RAP PUBLICATION 2006/20

Arsenic contamination of irrigation water, soil and crops in Bangladesh:

Risk implications for sustainable agriculture and food safety in Asia

RAP PUBLICATION 2006/20

Arsenic contamination of irrigation water, soil and crops in Bangladesh:

Risk implications for sustainable agriculture and food safety in Asia

Alex Heikens

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
REGIONAL OFFICE FOR ASIA AND THE PACIFIC
Bangkok, 2006

ISBN 974-9746-88-2

Photo credits cover page: Dr G.M. Panaullah, CIMMYT, Bangladesh

For copies write to:
Zhijun Chen
Water Resources Development and Conservation Officer
FAO Regional Office for Asia and the Pacific
Maliwan Mansion, 39 Phra Atit Road
Bangkok 10200
THAILAND
Tel: (+66) 2 697 4000
Fax: (+66) 2 697 4445
E-mail: Zhijun.Chen@fao.org

Contents

Foreword

Natural arsenic contamination of groundwater resources in parts of Asia is posing a serious threat to the health of millions of people. The issue of contaminated drinking-water has been taken up by governments and development partners and many efforts are under way to mitigate the problem. However, the same water resources are used extensively for irrigation purposes throughout the region too. Over the last few years, questions have been raised by governments, the general public and development partners about the possible risks of irrigating with arsenic-contaminated water and about mitigation options.

The Food and Agriculture Organization of the United Nations – Regional Office for Asia and the Pacific (FAO–RAP) has therefore prepared this technical report to provide insight into the behaviour of arsenic in food and agriculture, and to evaluate available knowledge of the effects of irrigating with arsenic-contaminated water on crop production and food safety in Asia. The focus is on Bangladesh, where most studies on arsenic contaminated irrigation water have been carried out. It is also the country with the most serious arsenic contamination crisis in the drinking-water sector. Throughout the report, specific attention is given to rice because it is the most important staple crop in Asia, and it is one of the crops most sensitive to arsenic contamination.

Information provided in this report is based on peer-reviewed publications in international journals and on discussions with experts from various organizations. In particular, Professor A.A. Meharg, University of Aberdeen, and Dr G.M. Panaullah, The International Maize and Wheat Improvement Centre (CIMMYT), Bangladesh, are acknowledged. From FAO, Sasha Koo-Oshima (Water Quality and Environment Officer), Zhijun Chen (Water Resources Development and Conservation Officer) and Thierry Facon (Senior Water Management Officer) are acknowledged for their valuable technical inputs and reviewing the draft report.

He Changchui
Assistant Director-General and
FAO Regional Representative for Asia and the Pacific

Acronyms

As	arsenic
AsB	arseno-betaine
AsIII	arsenite
AsV	arsenate
AusAID	Australian Agency for International Development
BARI	Bangladesh Agricultural Research Institute
BAU	Bangladesh Agricultural University
BRRI	Bangladesh Rice Research Institute
BUET	Bangladesh University of Engineering and Technology
CGIAR	Consultative Group on International Agriculture Research
CIMMYT	International Maize and Wheat Improvement Centre
CRM	certified reference material
DMA	dimethylarsenic acid
DTW	deep tubewell
dw	dry weight
FAO	Food and Agriculture Organization of the United Nations
Fe	iron
FeOOH	iron(hydr)oxides
IRRI	International Rice Research Institute
IWMI	International Water Management Institute
MMA	monomethylarsenic acid
MPC	maximum permissible concentration
NIPSOM	National Institute of Preventive & Social Medicine
NIST	National Institute of Standards and Technology
O_2	oxygen
OM	organic matter
pH	acidity
PMTDI	provisional maximum tolerable daily intake
PO_4	phosphate
QA/QC	quality assurance/quality control
SOP	standard operating procedure
SRM	secondary reference material
STW	shallow tubewell
TFA	trifluoroacetic acid
UNDP	United Nations Development Programme
UNICEF	United Nations Children’s Fund
USAID	United States Agency for International Development
WHO	World Health Organization
ww	wet weight

Executive summary

Arsenic (As) in groundwater is a major health concern in Asia and the risks from using shallow tubewells (STWs) for drinking-water are well-known. At present, twelve countries in the region have reported high As levels in part of their groundwater resources. Bangladesh has the highest percentage of contaminated STWs (~20 percent) and an estimated 30 million people are dependent on those wells for domestic purposes. Since an initial investigation on As accumulation in rice undertaken by FAO with support from the United Nations Development Programme (UNDP) in 2001, further scientific studies in the last couple of years have reported potential risks from As in irrigation water because of land degradation affecting agro-ecosystem services. The most well-known concern is As entering the food chain, affecting food safety. This poses a potential dietary risk to human health in addition to the risk from drinking contaminated groundwater. Less well-known but potentially more serious is the risk of As to crop production. Continuous build up of As in the soil from As-contaminated irrigation water may reduce crop yields, thus affecting the nutritional status and incomes of rural farming communities.

As part of the green revolution, millions of STWs have been installed throughout Asia over the last three decades. This has resulted in a sharp increase of groundwater extraction for irrigation. In Bangladesh, of the four million ha under irrigation, 2.4 million ha are irrigated with approximately 900000 STWs. Considering that there are many contaminated drinking-water STWs, a high percentage of contaminated irrigation STWs can also be expected. It has been estimated that water extraction from the shallow aquifer for irrigation adds 1 million kg of As per year to the arable soil in Bangladesh, mainly in the paddy fields.

The form and behaviour of As vary greatly between flooded soils, such as paddy fields, and non-flooded soils. The most important As species are arsenate (AsV) under non-flooded conditions and arsenite (AsIII) under flooded conditions. AsIII has a higher solubility than AsV, resulting in a higher mobility of As in flooded soils. Soil texture is another important factor, with clay soils having a much greater capacity to bind As than sandy soils. Worldwide, natural soil concentrations are around 5 mg/kg, but this can vary depending on the origin of the soil. Data on As in soils in Bangladesh are limited both in number and quality, but there are indications that 5–10 mg/kg is the background level. Various reports indicate that soil concentrations are increasing because of As input via irrigation water, and is a major concern. At this stage, the conditions under which As is building up in the soil and the time frame involved are still uncertain.

One of the major factors determining uptake and toxicity to plants is the form of As. The two most important forms, AsV and AsIII, are taken up by completely different mechanisms. Uptake, accumulation and toxicity vary within and between plant species. In general, more As in the soil leads to higher concentrations in plants, but this depends on many factors. It is not yet possible to predict As uptake and/or toxicity in plants from soil parameters. Toxicity data are limited in number and in terms of the experimental setups used to obtain them. At this stage, available data cannot be extrapolated to the field situation. Reliable and representative plant uptake and toxicity data are essential to evaluate current and future soil concentrations in Bangladesh and other affected countries.

There are some potential human health risks related to livestock and fresh water fisheries as these can be exposed to As via drinking-water, pond water, and feeds. Although As is less toxic to animals and concentrations of inorganic As in animal products are expected to be relatively low, very limited information is available, and research is thus needed to examine, for example, the transference of As to milk. Because of the lack of information, impacts on livestock and fisheries are not covered in this technical report.

To date, little field research on the relation between As in irrigation water and crop yield has been conducted. It is of concern that a number of studies from Bangladesh and West Bengal (India) have reported increased concentrations in soils and crops because of irrigation with As-contaminated groundwater. Similar situations can be expected elsewhere in the region where As-contaminated irrigation water is used. The increase in soil concentrations may finally result in a reduction of soil quality and crop yields. Assessment of risks to crop production is difficult because of the limited information on current and future As soil concentrations and the lack of reliable plant toxicity data. These gaps need to be addressed urgently.

Recent data on total and inorganic As in rice and vegetables from Bangladesh indicate that rice contributes significantly to the daily intake. A positive correlation between As in groundwater resources, soil and rice has been reported, indicating that food chain contamination takes place because of prolonged irrigation with contaminated water. In order to refine the risk assessment, more scientific data are needed on As in foods, and on food and water consumption patterns. With limited technical capacity and the continuation of current agricultural practices, it can be expected that As in the food chain will further increase. This would, unfortunately, offset the ongoing activities in the drinkingwater sector to reduce As exposure.

The risks of land degradation are likely to increase with the accumulation of As in the soil. Management options should therefore focus on preventing and minimizing As input to soils. While the risks need to be quantified in detail, various management options could be further explored. For example, farmers often use more irrigation water than needed. Optimizing water input would be a sound option to reduce As input while saving water. Furthermore, aerobic growth conditions in paddy fields may reduce bioavailability and uptake of As in rice. Other possible options include breeding crops tolerant to As and/or low accumulation of As in grains, and shifting from rice in the dry season to crops that demand less water, where feasible.

Based on the information available as presented in this literature review, it can be concluded that As in irrigation water can result in land degradation that in turn affects food safety and crop production. A number of reports found evidence that irrigating with As-contaminated water, causes accumulation of As in the soil and that this is reflected in higher levels of As in the edible parts of crops. Major gaps in our knowledge still need to be filled and this currently hampers a comprehensive risk assessment. Action is urgently needed in a concerted manner from multiple stakeholders, particularly from governments and development partners. The urgency of the need to take up this issue is emphasized by the large scale of As-contaminated groundwater resources, by the extensive use of groundwater for irrigation, and by rice being the staple crop in Asia.

It is recommended to initiate an integrated programme to quantify the scale of the problem in combination with the development of a water/soil/crop quality monitoring system for land degradation in agro-ecosystems. This should not only include As, but a range of physical, chemical (nutrients and contaminants) and biological parameters. Further, management options to prevent and mitigate As contamination need to be explored.

1. Background

In the last three decades, the number of STWs has increased dramatically in the Asian region. These STWs are providing a reliable and inexpensive source of irrigation water, which allows farmers to grow additional crops during the dry season, and ensures them of water security during periods of drought. Furthermore, STWs are an inexpensive source of drinking-water mostly free of waterborne diseases. The installation of STWs to provide drinking-water has significantly contributed to the reduction of diarrhoea and has saved millions of lives. And access to groundwater resources has been a major contributor to the green revolution in Asia.

Since the 1980s, evidence has gradually unfolded that As is present in elevated levels in part of tapped groundwater resources, and the World Health Organization (WHO) has set a drinking-water standard of 10 µg/l (or 0.01 mg/l). At present, countries in the region have reported high levels of As in part of their groundwater resources (Afghanistan, Bangladesh, Cambodia, China, India, Lao PDR, Mongolia, Myanmar, Nepal, Pakistan, Thailand, Viet Nam) and more cases are being reported and published (Berg *et al.*, 2001; Chakraborti *et al.*, 2002; Mandal and Suzuki, 2002; Ng *et al.*, 2003; Polya *et al.*, 2005). The high levels of As in groundwaters in the affected countries are predominantly of geogenic origin. Reductive dissolution of iron(hydr)oxides (FeOOH) stimulated by microbial activity and organic materials is regarded as the most important mechanism releasing As into the aquifer (Ahmed *et al.*, 2004; McArthur *et al.*, 2004; Mukherjee and Bhattacharya, 2001; Ravenscroft *et al.*, 2001; Smedley and Kinniburgh, 2002; Smedley *et al.*, 2003; Zheng *et al.*, 2004). Anthropogenic sources of As include various industrial activities, pesticides, herbicides, and fertilizers. Natural contamination is generally regarded as the main mechanism causing the high levels of As in the groundwater in Asia and this will therefore be the focus of this report.

To illustrate the scale of the As problems in the region, a brief description of the As situation in a few countries in the region with regard to As in groundwater resources and people at risk of consuming contaminated drinking-water are presented below.

Bangladesh

In Bangladesh, groundwater from the shallow aquifer is the main source of drinking-water. Part of the shallow aquifer contains As concentrations above the national drinking-water standard of 0.050 mg/l, particularly in the south and southwestern part of the country. The latest data indicate that approximately 20 percent of the STWs exceed the standard and 10000 to 30000 people have been diagnosed with arsenicosis to date (R. Johnston and G. Howard, personal communication, 2005). An estimated 30 million people consume water which exceeds the Bangladesh drinking-water standard for As. The shallow aquifer is also the main source of irrigation water during the dry Boro season. Approximately 95 percent of all groundwater extracted is used for irrigation, mainly for Boro rice production. More detailed information on the situation in Bangladesh can be found in, for example, Ahmad *et al.*, 1997; Alam *et al.*, 2002; Chakraborti *et al.*, 2002; Chowdhury *et al.*, 2000; Mukherjee and Bhattacharya, 2001.

China

During the 1980s, endemic arsenicosis was found successively in many areas in mainland China. At present, the population exposed to As levels in drinking-water exceeding the national standard of 0.050 mg/l is estimated to be over two million and more than 10000 arsenicosis patients were confirmed by 2001. As-contaminated groundwater resources are mainly located in west China and north China. By 2004, high As levels in groundwater had been reported in the following provinces: Xinjiang, Inner Mogolia, Shanxi, Ningxia, Jilin, and Qinghai. In Qinghai, arsenicosis is mainly related to burning As-rich coal indoors. Based on geochemical and hydrological characteristics, more areas with high As concentrations can be expected within these and other provinces in China (Sun, 2004; Xia and Liu, 2004).

India

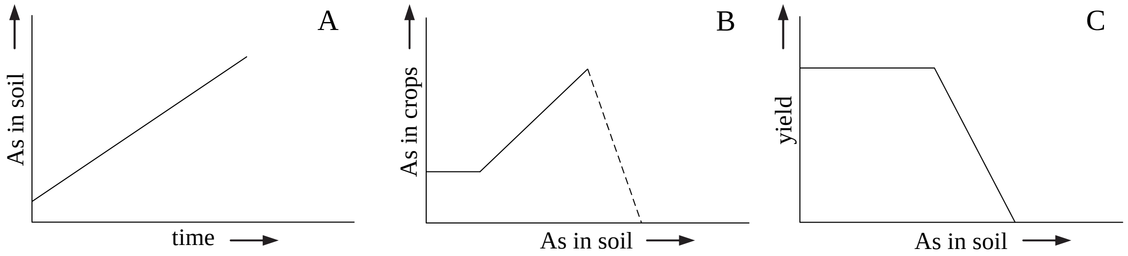
The presence of naturally elevated levels of As in groundwater was confirmed in seven Indian states, namely West Bengal, Bihar, Uttar Pradesh, Assam, Jharkland, Chattisgarh and Madhya Pradesh. Except for West Bengal, the extent of the problem is not fully known and the number of people at risk is impossible to estimate with any degree of confidence. In West Bengal, investigations suggest that eight districts show As content in well-water to be above 0.050 mg/l with, according to United Nations Children's Fund (UNICEF), over 13.8 million people at risk (R. Nickson, personal communication, 2006). More detailed information can be found in, for example, Ahmad *et al.*, 1997; Alam *et al.*, 2002; Chakraborti *et al.*, 2002; Chowdhury *et al.*, 2000.

Nepal

As contamination in Nepal has been detected in the Terai region of southern Nepal, where nearly half of Nepal's total population is living. In this region bordering India, 90 percent of the people served by approximately 200000 STWs, use groundwater for drinking purposes. Of the 15000 STWs tested, 23 percent exceeds the WHO drinking-water guideline of 0.010 mg/l, whereas 5 percent exceeds the Nepal interim As guideline of 0.050 mg/l. An estimated 0.5 million people are consuming drinking-water with As levels exceeding 0.050 mg/l (Shrestha *et al.*, 2003).

1.1 Arsenic contaminated irrigation water: the risks

To date, only limited attention has been paid to the risks of using contaminated groundwater for irrigation. Irrigation water with high levels of As may result in land degradation in terms of crop production (loss of yield) and food safety (food chain contamination) (Brammer, 2005; Duxbury and Zavala, 2005). Long-term use of As-contaminated irrigation water could result in As accumulation in the soil. If absorbed by the crops, this may add substantially to the dietary As intake, thus posing additional human health risks. Over time, As accumulation in the soil could reach soil concentrations toxic to crops, thus reducing yields (Figure 1.1).



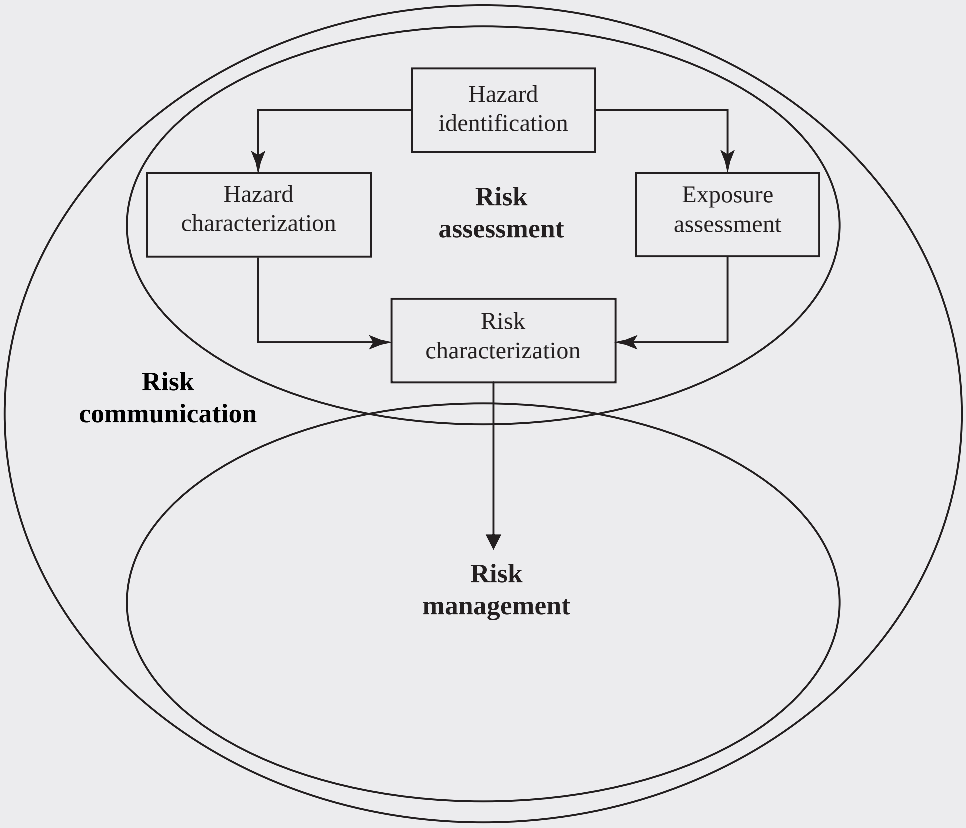
Note: A: input of As via irrigation water can lead to accumulation of As in the soil over time. B: depending on bioavailability, uptake and transport within the plants, higher soil concentrations may be reflected in higher concentrations in crops. The dotted line indicates that at a certain level the plant growth becomes severely inhibited and As concentrations in the plants are then no longer relevant. C: with an increase in soil concentration, yields are expected to stay more or less constant until a threshold level is reached, after which yield will decline.

Figure 1.1 The possible risks of using As-contaminated irrigation water over time

Reliable and representative data are therefore needed to assess and manage the risks of As-contaminated irrigation water. With millions of irrigation STWs tapping water from the same As-contaminated aquifer as the STWs for drinking-water, the extent of possible risks can be substantial. An overview of the risk analysis paradigm is presented in Box 1.

BOX 1: RISK ANALYSIS PARADIGM

The terms and definitions presented here are taken from the Codex Alimentarius Commission (Codex, 2004). Although the definitions refer to food safety, with minor adaptations they are also applicable to the risks of As to crop production. For a detailed description refer to http://www.codexalimentarius.net/web/procedural_manual.jsp (Codex, 2004).



Definitions of risk analysis terms related to food safety as used in the Codex Alimentarius (Codex, 2004)

Hazard	A biological, chemical or physical agent in, or condition of, food with the potential to cause an adverse health effect.
Hazard characterization	The qualitative and/or quantitative evaluation of the nature of the adverse health effects associated with biological, chemical and physical agents that may be present in food. For chemical agents, a dose–response assessment should be performed. For biological or physical agents, a dose–response assessment should be performed if the data are obtainable.
Hazard identification	The identification of biological, chemical and physical agents capable of causing adverse health effects and which may be present in a particular food or groups of foods.
Dose–response assessment	The determination of the relationship between the magnitude of exposure (dose) to a chemical, biological or physical agent and the severity and/or frequency of associated adverse health effects (response).
Exposure assessment	The qualitative and/or quantitative evaluation of the likely intake of biological, chemical and physical agents via food as well as exposures via other sources if relevant.
Risk	A function of the probability of an adverse health effect and the severity of that effect, consequential to a hazard(s) in food.

BOX 1: *(continued)*

Risk analysis	A process consisting of three components: risk assessment, risk management and risk communication.
Risk assessment	A scientifically based process consisting of the following steps: 1) hazard identification, 2) hazard characterization, 3) exposure assessment, 4) risk characterization.
Risk characterization	The qualitative and/or quantitative estimation, including attendant uncertainties, of the probability of occurrence and severity of known or potential adverse health effects in a given population based on hazard identification, hazard characterization and exposure assessment.
Risk communication	The interactive exchange of information and opinions throughout the risk analysis process concerning risk, risk-related factors and risk perceptions, community and other interested parties, including the explanation of risk assessment findings and the basis of risk management decisions.
Risk management	The process, distinct from risk assessment, of weighing policy alternatives in consultation with all interested parties, considering risk assessment and other factors relevant for health protection of consumers and for the promotion of fair trade practices and, if needed, selecting appropriate prevention and control options.

2. Arsenic in agriculture: current knowledge

2.1. Soil chemistry

Low levels of As are naturally present in the soil (Matschullat, 2000). The background levels are around 5 mg/kg worldwide with substantial variation depending on the origin of the soil (Mandal and Suzuki, 2002). The behaviour of As is distinctly different under flooded (anaerobic) and non-flooded (aerobic) soil conditions, with flooded conditions being likely the most hazardous in terms of uptake by plants and toxicity, as will be explained in this chapter. Taking into consideration that rice is the staple crop in Asia, that its cultivation largely takes place under flooded conditions, and that its high demand for irrigation water, often from groundwater resources, understanding the behaviour of As under flooded soil conditions is of particular importance.

As speciation in the soil

As exists in the environment in various organic and inorganic forms (species). The most important inorganic species are arsenate (AsV) and arsenite (AsIII). Monomethylarsenic acid (MMA) and dimethylarsenic acid (DMA) are the most common organic species in the soil, but their natural presence is low compared to inorganic As (Abedin *et al.*, 2002c; Fitz and Wenzel, 2002).

Speciation of inorganic As in the soil is largely controlled by reduction and oxidation processes (redox). Under aerobic (oxidizing) conditions AsV predominates, whereas AsIII predominates under anaerobic (reducing) conditions (Fitz and Wenzel, 2002; Takahashi *et al.*, 2004). For example, in an experimental paddy field 30 percent of the As was present as AsIII under non-flooded conditions and up to 70 percent was present as AsIII under flooded conditions (Takahashi *et al.*, 2004). Masscheleyn *et al.*, (1991) reported that under oxidizing conditions, As was mainly present as AsV (>95 percent of the total soluble As) with a relatively low solubility. Under more reducing conditions, AsIII became by far the predominant species and the solubility of As increased sharply. Microbial activity can influence As speciation via various mechanisms such as redox reactions with Fe and As and via (de)methylation of As species (Fitz and Wenzel, 2002; Mahimairaja *et al.*, 2005).

The role of iron hydroxides

AsV and AsIII adsorb mainly to iron(hydr)oxides (FeOOH) present in the soil and AsV is bound strongest. The behaviour of FeOOH is highly dependent on redox conditions, making Fe redox chemistry the most important factor in regulating As behaviour (Fitz and Wenzel, 2002; Takahashi *et al.*, 2004). Under anaerobic conditions, FeOOH readily dissolves and As is released into the soil solution, where As will be present mainly as AsIII (Takahashi *et al.*, 2004; Masscheleyn *et al.*, 1991). Microbial activity is closely involved in this process (Islam *et al.*, 2004a; Zobrist *et al.*, 2000; Masscheleyn *et al.*, 1991). Under aerobic conditions FeOOH is relatively insoluble and serves as a sink for As. Fe and As behaviour is therefore dynamic and closely related in lowland paddy fields.

The As concentrations in the irrigation water usually differ from those in the soil water. For example, Takahashi *et al.* (2004) reported that As concentrations in irrigation water were higher compared to the soil water concentrations during the non-flooded period because of sorption to FeOOH. Under flooded conditions, soil water concentrations increased because of remobilization and, important to note, became higher than the irrigation water concentrations. Under flooded conditions, plants can therefore be exposed to much higher concentrations in the soil water than would be expected based on the concentrations in the applied irrigation water.

FeOOH is mainly present in the clay size soil fraction (<2 µm) and clayey soils therefore generally have a higher As content compared to more sandy soils (Fitz and Wenzel, 2002; Mahimairaja *et al.*, 2005). At the same total soil concentration, clayey soils are less toxic compared to sandy soils because As is more strongly bound in the clayey soils. Under specific soil conditions, other sorption substrates such as carbonate minerals and manganese oxides (MnO) can also be relevant (Mahimairaja *et al.*, 2005).

Phosphate

Phosphate (PO_4) is an analogue of AsV, making it an important factor in the behaviour of As in aerobic soils (Lambkin and Alloway, 2003; Mahimairaja *et al.*, 2005; Williams *et al.*, 2003). Both ions compete for sorption sites on FeOOH and for uptake by plants. The effect of PO_4 additions to aerobic soils on the uptake of As will therefore depend on the balance between competition for sorption sites and competition for uptake.

AsIII is not an analogue of PO_4, making the presence of PO_4 probably less relevant to As behaviour under flooded soil conditions (Takahashi *et al.*, 2004). It is not known if PO_4 plays a role in the rhizosphere (the microenvironment around the roots), where aerobic conditions can occur under flooded conditions. Other ions may also influence As behaviour, but the impact seems to be less compared to PO_4 (Cornu *et al.*, 2003; Mahimairaja *et al.*, 2005; Williams *et al.*, 2003).

pH

AsV adsorption decreases with increasing pH, in particular above pH 8.5, whereas the opposite occurs for AsIII. The extent to which pH influences As sorption differs between soils. The adsorption maximum for AsV on FeOOH lies around pH 4, whereas for AsIII the maximum is found at approximately pH 7–8.5 (Fitz and Wenzel, 2002; Mahimairaja *et al.*, 2005; Masscheleyn *et al.*, 1991).

Volatilization

As may be lost from the soil via the formation of volatile As components (Abedin *et al.*, 2002c; Mahimairaja *et al.*, 2005). As summarized by WHO (2001), this can contribute a removal of 12 to 35 percent per year. The extent to which this process is relevant to flooded paddy fields with their distinct soil conditions is however still unknown.

2.2 Crops

Rhizosphere

Conditions in the the rhizosphere may deviate substantially from the bulk soil. As summarized by Fitz and Wenzel (2002), plants will influence the pore water composition by uptake and excretion of substances. Micro-organisms in the rhizosphere will also influence its composition (Harvey *et al.*, 2002; Nicolas *et al.*, 2003). Because Fe and As behaviours in the soil are closely related to each other, it can be expected that plant processes related to Fe uptake may also influence As bioavailability and uptake. The same is true for PO_4.

When a paddy field is flooded, the rhizosphere can still be aerobic. The main reason is that rice plants can transport oxygen from the leaves to the roots, resulting in the transfer of O_2 to the rhizosphere. A number of micro-organisms is also capable of oxidizing the rhizosphere. The oxidized conditions can result in the precipitation of FeOOH around the roots, also known as Fe-plaque. Fe-plaque has been reported frequently on roots of wetland plants including rice (Meharg, 2004). It may influence As speciation, bioavailability and uptake and Fe reducing and oxidizing bacteria are likely to play a major role (Fitz and Wenzel, 2002; Meharg, 2004; Weiss *et al.*, 2003; Weiss *et al.*, 2004). The importance of Fe-plaque on As uptake by wetland plants including rice remains to be resolved (Liu *et al.*, 2004; Chen *et al.*, 2005).

Uptake

AsIII and AsV are taken up by different mechanisms. AsV is taken up via the high affinity phosphate uptake system (Meharg, 2004). PO_4 additions have therefore been suggested to reduce uptake because of competition between PO_4 and AsV for uptake.

For rice grown in pots with soil and irrigated with AsV contaminated water, no effect of PO_4 on As accumulation in rice plants was observed (Abedin *et al.*, 2002a; Abedin *et al.*, 2002b). Abedin *et al.* (2002a) suggested that the plants were effectively exposed to AsIII and not to AsV because of the reducing soil conditions. An alternative explanation is that PO_4 competes with AsV both for both sorption at Fe-plaque and for uptake, minimizing the overall effect of PO_4 (Chen *et al.*, 2005). As summarized in various papers, the addition of PO_4 to As-contaminated soils to minimize As uptake is controversial under non-flooded conditions (Abedin *et al.*, 2002c; Fitz and Wenzel, 2002).

AsIII is actively taken up by so-called water channels (aquaporins) in the roots (Meharg and Jardine, 2003). Laboratory experiments have shown that Boro (dry season) rice cultivars take up less AsIII and AsV than Aman (rainy season) rice cultivars. This may be related to physiological or morphological differences between the root systems (Abedin *et al.*, 2002c). However, this does not imply that Boro rice will accumulate less As than Aman rice under field conditions, because Boro rice is irrigated with As-rich groundwater whereas Aman rice is rainfed.

The uptake mechanism of organic As is largely unclear (Meharg, 2004). It seems that monomethylarsenic acid (MMA) and dimethylarsenic acid (DMA) are taken up by rice plants but that the rate of uptake is much lower compared to inorganic As (Abedin *et al.*, 2002c).

To date, it has not been possible to predict As uptake by plants from the soil. Most papers only include total As concentrations in the soil and the As concentration in the irrigation water. It has been suggested that total As can be regarded as potentially bioavailable in paddy fields, because most of it is bound to FeOOH (R. Loeppert, personal communication, 2004). Good correlations between total As in soil and plants are however not always found (see also Section 3.1) (Jahiruddin *et al.*, 2005; Miah *et al.*, 2005).

Translocation and accumulation

With the exception of hyperaccumulators such as certain ferns, the translocation of inorganic As from the roots to the above ground parts is limited. Organic As is more readily translocated but the uptake is much lower compared to inorganic As (Carbonell *et al.*, 1998; Carbonell-Barrachina *et al.*, 1998). In pot experiments with rice plants exposed to As added via AsV in irrigation water, plant parts were ranked according to the As concentrations as follows: root > straw > husk > grain. Concentrations in all plant parts increased with the exposure concentration (Abedin *et al.*, 2002a; Abedin *et al.*, 2002b). This is a common observation for other plants as well (Bleeker *et al.*, 2003; Carbonell *et al.*, 1998; Carbonell-Barrachina *et al.*, 1998; Carbonell-Barrachina *et al.*, 1997; Hartley-Whitaker *et al.*, 2001; Sneller *et al.*, 1999b).

Metabolism

After uptake, AsV is rapidly reduced to AsIII, causing oxidative stress. This induces the formation of certain antioxidants. This is regarded as a detoxification mechanism that is also activated by heavy metals such as cadmium (Meharg and Hartley-Whitaker, 2002; Sneller *et al.*, 1999a; Sneller *et al.*, 2000). On the contrary, exposure to AsIII does not induce this system. In spite of the rapid reduction of AsV to AsIII, high levels of AsV have been found in plant material. Abedin *et al.* (2002b) reported that more than 70 percent of the As in the straw of rice was present as AsV. Schmidt *et al.* (2004) found AsV in plants that were only exposed to AsIII, showing that oxidation of AsIII in plants took place. Many organic As species have been found in plants as well, but only in minor amounts (Dembitsky and Rezanka, 2003). It is unclear whether organic As species found in plants are taken up from the soil or are formed by the plants (Meharg and Hartley-Whitaker, 2002; Sneller *et al.*, 1999b).

Effects

AsV can compete with PO_4 within the plant cells disturbing the energy flow in the cell. AsIII reacts with a number of enzymes and tissue proteins that can cause inhibition of cellular function and finally death (Meharg and Hartley-Whitaker, 2002). Exposure to As also influences concentrations of other elements in plant tissue (Carbonell *et al.*, 1998).

A specific form of As toxicity to rice known as straighthead disease has been reported in the USA (Meharg and Hartley-Whitaker, 2002). Straighthead disease is a physiological disorder that causes panicle sterility. Visual symptoms are empty panicles standing upright instead of bending downward at maturity. This disease was related to rice production on former cotton fields heavily contaminated with MMA used as a pesticide. It was most frequently observed on sandy loam soils but seldom on clay soils. Affected plants were usually found in spots scattered throughout a field. Besides As, a high organic matter (OM) content seemed to play a role as well. Rice cultivars show a great variation in their tolerance to MMA, which was used to select/develop tolerant cultivars.

A generalized ranking of plant parameters according to sensitivity to metals including As is as follows: root length > root mass > shoot length > total mass (root plus shoot) > shoot mass > germination (Abedin and Meharg, 2002). This is in agreement with e.g. Abedin *et al.* (2002b) who found that root biomass production of rice plants was most sensitive to As whereas plant height was not very sensitive. Carbonell-Barrachina *et al.* (1998) reported for coastal marsh grasses that dry matter production of roots was most sensitive. Abedin and Meharg (2002) proposed that shoot height can be used in the field as an indicator. Abedin *et al.* (2002a) reported that shoot height is however (much) less sensitive than root length. Abedin and Meharg (2002) proposed the next chain of effects: reduced shoot height, reduced leaf area, reduced photosynthesis, reduced yield. It is likely that toxicity on the root system is actually the first step.

The relative toxicity of As species

The relative toxicity of different As species to plants depends on a range of factors including experimental conditions and plant species and examined plant parameters. Therefore, one should be cautious about using a generalized classification of As species according to toxicity. Taking that into account, inorganic As is generally regarded as being more toxic than organic As, with AsIII being the most toxic form (Dembitsky and Rezanka, 2003; Fitz and Wenzel, 2002; Liu *et al.*, 2004; Mahimairaja *et al.*, 2005; Meharg and Hartley-Whitaker, 2002).

Tolerance

AsV tolerance is related to PO_4 metabolism (suppression of the AsV/PO_4 uptake mechanism) (Bleeker *et al.*, 2003; Hartley-Whitaker *et al.*, 2001; Meharg, 2004; Meharg and Hartley-Whitaker, 2002; Sneller *et al.*, 1999b). Most AsV tolerant plants accumulate less AsV than non-tolerant plants and this could be used to develop/select rice cultivars with a low accumulation of As.

Tolerance to AsIII is largely unknown. If there is any relevant variation in AsIII tolerance, this may be found in variation in glutathione-levels and/or the activity of certain transporters that transport As complexes within the plant (H. Schat, personal communication, 2004). Schat also reported for *Holcus* (Velvet Grass) that AsV tolerant plants did not show any tolerance to AsIII at all.

To date, studies on the genetics behind As tolerance have focused only on AsV. In rice and wild grasses, the tolerance to AsV is under the control of a single gene (Dasgupta *et al.*, 2004). If AsV uptake is relevant in rice grown under flooded conditions, this can be an important finding to develop/select rice cultivars with a high tolerance and a low As uptake. Considering the indications that AsIII predominates the As uptake by rice, it is necessary to quantify the uptake of As species under (semi)field conditions. Based on this information, the research on genetics behind As tolerance and uptake can focus on the environmentally relevant As species.

Toxicity data

Most toxicity experiments have been carried out with plants grown in water only (hydroponics). Such a design can be useful to study, for example, uptake mechanisms, internal transport, metabolism, and toxic effects. The design is however not suitable to generate toxicity data to evaluate concentrations in the environment because all interactions with the soil matrix influencing bioavailability are neglected.

Toxicity experiments are also carried out with plants grown in soil to which a certain amount of As is added (spiked soil) shortly before the experiment. This setup has various limitations as well. Adding As to reach a certain soil concentration suggests that the results are representative of the field. However, in the field As is added over a number of years. The prolonged contact time between As and the soil in the field can result in a lower solubility of As and therefore lower uptake by plants in the field. Therefore, experiments with spiked soils often result in an overestimation of the adverse effects compared to the actual field situation (Duxbury and Zavala, 2005).

In other studies, As has been added via irrigation water to the soil during the experiments. This is more in agreement with the field situation in Bangladesh compared to hydroponics and spiked soils. However, this experimental setup neglects that As levels in irrigation water in the field are relatively constant and that As is slowly added to the soils over a period of many years. Ideally, experiments should be performed with naturally contaminated soils and constant As concentrations in irrigation water.

To date, toxic effects have only been related to the irrigation water concentration and/or the total soil concentration. The As concentration in soil water (pore water) will surely deviate from the irrigation water because of interactions with the soil matrix. Total As in the soil is also unlikely to be a good predictor of As uptake and toxicity for different soil types as only part of the As in the soil is likely to be potentially available to the plants. Dose–response relationships based on irrigation water concentrations or on total soil concentrations are only valid for the experiment from which they were derived and cannot be extrapolated to any other system. With all these limitations in mind, various studies of the above mentioned types will be summarized and discussed.

Hydroponics

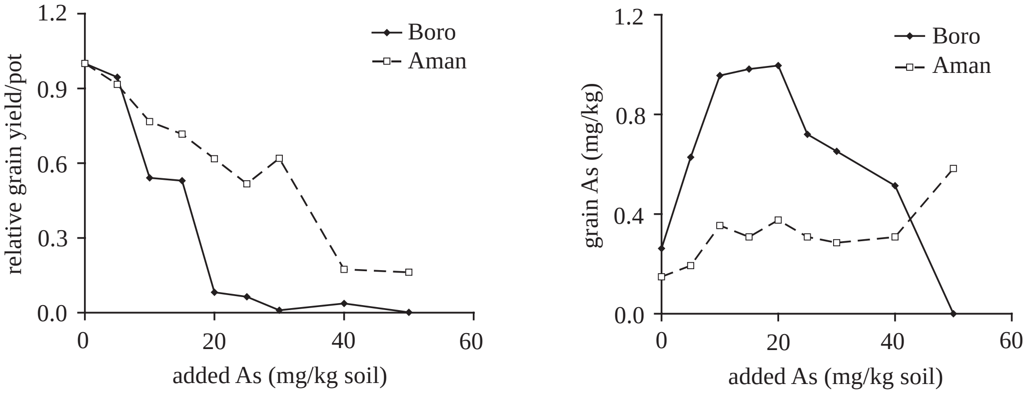
Abedin and Meharg (2002) exposed eight Bangladesh rice varieties to AsIII and AsV and tested for germination and seedling growth. Germination was slightly inhibited at 0.5 and 1 mg/l. At 2 mg/l, inhibition was more than 10 percent. AsIII was more toxic than AsV. No significant difference between Boro and Aman cultivars in terms of germination was observed. Root growth was inhibited by ~20 percent at 0.5 mg/l and AsV was more toxic than AsIII. Boro cultivars were slightly more tolerant than Aman cultivars. Shoot height was also affected. At 0.5 mg/l, the shoot height was reduced by ~30 percent with no significant difference between cultivars and As species.

Marin *et al.* (1992), cited in Abedin *et al.* (2002b), found a reduced shoot height at 0.8 mg/l AsIII and MMA but not with AsV. In contrast with the findings of Abedin and Meharg (2002), AsIII was more toxic to root growth (dry weight production) than AsV, with the first inhibition observed at 0.8 mg/l. Dasgupta *et al.* (2004) reported AsV a root growth inhibition of 90 percent for rice cultivar Azucena and 50 percent inhibition for Bala at 1 mg/l.

Spiked soil

Onken and Hossner (1995) spiked soil with 25 mg/kg AsIII or AsV. In the silt loam soil, a reduced dry matter was first observed after 40 days exposure. At the termination of the experiment (60 days exposure), the dry matter was reduced by approximately 50 percent with no significant difference between AsV and AsIII. In the clayey soil, no toxicity was observed, suggesting that a greater part of the added As was strongly bound to the soil. Taking into account the large uncertainties and fluctuations in soil water concentrations, water from the clayey soil contained 10 to15 times less As.

Jahiruddin *et al.* (2004) spiked silt loam soil with As. First, a Boro rice cultivar developed by the Bangladesh Rice Research Institute “BRRI *dhan* 29” and then an Aman cultivar “BRRI *dhan* 3” was grown. For Boro rice, the first significant effects occurred at 10 mg/kg soil, causing a grain yield reduction of more than 45 percent (Figure 2.1).



Source: Jahiruddin *et al.* (2004)

Note: Soil was contaminated just before the experiments.

Figure 2.1 The effect of As on grain yield and on As concentrations in grains of Boro and Aman rice cultivars consecutively grown in the same pots

The As concentration in grains of Boro rice first increased with the exposure level but then decreased. A possible explanation is that the toxic effects became so severe that As was hardly translocated anymore to the few grains that were produced at 25 mg/kg soil and higher (see also Figure 2.1). For Aman rice, the first significant adverse effects were on the number of grains per panicle and straw yield at 10 mg/kg. At 20 mg/kg soil, grain yield became affected whereas the other parameters were not significantly affected below 40 mg/kg soil.

A shortcoming of Jahiruddin *et al.* (2004) was that no measures to avoid or remove contamination like the dust of samples during sample preparation before digestion were described. This may explain the unlikely high Fe concentrations in grains of ~100 mg/kg whereas concentrations in rice are usually around 5 mg/kg. The chemical analysis did not include a certified reference material (CRM). The reported concentrations can therefore only be regarded as indicative.

Soil culture irrigated with As-contaminated water

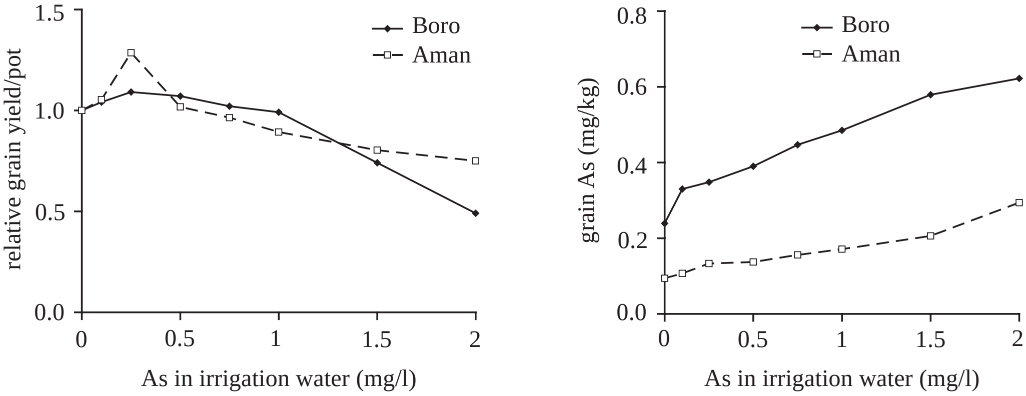
Abedin *et al.* (2002b) exposed rice cultivar BR11 to AsV and studied growth and As uptake. The first observed adverse effect was a reduced root biomass at 0.2 mg/l. Other effects including reduction of plant height, spiklet weight, number of spiklets and grain yield started at 2 mg/l. In an almost similar experimental setup, a reduced root biomass, grain number and grain weight (g/pot; 26 percent reduction) was found at ≥1 mg/l (Abedin *et al.*, 2002a).

Comparing the two studies suggests that the lowest As concentrations associated with toxic effects deviated substantially despite the similar setup. The main reason is probably the difference in the lowest As concentrations used in the irrigation water, namely 0.2 mg/l in Abedin *et al.* (2002b) and 1.0 mg/l in Abedin *et al.* (2002a). In both studies, first effects occurred already at those levels. This indicates that the range of exposure concentrations did not include a concentration so low that it did not cause any effect. It seems that for this particular experimental setup, the lowest concentration causing adverse effects is equal to or below 0.2 mg/l.

Smith *et al.* (1998), cited in Abedin *et al.* (2002a), reported that rice, bean, oats can suffer from phytotoxicity at a soil concentration of 20 mg/kg, whereas for maize and radish this is 100 mg/kg. According to Sheppard (1992), also cited in Abedin *et al.* (2002a), soil type is the most important variable for toxicity of inorganic As to plants, with soil texture one of the most important factors. Inorganic As was five times more toxic in a sandy soil (40 mg/kg) than in a clayey soil (200 mg/kg). Yan-Chu (1994), also cited in Abedin *et al.* (2002a) found a rice yield reduction of ten percent at 13 and 23 mg/kg soil. In sandy soil with 47–52 mg/kg, rice growth was reduced by up to 50 percent and completely inhibited at 109–157 mg/kg soil.

Islam *et al.* (2004b) carried out a similar experiment with the same soil and rice cultivars as Jahiruddin *et al.* (2004) with the difference being that AsV was added via irrigation water during Boro rice cultivation in the Islam *et al*. experiment. During the Aman cultivation As-free irrigation water was used, resembling the field situation. With an increase in As concentration in the irrigation water, first an increase in grain yield was observed, both for Boro rice and Aman rice. After that, yields declined (Figure 2.2). As concentrations in grains steadily increased with As levels in irrigation water (Figure 2.2).

Within the tested range of As concentrations in irrigation water, the observed toxic effects and As accumulation in grains reported by Islam *et al*. (2004b) were far less compared to the observations within the range of soil concentrations used by Jahiruddin *et al.* (2004). At first, the patterns seem to differ, but a closer look reveals that it is most likely that the range of concentrations used by Islam *et al.* (2004b) was narrower than that used by Jahiruddin *et al.* (2004). Comparing the two sets of results for 0–10 mg/kg As in soil shows a similar pattern. In spite of this, it is not known what the true exposure concentrations were and the results cannot be extrapolated to the field. The reports of both sets of authors had the same shortcomings regarding chemical analysis and the overall description of the methodology.



Source: Islam *et al.* (2004b)

Note: Pots were irrigated with contaminated water only during the Boro cultivation. As-free irrigation water was used during the following Aman cultivation.

Figure 2.2 The effect of As on grain yield and on As concentrations in grains of Boro and Aman rice cultivars consecutively grown in the same pots

In conclusion, none of the existing toxicity data can be regarded as representative of the field situation and extrapolations are not yet possible. A better understanding of As in the soil in relation to uptake and toxicity is therefore urgently needed. Ideally, soil parameters should be identified that correlate with uptake and toxicity. The development of a methodology for toxicity experiments that give results representative of field conditions has to be emphasized.

With the elevated As levels found in various paddy fields because of long–term irrigation with contaminated water, it may be possible to study phytotoxicity at the field level. Results from such studies would by definition be representative of the field situation, but a thorough understanding of the critical parameters involved would still be necessary in order to extrapolate the data to locations with other environmental conditions.

Toxicity to micro-organisms

Soil micro-organisms may also be affected by As toxicity (Mahimairaja *et al.*, 2005). Effects of As and on the soil microbial community can be expected with AsIII being more toxic than AsV. Microbes can adapt to As contamination, but this can be accompanied by a change in density and structure of the community. Ghosh *et al.* (2003) reported that microbial biomass and activity were negatively correlated with total and bioavailable As in soil samples from West Bengal. However, the description of the used soils was limited, making it difficult to assess if there were any other reasons like different soil types and land use that could explain the results.

2.3 Arsenic speciation in foods

It is generally recognized that inorganic As is far more toxic to humans than organic As. A well-balanced evaluation of As in foods should thus be based on inorganic As and not on total As to avoid an overestimation of the human health risks.

The methodology to assess As speciation in plant and animal tissue is complicated. To date, the methodology is not yet standardized and certified reference materials (CRMs) for inorganic As are not available. As speciation measurements depend on the pretreatment, extraction technique and storage (Heitkemper *et al.*, 2001; Norra *et al.*, 2005; Pizarro *et al.*, 2003a; Pizarro *et al.*, 2003b). Available values should be regarded as experimentally defined levels of inorganic As species. In Table 2.1, an overview of speciation data on rice has been presented. In summary, rice mainly contained AsV, AsIII and DMA.

Table 2.1 As speciation data for rice

Country	Type	Total As (µg/kg)	Inorganic As (µg/kg)	Extraction efficiency (%)	% inorganic As	Reference
Spain		[a]	0.062	95	[a]	1
Spain	Paella rice	0.17	0.08	78	48	9
Spain	White	0.149	0.126	91	85	
Italy		[a]	0.061–0.069	95	[a]	1
Italy	Various	0.19–0.22	0.10–0.14	77–103	53–65	9
USA	White rice	[a]	<0.025–0.271	[a]	[a]	2
USA		0.303	0.074	[a]	[a]	3
USA	White rice	0.21–0.34	0.021–0.095	86–97	11–35	4
USA	Brown rice	0.160	0.098	99	62	4
USA	Various	0.11–0.40	0.05–0.14	59–90	20–59	9
India	Various	0.03–0.08	0.02–0.05	62–88	36–67	9
Bangladesh	Aman	0.03–0.30	0.01–0.21	51–98	34–86	9
	Aman	0.18–0.31	0.11–0.22	69–81	85–94	10
	Boro	0.21–0.27	0.17–0.22	84–90	81–83	10
China	Unknown	0.22	0.07	85	37	10
	Unknown	0.180	0.164	[a]	91	11
Thailand	Various	0.11–0.20	0.06–0.10	72–97	44–74	9
Unknown	Unknown	0.410	0.367	94	90	5
	NIST 1568a	0.286	0.088	99	31	6
	NIST 1568a	[a]	0.087	[a]	31	6
	NIST 1568a	0.280	0.092	94	34	4
	NIST 1568a	0.283	0.085	99	35	7
	NIST 1568a	0.290	0.080	83	33	9
	NIST 1568a	0.29	0.10	84	33	10

[a] No data.

[b] Certified reference material for total As in rice.

1 = (Pizarro *et al.*, 2003b); 2 = (Lamont, 2003); 3 = (Schoof *et al.*, 1999); 4 = (Heitkemper *et al.*, 2001); 5 = (Kohlmeyer *et al.*, 2003); 6 = (Pizarro *et al.*, 2003a); 7 = (Pizarro *et al.*, 2003b); 8 = (Cava-Montesinos *et al.*, 2003); 9 = (Williams *et al.*, 2005); 10 = (Williams *et al.*, 2006); 11 = (Zhu and Meharg, 2006).

Total As concentrations ranged from 0.03 to 0.4 mg/kg and inorganic As ranged from 0.01 to 0.363 mg/kg. The percentage of inorganic As was highly variable (11–90 percent). Although the results for As speciation in CRM NIST 1568a (rice) showed consistent results, this CRM is only certified for total As.

Williams *et al.* (2005) and Williams *et al.* (2006) presented the first speciation data for rice from Bangladesh and India. Data for Bangladesh indicated inorganic As comprised about 80 percent of the total As present in rice. The results will be discussed in more detail in Section 3.2.

Zhu and Meharg (2006) analysed 600 rice samples from China, mainly Hunan province, for total As, and randomly analysed 17 of those for inorganic As. The average percentage of inorganic As was 91 percent, which was three times higher than that reported by Williams (2006). Assuming a similar percentage of inorganic As in all 600 samples, approximately 50 percent of all samples exceeded the Chinese food safety standard for inorganic As in rice, 0.15 mg/kg.

Kohlmeyer *et al.* (2003) analysed numerous rice and seafood samples. The percentage of inorganic As was usually at least 50 percent with maximum values of more than 90 percent. Of the 180 rice samples analysed, total concentrations were between 0.08 and 0.5 mg/kg fresh weight. A typical raw rice sample contained 0.170 mg/kg AsIII, 0.193 mg/kg AsV, and 0.023 mg/kg DMA, whereas MMA was below detection limit. A typical parboiled rice sample contained 0.102 mg/kg AsIII, 0.010 mg/kg AsV, 0.044 mg/kg DMA, whereas MMA was below detection limit. Raw rice and brown rice had higher totals of As and higher percentages of inorganic As compared to white and parboiled rice. This may suggest that parboiling and/or polishing remove As from the rice and that the As is mainly present in the outer husk and bran layer. Marine fish mainly contained arseno-betaine (AsB) (90–100 percent) and no inorganic As was found. Arsenosugars were predominant in marine algae. High concentrations of AsV were found in some brown algae like *Hizikia* (25.6 mg/kg dw, which is 60 percent of the extractable As). High AsV was also found in a sample of roasted seaweed (12 mg/kg dw, which is 86 percent of the extractable As).

Li *et al.* (2003) reported that the total As concentrations for seafood were 1.7–19.3 mg/kg dw in red algae, 14.6–38.7 mg/kg dw in brown algae and 0.086–7.54 mg/kg ww in marine fish and shellfish. Fish and shellfish contained less than 2 percent inorganic As, whereas inorganic As was not detected in marine algae (both brown and red algae). Fish mainly contained AsB, whereas As-sugars were predominant in algae.

Huang *et al.* (2003) studied As speciation in farmed fish (*Oreochromis mossambicus*) in As-affected areas in Taiwan Province of China. The fish prefer brackish waters and are cultivated in ponds. Although contaminated groundwater is not used anymore for drinking-water, it is still a source of water for aquaculture. Results showed that there was a positive correlation between As in the water and fish. The water mainly contained AsV, whereas AsB predominated in the fish. In fish, total As concentrations were in the range of 18 to 329 mg/kg dw and inorganic As was ~5 mg/kg dw (ranging from 1.7 to 26.1 mg/kg dw).

3. Bangladesh

3.1 Arsenic in irrigation water, soil and crops

Irrigation water

Bangladesh is mainly known for periods of flooding and not so much for drought. However, a lack of water during the dry season and spells of drought at the beginning and end of the rainy season are a threat to agricultural production in Bangladesh. It is also feared that more areas will become drought prone as a result of climate change associated with global warming. During the last three decades, many hectares (ha) of land have been brought under Boro rice cultivation in the dry season by using STWs for irrigation. This is one of the main reasons that the country is self-sufficient in rice production. Boro rice receives the most irrigation water of all crops, with an estimated amount of 1000 mm/cycle. The total area under irrigation is 4 million ha and 75 percent is covered by groundwater resources: 2.4 million ha via 924 000 STWs and 0.6 million ha via 23 000 deep tubewells (DTWs). In the dry season, 3.5 million ha is used for Boro rice, 0.23 million ha for wheat and 0.27 million ha for other crops. Classifying the divisions according to the area under irrigation gives the following ranking: Rajshahi (39 percent) > Dhaka (27 percent) > Chittagong (13 percent) and Khulna (12 percent) > Sylhet (7 percent) and Barisal (2 percent). The area under Boro rice production follows the same pattern. Wheat and other crops follow a somewhat different pattern with Rajshahi being the most important area followed by Dhaka and Khulna (BADC, 2004).

With regard to drinking-water, the As-affected areas are mainly located in the south and southwest, i.e. Khulna, Dhaka and north Chittagong. With an estimated 20 percent of the drinking-water STWs having As concentrations above the Bangladesh drinking-water standard of 0.050 mg/l, it can be expected that a substantial percentage of irrigation STWs also have high As levels. The exact percentage is unknown because the spatial distribution of irrigation STWs is not similar to that of drinking-water STWs. In groundwater, only AsIII and AsV have been found and levels are within the same order of magnitude.

Data from Jessore showed that 87 percent (74 out of 85 tested wells) of irrigation DTWs contained more than 0.050 mg/l (JICA/AAN, 2004). The average As concentration in those DTWs was 0.21 mg/l. This value is very high because DTWs generally have As concentrations below 0.050 mg/l. One of the probable reasons for this is that those irrigation DTWs are tapping water from shallower depths than the drinking-water DTWs in the same area (~100 versus 200 m). Of the irrigation STWs that were at the same depth as the drinking-water STW, 24 percent (59 out of 246 tested wells) contained more than 0.050 mg/l. The average concentration of those wells was 0.07 mg/l.

The terminology STW and DTW is used both in the drinking-water and agricultural sector but it is important to realize that tubewells for irrigation do not necessarily tap water from the same depth as tubewells for drinking-water. The distinction between DTW and STW is first of all based on the capacity of the pump and, although related, not on the depth from which the water is withdrawn. STW pumps use suction-mode (centrifugal) pumps that have a maximum lift of about 7–8 m. The pipes used are less than 10 cm in diameter and irrigate about 4 ha. DTW pumps can, in principle, tap groundwater from any depth, use force-mode pumps, have pipes up to 25 cm in diameter and irrigate up to about 25 ha.

Farid *et al.* (2005) studied seasonal and temporal variation in As concentrations in a single STW. During the monitored Boro season, a small seasonal effect was observed: Starting in January, the concentrations slowly increased reaching the highest concentrations in early March and then declining again to reach the original level in June. The difference between the highest and lowest concentration was only 5 percent. No diurnal variation was found.

Based on available data on drinking-water STWs, it has been estimated that 900 000–1 360 000 kg As per year is brought onto the arable land via groundwater extraction for irrigation (Ali, 2003). The deposition of As on the arable land is high, especially in southwest and south Bangladesh. The northwestern part of the country, which has relatively low As concentrations in the shallow aquifer but has a very high intensity of using irrigation STWs, is also extracting a considerable amount of As from the aquifer. Other sources like P-fertilizer and manure are likely to be minor sources of As, but this needs confirmation.

According to Meharg and Rahman (2003), 150–200 (up to 900) mm water is used for land preparation before planting, and crop growth requires 500–3 000 mm. Conservatively, they assumed 1 000 mm groundwater/year (1 000 l/m^2/year). If the irrigation water would contain 0.1 mg/l and the As would retain in the first 10 cm of soil (assuming soil density of 1 kg/l), the water input would cause a yearly increase of 1 mg As per kg soil. These figures depend strongly on the permeability of the soil. A clayey paddy field may only need water every three days, whereas a sandy field needs water every day. Irrigation of a clayey paddy field is usually stopped a few weeks up to a month before harvest, whereas water input on sandy soil is continued until a few days before harvest. Wheat, maize and vegetables are produced on a smaller scale and require much less water.

Duxbury and Zavala (2005) estimated that ten years of irrigating paddy fields with As-contaminated water would add 5–10 mg/kg soil to 41 percent of the 456 study sites included in their study. Based on existing national data for As in STWs used for drinking-water and the distribution of Boro rice production, Ross *et al.* (2005) estimated that 76 percent of the Boro rice is grown in areas where STWs usually contain less than 0.050 mg/l, 17 percent in areas with 0.050–0.100 mg/l, and 7 percent in areas with more than 0.100 mg/l. In a case study in West Bengal (India), data on As in irrigation water and the paddy soil profile indicated a yearly As input of 1.1 mg/kg to the top soil (Norra *et al.*, 2005).

The bulk production of Boro rice, which is mainly distributed to Dhaka, seems to take place in areas with low As in the shallow aquifer. Although less rice production takes place in the areas with high As in the shallow aquifer, the rice that is being produced is likely to be used for personal/local consumption (A.A. Meharg, personal communication, 2005).

Soil and crops

Meharg and Rahman (2003) carried out a preliminary survey of As in rice and soil from Bangladesh. A total of 71 soil samples was collected throughout the country. The highest measured soil concentration was 46 mg/kg, whereas less than 10 mg/kg was found in areas with low As in irrigation water. The western part of Bangladesh seems to have the highest soil concentrations (>30 mg/kg), followed by the central belt, which is in agreement with groundwater concentrations. At various locations with high As levels in groundwater, low concentrations were found in the soil. However, they did find a correlation between soil concentrations and irrigation water concentrations when the age of the water-well is taken into account. A positive correlation between As concentrations in rice and soil was also found.

Following up on the Meharg and Rahman (2003) study, Williams *et al.* (2006) did an extensive sampling of rice throughout Bangladesh, collecting 330 samples of Aman rice and Boro rice. Importantly, a positive correlation was found between As in the groundwater and As in the rice. This correlation was stronger for Boro rice than for Aman rice. Highest As concentrations in rice were all from districts in the southwest, namely Faridpur > Satkhira > Chuadanga > Meherpur. For detailed results on reported As concentration in rice refer to paragraph 3.2.

In agreement with Meharg and Rahman (2003), data from a preliminary nationwide survey of As in soil, crops and irrigation water indicate that the soils in the west and southwest part of Bangladesh contain the highest As concentrations (Miah *et al.*, 2005). In these parts, irrigated soils had higher levels of As compared to adjacent non-irrigated soils. In the irrigated soils, the first 0–15 cm had the highest levels of As. In other parts of the country, irrigated and non-irrigated soils did not differ in As concentrations. The differences in soil concentrations were, however, not reflected by As levels in the rice plants.

Islam *et al.* (2005) studied As levels in water, soil and crops at 456 locations in five *upazilas*. The average As concentration in the soil was 12.3 (ranging from 0.3 to 49 mg/kg) and the *thanas* were classified according to soil concentrations: Faridpur > Tala > Brahmanbaria > Paba > Senbag. Of all soil samples, 53 percent contained less than 10 mg/kg, 26 percent contained between 10.1 and 20 mg/kg, and 18 percent contained more than 20 percent. Concentrations both between and within *thanas* were highly variable. The same was observed at the command area and paddy field level. In some cases this correlated with the distance to the tubewell used, in other cases the variation seemed to be random or related to micro-elevation. They also found a high seasonal variation in As soil concentrations. At the end of the Boro (dry) season the soil concentration had increased sharply when irrigated with As-rich water. Most of it was again removed after the Aman season, i.e. after flooding. There are various explanations for this phenomenon: 1) As desorbs to the standing water and is then removed laterally; 2) the top layer may be eroded and run off during heavy rainfall; 3) volatilization of As during prolonged periods of flooding; and 4) leaching of standing water desorbing and transporting As from the topsoil to deeper layers. These different processes have not been quantified yet. The general opinion is that leaching is an unlikely process because of the slow percolation rate, 2–4 cm/day (Brammer, 2005; Islam *et al.*, submitted; Islam *et al.*, 2005). This explains why soil concentrations in the first 15 cm are generally highest compared to the rest of the soil column.

Islam *et al.* (2000) reported total As concentrations of 5–33 mg/kg with an average of 17 mg/kg for some soil samples from Nawabganj, Rajarampur, Jessore, Jhenidah and Comilla. However, the study has some limitations: No information was provided about the use of the sampling locations (e.g. fallow land, paddy field, other crops, residential area, etc.); the chemical analyses were not described in detail; and the use of CRMs was not mentioned.

Islam *et al.* (submitted) collected 100 samples of irrigation water, soil (composite sample of five randomly taken samples at 0–15 cm depth) at 100 STW command areas in Chapai Nawabganj (Sadar *upazila*) during the Boro season of 2003. The irrigation water contained 0.025–0.352 mg/l (mean 0.075 mg/l).

Soil concentrations were 5.8–17.7 mg/kg (mean 11.2 mg/kg), straw contained 1.48–17.6 mg/kg (mean 5.88 mg/kg) and rice grain contained 0.241–1.298 (mean 0.759). Poor correlations were found between grain and soil and between grain and water only. A good correlation was found between grain and straw. The total soil As was correlated with As in irrigation water indicating As accumulation in soil because of As rich irrigation water input. In terms of quality, the study had some limitations: Rice cultivars were not identified; pretreatment of straw and grains (e.g. rinsing with As free water to remove dust) was not mentioned; and CRMs were not included.

Rapid adsorption of As from irrigation water to soil may explain the spatial patterns found in irrigation canals and some paddy fields (Farid *et al.*, 2005). An alternative hypothesis is that Fe^{2+} present in irrigation water is rapidly oxidized to Fe^{3+} when the water is exposed to air, resulting in the precipitation of AsV (Islam *et al.*, submitted).

Five soil profiles of 15 m depth were collected at Deuli village (near Samta village, southwest Bangladesh) (Yamazaki *et al.*, 2003). Although not specifically mentioned in the paper, it seems that samples were collected from fallow land, not used for agriculture. Results showed that the soil concentrations were dependent on the type of sediment. Sandy sediments contained 3–7 mg/kg (median: 5 mg/kg), clayey sediments contained 4–18 mg/kg (median: 9 mg/kg), whereas peaty and peaty clay sediments contained 20–111 mg/kg. The first 6 m contained sandy and clayey layers whereas the peaty layers were found at a depth of 7–10 m. Below 10 m, a sandy layer was dominant. The general pattern was that the As concentrations were relatively constant except for the 7–10 m layer, which contained a much higher total As.

Twenty-five locations in five *thanas* (Chapai Nawabganj Sadar, Kushtia Sadar, Bera, Ishurdi and Saishabari) of four districts were sampled (Alam and Sattar, 2000). Soil concentrations ranged from below detection limit to 56.7 mg/kg. Ten out of 25 locations contained As concentrations of more than 20 mg/kg. The As concentrations in the adjacent water-wells ranged from below detection limit to 0.071 mg/l, i.e. low to moderate. A positive correlation was found between As concentrations in the soil and water. In terms of quality, the paper had a number of limitations: Land use of the sampling locations was not described, which is an important feature because irrigation varies according to the type of crop; there was no mention of whether the sampled wells were for drinking-water or irrigation water; and in the methodology section quality assurance/quality control (QA/QC) was not described and CRMs were not used.

Das *et al.* (2004) collected soil samples (n = 18) in three *upazilas*: Kachua, Hajiganj (both in Chandpur district) and Sharishabari (in Jamalpur district). Composite soil samples (15–45 cm depth) were probably taken from arable land, but specific land use was not mentioned. A CRM for soil was not included. Soil concentrations ranged from 7.3 to 27.3 mg/kg with an average of 15.7 ± 6.6 mg/kg. A positive correlation was found between the As in STWs and soil.

Except for some experimental work with rice exposed to artificially contaminated soils (see Section 2.2), hardly any work has been done on the potential risk of As in irrigation water to crop production. Duxbury *et al.* (2003) studied As concentrations in rice in relation to yield and panicle sterility in Bangladesh. They did not find any correlation and concluded that there was no indication of toxic effects under the current field conditions. Regarding the study design, the authors acknowledged that a potential important factor, the effect of different rice varieties, was not studied. This may have hidden a possible correlation between yield and As concentrations in grains. The authors stated that because of the continuous input of As-rich irrigation water, toxic effects cannot be excluded in the future.

In agreement with Takahashi *et al.* (2004), As concentrations in soil water from flooded paddy fields increased with the duration of flooding reaching levels above 3 mg/l (Loeppert *et al.*, 2005). During both Boro and Aman seasons, the soil water concentrations thus largely exceeded the irrigation water concentrations. This is important to realize, particularly for Aman rice, which is rainfed and contains As below the detection limit. It has been hypothesized that As uptake and toxicity to rice is much better correlated with As in the soil water than with As in irrigation water or total As in the soil.

In the 2006 Boro season, a small but detailed pilot study was conducted in which phytotoxicity to rice was studied at the field level in a paddy field in Faridpur contaminated by twenty years of irrigation. Preliminary data indicate a clear negative correlation between As in the soil water and plant growth (G.M. Panaullah, personal communication, 2006). These results emphasize the need to further investigate the possible risks of As in irrigation water to crop production.

Two studies from West Bengal have provided strong evidence for accumulation of As in topsoil because of irrigation with contaminated groundwater (Norra *et al.*, 2005; Roychowdhury *et al.*, 2002a). Roychowdhury *et al.* (2002a) reported for Domkal block that fallow lands contained 5.31 mg/kg, whereas adjacent lands irrigated with 0.082 mg/l and 0.17 mg/l contained 11.5 and 28.0 mg/l, respectively. The calculated input of As was approximately 1.6–16.8 kg/ha/yr. The age of the tubewells was not given and an estimation of the total amount of As deposited on these lands is therefore not possible. The suggestion that the samples from the fallow land and irrigated land did not originate from the same parent material, which would (partially) explain the observed differences, cannot be excluded. However, the good correlations between concentrations in STWs and soils clearly showed the effect of the contaminated irrigation water on As concentrations in the soil. On a smaller scale, measurements of As in paddy fields with increasing distance from the tubewells also gave a good correlation, again showing that soil concentrations increased because of the application of contaminated water. The same was reported by Hossain (2005) for a number of locations in Faridpur, Bangladesh. This study also found a relationship between micro-elevation and As soil concentration.

Norra *et al.* (2005) collected water, soil and plant samples from three fields in Kaliachack I block: one paddy field and one adjacent wheat field both irrigated with water containing 0.5–0.8 mg/l, and one reference paddy field not contaminated with As. Soil profiles were collected down to a depth of 110 cm. The upper topsoil in the contaminated paddy field contained 38 mg/kg, the less intensively irrigated wheat field grown with wheat contained 18 mg/kg, whereas a reference paddy field contained 7 mg/kg. The soil profiles of the contaminated paddy field and wheat field clearly showed a decreasing As level with increasing depth. Although, to a lesser extent, it is important to note that As did not only build up in the paddy field but also in the wheat field. This has not been reported before. According to the author, continuation of irrigation with As-rich water could result in alarmingly high soil concentrations within a few decades.

Conclusions: irrigation water, soil and crops

There are indications that soil concentrations are increasing over time because of irrigation with As-contaminated water. Data are, however, insufficient in terms of quantity and quality. It is thus still unclear under what specific conditions and over what period of time As is accumulating in the soil. Major problems with many data available in Bangladesh are the quality of the chemical analysis and the description of the methodology in general. The international symposium on the behaviour of arsenic in aquifers, soils and plants, held in January 2005 in Dhaka, recommended long-term monitoring of As in water, soil, and crops under the various conditions present in Bangladesh, along with detailed studies on the behaviour of As in paddy fields.

The risk of As-contaminated irrigation water to crop production has received little attention until now. To evaluate current and future soil concentrations, representative toxicity data for crops are needed, both for flooded and non-flooded soil conditions. Thus, field studies to test if As is one of the factors limiting growth in the field should be emphasized. Further, it should become clear what soil parameters correlate with uptake and toxicity and, based on that information, a toxicity database for different rice cultivars and other crops could be developed to set standards for As in flooded and non-flooded soils.

3.2 Human exposure

Food safety standards

When properly applied, food safety standards are a useful tool to evaluate levels of contaminants in foods. This section will therefore briefly discuss available As food safety standards before presenting published As concentrations in foods from Bangladesh.

Many papers refer to the Australian maximum permissible concentration (MPC) for As in foods (1 mg/kg) to evaluate their results (Abedin *et al.*, 2002a; Abedin *et al.*, 2002b; Das *et al.*, 2004; Islam *et al.*, 2004b; Jahiruddin *et al.*, 2004; JICA/AAN, 2004). The Australian MPC has however various limitations in the context of Bangladesh and other countries where rice is a staple food. Australia does not have a rice based diet, and thus the Australian food safety standard is probably too high for countries where rice is a staple food. Further, the MPC has been set for total As only, which does not take into account the great differences in toxicity between organic and inorganic As species. It is therefore advised not to use the Australian MPC for evaluating As concentrations in rice and other foods in the Bangladesh or Asian context. To evaluate As in foods, it is recommended to develop a guideline value for inorganic As in foods and specifically for rice taking into account dietary consumption patterns.

Table 3.1 Chinese food safety standard for inorganic As (mg/kg) in various products

Product	Inorganic As
Rice	0.15
Flour	0.10
Other cereals	0.20
Vegetables	0.05
Fruit	0.05
Poultry	0.05
Egg	0.05
Milk powder	0.25
Fresh milk	0.05
Beans/pulses	0.10
Fish	0.10
Algae	1.50
Shellfish	0.50

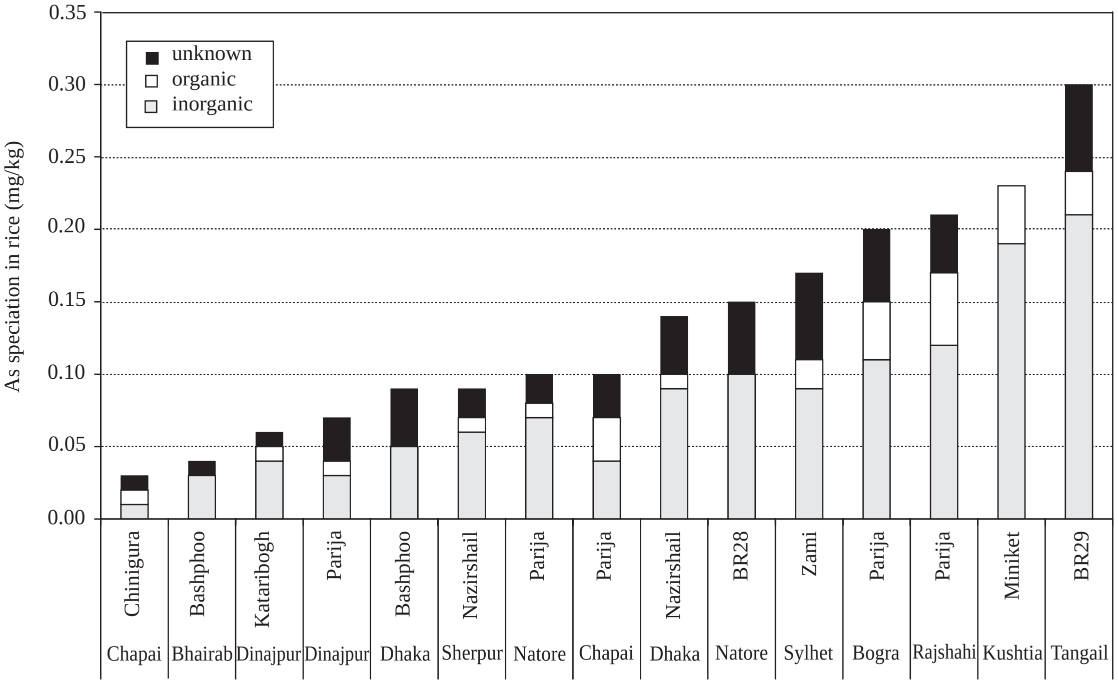
In October 2005, the Ministry of Health of China adopted a new food safety standard on As in foods, GB 2762-2005 (Table 3.1). This standard, specified for a variety of food products, has been set for inorganic As, and not for total As. This is an important step as it recognizes that total As in foods is not appropriate for evaluating food safety. This standard may serve as a guideline for other countries in the region that could derive their respective national standards from it. A re-examination of the data of Abedin *et al.* (2002a), Abedin *et al.* (2002b), Das *et al.* (2004), Islam *et al.* (2004b), Jahiruddin *et al.* (2004), JICA/AAN (2004), which used the Australian food safety standard, shows that most of their results exceed the Chinese food safety standard.

Arsenic in foods from Bangladesh

Inorganic As

The first data on As speciation in rice from Bangladesh have been published recently (Williams *et al.*, 2006; Williams *et al.*, 2005). Williams *et al.* (2005) collected 15 samples of various rice cultivars from the wholesale market in Dhaka and analysed for total As, DMA and inorganic As (Figure 3.1). The average total As concentration was 0.13 ± 0.02 mg/kg (ranging from 0.03 to 0.30 mg/kg). The method to extract the As species trifluoroacetic acid (TFA) had an efficiency of approximately 80 ± 12 percent. This means that a relatively small portion of the As is unaccounted for and its speciation is unknown. Assuming that the speciation pattern of this portion is equal to the other portion (A. Meharg, personal communication, 2005), the average percentage of inorganic As was 80 ± 3 percent. It also implies that three out of 15 samples exceeded the Chinese food safety standard.

Compared to other countries, rice from Bangladesh (and India) had the highest percentage of inorganic As (80 percent), against 42 percent in rice from the USA. This indicates that the percentage of inorganic As in rice is not a constant factor geographically and probably depends on cultivar and growth conditions.



Source: Williams *et al.* (2005)

Note: All samples bought at the wholesale market in Dhaka. Cultivars are shown vertically below the x-axis whereas districts are shown horizontally below the x-axis.

Figure 3.1 As speciation in 15 rice samples of various cultivars and districts

Williams *et al.* (2006) analysed another 21 rice samples from Bangladesh (seven different cultivars, both Boro and Aman) for As speciation of which approximately half of the samples exceeded the Chinese food safety standard. The observed As speciation pattern was similar to Williams *et al.* (2005). Combining the data from Williams *et al.* (2005) and Williams *et al.* (2006) shows a strong positive correlation between total As and inorganic As in rice from Bangladesh, and indicates that 80 percent of inorganic As in rice may be representative of use within the country (Figure 3.2).

As speciation analysis on a number of vegetables (arum stolon and tuber, potato, bitter gourd, ribbed gourd, pointed gourd, teasel gourd, plantain banana and long yard bean), pulses and spices indicated that all As was present in the inorganic form (Williams *et al.*, 2006). The extraction efficiency with TFA varied from a reasonable 79 percent to 128 percent for vegetables, 70 percent for spices, to a low 45 percent for pulses. All detected As was in the inorganic form.

Total arsenic

With only two studies published on inorganic As in rice and vegetables from Bangladesh, some data on total As in foods from Bangladesh and its neighbour West Bengal, India are presented here as well.

Williams *et al.* (2006) collected a large number of samples (rice: 330, vegetables: 94, pulses and spices: 50) throughout the country. For rice, the results clearly showed that the highest levels of As were found in the southwestern part of the country, and there was a positive relationship between As levels in rice and As levels in groundwater (Figure 3.3). With a factor of 1.3, Boro rice contained significantly more As than Aman rice. This could be caused by a difference in rice cultivars grown during the Boro and Aman seasons. An alternative explanation is that Aman rice is mainly rain fed, while Boro rice is irrigated with groundwater containing As. Concentrations in Boro rice were in the range of 0.04 to 0.91 mg/kg, whereas Aman rice contained <0.04 to 0.92 mg/kg. Assuming that 80 percent of the total As was in the inorganic form, a substantial number of samples exceeded the Chinese food safety standard.

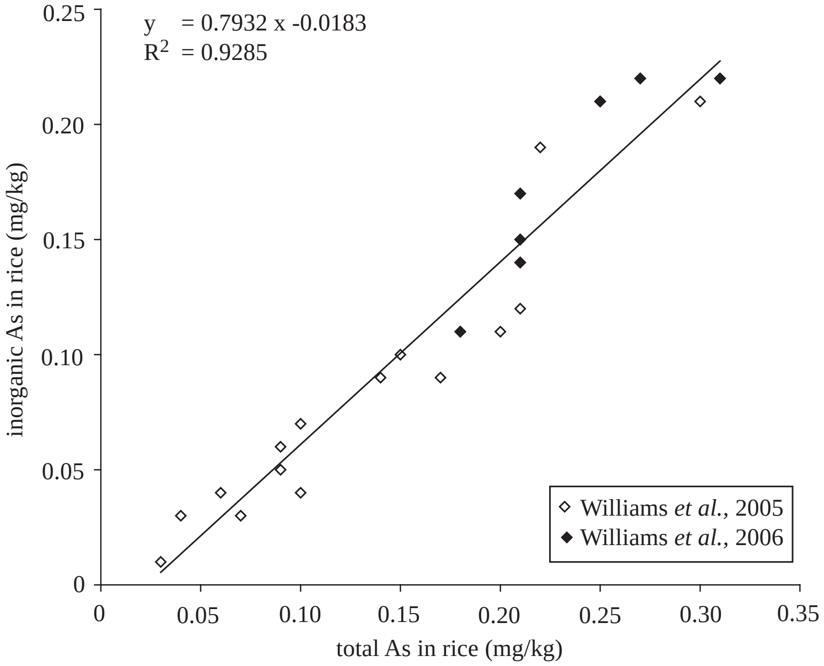
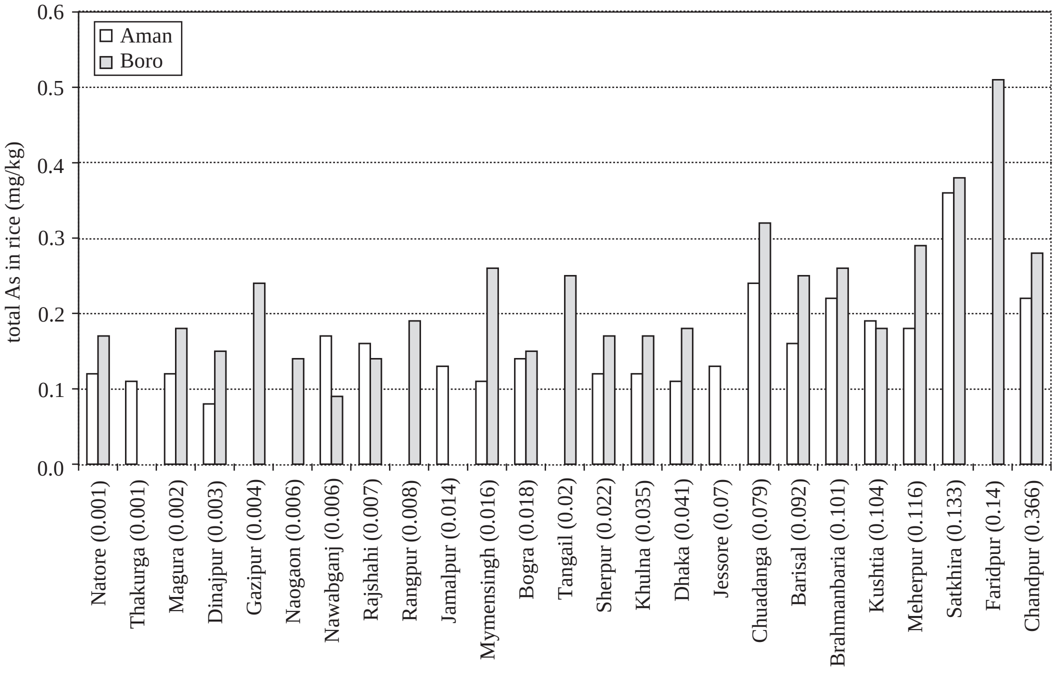


Figure 3.2 The correlation between total As and inorganic As in rice from Bangladesh



Source: Williams *et al.* (2006)

Note: Figures in parentheses represent the mean As concentrations in the shallow aquifer of each district given in mg/l.

Figure 3.3 The average concentrations of total As in rice (both Boro and Aman cultivars) collected from 25 districts

In Table 3.2, other data on total As concentrations in rice from Bangladesh and West Bengal have been summarized. Roychowdhury *et al.* (2002b) found that cooked rice had approximately twice the level of As as raw rice. This is likely to be because of parboiling and/or boiling the rice in As-contaminated water. Bae *et al.* (2002) reported that As concentrations in rice after boiling in As-contaminated water (0.223–0.373 mg/l) were increased from 0.178 mg/kg dw to 0.228–0.377 mg/kg dw. On the other hand, Duxbury *et al.* (2003) found that processing (parboiling and milling) of rice grown and processed in low and high As areas reduced As concentrations by ~20 percent. Unpublished data from Meharg and co-workers showed the same pattern: processed rice bought at markets had lower As concentrations than rice sampled in the field (A. Meharg, personal communication, 2005).

Table 3.2 Total As concentrations (mg/kg dw) in rice from Bangladesh and West Bengal

Country	Location	Rice (mg/kg)	Soil (mg/kg)	Remarks	Ref.
Bangladesh	Gazipur	0.092 (0.043–0.206)	10.9; 14.6	11 cultivars[a]	1
	Bogra	0.058–0.104	4.9–15.5	4 cultivars	1
	Dinajpur	0.203	11.7	BR11	1
	Naogaon	1.835	24.3; 26.7	BR11	1
	Nawabganj	1.747; 1.775	15.7; 20.9	BR11	1
	Mymensingh	0.078	6.0–25.4	BR8	1
	Rangpur	0.185	6.5–11.5	BR11	1
	Rajshahi	0.075–0.117	7.8	3 cultivars	1
	Various	0.183 (0.108–0.331)		Raw Boro rice (n = 78); 14% water	2
	Various	0.117 (0.072–0.170)		Raw Aman rice (n = 72) 14% water	2
	Various	0.125		Processed rice (n = 21)	2
	Kachua, Hajiganj, Sharishabari	0.14 (0.04–0.27)	15.68 (7.31–27.28) (15–45 cm depth)	(n = 10)	3
	Northwest	0.173		As-affected area	7
	Chapai Nawabganj	0.759 (0.241–1.298)	11.2 (5.8–17.7)	As-affected area; n = 100	8
	Market	0.13 (0.3–0.30)		Various Aman cultivars (n = 15)	9
West Bengal	Jalangi & Domkal	0.239 (0.043–0.662)		Raw rice (n = 34)	4
	Jalangi & Domkal	0.569 (0.198–1.930)		Cooked rice (n = 18)	4
West Bengal	South 24-Parganas	0.072 ± 0.010		Precooked rice	5

[a] Lowest concentration in BR11: 0.043 mg/kg dw.

[b] Only inorganic As and DMA were found.

1 = (Meharg and Rahman, 2003); 2 = (Duxbury *et al*., 2003); 3 = (Das *et al*., 2004); 4 = (Roychowdhury *et al*., 2002b); 5 = (Mandal *et al*., 1998); 7 = (Watanabe *et al*., 2004); 8 = (Islam *et al*., submitted); 9 = (Williams *et al*., 2005).

A summary of the results of Williams *et al.* (2006) on total As in vegetables, roots and tubers, pulses, and spices is presented in Table 3.3. Recalculating the presented concentrations to wet weight shows that a number of samples exceed the Chinese food safety standard. More specifically, the mean total As concentration of the following items exceeds the standard: radish leaf, plantain banana, long yard bean, arum tuber, giant taro, potato, and arum stolon. The authors do not give a clear indication of any correlation between As in these products and As levels in the shallow aquifer in the area where the samples were collected.

Table 3.3 Total As concentrations in vegetables, roots and tubers, pulses, and spices

Group	Number of different food items per group	Total number of samples per group	Range of means of different items (mg/kg dw)	Min-max per group (mg/kg dw)
Leafy vegetables	5	9	0.13–0.79	0.10–0.79
Fruit vegetables	16	64	0.11–0.62	0.05–1.59
Roots and tubers	6	21	0.20–0.74	<0.04–1.93
Pulses	5	25	0.03–0.10	<0.04–0.20
Spices	5	25	0.04–0.49	<0.04–0.98

Source: Williams *et al.* (2006)

Three other papers were found on total As in non-rice foods from Bangladesh and West Bengal, namely Alam *et al.* (2003), Das *et al.* (2004) and Roychowdhury *et al.* (2002b). They all described the methodology reasonably well and certified reference materials were included. Most samples were collected from a few locations known for high As in the shallow aquifer. Concentrations in vegetables, fruits, spices, and freshwater fish ranged from less than 0.04 mg/kg dry weight (dw) to 3.99 mg/kg dw, with most samples having less than 0.5 mg/kg dw. Das *et al.* (2004) reported that total As concentrations in fish were below 1 mg/kg dw. However, data from Taiwan Province of China showed that fish cultivated in As-rich water may lead to high levels of inorganic As (Huang *et al.*, 2003). Alam *et al.* (2003) mistakenly did not convert dry weight concentrations to wet weight before data interpretation, including the estimation of the daily As intake. This has caused a fivefold overestimation of the daily exposure to As and other metals that were included (corresponding author E. Snow, personal communication, 2004).

On a dry weight basis, a number of vegetable samples contained higher As concentrations than rice. However, this does not necessarily mean that As in vegetables poses a higher risk to human health than As in rice. From a food safety perspective, water contents and food consumption data need to be taken into account. Usually, food consumption data are on a raw weight basis, i.e. fresh vegetables (usually containing 70 to 90 percent water) and uncooked rice (containing approximately 13 percent water). Comparing As concentrations in rice and vegetables on a raw weight basis shows that As levels in rice are usually higher.

Concerning dairy products and meat, various researchers have expressed concern about possible transfer of As from water and straw to cattle (Abedin *et al.*, 2002a; Jahiruddin *et al.*, 2005; Panaullah *et al.*, 2005). However, no peer-reviewed publications on this issue have been found, indicating a need to investigate this issue.

Food consumption in Bangladesh

Only one study reported food consumption on a gram/capita/day basis in Bangladesh (Hels *et al.*, 2003). That study reported data from two villages, Falshatia (Manikganj) and Jorbaria (Mymensingh), covering 1981/1982 and 1995/1996. Twenty-four hour food weighing data were collected from October to November and from January to March. Data were collected at household level from which consumption

Table 3.4 Estimated food consumption for two villages in Manikganj and Mymensingh (g/capita/day) in 1995/1996

Food group	Manikganj		Mymensingh		Average
	Oct-Nov n = 152[1]	Jan-Mar n = 145	Oct-Nov n = 152	Jan-Mar n =143	
Rice	399	454	446	502	450
Non-rice cereals	20	20	2	12	14
Green leafy vegetables	28	49	46	37	40
Other vegetables	106	166	59	221	138
Roots and tubers	43	101	64	37	61
Fish	52	37	37	30	39
Animal products excluding fish	23	25	35	39	31
Fats and oils	7	5	10	9	8
Spices	5	6	8	6	6
Fruits	7	7	6	9	7
Miscellaneous	10	13	5	11	10
Total intake	701	887	719	914	804

[1] n = number of surveyed households; average household size = 5; average consumption unit is 4.5.

Source: Hels *et al.* (2003)

per capita values were derived. Corrections were made for the number of meals consumed outside the house. All members of a household were treated equally. In Table 3.4, food consumption data are presented for the period 1995/1996. For most data, the standard error was less than 20 percent. Seasonal variation in food consumption was observed, particularly for rice and vegetables. There were also differences between the villages.

Drinking-water consumption in Bangladesh

Watanabe *et al.* (2004) studied water intake by adult men and women in two As-affected areas and reported a total water intake of 4.6 and 4.2 l/day, respectively. Two methods were used to assess drinking-water intake, namely 24-hour self report and interviews with frequent visits. The range of water intake was 1 to 6 l/day. There was no difference between direct intake of drinking-water between men and women (both ~3 l/day). Water intake via food preparation determined by field experiments added another 1.6 l/day (men) and 1 l/day (women) per day.

Mandal et al. (1998) estimated the drinking-water intake of a small number of people in an As-affected village and the average intake by adults was 4 l/day. The highest intake values for some individuals were 7 and 8 l/day. However, reliability of the data is unknown because the methodology was not described. In contrast with Watanabe *et al.* (2004), a great difference in water consumption between the sexes was reported. The general impression is that adults consume 3 l/day and an additional 1 l/day from foods.

Dietary exposure to arsenic

Williams *et al.* (2006) concluded that rice is the predominant source of inorganic As from foods. This was based on a daily consumption of 500 g rice, 130 g vegetables, 12 g pulses and 5 g spices (all weights based on unprepared products) and data on inorganic As and total As in a range of food items from Bangladesh. Most of the Boro rice samples collected contributed at least 50 percent to the provisional maximum tolerable daily intake (PMTDI) for inorganic As (0.126 mg/day for a 60 kg person). That leaves only 0.66 mg/day or less to other sources of exposure including drinking-water. Assuming a realistic level of inorganic As of 0.2 mg/kg in rice, a drinking-water concentration of 0.050 mg/l (Bangladesh drinking-water standard) and a water consumption of 3 l/day, the total daily intake of inorganic As would be 0.25 mg/day, exceeding the PMTDI by a factor of two. Rice would contribute 40 percent of total daily intake of As.

Food items other than rice only make a minor contribution. Even for a worst case scenario (consumption of 130 g/day of a vegetable with the highest As level on a wet weight basis, which in this study is potato with 0.23 mg/kg ww), the contribution is only 0.03 mg/day (25 percent to the PMTDI). In a number of cases, arum stolon has been receiving particular attention because of the reportedly high levels of As. The data show that high levels of As in arum stolon (in this study 1.93 mg/kg dw, i.e. 0.193 mg/kg wet weight (ww)) would only contribute 0.025 mg/day. This emphasizes the need to consider As concentrations in food items from the perspective of the overall dietary intake of inorganic As. Also the important nutritional value of vegetables like arum should be taken into account before conclusions are drawn on the risks of As in such food items.

Roychowdhury *et al.* (2002b) estimated the daily intake of total As via water and food for two locations in West Bengal. The intake via foods was approximately 180 and 97 µg/day for adults and children (10 years old) respectively. Adults and children were exposed to approximately 400 and 200 µg/day via drinking-water. Drinking-water counted for ~70 percent of the exposure whereas rice contributed ~30 percent. The authors poorly described the method of collecting data on water and food consumption, and compared to Hels *et al.* (2003), they used substantially higher values for food consumption.

Watanabe *et al.* (2004) estimated that the daily intake of total As by adults was approximately 600 µg/day (male: 674 µg/day, female: 515 µg/day) with 70 percent via drinking-water and 10 percent via rice. The food consumption data seemed to be a rough estimation only. Neither Roychowdhury *et al.* (2002b) nor Watanabe *et al.* (2004) included any variation in concentrations, consumption and seasonal effects in their exposure assessment.

Duxbury *et al.* (2003) analysed 150 rice samples from Bangladesh. Assuming a rice consumption of 400 g/day with 0.250 mg/kg and a water intake of 4 l/day with 0.050 mg/l As (drinking-water standard Bangladesh), the total daily intake would be 0.3 mg/day. Rice would contribute 33 percent. Fourteen percent of their rice samples contained ≥ 0.250. Meharg and Rahman (2003) assumed a rice consumption of 420 g/day with 0.5 mg/kg and a water intake of 2 l/day (the WHO default value) with 0.1 mg/l. The calculated total daily intake was 0.41 mg/day and rice contributed 50 percent. However, taking into account the climate in Bangladesh and the high consumption of rice, 2 l/day is likely to be an underestimation (Watanabe *et al.*, 2004).

After assessing the exposure levels, the results need to be compared to a reference value such as a tolerable daily intake (TDI) value. For As, only a provisional maximum tolerable daily intake (PMTDI) is available. This provisional value of 0.0021 mg/kg body weight/day for inorganic As was established in 1988 and is commonly used to evaluate dietary intake (WHO, 1996). After almost two decades, the PMTDI still has not been ratified.

When evaluating risks to human health associated with As in foods, other sources of exposure such as drinking-water have to be taken into account as well. The WHO guideline value is 0.010 mg/l and the Bangladesh drinking-water standard is 0.050 mg/l (Duxbury and Zavala, 2005; Williams *et al.*, 2005). Assuming a body weight of 60 kg, the PMTDI is 0.126 mg/day. A water consumption of 3 l/day with 0.050 mg/l would already exceed the PMTDI, regardless the levels of As in foods. This suggests that the PMTDI and the Bangladesh drinking-water standard need to be evaluated so that a proper assessment of As in foods can be made.

The human health risk assessment is likely to be more complicated because of the prevalence of micronutrient deficiency in Bangladesh and many other Asian countries, particularly among women and children. For example, studies have reported on the selenium and As interaction. Arsenic is toxic by itself and it also interacts with selenium, resulting in excretion of their mutual metabolite (Gailer *et al.*, 2000). As selenium is an essential micronutrient, this confounding excretion of selenium can aggravate further micronutrient deficiency among the most vulnerable subpopulations and can thus be a health concern.

Conclusions: human exposure

It has become clear that dietary exposure can contribute significantly to the total daily intake of inorganic As. More data on food and water consumption patterns and As levels in foods are needed to refine the human exposure assessment. It is further recommended to review the status of the PMTDI and to propose a food safety standard for inorganic As.

It is important to realize that there are clear indications that As concentrations in rice are increasing over time because of the prolonged input of As-contaminated irrigation water. This may offset the efforts in the drinking-water sector to reduce human exposure to As through water consumption. This emphasizes the need to further investigate As in the food chain and develop appropriate management options.

3.3 Agricultural management options

Even though the risks to food safety and, in particular, to crop production are not yet fully understood, it can be stated generally that the input of contaminants to the environment should be avoided or, at least, minimized, and that natural resources such as groundwater should be used in a sustainable way. From this perspective, there are various topics that can be explored to address management options (Brammer, 2005; Duxbury *et al.*, 2003; Lauren and Duxbury, 2005; Meharg, 2004; Panaullah *et al.*, 2005; Ross *et al.*, 2005):

- Reduce irrigation water use in rice cultivation. This will reduce As input, reduce leaching of nutrients and minimize extraction of groundwater resources. The overall aim of optimizing groundwater use should be to improve sustainable agricultural production. According to BRRI, farmers could apply 40 percent less irrigation water without any yield losses compared to current practices. If water input could be reduced to such an extent that the soil conditions become more aerobic, the solubility of As and, therefore, the uptake of As will be minimized as well. Less irrigation water means fewer costs for the farmer, but it requires a reliable water supply.
- Promote cropping patterns that require less irrigation water in areas with high As in soil and/or irrigation water. For example, depending on soil conditions, replace Boro rice with crops like wheat and maize.
- Select/breed rice cultivars that are tolerant to As and have a limited uptake of As. This could be useful only in combination with a minimized As input in the soil to avoid As buildup in the soil. Rice cultivars show variation in their response to As exposure, but only a limited number of cultivars has been screened to date. A systematic screening of the numerous cultivars could reveal differences in As uptake, tolerance, translocation and speciation.

Phytoremediation has been suggested as a means to remove As from soil. There are two main reasons why this is an unlikely option. First, there may be no need to remove As actively from the soil. Second, phytoremediation is a very slow process and thus not a pragmatic approach for agriculture.

4. Summary and recommendations

4.1 Knowledge and gaps in knowledge

This report has gathered sufficient evidence showing that As in irrigation water can result in land degradation, adversely affecting incomes and agro-ecosystem services in terms of their ability to provide a sustainable source of sufficient and safe foods. The soil contamination can cause contamination of crops and foods, resulting in risks to food safety and thus to human health. The continuous contamination of soil is a growing threat to crop production itself, and thus to sustainable agriculture, because As in soils will become toxic to plants and other organisms at a certain level. This would result in reduced crop yields and thus pose a risk to incomes and the nutritional status of rural farming communities. To date, the risks of using As-contaminated groundwater resources for irrigation have not received sufficient attention and have not been addressed within the framework of land degradation. The current and future extent of land degradation caused by As-contaminated irrigation water is still unknown.

An overview of our knowledge and gaps in our knowledge is given below. It should be remembered that a number of points are based on results from Bangladesh only. These need to be validated in other countries, taking into account differences in agricultural practices, environmental conditions, food habits and other factors.

CURRENT KNOWLEDGE

- Twelve countries in the region have reported high levels of As in parts of their groundwater resources.
- All these countries use groundwater resources for irrigation, but the extent differs substantially between and within countries.
- Depending on local conditions, irrigation with As-contaminated water can result in land degradation through As accumulation in topsoils.
- Depending on local conditions, an increase in soil concentrations can be reflected in concentrations in crops, including the edible parts.
- A substantial number of rice samples from Bangladesh contained As levels exceeding the Chinese food safety standard for inorganic As.
- Rice cultivars and other crops can differ substantially in As accumulation and in the percentage of inorganic As.
- The limited number of rice samples analysed for inorganic As indicates that various rice cultivars from Bangladesh, China and India contain a high percentage of inorganic As.
- Rice can contribute significantly to the total daily intake of inorganic As through water and foods in Bangladesh because of the high rice consumption and the relatively high levels of inorganic As in rice.
- Of all foods in rice based diets, rice is likely to be the main contributor to the daily intake of inorganic As.
- Although some vegetables have been reported to have high levels of As, their contribution to the total daily intake is low because of their low consumption rate. In addition, many vegetables are only available during specific seasons.

IDENTIFIED GAPS

- The extent of using As-contaminated groundwater resources for irrigation in Asia has not been quantified.
- The scale of As accumulation in topsoils through As-contaminated irrigation water in Asia is unknown.
- The scale of land degradation caused by irrigation with As-contaminated water is unknown.
- Factors determining As accumulation in soils are not sufficiently understood and quantified.
- The relationship between As in water, soil and plants has not been quantified.
- Management options to prevent and mitigate As-contamination of agricultural land have not been developed.
- Uptake and toxicity of As in crops cannot yet be predicted.
- Limited knowledge is available about the differences between plant species and cultivars in As uptake, sensitivity, translocation, and speciation.
- There are no plant toxicity data representative of the field situation.
- There is no insight into the risks of As in water and fodder to livestock and their food products.
- There are no policies concerning the use of As-contaminated groundwater for irrigation.
- Only limited data on inorganic As in rice, vegetables and other foods are available.
- The uptake efficiency/bioavailability of As in rice and other foods after consumption is largely unknown.
- The provisional tolerable daily intake for dietary inorganic As intake is still provisional 18 years after issuance, indicating uncertainties about the acceptable level.
- Globally, except for China, no food safety standards for inorganic As in foods have been found.
- A reliable and representative human health risk assessment for As in foods cannot be made at this stage.
- Data from countries other than Bangladesh on (inorganic) As in irrigation water, soil, crops and foods are even more limited.
- Data on As in livestock and freshwater fisheries are by far insufficient to make any statement on the risks of As to animal health and the safety of food products from these sectors.

Any risk to crop production is of serious concern as raising crop production is necessary to keep up with population growth. Also, raising crop production is regarded as one of the key elements in rural poverty alleviation. Land degradation caused by As-contamination could thus pose a threat to sustainable development.

With the continuation of uncontrolled use of contaminated water in agriculture, it is expected that the risks will increase over time. In the long run, this may offset the ongoing efforts in the drinking-water sector to reduce the adverse impacts of As.

Rice production systems are of particular concern because flooded soil conditions are most favourable for As uptake and rice is the most important staple food in the region. Other crops under irrigation with contaminated water need to be addressed as well as there are great uncertainties about the As mass balance under different environmental conditions.

It is important to note that once arable soils are polluted to an unacceptable level, rehabilitation is unlikely to be cost effective. Preference should thus be given to prevention and control over rehabilitation.

4.2 Recommendations

Considering the potentially serious consequences of As-contamination in agriculture, the identified gaps need to be filled, for example through an integrated regional programme covering both crop production and food safety aspects within the framework of land degradation.

Most importantly, the scale of the problem needs to be quantified. This should be based on scientifically justified methodologies resulting in reliable results, conclusions and recommendations. Close involvement of stakeholders of different sectors is necessary to optimize integrated and cross-sectoral programme coordination and implementation, including data sharing, human resources and funding, and to optimize dissemination and integration of the outcomes in strategic planning and programming, thus ensuring sustainability.

Crop production

Agricultural production is under pressure from multiple factors including As. In Bangladesh, it has recently been noted that rice production per ha is stagnating and declining. Deterioration of soil quality has been mentioned as one of the main factors, including depletion of nutrients and organic matter. Another increasing problem is access to water. Boro rice production depends heavily on irrigation with water from the shallow aquifer, but the water table is decreasing rapidly at many locations because of overextraction. The high salinity of irrigation water is also hampering crop production in the coastal areas. Similar problems are occurring elsewhere in the region. The continuous addition of As to the topsoil may pose additional risks to soil quality and crop production and therefore to nutrition and income.

The extent of land degradation caused by As in irrigation water needs to be quantified and weighted against other factors causing land degradation. This will require in-depth understanding of As levels in irrigation water, soil and crops, the behaviour of As in soil, uptake and toxicity in crops in the relevant agro-ecosystems, the influence of agricultural practices including irrigation water management on As, and the establishment of standards for As in irrigation water, soil and crops. This information would result in an evaluation of the current situation and would serve as a baseline for the future. Further, it will be of great importance to develop an effective and efficient water/soil/crop quality monitoring system for long-term monitoring of land degradation in agro-ecosystems. This system should include both As and other physical, chemical (nutrients and contaminants) and biological parameters that together determine the quality of agro-ecosystems.

The integrated approach to address the issue of using As-contaminated groundwater resources for irrigation offers substantial opportunities to assess the overall status of land degradation in agro-ecosystems, develop a system to monitor future trends in land degradation, and assist governments and development partners in developing policies and setting priorities.

Depending on the outcome of the initial evaluation, management options to prevent or mitigate As input to the agricultural system can be developed. Exploratory studies are needed to assess the potential risks of As in the environment to livestock and fisheries.

Food Safety

Arsenicosis is commonly observed in areas with high As levels in STWs used for drinking-water, with thousands of people suffering in Asia. It is generally accepted that drinking-water is the main source of exposure, but foods could be an important and possibly growing source as well. However, foods are not only a potential source of As, they are also the main source of micronutrients. Micronutrient deficiency is prevalent in Bangladesh, particularly among poor women and children. Improved nutrition is therefore high on the priority list in most Asian countries. Most well-known deficiencies are vitamin A and iron deficiencies. In general, nutrient deficiencies make people more sensitive to diseases and toxicants. There are indications that this is also the case for arsenicosis.

To determine the risk to human health posed by As in foods a reliable and representative database on inorganic As in foods is a necessary first step. This database can then be used with data on food consumption to estimate the dietary intake of As through foods and the resulting exposure levels can be compared to health guidelines to assess the risks. However, health guidelines may need to be evaluated first as there are some concerns about their reliability and applicability. Considering the high prevalence of nutrient deficiency in Asia, it will be of great importance to take the nutritional value of foods into consideration when evaluating the risks of As in foods.

Depending on the outcome of the human health risk assessment, options to minimize As exposure can be identified at field level, e.g. adapting dietary habits and/or food preparation techniques. Such measures can be combined with efforts to improve nutrient intake. An improved nutritional status is regarded as one of the keys to strengthening the human immune system, thus making people less vulnerable to many diseases.

Technical capacity

The technical capacity of many national laboratories involved needs to be strengthened to ensure the quality of the data. Within a laboratory, the quality of data depends on all activities that take place including pretreatment of samples, maintenance of facilities and equipment, training of staff, chemical analysis, administration, data processing, etc. All activities should therefore be described in standard operating procedures (SOPs) to ensure maximum standardization and transparency. This should result in a high level of QA/QC. QA/QC does add additional costs but unreliable data are more expensive. One specific requirement now is the use of CRMs during chemical analysis. This is not yet common practice in, e.g. Bangladesh, partially because of the high costs of CRMs. To overcome this, secondary reference materials (SRMs) representative of As in food and agriculture should be developed. As a start, SRMs for water, soil, rice grains and rice straw are suggested. This should be a mutual effort from a number of national research institutes and universities both from the health sector, the agricultural sector and the water sector, assisted by at least one established laboratory from abroad. Such an effort will not only upgrade chemical analysis, but can also strengthen inter-institutional planning and collaboration as a whole.

The behaviour and toxicity of As depends on the form in which it is present. In most cases, various forms of As co-exist in the environment. To understand the behaviour of As and assess its risks, it is therefore necessary to segregate As species during chemical analysis. The necessary techniques are, however, complicated and hardly available in the region. It is recommended that at least one institute in each of the affected countries develop As speciation capacity.

4.3 Stakeholders

A first inventory reveals a substantial number of stakeholders. This can be attributed to the cross-sectoral and multidisciplinary character of the issue. Below is a brief overview of the key stakeholders.

Governments

Governments are the key stakeholder. The issue requires involvement from various sectors including agriculture, water, food, health, and environment, and thus the responsible ministries. Considering the many gaps in our knowledge, a number of national research institutes under the relevant ministries need to play an important role, and collaborate with the national and international scientific community. With most of the countries in Asia considered to be developing countries and countries in transition, risks to food safety and agricultural sustainability should be seen in the light of governments' development agendas. Providing governments with reliable and representative information, for example through the national research institutes, should stimulate governments to take appropriate action. At the same time, commitment from governments is needed to provide an enabling environment in which the necessary work can take place.

UN agencies

Since 2000, FAO, UNICEF and WHO have been involved in the issue of As in food and agriculture. With the Millennium Development Goals as a target, the agencies are promoting sustainable agricultural production and improved access to safe and nutritious foods, particularly for the poor and vulnerable groups. Since 1999, FAO has been involved in the issue of As in irrigation water. The issue is now part of the Regular Programme and FAO experts regularly provide information, advice and inputs on the issue to governments, development partners and the scientific community. UNICEF is strongly involved in providing safe drinking-water in Asia, including mapping of groundwater quality. Although the aim is to provide safe drinking-water in terms of numerous parameters, As has received special attention. WHO is strongly involved in water quality issues and health aspects of arsenicosis, and together with UNICEF, provides guidance on the supply of sufficient and safe drinking-water. Both WHO and UNICEF have expressed concerns about possible risks of using As-contaminated irrigation water. Recently, WHO, FAO and UNICEF started the preparation of education, information and communication materials on As in drinking-water, and As in food and agriculture.

Donors

There is some concern about As-contaminated irrigation water among donors, as can be seen by the few projects funded by the Australian Agency for International Development (AusAID), the United States Agency for International Development (USAID) and the United Nations Development Programme (UNDP) in Bangladesh. However, the potential impacts of As on food safety and in particular on agricultural sustainability should be made clearer to the donor community through awareness raising activities so as to raise funds necessary to address the issue systematically and in an integrated manner. It is important for donors to realize that the continuation of As-contamination of agricultural land may in the long run offset the progress made in the drinking-water sector, which they are heavily supporting.

Universities & international research institutes

Universities from the region and elsewhere have been among the first to raise the issue and, mostly through their own initiatives and commitment, have developed and funded As-related projects. They will need to continue to play a key role as there are still many scientific gaps to be filled. A major challenge for the scientific community will be to improve the translation of the knowledge obtained through research into messages understandable by policy makers, donors, and the general public. Most of the countries in Asia facing As problems are classified as developing countries. It is therefore recommended to put the research outcomes into the perspective of the development agenda of governments and development partners. Part of the work to be done is challenging and will best be implemented through consortia of national and international research institutes. Consultative Group on International Agriculture Research (CGIAR) institutes such as the International Water Management Institute (IWMI), the International Rice Research Institute (IRRI) and the International Maize and Wheat Improvement Centre (CIMMYT) could play an important role in such consortia by providing leadership and/or contributing their technical expertise. Both IRRI and CIMMYT have already been involved in As-related activities in Bangladesh. IWMI has substantial expertise and experience in irrigation water quality issues in the region and elsewhere.

Beneficiaries

Farmers are facing multiple problems with regard to agricultural production. Strengthening their understanding and capacity to improve agricultural practices and minimize land degradation will enhance the sustainability of crop production and thus their livelihoods. This is of great importance to them and rural farming communities as a whole because agriculture is generally regarded as one of the keys to poverty alleviation in the region.

4.4 Lessons learned

Throughout the course of preparing this report, a number of shortcomings and suggestions for improving future activities have been identified and are briefly outlined below.

LESSONS LEARNED FOR FUTURE ACTIVITIES

- Strengthen research methodology and technical capacity
- Improve data interpretation
- Include risks to crop production
- Include long-term risks
- Use an integrated approach within the framework of land degradation
- Improve dissemination of information
- Improve funding and coordination mechanisms

Methodology – Limitations in study designs including a localized approach and a lack of detail do not allow extrapolation of results and conclusions. Detailed information on the conditions under which As accumulates in the soil is still not available. Chemical analyses often do not comply with basic quality control principles, reducing the reliability of the results and conclusions. For Bangladesh, the limited technical capacity currently present in the country is a major constraint in delivering scientifically justified information on which policy can be based.

Data interpretation – The large difference in toxicity of the various forms of As present in foods has mostly been ignored. To evaluate obtained data on As in soil and foods, reference is often made to standards from other countries without evaluating the quality and suitability of that specific standard. In many cases, the use of those standards cannot be justified.

Crop production – By far, projects on As in irrigation water have only focused on As in the food chain, neglecting the threat to crop production. The issue has not been put in the necessary broader framework of land degradation. Management aspects related to the use of contaminated groundwater for irrigation have only received limited attention.

Long-term effects – Most projects have only evaluated the current levels of As in water, soil and crops, and have not considered future levels. Contamination via irrigation is likely to continue as long as contaminated sources are being used for irrigation. The problems are thus expected to increase over time, regardless of whether current levels in soil and foods are acceptable or not. This highlights the need for a strategic approach and the incorporation of the issue in long-term monitoring activities on land degradation.

Funding and coordination – The number and scale of activities/projects related to addressing As in irrigation water have been limited and there has been a lack of coordination between agencies and between sectors. This situation can be explained partly by a lack of awareness raising initiatives, limited project outcomes and little dissemination of these, and the low priorities of governments and donors in terms of addressing As related issues. It can also be explained by the overall management of the As crisis, for example, in Bangladesh. As was first identified as a drinking-water and health problem, logically leading to the establishment of coordinating and funding mechanisms in the water and health sectors. However, this focus on water and health is now hampering activities designed to address As in the food and agriculture sectors and so far there has been little collaboration between the water and health sectors and the food and agricultural sectors and little interest from the major donors in tackling the problem.

References

Abedin M.J., Cotter-Howells J. & Meharg A.A. 2002a. Arsenic uptake and accumulation in rice (*Oriza sativa L.)* irrigated with contaminated water. *Plant Soil*, 240: 311-319.

Abedin M.J., Cressner M.S., Meharg A.A., Feldmann J. & Cotter-Howells J. 2002b. Arsenic accumulation and metabolism in rice (*Oryza sativa L.*). *Environ Sci Technol*, 36: 962-968.

Abedin M.J., Feldmann J. & Meharg A.A. 2002c. Uptake kinetics of Arsenic species in rice plants. *Plant Physiol*, 128: 1120-1128.

Abedin M.J. & Meharg A.A. 2002. Relative toxicity of arsenite and arsenate on germination and early seedling growth of rice (*Oryza sativa L.). Plant Soil*, 243: 57-66.

Ahmad S.A., Bandaranayake D., Khan A.W., Hadi S.A., Uddin G. & Halim M.A. 1997. Arsenic contamination in groundwater and arsenicosis in Bangladesh. *Int J Environ H Res*, 7: 271-276.

Ahmed K.M., Bhattacharya P., Hasan M.A., Akhter S.H., Alam S.M.M., Bhuyian M.A.H., Imam M.B., Khan A.A. & Sracek O. 2004. Arsenic enrichment in groundwater of the alluvial aquifers in Bangladesh: an overview. *Appl Geochem*, 19: 181-200.

Alam M.B. & Sattar M.A. 2000. Assessment of arsenic contamination in soils and waters in some areas of Bangladesh. *Water Sci Technol*, 42: 185-193.

Alam M.G.M., Allinson G., Stagnatti F., Tanaka A. & Westbrooke M. 2002. Arsenic contamination in Bangladesh groundwater: a major environmental and social disaster. *Int J Environ H Res*, 12: 236-253.

Alam M.G.M., Snow E.T. & Tanaka A. 2003. Arsenic and heavy metal contamination of vegetables grown in Samta village, Bangladesh. *Sci Tot Environ*, 308: 83-96.

Ali M.A. 2003. Fate of Arsenic in the environment. *In: Arsenic contamination: Bangladesh perspective*, Ed M F Ahmed. pp. 84-100. ITN-Bangladesh, Dhaka.

BADC 2004. *Survey report on irrigation equipment and irrigated area in Boro/2003 season*. BADC, Dhaka. 200 p.

Bae M., Watanabe C., Inaoka T., Sekiyama M., Sudo N., Bokul M.H. & Ohtsuka R. 2002. Arsenic in cooked rice in Bangladesh. *Lancet*, 360: 1839-1840.

Berg M., Tran H.C., Nguyen T.C., Schertenleib R. & Giger W. 2001. Arsenic contamination of groundwater and drinking water in Vietnam: a human health threat. *Environ Sci Technol*, 35: 2621-2626.

Bleeker P.M., Schat H., Vooijs R., Verkleij J.A.C. & Ernst W.H.O. 2003. Mechanisms of arsenate tolerance in *Cytisus striatus*. *New Phytol*, 157: 33-38.

Brammer H. 2005. The Arsenic Problem in Bangladesh. *Trop Agric Associat News*.

Carbonell A.A., Aarabi M.A., DeLaune R.D., Gambrell R.P. & Patrick Jr W.H. 1998. Arsenic in wetland vegetation: Availability, phytotoxicity, uptake and effects on plant growth and nutrition. *Sci Tot Environ*, 217: 189-199.

Carbonell-Barrachina A.A., Aarabi M.A., DeLaune R.D., Gambrell R.P. & Patrick Jr W.H. 1998. The influence of arsenic chemical form and concentration on Spartina putens and Spartina alterniflora growth and tissue arsenic concentration. *Plant Soil*, 198: 33-43.

Carbonell-Barrachina A.A., Burló F., Burgos-Hernández A., López E. & Mataix J. 1997. The influence of arsenite concentrations on arsenic accumulation in tomato and bean plants. *Sci Hortic*, 71: 167-176.

Cava-Montesinos P., De la Guardia A., Teutsch C., Cervera M.L. & De la Guardia M. 2003. Non-chromatographic speciation analysis of arsenic and antimony in milk hydride generation atomic fluorescence spectrometry. *Anal Chim Act*, 493: 195-203.

Chakraborti D., Rahman M.M., Paul K., Chowdhury U.K., Sengupta M.K., Lodh D., Chanda C.R., Saha K.C. & Mukherjee S.C. 2002. Arsenic calamity in the Indian subcontinent: What lessons have been learned? *Talanta*, 58: 3-22.

Chen Z., Zhu Y.G., Liu W.J. & Meharg A.A. 2005. Direct evidence showing the effect of root surface iron plaque on arsenite and arsenate uptake into rice (*Oryza sativa*). *New Phytol*, 165: 91-97.

Chowdhury U.K., Biswas B.K., Chowdhury T.R., Samanta G., Mandal B.K., Basu G.C., Chanda C.R., Lodh D., Saha K.C., Mukherjee S.K., Roy S., Kabir S., Quamruzzaman Q. & Chakraborti D. 2000. Groundwater Arsenic Contamination in Bangladesh and West Bengal, India. *Environ Health Persp*, 108: 393-397.

Codex A.C. 2004. *Procedural manual*. FAO/WHO, Rome.

Cornu S., Breeze D., Saada A. & Baranger P. 2003. The influence of pH, electrolyte type, and surface coating on arsenic(V) adsorption onto kaolinites. *Soil Sci Soc Am J*, 67: 1127-1132.

Das H.K., Mitra A.K., Sengupta P.K., Hossain A., Islam F. & Rabbani G.H. 2004. Arsenic concentrations in rice, vegetables, and fish in Bangladesh: a preliminary study. *Environ Int*, 30: 383-387.

Dasgupta T., Hossain S.A., Meharg A.A. & Price A.H. 2004. An arsenate tolerance gene on chromosome 6 of rice. *New Phytol*, 163: 45-49.

Dembitsky V.M. & Rezanka T. 2003. Natural occurrence of arseno compounds in plants, lichens, fungi, algal species, and microorganisms. *Plant Sci*, 165: 1177-1192.

Duxbury J.M., Mayer A.B., Lauren J.G. & Hassan N. 2003. Food chain aspects of arsenic contamination in Bangladesh: effects on quality and productivity of rice. *J Environ Sci Health*, A38: 61-69.

Duxbury J.M. & Zavala Y.J. 2005. What are safe levels of arsenic in food and soils? *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, International Symposium, Dhaka, 2005.

Farid A.T.M., Sen R., Haque M.A., Hossain K.M., Panaullah G.M., Meisner C.A., Loeppert R.H. & Duxbury J.M. 2005. Arsenic status of water, soil, rice grain and straw of individual shallow tube well command area of Brahmanbaria. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Fitz W.J. & Wenzel W.W. 2002. Arsenic transformations in the soil-rhizosphere-plant system: fundamentals and potential application to phytoremediation. *J Biotechnol*, 99: 259-278.

Gailer J., George G.N., Pickering I.J., Prince R.C., Ringwald S.C. 2000. A metabolic link between arsenite and selenite: the seleno-bis(S-glutathionyl) arsinium ion. *J Am Chem Soc*, 122: 4637-4639.

Ghosh A.K., Bhattacharyya P. & Pal R. 2003. Effect of arsenic contamination on microbial biomass and its activities in arsenic contaminated soils of Gangetic West Bengal, India. *Environ Int*, 30: 491-499.

Hartley-Whitaker J., Ainsworth G., Vooijs R., Ten Bookum W.M., Schat H. & Meharg A.A. 2001. Phytochelatins are involved in differential arsenate tolerance in *Holcus lanatus*. *Plant Physiol*, 126: 299-306.

Harvey C.F., Swartz C.H., Badruzzaman A.B.M., Keon-Blute N., Yu W., Ashraf Ali M., Jay J., Beckie R., Niedan V., Brabander D.J., Oates P.M., Ashfaque K.N., Islam S., H.F.H. & Ahmed M.F. 2002. Arsenic Mobility and Groundwater Extraction in Bangladesh. *Science*, 298: 1602-1606.

Heitkemper D.T., Vela N.P., Stewart K.R. & Westphal C.S. 2001. Determination of total and speciated arsenic in rice by ion chromatography and inductively coupled mass spectrometry. *J Anal At Spectrom*, 16: 299-306.

Hels O., Hassan N., Tetens I. & Thilsted S.H. 2003. Food consumption, energy and nutrient intake and nutritional status in rural Bangladesh: changes from 1981-82 to 1995-96. *Eur J Clin Nutr*, 57: 586-594.

Hossain M.B. 2005. Arsenic distribution in soil and water of a STW command area. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Huang Y.K., Lin K.H., Chen H.W., Chang C.C., Liu C.W., Yang M.H. & Hsueh Y.M. 2003. Arsenic species contents at aquaculture farm and in farmed mouthbreeder (Oreochromis mossambicus) in blackfoot disease hyperendemic areas. *Food Chem Toxicol*, 41: 1491-1500.

Islam F.S., Gault A.G., Boothman C., Polya D.A., Charnock J.M., Chatterjee D. & J.R. L. 2004a. Role of metal-reducing bacteria in arsenic release from Bengal delta sediment. *Nature*, 430: 68-71.

Islam M.R., Islam S., Jahiruddin M. & Islam M.A. 2004b. Effects of irrigation water arsenic in the rice croppping system. *J Bio Sci*, 4: 542-546.

Islam M.R., Jahiruddin M. & Islam S. submitted. Arsenic linkage in the irrigation water-soil-rice plant system. *Chemosphere*.

Islam M.R., Jahiruddin M., Rahman G.K.M.M., Miah M.A.M., Farid A.T.M., Panaullah G.M., Loeppert R.H., Duxbury J.M. & Meisner C.A. 2005. Arsenic in paddy soils of Bangladesh: levels, distribution and contribution of irrigation and sediments. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Islam M.R., Salminen R. & Lahermo P.W. 2000. Arsenic and other toxic elemental contamination of groundwater, surface water and soil in Bangladesh and its possible effects on human health. *Environ Geochem Health*, 22: 33-53.

Jahiruddin M., Islam M.A., Islam M.R. & Islam S. 2004. Effects of arsenic contamination on rice crop. *Environtropica*, 1: 204-210.

Jahiruddin M., Islam M.R., Shah M.A.L., Rashid M.A. & Ghani M.A. 2005. Arsenic in the water-soil-crop systems: PETRRA-BRRI-BAU-AAS study. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

JICA/AAN 2004. *Arsenic contamination of irrigation tubewells in Sharsha upazila, Jessore*. Asia Arsenic Network, Dhaka. 13 p.

Kohlmeyer U., Jantzen E., Kuballa J. & Jakubik S. 2003. Benefits of high resolution IC-ICP-MS for the routine analysis of inorganic and organic arsenic species in food products of marine and terrestrial origin. *Anal Bioanal Chem*, 377: 6-13.

Lambkin D.C. & Alloway B.J. 2003. Arsenate-induced phosphate relaese from soils and its effect on plant phosphorus. *Water Air Soil Pollut*, 144: 41-56.

Lamont W. 2003. Concentrations of inorganic arsenic in samples of white rice from the United States. *J Food Comp Anal*, 16: 687-695.

Lauren J.G. & Duxbury J.M. 2005. Management strategies to reduce arsenic uptake by rice. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Li W., Wei C., Zhang C., Van Hulle M., Cornelis R. & Zhang X. 2003. A survey of arsenic species in chinese seafood. *Food Chem Toxicol*, 41: 1103-1110.

Liu W.J., Zhu Y.G., Smith F.A. & Smith S.E. 2004. Do phosphorus nutrition and iron plaque alter arsenate (As) uptake by rice seedlings in hydroponic culture? *Do phosphorus nutrition and iron plaque alter arsenate (As) uptake by rice seedlings in hydroponic culture?*, 162: 481-488.

Loeppert R.H., White N., Biswas B.K. & Drees R. 2005. Mineralogy and arsenic bonding in Bangladesh rice paddy soils. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Mahimairaja S., Bolan N.S., Adriano D.C. & Robinson B. 2005. Arsenic contamination and its risk management in complex environmental settings. *Adv Agron*, 86: 1-82.

Mandal B.K., Chowdhury T.R., Samanta G., Mukherjee D.P., Chanda C.R., Saha K.C. & Chakraborti D. 1998. Impact of safe water for drinking and cooking on five arsenic-affected families for 2 years in West Bengal, India. *Si Tot Environ*, 218: 185-201.

Mandal B.K. & Suzuki K.T. 2002. Arsenic around the world: a review. *Talanta*, 58: 201-235.

Masscheleyn P.H., Delaune R.D. & Patrick W.H. 1991. Effect of redox potential and pH on arsenic speciation and solubility in a contaminated soil. *Environ Sci Technol*, 25: 1414-1419.

Matschullat J. 2000. Arsenic in the geosphere – a review. *Sci Tot Environ*, 249: 297-312.

McArthur J.M., Banjeree D.M., Hudson-Edwards K.A., Mishra R., Purohit R., Ravenscroft P., Cronin A., Howarth R.J., Chatterjee A., Talukder T., Lowry D., Houghton S. & Chadha D.K. 2004. Natural organic matter in sedimentary basins and its relation to arsenic in anoxic ground water: the example of West Bengal and its worldwide implications. *Appl Geochem*, 19: 1255-1293.

Meharg A.A. 2004. Arsenic in rice – understanding a new disaster for South-East Asia. *Tr Plant Sci*, 9: 415-417.

Meharg A.A. & Hartley-Whitaker J. 2002. Arsenic uptake and metabolism in arsenic resistant and nonresistant plant species. *New Phytol*, 154: 29-43.

Meharg A.A. & Jardine L. 2003. Arsenite transport into paddy rice (*Oryza sativa*) roots. *New Phytol*, 157: 39-44.

Meharg A.A. & Rahman M.M. 2003. Arsenic contamination of Bangladesh paddy field soils: implications for rice contribution to arsenic consumption. *Environ Sci Technol*, 37: 229-234.

Miah M.A.M., Rahman M.S., Islam A., Paul D.N.R., Farid A.T.M., Jahiruddin M., Sattar M.A., Panaullah G.M., Meisner C.A., Loeppert R.H. & Duxbury J.M. 2005. Nationwide survey of arsenic in soils, water and crops in Bangladesh. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Mukherjee A. & Bhattacharya P. 2001. Arsenic in groundwater in the Bengal Delta Plain: slow poisoning in Bangladesh. *Environ Rev*, 9: 189-220.

Ng J.C., Wang J. & Shraim A. 2003. A global health problem caused by arsenic from natural sources. *Chemosphere*, 52: 1353-1359.

Nicolas D.R., Ramamoorthy S., Palace V., Spring S., Moore J.N. & Rosenzweig R.F. 2003. Biogeochemical transformations of arsenic in circumneutral freshwater sediments. *Biodegradation*, 14: 123-137.

Norra S., Berner Z.A., Agarwala P., Wagner F., Chandrasekharam D. & Stuben D. 2005. Impact of irrigation with As rich groundwater on soil and crops: A geochemical case study in West Bengal Delta Plain, India. *Appl Geochem*, 20: 1890-1906.

Onken B.M. & Hossner L.R. 1995. Plant uptake and determination of arsenic species in soil solution under flooded conditions. *J Environ Qual*, 24: 373-381.

Panaullah G.M., Ahmed Z.U., Duxbury J.M., Loeppert R.H. & Meisner C.A. 2005. Arsenic in the irrigation water – soil environment and flows of arsenic in the food system. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Pizarro I., Gomez M., Camara C. & Palacios M.A. 2003a. Arsenic speciation in environmental and biological samples Extraction and stability studies. *Anal Chim Act*, 495: 85-98.

Pizarro I., Gomez M., Palacios M.A. & Camara C. 2003b. Evaluation of stability of arsenic species in rice. *Anal Bioanal Chem*, 376: 102-109.

Polya D.A., Gault A.G., Diebe N., Feldman P., Rosenboom J.W., Gilligan E., Fredericks D., Milton A.H., Sampson M., Rowland H.A.L., Lythgoe P.R., C. J.J., Middleton C. & Cooke D.A. 2005. Arsenic hazard in shallow Cambodian groundwaters. *Mineral Mag*, 69: 807-823.

Ravenscroft P., McArthur J.M. & Hoque B.A. 2001. Geochemical and palaeohydrological controls on pollution of groundwater by arsenic. *In: Arsenic exposure and health effects IV*, Eds W.R. Chappel, C.O. Abernathy and R. Calderon. pp. 53-78. Elsevier Science Ltd. Oxford.

Ross Z., Duxbury J.M., Paul D.N.R. & DeGloria S.D. 2005. Potential for arsenic contamination of rice in Bangladesh: spatial analysis and mapping of high risk areas. *In: Behavior of arsenic in aquifers, soils and plants (Conference Proceedings)*, Dhaka, 2005.

Roychowdhury T., Uchino T., Tokunaga H. & Ando M. 2002a. Arsenic and other heavy metals in soils from an arsenic-affected area of West Bengal, India. *Chemosphere*, 49: 605-618.

Roychowdhury T., Uchino T., Tokunaga H. & Ando M. 2002b. Survey of arsenic in food composites from an arsenic affected area of West Bengal, India. *Food Chem Toxicol*, 40: 1611-1621.

Schmidt A.C., Mattusch J., Reisser W. & Wennrich R. 2004. Uptake and accumulation behaviour of angiosperms irrigated with solutions of different arsenic species. *Chemosphere*, 56: 305-313.

Schoof R.A., Yost L.J., Crecelius E.A., Cragin D.W., Meacher D.M. & Menzel D.B. 1999. A market basket survey of inorganic arsenic in food. *Food Chem Toxicol*, 37: 839-846.

Shrestha R.R., Shrestha M.P., Upadhyay N.P., Pradhan R., Khadka R., Maskey A., Maharjan M., Tuladhar S., Dahal B.M. & Shrestha K. 2003. Groundwater arsenic contamination, its health impact and mitigation program in Nepal. *J Environ Sci Health*, A38: 185-200.

Smedley P.L. & Kinniburgh D.G. 2002. A review of the source, behaviour and distribution of arsenic in natural waters. *Appl Geochem*, 17: 517-568.

Smedley P.L., Zhang M., Zhang G. & Luo Z. 2003. Mobilisation of arsenic and other trace elements in fluviolacustrine aquifers of the Huhhot Basin, Inner Mongolia. *Appl Geochem*, 18: 1453-1477.

Sneller F.E.C., Noordover E.C.M., Ten Bookum W.M., Schat H., Bedaux J.J.M. & Verkleij J.A.C. 1999a. Quantitative relationship between phytochelatin accumulation and growth inhibition during prolonged exposure to cadmium in *Silene vulgaris*. *Ecotoxicology*, 8: 167-175.

Sneller F.E.C., Van Heerwaarden L.M., Kraaijeveld-Smit F.J.L., Ten Bookum W.M., Koevoets P.L.M., Schat H. & Verkleij J.A.C. 1999b. Toxicity of arsenate in *Silene vulgaris*, accumulation and degradation of arsenate-induced phytochelatins. *New Phytol*, 144: 223-232.

Sneller F.E.C., Van Heerwaarden L.M., Schat H. & Verkleij J.A.C. 2000. Toxicity, metal uptake, and accumulation of phytochelatins in *Silene vulgaris* exposed to mixtures of cadmium and arsenate. *Environ Toxicol Chem*, 19: 2982-2986.

Sun G. 2004. Arsenic contamination and arsenicosis in China. *Toxicol Appl Pharmacol*, 198: 268-271.

Takahashi Y., Minamikawa R., Hattori K.H., Kurishima K., Kihou N. & Yuita K. 2004. Arsenic behavior in paddy fields during the cycle of flooded and non-flooded periods. *Environ Sci Technol*, 38: 1038-1044.

Watanabe C., Kawata A., Sudo N., Sekiyama M., Inaoka T., Bae M. & Ohtsuka R. 2004. Water intake in an Asian population living in arsenic-contaminated area. *Toxicol Appl Pharmacol*, 198: 272-282.

Weiss J.V., Emerson D., Backer S.M. & Megonigal J.P. 2003. Enumeration of Fe(II)-oxidizing and Fe(III)-reducing bacteria in the root zone of wetland plants: implications for a rhizosphere iron cycle. *Biogeochemistry*, 64: 77-96.

Weiss J.V., Emerson D. & Megonigal J.P. 2004. Geochemical control of microbial Fe(III) reduction potential in wetlands: comparison of the rhizosphere to non-rhizospere soil. *FEMS Microbiol Ecol*, 48: 89-100.

WHO 2001. *Environmental Health Criteria 224: Arsenic and Arsenic compounds*. WHO, Geneva.

WHO 1996. *Trace elements in human nutrition and health*. WHO, Geneva.

Williams L.E., Barnett M.O., Kramer T.A. & Melville J.G. 2003. Adsorption and transport of Arsenic(V) in experimental subsurface systems. *J Environ Qual*, 32: 841-850.

Williams P.N., Islam M.R., Adomako E.E., Raab A., Hossain S.A., Zhu Y.G. & Meharg A.A. 2006. Increase in rice grain arsenic for regions of Bangladesh irrigating paddies with elevated arsenc in groundwater. *Environ Sci Technol*, 40: 4903-4908.

Williams P.N., Price A.H., Raab A., Hossain S.A., Feldmann J. & Meharg A.A. 2005. Variation in arsenic speciation and concentration in paddy rice related to dietary exposure. *Environ Sci Technol*, 39: 5531-5540.

Xia Y. & Liu J. 2004. An overview on chronic arsenism via drinking water in PR China. *Toxicology*, 198: 25-29.

Yamazaki C., Ishiga H., Ahmed F., Itoh K., Suyama K. & Yamamoto H. 2003. Vertical distribution of arsenic in ganges delta sediments in Deuli Village, Bangladesh. *Soil Sci Pant Ntr*, 49: 567-574.

Zheng Y., Stute M., van Geen A., Gavrieli I., Dhar R., Simpson H.J., Schlosser P. & Ahmed K.M. 2004. Redox control of arsenic mobilization in Bangladesh groundwater. *Appl Geochem*, 19: 201-214.

Zhu Y.G. & Meharg A.A. 2006. 2006. Arsenic in rice: potential health problem and solution. *In: FAO-Chinese Academy of Sciences joint workshop on arsenic in water, soil and crops*, Beijing, 2006. Ed Y.G. Zhu.

Zobrist J., Dowdle P.R., Davis J.A. & Oremland R.S. 2000. Mobilization of arsenite by dissimilatory reduction of adsorbed arsenate. *Mobilization of arsenite by dissimilatory reduction of adsorbed arsenate*, 34: 4747-4753.

Appendix I

Acknowledged are the valuable contributions of the following resources persons and many others that made this report possible:

Prof. A.A. Meharg	University of Aberdeen, United Kingdom
Dr P.N. Williams	University of Aberdeen, United Kingdom
Dr H. Schat	Vrije Universiteit Amsterdam, the Netherlands
Prof. Y.G. Zhu	Chinese Academy of Sciences
Prof. W. Seinen	University of Utrecht, the Netherlands
Prof. R. Naidu	CERAR, Australia
Prof. R. Loeppert	Texas University, United States of America
Prof. J. Duxbury	Cornell University, United States of America
Dr R. Ritsema	RIVM, the Netherlands
Dr G. Howard	DFID
Mr T. Facon	FAO
Mr Z. Chen	FAO
Ms S. Koo-Oshima	FAO
Dr Y. Niino	FAO
Mr R. Johnston	UNICEF
Mr R. Nickson	UNICEF
Ms A. Currie	UNICEF
Mr H. Heijnen	WHO
Dr R. Rooney	WHO
Dr R. Boerschke	WHO
Dr C. Meisner	CIMMYT
Dr G.M. Panaullah	CIMMYT
Dr A. Noble	IWMI
Dr R. Simmons	IWMI
Dr M.A.M. Miah	BRRI
Dr A.T.M. Farid	BARI
Prof. Sk. Akhtar Ahmad	NIPSOM
Dr M.H. Salim Ullah Sayed	NIPSOM
Prof. R.A. Islam	BAU
Dr A. Ali	BUET
Prof. M.F. Ahmed	BUET
Prof. M.M. Rahman	BUET
Prof. S.M.I. Huq	Dhaka University
Dr S.A. Hossain	Dhaka University
Ms D. Panagides	Helen Keller International
Ms G. Stallkamp	Helen Keller International
Ms N. Akther	Helen Keller International