

Geochemical migration of arsenic and its environmental effect

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Abstract—This paper presents the results of a field study regarding the transport and environmental fate of arsenic. Concentrations of arsenic and various forms of its compounds in rock, soil, river water, and river bottom deposits in an area near Hangzhou, China, were measured and analyzed. On the basis of these field data, the source and interface transfer of arsenic were discussed. A few very interesting findings were reported. This paper indicates that there was relatively low arsenic concentration in Jing-Hang Canal, whereas, rather high arsenic concentration in the river bottom mud was found. The paper attributed this phenomenon to mechanism of arsenic deposition and absorption in the water column, the canal has a rather high self-purification ability relative to the arsenic input.

Keywords: arsenic; geochemical migration; environmental effect.

1 Introduction

Arsenic is a considerably complex element. As for its ionic character, arsenic not only can form cation or anion, but can transform from cation or anion to neutral atom. Therefore, it exists in nature as various species. Arsenic and its compounds are used widely in agriculture, pottery and glass industries, and even in medicine. However, it is well known that As is toxic to human health. In the course of natural circulation, As may enter into the ecosystem and ultimately affect human health. Therefore, it is important to study the geochemical migration of As and its environmental effect for understanding the laws of As migration and transformation in nature and for protecting the environment against As pollution.

The behavior and circulation of As in local natural ecosystems, such as water system, soil and plants, have been reported (Ferguson, 1972; Xiaoshanxiangsen, 1975; Yang, 1986). However, the geochemical behaviors of As are synthetically affected by various-natural and man-induced factors. The geochemical migration of As in natural bodies is a continuous process. The properties of As in every natural body will exert an imperceptible influence on environment. In this paper, with the area north of Hangzhou in China as the suitable area of study, the geochemical behaviors of As in rock, soil, river, ground water and the entire ecosystem are described focusing on As migration, transformation and accumulation. Also, As effects on environment and human body health are discussed.

2 Setting of the study and methods

The study area is located in southeast of China and consists of plains and hill lands; in topography, the southwest is higher than the northeast region. The plain area is covered by Quaternary

System with a thickness of 10 to 70 m. The studied shallow Quaternary System sediments, in depths of up to 15m, came from a single source and were deposited in Holocene. They were made up of fine-grained river lacustrine facies, limnological facies and littoral facies and littoral facies sediments. The water system includes the Jing-Hang Canal, following from south to north and has the width of 100m and the depths of 2 to 3 m. Such deposition conditions present many opportunities for chemical and physical fractionation, so that the suspended matter precipitates from the water to riverbed with various chemical and physical species. The biota in the river is too little to affect the matter migration. The river water chemical type is $\text{HCO}_3\text{-Cl-Ca-Na}$. The river flows through three different geographic environments including the densely populated city proper, the factory distributive region and agriculture fields. The ground water level in the plain area is about 0.5 m. The river and ground water are mainly supplied by atmospheric precipitation.

The cropping bedrocks in the hill area, included deposition rocks (e.g. sandstone, limestone and mud stone), volcanic rock (e.g. tuff) and magmatic rock (e.g. granite-porphyry), formed Sinian Period to Cretaceous Period. The fold and crack structures of rock were greatly development. The geochemical process in the hypogene zone was strong and the weathering denudation process was remarkable.

Rock, soil, water and sediment samples for this study were collected in the study area. The 64 bedrock samples were collected from the hill area at intervals of 250 m along the vertical direction of the stratum strike. The cropping area and thickness of stratum, and also the typical stratum with less thickness were considered during rock sampling. The 23 samples were collected from the weathering crust. The 53 Quaternary System sediment samples were collected from different depths of up to 15 m at 11 drill holes in the plain area. In addition, the 78 soil samples were respectively collected from upperlie and underlie at two depths of 0—20 cm and 20—40 cm after soil types were defined. All samples as mentioned above were analyzed for As concentration.

The river sediment and water volume samples were collected from 11 sites at intervals of similar distance as in Jing-Hang Canal. At every site, the sediment samples were collected from right, middle and left drill holes, and the sampling depths were from surficial riverbed to about 5 m. The water samples were collected at about 50cm after mixing to obtain a representative sample.

The quantity of coal burned and waste water draining from major factories in this area was investigated and estimated. As concentration and species in waste water were determined. In order to test the influence of As migration on the environment, the concentration and distribution of As in ground water on both sides of Jing-Hang Canal were evaluated, and the bacteriolytic enzyme in saliva of children who lived in a control area.

3 Results and discussion

3.1 Variation of As in various natural bodies

The background content of As in outcropping bedrock in the study area is higher [compared to that reported by Taylor (Taylor, 1964) who estimated that 1.8 ppm of As abundance in the crust], ranged from 2 to 72 ppm with an average of 9.2 ppm (Table 1). This value was derive from 64

samples collected in the southwest of study area. Original As contents in rocks are closely related to the volcanic effect and magmatic activeness (Goldschmit, 1931). The cropping volcanic and magma rocks in this area showed that volcanic effect and magmatic activeness had occurred in a past geological period, which resulted in the higher As background content in bedrock than the Taylor (Taylor, 1964) value. As concentrations are different in the various rocks (Table 1). On the basis of Turkian and Wedepohl's (Turkian, 1961) estimates, the average contents of As in limestone and sandstone were 1.1 ppm, whereas those in the current study area were 12 and 5.17 ppm, respectively.

Table 1 As concentration in various natural bodies

(Unit: ppm)

	Bedrock				Weathering crust			Quaternary system		
	Sandstone	Limestone	Tuff	Total	Sandstone	Limestone	Tuff	Total	Sediment	Soil
No. of samples	19	12	12	64	2	3	11	15	53	78
Range	2—12	3—59	3—33	2—72	5—10	10—960	3—14	3—960	3.45—19.8	1.33—15.25
Mean content	5.17	12	9.25	9.2	7.5	280	12.9	99.3	8.9	7.86

These findings significantly higher than the previous value of 1.1 ppm, particularly in limestone. Because of the stronger affinity between arsenic and sulphur, As may go in sulfides substituting for S. The pyritization of limestone in this area was obvious, which is probably the cause of the significantly higher As concentration in limestone. Arsenic concentration in the crust of weathering is remarkably higher than that in bedrock. The concentration ranged from 3 to 960 ppm with an average of 99.3 ppm, which indicates the stronger trend in As enriched in the process of weathering. However, there are clearly differences in the degree of As enriched in various crusts of weathering (Table 1). In contrast to the As concentration in bedrock, As strongly enriched in limestone weathering crust, and the degree of As enriched in sandstone weathering crust is not remarkable.

As may exist in the following forms in rock minerals: (1) natural arsenic and its alloy; (2) simple sulfides and oxides of As^{3+} ; (3) As^{5+} forms $[\text{AsO}_4]^{3-}$ and various As-bearing minerals combined with Fe^{3+} , Cu, Pb and Zn etc.; (4) As forms $[\text{AsS}_n]^{x-}$ combined with S and then form sulfuric minerals combined with heavy metals; (5) As^{3-} or As^{n-} may substitute for S^{2-} in minerals. However, when the rock minerals were weathered, As was transformed from lower valence into higher because of the thermodynamic condition change.

This resulted in As being decomposed from minerals and gradually becoming enriched in the crust of weathering with other dissolvable matter transported. It was known from Table 1 that As enriched degree was dominated by two factors (1) original As content in bedrock; and (2) the bedrock weathered degree. Under riched oxygen condition, iron pyrite in limestone was oxidated and transformed ultimately into $\text{Fe}_2\text{O}_3 \cdot n\text{H}_2\text{O}$ (Zhao, 1988). This was strongly enriched in the crust of weathering simultaneously S and As in iron pyrite were also oxidated and transformed into

SO_4^{2-} and As^{5+} , respectively. The SO_4^{2-} was easily washed away whereas As^{5+} was strongly adsorbed by $\text{Fe}_2\text{O}_3 \cdot n\text{H}_2\text{O}$, which was accounted for by the tremendous As concentration in limestone weathering crust. Tuff was significantly weathered as well as having higher original As content. Thus As enriched in the crust of weathering was obvious. Because of sandstone's relative stronger antiweather capacity and lower original As content, the trend in As enriched in sandstone weathering crust is not remarkable.

As average content in shallow Quaternary System sediment is 8.90 ppm and ranged from 3.45 to 19.8 ppm (Table 1). This value was derived from 54 samples collected from different depths at 11 drill holes. This value is significantly higher than the background content of As in the bedrock and is similar with that in the crust of weathering (average of 8.95 ppm excluding tremendous high As concentration in the limestone weathering crust).

The profile for As in Quaternary System sediments is highlighted by two key observation (Fig. 1): (1) Distinct differences in As concentration from samples collected at terrestrial facies and sea facies sediments, trend in terrestrial facies sediments As content as a function of sampling phase; (2) Extremely high As concentration averaged 20.25 ppm and a maximum of 32.4 ppm at depths of 3.5—5m where there is littoral facies sediment; As concentration ranged from 3.6 to 11.0 ppm with an average of 7.23 ppm at depths from surface to 3.6m where there is river-lacustrine facies sediment; As concentration ranged from 5.8 to 12.2 ppm with an average 8.49 ppm at depths of 5 to 10 m where there is limnological facies sediment; and As concentration ranged from 8.4 to 12.6 ppm with an average of 9.84 at depths of 10 to 15 m where there is estuarial facies sediment. Variations in As concentration in various facies sediments show that As accumulated in sediments due to deposition environment.

The coacervate speed of Fe-hydroxides exerted clearly an influence on As concentration in sediments. The quicker the coacervate speed, the more As was adsorbed by Fe-hydroxides. So, As concentrations in sediments was related to whether there was an adequate coacervate agent to bring about Fe-hydroxides coacervated (Liu, 1984). There is a great deal of the strong coacervate agent MgSO_4 and NaCl in sea water, in addition, there is remarkable variation in pH and Eh value, which could foster Fe-hydroxides colloid to quickly coacervated and deposit. This process resulted in Fe-hydroxides in conjunction with As trapped in littoral sediments. When the littoral sediments became an aquifer, As and Fe would be released from sediments to ground water through the desorption effect. This might result in increased As and Fe concentrations in ground water

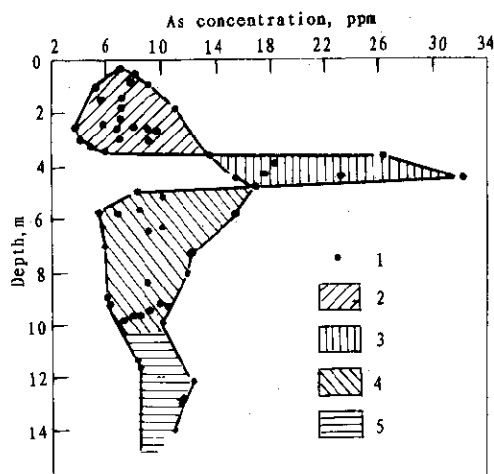


Fig. 1 Profile for As in Quaternary System sediments

1. show samples collected from 11 drill holes;
2. river-lacustrine facies sediment; 3. littoral facies sediment; 4. limenological facies sediment;
5. estuarial facies sediment

(Weng, 1990).

Arsenic concentration in soil was not only dominated by original content in the mother matter but was also affected by the factors of living things, climate, pedogenic years and human activity. As average concentration in soil is similar to that in shallow Quaternary System sediments and ranged from 1.33 to 15.25 ppm with an average of 7.86 ppm (Table 1). This shows that As in shallow Quaternary System sediments was the major source of As in soil. Table 2 exhibits that As concentrations in upperlie soil, except green powdery mud, are higher than those in underlie soil.

Table 2 As average concentration in upperlie and underlie of various types

Unit : ppm

	Green violet mud	Green powdery mud	Small powdery soil	Yellow loose soil	Yellow stain field
No. of samples Upperlie	17	23	20	9	5
Concentration	8.70	1.32	7.79	7.71	8.94
No. of samples Underlie	17	23	20	9	5
Concentration	7.67	9.21	7.57	7.51	8.00

From the point of view of soil development, As may be accumulated in the illuvium (underlie) in conjunction with Fe. First of all, however, the illuvium (upperlie) must be changed into dis-solvable. In a dry field, As trend towards accumulated in the upperlie with As acid species because of oxidation, and in a paddy field, As may be reduced into arsenite and accumulated in the underlie because of reduction. When the soil was alternately cultivated, by dry and water, the process of As migration and accumulation between upperlie and underlie illustrated dynamic equilibrium. Although the soil in this area were principally cultivated as paddy fields, As still trend toward accumulated in upperlie. This case is probably related to man-induced As added to soil. In addition to Eh and pH values (Zhou, 1986; Yang, 1986), As migration and accumulation in soil are associated closely with soil grain. The higher the viscograin content, the easier As was adsorbed by soil (Liu, 1984). The capability in adsorbing As of green powdery mud is relatively weaker because of lower viscograin content, so that As migrated from the upperlie to the underlie and accumulated in the underlie soil.

3.2 Migration of As in the river

The sources of As in Jing-Hang Canal, on the basis of their genesis, were divided into two groups. One group is natural genesis, coming principally from As in Quaternary System sediments and soils. Another group came from man-induced source. A great deal of waste water was drained into the river by the many factories located both sides of the river (1.2315×10^8 t/a). In drained waste water, As and its inorganic compounds reached 0.966 kg/day and 0.7999 t/a, respectively. The coal quantity burned by factories reached 1350000 t/a. Bertine *et al.* (Bertine, 1971) estimated that As concentration in various coals ranged from 3 to 45 mg/kg. They estimated that if it averaged 5 mg/kg and half of total As was volatilized in burning process, then 2.5 g/t coal As might enter the atmosphere from burned coal in this area. The partial As in the atmosphere descended directly into the river with descension dust or rainfall, and the rest descended on the soil. As in soil included the remains of As-bearing pesticides that might also indirectly enter the river through a

scouring process.

The As concentrations in water samples collected from various river sites are less than 0.01mg/L, except in individual sections. As concentration in section 2, 3, 4 and 9 are below determined limit (0.001 ppm). Previous works (Biws, 1982) showed that As concentration in the HCO_3^- type river ranged from 0 to 10 $\mu\text{g/L}$, as compared with this value, As concentration in Jing-Hang Canal is not higher.

Table 3 summarizes As concentration in riverline sediments, in which surficial sediment is jet black and highly unconsolidated, that represent the modern riverline deposition and bottom mud is Quaternary System sediment make up the riverbed. As concentration in surficial sediment of Jing-Hang Canal varied from 7.5 to 65.77 ppm, and averaged 20.58 ppm which is 2 times and 1.6 times that in bottom mud and shale, respectively (Table 3). It is shown that As was remarkably accumulated in surficial sediment, and likely resulted from man-induced pollution rather than from natural genesis because As concentration in terrestrial facies sediments without pollution always is less than As average concentration (13ppm) in shale (Zhen, 1983).

Table 3 As concentration in sediment of Jing-Hang Canal

Unit: ppm

	No. of samples	Range	Average concentration
Surficial sediment	11	7.5—65.77	20.58
Bottom mud	90	3.49—18.68	10.09
Shale (Biws, 1982)	—	—	13

Arsenic in bottom mud averaged 10.09 ppm in 3.49 to 18.68 ppm range, which is less than that in shale. This value was derived from 90 samples collected at different depths (0.3—4 m) at 11 sampling sites. Comparing the vertical distribution of As in bottom mud at section 7 and in the corresponding depth of near drill hole shows that As concentration varies with depth in both are uniform. This indicates that As in bottom mud has resulted from natural genesis.

It can be seen from Table 3 that, average As concentration in surficial sediment was significantly higher than that in bottom mud, and As concentration range in surficial sediment (from 7.5 to 65.77 ppm) is clearly larger than that in bottom mud (from 3.49 to 18.68 ppm). This shows that there is remarkable variation in content features between As accumulated in surficial sediment and natural genesis As in bottom mud. As in surficial sediment shows a maximum of 65.77 ppm in section 4 (Fig. 2), that is 4 times higher than As in shale, in addition As in section 5, 6 and 8 also are higher. These sections occur either close to or several decade meters downstream for the drain outfall's location coming from the printing and dyeing, ship making and steel factories. Significantly higher As in surficial sediment at these sites are closely associated with As in waste water drained by those factories, although As concentrations in water volume at these sections are not higher than those at other sections. The horizontal distribution of As in surficial sediment not only reflect the degree of As pollution, but also indicates the directive sources of As contaminant.

The research results as mentioned earlier show that, for the most part, various genesis sources, As entered the river quickly and was transferred from water volume to riverbed and en-

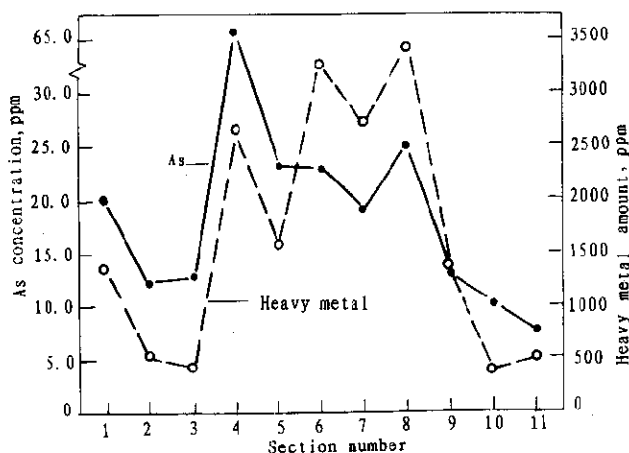


Fig. 2 Horizontal distribution of As and heavy metal in surficial sediment

riched in surficial sediment. Two mechanisms could cause As transferred from water volume to sediment: (1) various species as acid radical combine with cation Ca^{2+} , Mg^{2+} and Fe^{3+} etc. in water and deposit with insoluble arsenates e. g. FeAsO_4 , $\text{Ca}_3(\text{AsO}_4)_2$ and $\text{Mg}_3(\text{AsO}_4)_2$; (2) various species As compounds are adsorbed by suspended matter or colloid and deposited. To test mechanism(1), the existing species of As in water volume must be confirmed. The major As species are a function of E_h and pH values (Ferguson, 1972). Based on the E_h and pH values in Jing-Hang Canal, water volume ranged from 0.68 to 0.71 V and from 6.6 to 7.6, respectively, As should assume positive 5 valence and As exist as H_2AsO_4^- or HAsO_4^- species. Then, the solubility products of arsenates were calculated according to As and cations concentrations in water volume. The ionic strength affecting As dissolvability had been considered because the river water really is a complex mixing dielectric solution. Table 4 summarized the calculated results, which show that only the product of Fe^{3+} and HAsO_4^{2-} is larger than FeAsO_4 solubility product and the product of Ca^{2+} and Mg^{2+} ions and HAsO_4^{2-} are less than that of their arsenates. This indicates that partial As in river water volume could possibly form insoluble arsenate combining with Fe^{3+} and then be deposited. However, Fe^{3+} concentration in the river water volume was significantly lower, ranging from 0.23 to 0.63 ppm. For the most part, Fe^{3+} strongly trend toward combining with OH^- in water volume and was deposited as $\text{Fe}(\text{OH})_3$ because the solubility product of $\text{Fe}(\text{OH})_3$ ($K_{sp} = 4 \times 10^{-38}$) is remarkably less than that of FeAsO_4 ($K_{sp} = 5.7 \times 10^{-21}$). Only when OH^- was in water volume depleted, the remaining Fe^{3+} could then combine with As^{5+} to form FeAsO_4 which, however, is hardly ever possible because of significantly lower Fe^{3+} concentration. Therefore, it is insignificant that As formed insoluble compound combining with Fe^{3+} and accumulated in river sediment.

Not only was As enriched in surficial sediment of Jing-Hang Canal, but heavy metals were also. At sections where As concentration was higher, heavy metal concentrations were correspondingly higher. Fig. 2 also shows that the horizontal distribution of As in surficial sediment is in agreement with that of heavy metal total amount. Weng(Weng, 1990) had reported that the heavy

metal total amount maximum coincided with the total suspended matter peak in the river. In addition there is significant correlativity ($\alpha = 0.001$) because As and heavy metal concentrations, all correlative coefficients are above 0.8 (Table 5), which accounts for a causal relationship between them. For the great part heavy metals, went in the river by various ways, migrated either as the species adsorbed by suspended matter or as the colloid and gradually transferred from water to sediment (Wu, 1987). The significant correlative coefficients between As and heavy metals show that the As accumulative process in surficial sediment was similar to heavy metals, i.e. was transferred from water volume to surficial sediments principally through adsorbed suspended matter or colloid.

Table 4 Solubility product values of forming major arsenates

Sampling section	Ionic concentration in water, mol/L				Solubility product, K_{sp}			Solubility product content, K_{sp}
	Ca^{2+}	Mg^{2+}	Fe^{3+}	HAsO_4^-	$\text{Ca}_3(\text{AsO}_4)_2$	$\text{Mg}_3(\text{AsO}_4)_2$	FeAsO_4	
6	0.86×10^{-3}	0.28×10^{-3}	0.23×10^{-5}	$< 0.71 \times 10^{-7}$	$< 4.07 \times 10^{-22}$	$< 1.15 \times 10^{-23}$	$< 1.63 \times 10^{-2}$	$K_{sp}\text{Ca}_3(\text{AsO}_4)_2 = 6.8 \times 10^{-19}$
7	0.70×10^{-3}	0.25×10^{-3}	0.63×10^{-6}	0.71×10^{-7}	1.89×10^{-22}	8.32×10^{-24}	4.44×10^{-13}	$K_{sp}\text{Mg}_3(\text{AsO}_4)_2 = 2.1 \times 10^{-20}$
9	0.60×10^{-3}	0.20×10^{-3}	0.63×10^{-5}	$< 1.28 \times 10^{-7}$	$< 1.28 \times 10^{-22}$	$< 8.32 \times 10^{-24}$	4.44×10^{-13}	$K_{sp}\text{FeAsO}_4 = 5.7 \times 10^{-21}$

Table 5 Correlative coefficients between As and heavy metals

	Cu	Pb	Zn	Cd	Ni	Cr
As	0.887	0.888	0.822	0.826	0.824	0.839

Note: level of significance; $\alpha = 0.001$

3.3 Arsenic accumulated in surficial sediment and its environmental effect

The contrary process that As was desorbed from sediment to water volume existed simultaneously when As transferred from water volume to sediment. The desorption effect exerts a remarkable influence on the water quality, although this contrary process is minor. Bottom water and sediment become oxygen depleted because dissolved oxygen decreased drastically and organic matter decomposed, and the redox potential (E_h) in surficial sediment was quickly reduced. Under this reducing geochemical condition the changeable valence element varied from high valence to low valence, such as Fe^{3+} to Fe^{2+} and As^{5+} to As^{3+} so far as to As or As^{3-} . This resulted in Fe-arsenate being decomposed and the As adsorbed capacity was also greatly reduced because of charge decrease. Although for the most part As released from anoxic sediment was trapped in the interstitial water, partial As^{3+} could diffuse into oxygen-poor bottom water and be advected upward. This may pollute river water, and be harmful on water quality because As^{3+} is more toxic than other As species (Mochunxiuyi, 1981).

The riverline mud, because of its involved abundant organic matter, is usually used as fertilizer for nearby agricultural fields in the south China. This is an important way that As enriched in river surficial sediment exerts an influence on environment pollution. In addition to abundant organic matter, the riverline mud accumulated significantly higher As. It may result in increase in As concentration in soil at both sides of the river, because As in surficial sediment incorporated with remaining pesticides in soil when the riverline mud was transferred from river to agricultural fields by

the farmer. To determine if As in soil showed a significantly higher As concentration, samples were usually found at both sides of river sections where As in surficial sediment was significant higher. The consistency in space distribution of higher As concentration in soil and riveline mud showed that the As concentration increase in local soil closely associated with As enriched in surficial sediment of river. Because of conditions varying from reduction to oxidation, As transferred from river to soil was easily oxidated from As^{3+} (or lower valence) to As^{5+} . One of the reasons that As concentration in upperlie soil was higher than in underlie soil as mentioned earlier (Table 2) is because As^{5+} may be strongly adsorbed by Fe-hydroxides in soil. When the agricultural field were deeply ploughed, the partially adsorbed As might be ploughed below the depth of 0.5 m where As immediately contacted with ground water. Under this reducing condition, As was reduced again from higher valence to low valence and released from soil to ground water. This process might result in increased As concentration in ground water. 130 ground water samples collected from both sides of the river section 1 to 9, where As was remarkably enriched in surficial sediment, were analyzed for As concentration. It was discovered that As in ground water near these river sections was higher than that in other locations (Fig. 3). This shows that As enriched in the surficial sediment of the river have certain influences on ground water near both sides of the river. To drink this ground water for a long-time will harm on the human body health. Although As concentration in plants was not measured in this paper, soil As pollution might bring about As accumulated in the plants (Xiaoshanxiang, 1975).

It has been directly confirmed, through determining children's bacteriolytic enzyme in saliva, that the environmental affects brought about by the geochemical migration of As did exert potential harmful effects on human body health. The results showed that the bacteriolytic enzyme level of children living in the study area was significantly lower than that of children living in the contrast area. The bacteriolytic enzyme is one of non-idiosyncratic immunity protective screen. The variance in level of bacteriolytic enzyme in saliva can reflect general resistance to disease of organism cases. The level of bacteriolytic enzyme in children's saliva, children being the most sensitive, can reflect environmental pollution affecting the crowd's resistance to disease function, because the excessive As can stifle normal function of many enzymes in human body interior (Zhu, 1980).

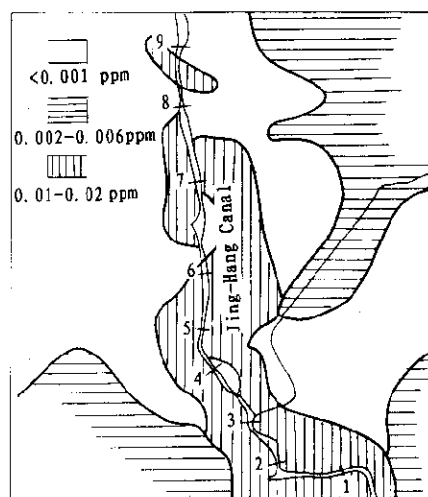


Fig. 3 Distribution of As in ground water near the river section 1 to 9

4 Conclusion

Arsenic trend toward enriched in weathering crust after some dissolvable components were washed away. The higher As background content in the rocks and the stronger the rocks were

weathered, the more obvious was As enriched in the weathering crust. When weathering crust matter were disintegrated, transported and deposited again, the enrichment degree of As in sediments was closely associated with the deposition environment. As concentration in terrestrial facial sediments (averaged 8.90 ppm) was similar with in weathering crust (averaged 8.95 ppm), whereas As in littoral facial sediment was significant higher (Averaged 20.25 ppm with maximum 32.4 ppm). As accumulated in littoral facial sediments, when they became aquifer, might be reduced and released to ground water under the reducing condition.

Various sources of As in the river principally showed high valence oxidic species (AsO_4^{3-} or HAsO_4^{2-}). The partial As might possibly form insoluble compounds combining with some cations (e.g. Fe^{3+}) in the water volume, whereas for the most part As was adsorbed by suspended matter or colloid and quickly separated out from the water volume and enriched in surficial sediment. Under reducing conditions, As accumulated in surficial sediment which might bring about second time river pollution through decomposed arsenates and desorption effects. Particularly, it might result in a significant increase in local soil As from river mud that was transferred from the riverbed to the agricultural fields as fertilizer. As in soil might directly and indirectly go in ground water and also absorbed by certain plants. People drink and eat this ground water and plants, and are potentially harming their health.

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