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Rockwell Hanford Operations

**Relationships Between Properties
of Hanford Area Soils and the
Availability of ^{134}Cs and ^{85}Sr for
Uptake by Cheatgrass and
Tumbleweed**

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ABSTRACT

The relationships between plant root absorption mechanisms and basic soil parameters which effect the concentration of Sr and Cs, and analog ions in soil solution are reviewed. Based on these relationships, studies were undertaken to determine the relative availability of Cs and Sr for uptake by plants grown on Hanford area soils, and the feasibility of developing a predictive relationship between readily measurable soil parameters and plant availability of Cs and Sr. Concentration ratios were determined for cheatgrass and tumbleweed grown on Hanford area soils representing a range in properties (Burbank, Ritzville, Licksillet, Rupert and Warden). Concentration ratios ranged from 0.0078 to 0.066 and 3.5 to 16 for ^{134}Cs and ^{85}Sr , respectively. Soils were analyzed for physical properties, mineralogy, extractable cations and extractable ^{134}Cs and ^{85}Sr . Simple correlation analyses showed Cs and Sr uptake in cheatgrass and Sr uptake by tumbleweed to be related to cation exchange capacity, and extractable Sr, Ba and Mg. However, in the case of Cs, this correlation may be coincidental since these divalent cations are not chemical analogs of Cs. Uptake of Cs by tumbleweed showed weak correlations with extractable and exchangeable K. Factor analysis and principle components analysis did not assist in further quantitation of the relationships between plant uptake and soil parameters.

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INTRODUCTION

Cesium-137 (Cs) and strontium-90 (Sr) are two high-yield fission products which represent a significant fraction of radionuclide inventories requiring long-term waste management. Currently, we have a qualitative understanding of the environmental behavior and fate of these two radionuclides. This not only results from the studies concerned with the fate of Sr and Cs contained in fallout and the potential radiological hazard to man, but also from their close chemical similarities to the analogs Ca and K. Much of what we have learned from these studies about the behavior and fate of ^{137}Cs and ^{90}Sr in the environment is applicable to waste management practices. The following represents a brief review of the behavior of Cs and Sr in soils as it relates to waste management. We have attempted to tie together available information on the behavior of Cs and Sr in soils, develop a set of readily measurable soil parameters which could be useful in predicting Cs and Sr availability in Hanford soils, and correlate these parameters with Cs and Sr uptake by cheatgrass and tumbleweed.

The fate and behavior of Cs and Sr in soil, with respect to both transport through the soil profile and availability for uptake by plants, is a function of the soil chemistry of Sr and Cs. Unlike most other trace contaminants, the chemical behavior of Sr and Cs is similar to stable analogs present in macroconcentrations in soil, namely Ca and K, respectively. Therefore, the physical and chemical processes that regulate the sorption and desorption of Ca and K may relate to the soil retention of Sr and Cs. The principle processes which regulate soil retention and thus the magnitude of the soluble component are cation exchange (on mineral surfaces and/or organic matter), precipitation and mineral replacement reactions. Each of these processes are, in turn, controlled by soil mineralogy, pH, particle size and concentrations of both the radionuclide and stable macroions.

In considering the disposal to the ground of radionuclides, it is necessary to consider: 1) the sorption reactions of waste components with Hanford sediments, and 2) composition of waste solutions disposed to the sediments. An evaluation and integration of data on the waste solutions and sediments relating to the above categories allows one to predict disposal conditions which will improve the retention of radioactive wastes and limit the availability of radionuclides for plant uptake.

SOIL SORPTION REACTIONS

When radioactive waste solutions are percolated through sediments, radionuclides react with, and are retained to a large extent by, the sediments. The sorption of radionuclides can be divided into five categories of reactions: 1) ion exchange, 2) precipitation, 3) replacement, 4) other sorption reactions, and 5) behavior of neutral and anionic species. The relative importance of the above reactions to the sorption of any given waste solution, in low organic matter soil as those at Hanford, is a function of both the waste composition and mineralogical composition of the sediments. For example, in disposal of a waste containing both rare earth and cationic components to sediments having both cation exchange capacity (CEC) and free CaCO_3 ion exchange may be the principal sorption reaction for the cationic component, while an increase in pH due to the reaction with CaCO_3 may cause the precipitation of the rare earths due to hydrolysis. Thus, it is difficult to generalize as to what sorption reaction is the most important with reference to any given waste or component of that waste.

Cation Exchange

Many solids, when placed in contact with solution, exchange ions with similar ionic species in solution. This mechanism is known as ion exchange. In ion exchange, the solid (exchanger) phase remains basically unchanged during the exchange reaction, and exchange is stoichiometric. Ion exchange can be divided into cation exchange and anion exchange

depending on whether cations or anions are exchanged with the solution phase. In the Hanford sediments, cation exchange is the major sorption reaction, while anion exchange is minor to nonexistent (McHenry, 1958; Bensen et al, 1963; and Hajek, 1965). Layer lattice silicates and zeolites of the silt and clay size fractions of sediments have relatively high cation exchange capacities because of: 1) the isomorphous substitution of lower valence for higher valence cations, 2) broken bonds at surfaces and particle edges, and 3) dissociation of structural OH^- groups (Carroll, 1957; Garrels and Christ, 1965). Much of the exchange capacity of layer lattice silicates and zeolites is due to isomorphous substitution with a lesser portion due to the other two mechanisms (Schofield, 1949; Ames, 1962, 1964, 1965, 1966). The CEC resulting from isomorphous substitution is constant at all pH values, while the CEC resulting from dissociation of $-\text{OH}$ is pH dependent. The higher the pH, the greater is the cation exchange capacity due to dissociation of H^+ from $-\text{OH}$ groups.

Anion exchange in low organic matter sediments is due primarily to the protonation of free iron and aluminum hydrous oxides at a pH below 7.0 (Schofield, 1949). The sediments of the Hanford project are generally basic ($\text{pH} > 7.0$) in reaction (McHenry, 1957). Thus, the sediments of the Hanford project would be expected to have limited anion exchange capacities as was found by Hajek (1965).

Precipitation

When the solubility product of any compound is exceeded in a solution, the compound precipitates. The rare-earth's and transition metal's hydroxides and hydrous oxides are relatively insoluble, and appreciable ionic concentrations of these elements can be maintained only in acid solutions. As a general example, consider the reaction:

If the above hydroxide is extremely insoluble, the reaction will tend to be displaced to the right side of the equation, and any reaction that

tends to make the solution more basic (remove H^+ ions) will drive the reaction further to the right. When an acid waste solution enters a calcareous soil, the free $CaCO_3$ in the soil tends to neutralize the waste, and precipitation of radionuclides which form insoluble hydroxides will occur (Kraus and Nelson, 1950; Bensen, 1960).

Replacement

Replacement is a special case of precipitation. In a replacement reaction, at least one of the components of the newly precipitated phase is supplied by a previous solid phase, and the newly precipitated phase precipitates (replaces) in place of the previous solid phase. The solubility product of the new phase would maintain the concentration of the component supplied by the previous solid phase at a lower concentration in the replacing solution than would the previous solid phase. Thus, the new solid phase is more stable in the replacing solution than the original solid phase (Ames, 1960, 1961, 1962, 1963).

Other Sorption Reactions

Sorption reactions are generally divided into chemical and physical sorption. These are usually differentiated on the basis of whether the sorption increases (chemical sorption) or decreases (physical sorption) with increasing temperature. Some research workers have empirically measured sorption reactions that would seem to be neither an ion exchange nor a precipitation reaction, but the exact sorption mechanism was not determined (Tamura, 1962, 1963a, 1963b, 1964; Tamura and Struxness 1963). These other sorption reactions are best considered as a separate category of reactions.

Behavior of Neutral and Anionic Species

Some radionuclides fail to react with sediments to any significant degree and percolate to the groundwater along with the waste solution. When a nonreacting, long-lived radionuclide is present in a waste at a concentration greater than 10% of MPC, the waste is disposed on a specific retention basis. Such radionuclides are present in wastes as an uncharged complex, anionic complexes, or as an anion (Haney, 1957; Hajek, 1965; Rhodes and Nelson, 1957; Hajek and Knoll, 1966).

STRONTIUM-90 SORPTION

Strontium is an alkaline earth metal which usually exists in solution as a divalent cation. Strontium-90 is generally considered to be sorbed on sediments by cation exchange reactions (McHenry, 1955; Klechkovsky, 1957; Schultz et al., 1958). Certain experimental findings, however, suggest that ion exchange may not be the sole sorption reaction of Sr.

McHenry (1955) found an extreme pH dependence of Sr uptake in a calcareous Hanford sediment, and Sr sorption was enhanced by the addition of PO_4^{--} to waste solutions in laboratory studies. This pH dependence was greater than would be expected from a simple ion exchange reaction. Ames and Hajek (1966a, 1966b, 1966c) and Hajek and Ames (1968) in the statistical experiments found a large Sr-sorption dependence upon the presence of calcite in calcareous Hanford sediments. Tamura (1962, 1963a, 1963b, 1964) and Tamura and Struxness (1960, 1963) found selective sorption of ^{90}Sr on low exchange capacity Fe and Al oxides (hematite, limonite, corundum). Tamura also found that the Sr-sorption capacity of some soils was decreased due to the removal of free hydrous Fe and Al oxides. All of the above findings seem to indicate the Sr can be sorbed by reactions other than cation exchange.

The elucidation of the increased sorption of ^{90}Sr from wastes in which phosphate is added to the waste solutions was made by Ames (1959, 1960, 1961). In laboratory studies, it was found that solutions containing phosphate replaced calcite (CaCO_3) in the Hanford sediment samples with the more stable mineral apatite at moderate and high pH. Apatite, $\text{Ca}(\text{F}, \text{Cl}, \text{OH})(\text{PO}_4)_3$, is a calcium phosphate; and Sr, which is chemically analogous to and a size similar to Ca, was found to substitute for Ca in the apatite structure.

Increased sorption of Sr in calcareous soils in the absence of phosphate is more difficult to explain because of some, seemingly, conflicting evidence. For example, it was found that the CaCO_3 neutralization of wastes decreased the sorption of Sr relative to NaOH neutralization (Brandt and Reisenauer, 1958). In addition, Spitsyn et al.

(1963) reported that Ca competed with Sr for ion exchange sites which had the effect of reducing Sr sorption. Thus, increased solute Ca is not the cause of increased sorption, but actually decreases Sr sorption by cation exchange. Increased sorption then must be due to the presence of solid phase calcite.

Halevy and Tzur (1964) reported the precipitation of SrCO_3 , by CaCO_3 in cation exchange studies. This precipitation (replacement) of CaCO_3 by SrCO_3 may be the key to the understanding of increased Sr sorption. The increased sorption reported by Ames and Hajek (1966c) was found in an open flow system (through small soil columns) with a concentrated Na^+ influent. In this open system, the concentrated Na^+ probably maintained the solution Ca^{2+} concentration at a low level, thus preventing a strong Ca^{2+} competition with Sr for ion exchange sites, and a SrCO_3 phase may have replaced CaCO_3 on the surface of calcite. Thus, replacement would account for the increased sorption. In batch studies (closed system) with the same sediment, decreased sorption was found, probably due to the overwhelming effect of Ca^{2+} competition with Sr^{2+} for ion exchange sites. The above reaction also explains the strong pH dependence of Sr sorption in a calcareous soil since a low pH would result in the dissolution of CaCO_3 (McHenry, 1955, 1958).

Precipitation processes can be induced by the presence of soil solution excesses of calcium, phosphate (Rhodes, 1957; Kunishi and Taylor, 1972) oxalate (McHenry, 1958), and sulfates (Menzel, 1960). Mokady and Gal (1964) have shown strontium (0.01 N SrCl_2) passed into soils containing CaCO_3 to be retained by replacement of Ca and by Sr. This type of soil fixation process should play a major role in limiting both the transport of Sr in soils and its bioavailability to plants in calcareous soils.

Bensen (1960) in review of the soil chemistry of calcareous Hanford soils describes the specific behavior of strontium with respect to soil retention under various conditions. The percentage retention of Sr on soil was both concentration-dependent and pH dependent. Retention generally increases logarithmically with pH between 1 to 12 with strontium

retention by soil near maximum (>80%) at pH 8. At relatively high strontium concentrations (0.01 to 0.1 $\underline{S}$; $\underline{S} \equiv$ cation concentration equivalent to CEC) the percentage of Sr sorbed decreased with increasing Sr concentration. At constant pH (7-8) and strontium concentration equivalent to 1% of CEC, equivalent concentrations of La, Ba, Cs, NH_4 , K, Na, and Li had only negligible effects on strontium retention. However, at strontium and accompanying ion concentration of 10% of CEC, retention of strontium was depressed in the order $\text{La} \gg \text{Ba} > \text{Cs} \gg \text{NH}_4 = \text{K} > \text{Na} > \text{Li}$ (McHenry, 1955). Calcium was found to be more effective in displacing Sr than other divalent ($\text{Ca} > \text{Ba} > \text{Cd} > \text{Mg}$) and monovalent cations studied. The anions, chloride and sulfate, had no effect at 10 times the strontium concentration (0.1 and 0.01 $\underline{S}$). Nitrate at equivalent Sr concentrations decreased Sr retention while oxalate and phosphate ions increased Sr to nearly 100% of that added, but only at 0.01 $\underline{S}$ Sr. The inhibition in Sr retention induced by accompanying cations as Na (4 $\underline{M}$) is reversed by PO_4^{3-} (0.01 $\underline{M}$). At pH 8, the K_d of Sr in the presence of Na^+ is increased to ~ 80 by the addition of 0.01 $\underline{M}$ PO_4^{3-} , while at pH 9.0, K_d increases from 7 to 250.

CESIUM-137 SORPTION

Cesium is an alkali metal element which usually exists in solution as a monovalent cation. Cesium is certainly sorbed by ion exchange; however, it is also sorbed by a more selective reaction when present in trace concentrations (Jacobs and Tamura, 1960; Tamura and Jacobs, 1960, 1961; Schultz et al., 1960; Sawhney, 1964; Gaudette et al., 1966). It was demonstrated by Tamura and Jacobs (1960) that mica-like minerals (illite) selectively-fixed Cs^+ . The above authors further demonstrated that when expandable layer silicates were collapsed to a 10 Å mica spacing by K^+ saturation and/or heating, an increasing Cs selectivity by the collapsed minerals resulted (Jacobs and Tamura, 1960; Tamura, 1963a; Tamura and Jacobs, 1960, 1961). It was suggested that the increased Cs sorption is due to the creation of a small number of selective sites around the edges of the collapsed mica-like structure.

Cesium in these selective sites is usually considered to be fixed and not to conform to normal ion exchange equilibria treatments, because the

process is assumed to be irreversible (Spitsyn et al., 1963). This may not be the case. The reaction may still be reversible, but has different reaction rates depending upon the direction of equilibrium attainment. Equilibrium attainment from a previously Cs-saturated system would be expected to be much slower than would equilibrium attainment from a previously Cs-unsaturated system. This would be true because the previously Cs-saturated system would be collapsed which would sterically hinder the exchange. Thus, the Cs-selective reaction may only be a special case of an ion exchange reaction at selective Cs sites.

In studies with Hanford soils, Cs sorption, in the absence of other salts, has been shown to increase with pH (2-12) and decrease with Cs concentration over a range of 0.01 $\mu\text{Ci/ml}$ to 1.0 $\underline{\text{S}}$ (Bensen, 1960). Added salts tended to depress Cs sorption with a corresponding lack of pH effect (McHenry, 1954). The ability of macroions to depress Cs sorption in equilibrium systems decreased in the order: $\text{H} > \text{La} \geq \text{Sr} > \text{Na} > \text{NH}_4$ when Cs and macroions concentrations were each 0.01 $\underline{\text{S}}$ and 0.5 $\underline{\text{S}}$. Displacement of exchangeable cesium from soils with 0.1 N solutions is in the order, $\text{K}^+ \geq \text{La}^{3+} \geq \text{Na}^+ > \text{Sr}^{2+} > \text{NH}^+$. The apparent differences in the interaction of specific ions in the sorption and desorption of Cs appears to be related to valence and also to atomic radius, with the latter being more important in the desorption of Cs associated with the unit layers (other than external surfaces) of mica-like clay minerals.

FACTORS INFLUENCING PLANT UPTAKE OF CESIUM AND STRONTIUM

Uptake of ions, both nutrient (K, Ca) and nonnutrient (Cs, Sr) species by plants is mediated by metabolic transport processes. The rate at which this process proceeds is dependent on the concentration of the specific element in soil solution, its rate of equilibration from soil solid phases, and the presence of competing or analog elements. Therefore, when considering the availability of a given radionuclide for plant uptake, one needs to be aware of its concentration in soil solution and the factors which influence the equilibrium concentration such as pH, organic matter, soil amendments, and overall mineralogy as described earlier.

Soil pH (4-3) has been shown to have pronounced effect on soil retention of Sr and to a lesser extent, on Cs (Juo and Barber, 1970; Rhodes, 1957). However, there is little direct information as to the effect of pH on plant uptake for a given soil over a range of pH. Available data for a range of soils from Finland indicated that Sr uptake by timothy decreased with increasing pH (Lakanen and Paasikallio, 1968). Similarly, Jones and Haghiri (1962) found that increasing the pH of agricultural soils from 5 to 7 results in a 50% reduction in Sr uptake by agricultural plants. From a waste management standpoint, localized changes in soil pH resulting from liquid wastes can be expected to have an effect on plant availability of both Sr and Cs. The magnitude of the effect will depend on pH, radioisotope concentration (Cs or Sr), and the concentrations of macroions (K , Ca^{2+} , Na^+ , PO_4^{3-}), with availability being inversely related to soil sorption processes.

Soil organic matter, although not particularly relevant to the younger, low organic matter Hanford soils, is important in principle since large quantities of organic compounds are employed and disposed of in the processing of nuclear materials. Uptake of ^{137}Cs by ryegrass was shown to be highly correlated with soil organic matter (correlation coefficient, $r = 0.94$) and CEC or organic matter present ($r = 0.95$) by Barber (1964). Similarly, Fredriksson et al. (1969) showed ^{137}Cs uptake by barley and peas to increase with decreasing clay content and increasing soil organic matter in 12 Swedish soils. Barber and Mitchell (1964) reported an increased uptake of both ^{137}Cs and ^{90}Sr by pasture grasses with increasing levels of soil organic matter. Juo and Barber (1969, 1970) have investigated the sorption of Sr by soil organic matter. They found that Sr tended to form insoluble coordination linkages with phenolic hydroxyl, ketonic carboxyl, and carboxyl groups. These reactions appear to be formed under slightly basic and high salt conditions. The relative affinity of ion exchange mechanisms in soil (clay minerals), chelation mechanisms (with organic compounds), and their relative equilibration dynamics may increase availability of both Cs and Sr for plant uptake in

high organic matter soils. The question arises as to whether the presence of synthetic chelates can have an appreciable effect on Cs and Sr uptake by plants on high pH calcareous soils such as are found at Hanford. L'Annunziata and Fuller (1968) have demonstrated using Sr-complexed chelates that soil sorption is greatly reduced on a calcareous soil, therefore, plant availability should be increased. Essington et al. (1962) failed to significantly increase the uptake of Sr and Cs by bushbean following the addition of several synthetic chelates to a calcareous soil. However, since these chelates (Na or H form) were amended to soils already contaminated with radionuclides, it is possible that the kinetics of the system did not allow for formation of significant quantities of Sr- or Cs-chelate.

Much of the experimental work done on Sr and Cs relate either to the effect of soil amendments on plant availability or the behavior of Sr and Cs with respect to major stable soil analogs, Ca and K, respectively. The availability of radionuclides to plants grown in contaminated soils will be dependent on the extent and strength of sorption to soil and its affinity for plant uptake and transport processes. Both of these processes are subject to interactions caused by ionic composition of endogenous soil solution and ions entering the system, particularly in waste process situations. Several studies have investigated the affinity of plant uptake and transport processes for Cs and Sr. Bange and Overstreet (1960) found ^{137}Cs uptake by excised barley roots to be concentration-dependent over a range of 0.01 to 75 meq/l. Cs uptake by plant roots is reduced by the presence of chemical analogs over a broad range. Handley and Overstreet (1961) showed uptake of carrier-free ^{137}Cs (1×10^{-4} meq/l) to be reduced 50% by 0.01 and 0.03 meq/l of K or Rb and NH_4 , respectively. Lithium and sodium exhibited a similar inhibitory effect at concentrations of 1 and 5 meq/l, respectively. Subsequent studies by Jackson et al. (1966) demonstrated a similar inhibition in uptake by K, Rb, and NH_4 , and the concentration of Cs in root and shoots to be dependent both on pH and Ca concentration of the uptake solution. The

behavior of Sr and Cs uptake by corn was investigated by Hutchin and Vaughan (1967). They showed Ca to have a marked effect on Sr uptake (0.05 to 5 mM) at Ca concentrations above 0.05 mM with 50% reduction in Sr uptake rate at equimolar concentration of Sr and Ca (2.5 mM). Transport rates for both Ca and Sr were comparable over a broad concentration range, with no apparent discrimination against Sr in favor of Ca. Radionuclide concentrations in soil can potentially be quite variable. Routson (1975) evaluated the effect of Sr and Cs concentration on uptake by tumbleweed from a Hanford soil. He found tissue concentrations of ^{90}Sr and ^{137}Cs to be constant over a broad range in soil concentration (8.5×10^{-4} to 1.6×10^2 and 3.9×10^{-2} to 1.4×10^2 $\mu\text{Ci/g}$ soil, respectively). The lack of a concentration effect on plant uptake, which is a functional respect of plant uptake processes, was believed to result from insignificant soil concentration of amended Sr and Cs compared to the indigenous extractable Sr and Cs macroion analogs (Ca and K, respectively). Thus, with constant radioactive addition, the soil-amended radionuclides of Cs and Sr are tracing large endogenous elemental pools with respect to plant uptake.

Many soil studies have been undertaken to determine the effect of soil amendments on Cs and Sr uptake by plants. Increased strontium uptake by plants is well correlated with decreasing exchangeable soil Ca (Anderson, 1967; Fredriksson et al., 1970), as is evident from nutrient culture experiments. Similarly, Cs uptake from soils has been shown to be inversely proportioned to exchangeable soil K (Fredriksson, 1970; Fredriksson et al., 1966). The use of soil amendments have been shown to affect Sr and Cs uptake by plants. Liming has been shown to be of marginal value in reducing Sr uptake from soils of moderate to high Ca content (Ellis et al., 1969). Nitrogen additions to soils have been shown to increase the uptake of both Cs and Sr (Jackson et al., 1968; Anderson and Jackson, 1972). Anderson (1971) showed 1 g P/pot (22 kg soil) to increase total uptake of ^{90}Sr ; this resulted from an increase in dry matter production since plant tissue concentrations were unaffected. At P concentrations above 1 g/pot, dry matter production was not increased

further, but Sr uptake and concentration decreased, presumably as a result of Sr and Cs precipitation as phosphates. Addition of N to pots (1 to 8 g/pot) resulted in increased Sr uptake by plants; addition of P to N amended soils, yielded maximum values at 1 g P/pot.

The above studies briefly describe the relationship between plant root absorption mechanisms and basic soil parameters which affect and control the concentration of Sr and Cs and analog ions in soil solution.

However, in addition to soil parameters such as pH, soil organic matter and macroion concentrations, the fate of radionuclides contained in wastes is very dependent on soil mineralogy and the equilibrium kinetics of the individual radionuclides and their functional analogs.

The following study was undertaken to determine the comparative uptake of strontium and cesium by tumbleweed and cheatgrass grown on five Hanford soils (Rupert, Burbank, Warden, Ritzville and Licksillet). Relevant soil characteristics were determined for each of the soils to determine the feasibility of developing a predictive model for the uptake of specific radionuclides by plants indigenous to the Hanford Reservation based on physicochemical properties of soils.

METHODS AND MATERIALS

PLANT MATERIAL AND CULTURE

Seeds of the tumbleweed (Salsola kali) and cheatgrass (Bromus tectorum) were planted in quart cartons (0.946 liters) containing 1 kg of soil. The surface of the soil was covered with 2 cm (160 g) of coarse, distilled water-washed sand to reduce water evaporation and prevent airborne suspension of dry surface soil. Four replicates of each treatment (amended with ^{134}Cs and ^{85}Sr) and controls were prepared and placed in a controlled environment growth chamber with a 27°C day and 15°C night temperature, 14 hour photoperiod and ambient humidity. Seedlings were thinned to three and four plants per pot for the tumbleweed and cheatgrass, respectively. Plants were irrigated three times a week with distilled water to maintain soil moisture near 13, 13, 23, 21 and 29% by weight on an oven dry basis for the Rupert, Burbank, Warden, Ritzville and Licksillet soils, respectively; nitrogen (NH_4NO_3) was supplied during the growing period, total amount of N added was 89 $\mu\text{g/g}$ soil.

Above-ground plant material was harvested after 90 days by pouring off the coarse sand and cutting the plants at the soil surface. Plant tissues were dried in a forced draft oven at 60°C for 24 hr. The dried tissue was weighed in 250 ml beakers after cooling to room temperature in a desiccator. The samples were ground, and weighed subsamples were counted on a 400-channel analyzer with a 4-in. NaI well crystal detector to obtain the radioactivity per gram of plant material. Concentration ratios (CR values) were calculated for shoots: $\text{CR} = \text{nCi/g dry wt shoot} \div \text{nCi/g dry wt soil}$.

SOIL AMENDMENT AND CHARACTERIZATION

The five soils employed in this study represent the range of soils and soil properties found on the Hanford Project (Rupert, Burbank, Warden, Ritzville, and Licksillet), with only Rupert and Burbank being indigenous to waste management areas. Soils were amended with ^{134}Cs and

^{85}Sr as described previously (Routson, 1975). In the five soils, final concentrations of ^{134}Cs and ^{85}Sr ranged from 3.11 to 6.30 and 0.165 to 4.03 nCi/g soil, respectively.

Soils were characterized and physical-chemical parameters determined. Soil pH was determined potentiometrically using a glass electrode, cation exchange capacity determined by the method of Routson et al. (1973), particle size distribution determined by standard methods (Bouyoucos, 1926), and total cations determined by atomic absorption spectrometry. Percentage of N and C in soils was determined by combustion and modified Kjeldahl, respectively, on <2 mm size fraction. The rate of equilibration of ^{85}Sr and ^{134}Cs , and stable cations was determined for Burbank and Rupert using solutions containing 0.01 M calcium chloride, 1 M ammonium acetate, 0.2 M ammonium oxalate, and distilled water (soil to solution ration = 1:10). Subsamples were removed at various time intervals, centrifuged at 20,000 x g for 10 minutes and filtered through 0.01 μ membrane filter. Cesium-134 and strontium-85 content in the equilibration solution was determined by gamma spectrometry: soils not amended with ^{134}Cs and ^{85}Sr were incubated with distilled water, 0.01 M calcium chloride, 1.0 M ammonium acetate and 0.2 M ammonium oxalate for 1 to 10 days, centrifuged at 20,000 x g and filtered through 0.45 μ membrane filter. Extractable soil cations were determined by induction coupled argon-plasma spectroscopy.

RESULTS

Table 1 gives CR values for the uptake of ^{134}Cs and ^{85}Sr by cheatgrass and tumbleweed grown on five Hanford Project soils. CR values for cheatgrass ranged from 0.015 to 0.064 and 3.5 to 12.0 for ^{134}Cs and ^{85}Sr , respectively. Highest CR values were obtained with plants grown on Rupert and Burbank. Interestingly, the order of availability of both ^{134}Cs and ^{85}Sr is similar for cheatgrass with Rupert = Burbank > Ritzville > Warden > Lickskillet. In contrast, although CR values for tumbleweed exhibit a comparable range (0.0078 to 0.066 and 4.4 to 16.0 for ^{134}Cs and ^{85}Sr respectively), the order of availability with respect to soil type differs from the response seen for cheatgrass.

TABLE 1. Plant Uptake Data for ^{134}Cs and ^{85}Sr Amended to Five Hanford Area Soils^a

	<u>Cheatgrass</u>		<u>Tumbleweed</u>	
	^{134}Cs	^{85}Sr	^{134}Cs	^{85}Sr
	-----		CR ^{b,c}	-----
Rupert	0.064 ± 0.002	12 ± 0.6	0.014 ± 0.001	16 ± 0.3
Burbank	0.059 ± 0.004	12 ± 0.2	0.024 ± 0.003	8.7 ± 0.4
Warden	0.018 ± 0.002	5.8 ± 0.5	0.027 ± 0.007	10 ± 0.2[2]
Ritzville	0.031 ± 0.004	7.4 ± 0.4	0.066 ± 0.021[2]	6.1 ± 0.2[3]
Lickskillet	0.015 ± 0.002[3]	3.5 ± 0.2[3]	0.0078 ± 0.0033[3]	4.4 ± 0.6[3]

^aSoil amendment levels were 3.11 to 6.30 nCi/g and 0.165 to 4.03 nCi/g for ^{134}Cs and ^{85}Sr , respectively.

^bConcentration ratio = nCi/g dry wt shoot/ nCi/g dry wt soil.

^cEach value represents $\bar{x} \pm \text{SD}$ for 4 replicate samples, except where noted in brackets.

In an effort to use readily measurable soil parameters to predict the extent of plant availability, physical and chemical properties of soils were determined. Tables 2 and 3 list the chemical and physical properties of the bulk soils. These soils range in pH from 5.7 to 7.0 and, with the exception of Lickskillet (1.8%), have comparable organic C content (0.42 - 0.63%). They vary only slightly in clay content (4.0 to 11.1%). The major parameters which differentiate soils indigenous to waste management areas (Burbank and Rupert) from other Hanford soils are their lower cation exchange capacity, and their lower silt content. An analysis of those soil cations (total) known to interact with soil and plant processes with respect to Cs and Sr transport and plant uptake is given in Table 3. Although there are minor differences in macro-elemental content (K, Ca, Mg) among the soils studied there does not appear to be any consistent trends which would set individual soils apart. The micro-elements (Cs, Rb, Sr and Ba) exhibit more variability between soils; correlations with ^{134}Cs and ^{85}Sr uptake by plants is discussed later.

Since plant uptake processes are dependent on the soil solution or available concentration of a specific element and the concentration of chemically similar elements (analogues), soils were extracted under various conditions to determine the available or soluble component of specific elements.

Analyses were made of extractable ^{134}Cs and ^{85}Sr amended soils, and 29 endogenous soil elements, using a number of extraction systems. Unamended subsamples of Rupert, Burbank, Warden, Ritzville and Lickskillet were extracted with distilled water, 0.01 M calcium chloride, 1 M ammonium acetate, and 0.2 M ammonium oxalate. Soils were extracted for 1, 2, 4, 7 and 10 days. Data generated using ammonium acetate and ammonium oxalate are not presented due to interference of the extractant with the analytical procedure employed for the analysis of cations. Table 4 and 5 give extractable cations for soils extracted with distilled water and 0.01 M CaCl_2 , respectively. Results obtained with water as the extractant are questionable since high concentrations of extractable Ti, Fe, and Al indicate dispersion of soil minerals of colloidal size that were not removed by centrifugation and filtration procedures. Mg, Sr, Ca, Ba, Zn, Na, Mn and K appear to have reasonable distributions.

TABLE 2. Properties of Hanford Project Soils

<u>Soil</u>	<u>Textural Class</u>	<u>Silt %</u>	<u>Clay %</u>	<u>Sand %</u>	<u>CEC</u>		<u>Organic Carbon %</u>
					<u>Meq/ 100 g</u>	<u>pH</u>	
Rupert	Loamy Sand	8.5	7.4	84.1	7.1	6.8	0.42
Burbank	Loamy Sand	17.8	4.0	78.2	5.5	7.0	0.52
Warden	Silt Loam	57.6	8.5	33.9	13.4	5.7	0.63
Ritzville	Silt Loam	55.0	9.8	35.2	14.4	6.2	0.54
Lickskillet	Loam	4.15	11.1	47.4	23.7	6.3	1.8

TABLE 3. Elemental Analysis of Hanford Project Soils

<u>Element</u>	<u>Soils</u>				
	<u>Rupert</u>	<u>Burbank</u>	<u>Warden</u>	<u>Ritzville</u>	<u>Lickskillet</u>
	----- ppm -----				
K	15,200	17,200	14,200	16,300	15,400
Cs	< 0.1	7.7	1.8	1.8	0.70
Rb	57	111	112	112	32
Sr	697	698	386	264	164
Ba	1,010	1,420	753	488	277
Ca	26,300	25,300	13,300	18,300	24,600
Mg	13,100	14,200	11,200	11,200	14,400

TABLE 4. Extraction of Soil Cations from Hanford Soils Using
Distilled Water ^a

		Mn	Cr	Zr	Ba	Zn	Ti	Fe	Al	Ca	Sr	Mo	P	Hg	Na	K	Cu	Si
		<0.10 ^b	<0.01	<0.01	<0.10	<0.01	<0.05	<0.01	<0.1	<0.005	<0.001	<0.1	<0.1	<0.01	<0.01	<0.01	<0.01	<0.5
	Day	(µg/ml)																
Rupert	1	0.26	0.021	0.025	0.16	0.030	0.22	8.4	5.6	3.3	0.014	0.14	1.2	2.1	4.4	10.3	0.019	22.2
	2	0.29	0.029	0.038	0.18	0.051	0.32	12.7	8.7	5.1	0.023	0.15	1.1	3.2	4.2	12.7	0.031	31.4
	4	0.39	0.044	0.055	0.23	0.055	0.61	22.9	16.7	7.2	0.031	0.16	1.2	5.1	4.0	15.4	0.020	57.8
	7	0.34	0.039	0.047	0.18	0.061	0.60	19.3	14.0	7.4	0.031	0.13	1.1	4.7	3.0	15.2	0.025	49.9
	10	0.26	0.035	0.041	0.18	0.052	0.49	17.8	12.7	8.3	0.035	0.09	1.1	4.7	3.3	16.3	0.022	48.0
Burbank	1	0.72	0.021	0.042	0.15	0.030	0.13	4.7	3.6	6.3	0.027	0.42	0.73	1.8	3.0	7.8	0.034	13.4
	2	0.28	0.015	0.024	0.14	0.037	0.14	5.0	3.7	9.6	0.038	0.15	0.77	2.3	3.2	8.9	0.023	15.9
	4	0.44	0.021	0.033	0.16	0.062	0.22	7.5	5.9	10.9	0.044	0.21	0.75	2.9	3.1	10.6	0.035	22.1
	7	0.29	0.014	0.029	0.15	0.029	0.17	5.3	4.1	13.2	0.052	0.13	0.63	2.9	3.4	11.2	0.022	20.0
	10	0.31	0.017	0.028	0.13	0.039	0.19	5.3	4.1	11.8	0.048	0.16	0.68	2.8	2.7	11.0	0.025	19.9
Warden	1	0.23	0.022	0.025	0.15	0.29	0.23	10.5	7.4	5.6	0.023	0.11	1.1	3.1	4.2	4.0	0.032	28.8
	2	0.20	0.031	0.033	0.16	0.17	0.33	15.1	11.2	6.3	0.030	0.08	1.1	4.1	4.2	5.2	0.041	41.0
	4	0.41	0.041	0.040	0.17	0.11	0.42	19.1	14.6	7.9	0.038	0.17	1.4	5.2	4.1	5.5	0.038	54.0
	7	0.30	0.030	0.031	0.20	0.10	0.41	18.5	14.3	8.5	0.039	0.07	1.3	5.3	4.5	5.5	0.030	55.0
	10	0.29	0.036	0.033	0.17	0.091	0.40	17.6	13.4	8.3	0.039	0.10	1.1	5.2	3.7	5.1	0.030	54.2
Ritzville	1	0.24	0.015	0.020	0.15	0.18	0.14	6.1	4.4	10.4	0.040	0.08	0.68	3.5	2.6	6.4	0.024	20.1
	2	0.19	0.008	0.011	0.12	0.045	0.08	3.6	2.5	11.0	0.043	0.05	0.57	3.4	2.9	5.5	0.025	15.5
	4	0.19	0.006	0.016	0.15	0.039	0.06	1.8	1.3	20.8	0.071	0.06	0.63	5.0	3.8	8.1	0.037	13.6
	7	0.26	0.012	0.018	0.14	0.13	0.14	5.0	3.6	14.7	0.056	0.08	0.60	4.4	2.7	7.6	0.023	21.5
	10	0.21	0.016	0.023	0.14	0.082	0.17	6.3	4.6	13.5	0.054	0.11	0.61	4.3	2.4	7.2	0.024	25.4
Licksillet	1	0.19	0.015	0.027	0.13	0.073	0.15	5.8	4.6	9.5	0.035	0.09	0.60	3.6	3.2	10.6	0.026	18.1
	2	0.15	0.018	0.022	0.13	0.11	0.21	8.3	6.7	11.9	0.048	0.09	0.65	4.7	2.7	12.0	0.020	23.8
	4	0.19	0.022	0.025	0.13	0.12	0.24	9.4	7.5	13.4	0.049	0.09	0.66	5.3	2.5	12.6	0.026	27.1
	7	0.16	0.021	0.027	0.13	0.07	0.26	9.7	7.8	14.6	0.054	0.07	0.66	5.7	2.9	13.7	0.026	29.0
	10	0.20	0.019	0.021	<0.10	0.04	0.21	7.8	6.0	14.4	0.053	0.13	0.53	5.5	2.6	13.1	0.025	25.7

^aSoil to extractant ratio was 1 g : 10 ml.

^bDetection limits. Cations below detection limits were: Pb, Cd, Ni, Ag, Co (< 0.01); La, Nd, Pd (< 0.1); Ce (< 0.2); Bi (< 0.5); B, Sn (< 1.0).

TABLE 5. Extraction of Soil Cations from Hanford Soils Using
0.01 M CaCl_2 ^a

		Mn	Cr	Zr	Ba	Zn	Tl	La	Al	Ca	Sr	Mo	P	Nd	Pd	Mg	Na	K	Cu	Si
		<0.10 ^b	<0.01	<0.01	<0.1	<0.01	<0.05	<0.01	<0.1	<0.005	<0.001	<0.1	<0.1	<0.1	<0.1	<0.01	<0.01	<0.01	<0.01	<0.5
	Day	-(µg/ml)-																		
Rupert	1	0.55		0.081	0.38	0.044	0.089	0.79	0.50	303.0	0.31	0.28	0.27	0.28	0.25	13.4	3.2	20.4	0.006	6.1
	2	0.48		0.080	0.34	0.10	0.087	0.79	0.48	278.0	0.31	0.22	0.29	0.24	0.22	13.5	2.7	20.4	0.011	6.8
	4	0.56		0.083	0.36	0.034	0.088	0.82	0.52	306.0	0.32	0.26	0.22	0.28	0.25	14.5	2.9	22.3	0.015	8.9
	7	0.50		0.081	0.35	0.015	0.090	0.80	0.54	306.0	0.32	0.25	0.22	0.25	0.22	14.5	2.8	23.3	0.013	9.6
	10	0.42		0.079	0.33	0.021	0.091	0.80	0.50	304.0	0.32	0.21	0.27	0.21	0.23	14.7	2.2	23.3	0.007	9.6
Burbank	1	0.50		0.082	0.31	0.058	0.088	0.80	0.47	324.0	0.34	0.22	0.29	0.22	0.23	08.7	2.4	14.2	0.013	5.1
	2	0.48		0.078	0.30	0.019	0.092	0.79	0.48	297.0	0.33	0.23	0.28	0.25	0.21	08.5	2.3	13.2	0.012	5.9
	4	0.54		0.085	0.32	0.014	0.090	0.81	0.56	322.0	0.35	0.27	0.27	0.25	0.25	09.3	4.0	16.9	0.021	6.9
	7	0.54		0.086	0.33	0.013	0.094	0.82	0.53	314.0	0.36	0.24	0.23	0.25	0.24	09.4	2.3	15.1	0.018	7.6
	10	0.46		0.081	0.31	0.040	0.091	0.82	0.54	309.0	0.35	0.21	0.24	0.23	0.21	09.3	2.3	15.6	0.013	7.7
Warden	1	2.0		0.077	0.77	0.079	0.087	0.75	0.52	304.0	0.42	0.21	0.34	0.23	0.20	27.7	3.7	8.3	0.017	7.6
	2	0.68		0.075	0.83	0.083	0.088	0.76	0.46	276.0	0.43	0.22	0.37	0.20	0.23	28.3	4.0	9.2	0.015	9.1
	4	0.47		0.076	0.84	0.087	0.089	0.77	0.48	277.0	0.46	0.22	0.34	0.19	0.19	29.9	3.7	9.3	0.016	10.3
	7	0.50		0.081	0.95	0.041	0.086	0.78	0.50	291.0	0.45	0.21	0.33	0.25	0.25	29.5	4.2	9.2	0.016	11.7
	10	0.53	0.010	0.081	0.85	0.063	0.087	0.80	0.55	286.0	0.46	0.26	0.34	0.25	0.23	29.6	3.6	10.0	0.019	12.1
Ritzville	1	0.88		0.074	0.53	0.091	0.086	0.77	0.47	279.0	0.41	0.18	0.33	0.20	0.21	20.2	2.6	11.4	0.016	7.8
	2	0.83		0.080	0.50	0.053	0.093	0.79	0.53	301.0	0.42	0.23	0.35	0.24	0.26	20.3	3.1	11.4	0.020	7.8
	4	1.1		0.084	0.58	0.029	0.094	0.81	0.52	320.0	0.45	0.23	0.33	0.25	0.25	22.5	2.6	12.2	0.018	9.9
	7	1.1		0.084	0.65	0.024	0.086	0.82	0.54	311.0	0.46	0.23	0.36	0.25	0.22	22.6	2.5	12.7	0.014	11.2
	10	1.0		0.082	0.57	0.017	0.093	0.80	0.54	308.0	0.45	0.21	0.38	0.21	0.23	22.0	2.3	12.3	0.011	11.1
Licksfiliot	1	0.55		0.076	0.48	0.033	0.081	0.74	0.47	269.0	0.47	0.20	0.26	0.18	0.20	32.6	2.8	22.0	0.013	7.9
	2	0.69		0.079	0.46	0.086	0.083	0.76	0.50	277.0	0.47	0.27	0.35	0.25	0.22	32.4	3.3	21.5	0.018	8.0
	4	0.57		0.072	0.49	0.017	0.091	0.77	0.46	285.0	0.50	0.21	0.24	0.21	0.19	35.5	2.5	22.7	0.014	9.6
	7	0.68		0.080	0.49	0.029	0.090	0.77	0.51	307.0	0.49	0.23	0.37	0.24	0.22	34.0	2.6	22.4	0.015	9.8
	10	0.64		0.071	0.47	0.040	0.083	0.77	0.48	321.0	0.49	0.18	0.22	0.21	0.22	33.4	2.2	21.8	0.012	10.0

^aSoil to extractant ratio was 1 g : 10 ml. High concentration of Ca in extractant interfered with analysis of Zr, La, Nd, and Pd.

^bDetection limits. Cations below detection limits were: Pb, Cd, Fe, Ni, Ag, Co (< 0.01); Ce (< 0.2); Bi (< 0.5); B, Sn (< 1.0).

The most consistent and reproducible extraction data were obtained using 0.01 M CaCl_2 as the extractant (Table 5). Equilibrium of the extractable soil cations with the extracting solution was generally obtained within the first 24-48 hr. Exceptions include Si which tends to equilibrate by 10 days, Cu which increases over the first 2-4 day period and subsequently decreases by day 10. In addition, the equilibrium concentration of Mn, tends to decrease with extraction time for Warden soil, while increasing with time for Ritzville soil. The behavior of Si and Cu is qualitatively similar using distilled water as the extractant (Table 4). When considering the availability of Cs and Sr for uptake by plants, the elements which have been shown to have a significant influence include K and Na, and Ca, Sr, Ba and Mg, respectively. Of these, K, Mg, Sr and Ba exhibit differences in extractable levels for the five soils using CaCl_2 as the extractant; K, Na, Mg, Sr and Ca are shown to vary when water was used as the extractant.

The final parameter which was investigated in an effort to correlate readily measurable soil parameters with Cs and Sr uptake by cheatgrass and tumbleweed was extractable ^{134}Cs and ^{85}Sr . Only Rupert and Burbank were studied since these soils predominate in waste management areas. Soils amended with ^{134}Cs and ^{85}Sr were extracted over a 24-hr period using four extractants. Cesium-134 comes to equilibrium with the extraction solution within 24 hr for all four extraction systems, except for Rupert extracted with distilled water (Table 6). A comparison of the 4 and 24 hr extractable ^{134}Cs values (filtered and unfiltered) for Rupert shows a substantial portion of extracted cesium to be associated with dispersed soil colloids. This dispersion of soil colloids does not appear to occur with Burbank. Extraction efficiency for both soils was in the order ammonium acetate > ammonium oxalate > distilled water > calcium chloride. Strontium-85 also equilibrates rapidly with the extraction solutions (Table 7). Extraction with distilled water resulted in dispersion of soil minerals in both Burbank and Rupert, as evidenced by differences between filtered and unfiltered samples at 4 and 24 hr. Ammonium oxalate was much less effective in extracting ^{85}Sr , than ^{134}Cs . Extraction of ^{85}Sr was in the order, ammonium acetate > calcium chloride > ammonium oxalate > distilled water.

TABLE 6. Effects of Extractant and Extraction Time on Removal of ^{134}Cs from Burbank and Rupert Soils

Soil	Extractant ^a	Extraction Time, hr							
		0	2	4	4F ^b	6	22	24	24F ^b
		<div><div><div><div><div></div><div>¹³⁴Cs</div><div>Extracted</div><div>(% x 10⁻⁴)</div></div></div><div></div></div></div>							
Burbank	Dist. Water	0.934	26.9	46.0	51.9	30.7	10.9	12.8	15.0
	0.01 M Calcium Chloride	3.56	6.80	3.00	4.42	7.03	8.29	9.42	8.97
	1 M Ammonium Acetate	959	1590	1730	1700	1730	2020	1940	1980
	0.2 M Ammonium Oxalate	1220	1490	1550	----	1530	1780	1720	1710
Rupert	Dist. Water	8.85	58.9	82.2	8.73	85.5	157.0	366.0	67.8
	0.01 M Calcium Chloride	10.7	13.9	11.5	10.6	12.0	14.3	12.6	15.5
	1 M Ammonium Acetate	2100	2410	2490	2460	2540	2780	2750	2730
	0.2 M Ammonium Oxalate	1470	2160	2250	2230	2270	2360	2340	2340

^a Soil to extractant ratio was 1 g : 10 ml.

^b Replicate samples filtered using 0.01 μ membrane filters.

TABLE 7. Effect of Extractant and Extraction Time of Removal of ^{85}Sr from Burbank and Rupert Soils

Soil	Extractant ^a	Extraction Time, hr							
		0	2	4	4F ^b	6	22	24	24F ^b
		----- ⁸⁵ Sr Extracted (% x 10 ⁻²) -----							
Burbank	Water	2.11	3.30	3.78	2.47	4.19	4.58	4.60	2.26
	0.01 <u>M</u> Calcium Chloride	29.2	27.9	28.1	28.2	28.2	29.5	31.4	29.4
	1 <u>M</u> Ammonium Acetate	38.0	41.4	41.7	40.7	44.5	45.2	45.3	45.6
	0.2 <u>M</u> Ammonium Oxalate	1.77	1.62	1.94	----	2.99	1.78	2.27	2.38
Rupert	Water	0.431	0.829	1.37	0.131	1.37	2.23	4.29	0.199
	0.01 <u>M</u> Calcium Chloride	24.7	30.5	31.9	29.4	31.1	31.8	29.7	27.8
	1 <u>M</u> Ammonium Acetate	44.3	42.0	46.1	45.2	46.2	49.4	46.5	47.9
	0.2 <u>M</u> Ammonium Oxalate	0.989	1.62	4.82	4.20	4.07	4.63	3.30	5.26

^aSoil to extractant ratio was 1 g : 10 ml.

^bReplicate samples filtered using 0.01 μ membrane filters.

DISCUSSION

Statistical analysis of the data presented in Tables 1 through 7 was performed using simple correlations and factor analysis. Simple correlations were run between CR values, obtained for ^{85}Sr and ^{134}Cs in tumbleweed and cheatgrass, and 27 soil parameters derived from five soils. Tables 8, 9, 10 and 11 give the correlation coefficients (r) for a series of dependent soil variables; included are variables with r values ≥ 0.50 , and variables felt to play a role in Sr and Cs uptake by plants.

Correlation between ^{85}Sr uptake and soil variables from five Hanford soils are shown in Table 8. For cheatgrass, a number of soil parameters were found to account for the variability in ^{85}Sr CR values (Table 8, Group I). ^{85}Sr uptake is directly correlated with total soil Sr, % sand and pH, while being inversely correlated with CEC, % silt, % clay and % OM. The latter relationships appear to be related to soil factors which play a role in ion exchange. Similar trends are seen for the correlations of ^{85}Sr uptake by tumbleweed, but these are not as significant. No significant correlations were obtained between total soil cations (Cs, Ba, Ca, Mg, K, and Rb) and CR values for either tumbleweed or cheatgrass (Table 8, Group II).

Overall, relationships between soil parameters and ^{134}Cs uptake by cheatgrass (Table 9) were comparable to those obtained for ^{85}Sr uptake. However, relative low correlation coefficients were found between soil parameters and ^{134}Cs uptake by tumbleweed, including those broad parameters affecting ion exchange (CEC, etc.).

Since absorption of ^{134}Cs and ^{85}Sr in higher plants is regulated by the presence and concentration of co-ions and analog elements that can compete for uptake sites, the behavior of ^{134}Cs and ^{85}Sr in the five soils studied was investigated with respect to extractable and exchangeable ions (Table 10 and 11). Correlation coefficients for ^{85}Sr uptake by cheatgrass indicates an inverse relationship between Mg and Sr

extractable with 0.01 M CaCl_2 and water; although correlations for the latter are not as strong. Surprisingly, water extractable Ca does not correlate with ^{85}Sr uptake by cheatgrass. Assuming there are minimal dispersion effects for the elements listed for water extraction and that the CaCl_2 extractant involves both an extraction and Ca^{++} exchange phenomenon, an estimate can be made of the exchangeable cation concentration by subtracting the water extractable concentration from that obtained following 0.01 M CaCl_2 extraction. Under these conditions exchangeable Mg, Sr and Ba exhibit comparatively good inverse correlations with ^{85}Sr uptake by cheatgrass. This is a reasonable observation, since each of these ions can be shown to interact with Ca uptake. In addition, it is interesting to note that high correlations are found for major extractable cations and CEC (Tables 9 and 10).

Similar correlations for ^{85}Sr uptake by tumbleweed vary somewhat. While relatively high inverse correlations are found for water extractable Sr and Ca, extractable Ba exhibits a positive correlation. Data for exchangeable ions show only Sr concentrations to correlate with Sr uptake by tumbleweed. The positive correlations obtained for Ba are difficult to explain since exchangeable Ba does not show a similar trend. The positive correlation obtained for Na with tumbleweed, while not apparent in cheatgrass, may be related to preferential uptake of divalent cations (Mg, Ca, Sr, Ba) over monovalent cations (K, Cs, Rb, Na) characteristic of salt tolerant species.

Correlation coefficients for ^{134}Cs uptake by cheatgrass and tumbleweed and extractable and exchangeable ions are tabulated in Table 11. Surprisingly, exchangeable and extractable Mg, Sr and Ba are relatively well correlated with ^{134}Cs uptake by cheatgrass, while the Cs analogs K and Na are apparently not influencing ^{134}Cs uptake. ^{134}Cs uptake by tumbleweed was not strongly influenced by Mg, Sr and Ba but showed an inverse relationship to both extractable and exchangeable K (-0.62). Na was not correlated. High positive correlations with Mn are unexplainable at the present time.

TABLE 8. Simple Correlations Between ^{85}Sr CR Values for
Tumbleweed and Cheatgrass, and Various
Dependent Soil Variables

Group	Soil Variable	<u>Correlation Coefficient (r)</u>	
		Cheatgrass	Tumbleweed
I	Sr (total soil)	0.90	0.48
	CEC	-0.96	-0.72
	% Sand	0.83	0.70
	% Silt	-0.80	-0.70
	% Clay	-0.83	-0.45
	pH	0.77	0.41
	% OM	-0.75	-0.51
II	Cs (total soil)	0.44	-0.13
	Ba (total soil)	0.08	-0.34
	Ca (total soil)	-0.07	-0.41
	Mg (total soil)	0.17	0.08
	K (total soil)	0.45	-0.21
	Rb (total soil)	0.25	0.14

TABLE 9. Simple Correlations Between ^{134}Cs CR Values for
Tumbleweed and Cheatgrass, and Various Dependent
Soil Variables

Group	Soil Variable	Correlation Coefficient (r)	
		Cheatgrass	Tumbleweed
I	Sr (total soil)	0.85	0.26
	CEC	-0.89	0.05
	% Sand	0.90	-0.36
	% Silt	-0.87	0.37
	% Clay	-0.78	0.18
	pH	0.87	-0.17
	% OM	-0.63	-0.24
II	Cs (total soil)	0.41	-0.02
	Ba (total soil)	0.05	0.74
	Ca (total soil)	-0.06	-0.74
	Mg (total soil)	0.32	-0.47
	K (total soil)	0.53	0.25
	Rb (total soil)	0.10	0.39

TABLE 10. Simple Correlations Between ^{85}Sr CR Values for
Tumbleweed and Cheatgrass, and Extractable Ions

Extractant	Element	Correlation Coefficient (r)	
		Cheatgrass	Tumbleweed
0.01 M CaCl_2	Mg	-0.98	-0.53
	Sr	-0.97	0.41
	Ba	-0.65	-0.21
	Na	-0.28	0.23
	Mn	-0.34	-0.55
	K	0.11	0.18
Water	Mg	-0.83	-0.23
	Sr	-0.50	-0.92
	Ba	0.62	0.97
	Ca	-0.44	-0.89
	Na	0.22	0.76
	Mn	0.83	0.54
Exchangeable ^a	K	0.32	0.32
	Mg	-0.98	-0.56
	Sr	-0.98	-0.75
	Ba	-0.73	-0.33
	Mn	-0.53	-0.60
	K	-0.23	-0.06

^a CaCl_2 extractable - water extractable

TABLE 11. Simple Correlations Between ^{134}Cs CR Values for
Tumbleweed and Cheatgrass, and Extractable Ions

Extractant	Element	Correlation Coefficient (r)	
		Cheatgrass	Tumbleweed
0.01 M CaCl_2	Mg	-0.95	-0.11
	Sr	-0.74	0.19
	Ba	-0.76	0.30
	Na	-0.40	-0.13
	Mn	-0.36	0.85
	K	0.29	-0.66
Water	Mg	-0.77	-0.13
	Sr	-0.49	0.54
	Ba	0.57	-0.14
	Ca	-0.41	0.50
	Na	0.14	-0.23
	Mn	0.13	-0.15
Exchangeable ^a	K	0.49	-0.64
	Mg	-0.96	-0.10
	Sr	-1.00	0.15
	Ba	-0.83	0.31
	Mn	-0.52	0.71
	K	-0.04	-0.62

^a CaCl_2 extractable - water extractable

The simple correlations performed indicate that uptake of both Cs and Sr by cheatgrass is correlated with extractable and exchangeable cations (Sr, Mg and Ba) and soil parameters affecting CEC. This would suggest that the availability and uptake of ^{134}Cs and ^{85}Sr may be controlled by simple sorption and desorption kinetics within the soils. A similar situation appears to exist for ^{85}Sr uptake by tumbleweed, although the correlations are not as high. An interesting aspect of the statistical analyses is the apparent lack of correlation between soil parameters and the uptake of ^{134}Cs by tumbleweed. One could speculate that the lack of correlation was due to inadequate selection of soil parameters, complex interactions between soil parameters, or insufficient variability between treatments. It should be emphasized that it is difficult to obtain reliable correlations using a limited number of soils unless those parameters studied have a normal distribution. Another plausible explanation could be that soil parameters are not limiting the uptake of ^{134}Cs by tumbleweed, but uptake is limited by specific ion absorption mechanisms. Salt tolerant plants such as tumbleweed possess a number of mechanisms which allow them to persist in high salt environments (Rains, 1972). The mechanism of interest with respect to the present data is the ability of plants to selectively exclude or alter the ratio of monovalent cations transported across the root membrane. Although this selectivity is shown to occur with K and Na, the possibility exists that Cs (by virtue of its analog behavior with K) may be under a similar regulation and therefore membrane selectivity for Cs may be the limiting factor in tumbleweed.

Factor analysis was undertaken across species to uncover general correlations between availability of radionuclides (^{134}Cs and ^{85}Sr) and soil parameters. No simple structure or causal factors underlying such structures were ascertained. Principle components' analysis was attempted to reduce the measured independent parameters to fewer artificial variables. The major purpose of the principle components' analysis was to summarize information on soil variants in a set of artificial scores which can account for complex interrelationships among original variants. However, these statistical analyses failed to generate the necessary information to develop a predictive relationship between bioavailability and measured soil parameters.

CONCLUSIONS

The relative availability of ^{134}Cs and ^{85}Sr amended to five Hanford soils was determined for cheatgrass and tumbleweed. Concentration ratios were shown to range from 0.0078 to 0.056 for ^{134}Cs amended soils and 3.5 to 16 for soils amended with ^{85}Sr . The relative order of availability of both ^{134}Cs and ^{85}Sr for cheatgrass was Rupert = Burbank > Ritzville > Warden > Licksillet. Tumbleweed exhibited different orders of availability for the two radionuclides. The order of availability of ^{134}Cs was Ritzville > Warden = Burbank > Rupert > Licksillet, while the order for ^{85}Sr was Rupert > Warden > Burbank > Ritzville > Licksillet.

Simple correlations between measurable soil parameters for five Hanford area soils and corresponding CR values for cheatgrass and tumbleweed suggest that a number of soil factors correlate with the bioavailability of Cs and Sr. In cheatgrass, both Cs and Sr uptake correlated with extractable Sr, Ba and Mg, total soil Sr, and CEC. Although this relationship appears reasonable for Sr uptake, correlations with parameters other than CEC with Cs uptake may be erroneous. Surprisingly, there was a lack of correlation between Sr uptake by cheatgrass and extractable Ca; similarly, extractable K was not correlated with Cs uptake. Simple correlations for Sr uptake by tumbleweed showed relationships similar to those for Sr uptake by cheatgrass, although correlation coefficients were substantially lower. No significant correlations were found for Cs uptake by tumbleweed and soil parameters, except for weak correlations with extractable and exchangeable K. A number of possible reasons for the latter have been discussed. Factor and principle components' analyses failed to demonstrate any general correlations between soil factors and plant availability. A major factor affecting the above correlation appears to be the lack of variability between individual data sets. This lack of variation for individual soil and plant parameters makes statistical analyses difficult. In addition to the close similarities in physical and chemical properties of these Hanford soils, there is some concern about the time required for soil-amended radionuclides to equilibrate with soils. This may represent a major factor as far as availability of Cs and Sr for plant uptake.

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