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Environmental aspects of selected trace elements associated with coal and natural waters of Pench Valley coalfield of India and their impact on human health

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Abstract

The Pench Valley coalfield extends over 32 km from west to east and is bounded by latitude 22°09'–22°24'N and longitude 78°38'30"–79°0'E in the Chhindwara district of Madhya Pradesh (M.P.), India. This coalfield produces about 181,000 tons per month of non-coking grade coal from 17 mines. For an environmental hazard study, composite samples of coals from Eklehra, Rawanwara Khas, and Shivpuri open cast collieries were prepared for spectrographic analysis of Co, Cu, Cr, Mn, Ni and Pb. The concentrations of these elements range from 17–36.5 ppm, 66–105 ppm, 55–58.5 ppm, 745–935 ppm, 59–78 ppm, and 18.5–22 ppm, respectively. These elements become mobile with the exploitation of coal and on combustion contaminate farms, forests, and soils, and affect the quality of surface and ground waters, and finally, human health. The study of trace elements in natural waters suggests that concentration of the above elements are above the safety limit in a majority of samples and are hazardous to human health. © 1999 Elsevier Science B.V. All rights reserved.

Keywords: environment; trace elements; Pench Valley; coalfield; water; human health

1. Introduction

In India it has been realized in recent years that the environment of coalfield areas is deteriorating at a very fast rate due to enhanced production of coal each year. It has been

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established by workers such as Stutzer (1940), Swaine (1962, 1972, 1977), Eskenazi (1977, 1980), Ruch et al. (1974), Gluskoter et al. (1977), Chou et al. (1982, 1983) and Bouska (1981), that most coals contain measurable quantities of many metals (trace elements). In the pre-mining stage these metals are immobile, but with the initiation of mining activities, they tend to be dispersed into the surrounding environment and contaminate the atmosphere, soil, and surface and ground water resources of the area.

Human health is the state of adjustment by an organism to its own internal environment and to its external environment. The relationship between the geologic environment and regional and local variations in chronic diseases such as cancer and heart disease has been observed for many years (Keller, 1979). Evidence suggesting an association between the geochemical environment and chronic diseases continues to accumulate.

Human health may be affected by the amount of certain chemical elements available from food, drinking water and the atmosphere. In developed countries, the relationship between local geochemistry and human intake of these chemicals in food and water may be of low order. On the contrary, in developing countries, where local communities live closer to the land, people are more exposed to their geochemical environment and suffer many health problems (Bowie and Thornton, 1985).

2. Area description

The Pench Valley coalfield occupies the southern portion of the Satpura-Gondwana basin, at the eastern extremity of the basin, and is located in the Chhindwara districts of Madhya Pradesh (Fig. 1). It extends over a length of about 32 km from Sukri in the west to Haranbhata in the east. Geologically, the area is comprised of Archaean granite and gneisses, the Talchir, Barakar and Motur formations of the Lower Gondwana Group and the Jabalpur formation of the Upper Gondwana Group, followed by Deccan Trap lava flows of Cretaceous–Eocene age. The Pench Valley coalfield comprises about 13 underground and 4 opencast working coal mines in the area, whereas 6 mines are abandoned. These working mines are producing about 181,000 tons of coal per month and are discharging about 10,826.7 kl of mine water per day on the surface (Environmental Audit Report, 1993).

In the Pench Valley area, there occur about five seams with their split sections. The thickness of the seam/sections varies from 0.04 m to 3.39 m. These coal seams are confined to the Barakar sediments of the Lower Gondwana Group and are of Lower Permian age. The produced coal is non-coking grade, in which moisture varies from 4.0 to 10.50%, ash ranges from 15.0 to 32.00%, volatile matter 24.0 to 32.00%, fixed carbon 44.0 to 51.00%, with useful heat value (UHV) 4900–5900 k cal/kg at 60% RH and 40°C.

Coal production in the Pench Valley area has shown a three-fold increase in a decade or so. Due to increased production, the environmental quality in the valley has greatly deteriorated and hence, air, water and soil resources of the region are contaminated either by coal dust or by heavy metal pollution from coal utilization. The coal mining

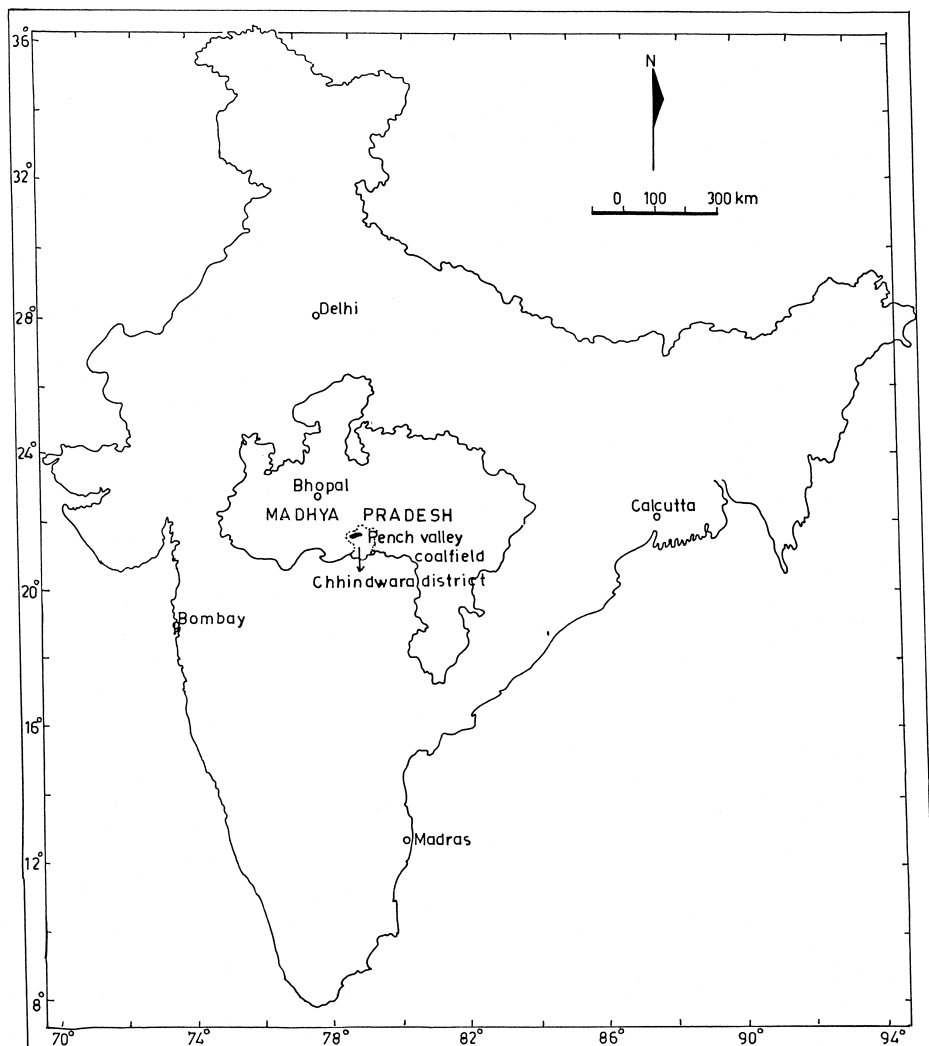


Fig. 1. Location map of the study area.

activity together with coal handling and vehicular emission has produced a large quantity of dust in the area. The suspended particulate matter (SPM) concentration ranges from $232.0 \mu\text{g}/\text{m}^3$ to $445 \mu\text{g}/\text{m}^3$, showing the highly contaminated nature of the atmosphere (Environmental Audit Report, 1993).

The Pench River is the main source of water in the area for industrial, agricultural, and domestic purposes. A huge amount of mine water is discharged in the Pench River. The mine water is mixed with river water and after treatment (slow sand filtration method) supplied to the public for drinking purposes. Due to extensive mining, the

ground water table has been depleted, therefore, the people of the area are entirely dependent upon surface water resources.

3. Methodology

Coal samples have been collected from all working phases of the Eklehra, Rawanwara Khas, and Shivpuri mines, and mixed thoroughly to make one composite sample for each mine. The coal samples were ashed at 450°C by the procedure given by Stadnichenko et al. (1953). The samples were spectrographically analyzed using the total energy method of Slavin (1938). By using a synthetic base, standards of different concentration are prepared by solid dilution techniques. The trace element content of coals of the area is given in Table 1. In order to work out the environmental impact of coal, 35 water samples have been collected, which include 14 samples of the Pench River, 13 samples of ground water and 8 samples of mine water. The water samples were analyzed by atomic absorption spectrophotometry (AAS; model GBC-902, double beam) in the laboratory of the Centre of Mining Environment, Indian School of Mines, Dhanbad. The degree of precision of analytical data is worked out by preparing two standards having absorption similar to the sample in the range 0.4 to 0.6 Abs. Zero position is set by using the low standard and 1.00 position with high standard. The observed difference between these two standards is about 0.1 Abs indicate the degree of accuracy of 0.1% to 0.2% of the result. The results of chemical analyses of natural waters in the area are given in Table 2.

4. Results and analyses

The chemical analysis of the three coals from the Pench Valley shows that trace elements like Co, Cu, Cr, Mn, Ni and Pb are present and their concentration ranges from 17–36.5 ppm, 66–105 ppm, 55–58.5 ppm, 745–935 ppm, 59–78 ppm and 18.5–22 ppm, respectively. Chemical analyses of natural waters give measurable contents of As

Table 1
Chemical analysis of coals of Pench Valley coalfield area

S. no.	Name of the mine	Nature of coal sample	As*	Co*	Cu*	Cr*	Mn*	Ni*	Pb*
1	Shivpuri open cast mine	Composite	BDL	36.5	66	58.5	919	74	18.5
2	Rawanwara Khas colliery	Composite	BDL	19	105	57	745	78	28
3	Eklehra colliery	Composite	BDL	17	72	55	935	59	22
Average				24.17	81	56.83	866.33	70.33	22.83

* Indicates all values in ppm on coal ash basis.

Table 2

Chemical analysis of natural waters of Pench Valley coalfield area. Figures in bold show the concentration above the desirable limits as per ISI: 10500 (1983) specifications

S. no.	Sample no.	Type of water	pH	As*	Co*	Cu*	Mn*	Ni*	Pb*
1	PV-1	MW	7.3	BDL	BDL	0.469	0.107	BDL	BDL
2	PV-2	MW	7.4	BDL	BDL	1.537	2.996	BDL	0.194
3	PV-3	MW	7.3	BDL	0.010	1.390	3.920	BDL	BDL
4	PV-4	MW	7.2	BDL	0.054	0.662	3.875	BDL	BDL
5	PV-5	MW	7.5	BDL	BDL	1.266	0.263	BDL	BDL
6	PV-6	MW	7.6	BDL	BDL	0.872	0.029	BDL	BDL
7	PV-7	MW	7.8	BDL	0.036	0.234	BDL	BDL	BDL
8	PV-8	MW	7.7	BDL	BDL	1.954	0.094	BDL	0.095
9	PV-9	DWW	7.1	BDL	0.019	BDL	0.028	BDL	BDL
10	PV-10	DWW	7.1	0.400	BDL	0.251	0.020	BDL	BDL
11	PV-11	DWW	7.2	0.360	BDL	0.394	0.047	BDL	BDL
12	PV-12	DWW	7.0	BDL	0.015	0.721	BDL	BDL	BDL
13	PV-13	DWW	7.3	BDL	BDL	BDL	BDL	BDL	BDL
14	PV-14	DWW	7.3	BDL	BDL	BDL	0.039	BDL	0.121
15	PV-15	DWW	7.2	BDL	BDL	0.483	0.035	BDL	0.131
16	PV-16	TWW	7.4	BDL	0.012	0.595	0.034	BDL	0.074
17	PV-17	HPW	7.3	0.067	0.013	1.185	0.035	BDL	BDL
18	PV-18	HPW	7.2	BDL	BDL	1.028	0.656	BDL	0.155
19	PV-19	HPW	7.3	BDL	0.052	BDL	0.137	BDL	0.014
20	PV-20	HPW	7.5	BDL	BDL	BDL	0.026	BDL	BDL
21	PV-21	HPW	7.2	0.580	BDL	0.494	0.022	BDL	BDL
22	PV-22	PRW	6.8	0.550	0.010	0.335	0.016	BDL	0.134
23	PV-23	PRW	6.5	0.180	BDL	BDL	0.023	BDL	BDL
24	PV-24	PRW	6.8	BDL	BDL	0.218	0.028	BDL	BDL
25	PV-25	PRW	7.1	BDL	BDL	BDL	BDL	BDL	0.035
26	PV-26	PRW	6.5	BDL	BDL	0.159	0.618	BDL	BDL
27	PV-27	PRW	6.8	BDL	BDL	0.092	0.016	BDL	BDL
28	PV-28	PRW	7.0	BDL	BDL	0.469	0.038	BDL	BDL
29	PV-29	PRW	7.5	BDL	0.014	0.402	0.108	BDL	0.534
30	PV-30	PRW	7.4	BDL	BDL	0.992	0.024	BDL	0.130
31	PV-31	PRW	6.0	BDL	BDL	0.664	0.070	BDL	0.681
32	PV-32	PRW	6.5	BDL	BDL	1.425	0.018	BDL	0.089
33	PV-33	PRW	6.8	BDL	BDL	0.963	0.014	BDL	0.072
34	PV-34	PRW	7.2	BDL	BDL	0.574	BDL	BDL	BDL
35	PV-35	PRW	7.2	BDL	BDL	BDL	BDL	BDL	BDL

*Indicates all values in ppm. BDL = below detection limit, MW = mine water, DWW = dugwell water, TWW = tubewell water, HPW = handpump water, PRW = Pench River water.

in 17% of the samples, Co in 29%, Cu in 77%, Mn in 83% and Pb in 40% of the samples. In natural water, the As contents range from 0.067 to 0.580 ppm, Co varies from 0.010 to 0.054 ppm whereas Cu, Mn, and Pb ranges from 0.092 to 1.954 ppm, 0.014 to 3.920 ppm and 0.014 to 0.681 ppm, respectively. However, Ni is found to be below the detection limit in all the water samples of the area. The data reveal that the trace element quantities in the majority of sites are above the drinking water specifications.

Table 3

Drinking water quality criteria as proposed by World Health Organization (WHO), US Public Health Service (USPHS), South African Bureau of Standards (SABS), Russia (USSR), US National Academy of Sciences (NAS), Australia, Japan and the Environmental Protection Agency (EPA) of the USA, compiled by Hattingh (1977), except for FRG data (Schottler, 1977) and ISI: 10500 (1983)

Parameters	USPHS (1962)	Japan (1968)	USSR (1970)	WHO European (1970)	WHO Int. (1971)	SABS (1971)	NAS (1971)	Australia (1973)	USEPA (1975)	FRG (1975)	ISI: 10500 (1983)
As	0.01	0.05	0.05	0.05	0.05	0.05	0.10	0.05	0.05	0.04	0.05
Ba	1.0	–	4.0	1.0	–	–	1.0	1.0	1.0	–	–
Cd	0.01	–	0.01	0.01	0.01	0.05	0.01	0.01	0.01	0.006	0.01
Cr	0.05	0.05	0.10	0.05	–	0.05	0.05	0.05	0.05	0.05	0.05
Cu	1.0	1.0	0.1	0.05	0.05	1.0	1.0	10.0	–	–	0.05
Pb	0.05	0.10	0.10	0.10	0.10	0.05	0.05	0.05	0.05	0.04	0.05
Mn	–	–	–	–	–	–	–	–	–	–	0.10
Hg	–	0.001	0.005	–	0.001	–	0.002	–	0.002	0.004	0.001
Se	0.01	–	0.001	0.01	0.01	–	0.01	0.01	0.01	0.008	0.01
Ag	0.05	–	–	–	–	–	–	0.05	0.05	–	–
Zn	5	0.1	1.0	5	5	5	5	5	–	2	5

5. Discussion

Study of the literature reveals that there is great variation in the trace element concentrations in coals of different parts of the globe. These differences exist due to differences in the factors controlling the dispersal and accumulation of chemical elements in coal forming basins. A review of the literature on coals reveals that nearly every element has been found in coal. The concentration of these elements in coal ash ranges from parts per trillion to more than 50 wt.% (Finkelman, 1993). Mukherjee et al. (1982) have reported a range of trace elements from major Tertiary and Gondwana coalfields of India.

Wood (1974) has classified the metals such as As, Co, Cu, Ni and Pb into the category very toxic and relatively accessible, from environmental pollution point of view. Sahoo (1991) suggests that trace elements such as Co, Cd, Mn, Ni and Zn are present in high concentration in respirable coal mine dust. Chemical analysis of coals of the Pench Valley reveals that trace elements such as Co, Cu, Cr, Mn, Ni and Pb are also found in coals of the area (Table 1).

Coal mining and related activities such as coal transportation, burning of coal, and subsequent fallout of ash has dispersed trace elements into different parts of the environment. The analytical results reveal (Table 2) that As, Co, Cu, Mn, and Pb concentrations in many waters are above drinking water specifications (Table 3). It is believed that concentration of these elements has increased due to coal mining and combustion in the area rather than other anthropogenic activities.

Workers such as Zubovic et al. (1979, 1980), Bouska (1981), Swaine (1985) and Coleman et al. (1986) have reported appreciable concentration of arsenic in coals from different parts of the globe. No general account of arsenic concentrations in Indian coal is available. Chemical analyses of Pench Valley coal reveal that arsenic concentration is below the detection limit. On the contrary, 17% of natural water samples contain arsenic at concentrations ranging from 0.067 to 0.580 ppm. The sample locations (PV-10, PV-11, PV-17, PV-21, PV-22 and PV-23) are randomly distributed and represent two samples each of handpump, dugwell and Pench River water. Coleman and Bragg (1990) suggest that arsenic has a predominant inorganic affinity, therefore, it is associated with the sulfide minerals present in the coal. The most common sulfide minerals that occur in coal are pyrite, sphalerite, galena, chalcopyrite and pyrrhotite (O’Gorman and Walker, 1971; Finkelman, 1981; Davis et al., 1984; Harvey and Ruch, 1986). Field study reveals no prominent association of pyrite or other sulfides within the coal seams. Newland (1984) suggests that arsenic is released into the environment due to the burning of fossil fuels. However, human activities greatly influence the amount of arsenic in the environment.

Arsenic in natural water occurs in four oxidation states, which exist as different species under various conditions. In general, most forms of arsenic are soluble under oxidizing conditions. The movement of arsenic in the aquatic environment is affected by sorption to soils. However, sorption increases with increasing pH, clay and hydroxide content (Aller et al., 1987). The fact that arsenic levels in mine waters are below detection limit suggests that the arsenic content in coal itself is low, or the clay content

available for the attenuation of arsenic in mine water is low. The presence of arsenic in some surface and sub-surface waters suggests that arsenic is not derived by coal combustion in the area, rather, the high concentration of arsenic is due to human activities such as the use of fertilizers, insecticide and discharge of sewage water into the Pench River. Anastasia and Kender (1975) have shown that arsenic can accumulate in the soil when arsenic compounds are repeatedly applied to crops. Appreciable amounts of arsenic could also move down in the soil profile with the percolating soil water (Richardson et al., 1978). Such activities, locally enhance the arsenic level in surface and sub-surface water, which then can be sorbed onto clay/organic colloids and increase the arsenic concentration.

The cobalt concentration in Indian coals is reported by many workers. Rishi (1970) estimated a concentration of 10–100 ppm cobalt in Umaria coal ash of M.P. The cobalt content of Lower Gondwana coals varies from 10 to 70 ppm, and in northwest Indian coals, it ranges from 18 to 80 ppm (Mukherjee et al., 1982). Pareek and Bardhan (1985) determined cobalt concentrations of 10–120 ppm from East Bokaro coalfields of India. In Pench Valley coals, the cobalt concentration ranges from 17 to 36.5 ppm. Cobalt occurs in 29% of water samples and its concentration varies from 0.010 to 0.054 ppm. Thomson (1977) suggests that mobility of cobalt ions in the aquatic environment is significantly lower in the presence of sulfide ions in pore waters that fix the cobalt under reducing conditions. However, Förstner and Wittmann (1981) suggest that enrichment of cobalt in natural water occurs with decreasing pH values. The pH value of natural waters of the study area varies from 6–7.8, which may determine the low concentration of cobalt. However, the variation in cobalt content from 0.54 ppm to below the detection limit may be due to the presence of varying chemical conditions and sorption of cobalt on clay minerals in aquatic environment.

Indian coals have high concentrations of copper. Mukherjee et al. (1982) reported copper concentrations of 27–321 ppm and 5–1264 ppm in Lower Gondwana and Tertiary coals of India, respectively. Singh et al. (1983) reported copper concentrations of 110–325 ppm in the Ghugus coalfield of Wardha valley. The copper concentration in Pench Valley coal ranges from 66 to 105 ppm. Copper contamination of water resulted from the discharge of mine tailings and flyash (Krishnan, 1995). Copper is present in 77% of natural water samples and varies from 0.092 to 1.945 ppm. This value is well above the drinking water specifications. The analytical results reveal the copper concentration in mine water is high, followed by surface and sub-surface water. Aller et al. (1987) suggest that the sorption of copper on suspended matter is a function of pH, and sorption increases at high pH values. Hart and Davies (1977) suggest that copper ions in bulk water are present in the filterable fraction of labile form, while in interstitial water, copper is evenly distributed between labile species and more strongly bound complexes. The study reveals that pH of mine water is slightly higher than natural surface and subsurface waters, which explains the higher copper content in mine waters. However, variation of copper concentration in surface and sub-surface waters suggests that copper ions occur in the labile phase and form labile species on suspended sediments in river water, while in ground water, the organic matter present in the soil forms stable complexes with copper (Brown et al., 1983), and provides cation exchange on organic and inorganic substances.

The manganese concentration of Indian coals varies widely. Rishi (1970) reported 80–1000 ppm manganese in coal ash from Shahdol. Chandra et al. (1981) found 50–150 ppm manganese in coal ash of G-III seam of the Korba coalfield. Mukherjee et al. (1982) reported 20–1240 ppm manganese in coal ash of Lower Gondwana coalfields. Pareek and Bardhan (1985) recorded 20–1000 ppm manganese in coal ash of the East Bokaro coalfield. The manganese concentration in Pench Valley coal ranges from 745 to 935 ppm. The manganese is mobilized due to the burning of coal in the environment (Krishnan, 1995). However, manganese-bearing fertilizers also contribute to water pollution. The manganese concentration of natural waters in the area occurs in 83% of water samples and ranges from 0.014 to 0.618 ppm. The study reveals that the manganese concentration in five samples of mine water and two each of surface and sub-surface water is higher than the drinking water specifications.

The chemistry of manganese in the aquatic environment is complex due to the different oxidation states of manganese, however, manganese is a predominant participant in aquatic redox processes (Förstner and Wittmann, 1981). Mn^{+2} is soluble and mobile, while Mn^{+4} is insoluble and immobile. Lee (1975) suggests that oxidation of manganese led to an increase in the adsorption of Mn^{+2} from solution. Hem (1964) pointed out that oxidation of Mn^{+2} is influenced by inorganic ions such as HCO_3^- and SO_4^{2-} . Microorganisms also enhance the oxidation of Mn^{+2} (Schweisfurth, 1972). Given the complex chemistry of manganese, it is difficult to work out whether oxidation or reducing conditions are responsible for the variation in manganese concentrations in natural waters of the area. It is, therefore, suggested that variation in manganese concentrations may be attributed to both of the above conditions. Under reducing conditions, the Mn^{+2} is strongly sorbed onto clay minerals and organic matter, and become less soluble as pH increases, while under oxidizing conditions, several stable manganese compounds are formed (Brown et al., 1983). Biologically initiated oxidation probably causes a downward depletion of oxygen in sediments which may lower the pE value; as a consequence, the concentration of soluble Mn^{+2} increases at the expense of Mn^{+4} with depth (Stumm and Morgan, 1970).

In Indian coals, Rishi (1970) reported 60–180 ppm nickel in coal ash of the Umaria coalfield. Mukherjee et al. (1982) have recorded 250 ppm and 754 ppm nickel in coal ash from Rajmahal basin, and Tertiary coals of northeast India, respectively. In Pench Valley coal, nickel concentrations range from 59 to 78 ppm. Analytical results for natural waters show that nickel is below detection limit in all the water samples of the area. Nickel tends to be mobilized into the environment due to the burning of fossil fuels (Dara, 1993). Studies related to sorption of nickel in the aquatic environment are less well known. However, Swanson et al. (1966) suggest that organic matter can adsorb between 1–10% dry weight of nickel. The experimental data of Lu and Chen (1977) suggest that migration of nickel under reducing conditions in interstitial water is controlled by an organic complex for nickel. The presence of humic substances in aquatic water suggests that nickel tends to concentrate under reducing conditions. The aforementioned study points out that nickel concentrations of natural water are dependent on the availability and non-availability of organic substances under reducing conditions. The nickel content below detection limit in natural waters of the area, most

probably attributed to the high oxidation state, do not allow to set up reducing condition to enrich nickel in available humic substances.

Chandra et al. (1981) have found that the lead concentration in coal ash of the Korba coalfield varies from traces to 100 ppm. Mukherjee et al. (1982) reported 8–89 ppm lead in coal ash of the Lower Gondwana coalfield and 5–280 ppm and 15–45 ppm lead in Tertiary coals of northeast and northwest India, respectively. The lead concentration in Pench Valley coal ranges from 18.5 to 28 ppm. The main source of lead contamination in the aquatic environment is from atmospheric fallout due to fossil fuel combustion (Pillai, 1983). Klein and Andrew (1975) found that lead-enriched flyash is not collected by electrostatic precipitators and generally appears as fallout. Other sources of lead contamination include industrial discharge, vehicular exhaust on the roadside and combustion of lead products. Study of analytical data for natural waters in the area reveals that lead occurs in 40% of water samples and ranges from 0.72 to 0.681 ppm. Hildebrand and Blum (1974) suggest that pH influences sorption of lead onto organic matter. At alkaline conditions, Pb^{+2} is less soluble and is preferentially sorbed onto clay surfaces. Under reducing conditions, lead becomes mobile and may form insoluble complexes with organic compounds (Brown et al., 1983). Thus, the variation in lead concentration in mine and ground water is primarily controlled by the pH of solutions, perhaps allowing sorption of Pb^{+2} ions onto available clays. However, a systematic variation has been observed in the lead concentration of Pench River water. At sample points PV-22, PV-29, and PV-31, the lead values are 0.134 ppm, 0.534 ppm, and 0.681 ppm, respectively. These values gradually decrease as the river water flows down stream. It is important to mention here that PV-22, 29 and 31 are the point of mine water discharge, therefore, high concentrations of lead have been formed in these samples. The gradual dilution of Pb^{+2} with flow direction may be accounted for by a change in pH of the solution and in lead speciation, forming stable complexes on clay or organic matter available in river water of the area.

5.1. Potential health effects

Modern research has shown that trace elements are essential for normal growth in plants and human beings. It has been established that the role of heavy metal ions in living systems follows natural availability (Willams, 1967; Vahrenkamp, 1973; Wood, 1975). A metal in trace amounts (smaller than 0.01% of the mass of the organism) is essential when an organism fails to grow or complete its life cycle in the absence of that metal. However, the same trace metal is toxic when concentrations exceed those required for correct nutritional response (Venugopal and Luckey, 1975). Therefore, studies pertaining to the toxicity of trace metals follow the general trend that an undersupply leads to a deficiency, sufficient supply results in optimum conditions, but an oversupply produces toxic effects, followed by lethality (Förstner and Wittmann, 1981). Therefore, an attempt has been made to know the potential health problems, associated with the above elements found in natural waters of the area.

There are about 25 trace elements (e.g., Al, Sb, Hg, Cd, Ge, V, Si, Rb, Ag, Pb, Bi and Ti) which are not needed for human growth, development, achievement and reproduction, and occur more or less constantly in living tissues. These elements are

believed to be acquired by the body as environmental contaminants due to contact with the environment. Therefore, trace quantities of these elements are already available in the human body.

Trace elements find their way into the human body either by direct absorption, via the air (inhaling) or drinking water, or via the food chain. Thus, even a small additional 'pollution' component can pose a serious health problem under some circumstances. However, the hazardous impacts of these elements are dependent upon the period of exposure, concentration and quantity of material used. In the foregoing discussion, an attempt has therefore been made to discuss the potential health hazard problems related to trace elements when they accumulate in the human body.

Arsenic is widely distributed in the biosphere and occurs in the air wherever coal is burnt. Natural release of arsenic on a global scale is estimated to be about 8×10^4 metric tons per year, whereas human activities account for about 24×10^4 metric tons of arsenic per year (Dara, 1993). The tolerance limit of arsenic in drinking water is 0.05 ppm (Table 3) and the available arsenic concentration in natural waters of the area is high and range from 0.067 to 0.58 ppm. Arsenic is a protoplasmic poison that affects the cardiovascular system, gastrointestinal tract, kidney, nervous system, skin, blood and liver.

Pillai (1983) suggests that the toxicity of arsenic in trivalent form is higher than that of pentavalent form. However, the experimental evidence has shown a high degree of gastrointestinal absorption of both trivalent and pentavalent forms of arsenic (Tam et al., 1979). Arsenic is stored mainly in the liver, kidney, lung and heart. Arsenic readily crosses the placental barrier and causes fetal damage (Klaassen, 1991). The acute hepatocellular injury, anemia, developmental disabilities, embrotoxicity, heart disease, hyperpigmentation and peripheral neuropathies are some typical human health effects that may be produced due to arsenic toxicity (Asante-Duah, 1993). Small doses of inorganic arsenic induce mild vasodilatation, splanchnic hyperemia, hematemesis, renal damage, encephalopathy and bone marrow in blood. However, long term exposure may cause myocardial damage, hypertension, milk and rose complexion and anisocytosis of blood (Klaassen, 1991).

The chronic ingestion of arsenic causes muscle weakness and aching, skin pigmentation (especially of the neck, eyelids, nipple and axillae), hyperkeratosis, edema and peripheral arteriosclerosis, commonly known as 'blackfoot disease' (Klaassen, 1991). Humans exposed to arsenic-containing dust may suffer non-allergic contact dermatitis and conjunctivitis, whereas continued inhalation of arsenic dust may cause perforation of nasal septum (Dara, 1993). There is overwhelming epidemiological evidence that long-term ingestion of arsenic in drinking water may be teratogenic to animals and carcinogenic to humans.

Cobalt is an essential element for humans, plants and animals because it is associated with the synthesis of vitamin B₁₂. There is very little information about the environmental status of cobalt and related health hazards. However, the recommended maximum concentration of cobalt in irrigation waters is 0.05 mg/l (Ayers and Westcott, 1976) and in livestock drinking water it is 1.0 mg/l (NAS, 1972). The maximum permissible limits of cobalt in drinking water for human consumption are not available. However, studies suggest that ingestion of an excessive amount of cobalt causes polycythemia and

intercellular hypoxia. Chronic exposure to cobalt at higher concentrations produced goiter (Krishnan, 1995).

Copper is also an essential element for normal biological activities in humans. Sources of atmospheric copper include burning of coal, fertilizer and iron/steel production. Water pollution by copper results from the discharge of mine tailings and flyash (Krishnan, 1995). WHO (1972) recommends 0.05 ppm as the maximum tolerance limit for copper. The copper concentrations in 77% of waters of the study area are above the safety limit. A daily dietary intake of 2 to 3 mg of copper is recommended for human adults.

Copper is mainly absorbed from the duodenum in man and its retention depends upon the chemical form in which the metal is ingested. Ingestion of 15–75 mg of copper a day causes gastrointestinal disorders. Continuous ingestion of copper from food and water may cause chronic copper poisoning and a disease called heptolenticular degeneration or 'Wilson's disease' may develop. In addition, excessive intake of copper may cause hemolysis, heptotoxic and nephrotoxic effects. Other conditions which manifest increased copper contents in liver include thalassemia (Mediterranean anemia), hemochromatosis, cirrhosis, atrophy of the liver, tuberculosis and carcinoma (Dara, 1993). However, impairment of the ability to absorb copper resulting in its deficiency is called Manke's disease (Krishnan, 1995). Inhalation of airborne copper causes irritation of the respiratory tract and metal fume fever.

The presence of chromium in the air is mostly due to burning of fossil fuels, which contribute about 1450 metric tons of chromium every year. Chromium occurs in several valance states. Only trivalent and hexavalent forms are biologically important. Hexavalent chromium is 100 times more toxic than its trivalent form (Krishnan, 1995). Trivalent chromium is also known to be toxic, therefore, the limits for chromium in water are based on total chromium contents rather than on hexavalent chromium alone (Schroeder and Lec, 1975). The US EPA (1976) suggests a total chromium limit of 0.05 ppm in drinking water. The daily dietary intake of chromium in India is about 150 μg per person. Chromium deficiency is characterized by impaired growth and longevity and by disturbance in glucose-lipid and protein metabolism (Dara, 1993).

Orally administered trivalent chromium is not as readily absorbed as the hexavalent form. Absorbed anionic hexavalent chromium readily passes through the membrane of red blood cells and binds to the globin fraction of hemoglobin, whereas cationic trivalent chromium is unable to pass through this membrane. Exposure to hexavalent chromium causes allergic skin irritations, dermatitis, irritation to mucous membranes and conjunctiva and gastrointestinal ulcers, chrome holes, i.e., penetrating ulcer which occurs around fingernails, eyelid and occasionally on forearms. Lesions on the nasal mucosa may lead to perforation of the septum (Dara, 1993). Chronic exposure to chromate dust has been correlated with increased incidence of lung cancer. Hexavalent chromium may be carcinogenic, teratogenic, and mutagenic.

Manganese is an essential trace element at low concentrations but toxic at higher concentrations. Manganese is introduced into the environment due to the burning of coal (Krishnan, 1995). The use of manganese in some fertilizers also contributes to water and air pollution (Dara, 1993). In 31% of natural water samples, the manganese concentration is above the recommended limit (Table 3). A daily allowance of 2.5–5 mg of

manganese is recommended for humans, which acts as a cofactor in enzymatic reactions such as phosphorylation, synthesis of fatty acids and cholesterol. However, when exposed to higher levels of manganese, it accumulates in the kidney, liver and bones. Some typical health effects that may develop in human include cirrhosis, influenza, (metal-fume fever), and bronchitis (Asante-Duah, 1993). Chronic exposure to manganese causes 'Manganese Psychosis', which is an irreversible brain disease characterized by uncontrollable laughter, euphoria, impulsiveness, sexual excitement followed by impotency and speech disability (Dara, 1993). The inhalation exposure to manganese causes another fatal disease called 'Manganese Pneumonia' (Krishnan, 1995). However, there is no conclusive evidence on the causal relationship between manganese pollution and prostate cancer, teratogenic and mutagenic effects of these elements.

Nickel is mainly available in the coalfield environment due to burning of coal. Emission of nickel from fossil fuels amounts to about 70,000 tons per year (Dara, 1993; Krishnan, 1995). Nickel is an essential micronutrient for some microorganisms and animals, but not to plants. It is associated with the synthesis of vitamin B₁₂. It is toxic at higher concentrations (Dara, 1993). A recommended daily dietary intake of nickel is about 300–600 µg per day. Only 1–10% of the dietary ingested nickel is absorbed (Dara, 1993). Asthma, CNS effects, gastrointestinal effects, headache, neoplasia of lung and respiratory tract are some common health effects attributed to nickel exposure (Asante-Duah, 1993).

One of the most dangerous sources of nickel pollution is nickel carbonyl, which is the airborne effluent from industrial units. Initial symptoms of toxicity due to nickel carbonyl are nausea, dizziness, headache, and chest pain. After 1 to 5 days, severe pulmonary symptoms, tachycardia and extreme weakness develop and even death may occur within 4 to 13 days. Nickel dust is reported to be carcinogenic.

The natural dispersal of lead is somewhat restricted due to its insolubility. In coalfield areas, the major source of lead pollution is atmospheric fallout due to fossil fuel combustion. However, other anthropogenic sources contribute a major fraction to the aquatic environment and atmosphere. The overall human exposure to lead is primarily from food; however, most of the overt toxicity of lead results from fossil fuel and industrial exposure. ISI: 10500 (1983) and US EPA (1975) have given a maximum permissible limit of 0.05 ppm for lead in drinking water (Table 3). It is estimated that 34% of samples in the area are contaminated with lead.

Estimated typical daily intake ranges from 100 to over 500 µg of lead (WHO, 1972; US EPA, 1977, 1979). The worldwide average for an adult is about 200 µg/day. The positive lead balance begins at a daily intake of 0.6 mg. This amount will not ordinarily produce overt toxicity within a lifetime. However, the time required to accumulate toxic amounts shortens disproportionately as the amount ingested increases. For example, a daily intake of 2.5 mg of lead requires 4 years to reach a toxic burden, whereas a daily intake of 3.5 mg only requires a few months (Klaassen, 1991).

The major routes of absorption of lead are the gastrointestinal tract and the respiratory system. Gastrointestinal absorption of lead varies with age: adults absorb approximately 10% of ingested lead while children absorb up to 40%. Absorption of inhaled lead varies with the form (vapors vs. particles) as well as with concentration. Approximately 90% of inhaled lead particles from ambient air are absorbed (Goyer, 1985). If a

large amount of lead is absorbed rapidly, a shock syndrome may develop due to loss of gastrointestinal fluids. An acute hemolytic crisis sometimes occurs and causes severe anemia and hemoglobinuria. The kidneys are damaged and oliguria and urinary changes are evident and death may occur in 1 or 2 days. Acute central nervous system (CNS) symptoms such as paresthesias, pain and muscle weakness develop due to acute lead poisoning.

Chronic lead poisoning often begins with vague symptoms such as anorexia, muscle discomfort, malaise and headache as well as gastrointestinal effects. The chronic CNS effects (lead encephalopathy) may include symptoms like clumsiness, vertigo, ataxia, headache, insomnia, restlessness and irritability. As the encephalopathy develops, the patient may first become excited and confused, lethargic followed by coma.

The hematological effects of chronic lead intoxication are hypochromic microcytic anemia. The renal chronic toxicity occurs in two forms—a reversible renal tubular disorder and an acute irreversible intestinal nephropathy (Goyer, 1985). Clinically, a 'Fanconi' like syndrome is seen with proteinuria, hematuria, and casts in urine (Craswell, 1987; Bernard and Becker, 1988). Other signs and symptoms of plumbism are an ashen color of the face and pallor of the lips, retinal stippling, appearance of 'premature aging' with stooped posture, poor muscle tone, emaciation, and a black or grayish so-called lead line along the gingival margin (Klaassen, 1991). Todd et al. (1996) suggest that lead can hypothetically be mobilized from the skeleton and increase bone mineral turnover, which affects pregnancy, lactation, menopause, and produces hyper-metabolic states. Lead mobilization during pregnancy is potentially very hazardous to the fetus.

The aforementioned discussion demonstrates many potential health problems related to toxicological concentrations of various trace elements. All of the above problems are, however, not known to occur in the Pench Valley area, but high concentration of trace elements found in some coals could be a major source of these elements in the coalfield environment. These elements in turn find their ways into natural water and finally become ingested, and may affect the public health in the Pench Valley area. Chronic exposure to toxic concentration of these elements will be very severe, the cumulative effect of which may produce health hazards in the area.

6. Conclusions

The above study suggests that in Pench Valley coal, trace elements such as Co, Cu, Cr, Mn, Ni and Pb are present. The extensive mining and burning of coal disperses these elements into the environment and contaminates the surface and ground water resources of the region. Concentration of As, Cu, Mn, and Pb in some of the samples of surface and sub-surface waters are above the desirable limit of drinking water specifications. The variation in the concentration of these elements may be due to chemical speciation, pH of solution, sorption onto clay and organic matter, cation exchange capacity and relative solubility of different ions under various chemical conditions. The ingestion of these elements either through inhalation or orally allows them to accumulate in the body, and the cumulative effect of these elements over a long time of exposure may produce

gastrointestinal disorder, cardiovascular disease, pneumoconiosis, carcinogenic, teratogenic and mutagenic effects and other diseases as discussed above.

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