action, root action, and burrowing animals. Current modelling techniques do not adequately allow for such disruptions. The comparative reduction in oxygen entry would be greatest for covers over very coarse waste dumps where 'chimney' and 'lung effects' would be greatest. There is considerable doubt that shallow soil covers will provide sufficient oxygen transfer control to abate acid generation. They reduce the infiltration of water and hence reduce

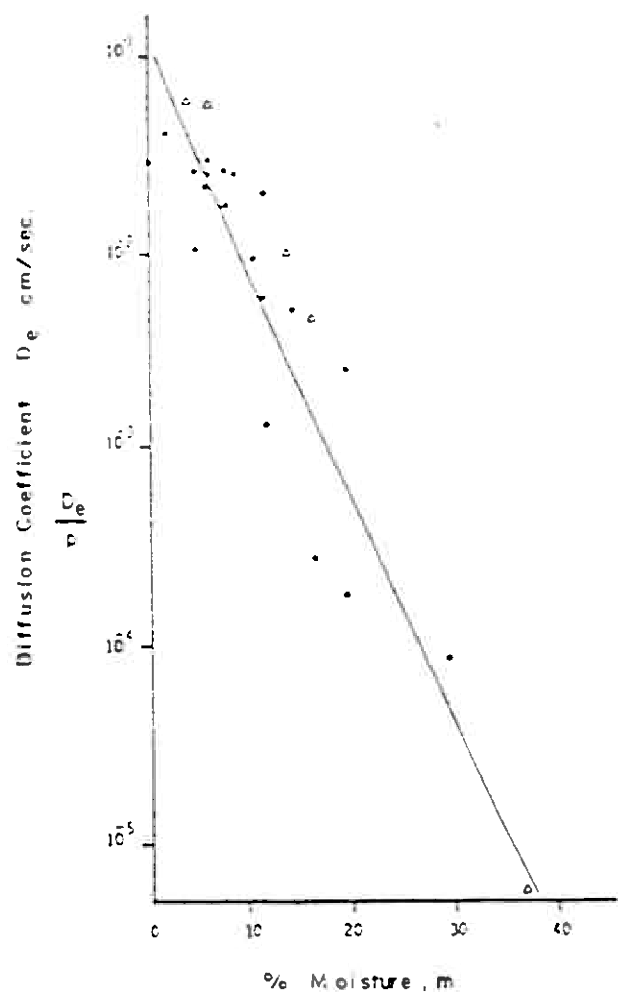

FIGURE 7 : MOISTURE DEPENDENCE OF THE DIFFUSION COEFFICIENT (ROGERS AND NIELSON, 1981)

the acid transport mechanism as discussed in the next section. Mathematical models and computer codes (such as the RATAP model, SENES et al. 1986.) have been developed to evaluate the effect of soil covers on acid generation rates.

Soil covers all have a variable but finite permeability. Depending on the nature of the cover, cover slope and precipitation pattern the percentage runoff varies dramatically. Surface slopes which permit ponding considerably increases infiltration. Vegetation increases evapotranspiration but inhibits runoff and may or may not result in a net reduction in infiltration.

Complex covers (layers of high and low permeability) may be employed to promote drainage, reduce infiltration, facilitate vegetation growth, reduce erosion or resist frost action. Alternative cover designs and their long term stability are reviewed by Steffen, Robertson & Kirsten, 1986a.

The Hydrologic Evaluation of Landfill Performance (HELP) amd TRUST (Reisenauer et al, 1982) programs may be used to estimate surface runoff, sub-surface drainage, and seepage that may be expected from a wide variety of cover designs.

Bog covers are a combination of water and soil covers, involving a layer of non acid generation soil placed over the acid generation waste together with shallow flooding, such that a layer of the soil cover is always saturated. The bog vegetation and saturated soil provides the barrier to the oxygen transfer (Steffen, Robertson and Kirsten, 1986a).

4.0 INFILTRATION

4.1 WATER FLOW IN UNSATURATED SOILS

Water transport in the cover material and in the waste underneath takes place under generally unsaturated conditions (Collin, 1987). This implies that the porous material is partly filled with air. Flow under unsaturated conditions is considerably smaller than under saturated conditions. Water statics and dynamics in the unsaturated zone are also of utmost importance for the diffusional transport of oxygen from the surrounding air since a high moisture content in the cover material is needed to restrict this transport. In the following, a general description of water flow in unsaturated materials is given (substantially summarized from Siwik et al. 1987). For a more detailed account see Collin (1987), Steffen, Robertson and Kirsten (1986a), Rasmuson (1978), or Bear (1972 and 1979).

One of the fundamental relationships that describes fluid flow through porous media is Darcy's Law:

$$v = -k \frac{dh}{dx}$$

Darcy's Law is a linear relationship between the specific discharge (v) and the driving force, the hydraulic gradient (dh/dx), with the proportionality coefficient being the hydraulic conductivity (k). This relationship holds true for both saturated and unsaturated conditions.

Whereas the water in a saturated soil is under positive pressures relative to atmospheric (gauge pressure), the water in an unsaturated soil is held in the pores under negative pressures (tension) caused by the surface tension between water and air (see Figure 8). This soil-water pressure head varies with water content (see Figure 9) because smaller pores hold water under greater tension than larger pores, due to the greater curvature of the water menisci. Figure 10 shows the result of this phenomena relative to the water table. In the saturated soil, the hydraulic conductivity (k) is a constant; however, in the unsaturated zone, k varies with pore water pressure head, p_w) (see Figure 11) and thus with the water content (w). As the pore water pressure head becomes more negative (drier soil), more pores drain and no longer contribute to the flow of water, thereby reducing the hydraulic conductivity of the soil.

A clearer conceptual appreciation of the difference between water flow in saturated and unsaturated soil can be achieved by inspecting Figure 11. This diagram shows the variation of the hydraulic conductivity (k), the pore water pressure head (or simply noted as pressure head (p_w) and the water content (w), along a horizontal soil core conducting water under steady-state conditions.

The non-linear characteristics of the unsaturated flow system result in it being more difficult to accurately model water flux.

$$\frac{\partial w}{\partial_t} = \frac{\partial}{\partial x} \left[k(p_w) \frac{\partial p_w}{\partial x} \right]$$

where:

w	=	the volumetric water content
p_w	=	the pressure head
k	=	the unsaturated hydraulic conductivity.

Traditionally, the subject of unsaturated flow of water in soil has been dealt with by subdivision into narrower topics with infiltration, drainage and evapotranspiration being prominent. The following is a brief discussion of these three topics for uniform soil profiles (such as a single layer engineered

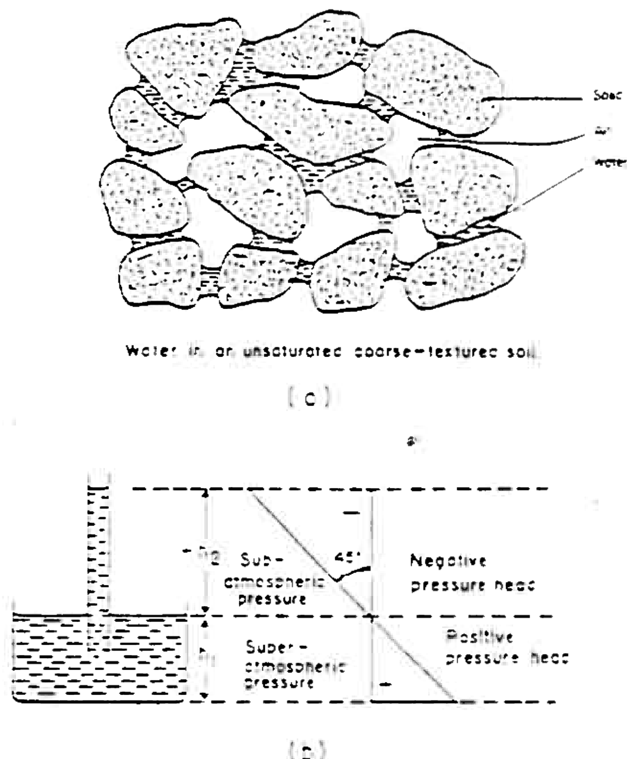

FIGURE 8 WATER IS HELD IN UNSATURATED POROUS MEDIA UNDER NEGATIVE PRESSURE HEAD (after Hillel, 1980)

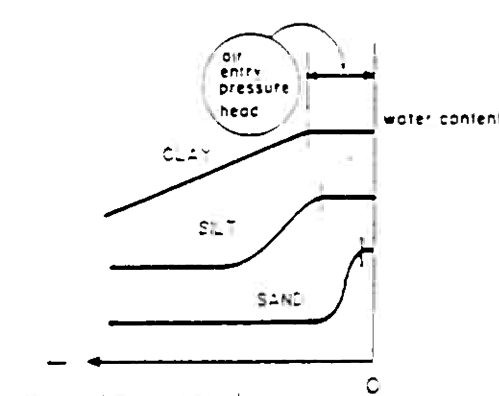

FIGURE 9 SOIL-WATER CONTENT VERSUS PRESSURE HEAD (IGNORING HYSTERESIS) (after Freeze and Cherry, 1979)
a) uniform sand
b) silty sand
c) silty clay

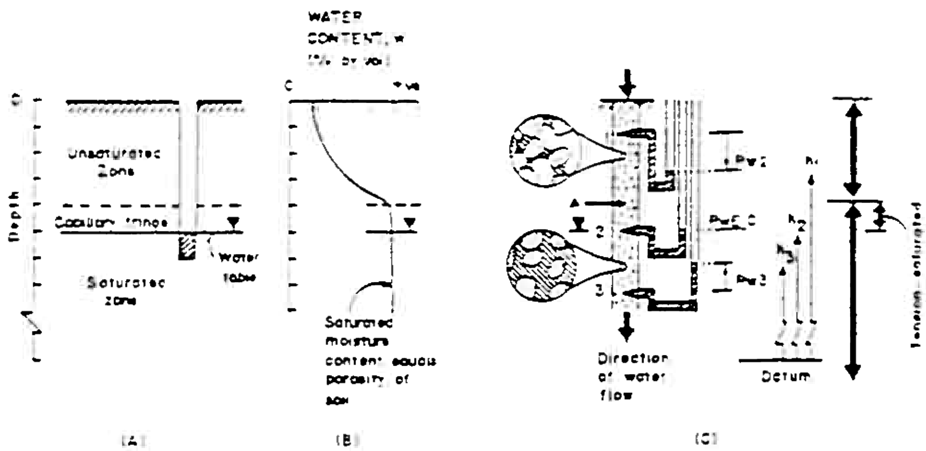

FIGURE 10 AN EXAMPLE OF WATER CONTENT, PRESSURE HEAD AND HYDRAULIC HEAD FOR THE CASE OF RECHARGE TO A WATER TABLE (after Freeze and Cherry, 1979)

a) saturated and unsaturated zones
b) profile of moisture content versus depth
c) pressure head and hydraulic-head relationships; insets: water retention under pressure heads less than (top) and greater than (bottom) atmosphere

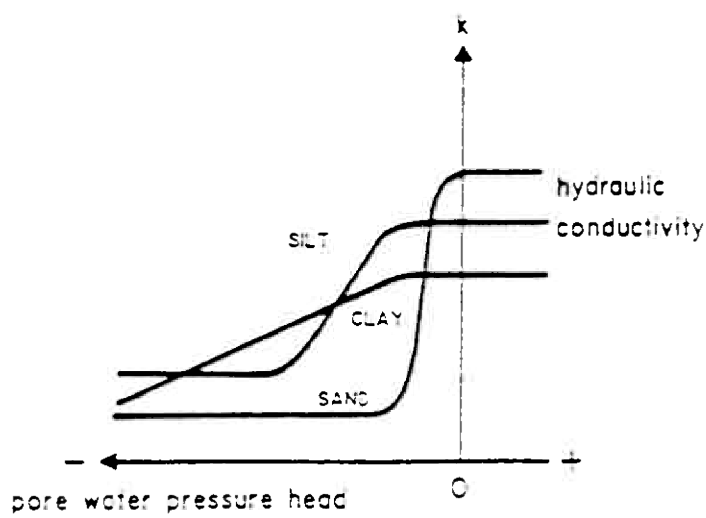

FIGURE 11 HYDRAULIC CONDUCTIVITY VERSUS PORE-WATER PRESSURE HEAD (after Freeze and Cherry, 1979)

a) uniform sand
b) silty sand
c) silty clay

soil cover). The textbooks by Hillel (1980a,b) provide a more in-depth discussion than will be presented here and are good sources of further references on these topics.

The rate at which water infiltrates into an unsaturated soil when water is applied to the soil surface at atmospheric pressure (infinitesimal ponding depth) is termed the infiltrability. If water is ponded to some depth on the surface of the soil, the infiltration rate will exceed the infiltrability. The infiltration rate will be less than the infiltrability if the water is applied at a rate lower than the infiltrability. Even when water is applied at atmospheric pressure, the infiltrability is not constant. As shown in Figure 12, the infiltrability decreases with time until, at long time, a steady infiltrability value is approached. A further complication arises in that the infiltrability depends on the initial water content of the soil as shown in Figure 13. For the purpose of introducing concepts of infiltration into a uniform soil layer, we will restrict the remainder of the section to infiltration of shallow ponded water on the soil surface. Figure 14 shows plots of hydraulic head and degree of saturation versus depth for continuous shallow ponding of water on the soil surface.

As the wetting front reaches a given depth in the soil, the hydraulic head increases with time, as does the degree of saturation. When time is relatively large, the hydraulic gradient approaches the unit gradient with the pressure head equal to zero throughout the profile and the change in hydraulic head is due to the change in elevation head only. At that time, the water flux density (specific discharge) is numerically equal to the hydraulic conductivity. Under these conditions water will not infiltrate at a rate greater than the hydraulic conductivity unless significant ponding occurs on the surface.

When water is applied to the soil surface for a relatively short period of time, the water redistributes in the soil profile as shown in Figure 15. The actual water content at a given depth and time is dependent on the soil water characteristic curve of the particular soil in question. This type of infiltration and redistribution would occur in a waste cover after a rainfall event.

The changing soil moisture distribution during drainage of a uniform soil profile that is initially saturated is shown in Figure 16. Comparing the sand, loam and clay soils, we see that fine grained soils maintain higher water contents (therefore lower air filled porosity) than coarse grained soils drained for the same period of time. This is due to both the lower conductivity and smaller pores in the clay soil.

This relationship between soil textures is also apparent in plots of volumetric water content versus time at a depth of 41 cm (arbitrary) shown in Figure 17. Figure 17 also shows the additional effect of simultaneous evaporation from the soil surface. Covers in Canada and Norway would be saturated in the spring during and immediately after snow-melt, and would likely exhibit the type of drainage shown in Figures 16 and 17.

The water content versus soil depth profile caused by evaporation at a bare soil surface is shown in Figure 18. If vegetation was present, the plant roots would be extracting water form the soil to be transpired to the atmosphere. The combined effect of evaporation and transpiration is termed evapotranspiration. The presence of vegetation results in a greater loss of water form the soil and drying of the soil to a greater depth, dependent on plant type and time of year. It also tends to disrupt drainage and this increases ponding and filtration. There are a number of other factors which influence the infiltrate rate, such as the texture and compactness of the soil surface, texture and irregularity and slope and the resulting ponding, duration and intensity of rainfall. Prediction of infiltration rates through uniform soils is therefore extremely complex. Cracks, root holes, borrows and soil variability makes this even more complex.

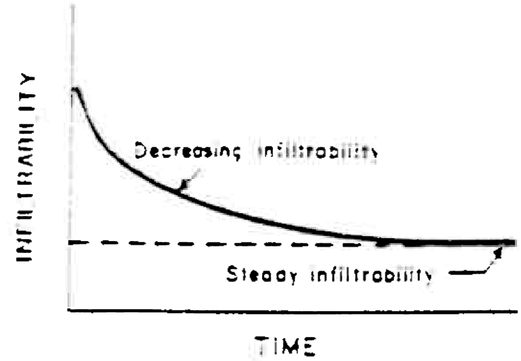

FIGURE 12 INFILTRABILITY VERSUS TIME (after Hillel, 1980b)

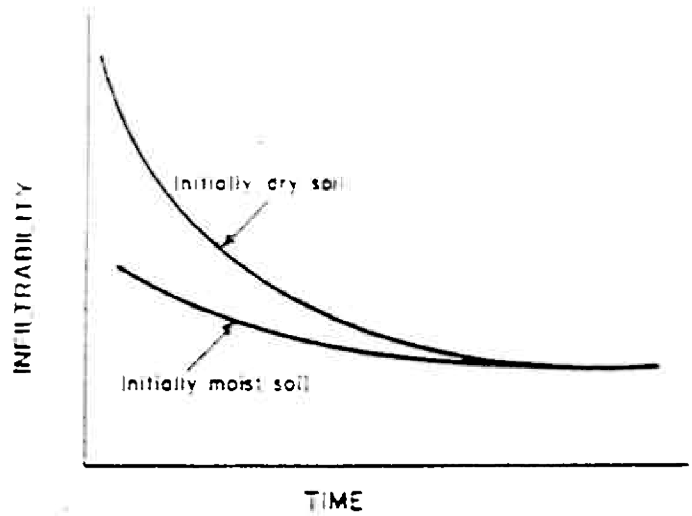

FIGURE 13 INFILTRABILITY VERSUS TIME FOR AN INITIALLY DRY AND INITIALLY WET SOIL (after Hillel, 1980b)

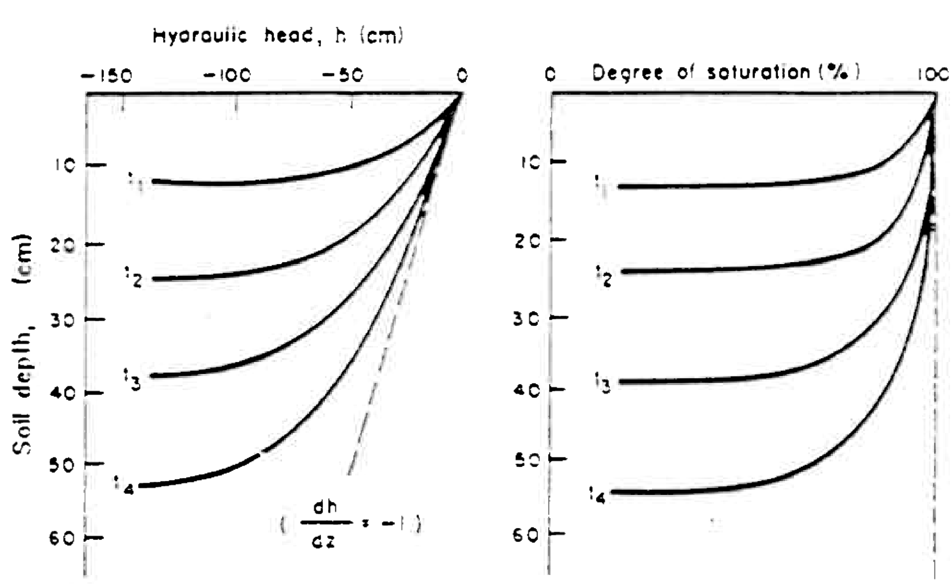

FIGURE 14 WATER CONTENT PROFILES (AT RIGHT) AND HYDRAULIC HEAD PROFILES (AT LEFT) AT DIFFERENT TIMES DURING INFILTRATION (after Hillel, 1980b)

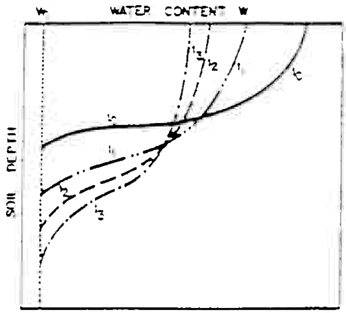

FIGURE 15 THE CHANGING MOISTURE PROFILE IN A MEDIUM-TEXTURED SOIL DURING REDISTRIBUTION FOLLOWING AN IRRIGATION. The moisture profiles shown are for T = 0, 1, 4 and 14 days after irrigation. w_i is pre-irrigation (antecedent) water content (after Hillel, 1980b).

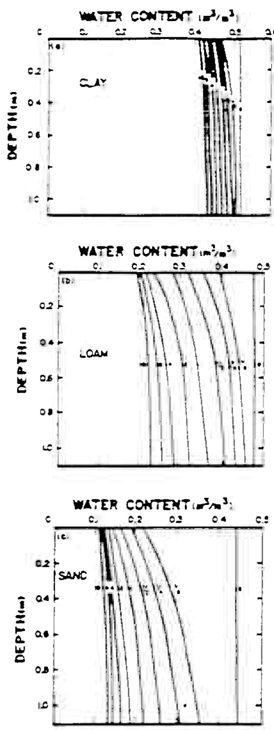

FIGURE 16 THE CHANGING SOIL MOISTURE DISTRIBUTION DURING DRAINAGE FROM INITIALLY-SATURATED UNIFORM PROFILES OF (A) CLAY, (B) LOAM, AND (C) SAND. THE NUMBERS INDICATE DURATION OF THE PROCESS (DAYS) (after Hillel, 1980b).

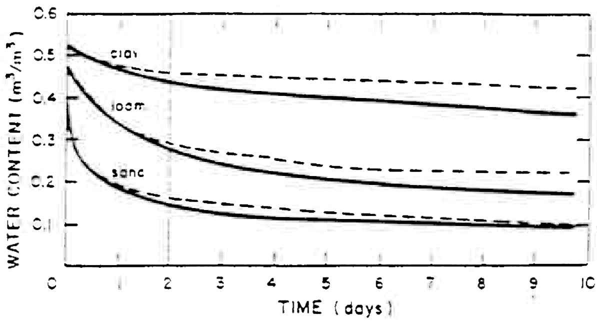

FIGURE 17 WATER CONTENT (W) AT DEPTH OF 41 CM AS FUNCTION OF TIME IN INITIALLY-SATURATED UNIFORM PROFILES OF SAND, LOAM, AND CLAY. DASHED LINES-DRAINAGE WITHOUT EVAPORATION; SOLID LINES-SIMULTANEOUS DRAINAGE AND EVAPORATION (after Hillel, 1980b)

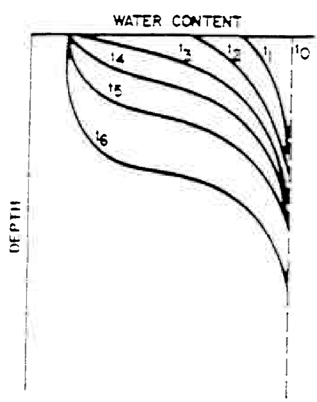

FIGURE 18 WATER CONTENT VERSUS DEPTH SHOWN FOR VARIOUS TIMES DURING EVAPORATION FROM AN INITIALLY SATURATED SOIL

4.2 FLOW THROUGH LAYERED SOILS

By placing layers of coarse and fine grained soils above each other it is possible to develop infiltration barriers and drainage layers. These complex engineered covers are reviewed in greater detail in Section 10 of this report.

A concept for a complex layered soil cover for sulphidic mine water deposits is described by Magnusson and Rasmuson (1983). The waste is covered with, from below, a coarse-grained layer, a fine-grained layer, another coarse-grained layer and a till layer (Figure 19). This cover system includes capillary barriers which are formed at the interfaces between coarse-textured and fine-textured soils. The capillary barrier concept involves the principles of unsaturated flow between materials of different textures. If a fine-textured soil overlies a coarse-textured soil and both materials are initially dry, water infiltrating from the surface will not move into the coarse-textured soil until the capillary potential in the fine-textured soil approaches zero (i.e., until the fine-textured soil is at, or near, saturation). The capillary barrier can also be placed above the fine-grained barrier layer thereby preventing capillary water transport upwards from the fine-textured soil.

Drying of the fine-grained layer, caused by capillary water transport upwards and downwards from the fine-grained layer, is prevented. The lower coarse-grained layer may also form a capillary barrier at the top of the waste sand, thereby preventing capillary transport of contaminated water from the waste sand to the soil cover. The upper coarse-grained layer may form a capillary barrier at the bottom of the till layer. Furthermore, the upper coarse-grained layer may act as a drainage layer resulting in an increased lateral water drainage and hence a decrease of infiltration.

A conceptual moisture content profile in such a cover system is shown in Figure 20. This design of the cover gives very good possibilities for the reduction of both the oxygen transport and the water percolation through the cover. Fore economical reasons, however, a simpler soil cover would be preferred.

4.3 MODELLING INFILTRATION

Modelling of water flow in unsaturated soils is complicated by the non-linear nature of the interrelationships of the soil water properties, and by the fact that steady-state flow conditions are seldom achieved in the field.

Alternative methods for the modelling of infiltration has been reviewed by Steffen, Robertson and Kirsten, 1986a.

The Hydrological Evaluation of Landfill Performance (HELP) model developed by the U.S. Army Corps of Engineers is an extremely useful tool for a first evaluation of the relative benefits of alternative cover layers with and without vegetation cover. Figure 21 indicates the results of the evaluation of a number of alternative cover combinations, using the HELP model, for the acid generating tailings at Elliot Lake (Steffen, Robertson and Kirsten, 1987). The large potential differences in infiltration between the different cover types is apparent. The HELP model assumes saturated soil conditions at the commencement of precipitation. Clearly this is an approximation leading to considerable inaccuracies under certain circumstances.

The TRUST model was developed at Lawrence Berkeley Laboratories and is well described by Reisenauer et al, 1982. This suite of programs has been extended to allow more effective and convenient applications to cover design by McKeon et al, 1983. It was used by Collin, 1987, for the evaluation of the effectiveness of the complex cover shown in Figure 19. While the program allows for partially saturated flow the determination of input variables to adequately allow for the seasonal and extreme variations found in practice still renders the accuracy of the answers questionable.

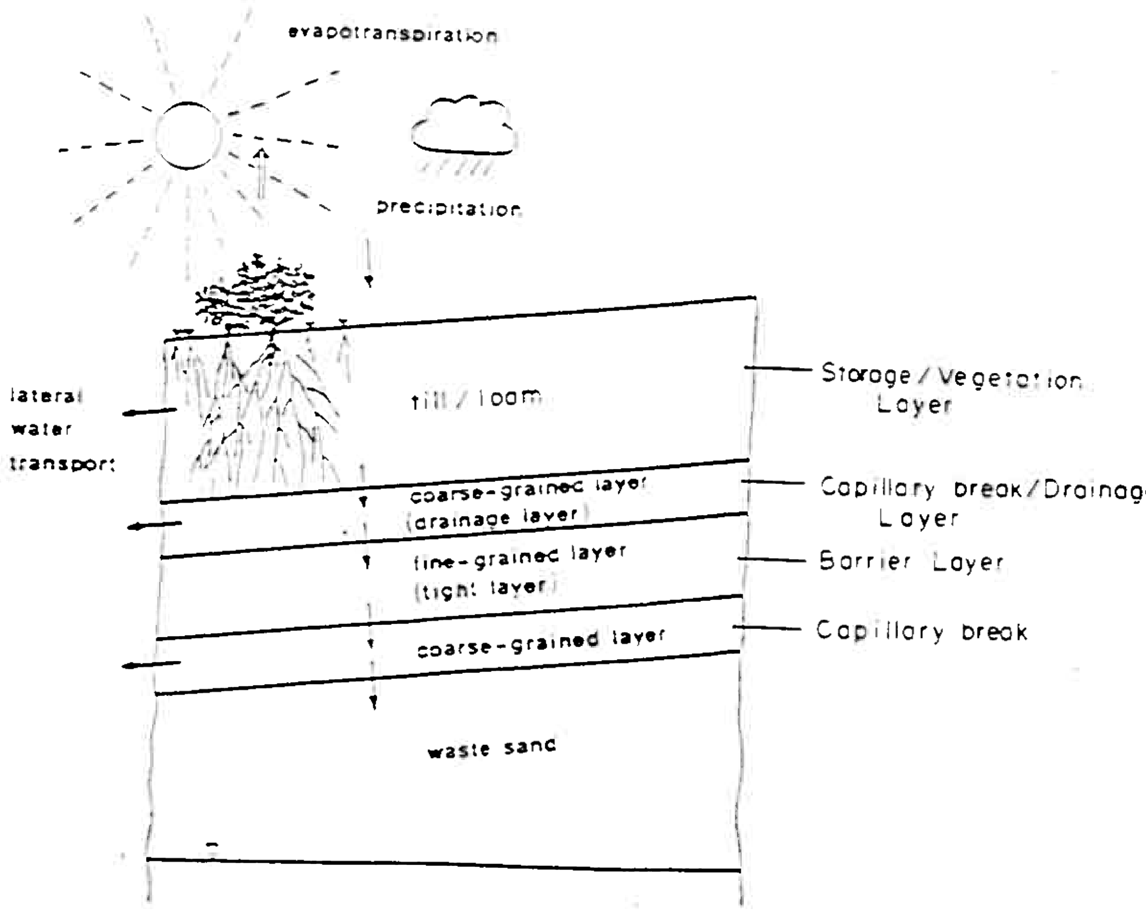

Fig. 19 Concept for a Soil Cover for Reactive Tailings

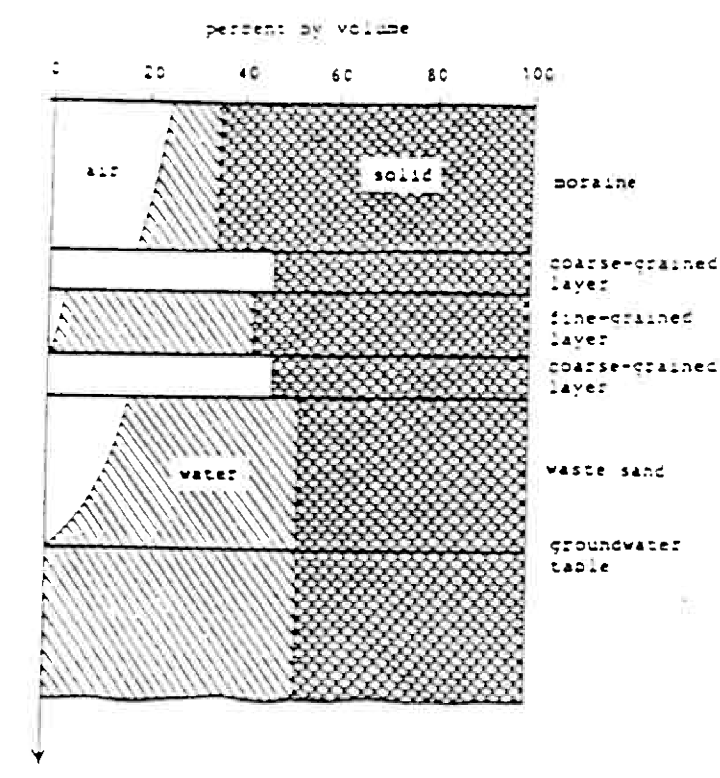

Fig. 20 Principal Profile Through the Soil Cover and Waste Sand Deposit

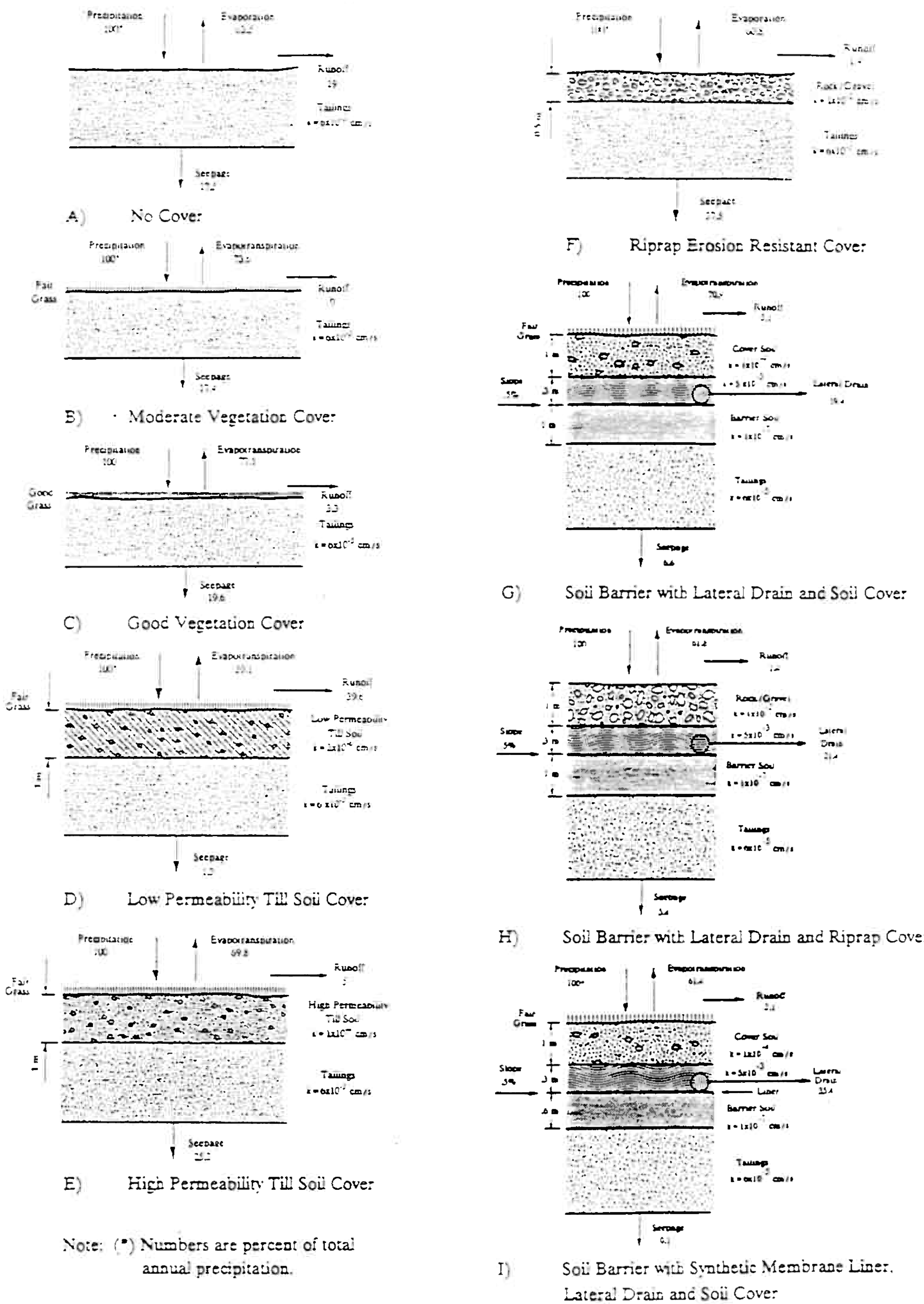

Figure 21 Infiltration Barriers for Engineered Cap - Elliot Lake

A model that simulates unsaturated conditions, called SOILMOIST, has been developed for the US Uranium Mill Tailings Remedial Action Program and this may yield more realistic results under certain circumstances.

A limitation that all the models suffer from is the inability to anticipate and include in the models the effects of layer disruptions such as:

- settlement causing drainage disruption and ponding
- cracking due to settlement or dessication
- root holes
- burrowing channels formed by insects, animals and man
- frost action effecting permeability and drainage
- erosion
- vegetation disruption of drainage
- clogging of drains due to frost or root action

These disruptions may be more severe with complex layered covers than with simple homogeneous covers.

5.0 OXYGEN DIFFUSION

There are a number of potential mechanisms contributing to the mass transport of oxygen through an unsaturated porous medium. These processes can generally be classified as either diffusive or advective transfer. Advective and diffusive transfer occur both in the gas phase and as a dissolved constituent in porewater, and thus transport can occur in both phases.

5.1 ADVECTIVE TRANSPORT WITH PERCOLATING WATER

The solubility of oxygen in water in equilibrium with air is 0.45 mole O_2/m^3 at 0°C and 0.35 mole/m^3 at 10°C (Landolt-Bornstein, 1968). Assuming a percolation of 500 mm/year, less than 0.225 moles O_2/m^2/year may be transported into the waste dump by this mechanism. This amount of oxygen may oxidize 8 g pyrite/m^2, year (Collin, 1987).

5.2 ADVECTIVE TRANSPORT IN AIR

Changes in the atmospheric pressure cause a considerable transport of oxygen into the upper parts of the ground and is an important mechanism in soil aeration. The effect on covered mine tailings is investigated below. The diurnal changes in the atmospheric pressure is less than 4% (4 kPa) (Glinski and Stepniewski, 1983). The oxygen transport rate is limited either by the accumulation of air in the waste dump (rapid pressure equilibration) or by the transport capacity in the soil cover. The maximum oxygen transport rates for these two limitation mechanisms are calculated below.

Rapid Pressure Equilibration

Assuming rapid pressure equilibration in the dump, the air flow into the dump is given by the pressure increase and the volume of gas in the dump.

For a waste sand dump with porosity 0.5, degree of saturation 0.70 and 1 m depth to the groundwater table, a 4 kPa pressure increase in the atmosphere gives a transport of $5.4 \cdot 10^{-2}$ moles O_2/m^2 into the dump (Collin, 1987). The oxygen concentration in the air flowing into the dump is assumed to be equal to the concentration in the atmosphere. Assuming such an atmospheric pressure increase every day and neglecting pressure decrease leads to a transport of 19.5 moles O_2/m^2, year into the dump. This amount of oxygen may oxidize 670 g pyrite/m^2, year. A lower degree of saturation gives a higher oxidation rate, for example S=0.5 gives 1 100 g pyrite/m^2, year. For large coarse rock waste dumps the depth to the groundwater table would be much greater resulting in much greater oxygen transport rates. Collins, 1987, notes that this rough calculation gives an overestimation of the oxygen transport because:

1) The maximum pressure increase is used.
2) Pressure decrease is neglected.
3) The maximum oxygen concentration is used for the air flowing into the dump.

4) The resistance against air flow is neglected.

In addition to advective flow due to pressure changes there is flow due to temperature differences between the dump materials and the ambient air. Such temperature differences change diurnally and seasonally. Increased temperature differences may result, particularly along an air flow path in a coarse hard rock waste dump, as a result of the rapid exothermic reactions which may occur. No estimation of the oxygen transfer due to such convection is known to the authors. It is anticipated that advective flow of this type may result in considerably more oxygen entry to a coarse rock waste dump than is estimated for changes in atmospheric pressure.

It is self evident that the placement of an intact cover layer over coarse rock waste will have a large effect on the potential for advective oxygen transport through such a waste dump. The effectiveness is reduced for a tailings where the lower permeability of the tailings act as a natural inhibiter of such advective flow.

Holes, which may develop through a cover over coarse waste, will provide passage ways for large increases in such advective flows. Thus the continuity of a cover is very important to the long term control of oxygen entry.

5.3 DIFFUSION IN WATER

The diffusion rate for oxygen in water is extremely slow, of the order of 0.02 to 0.05 meters squared per year. Thus, in the absence of convective transport, the oxygen transport rate through water is sufficiently slow to not be a concern for acid generation. This aspect is considered further in Section 8 of this report.

5.4 DIFFUSION IN AIR

The following general description has been summarized from Siwik et al, 1987.

In natural soils that are well-drained, it is expected that most of the gaseous exchange between soil gas and the atmosphere is by gaseous diffusion (Evans, 1965; Kimball and Lemon, 1971; Kirkham and Powers, 1972). Under other conditions, where pore spaces are almost entirely filled with water (i.e., almost saturated), the flux of oxygen will be low and limited by the lower diffusivity in water. Nicholson (1984) found that the infiltration of dissolved oxygen could become an important or dominant component of the downward oxygen flux if diffusion through the pore network is sufficiently restricted by high water content. In either case, calculation can be performed to determine the approximate relative importance of each of these mechanisms to oxygen transport.

A number of models have been used to describe oxygen transport through a porous medium. The common basis for these is the inclusion of a diffusion term to describe oxygen movement. Models have been developed to describe oxygen transport into reactive tailings (Nicholson, 1984; SENES-BEAK, 1986) and into ore leaching piles (Cathles and Apps, 1975). The emphasis on transport mechanisms differs for each of these cases, but the fundamental principles are the same. The following discussion introduces a relatively comprehensive model for oxygen transport that could be applied to tailings covers and/or laboratory experiments designed to determine the behaviour of the potential cover materials.

Oxygen transport is assumed to occur as a result of diffusion through the gas phase and as dissolved water infiltrating (at rate q). The differential equation describing oxygen movement can be written as:

$$D_e E \frac{\partial^2 C}{\partial x^2} - \frac{q \partial C^*}{\partial x} = E \frac{\partial C}{\partial t} + \rho_w n \frac{\partial C^*}{\partial t} + G$$

where:

- D_e is the effective diffusion coefficient of oxygen,
- E is the air-filled porosity,
- q is the infiltration rate of water,
- C is the concentration of oxygen in the gas phase,
- C^* is the concentration of dissolved oxygen in the water,
- x is depth,
- w is the density of water,
- G is the reaction term for oxygen (possibly a function of C),
- n is the porosity, and
- t is time.

Assuming that oxygen dissolution in water is an equilibrium process, then the expression can be simplified to:

$$D_e E \frac{\partial^2 C}{\partial x^2} - \frac{q \partial C}{K_w \partial x} - G = E + \left[\frac{\rho_w n}{K_w}\right] \frac{\partial C}{\partial t}$$

in which Kw is the dissolution constant for oxygen in water (Henry's Law Constant). This equation is simplified at steady-state when the right-hand side is equal to zero.

This equation describes the oxygen movement into tailings or other material which consumes oxygen. When oxygen migrates through a cover material that does not consume oxygen, G = 0 and an appropriate solution can be found.

Oxygen movement within the porous medium is controlled by both the infiltration of water containing dissolved oxygen and the diffusion of oxygen in the gas phase and water phase of the pores. It is possible to estimate both the maximum and minimum rates of oxygen movement through a cover material on tailings. The flux of oxygen through such a cover can be estimated for both diffusion and advection separately to show the relative influence of each process.

Oxygen flux by advection (entering as dissolved oxygen with the natural infiltration) in water is calculated in the following way.

$$Fa = qC^*$$

where: Fa is the advective flux of oxygen (in mass per unit area per unit time),
q is the rate of infiltration through the cover (length per time), and
C^* is the concentration of dissolved oxygen in the infiltrating water (mass per unit volume).

The average annual rate of infiltration is usually known within reasonable limits. For instance, in northern Ontario, infiltration in a vegetated area would likely be on the order of 0.01 to 0.25 m a^{-1} (i.e., m^3 m^{-2} a^{-1}). Higher rates of infiltration are, of course, possible, but an absolute upper maximum equal to the amount of annual precipitation must hold. With a maximum precipitation of about 10.0 m a^{-1}, it is not likely that annual cumulative infiltration will be greater than about 0.5 m a^{-1}. For the purposes of the following calculation, the range of infiltration values is assumed to be 0.01 to 0.10 m a^{-1}. The value of C^* (dissolved oxygen concentration) is constant for a specified temperature. Therefore, the flux of oxygen through cover, due to transport in water, can be calculated within at least an order of magnitude range.

When oxygen is consumed rapidly in the underlying tailings, oxygen diffusion through a cover material can be described by steady-state diffusion (Nicholson et al., 1987). When the tailings consume oxygen at a moderate to high rate such as that expected in reactive tailings, then the flux relationship reduces to:

$$F_d = (D_e/L)\, C_o$$

where: F_d is the flux of oxygen,
D_e is the effective diffusion coefficient of oxygen in gas-filled pore spaces,
L is the thickness of the cover, and
C_o is the concentration of oxygen in the atmosphere.

This relationship represents Fick's first law when the concentration of oxygen at the bottom of the cover is zero. It is convenient to take the ratio of D_e/L as a mass transfer term that varies over a wide range of values. The equation can then be used to show the numerical relationship between the diffusive flux F_d and the mass transfer value (D_e/L). It is known that the value of D_e will vary form about 1×10^{2} to 1×10^{-2} m^2 a^{-1} for diffusion through air to diffusion through water, respectively. The effective diffusion coefficient in a water-saturated porous medium may be somewhat lower than 1×10^{-2} m^2 a^{-1}. The value of L will not likely be more than about 2 m, and will probably be close to 1 m for practical purposes. Therefore, the range of values for D_e/L are close to the range of values for D_e alone.

The relationship between the diffusive flux and the mass transfer term is shown graphically in Figure 22. In this figure, the flux of oxygen has been converted to the equivalent flux of acid that would be produced by reaction of the oxygen with pyrite. This acid flux represents a total available acidity (as H_2SO_4) that would be produced, and is based on the stoichiometric relationship between oxygen and pyrite in the production of acid. This figure shows that the maximum flux of oxygen is equivalent to about 10 kg of H_2SO_4 per m_2 of tailings per year, when oxygen diffuses through a very dry cover material. When the material has a greater moisture content, and diffusion is the controlling process, values are considerably reduced. The resulting flux can be as low as 1 g or less of H_2SO_4 per m^2 per year. However, on the same diagram, it can be seen that the advective flux of oxygen occurs within the range of about 1 to 10 g of H_2SO_4 per m^2 per year. Because the infiltration rate is independent of diffusion, the flux of dissolved oxygen will represent the maximum limit for oxygen transport through the cover.

This lower limit to oxygen flux is not practically affected by low permeability cover materials. The lower limit considers an infiltration rate of about 0.01 m a^{-1}, which is equivalent to the flow under unit gradient through a material having a hydraulic conductivity of about 3×10^{-10} m s^{-1}. This value would be typical of a clay till or marine clay (Freeze and Cherry, 1979), and it would be difficult to reproduce this property under field conditions. Therefore, the lower limit for the flux of dissolved oxygen will depend on the net infiltration rate and, therefore, will not likely be much lower than that shown in Figure 22.

As noted above, covers are effective but only eliminate acid generation when the water table rises into the cover. Figure 23 developed from the RATAP model (SENES et al, 1986, Steffen, Robertson and Kirsten, 1987) illustrates the effects of depth of earthen cover (glacial till) on a typical Elliot Lake tailings area for various depths of unsaturated tailings. As shown on the Figure, tailing with a 500 mm unsaturated zone would initially produce about 50% as much acid as those with a 2.5 metre oxidizing zone. Two metres of the cover over these tailings would reduce the oxidation rates by 60 and 70 percent respectively. Greater reductions would occur if the water table was to rise as a result of cover addition. It is worthy to note that the depth of the unsaturated zone is often not a major factor in determining the initial acid generation rate. This is because oxygen depletion occurs in the near surface zone thus being the rate limiting factor. However, the deeper the unsaturated zone, the longer acid generation will continue. Eventually, virtually all pyrite would be oxidized from the unsaturated zone.

The effect on diffusion of a small air filled gap (crack) in a cover layer has been investigated by Collin, 1987. Her findings are summarized in Figure 24. This indicates the dramatic effect of small imperfections in the cover on oxygen entry, and reinforces the conclusion that oxygen exclusion is dependent on having a cover which is maintained at a high moisture content and is essentially crack and hole free.

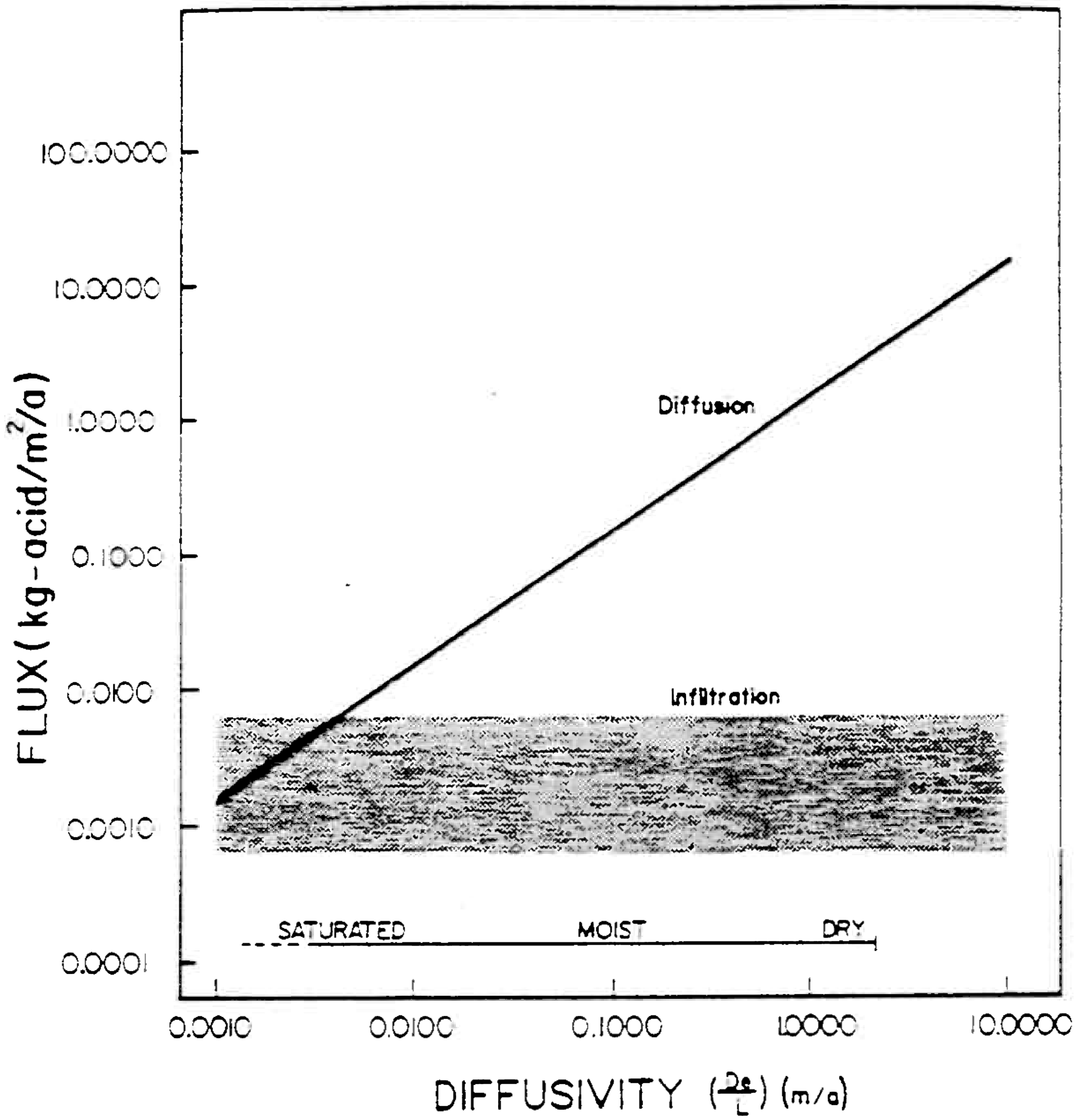

FIGURE 22 OXYGEN FLUX EXPRESSED AS H_2SO_4 ACID PRODUCED AS A FUNCTION OF ANNUAL INFILTRATION AND THE EFFECTIVE DIFFUSIVITY OF THE COVER

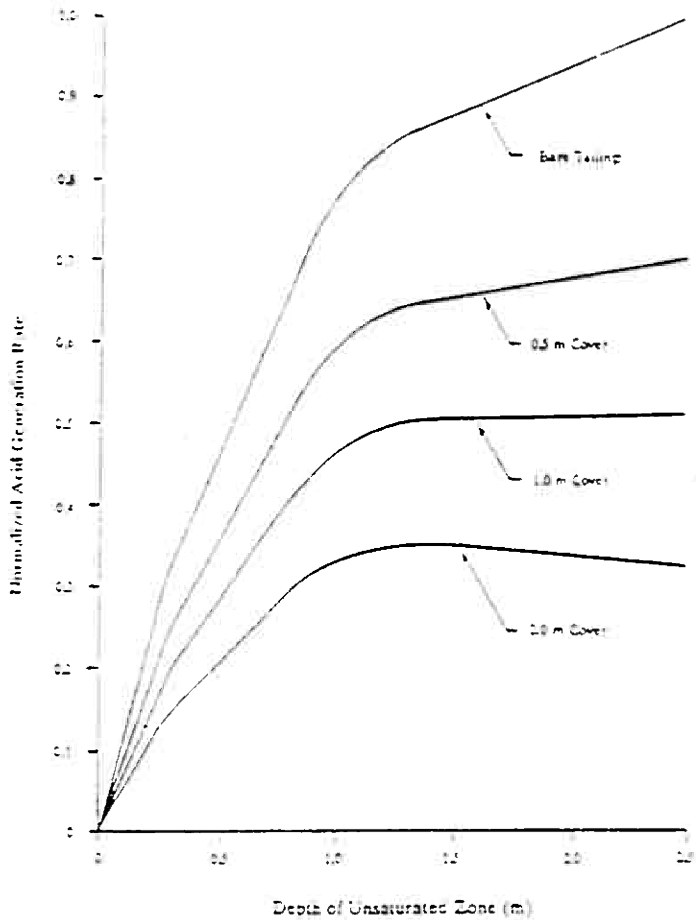

Figure 23 The Effects of Till Cover on Acid Generation Rates

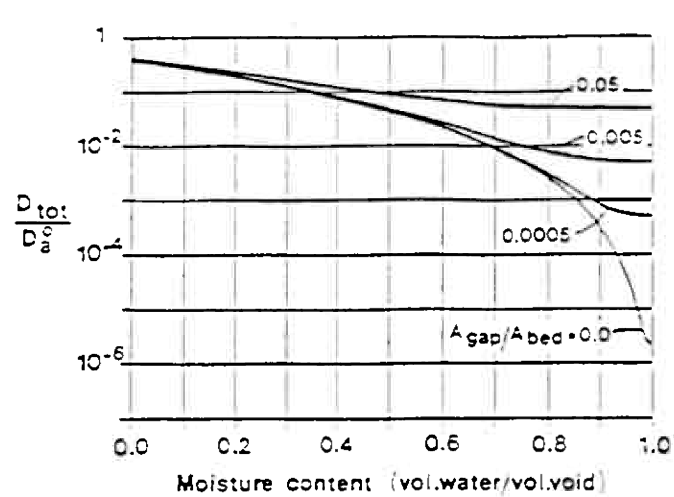

Figure 24 The influence of a thin gas-filled gap through the sample on the measured effective diffusivity. A_{gap}/A_{bed} = 0.05, 0.005, 0.0005, 0.

5.5 PARAMETER INTERACTION

This section reviews some of the functional relationships between important parameters to enable the the effectiveness of different cover materials to be assessed. It draws mainly from the writings of Siwik et al. 1987.

The water content within the cover will be related to the pressure head distribution and grain size distribution characteristics for a specified cover material. When the water table is relatively shallow, the pressure head will represent the distance between a specified elevation in the cover and the water table. The typical relationship between water content and pressure head has been shown in Figure 9. This relationship can be determined in the laboratory for a specified material. The relationship is hysteretic such that different values of moisture content are obtained at the same pressure head value depending on whether the material is going through a wetting or a drying cycle. Hysteresis does not affect the upper end of the curve where the medium is saturated under low to moderate pressure head values. In any cover layer design, therefore, it would be desirable to use as fine grained a material as possible to provide a large range of pressure heads for which the material remains saturated or near saturated.

The hydraulic conductivity of a medium is also a function fo the water content as discussed earlier. This relationship is shown graphically in Figure 25. This relationship is not hysteretic.

The effective diffusion coefficient for oxygen in a cover layer can be shown to be a function of the water content. This relationship is shown graphically for a sample of tailings from Elliot Lake in Figure 26. It is evident that the value of D_e is quite sensitive to the moisture content (and therefore air-filled porosity), especially when the medium approaches saturation. The relationships between D_e and the air-filled porosity is given by the following empirical equation (Troeh, et al, 1982):

$$D = ((E-a)/(1-a))^b$$

in which a and b are experimental coefficients. It is necessary to determine the value of the coefficients for porous media with different textures.

Selection of Critical Parameters

The overall selection of critical parameters should ideally identify a small number of controlling variables. Key parameters that control the system overall, and those that can be related directly to field and/or laboratory measurements, are desirable.

The key parameters of interest for oxygen transport through a cover are; the volumetric water content (w) the soil water pressure head (p_w), the effective gas diffusion coefficient (D_e), and the saturated hydraulic conductivity (k_s). Of these, the volumetric water content is the least difficult to measure. The three other parameters are a function of water content.

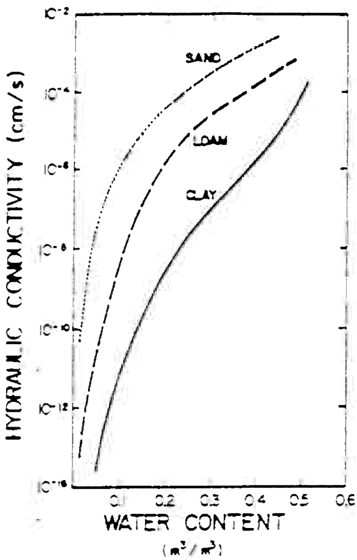

FIGURE 25 HYDRAULIC CONDUCTIVITY AS A FUNCTION OF WATER CONTENT

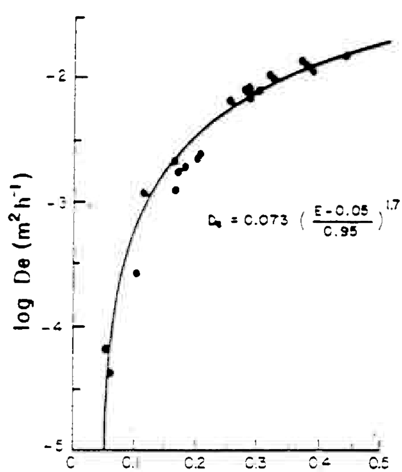

FIGURE 26 EFFECTIVE DIFFUSION COEFFICIENT AS A FUNCTION OF AIR-FILLED POROSITY FOR A SAMPLE OF ELLIOT LAKE URANIUM TAILINGS (after Reardon and Moddle, 1985)

The interdependence of the water content and pressure head is expressed by the soil water characteristic curve. Each cover material will have a unique soil water characteristic curve. The water content and associated pressure head can be measured in the field and monitored and controlled in the laboratory. Selection of an appropriate cover material will be based on the ability of the material to withstand significant negative pressure head values while remaining at or near saturation (i.e., near or above the air entry value). Because the ideal cover should remain near saturation to reduce diffusion of oxygen, the water content at a number of selected pressure head values should be used as criteria for ranking proposed cover materials.

Water content is also known to be the key parameter controlling the effective gas diffusion coefficient. Because the gas diffuses primarily through continuous gas-filled pores, higher water content results in lower effective gas diffusion coefficients. The effective diffusivity of the cover can be predicted from known values of water content after experimentally obtaining the two parameters that are related to grain size and texture of the specific material.

The saturated hydraulic conductivity is a critical parameter for the control of water movement (with dissolved oxygen) through potential cover materials. However, it is of lesser importance than the three parameters listed above for two important reasons. First, the mass of oxygen transported in dissolved form in water is usually small relative to that which would diffuse through the cover if it were dry. Second, only soils with large percentages of clay can significantly impede natural infiltration. If clay soils are disregarded for practical reasons, such as lack of availability, then it is fair to assume that most of the natural rainfall and snowmelt, less natural evapotranspiration, will infiltrate the cover material.

6.0 PLACEMENT DIFFICULTIES

The establishment of covers on mine waste and tailings is complicated by the difficulties of access, trafficability and stability of the surfaces onto which the cover is to be placed. These difficulties often render a particular cover type impractical or prohibitively expensive. Placement of some cover types requires access of wheeled vehicles working on fairly flat surfaces (asphalt covers), others require careful bed preparation and moderate slopes (synthetic membranes) and others a firm surface against which to compact (clay layers).

6.1 WASTE DUMP SURFACES

Rock waste surfaces are conveniently subdivided into the dump surface and the dump slopes; with different conditions applying to each.

Dump Surface

The upper surface of a rock waste dump is usually readily accessible, trafficable and nearly flat. The placement of any type of surface cover, except a water cover, is usually not difficult.

During dump development the material on the upper surface of dumps placed by trucking is often broken down and compacted under the wheel traffic of the dump trucks. This results in a fairly compact lower permeability upper surface. This surface reduces infiltration and ponding is often experienced on such surfaces. Despite the initial coarse nature of the material in such dumps it may be necessary to install a suction breaking layer to prevent downward suction on low permeability cover layers.

The dumps are subject to long term consolidation and settlement under the self weight of the dump and as the dump rock weathers. These settlements are large (a few percent of the dump height), and uneven, reflecting the natural variation of the waste rock and dumping procedures. Differential settlements result in disruption of the drainage pattern on the dump surface and cracking of cover materials. Settlement and crack patterns are often such that drainage is towards cracks resulting in considerably increased infiltration.

Dump Slopes

Dump slopes are usually placed at their angle of repose. At this angle slopes are inaccessible, untrafficable and marginally stable. Cover placement on such steep slopes is essentially impractical. Crest dumping of cover materials has been attempted at some sites creating uncompacted (permeable) uneven covers of questionable stability.

For dump slopes to be accessible it is necessary to first reslope. At a slope of 3 horizontal to 1 vertical (3:1), the slopes are trafficable by tracked vehicles and it is possible to place soil type cover materials. At this slope it is possible to also place synthetic membrane liners but the stability of cover layers over such membranes is questionable. The cost of resloping large dumps to 3:1 is very large, requiring large expenditures of dozer time, unless the resloping had been planned for and the dumps constructed with a staggered dump slope.

At its natural angle of repose a slope is just stable. Weathering of the dump materials, or increases in the water table with time, may result in slope failure (HMSO, 1967, Campbell and Shaw, 1980, Bishop, 1973). Such slope failure may not be catastrophic but take the form of slope creep. Either will result in the disruption of any cover layers.

Dump slopes are subject to the same concerns regarding differential settlement as are the dump surfaces.

Erosion on the steep dump slopes is a major long term concern. This is considered in greater detail in Section 7.

6.2 TAILINGS SURFACES

The access, trafficability and stability of a tailings surface varies tremendously, depending on the nature and age of the tailings deposit and location on the impoundment surface. The effects of placement conditions on the properties of tailings deposits has been reviewed by Robertson, 1987b. The difficulties of placing covers on tailings and the deformations of tailings surfaces are reviewed in Steffen, Robertson and Kirsten, 1986a.

Wet unconsolidated tailings always represent difficult access conditions. Access improves as the tailings are drained and consolidate. Where tailings have been spigotted onto beaches, the sand fraction is deposited nearest the spigots and drain more freely than the slimes which accumulate near and in the pond. Drained sandy beaches may be trafficable within days of deposition while pond areas may never achieve this condition. Thus it is possible to place and compact covers on the beach areas with little preparation. In the pond areas it may be necessary to apply drainage measures to remove free and near surface water, and use geofabrics onto the slimes followed by thin layers of the cover. The authors have successfully used this technique to place a cover over slimes which could not be walked on at the start of cover placement. Covers may also be placed during winter when freezing conditions allow access, as was done for the cover placement over wet tailings at the Beaverlodge mine (Steffen Robertson and Kirsten, 1983, Eldorado Resources, 1986).

To prevent capillary suctions in covers it may be necessary to utilize capillary barriers over tailings fines and slimes as discussed in Sections 5 and 10.

Prior to placing covers on tailings it is necessary to first develop a tailings surface which has an adequate slope and drainage pattern. Requirements are reviewed in Steffen, Robertson and Kirsten, 1987. Much can be done to achieve such slopes by adopting an appropriate tailings placement and management method (Robertson, 1987b). Reshaping of tailings surfaces after closure, with earth moving equipment, may be difficult and prohibitively costly. Trafficability of dewatered tailings are reviewed by Robertson et al. 1982.

After closure, tailings continue to consolidate and settle as a result of dissipation of pore pressures and thawing of included ice. The effects have been reviewed for Canadian tailings impoundments by Steffen, Robertson and Kirsten, 1986a and 1987. These settlements can be a substantial portion of the total tailings depth and result is disruption of the drainage pattern, leading to extensive ponding on the tailings surface, and cracking of covers.

6.3 ROCK SURFACES

The placement of covers on steep rock surfaces, such as pit walls, poses a particular problem. Two approaches can be used.

The first requires the construction of thick self supporting covers (Robertson et al. this conference).

The second requires adherence of the cover to the rock face and relies on the rock face to support the cover. The use of gunite or shotcrete methods is appropriate for the second. Both asphalt and concrete materials can be considered. Because of the corrosive nature of AMD to cement and steel, the use of synthetic fibres and silica fume concrete is appropriate.

7.0 LONG TERM STABILITY

Waste facilities can usually be designed to adequately achieve the design and abandonment plan objectives at the time of close-out. However, in the long term, the waste deposit and its control structures are subject to two classes of disruptive forces.

1) Short duration extreme events such as floods, fires, earthquakes and tornadoes which apply forces to the structures in excess of values for which they were originally designed.

2) The slow but perpetual action of forces which bring about deterioration, such as water and wind erosion, frost action, the weathering and chemical change of wastes, covers liners or structural materials, and intrusion by roots, animals and man.

Under the action of these forces, failure is inevitable within geologic time. Periodic maintenance can serve to repair the ravages of many of the perpetual forces. For a well designed facility the required interval between maintenance may be decades or centuries, imposing minimal cost on future generations. Remedial measures may be required after extreme events. The level of periodic maintenance and risk of remedial action which may be appropriate to pass on to future generations must be balanced by the present value to society of the resource.

During their evaluation of the long term stability of uranium tailings structures and surfaces, Steffen Robertson and Kirsten (1986a, 1986b), developed a summary of potential causes and associated risk of long term instability of 'generic' Canadian tailings impoundments surfaces. The conclusions from these evaluations are summarized in Table 3.

Extreme Events

Because of the long period of interest, the likelihood of extreme events is proportionately large. This likelihood is determined from probability:frequency relationships based on the historical record of events.

The period of record is for an interval in which a particular climate applied. Evidence suggests (McInnis, 1985), that climatic cycles occur regularly, varying from relatively minor 30 year cycles to major glacial and inter glacial cycles of tens to hundreds of thousands of years. The weather in several years, or a few centuries from now is unknown (Hare and Thomas, 1974). While it is questionable whether man has altered world climate to date, the potential for material alteration increases and his influence may rival or overrule natural climatic changes in the early twenty-first century or sooner. While the nature of the changes are unknown, it is certain that the extremes to which the impoundments will be subjected will be greater than that predicted from existing records.

TABLE 3

SUMMARY OF EVALUATION OF POTENTIAL INSTABILITY OF IMPOUNDMENT SURFACES

COVER TYPE			MODE OF INSTABILITY							
			Sheet Erosion	Gully Erosion	Wind Erosion	Salt Migration	Differential Settlement	Infiltration	Frost Heave	Radon Emanation
A.	Generic Elliot Lake Site									
	(i)	no cover action	s	s	s	M	m	s	m	s
	(ii)	contour and revegetate*	m-M	s	M	m-M	m	M	m	M
	(iii)	1 m pyrite free tailings and vegetation**	m-M	s	m-M	m-M	m	M	m	m
	(iv)	thin soil cover (< 0.6 m)	m-M	s	m-M	m-M	m	M	m	m
	(v)	thin rock cover	m	M	m	M	m	s	m	m
	(vi)	think water cover	n	n	M-s	n	m	s	m	m
	(vii)	thick soil cover (> 2.0 m)	m	M	m	m	m	M	m	m
	(viii)	thick zoned cover	m	M	m	m	M	M	m	m
	(ix)	thick water cover	n	n	n	n	m	s	n	m
B.	Generic Saskatchewan WT Site									
	(i)	no cover action	s	s	s	m	m	s	s	s
	(ii)	contour and revegetate	m	s	m	m	m	M	M	M
	(iii)	thin soil cover	m	s	m	m	m	M	M	m
	(iv)	thin rock cover	m	M	m	m	m	s	M	M
	(v)	thick soil cover	m	M	m	m	m	M	m	m
	(vi)	thick zoned cover	m	m	m	m	M	M	m	m
C.	Generic Saskatchewan SD Site									
	(i)	no cover action	s	s	s	s	m	M	s	s
	(ii)	contour and revegetate***	s	s	s	s	m	M	s	s
	(iii)	thin soil cover	M	s	M	M	m	M	s	M
	(iv)	thin rock cover	m	M	m	s	m	M	s	s
	(v)	thick soil cover	m	M	m	m	m	M	m	m
	(vi)	thick zoned cover	m	M	m	m	M	M	m	m

KEY:

1. Potential Instability Code

n = none
m = minor
M = moderate
s = severe
N = not applicable

2 x = cost of producing pyrite free tailings

REMARKS

* Stability of vegetation questioned. Evaluation assumes success.
** Feasibility of producing pyrite free tailings unknown.
*** Direct vegetation presumed inappropriate. Evaluation assumes failure.

High Precipitation and Floods

Large precipitation events represent one of the most likely causes of waste impoundment failure. Failure during such events are also likely to result in large losses of tailings or waste fines to the environment.

Methods for the estimation of high precipitation events, and for the calculation of the resulting flood flows are well developed for dam design purposes. Current practice is usually to design operating facilities to withstand the one in two hundred year precipitation event. Design for the Probable Maximum Precipitation (PMP) is probably more appropriate for long term conditions.

Earthquakes

Dynamic loads, due to earthquakes, may result in the liquefaction of low density saturated tailings or uncompacted, saturated portions of granular embankments or embankment foundation materials. Failure of the El Cobre tailings dam in Chile, (Dobry and Alvarez, 1967) and Mochi Koshi tailings dam in Japan, (Okusa and Anma, 1980), are ample demonstration. New dams can be designed to appropriate standards. In populated areas, it may be appropriate to design for the long term to cope with the maximum credible earthquake. Methods of earthquake loading probability estimation is well developed for dam design.

Numerous older tailings impoundments have been constructed in a manner which make their embankments susceptible to liquefaction during extreme earthquake events. Abandoned impoundments of this type, if located in an earthquake potential area, may require remedial works to render them stable in the long term.

High Winds and Tornadoes

Tornadoes, tropical cyclones and low pressure systems are sources of high winds. Risk of failure from the last two sources is associated more with the precipitation that accompanies them than from the high winds themselves. Tornadoes, though the most destructive of all winds, are of such a short duration at an impoundment site that they do not have a significant potential of effecting the stability (Kolousek, 1984).

Forest Fires

Forest fires are expected to occur several times in the period of interest. Of themselves, they do not pose a significant threat to waste impoundment stability. However, loss of vegetation cover may lead to accelerated erosion by wind and water.

Perpetual Disruptive Forces

Erosion

Erosion may occur as a result of either wind or water action. Both are potentially severe causes of instability of surfaces and covers.

- Wind Erosion

Wind erosion has been observed to be a major release mechanisms at some existing waste impoundments. Control of this mechanism, for the long term, depends on the successful establishment and maintenance of a wind erosion resistant cover, such as vegetation, waste rock or surface crusting. Methods for the determination of the wind erosion potential and release rates of a particular cover type are reviewed by Steffen Robertson and Kirsten (1986a). They conclude:

1) Unacceptably high wind erosion rates occur on tailings surfaces and soil covers (without gravel) unless a well developed vegetation cover is established. Loss of vegetative cover (due to forest fires, flooding, or salt migration) could result in unacceptably high rates of wind erosion releases.

2) till, with a substantial gravel percentage, and rock waste will form effective wind erosion resistant layers when placed in thin (600 mm) cover layers.

- Water Erosion

This is probably the single most severe cause of impoundment instability. Erosion can take the form of flood erosion of the diversion works, or sheet and gully erosion of the impoundment surface and embankment slopes.

i) flood erosion

A substantial portion of total erosion occurs during extreme precipitation and flood events. The probability of failure will depend on the criteria used to design the structure, and the degree of scour, sedimentation and/or blockage which has occurred. Sedimentation, ice, vegetation growth and debris blockage are extremely difficult to avoid in the long term; though easily and inexpensively cleared through maintenance.

Methods for the evaluation of erosion risk, and appropriate methods of design are reviewed in Steffen Robertson and Kirsten (1986a, 1986b). Appropriate control structure design would involve:

1) Design and construction of diversion structures to accommodate the Probable Maximum Flood (PMF) with ample width and size to allow for partial blockage or sedimentation.

2) Use of heavy riprap (or waste rock) armouring along flow channels and on adjacent slopes. Methods for erosion protection design are provided in Walters and Skaggs (1984).

ii) sheet and gully erosion

The most suitable methods for the prediction of sheet and rill erosion are the Unified Soil Loss Equation (Wischmeier and Smith, 1978), and Modified Universal Soil Loss Equation (Williams, 1975). After using these methods for erosion rate estimation, Steffen Robertson dn Kirsten (1986a) concluded that:

1) Tailings and bare soil would result in excessive sheet and rill erosion unless the soil contained a high percentage of coarse gravel.

2) Good grass cover does much to control this type of erosion which becomes insignificant with continuous forest cover.

3) Rock waste and cobble riprap are effective controls.

4) Discontinuous cover or periodic cover loss will result in unacceptable erosion rates.

iii) gully erosion

Gully erosion has been observed to be a major cause of instability of tailings surfaces and embankments. The only available method of gully erosion estimation is that proposed by Falk et al (1985). This method is based on limited data for American climatic conditions, does not account for vegetation cover and is considered invalid for long period estimation. Nevertheless, it has been used to demonstrate that the only effective gully erosion control for gravel free embankments and tailings slopes is riprap. This conclusion is in agreement with field experience.

Biotic Activity

i) root penetration

In general root action is considered to have an overall beneficial effect on covers, embankments and other structures in providing a binding effect and resistance to soil erosion, as well as minimizing infiltration. Two possible destabilizing effects have been observed:

1) Roots may penetrate low permeability layers and, on decomposing, provide seepage channels which increases infiltration through covers, or piping in embankments.

2) extensive root development in moist, permeable drains or drainage layers may ultimately result in clogging of the drains, in a manner similar to that experienced with residential drains.

Methods to reduce the potential for drain blockage include ensuring that the drains operate in a flooded condition and use of large drains with surplus drainage void space.

ii) burrowing intrusion

Burrowing intrusion by insects and animals have the potential, in the long term, of significantly altering the permeability of low permeability capping layers.

Intrusion by man, principally to obtain tailings for use in construction, has proved to be a material transport mechanism at some acid generating waste sites. Vehicle and large animal traffic can also be a major cause of erosion. The prevention of intrusion by man, in the long term, can only be achieved through institutional control and enforcement.

Frost action

Recent studies and experience regarding frost action in tailings impoundments (Knight and Piesold, 1986; Geocon, 1986; Steffen Robertson and Kirsten, 1987), have demonstrated that the effects of frost on the engineering properties of tailings and their containment structures can be large. In those areas where continuous or discontinuous permafrost develops, and in areas of severe winter cold, frost action may be a major cause of long term instability.

The effects of freezing temperatures on tailings impoundment stability can be divided into two broad groups.

i) annual ice accumulation

Water flow in channels or drains may freeze in successive layers resulting in large accumulations of ice in a single winter, ice accumulations may result in blockage of the diversion structures or outlet works, with a consequential risk of erosion along the displaced flow channel during the early spring melt. Freezing of drains may result in a build up of pore pressures in embankments resulting in slope failure.

Ice accumulation can and does occur during tailings placement, where tailings are discharged onto beaches. Depending on the rate of rise and the sequencing of the beach development, annual accumulations may be many metres thick. Incumbent layers of tailings serve as a thermal insulation layer resulting in successive annual accumulations of frozen tailings. These layers of frozen tailings prevent drainage and hence both the dissipation of pore pressures and the consolidation of the tailings.

As a result, large consolidation settlements may occur after close-out. Such settlement will affect the drainage pattern on the surface of the impoundment and may result in cracking of any cover layers placed on the tailing.

ii) seasonal frost penetration

Knight and Piesold (1986) found that the effect of freezing low density tailings is to create a lattice work of ice crystal with consolidated tailings in between. The effect on the engineering properties of the tailings, (Robertson, 1987a), is to increase the permeability of the tailings at the same time as increasing the density. Seepage and consolidation rates (after thawing) are therefore increased. Frost susceptible cover materials subjected to freezing will develop ice lenses and a fissured structure which, on thawing, increases the impermeability of the cover. Where the cover is designed to limit infiltration this represents a severe failure mechanism. Frost penetration may also block subsurface drains, preventing drainage and causing pore pressure increases.

With annual accumulations of ice, there is an associate desegregational frost heave. Conditions have been identified where up to 200 mm of heave can occur annually (Steffen Robertson and Kirsten, 1987). This seasonal heave is uneven, depending on surface and deposit conditions. Heave, with its consequential effects on drainage, wind and water erosion and cover disturbance, further affects differential frost heave conditions. The end result is a hummocky, irregular surface with greatly disrupted drainage and cover layers. Small variations in seasonal temperature, snow cover and vegetation cover conditions will materially effect rates of aggradation and degradation of the frost. Frost induced creep (solifluction) also occurs on steeper surfaces such as embankment and waste pile slopes.

Weathering

The chemical and physical change of cover materials due to weathering may result in long term degradation of the protective layers. Factors to consider include the effects of alternate wetting and drying, alternate temperature changes, ultraviolet radiation, and chemical reaction with pore fluids. Methods for the selection and evaluation of riprap durability are reviewed in Steffen Robertson and Kirsten (1986a, 1986b). Long term durability of natural materials for use as liners has been studied by Clifton Associates Ltd. (1986), Uziembio et al (1981), Nasiatka et al (1981) and Buelt and Barnes (1981), these results should be applicable to the durability of soil covers as well. Golder Associates (1984) and Buelt and Barnes (1981) have evaluated the durability of synthetic materials.

8.0 WATER COVERS

Techniques to control acid generation include the exclusion of oxygen from the sulphidic waste as described in Section 2.3. Underwater disposal is currently the technique that is the most successful in achieving this objective and is therefore the most secure acid generation control measure. There is steadily accumulating evidence that storage of acid generating wastes under water reduces acid generation levels to negligible amounts. (Nolan, Davis & Associates, 1987; Robertson, 1987; McCready, 1987; Senes Consultants Ltd., 1984). Although oxidation of sulphides and resultant acid generation may not be halted entirely by placing wastes underwater, the rate of acid generation is generally reduced sufficiently to make the impact negligible.

Water cover may be achieved by the disposal of waste into natural waters or into man-made impoundments, and the flooding of underground mine workings and open pits.

8.1 WASTE DISPOSAL INTO NATURAL WATERS

There are a number of sites in Canada where acid generating wastes have been and are being placed in lakes or coastal waters (Nolan, Davis & Associate, 1987; Steffen Robertson & Kirsten, 1988). An example of copper tailings deposited in a shallow lake about 40 years ago at Mandy Lake, Manitoba, provides interesting information (Senes Consultants Ltd., 1984; Hamilton and Frazer, 1978). A study carried out showed the pyritic tailings had a high sulphide content (15 to 17 percent sulphur) and appreciable amounts of zinc (1.7 percent) and copper (0.9 percent). Submerged tailings under a water cover of 0.2 to 1.1 m showed little evidence of oxidation or acidification. Typically, submerged tailings porewater pH values were around 6.9 in comparison to a surface water pH of 7.7 above the tailings. Only in partially exposed tailings deposited along the shoreline had the pH declined to 4.5. The authors reported vigorous revegetation and a healthy benthic community on the underwater tailings deposit.

In contrast, nearby exposed dry tailings of similar composition were found by these same investigators to have undergone substantial oxidation. A significant fraction of the sulphide minerals in the top 15 cm has been oxidized to iron oxides, sulphate and dissolved heavy metals. Soluble copper and zinc concentration measured 797 $mg.L^{-1}$ and 218 $mg.L^{-1}$, respectively, and the pH value was approximately 2.5. The entire surface of the tailings remained devoid of vegetation (Hamilton and Frazer, 1978).

There are other examples of tailings discharged into natural lakes that indicate a general lack of oxidation under these conditions (Nolan, Davis & Associates, 1987). The concept of flooding pyritic mine tailings has been evaluated by many researchers (B.C. Research 1970, 1974 and 1980, Hawley 1975, Nanisivik Mines 1981, Halbert et al, 1982). An extensive search of the literature on this specific topic identified some twenty-five tailings disposal schemes involving discharge directly under a water cover (SENES and Golder, 1981). The name, location, mine and disposal type for these schemes are shown in Table 4.

Table 4 APPLICATIONS OF WASTE DISPOSAL INTO NATURAL WATERS*

Operation Name/Location/Type	Disposal Location/Type	Tailings Treatment/Discharge
F R E S H W A T E R		
Cominco Benson Lake, B.C. copper	· Benson Lake vertical discharge from raft at depth of 30 m	· in operation 1962 to 1973
Eldorado Beaverlodge, Saskatchewan uranium	· Fookes Lake shore discharge has created delta	· 10% solids in slurry · all particles in tailings 45 µm diameter
Grong Gruber Norway iron	· Huddingvann horizontal submerged outfall	· de-aeration tower at shore
Iron Ore Company Labrador City, Newfoundland iron	· Wabush Lake shore discharge	
Mandy Mine Mandy Lake, Manitoba copper, silver, gold	· Mandy Lake tailings deposited in shallow water at shore	· in operation 1917 to 1920 and 1943 to 1944
Nanisivik Baffin Island, N.W.T. lead, zinc	· West Twin Lake horizontal submerged outfall	
Newmont Mines Stewart, B.C. copper	· headwaters of river	

*After Senes Consultants Ltd. and Golder Associates, 1981.

Table 4 (cont'd)

Operation Name/Location/Type	Disposal Location/Type	Tailings Treatment/Discharge
Polaris Mine Little Cornwallis Island, N.W.T. lead, zinc	· Garrow Lake (saline) proposed, horizontal submerged outfall	· premixed with saline water · thickened to 50% solids
Reserve Mining Company Silver Bay, Minnesota iron	· coastal bay shore discharge	· in operation 1947 to 1980
Western Mines Campbell River, B.C. copper, lead, zinc	· Buttle Lake vertical submerged outlet	· Alkaline chlorination. · 7% to 10% solids in tailings slurry
MARINE		
Amax Alice Arm, B.C. molybdenum	· Alice Arm, fjord submerged pipeline	· mixed with sea water · de-aerated
A/S Sydvaranger Kirkenes, Norway iron ore processing	· fjord pipeline with outlet near shore	
Atlas Consolidated Philippines copper	· coastal bay vertical submerged pipeline	
British Aluminum Marseille, France iron	· coastal bay submerged discharge	
Cleveland Potash Yorkshire, U.K. potassium	· coastal bay	· premixed with sea water

Table 4 (cont'd)

Operation Name/Location/Type	Disposal Location/Type	Tailings Treatment/Discharge
Folidal Verk Norway copper	· fjord horizontal submerged pipelines	
Fosdalens Bergverks Norway iron	· fjord coarse tailings are barged, fine tailings are discharged from a launder at the shore	
Greenex Greenland lead, zinc, silver, copper	· fjord horizontal submerged pipeline	· lime and flocculant added · mixed with sea water
Island Copper Vancouver Island, B.C. copper, molybdenum	· fjord horizontal submerged outfall along sea bed	· lime, coagulants added · mixed with sea water · 40% solids
Marcopper Philippines copper	· coastal bay submerged discharge	
Norsk-Nefelin Norway iron	· fjord submerged pipeline	
Rana Gruber Ranafjord, Norway iron ore processing	· fjord vertical submerged pipeline	

Table 4 (cont'd)

Operation Name/Location/Type	Disposal Location/Type	Tailings Treatment/Discharge
Sunro Mine Jorden River, B.C. copper	· coastal bay tailings discharged from platform 700 m off-shore	· in operation 1960 to 1974
Titania Tellnes, Norway titanium	· fjord horizontal submerged tunnel for discharge	· tailings are thickened and "degassed" before release
Westfrob Mines Tasu, B.C. copper, iron	: ocean inlet	

There appears to be very limited information available on the results of monitoring of the impact of waste rock disposal in natural waters. A detailed monitoring program is underway at the Island Copper Mine on Vancouver Island, British Columbia, where waste rock is being placed in the sea.

A serious limitation of waste disposal in natural water bodies is that there may be other environmental impacts. This is reflected in the general concern on the part of the public and many regulatory agencies over the use of natural water bodies for any form of waste disposal. The environmental concerns include the following:

- Toxicity from reagents and heavy metals in mill effluent;
- Excessive nutrient additions;
- Increased turbidity causing a reduction in light penetration;
- Direct impact on the habitat, e.g., smothering of bottom organisms.

In circumstances where natural water bodies do exist in reasonable proximity to mining operations, the relative merits of using them for reactive waste disposal should be weighed against the cost and long-term environmental implications. This evaluation should then be carried out for land-based disposal alternatives.

8.2 WASTE DISPOSAL INTO MAN-MADE IMPOUNDMENTS

Since available evidence indicates that water cover provides the most secure method of acid generation control, consideration should be given to the construction of a water retention facility should natural waters be unavailable or unsuitable. The practicality and cost of a man-made reservoir relative to alternative measures is clearly dependent on site specific criteria, for example topography and volume of waste to be stored. The cost of flooding existing waste facilities is likely to be very high. A proposal for the construction of a dam to flood the existing reactive tailings at the Curragh Resources property in the Yukon, Canada, was estimated would cost approximately $50 million (Nolan, Davis & Associates, 1987).

The design of a facility to provide water cover to combined tailings and waste rock may prove beneficial and cost effective for proposed developments. A tailings and waste rock impoundment system for permanent underwater storage of waste has been designed for the proposed Cinola project in Canada (Robertson et al, 1988). Combined tailings and waste rock disposal may have definite advantages in terms of acid generation control, particularly if the tailings are not acid generating and are discharged at elevated pH (greater than 7). If intimate mixing of tailings and waste rock can be achieved, the permeability of the coarse waste rock would be significantly less than if the rock were placed alone. This has the advantage of reducing potential water movement through the waste rock. These advantages are illustrated in the proposals for the Cinola project (Robertson et al, 1988; Norecol Environmental Consultants, 1988).

There are however, limitations and design considerations that may be a disadvantage to man-made water cover facilities. These include:

There are however, limitations and design considerations that may be a disadvantage to man-made water cover facilities. These include:

- Water retention dams require detailed design of embankment and spillway facilities, careful construction control and maintenance in the long-term. Depending on the site-specific conditions this may prove uneconomical.

- Reliable water sources must be available to provide a continuous water cover of sufficient depth to avoid generation of the waste and erosion due to wave action or water flow.

- Minimum water cover needs to be maintained in low precipitation and drought periods.

- Water reservoirs may induce unacceptable seepage. If there are other soluble deleterious products in the wastes this may result in increased contaminant loading of the environment. Whether these are significant for the specific project and site conditions has to be determined.

8.3 FLOODED MINE WORKINGS

Flooding underground mine workings and open pits is a means of controlling acid generation from the exposed rock faces in these facilities. This method also provides a potential disposal area for acid generating waste.

Flooding of worked out coal mines has been successful in the control of acid mine drainage in several instances with acidity reductions of 45 to 99% being reported (Gleason et al 1978). The potential benefits to be derived from flooding underground mine workings have been reported for several anthracite coal mines in eastern Pennsylvania (Ladwig et al 1984). Field investigations at the mines, which were allowed to flood some 14 to 20 years ago, revealed that the mine waters which were formerly highly acidic are now slightly alkaline. Sulphate reductions of approximately 54 and 74 percent in mine waters were seen in comparison to 1960's data. In addition, marked decreases in the iron, aluminum, manganese, calcium and magnesium levels were observed. The proposed Cinola gold mine in British Columbia is an example of a plan to backfill and flood the open pit on closure of the mine (Robertson et al, 1988).

The disadvantages associated with storage of waste in flooded mine workings are as follows:

- At single pit operations it is necessary to store reactive waste rock for the life of the mine and to then incur rehandling costs in moving the material back to the pit at a time when the operation is producing no revenue.

- If all waste removed from the pit is reactive, the bulking factor (usually about 30%) will result in an excess volume of reactive waste to available underwater storage, particularly as the pit will likely flood to a point less than full.

- Any sulphides in the pit walls above the final water elevation will oxidize causing a deterioration in water quality unless preventative measures can be applied.

- Backfilling and flooding precludes future underground development that might be associated with the ore body.

9.0 SYNTHETIC COVERS

The use of synthetic membranes for liners for tailings impoundments has been reviewed by Golder Ass., 1984.

9.1 GEOMEMBRANES

Flexible membrane liners are commonly referred to as geomembranes. Common types are:

i) polyethelene (P.E.)
ii) high density polyethylene (HDPE)
iii) chlorinated polyetheiene (CPE)
iv) chlorosulphonated polyethylene (CSPE)
(commonly known by the Dupont trade mark - HYPALON)
v) polyvinyl chloride (PVC)
vi) ethylene propylene diene monomer (EPDM)
vii) butyl rubber

Occasionally neoprene and polyurethane are also used.

Collectively, synthetic membranes display a number of advantages and disadvantages which may be summarized as follows:

Advantages:

i) can contain a wide variety of fluids with minimum seepage due to low reported permeabilities of typically 1×10^{-10} cm/sec or less.
ii) have relatively high resistance to chemical and bacterial deterioration.
iii) are readily installed for many applications.
iv) are relatively economical to install and maintain.

Disadvantages:

i) are relatively vulnerable to attack from ozone and ultra-violet light.
ii) have limited ability to withstand stress from heavy machinery.
iii) have not been in service long enough to evaluate long term performance.
iv) are comparatively susceptible to laceration, abrasion and puncture.
v) some materials are prone to cracking and creasing at low temperatures or stretching and distorting at high temperatures.
vi) although readily installed there are often difficulties associated with seaming.

Polymeric membranes offer wide ranging chemical resistance and are readily inspected. However, they are susceptible to damage during installation largely due to improper subgrade preparation and vehicular traffic. They require very careful installation and their performance is dependent on careful and successful field seaming. Field seaming is, in general, a detailed and sensitive operation. Weather, including temperature and precipitation, is generally the governing factor. In this regard, the elastomeric liners namely, Butyl, Polychloroprene and EPDM would appear to present the most problems in field seaming. Of the remaining liner types considered, successful field seaming has been demonstrated with HDPE, PE, CSPE, CPE and PVC. It is noted, however, that there are serious concerns about the long-term weatherability of PVC and PE.

Proper subgrade preparation and construction is crucial for a successful liner installation and would typically consist of subexcavation of compressible materials, sterilization of the subgrade, removal of all roots, sticks, stones and debris, grading and proof-rolling, and installation of the sand cushion, liner and soil cover. Installation of the liner and field seaming should be carried out by approved installers meticulously following liner supplier instructions.

Soil cover is desirable but will require liner inclinations flatter than about 3 horizontal to 1 vertical. This is a sever limitation when applied to waste dumps.

With the exception of polyurethane the base polymeric resins and asphalt show promise for long term resistance to the major anticipated constituents of uranium tailings. Caution should be noted with regard to the following:

- CPE may be affected by weak sulphuric acid solutions

- kerosene is used for solvent extraction in some milling operations. Most of the liners will offer satisfactory resistance to the low kerosene concentration anticipated.

- most liners would be adversely affected by high kerosene concentrations associated with accidental spillage

Compatibility testing carried out by Pacific Northwest Laboratories (PNL) on CSPE, PVC, HDPE and Catalytic Airblown Asphalt in the presence of simulated acidic uranium tailings leachate indicates that, with the exception of PVC, these materials were relatively unaffected by the leachate for the short test periods.

Thin flexible membrane liners are susceptible to overstressing by strains associated with large differential deformations in the subgrade. It may be necessary to subexcavate and replace compressible materials encountered over the subgrade prior to liner installation. Similar concerns exist for liners placed on slopes and where there is a potential for excess hydrostatic or gas pressure buildup beneath the liner.

Seepage through liners is primarily through liner defects. A rational approach to evaluating apparent or field liner permeabilities is through detailed monitoring of existing installations.

Estimates of liner release rates were undertaken by Golder Associates, 1984, assuming permeable subsurface conditions (permeability greater than 1×10^{-4} cm/sec) and tailings permeability of 1×10^{-5} cm/sec. The analyses indicate that an asphaltic membrane would reduce seepage to about 50 percent of an unlined basin for a field liner permeability of 1×10^{-8} centimetres per second. Polymeric liners with an effective permeability of 1×10^{-10} centimetres per second would reduce seepage to less than 10 per cent of an unlined basin.

In West Virginia, a polyvinyl chloride liner was used to cover a 45 acre backfilled site to prevent seepage into acid producing materials (Caruccio and Geidel, 1983). Results showed substantial decreases in flow and acidity from associated seeps (Caruccio and Geidel, 1986).

Used as the barrier layer in combination with soil material layers in a complex cover, geomembranes should prove to be very effective in limiting oxygen and water transport.

9.2 ASPHALTIC AND SPRAY ON SEALS

Asphaltic and spray on surface sealants can be applied to the surface of the waste to form a barrier to infiltration and oxygen diffusion. A number of products are available, including:

- Alkyd
- Asphalt
- Concrete
- Epoxy
- Polyester
- Polysulfide
- Polyurethane
- Silicone
- Synthetic Rubber
- Thermoplastic Molten Sulphur
- Vinyl

These materials have, in general, been developed for applications such as caulking sealants, soil stabilizers, waterproof barriers, and corrosion protective coatings. Their application to date in mine waste covers is limited.

Surface sealants can be formulated to produce either flexible or rigid lining structures. As a class, these materials do not interact with the existing subgrade of the tailings ponds, but provide a surface coating over the prepared subgrade.

Surface sealants can be installed with three basic techniques:

i) In-Situ Chemical Cure

The materials chemically cure or harden after being applied to the surface. These materials usually involve more than one specific chemical.

ii) Heat Application

Materials which are solid in the desired operating temperature range are applied at elevated temperatures to improve ease of application.

iii) Surface Drying

The material is formulated in a water emulsion or diluted in a solvent carrier for application. The carrier evaporates leaving a solid coating.

Combinations of the above techniques are also feasible in many cases. The object is to prepare the material for ease of application usually with conventional spraying equipment. The actual technique for application is a function of the specific material.

The primary advantages and disadvantages of surface sealants are:

Advantages

a) either sufficient flexibility to conform with or sufficient strength to support the design load bearing (pedestrian or vehicle traffic for example),

b) good weatherability and service life,

c) compatibility with the stored product,

d) immunity to biological attack,

e) sufficient puncture and abrasion resistance,

f) capability of being placed with minimal defects

g) easily repairable, and

h) ease of application and produces an integral liner with no joints.

Disadvantages

a) Relatively difficult to regulate the rate of application and thus the thickness and uniformity of the sealant.

b) As a class these materials are relatively expensive. The high initial cost versus relative ease of application for the spray-ons should be considered for specific applications.

Pacific Northwest Laboratories have investigated the effectiveness of various asphaltic seals for use as radon barriers on uranium tailings deposit (Baker, Hartley and Freeman, 1984). Types of seals tested included sprayed-on asphalt emulsion, in situ addition to asphalt emulsion, hot rubberized asphalt seal, and asphalt emulsion admixes. They found that, though all applications significantly reduced radon flux, admix seals such as those applied with a cold mix paver appeared to be significantly more stable than the other seals. Admix seals could not be applied to surfaces sloping more than 12%, however.

Test results on radon barrier systems using an asphalt emulsion admix seal are described by Hartley et al (1982). The barrier system is shown in Figure 27.

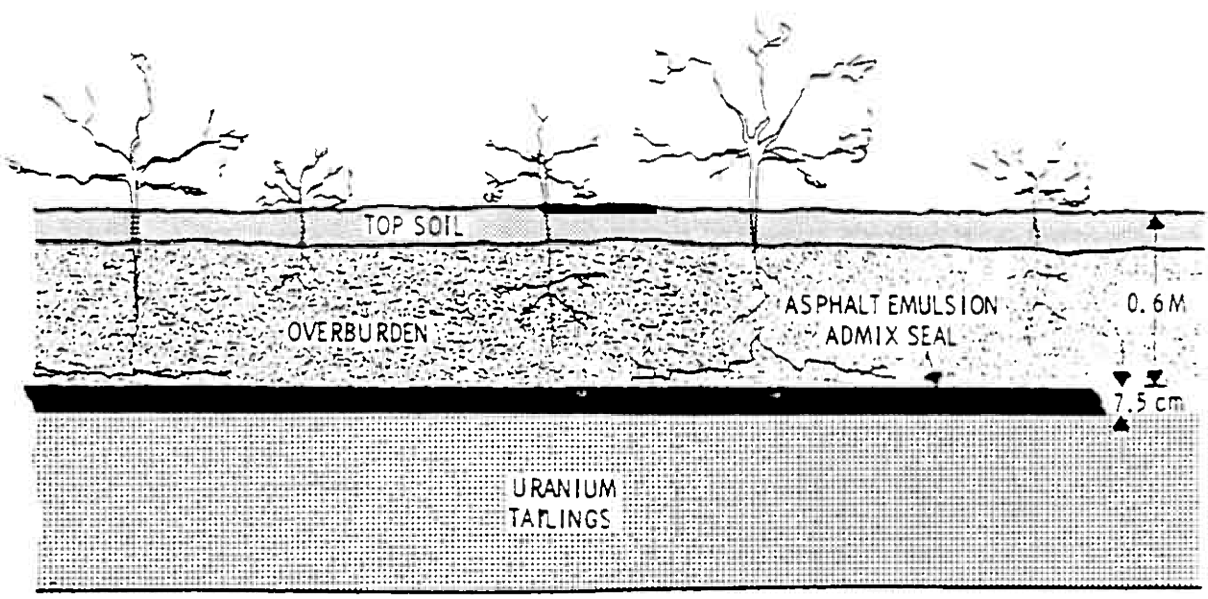

FIGURE 27 Asphalt Emulsion Radon Barrier System

The asphalt emulsion system consists of an 8 cm thick asphalt emulsion/aggregate admix seal that his covered with ~0.6 m of overburden. The admix seal, which forms the radon-impermeable diffusion barrier, is a mixture of cationic asphalt emulsion and an aggregate such as concrete sand. When cured, it contains ~ 22 wt% asphalt and less than 1 wt% water. The overburden stabilizes the admix seal and protects it from UV exposure, rain, extreme temperatures, and sudden temperature changes. Filter/drainage layers would have to be added above and below the seal to prevent degradation due to frost action in Norwegian climates.

Tests performed at Pacific Northwest Laboratories showed that asphalt seals are very effective in reducing radon flux. Asphalt seals were shown to have very low diffusion coefficients to radon (see Table 5); and, field tests showed that asphalt seals could reduce radon flux by more than 99%. Asphalt seals should, therefore, be very effective in reducing oxygen diffusion as well. They also provide very good barriers to infiltration, their hydraulic conductivities being around 10^{-20} m/s (Bell, 1987).

Table 5

Effective Diffusion Coefficients for Radon Through Asphalt Seals
(after Hartley et al, 1982)

Seal Description	De, $cm^2/s \times 10^{-6}$
Asphalt Cement	0.44 to 1.5
Rubberized Asphalt	0.12 to 0.15
Laboratory Prepared Asphalt Emulsion Admix Seals	1.6 to 31
Grand Junction Field Test Asphalt Emulsion Admix Seal Core Samples	2.2 to 10
Typical Earthen Cover	10,000 to 20,000

The concern with asphalt seal is their long-term stability. Hartley et al (1982) stated that aqueous leaching and oxidation should not be expected to degrade the seal, and that the mechanical properties of the asphalt seal are more important in assessing its long-term stability. Further testing of the long-term mechanical properties of asphalt seals will be required.

9.3 GEOPOLYMERS

Geopolymer is the term given to a compound of minerals, principally containing silica, phosphate and oxygen that bond to form a ceramic type product. The suitability of using this product as a control measure for acid generation is currently being investigated in Canada by CANMET. It is anticipated that geopolymers may be mixed with tailings to form a solid mass preventing oxygen and/or water access to sulphides. A possible alternative is to mix the geopolymer with soil or other material and apply this as a cover to the waste. The behaviour of geopolymers when mixed with different waste materials is not fully known at this stage, nor the resistance to natural processes such as freezing and thawing. Geopolymers are still in the development stage and require extensive research to establish their suitability as a control measure.

9.4 SHOTCRETE

Shotcrete is the name given to concrete pneumatically delivered through a hose and applied onto a surface at high velocity. Shotcrete may be effective in the control of acid generation when applied as a cover to certain wastes. The advantage of shotcrete is that it can be applied to steep rock slopes or other surfaces which may be difficult to cover using other methods.

The effective use of shotcrete as a cover is dependent on the stability of the underlying material. This method has been used very successfully on rock faces and on compacted materials.

However, if the material to which the shotcrete is applied undergoes consolidation or settlement causing relative displacement at the surface, cracking of unreinforced shotcrete will occur. Once the shotcrete liner has suffered cracking, the effectiveness of the cover is lost. Experience has shown that displacement of uncompacted waste dumps often occurs and for this reasons unreinforced shotcrete is not appropriate as a cover to these materials. High temperature induced stresses in a shotcrete cover on the surface of a waste facility may also result in cracking.

The resistance of shotcrete to cracking may be increased by providing reinforcing. Conventional steel mesh reinforcing is expensive, difficult to handle and subject to corrosion in the long-term. Steel fibre reinforcement is easier to apply, however, is also vulnerable to corrosion. A method of reinforcement using high volume polypropylene fibre reinforcement, which is corrosive resistant and relatively flexible, shows promise. A benefit of the fibre type reinforcement is that it reduces crack widths in the shotcrete. This gives the shotcrete cover the facility to accommodate larger movements than mesh reinforced shotcrete.

10.0 SOIL COVERS

10.1 SIMPLE COVERS

To minimize cost, one would prefer to cover the mine waste with a soil layer. A fine textured soil, such as clay or clay-till would be required to limit infiltration. To limit oxygen transport it would be necessary to maintain the layer at a high moisture content. A single soil layer, however, would be limited in its effectiveness for the following reasons.

Without suction barriers, a simple soil cover would be subject to large seasonal variations in moisture content. This could result in dessication cracking, increasing permeability. In addition, decreasing the moisture content of the soil increases the rate of oxygen diffusion, as discussed in Section 5.0. These seasonal variations are greatest near the surface and are therefore greatest for thin cover layers. For single cover layers to be effective they must be thick - generally greater than 2 m.

The fine-grained soils required to limit infiltration would be frost susceptible. Ice segregation would result in degradation of the cover and increased permeability. Frost heave would also make the surface of the cover irregular, allowing ponding and increasing infiltration.

A simple soil cover does not have the ability to prevent moisture being sucked up from underlying tailings by capillary action. Likewise, it could not limit the migration of salts from the tailings to the surface due to surface evaporation and transpiration.

A simple fine-grained soil cover would not be able to adequately withstand wind and water erosion or burrowing and root action. Some form of erosion protection, such as vegetation or riprap would normally be required.

These limitations on the effectiveness of a single soil layer can be overcome by using complex covers, as described below.

10.2 COMPLEX COVERS

The effectiveness of a soil cover would be greatly improved by adopting a complex cover design consisting of several layers, each performing specific functions to improve water and oxygen exclusion effectiveness and long-term stability. These layers and their specific functions are described below. A typical complex cover design is illustrated in Figure 19.

1) Erosion Control Layer

Erosion protection can be provided by vegetation or by a layer of coarse gravel or riprap.

The establishment of vegetation on the waste dumps is desirable for aesthetic and land use reasons. Therefore, revegetation is usually the most desirable method of providing erosion control. However, where revegetation is not practical or will not sufficiently control erosion coarse gravel or riprap may be required.

Studies for uranium tailings deposits in Canada (Steffen, Robertson and Kirsten, 1986a) indicated that forest cover would adequately control sheet and rill erosion, and wind erosion, but no analysis methods are available to assess the effectiveness of vegetation on gully erosion. A vegetation cover design guide has been prepared for U.S. uranium tailings deposits by Beedlow, 1984. While this design guide illustrates and discusses many of the aspects relevant to vegetation cover design, it is not directly applicable to Norwegian mine waste deposits. Studies specific to Norwegian conditions will be required. A similar guide for the design of riprap erosion protection has been prepared by Walters (1982).

2) Moisture Retention Zone

The purpose of the moisture retention zone is to provide a zone for moisture retention to limit the effects of dessication. It also provides a growth medium to support vegetation. Moisture retention is desirable for two reasons:

i) It helps to keep the infiltration/oxygen barrier moist. This helps prevent dessication cracking and reduces oxygen diffusion.

ii) By retaining moisture after a precipitation event it allows time for evapotranspiration.

The soil used to construct the moisture retention zone would general be a loam soil with a substantial sand fraction.

3) Upper Drainage/Suction Break Layer

The upper drainage/suction break layer serves two primary purposes:

i) to drain water laterally from the surface of the infiltration barrier, preventing ponding

ii) to prevent moisture loss from the infiltration barrier due to upward capillary suction

Prevention of ponding reduces infiltration. Keeping the infiltration barrier moist helps to reduce oxygen diffusion and prevents dessication cracking.

This layer can also be designed to prevent intrusion by burrowing animals if it incorporates large gravel. For drainage to be effective it must be constructed with a cross fall of 1% or greater.

The effectiveness of this layer would be expected to decrease with time as becomes clogged with roots and organic debris and in-washed fines, and as the drainage slope is modified by long-term settlement of the underly tailings or rock waste.

4) Infiltration Barrier

This is a low-permeability layer consisting of fine-grained soil or synthetic materials (or a combination of both). Its purpose is to prevent the downward infiltration of moisture and the diffusion of oxygen into the tailings.

Rasmuson and Eriksson (1987) have investigated the use of capillary barriers, containing fine and coarse soil layers, to reduce infiltration. They found that the capillary effect of a fine-grained soil overlying a coarse-grained soil would not be practical for preventing infiltration. A low-permeability infiltration barrier is required. They did conclude, however, that "the low conductivity of the fine layer, together with higher conductivity layers above it, could diminish infiltration to the waste rock significantly".

5) Lower Suction Break Layer

Finally, a suction break layer is placed between the infiltration barrier and the tailings surface to prevent suction of moisture downwards into the tailings.

The long-term performance of a complex soil cover could be greatly reduced if fine-grained materials are allowed to migrate into the coarse-grained layers. Filter layers could be added.

6) Basic Layer

A basic layer could be incorporated into the design to reduce the pH of infiltrating water and therefore acid generation rates. Alkaline materials such as limestone could be spread over the surface of the waste before placing the cover or mixed into the cover layers.

Limestone is commonly mixed with waste rock during placement at coal mines with great success and research is being done on the addition of phosphate rock (Chiado et al, 1988). However, the potential for acid generation control by surface applications of alkaline materials is less attractive than mixing them with the waste. Limestone has a low solubility in near neutral water, and the resulting alkaline charge is therefore small and insufficient to control AMD. Surface inflows tend to be concentrated at isolated locations such as depressions, cracks, permeable zones, etc. At these locations the available alkaline materials are quickly exhausted. The addition of a basic layer would not significantly reduce acid mine drainage where unsaturated conditions predominate, such as in waste piles. It would be more beneficial in saturated tailings, and might be usefully employed in tailings impoundment covers.

10.3 THE EFFECTIVENESS OF SOIL COVERS

Information on the effectiveness of soil covers in controlling acid mine drainage is available as results from mathematical model simulations of covers and from monitoring results from a limited number of actual covers.

Acid flux reductions calculated in Steffen, Robertson and Kirsten (1987) using the RATAP model for some simple cover types are tabulated in Table 6, and illustrate the theoretical effects of riprap, soil and limestone covers.

Table 6
Effect of Closeout Options on Acid Generation

Tailings/Cover	Depth (m)	Sulphate Flux (moles/m^2/year)	Reduction of Acid Flux (%)
None	-	76.65	-
Waste Rock	0.6	59.37	22.5
Depyritized Tailings (soil)	2.0	21.06	72.5
Depyritized Tailings (soil) + Limestone	2.0	7.9	89.7

The effect of cover thickness is clearly demonstrated in Figure 23.

Figure 21 shows the effects of various types of covers on infiltration rates as predicted by the HELP model for uranium tailings deposits in Canada. The results show considerable benefit of a complex cover design over simple covers, and benefit of adding a synthetic liner.

The results of the infiltration modelling runs are discussed below.

i) Bare Tailings

Bare tailings can be expected to have high runoff rates, modest evapotranspiration losses and substantial net infiltration or seepage. With an unvegetated surface, the run-off can be expected to be quite high. For the example model runs, runoff, evaporation and seepage rates account for 19%, 63.5% and 17.5% of the annual precipitation respectively. When the tailings permeability was increased by a factor of 3, runoff rates decline by 3% and seepage rates increase by 3%.

ii) Vegetated Tailings

Vegetation has a marked effect on the water balance at a tailings site. With the growth of vegetative cover, runoff rates decrease from 19% to 9% of the annual precipitation while evapotranspiration increases from 63.5% to 73.6%. The major finding is that seepage rates are not changed. With a good vegetative cover, runoff rates are again reduced further to 3.3% of the annual precipitation. Although evapotranspiration rates are increased, this may not offset the reduced runoff. This is not to conclude that this phenomenon is universally applicable to all sites.

iii) Soil Cover

Direct application of soil to the tailings area surface may have mixed effects. If the soil retains its low permeability, runoff will increase substantially and seepage rates will be greatly reduced. A compacted till cover with a permeability of 2×10^{-8} m/s will reduce seepage rates to less than 2% of the annual rainfall. If this cover cracks and weathers (as is expected) infiltration rates increase substantially. The example indicates that if the effective permeability of the cover increases to 1×10^{-6} m/s, seepage rates exceed those for bare tailings. The increased permeability results in a major reduction in the rate of surface runoff.

iv) Rock/Gravel Cover

Rock or gravel is often applied to stabilize the surface of a tailings area. This pervious layer effectively eliminates runoff and therefore can substantially increase infiltration rates. For the modelled case, the rock/gravel cover increases seepage rates from 17.5% to 37.8%. This is more than a factor of 2 and further demonstrates how the permeability of the surface layer can affect the overall amount of seepage produced.

v) Engineered Cap

A properly constructed engineered cap can greatly reduce infiltration rates. The example modelled includes a cap with one metre of soil for frost protection and vegetation, 0.3 metres of lateral drainage layer, and one metre of a low permeability seepage barrier. This cap reduces surface runoff to 3.1% of the annual rainfall. The lateral drain intercepts 19.4% while evapotranspiration accounts for 70.9% leaving 6.6% as seepage. This is a 62% reduction in the total seepage as compared with bare or vegetated tailings. The major finding is that these layers are effective but not 100% efficient in limiting seepage. At 6.6% infiltration this represents approximately 60 mm of precipitation or 60,000 m^3/yr from a 100 ha disposal site.

Rock/gravel surface layers have a major effect, increasing the infiltration. With a pervious surface zone, the lateral drains become more efficient reducing the seepage rates.

An engineered cap with a synthetic membrane liner is by far the most effective short-term infiltration barrier. The seepage rates predicted for an engineered cap with a liner that was 99 percent efficient are 0.1% of the annual rainfall. The life of the liner, however, needs to be considered.

Although theoretical simulations are useful in comparing alternate cover types, the true effectiveness of covers in controlling acid mine drainage can only be determined from monitoring the performance of actual covers in the field. Unfortunately, monitoring results are limited.

The best documented case of a soil cover in use on an actual mine waste dump is that of the Rum Jungle uranium and copper mine in Australia (NTDME, 1986). Composite covers were placed on three acid-generating overburden heaps. The covers consisted of a 225 mm compacted clay layer, overlain by a 250 mm sandy clay loam retention zone layer, overlain by a 150 mm gravelly sand erosion layer. Rehabilitation of the heaps also included reshaping their surfaces and providing surface drainage systems. A typical cross section of the rehabilitated heaps is shown in Figure 28. The largest of the three heaps, Whites Heap was covered in 1983/84. The other two were covered in 1984/85.

Measurements of oxygen concentrations in the pore gas in the heaps show a marked reduction in oxygen concentrations after installation of the compacted clay cover (Bennett et al, 1988). Although measurements indicate that the transition rate of gas through the seal has increased since its initial placement, due to dessication cracking in the dry season, the oxygen concentrations in the heaps are still much less than they were before rehabilitation (NTDME, 1986). The effect of this reduction of oxygen concentration on oxidation rates has not been quantified, (NTDME, 1986).

Pre and post rehabilitation measurements on and near the heaps indicate that the cover has provided some reduction in infiltration (NTDME, 1986). However, the amount of that reduction remains in question. Estimates based on lysimiter measurements indicate reductions greater than 90%, while others based on groundwater estimates indicate only a 50% reduction (NTDME, 1986).

The final measure of the effectiveness on the covers, though, is reduction of metals loads in the local river system. Precipitation and flow data are summarized in Tables 7 and 8. The reductions in post-reclamation metal loads determined by NTDME, 1986 are indicated in Table 9. Although samples taken from the East Finniss River show large reductions in metals loading it is unclear whether these reductions are due to covering the heaps, or due to variations in precipitation rates, i.e., it is not clear whether the reduction in the loads is due to reduced acid generation and migration resulting from cover placement or merely due to reduced migration resulting from low precipitation in the years following cover placement. To better evaluate this the Rum Jungle results were reanalyzed as follows.

The results of the East Finniss River flow rates and metals loading monitoring are shown in Figures 29, 30 and 31. Figure 29 shows the relationship of river flow to rainfall. Figures 30 and 31 show plots measured metals loads as a function of annual rainfall. Both flow and metal load values have been normalized to the maximum measured flow year (73/74) to compensate for relative differences in concentrations of the different metals measured. Comparing the normalized metals loads against the flow/rainfall curve permits evaluation of the metals load measured in a year against the metals loads to be expected given the amount of rainfall for that year. Although the data from the 1983/1984 season (the year Whites Heap was covered) appear to show some reduction in metals loads from those expected, the data from the 1984/1985 seasons and the 1985/86 seasons show no improvement. More monitoring results are required to conclusively assess the effectiveness of the rehabilitation measures. Metals loads measured during a high-precipitation year, such as the 1973/1974 season, would be particularly useful.

Table 7
East Branch of the Finniss River
Revised Pollution Loading Values
(after NTDME, 1986)

Season	1971/72	1972/73	1973/74
Rainfall (mm)	1542	1545	2000
Total Flow $m^3 \times 10^6$	31	22	69
Metal Load (t)			
Copper	77	67	106
Manganese	84	77	87
Zinc	24	22	30

Table 8
Summary of Monitoring Results for the East Branch of the Finniss River at GS 815097
(after NTDME, 1986)

Season	1982/83	1983/84	1984/85	1985/86
Rainfall	1121	1704	1112	910
Total Flow $m^3 \times 10^6$	9.5	48	11.7	11.4
Metal Load (t)				
Copper	23	28	9	4
Manganese	6	21	7	8
Zinc	5	9	4	3

Table 9

Percentage Reduction in Pollution of the East Branch of the Finniss River

(after NTDME, 1986)

Season	1983/84 (Stage 1 & part Stage 2 complete)	1984/85 (Stage 3 complete)	1985/86	Target
Metal				
Copper	70%	80%	91%	70%
Manganese	76%	88%	85%	56%
Zinc	67%	73%	79%	70%

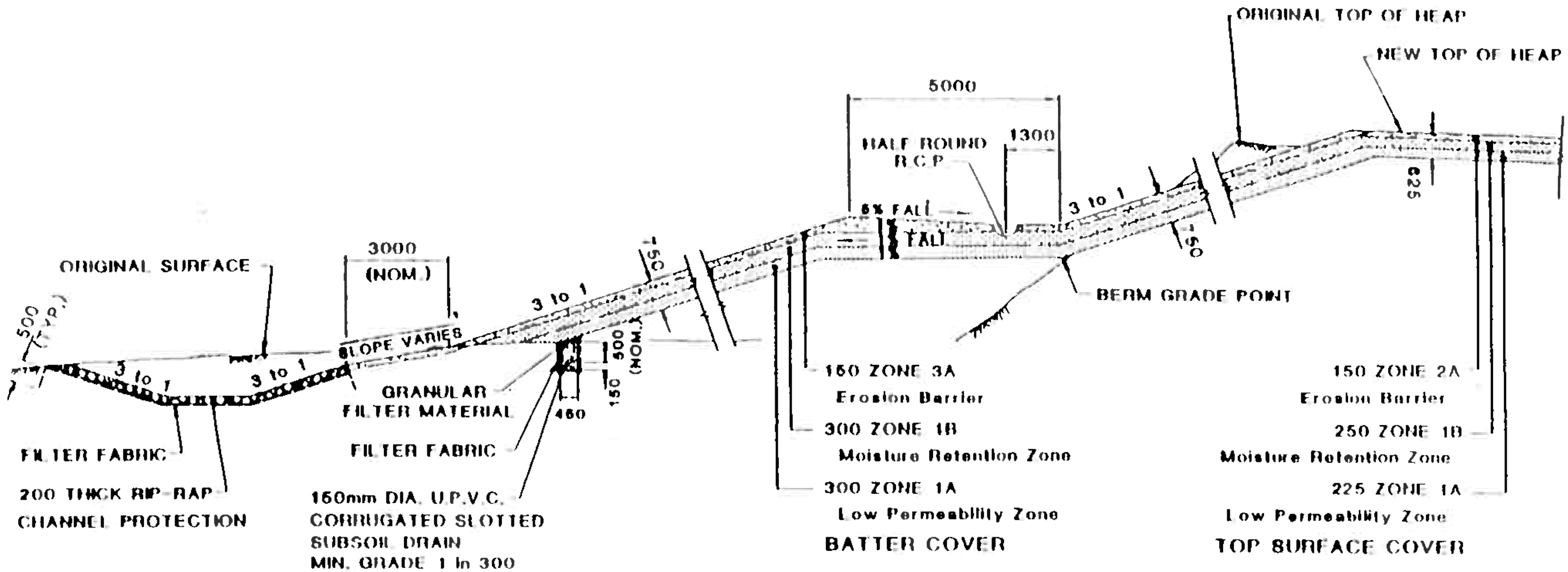

FIG. 28 TYPICAL CROSS SECTION OF REHABILITATED HEAP (after NTDME, 1986)
RUM JUNGLE

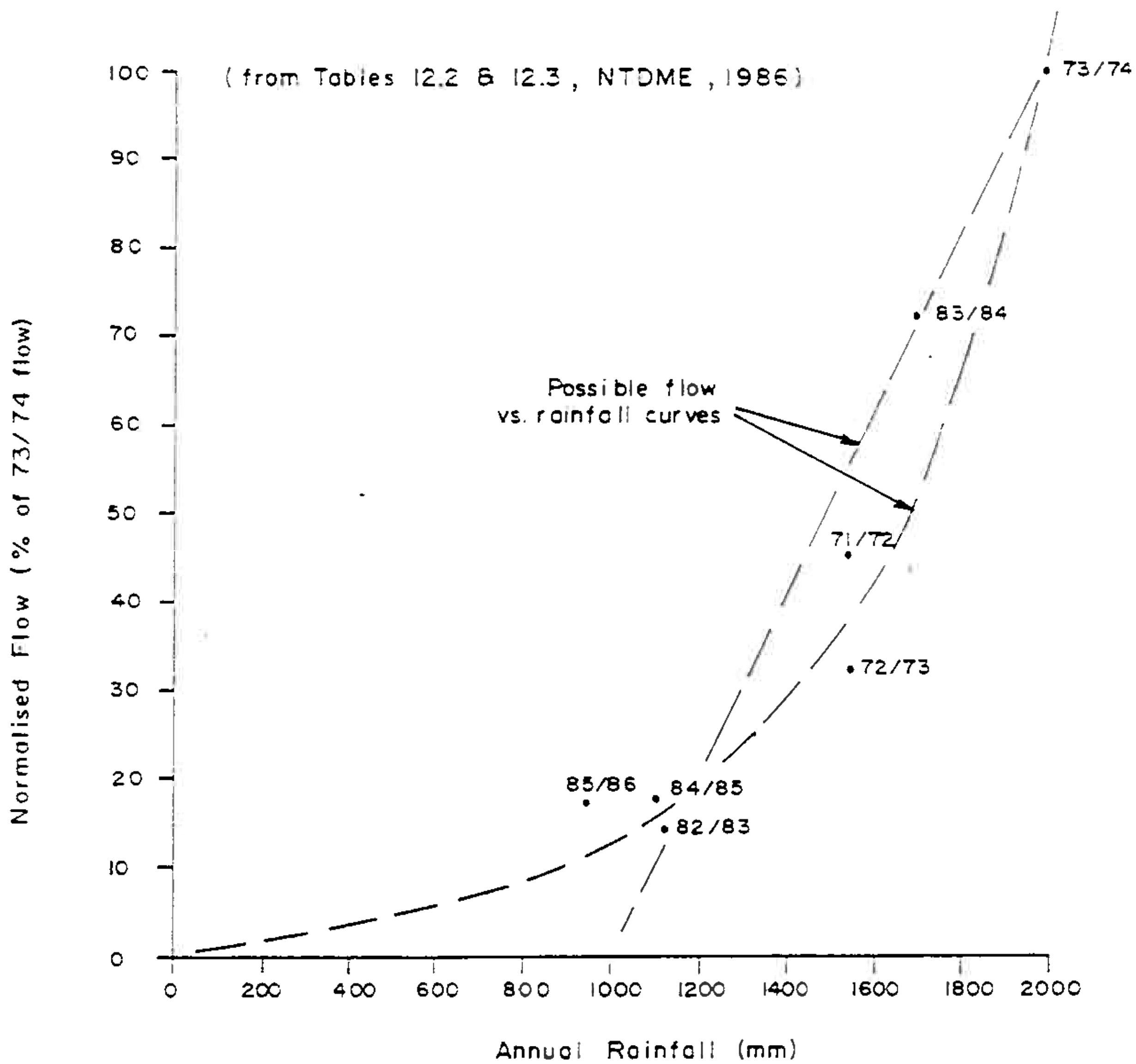

FIG. 29 RUM JUNGLE REHABILITATION PROJECT
EAST FINNISS RIVER
FLOW vs. RAINFALL RELATIONSHIP

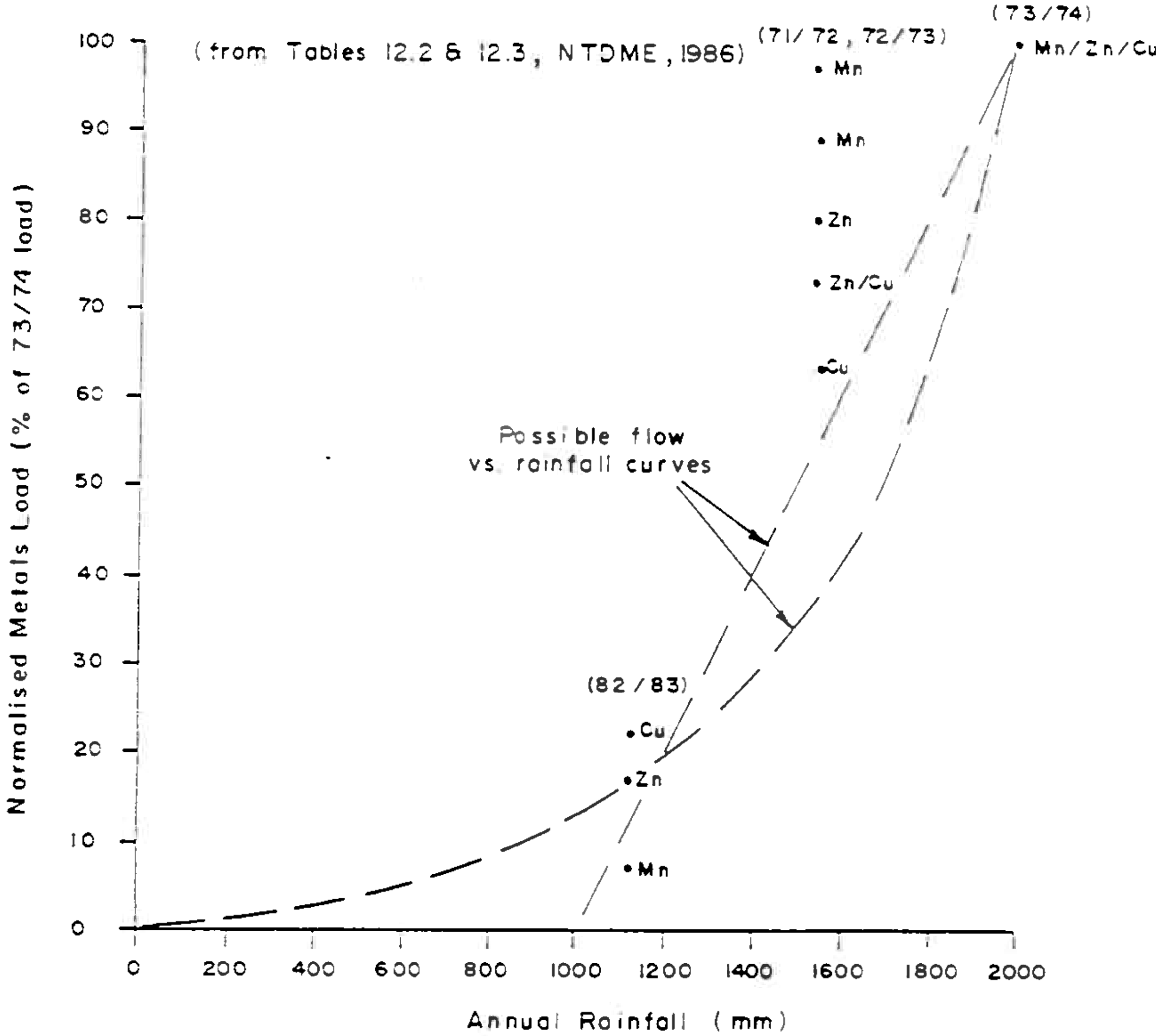

FIG. 30 RUM JUNGLE REHABILITATION PROJECT
METAL LOADING vs. RAINFALL
PRIOR TO COVER PLACEMENT

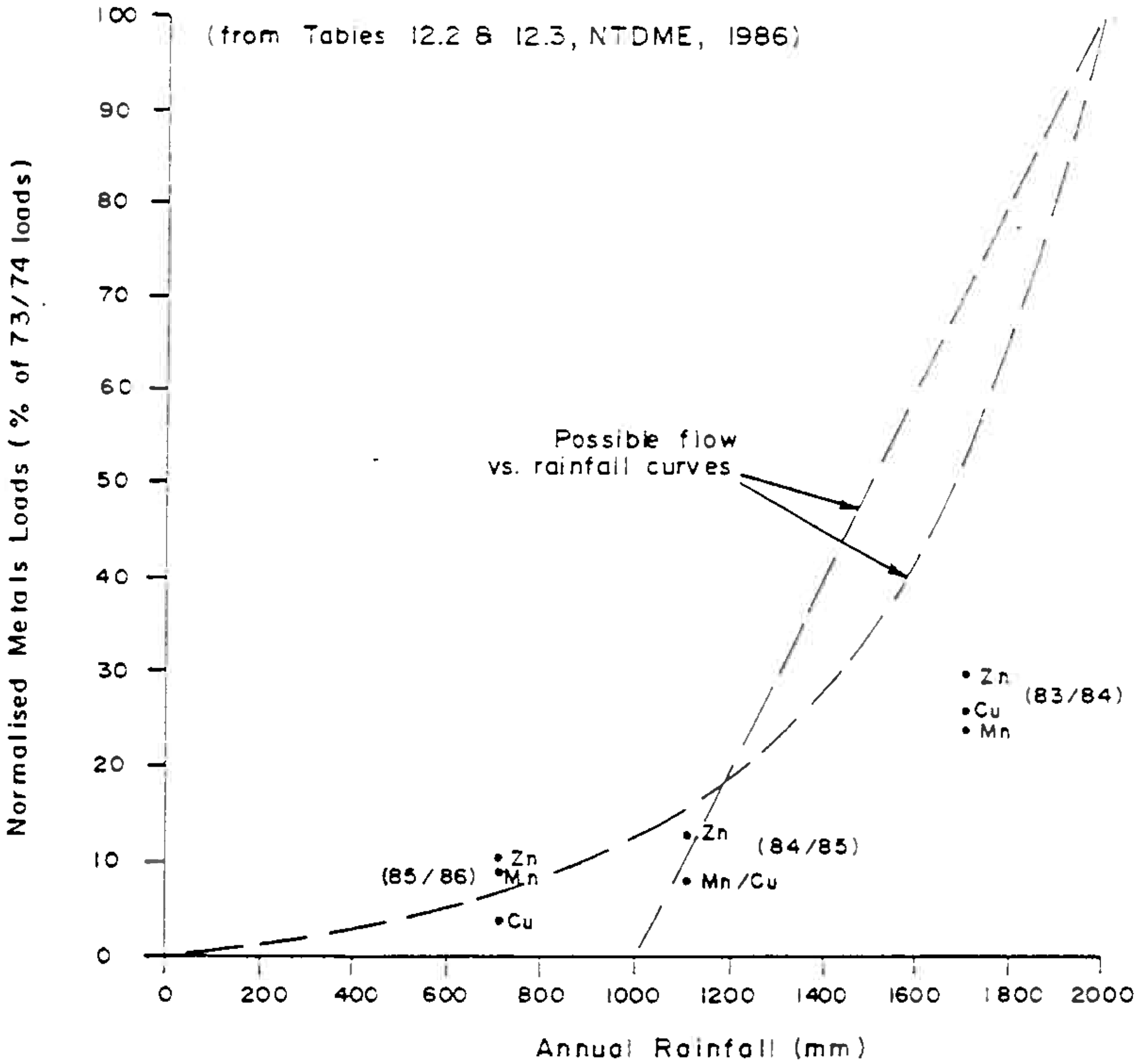

FIG. 31 RUM JUNGLE REHABILITATION PROJECT
METALS LOADING vs. RAINFALL
AFTER COVER PLACEMENT

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Armstrong and Beckett. (1985)

Extensive modelling of oxygen diffusion in plant roots is presented, along with the oxygen concentration in the water film around the root.

Bakker and Hidding. (1970)

A non-steady state method of determining the effective diffusion coefficient for oxygen into undisturbed soil cores is described. Calculations are made of the errors involved when neglecting the storage term, temperature and pressure variations, and the oxygen consumption and resistance in the ambient air. An extensive table reviewing the literature on the relation between air porosity and gas diffusion in soils is given.

Bornstein, Benoit, Scott, Hepler and Hedstrom. (1984)

Platinum electrodes and neutron moderation techniques were used to determine the oxygen diffusion rate (ODR) and soil-water content, respectively, in packed laboratory columns of silty clay loam. A linear relationship between ODR and soil-water content was observed for the wet range (16 to 47% water by volume). Graphs are presented that clearly show the relationship between ODR and soil-water content when water is applied to the column.

Brown. (1970)

A laboratory study, using a closed system, is used to assess the effectiveness of thin (a few inches) cover layers of non-reactive natural porous media for the reduction of oxygen consumption by reactive pyritic tailings. A steady-state finite difference model was used. It was concluded that Penman's (1940) equation ($D/D_o = 0.66E$) relating the relative diffusion coefficient to air-filled porosity cannot be used to estimate oxygen diffusivities of pyritic refuse. An air pycnometer was used to measure air-filled porosity. The amount of oxygen consumed was calculated from the change in pressure within the closed system. Occasional injection of nitrogen was used to offset pressure differentials within the chamber. The soil moisture characteristic curve was not mentioned.

Callebant, Gabriels, Minjnuw and deBoodt. (1982)

Laboratory columns of sandy loam and loamy sand were used. Gas sampling and platinum microelectrodes were used to determine oxygen concentrations and diffusion rates, while the soil-water pressure was monitored using mercury tensiometers. The relationship between depth to watertable and the monitored parameters was investigated, and showed an increase in oxygen diffusion with a deeper watertable.

Colvin. (1977)

Colvin cites Shumate (1971) (EPA Dast-42 14010 FRR 04/71) as saying that 10 to 15 feet of dry, granular soil would be needed to make a significant difference on the rate of oxygen transport through a cover. Used Taylor's (1949) transient diffusion method. Like most studies, the oxygen diffusion coefficient and oxygen uptake rate (i.e., reaction rate) are considered to be constant throughout the soil layer, but it is acknowledged that this may no in fact be a valid assumption. One, two and three layer cover designs are simulated using a steady-state mixing cell model.

Currie. (1960a)

A laboratory apparatus using a modified thermal conductivity gas analyzer and soil columns is described. Hydrogen gas diffusing into air is studied and the transient solution for a gas diffusing from a closed chamber into a porous soil column is given.

Currie. (1960b)

An equation based on theoretical considerations for porosity and the tortuosity of pores is given that equates the effective diffusion coefficient to gas filled porosity. A large variety of porous media is tested, including sand, talc, steel wool, ceramic beads, mica and salt, to name a few. An extensive table of empirical parameters for fitting a curve to the effective diffusion coefficient versus porosity data is given for the full range of materials tested.

Currie. (1961a)

The theory concerning oxygen diffusion into spherical soil aggregates in a soil is discussed and accompanying equations are given. The concept of a critical aggregate radius under which there is no anaerobic centre in the aggregate is introduced.

Currie. (1961b)

A non-steady state technique was used to investigate the relationship between gas (hydrogen) diffusion coefficients and air-filled porosity for a number of materials including glass beads, sintered glass, sand, carborundum, pumice and soil crumbs. The effect of inter- and intra-aggregate porosity is discussed. Numerous graphs of relative diffusion coefficients versus air-filled porosity are given. No single expression fits all data.

Enoch and Falkenflug. (1969)

Modification to the design of a Clark-type membrane-covered oxygen probe improved the probe so that only one calibration curve is needed for operation of the probe in air or water. The double membrane modification did increase the response time of the probe to five to eight minutes.

Grable and Siemer. (1968)

The diffusion of oxygen in soil was found to be determined primarily by air-filled porosity. Bulk density and aggregate size per se had little effect on the rate of diffusion or on the concentration of oxygen at rooting depth.

Gradwell. (1961)

A study of diffusion of oxygen through both packed and undisturbed soil cores. The transient method of Taylor (1949) was used as the basis of the analysis. The relationship between effective diffusivity and air-filled porosity was not found to fit the models in the literature (e.g., DeVries, 1950) for the undisturbed cores. This was attributed to soil structure, i.e., macropores. An attempt to experimentally quantify the oxygen sink term is presented.

Jaynes and Rogowski. (1983)

Fick's Law is compared with the Stefan-Maxwell equations for gas diffusion. It is concluded that the Fickian diffusion coefficient is not a constant. Rather, it is dependent on the diffusional fluxes of other gases and on the mole fraction of the gas in question. Examples of the cases when the Fickian gas diffusion coefficient can be considered constant are given as: the diffusion of a trace concentration of a gas through a gas mixture of any composition, equimolar counter-current diffusion in a binary gas mixture, and diffusion in a tertiary mixture where one gas is stagnant.

Jellick and Schnabel. (1986)

An <u>in situ</u> method, based on spherical diffusion of gas from an injection point is evaluated for the determination of diffusion coefficients of N_2O in field soils. A finite difference model was developed that incorporated initial conditions more consistent with the experimental data, than was possible using analytical solutions. The method involves the injection, through a needle, of a slug of gas at time zero, followed by periodic microsampling of the gas back through the needle. A linear relationship between relative gas diffusion coefficient and air-filled porosity is observed.

Kimball and Lemon. (1971)

A device that measured the rate of evaporation of liquid heptane from a porous steel plate buried in the soil is described. It is concluded that, at very shallow depths (top few cm), air turbulence and pressure fluctuation have a significant effect, but that soil aeration is mostly a diffusive process.

Kowalik, Barnes and Smiles. (1979)

Soil columns were equipped with small access ports through which small gas samples were drawn. The oxygen content in the gas was measured using an oxygen electrode. Using this set-up, a theory of oxygen diffusion in soil in which the rate of oxygen consumption varied with time was applied to describe the oxidation of a slurry of pig manure.

Lai, Tiedje and Erickson. (1976)

A method based on radial diffusion from a point source injection was used. A needle is inserted into the soil. Fifty ml of gas is injected and, at given time intervals, 0.25 mL samples are drawn back through the needle and analyzed using a gas chromatograph. The duration of each experiment was only four minutes; therefore, sink and source terms were considered negligible. A least squares fit of the theoretical solution is applied to the data to solve for the diffusion coefficient. A plot of effective diffusion coefficient versus air-filled porosity is given which fits the expression $D/D_o = E^{2/3}$.

Lemon and Erickson. (1952)

The original paper on the use of a platinum electrode to measure oxygen diffusion in soils. Data given show increasing oxygen diffusion rates with larger soil aggregates and greater soil porosity, along with decreasing oxygen diffusion rates with increasing depth in the soil profile.

Marshall. (1959)

Theory based on the bundle of capillary tubes model of flow through porous media is used to develop a non-linear expression for the relationship between the effective gas diffusion coefficient and air-filled porosity.

McIntyre. (1966)

The platinum microelectrode is stated to give reliable data only over the range of soil-water contents that maintain a water film on the electrode surface. This minimum water content, for which accurate oxygen flux rates could be measured with a 1.5 mm platinum electrode, was achieved after two days of drainage from the saturated state for a well-drained soil. In a poorly drained soil, the platinum electrode was still operating after six days. A table of oxygen flux rate after one, two and three days of drainage, along with infiltration time for five inches of water, is given for a wide range of soil textures.

McIntyre. (1966)

The platinum microelectrode is tested on glass beads, a sandy loam and a loam. The dependence of current on the rate of oxygen flux to the electrode was found to be better for constant effective voltage relative to a reference electrode, rather than constant applied voltage.

Millington. (1959)

Theory is presented to support the relationship $D/D_o = E^{4/3}$, where D/D_o is the relative gas diffusion coefficient, and E is the air-filled porosity. The data of Taylor (1949), Penman (1940) and VanBave (1952) are compared to the theoretical curve. The agreement between theory and data is marginal.

Millington and Shearer. (1971)

Theory and an equation that relates the effective diffusion coefficient to the degree of saturation and the inter- and intra-aggregate water contents is given. A number of graphs of effective diffusivity versus air-filled porosity are given that compare the theory to experimental results for a number of aggregated soils and pumice. The agreement between the theory and experimental data is good.

Nielson, Rogers and Gee. (1984)

Theory is presented for calculating the diffusion coefficient of radon gas from soil-water content and pore size distribution data. Diffusion in both the air-filled and water-filled pores is considered. The model results compare well with measured values.

Papendick and Runkles. (1966)

The necessity of using a transient analysis because of non-constant oxygen consumption rates (sink terms) is discussed. A laboratory apparatus incorporating platinum oxygen electrodes, and a hanging water column (tension table) to maintain constant water contents is described. Plots of oxygen consumption rate versus time for a silty clay loam over the period of eight days is shown. An exponential decay in the oxygen consumption rate is observed.

Phene. (1986)

A detailed discussion of the theory and method of construction of platinum electrodes is presented, along with the design specifications and considerations for a basic oxygen diffusion rate measurement system. A fully automated field installation is discussed and a schematic diagram provided. In addition, the theory and application of the membrane electrode method is discussed with detailed schematics of probes and electronic curcuits.

Phene, Campbell and Doty. (1976)

A fully automated field system of measurement of oxygen diffusion rates (ODR) in a soil profile is described. Platinum microelectrodes are incorporated into the system design. The ODR at various depths in a sandy loam soil are monitored, and a distinct response to rainfall and irrigation events is shown.

Pilot and Patrick. (1972)

The method of Taylor (1949) was used to measure oxygen diffusion into soil with 200 ppm nitrate. A polarographic Clark type oxygen electrode was used in conjunction with a Beckman Model D oxygen analyzer. The redox potential of the soil was measured using 18-gauge platinum electrodes. At low air-filled porosities, no oxygen diffused all the way through the cores. At air-filled porosities above 11%, 12% and 14% for loamy sand, very fine sandy loam and silty clay loam respectively, oxygen diffused completely through the soil cores.

Pritchard and Currie. (1982)

A steady-state method in which the gas is injected at a steady but low rate into the end of a column is presented. Small (0.10 to 0.35 cm^3) samples are removed through sample ports in the wall of the column and then analyzed using a gas chromatograph. Tables comparing measured diffusion coefficients to those found in the literature are given.

Raney. (1949)

A robust in situ diffusion chamber is described for sampling gas in the unsaturated zone. A Beckman oxygen analyzer is incorporated into the design to determine oxygen diffusion coefficients. The effect of different tillage practices on diffusion rate is investigated.

Reardon and Moddle. (1986)

A steady-state method for meauring CO_2 diffusion in unsaturated pyritic tailings is presented. A sodium hydroxide solution is used as a sink for CO_2. The mass flux of CO_2 into the NaOH solution causes a change in electrical conductance. This change is used to measure the CO_2 discharge from the core. Oxygen diffusion coefficients are calculated using the CO_2 data and the fact that the relative diffusion coefficient (i.e., measured divided by the free space value) is independent of the gas used.

Rickman. (1968)

The effect of salts and soil solution electrical conductivity on platinum oxygen electrodes is discussed. The conditions under which a platinum electrode should work most satisfactorily or have minimum interference from salts are described.

Rogowski, Pionke and Broyan. (1977)

The lack of literature on spoil water flow, oxygen diffusion, surface runoff, erosion, evapotranspiration and temperature distributions within the spoil banks is pointed out.

Rolston. (1986a)

Recommended laboratory and field methods for the determination of gas diffusion coefficients of soils. The methods are transient and assume a non-reactive system, i.e., no source or sink terms.

Rolston. (1986b)

Three methods: (1) flux calculated from Ficks Law, (2) closed chamber method and (3) flow-through chamber method, are described for the quantification of the gas flux at and near the soil surface.

Rolston and Brown. (1977)

A laboratory column fitted with gas sampling ports is used to observe the diffusion of nitrogen gas into an Argon gas flushed air-dry loam soil. A field experiment is also described. A sealed chamber fitted to the soil surface is flushed with Argon. Nitrogen was then pumped through the chamber. The rate of flow and nitrogen concentration of the gas flowing out of the chamber through exit ports is used to calculate the flux of nitrogen into the soil. A plot of gas diffusion coefficient versus air-filled porosity is shown.

Rust, Klute and Gieseking. (1957)

A transient solution for diffusion from a closed reservoir into a soil core is given; however, no sink or source terms are included. The problem of mass counter flow caused by the differing diffusion coefficients of gases is discussed. To overcome the possibility of mass counter flow "self diffusion", using radioactive and non-radioactive carbon dioxide is used. A detailed schematic of the apparatus is given which incorporated the use of an air pycnometer for determining air-filled porosity.

Sallam, Jury and Letey. (1984)

A laboratory study was conducted to measure gas diffusion through soil at low air-filled porosities, using the inert gas Freon. Penman's (1949) model greatly overestimated, while the Millington-Quirk (1961) model slightly underestimated the diffusion coefficient. However, the Millington and Quirk model could be made to fit the data by using a slightly smaller exponent on the air-filled porosity term.

Shearer, Millington and Quirk. (1966)

An improved oxygen probe is presented along with a proposed laboratory apparatus for measuring oxygen diffusion through porous media. Design specifications are discussed and experimental results for capillary tube bundles are given. No sand data are presented.

Smiles and Griffin. (1966)

Methods of measuring oxygen diffusion in saturated porous media are examined relative to the platinum microelectrode technique. The effect of microbial respiration is discussed.

Taylor. (1949)

One of the earliest papers on oxygen diffusion in soils. This paper is frequently referenced as the source for equipment design when a closed chamber, transient gas concentration method is used. A Beckman oxygen analyzer is used with continuous gas sampling of the diffusion source chamber. Theory is presented for diffusion through porous media with no sink or source terms.

Troeh, Jabro and Kirkham. (1982)

The commonly used equations relating the gas diffusion coefficient to air-filled porosity are examined. The commonly used linear $D/D_o = a\ (E-b)$ and curvilinear $D/D_o = kE^m$ are discussed in terms of their advantages and limitations. A more accurate and general equation is presented, i.e., $D/D_o = ((E-u)/(1-u))^v$. An extensive table of values for the curve fitting parameters, derived from the data available in the literature, is given.

vanBavel. (1954)

A simple straight cylindrical tube inserted into the soil is flushed with nitrogen and the oxygen diffusion coefficient is calculated from the rate at which oxygen diffuses into the cylinder. Charts for computing purposes are given. The physical and mathematical aspects of the method are presented, along with experimental data.

Willey and Tanner. (1963)

Continuous in situ measurement of oxygen concentration of soil-air is accomplished using a temperature-compensated membrane-covered polarographic electrode. A robust design permits sealing in a soil access tube. The time constant is on the order of 0.25 minutes, and the relative error in the measurement is approximately 2%. Geneal design features of the probe and example measurements are presented. A plot of percent oxygen versus time over a six-day period is presented, which shows response to soil surface cracking and rainfall events for a probe 12 inches deep in a silty clay loam.

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