Geochemical Reconnaissance Study of Vassar Meadow (Adams Rib) Wetlands and Vicinity, Eagle County, Colorado
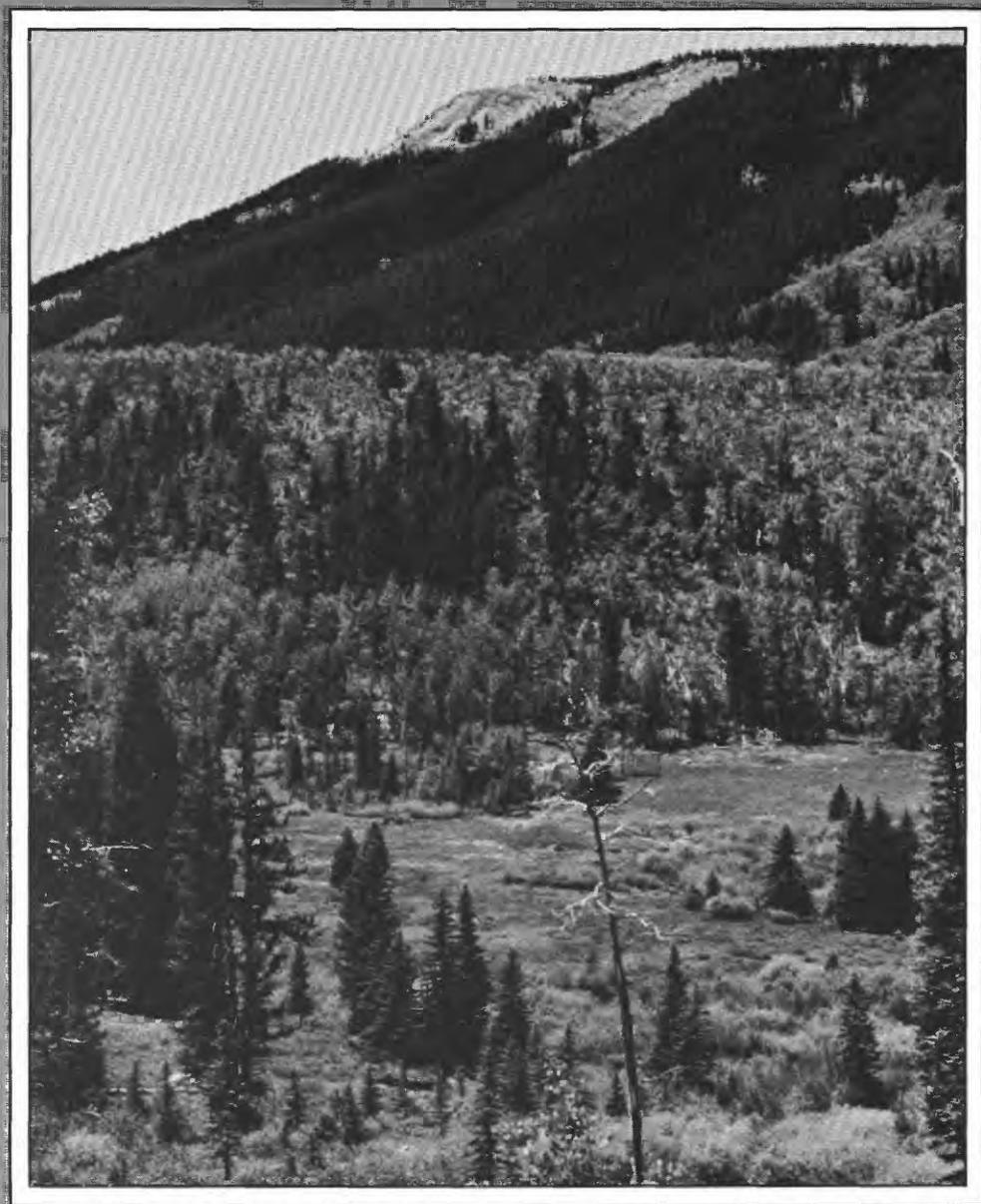
GEOCHEMICAL RECONNAISSANCE STUDY OF VASSAR MEADOW (ADAMS RIB) WETLANDS AND VICINITY, EAGLE COUNTY, COLORADO

By Douglass E. Owen *and* George N. Breit

ABSTRACT

Wetlands are known to be efficient filters of metals dissolved in ground and surface waters. The wetlands at the proposed base area of Adams Rib Recreation Area (Adams Rib) were sampled at the request of the U.S. Environmental Protection Agency to determine if the wetlands are concentrating any metals in sufficient quantities to document a significant filtration function. Out of the many wetland areas present at Adams Rib, time permitted sampling 10 sites using a hand auger.

The 10 sampled wetlands were found to be variously enriched in chromium, molybdenum, and uranium. Four of the wetlands sampled show enrichment in chromium. Three of the wetlands sampled show enrichment in molybdenum. Two of the wetlands sampled contain concentrations of uranium far in excess of what can be explained by the inclusion of rock and mineral fragments within the wetland sediments. The uranium and chromium concentrations, and to a lesser extent molybdenum, represent an environmental concern should they be released as the result of anthropogenic disturbance. The metal accumulation in these wetlands documents that the wetlands have been functioning as filters that protect water quality in East Brush Creek by lowering the dissolved metal content in water.

INTRODUCTION

Adams Rib Recreation Area is a proposed ski area located in Eagle County, Colorado, approximately 17 miles south of the town of Eagle. The site, also known as Vassar Meadow, is immediately north of Yeoman Park (fig. 1). The development of Adams Rib would impact some of the wetlands present in the East Brush Creek ecosystem. This reconnaissance study was performed at the request of the U.S. Environmental Protection Agency (EPA), Region VIII. The EPA believes that the aquatic ecosystem in East Brush Creek constitutes an aquatic resource of national importance (G.R. Reetz, U.S. EPA, written commun., 1992) because the special aquatic sites in the area are essentially undisturbed and provide an excellent example of a diversity of wetlands with a wide range of wetland functions and values. This study focused on sampling wetland sediments to determine if any metals are concentrated in quantities that clearly document that the wetlands have performed the function of filtration. It would follow from such documentation that the wetlands serve to protect water quality in East Brush Creek.

Unusually high concentrations of trace elements in wetland sediments can be the result of several processes. Addition of elements dissolved in ground and surface waters to the sediment is the most common explanation. This enrichment is largely due to geochemical reactions caused by the reducing and complexing capability of wetland sediments. Alternatively, the enrichments could result from the deposition of sediments containing minerals with high trace-element concentrations.

The next three sections will explain some of the reasons why wetlands are efficient filters, provide some comparison data from reconnaissance in other wetlands in Colorado, and describe some of the possible health and environmental implications of metalliferous wetlands. The remaining sections of this paper examine these topics in light of the results of the geochemical study at the Adams Rib wetlands.

WHY WETLANDS ARE EFFICIENT FILTERS OF METALS

Organic matter in wetlands is an effective sorber of uranium and other metals. Organic matter degradation greatly increases the surface area available for sorption and yields humic material, humic acids, and fulvic acids, all of which facilitate geochemical enrichment (Robbins and others, 1990). Moore (1954), in laboratory experiments with uranyl sulfate, found peat could remove as much as 98 percent of added uranium (U) from solution. Research since 1954 has continued to show that organic-rich sediments are
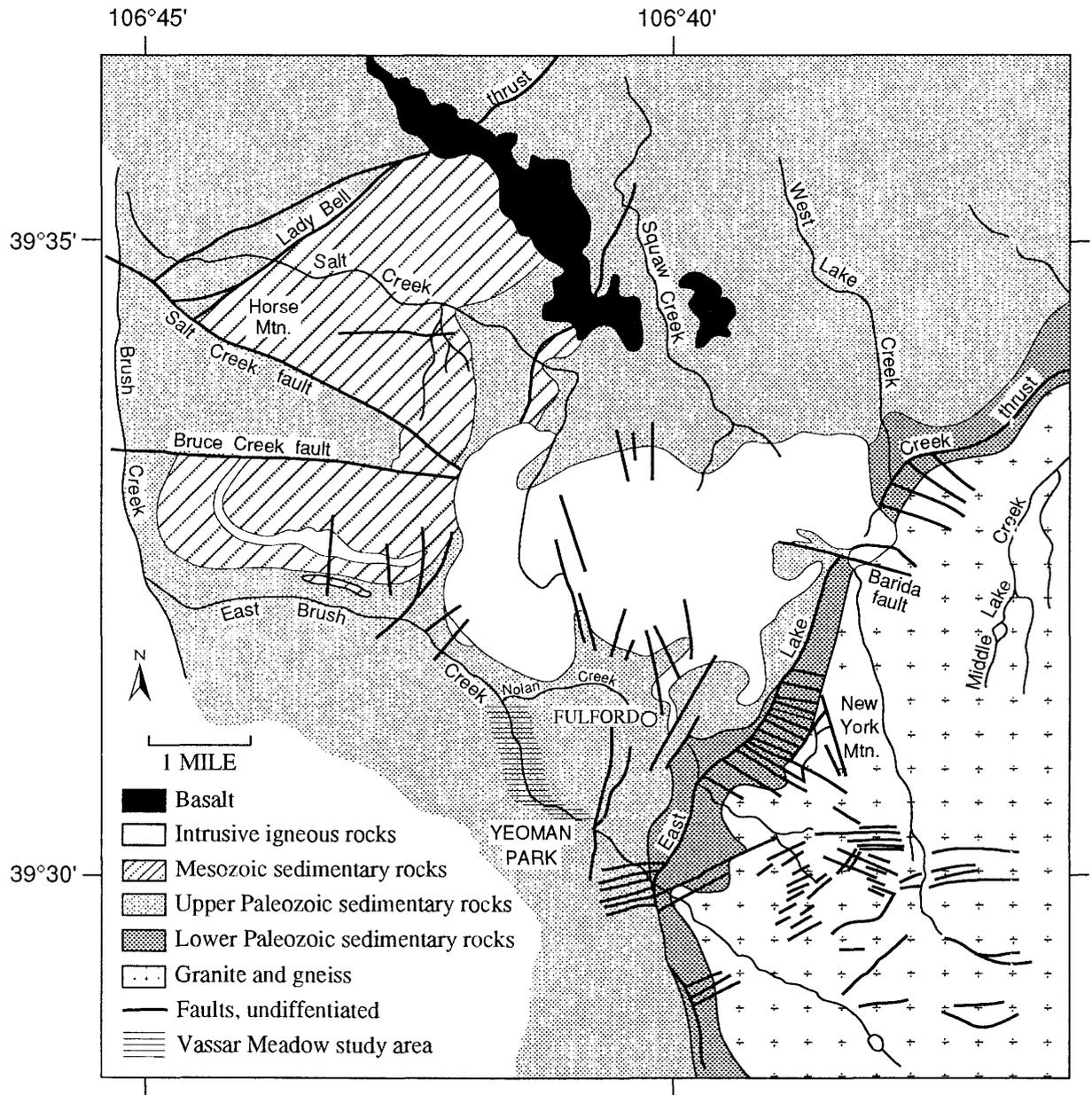
Figure 1.   Bedrock geology map of vicinity around Adams Rib (modified from Gabelman, 1950).

efficient in removing U from water and has shown that the majority of the U becomes bound or complexed with humic and fulvic acids, which typically comprise 6 to 40 percent of the total carbon in peats, peaty mucks, and mucks (Titayeva, 1967; Borovec and others, 1979; Kribek and Podlaha, 1980; Shanbhag and Choppin, 1981; Mathur and Farnham, 1985; Idiz and others, 1986). For uranium, various investigators have calculated effective geochemical enrichment factors, GEF (the ratio of U concentrations in the organic-rich sediments to U concentrations in the ground water), that range from 500 to as high as 2×10^6 (Kochenov and others, 1965; Lopatkina, 1967; Szalay, 1974; Idiz and others, 1986). One wetland currently under

study by the U.S. Geological Survey (USGS) in northern Colorado shows U enrichment factors between 10,000 and 20,000 (R.A. Zielinski, written commun., 1993).

Similar to uranium, other polyvalent cations of high atomic weight have large GEF's in relation to peat (Szalay, 1974). Two early studies in the United States, Lovering (1928) and Eckel (1949) reported on copper being found in a wetlands in Montana and Colorado, respectively. Numerous other investigators have reported on accumulations of metals other than uranium in wetland sediments (for example: Glooschenko and Capobianco, 1978; Domman, 1979; Pakarinen and others, 1983; Taylor and Crowder, 1983; Lee and others, 1984; Stone and Pesavento, 1985; Andrews and

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Fuge, 1986; Dissanayake, 1987). Ibarra and others (1979) reported that heavy metals show a large GEF with humic acids and that high pH, large atomic weights, and high valences all favor metal retention by organic acids. Cation exchange capacity and organic content are positively correlated (Mitsch and Gosselink, 1986). Peat and peaty muck, because they have high organic contents, exhibit a large cation exchange capacity. Stednick (1988) pointed out that the pH of most riparian-wetland systems is near neutral (the relatively few measurements we have made support this observation), which helps limit metal solubility. Ibarra and others (1979) concluded that humic acids produced from peat-forming processes, as well as those in existing peats, can exert a strong concentrating-accumulating effect on heavy metals being transported by natural waters, even waters low in metal concentrations. Tannins (water-soluble secondary plant products that impart some of the "tea" or brown coloration to streams) also form complexes with ions in solution (Crum, 1988).

Bacteria and fungi also play a role in concentrating metals in wetlands. Besides being prime degraders of vegetation in the peat-forming process (Waksman, 1930; Moore and Bellamy, 1974), and thus being partially responsible for formation of sorbents such as humic acids, bacteria themselves may trap metals in or on their cell walls. Beveridge (1978) found that most transition elements have a high affinity for the cell wall of the bacteria *Bacillus subtilis*. Beveridge (1984) found that *Sporosarcina urea* bacteria survive and grow in toxic environments by using their surface arrays to bind and immobilize heavy metals; he proposed that initially bound metal acts as nucleation sites for the growth of metal aggregates that can sequester significant additional amounts of metal from solution. Lovley and others (1991) found that a Fe(III)-reducing bacteria, which he designated GS-15, and *Shewanella putrefaciens* obtain energy for growth by coupling the oxidation of various electron donors to the reduction of U(VI) to U(IV).

Sikora and Keeney (1983) found that fen peats contain more bacteria than bog peats. True bogs are rain-formed and rain-fed (ombrogenous and ombrotrophic); because of low humidity and low rainfall, true bogs are rare in the Rocky Mountain region. Fens, which are sedge-, grass-, or reed-dominated minerotrophic peatlands (Windell and others, 1986), are the dominant peatlands in the Rocky Mountain region and are favorable sites for microbially assisted geochemical enrichment. In the laboratory, Mohagheghi and others (1985) demonstrated that sulfate-reducing bacteria can be effective in the concentration and deposition of uranium. The common fungus *Rhizopus arrhizus* has been reported to be very efficient (180 mg U^{6+}/g) in biosorption of uranium (Shumate and others, 1980; Tsezos and Volesky, 1981, 1982). Although the specific contribution of microbes to geochemical enrichment of wetlands remains to be determined, it is certain that microbial populations contribute to metal enrichment.

PREVIOUS RECONNAISSANCE STUDIES OF WETLANDS IN COLORADO

More than 100 wetlands previously have been sampled for uranium and other metals by the USGS in the Front Range of Colorado (Owen and others, 1992); U data for these wetlands is summarized in figure 2. The highest uranium concentration found at each of the 145 wetland sample stations is used in figure 2 because these concentrations represent the worst case scenarios from both environmental and health standpoints should U be released. Nearly one-half (46 percent) of the wetlands investigated showed uranium concentrations greater than 20 ppm; in light of the newly proposed EPA standards (U.S. EPA, 1991), these U concentrations represent an environmental concern. During the same reconnaissance as that for the uranium, about one-third of the wetlands were also studied for other elements, and their concentrations were found to be quite variable from wetland to wetland.

HEALTH AND ENVIRONMENTAL IMPLICATIONS OF METALLIFEROUS WETLANDS

Processes that liberate uranium or other potentially toxic elements from enriched wetlands are an important concern because of their possible impact on water supplies. Experiments by Kochenov and others (1965) show that sorption of uranium on peat is a reversible process under oxidizing conditions. This reversibility has implications for natural and human-induced alterations of the water table or the water supply to wetlands; these changes may convert the sediment-water system from an anoxic to an oxygenated environment. If a wetland is partly or completely drained, the subsequent oxidation of the organic-rich sediments may liberate metals that have been accumulating from very dilute solutions for thousands of years; this could result in increased metal concentrations in ground and surface waters. Three examples of anthropogenic activities that can influence the water table are road construction, pumping to keep mines dry, and deliberate draining of wetlands to harvest peat or to reclaim the wetlands for other uses. Natural seasonal variations of the water table in uraniferous wetlands can also impact domestic water supplies by influencing whether uranium is being stored or released by organic-rich sediments (Zielinski and Otton, 1989).

Anthropogenic activities that can affect wetland geochemistry and result in the release of uranium or other metals include the introduction of carbonates, sulfates, or phosphates, which can complex and mobilize uranium (Langmuir, 1978; Zielinski and Meier, 1988). These substances are constituents of lime, gypsum, and fertilizer that may be applied to a wetland being used for agriculture. Carbonate is particularly effective and has been tested and
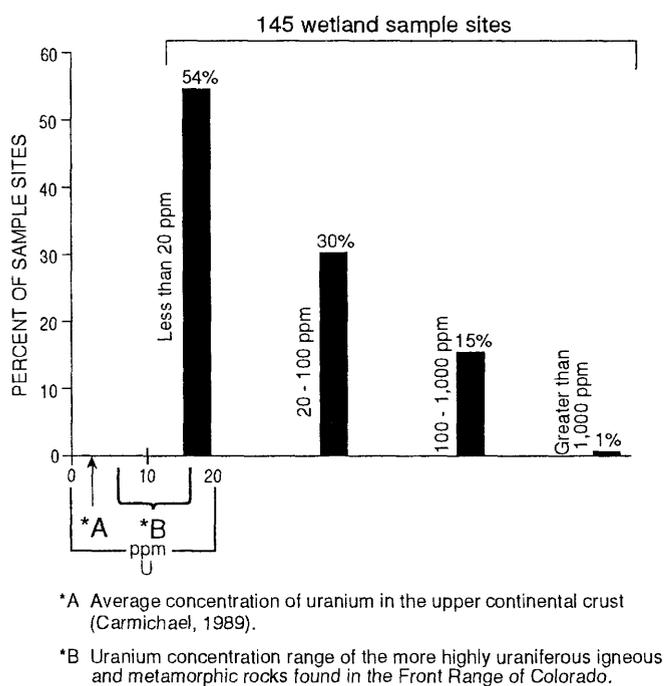
Figure 2. Maximum uranium (U) concentrations found in samples from wetland sites in Colorado (from Owen and others, 1992).

suggested as a leaching solution for mining uranium from wetlands (Hunkin Engineering, 1979).

Changes in pH, particularly introduction of acidic solutions to wetlands, may also effect the release of uranium or other metals from wetlands. Szalay (1974) found that a 1 percent HCl solution totally liberates uranyl (UO_2^{2+}) from peat and that a 1 percent acetic acid solution partially removes uranyl when the peats have a pre-experiment pH of 3–7. In laboratory experiments, Zielinski and Meier (1988) found that concentrated sulfuric acid is a very effective solution to leach uranium from peats. Because these experiments demonstrate that acid solutions can liberate metals held in peats, the effects of acid rain and acid mine drainage entering a wetland need to be better understood. Studies are being conducted in the Southern Rocky Mountains of Colorado to determine the effects on wetlands and the capacity of these wetlands to mitigate the influence of acid mine drainage on water quality (for example: Cooper and Emerick, 1987; Emerick, 1988; Walton-Day and others, 1989; Wildeman and Landon, 1989). The results so far indicate that wetlands receiving acid mine drainage can remove some metals from solution; however, a point is reached when the capacity of the natural filter system is exceeded.

Metal uptake by living plants in wetlands is another issue of health concern. Erickson and others (1991) found that some of the forage in a wetland receiving acid mine drainage has metal concentrations exceeding dietary intake standards for cattle. The possibility also exists that metals may be further concentrated in organisms higher up the food chain. Because some wetlands contain accumulations of peat that are mined for agricultural purposes, not only may disturbing wetlands to mine peat release metals from the peat to the environment, but the use of uranium-bearing or metal-bearing peat for agricultural purposes may cause foodstuffs to accumulate elevated levels of uranium or other metals.

Many metals that are needed by the human body in trace amounts are toxic in high concentrations. Zinc, for example, is needed as a trace element, but high concentrations cause poisoning and can even be lethal. The following discussion will center on uranium because it appears to be concentrated in many wetlands in the Rocky Mountain region. Uranium ingested by humans has an affinity for both kidney and bone tissues. It is known that 1–5 percent of a dose of ingested uranium goes to bone; however, no direct epidemiological study of the radiotoxicity of uranium has been made (Cothern, 1987). The two most common isotopes of uranium found in nature have such long half-lives (U-238, half-life 4.5 billion years; and U-234, half-life 247,000 years) that radiotoxicity is extremely difficult to demonstrate. The primary toxic effect of natural uranium is as a chemical poison to the kidneys. Inflammation of the kidneys (nephritis) is one of the symptoms of uranium poisoning; continued uranium poisoning results in total kidney failure and death. For a discussion of the toxicity of other elements and additional information on uranium toxicity, see Wrenn and others (1985), Cothern and others (1983), Gough and others (1979), and Hem (1985). The U.S. Environmental Protection Agency has proposed a maximum contaminant level for uranium in drinking water of 20 μg/L or 20 ppb (U.S. EPA, 1991). Studies of uraniferous wetlands are yet to be performed to determine possible release rates based on different kinds of disturbances (for example, physical or chemical), the magnitude of the disturbance (for example, how far the water table was dropped or how much acid mine drainage was received), or the duration of the disturbance. Wetlands are a part of the watersheds for many mountain communities, and because some of the wetlands may be enriched in uranium or other metals, any disturbance or alteration of the wetlands should be carefully evaluated beforehand.

PHYSICAL AND GEOLOGIC SETTING

Adams Rib is in the central portion of Eagle County, Colorado. The location of the study area (Vassar Meadow Study Area) is shown on figure 1. On the Fulford 7 1/2-minute quadrangle, the study area is located in T. 6 S., R. 83 W., secs. 22 and 27. Figure 3 is a detailed map that shows the locations of the sample sites in reference to the wetlands delineated by Wright Water Engineers (Denver, Colo.), the road, and East Brush Creek. The property
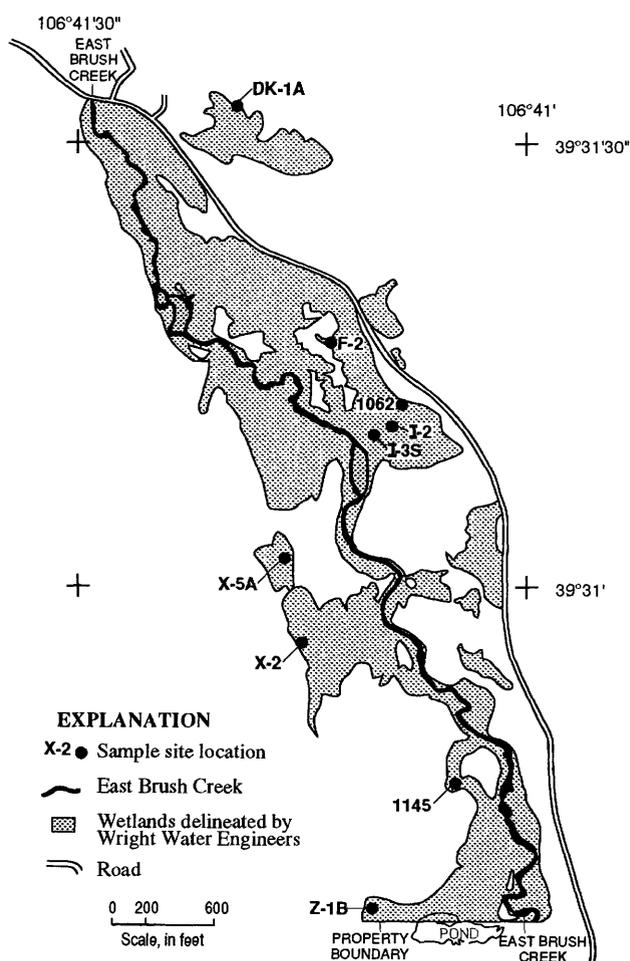
Figure 3. Location of sample sites at Adams Rib.

boundary shown at the south edge of figure 3 corresponds to the north edge of the U.S. Forest Service land at Yeoman Park (a landmark easy to pick out if you drive by the area). The western edge of the Holy Cross Wilderness lies about 3 miles to the east of the study area.

No geologic map for the Fulford 7 ½-minute quadrangle is available; however, a geologic map of the Leadville 1°×2° quadrangle (Tweto and others, 1978) at a scale of 1:250,000 is available that includes the area. Gabelman (1950) studied the geology of the Fulford and Brush Creek mining districts, which includes Yeoman Park and vicinity where the study area is located. Figure 1 is a generalized bedrock geology map of the vicinity around Adams Rib modified from Gabelman (1950). The bedrock in the study area consists of upper Paleozoic sedimentary rocks (fig. 1).

The study area lies completely within the portion of East Brush Creek that has been glaciated, and the wetland sample sites rest on glacial material. The glacial deposits are mapped as young glacial drift, Bull Lake and younger, and consist of unsorted bouldery glacial deposits (till) and associated sand and gravel deposits (Tweto and others, 1978). Terminal/recessional moraines lie just down valley (north)

from the study area shown in figure 3. Glacial moraines form the "ribs" on both sides of the valley. Ground water passing through these moraines forms extensive zones of seeps and springs along the flanks of the valley. These seeps and springs have allowed many wetlands to form away from the immediate stream channel. East Brush Creek throughout the study area is dominantly a cobble- and boulder-clad entrenched stream, except, where it has been modified by beaver activity.

Castor canadensis (American beaver) is active within the valley. The pond shown at the property boundary on the south edge of the study area (fig. 3) is an active beaver pond. There is current beaver activity (dams and a lodge) on the east side of the road just north of the study area along the switchback section of the road that climbs into Vassar Meadow. Beaver were also recently removed from the study area itself (S. Fowler, U.S. EPA, oral commun., 1992). Besides breached and abandoned beaver-pond areas, there are also remnants of pond areas constructed by early residents of the valley.

Several kinds of wetlands are present within the study area. Wet meadows and fens are primarily associated with the seeps and springs. The wet meadows and fens are dominated by sedges; some have moss, and some also are heavily covered by shrubs (primarily willows and bog birch). The wet meadows and fens sampled have organic-rich to peat soils. The riparian area wetlands are dominated by willows and other hydrophytic trees and shrubs. Limited observations indicate that soils within the riparian area are poorly developed and consist of coarse-grained sediments unless within an old pond area.

METHODS

SAMPLING

Sampling was conducted over parts of 2 days in late September 1992. A hand auger with a diameter of 1.25 inches, constructed from ship's auger bits, was used for subsurface sampling. A drill-rod coupling welded to the top of the bit allowed 3-ft-long sections of lightweight aluminum drill rod to be screwed on as necessary. A T-handle fabricated from welded pipe sections was used to turn the auger. The auger was twisted into the ground in 1-ft drives and then pulled straight up out of the hole. The sample recovered on the bit was examined carefully, and any slough above the sample was discarded. As the sample was removed from the bit, examined and bagged, a sample description was made. The sample descriptions are in Appendix I, and will not be repeated here. Successive 1-ft drives were made and additional sections of drill rod added as needed.

Most sample sites selected were at or near springs or prominent seep zones (see Appendix I). These locations were selected because wet meadows and fens that form in

Table 1. Concentrations of selected elements in wetlands sediments.

[Values in percent]

Sample site	Sample interval (ft)	Al	Ca	Fe	K	Mg	Na	P	Ti
X-5A	0–1	0.72	2.4	0.33	0.26	0.43	0.13	0.14	0.03
X-5A	1–2	1.2	4.7	0.35	0.36	0.60	0.17	0.15	0.06
X-5A	2–3	2.0	3.5	2.9	0.54	0.56	0.34	0.11	0.11
X-5A	3–4	6.8	2.4	3.6	1.7	1.3	1.2	0.14	0.43
X-5A	4–4'4"	2.4	3.0	0.82	0.70	0.64	0.41	0.12	0.12
X-5A	4'4"–5'	6.3	1.4	2.5	1.9	0.94	1.2	0.08	0.33
X-5A	5–5'2"	6.3	0.95	2.7	2.1	1.1	1.4	0.07	0.24
Z-1B	0–1	3.3	2.1	1.5	1.1	0.76	0.57	0.15	0.16
Z-1B	1–2	4.8	1.9	2.0	1.5	0.91	0.88	0.12	0.21
Z-1B	2–3	5.5	2.0	2.0	1.9	0.96	0.97	0.14	0.23
Z-1B	3–4	6.5	1.5	2.6	2.0	1.1	1.3	0.09	0.29
X-2	0–1	5.6	0.90	2.7	1.9	0.99	1.2	0.07	0.25
1145	0–9"	5.4	1.1	2.6	1.7	1.0	0.93	0.10	0.24
I-2	0–1	0.73	3.2	2.4	0.27	0.37	0.13	0.13	0.03
I-2	1–2	2.0	3.5	2.9	0.54	0.56	0.34	0.11	0.11
I-2	2–3	6.8	2.4	3.6	1.7	1.3	1.2	0.14	0.43
I-3S	0–1	1.2	3.6	0.45	0.38	0.62	0.20	0.18	0.06
I-3S	1–2	1.1	4.4	0.32	0.34	0.57	0.16	0.18	0.05
I-3S	2–3	1.2	4.7	0.34	0.36	0.60	0.17	0.15	0.06
I-3S	3–4	1.9	4.8	0.49	0.52	0.66	0.26	0.15	0.10
I-3S	4–5	5.3	5.1	2.1	1.0	1.2	1.0	0.22	0.45
I-3S	5–6	5.1	2.8	1.7	1.4	1.0	0.97	0.16	0.25
F-2	0–1	6.6	1.3	3.5	2.0	1.1	1.4	0.09	0.33
DK-1A	0–1	5.6	1.5	2.6	1.8	0.86	1.3	0.09	0.25
DK-1A	1–2	6.1	1.3	2.9	1.9	0.93	1.4	0.09	0.28
DK-1A	2–3	6.2	1.4	2.9	1.9	0.97	1.4	0.0S	0.28
BEV-1	0–1	5.1	1.4	2.3	1.7	0.92	1.0	0.11	0.21
BEV-1	1–2	5.5	1.2	2.4	1.8	1.0	1.2	0.09	0.21
1062	0–1	2.3	3.6	1.2	0.77	0.64	0.43	0.16	0.10
1062	1–2	5.4	2.0	2.5	1.8	1.0	1.2	0.12	0.24
1062	2–3	6.3	1.0	3.0	2.0	1.2	1.5	0.09	0.27
1062	3–4	6.5	0.95	3.1	2.2	1.2	1.6	0.08	0.27
1062	4–4'8"	6.6	0.96	3.2	2.2	1.2	1.5	0.08	0.29

association with springs and seeps provide the organic-rich environment favorable for filtration of the ground water entering them. At each sample site, two holes were augered, generally within 2 to 3 feet of each other, to obtain sufficient sample material for analysis. A total of 10 sites were sampled. At some sites only a surface sample was obtained, whereas at others several intervals from 0- to 6-ft in depth were sampled (see sample interval column, table 1). All of the sample site locations are shown on figure 3, except for BEV-1, which was located about 1/4 mile northwest of Adams Rib in an active beaver area. All samples were dried at 35°C to constant weight on return from the field. After drying, the samples were submitted to USGS analytical laboratories in Denver for grinding and analysis.

ANALYSES

Table 1 shows the results of trace- and major-element analyses performed on the sediment samples. Uranium concentrations were determined at the USGS-TRIGA reactor facility, Denver, Colo., using the delayed neutron (DN) activation counting technique described by Millard and Keaten (1982) and McKowan and Millard (1987). The detection limit for uranium is 0.1 ppm. Uranium concentrations reported in table 1 are on a dry-weight (whole-sample), unashed basis. The other 43 elements listed in table 1 were determined on a split from the ground auger sample that was ashed and analyzed using inductively coupled plasma-atomic emission spectrometry (ICP) described by

Table 1. Concentrations of selected elements in wetlands sediments—*Continued.*

[Values in parts per million; --, not detected]

Sample site	Sample interval (ft)	Mn	Ag	As	Au	B	Ba	Be	Bi	Cd	Ce	Co	Cr
X-5A	0–1	34	<2	29	<8	--	81	<1	<10	<2	8	2	20
X-5A	1–2	41	<2	<10	<8	--	130	<1	<10	<2	11	2	230
X-5A	2–3	720	<2	<10	<8	--	100	<1	<10	<2	17	9	38
X-5A	3–4	510	<2	<10	<8	--	1000	2	<10	<2	62	19	130
X-5A	4–4'4"	61	<2	31	<8	--	270	<1	<10	<2	21	5	69
X-5A	4'4"–5'	180	<2	23	<8	--	660	2	<10	<2	53	12	78
X-5A	5–5'2"	200	<2	20	<8	--	500	2	<10	<2	55	12	110
Z-1B	0–1	230	<2	<10	<8	--	480	1	<10	<2	31	8	68
Z-1B	1–2	180	<2	<10	<8	--	620	2	<10	<2	38	12	93
Z-1B	2–3	200	<2	<10	<8	--	680	2	<10	<2	46	10	75
Z-1B	3–4	300	<2	<10	<8	--	730	2	<10	<2	78	12	82
X-2	0–1	250	<2	<10	<8	--	670	2	<10	<2	42	11	72
1145	0–9"	4300	<2	<10	<8	--	590	2	<10	<2	49	10	83
I-2	0–1	4300	<20	<10	<8	--	360	<1	<10	<2	8	4	15
I-2	1–2	730	<2	<10	<8	--	110	<1	<10	<2	19	9	28
I-2	2–3	490	<2	<10	<8	--	950	2	<10	<2	65	18	110
I-3S	0–1	54	<2	<10	<8	--	150	<1	<10	<2	11	3	180
I-3S	1–2	35	<2	<10	<8	--	130	<1	<10	<2	10	2	210
I-3S	2–3	39	<2	<10	<8	--	130	<1	<10	<2	11	2	220
I-3S	3–4	36	<2	<10	<8	--	200	<1	<10	<2	16	3	280
I-3S	4–5	240	<2	10	<8	--	290	1	<10	<2	44	10	220
I-3S	5–6	190	<2	<10	<8	--	470	1	<10	<2	38	7	190
F-2	0–1	660	<2	<10	<8	--	750	2	<10	<2	69	14	64
DK-1A	0–1	520	<2	<10	<8	--	680	2	<10	<2	42	11	58
DK-1A	1–2	690	<2	<10	<8	--	720	2	<10	<2	57	12	78
DK-1A	2–3	460	<2	<10	<8	--	730	2	<10	<2	71	12	79
BEV-1	0–1	500	<2	<10	<8	--	790	1	<10	<2	38	10	58
BEV-1	1–2	390	<2	10	<8	--	720	2	<10	<2	44	11	58
1062	0–1	260	<2	<10	<8	--	460	<1	<10	<2	21	6	57
1062	1–2	210	<2	<10	<8	--	590	2	<10	<2	47	10	130
1062	2–3	250	<2	<10	<8	--	570	2	<10	<2	64	11	82
1062	3–4	240	<2	<10	<8	--	590	2	<10	<2	67	11	73
1062	4'–4'8"	270	<2	<10	<8	--	590	2	<10	<2	73	11	79

Lichte and others (1987). The results of the ICP analyses (table 1) are reported on a dry-weight (whole-sample) unashed basis.

ADDITIONAL ANALYSES OF CHROMIUM

Because of the relatively high chromium concentrations measured during the ICP analyses, additional chemical analyses were conducted to determine chromium residence in the samples collected from the second auger hole at the sites of most interest. Determination of the residence of chromium can aid in explaining its high concentration in the wetland samples. Residence can be assessed partially by measuring the amount of chromium dissolved in response to treatment of the sediment with chemical reagents. Extractability of chromium from the sediments also can be applied to estimating its mobility if conditions in the wetlands are altered.

The selected samples (previously dried at 35°C) were gently ground in a ceramic mortar and pestle and sieved through an 18-mesh nylon sieve. More than 80 percent of each sample passed through the sieve; materials that did not pass through the sieve (primarily plant and rock fragments) were discarded. The sieved samples were sequentially treated with six solutions of increasing reactivity: (a) 1 M KCl, (b) 0.2 M KH_2PO_4, (c) 1 M acetic acid, (d) 15 percent H_2O_2 and 0.01 N HNO_3, (e) 2 N HNO_3, and (f) aqua regia

Table 1. Concentrations of selected elements in wetlands sediments—*Continued.*

[Values in parts per million; --, not detected]

Sample site	Sample interval (ft)	Cu	Eu	Ga	Ge	Ho	La	Li	Mo	Nb	Nd	Ni	Pb
X-5A	0–1	21	<2	<4	--	<4	4	4	7	<4	5	5	12
X-5A	1–2	16	<2	4	--	<4	7	6	16	<4	6	12	<4
X-5A	2–3	18	<2	5	--	<4	10	12	7	<4	9	15	5
X-5A	3–4	34	<2	19	--	<4	34	44	3	9	31	45	13
X-5A	4'–4'4"	10	<2	5	--	<4	12	14	9	<4	11	18	5
X-5A	4'4"–5'	17	<2	15	--	<4	28	41	3	7	25	29	14
X-5A	5'–5'2"	13	<2	16	--	<4	29	33	2	4	24	31	16
Z-1B	0–1	24	<2	9	--	<4	17	20	3.	<4	16	21	14
Z-1B	1–2	29	<2	13	--	<4	21	27	3.	<4	19	31	12
Z-1B	2–3	30	<2	14	--	<4	26	29	<2	<4	22	27	12
Z-1B	3–4	25	<2	16	--	<4	42	39	<2	7	38	30	16
X-2	0–1	12	<2	13	--	<4	22	31	<2	4	19	25	13
1145	0–9"	17	<2	13	--	<4	28	34	<2	<4	26	28	14
I-2	0–1	10	<2	7	--	<4	4	4	5	<4	<4	5	11
I-2	1–2	18	<2	6	--	<4	11	12	6	<4	10	12	6
I-2	2–3	35	<2	17	--	<4	36	44	<2	9	33	38	13
I-3S	0–1	21	<2	4	--	<4	7	6	14	<4	6	8	5
I-3S	1–2	14	<2	4	--	<4	7	6	14	<4	7	6	<4
I-3S	2–3	14	<2	5	--	<4	7	6	15	<4	7	6	<4
I-3S	3–4	23	<2	6	--	<4	10	9	14	<4	9	11	4
I-3S	4–5	35	<2	13	--	<4	25	21	12	<4	26	26	7
I-3S	5–6	29	<2	13	--	<4	22	25	6	5	20	21	11
F-2	0–1	28	<2	16	--	<4	40	45	<2	6	36	27	13
DK-1A	0–1	22	<2	13	--	<4	23	31	<2	4	19	21	14
DK-1A	1–2	22	<2	15	--	<4	31	35	<2	5	27	28	12
DK-1A	2–3	24	<2	15	--	<4	39	36	<2	6	32	29	13
BEV-1	0–1	19	<2	12	--	<4	21	30	<2	<4	20	22	13
BEV-1	1–2	19	<2	14	--	<4	24	32	<2	<4	21	23	12
1062	0–1	42	<2	7	--	<4	12	13	2	<4	11	13	11
1062	1–2	49	<2	14	--	<4	27	28	<2	<4	26	26	11
1062	2–3	45	<2	15	--	<4	35	33	<2	5	30	25	11
1062	3–4	43	<2	15	--	<4	37	34	<2	6	33	24	12
1062	4–4'8"	47	<2	15	--	<4	40	33	<2	6	35	25	13

(9 N HCl: 4 N HNO$_3$). The procedure was modified from procedures described by Campbell and Tessier (1987) and Chao and Sanzalone (1989). These reagents are intended to dissolve the (a) readily soluble, (b) anion exchangeable, (c) weakly acid soluble, (d) oxidizable, (e) strong acid soluble, and (f) resistate phases.

First, approximately 2 g of each sample were combined with 25 mL of 1 N KCl in a polyethylene centrifuge tube; each suspension was agitated for 2.5 hours. The tubes were centrifuged and the solutions decanted and refrigerated. The solids were resuspended in distilled water, shaken for 1 hour, centrifuged, and the resulting solutions were combined with the original KCl solution. All subsequent rinse solutions were treated similarly. (2) Each residue was mixed with 25 mL of 0.2 M KH$_2$PO$_4$ (pH = 4.5). After shaking for 3 hours, the suspensions were centrifuged and decanted; the solutions were refriger-

ated. The solid residues were rinsed with distilled water. (3) Next, 25 mL of 1 M acetic acid (pH = 2.4) was added to each of the residues. Solutions were shaken for 2.5 hours, then centrifuged and decanted. The remaining solids were rinsed with distilled water. (4) Each residue was combined with 15 mL of a solution containing 15 percent H$_2$O$_2$ and 0.01 N HNO$_3$ (pH = 1.8) and allowed to react overnight. The next day, the samples were heated at 50°C for 1 hour. An additional 5 mL of the reagent solution was added to each sample, and the samples were allowed to react again overnight. The following day, the samples were shaken for 15 minutes, centrifuged, and the solutions decanted and refrigerated; solids were rinsed with distilled water. (5) Each residue was then combined with 25 mL of 2 N HNO$_3$, shaken, and allowed to react for 2 days. The solutions and samples were shaken for 30 minutes, centrifuged, and the solutions decanted. Again the residual sol-

Table 1. Concentrations of selected elements in wetlands sediments—*Continued.*

[Values in parts per million; --, not detected]

Sample site	Sample interval (ft)	Sc	Sn	Sr	Ta	Th	U	V	W	Y	Yb	Zn	Zr
X-5A	0–1	<2	<5	360	<40	<4	24	210	--	3	<1	21	--
X-5A	1–2	2	<5	670	<40	<4	56	77	--	11	<1	10	--
X-5A	2–3	4	<5	200	<40	<4	47.7	48	--	7	<1	50	--
X-5A	3–4	14	<5	310	<40	8	36.2	100	--	24	2	66	--
X-5A	4–4′4″	4	<5	490	<40	<4	33.1	210	--	7	<1	35	--
X-5A	4′4″–5	10	<5	320	<40	7	32.3	150	--	17	2	64	--
X-5A	5–5′2″	9	<5	220	<40	9	6.29	110	--	17	1	51	--
Z-1B	0–1	6	<5	90	<40	5	7.62	71	--	13	1	55	--
Z-1B	1–2	8	<5	130	<40	6	6.93	110	--	16	2	52	--
Z-1B	2–3	9	<5	150	<40	8	11.4	66	--	18	1	53	--
Z-1B	3–4	11	<5	190	<40	13	4.14	68	--	22	2	55	--
X-2	0–1	8	<5	160	<40	7	3.51	75	--	12	1	49	--
1145	0–9″	10	<5	95	<40	7	3.85	69	--	24	2	52	--
I-2	0–1	<2	<5	150	<40	<4	7.53	21	--	2	<1	30	--
I-2	1–2	4	<5	200	<40	<4	19.4	47	--	7	<1	50	--
I-2	2–3	14	<5	300	<40	9	5.21	100	--	25	2	65	--
I-3S	1–0	2	<5	520	<40	<4	45.4	77	--	17	1	18	--
I-3S	1–2	2	<5	630	<40	<4	70.5	70	--	13	<1	10	--
I-3S	2–3	2	<5	670	<40	<4	68.7	77	--	11	<1	10	--
I-3S	3–4	3	<5	710	<40	<4	56.2	70	--	14	1	13	--
I-3S	4–5	13	<5	780	<40	5	33	97	--	23	2	37	--
I-3S	5–6	10	<5	470	<40	6	23.4	66	--	21	2	44	--
F-2	0–1	11	<5	210	<40	10	3.67	84	--	24	2	61	--
DK-1A	0–1	8	<5	190	<40	7	3.68	63	--	12	1	47	--
DK-1A	1–2	10	<5	200	<40	9	3.79	70	--	14	1	48	--
DK-1A	2–3	9	<5	210	<40	11	3.61	70	--	17	1	49	--
BEV-1	0–1	8	<5	110	<40	7	4.66	67	--	16	2	48	--
BEV-1	1–2	9	<5	110	<40	7	47	67	--	16	1	48	--
1062	0–1	4	<5	150	<40	<4	14.4	39	--	12	<1	33	--
1062	1–2	9	<5	170	<40	9	6.49	60	--	21	2	44	--
1062	2–3	11	<5	160	<40	10	3.62	70	--	23	2	50	--
1062	3–4	11	<5	160	<40	10	3.39	70	--	22	2	51	--
1062	4–4′8″	11	<5	170	<40	11	3.23	72	--	23	2	52	--

ids were rinsed with distilled water. (6) The remaining solids along with 30 mL of aqua regia were heated at 80°C for 4 hours covered, then uncovered and heated to dryness. The residues were mixed with another 30 mL aliquot of aqua regia, heated for another 4 hours covered, then uncovered and heated to dryness. The dried residues were then dissolved in 50 mL of 2 N HCl and stored for analysis. A few undissolved mineral grains and some organic carbon were common after the aqua regia treatment.

Chromium concentrations in all solutions were measured by flame atomic absorption spectrophotometry. Standards were prepared in the reagents used for each extraction. Samples extracted by H_2O_2 were analyzed by standard addition because of probable interference from dissolved organic compounds as indicated by the dark-brown extracting solutions. Two samples were analyzed in duplicate through the entire extraction procedure. Average relative precision of

the duplicates was 5 percent. Total chromium extracted for the duplicate samples was reproduced within 3 percent. A sample known to contain chromium-rich illite-smectite (CR-019) was also treated with the sequential extraction scheme to evaluate the behavior of Cr-rich silicates relative to the various reagents.

Because the auger used for sampling contained Cr, the potential of Cr contamination from the steel auger was investigated. Two samples were filed off the steel auger, dissolved in aqua regia, and analyzed and found to contain 3,000 and 5,000 ppm chromium respectively; both samples were 93.5 weight percent iron. Although this chromium concentration is high, the abundance of iron in the wetland samples (table 1) indicates that contamination from the auger is not significant. Several wetland samples with chromium contents greater than 100 ppm have less than 0.5 weight percent iron; the iron value

limits the maximum amount of chromium attributable to contamination to less than 30 ppm (based on a potential contribution of 60 ppm Cr for 1 percent Fe added from the auger). Again, this estimate assumes that all iron present in the sediment sample has come from auger contamination and that none of the iron in the sample is from natural sources; however, iron is a ubiquitous natural component in soils. Therefore, even assuming that all the iron in wetland samples came from the auger, the Cr values cannot be explained by contamination.

RESULTS AND DISCUSSION

DN AND ICP DATA

The results of the DN and ICP analyses of the auger samples collected during this reconnaissance study are shown in table 1. For comparative purposes, table 2 from Shacklette and Boerngen (1984) is shown here; it lists mean concentrations, deviations, and ranges of element concentrations in soils and other surficial materials in the conterminous United States. Comparing table 1 with table 2, three metals standout as being enriched in various wetlands in Vassar Meadow: chromium, molybdenum, and uranium.

Sites X-5A, I-2, I-3S, and 1062 all have chromium concentrations greater than 100 ppm; the mean for the Western United States is 41 ppm (table 2). As is shown in the next section, Cr enrichment is in a form that is potentially soluble. The maximum contaminant level for chromium in drinking water is 0.1 mg/L (100 ppb). Sites X-5A, I-2, and I-3S have molybdenum concentrations greater than 5 ppm, whereas, the mean for the Western United States is only 0.85 ppm (see tables 1 and 2). Though the molybdenum concentrations indicate transport in ground water and filtration by the wetlands, the concentrations do not likely represent any hazard to man should release due to disturbance occur (see Gough and others, 1979, p. 36–38).

The uranium concentrations, which are as high as 56 ppm in X-5A and 70 ppm in I-3S, are considerably higher than the average concentration of uranium in the upper continental crust (2.5 ppm; Carmichael, 1989) or the 2.5 ppm in surficial materials (table 2). There are no known uranium occurrences within the drainage basin of East Brush Creek (Collins and others, 1982, plates 2 and 10). The highest uranium concentration within many miles of the study area on the uranium contour map of the Leadville 1°×2° quadrangle (Collins and others, 1982, plate 4a) is 2.3 ppm. The uranium concentrations found in X-5A and I-3S are much higher than would be expected, even if a high percentage of the detrital component were made up of the more uraniferous igneous and metamorphic rocks that are found much farther east in the Front Range of Colorado (fig. 2). The uranium content of X-5A and I-3S shown on figure 2 in comparison to the

Front Range wetlands places them in the upper 46 percent by concentration. Because available source rocks contain insufficient uranium, the uranium concentrations found in the wetland sediments can only minimally be attributed to the detrital component. The uranium concentrations indicate that the wetland sediments are removing and trapping uranium from waters passing through them.

Potential release of uranium from the sediments into ground and surface waters, based on the U concentrations found in the wetland samples, represents a possible hazard should the wetlands be disturbed or removed. The U.S. Environmental Protection Agency has proposed a maximum contaminant level for uranium in drinking water of 20 μg/L or 20 ppb (U.S. EPA, 1991). These sediments contain concentrations of uranium that are as much as 3,500 times higher than the proposed water standard. As mentioned previously in this paper, uranium can be released from organic-rich sediments by several processes. For example, draining the sediments, which changes them from an anoxic to oxygenated environment, is likely to release uranium. If X-5A and I-3S must be disturbed, the release of uranium to East Brush Creek should be guarded against.

ADDITIONAL CHROMIUM ANALYSES

The amount of chromium in the extracting solutions varied with the strength of the reagent (table 3). No measurable chromium was released from the sediments by treatments intended to dissolve readily soluble and anion-exchangeable forms of chromium (KCl and KH_2PO_4, respectively). Between 48 and 63 percent of the chromium dissolved after oxidation at low pH (H_2O_2 plus 0.01 N HNO_3). Less than 6 percent of the chromium dissolved in weak acid (1 M acetic acid), and less than 25 percent dissolved in strong acid (2 N HNO_3) treatments. Aqua regia dissolved relatively small amounts. Extraction treatments of the standard CR-019, except for aqua regia, were relatively ineffective at dissolving silicate-bound chromium. The different response of CR-019 relative to the wetland sediments indicates that chromium extracted from the wetland sediments is not contained in clay minerals.

The total concentration of chromium measured by the sum of the extracted amounts are consistently different from those reported in table 1. The samples analyzed by extraction were collected from separate auger holes within about 0.5 m of the samples collected for the DN and ICP analyses. Therefore, some of the variation can be attributed to heterogeneity in the wetland. Most extraction totals are less than the ICP values and can best be explained by the inability of aqua regia to completely dissolve resistate minerals, such as spinels and some silicates, that could contain chromium. Dissolution for ICP analyses typically includes hydrofluoric acid to dissolve silicates.

Table 2. Mean concentrations, deviations, and ranges of elements in samples of soils and other surficial materials in the conterminous United States.

[Means and ranges are reported in parts per million (μg/g), and means and deviations are geometric, except as indicated. Ratio, ratio of number of samples in which the element was found in measurable concentrations to number of samples analyzed. Leaders (--) indicate no data available. From Shacklette and Boerngen (1984)]

| Element | Conterminous United States | | | Western United States (west of 96th meridian) | | | | | Eastern United States (east of 96th meridian) | | | | |
	Mean	Deviation	Estimated arithmetic mean	Ratio	Mean	Deviation	Observed range	Estimated arithmetic mean	Ratio	Mean	Deviation	Observed range	Estimated arithmetic mean
Al (%)	4.7	2.48	7.2	661:770	5.8	2.00	0.5->10	7.4	450:477	3.3	2.87	0.7->10	5.7
As	5.2	2.23	7.2	728:730	5.5	1.98	<0.10–97	7.0	521:527	4.8	2.56	<0.1–73	7.4
B	26	1.97	33	506:778	23	1.99	<20–300	29	425:541	31	1.88	<20–150	38
Ba	440	2.14	580	778:778	580	1.72	70–5,000	670	541:541	290	2.35	10–1,500	420
Be	0.63	2.38	0.92	310:778	0.68	2.30	<1–15	0.97	169:525	0.55	2.53	<1–7	0.85
Br	0.56	2.50	0.85	113:220	0.52	2.74	<0.5–11	0.86	78:128	0.62	2.18	<0.5–5.3	0.85
C (%)	1.6	2.57	2.5	250:250	1.7	2.37	0.16–10	2.5	162:162	1.5	2.88	0.06–37	2.6
Ca (%)	0.92	4.00	2.4	777:777	1.8	3.05	0.06–32	3.3	514:514	0.34	3.08	0.01–28	0.63
Ce	63	1.78	75	81:683	65	1.71	<150–300	75	70:489	63	1.85	<150–300	76
Co	6.7	2.19	9.1	698:778	7.1	1.97	<3–50	9.0	403:533	5.9	2.57	<0.3–70	9.2
Cr	37	2.37	54	778:778	41	2.19	3–2,000	56	541:541	33	2.60	1–1,000	52
Cu	17	2.44	25	778:778	21	2.07	2–300	27	523:533	13	2.80	<1–700	22
F	210	3.34	430	598–610	280	2.52	<10–1,900	440	390:435	130	4.19	<10–3,700	360
Fe (%)	1.8	2.38	2.6	776:777	2.1	1.95	0.1->10	2.6	539:540	1.4	2.87	0.01->10	2.5
Ga	13	2.03	17	767:776	16	1.68	<5–70	19	431:540	9.3	2.38	<5–70	14
Ge	1.2	1.37	1.2	224:224	1.2	1.32	0.58–2.5	1.2	130:131	1.1	1.45	<0.1–2.0	1.2
Hg	0.058	2.52	0.089	729:733	0.046	2.33	<0.01–4.6	0.065	534:534	0.081	2.52	0.01–3.4	0.12
I	0.75	2.63	1.2	169:246	0.79	2.55	<0.5–9.6	1.2	90:153	0.68	2.81	<0.5–7.0	0.12
K (%)[1]	1.5	0.79	None	777:777	1.8	0.71	0.19–6.3	None	537:537	1.2	0.75	0.0005–3.7	--
La	30	1.92	37	462:777	30	1.89	<30–200	37	294:516	29	1.98	<30–200	37
Li	20	1.85	24	731:731	22	1.58	5–130	25	479:527	17	2.16	<5–140	22
Mg (%)	0.44	3.28	0.90	777:778	0.74	2.21	0.03->10	1.0	528:528	0.21	3.55	0.005–5	0.46
Mn	330	2.77	550	777:777	380	1.98	30–5,000	480	537:540	260	3.82	<2–7,000	640
Mo	0.59	2.72	0.97	57:774	0.85	2.17	<3–7	1.1	32:524	0.32	3.93	<3–15	0.72
Na (%)	0.59	3.27	1.2	744:744	0.97	1.95	0.05–10	1.2	363:449	0.25	4.55	<0.05–5	0.72
Nb	9.3	1.75	11	418–771	8.7	1.82	<10–100	10	322:498	10	1.65	<10–50	12
Nd	40	1.68	46	120:538	36	1.76	<70–300	43	109:332	46	1.58	<70–300	51
Ni	13	2.31	19	747:778	15	2.10	<5–700	19	443:540	11	2.64	<5–700	18
P	260	2.67	430	524:524	320	2.33	40–4,500	460	380:382	200	2.95	<20–6,800	360
Pb	16	1.86	19	712:778	17	1.80	<10–700	20	422:541	14	1.95	<10–300	17
Rb	58	1.72	67	221:224	69	1.50	<20–210	74	107:131	43	1.94	<20–160	53
S (%)	0.12	2.04	0.16	34:224	0.13	2.37	<0.08–4.8	0.19	20:131	0.10	1.34	<0.08–0.31	0.11
Sb	0.48	2.27	0.67	35:223	0.47	2.15	<1–2.6	0.62	31:131	0.52	2.38	<1–8.8	0.76
Sc	7.5	1.82	8.9	685:778	8.2	1.74	<5–50	9.6	389:526	6.5	1.90	<5–30	8.0
Se	0.26	2.46	0.39	590:733	0.23	2.43	<0.1–4.3	0.34	449–534	0.30	2.44	<0.1–3.9	0.45
Si (%)[1]	31	6.48	None	250:250	30	5.70	15–44	None	156:156	34	6.64	1.7–45	--
Sn	0.89	2.36	1.3	218:224	0.90	2.11	<0.1–7.4	1.2	123:131	0.86	2.81	<0.1–10	1.5
Sr	120	3.30	240	778:778	200	2.16	10–3,000	270	501:540	53	3.61	<5–700	120
Ti (%)	0.24	1.89	0.29	777:777	0.22	1.78	0.05–2.0	0.26	540:540	0.28	2.00	0.0007–1.5	0.35
Th	8.6	1.53	9.4	195:195	9.1	1.49	2.4–31	9.8	102:102	7.7	1.58	2.2–23	8.6
U	2.3	1.73	2.7	224:224	2.5	1.45	0.68–7.9	2.7	130:130	2.1	2.12	0.29–11	2.7
V	58	2.25	80	778:778	70	1.95	7–500	88	516:541	43	2.51	<7–300	66
Y	21	1.78	25	759:778	22	1.66	<10–150	25	477:541	20	1.97	<10–200	25
Yb	2.6	1.79	3.1	754:764	2.6	1.63	<1–20	3.0	452:486	2.6	2.06	<1–50	3.3
Zn	48	1.95	60	766:766	55	1.79	10–2,100	65	473:482	40	2.11	<5–2,900	52
Zr	180	1.91	230	777:778	160		1.77	<20–1,500	190	539:541	220	2.01	<20–2,000	290

[1] Means are arithmetic; deviations are standard.

Table 3.   Concentrations of chromium extracted from sediments from Vassar Meadow.

[All values are in parts per million (ppm) chromium in the sediment on a dry-weight basis]

Reagent	Sample identification						
	X-5A 1'-2'	X-5A 2'-3'	X-5A 3'-4'	X-5A 4'-5'	I-3S 2'-3'	I-3S 3'-4'	CR-019
1 M KCl	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
0.2 M KH$_2$PO$_4$	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
1 M acetic acid	0.8	0.5	<0.5	<0.5	1.1	1.4	0.6
15% H$_2$O$_2$ + 0.01 N HNO$_3$	6.4	9	16	22	123	410	1.3
2 N HNO$_3$	2.4	2.3	3.3	8.8	47	150	2.0
Aqua regia	3.5	3.7	7.2	15	32	87	44
Total	13	16	27	46	200	650	48

The results of the chemical extractions are consistent with chromium contained mainly as a reduced phase in the wetland sediments. In most surface environments, chromium may exist in two oxidation states, Cr(III) and Cr(VI) (fig. 4). Chromium (VI) forms the relatively soluble chromate ion (H$_n$CrO$_4^{n-2}$). Reductants, such as ferrous iron, organic matter, and hydrogen sulfide, are known to reduce Cr(VI) to Cr(III). Chromium(III) species are relatively insoluble in the pH range of most natural waters (Rai and others, 1989). The wetland sediments contain abundant organic matter and one sample contains iron sulfide minerals, which are indicative of strongly reducing conditions. Therefore, chromium (III) species should be the dominant form in the sediments. Depending on the pH of waters in the wetland and composition of the organic matter, the chromium may be in the form of insoluble hydroxides or may be bound to the organic matter. The increased abundance of soluble chromium after oxidative treatment of the sediments may be due to either the conversion of adsorbed chromium (III) to chromium (VI) species or the destruction of organic matter that occluded the chromium-containing phase.

The release of chromium from the sediments by oxidative treatment is consistent with accumulation of chromium from water that passed through the wetland and not an accumulation of chromium-rich detrital minerals. Detrital minerals derived from rocks surrounding the drainage may have included grains of mica, spinel, and clays that could contain large amounts of chromium. However, these minerals are expected to be relatively unreactive to the reagents used in the sequential extraction procedure.

Modification of conditions in the wetland could cause oxidation of the sediments and release chromium to surface waters. The rate of release and amount of chromium eventually dissolved will be a strong function of redox and geochemical conditions. Studies of chromium oxidation in soils (Bartlett and James, 1988; Saleh and others, 1989) determined that the rate of oxidation of Cr(III) to Cr(VI) is slow in the absence of manganese oxides (oxidation half-lives ranged to as long as 40 years). Considering the
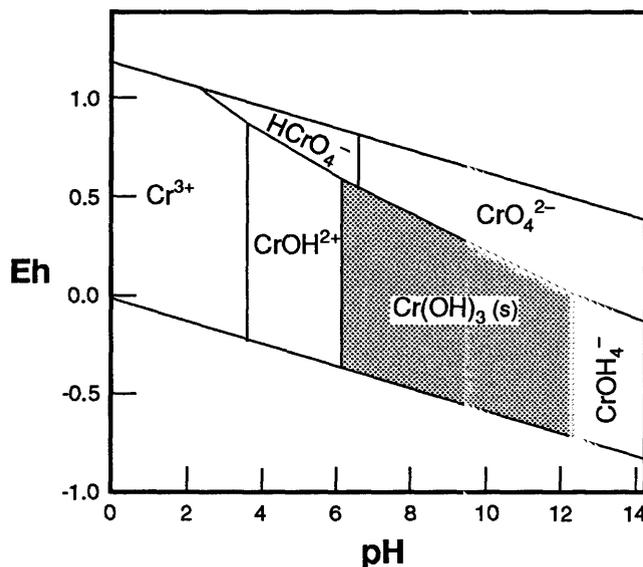

Figure 4. Eh-pH diagram of the system Cr-O-H at 25°C. Shaded area is stability field of solid chromium hydroxide (Cr(OH)$_3$ (s)) at a concentration of 50 ppb total dissolved chromium. Diagram was constructed using the thermodynamic data of Schmidt (1984) and Rai and others (1987).

reducing conditions of the wetland sediments, manganese oxides are unlikely to be stable in existing sediments. In addition, the capacity of the sediments to form and maintain iron in the reduced (ferrous) state would have to be lost to prevent the re-reduction of Cr(VI) species (Anderson and others, 1994) formed by oxidation.

CONCLUSIONS

The metal trapping behavior exhibited by wetlands at the Adams Rib site clearly documents that they have performed the wetland function of filtration. Some of the wetlands are enriched in chromium, molybdenum, and uranium. By removing these metals from water passing through them, they have helped protect water quality in East Brush Creek.

Any mitigation for wetland loss due to development of the site should take loss of function into consideration. Any removal or draining of metalliferous wetland sediments should have safeguards in place to prevent escape of metals (particularly chromium and uranium) to ground or surface waters.

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APPENDIX I—FIELD NOTES

SAMPLE SITE X-5A:

Auger hole located 16.8 m from X-5A along a bearing of 260°. Sulfur bacteria (*Thiothrix* sp.?). Slight sulfur smell. Seep area, slightly domed fen with moss and sedge dominant. Water: pH 7.5; temperature 10°C.

0–1′	Dark brown (10YR 3/3) fiber-rich peaty muck.
1–2′	Yellowish brown (10YR 5/6) fibric peat.
2–3′	Dark brown (10YR 3/3) peaty muck.
3–4′	Dark brown (7.5YR 3/2) peaty muck.
4–4′4″	Peaty muck.
4′4″–5′	Dark brown sandy, gritty silt.
5–5′2″	Grayish grus.

SAMPLE SITE Z-1B:

Spring ≅ 6 ft deep. Fen area—sedges, moss, willows, and bog birch. Area domed (spring mound). Water: pH 6.5; temperature 8°C.

0–1′	Very dark grayish brown (10YR 3/2) peaty muck with fibers.
1–2′	Very dark grayish brown (10YR 3/2) peaty muck with fibers and some grit.
2–3′	Very dark grayish brown (10YR 3/2) peaty muck with fibers, grades into muck with grit.
3–4′	Peaty muck.

HAT CREEK:

Water: pH 7.5; temperature 8°C.

SAMPLE SITE X-2:

Auger hole located 14.6 m from X-2 along a bearing of 320°. Seep located at the edge of a carr filling an old pond. Sedges and moss. Water: pH 6.5; temperature 9°C.

0–1′	Dark grayish brown (10YR 4/2) organic-rich, gritty silt.

SAMPLE SITE 1145:

Auger hole located 11.3 m from stake (backsight to stake 230°). Seep/spring area appears to have perennial flow. Willows, sedge, moss, and watercress. Water: pH 7; temperature 9°C.

0–9″	Dark brown peaty muck. Cobbles at base.

SAMPLE SITE I-2:

Good seep area with iron bacteria (red flocs and also *Leptothrix*). Sedges, willow and moss. Water: pH 7–7.5; temperature 13°C.

0–1′	Very dark grayish brown (10YR 3/2) peat.
1–2′	Very dark grayish brown (10YR 3/2) peat with wood fragments grading to peaty muck at base.
2–3′	Very dark grayish brown (10YR 3/2) muck.

SAMPLE SITE I-3S:

Spring ≅ 6 ft deep. Spring mound ≅ 8 m in diameter. Sulfur bacteria, stinks. Sedges, moss, watercress. Water: pH 7.5; temperature 9.5°C.

0–1′	Brown to dark brown (7.5YR 4/2) peat.

1–2′	Brown to dark brown (7.5YR 4/2) peat, a little more humic.
2–3′	Peat.
3–4′	Very dark grayish brown (10YR 3/2) muck with charcoal fragments and wood fragments.
4–5′	Very dark grayish brown (10YR 3/2) muck—silty with wood fragments, and organic-rich silt with charcoal fragments.
5–6′	Organic-rich silt.

SAMPLE SITE F-2:

Auger hole located 20 m from F-2 along a bearing of 280°. Seep zone with sedges, moss, willows, and horsetail. Major seep/spring area with flowing springs late in the year. Water: pH 7.5; temperature 13°C.

0–1′	Very dark grayish brown (10YR 3/2) muck with grit and twigs.

SAMPLE SITE DK-1A:

Auger hole located 50.5 m from soil pit DK-1A (backsight 230°). Broad leaf sedges dominant. Wetland below has a lot of scouring rush in with the sedge. Water: pH 6.5; temperature 7°C.

0–1′	Very dark grayish brown (10YR 3/2) muck.
1–2′	Muck.
2–3′	Muck.

SAMPLE SITE BEV-1:

Auger hole located above the new beaver pond that has a lodge present in it. The beaver pond is located up valley from the first switchback in the road. Seep area with sedges dominant, some moss. Water: pH 7–7.5; temperature 9°C.

0–1′	Dark brown muck with some grit.
1–2′	Muck with some grit.

SAMPLE SITE 1062:

Auger hole located 16.2 m from stake, backsight 170°. Sampled just below a good spring. Spring ≅ 5 ft deep. Water: pH 7.5; temperature 6°C.

0–1′	Very dark grayish brown (10YR 3/2) peaty muck.
1–2′	Peaty muck with some grit.
2–3′	Dark brown (7.5YR 3/2) muck with grit and charcoal fragments grading to gritty silt.
3–4′	Gritty muck with wood fragments.
4–4′8″	Dark brown (7.5YR 3/2) gritty silt.