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## Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants

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Title: Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants

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Abstract: This review is on arsenic in agronomic systems, and covers processes that influence the entry of arsenic into the human food supply. The scope is from sources of arsenic (natural and anthropogenic) in soils, biogeochemical and rhizosphere processes that control arsenic speciation and availability, through to mechanisms of uptake by crop plants and potential mitigation strategies. This review makes a case for taking steps to prevent or limit crop uptake of arsenic, wherever possible, and to work toward a long-term solution to the presence of arsenic in agronomic systems. The past two decades have seen important advances in our understanding of how biogeochemical and physiological processes influence human exposure to soil arsenic, and this must now prompt an informed reconsideration and unification of regulations to protect the quality of agricultural and residential soils

Response to Reviewers: Response to Reviewers

We thank both reviewers for taking the time to give our paper a rigorous and thoughtful peer review, and for ultimately improving the quality of the writing and the information. We include our responses to their comments below.

Reviewer 1 Comment 1 (R1C1): Page 3 L15-18: This sentence is a bit misleading. Importantly, As uptake by rice readily occurs in non-contaminated soils because naturally present As is liberated and plant available under reducing soil conditions. The sentence as written suggests that As uptake into rice is due to contamination.

Response to R1C1: We agree; this is an important point. To make it clearer that arsenic contamination of soil is not necessarily a pre-requisite for its entry into food crops, we have added after L15-18: "The presence of elevated concentrations of arsenic in the soil is not a pre-requisite in dietary arsenic exposure as seen in the effective accumulation of arsenic by rice grown in uncontaminated soils."

R1C2: In addition, I would argue that "in many cases" the route of As exposure via food is not more significant than drinking water. In the US, As ingestion via food is a primary route of exposure because most drinking water in the US has low As. However, in other more populated regions (e.g. S and SE Asia) drinking water is the primary route of exposure and ingestion via food is secondary.

Response to R1C2: We agree: this generalization is a US-centric view. In place of "In many cases", we have specified that "When drinking-water arsenic concentrations are low, dietary arsenic can be a significant exposure route."

R1C3: Page 4 L6 and L8 and elsewhere: Is the use of in text URLs suitable for STODEN? Please check. I suggest to cite them in the references instead.

Response to R1C3: Agreed. We have replaced full URLs with citations and moved them to the bibliography.

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Response to R1C5: We have changed this statement to read, "Below toxic concentrations, the higher the total soil arsenic concentration (the sum of all arsenic species, regardless of bioavailability) the higher the crop uptake of arsenic, including..." and have added the Panaulla and Syu references suggested by Reviewer 1.

R1C6: Page 6 L13: Suggest to change to "most soils range < 7.5 - 20 mg/kg arsenic" for clarity

Response to R1C6: Done.

R1C7: Page 11 L11 and elsewhere: Are edible plant tissue concentrations reported in mg/kg on a fresh weight or dry weight basis? This should be clearly stated to avoid confusion. I assume dry weight, but many of the market basket surveys (especially the FDA total diet study) report on fresh weight. Plants are generally 90% water, so this detail is important when comparing concentrations.

Response to R1C7: Done. This has been stated clearly at first mention.

R1C8: Page 12 L3: There is a hyperlink for "130 million tons" which should instead be removed and referenced.

Response to R1C8: Done.

R1C9: Page 12 L7: change "since" to "because"

Response to R1C9: Done

R1C10: Page 13 L13: Please define DMA and MMA. Not all STODEN readers will know what those compounds are.

Response to R1C10: Done

R1C11: Page 14 L8-15: It should be considered that the microbial communities also differ, and that microbes are known to methylate inorganic As as a detoxification mechanism. DMA and inorganic As differ in their in planta mobility, which is likely a contributing factor in enhanced total As (as DMA) in US vs Bangladeshi rice.

Response to R1C11: To our knowledge there are no comparative studies on the soil microbial communities between (for instance) the USA and Bangladesh that would support this statement, although it is a well-informed hypothesis. We are wary - particularly on what seems to have been a contentious issue - of hypothesizing about why USA rice contains more arsenic, or why the arsenic speciation differs. We have stated however, in response to this comment, what factors may be involved and what the uncertainties are, which will hopefully embrace the comment of Reviewer 1. We have also referenced the statement that plants themselves cannot methylate arsenic. This paragraph now reads:  
"Former pesticide application has been suggested be a factor in the presence of higher levels of total arsenic found in rice grown in the south-central regions of the USA<sup>65</sup>, <sup>66</sup> compared to other areas of the USA and to other countries, such as Bangladesh<sup>67</sup>. Evidence on varietal differences in arsenic uptake, speciation and distribution within rice grain (See also Section 6) strongly suggest that soil arsenic concentration is not the sole, nor particularly the main driver of this phenomenon. Other factors likely to be influential include the differences in the soil microbial community composition between geographical regions that affect arsenic methylation, which - considering that the ability to methylate arsenic has not been found in plants - may be another driving factor."

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Response to R1C12: Done.

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Response to R1C13: Done.

R1C14: Page 19 L5: What is the reference for the "animal waste products" part of the sentence?

Response to R1C14: Citation has been added.

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Response to R1C18: Thank you. This statement has been deleted, and the statement "In rice, Low Silicon 1 (OsLsi1) and OsLsi2 are silicic acid transporters and arsenite, MMAV, and DMAV are among their unintended targets<sup>148, 149</sup>." has been moved up and the paragraph edited accordingly.

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Response to R1C19: This statement now reads, "Despite having a lower affinity for transporters into the plant than inorganic forms, organic arsenic species are more efficiently transported toward the shoot than inorganic forms."

R1C20: Also, there was no earlier mention of the relative differences between inorganic and organic As for uptake. Consider revising.

Response to R1C20: Statements about the relative differences in uptake of arsenic species are included in two statements in Section 6.2: "In magnitude, plants take up arsenicals from the soil in the order arsenite > arsenate > DMA > MMA" and "The arsenic uptake specificity of OsLsi1 is arsenite >> MMA > DMA".

R1C21: Page 31 L7-19: It should also be mentioned that increased exogenous Si can down-regulated the Lsi1 transporter, which further decreases the potential for As uptake.

Response to R1C21: Added. Thank you.

R1C22: Page 31 L18-19. Suggest to change to "...practice for smallholder farmers), can provide silicon without increasing methane production and decreases either total or inorganic As in grain."

Response to R1C22: Added.

Reviewer 2:

Reviewer 2 Comment 1 (R2C1): As the authors discuss the regulated limits of arsenic in rice, I would have thought a section relating these exposures to arsenic to health impacts was needed. Some indication from the scientific literature on the impacts that arsenic rich water and food has on human health is needed, especially in relation to the concentrations given as the regulated limits.

Response to R2C1: This is a great comment. The reviewer refers to exposure epidemiological studies of arsenic, which is a huge area of science that is outside the scope of this synthesis paper. This topic is the subject of another excellent article submitted to this special issue. We have added a statement referring the reader to that article at the end of the first paragraph of the introduction:

"An in-depth review of the current findings on the relationship between dietary arsenic exposure and human health is provided by Davis et al. (this issue)..."

R2C2: Page 4 line 2. "but no safe level of arsenic has been found" what do the authors mean by this statement? The next sentence discusses impacts of arsenic in compared to MCLs, but does not go as far as saying that any concentrations are harmful.

Response to R2C2: We agree that the statement "no safe level of arsenic has been found", suggests that any arsenic level causes actual harm, when in fact observations are that any level causes an effect on biological model systems, such as cell lines. We recognize that this is an important distinction. This statement has been deleted. We have clarified the message here and the statement now reads:

"but given that measurable biological effects occur in at levels below the current maximum contaminant level (MCL) for arsenic in drinking water<sup>2</sup>, these low levels can still translate into significant exposures, particularly in children and presumably in adults who consume a lot of rice."

R2C3: Page 4 line 18. "the majority of other nations", some clarification needed on this statement as the statement above this indicates that the 10ug/l is supported by WHO, EU and Canada. Does that mean that all other nations except those in the EU, Canada and the couple of other mentioned nations have a limit of 50 ug/l?

Response to R2C3: We have named the countries that still adopt a standard of 50 µg/L, and included a citation.

R2C4: Page 5 line 2. Reference needed for this statement.

Response to R2C4: Done.

R2C5: Page 6 line 21. "USGS" I think this is the first use of this abbreviation. Please spell out in full.

Response to R2C5: Done.

R2C6: Page 9 line 20. A reference is needed for this statement.

Response to R2C6: Done.

R2C7: Page 12 line 3. Formatting of "130 million tons"

Response to R2C7: Done.

R2C8: Page 18 line 8. Can the authors expand on the bind of arsenic to soils particles.

Response to R2C8: We have added the following text to expand on this statement:

"Arsenite is less stably bound to aluminum hydroxides and aluminosilicate clay minerals in the soil than arsenate, for which they exhibit a much stronger binding preference<sup>30</sup>.

R2C9: Page 25. When discussing toxicity it would be good to indicate what this measurement actually is, is it root assays in hydroponics or soil based systems. As hydroponics is more artificial than the natural soil systems.

Response to R2C9: In our opinion, the discussion about arsenic toxicity in plants is clearly concerned with field studies where reduced yield has been a result. We have clarified this statement, and have ensured that comparative toxicity tests from hydroponic-based assays are not cited.

R2C10: Page 26 line 12. The references related to phosphate transporters, are these the transporters known to be involved in arsenic transport or those involved in phosphate transport and possibly arsenate? It is unclear from the way that it is written.

Response to R2C10: It is our opinion that the statement about arsenate being transported via phosphate transporters is very clear here. We state "Arsenate enters root cells through phosphate transporters" and then we have referenced experimental evidence for this in two model plant species.

R2C11: Page 31, section 7.2. This section seems on the small side given all the potential mechanisms that could mitigate arsenic accumulation by the alteration of arsenic chemistry in the soil. This could be expanded further. Additionally, this is very rice focused (actually all of section 7 is). While I understand that rice is the main target for reducing arsenic due to it being a dietary staple and having very high concentrations, you started the review by discussing not just rice but products like apple juice and vegetables and fruits. I think the mitigation strategies to be balanced need to also focus on non-rice plants to.

Response to R2C11: We agree that this section is short and rice-focused. After re-reviewing the current literature, we added missing information on the use of iron-based amendments, and have expanded the section. We have restricted our discussion to agricultural settings, and to mesocosms or field-scale studies to highlight technologies closer to deployment, for the article's target audience of stakeholders and regulators. Although we are aware of other amendment types (biochar, kaolin and various other organic matter additions) field scale testing of these materials has been minimal. Another issue - which we have added to the discussion - is that development of amendments proceeds without input from or consideration for the smallholder farmers or commercial entities that would use them, and almost always remain prohibitively priced, or without long term supporting data. We have made reference to this in our conclusions.

In addition, the literature is heavily rice-focused, because these agronomic approaches have been developed in response to the Codex Commission's adoption of a maximum level of 0.2 mg/kg inorganic arsenic in polished rice, and because it is primarily in anaerobic flooded systems that arsenic uptake by plants is an issue. We have tried to include studies on other cropping systems where available, but these are inevitably fewer than the rice-based studies.

R2C12: You discuss how water management, fertilisers and the plant can be altered for growing on arsenic soils, I think this could be expanded into possible remediation strategies for the soil as well. A lot of the information that you present in the opening sections is about the addition to arsenic to land / soil, a section on mitigation of this and then dealing with high arsenic soils would be good.

Response to R2C12: This is an interesting topic, but we feel an additional section on phytoremediation of highly arsenic contaminated soils is well beyond the scope of this article. Our focus is on the route arsenic takes into the human food supply, from soils not necessarily classified as highly arsenic contaminated, and how this can be managed to reduce dietary arsenic exposure. A section on mitigation of arsenic input into agricultural soil is also not warranted, since much of the arsenic contamination relevant to food production is either natural, legacy (former pesticide application), or the result of efficient plant-based arsenic accumulation mechanisms. We have dealt with some of these aspects in our conclusion, where we state that information on arsenic levels in soils be made more available so the appropriate action can be taken to avoid human exposure.





August 19 2016

To Whom It May Concern;

This letter accompanies the submission of our research article, “**Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants**”. This is an original work. It has never been previously published, in whole or in part, nor is it currently under consideration for publication elsewhere. We declare that there are no actual or potential competing interests, and that all authors have read and approved the manuscript and that they accept responsibility for its contents and agree it is ready for submission.

This paper is part of a series of reviews that resulted from the Collaborative on Food Arsenic and Associated Risk and Regulation, which took place at Dartmouth College, on November 2 2015. The workshop was attended by national and international scientists who lead the field on the study of arsenic in the environment and its affects on our health. The aim of the C-FARR workshop were to identify and synthesize information necessary to advance regulation of arsenic as a food supply contaminant.

This particular review is on arsenic in agronomic systems, and covers processes that influence the entry of arsenic into the human food supply. The scope is from sources of arsenic (natural and anthropogenic) in soils, biogeochemical and rhizosphere processes that control arsenic speciation and availability, through to mechanisms of uptake by crop plants and potential mitigation strategies. This review makes a case for taking steps to prevent or limit crop uptake of arsenic, and to work toward a long-term solution to the presence of arsenic in agronomic systems. The past two decades have seen important advances in our understanding of how sources of arsenic to the soil are influenced by biogeochemistry and result in human exposure, which must now prompt an informed reconsideration and unification of regulations that protect the quality of agricultural and residential soils.

Please note that the review is written to be accessible to a wide range of audiences, from the expert to the informed layperson.

I look forward to a constructive review of our work at Science of the Total Environment

Sincerely,

A handwritten signature in dark ink, appearing to be 'J. M. ...', followed by a long horizontal flourish.

## Response to Reviewers

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## **Reviewer 2:**

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Response to R2C3: We have named the countries that still adopt a standard of 50 µg/L, and included a citation.

R2C4: Page 5 line 2. Reference needed for this statement.

Response to R2C4: Done.

R2C5: Page 6 line 21. "USGS" I think this is the first use of this abbreviation. Please spell out in full.

Response to R2C5: Done.

R2C6: Page 9 line 20. A reference is needed for this statement.

Response to R2C6: Done.

R2C7: Page 12 line 3. Formatting of "130 million tons"

Response to R2C7: Done.

R2C8: Page 18 line 8. Can the authors expand on the bind of arsenic to soils particles.



Response to R2C8: We have added the following text to expand on this statement:  
"Arsenite is less stably bound to aluminum hydroxides and aluminosilicate clay minerals in the soil than arsenate, for which they exhibit a much stronger binding preference<sup>30</sup>.

R2C9: Page 25. When discussing toxicity it would be good to indicate what this measurement actually is, is it root assays in hydroponics or soil based systems. As hydroponics is more artificial than the natural soil systems.

Response to R2C9: In our opinion, the discussion about arsenic toxicity in plants is clearly concerned with field studies where reduced yield has been a result. We have clarified this statement, and have ensured that comparative toxicity tests from hydroponic-based assays are not cited.

R2C10: Page 26 line 12. The references related to phosphate transporters, are these the transporters known to be involved in arsenic transport or those involved in phosphate transport and possibly arsenate? It is unclear from the way that it is written.

Response to R2C10: It is our opinion that the statement about arsenate being transported via phosphate transporters is very clear here. We state "Arsenate enters root cells through phosphate transporters" and then we have referenced experimental evidence for this in two model plant species.

R2C11: Page 31, section 7.2. This section seems on the small side given all the potential mechanisms that could mitigate arsenic accumulation by the alteration of arsenic chemistry in the soil. This could be expanded further. Additionally, this is very rice focused (actually all of section 7 is). While I understand that rice is the main target for reducing arsenic due to it being a dietary staple and having very high concentrations, you started the review by discussing not just rice but products like apple juice and vegetables and fruits. I think the mitigation strategies to be balanced need to also focus on non-rice plants to.

Response to R2C11: We agree that this section is short and rice-focused. After re-reviewing the current literature, we added missing information on the use of iron-based amendments, and have expanded the section.

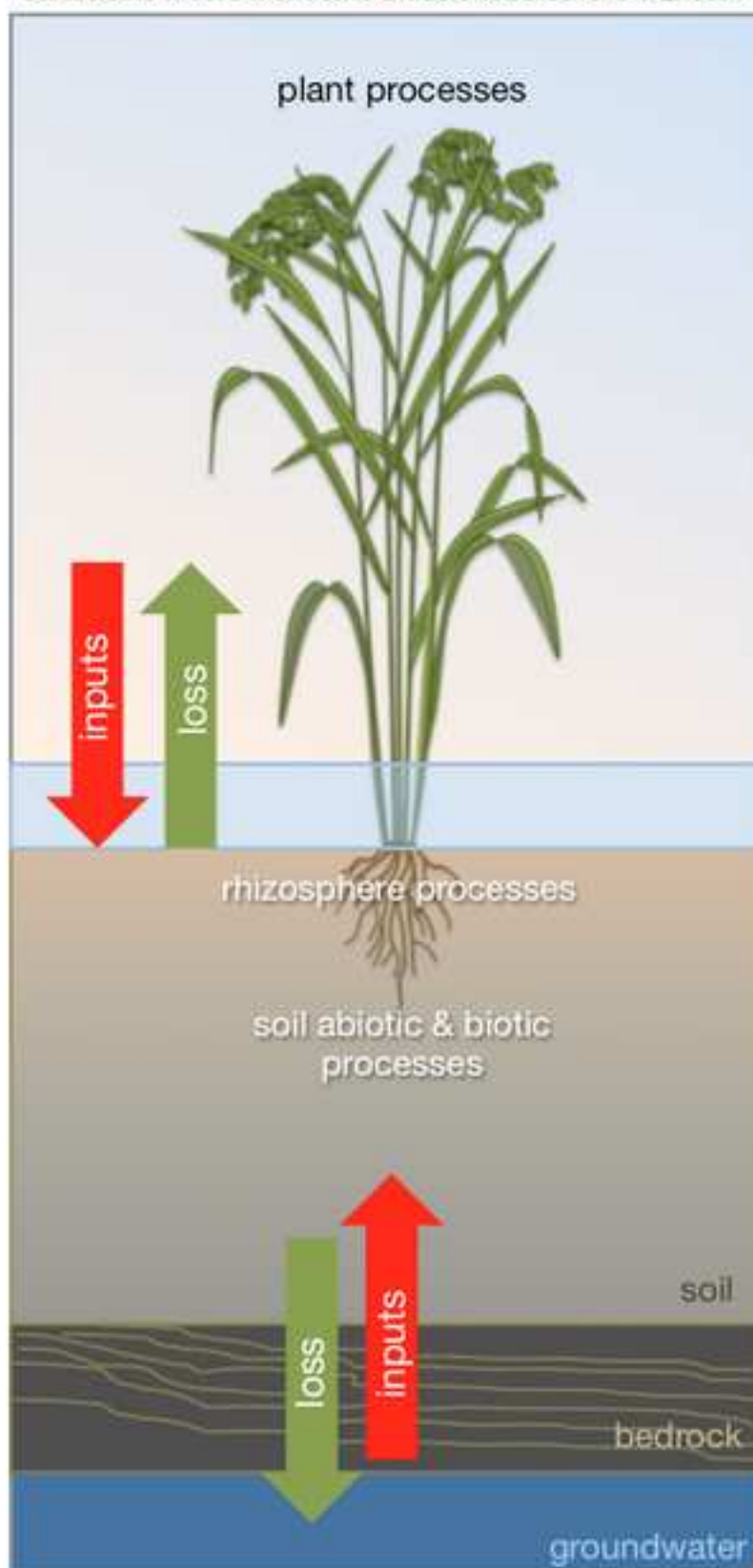
We have restricted our discussion to agricultural settings, and to mesocosms or field-scale studies to highlight technologies closer to deployment, for the article's target audience of stakeholders and regulators. Although we are aware of other amendment types (biochar, kaolin and various other organic matter additions) field scale testing of these materials has been minimal. Another issue – which we have added to the discussion – is that development of amendments proceeds without input from or consideration for the smallholder farmers or commercial entities that would use them, and almost always remain prohibitively priced, or without long term supporting data. We have made reference to this in our conclusions. In addition, the literature is heavily rice-focused, because these agronomic approaches have been developed in response to the Codex Commission's adoption of a maximum level of 0.2 mg/kg inorganic arsenic in polished rice, and because it is primarily in anaerobic flooded systems that arsenic uptake by plants is an issue. We have tried to include studies on other cropping systems where available, but these are inevitably fewer than the rice-based studies.

R2C12: You discuss how water management, fertilisers and the plant can be altered for growing on arsenic soils, I think this could be expanded into possible remediation strategies for the soil as well. A lot of the information that you present in the opening sections is about the addition to arsenic to land / soil, a section on mitigation of this and then dealing with high arsenic soils would be good.

Response to R2C12: This is an interesting topic, but we feel an additional section on phytoremediation of highly arsenic contaminated soils is well beyond the scope of this article. Our focus is on the route arsenic takes into the human food supply, from soils not necessarily classified as highly arsenic contaminated, and how this can be managed to reduce dietary arsenic exposure. A section on mitigation of arsenic input into agricultural soil is also not warranted, since much of the arsenic contamination relevant to food production is either natural, legacy (former pesticide application), or the result of efficient plant-based arsenic accumulation mechanisms. We have dealt with some of these aspects in our conclusion, where we state that information on arsenic levels in soils be made more available so the appropriate action can be taken to avoid human exposure.

# C-FARR

Collaborative on Food with Arsenic and associated Risk and Regulation



## Highlights

Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants

- Consumption of staple foods such as rice, apple juice and vegetables grown in contaminated soil is now recognized as a tangible route of human exposure to arsenic
- Arsenic occurs in food because it is present in the soil and water and is taken up by crop plants.
- Understanding the sources of arsenic to crop plants and influence the dynamics of the agronomic arsenic cycle are key to reducing crop uptake of arsenic now, and preventing exposure in future.
- This review considers natural and anthropogenic sources of arsenic to the soil, biogeochemical cycling, rhizosphere processes, plant processes, and mitigation strategies
- There must be strategies in place that protect human health from soil contamination by arsenic.
- This review recommends: mobilizing existing soil data so that it is readily accessible to commercial and private growers; expanding detailed soil monitoring; reconsideration, unification and enforcement of action levels for agricultural soil arsenic based on updated science, community outreach and education about the potential for arsenic in the soil, as necessary steps to protecting valuable soil resources.

**Understanding arsenic dynamics in agronomic systems to predict and prevent uptake by crop plants.**

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1    **Keywords:** arsenic, sources, soil, agriculture, plants, mitigation

## Abstract

This review is on arsenic in agronomic systems, and covers processes that influence the entry of arsenic into the human food supply. The scope is from sources of arsenic (natural and anthropogenic) in soils, biogeochemical and rhizosphere processes that control arsenic speciation and availability, through to mechanisms of uptake by crop plants and potential mitigation strategies. This review makes a case for taking steps to prevent or limit crop uptake of arsenic, wherever possible, and to work toward a long-term solution to the presence of arsenic in agronomic systems. The past two decades have seen important advances in our understanding of how ~~sources of arsenic to the soil are influenced by~~ biogeochemical ~~calstry~~ and physiological processes influence ~~result in~~ human exposure to soil arsenic, ~~which and this~~ must now prompt an informed reconsideration and unification of regulations ~~that to~~ protect the quality of agricultural and residential soils.

## 1. Introduction

Consumption of staple foods such as rice, beverages such as apple juice, or vegetables grown in historically arsenic-contaminated soils ~~are~~<sup>is</sup> now recognized as a tangible route<sup>s</sup> of arsenic exposure~~-~~. The presence of elevated concentrations of arsenic in the soil is not a pre-requisite for dietary arsenic exposure; seen in the accumulation of arsenic by rice grown in uncontaminated soils<sup>1</sup>.~~that~~, When drinking-water arsenic concentrations are low, dietary arsenic can be a significant exposure~~in many cases, is more significant than exposure from drinking water~~<sup>2</sup>. Understanding the sources of arsenic to crop plants and the factors that influence them is key to

reducing human exposure now and preventing exposure in future. In addition to the abundant natural sources of arsenic, there are a large number of industrial and agricultural sources of arsenic to the soil; from mining wastes, coal fly ash, glass manufacturing, pesticide application, wastewater sludge, pharmaceutical waste, livestock dips, smelting activities to phosphate fertilizers. Plant uptake of arsenic was previously assumed to be too low to merit setting limits for arsenic in food crops, but ~~given that no safe level of arsenic has been found,~~ measurable biological effects occur in at arsenic levels below the current maximum contaminant level (MCL) for ~~arsenic in~~ drinking water<sup>3</sup>, these low levels can still translate into significant exposures, particularly in children<sup>4</sup> and presumably in adults who consume a lot of rice. In response, the World Health Organization (WHO) ~~who~~ set an advisory MCL for inorganic arsenic in white (polished) rice of 0.2 mg/kg<sup>5</sup> (~~[http://www.who.int/ipcs/assessment/public\\_health/arsenic/en/](http://www.who.int/ipcs/assessment/public_health/arsenic/en/)~~) along with the limit of 10 µg/L in water, and the European Union set similar standards that included a lower MCL (0.1 mg/kg) for rice-containing baby foods<sup>6</sup> (~~[http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=urisrv%3AOJ.L\\_2015.161.01.0014.01.ENG](http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=urisrv%3AOJ.L_2015.161.01.0014.01.ENG)~~). Currently, dietary arsenic exposure is suspected to play a role in cardiovascular disease in adults<sup>7</sup>, and to disrupt the glucocorticoid system (involved in learning and memory) to those exposed *in utero*<sup>8</sup>. An in depth review of the current findings on the relationship between dietary arsenic exposure and human health is provided by Davis et al. (this issue).



1 In the United States, regulations on arsenic are distributed to several agencies. The  
2 Environmental Protection Agency (EPA) developed the MCL for arsenic in drinking  
3 water (10 µg/L) in 2006; a level supported by the World Health Organization, Canada  
4 and the European Union. In the state of New Jersey (USA) the limit is 5 µg/L, and in  
5 Australia, 7 µg/L. ~~The majority of~~Many other nations still adopt a level of 50 µg/L  
6 (Bahrain, Bangladesh, Bolivia, China, Egypt, India, Indonesia, Oman, Philippines, Saudi  
7 Arabia, Sri Lanka, Vietnam, Zimbabwe)<sup>9</sup>, with the exception of Mexico (35 µg/L). In  
8 the USA, The Food and Drug Administration (FDA) is responsible for setting action  
9 levels for arsenic in food, which includes apple and pear juice at 10 µg/L, in line with  
10 EPA's drinking water MCL. In Canada, the Canadian Food Inspection Agency issued  
11 alerts on excessive arsenic in rice and pear products in 2014. Consistent with the  
12 European Commission's limit for arsenic in rice used in food production for infants  
13 and young children, the FDA is proposing an action level of 0.1 mg/kg for inorganic  
14 arsenic in infant rice cereal<sup>10</sup>. Foods in Australia and New Zealand may not contain  
15 more than 1 mg/kg dry mass of arsenic, and salt for food use must not contain more  
16 than 0.5 mg/kg. Japan has a limit of 15 mg/kg of arsenic in paddy soils  
17 (<http://www.env.go.jp/en/water/soil/sp.html><sup>11</sup>). ~~Likewise~~Likewise, Thailand has an  
18 agricultural arsenic soil quality standard of 3.9 mg/kg. Within the USA, states differ  
19 widely in their action levels for arsenic in soil, for instance New Jersey has a cleanup  
20 criterion of 20 mg/kg and Florida has a cleanup target level of 2.1 mg/kg and 12  
21 mg/kg for industrial sites<sup>12</sup>.

Arsenic occurs in food because it is present in soil and water and is taken up by plants. This review article brings together the latest scientific information on arsenic in agronomic systems, describing its sources in soils and the processes that influence the uptake of arsenic by crop plants. The intention of this review is to prompt a reconsideration and unification of government regulations on action levels for arsenic in agricultural soil; raise awareness of how both former and ongoing inputs of arsenic to soil can result in food contamination and impacts to human health and finally, to indicate the way forward for mitigation strategies that safeguard valuable soil resources.

## 2. Natural sources of arsenic in soil

Below toxic concentrations, the higher the total soil arsenic concentration (the sum of all arsenic species, regardless of bioavailability) the higher the crop uptake of arsenic, ~~including~~ This is true of anaerobic cultivation systems such as rice<sup>13-15</sup>, aerobic horticultural systems<sup>16</sup> as well as conventional (aerobic) agriculture<sup>15</sup>. The global average total soil arsenic concentration is 5 mg/kg, (equivalent to parts per million), but there is large variation between and within geographical regions<sup>17</sup>. Where soils have formed on arsenic-rich bedrock, or downstream of these bedrocks, very high concentrations of natural arsenic can result. Concentrations of up to 4000 mg/kg arsenic have been measured in soils from the arsenopyrite belt (iron arsenic sulphide, FeAsS) in Styria, Austria<sup>18</sup>, for instance. There are approximately 568 known minerals that contain arsenic as a critical component<sup>19</sup>. Arsenic is present in many rock-forming minerals because it can chemically substitute for phosphorus (V),

silicate (IV), aluminum (III), iron (III) and titanium (IV) in mineral structures. Global mapping data of total arsenic concentrations in topsoil is not available, although large-scale regional maps are available for soil arsenic concentrations in Europe<sup>20</sup> and the USA <sup>21</sup>. European data predicts that most soils ~~are range below~~ < 7.5 - 20 mg/kg arsenic, with a median of 6 mg/kg<sup>20</sup>. This prediction comes from block regression-kriging; a spatial prediction technique based on regressing soil arsenic concentrations against auxiliary variables, and is useful because it uses a particularly high resolution (block size of 5 km<sup>2</sup>). On a continental scale, large zones of soils with approximately 30 mg/kg arsenic have been found in southern France, the ~~northeastern~~ north-eastern Iberian Peninsula and south-west England, with the two latter being zones of extensive natural mineralization associated with base and precious metal mining activities. The United State Geological Survey (USGS) soil sampling of the contiguous USA reports a mean USA soil arsenic concentration of approximately 5 mg/kg with 5 and 95 percentile values of approximately 1.3 and 13 mg/kg respectively<sup>22</sup>. Large regional patterns are apparent in the data, for example the soils of New Hampshire have soil arsenic concentrations of approximately 10 mg/kg arsenic, and Florida, 3.5 mg/kg. The sampling density goal for the USA surface soils and stream sediments database is 1 per 289 km<sup>2</sup> <sup>23</sup>, but is currently at only 1 sample per 1600 km<sup>2</sup>. This contrasts with smaller regional surveys such as the recently published Tellus database for Northern Ireland that has a sampling density of 2 km<sup>2</sup> <sup>24</sup> (median total soil arsenic concentration 8.7 mg/kg). At this sampling density, fine-scale data for factors shown to affect soil arsenic, such as bedrock type,

altitude and organic matter for instance, can be observed, providing the opportunity to make predictions about arsenic bioavailability and mobility.

Soil or sediment arsenic concentrations are the result of the complex and dynamic interplay between inputs and outputs<sup>25</sup>. Natural sources of arsenic to agronomic catchments are dominated by bedrock weathering (mechanical, chemical and biological) and depositional inputs, with the ultimate sinks at the base of catchments often being a significant distance away from sources<sup>26</sup>. Outputs include leaching into water bodies (vertically and horizontally), soil erosion<sup>25</sup> and biovolatilization<sup>27</sup>. In arid regions surface evaporation of water can lead to arsenic enrichment from the draw up of subsurface water<sup>25</sup> and from waters used in crop irrigation<sup>28</sup>. Mass-balances (accounting for all inputs and outputs for a particular ecosystem) are rarely conducted for arsenic fluxes within catchment areas, but a good example is from a mining-impacted catchment area<sup>29</sup>, where chemical weathering, followed by mechanical weathering dominated arsenic inputs, which were primarily from arsenopyrite. Similarly, in a gold-mining region, weathering contributed an estimated 95% of the arsenic<sup>30</sup>. In a forested catchment area, where atmospheric arsenic inputs were the dominant source to highly organic soil (soils with more than 10% organic matter), inputs of arsenic via precipitation were ~6 g/ha/y<sup>31</sup>, and organic soils were a net source of arsenic, while mineral soils (~~soils with~~ less than 10% organic matter) were a sink. This agrees with depositional inputs of arsenic measured in the UK, which ranged from ~1 to ~10 g arsenic/ha/y<sup>32</sup>. UK regional scale maps show that arsenic deposition is highest at altitude and in the west of the country; the least

1 polluted regions with air masses originating in the Atlantic. This suggests a marine  
2 source of arsenic. Depositional maps relate well to soil arsenic maps such as in maps  
3 of Northern Ireland and England<sup>33</sup> that show highest arsenic concentrations in peat  
4 soils at higher altitude, along with bedrock geological anomalies. Peat soils at higher  
5 altitude are sinks for arsenic, and become sources if the peat is mineralized or  
6 eroded. The topic of upland organic soils acting as sinks and sources of arsenic is  
7 receiving more research attention<sup>34</sup>, and could be important on a regional scale as a  
8 source of arsenic to downstream sediments.

9  
10 In large catchment areas of continental importance, such as the deltas that form to  
11 the south and east of the Himalayas, plate tectonic-derived mechanical weathering is  
12 thought to be the most important source of arsenic. One theory is that the  
13 mechanical weathering caused by Pleistocene tectonic uplift in the Himalayas is the  
14 key to understanding why arsenic is so elevated in Holocene aquifers, such as those  
15 of SE Asia, and in the glacial tills of Europe and North America<sup>26</sup>. Mechanical  
16 weathering of bedrock exposes previously inaccessible mineral surfaces, and the finer  
17 grinding leads to enhanced surface areas for chemical and microbial weathering to  
18 take place, causing greater solubilisation of arsenic<sup>25, 26, 35</sup>. Chemical and microbial  
19 weathering can take place at or near the source, or in sediment sinks. For  
20 ~~instance~~instance, bacteria isolated from Bay of Bengal aquifers can mobilize arsenic  
21 from apatite<sup>35</sup> (See Section 5). Invariably, the arsenic loadings into soil will be  
22 dependent on arsenic in the bedrock, and the extent of the weathering of that  
23 bedrock-derived material along the route from source to sink. Soils with basalt

bedrock had the lowest median arsenic content, while those with psammite, semipelite, and lithic arsenite bedrocks had the highest. Interpretation of such fine-scale mapping can ultimately lead to predictions of soil arsenic concentrations where detailed maps are not available. Combined with an understanding of soil chemistry, this will enhance the ability to predict elevated concentrations of arsenic in crops<sup>36</sup>.

### **3. Anthropogenic sources of arsenic to soil**

Many anthropogenic activities have increased soil arsenic concentrations above the natural, background levels mentioned in Section 2 above, and they have the potential to increase the arsenic concentration in food. This is especially the case in the USA where the widespread use of arsenic-based herbicides, pesticides and livestock antibiotics throughout the 20<sup>th</sup> century has ultimately increased the arsenic concentrations of current productive USA agricultural soils<sup>37-39</sup>.

#### *3.1. Base and precious metal mining*

The dominant mineral source of arsenic is thought to be pyrite (iron sulfide,  $\text{FeS}_2$ )<sup>40</sup>, an economically important ore deposit. High arsenic concentrations are found in many oxide minerals and hydrous metal oxides, either part of their structure or as sorbed and occluded species<sup>41</sup>. Iron oxides accumulate arsenic up to concentrations of several weight percent (1 weight % being equivalent to 10,000 mg/kg), and arsenic tends to bind to iron (III) (hydr)oxides whenever they are present. Arsenic is found predominantly as arsenopyrite but also can occur as orpiment (arsenic trisulphide  $\text{As}_2\text{S}_3$ ), realgar ( $\alpha\text{-As}_4\text{S}_4$ ) and other arsenic sulfide minerals<sup>42, 43</sup>. Arsenic is a

1 byproduct of most mining operations and is present at high concentrations in the  
2 mine waste, and, because arsenic sulfides are particularly prone to oxidation in  
3 surface environments, in mining wastewaters<sup>42, 44</sup>. Arsenic can constitute 1% or more  
4 of the ore and solid waste, and wastewaters and impacted streams often contain  
5 dissolved arsenic concentrations ranging from 0.01 to over 10 mg/L. Because mining  
6 and smelting operations are localized, arsenic contamination of soils exists around  
7 the mine site with the concentration decreasing with distance from the source.  
8 Windblown dispersion of fine particulate material is a particular problem, spreading  
9 contamination greater distances from the mine site. This fine material - which is not  
10 completely removed by washing<sup>16</sup> - can directly contaminate plant material;  
11 especially leafy material with high surface area. This presents a tangible risk to  
12 residents and home gardeners in the vicinity of areas with significant surface soil  
13 arsenic contamination. A comparison of arsenic concentrations in vegetables grown  
14 in SW England (the site of historic mining activities) with those from a pristine site in  
15 NW Scotland found a generally good correlation between total plant arsenic and soil  
16 arsenic concentrations. Increased arsenic concentrations were measured in produce  
17 from SW England where soil arsenic concentrations ranged from 120 – 1130 mg/kg.  
18 Arsenic concentrations were high in leafy greens (kale, spinach, lettuce) and some  
19 unpeeled vegetables (potatoes, swedes, carrots) were higher than when peeled,  
20 which, in both cases, points to contamination from windblown soil particles and soil  
21 adhesion to below ground biomass, rather than from root uptake. In this particular  
22 study, the majority of arsenic was present as the inorganic form<sup>16</sup>. Similar results  
23 were obtained from home gardens near the Iron King Mine Superfund Site in Arizona,

USA<sup>45, 46</sup>. Here the tailings had arsenic concentrations of 3,710 mg/kg and residential soil sampled adjacent to the site ranged from 120 – 633 mg/kg. Edible plant tissue concentrations ranged from < 0.01 – 1.96 mg/kg (plant concentrations are expressed as dry weight throughout), and were generally positively correlated with soil arsenic concentrations. Leafy and high surface area vegetables such as lettuce, kale, broccoli and cabbage accumulated higher arsenic concentrations than beans, tomatoes, cucumbers and peppers. Arsenic in mine-affected vineyard soils in Italy ranged from 4 – 283 mg/kg and positive correlations were observed between soil concentrations and arsenic levels in vine leaves and grapes, however, levels in wine were low (< 1.62 µg/L)<sup>47</sup>. In the Hunan province, China, the high levels of inorganic arsenic in rice have been traced to mining activities in the area<sup>48-50</sup>.

### 3.2. Coal combustion for energy

The concentration of arsenic in USA coal ranges from 1 – 71 mg/kg with an average concentration of 24 mg/kg<sup>51</sup>. Fly ash, the major byproduct of the coal combustion process, consists of fine particles that are driven out with the flue gases, and is a major source of arsenic to the wider environment. Coal ash is one of the most abundant of industrial wastes; close to 130 million tons<sup>52</sup> ~~130 million tons~~ of coal fly was generated in the USA in 2014, with 100 million tons estimated from the European Union in 2011<sup>53</sup>. Arsenic concentrates in the fly ash during combustion of coal for energy; the median arsenic concentration in USA fly ash is 71 mg/kg<sup>54</sup>. Fly ash is often sluiced into settling basins, and since-because arsenic in fresh ash is quite soluble, wastewater arsenic concentrations can consequently be quite high.

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Arsenic can build up in the sediments of coal fly ash settling basins and reach concentrations of over 1000 mg/kg. Catastrophic failures of these setting basins have caused severe environmental problems and contaminated surface waters with arsenic<sup>55</sup>. There is a well-founded concern that arsenic from coal combustion wastes can contaminate soil and enter the food supply. The use of coal fly ash as a soil amendment can lead to elevated arsenic concentrations in crops (as well as boron, selenium and molybdenum), although its lack of soil macronutrients and the potential for arsenic toxicity prevents the sole application of coal fly ash as a soil amendment<sup>56, 57</sup>. Formulating ash/organic waste mixtures that conform to USEPA regulations for total arsenic application and meet soil and plant fertility requirements has been shown to safe and effective for agronomic use<sup>58</sup>.

### *3.3. Pesticides*

Perhaps the largest anthropogenic input of arsenic to agricultural soils in the USA is from the agricultural use of arsenic-based pesticides and herbicides for most of the 20<sup>th</sup> century. Calcium arsenate and lead arsenate were used extensively up to the 1950s, mostly on orchard soils to combat the codling moth. At peak, 132,000 metric tons of each pesticide compound was applied annually between 1930-1940<sup>37</sup>. In addition to apples, inorganic arsenic pesticides were used on a range of crops including essentially all fruit trees, vine berries, sweet potatoes, white potatoes, most vegetables and cotton<sup>37</sup>. Both lead and arsenate have long residence times in soils and high concentrations (often >100 mg/kg) of these two elements have been reported in old orchard soils in Washington<sup>59</sup>, North Carolina<sup>60</sup>, New Hampshire<sup>61</sup>,

1 New Jersey<sup>62</sup> and Virginia<sup>63</sup>. There is some evidence of greater mobility for arsenic  
2 (than lead)<sup>61, 64</sup>, and retention of both elements depends on soil type and other  
3 environmental factors but most of this legacy contaminant remains in the soil<sup>62</sup>. Use  
4 of lead arsenate decreased after 1950s and was finally banned in 1988. The organic  
5 arsenic compounds dimethylarsinic acid (DMA) and monomethylarsonic acid (MMA)  
6 were used as pesticides on cotton and herbicides for golf courses and right-of-ways  
7 until they too were withdrawn from use in 2013. High levels of MMA were reported  
8 in transient surface waters adjacent to a crop sprayer operation<sup>65</sup>. Legacy soil arsenic  
9 contamination resulting from organic arsenical pesticides plays a major role in  
10 straighthead disease of rice<sup>66</sup> (See Section 6.1). It may be that arsenical pesticides  
11 have leached to groundwater, as has been suggested for the Texas High Plains  
12 Aquifer<sup>67</sup>, although a study of the Ogallala aquifer in the High Plains in Texas found  
13 no evidence of anthropogenic arsenic in the groundwater<sup>68</sup>. Similarly, there was no  
14 relationship between groundwater arsenic and past (inorganic) arsenic pesticide  
15 usage in a comprehensive study of New Hampshire groundwater sources<sup>69</sup>. About  
16 10%, depending on soil substrate, of monosodium methyl arsenate applied to sandy  
17 soils (simulated golf course greens) leached into percolating water. ~~and~~  
18 ~~De~~demethylation and methylation occurred because both inorganic arsenic species  
19 and DMA were also detected in the percolating water<sup>70</sup>. As with mining-impacted  
20 soils, plants grown on soils that are high in arsenic from arsenical pesticide  
21 contamination take up higher levels of arsenic into their edible tissues, observed for  
22 example in potatoes<sup>71</sup>, carrots<sup>72</sup> and leafy green vegetables<sup>73, 74</sup>.

Former pesticide application has been suggested be a factor in the presence of higher levels of total arsenic found in rice grown in the south-central regions of the USA<sup>75, 76</sup> compared to other areas of the USA and to other countries, such as Bangladesh<sup>77</sup>. ~~A survey of arsenic concentrations in whole grain rice products purchased from various countries found that rice grown in the USA and EU contained more total arsenic than rice grown with arsenic contaminated groundwater irrigation in Bangladesh<sup>67</sup>. Evidence on varietal differences in arsenic uptake, speciation and distribution within rice grain (See also Section 6) strongly suggest that soil arsenic concentration is not the sole, nor particularly the main driver of this phenomenon. (Meharg, 2004 #448)(Zavala, 2008 #449) Other factors likely to be influential include the differences in the soil microbial community composition between geographical regions that affect arsenic methylation, considering that plants themselves cannot methylate arsenic<sup>78</sup>. The distribution of arsenic forms found in the grains differed between cultivars, and is discussed in more detail in Section 6.~~

#### 3.4. Wood preservatives

Chromated copper arsenate (CCA) is used as a wood preservative and was extensively used on decking, and other residential usages until a voluntary manufacturer withdrawal in 2003. The primary health concern is for young children in direct contact with CCA-treated wood, but localized leaching of arsenic (as well as chromium and copper) also occurs to surrounding soil. Soil arsenic concentrations of 37 – 250 mg/kg have been reported for soils sampled near CCA-treated utility poles (N=12)<sup>79</sup> and mean arsenic concentrations for soils collected below decks and

footbridges in Florida, USA was reported to be 28.5 mg/kg compared with a control concentration of 1.3 mg/kg (N= 65)<sup>80</sup>. Arsenic from CCA contaminated soils appears to be more bioavailable than from other anthropogenic sources to soil<sup>81</sup>.

#### *3.4. Organic manures*

Land application of sewage sludge (biosolids) in the USA is regulated by Environmental Protection Agency Part 503 Biosolids rule; which set the maximum arsenic concentration of the sludge at 75 mg/kg, an annual pollutant-loading rate of 2.0 kilograms arsenic/hectare (kg/ha) and a cumulative pollution-loading rate of 41 kg/ha over the lifetime of applications. Assuming a plow layer of 17 cm, application at the maximum annual rate implies an approximate 1.2 mg/kg maximum increase in soil arsenic, while the cumulative maximum loading rate could increase soil arsenic concentrations by approximately 24 mg/kg over the lifetime of application and assuming no loss from the soil profile. This cumulative loading rate of 24 mg/kg is significant when considered against an average soil arsenic concentration of 5 mg/kg (See Section 2), however, relative to mine-impacted or inorganic arsenic pesticide impacted soils where arsenic concentrations are frequently > 100 mg/kg, it is of lesser concern. Also, sewage sludge is often high in aluminum and iron oxide phases, used in the flocculation process, which are efficient scavengers of inorganic arsenic thus lowering the arsenic bioavailability<sup>82</sup>.

Arsenic occurs in animal wastes primarily because of the former use of arsenic antibiotics in poultry and turkey feed; until 2015 four drugs, roxarsone, p-arsanilic

acid, carbarsone and nitrosone, were regulated for use, with roxarsone being the most prevalent. As of 2016 all four of these compounds have all been withdrawn from use<sup>83</sup>. All four are organic arsenic compounds with an arsenate functional group attached to a benzene ring, and differ by other substituents on the ring. The compounds are not readily adsorbed or metabolized and so occur at concentrations up to 40 mg/kg in animal manures. This provides three points of entry to the human food chain; directly through arsenic in chicken and turkey meat<sup>38, 39, 84</sup>, from plant uptake after land application of manure, and runoff to surface water or groundwater. A number of studies have shown that these organic arsenic compounds can be degraded by both photolytic<sup>85</sup> and microbial<sup>86</sup> processes and that this degradation happens both during composting of stockpiled litter<sup>87, 88</sup> and after land application<sup>89, 90</sup>. Long term application of poultry litter to Upper Coastal Plain soils increased soil arsenic concentrations; from 2.7 to 8.4 mg/kg ~~for an Upper Coastal Plain soil~~ after 25 years of application<sup>91</sup>. ~~and~~ similar increases have been reported for other southern states of the USA<sup>92</sup>. There is evidence to suggest that other soluble constituents of the litter, for example phosphate and dissolved organic carbon compounds, facilitate arsenic solubility and leaching<sup>89, 92, 93</sup>.

### 3.5. Seaweed fertilizers

Seaweeds can contain far higher concentrations of arsenic than crop plants: up to 100 mg/kg (~~See~~ Taylor et al, this issue). In most cases the arsenic is present as arsenosugars, which are of low toxicity to humans (~~See~~ Taylor et al, this issue). However, as in the case of poultry litter, these compounds degrade (ultimately) to

inorganic arsenic after land application<sup>94</sup>. Although seaweeds are a 'niche' soil amendment, their use agriculture is increasing and has been adopted by many organic farms as a soil fertilizer as well as a feed additive in organic dairy farming<sup>95</sup>.

#### **4. Biogeochemical cycling within terrestrial agronomic ecosystems**

Arsenic cycles within the soil surface and near-surface environment<sup>96</sup>, influenced by mineralogy, abiotic factors such as pH and redox potential ( $E_H$ ), and biotic factors such as microbially-mediated biomethylation.

##### *4.1. Redox regulation*

The most important biogeochemical step in the exposure of humans to arsenic is its release from soils and sediments into pore water; the water contained within soil pores and/or rock<sup>40</sup>. With the exception of extreme pH conditions (<4 or >9), or high concentrations of competing ions (e.g. phosphate, silicic acid or silicate<sup>97</sup>) the release of arsenic from its strong bonds with soil particles depends upon redox potential ( $E_H$ ); the extent of aeration of the soil<sup>40</sup>. As  $E_H$  falls, electron acceptors are depleted and anoxic conditions develop, causing iron oxides and oxyhydroxides to be reduced and dissolve, releasing sorbed arsenic into the soil solution<sup>98</sup> where it can be taken up by plant roots, or leached into groundwater.

Agronomic cropping systems can be divided with respect to arsenic mobilization on the basis of their redox status. Dominant biogeochemical processes influencing aerobic systems, specifically cereals, upland rice, fruit tree orchards, and community

gardens, differ from those that dominate in anaerobic systems, predominantly in flooded rice paddies. In aerobic soils, arsenic speciation is predominantly arsenate (arsenic (V)), and is tightly bound to soil particles. Under anaerobic or flooded conditions, arsenic is reduced, and arsenite (arsenic (III)) is the dominant species<sup>40</sup>.

Arsenite is less stably bound to aluminum hydroxides and aluminosilicate clay minerals in the soil ~~particles~~ than arsenate, for which they exhibit a much stronger binding preference<sup>40</sup>. With few exceptions (such as under conditions of sulfur release), transition of arsenic speciation from arsenate to arsenite is the most influential factor to arsenic bioavailability; and it is under anaerobic conditions where arsenic becomes an imminent human health concern. Influential biogeochemical processes in aerobic systems are ageing and accumulation of arsenic in soil, and in anaerobic systems reductive dissolution of iron-bearing minerals is the dominant process.

#### *4.2. Biotransformation to methylated and volatile species*

Volatile arsenicals are arsenic species with a boiling point below 150°C; the most volatile of which is arsine gas ( $\text{AsH}_3$ ), followed by monomethylarsine ( $\text{MeAsH}_2$ ), dimethylarsine ( $\text{Me}_2\text{AsH}_2$ ) and finally completely methylated trimethylarsine (TMA). Volatile arsenic species can be formed either biotically – by fungi, bacteria and algae<sup>27, 99, 100</sup> or abiotically<sup>99</sup>. In natural systems arsines readily react with oxygen to form non-volatile oxidation products, with  $\text{AsH}_3$  most rapidly oxidized and challenging to detect in environmental samples. Oxidation of the arsine gases to inorganic arsenic species completes the arsenic cycle, with arsenic returned to the soil by rain or dry deposition<sup>101</sup>.

Arsenic methylation in soils increases with decreasing redox potential<sup>102</sup>, and addition of organic matter. Increased arsenic volatilization was measured in soil after the addition of rice straw<sup>103</sup>, and animal waste products<sup>104</sup>. Inoculation of fungi (*Penicillium* and *Ulocladium* spp.) increased arsenic volatilization up to 8 fold in heavily contaminated and spiked soils<sup>105</sup>. Microbially mediated arsenic volatilization remains very inefficient, which hinders attempts to use it in soil remediation. Gaseous arsines are volatilized from arsenic contaminated soils into the atmosphere at very low rates: a microcosm study found 0.5 – 70 µg of arsenic kg<sup>-1</sup> soil year<sup>-1</sup> was volatilized from a range of soils and a range of arsenic levels<sup>27</sup>, and field measurements of arsenic volatilization are 1-2 orders of magnitude lower than those made in laboratory mesocosms<sup>98</sup>. Genetic transformation of bacteria, using genes encoding for the protein product arsenite S-adenosyl methyltransferase (*arsM*) is an attempt to enhance arsenic methylation and volatilization. The *arsM* from *Rhodopseudomonas palustris* was expressed in *Sphingomonas desiccabilis* and *Bacillus idriensis* grown in an aqueous system, resulting in a 10-fold increase in arsenic volatilization compared to the wild type strains. In a soil-based system, 2.2 – 4.5% of arsenic was removed via microbially-mediated volatilization over an incubation period of 30 days<sup>106</sup> (See also section 5.3).

#### 4.3. Changes in soil arsenic bioavailability due to ageing

Although arsenic in aerobic soils has a lower bioavailability and presents less of an immediate concern for crop uptake, aerobic soil can accumulate arsenic from human inputs, retain them for long periods of time, and release them when redox conditions



change (See Section 3.3). Human inputs of arsenic, as discussed in Section 3, are diverse; biosolids, sewage sludge, coal fly ash, poultry litter, industrial waste, arsenical pesticides and from irrigation with naturally arsenic-enriched groundwater. For aerobic soils, ageing – where binding stability of arsenic to soil particles increases over time, is a particularly important part of arsenic cycling. Factors controlling ageing of arsenic include soil type, organic matter content and arsenic species. Both inorganic and organic arsenic species are subject to ageing, with studies indicating a slow oxidation process from arsenite to arsenate over time<sup>107</sup>.

## 5. Rhizosphere processes

Processes occurring in the rhizosphere (the boundary layer of soil under the influence of plant roots) dramatically influence arsenic concentrations and bioavailability because they involve local alterations in redox potential, pH and organic matter content. Rhizosphere acidification occurs during iron uptake by all plant species ~~except grasses, whereas grasses secrete chelating agents (phytosiderophores) to acquire iron<sup>96</sup>~~during cation uptake and charge balance, when protons are released into the rhizosphere. Plants release anywhere from 10 to 250 mg of carbon per gram of root tissue into the rhizosphere; about 10-40% of their total photosynthetically fixed carbon<sup>108</sup>, making the rhizosphere particularly rich in organic carbon compared to bulk soil, which in turn exerts an influence on arsenic solubility by stimulating microbially-mediated reductive dissolution of soil minerals. Large differences have been found in the arsenic concentration of rhizosphere soils compared with bulk

soils in highly arsenic-contaminated areas, with higher concentrations of arsenic in rhizosphere soils compared to bulk soils<sup>109</sup>.

In anaerobic soils, the iron plaque that develops on the submerged stem and roots of rice plants dominates rhizosphere dynamics of arsenic. In flooded environments such as paddy fields, plants oxygenate the rhizosphere through specialized tissues called aerenchyma, which are found in many aquatic plants and emergent macrophytes such as rice. This radial oxygen loss creates an oxidized layer around plant tissue that stimulates aerobic microbial activity and the oxidation of iron, which precipitates and forms a visible iron plaque on the root surface <sup>110-114</sup>. Formation of an iron (oxyhydr)oxide plaque on root surfaces can alter the uptake of arsenic by rice, acting as a sorbent for excess nutrients such as ferrous iron (reduced iron) as well as arsenic and aluminum<sup>115</sup>. Rates of oxygen loss influence iron plaque formation <sup>115</sup>, and vary between rice cultivars<sup>116, 117</sup>. ~~However, s~~Studies conducted over the last forty years are inconsistent on whether iron plaque prevents or enhances arsenic uptake by plants<sup>111</sup>, and the hypothesis that arsenic influences the quality and amount of the iron plaque<sup>113</sup>. Profound differences in mineral composition and quantity of laboratory-created iron plaques has been demonstrated experimentally<sup>111</sup>, which may have contributed to these inconsistencies.

### *5.1. Microbial activity*

Microbes directly and indirectly influence arsenic speciation in rhizosphere soil, and are widely considered to play a key role in arsenic biogeochemistry<sup>118</sup>. Under certain

1 nutrient-limited conditions, microbes actively weather minerals to access nutrients for  
2 cellular growth, which releases arsenic<sup>35</sup>, as well as creating abiotic conditions that  
3 induce changes in arsenic speciation via production of organic acids, polysaccharides  
4 and ligands. Soil microorganisms can strongly affect soil redox, regulating arsenic  
5 release into pore water<sup>119</sup>. A number of strains of bacteria have also been shown to  
6 contribute to the formation of arsenic minerals by using arsenic as a terminal  
7 electron acceptor, such as *Desulfosporosinus auripigmentum*<sup>120</sup>, *Desulfovibrio* strain  
8 Ben-RB<sup>121</sup>, *Shewanella oneidensis*<sup>122</sup> and *S. putrefaciens* CN32<sup>123</sup>. These  
9 microorganisms also differ in their capabilities for liberating arsenic from specific  
10 arsenate-bearing minerals<sup>119</sup>.

12 Microbial transformation can mobilize arsenic by converting inorganic to organic  
13 forms, including MMA and DMA<sup>124, 125</sup>. Plants translocate organic arsenicals from  
14 roots to the (frequently edible) above-ground parts more efficiently than inorganic  
15 arsenic<sup>126-128</sup> (See Section 6), therefore microbial transformation to organic arsenicals  
16 can increase human dietary exposure.

18 Plants, green algae and microbes can all enzymatically transform arsenic species<sup>124, 129</sup>,  
19 but methylated forms of arsenic detected in plants are a product of rhizosphere  
20 bacteria; plants cannot methylate arsenic<sup>78, 124, 130, 131</sup>. The genomes of more than 85  
21 arsenic-metabolizing archaea and bacteria have been sequenced for genes involved  
22 in arsenic metabolism<sup>132</sup>. In bacteria, archaea and fungi, arsenic methylation is  
23 catalyzed by homologs of arsM, (See Section 4)<sup>124</sup>. Resistance to arsenite and

arsenate exists in nearly all microbes, which also confers the ability to transform arsenate into volatile arsine gases<sup>133</sup>, a particularly effective way of removing arsenic.

Profiling the transcriptome, proteome and metabolome of arsenic contaminated soils offers way of understanding microbially-mediated rhizosphere arsenic processes<sup>132</sup>. This approach measures the presence and expression of specific genes, rather than attempting to isolate and study the microbes that carry them, 98% of which – it is estimated - do not grow in culture<sup>134</sup>. Microbially mediated arsenic metabolic processes that play a major role in arsenic cycling in agronomic systems include arsenite oxidation (via the  *aio*  genes), arsenate respiration (via the  *arr*  genes), arsenate reduction (via the  *ars*  genes) and arsenite methylation (via the  *arsM*  genes)<sup>135</sup>. Interested readers are referred to the recent excellent work of Andres and Bertin<sup>132</sup> for a comprehensive review of this subject. Microbially mediated redox processes strongly influence arsenic uptake in rice, involving  *aioA* ,  *arsC*  and  *arrA* <sup>124</sup>, with pH emerging as an important factor in the distribution of microbes in paddy soils. Testing a variety of soils has shown that bacteria possessing the  *arsM*  gene for methylating arsenic are widespread and phylogenetically diverse, and even in paddy soils with low concentrations of arsenic, genes for arsenic metabolism are abundant<sup>124</sup>.

## 6. Arsenic and crop plants

Much of our understanding about the physiological mechanisms of arsenic uptake in plants comes from the study of a limited number of ~~plant~~ species. Called *model*

*plant* species, they are extensively studied, well described, easy to grow, and the results can be compared between studies. The understanding is that the information gained from studying model plants is applicable to other plant species. From a genetic perspective, orthologous genes exist in different plant species having evolved from a common ancestral gene, and they usually retain the same function. Characterization of arsenic-related genes in a model plant strongly suggests that they exist and perform similar functions in other species. Caveats to this are their levels of expression, which makes some plants more adept at accumulating arsenic than others. In this section, much of the knowledge gained on arsenic uptake and metabolism of plants comes from the study of mouse-eared pennycress (also called thale cress or rockcress) (*Arabidopsis thaliana* Heynh.) and rice (*Oryza sativa* L.); model plants with fully sequenced genomes. These species represent dicotyledonous (e.g. flowers, vegetables, deciduous trees) and monocotyledonous plant species (e.g. grasses, palm trees) respectively, thereby representing much of the edible crop species. An exception to this is the study of the arsenic hyperaccumulating fern (Chinese Brake fern, *Pteris vittata*), a seedless plant that is able to accumulate up to 22,630 mg/kg (dry weight) arsenic in its fronds<sup>136</sup>.

#### 6.1. Phytotoxicity of arsenicals

Arsenic is toxic to plants<sup>137</sup>. Despite lower acute human toxicity of the organic arsenicals (median lethal dose is 700-1,600 mg/kg and 700-2,600 mg/kg for MMA and DMA respectively compared to 10-20 mg/kg for inorganic forms)<sup>138</sup> no one form of arsenic is consistently more toxic to plants<sup>139</sup>. Soybean yields are affected when

tissue arsenic levels exceed 1 mg/kg, and 4 mg/kg limits cotton yields<sup>140</sup>, whereas in barley tissue concentrations of 20 mg/kg inhibited growth<sup>141</sup>. Higher yield-limiting arsenic levels have been recorded in rice: 20-100 mg/kg in above ground biomass, and 1000 mg/kg in root tissue<sup>142</sup>. By contrast, potatoes (*Solanum tuberosum* L.) suffered no growth inhibition in soils containing 290 mg/kg arsenic<sup>71</sup>. In some plants species, organic forms are more toxic than inorganic, for example in rice (order of toxicity: MMA > arsenite > arsenate = DMA)<sup>143</sup>, and in smooth cordgrass (*Spartina alterniflora* Loisel) (DMA = MMA > arsenite > arsenate)<sup>144</sup>.

Plants vary in their tolerance to arsenic, and the stress response differs for each arsenic species<sup>145-147</sup>. The chemical similarities between arsenate and phosphate means that arsenic can replace phosphate in biomolecules like ATP (adenosine triphosphate, a molecule used for intercellular energy transfer), with negative impacts on growth and metabolism<sup>148</sup>. In rice in particular, DMA and MMA induce straighthead disease (arsenic-associated straighthead disease), significantly lowering yield of certain rice varieties<sup>66</sup>. Straighthead is a physiological disorder of rice characterized by sterile florets, which remain upright at maturity instead of bending over under the weight of the filled grain. The exact cause of straighthead is unknown, but consistent flooding, low soil pH, high iron availability and high organic matter content have all been implicated in naturally-occurring straighthead disease<sup>66</sup>. Arsenic's suspected role in straighthead comes from observations of more frequent outbreaks in rice grown in soil where arsenical herbicides such as monosodium

methanearsonate (MSMA) – used in cotton production in the USA – have been historically applied.

## 6.2. Arsenic uptake mechanisms

In magnitude, plants take up arsenicals from the soil in the order arsenite > arsenate > DMA > MMA<sup>149, 150</sup>, with the various arsenic species entering via different root membrane transport proteins in the root plasma membrane that allow ions and molecules to cross with varying levels of selectivity, or target specificity. Similarities in chemical structure between arsenate and phosphate, and between arsenite and silicic acid, govern their entry into root cells. Arsenate enters root cells through phosphate transporters (the Phosphate Transporter 1 family of proteins; PHT1) in both the

model plant *Arabidopsis thaliana*<sup>151, 152</sup> and in rice<sup>153-155</sup> (Figure 1). In rice, Low Silicon 1 (OsLsi1) and OsLsi2 are silicic acid transporters and arsenite, MMA<sup>V</sup>, and DMA<sup>V</sup> are among their unintended targets<sup>156, 157</sup>.

~~The transporter that allows arsenite initial entry to root cells remains unidentified, but is hypothesized to be a member of the Plasma Membrane Intrinsic Protein family. Inside the root, These~~ Nodulin 26-like Intrinsic

Proteins (NIPs)<sup>158</sup>, which are members of the aquaporin water channel superfamily of proteins<sup>159</sup> embedded in the exodermal cell membranes of rice roots, move arsenic

from the soil into the vascular system for distribution to the stem and leaves. ~~In rice,~~

~~Low Silicon 1 (OsLsi1) and OsLsi2 are silicic acid transporters and arsenite, MMA<sup>V</sup>, and DMA<sup>V</sup> are among their unintended targets~~<sup>148, 149</sup>. OsLsi2 works in tandem with

OsLsi1 to transport arsenite inward toward the xylem<sup>160, 161</sup> (vascular tissue that conducts water and dissolved nutrients up from the roots). The arsenic uptake

Field Code Changed

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specificity of OsLsi1 is arsenite >> MMA > DMA<sup>158</sup>. These bidirectional NIP transport proteins also efflux arsenite back in to the soil, but since OsLsi1 effluxes only 15-20% of the arsenite in roots cells<sup>162</sup>, there may be other unidentified arsenite efflux transporters contributing to this process.

### 6.3. Arsenic transport and metabolism in plants

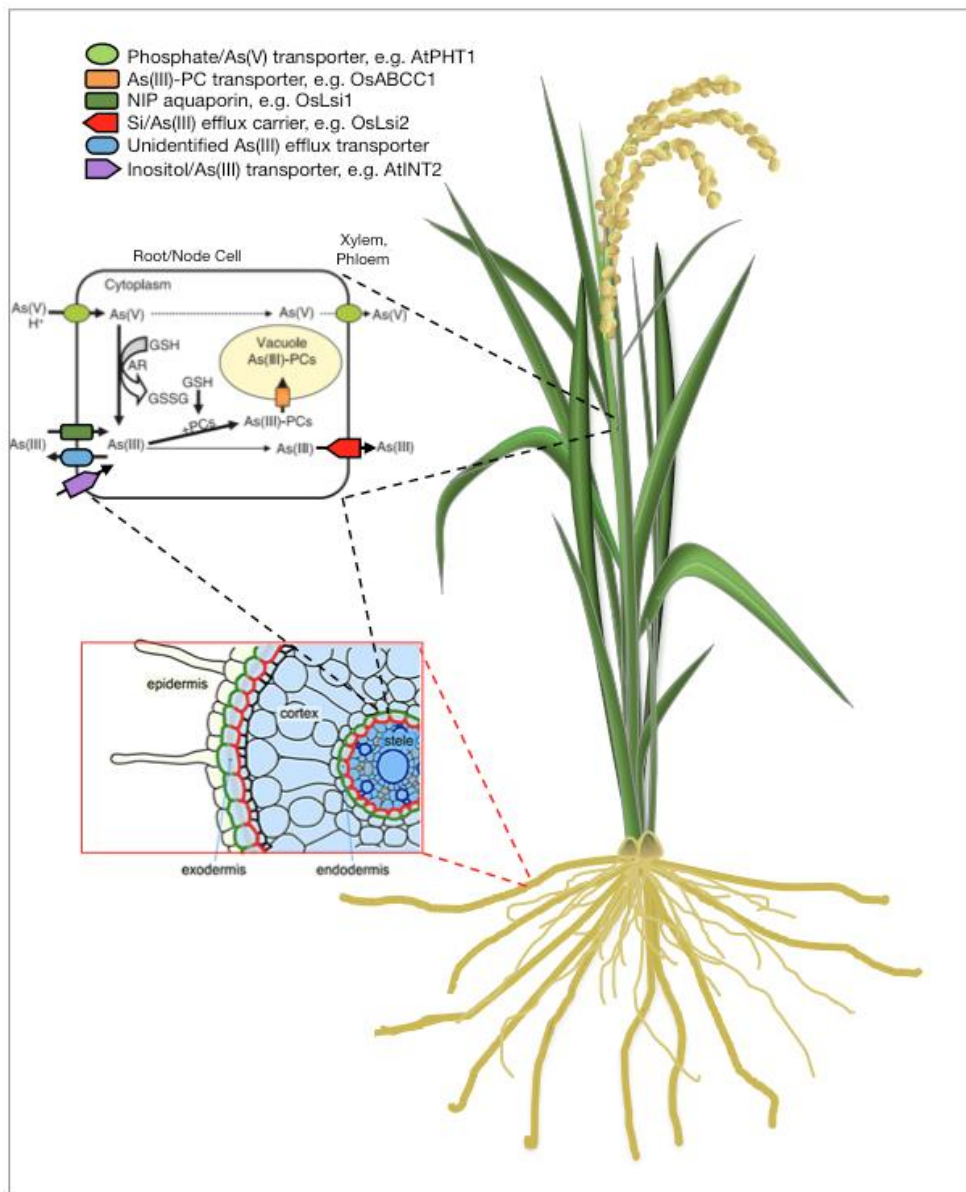
Transport of arsenite into the xylem for delivery to the shoot is less well characterized than its uptake from the soil. Arsenic is transported to the grain mainly via the phloem<sup>126-128</sup> (vascular tissue that conducts sugars and metabolic products from the leaves), by transporters in the nodes<sup>163</sup>, but their characterization is still in the early stages. Transporters for myo-inositol (Inositol Transporter 2 and 4); an important sugar for developing rice grains, also transport arsenite into the phloem companion cells<sup>164, 165</sup>. In Arabidopsis, AtINT2 or AtINT4 load about 45-64% arsenite into the grain<sup>166</sup>. The identity of transporters that move arsenite out of the phloem and into the grain are also unknown, but manipulating the target specificity of the INT genes might show promise in molecular genetic or plant breeding mitigation efforts as a way to prevent arsenite from reaching the grain.

~~Once inside the cell~~ Despite having a lower affinity for transporters into the plant than the inorganic forms, the rate of arsenic transport up towards shoot tissue is reversed, with organic forms-arsenic species are more efficiently transported towards the shoot than inorganic forms<sup>149, 150</sup> because they are not complexed by phytochelatins (PCs); sulphydryl-rich glutathione (GSH) polymers<sup>167, 168</sup>. Likewise, in broad beans (*Vicia faba*



1 L.) grown in a soil containing 90% inorganic arsenic, DMA and MMA were the  
2 dominant arsenic forms in the bean (68%)<sup>169</sup>. In root vegetables, carrot (*Daucus*  
3 *carota* L.) and beet (*Beta vulgaris* L.) grown on arsenic-contaminated soils, arsenic  
4 | forms were predominantly inorganic, ~~and~~ but for beets in particular were not readily  
5 identified using the typical standards (arsenate, arsenite, MMA and DMA)<sup>170</sup>.

6



**Figure 1.** Generalized diagram of arsenic uptake, transport and metabolism in plants.

GSH, glutathione; AR, arsenate reductase; GSSG, oxidized glutathione; PC, phytochelatins. Modified from Zhao et al.<sup>171</sup> and Ma et al.<sup>172</sup>.

1 The arsenic species composition of rice grain is influenced by the arsenic transport  
2 rate of the particular cultivar<sup>173, 174</sup>. Rice cultivars currently grown in the USA have an  
3 arsenic speciation split approximately equally between inorganic arsenic and DMA,  
4 while cultivars grown in Bangladesh contain mostly inorganic arsenic<sup>174</sup>. While lower  
5 inorganic arsenic in rice grain seems favorable for avoiding human health effects, the  
6 assumed safety of DMA is contentious<sup>175</sup>, being based on acute toxicity data, and not  
7 on genotoxicity or carcinogenicity, which are equally relevant in long term safety  
8 considerations.

9  
10 Arsenic detoxification inside cells uses a multi-step process beginning with reduction  
11 of arsenate to arsenite using an arsenate reductase enzyme<sup>176, 177</sup>. In Arabidopsis, the  
12 protein High Arsenic Content1 (~~At~~HAC1; also called Arsenate Reductase QTL1; ARQ1)  
13 reduces arsenate<sup>177</sup>. Even though arsenite is more toxic than arsenate<sup>158, 178, 179</sup>, it is  
14 hypothesized that ancestral organisms to plants were exposed almost exclusively to  
15 arsenite before atmospheric oxygen enabled arsenate formation<sup>180</sup>, and this  
16 mechanism persisted through natural selection. Arsenite is then complexed by PCs,  
17 and transported in to the vacuole<sup>167</sup> via ATP Binding Cassette-~~type~~ transporters<sup>181, 182</sup>.  
18 This process depletes glutathione availability, rendering the plant more susceptible to  
19 other oxidative stresses, which ~~negatively impact~~inhibits photosynthesis, pigment  
20 production, and the integrity of cell membranes<sup>183-186</sup>.

## 22 **7. Limiting arsenic uptake by crops**

### 23 *7.1. Water management*

1 Although the traditional method for cultivating rice involves flooding leveled, tilled  
2 fields before or shortly after planting germinated seedlings, ~~this~~flooded soil is not a  
3 biological requirement of rice plants. Flooding is used for weed and vermin control,  
4 for mobilization of key nutrients such as iron, phosphate and zinc, and importantly,  
5 flooding discourages the buildup of root nematodes over multiple years of rice  
6 growth. As mentioned earlier, flooded conditions mobilize soil-bound arsenic  
7 through reductive dissolution of Fe (oxyhydr)oxides, and the reduction of arsenate to  
8 the more mobile arsenite<sup>187</sup>. Water management strategies that involve periods of  
9 oxic soil conditions can decrease arsenic uptake in rice by limiting dissolution of  
10 arsenic. Rice grown in non-flooded or aerobic conditions has a lower yield than  
11 intermittently or constantly flooded rice<sup>188-190</sup>. Intermittent flooding (flooding  
12 maintained until full tillering, followed by intermittent irrigation) is a promising  
13 management technique to reduce arsenic levels, and can potentially produce higher  
14 grain yields than either non-flooded or constantly flooded conditions<sup>191</sup>. However,  
15 oxic conditions increase cadmium concentrations in the grain when grown in acidic  
16 soils<sup>191-193</sup>, and cadmium is also a highly toxic metal. The observed increases in  
17 cadmium were also a cultivar-specific trait, but the increase in cadmium uptake  
18 between rice grown under aerobic conditions were approximately an order of  
19 magnitude greater than their flooded counterparts. Pot experiments suggest that  
20 water management strategies implemented during the heading period of rice growth  
21 (when the rice panicle has emerged from the stem and is fully visible, just before  
22 flowering) can regulate both arsenic and cadmium concentration in the grain<sup>190, 192</sup>.

## 7.2. Amendment and fertilization practices

Soil amendment involves incorporating substances into the plow layer that either add missing nutrients, reduce the bioavailability of existing potentially toxic substances (to prevent crop uptake), or both. Soil amendments that have shown potential in reducing arsenic uptake by plants include iron-, and silica-based additives. The use of iron-based amendments increases in the concentration of free iron oxide in the soil, retarding the release of arsenite from the solid phase into soil solution, (mentioned in Section 4.1 and discussed in Section 5), whereas silica fertilization, inhibits arsenic uptake by competitive inhibition at the plant root surface while adding an essential nutrient.

Zero valent iron powder (90% iron) and iron oxide (56% iron) incorporation prevented uptake of arsenic in to the grain of rice grown on soil containing 39.5 mg/kg total arsenic by approximately 45%, and corresponded with a reduction in bioavailable arsenic in the soil<sup>194</sup>. Amendment with iron oxides (at a rate of 2%) was also more effective at reducing grain arsenic than phosphate amendment<sup>195</sup>. Amendments have also been used in combination with water management strategies to try and reduce both arsenic and cadmium concentrations in rice simultaneously<sup>193</sup>, without success. Reduction of arsenic in the grain was achieved with iron oxide addition and constant flooding, whereas cadmium reduction was achieved with converter furnace slag addition and rain water management (no irrigation after midseason drainage until harvest).

A combination of ethylenediaminetetraacetic acid ferric sodium salt (iron EDTA) and calcium peroxide was effective for reducing arsenic uptake by vegetable crops

(lettuce, Chinese cabbage and radish) from soils containing 14 mg/kg total arsenic<sup>196</sup>, again by increasing amorphous aluminum and iron oxides. It is likely that this this level of arsenic contamination would be deemed too high for commercial vegetable production, so these amendments may only be feasible for use in private vegetable gardens. Questions remain about whether iron oxide amendment application only temporarily reduces arsenic bioavailability<sup>197</sup>. In addition, the suitability for arsenic immobilization is highest at the lower soil pH range, and is strongly affected by soil phosphorus concentration, which strongly competes with arsenic.

Rice plants take up high concentrations of silica, constituting up to 10% of dry matter in the straw and husk of the plant<sup>198</sup>. As mentioned earlier (Section 6) the silicon membrane transporter (Lsi1) is the main route of arsenite entry in to rice root cells, and provision of silicon causes competitive inhibition of arsenite uptake. Increasing silicon availability in the soil also reduces the expression of the Lsi1 transporter in the plant, which further decreases the potential for arsenic uptake. -

Fertilization of rice paddy soils with silicon is a potential mitigation strategy for preventing or reducing arsenic uptake by rice through competitive inhibition of arsenite uptake<sup>199</sup>. The use of synthetic silicon fertilizers, such as calcium silicate or silica gel is prohibitively expensive for smallholder farmers in developing countries, however reusing the silicon-rich parts of the rice plant that remain after harvesting and grain processing may provide a sustainable solution that also addresses the ongoing issue of silicon depletion of the soil<sup>198</sup>. Soil incorporation of fresh rice husks, or the ash that remains after burning the husk and straw for energy (which is a common practice ~~in small holding farms~~ for smallholder farmers), can provide silicon

1 without increasing ~~arsenic or~~ methane production and decreases either total or  
2 inorganic arsenic in rice grain<sup>200</sup>.  
3 Despite the potential of soil amendment with iron oxides, or silica to reduce arsenic  
4 bioavailability or prevent plant uptake of arsenic, the high cost of these amendments  
5 inevitably prevents their use, especially by smallholder farmers. Large rice producers  
6 in the US or Europe have not so far adopted widespread use of these soil  
7 amendments to reduce rice grain arsenic concentrations. It is reasonable to assume  
8 that use of such costly amendments would also drive up the cost of rice. It may also  
9 be that iron amendments are essentially untested in a diverse range of large-scale  
10 agricultural settings and their performance will vary from soil to soil. In non-rice  
11 agricultural systems arsenic is tightly bound to the solid phase; significant crop  
12 uptake from oxidized soil is likely to be a result of extreme contamination, in which  
13 case effective mitigation is restricted to redirecting land use away from edible crops.  
14 In systems subject to periodic flooding, improving drainage remains the best  
15 mitigation strategy.

### 17 *7.3. Mitigation using plant breeding approaches*

18 The development of crops that accumulate high levels of arsenic and yet remain  
19 healthy, while preventing arsenic from reaching the edible grain is thought to hold  
20 great potential as a strategy for reducing human exposure to dietary arsenic. The use  
21 of molecular genetics techniques such as alterations in gene expression  
22 characteristics, gene editing to alter target specificity, or alternately, using traditional  
23 plant breeding techniques are both tangible approaches. Both use knowledge of the

arsenic uptake and tolerance characteristics of plants to develop varieties with desired characteristics. These characteristics include lower arsenic uptake<sup>201</sup>, higher arsenite efflux<sup>202</sup> and increased vacuolar arsenic sequestration<sup>203</sup>. For instance, many rice cultivars have now been screened to identify those that accumulate lower levels of arsenic in their grain and efforts are underway to identify the genes underlying this trait<sup>201, 204</sup>. Overexpressing Arabidopsis ABC-type transporters that sequester arsenite-PC complexes in the cell vacuole results in plants able to grow in otherwise toxic concentrations of arsenic<sup>182</sup>. Conversely, knocking out the function of the related rice ABC transporter OsABCC1 results in higher levels of grain arsenic. The OsABCC1 transporter limits arsenic transport to grains by sequestering arsenic in the vacuoles of the phloem companion cells directly connected to the grain. By combining what we have learned from the overexpression studies in Arabidopsis and the loss-of-function study in rice, overexpression of OsABCC1 can be used as a strategy to breed arsenic tolerance and low-arsenic accumulating rice cultivars. Another promising strategy is based on expressing the arsenate efflux transporter from yeast (*Saccharomyces cerevisiae*) in rice, which can reduce arsenic accumulation in brown rice by 20%. A less successful idea to methylate sodium arsenite to DMA by expressing an algal *arsM* gene in Arabidopsis resulted in lethal phytotoxicity<sup>205</sup>, suggesting that arsenic methylation in plants can only be an effective detoxification strategy if volatile arsines ~~were~~are the end point of the methylation.

## 8. Conclusions



1 The discovery of arsenic in staple foods, beverages and other products has increased  
2 awareness and stimulated research on the sources and the processes involved. The  
3 information brought together here illustrates the ~~wide diversity of natural and human~~  
4 ~~processes that can contribute arsenic to the soil and the~~ numerous geochemical and  
5 biological processes that ~~can move~~influence the movement of arsenic into the food  
6 supply.

7 It is clear there must be strategies for ~~protecting preventing human health to~~ arsenic  
8 exposure, that operate in both the short term – to protect consumers from existing  
9 contamination – and in the long term ~~;~~ to prevent further contamination. This  
10 requires government regulation on the permissible levels of arsenic food, with lower  
11 levels for infant foods (see Nachman et al, this issue), which must work in tandem  
12 with long term goals to ~~remediate address~~ arsenic in agricultural soils, actively  
13 prevent further inputs and identify contaminated areas. Our recommendations are  
14 that the information in this review is used to inform a reconsideration and a  
15 unification of regulations on the action levels of agricultural soil arsenic, which in the  
16 USA for example, exist only at the state level, vary widely from state to state, and  
17 have no formal channels of enforcement. We recommend that educating the  
18 community and garnering their support and involvement for lowering exposure to  
19 arsenic through food is an approach already shown to hold enormous potential.

20 Direct involvement of the commercial rice growing community in research and  
21 development of arsenic mitigation strategies and amendments is needed.  
22 Community-based participatory research should extend to the agricultural community,  
23 leading to partnerships that will make longer-term field-scale testing accessible.

Much effort is given to short-term, greenhouse-scale testing of amendment formulations that will ultimately be too expensive, impractical, or ineffective in the long term. Feasibility should be the first consideration in arsenic mitigation research.

Community outreach efforts targeted to commercial growers or the home gardener specifically ~~could~~must raise awareness of the significance and potential impacts of the former land ~~uses of the land~~, encouraging testing for the presence of arsenic in the soil and educating growers on the kinds of crops shown to accumulate arsenic in their edible parts. Information gathering on former arsenic input into the soil from pesticides and from proximity to various waste sites is of paramount importance ~~to~~ ~~this effort~~, and will allow monitoring to be targeted to where it is needed most. This information needs to be accessible to everyone, everywhere; currently there is no readily available source of soil quality information at a sufficient resolution to inform commercial producers or homeowners. Going forward, management and remediation of arsenic contaminated soils is essential both for human health and food security, and innovative technologies are urgently needed that will expedite this process. Innovative solutions such as the use of rice husks to add silicon to the soil to offset arsenic uptake, and the use of cultivars with low-arsenic accumulating characteristics point the way forward for sustainable solutions.

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- 7

1     **Understanding arsenic dynamics in agronomic systems to predict and prevent**  
2     **uptake by crop plants.**

3

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1    **Keywords:** arsenic, sources, soil, agriculture, plants, mitigation

## 1   **Abstract**

2   This review is on arsenic in agronomic systems, and covers processes that influence  
3   the entry of arsenic into the human food supply. The scope is from sources of  
4   arsenic (natural and anthropogenic) in soils, biogeochemical and rhizosphere  
5   processes that control arsenic speciation and availability, through to mechanisms of  
6   uptake by crop plants and potential mitigation strategies. This review makes a case  
7   for taking steps to prevent or limit crop uptake of arsenic, wherever possible, and to  
8   work toward a long-term solution to the presence of arsenic in agronomic systems.  
9   The past two decades have seen important advances in our understanding of how  
10   biogeochemical and physiological processes influence human exposure to soil arsenic,  
11   and this must now prompt an informed reconsideration and unification of  
12   regulations to protect the quality of agricultural and residential soils.

13

## 14   **1. Introduction**

15   Consumption of staple foods such as rice, beverages such as apple juice, or  
16   vegetables grown in historically arsenic-contaminated soils are now recognized as  
17   tangible routes of arsenic exposure. The presence of elevated concentrations of  
18   arsenic in the soil is not a pre-requisite for dietary arsenic exposure; seen in the  
19   accumulation of arsenic by rice grown in uncontaminated soils<sup>1</sup>. When drinking-water  
20   arsenic concentrations are low, dietary arsenic can be a significant exposure<sup>2</sup>.

21   Understanding the sources of arsenic to crop plants and the factors that influence  
22   them is key to reducing human exposure now and preventing exposure in future. In  
23   addition to the abundant natural sources of arsenic, there are a large number of



1 industrial and agricultural sources of arsenic to the soil; from mining wastes, coal fly  
2 ash, glass manufacturing, pesticide application, wastewater sludge, pharmaceutical  
3 waste, livestock dips, smelting activities to phosphate fertilizers. Plant uptake of  
4 arsenic was previously assumed to be too low to merit setting limits for arsenic in  
5 food crops, but given that measurable biological effects occur in at arsenic levels  
6 below the current maximum contaminant level (MCL) for drinking water<sup>3</sup>, these low  
7 levels can still translate into significant exposures, particularly in children<sup>4</sup> and  
8 presumably in adults who consume a lot of rice. In response, the World Health  
9 Organization (WHO) set an advisory MCL for inorganic arsenic in white (polished) rice  
10 of 0.2 mg/kg<sup>5</sup> along with the limit of 10 µg/L in water, and the European Union set  
11 similar standards that included a lower MCL (0.1 mg/kg) for rice-containing baby  
12 foods<sup>6</sup>. Currently, dietary arsenic exposure is suspected to play a role in  
13 cardiovascular disease in adults<sup>7</sup>, and to disrupt the glucocorticoid system (involved  
14 in learning and memory) to those exposed *in utero*<sup>8</sup>. An in depth review of the  
15 current findings on the relationship between dietary arsenic exposure and human  
16 health is provided by Davis et al. (this issue).

17  
18 In the United States, regulations on arsenic are distributed to several agencies. The  
19 Environmental Protection Agency (EPA) developed the MCL for arsenic in drinking  
20 water (10 µg/L) in 2006; a level supported by the World Health Organization, Canada  
21 and the European Union. In the state of New Jersey (USA) the limit is 5 µg/L, and in  
22 Australia, 7 µg/L. Many other nations still adopt a level of 50 µg/L (Bahrain,  
23 Bangladesh, Bolivia, China, Egypt, India, Indonesia, Oman, Philippines, Saudi Arabia,

1 Sri Lanka, Vietnam, Zimbabwe)<sup>9</sup>, with the exception of Mexico (35 µg/L). In the USA,  
2 The Food and Drug Administration (FDA) is responsible for setting action levels for  
3 arsenic in food, which includes apple and pear juice at 10 µg/L, in line with EPA's  
4 drinking water MCL. In Canada, the Canadian Food Inspection Agency issued alerts  
5 on excessive arsenic in rice and pear products in 2014. Consistent with the European  
6 Commission's limit for arsenic in rice used in food production for infants and young  
7 children, the FDA is proposing an action level of 0.1 mg/kg for inorganic arsenic in  
8 infant rice cereal<sup>10</sup>. Foods in Australia and New Zealand may not contain more than 1  
9 mg/kg dry mass of arsenic, and salt for food use must not contain more than 0.5  
10 mg/kg. Japan has a limit of 15 mg/kg of arsenic in paddy soils<sup>11</sup>. Likewise, Thailand  
11 has an agricultural arsenic soil quality standard of 3.9 mg/kg. Within the USA, states  
12 differ widely in their action levels for arsenic in soil, for instance New Jersey has a  
13 cleanup criterion of 20 mg/kg and Florida has a cleanup target level of 2.1 mg/kg  
14 and 12 mg/kg for industrial sites<sup>12</sup>.

15  
16 Arsenic occurs in food because it is present in soil and water and is taken up by  
17 plants. This review article brings together the latest scientific information on arsenic  
18 in agronomic systems, describing its sources in soils and the processes that influence  
19 the uptake of arsenic by crop plants. The intention of this review is to prompt a  
20 reconsideration and unification of government regulations on action levels for arsenic  
21 in agricultural soil; raise awareness of how both former and ongoing inputs of arsenic  
22 to soil can result in food contamination and impacts to human health and finally, to

1 indicate the way forward for mitigation strategies that safeguard valuable soil  
2 resources.

## 4 **2. Natural sources of arsenic in soil**

5 Below toxic concentrations, the higher the total soil arsenic concentration (the sum of  
6 all arsenic species, regardless of bioavailability) the higher the crop uptake of arsenic.

7 This is true of anaerobic cultivation systems such as rice<sup>13-15</sup>, aerobic horticultural  
8 systems<sup>16</sup> as well as conventional (aerobic) agriculture<sup>15</sup>. The global average total soil  
9 arsenic concentration is 5 mg/kg, (equivalent to parts per million), but there is large  
10 variation between and within geographical regions<sup>17</sup>. Where soils have formed on  
11 arsenic-rich bedrock, or downstream of these bedrocks, very high concentrations of  
12 natural arsenic can result. Concentrations of up to 4000 mg/kg arsenic have been  
13 measured in soils from the arsenopyrite belt (iron arsenic sulphide, FeAsS) in Styria,  
14 Austria<sup>18</sup>, for instance. There are approximately 568 known minerals that contain  
15 arsenic as a critical component<sup>19</sup>. Arsenic is present in many rock-forming minerals  
16 because it can chemically substitute for phosphorus (V), silicate (IV), aluminum (III),  
17 iron (III) and titanium (IV) in mineral structures. Global mapping data of total arsenic  
18 concentrations in topsoil is not available, although large-scale regional maps are  
19 available for soil arsenic concentrations in Europe<sup>20</sup> and the USA <sup>21</sup>. European data  
20 predicts that most soils range < 7.5 - 20 mg/kg arsenic, with a median of 6 mg/kg<sup>20</sup>.  
21 This prediction comes from block regression-kriging; a spatial prediction technique  
22 based on regressing soil arsenic concentrations against auxiliary variables, and is  
23 useful because it uses a particularly high resolution (block size of 5 km<sup>2</sup>). On a

1 continental scale, large zones of soils with approximately 30 mg/kg arsenic have  
2 been found in southern France, the north-eastern Iberian Peninsula and south-west  
3 England, with the two latter being zones of extensive natural mineralization  
4 associated with base and precious metal mining activities. The United State  
5 Geological Survey (USGS) soil sampling of the contiguous USA reports a mean soil  
6 arsenic concentration of approximately 5 mg/kg with 5 and 95 percentile values of  
7 approximately 1.3 and 13 mg/kg respectively<sup>22</sup>. Large regional patterns are apparent  
8 in the data, for example the soils of New Hampshire have soil arsenic concentrations  
9 of approximately 10 mg/kg arsenic, and Florida, 3.5 mg/kg. The sampling density  
10 goal for the USA surface soils and stream sediments database is 1 per 289 km<sup>2</sup> <sup>23</sup>, but  
11 is currently at only 1 sample per 1600 km<sup>2</sup>. This contrasts with smaller regional  
12 surveys such as the recently published Tellus database for Northern Ireland that has  
13 a sampling density of 2 km<sup>2</sup> <sup>24</sup> (median total soil arsenic concentration 8.7 mg/kg). At  
14 this sampling density, fine-scale data for factors shown to affect soil arsenic, such as  
15 bedrock type, altitude and organic matter for instance, can be observed, providing  
16 the opportunity to make predictions about arsenic bioavailability and mobility.

17  
18 Soil or sediment arsenic concentrations are the result of the complex and dynamic  
19 interplay between inputs and outputs<sup>25</sup>. Natural sources of arsenic to agronomic  
20 catchments are dominated by bedrock weathering (mechanical, chemical and  
21 biological) and depositional inputs, with the ultimate sinks at the base of catchments  
22 often being a significant distance from sources<sup>26</sup>. Outputs include leaching into water  
23 bodies (vertically and horizontally), soil erosion<sup>25</sup> and biovolatilization<sup>27</sup>. In arid

regions surface evaporation of water can lead to arsenic enrichment from the draw up of subsurface water<sup>25</sup> and from waters used in crop irrigation<sup>28</sup>. Mass-balances (accounting for all inputs and outputs for a particular ecosystem) are rarely conducted for arsenic fluxes within catchment areas, but a good example is from a mining-impacted catchment area<sup>29</sup>, where chemical weathering, followed by mechanical weathering dominated arsenic inputs, which were primarily from arsenopyrite. Similarly, in a gold-mining region, weathering contributed an estimated 95% of the arsenic<sup>30</sup>. In a forested catchment area, where atmospheric arsenic inputs were the dominant source to highly organic soil (soils with more than 10% organic matter), inputs of arsenic via precipitation were ~6 g/ha/y<sup>31</sup>, and organic soils were a net source of arsenic, while mineral soils (less than 10% organic matter) were a sink. This agrees with depositional inputs of arsenic measured in the UK, which ranged from ~1 to ~10 g arsenic/ha/y<sup>32</sup>. UK regional scale maps show that arsenic deposition is highest at altitude and in the west of the country; the least polluted regions with air masses originating in the Atlantic. This suggests a marine source of arsenic. Depositional maps relate well to soil arsenic maps such as in maps of Northern Ireland and England<sup>33</sup> that show highest arsenic concentrations in peat soils at higher altitude, along with bedrock geological anomalies. Peat soils at higher altitude are sinks for arsenic, and become sources if the peat is mineralized or eroded. The topic of upland organic soils acting as sinks and sources of arsenic is receiving more research attention<sup>34</sup>, and could be important on a regional scale as a source of arsenic to downstream sediments.

1 In large catchment areas of continental importance, such as the deltas that form to  
2 the south and east of the Himalayas, plate tectonic-derived mechanical weathering is  
3 thought to be the most important source of arsenic. One theory is that the  
4 mechanical weathering caused by Pleistocene tectonic uplift in the Himalayas is the  
5 key to understanding why arsenic is so elevated in Holocene aquifers, such as those  
6 of SE Asia, and in the glacial tills of Europe and North America<sup>26</sup>. Mechanical  
7 weathering of bedrock exposes previously inaccessible mineral surfaces, and the finer  
8 grinding leads to enhanced surface areas for chemical and microbial weathering to  
9 take place, causing greater solubilisation of arsenic<sup>25, 26, 35</sup>. Chemical and microbial  
10 weathering can take place at or near the source, or in sediment sinks. For instance,  
11 bacteria isolated from Bay of Bengal aquifers can mobilize arsenic from apatite<sup>35</sup> (See  
12 Section 5). Invariably, the arsenic loadings into soil will be dependent on arsenic in  
13 the bedrock, and the extent of the weathering of that bedrock-derived material along  
14 the route from source to sink. Soils with basalt bedrock had the lowest median  
15 arsenic content, while those with psammite, semipelite, and lithic arsenite bedrocks  
16 had the highest. Interpretation of such fine-scale mapping can ultimately lead to  
17 predictions of soil arsenic concentrations where detailed maps are not available.  
18 Combined with an understanding of soil chemistry, this will enhance the ability to  
19 predict elevated concentrations of arsenic in crops<sup>36</sup>.

### 21 **3. Anthropogenic sources of arsenic to soil**

22 Many anthropogenic activities have increased soil arsenic concentrations above the  
23 natural, background levels mentioned in Section 2 above, and they have the potential

1 to increase the arsenic concentration in food. This is especially the case in the USA  
2 where the widespread use of arsenic-based herbicides, pesticides and livestock  
3 antibiotics throughout the 20<sup>th</sup> century has ultimately increased the arsenic  
4 concentrations of current productive USA agricultural soils<sup>37-39</sup>.

### 6 *3.1. Base and precious metal mining*

7 The dominant mineral source of arsenic is thought to be pyrite (iron sulfide,  $\text{FeS}_2$ )<sup>40</sup>,  
8 an economically important ore deposit. High arsenic concentrations are found in  
9 many oxide minerals and hydrous metal oxides, either part of their structure or as  
10 sorbed and occluded species<sup>41</sup>. Iron oxides accumulate arsenic up to concentrations  
11 of several weight percent (1 weight % being equivalent to 10,000 mg/kg), and  
12 arsenic tends to bind to iron (III) (hydr)oxides whenever they are present. Arsenic is  
13 found predominantly as arsenopyrite but also can occur as orpiment (arsenic  
14 trisulphide  $\text{As}_2\text{S}_3$ ), realgar ( $\alpha\text{-As}_4\text{S}_4$ ) and other arsenic sulfide minerals<sup>42, 43</sup>. Arsenic is a  
15 byproduct of most mining operations and is present at high concentrations in the  
16 mine waste, and, because arsenic sulfides are particularly prone to oxidation in  
17 surface environments, in mining wastewaters<sup>42, 44</sup>. Arsenic can constitute 1% or more  
18 of the ore and solid waste, and wastewaters and impacted streams often contain  
19 dissolved arsenic concentrations ranging from 0.01 to over 10 mg/L. Because mining  
20 and smelting operations are localized, arsenic contamination of soils exists around  
21 the mine site with the concentration decreasing with distance from the source.  
22 Windblown dispersion of fine particulate material is a particular problem, spreading  
23 contamination greater distances from the mine site. This fine material - which is not

1 completely removed by washing<sup>16</sup> - can directly contaminate plant material;  
2 especially leafy material with high surface area. This presents a tangible risk to  
3 residents and home gardeners in the vicinity of areas with significant surface soil  
4 arsenic contamination. A comparison of arsenic concentrations in vegetables grown  
5 in SW England (the site of historic mining activities) with those from a pristine site in  
6 NW Scotland found a generally good correlation between total plant arsenic and soil  
7 arsenic concentrations. Increased arsenic concentrations were measured in produce  
8 from SW England where soil arsenic concentrations ranged from 120 – 1130 mg/kg.  
9 Arsenic concentrations were high in leafy greens (kale, spinach, lettuce) and some  
10 unpeeled vegetables (potatoes, swedes, carrots) were higher than when peeled,  
11 which, in both cases, points to contamination from windblown soil particles and soil  
12 adhesion to below ground biomass, rather than from root uptake. In this particular  
13 study, the majority of arsenic was present as the inorganic form<sup>16</sup>. Similar results  
14 were obtained from home gardens near the Iron King Mine Superfund Site in Arizona,  
15 USA<sup>45, 46</sup>. Here the tailings had arsenic concentrations of 3,710 mg/kg and residential  
16 soil sampled adjacent to the site ranged from 120 – 633 mg/kg. Edible plant tissue  
17 concentrations ranged from < 0.01 – 1.96 mg/kg (plant concentrations are expressed  
18 as dry weight throughout), and were generally positively correlated with soil arsenic  
19 concentrations. Leafy and high surface area vegetables such as lettuce, kale, broccoli  
20 and cabbage accumulated higher arsenic concentrations than beans, tomatoes,  
21 cucumbers and peppers. Arsenic in mine-affected vineyard soils in Italy ranged from  
22 4 – 283 mg/kg and positive correlations were observed between soil concentrations  
23 and arsenic levels in vine leaves and grapes, however, levels in wine were low (< 1.62



1  $\mu\text{g/L}$ )<sup>47</sup>. In the Hunan province, China, the high levels of inorganic arsenic in rice  
2 have been traced to mining activities in the area<sup>48-50</sup>.

### 3 4 *3.2. Coal combustion for energy*

5 The concentration of arsenic in USA coal ranges from 1 – 71 mg/kg with an average  
6 concentration of 24 mg/kg<sup>51</sup>. Fly ash, the major byproduct of the coal combustion  
7 process, consists of fine particles that are driven out with the flue gases, and is a  
8 major source of arsenic to the wider environment. Coal ash is one of the most  
9 abundant of industrial wastes; close to 130 million tons<sup>52</sup> of coal fly was generated in  
10 the USA in 2014, with 100 million tons estimated from the European Union in 2011<sup>53</sup>.  
11 Arsenic concentrates in the fly ash during combustion of coal for energy; the median  
12 arsenic concentration in USA fly ash is 71 mg/kg<sup>54</sup>. Fly ash is often sluiced into  
13 settling basins, and because arsenic in fresh ash is quite soluble, wastewater arsenic  
14 concentrations can consequently be quite high. Arsenic can build up in the  
15 sediments of coal fly ash settling basins and reach concentrations of over 1000  
16 mg/kg. Catastrophic failures of these setting basins have caused severe  
17 environmental problems and contaminated surface waters with arsenic<sup>55</sup>. There is a  
18 well-founded concern that arsenic from coal combustion wastes can contaminate soil  
19 and enter the food supply. The use of coal fly ash as a soil amendment can lead to  
20 elevated arsenic concentrations in crops (as well as boron, selenium and  
21 molybdenum), although its lack of soil macronutrients and the potential for arsenic  
22 toxicity prevents the sole application of coal fly ash as a soil amendment<sup>56, 57</sup>.

23 Formulating ash/organic waste mixtures that conform to USEPA regulations for total

arsenic application and meet soil and plant fertility requirements has been shown to safe and effective for agronomic use<sup>58</sup>.

### *3.3. Pesticides*

Perhaps the largest anthropogenic input of arsenic to agricultural soils in the USA is from the agricultural use of arsenic-based pesticides and herbicides for most of the 20<sup>th</sup> century. Calcium arsenate and lead arsenate were used extensively up to the 1950s, mostly on orchard soils to combat the codling moth. At peak, 132,000 metric tons of each pesticide compound was applied annually between 1930-1940<sup>37</sup>. In addition to apples, inorganic arsenic pesticides were used on a range of crops including essentially all fruit trees, vine berries, sweet potatoes, white potatoes, most vegetables and cotton<sup>37</sup>. Both lead and arsenate have long residence times in soils and high concentrations (often >100 mg/kg) of these two elements have been reported in old orchard soils in Washington<sup>59</sup>, North Carolina<sup>60</sup>, New Hampshire<sup>61</sup>, New Jersey<sup>62</sup> and Virginia<sup>63</sup>. There is some evidence of greater mobility for arsenic (than lead)<sup>61, 64</sup>, and retention of both elements depends on soil type and other environmental factors but most of this legacy contaminant remains in the soil<sup>62</sup>. Use of lead arsenate decreased after 1950s and was finally banned in 1988. The organic arsenic compounds dimethylarsinic acid (DMA) and monomethylarsonic acid (MMA) were used as pesticides on cotton and herbicides for golf courses and right-of-ways until they too were withdrawn from use in 2013. High levels of MMA were reported in transient surface waters adjacent to a crop sprayer operation<sup>65</sup>. Legacy soil arsenic contamination resulting from organic arsenical pesticides plays a major role in

1 straighthead disease of rice<sup>66</sup> (See Section 6.1). It may be that arsenical pesticides  
2 have leached to groundwater, as has been suggested for the Texas High Plains  
3 Aquifer<sup>67</sup>, although a study of the Ogallala aquifer in the High Plains in Texas found  
4 no evidence of anthropogenic arsenic in the groundwater<sup>68</sup>. Similarly, there was no  
5 relationship between groundwater arsenic and past (inorganic) arsenic pesticide  
6 usage in a comprehensive study of New Hampshire groundwater sources<sup>69</sup>. About  
7 10%, depending on soil substrate, of monosodium methyl arsenate applied to sandy  
8 soils (simulated golf course greens) leached into percolating water. Demethylation  
9 and methylation occurred because both inorganic arsenic species and DMA were also  
10 detected in the percolating water<sup>70</sup>. As with mining-impacted soils, plants grown on  
11 soils that are high in arsenic from arsenical pesticide contamination take up higher  
12 levels of arsenic into their edible tissues, observed for example in potatoes<sup>71</sup>, carrots<sup>72</sup>  
13 and leafy green vegetables<sup>73, 74</sup>.

14  
15 Former pesticide application has been suggested be a factor in the presence of  
16 higher levels of total arsenic found in rice grown in the south-central regions of the  
17 USA<sup>75, 76</sup> compared to other areas of the USA and to other countries, such as  
18 Bangladesh<sup>77</sup>. Evidence on varietal differences in arsenic uptake, speciation and  
19 distribution within rice grain (See also Section 6) strongly suggest that soil arsenic  
20 concentration is not the sole, nor particularly the main driver of this phenomenon.  
21 Factors likely to be influential include the differences in the soil microbial community  
22 composition between geographical regions that affect arsenic methylation,  
23 considering that plants themselves cannot methylate arsenic<sup>78</sup>.

1

### 2 *3.4. Wood preservatives*

3 Chromated copper arsenate (CCA) is used as a wood preservative and was  
4 extensively used on decking and other residential usages until a voluntary  
5 manufacturer withdrawal in 2003. The primary health concern is for young children in  
6 direct contact with CCA-treated wood, but localized leaching of arsenic (as well as  
7 chromium and copper) also occurs to surrounding soil. Soil arsenic concentrations of  
8 37 – 250 mg/kg have been reported for soils sampled near CCA-treated utility poles  
9 (N=12)<sup>79</sup> and mean arsenic concentrations for soils collected below decks and  
10 footbridges in Florida, USA was reported to be 28.5 mg/kg compared with a control  
11 concentration of 1.3 mg/kg (N= 65)<sup>80</sup>. Arsenic from CCA contaminated soils appears  
12 to be more bioavailable than from other anthropogenic sources to soil<sup>81</sup>.

13

### 14 *3.4. Organic manures*

15 Land application of sewage sludge (biosolids) in the USA is regulated by  
16 Environmental Protection Agency Part 503 Biosolids rule; which set the maximum  
17 arsenic concentration of the sludge at 75 mg/kg, an annual pollutant-loading rate of  
18 2.0 kilograms arsenic/hectare (kg/ha) and a cumulative pollution-loading rate of 41  
19 kg/ha over the lifetime of applications. Assuming a plow layer of 17 cm, application  
20 at the maximum annual rate implies an approximate 1.2 mg/kg maximum increase in  
21 soil arsenic, while the cumulative maximum loading rate could increase soil arsenic  
22 concentrations by approximately 24 mg/kg over the lifetime of application and  
23 assuming no loss from the soil profile. This cumulative loading rate of 24 mg/kg is

significant when considered against an average soil arsenic concentration of 5 mg/kg (See Section 2), however, relative to mine-impacted or inorganic arsenic pesticide impacted soils where arsenic concentrations are frequently > 100 mg/kg, it is of lesser concern. Also, sewage sludge is often high in aluminum and iron oxide phases, used in the flocculation process, which are efficient scavengers of inorganic arsenic thus lowering the arsenic bioavailability<sup>82</sup>.

Arsenic occurs in animal wastes primarily because of the former use of arsenic antibiotics in poultry and turkey feed; until 2015 four drugs, roxarsone, p-arsanilic acid, carbarsone and nitrosone, were regulated for use, with roxarsone being the most prevalent. As of 2016 all four of these compounds have all been withdrawn from use<sup>83</sup>. All four are organic arsenic compounds with an arsenate functional group attached to a benzene ring, and differ by other substituents on the ring. The compounds are not readily adsorbed or metabolized and so occur at concentrations up to 40 mg/kg in animal manures. This provides three points of entry to the human food chain; directly through arsenic in chicken and turkey meat<sup>38, 39, 84</sup>, from plant uptake after land application of manure, and runoff to surface water or groundwater. A number of studies have shown that these organic arsenic compounds can be degraded by both photolytic<sup>85</sup> and microbial<sup>86</sup> processes and that this degradation happens both during composting of stockpiled litter<sup>87, 88</sup> and after land application<sup>89, 90</sup>. Long term application of poultry litter to Upper Coastal Plain soils increased soil arsenic concentrations from 2.7 to 8.4 mg/kg after 25 years of application<sup>91</sup>. Similar increases have been reported for other southern states of the USA<sup>92</sup>. There is

evidence to suggest that other soluble constituents of the litter, for example phosphate and dissolved organic carbon compounds, facilitate arsenic solubility and leaching<sup>89, 92, 93</sup>.

### *3.5. Seaweed fertilizers*

Seaweeds can contain far higher concentrations of arsenic than crop plants: up to 100 mg/kg (Taylor et al, this issue). In most cases the arsenic is present as arsenosugars, which are of low toxicity to humans (Taylor et al, this issue). However, as in the case of poultry litter, these compounds degrade (ultimately) to inorganic arsenic after land application<sup>94</sup>. Although seaweeds are a 'niche' soil amendment, their use agriculture is increasing and has been adopted by many organic farms as a soil fertilizer as well as a feed additive in organic dairy farming<sup>95</sup>.

## **4. Biogeochemical cycling within terrestrial agronomic ecosystems**

Arsenic cycles within the soil surface and near-surface environment<sup>96</sup>, influenced by mineralogy, abiotic factors such as pH and redox potential ( $E_H$ ), and biotic factors such as microbially-mediated biomethylation.

### *4.1. Redox regulation*

The most important biogeochemical step in the exposure of humans to arsenic is its release from soils and sediments into pore water; the water contained within soil pores and/or rock<sup>40</sup>. With the exception of extreme pH conditions (<4 or >9), or high concentrations of competing ions (e.g. phosphate, silicic acid or silicate<sup>97</sup>) the release

of arsenic from its strong bonds with soil particles depends upon redox potential ( $E_H$ ); the extent of aeration of the soil<sup>40</sup>. As  $E_H$  falls, electron acceptors are depleted and anoxic conditions develop, causing iron oxides and oxyhydroxides to be reduced and dissolve, releasing sorbed arsenic into the soil solution<sup>98</sup> where it can be taken up by plant roots, or leached into groundwater.

Agronomic cropping systems can be divided with respect to arsenic mobilization on the basis of their redox status. Dominant biogeochemical processes influencing aerobic systems, specifically cereals, upland rice, fruit tree orchards, and community gardens, differ from those that dominate in anaerobic systems, predominantly in flooded rice paddies. In aerobic soils, arsenic speciation is predominantly arsenate (arsenic (V)), and is tightly bound to soil particles. Under anaerobic or flooded conditions, arsenic is reduced, and arsenite (arsenic (III)) is the dominant species<sup>40</sup>. Arsenite is less stably bound to aluminum hydroxides and aluminosilicate clay minerals in the soil than arsenate, for which they exhibit a much stronger binding preference<sup>40</sup>. With few exceptions (such as under conditions of sulfur release), transition of arsenic speciation from arsenate to arsenite is the most influential factor to arsenic bioavailability; and it is under anaerobic conditions where arsenic becomes an imminent human health concern. Influential biogeochemical processes in aerobic systems are ageing and accumulation of arsenic in soil, and in anaerobic systems reductive dissolution of iron-bearing minerals is the dominant process.

#### *4.2. Biotransformation to methylated and volatile species*

1 Volatile arsenicals are arsenic species with a boiling point below 150°C; the most  
2 volatile of which is arsine gas ( $\text{AsH}_3$ ), followed by monomethylarsine ( $\text{MeAsH}_2$ ),  
3 dimethylarsine ( $\text{Me}_2\text{AsH}_2$ ) and finally completely methylated trimethylarsine (TMA).  
4 Volatile arsenic species can be formed either biotically – by fungi, bacteria and  
5 algae<sup>27, 99, 100</sup> or abiotically<sup>99</sup>. In natural systems arsines readily react with oxygen to  
6 form non-volatile oxidation products, with  $\text{AsH}_3$  most rapidly oxidized and  
7 challenging to detect in environmental samples. Oxidation of the arsine gases to  
8 inorganic arsenic species completes the arsenic cycle, with arsenic returned to the  
9 soil by rain or dry deposition<sup>101</sup>.

10 Arsenic methylation in soils increases with decreasing redox potential<sup>102</sup>, and addition  
11 of organic matter. Increased arsenic volatilization was measured in soil after the  
12 addition of rice straw<sup>103</sup>, and animal waste products<sup>104</sup>. Inoculation of fungi  
13 (*Penicillium* and *Ulocladium* spp.) increased arsenic volatilization up to 8 fold in  
14 heavily contaminated and spiked soils<sup>105</sup>. Microbially mediated arsenic volatilization  
15 remains very inefficient, which hinders attempts to use it in soil remediation. Gaseous  
16 arsines are volatilized from arsenic contaminated soils into the atmosphere at very  
17 low rates: a microcosm study found 0.5 – 70  $\mu\text{g}$  of arsenic  $\text{kg}^{-1}$  soil  $\text{year}^{-1}$  was  
18 volatilized from a range of soils and a range of arsenic levels<sup>27</sup>, and field  
19 measurements of arsenic volatilization are 1-2 orders of magnitude lower than those  
20 made in laboratory mesocosms<sup>98</sup>. Genetic transformation of bacteria, using genes  
21 encoding for the protein product arsenite S-adenosyl methyltransferase (*arsM*) is an  
22 attempt to enhance arsenic methylation and volatilization. The *arsM* from  
23 *Rhodopseudomonas palustris* was expressed in *Sphingomonas desiccabilis* and



*Bacillus idriensis* grown in an aqueous system, resulting in a 10-fold increase in arsenic volatilization compared to the wild type strains. In a soil-based system, 2.2 – 4.5% of arsenic was removed via microbially-mediated volatilization over an incubation period of 30 days<sup>106</sup> (See also section 5.3).

#### *4.3. Changes in soil arsenic bioavailability due to ageing*

Although arsenic in aerobic soils has a lower bioavailability and presents less of an immediate concern for crop uptake, aerobic soil can accumulate arsenic from human inputs, retain them for long periods of time, and release them when redox conditions change (See Section 3.3). Human inputs of arsenic, as discussed in Section 3, are diverse; biosolids, sewage sludge, coal fly ash, poultry litter, industrial waste, arsenical pesticides and from irrigation with naturally arsenic-enriched groundwater. For aerobic soils, ageing – where binding stability of arsenic to soil particles increases over time, is a particularly important part of arsenic cycling. Factors controlling ageing of arsenic include soil type, organic matter content and arsenic species. Both inorganic and organic arsenic species are subject to ageing, with studies indicating a slow oxidation process from arsenite to arsenate over time<sup>107</sup>.

### **5. Rhizosphere processes**

Processes occurring in the rhizosphere (the boundary layer of soil under the influence of plant roots) dramatically influence arsenic concentrations and bioavailability because they involve local alterations in redox potential, pH and organic matter content. Rhizosphere acidification occurs during iron uptake by all plant species

1 during cation uptake and charge balance, when protons are released into the  
2 rhizosphere. Plants release anywhere from 10 to 250 mg of carbon per gram of root  
3 tissue into the rhizosphere; about 10-40% of their total photosynthetically fixed  
4 carbon<sup>108</sup>, making the rhizosphere particularly rich in organic carbon compared to  
5 bulk soil, which in turn exerts an influence on arsenic solubility by stimulating  
6 microbially-mediated reductive dissolution of soil minerals. Large differences have  
7 been found in the arsenic concentration of rhizosphere soils compared with bulk  
8 soils in highly arsenic-contaminated areas, with higher concentrations of arsenic in  
9 rhizosphere soils compared to bulk soils<sup>109</sup>.

10

11 In anaerobic soils, the iron plaque that develops on the submerged stem and roots  
12 of rice plants dominates rhizosphere dynamics of arsenic. In flooded environments  
13 such as paddy fields, plants oxygenate the rhizosphere through specialized tissues  
14 called aerenchyma, which are found in many aquatic plants and emergent  
15 macrophytes such as rice. This radial oxygen loss creates an oxidized layer around  
16 plant tissue that stimulates aerobic microbial activity and the oxidation of iron, which  
17 precipitates and forms a visible iron plaque on the root surface <sup>110-114</sup>. Formation of  
18 an iron (oxyhydr)oxide plaque on root surfaces can alter the uptake of arsenic by rice,  
19 acting as a sorbent for excess nutrients such as ferrous iron (reduced iron) as well as  
20 arsenic and aluminum<sup>115</sup>. Rates of oxygen loss influence iron plaque formation <sup>115</sup>,  
21 and vary between rice cultivars<sup>116, 117</sup>. Studies conducted over the last forty years are  
22 inconsistent on whether iron plaque prevents or enhances arsenic uptake by plants<sup>111</sup>,  
23 and the hypothesis that arsenic influences the quality and amount of the iron

1 plaque<sup>113</sup>. Profound differences in mineral composition and quantity of laboratory-  
2 created iron plaques has been demonstrated experimentally<sup>111</sup>, which may have  
3 contributed to these inconsistencies.

#### 4 5 *5.1. Microbial activity*

6 Microbes directly and indirectly influence arsenic speciation in rhizosphere soil, and  
7 are widely considered to play a key role in arsenic biogeochemistry<sup>118</sup>. Under certain  
8 nutrient-limited conditions, microbes actively weather minerals to access nutrients for  
9 cellular growth, which releases arsenic<sup>35</sup>, as well as creating abiotic conditions that  
10 induce changes in arsenic speciation via production of organic acids, polysaccharides  
11 and ligands. Soil microorganisms can strongly affect soil redox, regulating arsenic  
12 release into pore water<sup>119</sup>. A number of strains of bacteria have also been shown to  
13 contribute to the formation of arsenic minerals by using arsenic as a terminal  
14 electron acceptor, such as *Desulfosporosinus auripigmentum*<sup>120</sup>, *Desulfovibrio* strain  
15 Ben-RB<sup>121</sup>, *Shewanella oneidensis*<sup>122</sup> and *S. putrefaciens* CN32<sup>123</sup>. These  
16 microorganisms also differ in their capabilities for liberating arsenic from specific  
17 arsenate-bearing minerals<sup>119</sup>.

18  
19 Microbial transformation can mobilize arsenic by converting inorganic to organic  
20 forms, including MMA and DMA<sup>124, 125</sup>. Plants translocate organic arsenicals from  
21 roots to the (frequently edible) above-ground parts more efficiently than inorganic  
22 arsenic<sup>126-128</sup> (See Section 6), therefore microbial transformation to organic arsenicals  
23 can increase human dietary exposure.

1

2 Plants, green algae and microbes can all enzymatically transform arsenic species<sup>124, 129</sup>,  
3 but methylated forms of arsenic detected in plants are a product of rhizosphere  
4 bacteria; plants cannot methylate arsenic<sup>78, 124, 130, 131</sup>. The genomes of more than 85  
5 arsenic-metabolizing archaea and bacteria have been sequenced for genes involved  
6 in arsenic metabolism<sup>132</sup>. In bacteria, archaea and fungi, arsenic methylation is  
7 catalyzed by homologs of *arsM*, (See Section 4)<sup>124</sup>. Resistance to arsenite and  
8 arsenate exists in nearly all microbes, which also confers the ability to transform  
9 arsenate into volatile arsine gases<sup>133</sup>, a particularly effective way of removing arsenic.

10

11 Profiling the transcriptome, proteome and metabolome of arsenic contaminated soils  
12 offers way of understanding microbially-mediated rhizosphere arsenic processes<sup>132</sup>.  
13 This approach measures the presence and expression of specific genes, rather than  
14 attempting to isolate and study the microbes that carry them, 98% of which – it is  
15 estimated - do not grow in culture<sup>134</sup>. Microbially mediated arsenic metabolic  
16 processes that play a major role in arsenic cycling in agronomic systems include  
17 arsenite oxidation (via the *aio* genes), arsenate respiration (via the *arr* genes),  
18 arsenate reduction (via the *ars* genes) and arsenite methylation (via the *arsM*  
19 genes)<sup>135</sup>. Interested readers are referred to the recent excellent work of Andres and  
20 Bertin<sup>132</sup> for a comprehensive review of this subject. Microbially mediated redox  
21 processes strongly influence arsenic uptake in rice, involving *aioA*, *arsC* and *arrA*<sup>124</sup>,  
22 with pH emerging as an important factor in the distribution of microbes in paddy  
23 soils. Testing a variety of soils has shown that bacteria possessing the *arsM* gene for

1 methylating arsenic are widespread and phylogenetically diverse, and even in paddy  
2 soils with low concentrations of arsenic, genes for arsenic metabolism are  
3 abundant<sup>124</sup>.

## 5 **6. Arsenic and crop plants**

6 Much of our understanding about the physiological mechanisms of arsenic uptake in  
7 plants comes from the study of a limited number of species. Called *model plant*  
8 species, they are extensively studied, well described, easy to grow, and the results  
9 can be compared between studies. The understanding is that the information gained  
10 from studying model plants is applicable to other plant species. From a genetic  
11 perspective, orthologous genes exist in different plant species that have evolved from  
12 a common ancestral gene, and they usually retain the same function. Characterization  
13 of arsenic-related genes in a model plant strongly suggests that they exist and  
14 perform similar functions in other species. Caveats to this are their levels of  
15 expression, which makes some plants more adept at accumulating arsenic than  
16 others. In this section, much of the knowledge gained on arsenic uptake and  
17 metabolism of plants comes from the study of mouse-eared pennycress (also called  
18 thale cress or rockcress) (*Arabidopsis thaliana* Heynh.) and rice (*Oryza sativa* L.);  
19 model plants with fully sequenced genomes. These species represent dicotyledonous  
20 (e.g. flowers, vegetables, deciduous trees) and monocotyledonous plant species (e.g.  
21 grasses, palm trees) respectively, thereby representing much of the edible crop  
22 species. An exception to this is the study of the arsenic hyperaccumulating fern

(Chinese Brake fern, *Pteris vittata*), a seedless plant that is able to accumulate up to 22,630 mg/kg (dry weight) arsenic in its fronds<sup>136</sup>.

#### 6.1. Phytotoxicity of arsenicals

Arsenic is toxic to plants<sup>137</sup>. Despite lower acute human toxicity of the organic arsenicals (median lethal dose is 700-1,600 mg/kg and 700-2,600 mg/kg for MMA and DMA respectively compared to 10-20 mg/kg for inorganic forms)<sup>138</sup> no one form of arsenic is consistently more toxic to plants<sup>139</sup>. Soybean yields are affected when tissue arsenic levels exceed 1 mg/kg, and 4 mg/kg limits cotton yields<sup>140</sup>, whereas in barley tissue concentrations of 20 mg/kg inhibited growth<sup>141</sup>. Higher yield-limiting arsenic levels have been recorded in rice: 20-100 mg/kg in above ground biomass, and 1000 mg/kg in root tissue<sup>142</sup>. By contrast, potatoes (*Solanum tuberosum* L.) suffered no growth inhibition in soils containing 290 mg/kg arsenic<sup>71</sup>. In some plants species, organic forms are more toxic than inorganic, for example in rice (order of toxicity: MMA > arsenite > arsenate = DMA)<sup>143</sup>, and in smooth cordgrass (*Spartina alterniflora* Loisel) (DMA = MMA > arsenite > arsenate)<sup>144</sup>.

Plants vary in their tolerance to arsenic, and the stress response differs for each arsenic species<sup>145-147</sup>. The chemical similarities between arsenate and phosphate means that arsenic can replace phosphate in biomolecules like ATP (adenosine triphosphate, a molecule used for intercellular energy transfer), with negative impacts on growth and metabolism<sup>148</sup>. In rice in particular, DMA and MMA induce straighthead disease (arsenic-associated straighthead disease), significantly lowering

yield of certain rice varieties<sup>66</sup>. Straighthead is a physiological disorder of rice characterized by sterile florets, which remain upright at maturity instead of bending over under the weight of the filled grain. The exact cause of straighthead is unknown, but consistent flooding, low soil pH, high iron availability and high organic matter content have all been implicated in naturally-occurring straighthead disease<sup>66</sup>.

Arsenic's suspected role in straighthead comes from observations of more frequent outbreaks in rice grown in soil where arsenical herbicides such as monosodium methanearsonate (MSMA) – used in cotton production in the USA – have been historically applied.

## *6.2. Arsenic uptake mechanisms*

In magnitude, plants take up arsenicals from the soil in the order arsenite > arsenate > DMA > MMA<sup>149, 150</sup>, with the various arsenic species entering via different root membrane transport proteins in the root plasma membrane that allow ions and molecules to cross with varying levels of selectivity, or target specificity. Similarities in chemical structure between arsenate and phosphate, and between arsenite and silicic acid, govern their entry into root cells. Arsenate enters root cells through phosphate transporters (the Phosphate Transporter 1 family of proteins; PHT1) in both the model plant *Arabidopsis thaliana*<sup>151, 152</sup> and in rice<sup>153-155</sup> (Figure 1). In rice, Low Silicon 1 (OsLsi1) and OsLsi2 are silicic acid transporters and arsenite, MMA<sup>V</sup>, and DMA<sup>V</sup> are among their unintended targets<sup>156, 157</sup>. These Nodulin 26-like Intrinsic Proteins (NIPs)<sup>158</sup>, which are members of the aquaporin water channel superfamily of proteins<sup>159</sup> embedded in the exodermal cell membranes of rice roots, move arsenic

1 from the soil into the vascular system for distribution to the stem and leaves. OsLsi2  
2 works in tandem with OsLsi1 to transport arsenite inward toward the xylem<sup>160, 161</sup>  
3 (vascular tissue that conducts water and dissolved nutrients up from the roots). The  
4 arsenic uptake specificity of OsLsi1 is arsenite >> MMA > DMA<sup>158</sup>. These bidirectional  
5 NIP transport proteins also efflux arsenite back in to the soil, but since OsLsi1  
6 effluxes only 15-20% of the arsenite in roots cells<sup>162</sup>, there may be other unidentified  
7 arsenite efflux transporters contributing to this process.

### 8 9 *6.3. Arsenic transport and metabolism in plants*

10 Transport of arsenite into the xylem for delivery to the shoot is less well  
11 characterized than its uptake from the soil. Arsenic is transported to the grain mainly  
12 via the phloem<sup>126-128</sup> (vascular tissue that conducts sugars and metabolic products  
13 from the leaves), by transporters in the nodes<sup>163</sup>, but their characterization is still in  
14 the early stages. Transporters for myo-inositol (Inositol Transporter 2 and 4); an  
15 important sugar for developing rice grains, also transport arsenite into the phloem  
16 companion cells<sup>164, 165</sup>. In Arabidopsis, INT2 or AtINT4 load about 45-64% arsenite  
17 into the grain<sup>166</sup>. The identity of transporters that move arsenite out of the phloem  
18 and into the grain are also unknown, but manipulating the target specificity of the  
19 INT genes might show promise in molecular genetic or plant breeding mitigation  
20 efforts as a way to prevent arsenite from reaching the grain.

21  
22 Despite having a lower affinity for transporters into the plant than the inorganic  
23 forms, organic arsenic species are more efficiently transported towards the shoot



1 than inorganic forms<sup>149, 150</sup> because they are not complexed by phytochelatins (PCs);  
2 sulphhydryl-rich glutathione (GSH) polymers<sup>167, 168</sup>. Likewise, in broad beans (*Vicia faba*  
3 L.) grown in a soil containing 90% inorganic arsenic, DMA and MMA were the  
4 dominant arsenic forms in the bean (68%)<sup>169</sup>. In root vegetables, carrot (*Daucus*  
5 *carota* L.) and beet (*Beta vulgaris* L.) grown on arsenic-contaminated soils, arsenic  
6 forms were predominantly inorganic, but for beets in particular were not readily  
7 identified using the typical standards (arsenate, arsenite, MMA and DMA)<sup>170</sup>.  
8



1 The arsenic species composition of rice grain is influenced by the arsenic transport  
2 rate of the particular cultivar<sup>173, 174</sup>. Rice cultivars currently grown in the USA have an  
3 arsenic speciation split approximately equally between inorganic arsenic and DMA,  
4 while cultivars grown in Bangladesh contain mostly inorganic arsenic<sup>174</sup>. While lower  
5 inorganic arsenic in rice grain seems favorable for avoiding human health effects, the  
6 assumed safety of DMA is contentious<sup>175</sup>, being based on acute toxicity data, and not  
7 on genotoxicity or carcinogenicity, which are equally relevant in long term safety  
8 considerations.

9  
10 Arsenic detoxification inside cells uses a multi-step process beginning with reduction  
11 of arsenate to arsenite using an arsenate reductase enzyme<sup>176, 177</sup>. In Arabidopsis, the  
12 protein High Arsenic Content1 (HAC1; also called Arsenate Reductase QTL1; ARQ1)  
13 reduces arsenate<sup>177</sup>. Even though arsenite is more toxic than arsenate<sup>158, 178, 179</sup>, it is  
14 hypothesized that ancestral organisms to plants were exposed almost exclusively to  
15 arsenite before atmospheric oxygen enabled arsenate formation<sup>180</sup>, and this  
16 mechanism persisted through natural selection. Arsenite is then complexed by PCs,  
17 and transported in to the vacuole<sup>167</sup> via ATP Binding Cassette tranporters<sup>181, 182</sup>. This  
18 process depletes glutathione availability, rendering the plant more susceptible to  
19 other oxidative stresses, which inhibits photosynthesis, pigment production, and the  
20 integrity of cell membranes<sup>183-186</sup>.

## 21 22 **7. Limiting arsenic uptake by crops**

### 23 *7.1. Water management*

1 Although the traditional method for cultivating rice involves flooding leveled, tilled  
2 fields before or shortly after planting germinated seedlings, flooded soil is not a  
3 biological requirement of rice plants. Flooding is used for weed and vermin control,  
4 for mobilization of key nutrients such as iron, phosphate and zinc, and importantly,  
5 flooding discourages the buildup of root nematodes over multiple years of rice  
6 growth. As mentioned earlier, flooded conditions mobilize soil-bound arsenic  
7 through reductive dissolution of Fe (oxyhydr)oxides, and the reduction of arsenate to  
8 the more mobile arsenite<sup>187</sup>. Water management strategies that involve periods of  
9 oxic soil conditions can decrease arsenic uptake in rice by limiting dissolution of  
10 arsenic. Rice grown in non-flooded or aerobic conditions has a lower yield than  
11 intermittently or constantly flooded rice<sup>188-190</sup>. Intermittent flooding (flooding  
12 maintained until full tillering, followed by intermittent irrigation) is a promising  
13 management technique to reduce arsenic levels, and can potentially produce higher  
14 grain yields than either non-flooded or constantly flooded conditions<sup>191</sup>. However,  
15 oxic conditions increase cadmium concentrations in the grain when grown in acidic  
16 soils<sup>191-193</sup>, and cadmium is also a highly toxic metal. The observed increases in  
17 cadmium were also a cultivar-specific trait, but the increase in cadmium uptake  
18 between rice grown under aerobic conditions were approximately an order of  
19 magnitude greater than their flooded counterparts. Pot experiments suggest that  
20 water management strategies implemented during the heading period of rice growth  
21 (when the rice panicle has emerged from the stem and is fully visible, just before  
22 flowering) can regulate both arsenic and cadmium concentration in the grain<sup>190, 192</sup>.

23

## 1    *7.2. Amendment and fertilization practices*

2    Soil amendment involves incorporating substances into the plow layer that either add  
3    missing nutrients, reduce the bioavailability of existing potentially toxic substances (to  
4    prevent crop uptake), or both. Soil amendments that have shown potential in  
5    reducing arsenic uptake by plants include iron-, and silica-based additives. The use of  
6    iron-based amendments increases in the concentration of free iron oxide in the soil,  
7    retarding the release of arsenite from the solid phase into soil solution, (mentioned  
8    in Section 4.1 and discussed in Section 5), whereas silica fertilization inhibits arsenic  
9    uptake by competitive inhibition at the plant root surface while adding an essential  
10   nutrient.

11   Zero valent iron powder (90% iron) and iron oxide (56% iron) incorporation  
12   prevented uptake of arsenic in to the grain of rice grown on soil containing 39.5  
13   mg/kg total arsenic by approximately 45%, and corresponded with a reduction in  
14   bioavailable arsenic in the soil<sup>194</sup>. Amendment with iron oxides (at a rate of 2%) was  
15   also more effective at reducing grain arsenic than phosphate amendment<sup>195</sup>.

16   Amendments have also been used in combination with water management strategies  
17   to try and reduce both arsenic and cadmium concentrations in rice simultaneously<sup>193</sup>,  
18   without success. Reduction of arsenic in the grain was achieved with iron oxide  
19   addition and constant flooding, whereas cadmium reduction was achieved with  
20   converter furnace slag addition and rain water management (no irrigation after  
21   midseason drainage until harvest).

22   A combination of ethylenediaminetetraceetic acid ferric sodium salt (iron EDTA) and  
23   calcium peroxide was effective for reducing arsenic uptake by vegetable crops

1 (lettuce, Chinese cabbage and radish) from soils containing 14 mg/kg total arsenic<sup>196</sup>,  
2 again by increasing amorphous aluminum and iron oxides. It is likely that this this  
3 level of arsenic contamination would be deemed too high for commercial vegetable  
4 production, so these amendments may only be feasible for use in private vegetable  
5 gardens. Questions remain about whether iron oxide amendment application only  
6 temporarily reduces arsenic bioavailability<sup>197</sup>. In addition, the suitability for arsenic  
7 immobilization is highest at the lower soil pH range, and is strongly affected by soil  
8 phosphorus concentration, which strongly competes with arsenic.

9 Rice plants take up high concentrations of silica, constituting up to 10% of dry  
10 matter in the straw and husk of the plant<sup>198</sup>. As mentioned earlier (Section 6) the  
11 silicon membrane transporter (Lsi1) is the main route of arsenite entry in to rice root  
12 cells, and provision of silicon causes competitive inhibition of arsenite uptake.

13 Increasing silicon availability in the soil also reduces the expression of the Lsi1  
14 transporter in the plant, which further decreases the potential for arsenic uptake.

15 Fertilization of rice paddy soils with silicon is a potential mitigation strategy for  
16 preventing or reducing arsenic uptake by rice through competitive inhibition of  
17 arsenite uptake<sup>199</sup>. The use of synthetic silicon fertilizers, such as calcium silicate or  
18 silica gel is prohibitively expensive for smallholder farmers in developing countries,  
19 however reusing the silicon-rich parts of the rice plant that remain after harvesting  
20 and grain processing may provide a sustainable solution that also addresses the  
21 ongoing issue of silicon depletion of the soil<sup>198</sup>. Soil incorporation of fresh rice husks,  
22 or the ash that remains after burning the husk and straw for energy (which is a

common practice for smallholder farmers), can provide silicon without increasing methane production and decreases either total or inorganic arsenic in rice grain<sup>200</sup>. Despite the potential of soil amendment with iron oxides or silica to reduce arsenic bioavailability or prevent plant uptake of arsenic, the high cost of these amendments inevitably prevents their use, especially by smallholder farmers. Large rice producers in the US or Europe have not so far adopted widespread use of these soil amendments to reduce rice grain arsenic concentrations. It is also reasonable to assume that use of expensive soil amendments would drive up the cost of rice. Their lack of use may also be attributable to the fact that iron amendments are essentially untested in a diverse range of large-scale agricultural settings and their performance will vary between soil types. In non-rice agricultural systems arsenic is tightly bound to the solid phase; significant crop uptake from oxidized soil is likely to be a result of extreme contamination, in which case effective mitigation is restricted to redirecting land use away from edible crops. In systems subject to periodic flooding, improving drainage remains the best mitigation strategy.

### *7.3. Mitigation using plant breeding approaches*

The development of crops that accumulate high levels of arsenic and yet remain healthy, while preventing arsenic from reaching the edible grain is thought to hold great potential as a strategy for reducing human exposure to dietary arsenic. The use of molecular genetics techniques such as alterations in gene expression characteristics, gene editing to alter target specificity, or alternately, using traditional plant breeding techniques are both tangible approaches. Both use knowledge of the

1 arsenic uptake and tolerance characteristics of plants to develop varieties with  
2 desired characteristics. These characteristics include lower arsenic uptake<sup>201</sup>, higher  
3 arsenite efflux<sup>202</sup> and increased vacuolar arsenic sequestration<sup>203</sup>. For instance, many  
4 rice cultivars have now been screened to identify those that accumulate lower levels  
5 of arsenic in their grain and efforts are underway to identify the genes underlying  
6 this trait<sup>201, 204</sup>. Overexpressing Arabidopsis ABC-type transporters that sequester  
7 arsenite-PC complexes in the cell vacuole results in plants able to grow in otherwise  
8 toxic concentrations of arsenic<sup>182</sup>. Conversely, knocking out the function of the  
9 related rice ABC transporter OsABCC1 results in higher levels of grain arsenic. The  
10 OsABCC1 transporter limits arsenic transport to grains by sequestering arsenic in the  
11 vacuoles of the phloem companion cells directly connected to the grain. By  
12 combining what we have learned from the overexpression studies in Arabidopsis and  
13 the loss-of-function study in rice, overexpression of OsABCC1 can be used as a  
14 strategy to breed arsenic tolerance and low-arsenic accumulating rice cultivars.  
15 Another promising strategy is based on expressing the arsenate efflux transporter  
16 from yeast (*Saccharomyces cerevisiae*) in rice, which can reduce arsenic accumulation  
17 in brown rice by 20%. A less successful idea to methylate sodium arsenite to DMA by  
18 expressing an algal arsM gene in Arabidopsis resulted in lethal phytotoxicity<sup>205</sup>,  
19 suggesting that arsenic methylation in plants can only be an effective detoxification  
20 strategy if volatile arsines are the end point of the methylation.

## 22 8. Conclusions



1 The discovery of arsenic in staple foods, beverages and other products has increased  
2 awareness and stimulated research on the sources and the processes involved. The  
3 information brought together here illustrates the numerous geochemical and  
4 biological processes that influence the movement of arsenic into the food supply.  
5 It is clear there must be strategies for preventing arsenic exposure, that operate in  
6 both the short term – to protect consumers from existing contamination – and in the  
7 long term, to prevent further contamination. This requires government regulation on  
8 the permissible levels of arsenic food, with lower levels for infant foods (see  
9 Nachman et al, this issue), which must work in tandem with long term goals to  
10 address arsenic in agricultural soils, actively prevent further inputs and identify  
11 contaminated areas for mitigation. Our recommendations are that the information in  
12 this review is used to inform a reconsideration and a unification of regulations on the  
13 action levels of agricultural soil arsenic, which in the USA for example, exist only at  
14 the state level, vary widely from state to state, and have no formal channels of  
15 enforcement. We recommend that educating the community and garnering their  
16 support and involvement for lowering exposure to arsenic through food is an  
17 approach already shown to hold enormous potential. Direct involvement of the  
18 commercial rice growing community in research and development of arsenic  
19 mitigation strategies and amendments is needed. Much effort has been given to  
20 short-term, greenhouse-scale testing of amendment formulations that will ultimately  
21 be too expensive, impractical, or ineffective in the long term. Community-based  
22 participatory research should extend to the agricultural community, leading to  
23 partnerships that will make longer-term field-scale testing of mitigation strategies

1 accessible. Feasibility should be the first consideration in arsenic mitigation research.  
2 Community outreach efforts targeted to commercial growers or the home gardener  
3 specifically must raise awareness of the significance and potential impacts of former  
4 land uses, encouraging testing for the presence of arsenic in the soil and educating  
5 growers on crops shown to accumulate arsenic in their edible parts. Information  
6 gathering on former arsenic input into the soil from pesticides and from proximity to  
7 various waste sites is of paramount importance, and will allow monitoring and  
8 mitigation to be targeted to where it is needed most. Currently there is no readily  
9 available source of soil arsenic concentration information at a sufficient resolution to  
10 inform commercial producers or homeowners: this information needs to be  
11 accessible to everyone, everywhere. Going forward, management and remediation of  
12 arsenic contaminated soils is essential both for human health and food security, and  
13 innovative technologies are urgently needed that will expedite this process.  
14 Innovative solutions such as the use of rice husks to add silicon to the soil to offset  
15 arsenic uptake, and the use of cultivars with low-arsenic accumulating characteristics  
16 point the way forward for sustainable solutions.

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Figure

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