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Inorganic arsenic in Chinese food and its cancer risk

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ABSTRACT

Even moderate arsenic exposure may lead to health problems, and thus quantifying inorganic arsenic (iAs) exposure from food for different population groups in China is essential. By analyzing the data from the China National Nutrition and Health Survey (CNNHS) and collecting reported values of iAs in major food groups, we developed a framework of calculating average iAs daily intake for different regions of China. Based on this framework, cancer risks from iAs in food was deterministically and probabilistically quantified. The article presents estimates for health risk due to the ingestion of food products contaminated with arsenic. Both per individual and for total population estimates were obtained. For the total population, daily iAs intake is around 42 μg day^{-1}, and rice is the largest contributor of total iAs intake accounting for about 60%. Incremental lifetime cancer risk from food iAs intake is 106 per 100,000 for adult individuals and the median population cancer risk is 177 per 100,000 varying between regions. Population in the Southern region has a higher cancer risk than that in the Northern region and the total population. Sensitive analysis indicated that cancer slope factor, ingestion rates of rice, aquatic products and iAs concentration in rice were the most relevant variables in the model, as indicated by their higher contribution to variance of the incremental lifetime cancer risk. We conclude that rice may be the largest contributor of iAs through food route for the Chinese people. The population from the South has greater cancer risk than that from the North and the whole population.

1. Introduction

Arsenic (As) is one of the first chemicals designated as a group I carcinogen (IARC, 1973), with its severe impacts on human health due to chronic exposure having been widely acknowledged for many years. The epidemiological studies of As disease have arisen in the main from incidences of potable water contamination, which have been linked not only to increases in skin, bladder and lung cancers but also to developmental, cardiovascular and metabolic disorders (Abernathy et al., 1999; Chen et al., 2009; Lubin et al., 2007; NRC, 2001; Smith and Steinmaus, 2009). For example, a study conducted in Inner Mongolia (Northwest China) found that cancer mortality and all-cause mortalities were associated with well-water As exposure, subject to candidates having used the As elevated water sources for over 10 years (Wade et al., 2009). Similarly, in Bangladesh chronic As exposure via tubewell water is also associated with an increase in all-cause and chronic disease mortality rate (Argos et al., 2010). Whereas a strong dose–response relationship between As in tubewell water and skin lesion development, a precursor of skin cancer, has been observed, with even low exposure to As was shown to be problematic (Ahsan et al., 2006). Low level As intake may also play a role in diabetes prevalence, based on findings from a cross-sectional study of 788 adults in the USA (Navas-Acien et al., 2008), in addition to impairing H1N1 infection immune system in responses (Kozul et al., 2009). These research findings amongst others have prompted critical re-evaluation of exposure thresholds for As (WHO/FAO, 2010).

The principal exposures routes for As are via drinking water, foods and inhaled particulates (Mondal et al., 2010), with many studies detailing the importance of the food pathway to overall As body burdens (Georgopoulos et al., 2007; Meacher et al., 2002; Mondal et al., 2010; Schoof et al., 1999; Xue et al., 2010). In West Bengal (India) it has been shown that even for populations exposed to high As levels in the drinking water, rice constitutes a major source of iAs in the diet (Mondal and Polya, 2008). Yet, rice is the dominant exposure route in all scenarios with seafood also being of importance, especially in those diets without a preference for the grain (Baeyens et al., 2009; Carbonell-Barrachina et al., 2009; ESFA, 2009). Indeed understanding iAs exposure

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from rice relies upon the interplay from rice consumption rates, variability in grain As concentrations and the proportion of iAs in the grain (Meharg et al., 2009; Mondal and Polya, 2008).

Rice has been shown to be particularly susceptible to As uptake in comparison with other cereal crops (Williams et al., 2007), reflecting the As levels in the environments in which it is grown. China annually discharges around 195 t of arsenic into the atmosphere from coal power stations alone (Luo et al., 2004). Other prolific, industrial emission sources include mining, non-ferrous metal ores processing and chemical manufacturing (Li et al., 2006a). Far from abating, emission trends will likely be sustained if not increase further in the next few years given China's rapid economic expansion, thereby continuing to threaten food supply chains. We have reported a nationwide survey of As in rice, and found that over 95% of market rice samples contained less than the national food safety standard of 0.15 iAs mg/kg (China Food Standard Agency, 2005); but for rice collected from mining-impacted sites in Hunan and Guangdong provinces a large percentage of the samples failed the national food safety standard (Williams et al., 2009; Zhu et al., 2008a) and this needs to be also considered.

Estimates of dietary iAs exposure for the Chinese population have been attempted, for example Li et al. (2006b) reported that the iAs intake from food by adult males was 1.26 μg/kg body weight, however, rice was not separated from other grains in this analysis. More recently, Meharg et al. (2009) estimated global iAs intakes from rice, calculating the associated excess cancer risk for the Chinese population due to rice consumption to be 152 per 100,000. This was derived from a sample base of 124 rice samples and modeled using average rice consumption values for China and United States Environmental Protection Agency (USEPA) excess internal cancer risk slopes. However, given the wide ranging variability in dietary habits of sub-populations within China, coupled with the increasing use of rice based ingredients being used in other products such as health-foods, non-dairy alternatives to milk and snack foods (ESFA, 2009; Meharg et al., 2008; Sun et al., 2009), the range of iAs exposures via all food groups and therefore associated cancer risks could be considerable and is currently unclear.

The present study therefore aims to address this issue, by compiling a database of iAs in Chinese food, to quantify at the national and regional level trends in average iAs intake. Furthermore, we propose to model increasing lifetime cancer risks for individuals and the total population due to the ingestion of iAs via food.

2. Materials and methods

2.1. Food consumption data

Trends in regional and national dietary habits were taken from the 2002 China National Nutrition and Health Survey (CNNHS) (Jin, 2008). This database is China's principal nutrition reference source, containing the dietary patterns of 68,962 individuals from 31 provinces (excluding Hong Kong, Taiwan and Macau). Based on consumption frequency in order to reflect the most commonly consumed products, the food types are categorized into one of 10 groupings: rice, flour, coarse cereal, pulses, vegetables, fruit, meat, milk, eggs, and aquatic products. Categorization of sub-population followed three themes: i) urban vs. rural ii) geographic segregation, i.e. Northern vs. Southern China iii) proximity to the sea, i.e. coastal vs. inland (Further details are provided in the Supplementary data, Table S1). In brief, standard economic development criteria like like gross industrial product, gross domestic product and local financial revenues were used to differentiate, urban and rural areas (Liu et al., 2003). Classification of North and South China was achieved using a geographical divide running from the Huai river to the Qinling mountains (Han et al., 2009), while the coastal and inland regions were categorized by the location of the province. The food consumption rates of the whole nation and the urban and rural categories were directly obtained from the CNNHS survey report (Jin, 2008); those of the other two regions (north and south, coastal and inland) were calculated by weighting the population proportion of the involved provinces (Tables S1–S3). For example, the food consumption of the north region was obtained by the following formula: food consumption rate $= \sum R \times P$ R: food consumption rate of the province in north region; P: the proportion of the province in the north region.

2.2. iAs concentrations in different types of food

The database of iAs concentrations in different food types was compiled from the published literature. As shown in Table S4 as Supplementary data, a total of 13,684 data points for iAs concentrations were collected. Due to the lack of original data, statistics for iAs concentration for food types other than rice was used for arsenic exposure.

Data for total arsenic concentrations in rice from our dataset (494 samples, Table S4, Supplementary data) was converted to inorganic arsenic concentration by using the regression equations reported in the Supplementary data, Fig. S1 in accordance with the method of Meharg et al. (2009). Because a portion of the data featuring total and inorganic arsenic concentrations in rice was obtained from the literature and based on dry weight, while rice consumption rate is on fresh weight, we converted the concentrations based on dry weight into equivalent fresh weight concentration, with the assumption that water content of rice grain is 10% (Williams et al., 2005).

The iAs concentrations in vegetables, fruit, meat, milk, eggs and aquatic products were based on wet weight. Milk powder was converted into fresh milk during the calculation. Meanwhile, for the variation of iAs in meat and aquatic products, the mean iAs concentrations in them were calculated by weighting their supply from FAO statistical databases (FAOSTAT, 2005).

2.3. Calculation of estimated daily intake of iAs (EDI)

To determine the iAs exposure through food, we calculated the estimated daily intake of iAs by multiplying daily food consumption rate with corresponding iAs residues according to the following equation:

$$EDI_{iAs} = \sum (C_i \times IR_i{}^i)$$

Where EDI is estimated daily intake of iAs (μg/day); C_i is inorganic As concentration in subscripted food (mg/kg); i refers to different types of food (rice, flour, coarse cereal, pulses, vegetables, fruit, meat, milk, eggs, and aquatic products); IR_i is the ingestion rate for the subscripted food (g/day), which is the amount of food item consumed per day (Jin, 2008).

We calculated the daily iAs intake with the assumption that the contribution to iAs of some food types like oil, salt, sugar and pastry was neglected, and the main food types were involved, which accounted for about 85% of the total intake amount (Jin, 2008).

Table 1
Variables used in the deterministic risk model.

Variables	Parameter characteristics
ED (year)	Constant = 70
EF (day)	Constant = 365
C (mg/kg)	From the published literature listed in Supplementary data, Table S4
IR (g/day)	From Table 3 obtained from Jin (2008)
LT (year)	Constant = 25,550
BW (kg)	Constant = 60
CSF (per mg/kg/d)	Constant = 1.50 (ATSDR, 2010; USEPA, 2010)