Review of the current contamination status of Potentially Toxic Elements (PTEs) in the Greater Mekong Sub-region

UNESCAP-IWMI-IAEA Training Workshop on the Development of effective Management and Decision-Making Tools for the Mitigation of Contamination of Soils, Crops and Water in the Greater Mekong Sub-region

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1. Introduction

The UNESCAP-IWMI-NISF workshop held in Hanoi, Vietnam in December 2002 (Simmons and Bakker, 2003) demonstrated the significant concern of countries within the Greater Mekong Sub-region (Thailand, Myanmar, Cambodia, Lao PDR, Vietnam and China) regarding Potentially Toxic Element (PTE) contamination of soil and water resources. Country specific concerns primarily focused on potential impacts on public health, the sustainable use of soil and water resources and national export security. These concerns are affectively addressed by the cross-sectoral Regional IWMI-UNESCAP Program of activities entitled "Protecting food security, human health, environmental integrity and livelihoods in rice-based agricultural systems from the detrimental impacts of PTEs in the Greater Mekong Sub-region" (www.iwmi.cgiar.org/southeastasia/index.asp). The continued comittment of in-country partners has resulted in the joint development and submission of full project proposals and the partner driven request for a follow up workshop. This review of the current contamination status of PTEs in the Greater Mekong Sub-region was undertaken as a pre-requisite to the UNESCAP-IWMI-IAEA follow up training workshop entitled "Development of Effective Management and Decision-making Tools for the Mitigation of Contamination of Soils, Crops and Water in the Greater Mekong Subregion" held in Bangkok, Thailand, 16-18 June, 2004.

Potentially toxic elements discussed in this report include arsenic, (As), fluoride (F), cadmium (Cd), thallium (Tl), selenium (Se), copper (Cu), lead (Pb), mercury (Hg), chromium (Cr), zinc (Zn) and nickel (Ni). The results of this review confirm that naturally elevated levels of arsenic (As) in groundwater aquifers with confirmed direct impacts on public health is an issue of regional significance that needs to continue to be comprehensively and systematically addressed. In addition, millions of rural poor in China are exposed to naturally elevated levels of F in groundwater aquifers. Further, the household use of high-As/F coal as a means of drying agricultural produce is a major pathway of arsenocosis and fluorosis in several provinces of China.

Numerous articles in international peer reviewed journals and reports by national agencies have identified Cd contamination of soils and agricultural produce in localized areas of Thailand, Vietnam and China resulting from the agricultural use of irrigation water contaminated with discharge from non-ferrous mining and ore processing activities, particulate deposition in areas adjacent to non-ferrous smelters and the agricultural use of untreated urban/industrial wastewater. With regards Cd contamination, it must be stressed that detrimental affects on crop quality and public health are predominantly at a local/community level. However, the impacts of Cd contamination take on a national significance due to the need to provide health services, establish effective crop quality monitoring programs and the implementation of effective management and remediation options.

Chronic thallium (Tl) poisoning resulting from food chain Tl contamination has also been reported at specific point source locations. Selenium deficiency and toxicity are also considered to be significant public health problems in several provinces in China. Localized Hg contamination of surface water, soils and crops has also been reported as a result primarily of coal combustion and gold mining although no confirmed impacts on public health have been reported.

On a regional basis, decision support tools namely national and international water and soil quality standards, causality chain indicator frameworks, and legislation in the form of land, water, mining and environmental laws exist (http://www.ecolex.org/ecolex/index.php). However, this review indicates that the exposure of urban and rural communities to elevated levels of PTEs is a reality. The results of this review further infer that the root cause of anthropogenic PTE contamination of soil and water resources, localized food chain Cd contamination and the exposure of millions of predominately rural poor to naturally elevated levels of As and F in groundwater aquifers is primarily due to a prior lack of awareness by decision makers.

This appears to have resulted in a lack of effective monitoring programs, lack of strategic decision support tools (namely PTE specific hazard maps), in-adequate enforcement of legislation and lack of compliance. These deficiencies are effectively addressed by the Regional IWMI-UNESCAP Program of activities entitled "Protecting food security, human health, environmental integrity and livelihoods in rice-based agricultural systems from the detrimental impacts of PTEs in the Greater Mekong Sub-region" (www.iwmi.cgiar.org/southeastasia/index.asp)

This report highlights priority PTE contamination issues in countries within the Greater Mekong Sub-region and identifies and briefly discusses potential management solutions, decision support tools, and remediation options.

1.1 Potential food security and economic implications

Agricultural produce cultivated on soils with elevated levels of heavy metals may not meet internationally recognized maximum levels (MLs) as established by the Joint FAO/WHO Food Standards Programme, Codex Alimentarius Commission (CAC), Codex Commission on Food Additives and Contaminants (CCFAC). The MLs established by CCFAC are based on the 'safe' lifetime consumption of agricultural produce and the *free movement* of products in international trade. MLs set by the CCFAC are used internationally as criteria to establish *non-tariff trade barriers*. In addition, ISO 14000 is being sought by many food importers to guarantee that food and fiber are produced using environmentally sustainable practices. The inability of countries within the Greater Mekong Sub-region to comply with CCFAC MLs and ISO 14000 would have a significant impact on export security. However, the results of this review indicate that in terms of crop quality, the issue is of a local/community scale. In terms of Cd, health implications are primarily associated with the long-term consumption of Cd contaminated rice. Heavy metals particularly copper (Cu), nickel (Ni), and zinc (Zn) are of potential concern due to direct and indirect effects on food security by reducing yields and crop quality. However, no reports of reduced yields due to phyto-toxic levels of Cu, Ni and/or Zn have been reported.

2. Vietnam

In general background levels of PTEs in Vietnamese soils are well within national and international standards. However, localized Cd and Zn contamination of irrigated urban and peri-urban rice-based agro-ecosystems of Vietnam has been identified (Table 1). Cadmium contamination is of concern due to potential impacts on public health and the long-term sustainable use of soil resources. It should be noted that the summarized tabulated data included in this section does not always indicate a statistical evaluation of the data quality. For further confirmation of the validity of the data the reader is directed to the original article.

Table 1. Mean soil PTE contamination (mg kg^{-1}) in irrigated rice-based production systems in Vietnam

		Soil total element concentration (mg kg^{-1})				
Contamination Source	Location	Cd	Zn	Pb	Cu	Cr
Domestic and/or Industrial wastewater	Nha Be & Binh Chanh Districts HCM City	Min 1.37 Max 1.58	Min 86.0 Max 146.0	Min 29.5 Max 50.8	Min 19.0 Max 46	Min 74.8 Max ***131***
Discharge from Ni/Cd battery and P-fertilizer factories	Hanoi	Min 0.68 Max 1.55	Min 72.5 Max ***2061***	Min 30.2 Max ***123***	Min 29.6 Max 49.5	Min 44.6 Max ***194***
*Domestic and Industrial wastewater	Thu Duc District Hanoi	Min ***6.30*** Max ***8.90***	Min ***159.0*** Max ***398***	Min 36.0 Max ***57.0***	Min 22.0 Max ***65.0***	Min 79.0 Max ***101***
EU MP levels (mg kg^{-1})		1.0 -3.0	150-300	50-300	50-140	100-150

EU Maximum Permissible (MP) levels for sludge amended soils (Directive 86/278/EEC).
Data provided by Dr. Nguyen Cong Vinh NISF. * Quang, (2000).

2.1 Background levels of heavy metals in agricultural soils of Vietnam.

Ha et al., (2003) investigated the distribution of selected heavy metals in fluvisols, acrisols and ferrasols in the North and Northern Central Vietnam and found that mean total soil Cu, Cd, Pb, Zn and Hg concentrations varies according to soil type (Table 2). In addition, 'background' levels of Zn and Cd in agricultural soils of Northern Vietnam were investigated by the ACIAR Project LWR119/1998 as compared to selected forest and polluted areas. With the exception of the maximum Cd concentration of polluted areas the ACIAR results indicate the non-contaminated status of agricultural soils of Vietnam (Table 3).

Table 2. Mean background PTE concentrations (mg kg^{-1}) in fluvisols, acrisols and ferrasols in North and Northern Central Vietnam

	Element Concentration (mg kg^{-1})				
Soil Type	Cu	Cd	Pb	Zn	Hg
Fluvisols	30.6	0.85	34.6	87	0.53
Acrisols	16.6	0.48	20.6	30.5	0.09
Ferrasols	46.8	1.24	21.8	107	0.26
EU MP levels (mg kg^{-1})	50-140	1-3	50-300	150-300	1-1.5
*VN Standard (TNVN 7209-2002)	50	2	70	200	/

EU Maximum Permissible (MP) levels for sludge amended soils (EEC, 1986. Directive 86/278/EEC)
*VN Standard (TNVN 7209-2002) for agricultural soils

Table 3. Soil Zn and Cd concentrations (mg kg^{-1}) of Northern Vietnam as related to selected land-uses (ACIAR Project LWR119/1998).

Land Use	Zn	Cd
Polluted areas (n= 25)	14.2 – 151.5 (**68.7**)	0.02 – 7.24 (**0.54**)
National parks (n=25)	10.97 – 204 (**57.19**)	0.01 – 0.24 (**0.07**)
Agricultural areas (n=156)	3.8 – 142.5 (**40.4**)	0.00 – 1.7 (**0.10**)
Vegetable production areas (n=38)	1.21 – 140.6 (**50.27**)	0.00 – 0.32 (**0.09**)
EU MP levels (mg kg^{-1})	150-300	1-3.0
VN Standard (TNVN 7209-2002)	200	2.0

*VN Standard (TNVN 7209-2002) for agricultural soils. EU Maximum Permissible (MP) levels for sludge amended soils (Directive 86/278/EEC). Values in parentheses indicates the arithmetic mean.

However, Ha et al., (2003) indicated that total soil Cd concentrations in Fluvisols sampled within the National Base Line Program as well as alluvial soils (mainly Fluvisols) of the Red River and Mekong River deltas utilized for rice and vegetable production are relatively high as compared to the EU MP levels (Table 4). For agricultural soils the EU MP level ranges is 1-3 mg Cd kg^{-1} and given as a function of soil pH which, is a major factor controlling the mobility and relative uptake of Cd in paddy soils. Iumura et al., (1981) observed that the relative uptake of Cd by rice plants was greatest within the pH range 4.5-5.5. In addition, Bingham *et al.,* (1980) suggested that Cd content of rice grain was highly dependent on soil pH with maximum uptake at pH 5.5. In general Cd mobility is limited above a soil pH of pH 7.5.

Table 4. Soil Cd concentration (mg kg^{-1}) in Fluvisols (National Base Line Program) as compared to alluvial soils of the Red River and Mekong River Deltas

	Fluvisols	Alluvial soils of the Red River delta	Alluvial soils of Mekong River delta
No. sampling locations	190	42	49
Range* (mg kg^{-1})	0.745-0.833	0.891-1.064	0.606-0.715
Mean (mg kg^{-1})	0.789 (± 0.307)	0.978 (± 0.278)	0.888 (± 0.280)
EU MP level for soil Cd (mg kg^{-1})	1-3.0		

*95% confidence interval
EU Maximum Permissible (MP) levels for sludge amended soils (Directive 86/278/EEC).

2.2 Evaluation of potential soil and crop PTE contamination

To date research into PTE contamination of soils in Vietnam has been primarily focused on urban and peri-urban areas of Hanoi and Ho Chi Minh City. In the peri-urban Hoc Mon District, Ho Chi Minh City, Quang, (2000) reported total Cu, Zn, Pb, Cr and Cd concentrations in alluvial soils ranging from 17.2-81, 64-168, 14.5-75, 10.5-41 and 0.48-1.05, respectively (Table 1). All of these values are below the Vietnamese (TCVN 7209:2002) and EU (Directive 86/278/EEC) permissible levels for agricultural soils. Further, Quang, (2000) reported total As and Hg concentrations in alluvial soils of 1.25-3.75 and 0.049-0.512 which, are both below the respective EU (Directive 86/278/EEC) permissible levels. In addition, Manh, (2002) identified levels of Hg ≥ 0.50 mg kg^{-1} in agricultural soils utilized for vegetable and rice cultivation in localized areas of Thanh Tri, Linh Nam and Tran Phu Districts of Hanoi. PTE contamination of soil, water and sediment surrounding industrial and urban areas of Ho Chi Minh City and Hanoi have also been reported by Khoa et al., (1999), Thu, (2001), Hien, (2000) and Quang, (2002). The uncontrolled release of untreated waste water from industries is a major source of contamination to agricultural soils. Khoa et al., (1999) investigated the heavy metal content in wastewater discharging from the Van Dien and Orion-Hanel industrial zones, and heavy metals in sediments collected from the Tolich river. Khoa et al., (1999) reported that the content of Hg and Pb in waste water discharging from the Van Dien battery factory were 0.09 mg l^{-1} and 1.12 mg l^{-1}, respectively significantly higher than the MP level outlined in the Vietnamese Industrial Wastewater Discharge Standard TCVN 5945/1995.

In Hanoi, PTE levels were investigated in the heavily industrialized and densely populated southwestern Districts of Thanh Tri and Tu Liem. The main cropping systems are rice-rice, rice–rice-corn and rice–rice–vegetables. Thanh Tri and Tu Liem districts provide a high proportion of the vegetables and food products for Hanoi City. The intensive peri-urban rice and vegetable production systems in Thanh Tri and Tu Liem Districts are irrigated primarily by the Tolich and Kimnguu Rivers. Nguyen, (1997) estimated that the Tolich and Kimnguu Rivers receive 24,900 m^3 and 130,000 m^3 of untreated combined industrial-urban wastewater on a daily basis. The industrial sector of Thanh Tri and Tu Liem Districts includes tanneries, Cd/Ni battery, phosphate and chemical manufacturing businesses and electro-plating and engineering activities. In 1996, soil samples (0-15cm depth) were collected from vegetable and rice fields within Thanh Tri and Tu Liem Districts (Ho *et al*., 1998). With the exception of one vegetable field within an area irrigated by wastewater discharging from the Vandien Phosphate Factory total soil Cd, Zn and Cu concentrations ranged from of 0.08-0.4, 22.5-89.9 and 12.1-70.4 mg kg^{-1}, respectively and were within the Vietnamese (TCVN 7209:2002) and EU (Directive 86/278/EEC) permissible levels for agricultural soils. Significantly elevated Cd and Zn levels of 1.33 and 1164 mg kg^{-1} were recorded for the area receiving wastewater discharging from the Vandien Phosphate Factory, respectively.

In a follow up survey conducted in 1997 additional soil samples were collected from previously un-sampled but spatially comparable sites within Thanh Tri and Tu Liem Districts (Ho and Egashira, 1999). The results indicate that Cd, Zn, Pb, Ni, Cu and Cr concentrations in soil ranged from 0.16 – 0.36, 98.2 – 102, 31.9 – 45.3, 33.6 – 66.0, 40.1 – 73.2 and 62.5 – 112.8, respectively. Cadmium and Cu levels are comparable to the values presented in Ho *et al*., (1998). In comparison, all the soil samples collected in the 1997 survey were associated with significantly higher levels of total Zn as compared to the 1996 survey. It is unclear whether this reflects a systematic increase in soil Zn concentration, a systematic analytical error or is a function of the sampling strategy.

Quynh and Ba, (2002) identified significant Cd and Zn contamination in soils of rice-based agricultural systems in selected Districts of Ho Chi Minh City as compared to Vietnamese (TCVN 7209:2002) and/or EU (Directive 86/278/EEC) MP levels for agricultural soils (Table 5). Levels of Cu, Pb and Hg are within acceptable levels. In 2000, Ho Chi Minh City was associated with 1,000 registered industrial factories, 28,500 handicraft foundations and 12 industrial zones (Quynh and Ba, 2002). In Nha Be and Binh Chanh Districts, irrigated rice-based agricultural systems receive irrigation from the Tan Hao, Lo Gom Te Doi, Tau Hu and Ben Nghe canals. A high proportion of the wastewater discharged from these industries is untreated and used directly for agriculture. It should be noted that Cd and Zn, exceed EU MP levels whilst all other PTEs fall below statutory levels. A detailed breakdown of heavy metal contamination in soils of Nha Be District is presented in Table 6.

Table 5. Mean total soil heavy metal concentrations (mg kg^{-1}) in soils of rice-based agricultural systems receiving untreated wastewater from Ho Chi Minh City.

		Element concentration (mg kg^{-1})					
District	No. Samples	Cd	Cu	Zn	Pb	Hg	Cr
Nha Be	88	**9.9**	28.6	110	61.7	0.09	125.3
Binh Chanh	10	**10.3**	31.0	197	58.0	0.21	119
Thu Duc	8	**6.8**	30.0	282	44.3	0.20	84.3
No. 2	10	**5.5**	33.1	435	43.6	0.34	44.3
No. 7	4	**4.7**	22.7	233	39.0	0.05	115
No. 9	6	**4.9**	29.5	568	40.5	0.03	54.3
EU MP levels (mg kg^{-1})		1-3.0	50-140	150-300	50-300	1-1.5	nv
*VN Standard (TNVN 7209-2002)		2.0	50	200	70	nv	nv

Adapted from Quynh and Ba, (2002). Values in bold exceed MP levels. nv = no standard for this element.
EU Maximum Permissible (MP) levels for sludge amended soils (Directive 86/278/EEC)
*VN Standard (TNVN 7209-2002) for agricultural soils

Table 6. Mean total PTE concentrations (mg kg^{-1}) in soils of rice-based agricultural systems receiving untreated wastewater in Nha Be District, Ho Chi Minh City.

	Element concentration (mg kg^{-1})					
Soil Type	Cu	Cd	Zn	Pb	Hg	Cr
Red-Yellow alluvial soil (Pfm)	27.5	16.7	130	78.1	0.19	133
Alluvial soil/potential acid sulphate soil (Ppm)	29.4	14.0	102	63.8	0.07	136
Acid sulphate soil (Sj2m)	28.7	14.5	102	66.8	0.15	133
Acid sulphate soil (Sj2Rm)	23.6	7.6	83.8	70.2	0.12	124
Acid sulphate soil at depth (Sp2m)	27.7	12.4	92.7	59.4	0.12	129
Acid sulphate soil organic horizonn (Sp2hm)	18.6	7.8	97.2	67.1	0.11	121
EU MP levels (mg kg^{-1})	50-140	1-3.0	150-300	50-300	1-1.5	/
*VN Standard (TNVN 7209-2002)	50	2.0	200	70	/	/

Adapted from Quynh and Ba, (2002) and Kuo et al., (1983). Values in bold exceed MP levels. EU Maximum Permissible (MP) levels for sludge amended soils (Directive 86/278/EEC) *VN Standard (TNVN 7209-2002) for agricultural soils

2.3 Sources of PTE contamination to rice-based agricultural systems in Vietnam

Potential sources of heavy metal/metalloid contamination to agricultural systems include; natural runoff from non-ferrous ore mineralized areas, uncontrolled runoff and leaching from the extraction and processing of non-ferrous ores; agricultural use of municipal sewage sludge; bio-solids from agro-industries; widespread use of untreated and/or partially treated industrial and/or domestic wastewater; pesticides; and phosphatic fertilizers. As indicated in Table 7. biosolids and inorganic phosphate fertilizers in Vietnam often contain Cd as a contaminant. However, when compared to the EU maximum permissible level of Cd in sewage sludge of 20-40 mg kg^{-1} (Directive 86/278/EEC) the levels of Cd in biosolids in northern Vietnam are negligible. Further, limit values for Cd in fertilizers in EU member states ranges from 21.5 – 90 mg Cd kg P_2O_5 (Oosterhuis, et al., 2000). In addition, Cupit et al., (2002) proposed a phased reduction of Cd in phosphate fertilizers from 60 mg Cd kg P_2O_5 in 2006 to 20 mg Cd kg P_2O_5 in 2015. The levels of Cd in inorganic phosphate fertilizers represented in Table 7 are well below the limit values proposed by Cupit et al., (2002).

Table 7. Cadmium concentration (mg kg^{-1}) in biosolids and inorganic phosphate fertilizers in sampled in Northern Vietnam

Material	**Cd**	**Location**
Cow manure	0.48	Vinh Phuc province
Chicken manure	1.50	Ha Tay provine, Hanoi
	1.48	ACIAR Project LWR/119/1998
Pig manure	0.54	Ha Tay province
Human manure	0.39	Ha Tay provine, Hanoi
Mineral organic fertilizer	0.70	Hanoi
Super phosphate	2.77	Lamthao town (BTPB)
Super phosphate	2.70	Lamthao town (BN)
FMP phosphate	2.53	Vandien town (BTPB)
FMP phosphate	2.63	Vandien town (BN)
Apatite mineral	4.25	Lao Cai (BTPB)
Apatite mineral	2.88	Thanh Hoa (BTPB)

(Source: ACIAR Project LWR119/1998 and NISF)

Traditionally, farmers in the Red River Delta utilize sediment from irrigation and drainage channels as an organic fertilizer. However, heavy metal concentrations in sediment from irrigation/drainage channels within industrial areas of Thanh Tri and Tu Liem districts of Hanoi are significantly contaminated with heavy metals (Table 8), (Ho et al., 1998; Ho and Egashira*,* 2001; Ho et al., 2000a b).

The mean Cd and Zn concentrations in sediment taken from the Tolich and Kimnguu Rivers in Thanh Tri and Tu Liem districts of Hanoi were 2.36 and 875 mg kg^{-1} and 2.71 and 2118 mg kg^{-1}, respectively. Elevated levels of Cu, Cr and Ni were also observed (Table 8).

In comparison, significantly lower levels of Cd, Cr, Cu Ni, Pb and Zn were observed in sediment from the Nhue River which in 2000 was not associated with industrial activities and sediment taken from a sample point 4km downstream of the confluence and main industrial zone on the Tolich and Kimnguu Rivers (Table 8). The per-urban farmers are effectively using the sediment as a soil amendment in the absence of sewage sludge. Therefore, for comparison the sediment PTE concentrations are evaluated against the EU MP levels of heavy metals in sewage sludge applied to agricultural soils (Directive 86/278/EEC). This comparison indicates that only Zn exceeds the EU MP levels. However, it is also important to note that the exchangeable Cd fraction in the sediment (air-dried) ranged from less that 5% to over 50% of the total Cd (Ho et al., 2001). Further, it is of note that the use of Kimnguu River sediment as a soil amendment was prohibited in 2000 (Ho et al., (2000b).

Table 8. PTE concentrations (mg kg^{-1}) in sediment derived from the Nhue, Tolich and Kimnguu Rivers within Thanh Tri and Tu Liem Districts of Hanoi.

River	Element concentration (mg kg^{-1})					
	Cd	Cr	Cu	Ni	Pb	Zn
Nhue	(n=4) 0.27 - 0.36 *0.30* (±0.05)	(n=4) 81 - 100 *92* (±9.84)	(n=4) 64 - 67 65 (±1.52)	(n=4) 38 - 47 *44* (±5.19)	(n=4) 76 - 159 *103* (±47.9)	(n=4) 93 - 163 *121* (±36.6)
Tolich	(n=3) 0.34 - 4.5 *2.43* (±1.88)	(n=3) 78 - 337 *190* (±113)	(n=3) 37 - 112 *65* (±32.4)	(n=3) 37 - 100 *61* (±28.2)	(n=3) 60 - 96 *73* (±15.7)	(n=3) 105 - 877 *458* (±351)
Kimnguu	(n=3) 2.52 - 2.94 *2.79* (±0.23)	(n=3) 110 - 517 *347* (±211)	(n=3) 77 - 309 *207* (±118)	(n=3) 59 - 174 *134* (±65.5)	(n=3) 157 - 361 *243* (±105)	(n=3) 1062 - **4950** *2478* (±2148)
	(n=1) 0.63*	(n=1) 96*	(n=1) 44*	(n=1) 46*	(n=1) 43*	(n=1) 103*
EU MP levels (mg kg^{-1})	20-40	nv	1000-1750	300-400	750-1200	2500-4000

Data derived Ho and Egashira, (2001). * 4km downstream of the confluence of the Tolich and Kimnguu Rivers. EU Maximum Permissible (MP) levels of heavy metals in sewage sludge applied to agricultural soils (Directive 86/278/EEC) nv = no standard for this element. Value in parentheses indicates ±1 STDEV.

2.4 Potential crop heavy metal uptake resulting from the land application of PTE contaminated river/canal sediment

Internationally recognized Maximum Levels (ML) for contaminants in foods are established by the Codex Committee on Food Additives and Contaminants (CCFAC) and the Joint FAO/WHO Expert Committee on Food Additives (JECFA, 2003). To assess potential public health risk associated with the use of ricer sediments within Thanh Tri and Tu Liem districts Ho et al., (2000a) investigated the uptake of Cd, Pb, Zn Cr and Ni to white cabbage and beet in soils amended with a composite, Tolich River sediment. The results indicated increased biomass production at low rates of application due to inherent the fertility of the Tolich River sediment. This confirms the farmer justification for the use of sediment as a soil amendment. However, at higher rates of sediment application (30-50 t ha^{-1}) equating to total soil Cd concentrations ranging from 2.94 - 4.69 mg kg^{-1}. This resulted in white cabbage leaf Cd concentrations ranging from 0.22 – 0.27 mg Cd kg^{-1} (Fresh Weight). This exceeds the Joint FAO/WHO Food Standards Programme, Codex Alimentarius Commission (CAC) Maximum Permissible (MP) level of Cd in leafy vegetables of 0.2 mg Cd kg^{-1}. In addition, at all rates of sediment application Pb levels in white cabbage leaf (0.7 ±0.17 mg kg^{-1}) exceeded the CAC MP level for Pb in leafy vegetables of 0.3 mg Pb kg^{-1}. Levels of Cr, Ni and Cu in white cabbage leaf irrespective of the rate of amendment application were negligible and within CAC MP levels.

In comparison, with the exception of Pb heavy metal concentrations in the beet tissue samples were within CAC MP levels. The mean Pb in beet tissue samples remained relatively constant (0.4 ±0.06 mg kg^{-1}) and above the CAC MP level for Pb in tuber vegetables of 0.1 mg Pb kg^{-1}, irrespective of the rate of sediment application. The results of Ho et al., (2000a, b) although limited in scope demonstrate the potential risks associated with the application of heavy metal contaminated sediment to agricultural soils.

2.5 Elevated levels of Cd in rice grain in Vietnam: Potential health impacts

The 34th Session CCFAC (Rotterdam, The Netherlands during the 11-15th March 2002) proposed a *draft* provisional Maximum Level (ML) for Cd in rice grain of 0.2 mg Cd kg^{-1}. This ML is based on the 'safe' lifetime consumption of rice and is established to ensure the free movement of rice in international trade. From a health perspective, the 61st JECFA agreed to maintain the Provisional Tolerable Weekly Intake (PTWI) of Cd at 7µg Cd per kg Body Weight (BW) per week (Rome, Italy, 10-19 June 2003). This PTWI value is established to prevent potential Cd-induced detrimental health impacts via dietary Cd.

From a public health perspective the long-term consumption of cadmium (Cd) contaminated rice results in human Cd-disease as manifested, as proximal tubular renal dysfunction (Nogawa et al., 1983; Kido et al., 1988). Cd-induced renal dysfunction in individuals' exposed to elevated dietary-Cd is irreversible and progressive despite decreases in exposure (Gambini and Leurini, 1992; Nogawa and Kido, 1993). In addition, several studies have confirmed that Cd-induced renal dysfunction interferes with Vitamin D metabolism, resulting in decreased Ca absorption and the occurrence of osteopenia and osteoporosis particularly, in multiparous women (Kido et al., 1990; Tsuritani et al., 1992). Previous research has established that levels of dietary Zn, Fe, and to a lesser extent Ca, are known to influence the absorption of Cd and its distribution in organs and tissues (Flanagan *et al.,* 1978; Koo et al., 1978; Fox, et al., 1979; Brzóska and Moniuszko-Jakoniuk, 1997; Reeves and Vanderpool, 1998; Berglund et al., 1994; Reeves and Chaney, 2001, Chaney et al., 2001). Cadmium contamination of rice has a strong gender bias and is also related to wealth and dietary intake. Iron (Fe) and Zn deficiency predisposes individuals to a higher Cd absorption and therefore women are a more vulnerable group since they are more prone to Fe deficiency (Berglund et al., 1994).

Cadmium related health risks associated with the long-term consumption of Cd contaminated rice grain result from several compounding factors Firstly, rice grain iron (Fe), zinc (Zn) and calcium (Ca) contents are insufficient for human needs (Hallberg et al., 1977; Pedersen and Eggum, 1983). Milling rice grain results in further Fe and Zn losses (Pedersen and Eggum, 1983; Zhang et al., 1997) whilst grain Cd levels remain un-affected (Yoshikawa et al., 1981). In addition, the rice plant, irrespective of total and bio-available soil Zn concentrations, effectively controls rice grain Zn without reciprocal control on grain Cd (Simmons et. al., 2003). In Laos, Cambodia and Vietnam, subsistence rice-based agrarian communities dominate the agricultural sector. Within the Mekong Basin, the WHO has documented widespread Fe and Zn deficiencies within rural communities (WHO, 1996). Since rice makes up 75%, 66% and 66% of total daily calorie intake in individuals from Cambodia, Lao PDR and Vietnam (IRRI Rice database 2000). This may constitute a significant potential dietary intake pathway for Cd and hence is a major long-term threat to food security and rural livelihoods if rice production is compromised.

2.6 Case studies in Vietnam: Elevated levels of Cd in rice grain in case study locations in Vietnam: Does it pose a potential public health risk? (Revised May 2005)

Preliminary results from the IWMI-DANIDA Project *'Waste Water reuse in Agriculture in Vietnam'* indicate a mean (n=53) soil Cd concentration of 0.91 (± 0.229) mg kg^{-1}. The IWMI-DANIDA study area is located in My Tan commune, south of Nam Dinh city and is split between a 30ha wastewater irrigated (Nam Dinh City municipal wastewater) sub-area in Hong Long Cooperative and a 16 ha Red River irrigated sub-area in Tan Tien Cooperative. No significant difference was observed in the soil Cd concentration between the wastewater and Red River irrigated sites with mean values of (n=41) 0.910 (± 0.036) and (n=12) 0.858 (± 0.049) mg Cd kg^{-1}. Further, the mean (n=53) soil Cd concentration of 0.91 (± 0.229) mg kg^{-1} is indicative of the 'background' Cd levels found in soils of the Red River Delta by Pham Quang Ha et al., (2003) (Table 4).
Cd concentrations in the concurrent rice grain samples (n=52) collected in 2003 ranged from 0.15 – 0.38 mg Cd kg^{-1} and are within the recently established JECFA MP level for Cd in rice grain of 0.4 mg Cd kg^{-1} (JECFA 2005).

Average daily rice intake in Vietnam is 0.466 kg d^{-1} (National Institute of Nutrition, 2004) and average body weight of a male and female aged 50 years is 61 kg and 55 kg, respectively. Consequently, based on the range in rice Cd concentration at the Nam Dinh study site, and the relatively high average daily rice intake for Vietnam (as compared to Thailand) WI values for men and women of 50yrs would range from 8.90 – 22.54 and 8.02 – 20.32 µg Cd per kg BW. In the My Tan commune, rice is grown on family plots primarily for home consumption. Excess is sold to local markets. The high estimated WI values (>7µg Cd per kg BW) would therefore suggest that residents of the aforementioned communes are exceeding their weekly intake of Cd. The elevated levels of Cd in rice grain may to a large extent be explained by the fact that the soils in the Nam Dinh study area have a mean (n=53) pH_{water} of 5.2 (± 0.15). This confirms the findings of Iumura et al., (1981) who observed that the relative uptake of Cd by rice plants was greatest within the pH range 4.5-5.5. Further, as previously mentioned, the mean (n=53) soil Cd concentration at the Nam Dinh study site are indicative of Cd concentrations in alluvial soils of the Red River and Mekong River Deltas. At similar soil pH (pH 5.0), to what extent are communities within these deltaic areas exposed to high 'background levels' of dietary Cd via the rice food chain?

Similarly, Quynh and Ba, (2003) identified rice grain Cd concentrations grown on Cd contaminated paddy soils of Binh Chanh District, Ho Chi Minh City ranging from 0.38 – 0.56 mg Cd kg^{-1} (Table 9). Therefore, based on the range in rice Cd concentration recorded for the Binh Chanh District by Quynh and Ba, (2003) and the daily rice intake for Vietnam of 0.466 kg d^{-1} (National Institute of Nutrition, 2004) WI values for men and women of 50yrs would range from 22.54 – 33.21 and 20.32 – 29.95 µg Cd per kg BW. As is the case for Nam Dinh, these values significantly exceed the JECFA PTWI value of 7µg Cd per kg BW.

Table 9. Cadmium concentration in soil and rice plant tissues (mg kg^{-1}) from selected areas of Binh Chanh District, Ho Chi Minh City.

	Cd concentration (mg kg^{-1})		
Sampling Location	Soil	Un-polished rice grain	Straw/Leaf
BC 3	7.6	0.38	1.26
BC 5	9.8	0.52	2.03
BC 9	14.5	0.55	2.37
BC 12	10.3	0.56	2.09
BC 13	9.6	0.55	1.96
BC 14	9.9	0.41	1.28
BC 32	10.3	0.48	2.33
Mean	10.28	**0.49**	1.90
STDEV	2.074	0.072	0.457
CV	20.17	14.71	24.04

Adapted from Quynh and Ba, (2003).

It must be noted that although based on robust laboratory analysis the WI values calculated for communities consuming rice cultivated at the Nam Dinh and Binh Chanh study sites are ‘estimated’ WI values only. Further detailed epidemiological and dietary studies must be undertaken to accurately determine actual public health risks. In this regard, as previously noted, adequate levels of dietary Fe and Zn have been shown to effectively protect against Cd-induced renal dysfunction. In 1995, 40% of Vietnamese women between 15 and 40 yrs were considered anaemic (Fe deficient). In 2004, the national average had through a national Fe supplementation program declined to 28% (National Institute of Nutrition in Vietnam (National Institute of Nutrition, 2004). However for pregnant women, the percentage with anemia is still high at 34% (National Institute of Nutrition, 2004).

2.7 Cadmium contamination associated with non-ferrous ore extraction, processing and smelting.

Reports of health effects of Cd in populations not occupationally exposed to Cd have centered on Japan (Watanabe et al., 1998; Kobayashi et al., 2002; Nogawa et al., 1983; Kido et. al., 1988; Nogawa and Kido, 1993; Kido *et al.,* 1990*;* Tsuritani *et al.,* 1992) and China (Cui et al., 2004; Jin et al., 2002; Wu et al., 2001; Jin et al., 1999; Cai et al., 1995; Cai et al., 1990; Nordberg, 2003) where many areas are contaminated with Cd as a result of discharges from non-ferrous mines and smelters. Vietnam has extractable proven resources for Zn and other base metals. In the colonial era Vietnam was a major Zn ore and metal producing center. From 1920 to 1930, approximately 500,000 metric tons of Zn ore were exported from Vietnam. This ore was of an exceptionally high grade of up to 40% Zn with a relatively high concentration of Cd. The main areas of activity were the Cho Dien-Cho Don and Bac Thai deposits located in Băc Can, Tuyên Quang and Thai Nguyen provinces. In addition, historically active non-ferrous ore extraction was undertaken in a further 15 provinces throughout Vietnam (Table 10). Historically active locations of ore extraction are of considerable concern due to a lack of continued maintenance and control over discharge/runoff from ore waste containment facilities.

Table 10: Location and status of non-ferrous base metal ore deposits in Vietnam

Province	Mineral Deposit	Status	Province	Mineral Deposit	Status
Lai Châu	Cu (x2) Pb,	Closed Closed	Yên Bái	Pb/Zn/Ag Cu	Closed Closed
Laò Cai	Cu Cu (x2) Pb/Zn	Closed Potential Area Closed	Băc Thai	Pb/Zn (x2) Zn/Pb Pb/Zn/Ag	Potential Area Active Potential Area
Hà Giang	Pb/Zn	Potential Area	Dà Năng	Cu	Potential Area
Cao Băng	Pb/Zn (x1) Zn/Pb/Sn Cu/Ni P	Closed Potential Area Potential Area Closed	Tuyên Quang	Pb/Zn/Au Pb/Sr/Ba Pb Pb/Zn/Ba	Potential Area Potential Area Potential Area Closed
Són La	Pb/Zn (x2) Pb/Zn (x2) Zn/Cu/Au Ni/Cu Ni/Co/Cu/Se Cu (x2) Cu (x2) Cu	Closed Potential Area Closed Potential Area Potential Area Potential Area Closed Active	Hòa Bình	Zn/Pb/Cd Pb/Zn Cu	Potential Area Potential Area Potential Area
Lang Son	Pb/Zn/Cu Pb/Zn/Cd P	Active Closed Potential Area	Hòa Bình	Zn/Pb/Cd Pb/Zn Cu	Potential Area Potential Area Potential Area
Thanh Hóa	Pb/Zn Zn/Au/Cu Pb/Zn P	Closed Active Potential Area Potential Area	Nghe An	Pb/Zn Pb/Zn/Sn Pb/Zn/Au P	Closed Potential Area Active Closed
Vinh Phu	Zn/Pb	Closed	Băc Thai	Zn/Pb/Ag/Cd	Closed
Ha Bác	Pb/Zn/Ba	Closed	Quang Ninh	P	Potential Area
Quang Binh	Pb/Zn P P	Closed Active Potential Area	Thua Thien Hue	Pb/Zn	Potential Area

Source: Atlas of Mineral Resources of the ESCAP Region: Volume 6, Viet Nam (ESCAP, 1990)

2.8 Heavy metal contamination associated with 'traditional' metal re-cycling villages.

Hai Hung Province in Vietnam is associated with numerous metal (primarily Cu and Al) re-cycling villages in which effective measures to control emissions, disposal of wastes and groundwater contamination are sub-optimal. Le Duc et al., (2002) investigated soil quality as impacted by Cu-recycling in Dai Dong village, Van Lam District, Hai Hung Province. Dai Dong village covers an area of 7,000 Ha and has an estimated resident population of 8,275. Household economy is based on agriculture with the main crops being paddy rice, corn, sweet potato and vegetables. Currently, 33% of the households are involved in Cu re-cycling. Initial results indicated that soil Cu contamination was directly related to the predominant wind direction and wastewater discharge channels. Maximum levels of total Cu in soils at a distance of 300m from the centre of the village ranged from 237 – 375 mg kg^{-1}. This is significantly higher than the Vietnamese (TCVN 7209:2002) and EU (EEC, 1986. Directive86/278/EEC) permissible levels for agricultural soils. However, the paper did not report potential phyto-toxic reductions in crop yields and associated declines in household income or health problems in the resident population.

2.9 Arsenic in ore deposits, surface and groundwater in Vietnam

In Vietnam, As occurs naturally in areas mineralized with sulfide minerals, gold and is also associated with areas of volcanism. The main areas of As concentration in Vietnam are in Pia Oac, Ngan Son, Pac Lang, Chay River, northern Tam Dao, Cho Don-Cho Dien, Dao Vien-Chiem, Hoa-Na Hang-Tung and Bac Me (Thanh, 2003).

Thanh (2003) identifies specific naturally occurring point sources of As concentration. These include As associated with gold ore in the basaltic Vien-nam formation, Doi Bu, Hoa Binh Province with As ranging from 50-204 mg kg^{-1} and As in a sulfur-gold-quartz mineralized zone of the Than Sa formation, Khau Au-La Hien, Bac Can, Thai Nguyen (Now known as Bac Thai Province) Province with an As concentrations in the ore range of 1292-1442 mg kg^{-1}. Further, in the Zn-Pb ore area of Cho Don, Bac Can (Bac Thai Province) Province, As content in the sericite shale and clay shale of the Coc Xo formation is 97.8 mg kg^{-1} with As concentration in the Zn-Pb ore ranging from 8.2-261mg kg^{-1}. In terms of water quality, Can, (2000) identified As levels of up to 730 µg l^{-1} in water originating from the gold ore zone east of the Pu Sam Cap region, Lang Vai, Lai Chau Province. In Bo Sinh-Moc Chau District, Son La Province the As level in springs from the right side of the Ma river ranged from 430-1130 µg l^{-1}. Tuyet (1998) identified naturally occurring As concentrations in karst water at Tay Bac in Son La Province ranging from 2290 - 3140 µg l^{-1} (Thanh, 2003). To put this in context, the Vietnamese Standard for As in drinking water is 50 µg l^{-1} (TCVN 5501-1991) and the WHO Standard for As in drinking water is currently 10 µg l^{-1}.

Anthropogenic sources of As are primarily associated with industrial activities. Duc (2001) recorded a maximum As concentration in groundwater of the Viet Tri industrial zone, Phu Tho (Vinh Phu Province) Province in northeast-western Vietnam of 320 µg l^{-1}. This significantly exceeds the Vietnamese and WHO Standards. Further, in the Thuong Dinh industrial zone, Hanoi area, As in untreated wastewater ranged from 145-346 µg l^{-1} (Duc, 2001). In 1999 with funding support from UNICEF, the As concentration in the groundwater of 7 provinces in the Red River Delta plain was investigated. A total of 1228 samples were analyzed and the results indicate that 60% of samples had an As concentration less than the WHO standard of 10 µg l^{-1}, whilst 87.5% of the samples analyzed had an As concentration less than the Vietnamese Standard for drinking water of 50 µg l^{-1} (TCVN 5501-1991). However, 12.5% of the samples had an As concentration >50 µg l^{-1} with a maximum value of 600 µg l^{-1}.

In 2000, UNICEF in collaboration with the Geological and Mineral Survey of Vietnam, (Northern Hydrogeochemical Engineering Geological Division) investigated As levels in groundwater in urban Hanoi and the surrounding districts of Soc Son, Dong Anh, Gia Lam, Thanh Tri and Tu Liem (Geological and Mineral Survey of Vietnam, 2001).

Groundwater quality was evaluated against the Vietnamese Standard for drinking water (TCVN 5501-1991), Vietnamese Standard (TCVN 5944-1995) for surface and wastewater and the WHO drinking water standard (WHO, 1993). Groundwater As pollution, was assessed in the shallow surface aquifers utilized for UNICEF wells namely the upper Holocene aquifer (qh) and the upper Pleistocene aquifer (qp_2) and the deep Pleistocene aquifer (qp_1) which is the primary source for industrial wells (public well fields and private wells).

The results indicate that for the upper aquifer, elevated levels of As were found in a high percentage of the dry season groundwater samples of Thanh Tri, Gia Lam Districts and within urban Hanoi (Table 11). In comparison, the rainy season was associated with a decrease in the percentage of samples with an As content >50 µg l^{-1} although Thanh Tri District urban Hanoi are still high risk areas (Table 11). Marked variations in As contamination between the upper and lower aquifer were also observed particularly with respect to season. The As contamination results for the lower aquifer indicate a potential public health risk to consumers of groundwater in Thanh Tri District.

Table 11. Arsenic levels in groundwater of the Hanoi area and surrounding districts as compared with the Vietnamese drinking water standard (TCVN 5501-1991)

Study area	Number of samples	Arsenic ($\mu g\ l^{-1}$)	
		>50	%
	Upper aquifer (qh)		
	I. Dry Season		
North area	66	4	6.1
Gia Lam area	20	8	40
Tu Liem area	55	8	14.5
Thanh Tri area	72	43	59.7
Urban area	47	18	38.3
	II. Rainy Season		
Gia Lam area	19	2	10.5
Tu Liem area	55	1	1.8
Thanh Tri area	72	29	40.3
Urban area	46	12	26.1
	Lower aquifer (qp)		
	I. Dry Season		
North area	46	4	8.7
Gia Lam area	72	13	18.1
Tu Liem area	25	9	36.0
Thanh Tri area	24	13	54.2
Urban area	43	17	39.5
	II. Rainy Season		
Gia Lam area	72	2	2.8
Tu Liem area	25	3	12.0
Thanh Tri area	23	9	39.1
Urban area	42	8	19.0

Adapted from Thanh, (2003)

Arsenic contamination of the Red River alluvial tract in Hanoi and surrounding districts was also reported by Berg et al., (2001) and Christen, (2001). Arsenic concentration in the groundwater samples analyzed ranged from 1-3050 $\mu g\ l^{-1}$ with a mean value of 159 $\mu g\ l^{-1}$. In a high risk area, the groundwater used directly as a household drinking water had a mean concentration of 430 $\mu g\ l^{-1}$. Analysis of untreated groundwater extracted from the lower aquifer for the Hanoi water supply yielded As levels of 240-320 $\mu g\ l^{-1}$ in 3 of the 8 treatment plants and 37-82 $\mu g\ l^{-1}$ in the remaining 5 plants. Aeration and sand filtration to remove excess Fe lowered the As concentration to 25-91 $\mu g\ l^{-1}$. However, 50% of the samples remained higher than the Vietnamese drinking water quality standard of 50 $\mu g\ l^{-1}$.

Berg et al., (2001) suggest that due to naturally occurring organic matter in the alluvial sediments, the groundwater aquifers are anoxic and rich in iron. Arsenic in the aquifer may be associated with Fe-oxyhydroxides released to the groundwater by reductive dissolution of Fe. This is indicated by the strong positive correlation ($R^2 = 0.700$) between total As and Fe concentrations in sediment.

This is corroborated by Nga et al., (2003) who observed that As was negatively correlated with SO_4^- and Oxidation Reduction Potential (ORP) indicating that As is leached from organic matter and/or iron oxyhydroxide under anaerobic conditions. Oxidation of sulphide phases could also release As to the groundwater but S concentrations in sediments were below 1 μg g^{-1} (Berg et al., 2001). Berg et al., (2001) suggest that the high As concentrations found in tube wells (48% >50 μg l^{-1} and 20% > 150 μg l^{-1}) indicate that several million people consuming untreated groundwater might be at a considerable risk of chronic arsenocosis. However, in 2001, no acute clinical symptoms of arsenocosis have yet been observed (Christen, 2001). Significantly elevated levels of Fe and Mn were recorded in groundwater samples irrespective of aquifer or season. Elevated levels of NH_4^+ in both the upper and lower aquifers were predominately found in Thanh Tri District and within urban Hanoi (Thanh, 2003). Nga et al., (2003) also reported levels of Fe and NH_4^+ in groundwater significantly exceeding the Vietnamese drinking water standard.

3. Thailand

3.1Background soil PTE concentrations

Zarcinas et al. (2004) assessed the PTE pollution of agricultural soils and crops in Thailand. A total of 318 soil (0-15cm) and 122 plant samples were collected from agricultural, forested and uncultivated sites in the central, eastern, northern, northeastern and southern regions of the country. The mean total soil As, Cd, Co, Cr, Cu, Hg, Ni, Pb and Zn concentrations in Thai soils are 7.5, 0.03, 6.0, 25.2, 14.1, 0.04, 13.5, 17.5 and 23.9, respectively (Table 12). However, at a few specific locations levels of heavy metals exceeded international standards and resulted in elevated levels of heavy metals in edible portions of the crops sampled (Zarcinas et al., 2004).

Table 12. PTE concentrations in Thai soils.

	Element concentration (mg kg^{-1})							
All Thai Soils (n=318)	As	Cd	Co	Cr	Hg	Ni	Pb	Zn
Mean	7.5	0.03	6.0	25.2	0.04	13.5	17.5	23.9
Median	2.7	0.01	2.8	15.3	0.03	6.2	9.0	14.0
Min	0.008	0.01	0.1	0.014	0.01	0.1	0.1	0.1
Max	124	1.3	113	295	0.27	270	550	140
95% Percentile 'Investigation Level' for Thai Soils	30	0.15	20	80	0.10	45	55	70

Adapted from (Zarcinas et al., 2004).

The results indicate that back-ground concentrations of As, Cd and Hg from non-agricultural soils were significantly lower than concentrations from agricultural soils while Co, Cr. Cu, Ni, Pb and Zn were not (Table 12). However, the number of non-agricultural soil samples was limited (n=33). Zarcinas et al., (2004), suggest that the adoption of the 95th percentile value of randomly selected agricultural and background soils be used as an 'Investigation Levels' above which Thai soils are considered to be contaminated (Table 12). Investigation levels do not indicate a potential hazard but rather that further investigation is needed to determine if the contamination is related to anthropogenic and/or could develop into a risk to the environment or public health.

3.2 PTE contamination and public health risks associated with point source contamination

In Thailand, heavy metal/metalloid documented contamination problems are primarily chronic As-poisoning in Ron Phibun District, Nakon Si Thammarat Province, chronic Pb-poisoning in Tong Phapum District, Kanchanaburi Province and Cd contamination of soils and crops in Tambon Phatat Padaeng, Mae Sot District, Tak Province.

3.2.1 Arsenic

Chronic human arsenism in Ron Phibun District, Nakon Si Thammarat Province was identified in 1987 by the Thai Ministry of Public Health (Timpatanapong et al., 1987). In 1988 a Thai Ministry of Public Health survey documented over 1000 cases of As-induced skin disorders. In addition, concentrations of As in hair and fingernails were found to be elevated in 80% of the school age population (Williams et al., 1998, Choprapawon and Ajjimangkul, 1999). Foy et al. (1992-1993) reported manifestations of palmoplantar keratoderma, hyper-pigmentation and two cases of Bowens carcinoma. Children with typical palmoplantar keratoderma were identified, the youngest being 4 years old. Foy et al., (1992-1993) reported As levels in shallow wells ranging from 20 - 2700 $\mu g\ l^{-1}$ with piped water having an As concentration of 70 $\mu g\ l^{-1}$. A follow-up study of 2400 school pupils in 1992 indicated that 89% of the study group had excess blood As concentrations with a 22% incidence of arsenical skin manifestations. (Williams et al., 1998). In 2000, Siripitaykunkit investigated chronic arsenism in Ron Phibun District and indicated that 24% of the population studied exhibited symptoms of arsenism (Siripitaykunkit, 2000). In a follow up studied conducted between 2000-2002 Pavittranon et al., (2003) report a prevalence of melanosis and hyperkeratosis of 5.99% and 8.67%, respectively.

The source of the As contamination was identified as leachate from As-rich mining and processing waste piles in the upper Hai Ron Na river, coupled with the naturally high abundance of disseminated arsenopyrite in alluvial deposits of the mid and lower catchment. Consequently, As concentrations in the shallow interstitial groundwater ranged from 100-5100 μg As l^{-1} (Williams et al., 1998). Fordyce et al. (1994) demonstrated a statistical correlation between the As concentration of the shallow well water at individual households and the body As burdens of residents. Further, Williams et al., (1996) demonstrate that As is present almost exclusively as the As(V) oxyanion, $H_2AsO^-_4$. This is significant with respect to risk assessment as $H_2AsO^-_4$ is significantly less toxic to humans than the arsenite oxyanion, $H_2AsO^-_3$ (Abernathy, 1993, Williams et al., 1998). In addition, Williams et al. (1998) suggest that the ingestion of As contaminated soil via the consumption of inadequately washed vegetables or as a result of Pica (direct ingestion of soil) in children may also be a significant exposure pathways that should not be ignored. Hazard mitigation programs initiated by the Thai Department of Mineral Resources (DMR) focused on the provision of an alternative 'safe' water supply. However, in a study investigating the use and provision of 'safe' water in over 3,000 households in Ron Phibun District in 1994 and 1997 Chongsuvivatwong et al. (2000) suggested that 'safe' water supply in the area was inadequate.

3.2.2 Lead

Tong Phapum District, Kanchanaburi Province is associated with extensive Galena (PbS) and Cerrussite ($PbCO_3$) deposits. Lead mining activities have been undertaken in the area for over 500 years. In 1997, heavy flooding in Tong Phapum District, Kanchanaburi Province resulted in the collapse of Pb-tailings impoundment dam adjacent to the Clitty Pb-ore concentrator. This resulted in significant Pb contamination of Clitty Creek (Tantanasrikul et al., 2002; Paijitprapapon, 2003). Following the collapse of the impoundment dam Pb levels in sediment upstream, onsite and downstream of the Clitty Pb-ore concentrator were reported to be 2,457, 12,941 and 40,053 mg kg^{-1}, respectively (Paijitprapapon, 2003). Mean Pb levels in water samples taken upstream, onsite and downstream of the Clitty Pb-ore concentrator were reported to be 0.017, 0.074 and 0.065 mg kg^{-1}, respectively (Paijitprapapon, 2003).

These levels are within the WHO water quality standards. The low levels of Pb in water may be attributed to immobilization and precipitation of $PbCO_3$. Following the accident, Pb levels in fish from Clitty Creek and in locally grown vegetables were reported to be > 2.0 mg kg^{-1} and 1.9 – 9.0 mg kg^{-1}, respectively (Paijitprapapon, 2003).

Mean concentrations of Pb in blood of children aged 0-14 yrs in the villages of Clitty Lang and Clitty Bon were 21.33 ±6.4 (n=15) and 36.26 ±6.5 (n=33), µg dl^{-1} respectively. With regards the WHO maximum permissible levels of Pb in blood, in Clitty Lang and Clitty Bon 13.3% and 81.8% of the children sampled (aged 0-14yrs) had blood Pb levels >25 µg dl^{-1} (Paijitprapapon, 2003). Similar levels were reported by Tantanasrikul et al., (2002). In 1998, environmental remediation measures were initiated. These proved ineffective at reducing the blood Pb levels in children monitored from 1998-2000 (Tantanasrikul et al., 2002). However, the use of chelation therapy with $CaNa_2EDTA$ lowered blood Pb in a group of pre-selected children by approximately 66.4% from the mean pre-trial value of 27.75 µg dl^{-1} to 12.38 µg dl^{-1}. In comparison, the use of the chelate DMSA reduced blood Pb by approximately 33.2%.

3.2.3 Cadmium

Collaborative research activities undertaken by the International Water Management Institute (IWMI) and the Thai Department of Agriculture (DOA) have identified significantly elevated levels of cadmium in soils, rice grain and rotation crops in Tambon Phatat Pha Daeng Ampher Mae Sot, Tak Province. Of the 524 rice fields sampled from 2000-2003 soil Cd concentrations range from 0. 5 – 284 mg kg^{-1}. In addition rice grain Cd concentrations ranged from 0.05 – 7.7 mg kg^{-1}. Over 90% of the rice samples analyzed contained cadmium at concentrations exceeding the Codex Commission on Food Additives and Contaminants (CCFAC) Maximum Permissible (MP) Level of 0.2 mg kg^{-1}. In addition, for the 76 fields sampled in May 2002 soybean Cd concentrations ranged from 0.34 – 3.37 mg kg^{-1}. All, of the soybean samples analyzed exceeded the CCFAC MP Level for cadmium in soybean of 0.2 mg kg^{-1} (Simmons et al., 2003; Simmons et al., 2004).

In Tambon Phatat Pha Daeng Ampher Mae Sot irrigation is sourced from Mae Tao Creek the upper stretches of which pass through an actively mined Zn-mineralized zone. Mae Tao Creek is utilized for irrigation by the eight communities with a combined resident population of 5,796 and an annual combined rice production of 7,592 t yr^{-1} (National Statistical Office, 2002). The total area under paddy rice for the 8 villages is 2,201 ha (National Statistical Office, 2002).

Farmer practice in the study area is to composite the harvested rice into a single bulk Field Group (FG) sample prior to threshing. To simulate this, for each field sampled, a sub-sample representing 10% of the total sample mass was collected and combined to form a bulk sample for each FG. These bulk samples were thoroughly mixed and a 500 g sub-sample taken. The Cd concentrations in the grain were then determined and used in the calculation of PTWI. For the demographic groups selected the estimated PTWI values based on rice intake alone ranged from 20-98 µg Cd kg body weight (Table 13 and Table 14). This poses a significant threat to public health.

In February 2004, the Pollution Control Department of the Thai Ministry of Natural Resources and Environment collected rice grain samples from 45 randomly selected households within Phatat Pha Daeng, sub-district. Rice grain Cd concentrations ranged from <0.05 – 5.0 mg kg^{-1} with a mean of 1.25 mg kg^{-1} (±0.92). Estimated PTWI values ranged from 2-244 µg Cd per kg body and confirm the findings of this study (Ministry of Natural Resources and Environment, 2004).
Cadmium levels in blood and urine samples collected from 250 residents within Phatat Pha Daeng, sub-district indicate potential renal dysfunction in 8% of the population studied (Ministry of Natural Resources and Environment, 2004).

Table 13. Estimated PTWI values of cadmium for selected demographic groups: Data derived from bulk samples of 2 Field Groups adjacent to Baan Pha Te Ampher Mae Sot District, Tak Province.

		Children 4-5yrs	Children 14yrs	Adults 20-29yrs	Adults 30-39yrs	Adults 40-49yrs
Body Weight (kg)*		16.2	42.4	56.2	57.8	59.3
Daily rice intake (kg)**		0.1	0.3	0.3	0.3	0.3
Field Group (FG) Bulk Rice Grain Samples	Rice grain Cd (mg kg^{-1})	Estimated PTWI value (µg Cd kg BW)				
Study Area B FG 1	1.777	87a	82a	62b	60b	59^b
Study Area B FG 2	2.014	98a	93a	70b	68b	67^b

Fourth National Food and Nutrition Status Survey Report, 1995; Thai Ministry of Public Health.** National Food and Nutrition Plan of the 8th National Economic and Social Development Plan, 1998. Within the same Field Group, mean values not followed by a common letter are significantly different at p<0.05 (two-tailed) as determined by t-Test: (Two Sample Assuming Equal Variances).

Table 14. Estimated PTWI cadmium values for selected demographic groups. Data derived from bulk samples from 7 of the 11 Field Groups within Phatat Pha Daeng, sub-district Ampher Mae Sot District, Tak Province sampled in November 2002.

		Children 4-5yrs	Children 14yrs	Adults 20-29yrs	Adults 30-39yrs	Adults 40-49yrs
Body Weight (kg)*		16.2	42.4	56.2	57.8	59.3
Daily rice intake (kg)**		0.11	0.28	0.28	0.28	0.28
Field Group (FG)	Rice grain Cd (mg kg^{-1})	Estimated PTWI value (µg Cd kg BW)				
Study Area D FG 2	0.959	47a	44a	34b	33b	32b
Study Area D FG 3	0.985	48a	46a	34b	33b	33b
Study Area D FG 4a	1.683	82a	78a	59b	57b	56b
Study Area D FG 5	0.685	33a	32a	24b	23b	23b
Study Area D FG 6	0.592	29a	27a	21b	20b	20b
Study Area D FG 7	1.291	63a	60a	45b	44b	43b
Study Area D FG 8	1.108	54a	51a	39b	38b	37b

* Fourth National Food and Nutrition Status Survey Report, 1995; Thai Ministry of Public Health.** National Food and Nutrition Plan of the 8th National Economic and Social Development Plan, 1998. Within the same Field Group, values not followed by a common letter are significantly different at p<0.05 as determined by t-Test: (Paired Two Sample for Means).

In Thailand, research published in international journals has focused on; remediation of As contaminated soils using native As-hyperaccumulator species (Francesconi et al., 2002; Visoottiviseth et al., 2002; Wongkomgkatep et al., 2003); elevated levels of As(V) in sediments, fish and crustaceans of the Pak Pa-Nang river estuary (Rattanacongkiat et al., 2004); elevated levels of Pb and Cd in soils utilized for grazing as a result of the long-term simultaneous application of P-fertilizers and manure (Parkpian et al., 2003); elevated levels of Cd, Pb and Hg in water spinach (Gothberg et al., 2002; Gothberg et al., 2003); elevated levels of Cd in the water and sediment of the Chao Phraya River and associated water ways (McLaren et al., 2004) and the bio-availability of heavy metals in sewage sludge-amended soils (Parkpian et al., 2000; Parkpian et al., 2002a; Parkpian et al., 2002b)

4. Cambodia

Information on PTE contamination in Cambodia is extremely limited and primarily associated with elevated levels of As in groundwater (Peng and Pichnimith, 2003; Raingsey, 2004) and Hg and As contamination of soils and surface waters due to artisanal gold mining (Sotham, 2003). To the author's knowledge, no comprehensive assessment of the current and potential magnitude and extent of PTE contamination in soils and crops of Cambodia has been undertaken (Personal Communication: Dr. Pak Sokharavuth, Deputy Director, Department of Pollution Control (DPC)). In Cambodia the opportunity may still exist to prevent this form of irreversible soil and water degradation at an early stage through the establishment of an effective legislative, regulatory, monitoring and enforcement framework supported by decision support tools and appropriate management solutions where applicable. Raingsey, (2004) reported on the results of a national field and laboratory based groundwater sampling program. This program investigated 5,000 groundwater samples and linked with existing studies conducted by UNICEF, JICA and WHO. The results indicate that 7.0% and 8.6% of water samples from open wells and hand pumps had an As concentration ranging from >10 - <50 $\mu g\ l^{-1}$ (Table 15.) The results also indicate that no open wells had an As concentration > 50 $\mu g\ l^{-1}$. However, 3.0% of water samples taken from hand pumps (Tube wells) had an As concentration ranging from >50 - <300 $\mu g\ l^{-1}$. In addition, 1.3% of all tube well water samples had an As concentration ranging from >300 - >500 $\mu g\ l^{-1}$. In terms of spatial distribution and aquifer, in Kandal, Svay Rieng and Prey Vang Provinces, groundwater As concentrations >10 $\mu g\ l^{-1}$ are primarily associated with an aquifer depth of 30-60m. In Kratie and Kompong Cham Provinces groundwater As concentrations >10 $\mu g\ l^{-1}$ are primarily associated with an aquifer depth of 30-50m (Figures 1-4).

Table 15. Groundwater As concentrations in Cambodia in relation to well type

	Percentage of samples within an As concentration within the pre-selected As water quality criteria	
As water quality criteria ($\mu g\ l^{-1}$)	Open Well	Hand Pump (Tube Well)
0	70%	63%
0 to <10	23%	24%
>10 to <50	7%	8.6%
>50 to <100	0.0%	1.4%
>100 to <300	0.2%	1.6%
>300 to <500	0.0%	0.9%
> 500	0.0%	0.4%
Cambodian Standard	50 $\mu g\ l^{-1}$	
WHO Standard	10 $\mu g\ l^{-1}$	

Adapted from Raingsey, (2004)

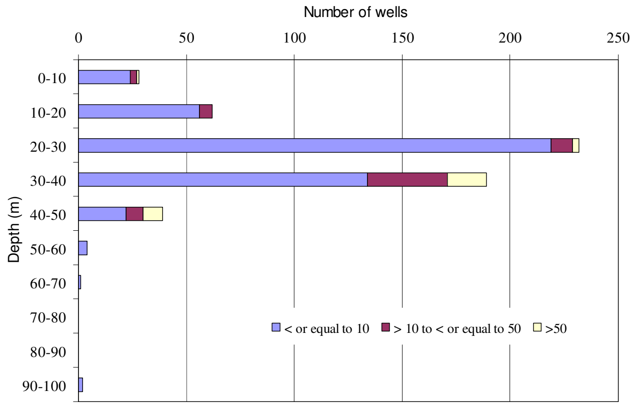

Figure 1. Arsenic concentration (μg l^{-1}) in groundwater as related to borehole depth: Kratie and Kompong Cham Provinces (Source: Raingsey, 2004)

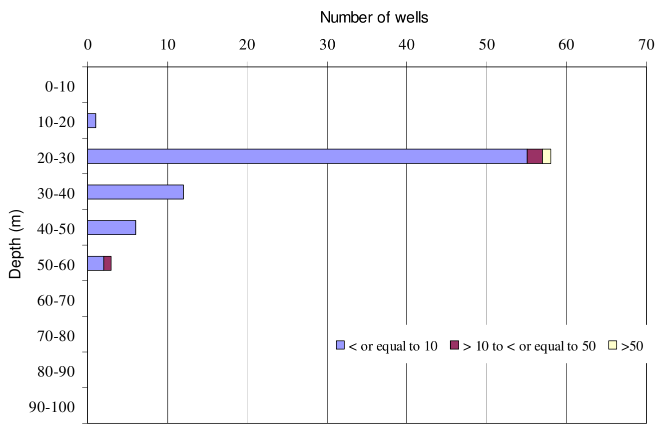

Figure 2. As concentration (μg l^{-1}) in groundwater as related to bore hole depth: Tonle Sap River, Kompong Chhnang Provinces (Source: Raingsey, 2004)

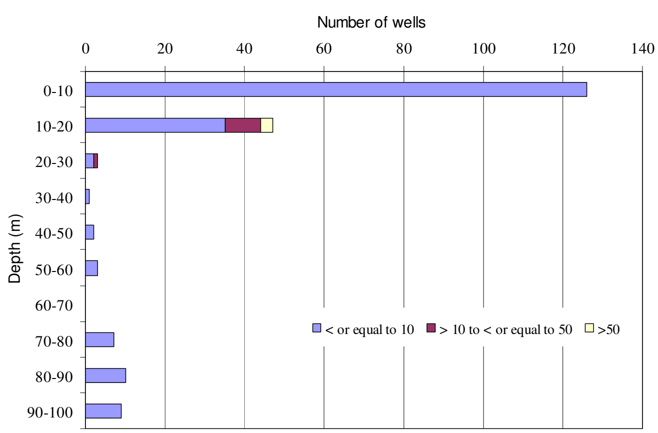

Figure 3. As concentration (μg l^{-1}) in groundwater as related to bore hole depth: Tonle Sap Lake, Kompong Thom Province (Source: Raingsey, 2004)

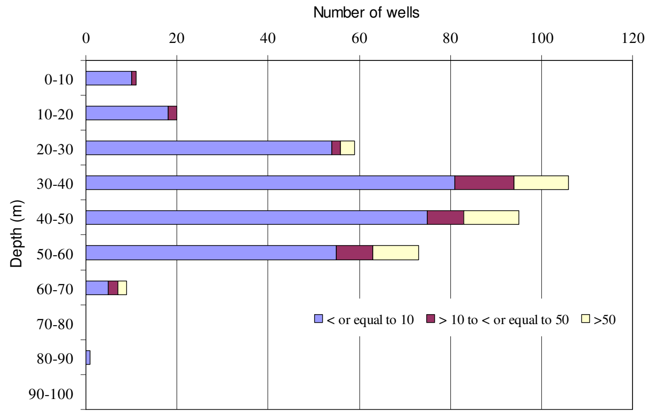

Figure 4. As concentration (µg l^{-1}) in groundwater as related to bore hole depth: Kandal, Svay Rieng and Prey Veng Provinces (Source: Raingsey, 2004)

The results of the national survey were used to produce a GIS-based 'As Risk Map'. High risk areas were characterized by 19% and 50% of tube well water having an As concentration > 50 µg l^{-1} and > 10 µg l^{-1}, respectively. Low risk areas were classified as areas with ~0.5% and 5% of tube well water having an As concentration > 50 µg l^{-1} and > 10 µg l^{-1} and 'Very Low Risk' areas characterized by ~0.5% and 3% of tube well water having an As concentration > 50 µg l^{-1} and > 10 µg l^{-1}, respectively (Raingsey, 2004). Raingsey (2004) estimated that a combined population of up to 30,080 people are exposed to groundwater with excess As levels in Kandal, Kampong Cham, Prey Veng and Kracheh Provinces. A detailed study of 1,470 households conducted in Kien Svay, Kandal Province indicated that of the 723 tube wells sampled, 35.63% had an As concentration > 50 µg l^{-1}. In addition this detailed survey identified 62 suspected arsenocosis patients. The mean age of tube wells was 4.5 yrs and mean tube well depth was reported as 43.5 m (Raingsey, 2004). It is important to note that As exposure via high-As groundwater in Cambodia is relatively short for the manifestation of visible symptoms of arsenocosis as compared to other countries (Bangladesh, West Bengal, Vietnam and China). However, the results reported by Raingsey, (2004) indicate the pro-active response of the Cambodian Government in collaboration with international organizations in mitigating this issue.

Sothem, (2003) briefly summarized the history and impact of artisanal gold mining in Cambodia. With reference to a case study of the Mermot gold deposit in Kampong province Sothem (2003) discussed the transition from panning to amalgamation methods with the subsequent domination of cyanidation. However, as a result of inadequate environmental controls during the period of amalgamation use soil Hg concentrations in Chaom Tamoa Commune of up to 25 mg kg^{-1} and As concentrations in the Antap River of up to 250 µg l^{-1} have been reported (Sothem, 2003). Although Sothem, (2003) refers to reduced cognitive ability in children no indication of Hg in surface or groundwater sources or dietary Hg intake via the ingestion of contaminated aquatic vertebrates is given.

5. Lao PDR

As is the case for Cambodia, information on PTE contamination in Lao PDR is extremely limited and primarily associated with elevated levels of arsenic in groundwater (Peng and Pichnimith, 2003) To the authors knowledge, no comprehensive assessment of the current and potential magnitude and extent of PTE contamination in soils and crops of Lao PDR has been undertaken (Personal Communication: Dr. Bounthoung Bouahom, NAFRI, Lao PDR).

6. Myanmar

No information available

7. China

Numerous international journal articles have reported heavy metal and metalloid contamination and, public health issues in China. In addition, several internet search engines including PubMed (http://www.ncbi.nih.gov/entrez/query.fcgi) have generated a considerable number of articles in Chinese Language Journals of which only the English abstracts are available. Of the over 100 articles referring to PTE contamination in China none have cited PTE contamination in Yunnan Province (within the Greater Mekong Sub-region). This is inconsistent with the intensive non-ferrous ore mining and processing activities and industrial base associated with Yunnan Province and the PTE contamination associated with non-ferrous ore mining and processing activities in other province of China. Yunnan Province, ranks first in the country in the deposits of zinc (Le and Kyle, 1997), lead, tin, cadmium, indium, thallium, and crocidolite.

Principal exports include tobacco, machinery and electrical equipment, chemical and agricultural products and non-ferrous metals. According to the Australia-China Chamber of Commerce and Industry of New South Wales http://www.accci.com.au/keycity/yunnan.htm Yunnan Province is one of the major production bases of copper, lead, zinc, tin and aluminum in China. Gejiu City is well known as "the kingdom of Zinc" with the reserves ranked first in the country. The Yunxi brand refined tin is one of the main products in Gejiu, which is registered on the London Nonferrous Metal Exchange. Other mineral resources include germanium, indium, zirconium, platinum, rock salt, nickel, phosphate, mirabilite, arsenic and blue asbestos.

7.1 Status of PTE contamination in soil, crops and water resources in China: Focus on Cd

Numerous authors have reported Cd contamination and several have identified health effects of Cd in populations not occupationally exposed to Cd in China (Cui et al., 2004; Jin et al., 2002; Wu et al., 2001; Jin et al., 1999; Cai et al., 1995; Cai et al., 1990; Nordberg et al., 2002; Nordberg, 2003; Zheng et al., 2003; Nordberg et al., 1997). Studies have primarily focused on rice producing areas contaminated with Cd as a result of discharges and/or emissions from non-ferrous mines, smelters and industrial zones.

Woo et al. (2000) identified elevated levels of Cd in surface and groundwater resources in the Hunchun Basin northeast China. All water samples contained Cd at concentrations exceeding the Chinese (0.010 mg l^{-1}) and WHO (0.005 mg l^{-1}) Drinking Water Standards. The average concentrations of Cd in surface and ground waters were 0.049 mg l^{-1} and 0.013 mg l^{-1}, respectively. In comparison, ECGC, (1993) reported Cd concentration in the shallow wells in the south-western part of the study area ranging from 0.0001 – 0.0023 mg l^{-1}. This implies an increase in Cd contamination of the shallow aquifer, from 1993-2000. Cadmium distribution in surface and ground waters of the Hunchun Basin were relatively uniform. Woo et al., (2000) suggest that this infers that Cd is derived from non-point sources and further state that agrochemicals are a possible source of Cd within the study area although this was not confirmed. Jing and Xue (2002) reported significant decadal loading of Cd and Pb to paddy soils of the Tai Lake region southern Jiangsu Province, China. This was achieved by the interpretation of archived data and re-analysis of archived soils in comparison to the laboratory analysis of field derived samples. The estimated loading rate of Cd primarily via Cd contaminated irrigation and fertilizer applications was reported as 0.3-3 μg kg^{-1} yr^{-1}. In addition, Wong et al. (2002) reported elevated levels of heavy metals including Cd in agricultural soils of the Pearl River Delta, South China. Further, Zhang et al. (2002) reported high levels of heavy metals in sludge from sewage discharge channels of Tianjin City, Tianjin Province, China.

Cui et al. (2004) investigated possible health risks to the local population living in close proximity to a non-ferrous smelter in Nanning, Guangxi Zhuang Autonomous Region, southern China. Results indicate that soil and vegetables were contaminated with Cd and Pb. The mean Cd and Pb concentrations in vegetables ranged from 0.15-0.24 mg Cd kg^{-1} and 0.38-0.45 mg Pb kg^{-1}, respectively. The CAC MP limit for Cd and Pb in leafy vegetables is of 0.2 mg Cd kg^{-1} and 0.3 mg Pb kg^{-1}. Cui et al. (2004) state that oral intake of Cd and Pb through vegetables poses a 'high health risk to local residents'. However, no indication of daily Cd and Pb intake via the vegetable food chain is given. In addition, no data is provided on rice grain Cd concentrations and levels of dietary Cd intake via the rice food chain.

Zheng et al. (2003) reported Cd contamination in soils and crops adjacent to a copper smelter in Daiya city, Hubei Province. This is the second largest Cu smelter in China. Of 98 soils samples collected, the maximum Cd concentration was 9.65 mg kg^{-1} for a paddy soil receiving irrigation contaminated with untreated discharge from the Cu-smelter. The soils in the area are predominantly acidic. As a result, the average percentage of exchangeable Cd (1M $MgCl_2$ extractable) to total Cd was 64 ±8.9 % with a maximum of 80%. A significant linear relationship (R^2=0.956, n=46 significant at 1%) was observed between soil Cd concentration and rice grain Cd content. Results indicate that when total soil Cd concentration was >0.261 mg kg^{-1} rice grain Cd concentration exceeded the Chinese National Food Sanitation Standard for rice of 0.2 mg Cd kg^{-1} (Zheng et al., 2003).
Further, Nan and Zhao, (2000) and Nan et al. (2002a, b) reported significant Cd, Zn, Pb and Cu contamination of soils within a 500 km^2 area receiving wastewater from a non-ferrous mining and smelting base in Baiyin, Gansu Province China. Nan et al., (2002a) reported elevated levels of Cd and Zn in agricultural soils of the Baiyin region Gansu Province utilized for wheat (*Triticum aestivum* L.) and corn (*Zea mays* L.) production that receive irrigation contaminated with waste from non-ferrous mining and smelting. Soil Cd and Zn concentrations ranged from 0.14-19.3 mg kg^{-1} and 43.5 – 565 mg kg^{-1}, respectively. However, low levels of grain accumulation were reported.

Zabowski et al. (2001) reported elevated levels of Cd, Pb, Cu, and Zn in soils, water and sediment resulting from the un-controlled discharge of wastes from Au and Fe mines in Hei, River Basin, Kansu Province, China. In addition, Zang et al, (2004) reported levels of Pb and Zn in water of 6780 and 324 μg l^{-1}, respectively close to main tailings derived from Shanshulin Pb-Zn mine in Guizhuo province China as compared to background levels of 3.71 and 11.6 μg l^{-1}, respectively. Thirty kilometers downstream Pb and Zn concentrations in water drop to 3.15 and 16.4 μg l^{-1}. In contrast Pb and Zn contents in sediment close to the tailings are 4553 and 7971 mg kg^{-1} and are still high 30 km downstream with concentrations of 3334 and 7268 mg kg^{-1}.

Elevated levels of Cd, Pb, Cu, Hg and Zn in water and sediments of the Xiang River, Zhuzhou City, Hunan Province, China were reported by Haiyan, (2003). Associated with these soil levels, elevated concentrations of heavy metals where observed in vegetables and rice grain exceeding the Chinese Standard.

7.2 Impacts of Cd on public health

Nordberg et al., (2002), investigated bone mineral density (BMD) and bio-indicators of renal dysfunction in population groups exposed to Cd via rice. Decreased BMD was found in postmenopausal women with elevated levels of urinary-Cd (Cd-U) and/or Cd in blood (Cd-B). In addition, clear and statistically significant dose-effect and dose-response relationships were found between Cd-B and Cd-U and renal dysfunction as indicated by increased excretion of retinol-binding protein. The authors suggest that this is the first report of bone effects among Cd-exposed population groups in Asia outside of Japan. Jin et al., (1999), Wu et al., (2001) and Jin et al., (2002) also reported renal dysfunction in populations consuming rice contaminated with Cd via non-ferrous smelting and ore processing activities.

Cai et al. (1990, 1995) investigated Cd exposure among residents in Dayu County, Jiangxi Province China. Irrigated arable land is contaminated with Cd from tailings and wastewater from tungsten ore dressing plants. Mean concentrations of Cd in irrigation water and soils were 0.05 mg l^{-1} and 1.0 mg kg^{-1}, respectively. The average oral intake of Cd was calculated to be 367-382 μg d^{-1}. Approximately 99.5% of the oral Cd intake came from Cd contaminated rice (Cai et al., 1995). Assuming a body weight (BW) of 70 kg this would equate to a PTWI value of 36.7-38.2 μg Cd per kg BW. This significantly exceeds the JECFA PTWI value of 7 μg Cd per kg BW and indicates a significant public health risk. Among residents exposed to elevated levels of dietary Cd for >25yrs cadmium adsorption as indicated by urinary Cd ≥10 μg g $creatine^{-1}$ was 60%. Further, an early indicator of renal dysfunction namely urinary Cd >15 μg g $creatinine^{-1}$ and urinary β-2-microglobulin >500 μg g $creatinine^{-1}$ were detected in 17% of the 433 people evaluated (Cai et al., 1990).

Han (1988) investigated the effects of Cd exposure on populations living in close proximity to Cd industries in five provinces in China. Mean Cd-U and the incidence of low-molecular-weight protein in urine was positively correlated with total dietary Cd intake via Cd contaminated rice. The incidence of low-molecular-weight protein was significantly elevated in the exposed groups with a total per capita daily Cd intake of 113 μg Cd d^{-1}. Based on their research Han (1988) suggest that the acceptable daily intake (ADI) of Cd be 100 μg Cd d^{-1}. However, at this ADI, assuming a body weight of 70 kg, this would generate a PTWI value of 10 μg Cd kg BW which exceeds the JECFA MP level (JECFA, 2003). Nordberg et al. (1997) collected rice and urine samples obtained from three areas in Zhejiang Province China representing a highly exposed, medium exposed and control area. Cadmium concentrations in rice grain were 3.70, 051 and 0.072 mg kg^{-1} for the highly exposed, medium exposed and control area, respectively. Nordberg et al. (1997) demonstrate a clear statistically significant dose-response relationship between cadmium in urine, β-2-microglobulin excretion and dietary Cd intake via rice.

In contrast to southern China, in northeast China the population relies not on rice but wheat and other cereals and to a lesser extent on pulses for calorific intake. Zhang et al. (1998) investigated Pb and Cd levels in cereals and pulses collected from open markets in north-eastern China. Among cereals, Pb concentrations in foxtail millet, maize, wheat and rice flour were 0.054, 0.035, 0.028 and 0.022 mg kg^{-1} respectively. In comparison, Zhang et al. (1998) reported Pb levels in kidney bean and soybean of 0.024 and 0.030 mg kg^{-1}. The mean Cd concentration in cereals and pulses was 0.09 and 0.057 mg kg^{-1} with a maximum mean Cd value of 0.073 mg kg^{-1} in soybean. It is important to note that the levels of Pb and Cd in cereals and pulses reported by Zhang et al. (1998) are significantly below the MP levels established by CCFAC and indicate, that the 'background' Pb and Cd concentrations in agricultural products of north-eastern China do not pose a risk to public health.

Watanabe et al. (2000) investigated background Pb and Cd exposure of adult women in Xian City and two farming villages in Shaanxi Province, China. The geometric mean daily Pb and Cd intake via foods was 30 μg d^{-1} and 6.1 μg d^{-1}.

Assuming a body weight of 56 kg this would equate to PTWI values of 3.75 and 0.763, respectively which are significantly lower than the JECFA PTWI values (JECFA, 2003). Cereals (wheat, rice, maize and foxtail millet) contributed 26 and 84% of dietary Pb and Cd intake. This demonstrates the significance of the dietary Cd pathway to human Cd exposure.

7.3 Arsenic

Chronic endemic arsenism via drinking water was first found in Taiwan in 1968 (Tseng et al., 1968) and reported in Xinjiang Province in mainland China in the 1980s (Huang, et al., 1985; Xia and Liu, 2004). Arsenic related skin lesions such as hyperkeratosis, pigmentation and depigmentation have been identified in Xinjiang Province (Wang Lianfang et al., 1994), Inner Mongolia (Wu Kegong et al., 1994a, b; Li Guojun et al., 1994; Wu Deqing et al., 1992; Luo Zhendong et al., 1994; Sun Yude et al., 1994; yang et al., 2002), Shanxi Province (Wang Jinhaua et al., 1998) and Ningxia Province (Hu Xingzhong et al., 1999), China (Table 16).

Table 16. Summary of chronic endemic arsenism via drinking water in mainland China.

Province/ Autonomous Region	No. of clinical cases of arsenism	Population exposed to elevated levels of As in water supply
Inner Mongolia	5,666 (MOH, 1998)	200,000 (Wu Kegong et al., 1994a) 2,931,800 (MOH, 1998) 300,000 (Guo et al., 2003)
Xinjiang	523 (Wang et al., 1994)	50,760 (Fang et al., 1995)
Shanxi	5,213 (MOH, 1998)	989,600 (MOH, 1998)
Ningxia	628 (Hu et al., 1999)	100,000 (Wang, 1997)
8 Provinces	7821 (Jin et al., 2003)	2,343,238 (Jin et al., 2003) 522,566 (As > 50 μg l^{-1})
National Survey (Sheng and Jiu, 1997)	10,000	9,200,000 (As 30-49μg l^{-1}) 3,340,000 (As 50-99μg l^{-1}) 2,290,000 (As >100 μg l^{-1}) 14,600,00 (As >30 μg l^{-1})

According to the Chinese National drinking water standard over 2 million people are exposed to high-As and over 10,000 people have been diagnosed with arsenism. A summary of As biogeochemistry and human health as related to elevated levels of As in groundwater and As in coal in the People's Republic of China according to a Pressure-State-Impact Response (PSIR) approach is given in Table 17 (Peterson et al., 2001).

The Inner Mongolia Autonomous Region contains the largest endemic arsenism area of China. Arsenic related skin lesions have been identified in 11 counties in Inner Mongolia (Linsheng et al., 2002). In 1998, a total of 2,666 patients had been identified (MOH, 1998). More than 200,000 people live in the disease affected areas and 10-37% of the wells are contaminated with As (Wu Kegong et al., 1994a). The distribution of arsenism is based upon the occurrence of As in a fault basin (Yu Xiaoying, 2001).

Guo et al. (2003) report that in Inner Mongolia, China more than 300,000 people are chronically exposed to As via drinking water. In the Hetao plain area, prevalence of arsenical dermatosis was as high as 40% (Guo et al., 2001, 2003) Results indicated that As contaminated ground water from tube wells of depths ranging from 15-30m was serious when compared with open wells with a depth of 3-5m (Guo et al., 2001). In the Wuyuan area, 96.2% of water samples from tube wells contained As >50 μg l^{-1}, with 69.3% of tube wells in the Alashan area, contaminated.

In Wuyuan and Alashan maximum recorded values of 1354 μg l^{-1} and 1088 μg l^{-1} As respectively. Prevalence of arsenical dermatosis in the Wuyuan and Alashan areas was 44.8% and 37.1%. Pi et al.

(2002) undertook epidemiological studies in Wuyuan, Inner Mongolia and reported evidence that chronic exposure to As from drinking water in humans' results in induction of oxidative stress as indicated by the reduction in nonprotein sulfhydryl (NPSH) in whole blood and the increase in lipid peroxides (LPO). Subjects in the high-As-exposure group (mean As in groundwater of 410 $\mu g\ l^{-1}$) had NPSH levels 56% lower than the low exposure group (mean As in groundwater of 20 $\mu g\ l^{-1}$). Epidemiological and clinical studies on the water from 102 wells containing high levels of F and As utilized by 50,760 people in Kuintun area, Xinjiang Province. Results confirmed that the long term consumption of naturally contaminated F and As drinking water resulted in fluorosis and arsenism (Fang et al., 1995). Further, Jin et al., (2003) evaluated arsenism in 21 provinces in China and reported that drinking water induced endemic arsenism is distributed in 8 provinces, 40 counties affecting 2,343,238 people with 522,566 people exposed to drinking water with As levels >50ug l^{-1} 7,821 arsenism patients have been diagnosed.

From the analysis of 28,000 water samples it was estimated that 9.02 million people were exposed to drinking water with an As content ranging from 30-49 $\mu g\ l^{-1}$ (Sheng and Jiu, 1997). Further 3.34 million people were exposed to drinking water with an As concentration of 50-99 $\mu g\ l^{-1}$ and 2.29 million people were exposed to drinking water with an As concentration >100 $\mu g\ l^{-1}$. A total of 14.6 million people were exposed to drinking water with an As content of >30 $\mu g\ l^{-1}$. It was estimated that 80% of high As drinking water was from ground water (Sheng and Jiu, 1997).

Guo et al., (2003) investigated the natural occurrence of As in shallow groundwater in Shanyin, Shanxi Province, China. Arsenic concentration in shallow groundwater exceeds the National and WHO guidelines for drinking water of 50 and 10 $\mu g\ l^{-1}$ with values up to 1932 $\mu g\ l^{-1}$ (MOH, 1985; WHO, 1981). Groundwater with high-As was characterized by higher pH, higher concentrations of phosphate and naphthenic acid and lower concentrations of sulphate and nitrate as compared to low-As groundwater.

The lower concentrations of sulphate and nitrate where attributed to microbial metabolism of sedimentary organic matter, which is present as high as 1.0% Org-C. This reaction decreases Eh and produces CO_2 which promotes the dissolution of carbonates, resulting in the observed high pH. The clay minerals and colloids including organic matter which scavenge As would release As to ground water in higher pH and low Eh conditions. In addition, the competitive adsorption between As and anions also high in the shallow groundwater, namely P and F contributes to the release of As from kaolinite, montmorillonite, illite and Fe oxyhydroxides. Naphthenic acid also promotes mobilization and translocation of As in groundwater.

Table 17. A summary of As biogeochemistry and human health as related to elevated levels of As in groundwater and high-As coal in the People's Republic of China (Peterson et al., 2001)

Pressure
Drinking water type
(As) in bed-rock and minerals
Gneiss: 12.40 μg g^{-1} (Wang et al., 1998)
Pyrite: 32.10~70.60 μg g^{-1} (Li and Li, 1994a)
Groundwater demand
2.5 1/person-day (Wang, 1997) for drinking and cooking
Disease identification
Affected people
Inner Mongolia: 5,666 (MOH, 1998)
Xinjiang: 523 (Wang et al., 1994)
Shanxi: 5,213 (MOH, 1998)
Ningxia: 628 (Hu et al., 1999)
Affected villages (countries)
Inner Mongolia: 151(12) (MOH, 1998)
Xinjiang: 77 (1) (Wang et al., 1994)
Shanxi: 72(4) (MOH, 1998)
Ningxia: 2(1) (Hu et al., 1999)
Population in affected countries
Inner Mongolia: 2,931,800 (MOH, 1998)
Shanxi: 989,600 (MOH, 1998)
Xinjiang: about 100,000 (Wang, 1997)
Coal-Smoke type
(As) in coal
381~9600 μg g^{-1} (Zheng et al., 1994)
Coal consumption : 145.5 kg/yr. person
(average in China)
Affected people
Guizhou : 2709 (MOH, 1998)
Affected villages
Guizhou : 42 (MOH, 1998)
People in affected countries
Guizhou : 2,265,100 (MOH, 1998)
State
Drinking water type
(As) in groundwater in disease areas
Inner Mongolia:0.05~1.86 μg ml^{-1} (Wu and Xing, 1993)
Ningxia: 0.02~0.17 μg ml^{-1} (Hu et al., 1999)
Xinjiang : 0.1~0.70 μg ml^{-1} (Wang et al., 1994)
Shanxi: 0.05~ >0.5 μg ml^{-1} (Wang et al., 1998)
Total As intake
684.4~3,422 mg/person.yr (Wang, 1997)

Table 17. Continued

Impact

(As) in urine as biomarker:

Inner Mongolia

Zhijiliang: 0.300 μ m^{-1} (Wu *et a*l., 1992)

Hanghou 0.254 μ ml^{-1} (Zhang e*t a*l., 1993)

Guizhou : 0.44 ~ 1.34 μ gml^{-1} (Zheng e*t a*l., 1994)

(As) in hair as biomarker

Inner Mongolia

Zhijiliang 4.733 μ g g^{-1} (Wu e*t a*l., 1992)

Linhe 2.07 μ g g^{-1} (He e*t a*l., 1994)

Azuo : 1.99~4.46 μ g g^{-1} (Yang e*t a*l., 2000)

Guizhou : 10.8 ~ 53.5 μ g g^{-1} (Zheng e*t a*l., 1994)

17.4~38.6 μ g g^{-1} (Zhong *et a*l., 1999)

Other biomarkers

Arsenism incidence

Inner Mongolia (Sun e*t a*l., 1995) :

Keqi : 66.89% ; Tuzuo : 5.80% ;

Tuyou : 12.04% ; Hanghou : 44.59% ;

Linhe : 13.04%; Wuyuan : 21.09% ;

Azuo : 15.28% ; Wuhou : 2.67%

Ningxia : 11.71% (Hu e*t a*l., 1999)

Shanxi : upto 50.6% (Wang e*t a*l., 1998)

Cancer incidence

Inner Mongolia : 12 cases lung cancer ;

133.30/100 thousand persons (Liang e*t a*l., 1999)

Xinjiang: 23 cases cancer, 1/3~1/2 of total deaths (Wang, 1997)

Guizhou: 1578.95/100 thousand persons (Wang, 1997)

Response

Community awareness raising from 0 to 100%

Health supplement expenditure

Inner Mongolia: $420,000 (calculated by Xing and Xie, 1997)

Water treatment expenditure

Inner Mongolia: $4,120,000 (calculated by Xing and Xie, 1997)

Re-design of local ovens and installation of chimneys expenditure

Scientific expenditure

$400,000 (calculated all endemic arsenism projects in China now)

Number of people changing water supply to low arsenic safe water

Inner Mongolia: 38,100

Shanxi: 3,300 (MOH, 1998)

People using safety ovens to reduce arsenic pollution: 12,800 (MOH, 1998)

Source: Peterson et al., 2001

7.4 Potential uptake of As to edible portions of crops and food chain As contamination

The impacts of elevated levels of As in groundwater on human health are well documented. However, of increasing concern is the potential contribution to dietary As intake of As derived from rice grown on As contaminated soils. As compared to other crops, rice is of primary concern due to the relative mobility of As under reducing conditions associated with paddy soils and the high levels of daily rice consumption. Meharg and Rahman (2003) indicate that in several areas of Bangladesh, the long term utilization of As contaminated groundwater for supplementary dry season irrigation has resulted in elevated levels of As in paddy soils with As levels in the 0-15 cm of paddy soils ranging from 3.1 – 42.5 mg As kg^{-1}. Ullah (1998) and Alam and Sattar (2000) reported similar values in the 0-15cm of paddy soils of up to 83 mg As kg^{-1} and trace – 31.8 mg As kg^{-1}, respectively.

Both Ullah, (1998) and Alam and Sattar, (2000) attributed the elevated levels of soil As to the use of As contaminated groundwater for irrigation. As a direct result, significantly elevated levels of As in rice grain have been identified. Rice grain As concentrations collected from western districts of Bangladesh ranging from 0.058 to 1.83 mg As kg^{-1} (Meharg and Rahman, 2003). The highest rice grain As concentrations were obtained from Nawabgonj and Naogaon districts. In addition, Xie and Huang (1998) observed rice grain As concentrations of up to 0.72 mg As kg^{-1} grown on soils containing 68 mg As kg^{-1}. Research in West Bengal India indicate elevated levels of As in soil and rice grain in areas dependent on groundwater for irrigation (Roychowdhury et al., 2002). In contrast, Alam et al. (2003) reported levels of As in snake gourd, ghotkol, taro, green papaya, elephant foot and bottle ground leaf of 0.489, 0.446, 0.440, 0.389, 0.338 and 0.306 mg kg^{-1}, respectively.

The As concentration in leafy vegetables was low (Alam et al., 2003). In comparison, data from a national survey of agricultural soils in Thailand identified paddy soil As concentrations ranging from 0.32 – 92.4 mg As kg^{-1} and rice grain As concentrations ranging from 0.22 – 1.78 mg As kg^{-1} (Zarcinas et al., 2003). The elevated levels of As were attributed to the long term application of As contaminated phosphate fertilizers (Zarcinas et al., 2004). Pesticides such as lead arsenate and sodium arsenite although banned in many countries are also major sources of As to the agricultural environment. The quantity of soluble or potentially soluble As in a soil varies with soil pH, Eh, presence of iron and aluminium oxides, clay minerals and organic matter. In general, there is an increase in As solubility and hence As bio-availability as soils become more acidic particularly at pH's below 5.0 when As binding species such as iron and aluminium oxy-compounds become more soluble. In addition, bio-available As content of soils has been shown to increase 25x when moving from an acidic oxidizing environment (Eh 500mV) at pH 4.0 to a reducing environment (Eh -200mV) at pH 6.9 (Masscheloyn et al., 1991). Consequently, As is likely to be more available in submerged paddy soils as compared with aerobic soils utilized for vegetable production. The Provisional Tolerable Weekly Intake (PTWI) value for arsenic as established by JECFA in 1988 is 0.015 µg As kg BW. For the rice grain As concentrations reported by Meharg and Rahman, (2003) of 0.058 – 1.835 mg As kg^{-1} and assuming a daily rice intake of 0.42kg (Baffes and Gautam, 2001) and body weight for an adult male (20-40 yrs) in rural areas of 55 kg (Bangladesh Bureau of Statistics, 2003), PTWI values would range from 3.10 – 98.09 µg As kg BW. This is from 206 to 6,539 times the JECFA PTWI value of 0.015 µg As kg BW. Clearly, therefore, the consumption of As contaminated rice grain makes a significant contribution to dietary As exposure.

However, to put this in context, as established by JECFA in 1988, the WHO drinking water standard for As is 0.01 mg As l^{-1}. At a daily rate of consumption of 2 liters, and body weight of 55 kg (Bangladesh Bureau of Statistics, 2003) this would correspond to a PTWI value of 2.55 µg As kg BW. This is over 169 times the JECFA PTWI value of 0.015 µg As kg BW. JECFA, (1988) noted that the margin between the PTWI and intakes reported to have toxic effects in epidemiological studies is narrow (WHO, 1996). In addition, in 1994 the 20th Session of the Codex Alimentarius Commission decided to discontinue the establishment of guidelines for As in cereals and pulses as arsenic did not appear to be a problem in international trade. Consequently, the draft guideline for As in cereals and pulses of 0.5 mg As kg^{-1} was withdrawn. In the light of the elevated As levels in rice grain discussed here, this decision needs to be re-evaluated. Arsenic is present in rice grain mainly as inorganic As (arsenate and arsenite) and methylated species (monomethyl arsenic acid and diemethyl arsenic acid (Schoof et al., 1998; Tao and Bolger, 1999; Heitkemper et al., 2001). All these forms of As, are toxic and readily assimilated into the blood stream (WHO, 1981). He et al. (2000) identified that As in rice grain is primarily accumulated in the endosperm and bound to proteins. In addition, He and Yang, (2002) observed that the protein-binding form of As in rice grain is easily decomposed under steam and digestive enzyme treatment. However, the bioavailability of rice derived arsenic in human stomachs is not fully understood (Meharg and Rahman, 2003). Elucidating this process is essential in order to determine the actual contribution of rice derived As to the total body burden.

7.5 Impact of burning coal with high-As on public health

In China several hundred million people commonly burn coal in unvented stoves that permeate their homes with high levels of As and F (Finkelman et al., 1999). In excess of 3,000 people in the southwest prefecture Guizhou Province in southwest China are suffering severe As poisoning as a result of consuming food dried using As-contaminated coal (Finkelman et al., 1999; Liu, et al., 2002). Skin lesions are common, including keratosis of the hands and feet, pigmentation on the trunk, skin ulceration and skin cancers. Toxicities to internal organs are also clinically evident (Liu, et al., 2002). Approximately 200,000 people are at risk of overexposure. Jin et al., (2003) suggest that coal endemic arsenism is associated with 2 Provinces and 8 counties affecting 333,905 people 48,438 people exposed to high As burning coal pollution with 2402 patients clinically diagnosed. Zhonghua et al., (1993) reported that 1548 villagers from 47 villages in Guizhuo Province suffered from chronic arsenic poisoning. Coal samples in the region contain up to 35,000 mg kg^{-1} As. Chilli peppers dried over high-As coal absorb As with a mean As content of 500 mg kg^{-1} (Zeng et al., 1996; Finkelman et al., 1999; Liu, et al., 2002). Health effects of As exposure from burning high As-containing coal in Guizhou, China were investigated in detail by Liu, et al., (2002) who estimated that sources of total arsenic exposure in the area are from As-contaminated food (50-80%), air 10-20%, water 1-5% and direct contact in coal mining workers (1%). Arsenic content in the drinking water supply was <50 µg l^{-1} (Zeng et al., 1996). Shraim et al., (2003) confirmed that in Changqing, Ghuizhou Province, 30% of the resident population demonstrated clinical symptoms of arsenocosis.

7.6 Fluoride

Endemic fluorosis is manifested either as dental flourosis with the appearance of yellowish to brownish striation and/or mottling of teeth, or skeletal flourosis where osteosclerosis, ligametous and tendinous calcification and extreme bone deformity result (WHO, 1984; Yang et al., 2003). As reported in Yang et al., (2003) in China flourosis occurs in all provinces, autonomous regions and municipalities except Shanghai (Tan Jian'an, 1989) (Table 18).

Table 18. Indicators of fluorosis in different provinces of China: Adapted from Yang et al. (2003)

Province	Population Exposed (%)[a]	Urinary Fluoride (mg l^{-1})[b]	Dental Occurrence (%)	Skeletal Occurrence (%)
Tianjin (TJ)	27.41	2.15	56.5	35.2
Shanxi (SX)	16.06	3.23	70.2	33.1
Shandong (SD)	9.47	0.87	30.9	27.3
Ningxia (NX)	11.94	2.92	75.1	13.1
Neimenggu (NM)	24.51	2.05	88.2	67.2
Liaoning (LN)	4.44	1.5	12.2	4.7
Jilin (JL)	6.84	3.16	51.3	22.9
Jiangsu (JS)	6.81	4.28	78.8	38.6
Henan (HN)	16.39	1.48	77.4	32.8
Heilongjiang (HL)	8.16	3.24	63.3	26.5
Hebei (HB)	13.58	1.08	40.2	15.9
Guangdong (GD)	0.92	3.02	12.2	35.5
Gansu (GS)	14.27	3.66	72.9	7.5
Anhui (AH)	11.90	3.75	31.8	16.5
Mean (STDEV)	12.3 (7.3)		54.4 (25.1)	26.9 (15.3)

[a] Percent of population was calculated using the population numbers in disease villages (MOH, 1998) divided by the total population in the province (NSB, 1999).

[b] Urinary fluoride concentration, dental fluorosis occurrence and skeletal fluorosis occurrence were the mean of the national monitoring stations in the different provinces in 1991. The data for Shanxi Province is 1992 (Sun Yufu *et a*l., 1990).

By the end of 1997, there were approximately 110 million people living is the disease areas, 45 million with dental flourosis and 2.6 million with skeletal flourosis (MOH, 1998). There are 2 main sources for endemic flourosis in China namely, high fluoride water and food chain F contamination resulting from the use of high-F coals (Wang and Huang, 1995). When the F concentration in drinking water increased to approximately 0.8-1.0 mg l^{-1}, dental flourosis rapidly increases (Dai Goujun et al., 1988). When F in drinking water increased to approximately 2 mg l^{-1} almost 100% of the exposed people developed dental flourosis and when water F concentration increased to over 4.0 mg l^{-1}, the incidence of skeletal fluorosis increases rapidly while crippling skeletal fluorosis develops where drinking water concentrations exceed 10 mg F l^{-1} (Yang et al., 2003). The Chinese drinking water guideline is 1.0 mg l^{-1} (Sun Yufu et al., 1996) as compared to the WHO guideline value of 1.5 mg l^{-1} (WHO, 1996). The principal route of F-excretion is via urine, hence urinary-F has been used to estimate dose (WHO, 1984). The west plain region of Jilin province is an endemic fluorosis area resulting from the long-term consumption of F-contaminated groundwater. Fluoride levels > 1000 µg l^{-1} resulted in dental fluorosis and skeletal fluorosis in local residents. Strong positive correlations were observed between F concentration in groundwater and morbidities of endemic fluorosis disease (R^2 = 0.872). Bo et al. (2003) determined that at F concentration >1000 µg l^{-1} exposed residents are at high risk of developing F related diseases. On this basis 68% of the residents of the west plain region of Jilin province are at risk. In northern China, high levels of F in groundwater's caused regional endemic fluoride poisoning resulting in fluoro-bone symptoms and endemic dental fluorosis (Liu and Zhu, 1991). Elevated levels of F in groundwater were also identified in Hunchun Basin by Woo et al. (2000) who reported F levels in groundwater ranging from 1.0 – 7.8 mg l^{-1}. Management options to mitigate endemic fluorosis in areas with high-F in groundwater have primarily focused on the provision of new water supply systems with F concentrations of < 1 mg l^{-1} (Table 19).

Table 19. Implementation of new water supply systems with F concentrations of < 1 mg l^{-1} in endemic flourosis areas of China: Adapted from Yang et al. (2003)

Province	New water supply Systems (%)	New water supply System with <1mg l^{-1} fluoride (%)	Years of new water supply
Tianjin (TJ)	36.1	67.74	<1
Shanxi (SX)	15	40.2	3
Shandong (SD)	100	85	5
Ningxia (NX)	30.4	95	4
Neimenggu (NM)	34.9	39.39	4
Liaoning (LN)	35	100	5
Jilin (JL)	64.7	90.72	8
Jiangsu (JS)	14.4	75.86	<1
Henan (HN)	59.8	100	4
Heilongjiang (HL)	42.5	30.38	10
Hebei (HB)	100	81.54	9
Guangdong (GD)	86	93.33	8
Gansu (GS)	25.2	41.18	5
Anhui (AH)	0.34	0	<1
Mean (STDEV)	46.0 (31.7)	67.1 (31.4)	4.8 (3.1)

7.7 Impact of burning coal with high-F on public health

More than 10 million people in Guizhou Province and surrounding areas suffer from dental and skeletal fluorosis as a result of food chain F-contamination (Zheng, and Huang, 1989; Zhang and Cao, 1996; Ando et al., 2001). Zheng and Huang (1989) demonstrated that endemic fluorosis resulted from the ingestion of F-contaminated corn dried over burning briquettes composed of high-F coals and high-F clay binders. The dried corn was found to have a mean F concentration >200 mg kg^{-1}. Ando et al., (1998) estimated that 97% of the F-exposure was from food with 2% from direct inhalation. Further, F contents in coal from two mines in Guizhou Province ranged from 559-802 mg kg^{-1}. Zhang and Cao (1996) reported mean F levels in coals from 11 regions of China ranging from 203-1513 mg kg^{-1} with a maximum recorded value of 3,762 mg kg^{-1}. In addition, Ando et al. (1998) reported endemic fluorosis in 13 provinces, autonomous regions, and municipalities in China.

7.8 Thallium

Zhou and Liu, (1985) reported chronic thallium (Tl) poisoning in a rural population within a naturally mineralized Tl-rich sulphide zone in the Lanmuchang area of southwest Guizhou Province China. The source of Tl poisoning was identified as being from vegetables following the land disposal of Hg/Tl-rich mining slag with a Tl concentration ranging from 25-106 mg kg^{-1}. Soil Tl concentrations ranged from 13-69 mg kg^{-1} (Xiao et al., 2003). However, Xiao et al., (2004) stress the need to pay heed to the geological context of "natural contamination" and reported ore Tl concentrations of 6-35,000 mg Tl kg^{-1}, and groundwater, surface water and soil Tl levels ranging from 0.005-1100 µg l^{-1}, 0.07-31 µg l^{-1} and 1.5-124 mg kg^{-1} respectively. Seasonal low soil pH was correlated with an increase in Tl concentration in green vegetables and the number of chronic Tl-patients. The Tl concentration in the edible portions of the crop species grown was in the order green cabbage>chilli>Chinese cabbage>rice>corn with a Tl concentration in green cabbage of up to 500 mg kg^{-1} (DW) (Xiao et al., 2003). The Brassicaceae are known to be a thallophilic species (Kazantzis, 2000). Clinical manifestations of Tl-poisoning included fatigue, paraesthesia, motor weakness and hair loss. Additional high risk areas include Yanshang area of southwestern Guizhou Province (Xiao et al., 2004).

7.9 Selenium

Selenium toxicity induced human health impacts (Selenosis) have also been identified in Enshi County, Huebi Province as a result of food chain Se contamination (Yang et al., 1983; Yang and Zhou, 1994). Further Zheng et al. (2002) reported >500 cases of human selenosis in southwest China and attributed this to the use of Se-rich carbonaceous shales as soil amendments with subsequent uptake in crops and excessive body burdens via ingestion. The Se-rich carbonaceous shales have Se concentrations up to 8,390 mg kg^{-1}. Yang and Zhou (1994) propose a maximum safe daily dietary Se intake of < 400 µg d^{-1}. However it is important to note that Se is also essential for human and livestock health. In humans Se deficiency is predominately manifested as Keshan Disease. In China, Se deficiency occurs in a wide area running from the northeast province of Heilongjiang to the Yunnan Province (Alloway, 1997). Specifically, human health impacts resulting from Se-deficiency have been identified in Linxian (Wei et al., 2004) Chounsang area, Yao County Shaanxi Province (Fang et al., 2003) and Jilin Province Ma and Zhang, (2000). Tan et al. (2002) highlight the role of total and water soluble soil Se on human endemic diseases namely Keshan disease, Kashin-Back diseases and selenosis.

7.10 Mercury

Feng et al., (2003) and Horvat et al., (2003) report significant mercury (Hg) contamination of air, soil, water and crops in Guizhou in southwestern China. The overall average emission of total gaseous Hg during the period April 2000 – November 2001 was 7.39 ng m^{-3}, which is significantly higher than the global background value of 1.5-2.0 ng m^{-3} (Feng et al., 2003). Horvat et al., (2003) suggest that Hg emissions from Guizhou account for 12% of global anthropogenic emissions. Mercury pollution in Guizhou originates from Hg mining and ore processing in the Wanshan area and in the Quingzhen area, Hg pollution originates from the chemical industry discharging through wastewaters and emissions to the atmosphere through coal combustion. Feng et al., (2002) reported a mean (n=48) Hg level in coal from Guizhou Province of 0.53 mg kg^{-1}. Total Hg emission from coal combustion in Guizhou Province was estimated as 8.3 tons in 1998 (Feng et al., 2002).

In 1999, the reactive gaseous mercury (RGM) concentration in Guizhou City was 450 pg m^{-3} as compared to 20 pg m^{-3} in pristine areas of China (Feng et al., 2002). High levels of Hg were found in soils, sediments and crops in both the Wanshan and Quingzhen areas (Horvat et al., 2003). The major source of Hg in both areas in inorganic-Hg. However, it was observed that active transformation of inorganic-Hg species to organic-Hg species namely methylmercury occurred in water, sediments and soil. The concentration of Hg in rice grains was up to 0.569 mg kg^{-1} total Hg of which 0.145 mg kg^{-1} was methylmercury. The percentage of methylmercury to total Hg in rice grain ranged from 5-83%.

The JECFA PTWI value for Hg is 5ug kg BW of which no more than 3ug kg BW can be as methylmercury (JECFA, 2003). With regards the data recorded by Horvat et al., (2003) for total Hg intake, assuming a daily rice intake of 400g (Chen et al., 2002) and a maximum total rice grain Hg concentration of 569 μg Hg kg^{-1} and a body weight of 70 kg this would result in a PTWI value of 56.9 μg Hg kg BW. Even assuming a total rice grain Hg concentration of 200 μg Hg kg^{-1} this would result in a PTWI value of 20 μg Hg kg BW. This demonstrates the potential health risks associated with the consumption of Hg contaminated rice as a result of un-controlled Hg emissions and/or discharge to irrigation waters. However, it is important to note that neither Feng et al., (2003), nor Horvat et al., (2003) conducted any epidemiological studies to evaluate actual health risks. Fang et al., 2001 investigated atmospheric particulate Hg [Hg(p)] concentration and its dry deposition flux in Changchun City, Jilin Province, China. Mean Hg(p) in the non-heating season and heating season were 0.145 ng m^{-3} and 0.461 ng m^{-3}, respectively. Coal burning and wind-blown soil were found to be the two most important sources of Hg(p). Environmental Hg contamination as a result of artisanal gold mining has also been identified in Indonesia (Limbong et al., 2003), Cambodia (Peng and Pichnimith, 2003) and the Philippines (Appleton et al., 1999)

8. Management and decision making tools to mitigate the impacts of PTEs

Causality chain risk assessment methodologies are widely accepted nationally and internationally because they are policy relevant. A widely adopted method is the Pressure-State-Response (PSR) causality chain indicator framework proposed by the Organization for Economic Co-operation and Development (OECD, 1994). The PSR framework is intended to highlight links and help identify effective policies to control and/or prevent human health impacts. The *pressures* describe human activities that can induce changes in the *state* of environmental or human health condition, while society *responds* to changes in pressure or state with policies to prevent or mitigate the pressures (Yang et al., 2003). The PSR model has also been refined to include impact (I), which can be seen as consequences of the changes in the state of the environment (Yang et al., 2002; Grandos and Peterson, 1999). Other causal chain models, such as the European Environment Agency's (EEA) driving force-pressure-state-impact-response (DFPSIR) model (EEA, 1998) and the WHO driving force-pressure-state-exposure-effect-action (DPSEEA) model (Corvalan et al., 1996), for detailed linking of environment and health, are variants of the PSR model. For further details refer to the relevant texts. With regards As, currently applied risk assessment models include the UK DOE CLEA (Ferguson and Denner, 1995) and US-EPA risk assessment packages (USEPA, 1988) which, incorporate As data for a range of exposure pathways from which a total human intake is derived. In addition, IWMI has developed an irrigation infrastructure based sampling protocol to rapidly evaluate Cd and Zn contamination in irrigated rice based agricultural systems (Simmons, 2004). Cupit et al. (2002) and de Meeûs et al., (2002) discuss decision making frameworks and legislative measures to manage the risks arising from exposure to cadmium in fertilizers. This risk assessment protocol was developed for use by EU member states with appropriate selection of input data to assess the risks to humans and the environment arising from exposure to Cd in fertilizers.

Particular risk groups can be characterized, namely children, smokers, women with low Fe stores, consumers of food items with high Cd content and extreme consumers of staple food items (de Meeûs et al., 2002). However, at present with the data sets available it is not possible to accurately characterize risk groups either at an EU level or Member State Level.

8.1International standards/guidelines

8.1.1 Soil and biosolids

Maximum Permissible (MP) levels of PTEs in sewage sludge applied to agricultural land and maximum levels of PTEs in sludge amended soils have been established by the USEPA (USEPA, 1992; USEPA, 1993) and the European Union (EU) (EEC, 1986). In addition, to ensure that the long-term application (100 years) of sewage sludge does not result in the MP levels for PTEs concentrations in agricultural soils being exceeded the EU has also established maximum annual average loading rates of PTEs to agricultural soils (g ha^{-1} yr^{-1}) (Table 20). It is important to note that these guidelines relate to temperate soils primarily under oxidizing conditions. Several authors consider that they are not applicable to tropical soils and more specifically to tropical soils under reducing/oxidizing environments associated with rice production.

Table 20. Guidelines for PTEs in European soils

Element	MP levels of heavy metals in sludge amended soils (mg kg^{-1})	MP levels of heavy metals in sludge (mg kg^{-1})	MP annual loading rates of heavy metals in sludge amended soils (g ha^{-1} yr^{-1})
Cd	1 - 3	20 - 40	150
Cr	-	-	-
Cu	50 - 140	1000 - 1750	12000
Hg	1 - 1.5	16 – 25	100
Ni	30 - 75	300 – 400	3000
Pb	50 - 300	750 – 1200	15000
Zn	150 - 300	2500 - 4000	30000

Source: EEC, (1986) Council Directive 86/278/EEC

8.1.2 Crop and agricultural product quality

Internationally recognized Maximum Levels (ML) for contaminants in foods are established by the Codex Alimentarious Commission, Codex Committee on Food Additives and Contaminants (CCFAC) http://www.fao.org/english/newsroom/news/2003/20363-en.html and the Joint FAO/WHO Expert Committee on Food Additives (JECFA) http://www.who.int/pcs/jecfa/jecfa.htm. The MLs are based on the 'safe' lifetime consumption of food products and are established to ensure the free movement of agricultural products in international trade.

8.1.3Water quality guidelines

Guidelines for drinking water quality including the maximum recommended levels of PTEs have been established by the WHO (WHO, 1993) and are used internationally as the 'bench mark' for most national government water quality assessment programs and for national water quality guidelines. In addition, irrigation water quality guidelines with regards PTEs have been established by the FAO (Pescod, 1992).

8.2 National standards/guidelines for soil, crop and water quality

As indicated in Table 21 national standards/guidelines for soil, crop and water quality exist within countries of Greater Mekong Sub-region. The extent to which these are applicable and available and, the ability of relevant government agencies to enforce these standards/guidelines are issues requiring further detailed investigation.

Table 21. National Standards/Guidelines for soil, crop and water quality within the Greater Mekong Sub-region.

	Standard/ Guideline						
Country	Drinking Water	Ground Water	Surface Water	Coastal Waters	*Industrial Wastewater	Soil	Food
Vietnam	Yes (ND)	TCVN 5944-1995	TCVN 5943-1995	TCVN 5943-1995	TCVN 5945-1995	TCVN 7209-2002	NS
Thailand	MOI No. 322 (1978)	Nat. Environ. Board No. 20 (2000)	Nat. Environ. Board No. 8 (1994)	Nat. Environ. Board No. 7 (1994)	MOSTE, No.3 (1996) PCC, No.3 (1996)	Draft standards (NEC, 2003) ILs (Zarcinas et al., 2004)	MOH, No. 98, 1986,
China	(MOH, 1985)	(MOH, 1985)	Yes (ND)	Yes (ND)	Yes (ND)	Yes (ND)	Yes (ND)
Cambodia	CoM No:27 ANRK.BK	Yes (ND)	Yes (ND)	Yes (ND)	CoM No:27 ANRK.BK	NS	NS
Lao PDR	MOH Draft Values	Yes (ND)	Yes (ND)	/	/	NS	NS
Myanmar	?	?	?	?	?	?	?

ILs = Investigation Levels; NEC = National Environment Committee; ND = Reference made to Standard/Guideline in literature but details not available. MOI = Ministry of Industry; PCC = Pollution Control Committee; MOH = Ministry of Health; CoM = Council of Ministers, Royal Government Kingdom of Cambodia; NS = No Standard; * Industrial wastewater for discharge to receiving bodies; ? Not able to confirm existence of Standard/Guideline.

8.3 Point source control of PTEs emissions and/or discharge

Numerous high and low technology treatment technologies are available for the point source control and on-site treatment of industrial and/or industrial wastes containing elevated levels of PTEs. It is not within the scope of this report to go into these treatment options in detail. However, the editors would refer the reader to the following UNIDO publications and websites (http://www.unido.org; UNIDO, 1993; UNIDO, 1994a,b,c).

9. Management and remediation technologies to mitigate soil contamination by PTEs

A summary of currently available techniques that have been successfully applied to PTE contaminated soils is given in Table 22. These techniques include engineering options, irrigation management options, in-situ immobilization using a range of organic and inorganic amendments, phyto-remediation, chelate enhanced phyto-extraction and the use of transgenic crops. These methods have their drawbacks in effectiveness, duration and economics (Iskandar and Adriano, 1997; Zaurov et al., 1999). In addition, these methods have primarily been developed to remediate PTE contamination in industrially contaminated sites in developed countries where the remediated land has a high value and where costs of remediation are met by the state or by the polluter.

Table 22. Soil remediation technologies to mitigate soil contamination by PTEs

Element/s	Method/Treatment/Amendment	References
Engineering options		
	Removal and replacement of contaminated soil	Iimura et al., (1981)
	Containment: Caps, Vertical Barriers,	USEPA, (1997)
	Solidification/Stabilisation: Cement-based ; Polymer-microencapsulation ; Vitrification	USEPA, (1997); Dutré et al., (1998)
	Separation/Concentration: Soil Washing; Soil Flushing	USEPA, (1997)
Cd, Zn, As, Tl, Pb, Cu, Cr	Electrokinetics	Virkutyte et al., (2002)
Cd, Mn, Tl, Cr,	Microwave immobilization	Abramovitch et al., (2003)
Cd, Cu, Pb and Zn	Sulfidisation pre-treatment and Denver floatation	Vanthuyne and Maes, (2002)
Irrigation management		
Cd	Redox management during the grainfill stage	Iimura et al., (1981) ; Chaney et al., (1996); Simmons et al., (2004)
In-situ immobilization		
Pb	Hydroxyapatite (HA)	Zhu et al., 2004; Chlopecka and Adriano, (1997)
Cd	Alkaline biosolids, lime stabilized-biosolids	Wong et al., (2004) Basta et al., (2001)
Cd/Zn	Sepiolite	Alvarez-Ayuso and Garcia-Sanchez, (2003)
Tl, Zn, Cd, Mn, Pb, Hg and Co	Zeolite (natural and synthetic)	Garcia-Sanchez et al., (1999) ; Haidouti, (1997) Chlopecka and Adriano, (1997); Malliou et al., (1994); Oste et al., (2002)
Pb	H_3PO_4 and $Ca(H_2PO_4)_2$	Melamed et al., (2003) Chen et al., (2003) Brown et al., (2004)
Cd + Pb	Iron oxide waste by-product	Chlopecka and Adriano, (1997)
Cd, Pb and Zn	Di-ammonium Phosphate (DAP)	McGowen et al., (2001)
Pb	Phosphate rock	Hettiarachchi et al., (2001) Basta et al., (2001)
Pb, Cd, Zn	Triple Super Phosphate (TSP)	Hettiarachchi et al., (2001) Hettiarachchi and Peirzynski, (2002)
Cd, Pb and Zn	Phosphate clay	Singh et al., (2001)
Pb	Mn oxide	Hettiarachchi and Peirzynski, (2002)
Cd	Liming	McLaughlin and Singh (1999)
Cr (Cr(VI) reduction to Cr(III))	Organic amendments	Bolan, et al., (2003)
Ni	Limestone	Kukier and Chaney, (2001)
As	Simultaneous addition of lime and $FeSO_4$	Warren et al., (2003) Warren and Alloway, (2003)
As	Goethite	Garcia-Sànchez et al., (1999)
As	Water treatment sludges and red mud	Lombi et al., (2004)

Table 22. Continued	Phytoremediation using hyperaccumulators	
As	Chinese brake fern (*Pteris vittata* L.) Silver back fern *Pityrogramma calomelanos*	Zhang et al., (2002) Tu and Ma, (2002) Francesconi et al., (2002) Visoottiviseth et al., (2002) Wongkomgkatep et al., (2003)
Cd/Zn	Thlaspi caerulescens	Brown et al., (1994) Brown et al., (1995a; 1995b) Lombi et al., (2001) Schwartz et al., (2003) Hutchinson et al., (2000)
Ni	Alyssum murale	Abou-Shanab et al., (2003)
Chelate enhanced phyto-extraction utilizing high biomass heavy metal tolerant plant species		
Pb Pb, Cd and Zn	Phyto-extraction of Pb using EDTA and high biomass plant species namely cabbage EDTA and Chinese cabbage	Shen et al., (2002) Greman et al., (2003)
Transgenic crop species		
Cd, Zn	Indian mustard (*Brassica juncea* (L.) Czern.	Bennett et al., (2003)

The effectiveness and practical applicability of these methods to remediate PTE contaminated agricultural soils in the Greater Mekong Sub-region whilst still facilitating the use of the land to sustain livelihoods in a critical issue.

10. Management and remediation technologies to remove PTEs from wastewater

Technical primarily engineering options for wastewater treatment include, waste stabilization ponds (WSP), wastewater storage and treatment reservoirs (WSTR), constructed wetlands (CW), chemically enhanced primary treatment (CEPT) and upflow anaerobic sludge blanket reactors (UASBs). Details of these technologies can be found at http://www.sanicon.net. In addition, Babel and Kurniawan, (2003) reviewed of over 100 journal articles relating to low cost adsorbents for heavy metals uptake from contaminated water. The results indicate that the traditional use of activated carbon to remove PTEs (namely, Hg, Cd, Cr and Pb) can be effectively replaced by the use of low cost options including chitosan (adsorption capacity of 815, 273 and 250 mg g^{-1} for Hg, Cd and Cr, respectively), zeolites (175 and 137 mg g^{-1} for Pb and Cd, respectively), waste slurries (adsorption capacity of 1030,560 and 540 mg g^{-1} for Pb, Hg and Cr, respectively) and lignin (1865 mg g^{-1} for Pb) (Babel and Kurniawan, 2003). Also of interest is the paper of Zhang and Frankenberger, (2003b), who demonstrate the effectiveness of rice straw to reduce Se(VI) to elemental Se using flow-through bio-reactor channel systems (BCSs). Results indicate that 89.5-91.9% of the input Se(VI) was reduced to red elemental Se, where 96.6 - 98.2% was trapped in the BCSs. In addition Gao et al., (2003) demonstrated effective Se removal from agricultural sub-surface drainage utilizing a flow-through constructed wetland. It is important to note that within the Greater Mekong Sub-region, increasing emphasis is being placed on economically viable, low technology de-centralized waste water treatment options. These may include constructed wetlands and land-based wastewater treatment systems namely, FILTER (Jayawardane et al., 2001; Xianjun et al., 2003) and Effluent irrigated plantations (CSIRO, 1999).

11. Management and remediation technologies to remove As and/or F from groundwater

Numerous high and low technology community and household based treatment technologies are available for the removal of As and/or F from contaminated groundwater sources. It is not within the scope of this report to go into these treatment options in detail. Detailed information can be obtained from the following publications and websites (www.who.org).

11.1 Chemotherapy and dietary treatment of arsenism

Chelating agents such as D-penicillamine, 2,3-dimercaptosuccinic acid (DMSA) and 2,3-dimercapto-1-propane sulfonate (DMPS) have been used to effectively reduce As stored in the body of chronic arsenism (synonymous with arseniasis, arsenocosis and arsenicism) patients (Aposhian et al., 1997, 2000; Mazumder et al., 1998). However, no clinical reductions in skin lesions have been reported Yang et al., (2002). Kosnett, (1999) suggested that Se merits attention as a potential therapeutic agent for patients suffering from arsenism Selenium is an antioxidant nutrient that antagonizes many of the effects of As in biological systems. Yang et al., (2002) reported the effects of a placebo-controlled Se-enriched yeast supplementation trial, conducted for 14 months, on arsenism in adult farmers living in Azuo County, Inner Mongolia, China. In addition, Spallholz et al., (2004) suggest that improved diet and/or dietary supplements may ameliorate As toxicity. Further, the dietary status of Se may be adversely affected by a chronic excessive ingestion of As. Excessive Se excretion due to Se/As complexation may increase the prevalence of As-induced carcinomas due to the oxidative stress imposed by the excessive As and low Se ingestion.

12. Summary of current and future IWMI activities relating to PTEs in the Greater Mekong Sub-region.

In 2002 IWMI-SEA in collaboration with in-country partners developed a sub-regional program of research activities to be initiated in Vietnam entitled "Protecting livelihoods, food security, human health and environmental integrity in irrigated rice-based agricultural systems of Vietnam from the detrimental impacts of heavy metals". Following initial contacts made by Mr. Tissa Bandaragoda (Former Regional Director IWMI-SEA) IWMI-SEA research staff met representatives of the United Nations Economic and Social Commission for Asia and the Pacific, (UNESCAP), to discuss potential regional collaboration. UNESCAP fully supported the proposed sub-regional program and co-organized/funded the sub-regional workshop on "Environmental and public health risks due to contamination of soils crops, surface and groundwater from urban, industrial and natural sources in South East Asia", Hanoi, Vietnam December 10-12th 2002. This workshop was attended by 60 participants from, Thailand, China, Cambodia, Lao PDR and Vietnam. The proceedings of the workshop are available on CD-ROM and the IWMI-SEA and UNESCAP Websites (Simmons and Bakker, 2003; www.iwmi.org/southeast asia/index.asp; www.unescap.org). Further, in 2003 IWMI-SEA research staff worked closely with partners from Lao PDR, Vietnam and Cambodia to further develop a component of the sub-regional program into a full proposal which was submitted to the CGIAR Challenge Program on Water and Food (www.iwmi.org/southeast asia/index.asp). Although un-successful the proposal is still fully supported by our country partners and will be re-formatted for sub-mission to in-country donors by the end of 2004.

In Thailand, IWMI-SEA is collaborating with the Ministry of Natural Resources and Environment (Pollution Control Department) and the Ministry of Agriculture and Cooperatives (Land Development Department), in their efforts to develop and implement socially and environmentally acceptable and practical management strategies that would eliminate Cd from the food chain. In this regard, a full proposal entitled "Development and dissemination of appropriate mitigation measures to prevent cadmium contamination of the rice food chain in Thailand" has been submitted via FAO for funding by Carrefour. Implementation of these activities is expected to commence in July, 2004. In addition, in March 2004 a 3-yr proposal to evaluate the effectiveness of *Thlaspi caerulescens* to Phytoremediate Cd contaminated soils in Tambon Phatat Padaeng was submitted to the USDA-Foreign Agric. Service/ICD/RSED. The research partnership for this proposal is formed by IWMI, University of Maryland, USDA Agricultural Research Service and the LDD. If successful this research will commence in November 2004. On a regional basis, future activities will focus of the implementation of the IWMI-UNESCAP Program of activities entitled "Protecting food security, human health, environmental integrity and livelihoods in rice-based agricultural systems from the detrimental impacts of PTEs in the Greater Mekong Sub-region" (www.iwmi.cgiar.org/southeastasia/index.asp).

13. Conclusions.

The results of this review confirm that naturally elevated levels of arsenic (As) in groundwater aquifers with confirmed direct impacts on public health is an issue of regional significance that needs to continue to be comprehensively and systematically addressed. In addition, millions of rural poor in China are exposed to naturally elevated levels of F in groundwater aquifers. Further, the household use of high-As/F coal as a means of drying agricultural produce is a major pathway of arsenocosis and fluorosis in several provinces of China.

Numerous articles in international peer reviewed journals and reports by national agencies have identified Cd contamination of soils and agricultural produce in localized areas of Thailand, Vietnam and China resulting from the agricultural use of irrigation water contaminated with discharge from non-ferrous mining and ore processing activities, particulate deposition in areas adjacent to non-ferrous smelters and the agricultural use of untreated urban/industrial wastewater. With regards Cd contamination, it must be stressed that detrimental affects on crop quality and public health are predominately at a local/community level. However, the impacts of Cd contamination take on a national significance due to the need to provide health services, establish effective crop quality monitoring programs and the implementation of effective management and remediation options.

Chronic thallium (Tl) poisoning resulting from food chain Tl contamination has also been reported at specific point source locations. Selenium deficiency and toxicity are also considered to be significant public health problems in several provinces in China. Localized Hg contamination of surface water, soils and crops has also been reported as a result primarily of coal combustion and gold mining although no confirmed impacts on public health have been reported.

On a regional basis, decision support tools namely national and international water and soil quality standards, causality chain indicator frameworks, and legislation in the form of land, water, mining and environmental laws exist (http://www.ecolex.org/ecolex/index.php). However, this review indicates that the exposure of urban and rural communities to elevated levels of PTEs is a reality. The results of this review further infer that the root cause of anthropogenic PTE contamination of soil and water resources, localized food chain Cd contamination and the exposure of millions of predominately rural poor to naturally elevated levels of As and F in groundwater aquifers is primarily due to a prior lack of awareness by decision makers.

This appears to have resulted in a lack of effective monitoring programs, lack of strategic decision support tools (namely PTE specific hazard maps), in-adequate enforcement of legislation and lack of compliance.

These deficiencies are effectively addressed by the Regional IWMI-UNESCAP Program of activities entitled "Protecting food security, human health, environmental integrity and livelihoods in rice-based agricultural systems from the detrimental impacts of PTEs in the Greater Mekong Sub-region" (www.iwmi.cgiar.org/southeastasia/index.asp)

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