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Ecohydrology Monitoring and Excavation of Semiarid Landfill Covers a Decade after Installation

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ABSTRACT

Landfill covers are intended to protect buried waste from water seepage and biointrusion for thirty to thousands of years, yet most cover studies are limited to a few years and do not directly investigate net changes in the soil profile that affect changing landfill performance. We evaluated water balances, vegetation cover, rooting patterns, and soil profiles of two landfill-cover designs (two plots each) more than a decade after installation at semiarid Los Alamos National Laboratory, NM, USA: a conventional design of 20 cm of topsoil over compacted crushed-tuff and an integrated design of 71 cm of topsoil over an engineered barrier designed to induce lateral flow (geotextile overlying 46 cm of gravel). Water balances for both designs had ~3% of precipitation as seepage; the integrated plots lost <1% of water as interflow, probably because the barrier interface had only a 5% slope. The conventional design had a net loss of stored soil water and proportionally more evapotranspiration than the integrated design. After more than a decade, (i) vegetation changes included increased biomass and species diversity on most plots, with proportionally fewer invading species and more extensive rooting in the integrated plots; (ii) the geotextile was largely unchanged; and (iii) infiltration and subsequent water penetration occurred primarily via macropores, including root channels and animal burrows. Both cover designs effectively minimized seepage during their initial decade, but observed effects of environmental processes such as succession and burrowing are expected to become progressively more important determinants of cover performance over additional decades.

A key strategy for isolating the effects of contaminants in the environment and mitigating associated human and ecological risks is to apply engineered covers over landfills used for disposal of hazardous and municipal solid waste (Reith and Thomson, 1993; Zornberg et al., 2003). Landfill covers, which include various combinations of soil layers, engineered barriers, and liners, are expected to exhibit long-term stability to effectively isolate wastes and to minimize the risks of exposure to the public (Albright et al., 2004; EPA, 1989; Reith and Thomson, 1993; Wing and Gee, 1994). The duration for monitoring and maintenance of landfill covers after closure varies but is generally not expected to exceed more than 30 to 50 years for cases in which institutional control is applied (Suter et al., 1993). However, risks associated with the waste frequently persist beyond institutional control; hence, the longer-term integrity of landfill covers is of concern.

Design of a landfill cover requires consideration of several tradeoffs. To keep the wastes dry, the cover is designed to minimize seepage. This can be achieved in part by storing soil water within the cover and by maximizing the subsequent removal of the stored water through evapotranspiration. The evaporative component of total evapotranspiration can be modified by orienting the cover to maximize incoming solar radiation (e.g., Nyhan et al., 1997), while the transpiration component can be modified through selection and management of plant species on the cover (e.g., Lopez et al., 1988). Seepage can be reduced further by increasing interflow (shallow subsurface lateral flow of water at the interfaces between layers) that is then diverted to a location away from the waste via engineering structures (e.g., Khire et al., 2000; Nyhan et al. 1997, 2001). Conversely, surface runoff from the cover should be minimized through establishment of ground cover sufficient to minimize water erosion, which degrades the integrity of the cover through time. Simultaneously, the cover must minimize biointrusion by plant roots and burrowing animals. The relative importance of these tradeoffs varies with climate (e.g., Albright et al., 2004; Khire et al., 2000). Caution must be used when extrapolating information from site to site because the seasonal occurrence of rain and snow varies with climate and impacts the dynamics of the amounts of soil water that can be stored in the landfill profiles (Gee et al., 1998).

Development of landfill cover designs appropriate for arid and semiarid environments is of particular interest given that hazardous waste disposal sites are numerous in the semiarid to arid western USA (Albright et al., 2004; see also Nyhan, 2005 and references therein). Evaluation of alternative cover designs for arid and semiarid environments has been the focus of several U.S. Department of Energy (DOE) sites (e.g., Albright et al., 2004; Dwyer, 1998; Link et al., 1995; Lopez et al., 1988; Nyhan et al., 1989a, 1989b; Nyhan et al., 1990; Nyhan et al., 1997), and a few on U.S. Department of Defense (DOD) sites (Hakonson et al., 1994; Paige et al., 1996; Warren et al., 1996;). Most studies have not evaluated cover performance for more than a few years (but see Andraski et al., 1995). The cover storage capacity at a site can be designed to be appropriate for a given site's precipitation by varying soil and thickness using relatively straightforward calculations (Reith and Thomson, 1993). The more challenging aspects of cover design involve determining vegetation effects on hydrology (e.g.

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Anderson et al., 1993; Waugh et al., 1994) and performance of engineered barriers designed to shunt downward moving water away from waste below and/or to prevent root penetration and animal burrowing into the waste (e.g., Nyhan et al. 1997, 2001). These topics require an improved understanding of the interface between ecology and hydrology, the focus of the emerging interdisciplinary area of ecohydrology. Importantly, to date there has been little investigation of changes in properties of the cover itself through direct investigation by excavating tested covers.

With time, engineered landfill covers are subject to modification by environmental processes, particularly after institutional control has ceased. The structure, bulk density, and effective permeability of cover layers can be altered through time by pedogenic processes and related disturbances by plants and animals. Environmental changes with time can result in rooting patterns, evapotranspiration, and erosion that are quite different from initial conditions. Climate changes may affect a site's water balance directly through increased or decreased precipitation and indirectly through influences on pedogenic and ecological factors. Numerous reports have pointed out the potential for environmental processes to modify landfill covers and liners (Gonzales et al., 1995; Johnson and Urie, 1985; Link et al., 1995; Nicholson and Safaya, 1993; Waugh and Smith, 1996). Several mechanisms whereby landfill barriers are likely to fail in >100 yr are discussed by Suter et al. (1993), who recommend that either perpetual care be required or that barriers be designed for long-term integrity. Construction materials may not be durable for periods of decades or longer (Pertusa, 1980). The National Research Council (1997) has highlighted the need for knowledge concerning the effective lifetimes of selected barrier materials and resultant barrier systems.

Despite the clear importance of designing landfill covers that will perform adequately over long time periods, most field-based studies of landfill liners and caps provide just a few years of data. Modeling environmental processes provides a means of projecting landfill performance further into the future, but the validity of such projections is limited by the quality and quantity of field data used for parameterization and testing of the models (Barnes and Rodgers, 1987; Nyhan, 1989; Paige et al., 1996). Fundamental ecological processes such as succession are not even factored into current models, yet they directly affect the integrity of landfill covers through biointrusion, erosion, and water balance (Nyhan et al. 1998). Waugh and Smith (1996) point out that natural analogs can sometimes be used to help project the effects of possible changes in climate, soil morphology, and ecology.

Here we focus on the next phase of evaluating long-term cover performance: integrating evaluation of hydrological and ecological processes both above- and below-ground through time series measurements of water balance and excavation of study covers to assess changes in vegetation, soil, and engineered materials. We evaluated the performance and the physical and ecological characteristics of two landfill-cover designs—a conventional design comprising a layer of topsoil over crushed bedrock and an integrated design that includes an engineered barrier—more than a decade after installation in a semiarid environment. The study was conducted at Los Alamos National Laboratory, where several existing landfills are expected to have covers placed on them in the future. Our objectives were to evaluate the status of these two cover designs with respect to water balance, selected soil properties, deterioration of geotextile liners, vegetation, root distributions, and preferential flow patterns and relative differences in infiltration and water penetration rates. On the basis of our findings, we hypothesize how the relative importance of engineering factors (e.g., slope, texture and thickness of layers) vs. environmental factors (e.g., succession, burrowing) may change with time in determining the integrity of landfill covers.

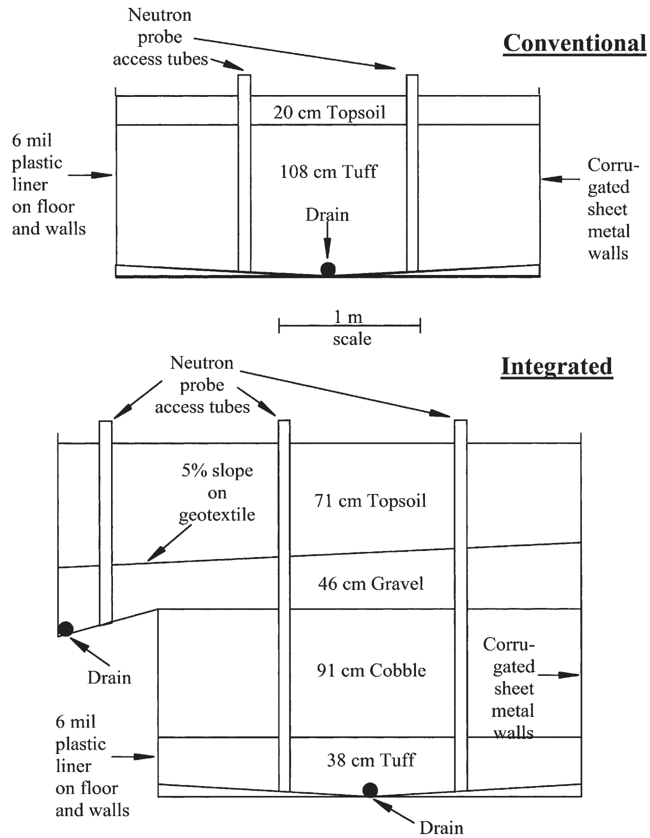

Fig. 1. Cross-sections showing design of landfill cover plots. A gravel mulch was applied to the surface of both designs.

METHODS

Two test plots for each of two landfill-cover designs were installed during the spring and summer of 1984 at Los Alamos National Laboratory (Nyhan et al., 1990). The plots were intended to provide data for comparing water balance of two landfill-cover designs in a semiarid environment. The plots were not situated over actual waste. The two cover designs, designated "conventional" and "integrated", are shown schematically in Fig. 1, and are described in detail by Nyhan et al. (1990). We use the term conventional for the design that Nyhan et al. (1990) referred to as "control" to be consistent with more recent studies (Nyhan et al., 1997) and to highlight that this design has been used conventionally at Los Alamos and elsewhere. Similarly, we use the term "integrated" for the design that Nyhan et al. (1990) referred to as "improved" to highlight that the design has an engineered barrier integrated within it to promote lateral subsurface flow, and to de-emphasize a priori presumed differences in design performance. The topsoil texture used in both designs was previously described as sandy loam (Nyhan et al., 1990) but during the excavation it became apparent that some gravel in the backfill pile had inadvertently been incorporated during construction, resulting in a topsoil texture of gravelly sandy loam. Both designs had a slope at the ground surface of 0.5% to minimize the amount of runoff from the plots so that percolation would be maximized, thereby more readily enabling estimation of the water budget (Nyhan et al., 1990).

The conventional design consisted of ~20 cm of topsoil (gravelly sandy loam) overlying 108 cm of compacted crushed tuff. The base of the plots sloped downward by 5% toward a perforated drain pipe running the length of the plots (whereas the slope at the surface of the plot was 0.5%, as noted above). These drain pipes were connected to collection stations (one for each plot), thereby allowing us to directly measure seepage from the plots. The vertical walls of the plots were made of corrugated sheet metal, and the walls and base of the plots were lined with 0-15-mm (6-mil) polyethylene. The conventional-design plots were each 3.0 by 10.7 m in area.

The integrated design consisted of 71 cm topsoil (gravelly sandy loam), 46 cm of 5- to 10-mm diameter gravel, 91 cm of 10- to 30-cm diameter cobble, and 38 cm of crushed tuff. The topsoil and gravel layers were separated by a high conductivity (0.24 m s^{-1}) geotextile (600X Brand, manufactured by MIRAFI, El Toro, CA) to maintain the distinct interface between these different textured layers. This boundary was constructed with a 5% slope toward the shoulder interflow drain (as noted above, the slope at the surface of the plot was 0.5%). The walls and floor of these plots were constructed in the same way as the conventional plots, with the addition of an overhanging shoulder with a perforated drain pipe for collecting interflow from the topsoil layer (see Fig. 1). Hence, the integrated plots were 3.7 by 10.7 m at the surface, the greater width due to the overhanging shoulder in the topsoil and gravel layers. The interface between topsoil and gravel with the associated geotextile was designed to cause lateral interflow of vertically penetrating water—because of the large difference in saturated hydraulic conductivities of these two layers. Hence, the cover was designed to convert vertical flow of penetrating water to lateral flow over to the side of the cover where there was a drain for capturing it. The cobble layer was designed to serve as a biobarrier that would minimize penetration by roots (e.g., Reynolds, 1990), because little soil water could be retained within the layer, and by burrowing animals, which cannot burrow effectively within cobble.

We obtained sufficient measurements to estimate the major components of the water balance for each cover. The water balance was estimated for one-dimensional movements of water (cm) in the soil profile using the following equation:

$$\Delta S = P - Q - \mathrm{ET} - L - I$$

where ΔS = change in soil water storage, P = precipitation, Q = runoff, ET = evapotranspiration, L = seepage (or percolation), and I = interflow (applicable to the integrated design only).

Data for estimating water balance on each plot were collected from 13 Aug. 1984 to 21 Nov. 1994 at variable intervals. For much of the study, we were able to obtain measurements approximately every 1 to 2 weeks; for other periods within the more than one decade of study, sampling intervals were longer due largely to issues associated with continuity of funding. We directly measured precipitation, soil water content at various depths (providing an estimate of change in soil water storage, as described below), interflow (applicable to the integrated plots only), and seepage. We assumed that there was no runoff because the plots had 0.5% slope and were bordered, preventing any water from leaving the plot. We estimated the remaining term—evapotranspiration—by difference using the water balance equation above. Note that in this approach all measurement errors impact the estimate of evapotranspiration. Precipitation was measured between the two central plots using a tipping bucket rain gauge with a heater and windscreen and a long-term event recorder (Weathermeasure Corp., Sacramento, CA).

Change in soil water storage was estimated from measurements of volumetric water content that were obtained using a Campbell Pacific Model 503 neutron moisture gauge (Campbell Pacific Nuclear Corp., Martinez, CA); the gauge was calibrated for crushed tuff backfill (Nyhan et al., 1983) and for topsoil (Nyhan et al., 1994). Each of the four plots included vertical aluminum access tubes for monitoring soil water content using a neutron probe. Soil water content was measured in the conventional plots in the topsoil (20 cm) just above the interface with crushed tuff, and at depths of 40, 80, and 100 cm in the crushed tuff, and in the integrated plots at two depths in the topsoil (20 and 40 cm, as well as at 60 cm only in the overhang area) and in the crushed tuff below the engineered barrier (220 cm; see Fig. 1). We used the calibration curve for topsoil, which was originally determined for topsoil that did not include any gravel. Excavations of the plot revealed that some gravel was inadvertently incorporated into the topsoil of plots, as mentioned above, and that some of the gravel applied to the ground surface migrated a few cm into the profile over the course of the study. However, this gravel influenced only a small fraction of topsoil, and previous studies indicated that the gravel would have a negligible effect on the neutron probe data (Nyhan et al., 1983). An additional issue for the integrated plots was that the 20-cm depth measurement was obtained directly above the interface between topsoil and crushed tuff. Although the hydrological properties of the topsoil and crushed tuff are quite different, they both had similar calibration curves for the neutron probe (Nyhan et al., 1994). Hence, we used the topsoil calibration curve alone rather than attempting to determine time-weighted calibrations that would vary with soil water content, as the latter required making additional assumptions. Depth-weighted averages of soil water content data were used in estimating change in soil water content. Soil water content was averaged by depth across six sampling locations for each sampling time and multiplied by the thickness (cm) of soil in each portion of the profile to determine soil water inventories, as described previously and in greater detail by Nyhan et al. (1990).

Seepage production in the integrated plots was originally determined (Nyhan et al., 1990) using only the measured flow occurring in the french drain located at the bottom of the crushed tuff layer beneath the biobarrier. Subsequently, we determined that this technique underestimated the total seepage produced, because it did not account for the increases in soil water inventory that occurred over time in the crushed tuff layer The current total seepage estimates take the changes in tuff water inventory into account by adding increases in the crushed tuff water content to the measurements of flow in the drains.

The plots were initially seeded with *Bouteloua gracilis* L. (blue grama) and *Pascopyron smithii* (Rydberg) Á. Löve (western wheatgrass; previously *Agropyron smithii*) in the summer of 1984. Vegetation establishment was aided by application of supplemental water during the summer months of that year (Nyhan et al., 1990). Subsequently the plots received only natural precipitation for the remainder of the 10-year study period. Estimates of biomass for August 1986 are reported in Nyhan et al. (1990). The vegetation was resurveyed in late July of 1995 (measurements were not obtained at intermediate periods of the study). Ground cover and species composition for each plot was measured using a point frame ($n = 12$ rows, each 3 m wide with measurements obtained every 6 cm). Biomass was estimated from quadrats (20 by 50 cm) located around the periphery of the plots ($n = 6$ per plot), around neutron probe access tubes ($n = 6$ per plot), and in the centers of the plots ($n = 8$ per plot). The proportions of biomass from *B. gracilis, P. smithii,* and other species were estimated from the point frame data. All vegetation within the quadrats was clipped at ground level and bagged, and the green leaf component (separated from standing dead biomass) was dried at 25°C for ≥24 hr and weighed. Leaf area index (LAI) was also determined for the green leaf component for each of the center quadrats by analyzing each sample with a LI-COR portable area meter (Model LI-3000, LI-COR, Lincoln, NE) twice and averaging the results. Leaf area and biomass were then correlated for all of the plots.

We tested relative rates and modes of infiltration and subsequent water penetration into the soil using dye to provide a relative comparison of the hydraulic behavior of the two cover designs under extreme conditions (i.e., a large precipitation input such as a snowmelt event), and to highlight preferential flow paths under those conditions. We used a modification of a falling head method that has been used to determine saturated hydraulic conductivity, similar to that described by Klute and Dirksen (1986). A determination of actual hydraulic conductivity was not attempted because of the difficulty in applying these methods at the plot scale, nor did we remove cores for applying such a test. On each plot, a sheet-metal border (0.9 by 1.9 m) was pressed 5 cm into the soil. A solution of red fluorescein dye in water was ponded to a depth of 25 cm inside each border. The depth of the solution was then recorded at regular time intervals until the ponded depth was less than 5 cm. The 25-cm depth of dye solution represents an extreme case, designed not to simulate actual precipitation but to force dye into the cover systems and highlight preferential flow pathways. Dye tests were conducted after the vegetation survey was complete in August 1995 to compare cover designs and evaluate the role of macropores. All dye tests of infiltration and subsequent water penetration were completed within a 24-hour period when antecedent moisture was quite low and similar on all plots, due to a lack of significant precipitation during the preceding several days.

All four plots then were excavated from the side to obtain pedologic descriptions, dye-penetration sampling, and root counts after vegetation surveys and infiltration testing were completed. Because pedologic descriptions were not recorded nor samples collected at the time of installation of the plots, changes in soil properties at the end of the study could only be inferred on the basis of assumptions of properties at the beginning of the study. For this reason, only abbreviated pedologic descriptions were recorded, consisting of field textures (including estimates of gravel content), observations of soil structure and horizonation, coloration patterns, and general root density patterns. Although we did not expect any major pedogenic development to occur over a single decade, the pedologic descriptions used in concert with the results of the dye infiltration and subsequent water penetration tests provide an unprecedented direct evaluation soil profile changes within the covers.

Root counts were obtained using the profile wall method (Böhm, 1979). Roots were counted on 50-cm long transects on vertical and horizontal surfaces at 10-cm depth intervals, beginning at a depth of 5 cm and extending to 105 cm for the conventional plots (into the crushed tuff) and to 75 cm for the integrated plots (base of topsoil). Counts were obtained on both vertical and horizontal surfaces at the same depths and locations. All fine (<2 mm diameter) and medium (2–5 mm diameter) roots intersecting the side of a 50-cm long steel wire frame were counted.

Bulk density cores were collected at plot walls and plot interiors on all plots, at depths dependent on cover design. For the conventional design, cores were collected in the topsoil near the surface (10 cm), in the topsoil just above the topsoil–crushed tuff interface (20 cm), in the upper portion of the crushed tuff (30 cm), and in the lower portion of the crushed tuff (100 cm). For the integrated design, cores were collected near the surface of the topsoil (5 cm), near the middle of the topsoil (30 cm), and slightly above the sloping interface between the topsoil and the gravel forming the capillary barrier (55 cm). Cores were obtained using 6.3-cm diameter by 7.5-cm long brass tubes. The cores were oven dried for 48 hours at 105°C and weighed. Bulk density was calculated by dividing the oven-dry sample weight by the known volume of the cores. We conducted three sets of one-way analysis of variance (ANOVA, Fisher's protected least significant difference test, $P < 0.05$) to test for differences in bulk density with respect to (i) cover design, (ii) depth within the conventional design, and (iii) depth within the integrated design.

RESULTS

Water Balance

The long-term measurements spanning 189 sampling intervals between 1984 and 1994 allowed us to estimate the average water balance for both designs (Table 1). Of the nearly 500 cm of precipitation that occurred over the course of the study, the vast majority was lost as evapotranspiration, as expected (Wilcox et al., 2003b). The conventional design lost ~98% of the precipitation, whereas the integrated design lost ~95%. The conventional plots were drier at the end than the start of the

Table 1. Water balance for conventional (C) and integrated (I) plots, August 1984 to November 1994 with respect to totals (cm) and as percentages of precipitation input (%; includes rounding error).

PLOT	Precipitation	Evapotranspiration†	Storage‡	Interflow§	Runoff¶	Seepage
C1	496.0 (100.00%)	485.9 (97.95%)	−5.7 (−1.16%)		0.0 (0.00%)	15.9 (3.20%)
C2	496.0 (100.00%)	488.1 (98.39%)	−6.3 (−1.27%)		0.0 (0.00%)	14.3 (2.88%)
I1	496.0 (100.00%)	472.7 (95.30%)	9.8 (1.97%)	0.04 (0.01%)	0.0 (0.00%)	13.5 (2.72%)
I2	496.0 (100.00%)	470.6 (94.87%)	6.9 (1.39%)	3.4 (0.69%)	0.0 (0.00%)	15.2 (3.06%)

† Evapotranspiration calculated by difference.
‡ Storage estimated from neutron probe soil water data as in Nyhan et al. (1990).
§ Interflow from side shoulder drains in integrated plots only.
¶ Runoff was eliminated by zero surface slope and metal flashing around plot borders to contain surface water.

study, resulting in a decrease of ~1.2% of total precipitation input, whereas the integrated plots were wetter at the end of the study by 1.4 to 2.0%. This difference in soil water storage accounted for most of the difference between the two designs in evapotranspiration. The integrated plots did indeed produce interflow, but the interflow accounted for <1% of the precipitation input. The amount of seepage was about ~3% and, in contrast to expectations, was not significantly less (t test, $P >$ 0.05) on the integrated plots (2.7–3.8%) than the conventional plots (2.9–3.2%).

The temporal dynamics of the water balance highlight fluctuations in soil water content that lead to relatively continuous increases in cumulative evapotranspiration and more sporadic increases in cumulative interflow and/or cumulative seepage (Fig. 2). During the earlier part of the study, fluctuations in the soil water inventory of the integrated design was more pronounced than on the

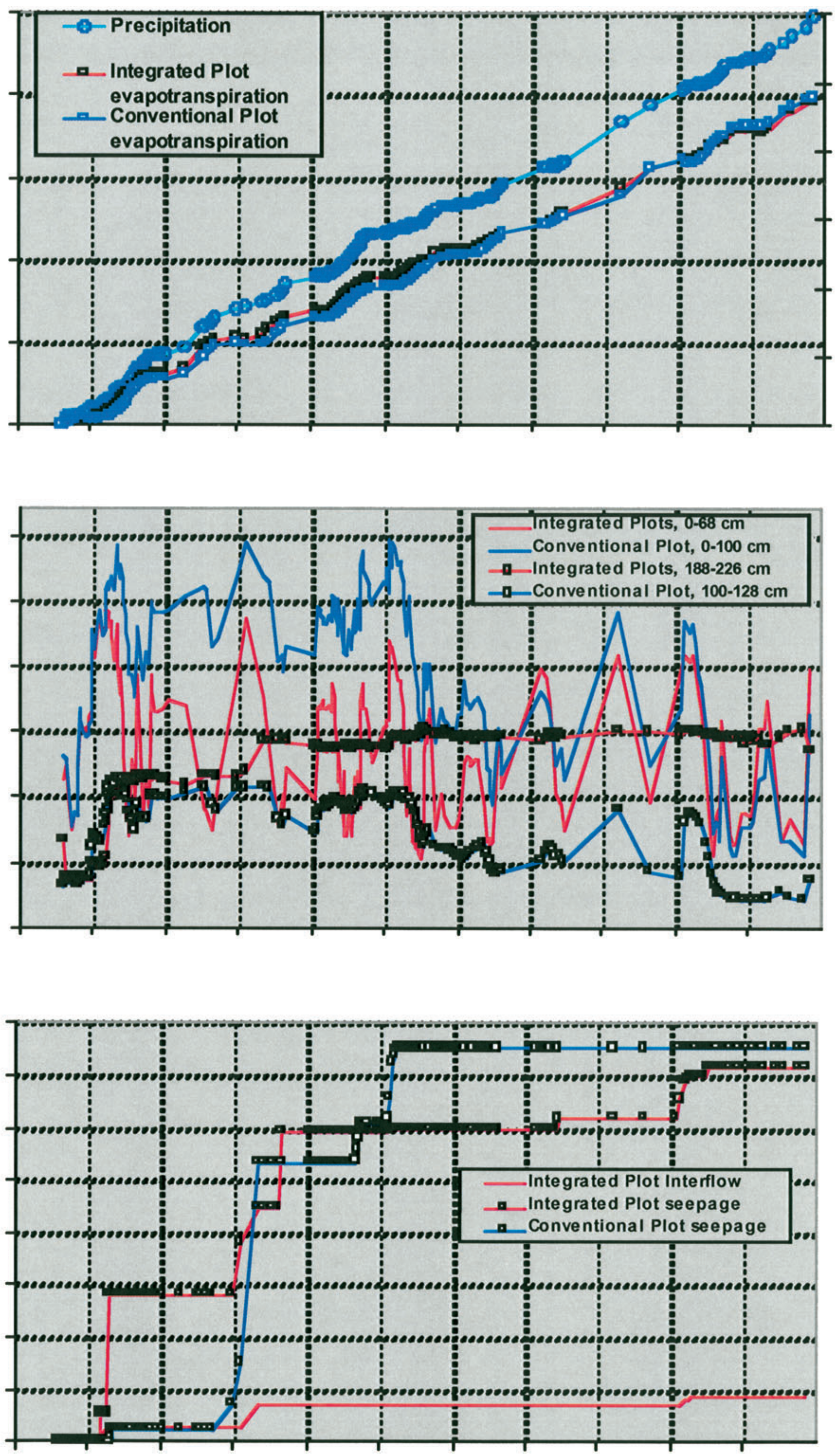

Fig. 2. Average volumetric water content in Conventional Plots and Integrated Plots from 1984 through 1994.

conventional design, resulting in higher evapotranspiration from the integrated plots during much of the study (e.g., 1987–1990). Towards the later part of the study, the soil water inventory on the conventional plot became more depleted, resulting in a cumulative loss of soil water storage and greater cumulative evapotranspiration for the conventional design compared to the integrated design. This decrease in soil water inventory for the conventional design was also reflected in changes in the deeper portion of the conventional design (100–128 cm) that were used to supplement direct measurements of seepage from the plot. In contrast, the soil water inventory below the biointrusion barrier of the integrated design exhibited a buildup of soil water content from 1984 to 1987 and then remained relatively stable. The integrated design produced interflow primarily during three short intervals (occurring one each in 1985, 1987, and 1993), all of which occurred shortly after soil water inventory for the upper portion of the integrated plots had exceeded 20 cm (although there was not a similar increase in interflow in 1989 after the soil water inventory exceeded 20 cm). Seepage on the integral plots occurred during the same intervals as when interflow occurred but also occurred during several other time intervals, all of which followed periods when soil water content in the top portion of the integrated design exceeded 15 cm. Surprisingly, seepage from the control design occurred at somewhat different intervals than the seepage from the integrated design, particularly as time progressed. Note that most of the seepage on the conventional plots occurred during three intervals (1985, 1987, and 1989) when soil water inventory in the upper portion of the profile exceeded 25 cm.

Soil Properties

Not unexpectedly, we found that little or no discernible changes in soil properties had taken place since installation of the plots. There was no horizonation (e.g., changes in soil color, texture, or structure) other than that designed for each plot. Macropores were present, but there was no corresponding, clearly-defined soil structure. Bulk density was variable both within and among plots, but did not show any patterns of pedogenesis that might be expected over longer time intervals. Bulk densities for the upper soils ranged from 1.17 to 1.45 Mg m^{-3} (Table 2), with an overall mean of 1.34 Mg m^{-3} (n = 28). There were no significant differences in bulk density with respect to sampling depth or plot type (Table 2; Fisher's protected least significant difference, $P > 0.05$). Because pedologic descriptions were not recorded when the plots were installed, assumptions have been made regarding the properties of the soils as installed. It is assumed that the topsoils were mixed such that texture and color were uniform throughout the topsoil layer. It is further assumed that the topsoil was in a relatively dry condition, so that no artificial soil structure was created during installation (i.e., the soil was essentially loose, single grains).

Table 2. Soil characteristics from vertical profiles of conventional (C) and integrated (I) plots.

Plot	Depth	Texture†	Structure‡	Bulk density, wall	Bulk density, interior
	cm			Mg m^{-3}	
C1	5	grSL	SG	–	–
	10	grSL	SG	1.17	1.43
	20	grSL	SG	1.20	1.43
	30	LS	M	1.17	1.33
	55	LS	M	–	–
	100	LS	M	1.22	1.35
C2	5	grSL	SG	–	–
	10	grSL	SG	1.43	1.33
	20	grSL	SG	1.44	1.30
	30	LS	M	1.29	1.33
	55	LS	M	–	–
	100	LS	M	1.30	1.38
I1	5	grSL	SG	1.32	1.38
	10	grSL	SG	–	–
	20	grSL	SG	–	–
	30	grSL	SG	1.35	1.21
	55	grSL	SG	1.38	1.45
	100	gravel	SG	–	–
I2	5	grSL	SG	1.41	1.33
	10	grSL	SG	–	–
	20	grSL	SG	–	–
	30	grSL	SG	1.34	1.35
	55	grSL	SG	1.40	1.40
	100	gravel	SG	–	–

† grSL, gravelly sandy loam; LS, loamy sand.
‡ SG, single-grain; M, massive.

Integrity of the Geotextile Liner

The geotextile that separated the soil from the biointrusion barrier in the integrated plots was found to be essentially intact (Fig. 3a). While some fine roots had penetrated the geotextile, apparently through pores in the liner, the geotextile itself did not appear to have lost integrity (i.e., there were no holes or tears). Moreover, the gravel–cobble layer contained a small amount of finer soil particles, primarily sand-sized grains, estimated to be less than 5% by volume. The spatial distribution of this sand suggested that it was probably present in the gravel and cobble fill at the time of installation of the plots. Hence, the observations indicate to us that the geotextile also apparently maintained functional integrity, with no discernable amount of soil passing through the geotextile into the gravel–cobble layer below.

Vegetation

Vegetation characteristics varied between cover design and by plot early in the project and at its conclusion, as shown for biomass in Fig. 4. In 1986, there was more establishment of the seeded grasses on the integrated plots than on the conventional plots. Total biomass increased between 1986 and 1995, except for integrated plot I1. In 1986, the deeper soil of the integrated plots supported much more biomass than did the shallow soil of the conventional plots. However, this pattern did not hold true in 1995 (t test, $P > 0.05$), as integrated plot I1 had less biomass than conventional plot C1 in 1995. We also estimated LAI, an important parameter in many hydrological models. The LAI in 1995 was 3.4 for C1, 6.1 for C2, 2.1 for I1, and 5.4 for I2. A significant portion of the variation in LAI was explained by differences in biomass [LAI = 132 + 34.2 × Biomass (g m^{-2}); $r^2 = 0.74$; $P < 0.0001$].

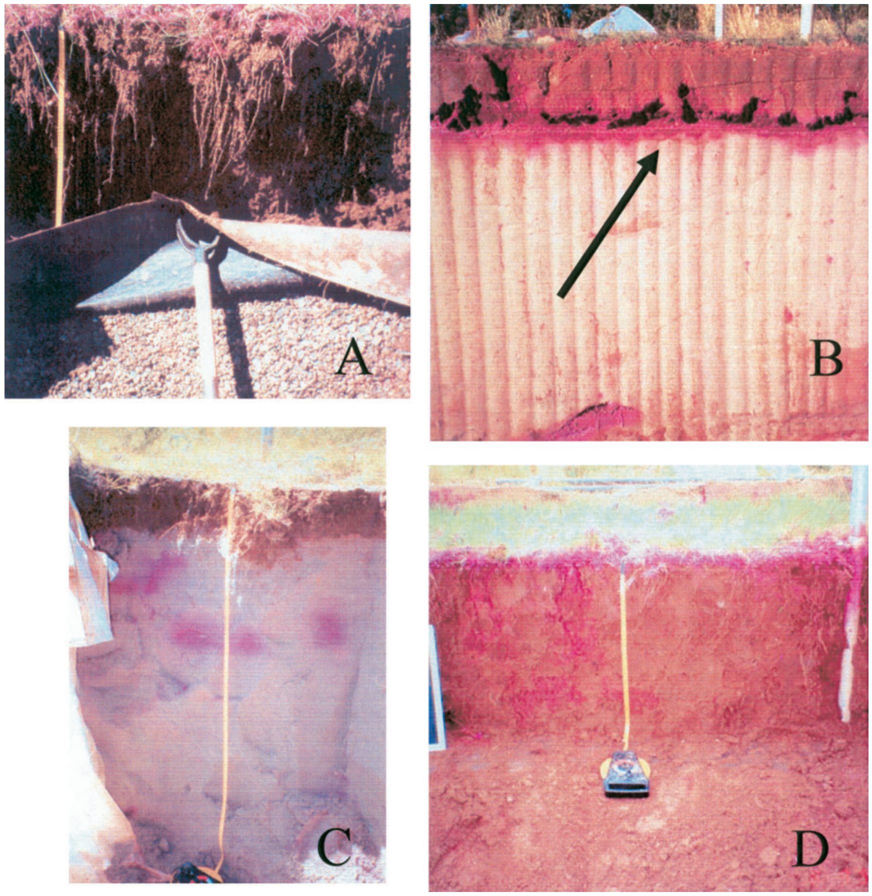

Fig. 3. (a) Geotextile of Integrated plot I1 intact with extensive rooting above in topsoil. (b) Conventional plot C1 showing the effects of pocket gophers. Gopher burrows are at the interface between the topsoil (20 cm thick) and crushed tuff (detailed in Fig. 1). (c) Conventional plot C2 showing dye-stained root channels in crushed tuff. (d) Integrated plot I1 showing dye-stained root channels in crushed tuff and preferential flow paths around access tube used to measure soil water content via neutron probe. Compare to Figure 1 for scale.

Ground cover was high on all four plots (C1 = 69%, C2 = 100%, I1 = 98%, I2 = 100%). However, species composition differed between the two cover designs. Declines in the magnitude and relative proportion of *B. gracilis* biomass occurred on the integrated plots from 1986 to 1995. In 1986, the integrated plots had a much greater proportion of *B. gracilis*, whereas the conventional plots had a greater proportion of *P. smithii* (Fig. 5). Other invading species also became established by 1995. On the integrated plots, the invading species were *Lactuca serriola* L., *Salsola kali* L., and a *Verbena* species. In contrast, invading species on the conventional plots included (in order of decreasing abundance) *Thelesperma megapotamicum* (Spreng.) Kuntze, *Ratibida columnifera* (Nutt.) Wooton & Standl., *Kochia scoparia* (L.) Schrad., *S. kali, Amaranthus graecizans* L., *Erigeron flagellaris* A. Gray, *Xanthium strumarium* L., *Medicago sativa* L., *Melilotus* species, *Chenopodium* species, *Solanum rostratum* Dunal, *Gutierrezia sarothrae* (Pursh) Britton & Rusby, *Chrysopsis villosa* (Pursh) Nutt., and *Artemisia dracunculus* L. The invading species accounted for about one third of the biomass on each of the conventional plots, but were a much smaller component of the integrated plot biomass (Fig. 5).

Rooting Patterns

Roots were found in almost all portions of the covers and decreased in abundance with depth (Fig. 6). In the integrated plots, a few fine and medium roots were observed in the gravel layer, although the root-count transects in those layers did not register any, and abun-

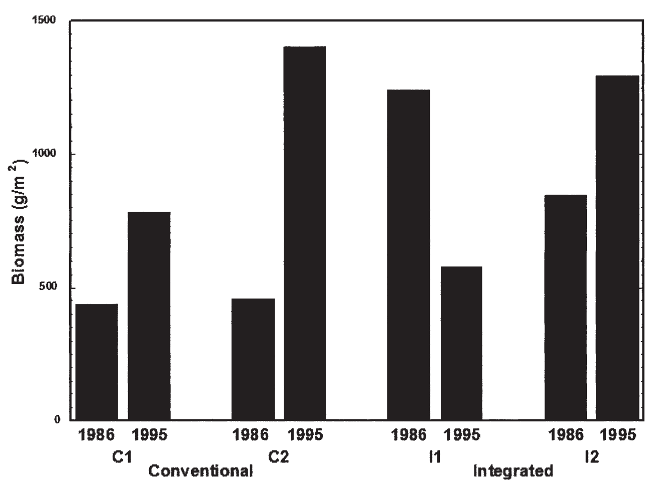

Fig. 4. Total biomass (g m^{-2}) for conventional and integrated plots in 1986 and in 1995.

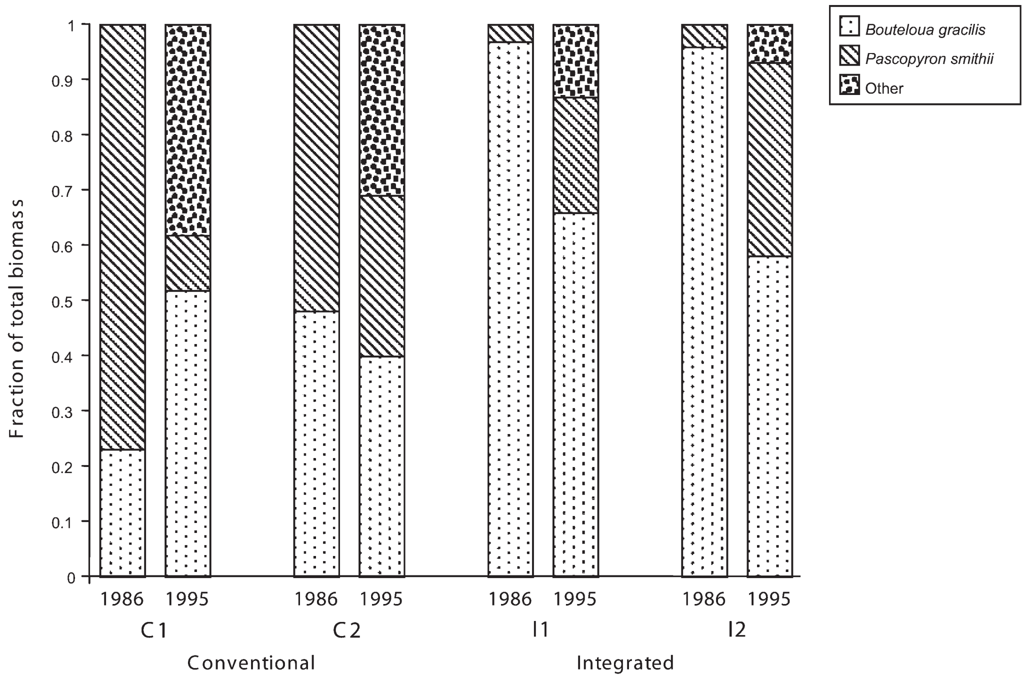

Fig. 5. ***Bouteloua gracilis*** **(blue grama),** ***Pascopyron smithii*** **(western wheatgrass, formerly** ***Agropyron smithii*****), and other species as percentages of total biomass on conventional (C) and integrated (I) plots in 1986 (reported in Nyhan et al., 1990) and in 1995.**

dant fine roots were concentrated along the geotextile separating the topsoil from the gravel layer. These roots formed a continuous root mat at the base of the topsoil and some of the fine roots penetrated pores in the geotextile. Only one instance of a larger root (2–3 mm diameter) penetrating the geotextile was observed, and that root extended into the gravel layer. Hence, the depth of root penetration for herbaceous species was generally limited by the geotextile barrier, and apparently water penetrated down to the geotextile interface sufficiently often for substantial rooting to develop above the interface.

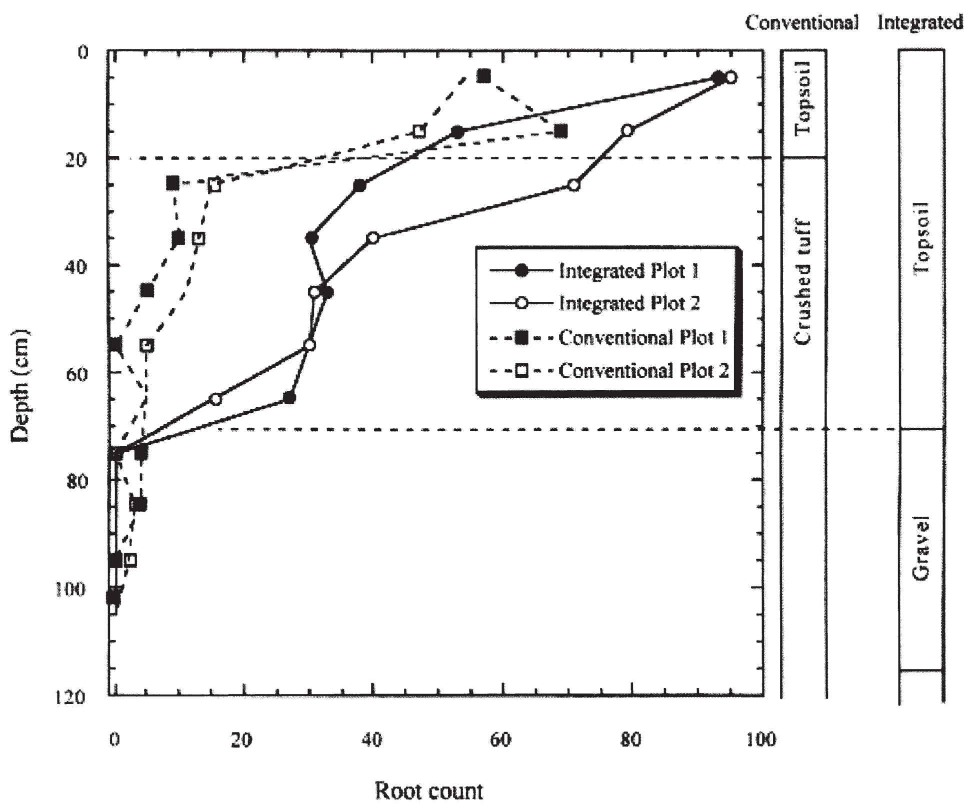

Fig. 6. Root abundance with depth in conventional and integrated plots.

Ponded Dye Infiltration and Water Penetration

During the ponded dye infiltration and water penetration tests, the depth of water above the soil surface on all plots decreased linearly with time, consistent with expected responses for macropore flow rather than nonlinear rates expected with infiltration into the bulk soil matrix alone. The changes in head for the conventional design were 5.4×10^{-2} cm s^{-1} (plot C1) and 2.5×10^{-3} cm s^{-1} (plot C2) and for the integrated design were 3.8×10^{-3} cm s^{-1} (plot I1) and 3.2×10^{-3} cm s^{-1} (plot I2). Conventional plot 1, which had a much greater infiltration rate than the other plots, contained extensive pocket gopher [*Thomomys bottae* (Eydoux & Gervais)] burrows that were observed during excavation of the plot (Fig. 3b).

There was significant staining from dye in root channels and other macropores and in the soil matrix in the topsoil of both designs. Excavation showed that much of the infiltration occurred via macropores in both the topsoil and crushed tuff layers (Fig. 3c and 3d). Matrix staining decreased significantly within 5 to 10 cm of the soil surface, while macropore staining extended through the entire thickness of the topsoil of both designs. Staining extended into the crushed tuff of the conventional design and followed root channels and animal (pocket gopher and ant) burrows, with essentially no staining of the matrix (Fig. 3c). In the integrated plots there was no staining in the gravel or cobble layers except in a few locations next to the plot walls and around neutron probe access tubes (Fig. 3d). In these cases it was clear that seepage had proceeded preferentially down the plot walls and access tubes.

DISCUSSION

Landfill Cover Performance over a Decade

Our study indicates that the integrity of both landfill-cover designs was relatively high after a decade. The water balance data indicate that both cover designs limited seepage to less than 4% of the total precipitation input for all plots. In contrast to expectations, the integrated design did not yield significantly less seepage on average than the conventional design over the >10-yr study period. This result contrasts with those reported for the first 3 yr of the study (Nyhan et al., 1990) that were used in initial calibration studies with the CREAMS model (Nyhan, 1990)—during that initial period seepage for conventional design was more than four times greater than that for the integrated design. This decrease in seepage through time on the conventional plots may be related to increases in evapotranspiration associated with the increase in biomass. In addition, other studies of landfill covers initiated after the plots we are studying were installed, indicate that slopes of greater than the 5% used in this study are required for effective generation of interflow and associated reductions in seepage (Nyhan et al., 1997). Hence, although the integrated design in this study generated limited interflow, the design did not maximize interflow.

The preferential flow patterns around the neutron access tubes may have resulted in elevated soil water content and seepage in our study. Studies can minimize the potential for preferential flow around tubes or along walls (Nyhan et al., 1997, 2001). For example, soil moisture measurement by time-domain reflectometry (TDR) rather than neutron probe likely diminishes the effect of preferential flow on soil water content. The preferential flow paths related to wall effects can be minimized by excluding areas near the wall in estimating seepage. Both of these methods have been applied in a subsequent studies that involved similar cover designs in the same environmental setting (Nyhan et al., 1997, 2001; Nyhan, 2005). The related studies reported smaller amounts of seepage than our estimate of ~3% (e.g., <1% for a 3.5-yr period, Nyhan et al., 1997, 2001; 1.7% for a 7-yr period, Nyhan, 2005), but our study was longer and may have spanned proportionally more wet periods than shorter studies. These other studies suggest that it is unlikely that our seepage estimates are off by more than 2% of the water budget. More importantly, the measurements reported by Nyhan et al. (1997, 2001) overlap with the measurements reported in this study between 1992 and 1994, during which the timing and amounts of seepage are similar across both studies. Hence, we believe that the estimates of seepage are relatively accurate and that the lack of difference in seepage between the conventional and integrated plots that we report is not the result of a measurement issue. Further, such artificial preferential flow paths would not likely be an issue for similarly designed full-scale landfill covers.

Not surprisingly, soil development was not significant more than a decade after plot installation. Although soil-forming processes are slow, particularly in semiarid and arid climates, it was important to evaluate soil changes because had the cover designs resulted in significant seepage, accelerated translocation of fine soil particles might have led to measurable changes in soil texture, with clay particles moving downward from the upper to the lower portion of the topsoil layer. This change might have been reflected in differences in field texture, bulk density, and color between the upper and lower topsoil, as well as the upper portion of the crushed tuff in the integrated plots.

Vegetation dynamics differed between the two cover designs. Biomass increased significantly after initial establishment, as expected. The thicker topsoil of the integrated plots appears to have allowed for better early establishment of the two species (*B. gracilis* and *P. smithii*) that were originally seeded (Nyhan et al., 1990). Species composition also was influenced by landfill cover design. In 1986, *B. gracilis* dominated the integrated plots, while *P. smithii* made up half or more of the biomass on the conventional plots. In 1995, the species composition of all plots had changed substantially, with other invading species accounting for about one-third of conventional-plot biomass, and a small but substantial proportion (7–13%) of integrated-plot biomass. The invading species on both plot types were limited to herbaceous plants; no woody plant establishment occurred on any of the plots. After more than a decade, then, the two designs had substantially different plant community composition, suggesting that these designs are progressing along different successional pathways. The conventional design was clearly more susceptible to establishment of invading species than was the integrated design, at least over the initial decade. This is likely related to the shallower depth of the topsoil in the conventional design, which provides less opportunity for soil moisture to be stored deeper in the profile. *Pascopyron smithii* may require a greater proportion of deep soil moisture to endure dry periods and this likely lead to the large decrease in the relative abundance of this species. It is also notable that the observed rooting patterns differed from more commonly observed and modeled patterns in which rooting depth decreases exponentially with depth (Canadell et al., 1996).

It is important to recognize the potential for burrowing animals to negate the effectiveness of landfill covers. In our case, pocket gopher burrows increased the rate of infiltration and subsequent water penetration by nearly an order of magnitude and moved a large amount of ponded water, analogous to an intense thunderstorm or rapid snowmelt, rapidly into the installed cover. Other work at this site has shown that pocket gophers can increase rates of infiltration and water penetration by 200 to 300% (Hakonson, 1998). Pocket gophers and other small mammals can displace large amounts of soil (Cox, 1990; Hakonson et al., 1982; Mielke, 1977; Spencer et al., 1985;), translocating it to the soil surface (Arthur et al., 1987; Gonzales et al., 1995; O'Farrell and Gilbert, 1975; Schuman and Whicker, 1986; Winsor and Whicker, 1980). Therefore, biointrusion barriers such as our gravel and cobble layers or gravel incorporated into the topsoil may be essential to even the short-term success of any cover design.

Our results suggest that both cover designs remained relatively intact after a decade. Based on the changes in the integrity of the cover, pathways associated with surface risks such as plant intrusion and animal intrusion may pose a greater potential for risk than subsurface risks such as those associated with contamination of groundwater by seepage. More generally, our results indicate that either design remains largely intact and effective over a decade, and if further deterioration does not occur for another subsequent 20 years, the designs may meet the requirements for the 30-year post-closure monitoring and maintenance period prescribed by RCRA (EPA, 1989). Our results also, however, provide evidence of initial degradation in cover performance within the first decade following installation that could become increasingly important with time. Although we are unable to project the longer-term effects of these changes on cover performance, we speculate, on the basis of our results and those of related studies, on how engineering vs. ecological factors may vary in importance through time.

Relating Landfill Cover Performance over a Decade to Longer Time Frames

An improved understanding of landfill cover performance over time requires studies over various time intervals. This study can be integrated with several related studies that include a conventional design in the same environmental setting to develop hypothesized relationships about changes in cover performance through time. The performance of a landfill cover is dependent on both the engineering factors that form the basis of the initial cover and the environmental factors that affect the cover through time (Fig. 7). Engineering factors are those factors that can be manipulated during the installation phase of the cover, such as soil horizonation (thickness, slope, layering, texture) and initial ground cover and vegetation cover. Here we refer to environmental processes as those factors that come into play following the installation of the cover, such as plant succession, soil profile development, erosion, and intrusion by plant roots and burrowing animals. With time, we hypothesize the relative importance of environmental factors in determining cover performance increases, while the relative importance of engineering factors should decrease, as discussed by Suter et al. (1993). The exact shape of the curves will of course depend on the cover design and local climate.

Because we have conducted numerous related studies at Los Alamos, we can highlight some time points along the hypothesized curve that are consistent with observations of a conventional design at Los Alamos (Fig. 7). Data point A in Fig. 7 corresponds to studies of initial conditions for cover performance, which were presented by Nyhan et al. (1990), demonstrating that the conventional covers remained intact over the first three years following installation. However, Nyhan et al. (1990) did note the important influence of plant community composition on water balance, as have others (e.g., Anderson et al., 1993; Nyhan et al., 1998), during this initial period. Data point B in Fig. 7 is supported by this study. Our results indicate that landfill covers largely remained intact after more than a decade. However, our results also provide evidence of landfill cover deterioration (e.g., higher infiltration due to animal burrowing) within the decade time frame. Data point C in Fig. 7 is derived from the investigation of a similar cover at Los Alamos that used a conventional design of topsoil over crushed tuff (Wenzel et al., 1987). It is notable that the topsoil was 90 cm thick in that design as opposed to 20 cm in the one we studied; nonetheless the comparison provides insights into landfill cover performance over longer time frames. The landfill had been closed for 34 years and was found to have a number of tree and shrub species growing on the cover, with some trees rooting directly into waste material. Pocket gophers had also exposed waste material, indicating penetration of the crushed tuff. Hence, the study of Wenzel et al. (1987) shows breakdown of a similar cover and dominance of environmental processes in determining landfill-cover performance in less than 35 years. This breakdown of the conventional cover is due in part to the establishment of woody plants and subsequent root intrusion (Tierney and Foxx, 1982), which have average and maximum root lengths that greatly exceed those for herbaceous plants (Canadell et al., 1996; Jackson et al., 1996). We speculate that landfill-cover performance for periods of a century

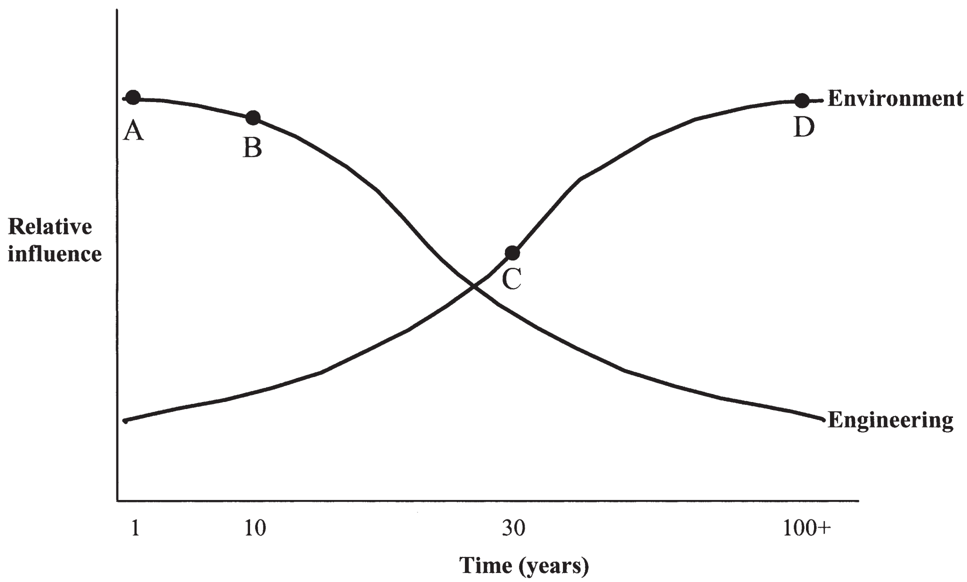

Fig. 7. Conceptual graph illustrating trends in the relative influence of engineering factors (e.g, slope, soil textures and thicknesses) vs. environmental factors (e.g., succession, burrowing) in the long-term effectiveness of engineered landfill covers. (A) Three years or less after installation (Nyhan et al., 1990). (B) Ten years after installation (this paper). (C) Thirty years after closure (Wenzel et al., 1987). (D) One century or more after closure (no studies of landfill covers; analog studies in ecosystems)

or longer, for which studies are lacking, becomes increasingly dominated by environmental processes, as highlighted by data point D in Fig. 7.

Recent studies at a woodland site at Los Alamos provide a basis for quantifying patterns and processes relevant to longer-term cover performance. In general, the proportions of herbaceous and woody plants have an important effect on spatial variability in fluxes of water and energy (Breshears et al., 1997b, 1998; Martens et al., 2000), thereby affecting seepage and other components of the water balance. Many woody evergreen species are able to remove soil water throughout much of the year, including times during which herbaceous species are senescent. For example, many of the landfill sites at Los Alamos are situated within piñon–juniper [*Pinus edulis* Engelm. and *Juniperus monosperma* (Engelm.) Sarg.] woodlands. Both *P. edulis* and *J. monosperma* transpire throughout the winter (Breshears, 1993), when a significant proportion of the annual precipitation at Los Alamos occurs and fluctuating temperatures can result in saturated soils as a result of snowmelt. The senescent herbaceous species cannot remove this excess soil water, but *P. edulis* and *J. monosperma* can help dry the topsoil during this period and prevent saturated conditions that could lead to seepage through the cover. Thus, these woody species can help minimize seepage through buried waste. Provided that an engineered barrier can prevent or minimize root intrusion, then, the presence of woody plants on a landfill cover can be used to decrease soil water storage and seepage. The relative proportions of woody and herbaceous plants are viewed as in large part being influenced by the vertical distribution of soil moisture (Breshears and Barnes, 1999; Breshears et al., 1997a; Martens et al., 1997). Consequently, the proportions of these two plant types can be optimized through varying soil textures and thicknesses in ways that determine the vertical profile of soil moisture, as well as by the relative proportions of the two plant types initially established on the cover.

Changes through time in vegetation also have important effects on runoff and erosion. Runoff in semiarid woodlands is relatively low when vegetation cover is high (Reid et al., 1999; Wilcox, 1994; Wilcox et al., 2003a, 2003c), but rapid changes in vegetation in response to climate can greatly increase erosion rates (Allen and Breshears, 1998; Davenport et al., 1998; Wilcox et al., 1996). Therefore, it is important to maintain a relatively high ground cover, particularly in the intercanopy spaces that separate woody plants (Davenport et al., 1998; Reid et al., 1999). Over longer time frames, disturbance factors such as fire and drought become more important to consider because they become more likely and can greatly accelerate wind and water erosion (Breshears and Allen, 2002; Breshears et al., 2003; Johansen et al. 2001, 2003; Whicker et al., 2002).

Our work in ecosystems has also identified another process that could impact the performance of landfill covers over longer times, particularly those using a clay liner, such as recommended by EPA (Nyhan et al., 1997). Other studies at Los Alamos have shown that tree roots penetrating clay layers can generate large amounts of interflow (Newman et al., 1998; Wilcox et al., 1997), which, if not mitigated, could be directly deposited into wastes and greatly increase associated risks (Wilcox and Breshears, 1997). In addition, on the time scales of hundreds or even thousands of years during which hazardous waste could pose a public health risk, soil development is expected to become more of a factor influencing water infiltration and erosion for covers. Collectively, these studies highlight the importance of integrating an understanding of environmental processes with engineering factors in assessing tradeoffs for different landfill-cover designs.

CONCLUSIONS

In conclusion, both conventional and integrated designs limited seepage to ~3% of precipitation input over more than a decade. The integrated plot generated a limited amount of interflow (<1% of precipitation input); a difference in seepage between the plots might have been evident if the slope of the engineered barrier was greater than 5%, based on other recent studies. There were numerous above- and below-ground changes in both designs more than a decade after installation, including: (i) biomass increased on both of the conventional plots and one of the integrated plots, (ii) species diversity had increased on all plots, (iii) integrated plots had less biomass from invading-species than conventional plots, (iv) rooting was more extensive in the integrated plots, (v) little change had occurred in the soil profiles of either plot type or in the geotextile of the integrated plot type, and (vi) infiltration occurred primarily via macropores (including root channels and animal burrows). The results that we report here indicate favorable performance of both the conventional and the integrated design over the first decade following installation and provide an important step in extending the evaluation of landfill covers from the first initial years to periods of decades. The results also highlight factors that may become increasingly important over longer periods if landfill covers are to effectively isolate contaminants and minimize human and ecological risks.

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