CONF-941124--18

PNL-SA-24159

THE ROLE OF PLANTS ON ISOLATION BARRIER SYSTEMS

S. O. Link
W. J. Waugh[a]
J. L. Downs

November 1994

Presented at the
Thirty-Third Hanford Symposium on Health and the
Environment Conference
November 7-11, 1994
Richland, Washington

Prepared for
the U.S. Department of Energy
under Contract DE-AC06-76RLO 1830

MASTER

Pacific Northwest Laboratory
Richland, Washington 99352

(a) UNC Chem-Nuclear Geotech, Grand Junction, Colorado

DISTRIBUTION OF THIS DOCUMENT IS UNLIMITED

DLC

DISCLAIMER

This report was prepared as an account of work sponsored by an agency of the United States Government. Neither the United States Government nor any agency thereof, nor any of their employees, make any warranty, express or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed, or represents that its use would not infringe privately owned rights. Reference herein to any specific commercial product, process, or service by trade name, trademark, manufacturer, or otherwise does not necessarily constitute or imply its endorsement, recommendation, or favoring by the United States Government or any agency thereof. The views and opinions of authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof.

DISCLAIMER

Portions of this document may be illegible in electronic image products. Images are produced from the best available original document.

Introduction

An isolation barrier is an earthen structure designed to prevent buried wastes from being dispersed in the environment. Most structures have been designed to prevent water from entering the waste and causing dispersal of wastes into groundwater. A major challenge in developing permanent isolation barriers is to ensure that buried wastes are isolated from dispersion by environmental forces for long periods of time (Gee *et al.*, 1992). Clay-lined landfills, as currently designed, are likely to fail after 100 years (Suter II *et al.*, 1993). In the case of radioactive wastes, the period of isolation needed has been suggested as being from at least 1000 (Wing, 1993) to 10,000 years (Nyhan *et al.* 1990) and 24,000 (Nativ, 1991) to up to millions of years (Winograd, 1974).

In the more mesic eastern part of the United States, sanitary landfill designs have been regulated by state laws in the last 20 years to facilitate leachate collection and minimize the release of contaminants to groundwater and gas to surrounding soils. Requirements specify clay-liner thickness and characteristics; plastic-liner characteristics; slopes; drainage media; leachate collection pipe network design; and clean-out and sump design. Final cover designs now required nationally (Ham, 1993) involve multiple layers to minimize water infiltration, promote lateral movement of water away from the retardation layer, and support vegetation. Landfills typically include a 0.3 to 1.5 m thick clay-liner, to inhibit subsurface leakage, covered with a compacted clay layer overlain by at least 0.15 m of topsoil to protect the clay cap from erosion and desiccation and support plants (Suter II *et al.*, 1993). Suter II *et al.* (1993) concluded that clay-lined landfills were likely to fail in the long

term. Such concerns have prompted recommendations to site waste facilities in arid areas in the U.S. to minimize problems associated with water.

In the arid West, considerable effort has been expended by the U.S. Department of Energy (DOE) to isolate radioactive and hazardous wastes from the environment (Nativ, 1991; Waugh *et al.*, 1991). These efforts include the Uranium Mill Tailings Remedial Action (UMTRA) Project (Portillo, 1992), the U.S. Ecology waste disposal facility near Beatty, Nevada (Hobson, 1992), the Integrated Test Plot (ITP) landfill demonstration at Los Alamos, New Mexico (Nyhan *et al.*, 1990), isolation barrier studies at the Idaho National Engineering Laboratory (INEL) (Anderson *et al.*, 1993), and isolation barrier studies at Hanford, Washington (Wing and Gee, 1993).

The UMTRA barriers are designed to prevent erosion, resist the exposure of the tailings, protect groundwater from tailings leachate, and minimize radon gas emissions. Efforts are progressing at 24 processing sites mainly in the western U.S. These barriers are constructed by placing a clay layer 60 to 210 cm thick over the tailings. On top of the clay is placed a loose cobble biobarrier to prevent root and animal intrusion into the clay radon barrier. Soil and plants are then placed on top of the cobble (Portillo, 1992).

The U.S. Ecology waste disposal facility near Beatty, Nevada, has been in operation for more than 25 years. Wastes are placed in 15 m deep trenches with a soil cap of at least 2.4 m thick. At grade, an additional 60-cm-thick soil cap is designed to direct precipitation away from the waste vaults. Only 114 mm of rain occurs annually at the site, which is lost subsequently to the atmosphere by evapotranspiration. The effects of

vegetation on soil water dynamics have been studied using neutron probes (Fischer, 1992; Gee, *et al.,* 1994). The risk of groundwater contamination is very low (Hobson, 1992).

The ITP landfill demonstration at Los Alamos, New Mexico (Nyhan *et al.*, 1990), compared a conventional design with an improved design. The conventional design consists of a vegetated topsoil 20 cm thick overlying 108 cm of crushed tuff backfill. The improved design consists of a vegetated topsoil 71 cm thick overlying 46 cm of gravel on 91 cm of cobble on 38 cm of crushed tuff (Nyhan *et al.*, 1990).

Isolation barrier studies at the INEL addressed radionuclide uptake by plants at the Subsurface Disposal Area (SDA). This facility buried radioactive wastes in trenches 4 m deep, which were covered by at least 0.6 m of compacted soils. Plants established on these surfaces, intruded into the wastes, and accumulated radionuclides (Arthur III, 1982). Subsequent work investigated the effect of native plants on soil water dynamics on a simulated waste trench 2.4 m deep filled with soils used to cap the SDA. Various native plants were established on the surface (Anderson *et al.*, 1987; Anderson *et al.*, 1993).

Research on the application of permanent isolation barriers to wastes at the arid Hanford Site, has been ongoing since the early 1980s (Gee *et al.*, 1993a; Link *et al.*, 1994a). The Hanford Site contains 10.3% of all low-level nuclear waste in the United States (Fisher, 1986). The minimization of infiltration into buried waste has been investigated using the capillary break concept. A capillary break is a soil structure that is a fine topsoil over loose rock (Richards 1950). Such a configuration limits downward water movement and tends to store water in the soil, where it can be recycled to the atmosphere through evapotranspiration. Using a

capillary break is one of the major design features of the permanent isolation barrier (Fig. 1) and has been investigated extensively at Hanford (Adams and Wing, 1986; Gee *et al.*, 1989; Gee *et al.*, 1993b).

The need for permanent isolation for extended periods of time means dispersal factors need to be carefully considered in the design of barriers. Elements that can disperse wastes into the environment include water, wind, plants, and animals. Plants will have significant effects on upper layers and can, potentially, compromise a barrier. Thus, it is important to determine how plants will affect the soil water balance, the stability of the surface subjected to wind and water erosion, and the potential for biointrusion into the waste (Wing, 1992; Gee *et al.*, 1993a). Our review of the role of plants on isolation barriers nationwide (Fig. 2) includes plant community dynamics, root intrusion, and how plants control wind erosion, water erosion, infiltration, and surface water budgets. Special attention is given to isolation barrier research and conditions at the Hanford Site (Fig. 3) in southcentral Washington.

Plant Community Dynamics

Plant community dynamics describe changes in the abundance of various plant species as well as the introduction and extinction of species. Short-term changes in species composition are related to disturbance and alien introductions. Long-term changes in plant communities in response to climate change could significantly alter long-term barrier performance, especially if the new conditions are outside of the design criteria of the barrier (Sackschewsky *et al.*, 1991). For example, if the climate were to become wetter, deep-rooted plants could

become established that might intrude into the buried waste in a barrier designed for shallow-rooted plants in the arid West .

The vegetation on engineered covers will likely change over time. Predicting community dynamics on engineered surfaces that are expected to function for hundreds to thousands of years becomes an important consideration. The plant community may change in response to climate or to disturbances such as fire or human disturbance. Climate change and disturbances can alter the numbers, types, and diversity of species, and may be accompanied by changes in water extraction rates. Even under the present climate, and without disturbances, species abundance, biomass production, and transpiration rates vary seasonally and from year to year in response to precipitation and temperature (Waugh *et al.*, 1994).

In eastern North America, succession on landfill caps is likely to go from grasses and weedy annual herbs to woody perennial shrubs and vines and pioneer trees to forest trees. At Oak Ridge, Tennessee, the depth of grass roots is less than 1 m, while trees can have maximum rooting depths of more than 60 m (Suter II, *et al.*, 1993). They suggest that landfill caps may be severely disturbed after 100 years because wind-thrown trees would tear holes in the surface.

In the arid west, succession may follow no inherent order or schedule in the context of species associations replacing each other over time. The arrival of alien species confounds predictability. For example, the dominance gained by alien species such as *B. tectorum* and *Salsola kali* in southeastern Washington over the last 150 years is common experience in the west. These species dominate not only cultivated and overgrazed lands at and next to the Hanford Site, but are supplanting native species

on undisturbed sites as well (Mack 1986). New aliens continue to appear, and in some cases, their potential ranges are quite large.

Alien species such as *B. tectorum* tend to foster their own persistence. Fire was important in pre-settlement ecology, but these relatively infrequent, low-temperature fires probably altered species composition little (Daubenmire 1975). In contrast, increased incidence and intensity of fire resulting from the several-fold greater litter production of *B. tectorum* (Rickard et al. 1977) may be having the long-term effect of eradicating native shrubs such as *A. tridentata.* Other alien species that are capable of dominating barrier surfaces have appeared at Hanford and include: *Centauria solstitialis*, *C. repens*, *C. diffusa*, *C. maculosa*, *Lactuca serriola, Lepidium latifolium,* and *Sisymbrium altissimum.*

The long-term development of plant communities on isolation barriers will be related to disturbance history, fire, invasive alien species, and microphytic crusts. Microphytic crusts alter the soil surface changing erosion, infiltration, evaporation, surface temperatures, nitrogen fixation, and microsites for seedling establishment compared with bare soils (West 1990). Cryptogamic crusts develop hexagonal cracking patterns that create "safe site" for plant establishment. There have been many reports of the synergistic effects of such "safe sites" on vascular plant establishment in arid and semi-arid regions. The growth of perennial bunchgrasses in southern Idaho was stimulated by moss litter (Schlatterer and Tisdale, 1969). Eckert, *et al.* (1986) found better establishment of a variety of vascular plant species in microsites of microphytic crusts than in bare interspaces in shrub-steppe ecosystems. The cracks create the microsites with improved moisture and nutrient

status, which aid in establishment. In addition, dark crusts increase soil temperature that would allow the plants to grow faster, aiding in establishment (Harper and Marble, 1988).

Change in plant community composition from 1980 to 1987 was observed on cover systems at the Grand Junction, CO, uranium mill tailings pile (Gee, *et al.*, 1989). They observed that by 1987 all cover types were dominated by annual grasses and forbs except one plot where perennial grasses had been established in 1980. They observed very few *S. kali*, which had been more common in 1980 (Beedlow, 1984). This reduction in *S. kali* presence on disturbed soils soon after revegetation and convergence to annuals has been observed in other arid systems (Allen, 1988). Beedlow (1984) examined plant communities for 40 UMTRA sites in the western U.S. and grouped them into six community types: perennial grass, shrub, forb/perennial grass, annual grass, salt-tolerant shrub, and woodland.

Plant communities of the Hanford Site that exist near isolation barrier sites are of the *Artemisia tridentata/Purshia tridentata/Poa sandbergii* and *A. tridentata/Bromus tectorum/P. sandbergii* types (Cline *et al.*, 1975). These areas also contain many other species, some of which are significant depending on the state of disturbance (Table 1). Plant community dynamics on barrier surfaces at Hanford will likely follow a chronosequence described in Waugh and Link (1988) for disturbed areas. Their analysis was based on observations contained in Rogers and Rickard (1977). The initial dominant plant found on disturbed burial sites is *S. kali*, which becomes less abundant as *B. tectorum* becomes dominant. Other species that established include *P. sandbergii*, *S. altissimum*, and *C. nauseosus*. The oldest disturbed surfaces were dominated by *B. tectorum*

and *C. nauseosus*. This is the likely outcome for barrier surfaces at Hanford if nothing is done to establish species such as *A. tridentata* that more closely resemble those of nearby native communities Current plans for revegetation of the prototype barrier surface (Gee *et al.*, 1993a) are to establish a perennial grass (*P. sandbergii* and *Sitanion hystrix*) and shrub (*C. nauseosus* and *A. tridentata*) community (Link *et al.*, 1994c).

Root Intrusion

Over time, plants will establish on barrier surfaces leading to the possible intrusion of roots into wastes. Root intrusion depends on initial plant establishment and subsequent succession, rooting depths of the plants, and the ability of roots to penetrate cap materials (Suter II *et al.*, 1993). Roots may transport wastes to the shoots (Hakonson *et al.*, 1992). Roots can modify wastes by increasing decomposition rates and by releasing exudates that mobilize metals (Browning and Molz, 1978; Cataldo *et al.*, 1987). Decomposing roots can leave channels for the movement of water and vapors through soil caps and into wastes. Roots can also dry clay layers causing shrinkage and cracking (Nyhan, 1989).

The intrusion of roots in buried landfill wastes in the mesic East is considered desirable in some instances because roots hasten the decomposition of refuse compared to fallow conditions (Molz and Browning, 1977). They found that deep-rooted trees reduced leachate volumes by transpiration and increased the rate of refuse decomposition.

The recommendation for the prevention of root intrusion into buried wastes in mesic landfills is simply to prevent the establishment of deep-

rooted trees and shrubs (Lutton, 1980). Landfills are not expected to be monitored for extended periods of time.

In the West maintenance of a deep surface soil cap sufficient to contain the rooting depths of perennial vegetation is considered an economical method of preventing roots from entering buried wastes (Anderson *et al.*, 1993). A depth of 2 m was considered sufficient for semi-arid perennial grasses at INEL. A 60 cm thick Scoria volcanic cinder or a gravel cobble placed at the bottom of a 50 cm deep high clay soil was effective in preventing root intrusion through the biobarrier for perennial grasses and *A.tridentata* at INEL (Reynolds, 1990).

In studies at Los Alamos, it was concluded that a gravel-cobble layer can keep plant roots from growing through a landfill cover and entering buried wastes (Hakonson, 1986).

The recommendation for root intrusion control for the UMTRA project is to maintain a thin cobble layer below the rooted surface soil. A thin layer consisting of rocks with a minimum diameter of 2.5 cm creating at least two void spaces was considered sufficient to inhibit root intrusion below the layer. This thin layer approach was suggested because a thick layer (20 cm) was considered impractical over the large surface areas of the UMTRA barriers (DOE, 1989).

Roots have been observed to intrude into clay radon barriers in six UMTRA project disposal cells in the arid West and in the mesic East (Pennsylvania). For example, at the Shiprock disposal cell, roots of *Tamarix pentrandra* were observed growing parallel to the interface of the bedding and radon barrier layers and up to 71 cm into the clay radon barrier. The plants had become established through at least 20 cm of coarse rock placed on top of the barrier as erosion control (Burt and Cox,

1993). Plants can transport dissolved radon gas and release radon gas to the atmosphere during transpiration (Lewis and MacDonell, 1986; 1990). Plant roots may increase radon flux by the creation of channels in the clay and drying out the underlying soil. Plant roots can deplete water from clay layers resulting in shrinkage and cracking enabling roots to grow deeper in the clay layer in search of water (Nyhan, 1989). This process will eventually compromise the clay barrier allowing roots to intrude into the tailings. Finally, Burt and Cox (1993) noted that continued plant and root growth into the clay barrier will change the performance of the barrier by the creation of a biologically active soil.

Burt and Cox (1993) felt that problems with plants were difficult to solve over long periods of time. Sand and dirt are likely to fill interstitial spaces in deep surface rock layers allowing plants to become established. Biointrusion layers will not be effective after 100 years (Van Voris *et al.* 1988; Nyhan, 1989). Deep (2 m) soil-surface layers would contain the rooting systems of shallow-rooted plants, but would be ineffective against shrubs such as *Chrysothamnus* sp. and *Sarcobatus vermiculatus*, which have maximum rooting depths of 4.5 and 9 m, respectively (Foxx *et al.*, 1984). Herbicide control was suggested as a reasonable approach for vegetation control, but only for short periods of time (Burt and Cox, 1993).

In experiments to test cover configurations for UMTRA sites at Grand Junction, Colorado, Beedow (1984) observed that roots would not penetrate a buried asphalt layer; thus, the use of an asphalt layer would form a root biointrusion layer.

In a study on vegetative covers for uranium mill tailings, Reith and Caldwell (1990) concluded that roots could be prevented from entering

buried wastes by placing a choked cobble capillary break below the rooting zone. This would prevent soils from falling into cracks in the cobble and, thus, not allow roots to enter. The capillary break would also maintain water in the upper fine soil zone.

The intrusion of roots into buried wastes has been studied at Hanford for many years. Observed rooting depths range from 35 to 300 cm for plants of the Hanford Site (Table 2). Deep-rooted plants [*Chrysothamnus nauseosus* (Klepper *et al.*, 1978); *S. kali* (Selders, 1950)] have been observed to accumulate fission products when growing over buried radioactive wastes. The presence of fission products in the shoot is a consequence of roots penetrating the radioactive wastes. Past workers have sought to prevent the intrusion of roots into buried wastes by maintaining a loose rock layer between the waste and the surface soils (Cline *et al.*, 1980). They were successful as long as an asphalt layer was present to prevent soils, and thus roots, from filling cracks in the rock layer. Caldwell (1992) suggests, from an engineering perspective, that roots can be kept out of waste zones by placing a filter between the rooted upper soil zone and a lower coarse rock layer or biointrusion layer. Roots will not grow in coarse rock where there is no water or soil. Others have prevented roots from entering wastes by keeping the surface barren of plants. This has been done by placing gravel on the surface and maintaining an herbicide program. This practice, unfortunately, leads to drainage because of the presence of the gravel and the lack of plants (Gee *et al.*, 1992; Burt and Cox, 1993).

On the Hanford prototype barrier, the chances that roots will enter buried wastes are small because there will be a loose rock layer between the waste and the surface soils. As long as this zone is dry, roots will not

enter. Even if this zone should become wetted, the asphalt layer below it should prevent roots from enter the waste below the asphalt.

Wind Erosion

Wind erosion can quickly degrade a barrier surface by mass loss of fine soils at the barrier surface and by the development of dunes near irregularities on the surface. Loss of surface soils will compromise barrier function (Suter II*et. al.*, 1993). Vegetation is the key to controlling wind erosion in semi-arid and arid lands (Leys, 1991). Dry bare soil surfaces are quickly eroded without plant or litter cover. The breakup of microphytic soil crusts because of physical disturbance increases wind erosion (Marshall, 1972). The loss of vegetative cover by fire is a leading cause of wind erosion in undisturbed ecosystems. It is anticipated that wind erosion will increase, through a decrease in plant cover, if climate conditions become hotter and dryer with global climate change (Muhs and Maat, 1993).

Wind erosion is caused by wind turbulence interacting with the soil. A wind velocity profile follows a logarithmic function with height. Vegetative canopies determine the displacement height of the wind profile. Canopies reduce wind erosion by decreasing the energy of air parcels passing through the canopy. Equations used to predict soil losses from wind erosion include vegetative cover, litter cover, and height of the canopy above the soil as predictors (Gregory, 1984). The use of wind tunnels to study wind erosion led to significant underestimates of erosional loss compared with field conditions (Leys, 1991).

There has been little research into the effects of vegetation on wind erosion for barrier surfaces in the arid West. It has been suggested that the best way to prevent wind erosion in arid regions with relatively little vegetative cover is to cover the surface with a rock rip-rap (Voorhees *et al.*, 1983). However, this would lead to increased infiltration to the waste because of reduced evapotranspiration. Wind erosion studies have been carried out at Hanford to determine the amount of admix gravel needed to minimize erosion. Admix gravels were found to significantly reduce wind erosion (Ligotke, 1993). This finding is significant as a means of protecting the surface after fires burn the vegetation off barrier surfaces or if vegetation is reduced because of hotter and drier climates.

Water Erosion

Water erosion can quickly degrade a barrier surface by sheet runoff loss of soil, and gully formation. Gully formation will lead to a severe barrier compromise. Water erosion can be controlled by placing a rock layer on the surface of the barrier. The use of surface rock layers has the negative effect of increasing infiltration. The best way to minimize water erosion is to maintain a vegetative cover. A vegetative cover is much less resistant to erosion than a rock cover and much more resistant than a cover of bare soil (DOE, 1989). Foliage and plant litter resist water erosion by intercepting and absorbing rainfall energy (Gregory, 1984) and prevents soil compaction from raindrops. Gregory (1984) developed equations to predict relative energy of raindrops reaching the soil surface that included terms for vegetative and litter cover as the primary predictors. Root systems bind soil particles while litter filters sediment

out of runoff. Above-ground plant litter increases surface roughness and slows runoff velocity. Roots and litter help maintain soil porosity and permeability, which increases infiltration and decreases runoff. Plants also dry the soil by transpiration, which delays the onset of soil saturation and runoff (Gray and Leiser, 1982).

Soil cryptogams (algae, mosses, and lichens) significantly influence water erosion especially in arid and semi-arid environments (West, 1990). Soil cryptogams reduce rainsplash erosion compared with bare soils (Tchoupopnou, 1989). The binding of soil surface particles by cryptogams reduces sediment production yet overland flow is increased because the soils are sealed (West, 1990).

Water erosion studies have been carried out at INEL to determine the effect of vegetation removal and soil disturbance in *A. tridentata* landscapes typical of SDA facilities. The study was done with a rainfall simulator with the conclusion that removal of *A. tridentata* and other vegetation down to a bare surface caused a cumulative soil loss of 100 to 1000 times that of vegetated controls (Goff *et al.*, 1993). These authors caution that their study was relevant only for short-term high-intensity summer storms. Studies are needed to assess snowmelt and the long-term effects of rainfall on the long-term erosion hazard to radioactive waste covers in shrub-steppe ecosystems.

Studies at Los Alamos compared water erosion with plots covered with natural vegetation and barley covered plots. The natural vegetation plots lost 20 times less soil than plots covered with barley (Nyhan *et al.*, 1984). Subsequent work at Los Alamos by Barnes and Warren (1988) found soil loss was minimized with a cover of shrubs and gravel mulch compared with a grass and gravel mulch cover.

Beedlow (1984) examined the effect of vegetation on water erosion for 40 UMTRA sites in the western U.S. using the Universal Soil Loss Equation (Wischmeier and Smith, 1978). He compared four vegetation types: grassland, shrub-steppe, woodland, and salt shrub and found that erosion increased going from grassland to salt shrub. Grass was most resistant to erosion because the more complete canopy reduces rainsplash energy more than the more open canopies of the salt shrub.

Water erosion studies at Hanford found vegetation significantly reduced erosion compared with a bare plot because of the reduction of raindrop energy (Walters and Gilmore, 1993).

Infiltration

Infiltration rates, the rate at which water enters the soil, are significantly related to vegetation and the effects of vegetation on soil hydraulic conductivity (Dunne, *et al.*, 1991). Infiltration characteristics are more complex in arid than in humid ecosystems. In humid areas, spatial patterns in infiltration are governed mainly by patterns in soil moisture (Yair *et al.*, 1980) while in arid ecosystems, spatial variability is strongly affected by variability in vegetation (Berndtsson and Larson, 1987). There has been little research into the effects vegetation may have on infiltration on barrier or landfill surfaces.

Dunne *et al.*, (1991) in reviewing the role of vegetation in infiltration rates noted that there have been very few studies of the effects of plants on infiltration. They did find that infiltration rates were higher under canopies than farther away from canopies (Lyford and Qashu, 1969; Johnson and Gordon, 1988; Glover *et al.*, 1962) and conclude

that increases were a consequence of the effect of vegetation on soil structure, hydraulic conductivity, and moisture characteristic curves (Blackburn, 1975).

Vegetation intercepts raindrop energy, reducing surface sealing, and thus, increasing infiltration rates. The presence of vegetation modifies soil hydraulic conductivity by creating macropores generated from root holes and the interpedal spaces between soil aggregates stabilized by organic material. Vegetation also encourages animal and insect activity, which creates macropores. The presence of macropores increases hydraulic local conductivity and infiltration rates. Macropores also allow for the escape of air, which can impede percolation (Jarrett and Hoover, 1985).

Cryptogamic mats under *A. tridentata* shrubs in semi-arid regions aided in infiltration (Johnson and Blackburn, 1989). In a review of the function of microphytic soil crusts, West (1990) notes that various researchers have found that microphytic crusts increase or decrease infiltration. Brotherson and Rushforth (1983) found less infiltration under lichens and algal crusts, but more under mosses compared with bare surfaces.

Surface Water Budgets

Vascular plants play a central role in surface water budgets of isolation barrier designs (Barnes and Rodgers, 1988). As discussed above, plants influence surface water runoff, infiltration, soil water storage, and the amount of precipitation that reaches the soil surface after evaporation in canopies. Water budgets can written as follows:

$$\Delta S = P + R - ET - G,$$

where ΔS is the change in the amount of water stored in the soil, P is precipitation, R is runon and runoff of water on the area of interest, ET is evapotranspiration, and G is drainage (Anderson *et al.*, 1993).

The amount of precipitation intercepted by canopies and prevented from entering the soil can be significant. Evans *et al.* (1976) observed that creosote bush intercepted 10% and bursage 32% of total rainfall in the Sonoran Desert. In a study of semi-arid shrubs, Návar and Bryan (1990) found that interception loss was 27% of total gross precipitation. Interception loss for a wheat canopy was measured at 33% of incoming precipitation (Leuning *et al.*, 1994). Water budget studies of isolation barrier systems have not taken into account canopy interception. This effect on P of the water budget equation will be more significant as plant communities develop and the canopy becomes denser and more structured.

The influence of vegetation in runon and runoff processes was reviewed in the section on water erosion with the conclusion that vegetation strongly reduces erosion by reducing runoff rates. The issue of transport of water onto and off of barrier caps has generally not been considered because isolation barriers are generally built on flat terrain or with small surface slopes and they have rip-rap rock or vegetated surfaces which greatly retard surface lateral flow. Of course, work has been done in the context of bare soils for barriers. In the arid West, on level sites with porous soils runoff is negligible (Anderson *et al.*, 1993).

The strongest effect of plants on the water budget is through evapotranspiration. Evapotranspiration is the combination of soil

evaporation and plant transpiration. In humid ecosystems, plants are not limited by water and, depending on soil type, substantial water can pass below the root zone and become drainage. This is why there has been a general move to site toxic waste dumps in arid regions (Nativ, 1991; Reith, 1992; Ham, 1993). In the arid West, depending on rooting depth and soil type, plants will remove all water stored in the soil column effectively minimizing drainage (Link *et al.*, 1990b; 1994b; Gee *et al.*, 1992; 1994).

Mesic Sites

In eastern humid sites, special efforts are needed to minimize infiltration of water into buried wastes. Most closure caps in the east use turf grasses as vegetative covers. At the Savannah River site, a study was conducted to determine if other types of grasses were more effective in removing water from the soil by transpiration than turf grasses (Cook and Salvo, 1993). They report that two species of bamboo successfully established on closure cap surfaces. The shallow-rooted bamboo was considered to be better than turfgrasses because they required less maintenance, removed much more water by transpiration, and reduced the chances of deep-rooted species establishing and entering the waste.

A water balance study at a low-level radioactive waste site in Illinois found that evapotranspiration was 67% of average annual precipitation, runoff was 17%, and deep percolation was 22% of average annual precipitation. The vegetation at the site was a pasture grass (*Bromus inermis*) and red clover (*Trifolium pratens*) (Healy *et al.*, 1989). This vegetation was not sufficient to prevent percolation or runoff.

Trees are very effective in drying sanitary landfills (Molz *et al.*, 1974). They found that pine and *Elaeagnus angustifolia* reduced leachate production to 6.32 cm compared with 21.82 cm of leachate produced in an unvegetated lysimeter. Recently, it has been demonstrated that vegetation in a bioengineered design can dewater or dry out lysimeters in Maryland (Schulz *et al.*, 1992). They conclude that plants can be used to dry out existing water-logged low-level waste disposal sites.

Arid Sites

In the arid West, considerable research has been conducted to determine what combinations of plant species, plant cover, and soils are best at maximizing evapotranspiration and minimizing drainage. Anderson, *et al.*, (1993), after several years of study at INEL, concluded it is possible to preclude water from reaching buried wastes by providing a sufficient cap of soil to store precipitation and establishing sufficient plant cover to deplete the stored water. They recommend that a top fine soil be 2 m deep and planted with any of four native deep-rooted perennial shrubs (*A. tridentata* subsp. *wyomingensis*) and grasses (*Agropyron desertorum, Leymus cinereus, Elymus lanceolatus*) to minimize the chances of drainage into wastes at INEL.

In studies at Los Alamos, Barnes and Warren (1988) report that a barrier cover composed of shrubs reduced soil moisture storage more than the usual grass cover. Another study at Los Alamos at the ITP site tested two barrier designs based on considerable experience at the site for their ability to minimize drainage (Nyhan *et al.*, 1990). A conventional design consisting of 20 cm of sandy loam over 108 cm of sandy silt backfill was

compared with an improved design consisting of 71 cm of topsoil over 46 cm gravel, 91 cm of river cobble, and 38 cm of sandy silt backfill. Both designs were planted with *Bouteloua gracilis* and *Agropyron smithii*. At the end of the experiment, the improved design reduced drainage through the cover by a factor of 4 compared with the conventional design. This was partially caused by the more than doubling of plant biomass on the improved design.

Beedlow (1984) examined the effects of vegetation and rocks on soil water storage at an UMTRA site near Grand Junction, Colorado. Covers with only rocks on the surface rapidly accumulated water while covers that were only vegetated or rocks with vegetation maintained the soil profile in a stable and dry condition. He suggests that vegetation will minimize the chance of drainage into the waste form.

The Nevada Test Site has been considered for long-term burial of hazardous wastes. Studies have been done to determine the role of native shrubs in the soil water balance to determine if there is a potential for drainage (Smith *et al.*, 1990). The Mojave Desert is so arid that there was no evidence of soil water moving deeper than 125 cm. All precipitation is lost by evapotranspiration during the dry part of the year with shrubs accounting for only 33% of the total evapotranspiration. In a soil water balance study at Beatty, a disturbed non-vegetated plot had increases in soil water storage while a vegetated (*Larrea tridentata)* plot had a reduction in soil water storage over a 3.3-year period (Gee, *et al.,* 1994). The implication is that a vegetated surface will minimize chances of drainage at Beatty.

It has been established at Hanford that areas dominated by shallow-rooted annuals such as *B. tectorum* (Link *et al.*, 1990a) can accumulate

water beneath the root zone that can, potentially, lead to recharge (Cline *et al.*, 1977; Gee, *et al.,* 1994). Variation in the rooting depth of deep-rooted perennials is associated with variation in soil water storage (Link *et al.*, 1990b). Soil water storage increased below the 125 cm depth in a *Pseudoroegneria spicata* dominated community in comparison with a more deeply rooted community dominated by *A. tridentata* and *P. spicata* after a fire. The presence of deeply (200 cm) rooted shrubs such as *A. tridentata* and *Grayia spinosa* at McGee Ranch has been demonstrated to extract more water from the soil profile than areas dominated by sparse vegetation. Areas dominated by these shrubs were also able to extract twice normal precipitation from the soil profile (Link *et al.*, 1994b). Soil water measurements in weighing lysimeters built to simulate the Hanford barrier design indicate that when plants (*A. tridentata*) are present twice as much water is removed from the soil column than when an area is unvegetated. In addition, vegetated lysimeters were capable of preventing drainage even under 3 times annual average precipitation conditions (Gee, *et al.,* 1993b).

Non-vascular plants or microphytic soil crusts also influence water budgets by influencing evaporation rates. Although there has been little research on the role of microphytic soil crusts on evaporation (West, 1990), Harper and Marble (1988) found that heavily crusted soils lost significantly more water than did bare areas in Utah. They concluded that the dark crusts were hotter than the light-colored soils, which led to higher evaporation.

Simulation of Water Budgets

Models for the prediction of soil water dynamics as influenced by vegetation have been developed at Los Alamos (Barnes and Rodgers, 1987; 1988), INEL (Laundre, 1990), and at Hanford (Fayer and Jones, 1990; Link *et al.*, 1993). Models can help test barrier concepts for long-term effects.

For vegetative conditions at the ITP landfill demonstration at Los Alamos, Barnes and Rodgers (1987; 1988) compared the CREAMS and HELP (Schroeder *et al.*, 1984) codes for the prediction of soil moisture. They concluded that the CREAMS code was more accurate than the HELP code. Deep-rooted shrub sites were drier than grass covered sites, which in turn, were drier than bare sites. Further studies with the CREAMS code at the ITP site indicated that drainage would be minimized with grasses (*B. gracilis, A. smithii*) growing on a trench cap 2 m thick (Nyhan, 1990). The model indicated that the tested design would still allow significant drainage through the barrier.

Simulation models were compared for vegetated shallow land burial sites at INEL (Laundre, 1990). He compared the CREAMS (Lane, 1984) and ERHYM-II (Wight, 1987) codes in a burial site with *A. tridentata* and a number of cool-desert perennial bunchgrasses for their ability to predict soil moisture, evapotranspiration, and runoff. Both models adequately predicted soil moisture and evapotranspiration but not runoff. These models were very sensitive to wilting point and vegetative cover.

Fayer and Jones (1990) describe the integration of *B. tectorum* into the UNSAT-H code to predict soil water dynamics. The parameterization of transpiration was based on potential evapotranspiration and simple equations to describe shoot and root characteristics. Link *et al.*, (1993) describe the use of the SWIM code (Ross, 1990) to simulate soil water dynamics in lysimeters containing *A. tridentata*. This effort adequately

simulated transpiration and soil water dynamics in irrigated and ambient precipitation lysimeters.

Conclusions and Recommendations

The role of vegetation on barriers and landfill surfaces has been studied extensively over the last 20 years. Our review of the role of plants on isolation barriers from the mesic East to the arid West included plant community dynamics, root intrusion, wind erosion, water erosion, infiltration, and surface water budgets. Special attention was given to isolation barrier research and conditions at the Hanford site in south-central Washington.

Plant community dynamics depend on the disturbance history of the cover. In the East, plant communities will change from alien-dominated annual communities to native and alien communities. In the arid West, plant communities will also change from alien-dominated annual communities to native and alien communities, but the rate will be slower, and final composition of plant communities less predictable. In the West, disturbed ecosystems have been dominated by alien annuals for decades. The ability of barriers to isolate wastes from the environment for thousands will depend strongly on plant community composition. Investigations into the prediction of plant community dynamics for long periods for barrier surfaces have not been carried out.

The issue of root intrusion has been intensively studied given the ability of roots to compromise barrier function and enter wastes. Many barrier and landfill designs have focused on preventing roots from intruding into wastes. Efforts to keep vegetation off barrier surfaces

have generally failed. Efforts to keep roots out of buried wastes have been most successful in the West where surface soil depth, rock, and asphalt layers have been used. Future research into this problem in the West will need to focus on the ability of barrier designs to maintain function over thousands of years during which deep-rooted plants could become established during wet periods. Investigations into paleoclimates and paleobotany will aid in barrier design for the prevention of root intrusion (Waugh *et al.,* 1994).

Numerous investigators have determined that vegetation will minimize wind and water erosion. Erosional processes are more significant in the arid West where vegetation cover is relatively low. There have been fewer studies of the relationship between vegetation cover and wind erosion than for water erosion for barriers. The role of vegetation in erosional processes for long periods of time is a difficult topic, but needs to be addressed to assess the stability of barrier surfaces for thousands of years.

There has been little research into the effects vegetation may have on infiltration on barrier or landfill surfaces. Infiltration depends on soil structure, which is determined by vegetation over time. This issue is most significant for sterile soils used on the surface of barriers. Over time these soils will become biologically active changing chemical makeup, hydrological characteristics and they will undergo pedoturbation (Boul *et al.*, 1980). There is a need to investigate the relationship between vegetation and soil development to properly understand soil physics and infiltration characteristics. Such investigations could be done in soil surfaces for which an age can be associated.

The role of vegetation in water budget studies for barrier surfaces has been relatively well studied. There has been a lack of research into vegetation's role on interception which can significantly affect the reliability of precipitation measurements in water balance studies. Continuing research is needed on the role of vegetation in soil water balance to assess the ability of barriers to function for thousands of years during which plant communities and water extraction abilities can change significantly. Models of surface hydrology will need to take into account plant community dynamics in the context of possible global climate change scenarios to increase our confidence in barrier function for thousands of years.

The role of non-vascular plants (soil cryptogamic crusts) in barrier function needs to be addressed in the context of vascular plant community dynamics, erosion, infiltration, and water budgets in arid regions.

Confidence in the ability of isolation barriers to isolate hazardous wastes from dispersion into the environment will require long-term integrated studies. Such studies will have to develop an understanding of how ecosystems develop over time and how this will influence surface stability and hydrologic properties of barriers.

Acknowledgements

Research was funded by the U.S. Department of Energy under Contract DE-AC06-76RLO 1830. We thank Glendon W. Gee for helpful comments on the manuscript and for long-term support of vegetation research. We thank George Anne O'Connor for editorial assistance.

Literature cited

Adams, M. R. and N. R. Wing. 1986. *Protective Barrier and Warning Marker System Development Plan.* RHO-RE-OL-35P, Rockwell Hanford Operations, Richland, WA.

Allen, E. B. 1988. Some Trajectories of Succession in Wyoming Sagebrush Grassland: Implications for Restoration. In: *The Reconstruction of Disturbed Arid Lands*, pp. 89-112., E. B. Allen (ed.). AAAS Selected Symposium, Westview Press. Boulder, CO.

Anderson, J. E., M. L. Shumar, N. L. Toft, and R. S. Nowak. 1987. Control of the soil water balance by sagebrush and three perennial grasses in a cold-desert environment. *Arid Soil Res. and Rehab.* 1: 229-244.

Anderson, J. E., R. S. Nowak, T. D. Ratzlaff, and O. D. Markham. 1993. Managing soil moisture on waste burial sites in arid regions. *J. Environ. Qual.* 22: 62-69.

Arthur III, W. J. 1982. Radionuclide concentrations in vegetation at a solid radioactive waste-disposal area in southeastern Idaho. *J. Environ. Qual.* 11: 394-399.

Barnes, F. J. and J. C. Rodgers. 1987. Hydrologic Modeling of Soil Water Storage in Landfill Cover Systems. In: *International Conference on Measurement of Soil and Plant Water Status*, Vol. 1, pp. 173-182., Utah State University, Logan, UT.

Barnes, F. J. and J. C. Rodgers. 1988. *Evaluation of Hydrologic Models in the Design of Stable Landfill Covers.* EPA/600/2-88/048, Cincinnati, OH.

Barnes, F. J. and J. L. Warren. 1988. Cover Technology Demonstration for Low-Level Radioactive Sites. In: *Proceedings of the Tenth Annual DOE Low-Level Waste Management Conference*, pp. 33-44., CONF-880839, Denver, CO.

Beedlow, P. A. 1984. *Revegetation and Rock Cover for Stabilization of Inactive Uranium Mill Tailings Disposal Sites: Final Report.* PNL-5105, Pacific Northwest Laboratory, Richland, WA.

Berndtsson, R. and M. Larson. 1987. Spatial variability of infiltration in a semi-arid environment. *J. Hydrol.* 90: 117-133.

Blackburn, W. H. 1975. Factors influencing infiltration and sediment production of semiarid rangelands in Nevada. *Water Resour. Res.* 11: 929-937.

Boul, S. W., F. D. Hole, and R. J. McCracken. 1980. *Soil Genesis and Classification.* The Iowa State University Press, Ames, Iowa.

Brotherson, J. D. and S. R. Rushforth. 1983. Influence of cryptogamic crusts on moisture relationships of a soil in Navaho National Monument, Arizona. *Great Basin Nat.* 43: 73-79.

Browning, V. D. and F. J. Molz. 1978. *Interaction of Root Growth and Refuse Decomposition in a Sanitary Landfill.* WRRI Bull. 32. Water Resour. Res. Inst., Auburn Univ., Auburn, AL.

Burt, C. J. and S. W. Cox. 1993. An assessment of plant biointrusion on six UMTRA project disposal cells. In: Proc. of the Waste Management Conf., Tucson, AZ. (in press)

Caldwell, J. A. 1992. Engineering Perspectives for Near-Surface Disposal. In: *Deserts as Dumps? The Disposal of Hazardous Materials in Arid Ecosystems*, pp. 161-198., C. C. Reith and B. M. Thomson (eds.). University of New Mexico Press. Albuquerque, NM.

Cataldo, D. A., C. E. Cowan, K. M. McFadden, T. R. Garland, and R. E. Wildung. 1987. *Plant Rhizosphere Processes Influencing Radionuclide Mobility in Soil.* PNL-6277. Pacific Northwest Laboratory, Richland, WA.

Cline, J. F., D. W. Uresk, and W. H. Rickard. 1975. *Characterization of Plant Communities Adjacent to the B-C Controlled Area and REDOX Pond Areas on the 200 Area Plateau*, BNWL-1916, UC-70, Pacific Northwest Laboratory, Richland, WA.

Cline, J. F., D. W. Uresk, and W. H. Rickard. 1977. Comparison of soil water used by a sagebrush-bunchgrass and a cheatgrass community. *J. Range Manage.* 30: 199-201.

Cline, J. F., K. A. Gano, and L. E. Rogers. 1980. Loose rock as biobarriers in shallow land burial. *Health Physics* 39: 497-504.

Cook, J. R. and S. K. Salvo. 1993. *Selection and Cultivation of Final Vegetative Cover for Closed Waste Sites at the Savannah River Site, SC,* WSRC-MS-92-513, Westinghouse Savannah River Company, Aiken, SC.

Daubenmire, R. F. 1975. Plant succession on abandoned fields, and fire influences, in a steppe area in southeastern Washington. *Northw. Sci.* 49: 36-48.

DOE (U.S. Department of Energy) 1989. Technical Approach Document: Revision II, UMTRA-DOE/AI-050425.0002, DOE UMTRA Project Office, Albuquerque Operations Office, Albuquerque, NM.

Downs, J. L., L. L. Cadwell, and S. O. Link. 1993. Root Intrusion, Root Distribution Studies. In: *Hanford Site Protective Barrier Development Program: Fiscal Year 1993 Highlights*, pp. 2.7-2.9, L. L. Cadwell, S. O. Link, G. W. Gee (eds.). PNL-8741, Pacific Northwest Laboratory, Richland, WA.

Dunne, T., W. Zhang, and B. F. Aubry. 1991. Effects of rainfall, vegetation, and microtopography on infiltration and runoff. *Water Resour. Res.* 27: 2271-2285.

Eckert, R. E. Jr., M. K. Wood, W. H. Blackburn, F. F. Peterson, J. L. Stephens, and M. S. Meurisse. 1986. Effects of soil-surface morphology on

emergence and survival of seedlings in big sagebrush communities. *J. Range Manage.* 39: 414-420.

Evans, D. D., T. W. Sammis, and J. B. Asher. 1976. *Plant Growth and Water Transfer Interactive Processes under Desert Conditions.* US/IBP Desert Biome Res. Memo. 76-33, Utah State Univ., Logan, UT.

Fischer, J. M. 1992. *Sediment Properties and Water Movement Through Shallow Unsaturated Alluvium at an Arid Site for Disposal of Low-Level Radioactive Waste Near Beatty, Nye County, Nevada.* U.S. Geological Survey Water-Resources Investigation Report. 92-4032.

Fisher, J. N. 1986. *Hydrogeologic Factors In The Selection Of Shallow Land Burial For The Disposal Of Low-Level Radioactive Waste.* U.S. Geological Survey Circular 973. 22 pp.

Fayer, M. J. and T. L. Jones. 1990. *UNSAT-H Version 2.0: Unsaturated Soil Water and Heat Flow Model.* PNL-6779, Pacific Northwest Laboratory, Richland, WA.

Foxx, T. S., G. D. Tierney, and J. M. Williams. 1984. *Rooting Depths of Plants on Low-Level Waste Disposal Sites.* LA-10253-MS. Los Alamos National Laboratory, Los Alamos, NM.

Gee, G. W., R. R. Kirkham, J. L. Downs, and M. D. Campbell. 1989. *The Field Lysimeter Test Facility (FLTF) at the Hanford Site: Installation and Initial Tests.* PNL-6810, Pacific Northwest Laboratory, Richland, WA.

Gee, G. W., M. D. Campbell, H. D. Freeman, and J. F. Cline. 1989. *Assessment of Cover Systems at the Grand Junction, Colorado, Uranium Mill Tailings Pile: 1987 Field Measurements.* PNL-6810, Pacific Northwest Laboratory, Richland, WA.

Gee, G. W., M. J. Fayer, M. L. Rockhold, and M. D. Campbell. 1992. Variations in recharge at the Hanford Site. *Northw. Sci.* 66: 237-250.

Gee, G. W., L. L. Cadwell, H. D. Freeman, M. W. Ligotke, S. O. Link, R. A. Romine, W. H. Walters Jr., and N. R. Wing. 1993a. *Testing and Monitoring Plan For The Permanent Isolation Surface Barrier Prototype.* PNL-8391, Pacific Northwest Laboratory, Richland, WA.

Gee, G. W., D. G. Felmy, J. C. Ritter, M. D. Campbell, J. L. Downs, M. J. Fayer, R. R. Kirkham, and S. O. Link. 1993b. *Field Lysimeter Test Facility Status Report IV: FY1993.* PNL-8911, Pacific Northwest Laboratory, Richland, WA.

Gee, G. W., P. J. Wierenga, B. J. Andraski, M. H. Young, M. J. Fayer, and M. L. Rockhold. 1994. Variations in water balance and recharge potential at three western desert sites. *Soil Sci. Soc. Am. J.* 58:63-72.

Glover, P. E., J. Glover, and M. D. Gwynne. 1962. Light rainfall and plant survival in E. Africa. II. Dry grassland vegetation. *J. Ecol.* 50: 199-206.

Goff, B. F., G. C. Bent, and G. E. Hart. 1993. Erosion response of a disturbed sagebrush steppe hillslope. *J. Environ. Qual.* 22: 698-709.

Gray, D. M. and A. T. Leiser. 1982. *Biotechnical Slope Protection and Erosion Control*, Von Nostrand Reinhold Co., Inc., New York, NY.

Gregory, J. M. 1984. Prediction of soil erosion by water and wind for various fractions of cover. *Trans. ASAE* 27: 1345-1350.

Hakonson, T. E. 1986. *Evaluation of Geologic Materials to Limit Biological Intrusion into Low-Level Radioactive Waste Disposal Sites.* LA-10286-MS, Los Alamos National Laboratory, Los Alamos, NM.

Hakonson, T. E., L. J. Lane and E. P. Springer. 1992. Biotic and Abiotic Processes. In: *Deserts as Dumps? The Disposal of Hazardous Materials in Arid Ecosystems*, pp. 101-146, C. C. Reith and B. M. Thomson (eds.). University of New Mexico Press. Albuquerque, NM.

Ham, R. K. 1993. Overview and implications of U.S. sanitary landfill practice. *Air and Waste* 43:187-190.

Harper, K. T. and J. R. Marble. 1988. A Role for Nonvascular Plants in Management of Arid and Semiarid Rangelands. In: *Application of Plant Sciences to Rangeland Management and Inventory*, pp. 135-169, P. T. Tueller (ed.). Martinus Nijhoff/W. Junk, Amsterdam.

Healy, R. W., J. R. Gray, M. P. de Vries, and P. C. Mills. 1989. Water balance at a low-level radioactive-waste disposal site. *Water Resour. Bull.* 25: 381-390.

Hobson, N. 1992. Disposal of Low-Level Radioactive and Hazardous Chemical Waste: U.S. Ecology's Beatty, Nevada Disposal Facility. In: *Deserts as Dumps? The Disposal of Hazardous Materials in Arid Ecosystems*, pp. 263-279, C. C. Reith and B. M. Thomson (eds.). University of New Mexico Press. Albuquerque, NM.

Jarrett, A. R. and J. R. Hoover. 1985. Evaluating the effect of increased concentrations of CO_2 on infiltration rate. *Trans. ASAE* 28: 179-182.

Johnson, C. W. and N. D. Gordon. 1988. Runoff and erosion from rainfall simulator plots on sagebrush rangeland. *Trans. ASAE* 31: 421-427.

Johnson, C. W. and W. H. Blackburn. 1989. Factors contributing to sagebrush rangeland soil loss. *Trans. ASAE* 32: 155-160.

Klepper, E. L., L. E. Rogers, J. D. Hedlund, R. G. Schreckhise, and K. R. Price. 1978. Radiocesium movement in a gray rabbit brush community, pp. 725-737. In: *Environmental Chemistry and Cycling Processes,* D. C. Adriano and I. L. Brisbin. (eds.), Proceedings of the Mineral Cycling Symposium. CONF-760429, Atlanta.

Klepper, E. L., K. A. Gano, and L. L. Cadwell. 1985. *Rooting Depth and Distributions of Deep-Rooted Plants in the 200 Area Control Zone of the Hanford Site.* PNL-5247, Pacific Northwest Laboratory, Richland, WA.

Lane, L. J. 1984. *Surface Water Management: A User's Guide to Calculate a Water Balance using the CREAMS Model.* LA-10177-M UC-70B, Los Alamos National Laboratory, Los Alamos, NM.

Laundre, J. W. 1990. Calibration and sensitivity testing of CREAMS and ERHYM-II computer models for use in a cold desert environment. *Arid Soils Res. and Rehab.* 4:205-222.

Leuning, R., A. G. Condon, F. X. Dunin, S. Zegelin, and O. T. Denmead. 1994. Rainfall interception and evaporation from soil below a wheat canopy. *Agr. and For. Meteor.* 67: 221-238.

Lewis, B. G. and M. M. MacDonell. 1986. Radon transport through a cool-season grass. *J. Environ. Radioact.* 4: 123-132.

Lewis, B. G. and M. M. MacDonell. 1990. Release of radon-222 by vascular plants: Effect of transpiration and leaf area. *J. Environ. Qual.* 19: 93-97.

Leys, J. F. 1991. Towards a better model of the effect of prostrate vegetation cover on wind erosion. *Vegetatio* 91: 49-58.

Ligotke, M. W. 1993. *Soil Erosion Rates Caused by Wind and Saltating Sand Stresses in a Wind Tunnel.* PNL-8478, Pacific Northwest Laboratory, Richland, WA.

Link, S. O., G. W. Gee, and J. L. Downs. 1990a. The effect of water stress on phenological and ecophysiological characteristics of cheatgrass and Sandbergs's bluegrass. *J. Range Manage.* 43: 506-513.

Link, S. O., G. W. Gee, M. E. Thiede, and P. A. Beedlow. 1990b. Response of a shrub-steppe ecosystem to fire: Soil water and vegetational change. *Arid Soils Res. and Rehab.* 4: 163-172.

Link, S. O., Kickert, R. N., Fayer, M. J., and Gee, G. W. 1993. *A Comparison of Simulation Models for the Prediction of Soil Water Dynamics in Bare and Vegetated Lysimeters.* PNL-8675, Pacific Northwest Laboratory, Richland, WA.

Link, S. O., N. R. Wing, and G. W. Gee. 1994a. The development of permanent isolation barriers for buried wastes in cool deserts: Hanford, Washington. (Submitted to the *Ann. New York Acad. Sci.*)

Link, S. O., W. J. Waugh, J. L. Downs, M. E. Thiede, J. C. Chatters, and G. W. Gee. 1994b. Effects of coppice dune topography and vegetation on soil water dynamics in a cold-desert ecosystem. *J. Arid Environ.* 27: (In press)

Link, S. O., L. L. Cadwell, C. A. Brandt, J. L. Downs, R. E. Rossi, G. W. Gee. 1994c. *Biointrusion Test Plan For The Permanent Isolation Surface Barrier Prototype.* PNL-9411, Pacific Northwest Laboratory, Richland, WA.

Lutton, R. J. 1980. *Evaluating Cover Systems for Solid and Hazardous Waste.* Interagency Agreement No. EPA-IAG-D7-01097, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Lyford, F. P. and H. K. Qashu. 1969. Infiltration rates affected by desert vegetation. *Water Resour. Res.* 5: 1373-1376.

Mack, R. N. 1986. Alien Plant Invasion into the Intermountain West: A Case History. In *Ecology of Biological Invasions of North America and Hawaii*, pp. 191-213, H. A. Mooney and J. A. Drake (eds). Ecological Studies, Vol. 58, Springer-Verlag, New York.

Marshall, J. K. 1972. Principles of Soil Erosion and its Prevention. In *The Use of Trees and Shrubs in the Dry Country of Australia*, pp. 90-107, N. Hall (ed.). Australian Government Publishing Service, Canberra.

Molz, F. J., S. R. van Fleet and V. D. Browning. 1974. Transpiration drying of sanitary landfills. *Ground Water* 12: 394-397.

Molz, F. J. and V. D. Browning. 1977. Effect of vegetation on landfill stabilization. *Ground Water* 15: 409-415.

Muhs, D. R. and P. B. Maat. 1993. The potential response of eolian sands to greenhouse warming and precipitation reduction on the Great Plains of the U.S.A. *J. Arid Environ.* 25: 351-361.

Nativ, R. 1991. Radioactive waste isolation in arid zones. *J. Arid Environ.* 20: 129-140.

Návar, J. and R. Bryan. 1990. Interception loss and rainfall redistribution by three semi-arid growing shrubs in northeastern Mexico. *J. Hydrol.* 115: 51-63.

Nyhan, J. W., G. L. DePoorter, B. J. Drennon, J. R. Simanton, and G. R. Foster. 1984. Erosion of earth covers used in shallow land burial at Los Alamos, New Mexico. *J. Environ. Qual.* 13: 361-366.

Nyhan, J. W. 1989. *Development of Technology for the Long-Term Stabilization and Closure of Shallow Land Burial Sites in Semiarid Environments.* LA-1128-MS. Los Alamos National Laboratory, Los Alamos, NM.

Nyhan, J. W. 1990. Calibration of the CREAMS model for landfill cover designs limiting infiltration of precipitation at waste repositories. *Haz. Waste & Haz. Materials* 7: 169-184.

Nyhan, J. W., T. E. Hakonson, and B. J. Drennon. 1990. A water balance study of two landfill cover designs for semiarid regions. *J. Environ. Qual.* 19: 281-288.

Portillo, R. S. 1992. Mill Tailings Remediation: The UMTRA Project. In: *Deserts as Dumps? The Disposal of Hazardous Materials in Arid Ecosystems*, pp. 281-302, C. C. Reith and B. M. Thomson (eds.). University of New Mexico Press. Albuquerque, NM.

Reith, C. C. and J. A. Caldwell. 1990. *Vegetative Covers for Uranium Mill Tailings*. Technical Contribution. Jacobs Engineering Group, Inc., Albuquerque, NM.

Reith, C. C. 1992. Introduction: Waste Management and the Arid-Land Disposal Concept. In: *Deserts as Dumps? The Disposal of Hazardous Materials in Arid Ecosystems*, pp. 3-19, C. C. Reith and B. M. Thomson (eds.). University of New Mexico Press. Albuquerque, NM.

Richards, L. A. 1950. Laws of soil moisture. Trans. Am. Geophys. Union 31: 750-756.

Rickard, W.H., D. W. Uresk, and J. F. Cline. 1977. Productivity Response to Precipitation by Native and Alien Plant Species. In *Proceedings of the Symposium on Terrestrial and Aquatic Ecological Studies of the Northwest*, pp. 1-4. R. D. Andrews, R. L. Carr, F. Gibson, B. Z. Lang, R. A. Soltero and K. C. Swedberg (eds.). Eastern Washington State College Press, Cheney, WA.

Reynolds, T. D. 1990. Effectiveness of three natural biobarriers in reducing root intrusion by four semi-arid plant species. *Health Physics* 59: 849-852.

Rogers, L. E. and W. H. Rickard. 1977. *Ecology of the 200 Area Plateau Waste Management Environs: A Status Report.* PNL-2253, Pacific Northwest Laboratory, Richland, WA.

Ross, P. J. 1990. *SWIM - a Simulation Model for Soil Water Infiltration and Movement.* CSIRO Division of Soils, Davies Laboratory, Townsville, Queensland, Australia.

Sackschewsky, M. R., J. C. Chatters, S. O. Link, and C. A. Brandt. 1991. *Protective Barrier Program: Test Plan for Plant Community Dynamics.* WHC-EP-0380, Westinghouse Hanford Company, Richland, WA.

Schlatterer, E. F. and E. W. Tisdale. 1969. Effects of litter of *Artemisia, Chrysothamnus* and *Tortula* on germination and growth of three perennial grasses. *Ecology* 50: 869-873.

Schroeder, P. R., J. M. Morgan, T. M. Walski, and A. C. Gibson. 1984. *The Hydrologic Evaluation of Landfill Performance (HELP) Model. Vol. I and II.* EPA/530-SW-84-010, U.S. Environmental Protection Agency, Washington, DC.

Schulz, R. K., R. W. Ridky, and E. O'Donnell. 1992. *Control of Water Infiltration into Near Surface LLW Disposal Units.* NUREG/CR-4918, U.S. Nuclear Regulatory Commission, Washington, DC.

Selders, A. A. 1950. *The Absorption and Translocation of Fission Elements by Russian Thistle.* HW-18034, General Electric Company, Richland, WA.

Smith, S. D.; Leary, K.; Herr, C; Hokett, S. 1990. Water relations and transpiration of native vegetation in the vicinity of Yucca Mountain, Nevada. In *Proceedings-Symposium on Cheatgrass Invasion, Shrub Die-Off, and Other Aspects of Shrub Biology and Management*, pp. 250-255. E. D. McArthur, E. M. Romney, S. D. Smith, and P. T. Tueller (eds.). Gen. Tech. Rep. INT-276. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT.

Suter II, G. W., R. J. Luxmoore, and E. D. Smith. 1993. Compacted soil barriers at abandoned landfill sites are likely to fail in the long term. *J. Environ. Qual.* 22: 217-226.

Tchoupopnou, E. (1989). Splash from microphytic soil crusts following simulated rain. M. Sc., Utah State Univ., Logan, UT

Van Voris, P., D. A. Cataldo, C. E. Cowman, N. R. Gordon, J. F. Cline, F. G. Burton, and W. E. Skeins. Long-term controlled release of herbicides, ACS Symposium Series 371, Pesticide Formulations, Innovations, and Developments; American Chemical Society, Washington D. C.

Voorhees, L. D., M. J. Sale, J. W. Webb, and P. J. Mulholland. 1983. *Guidance for Disposal of Uranium Mill Tailings: Long-Term Stabilization of Earthen Cover Materials.* NUREG/CR-3199, Oak Ridge National Laboratory, Oak Ridge, TN.

Walters, W. H. and B. G. Gilmore. 1993. Water Erosion. In: *Hanford Site Protective Barrier Development Program: Fiscal Year 1993 Highlights* , pp. 2.22-2.23, L. L. Cadwell, S. O. Link, G. W. Gee (eds.). PNL-8741, Pacific Northwest Laboratory, Richland, WA.

Waugh, W. J., and S. O. Link. 1988. *Barrier Erosion Control Test Plan: Gravel Mulch, Vegetation and Soil Water Interactions.* WHC-EP-0067, Westinghouse Hanford Company, Richland, WA.

Waugh, W. J., M. E. Thiede, L. L. Cadwell, G. W. Gee, H. D. Freeman, M. R. Sackschewsky, and J. F. Relyea. 1991. Small lysimeters for documenting arid site water balance, pp. 151-159. In: *Lysimeters for Evapotranspiration and Environmental Measurement*, R.G. Allen, T.A. Howell, W.O. Pruitt, I.A. Walter, and M.E. Jensen (eds.) Proceedings of the International Symposium on Lysimetry, American Society of Civil Engineers, New York.

Waugh, W. J., J.C. Chatters, G.V. Last, B.N. Bjornstad, S.O. Link, and C.R. Hunter. 1994. *Barrier Analogs: Long-Term Performance Issues, Preliminary Studies, and Recommendations.* PNL-9004, Pacific Northwest Laboratory, Richland, WA.

West, N. E. 1990. Structure and function of microphytic soil crusts in wildland ecosystems of arid and semi-arid regions. *Adv. Ecol. Res.* 20: 179-223.

Wight, J. R. 1987. *ERHYM-II: Model Description and users Guide for the Basic Version.* U.S. Department of Agriculture Research Service ARS-59.

Wing, N. R. 1992. *A Peer Review of the Hanford Site Permanent Isolation Surface Barrier Development Program.* WHC-MR-0392, Westinghouse Hanford Company, Richland, WA.

Wing, N. R. 1993. *Permanent Isolation Surface Barrier: Functional Performance.* WHC-EP-0650, Westinghouse Hanford Company, Richland, WA.

Wing, N. R. and G. W. Gee. 1993. The development of permanent isolation surface barriers: Hanford Site, Richland Washington, USA, pp 357-362. In: *Geoconfine 93*, M. Arnould, M. Barrès and B. Côme (eds.). Balkema. Rotterdam.

Winograd, I. J. 1974. Radioactive waste storage in the arid zone. EOS (Amer. Geophys. Union Trans.) 55: 884-894.

Wischmeier, W. H. and D. D. Smith. 1978. *Predicting Rainfall Erosion Losses - A Guide to Conservation Planning.* Agriculture Handbook No. 537. U.S. Department of Agriculture, Washington, DC

Yair, A., H. Lavee, R. Bryan and A. Adar. 1980. Runoff and erosion processes and rates in the Zin valley badlands, Northern Negev, Isreal. *Earth, Surf. Processes* 5: 205-225.

TABLES

TABLE 1. Plant Species Observed at Pit 29 and the PUREX Sand Pit (Waugh *et al.*, 1994).

Scientific Name	Common Names	Pit 29	PUREX Sand Pit
Ambrosia acanthicarpa	Bur ragweed	x	x
Amsinckia tessellata	Tessellate fiddleneck		x
Artemisia tridentata	Big sagebrush	x	x
Balsamorhiza careyana	Carey's balsamroot	x	x
Bromus tectorum	Cheatgrass	x	x
Chaenactis douglasii	Hoary false-yarrow	x	
Comandra pallida	Pale bastard toadflax	x	x
Chrysothamnus viscidiflorus	Green rabbitbrush		x
Cryptantha circumscissa	Matted cryptantha	x	
Cryptantha pterocarya	Winged cryptantha	x	
Cymopteris terebinthinus	Turpentine cymopteris		x
Descurainia pinnata	Western tansymustard	x	x
Erysimum asperum	Rough wallflower	x	
Lactuca serriola	Prickly lettuce	x	
Machaeranthera canescens	Hoary aster	x	
Oenothera pallida	Pale evening primrose	x	
Phlox longifolia	Longleaf phlox	x	
Poa sandbergii	Sandberg's bluegrass	x	x
Psoralea lanceolata	Scurf pea	x	
Purshia tridentata	Bitterbrush	x	

Salsola kali	Russian thistle	x	x
Sisymbrium altissimum	Tumblemustard	x	x
Sitanion hystrix	Squirreltail grass	x	
Sphaeralcea munroana	Munro's globe-mallow	x	x
Stipa comata	Needle-and-thread grass	x	
Tragopogon dubius	Yellow salsify	x	x

TABLE 2. Greatest observed rooting depth of various species at the Hanford site.

Site	Species	Soil type	Root Depth (cm)	Reference
Pit 29 Trench	*Artemisia tridentata*	sandy-loam to sand	230	Waugh *et al.*, (1994)
PUREX Trench	*Artemisia tridentata*	"	170	"
	Purshia tridentata		260	"
PUREX Pit Face	*Ambrosia acanthicarpa*	"	132	"
	Salsola kali		137	"
	Sisymbrium altissimum		97	"
Lower Snively	*Bromus tectorum*	silt-loam	60	Downs *et al.*, (1993)
McGee Ranch	*Artemisia tridentata*	silt-loam	200	Link *et al.*, (1994b)
	Grayia spinosa		200	
300 N Grass	*Bromus tectorum*	loamy-sand to sand	50	Link *et al.*, (1990a)
	Poa sandbergii	"	35	"
200 Area	*Chrysothamnus nauseosus*	sandy-loam to sand	240	Klepper *et al.*, (1978)

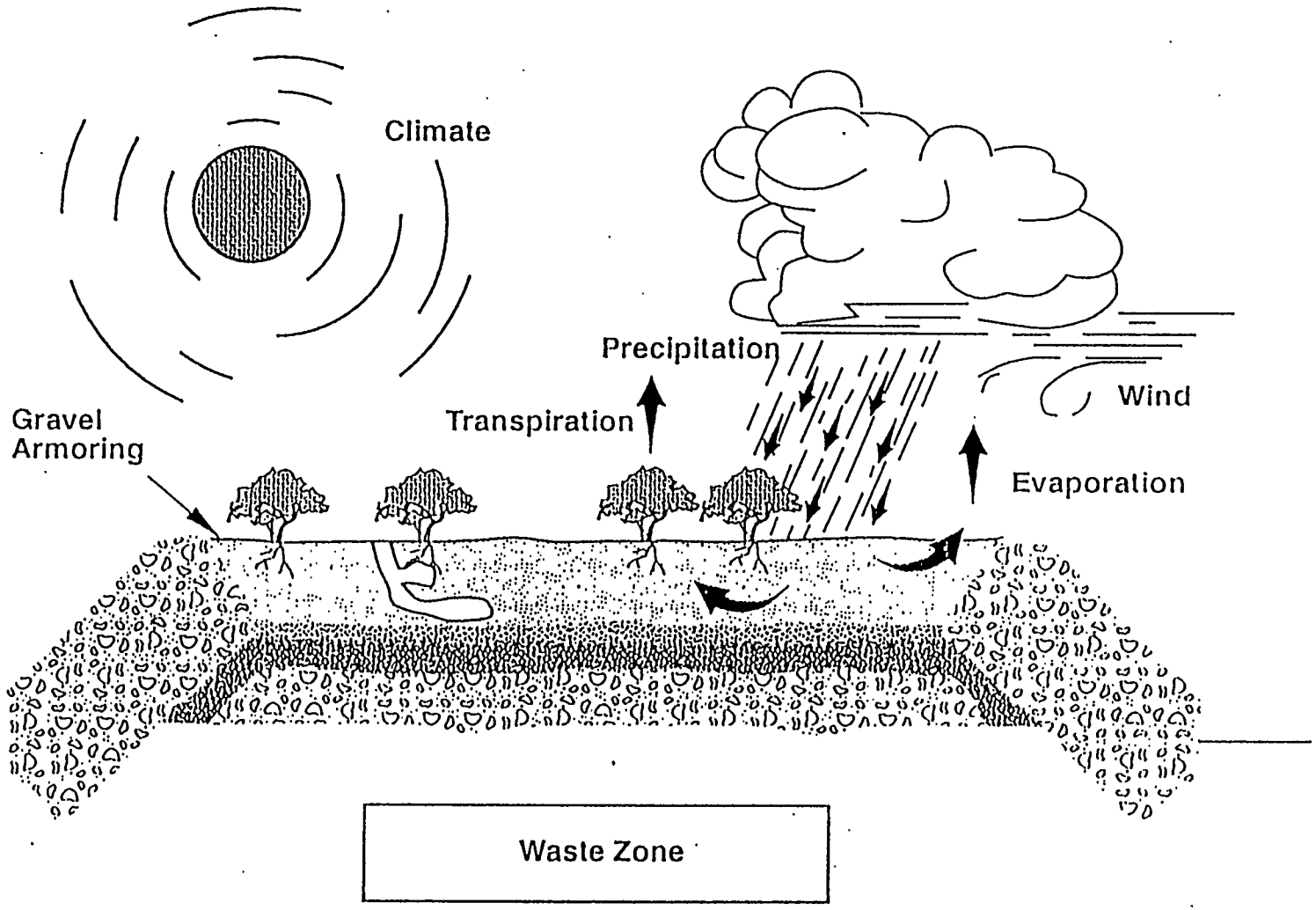
200 Area	*Chrysothamnus nauseosus*	sandy-loam to sand	183	Klepper *et al.*, (1985)
	A. tridentata	"	200	"
	S. kali	"	172	"
	A. acanthicarpa	"	162	"
	P. tridentata	"	300	"
	Aster canescens	"	155	"
	Stipa comata	"	160	"
	Oryzopsis hymenoides	"	125	"
	Cymopteris terebinthinus	"	145	"
	G. spinosa	"	195	"
	Eriogonum niveum	"	150	"
	Chrysothamnus viscidiflorus	"	160	"
	Lactuca serriola	"	85	"
	Balsamorhiza careyana	"	150	"

Figures

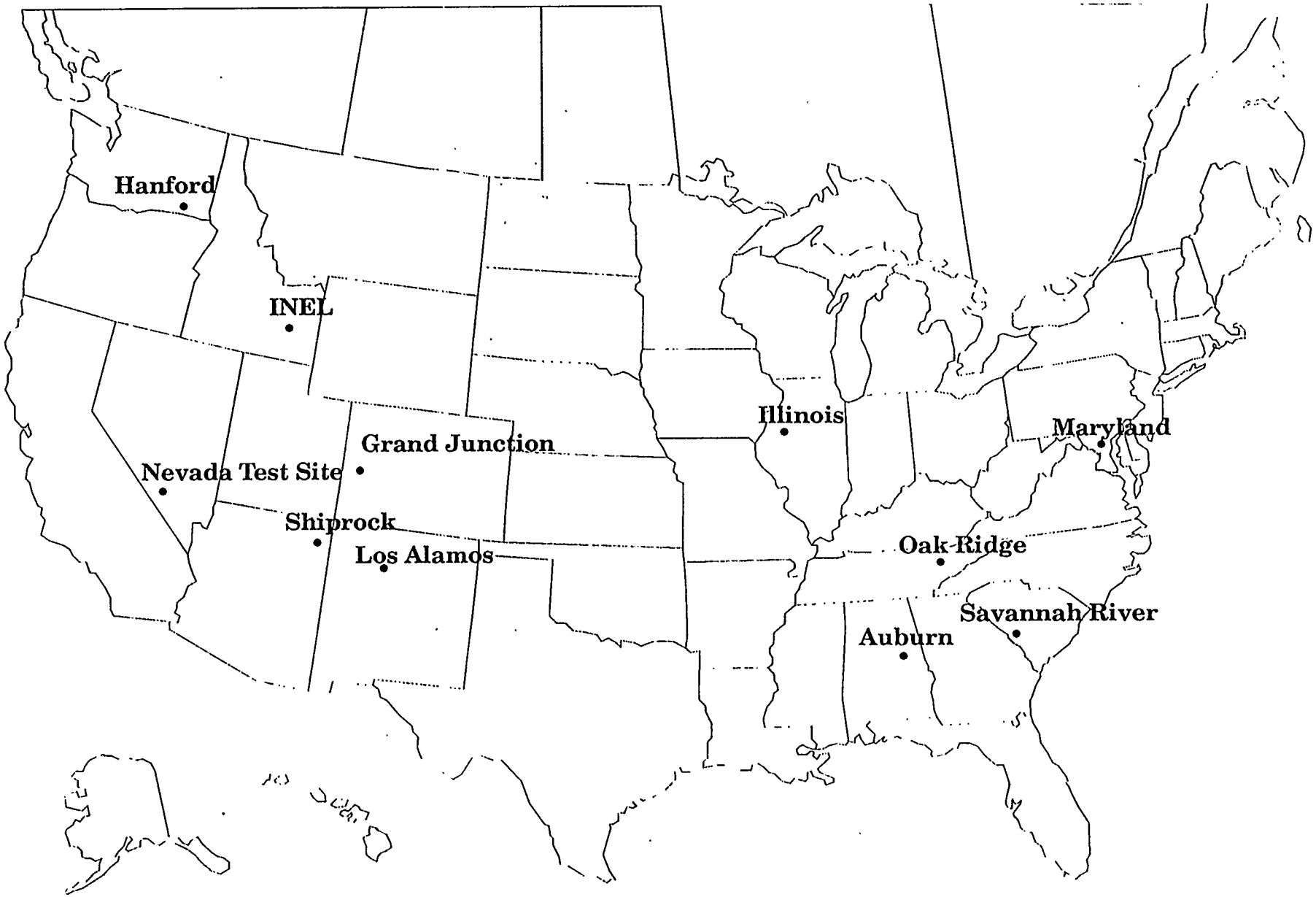
Figure 1. Typical isolation barrier.

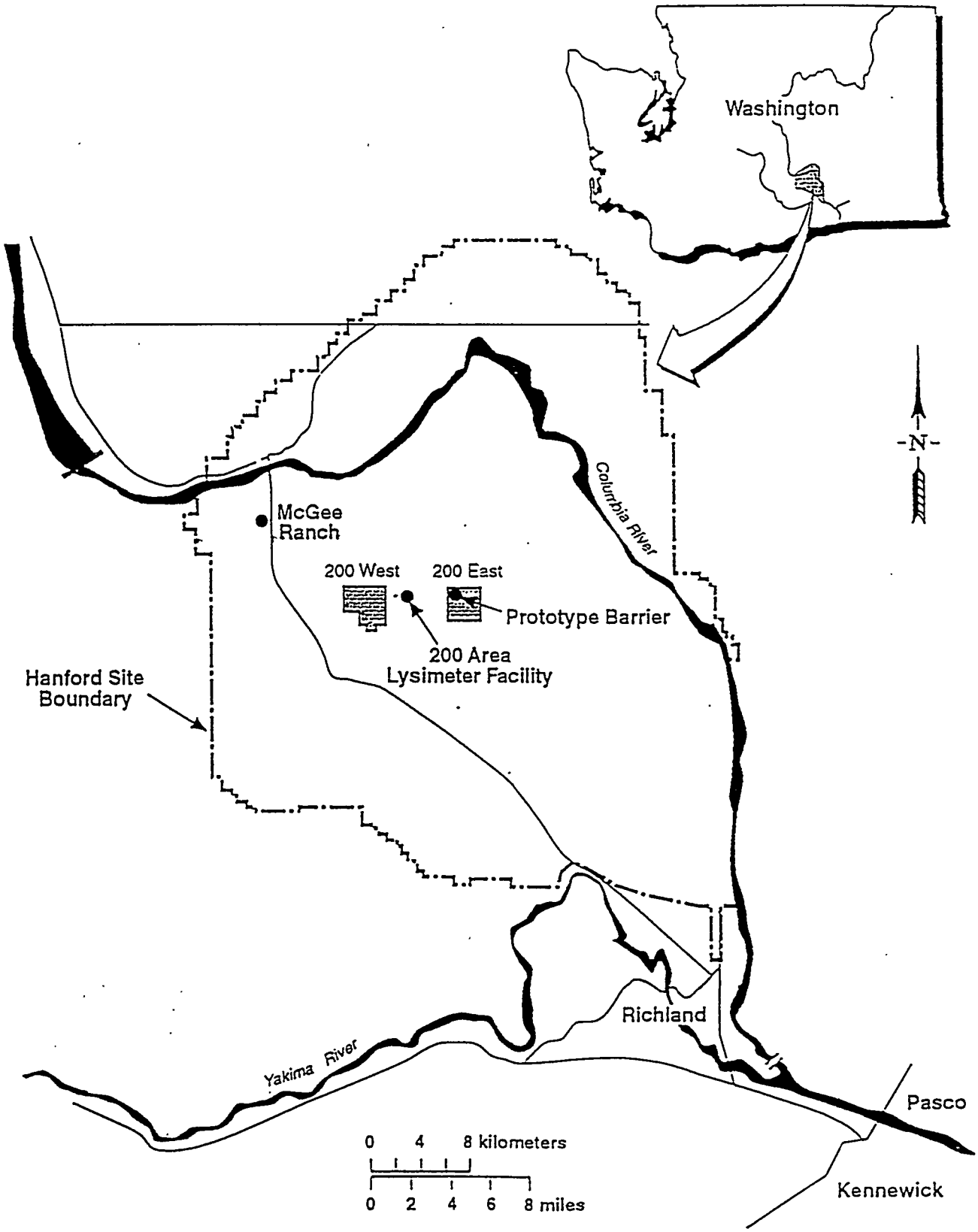
Figure 2. Location of barrier projects reviewed in this study.

Figure 3. Hanford Site map showing the location of McGee Ranch and the permanent isolation barrier in the 200 Area Plateau.


Climate
Precipitation
Wind
Transpiration
Gravel
Armoring
Evaporation
Waste Zone

S9311067.3


Hanford
INEL
Grand Junction
Nevada Test Site
Shiprock
Los Alamos
Illinois
Maryland
Oak Ridge
Savannah River
Auburn


Washington
N
Columbia River
McGee Ranch
200 West
200 East
Prototype Barrier
200 Area Lysimeter Facility
Hanford Site Boundary
Richland
Yakima River
Pasco
Kennewick
0 4 8 kilometers
0 2 4 6 8 miles

S9311067.1