STOLLER-ESER-56

Evapotranspiration Caps for the Idaho National Engineering and Environmental Laboratory: A Summary of Research and Recommendations

Jay E. Anderson and Amy D. Forman

January 2003

STOLLER-ESER-56

Evapotranspiration Caps for the Idaho National Engineering and Environmental Laboratory: A Summary of Research and Recommendations

Jay E. Anderson
Center for Ecological Research and Education
Department of Biological Sciences
Idaho State University
Pocatello, ID 83209

and

Amy D. Forman
S. M. Stoller Corporation
Idaho Falls, ID 83401

The exclusion of ecological principles is, in my opinion, the greatest failing of current practices in designing capping technology.

T. E. Hakonson (1994)

Prepared by:

Stoller

IDAHO STATE UNIVERSITY

Prepared for:

U. S. Department of Energy - Idaho Operations Office
Environmental Surveillance, Education and Research Program
Contract DE-AC07-00ID13658

Executive Summary

Shallow land burial is the most common method for disposing of industrial, municipal, and low-level radioactive waste. However, conventional landfill practices are often inadequate to preclude movement of hazardous materials to groundwater or biota. Generally, hydrologic processes account for most waste repository problems. Percolation of water into the waste zone may leach and transport toxic materials into groundwater. Water in the waste zone may also encourage growth of plant roots and transport of toxic materials to aboveground foliage.

Most final covers on hazardous waste landfills in the United States must meet performance standards specified under subtitle C of the Resource Conservation and Recovery Act (RCRA). The US Environmental Protection Agency's (EPA) recommended cover for arid and semiarid climates includes a compacted clay layer overlain by an impervious flexible membrane liner (FML), which, in turn, is overlain by vegetated topsoil. Such covers have been widely used, but they often fail in dry climates because the compacted clay layer dries and cracks. RCRA regulations allow EPA to consider alternative cap designs demonstrated to meet equivalency criteria. (EPA is currently drafting new guidance that permits the use of several alternative cap designs.)

In humid regions, keeping water received as precipitation from reaching interred wastes can be a formidable problem, but, in arid or semiarid areas, a natural ecosystem analogue provides a simple and elegant solution. Because the potential to evaporate water far exceeds the amount of water they receive as precipitation, many aridland ecosystems return all of the water received to the atmosphere via evapotranspiration (ET) each year. The soil serves as a reservoir, temporarily storing precipitation that is not immediately evaporated. In turn, plants extract that water from the soil and return it to the atmosphere. Hence, soil and plants are the principal components of an ET cap. The soil must be sufficiently deep to store water received, and a healthy stand of perennial plants must be present to empty that storage reservoir during each growing season.

This report summarizes the results of nearly two decades of research that demonstrate the effectiveness of ET caps at the Idaho National Engineering and Environmental Laboratory (INEEL). The products of this research are specific recommendations for construction and maintenance of ET caps at the INEEL.

The first phase of this research demonstrated that a soil cap 2 m in depth supporting a healthy stand of perennial, drought-tolerant plants should preclude water from reaching buried wastes at the INEEL. However, several issues related to the performance of ET caps were not addressed in the initial study, including:

1) the impacts of placing biological intrusion barriers and associated capillary breaks in an ET cap,

2) the potential effects of climate change on cap performance, and

3) the performance of a diverse community of native plants compared to that of monocultures.

To address these issues, the Protective Cap/Biobarrier Experiment (PCBE) was initiated in 1993. The ultimate goal was to confidently recommend an effective, economical ET cap for the INEEL and climatically similar repositories, a cap constructed of natural materials that will function with minimal maintenance over the long term as a natural ecosystem.

The PCBE was a field-scale experiment, consisting of three replicates of four cap configurations, two vegetation types, and three irrigation treatments. The four cap configurations were:

1) soil-only caps consisting of 2.0 m of homogeneous soil.

2) shallow-biobarrier caps that included a biobarrier consisting of 0.3 m of river cobble sandwiched between 0.1 m layers of crushed gravel. This biobarrier was placed at a depth of 0.5 m within 2 m of soil, for a total cap thickness of 2.5 m.

3) deep-biobarrier caps having a 0.5-m biobarrier at a depth of 1 m within 2 m of soil. The biobarrier was identical to that of shallow biobarrier caps.

4) RCRA caps having 1 m of soil overlying a flexible membrane liner and 0.6 m of compacted clay.

The biobarriers used in the cap designs described above were primarily intended to preclude animal burrowing; however, under some circumstances they may also restrict growth of plant roots.

Precipitation regimes were ambient precipitation, 200 mm of supplemental irrigation applied at four biweekly intervals during summer, and 200 mm of supplemental irrigation applied rapidly in late fall or early spring. The two vegetation types were a mix of twelve native species and a monoculture of crested wheatgrass. The period of study (1994-2000) included near extremes of both low and high annual precipitation for the 50 years of record at the INEEL.

Under ambient precipitation and summer irrigation treatments, all cap types performed satisfactorily. **Given the present and predicted climate for the upper Snake River Plain, a landfill cap constructed according to any of the cap configurations in this study should prevent water received as precipitation from reaching interred wastes.** The results indicate that even a large increase in summer precipitation would not adversely impact cap performance. **With increased winter precipitation (fall/spring irrigation treatment) the soil-only and biobarrier caps were still capable of storing and returning to the atmosphere far more moisture than the precipitation expected under current climate change scenarios.** Thus, the soil-only and biobarrier caps should preclude water from reaching buried wastes, even with a considerable increase in winter precipitation.

Despite generally satisfactory performance, there were important differences that translate to advantages or disadvantages of the various cap configurations. For the RCRA cap, 1 m of soil was inadequate to store ambient precipitation received during 1995, an exceptionally wet year. Furthermore, under fall/spring irrigation, RCRA caps often had little reserve storage capacity at the beginning of a growing season, and drainage off the FML was sometimes observed. Therefore, provisions would have to be made for disposing of water that would occasionally drain off the cap over the FML. This would increase construction complexity and cost of the RCRA cap.

Roots of numerous species were able to bridge the 0.5-m thick biobarrier and extract water from the underlying soil, so indeed it is possible to have a portion of the storage reservoir below a biobarrier. However, 0.5 m of soil above a biobarrier was insufficient to store the precipitation received in most years, so water routinely percolated into the soil below, providing a reservoir of deep soil moisture. Placement of the biobarrier at a shallow depth caused strong selection for gray rabbitbrush, a native shrub known to rely primarily on deep moisture reserves. Encouraging the growth of this deeply rooting species could result in intrusion of roots into buried wastes if any water was available in the waste zone. Although animals will not burrow through a biobarrier having a meter of overlying soil, we do not have definitive evidence that a biobarrier will preclude burrowing if the overlying soil is considerably thinner. Thus, another disadvantage of the shallow-biobarrier configuration is that it may not provide sufficient depth of soil to accommodate the needs of burrowing mammals and insects, which might encourage such species to burrow into and possibly through the biobarrier.

Under ambient precipitation and summer irrigation, there was seldom any change in water content below deep biobarriers, and 1 m of soil above the biobarriers was often sufficient to store ambient precipitation plus fall/spring irrigation. Aside from precluding burrowing animals, the greatest value of the biobarrier was the capillary break it created in the soil profile of the biobarrier cap designs. The capillary break created between fine textured soil overlying the biobarrier and the gravel at the top of the biobarrier maximized the amount of water stored in the overlying soil.

Using a combination of transplanting and seeding, we readily established diverse plant communities on the experimental plots. Shrubs, perennial grasses, and perennial forbs all grew vigorously. This study repeatedly demonstrated that a mixture of perennial species would use all of the plant available water in a 2-m storage reservoir each year, even when the soil in that reservoir was completely saturated early in the growing season. The monocultures of crested wheatgrass also established quickly and grew vigorously. However, after supplemental irrigation to facilitate establishment and a very wet growing season in 1995, the stands of crested wheatgrass were so dense that they became self-inhibiting, and live cover on those plots decreased by about 50%. Crested-wheatgrass caps receiving supplemental irrigation did not use all of the plant-available water each year. Higher end-of-season water content on crested-wheatgrass caps was likely attributable to both lower vegetative cover and the absence of shrubs. Shrubs such as sagebrush and the rabbitbrushes remain active late in the growing season, continuing to extract soil moisture after many grasses and forbs are senescent.

Given these results, we recommend two cap configurations: a soil-only cap consisting of a 2-m depth of homogeneous soil or a cap consisting of a 1.2-m depth of homogeneous soil overlying a 0.5-m thick gravel/cobble intrusion barrier. Caps constructed according to either of these configurations should preclude virtually any precipitation water from reaching interred wastes. A major advantage of a soil-only cap is simplicity of construction, but a relatively large amount of soil is required. Construction cost will depend largely on availability of soil and the distance it must be transported. If fill soil is limited and if gravel and cobble are readily available, then a cap incorporating a biobarrier and requiring less soil may be less expensive.

We recommend that, if a biobarrier is used, it should be placed at the bottom of the soil reservoir. Although 1 m of soil above a biobarrier was generally adequate to store precipitation received, water did percolate below the biobarrier on two of 18 deep-biobarrier subplots under ambient conditions during 1995, the wettest year on record at the INEEL. Therefore, we recommend a minimum of 1.2 m of soil overlying a biobarrier. A cap consisting of 1.2 m of soil overlying the capillary break at the top of a biobarrier should be more than adequate to store precipitation received, even during exceptionally wet years. Furthermore, this configuration should prevent intrusion by burrowing animals, and it should restrict root growth so long as the underlying materials have little or no plant available water.

For new burial sites, we recommend constructing a level cap on grade with surrounding terrain. This eliminates the necessity of accommodating drainage off cap layers, eliminates side slope problems, and reduces the potential for wind or water erosion. For ET caps constructed to cover existing landfills or contaminated soil, it may be necessary to construct the entire cap above grade. In such a case, it may be desirable to configure the cap with a slight slope on the surface (e.g., 2%) to help prevent pooling of water on the surface following snowmelt or heavy precipitation. For any cap constructed over an existing landfill or contaminated soil, we recommend placing a biobarrier on top of the existing cover or soil. The capillary break created by the biobarrier will help ensure that no moisture moves into the contaminated materials. The new cap should be constructed late in the growing season when the soil of the existing landfill or contaminated area is dry. This will reduce the likelihood of roots growing from the new cap into the contaminated zone.

For a cap to function effectively and consistently, a healthy stand of perennial, drought-adapted plants is essential. The plant community should be self-maintaining. An analogue to a natural sagebrush-perennial grass community performed better than a perennial grass monoculture. Additionally, diverse communities are generally better able to withstand disturbance, such as fire or global climate change, and maintain ecosystem function (i.e. remove water from the soil-storage reservoir) than monocultures. Therefore, we recommend establishing a diverse community of perennial species consisting of shrubs, perennial grasses, and perennial forbs on ET caps at the INEEL. Plants should be established as quickly as possible within the first growing season to ensure that water does not percolate into the dry material at the bottom of the cap and encourage root growth into the waste zone. Specific recommendations for plant materials and planting techniques are included.

We conclude that an ET cap constructed according to the recommendations above will preclude precipitation water from reaching interred wastes at the INEEL and climatically similar sites. The recommended cap configurations provide a low cost, low maintenance alternative to EPA's recommended RCRA cap and to more complex, highly engineered designs.

ACKNOWLEDGMENTS

The Protective Cap/Biobarrier Experiment (PCBE) was funded by the Department of Energy through contracts to Idaho State University issued by the Environmental Science and Research Foundation and by S. M. Stoller Corporation. The idea of using evapotranspiration (ET) caps at the Idaho National Engineering and Environmental Laboratory resulted from discussions between Drs. Doyle Markham and Tom Hakonson in the early 1980's. Dr. Markham's vision, encouragement, and continued support are gratefully acknowledged. The many intellectual contributions of Drs. Robert Nowak and Nancee Toft during our first ET-cap project influenced the design of the PCBE. Drs. Tim Reynolds, John Laundre, and Eric Limbach helped design the PCBE cap configurations and experimental layout. Dr. Reynolds was involved in many aspects of the project and deserves much credit for its success. Monica Thurston produced the blueprints for construction of the PCBE. Construction was supervised by Dr. Eric Limbach and Mark Deitrich. Robert Wilson, P. E., served as project engineer for design and construction. Steve Hansen and Bud Rasmuson assisted with construction, planting and maintenance of vegetation, and emplacement of the irrigation system. Teresa Ratzlaff and Eric Duffin were responsible for collection, management, and analyses of data, for maintenance of the site and equipment, and for applying the irrigation treatments. Dr. Amy Morris was responsible for collecting and analyzing vegetation data. Numerous students and summer interns assisted with construction, planting, and data collection. Doug Halford, Jake Jacobson, George Schnieder and Drs. Roger Blew, Jody Waugh, and Richard Inouye provided thoughtful reviews that substantially improved this report. We express our sincere thanks to all of these individuals.

TABLE OF CONTENTS

1.0 INTRODUCTION

Shallow land burial is the most common method for disposing of industrial, municipal, and low-level radioactive waste, but in recent decades it has become apparent that conventional landfill practices are often inadequate to prevent movement of hazardous materials into groundwater or biota (Suter et al. 1993, Daniel and Gross 1995, Bowerman and Redente 1998). Generally, most waste repository problems result from hydrologic processes. When wastes are not adequately isolated, water received as precipitation can move through the landfill cover and into the wastes (Nyhan et al. 1990, Nativ 1991). Presence of water may cause plant roots to grow into the waste zone and transport toxic materials to aboveground foliage (Arthur 1982, Hakonson et al. 1992, Bowerman and Redente 1998). Likewise, percolation of water through the waste zone may transport contaminants into groundwater (Fisher 1986, Bengtsson et al. 1994). If an impervious liner underlies the wastes, water moving through the wastes can pool on the liner and leach toxic or radioactive compounds from the wastes. Avoidance of such “bathtub” conditions is a major concern in designing radioactive waste landfills (10 CFR §61.51). Liners with leachate collection systems may prevent “bathtubs,” but require extensive maintenance.

Most final caps on hazardous waste landfills in the United States are required to meet performance standards specified under subtitle C of the Resource Conservation and Recovery Act (RCRA; 40 CFR §264). The US Environmental Protection Agency’s (EPA) recommended RCRA cap for arid and semiarid climates includes a compacted clay layer overlain by a synthetic nonpermeable liner, which, in turn, is overlain by vegetated topsoil (USEPA 1989). Such “traditional covers” have been widely used, but they often fail in dry climates because the compacted clay layer dries and cracks (Suter et al. 1993, Daniel and Gross 1995), often due to plant root intrusion into the clay layer. Plant roots dry the clay layer causing it to crack; the cracks then provide preferential pathways of flow for water into the waste zone. The EPA guidelines acknowledge that their recommended design may not be very effective in arid regions and that such covers are expensive and difficult to construct. RCRA regulations allow EPA to consider alternative designs demonstrated to meet the performance standards. (EPA is currently drafting new guidance that permits the use of several alternative cap designs.)

1.1 Evapotranspiration Caps

In humid regions, keeping water received as precipitation from reaching interred wastes can be a formidable problem, but in arid or semiarid areas, a simple and elegant “ecological engineering" solution exists. Under arid climates, the potential to evaporate water far exceeds the amount of water received from precipitation. To preclude water from percolating into buried wastes, the water must be stored on site until it can be evaporated. This is precisely how natural aridland ecosystems function. The soil serves as a reservoir, temporarily storing precipitation that is not immediately evaporated. In turn, plants extract that water from the soil and return it to the atmosphere. Hence, soil and plants are the principal components of what has become known as an evapotranspiration cap.

Evapotranspiration (ET) can be thought of as the opposite of precipitation. It includes water evaporated directly from the soil or other surfaces and water lost from plant leaves via transpiration. In natural arid ecosystems, plants typically use *all* of the available water in the soil each year. Thus, if a soil cap is sufficiently deep to store all water received and if a healthy stand of perennial plants is present to empty that storage reservoir during each growing season, no water will drain through the soil and into buried wastes. Such a natural ecosystem analog should be stable and require minimal maintenance over the long term.

Until recently, the literature concerning landfill closures often mentioned the role of plants in stabilizing a site to control erosion, but the more important role that plants play in removing water from throughout a soil profile received less attention (e.g., Nativ 1991, Suter et al. 1993). Direct evaporation removes water from relatively shallow depths of soil (only about 20 cm on vegetated caps), whereas plants typically extract water from the entire soil profile on a 2 m soil cap (Anderson et al. 1991). Earlier research at the Idaho National Engineering and Environmental Laboratory (INEEL) showed that soils without vegetation lose relatively little water over a growing season (Anderson et al. 1993). If plants are not present, most of the soil will reach field capacity within a year or so and then remain at that level (ibid.). Consequently, any significant precipitation event likely will cause drainage. Porro (2001) reported similar results from unvegetated test caps at the INEEL that were irrigated until drainage occurred. Following cessation of drainage, water content on all caps remained high, and melting snow resulted in drainage in subsequent years (ibid.). Researchers at the Hanford Site in eastern Washington found that drainage would eventually occur from unvegetated soil caps under very low annual precipitation (mean = 160 mm annually; Sackschewsky et al. 1995). Indeed, plants are essential on an ET cap to empty the soil's storage reservoir each year.

Over a growing season in an arid or semiarid climate, plants can use enormous amounts of water if it is plentiful. For example, alfalfa growing in southern Idaho can extract 12 mm of water from the soil in a single hot summer day. Average water use of an irrigated alfalfa crop over a growing season is typically about 8 mm per day (Wright and Jensen 1972), equivalent to roughly four times the average annual precipitation of the area. A stand of Great Basin wildrye (Le*ymus cinereus*), a native bunchgrass, used over 530 mm of water during one growing season at the INEEL (Anderson et al. 1987). That is 2.4 times the mean annual precipitation for the area. As we further document in this report, the native vegetation of the INEEL has the potential to use far more water than would be expected to fall under the present or foreseeable climates, provided that the soil is sufficiently deep to store the precipitation received.

This report summarizes the results of nearly two decades of research that demonstrate the effectiveness of ET caps at the Idaho National Engineering and Environmental Laboratory. In addition, we provide specific recommendations for their construction and maintenance at the INEEL.

2.0 Concepts, Terminology, and Calculations

2.1 Units of Measurement

Precipitation and ET are typically expressed as depths of water, in millimeters (mm) or inches (in.). The amount of water in the soil is commonly expressed as a percentage of the total soil volume. Soil water content can be estimated by weighing a soil sample before and after drying and then calculating the percent water by weight. This value is then multiplied by the bulk density of the soil (its undisturbed mass per unit volume) to convert percent water to a volume basis, referred to as volumetric water content. It is often convenient to express the amount of water in a soil profile as a depth of water in the same unit used for precipitation and ET. Percent water on a volumetric basis is readily converted to depths. For example, if a soil contains 30% water by volume, a meter depth of that soil will contain 300 mm of water.

2.2 Water Balance of Terrestrial Ecosystems

Water entering an ecosystem must be equal to that leaving plus the change in the amount stored in the soil or biota. Consider the water balance of a small plot of land (Figure 1). Water reaches the plot as precipitation (**P**) or as surface runoff from adjacent areas ($\mathbf{R_i}$). (We will ignore springs and groundwater moving to the surface). Water leaves the plot by **ET**, by surface runoff from the plot to adjacent areas (**R_o**), or by groundwater drainage (**G**). If inputs are greater than outputs, or vice versa, the amount of water stored in the soil (**S**) will change (**ΔS**). Thus, the water balance equation is merely a detailed statement of the law of conservation of matter.

These terms can be combined into a simple expression for the water balance of the plot:

$$P + R_i = ET + R_o + G + \Delta S \qquad (1)$$

All of the units are expressed as mm of water per unit time.

On relatively level sites having porous soils, surface runoff is negligible so that the terms R_i and R_o can be ignored. If no water passes below the rooting zone, G is equal to zero. Hence, the water balance equation then becomes:

$$ET = P + \Delta S \qquad (2)$$

We used this simplified form of the equation to estimate ET from experimental plots at the INEEL when we were certain that groundwater drainage was negligible (see Section 5.2.5 for a discussion of how we assessed groundwater drainage). We did not calculate ET for time periods in which drainage was occurring, such as during irrigation to breakthrough trials (see Section 5.2.6 for a description the irrigation to breakthrough trials).

2.3 Water-Storage Capacity of Soil.

Consider a uniform soil in which the upper part of the profile has been saturated with water and this overlies a relatively dry, unwetted zone. At first, redistribution of water within the profile is quite rapid because strong “suction gradients” from the dry soil augment the gravitational forces (Hillel 1998). With time, however, the downward flux slows as the suction gradients diminish and hydraulic conductivity of the soil decreases (Campbell and Norman 1998, Hillel 1998). Interaction of sorption and desorption at the ***wetting front*** further inhibit redistribution, causing the wetted zone to retain more water than would be expected at equilibrium (Hillel 1998). The overall result is a logarithmic decrease in the rate of distribution with time so that after several days the water content of the wetted zone becomes relatively constant. The amount of water remaining in the profile at that quasi constant state, assuming that no water has been removed by ET, is referred to as ***field capacity***. Hillel (1998, p. 465) stresses that field capacity is not an “intrinsic physical property” of a soil; redistribution is continuous and “exhibits no abrupt ‘breaks’ or static levels.” Thus, the decision of when the downward flux of water has become negligible is subjective. Despite these difficulties, however, reliable estimates of an operational field capacity (Campbell and Norman 1998) can be made using repeated measurements of soil-moisture content with depth *in situ*.

Plants differ in their capacity to extract water from a drying soil (Ritchie 1981). Perennial species native to arid regions may dry a soil more completely than will crops or species found in wetter areas. The extent to which a particular species depletes soil moisture is termed the ***lower limit of extraction***. The lower limit of extraction is estimated from the amount of water remaining in the soil when plant growth and activity completely stop (Ritchie 1981). Water remaining in the soil at that point is bound so tightly to soil particles that plants cannot remove it. At the lower limit of extraction, hydraulic conductivity of the dry soil is so low that drainage is negligible (see Section 7.7). Anderson et al. (1987) reported that the lower limit of extraction was very similar among drought-adapted species growing at the INEEL. Field capacity and the lower limit of extraction depend on soil texture, type of clay present, organic matter content, soil structure, and the kinds of plants present. Therefore, estimates of these values must be made *in situ* (Ratliff et al. 1983).

These concepts can be illustrated using soil moisture profiles. Soil moisture profiles depict the vertical distribution of moisture in the soil on various sampling dates. Each line shows volumetric soil moisture content as a function of depth for a particular sampling date. Differences in line position from date to date reflect the magnitude of change in moisture storage in the soil column. As moisture is extracted over the growing season, lines for consecutive dates move leftward across the graph. An increase in water storage is indicated by the profile line moving to the right from one sampling date to the next. When the position of lines from one sampling date to the next is virtually unchanged at particular depths, no appreciable change in moisture storage has occurred.

Soil moisture profiles are depicted in Figure 2, which shows changes in soil moisture through the 1999 growing season for an experimental plot at the INEEL. The line for 25 March depicts moisture content at the beginning of the growing season. Over the next few weeks there was some downward redistribution of water in the profile, and by May 20 the wetting front reached a depth of 1.2 m. Lines for subsequent dates show depletion of water until, by July 15, virtually all water available to plants has been extracted. The lines corresponding to August 12 or September 20 indicate the lower limit of extraction.

The difference between field capacity and the lower limit of extraction is the ***effective water-storage capacity*** of a soil. Examination of numerous soil-moisture profiles such as shown in Figure 2 indicates that field capacity for the experimental cap soil used at the INEEL is about 28% moisture by volume and that the lower limit of extraction is about 15%. The effective water-storage capacity is therefore 13%, which means that a 1-m depth of soil could store 130 mm of water annually. It should be noted that a considerable portion of the precipitation that falls annually occurs during the growing season, and is quickly returned to the atmosphere through evaporation and transpiration. Only water received from large precipitation events, or when plants are inactive and potential evapotranspiration is low, will be stored for any length of time. Thus, the water-storage capacity needed in the soil at any one time will be less than the total annual precipitation.

2.4 Biological Intrusion Barriers

Biological intrusion barriers (biobarriers) are structures incorporated into hazardous-waste covers to restrict burrowing by animals and, under some circumstances, growth of plant roots. Biobarriers typically consist of a layer or layers of cobble, gravel, or similar materials such as scoria (Reynolds 1990). Research has demonstrated that a layer of rock placed within a protective soil cap will restrict the depth to which mammals can burrow (Hakonson et al. 1983, Hakonson 1986, Reynolds 1990). Tunneling by ants can be obstructed by sandwiching a layer of cobble between layers of gravel placed in the soil (Johnson and Blom 1997, Gaglio et al. 1998).

2.5 Capillary Breaks

A capillary break is formed when a layer of fine-textured soil overlies a layer of coarse-textured sand or gravel. The matrix potential of the fine-textured soil prevents water from flowing into the larger pores of the sand or gravel until the fine soil approaches saturation at the interface (Hillel 1998). Therefore, capillary breaks function to increase the storage capacity of the overlying soil by limiting the movement of the wetting front. The gravel/cobble biobarriers used in our research at the INEEL create a capillary break at the bottom of the overlying soil.

3.0 Climate and Vegetation of the INEEL

The INEEL occupies some 2315 km^2 of the western edge of the upper Snake River Plain in southeastern Idaho, USA (43^o N, 112^o W). Average elevation of the area is about 1500 m.

Mean annual temperature is 5.6^{o}C, and the frost-free period averages about 90 days. During summer, low humidities and clear skies result in high temperatures and high evaporative demand during the day and relatively low temperatures at night due to rapid radiational cooling. Winters are cold, with several months having mean temperatures below freezing. Snow cover may persist for periods of a few weeks to over 2 months.

The INEEL lies in the rain shadow of the numerous mountain ranges of central Idaho. Average annual precipitation is 220 mm (Figure 3); however, there is substantial year-to-year variation in both annual and growing season precipitation, with total water-year (October – September) precipitation varying from 72 mm to 342 mm in the past century (Figure 4). Precipitation tends to be uniformly distributed throughout the year, except for a peak early in the growing season (Figure 3). On average, 37% of the annual precipitation falls during April, May, and June; May and June are the wettest months (Figure 3). Melting snow and spring rains account for most of the annual recharge of moisture into the soil (Caldwell 1985, Anderson et al. 1987). In a typical year, most plant growth occurs in spring and the water available to plants is largely depleted by early to mid summer (Anderson et al. 1987). However, some plants remain active through August and September if water is available.

The natural vegetation at the INEEL typically consists of a shrub overstory with an understory of perennial grasses and forbs (herbaceous plants other than grasses and sedges). The dominant shrub is Wyoming big sagebrush (*Artemisia tridentata* subspecies *wyomingensis*). Basin big sagebrush (*A. tridentata* subspecies *tridentata*) may be dominant, or co-dominant with Wyoming big sagebrush, on sites having deep soils or sand accumulation (Shumar and Anderson 1986). Other important shrubs include winterfat (*Ceratoides lanata*), green rabbitbrush (*Chrysothamnus viscidiflorus*), spiny hopsage (*Grayia spinosa*), prickly phlox (*Leptodactylon pungens*), horse-brush (*Tetradymia canescens*), and broom snakeweed (*Gutierrezia sarothrae*).

Common native grasses include thick-spiked wheatgrass (*Elymus lanceolatus*), bottlebrush squirreltail (*E. elymoides*), Indian ricegrass (*Achnatherum hymenoides*), and needle-and-thread (*Hesperostipa comata*). Bluebunch wheatgrass (*Pseudoroegneria spicata*) is common at higher elevations, especially on alluvial fans and the slopes of the buttes. Great Basin wildrye (Le*ymus cinereus*) occurs, often in nearly pure stands, on deep soils between lava ridges; it also is found in areas where sand accumulates or on disturbed sites such as mounds resulting from rodent burrowing.

Compared with much of the sagebrush steppe region, the INEEL supports a high diversity of forbs (Anderson et al. 1996). Some common native forbs are tapertip hawksbeard (*Crepis acuminata*), Hood's phlox (*Phlox hoodii*), false yarrow (*Chaenactis douglasii*), globemallow (*Sphaeralcea munroana*), bastard toadflax (*Comandra umbellata*) and various milkvetches (*Astragalus* sp.), buckwheats (*Eriogonum* sp.), paintbrushes (*Castilleja* sp.) and mustards (*Arabis* spp., *Stanleya viridiflora*, and *Lappula redowski*).

Numerous introduced annual and biennial species occur at the INEEL (Anderson et al. 1996). In 1995, those species contributed 11% of the vegetative cover on long-term vegetation plots at the INEEL (Anderson and Inouye 2001). Invasive weeds are among the most common of these introduced species. These include cheatgrass (*Bromus tectorum*), tumbling mustard (*Sisymbrium altissimum*), and Russian thistle (*Salsola kali*). Potential problems posed by these species to the performance of ET caps are addressed in Section 7.8.

4.0 Early Research on ET Caps at the INEEL

In 1983, ten simulated waste trenches were established at the INEEL Experimental Field Station. Each 3- by 10-m trench was excavated to a depth of 2.4 m and then filled with soil from Spreading Area B at the INEEL. The same soil was used for capping radioactive waste at the INEEL Subsurface Disposal Area; it consisted of 26% sand, 54% silt, and 20% clay.

This project examined the capacity of four species of drought tolerant perennial plants to deplete soil moisture on these plots. Three species of perennial grasses and one shrub were planted in monospecific stands on two trenches each. Sagebrush, crested wheatgrass (*Agropyron cristatum* and *A. desertorum*), and Great Basin wildrye were transplanted from natural stands in the area in the fall of 1983; streambank wheatgrass (*Elymus lanceolatus*) was seeded onto two plots in March of 1984. From 1984 through 1986, soil water content was monitored on all plots under natural precipitation. In 1987 and 1988, natural precipitation on one set of plots was supplemented via drip irrigation to simulate very high precipitation years. The results of this study are summarized below; additional details can be found in Anderson et al. (1987, 1991, 1993).

All four species established good cover during the first growing season, and ET from all eight plots exceeded the mean annual precipitation for the INEEL (Figure 5). Transplanted crested wheatgrass and Great Basin wildrye plants extracted water from the soil to depths of 1.6 m and 2.2 m, respectively during the first growing season. ET from the crested-wheatgrass plots and Great Basin wildrye plots averaged 443 mm and 385 mm, respectively (Figure 5). These ET values were a consequence of high precipitation (1983-84 water-year precipitation = 346 mm) and the fact that the fill soil was relatively moist when the plots were constructed. By the end of the second growing season, the entire soil profile of crested wheatgrass and Great Basin wildrye plots was at the lower limit of extraction, and the plants dried the soil to that level each subsequent season through 1988 when the project terminated.

Streambank wheatgrass, which was started from seed, extracted water to a depth of 1.2 m during the first growing season, removing 186 mm of water from the soil storage reservoir. Average ET from the two plots was 386 mm (Figure 5). Thus, in its first growing season, a species started from seed reduced the amount of water in storage sufficiently to provide ample storage for the next winter-spring recharge period. This species dried the soil to the lower limit of extraction in its third year and in each subsequent year of the experiment.

Sagebrush plants extracted water to a depth of 1 m during the first season after transplanting, removing 91 mm of water from storage. ET was 291 mm (Figure 5). Sagebrush roots extracted water from throughout the 2.2 m soil profile in the second season, and by the end of the third and subsequent seasons the soil was uniformly dry throughout.

These results indicated that any of the four species can remove water to a depth of at least 2.2 m in a waste cap. Furthermore, the results indicated that any of the species could use all of the water that might be stored in the soil during a very wet year. To gain an indication of the maximum amount of water that stands of these species could use, we supplemented precipitation on a crested wheatgrass and a Great Basin wildrye plot so that they received 600 mm of water in 1987 and 460-500 mm of water in 1988. Irrigated sagebrush and streambank wheatgrass received about 366 mm of water, equal to the maximum annual precipitation on record at the INEEL. With the exception of the streambank wheatgrass plot, which had suffered considerable plant mortality due to repair of subsidence, ET on all plots exceeded 366 mm (Figure 5). ET from the Great Basin wildrye plot was 636 mm, 2.8 times the mean annual precipitation.

5.0 THE PROTECTIVE CAP/BIOBARRIER EXPERIMENT – OBJECTIVES AND DESIGN

Although our first ET-cap project demonstrated that a soil cap 2 meters in depth supporting a healthy stand of perennial plants should be more than adequate to preclude water from reaching buried wastes at the INEEL, that project did not address several issues related to performance of ET caps. Knowledge gaps included the impacts of placing biological intrusion barriers in an ET cap, the potential effects of climate change on cap performance, and the performance of a diverse community of native plants compared to that of monocultures. The Protective Cap/Biobarrier Experiment (PCBE) was initiated in 1993 to address these issues. The ultimate goal of the PCBE was to confidently recommend an effective, economical ET cap for the INEEL and climatically similar repositories, a cap constructed of natural materials that will function with minimal maintenance over the long term as a natural ecosystem. Here, we summarize the rationale and objectives, methods, and results of the project. Results from two cap configurations that we recommend for the INEEL are emphasized. A comprehensive report including a discussion of experimental design, statistical analyses and associated power, and detailed results for all cap configurations is available (Anderson and Forman 2002).

5.1 Rationale and Objectives

Researchers at the INEEL and elsewhere have demonstrated that burrowing by small mammals (Hakonson et al. 1982, Laundre 1993) and ants (Blom et al. 1994) can increase water infiltration and percolation by decreasing the bulk density of soil or creating channels for preferential flow. Burrowing animals also may transport contaminants to the surface (Arthur and Markham 1983, Arthur et al. 1986, 1987; see Suter et al. 1993 for a summary of animal intrusion effects). A biobarrier consisting of a layer of cobble and gravel placed within a protective soil cap should restrict animal burrowing, but the barrier may also constrain growth of plant roots. If roots were restricted to the soil above an intrusion barrier, the effective water-storage reservoir of the soil cap would also be limited to the soil above the barrier. In this case, depth of emplacement of the intrusion barrier within a soil profile would be critical. On the other hand, if plant roots penetrated through the intrusion barrier and extracted water from the soil below it, depth of emplacement of the barrier might have little effect on the size of the water-storage reservoir. In addition, gravel/cobble biobarriers create capillary breaks within the soil profile, and depth of placement of those breaks may also affect the performance of a cap. Given these considerations, the first two objectives of the PCBE were:

Objective 1. To compare the hydrologic performance of caps having biobarriers with that of soil-only caps and that of caps based on EPA recommendations for RCRA caps.

Objective 2. To examine the effects of intrusion barriers placed at different depths on water percolation, water-storage capacity, plant rooting depths, and water extraction patterns.

Models of climatic change predict an increase in precipitation over the next 50 to 100 years for sagebrush steppe ecosystems, such as at the INEEL. Some models predict an increase in precipitation during summer, whereas others predict an increase during winter or early spring (Giorgi et al. 1994, Ferguson 1997). It is likely that a change in precipitation patterns, especially increased summer precipitation would change the composition of vegetation and, in turn, could affect the performance of an ET cap. To investigate the implications of such climate changes, we included two supplemental-irrigation treatments in addition to an ambient-precipitation control in the PCBE; one irrigation treatment augments summer precipitation by 200 mm and the other augments winter/spring precipitation by the same amount. These supplemental treatments are approximately equal to the average precipitation received at the INEEL (220 mm); thus, they roughly doubled the average ambient precipitation. This amount is much more than that predicted by climate change models (Giorgi et al. 1994, Ferguson 1997). Our intent was to augment precipitation sufficiently to cause measurable vegetation and hydrologic responses to address the following objective:

Objective 3. To evaluate the performance of caps receiving higher precipitation than expected under either the present climate or that anticipated in the foreseeable future.

Prior to the initiation of this project in 1993, most waste burial and other disturbed sites at the INEEL were planted with crested wheatgrass, species native to Europe and Asia. At the INEEL, crested wheatgrasses tend to form persistent monocultures (Marlette and Anderson 1986), which makes them attractive candidates for vegetation on landfill caps. Our earlier project (Anderson et al. 1987, 1993) demonstrated that crested wheatgrass grew well on simulated waste caps and would use all of the plant-available moisture in a 2.2-m soil cap, even during exceptionally wet years. Assuming that these species likely would be included in the plants established on ET caps at the INEEL, crested wheatgrass was planted in pure stands as one of the vegetation types.

Ecological theory predicts that a diverse plant community consisting of multiple life forms will be more stable and will more completely use resources such as soil moisture in comparison with a simple community (e.g., McNaughton 1977, 1993, Tilman et al. 1997b). Numerous recent studies support those predictions (e.g., Tilman and El Haddi 1992, Tilman and Downing 1994, Tilman et al. 1996, 1997a, Hector et al. 1999, Anderson and Inouye 2001). Analyses of long-term vegetation data from permanent plots at the INEEL indicate that areas having more species tend to maintain higher cover and fluctuate less in cover relative to the mean value, compared with areas supporting fewer species (Anderson and Inouye 2001). We postulated that such diversity would help ensure the functional integrity of a protective cap under threats from insect or pathogen outbreaks or disturbances such as fire. Furthermore, regardless of the kind of plants that are initially planted on a waste cap, common native species such as sagebrush (*Artemisia tridentata*) and rabbitbrush (*Chrysothamnus* sp.) likely will occupy the site eventually (Link et al. 1994). Hence, it is important to understand how a mixture of different species and different growth forms will perform. A mixture of 12 native species, including five shrubs, five perennial grasses, and two forbs was included as the second vegetation type in the experiment. Therefore, two vegetation types were included in the PCBE to address the following objective:

Objective 4. To compare the performance of a community of native species on ET caps to that of caps vegetated with a monoculture of crested wheatgrass.

Formatted

The Protective Cap/Biobarrier Experiment was a field-scale experiment, consisting of three replicates of four cap configurations, two vegetation types, and three irrigation treatments. To address Objectives 1 and 2, performance of two cap configurations with biobarriers at different depths was compared with that of caps consisting of soil only and with a cap design based on RCRA recommendations. Objective 3 was addressed by comparing cap performance under ambient precipitation with that under augmented fall/spring and augmented summer precipitation. Performance of crested-wheatgrass monocultures was compared to that of a diverse community of native species to meet PCBE Objective 4.

5.2 Design and Methods

Cap Configuration and Construction

The PCBE consisted of three replicates of four cap configurations, two vegetation types, and three irrigation treatments. Each replicate consisted of four, 16- x 24-m main plots representing the four cap configurations. Main plots were divided into six, 8- x 8-m subplots, representing the two vegetation types and three irrigation treatments. Thus, within one replicate, each 8- x 8-m subplot represented one of the 24 cap type, vegetation type, and irrigation treatment combinations.

The four cap configurations are illustrated in Figure 6. The **soil-only** cap consists of 2.0 m of homogeneous soil. The **shallow-biobarrier** cap includes a biobarrier consisting of 0.3 m of river cobble (0.1 – 0.2 m in diameter) sandwiched between 0.1-m layers of crushed gravel (5 – 15 mm in diameter). This biobarrier was placed at a depth of 0.5 m within 2-m of soil, for a total cap depth of 2.5 m. The **deep-biobarrier** cap includes a 0.5 m biobarrier at a depth of 1 m within 2 m of soil. The biobarrier is identical to that of the shallow-biobarrier cap. The **RCRA** cap was based on guidelines from the US Environmental Protection Agency for implementation of the waste disposal regulations in the Resource Conservation and Recovery Act (RCRA) of 1976 (USEPA 1989). For semi-arid areas, the EPA recommended a minimum of 0.6 m of soil overlying a sloped flexible membrane liner (FML) and a 0.6-m deep compacted clay layer. Our previous research showed that 0.6 m of soil would be inadequate to store ambient precipitation received during many years, so we increased the depth of soil overlying the FML to 1 m (Figure 6). The compacted clay layer and FML were emplaced with a 3% slope so that water would be transported off the cap and collected once the soil above the FML was saturated.

All experimental plots were constructed from the same fill soil, a silty clay loam soil obtained from Spreading Area B at the INEEL. Average soil consistency is 19% sand, 48% silt, and 33% clay. Each main plot was constructed in lifts of about 0.2 m and then compacted. Soils were compacted to a bulk density of approximately 1.29 g/cm^3. The clay soil underlying the FML on the RCRA plots was compacted to achieve a hydraulic conductivity of 1 x 10^{-7} cm/sec or less.

Vegetation Types

Crested-wheatgrass monocultures were established by drill-seeding the cultivar Nordan at 6.7 kg/ha in rows 0.36 m apart at the recommended seeding depth of about 20 mm. Planting was done in early March 1994. Gravel mulch creating about 75% ground cover was placed on the crested-wheatgrass subplots after they were seeded.

The native community analog established on the PCBE plots consisted of 12 species, including five shrubs, five perennial grasses, and two forbs (Table 1). Common shrubs and grasses were transplanted from local sagebrush communities onto the PCBE in mid-November, 1993. Small plants were hand excavated with their root-soil masses intact and transported in plastic pots. Transplants were placed into a 0.75- x 0.75-m grid pattern on each subplot. Growth forms were alternated within rows (shrub, grass, shrub, etc.) so that each subplot was as homogeneous as possible. The planting arrangement ensured that members of the same species were never closer than about 2.3 m and adjacent individuals of the same growth form were about 1 m apart. These spacing patterns were based on plant densities in natural sagebrush communities in the area.

Two species of forbs were drill seeded into rows midway between the transplant grid rows in early March 1994. Commercially obtained seeds of 'Appar' blue flax (*Linum perenne*) were drilled in rows parallel to the 16-m sides of main plots at a density of 1.66 kg/ha (about 100 seeds/m^2). Northern sweetvetch (*Hedysarum boreale*) was drilled in rows parallel to the 24-m sides of main plots at a density of 7.7 kg/ha (about 50 seeds/m^2). Following planting, all subplots received gravel mulch covering about 75% of the surface. Shrub and perennial grass transplants that did not survive were replaced during 1994.

Soil Moisture Measurements

A neutron hydroprobe access tube (20 gauge aluminum, 50.8 mm ID) was installed in the center of each subplot. Access tubes extend from the bottom of each subplot to 0.2 m above the soil surface. Soil water content (% by volume, θ) was estimated by neutron scattering (Schmugge et al. 1980) with Model 503 Hydroprobes (Campbell Pacific Nuclear Corp., Martinez, California). The neutron hydroprobes were calibrated to the soils of the PCBE. Measurements were made at depths of 0.2, 0.4, 0.6, 0.8, 0.9, 1.1, and 1.2 m, and then at 0.2 m, intervals to the bottom of the cap. Soil moisture was generally measured at biweekly intervals through the growing season, beginning in late March and continuing through early October. Intervals were sometimes longer toward the end of the growing season when most soil moisture was depleted.

The total amount of water in the soil profile (S, in mm), was estimated for individual access tubes for each sampling date as:

$$S = 2\,(1.5\ \theta_{0.2} + \Sigma\theta_i + 0.5\theta_{0.9} + 0.5\theta_{1.1} + \Sigma\theta_j) \qquad (3)$$

where, $\theta_{0.2}$ is the volumetric water content (%) at a soil depth of 0.2 m and θ_i is the volumetric water content at depths of 0.4, 0.6, and 0.8 m, $\theta_{0.9}$ and $\theta_{1.1}$ are volumetric water contents at depths of 0.9 and 1.1 m, respectively, and θ_j is volumetric water content at 1.2 m and subsequent 0.2-m depths to the bottom of the profile. Because water content at the soil surface is so variable and could not be estimated accurately with a neutron probe, we assumed uniform water content from the surface to 0.2 m.

Volumetric water content measured in mid to late September was used to estimate the lower limit of extraction of soil moisture by the vegetation. Because end-of-season moisture content indicates the extent to which the water-storage reservoir has been emptied, we compared end-of-season water content among cap types, vegetation types, and irrigation treatments.

Irrigation Treatments

Supplemental water was applied to the PCBE plots with a drip irrigation system, which allows for precise control and metering of the water applied. Emitters were placed at 0.5-m intervals; each subplot contains 264 emitters, which deliver water to a subplot at about 17.6 L/min. Solenoid valves controlled irrigation of each irrigation x vegetation treatment within each replicate. Standard water meters (Master Meter, Longview, Texas) were used to measure the amount of water applied. The irrigation system was installed in 1994 and became operational in August of that year.

The three irrigation/precipitation treatments were 1) ambient precipitation 2) summer irrigation, and 3) fall/spring irrigation. Ambient subplots received no supplemental irrigation after 1994, once plants were established. The summer-irrigation treatment consisted of four applications of 50 mm of water at biweekly intervals beginning in mid June. This treatment simulated an increase in summer rainfall events that would tend to only wet surface layers of soil. The fall/spring-irrigation treatment consisted of the application of 200 mm of water within a short time period (one to two weeks) in either April or October. This irrigation treatment simulated an increase in fall, winter, or early spring precipitation, which would result in deep recharge of soil water.

Assessing Cap Breakthrough

Breakthrough was defined as drainage of water from the bottom of the soil cap. We had planned to assess breakthrough by monitoring water draining from three drain pans underlying each subplot. However, we found that the drain pans were not a reliable technique for assessing drainage (see Anderson and Forman 2001 for details). Therefore, we used neutron-probe data to estimate when a subplot would drain. By examining soil-moisture profiles, it is easy to determine when the wetting front reaches the bottom of a cap. Because most plots were underlain by natural gravel that would create a capillary break, we assumed that no drainage would occur so long as the moisture content at the bottom is below field capacity for that soil. Our estimate of field capacity for this soil is 28% moisture by volume (see Section 2.3). Therefore, we used 28% volumetric water content at the bottom of a profile to indicate potential breakthrough of the cap on a subplot.

Irrigation to Breakthrough Trials

In April and May of 1999, we irrigated all fall/spring subplots until drainage was observed from the collection pans or the water content at the bottom of the subplot was estimated to be at or above field capacity (≥ 28%). For these trials, irrigation was applied as rapidly as possible without causing pooling of water on the subplots. We compared 1) the amount of water added before drainage occurred and, 2) the amount of water in the total soil profile when drainage occurred among cap and vegetation types.

6.0 Results from the Protective Cap/Biobarrier Experiment

6.1 Precipitation during Study Period

From1994 – 2000, water-year (October – September) precipitation ranged from a low of 129 mm in 1999-2000 to a high of 318 mm in 1994-1995 (Figure 7). The period included near extremes for the period of record at INEEL (1950 – 2000), representing well the recorded climate variability of the area (Figure 4). The 1994 – 1995 water-year stands out as exceptionally wet, the highest on record for the INEEL. Precipitation in May was above normal, and record amounts of rain fell in June (118 mm; Figure 7). In contrast, the 1993 – 1994 and 1999 – 2000 water years were very dry (Figure 7), with very little precipitation in June, normally the wettest month (Figure 3).

6.2 Vegetation Establishment and Development

Low precipitation in 1994 created unfavorable conditions for establishment of vegetation on the experimental plots. Mean soil water content at the beginning of the growing season averaged only 17%. No precipitation fell during March, precipitation in May was well below normal, and virtually no rain was received during June and July (Figure 7). As a consequence of the dry season, mean survival of shrubs that had been transplanted onto the plots in late fall, 1993, was only 55.4%, and that of grasses was only 65.9%. We transplanted 1,054 shrubs and

783 grasses in 1994 to replace those that had died. Because the drip irrigation system was not yet in place, we applied supplemental water periodically during the summer of 1994 with an old sprinkler system. Precise control of amounts applied was impossible, so the amounts received by individual plots was quite variable but sufficient to prevent further mortality of transplants and ensure germination and establishment of the seeded species. Wetting fronts from this supplemental irrigation averaged 1 m and never extended to the bottom of any of the soil-only and biobarrier plots. It did reach the FML in the RCRA caps on several subplots. By the end of the 1994 growing season, vegetation was established on all plots.

Plant cover on both native-vegetation and crested-wheatgrass subplots developed rapidly after initial establishment. Plant cover is the percentage of the total area, viewed from a vertical projection, covered by plant species. At the beginning of the 1995 growing season, plant cover on the native-vegetation subplots ranged from 10% to 13%; it doubled by the end of the growing season. Cover on the crested-wheatgrass subplots was measured at the end of the 1995 growing season; it ranged from 36% to 48%, roughly double that on the native-vegetation subplots. After the initial cover estimates in 1995, cover of crested wheatgrass was not measured again on all replicates until 2000. In the interim, cover on crested-wheatgrass subplots had been reduced by about 50%. This reduction was caused by high plant productivity in 1995 and 1996, which resulted in a lot of standing dead material and thick layers of litter that inhibited plant growth in subsequent years.

Although cover on the native-vegetation subplots at the end of the 1995 growing season was about half that of the crested-wheatgrass subplots, it continued to develop rapidly, peaking in 1997 (Figure 8). Under ambient precipitation, total cover of native-vegetation subplots on the four cap types was remarkably similar throughout the study period (Figure 8). Cover on the ambient subplots in 1997 averaged 53% and subsequently decreased to 29% in 2000. Such fluctuations in cover are to be expected in response to year-to-year and longer-term variation in precipitation (Anderson and Inouye 2001). The cover values for 1998 – 2000 under ambient precipitation (Figure 8) are similar to those of natural sagebrush steppe at the INEEL (Anderson and Inouye 2001).

Under summer irrigation, peak cover values in 1997 ranged from 55% to 79%, with the soil-only and deep-biobarrier subplots having somewhat higher cover than the shallow-biobarrier and RCRA subplots (Figure 8). However, by 2000, cover on all cap types was similar, averaging

39%. Thus, at the end of the study period, summer-irrigated subplots maintained about 10% more vegetative cover than did subplots receiving ambient precipitation.

Peak cover on fall/spring-irrigated subplots ranged from 69% on RCRA subplots to 106% on deep-biobarrier subplots (Figure 8). Cover greater than one hundred percent can be achieved when canopies of individuals overlap (i.e. a small shrub grows under the canopy of a larger shrub). Subsequently, cover decreased considerably on all cap types. By 2000, cover was similar on the soil-only and biobarrier subplots, averaging 51%, 12% higher than that on subplots receiving summer irrigation. Cover was lower ($p = 0.069$) on the RCRA subplot (36%), reflecting a decrease in plant-available water compared to the soil-only and biobarrier caps because of the difference in soil depth available to store water. Mean cover on the RCRA subplots in 2000 was identical under summer and fall/spring irrigation.

6.3 Performance of the Caps under Ambient Precipitation

Under ambient precipitation, the soil-only cap and the two biobarrier caps generally performed similarly. In the vast majority of cases, the wetting front never reached the bottom of a cap, and all moisture received was returned to the atmosphere via evapotranspiration. A representative example of soil-moisture dynamics on a soil-only subplot with native vegetation is shown in Figure 9. Over the six years shown, the depth to which wetting front reached varied from less than 0.2 m in 2000 to over 1.2 m in 1995 and 1999, reflecting the variability in precipitation received (Figure 9). In all years, all of the water available to plants on native-vegetation subplots was extracted by the end of the growing season.

Under ambient precipitation, water percolated below the biobarrier on at least one shallow-biobarrier subplot in each year of study except the very dry 2000. Nevertheless, there was no case where the wetting front reach the bottom of the soil cap, and in all cases water that had moved into the soil below a biobarrier was extracted by plants. Figure 10 depicts soil-moisture dynamics on one shallow-biobarrier subplot with native vegetation. In the spring of 1999, water percolated below the biobarrier of that subplot in the spring and was subsequently extracted (Figure 10).

Typically, there was no change in soil water content below deep biobarriers under ambient precipitation (Figure 11). In 1995, an exceptionally wet year, all of the soil above the biobarrier of the subplot shown in Figure 11 was at field capacity, but no water moved below the biobarrier. This demonstrates the effectiveness of the capillary break formed by the biobarrier in restricting the downward flux of water. These results also demonstrate that 1 m of soil above a biobarrier generally will suffice to store precipitation received under current climatic conditions at the INEEL.

RCRA caps were inadequate to store precipitation received in 1995, when drainage from the FML liner was observed from one of the three replicates. With the exception of 2000, soil moisture early in the growing season approached field capacity to a depth of 0.8 m on some RCRA caps each year under ambient precipitation (Figure 12), indicating little reserve storage capacity before drainage from the liner would occur.

Under ambient precipitation from 1995 through 2000, there was no significant difference in growing-season ET among cap types or between vegetation types. Mean ET ranged from 113 mm in 2000 to 338 mm in 1995. The value for 1995 is anomalously high since the total water-year precipitation for 1994-95 was 318 mm. This occurred because some of the PCBE plots had considerable residual moisture in the soil at the beginning of the growing season as a result of irrigation late in 1994; this was extracted and transpired. The other contributing factor was that much of the record 1995 precipitation fell during the growing season (Figure 6) and was simply returned to the atmosphere via evapotranspiration.

6.4 Performance of the Caps under Summer Irrigation

Performance of the caps under summer irrigation was generally similar to that under ambient precipitation. There was little difference in depths reached by the spring-infiltration wetting front between subplots receiving ambient precipitation and those receiving summer irrigation. The similarity between ambient-precipitation and summer-irrigated subplots reflects the fact that most water received during the growing season is returned immediately to the atmosphere, so long as plants remain active. Overall, the results indicate that a modest increase in summer precipitation, as predicted by some climate change models, would have little impact on the performance on an ET cap.

For the five years during which summer irrigation was applied, there was no significant difference among either cap type or vegetation type in total growing season ET. ET ranged from 329 mm in 2000 to 407 mm in 1997. These values are, of course, much higher than those under ambient precipitation (from 1.7 to 2.9 times higher), reflecting the addition of 200 mm of water during the growing season.

We also were unable to detect any statistical difference in end-of-season soil moisture between ambient and summer-irrigation treatments on native-vegetation subplots. Thus, addition of 200 mm of water during the summer did not decrease the size of the storage reservoir for the next water year for any of the cap types. However, end-of-season soil moisture was higher under summer irrigation than under ambient precipitation on crested-wheatgrass subplots in four of five years the irrigation treatment was applied ($P < 0.05$). Consequently, the storage reservoir of summer-irrigated/crested-wheatgrass subplots was lower than that of crested-wheatgrass subplots receiving ambient precipitation. This result was a consequence of reduced plant cover on the crested-wheatgrass subplots (see Section 6.7).

6.5 Performance of the Caps under Fall/Spring Irrigation

Subplots in the fall/spring-irrigation treatment received large applications of water on two occasions. The first occurred in August of 1995 when 550 mm of water were applied; the second occurred in 1999 during the "irrigation-to-breakthrough-trials." In both cases, the objective was to assess cap performance under a "worst case" scenario when the soil storage reservoir was essentially full. In 1996, 1997, 1998, and 2000, these subplots received 200 mm of water in the fall or (in 2000) early spring.

Following irrigation with 550 mm of water in 1995, the wetting front reached the bottom of all but three of the 18 subplots representing the soil-only, shallow-biobarrier, and deep-biobarrier caps (e.g., Figure 13). Moisture profiles indicated that soil moisture at the bottom of five subplots was above field capacity and water likely would have drained from them. Thus, despite adding 550 mm of water late in an exceptionally wet growing season, breakthrough occurred on only five of the 18 subplots. Sufficient plant cover was present and active to remove a substantial amount of the water added by late September, so by the end of the growing season moisture content in the upper 1 m of soil was well below field capacity.

Soil-moisture dynamics for representative subplots in 1995 and 1996 illustrate the ability of ET caps to recover storage capacity (Figure 13). Spring snowmelt and infiltration in 1995 (blue line) was followed by an abnormally wet spring (174 mm in April, May and June), which resulted in a continued increase in soil moisture throughout these months (cyan, green, and yellow lines). Irrigation in August of 1995 pushed water deep into the profile of all three subplots (red line). Subsequent redistribution of water during late summer and fall (gray line) resulted in the wetting front reaching the bottom of all three subplots. Water content at the bottom of the soil-only and deep-biobarrier subplots was above field capacity by early September. In 1996 (Figure 13, lower frames), plants extracted water first from shallow depths in the soil and then, as the season progressed, from increasing depths. By the end of the 1996 growing season, virtually all plant available water was removed from the entire soil profile, resetting the storage capacity of the caps.

Figure 13 (lower frames) demonstrates clearly that the change in water content below both biobarriers was due to extraction by plants and not to drainage, because water content below the biobarriers did not change until plants had extracted most of the plant-available water above the biobarriers. Had the change in water content below the biobarrier been due to drainage, change would have been seen at earlier dates. For example, in the case of the deep-biobarrier subplot, there is virtually no change in water content below the biobarrier between October of 1995 (black line) and 20 June 1996 (yellow line). By the latter date, most of the water available to plants had been extracted from the top 1 m of the profile; at the next sample date (16 July, orange line), there is evidence of removal of water below the biobarrier, and by the end of August, mean water content below the biobarrier was 20%. The seasonal pattern of extraction for the shallow-biobarrier subplot is similar.

Application of 200 mm of water in the falls of 1996, 1997, and 1998 caused percolation to depths of 0.8 m to 1 m in soil-only plots. Water from snowmelt and early spring rainfall pushed the wetting front to greater depths in those years, sometimes increasing water content in the soil at the bottom of a subplot. Regardless of the amount of water applied, plants generally used most of the available water by the end of the growing season, essentially emptying the storage reservoir.

Fall irrigation in 1996, 1997, and 1998 caused percolation through the shallow barrier on all subplots, but the wetting front reached the bottom of a subplot in only two cases during those three years. As with the soil-only subplots, snowmelt and early spring rainfall sometimes pushed the wetting front to greater depths, but in no case did moisture content at the bottom of a subplot reach field capacity. Water was extracted from below the biobarrier on all subplots. Native vegetation typically reduced water content throughout the profile to less than 20%, but end-of-season moisture below the biobarriers on the crested-wheatgrass subplots was about 25% (see Section 6.7).

Deep-biobarrier subplots generally were capable of storing the 200 mm of water applied in the fall above the biobarrier; although in a few cases there was a slight increase in soil moisture below the biobarrier (e.g., Figure 14). In some cases, snowmelt and early spring rainfall increased water content below the biobarrier, but, whenever that occurred, the water was extracted by the end of the growing season. As seen with the shallow-biobarrier plots, end-of-season water below the biobarrier was typically below 20% on the native-vegetation subplots but near 25% on the crested-wheatgrass subplots (Figure 14).

Following fall irrigation, the soil on RCRA caps typically was near saturation, so there was little reserve storage volume available to store winter and spring precipitation. The wetting front reached the FML liner on many subplots each year.

On native-vegetation subplots, we found no significant difference in mean end-of-season soil moisture, measured prior to fall irrigation, among cap types in any year under the fall/spring-irrigation treatments. Crested-wheatgrass subplots receiving fall/spring irrigation had significantly higher end-of-season soil moisture means than did subplots receiving ambient precipitation during every year of the study. No such differences in end-of-season soil moisture on native-vegetation subplots were statistically detectable. Thus, 200 mm of supplemental irrigation did not result in any accumulation of water in the soil of native-vegetation subplots.

6.6 Irrigation to Breakthrough and Post-Breakthrough Soil-Moisture Dynamics

Irrigation-to-breakthrough trials were conducted in April and May of 1999 on all fall/spring subplots. There was no significant difference among the soil-only and biobarrier caps or between vegetation types in the amount of water required to cause breakthrough. Means ranged from 607 to 727 mm; these estimates include water in the profile when the trials began plus the amount of irrigation and precipitation received. These values are surprisingly high, but because plants were transpiring and the soil surface was continually wet during the 32 days over which the trials were conducted, ET would have removed a substantial amount of that water. A conservative ET estimate of 5 mm/day over the period would account for 160 mm of water. Furthermore, some of the water in the profile at breakthrough ultimately would have drained from the plots.

Estimates of the amount of water in the profile at breakthrough were very similar among the soil-only and biobarrier caps, with an overall mean of 607 mm. These values are consistent with an estimated field capacity of 28%, which would translate to 560 mm of water in a 2-m depth of soil.

To examine recovery of moisture storage capacity after irrigation to breakthrough, we compared end-of-season water content for each cap-type/vegetation combination for 1998 (pre-irrigation to breakthrough), 1999, and 2000. For subplots having native vegetation, average moisture content in the total soil profile was reduced to 16% or less by the end of the 1999 season and were not significantly different from the 1998 values, except for the RCRA subplots which were drier in 1999 than in 1998. Furthermore, mean values were very similar among cap types, indicating that emplacement of a biobarrier did not significantly affect the effective storage capacity of a cap. Analyses of different portions of the soil profile revealed only one case where mean water content was significantly higher ($P < 0.05$) at the end of the 1999 growing season than it had been in 1998: Mean water content in the bottom 1.0 m of soil on the deep-biobarrier subplots was 18.3% in the fall of 1999 vs. 16.9% in 1998. Thus, with native vegetation, most caps had fully recovered their capacity to store water by the end of the same season in which that storage reservoir had been completely full.

On the crested-wheatgrass subplots, the end-of-season water contents for all but the RCRA subplots were considerably higher in all three years than on the native-vegetation subplots and on some subplots 2 years were required to reset the storage reservoir. Further comparisons of the two vegetation types are given in the next section.

6.7 Differences in Performance of the Two Vegetation Types

Plant cover on the plots planted to crested wheatgrass developed very rapidly during 1994 and 1995. By the end of the 1995 growing season, cover on the crested-wheatgrass subplots averaged 43%, double that on the native-vegetation subplots. Beginning during the winter of 1995-96, there was substantial lodging of dead tillers and production of litter on the PCBE crested-wheatgrass subplots, sharply reducing cover. Cover was not measured on those plots again until 2000 at which time it averaged about 50% of that in 1995; it was lower in 2000 than in 1995 for all cap types and under all precipitation/irrigation regimes. In 2000, cover on crested-wheatgrass subplots differed little among cap types or precipitation/irrigation treatments, whereas native-vegetation subplots maintained higher cover under both summer and fall/spring irrigation than under ambient precipitation (Figure 15). Furthermore, native-vegetation subplots had higher cover than crested-wheatgrass subplots under all precipitation/irrigation regimes and on all cap types in 2000 (Figure 15).

The consequences of reduced cover on the crested-wheatgrass subplots were evident as early as 1996, when significantly more water remained in the soil at the end of the growing season on subplots receiving supplemental irrigation (both summer and fall/spring) than on those that received only ambient precipitation (Table 2). Similar, highly significant differences were observed in each subsequent year of the study (Table 2). It should be noted that there was a substantial decrease in perennial grass cover on native-vegetation subplots in 1998, but increases in shrub cover offset those losses, so there was sufficient plant cover on the native-vegetation subplots to use all of the moisture available.

We used Student's *t*-tests to compare end-of-season soil moisture between crested-wheatgrass and native-vegetation subplots for each cap type under the three precipitation/irrigation regimes in 1998, a year in which the total amount of distribution of precipitation were close to long-term averages. There was no significant difference between means under ambient precipitation; in fact, all means were within 1.5% (Table 3). In contrast, under summer irrigation, the RCRA and deep-biobarrier caps planted to crested wheatgrass had significantly higher end-of-season moisture than those planted to native vegetation, and the difference for the shallow-biobarrier subplots was marginally significant (Table 3). Under fall spring irrigation, soil moisture was significantly higher on the crested-wheatgrass shallow- and deep-biobarrier subplots than on the native-vegetation shallow- and deep-biobarrier subplots (Table 3). It is noteworthy that, although differences were not always significant (probably due to small samples sizes), all means for crested-wheatgrass subplots were higher than those for native-vegetation subplots under irrigation treatments (Table 3).

7.0 Summary and Discussion of PCBE Results

7.1 Differential Performance of the Four Cap Configurations under Ambient and Augmented Precipitation (Objectives 1 and 3).

Under ambient precipitation and summer irrigation, all of the cap types performed satisfactorily and the soil-only and biobarrier caps performed quite similarly. Given the similar climatic conditions that have prevailed on the upper Snake River Plain for the last 10,000 years (Davis 1981, Davis et al. 1986, Beiswenger 1991), a landfill cap constructed according to any of the cap configurations included in this study likely would prevent water received as precipitation from reaching interred wastes, so long as the caps supported a healthy community of drought-tolerant perennial plants. This, of course, assumes no run on of water onto the cap and no subsidence that would cause pooling of water following snowmelt. We have demonstrated that even a very large increase in summer precipitation would not adversely impact cap performance. With increased winter precipitation (fall/spring-irrigation treatment), differences in cap performance became more important, but the soil-only and biobarrier caps were still capable of storing and returning to the atmosphere far more moisture than the precipitation expected under current climate change scenarios. Thus, the soil-only and biobarrier caps should preclude water from reaching buried wastes, even with a considerable increase in winter precipitation. Nevertheless, there are important differences that translate to advantages or disadvantages of the various configurations.

Soil-only Cap

This study confirms the conclusions of our first study at the INEEL (Anderson et al. 1991, 1993), that a cap consisting of a 2-m depth of soil would prevent percolation of water into interred wastes. This depth of soil is more than adequate to store the water received as precipitation under present or predicted future climates, so long as healthy perennial vegetation is present to empty that storage reservoir each year. Our results indicate that the soil wetting front would rarely reach the bottom of a 2-m soil cap. Hence, once plants extracted the available water from the entire soil profile, we would expect hydraulic conductivity of the dry soil at the bottom of the cap to remain very low (see Section 7.7). Furthermore, because root activity would be limited to those soil depths having plant-available water, we would not expect roots to grow beyond the depth of the wetting front each year. Thus, root intrusion into buried wastes should not be a problem once the vegetation initially dries the soil cap.

A 2-m cap of soil should provide sufficient depth to accommodate the maximum observed burrowing depths of small mammals (Reynolds and Wakkinen 1987, Reynolds and Laundre 1988, Laundre 1989, Laundre and Reynolds 1993, Pratt 2000) and harvester ants (Blom 1990, Gaglio et al. 1998). Laundre (1993) demonstrated that small mammal burrows increased water percolation into soils at the INEEL by very modest amounts. In addition, small mammals have been abundant on the PCBE since its inception; we have seen no evidence that their activities adversely affected cap performance. Gaglio et al. (1998) found that harvester ant nests increased percolation rates on PCBE soils, but those soils dried faster than undisturbed soils.

They concluded that nests of harvester ants “do not pose an immediate threat to the groundwater under low level nuclear waste buried under a 2-m protective cap.” Given these observations and results, we conclude that native animal threats to the integrity of a 2-m soil cap are minor.

Shallow Biobarrier Cap

Shallow-biobarrier caps generally performed as well as the other cap configurations. We found that roots of numerous species can bridge a 0.5-m thick biobarrier and extract water from the underlying soil, so indeed it is possible to have a portion of the storage reservoir below a biobarrier. However, this design has numerous disadvantages that make it the least favorable alternative to a RCRA cap. The results show that 0.5 m of soil above a biobarrier is insufficient to store the precipitation received in most years, so water will routinely percolate into the soil below, providing a reservoir of deep soil moisture. Placement of the biobarrier at a shallow depth also caused strong selection for gray rabbitbrush (*Ericameria nauseosus*), a native shrub known to rely primarily on deep moisture reserves (Anderson and Forman 2002). Encouraging the growth of this deeply rooting species could result in intrusion of roots into buried wastes if any water was available in the waste zone.

Although we have good evidence that animals will not burrow through a biobarrier having a meter of overlying soil, we do not have definitive evidence that a 0.5-m thick biobarrier is sufficient to preclude burrowing if the overlying soil is considerably thinner (Pratt 2000). Hence, another potential disadvantage of shallow biobarrier placement is that it would not provide sufficient burrowing depth to meet the needs of small mammals or ants, which, in turn, might encourage those species to burrow into and possibly through the biobarrier.

Deep Biobarrier Cap

Under ambient precipitation and summer irrigation, we seldom saw any change in water content below deep biobarriers; water typically did not percolate below the biobarriers during spring recharge, and there was no extraction of water (i.e. plant root activity) from the soil below the biobarriers. The stability of the moisture profiles below deep biobarriers over a growing season (e.g., Figure 14) reflects the very low hydraulic conductivity of these relatively dry soils. Under augmented fall/spring precipitation, water occasionally percolated below the biobarriers, but in many cases heavy irrigation in the fall coupled with ambient precipitation during the winter and spring did not result in any increase in water content below biobarriers (e.g., Figure

14). Given these results, a deep-biobarrier configuration is one of the caps recommended for isolating hazardous wastes at the INEEL (see below).

RCRA Cap

The minimum soil depth recommended for a RCRA cap is 0.6 m (USEPA 1989). We found that 1 m of soil overlying an impervious FML was inadequate to store the ambient precipitation received during 1995, an exceptionally wet year. Under fall/spring irrigation, there typically was little if any reserve storage capacity at the beginning of the growing season and drainage off the FML was sometimes observed. Thus, the main disadvantage of the RCRA-recommended cap is that provision would have to be made for disposing of water that would occasionally drain off the cap over the FML. This would substantially increase construction complexity and cost. Furthermore, it is possible that water so disposed of could run back under the cap, depending on the configuration of underlying substrata. We argue that, at sites such as the INEEL where potential evapotranspiration is so much higher than precipitation, it makes much more sense to design caps so that no provision for drainage is necessary.

Another concern with the RCRA cap is life of the FML. Research elsewhere has shown that, should an FML become damaged, we would expect plant roots to quickly extract water from the underlying compacted clay layer, resulting in it drying, cracking, and subsequently allowing deep water percolation (Daniel and Gross 1995). Thus, the long-term integrity of a RCRA cap in arid and semiarid environments is questionable (Suter et al. 1993).

A RCRA cap may also be problematic because it would be more expensive and difficult to construct than a soil-only or biobarrier cap. Soil with sufficient clay content to form the compacted clay layer would likely have to be imported at considerable cost, and it is difficult and time consuming to work with. In addition, great care must be taken to seal overlapping sheets of FML and to prevent damage to the FML as overlying soil is emplaced.

The only potential advantage that we see for a RCRA cap is that the FML might prevent drainage into wastes in the event of cap subsidence that caused local pooling of water, assuming that the FML remained intact. Given concerns about the long-term integrity of a FML and the increased complexity and cost compared to the alternative configuration that we evaluated, we cannot recommend the RCRA cap for the INEEL or similar semiarid environments.

7.2 Effects of Biological Intrusion Barriers on Soil Moisture Storage and Extraction (Objective 2).

The gravel/cobble biobarriers in this study interrupted the soil water-storage reservoir at depths of either 0.5 m or 1 m. The results demonstrate that it is feasible to have a portion of the storage reservoir below a biobarrier. Roots of several species bridged biobarriers (Anderson and Forman 2002), and the results show definitively that plants will extract water from all depths of soil below biobarriers (Figure 13). Indeed, on shallow-biobarrier plots, roots proliferated in and extracted water from a 1.5-m depth of soil below those biobarriers.

Aside from precluding burrowing animals, one of the greatest values of the biobarrier in the biobarrier cap designs was its function in creating a capillary break. Because of the capillary break between the fine textured soil above the biobarrier and the gravel at the top of the biobarrier, water content of the soil above the biobarrier must approach saturation before water will percolate through it (Sackschewsky et al. 1995, Hillel 1998, Porro 2001). This effect maximizes the amount of water stored in the overlying soil, as clearly shown in Figure 14 (see Porro 2001 for complementary data from INEEL). Consequently, 1 m of soil was often sufficient to store fall/spring irrigation plus ambient precipitation.

Several trends in soil-moisture dynamics emerge from our analyses of the effects of biobarriers. First, under fall/spring irrigation, end-of-season soil moisture was typically higher below shallow biobarriers than at similar depths in soil-only caps. These results indicate that plants extracted water more effectively from a continuous soil profile than from one interrupted by a biobarrier. Second, end-of-season soil moisture in soil overlying a gravel/cobble biobarrier tended to be lower than that of comparable depths in a continuous soil profile. This trend was especially apparent on shallow-biobarrier caps. This difference probably reflects the combined effects of more thorough extraction of water by plants in the soil above a biobarrier coupled with evaporation from a profile where depth is limited by a capillary break (see Porro 2001). Finally, caps planted to native vegetation tend to have lower end-of-season soil moisture than caps planted to crested wheatgrass both above and below biobarriers, particularly in caps receiving augmented precipitation. Differences in end-of-season volumetric soil moisture of even 3-5% in a cap with an effective water-storage capacity of 13% by volume affects the storage capacity of a cap, and could make the difference between a cap functioning effectively or failing during a series of wet years. A 3-5% increase in end-of-season soil moisture can reduce the effective storage capacity of a cap constructed with this soil to 8-10% by volume in subsequent years. Reasons for differences between the vegetation types are discussed in the next section.

In summary, we found no advantage of placing a biobarrier at a shallow depth in an ET cap. However, placing a gravel/cobble biobarrier at the bottom of an ET cap will take advantage of the capillary break at the soil/gravel interface and maximize the storage capacity of the overlying soil.

7.3 Differential Performance of Vegetation Types (Objective 4).

For reasons given in the introduction, we predicted that an analogue to a natural sagebrush-perennial grass community would perform better and require less maintenance than a perennial grass monoculture. The results support this prediction. Using a combination of transplanting and seeding, we readily established diverse communities on the experimental plots. Shrubs, perennial grasses, and perennial forbs all grew vigorously. All species planted became established, although over time some species became locally extinct on some subplots. Twelve species were originally planted; by 2000, some 27 species were recorded on the native-vegetation subplots. We expect such artificial communities to be dynamic, to vary in total plant cover and species composition through time just as natural sagebrush communities do (Anderson and Inouye 2001).

As expected, crested wheatgrass established quickly and grew vigorously on the subplots where it was planted, just as it had in our earlier study (Anderson et al. 1987). However, after supplemental irrigation to facilitate establishment and a very wet growing season in 1995, the stands of crested wheatgrass were so dense that they became self inhibiting. Live cover on those plots subsequently decreased by about 50%, and, on plots receiving supplemental irrigation, not all of the plant available water in the caps was withdrawn each year. The result was less capacity to store moisture received prior to the next growing season. Higher end-of-season water content on crested-wheatgrass subplots was likely attributable to both lower cover and the absence of shrubs. Shrubs such as sagebrush and the rabbitbrushes remain active late in the growing season, continuing to extract soil moisture after many grasses and forbs are senescent. As a consequence, native vegetation typically used all of the water available in the soil cap each year, maintaining a constant size of reservoir available to store precipitation.

7.4 Recommendations for Waste Cap Configurations at the INEEL

Based on the results of the PCBE and the considerations discussed above, we recommend two cap configurations: a soil-only cap consisting of a 2-m depth of homogeneous soil or a cap consisting of a 1.2-m depth of homogeneous soil overlying a 0.5-m thick gravel/cobble intrusion barrier. Caps constructed according to either of these configurations should preclude virtually any precipitation water from reaching interred wastes.

A major advantage of a soil-only cap is simplicity of construction. A disadvantage is the relatively large amount of soil required. Construction cost will depend largely on availability of soil and the distance it must be transported. If fill soil is limited and if gravel and cobble (or similar materials, see Reynolds 1990) are readily available, then a cap incorporating a biobarrier and requiring less soil might be a better choice.

Although 1 m of soil above a biobarrier was generally adequate to store precipitation received, during 1995, the wettest year on record at the INEEL, water percolated below the biobarrier on two of 18 deep-biobarrier subplots. Therefore, we recommend a minimum of 1.2 m of soil overlying a biobarrier. A cap consisting of 1.2 m of soil overlying the capillary break of the biobarrier should be more than adequate to store precipitation received during exceptionally wet years. Furthermore, this configuration should prevent intrusion by burrowing animals, and it should restrict root growth so long as the underlying materials are relatively dry.

Cap Construction and Surface Topography

The PCBE results demonstrate that an ET cap configured according to the recommendations above should prevent water from reaching buried wastes. Constructing the cap level and on grade with surrounding terrain eliminates any provision for drainage off cap layers, eliminates side slope problems, and reduces the potential for wind or water erosion. Thus, for a new burial site, we recommend this overall configuration. Each component of a cap should be horizontal. Soil should be emplaced in small, horizontal lifts (0.1 to 0.2 m) to avoid creating pitched layers that might provide pathways for preferential flow. Soils should be uniformly compacted to avoid subsequent subsidence that could cause pooling of water on the surface. The cap should be configured to minimize the potential for water to drain onto it from surrounding terrain.

For ET caps constructed to cover existing landfills or contaminated soil, it may be necessary to construct the entire cap above grade. In such a case, it may be desirable to configure the cap with a slight slope on the surface, which could prevent pooling of water on the surface following snowmelt or heavy precipitation. If the surface is sloped, it should be a very shallow slope (e.g., 2%) so that runoff from the cap is minimal (i.e., except for unusual circumstances, all of the precipitation received infiltrates the soil). This will ensure that sufficient moisture will be stored to maintain good vegetative cover and minimize any erosion problems associated with runoff.

For any cap constructed over an existing landfill or contaminated soil, we recommend placing a biobarrier on top of the existing cover or soil. This will help ensure that no moisture moves into the contaminated materials. The new cap should be constructed late in the growing season when the soil of the existing landfill or contaminated area is dry. This will reduce the likelihood of roots growing from the new cap into the contaminated zone.

7.5 Recommendations for Waste Cap Vegetation at the INEEL

Despite theoretical models that may indicate to the contrary (e.g., UNSAT-H, Fayer and Jones 1990), empirical evidence from early capping studies (e.g. Anderson et al. 1987, 1993) and the PCBE demonstrate without question that the bulk of water lost from an ET cap during the growing season is extracted and transpired by plants. For a cap to function effectively and consistently, a healthy stand of perennial, drought-adapted plants is essential. The objective is to establish a plant community that will be self-maintaining.

Our first ET-cap project showed that dominant native and introduced species at the INEEL differed little in their seasonal patterns of water use or in the extent to which they could dry a soil (Anderson et al. 1987). Thus, other ecological characteristics, such as persistence in a stand, ease of establishment, tolerance to pests, ability to resprout or re-colonize following disturbances such as fire, and potential for self-inhibition due to accumulations of standing dead materials and litter, are probably more important considerations in choosing species for cap vegetation. Species recommended for ET caps at the INEEL are shown in Table 4. These species all occur naturally at the INEEL, although commercially available cultivars have been developed from genetic stock derived elsewhere.

Because it is crucial to get vegetation established on ET caps as quickly as possible, it is best not to rely entirely on establishment from seed. The success of seeding varies greatly from year to year, depending on amounts and timing of precipitation. Therefore, we recommend transplanting shrubs and some of the perennial grasses. Although we have found that “wildings” transplanted from local communities survive well (Shumar and Anderson 1987), this technique is labor intensive and, because of impacts on the local vegetation, is only feasible for small revegetation projects. An alternative is to contract with a private firm to collect seed from desired species at the INEEL, grow seedlings in plastic tubes in a greenhouse, and then plant that “container stock” on the ET cap. Transplanting container stock can be combined with drill seeding of grasses and forbs known to establish well from seed, such as wheatgrasses and several forbs. Current cost for collecting seed, growing container stock, and planting the seedlings is about $1.00 per seedling.

The planting density used in the PCBE resulted in excellent vegetative cover. Therefore, we recommend that seedlings be planted into a grid spacing of approximately 0.75 m and in a pattern so that conspecifics are not adjacent to one another. One approach would be plant seeded species first with a conventional agricultural drill in which every other, or perhaps two out of three, drop tube(s) are blocked to increase the spacing between rows. Then, container stock could be planted at 0.75 m intervals between drill rows. Seeding and transplanting can be done either in fall (late October or early November) or early spring (April). Gravel mulch applied to the surface of the cap can retard evaporation, enhance seedling establishment, and reduce erosion (Winkel et al. 1991, Waugh et al. 1994, Sackschewsky et al. 1995). Gravel was applied to the surface of the PCBE plots to achieve about 75% surface cover.

If possible, arrangements should be made to provide some supplemental irrigation during the first growing season. This will help to ensure development of a vigorous stand of plants. Periodic irrigation (e.g., every other week from mid May through June) should suffice, depending on amounts of natural precipitation received. Sufficient water should be applied to drive the wetting front to a depth of 0.25 to 0.3 m each time. There is no need for concern about this irrigation causing cap breakthrough. Once plants are established, they will quickly use the supplemental water.

7.6 Monitoring and Maintenance of an ET Cap

As stated earlier, the objective to revegetation on an ET cap is to establish permanent vegetation and natural ecosystem processes that will function over the long term with minimal maintenance. We are confident that this objective can be met by developing an analogue of a natural plant community on the caps. However, care must be taken to ensure that good vegetative cover develops and that the surface of the cap remains free from depressions that could cause pooling of water and its subsequent drainage into the waste zone. During the first year or so, vegetation development should be closely monitored. If seedings fail, it may be necessary to repeat drilling of seed. Transplants that die should be replaced. Any sizeable depressions should be repaired and re-planted. Over the long-term, periodic monitoring to ensure that the surface remains free of depressions and well vegetated should be all that is necessary.

7.7 Meeting Equivalency Criteria

Demonstrating that the performance of an alternative ET cap design will be equivalent to a USEPA-prescribed cap design may be required for approval of the ET cap by regulatory agencies. Equivalency criteria are usually site specific and are based on an assumed percolation rate for an EPA prescribed cover (Benson et al. 2001). Because we did not measure percolation from the bottom of experimental caps directly, and because we used the water balance equation assuming no drainage to estimate ET, our water balance and ET estimates cannot be used to demonstrate equivalency. In general, water-balance methods are inadequate for demonstrating equivalency, even when ET is estimated from micrometeorological data (Benson et al. 2001). An alternative approach is to estimate percolation rates using Darcy's Law. Benson et al. (2001) indicate that this approach has a precision of one to two orders of magnitude and suggest that it can be used to demonstrate equivalency if the estimated percolation rate is at least two orders of magnitude lower than the equivalency criterion.

In our first ET-cap study, water potential of vegetated Spreading Area B soils was measured (Anderson et al. 1987). At the lower limit of extraction, water potentials in the upper 1 m of soil typically were about -3 MPa (-30 bars), while those in the bottom meter of soil were from -1.0 MPa to -1.5 MPa (-10 to -15 bars). To assess the potential for water to drain from these dry soils according to Darcy's Law, we used equation 9.2 from Campbell and Norman (1998):

$$K(\psi_m) = K_s\left(\frac{\psi_e}{\psi_m}\right)^{2+3/b} \qquad (4)$$

where, K_s is saturated hydraulic conductivity, ψ_m is matric potential, ψ_e is air entry water potential, and b is the exponent of the water release equation. We used the parameter estimates for a silty clay loam soil given in Table 9.1 of Campbell and Norman (1998) to estimate hydraulic conductivity. Hydraulic conductivity varied from 3.4 X 10^{-11} cm/s at a water potential of -1.0 MPa to 2.3 X 10^{-12} cm/s at a water potential of -3.0 MPa. These values are four to five orders of magnitude lower than the 10^{-7} required for a compacted clay layer in a RCRA cap (USEPA 1989). Clearly, a negligible amount of water would drain from the PCBE soils so long as the soil cap is sufficiently deep that soils near the bottom of the cap remain near the lower limit of extraction. The cap configurations that we have recommended should easily meet equivalency criteria

7.8 Potential Threats to Integrity of ET Caps at the INEEL

Wildfire

Concern is often expressed about the potential for wildfire to remove the vegetative cover from an ET cap, which subsequently could cause the cap to fail. The risk of wildfire is greatest late in the growing season when soil moisture has been depleted and many perennial grasses and forbs will have become dormant. Research at the INEEL and elsewhere in the region has shown that most of the perennial grasses and many shrubs and perennial forbs can resprout following fire (e.g., Ratzlaff and Anderson 1994, Patrick and Anderson 1999). Vegetative cover of INEEL areas burned in recent wildfires has recovered quickly (Patrick and Anderson 1999). Thus, if vegetation on an ET cap includes a diverse mix of species and life forms, including healthy populations of perennial grasses, cover on the cap can be expected to recover to prefire levels within two growing season (S. Patrick-Buckwalter 2002). It is likely that there would be sufficient cover in the first postfire season to use most of the precipitation received. In addition, the dry conditions that are likely to prevail when a fire occurs coupled with rapid vegetative recovery would likely prevent increase in soil moisture at the bottom of a cap, but additional research is recommended to confirm this.

Invasive Annual Plants

Associated with the concern about wildfire and the concern about inadequate vegetation establishment, is concern that postfire or post-revegetation vegetation on an ET cap may become dominated by invasive annual species such as cheatgrass. Research indicates that cheatgrass may not use all of the plant-available water in a deep soil (Cline et al. 1977, Anderson and Ratzlaff 1996). Accumulation of water might ultimately cause breakthrough of the cap. Postfire research at the INEEL and vicinity has shown that if vigorous populations of native perennial species are present when a wildfire occurs, the native community can recover and resist invasion by exotics (Ratzlaff and Anderson 1994, Patrick and Anderson 1999). Furthermore, cheatgrass does not establish well on fine-textured, clayey soils (Rasmuson 1996). On native-vegetation plots adjacent to the PCBE that were subjected to the same irrigation treatments as the PCBE plots, cheatgrass cover increased substantially in response to fall/spring irrigation (Morris 2001). However, on the PCBE plots, cheatgrass was rare and we observed no tendency for it to increase in response to irrigation. We conclude that if fine-textured soils are used for ET caps at the INEEL and climatically similar sites and that if those caps support a diverse community of native species, the risk of cheatgrass invasion is low.

Another invasive annual that may be problematic on disturbed sites having fine-textured soils is Russian thistle. Russian thistle is very common at the INEEL and was abundant on PCBE plots and surrounding disturbed areas for the first few years after plots were established. Because of its photosynthetic pathway (C_4), it requires relatively little water and therefore is not a desirable component of vegetation on an ET cap. Our experience at the PCBE site is that Russian thistle will persist only until perennial species become well established. Hence, if care is taken to ensure development of a diverse community of perennials on an ET cap, Russian thistle and other annuals should not pose a serious problem.

Burrowing Animals.

ET caps constructed according to our recommendations should provide sufficient depth of soil to meet the habitat needs of burrowing small mammals and ants. Although small mammal burrows and ant nests may increase infiltration and percolation of water, such increases are very modest and should not pose a problem on vegetated caps (Laundre 1993, Gaglio et al. 1998). We did not investigate the potential impacts of badgers (*Taxidea taxus*) on cap performance, but they would not be expected to burrow deeper than do ground squirrels (*Spermophilus* sp.), their major prey. Research at the Hanford site in eastern Washington indicated that, although badger burrows increased infiltration of rainfall, vegetation quickly removed the excess moisture. In fact, soils were consistently dryer beneath burrows than in non-burrow areas (Cadwell et al. 1989, Link et al. 1995).

7.9 Conclusion

We conclude that an ET cap constructed according to the recommendations above will preclude precipitation water from reaching interred wastes at the INEEL and climatically similar sites. The recommended cap configurations provide a low cost, low maintenance alternative to EPA's recommended RCRA cap and to more complex, highly engineered designs.

8.0 Literature Cited

Anderson, J. E., and A. D. Forman. 2002. The Protective Cap/Biobarrier Experiment: A Study of Alternative Evapotranspiration Caps for the Idaho National Engineering and Environmental Laboratory. Environmental Surveillance, Education, and Research report, Stoller Corporation and Idaho State University, STOLLER-ESER-46.

Anderson, J. E., and R. S. Inouye. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs 71:531-556.

Anderson, J. E., and T. D. Ratzlaff. 1996. Protective cap/biobarrier experiment II: a comparison of water use by annual and perennial species on simulated waste burial plots at the Idaho National Engineering Laboratory. Pages 45-48 *in* T. D. Reynolds and R. C. Morris, editors. Environmental Science and Research Foundation Annual Technical Report, Calendar Year 1995. Environmental Science and Research Foundation, Idaho Fall, Idaho.

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 Research and Rehabilitation **1**:229-244.

Anderson, J. E., R. S. Nowak, T. D. Ratzlaff, and O. D. Markham. 1991. Managing soil moisture on waste burial sites. Idaho Field Office, U.S. Department of Energy, Idaho Falls.

---. 1993. Managing soil moisture on waste burial sites in arid regions. Journal of Environmental Quality **22**:62-69.

Anderson, J. E., K. T. Ruppel, J. M. Glennon, K. E. Holte, and R. C. Rope. 1996. Plant communities, ethnoecology, and flora of the Idaho National Engineering Laboratory. Environmental Science and Research Foundation, Idaho Falls, Idaho.

Arthur, W. J. 1982. Radionuclide concentrations in vegetation at a solid radioactive waste disposal area in southeastern Idaho. Journal of Environmental Quality **11**:394-399.

Arthur, W. J., and O. D. Markham. 1983. Small mammal soil burrowing as a radionuclide transport vector at a radioactive waste disposal area in southeastern Idaho. Journal of Environmental Quality **12**:117-122.

Arthur, W. J., O. D. Markham, C. R. Groves, B. L. Keller, and D. K. Halford. 1986. Radiation does to small mammals inhabiting a solid radioactive waste disposal area. Journal of Applied Ecology **23**:13-26.

Arthur, W. J., O. D. Markham, C. R. Groves, and B. L. Keller. 1987. Radionuclide export by deer mice at a solid radioactive waste disposal area in southeastern Idaho. Health Physics **52**:45-53.

Beiswenger, J. M. 1991. Late Quaternary vegetational history of Grays Lake, Idaho. Ecological Monographs **61**:165-182.

Bengtsson, L., D. Bendz, W. Hogland, H. Rosqvist, and M. Akesson. 1994. Water balance for landfills of different age. Journal of Hydrology **158**:203-217.

Benson, C., T. Abichou, W. Albright, G. Gee, and A. Roesler. 2001. Field evaluation of alternative earthen final covers. International Journal of Phytoremediation 3:105-127.

Blom, P. E. 1990. Potential impacts on radioactive waste disposal situations by the harvester ant, *Pogonomyrmex salinus* Olsen (Hymenoptera: Formicidae). MS Thesis. University of Idaho, Moscow.

Blom, P. E., J. B. Johnson, B. Shafii, and J. Hammel. 1994. Soil water movement related to distance from three *Pogonomyrmex salinus* (Hymenoptera: Formicidae) nests in south-eastern Idaho. Journal of Arid Environments **26**:241-255.

Bowerman, A. G., and E. F. Redente. 1998. Biointrusion of protective barriers at hazardous waste sites. Journal of Environmental Quality **27**:625-632.

Cadwell, L. L., L. E. Eberhardt, and M. A. Simmons. 1989. Animal intrusion studies for protective barriers: status report for FY 1988. Pacific Northwest Laboratory, Richland, Washington.

Caldwell, M. 1985. Cold desert. Pages 198-212 *in* B. F. Chabot and H. A. Mooney, editors. Physiological ecology of North American plant communities. Chapman and Hall, New York.

Campbell, G. S., and J. M. Norman. 1998. An introduction to environmental biophysics. Springer-Verlag, New York.

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

Daniel, D. E., and B. A. Gross. 1995. Caps. Pages 119-140 *in* R. R. Rumer and J. K. Mitchell, editors. Assessment of barrier containment technologies: a comprehensive treatment for environmental remediation applications. National Technical Information Service, U.S. Department of Commerce, Springfield, Virginia.

Davis, O. K. 1981. Vegetation migration in southern Idaho during the late-Quaternary and Holocene. Ph.D. Dissertation. University of Minnesota, Minneapolis.

Davis, O. K., J. C. Sheppard, and S. Robertson. 1986. Contrasting climatic histories for the Snake River Plain, Idaho, resulting from multiple thermal maxima. Quaternary Research **26**:321-339.

Fayer, M. J., and T. L. Jones. 1990. UNSAT-H Version 2.0: Unsaturated soil water and heat flow model. Publication PNL-6779. Pacific Northwest Laboratory, Richland, Washington.

Ferguson, S. A. 1997. A climate-change scenario for the Columbia River Basin. *in* T. M. Quigley, editor. Interior Columbia Basin ecosystem management project: scientific assessment. U.S. Department of Agriculture Pacific Northwest Research Station, Portland, Oregon.

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 p.

Gaglio, M. D., E. A. Osorio, W. P. MacKay, and S. I. Watts. 1998. Effectiveness of biobarriers in preventing harvester ants from entering waste sites. Pages 34-37 *in* T. D. Reynolds and R. W. Warren, editors. Environmental Science and Research Foundation, Inc., annual technical report to DOE-ID: calendar year 1997. Environmental Science and Research Foundation, Idaho Falls, Idaho.

Giorgi, F., C. S. Brodeur, and G. T. Bates. 1994. Regional climate change scenarios over the United States produced with a nested regional climate model. Journal of Climate Change **7**:375-399.

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, New Mexico.

Hakonson, T. E., J. L. Martinez, and G. C. White. 1982. Disturbance of low-level waste burial site cover by pocket gophers. Health Physics **42**:868-871.

Hakonson, T. E., J. F. Clime, and W. H. Rickard. 1983. Biological intrusion barriers for large volume waste disposal sites. NUREG/CP-0028, Vol. 3. U.S. Nuclear Regulatory Commission, Silver Spring, Maryland.

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. (Eds: Reith,CC; Thomson,BM) University of New Mexico Press, Albuquerque, New Mexico:101-146.

Hector, A., B. Schmid, C. Beierkuhnlein, M. C. Caldeira, M. Diemer, P. G. Dimitrakopoulos, J. A. Finn, H. Freitas, P. S. Giller, J. Good, R. Harris, P. Hogberg, K. Huss-Danell, J. Joshi,

A. Jumpponen, C. Korner, P. W. Leadley, M. Loreau, A. Minns, C. H. P. Mulder, G. O. Donovan, S. J. Otway, J. S. Pereira, A. Prinz, D. J. Read, M. Scherer-Lorenzen, E.-D. Schulze, A.-S. D. Siamantziouras, E. M. Spehn, A. C. Terry, A. Y. Troumbis, F. I. Woodward, S. Yachi, and J. H. Lawton. 1999. Plant diversity and productivity experiments in European grasslands. Science **286**:1123-1127. (Science)

Hillel, D. 1998. Environmental Soil Physics. Academic Press, San Diego, California.

Johnson, J. B., and P. E. Blom. 1997. Studies on the effectiveness of biobarriers against harvester ant excavation of buried waste: laboratory experiments. Pages 55-60 *in* R. C. Morris and R. D. Blew, editors. Environmental Science and Research Foundation Annual Technical Report, Calendar Year 1996. Environmental Science and Research Foundation, Idaho Falls, ID.

Laundre, J. W. 1989. Burrows of least chipmunks in southeastern Idaho. Northwestern Naturalist **70**:18-20.

---. 1993. Effect of small mammal burrows on water infiltration in a semi-arid environment. Oecologia **94**:43-48.

Laundre, J. W., and T. D. Reynolds. 1993. Effects of soil structure on burrow characteristics of five small mammal species. Great Basin Naturalist **53**:358-365.

Link, S. O., W. J. Waugh, and J. L. Downs. 1994. The role of plants in isolation barrier systems. Pages 561-592 *in* G. W. Gee and N. R. Wing, editors. In-situ remediation: scientific basis for current and future technologies. Thirty-third Hanford symposium on health and the environment. Batelle Press, Richland, Washington.

Link, S. O., L. L. Cadwell, K. L. Petersen, M. R. Sackschewsky, and D. S. Landeen. 1995. The role of plants and animals in isolation barriers at Hanford, Washington. PNL-10788. Pacific Northwest Laboratory, Richland, Washington.

Marlette, G. M., and J. E. Anderson. 1986. Seed banks and propagule dispersal in crested-wheatgrass stands. Journal of Applied Ecology **23**:161-175.

McNaughton, S. J. 1977. Diversity and stability of ecological communities: a comment on the role of empiricism in ecology. American Naturalist **111**:515-525.

---. 1993. Biodiversity and function of grazing ecosystems. Pages 361-383 *in* E.-D. Shulze and H. A. Mooney, editors. Biodiversity and ecosystem function. Springer-Verlag, Berlin. (64)

Morris, A. N. 2001. The effects of changes in timing and amounts of precipitation on vegetation dynamics and nitrogen mineralization in a sagebrush-steppe ecosystem. Ph.D. Dissertation. Idaho State University, Pocatello.

Nativ, R. 1991. Radioactive Waste Isolation in Arid Zones. Journal of Arid Environments **20**:129-140.

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

Olsen, G. L., D. J. Jeppesen, and R. D. Lee. 1995. The staus of soil mapping for the Idaho National Engineering Laboratory. INEL-95/0051.

Patrick-Buckwalter, S. 2002. Postfire vegetation dynamics in sagebrush steppe on the Eastern Snake River Plain, Idaho. MS Thesis. Idaho State University, Pocatello.

Patrick, S. M., and J. E. Anderson. 1999. Fire ecology of the INEEL. Pages 66-69 *in* D. L. Weigmann and R. D. Blew, editors. Environmental Science and Research Foundation, Inc., Annual Technical Report to DOEID: Calendar Year 1998. Environmental Science and Research Foundation, Idaho Falls, Idaho.

Porro, I. 2001. Hydrologic behavior of two engineered barriers following extreme wetting. Journal of Environmental Quality 30:655-667.

Pratt, G.C. 2000. Components of habitat selection, predation risk and biobarrier design for use in managing populations of ground squirrels on low-level radioactive waste landfill caps. MS Thesis. Idaho State University, Pocatello.

Rasmuson, K. E. 1996. Population and individual responses of *Bromus tectorum* to environmental stresses: a study of factors that may limit its distribution in cold desert habitats. Ph.D. Dissertation. Idaho State University, Pocatello.

Ratliff, L. F., J. T. Ritchie, and D. K. Cassel. 1983. Field-measured limits of soil water availability as related to laboratory-measured properties. Soil Science Society of America Journal **47**:770-775.

Ratzlaff, T. D., and J. E. Anderson. 1994. Vegetal recovery following wildfire in seeded and unseeded sagebrush steppe. Journal of Range Management **48**:386-391.

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

Reynolds, T. D., and J. W. Laundre. 1988. Vertical distribution of soil removed by four species of burrowing rodents in disturbed and undisturbed soils. Health Physics **54**:445-450.

Reynolds, T. D. and W. Wakkinen. 1987. Characteristics of the burrows of four species of rodents in undisturbed southeastern Idaho soils. American Midland Naturalist 118:245-250.

Ritchie, J. T. 1981. Soil water availability. Plant and Soil **58**:327-338.

Sackschewsky, M. R., C. J. Kemp, S. O. Link, and W. J. Waugh. 1995. Soil water balance changes in engineered soil surfaces. Journal of Environmental Quality **24**:352-359.

Schmugge, T. J., T. J. Jackson, and H. L. McKim. 1980. Survey of methods for soil moisture determination. Water Resources Research **16**:961-979.

Shumar, M. L., and J. E. Anderson. 1986. Gradient analysis of vegetation dominated by two subspecies of big sagebrush. Journal of Range Management **39**:156-160.

---. 1987. Transplanting wildings in small revegetation projects. Arid Soil Research and Rehabilitation 1:253-256.

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

Tilman, D., and J. A. Downing. 1994. Biodiversity and stability in grasslands. Nature **367**:363-365.

Tilman, D., and A. El Haddi. 1992. Drought and biodiversity in grasslands. Oecologia **89**:257-264.

Tilman, D., D. Wedin, and J. Knops. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature **379**:718-720.

Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie, and E. Siemann. 1997a. The influence of functional diversity and composition on ecosystem processes. Science **277**:1300-1302.

Tilman, D., C. L. Lehman, and K. T. Thomson. 1997b. Plant diversity and ecosystem productivity: theoretical considerations. Proceedings of the National Academy of Sciences (USA) **94**:1857-1861.

USEPA. 1989. Technical guidance document: final covers on hazardous waste landfills and surface impoundments. U.S. Environmental Protection Agency Report 530-SW-89-047. Office of Solid Waste and Emergency Response, U. S. Environmental Protection Agency, Washington, D.C.

Walter, H., E. Harnickell , and D. Mueller-Dombois. 1975. Climate-diagram maps of the individual continents and the ecological climatic regions of the earth. Springer-Verlag, Berlin.

Waugh, W. J., M. E. Thiede, D. J. Bates, L. L. Cadwell, G. W. Gee, and C. J. Kemp. 1994. Plant cover and water balance in gravel admixtures at an arid waste-burial site. Journal of Environmental Quality **23**:676-685.

Winkel, V. K., B. A. Roundy, and J. R. Cox. 1991. Influence of seedbed microsite characteristics on grass seedling emergence. Journal of Range Management **44**:210-214.

Wright, J. L., and M. E. Jensen. 1972. Peak water requirements of crops in southern Idaho. Journal of the Irrigation Drainage Division, Society of Civil Engineering **98**:193-201.