

# **Small-Scale Gold Mining Related Mercury Contamination in the Guianas: A Review**

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**Purpose of the report:**

Numerous studies over several decades have examined different aspects of the impact of mercury use in mining across the Guianas. Globally, the growing awareness of the negative impacts of mercury use, both in terms of environmental and human health, has culminated in the drafting and ratification of the Minamata Convention which obliges signatories to phase out mercury use. As a contribution to decision making the aim of this review is to explore the extent of small scale gold mining (SSGM) related mercury contamination in the Guianas. It will focus on mercury trade; the presence of mercury in the air, freshwater, soil and marine environments; entry into the food chain and exposure of the human population.

As the title suggests, the focus of this document is on mercury contamination related to SSGM. The authors recognise that the relative contribution of human and natural sources (anthropogenic vs lithogenic) of mercury to contamination in the Guiana Shield and wider Amazon is the source of some controversy. Data speaking to this issue (e.g. core samples and isotope analyses) are included in this review, however, our principal intention is to detail the current levels and extent of contamination resulting from mercury use in SSGM.

This review is a comprehensive compilation of all available to clearly show the extent and severity of the issue. To that end a systematic literature search of online journals has been carried out (search terms: mercury OR Hg [title/abstract] + Suriname OR Guyana OR Guiana OR Guianas OR Guianan [title/abstract]) in addition to a review of the WWF Guianas' database of documents on mercury contamination and reports provided by partner organisations.

## **Executive Summary:**

Gold mining in the Guianas has a history that stretches back more than 150 years. Several international mining companies have productive working concessions in the region but the industry is still dominated, certainly in terms of geographic extent, numbers of miners and environmental impacts, by the small-scale sector. Small scale is something of a misnomer because these miners frequently work on a semi-industrial scale employing heavy duty digging equipment and high pressure hoses. They also often use very large quantities of mercury to help extract the gold.

Mercury has been linked to gold mining for centuries, however, a full awareness of its significant negative impacts has only come about in the last 50 years or so. Mercury is highly persistent in the environment and is highly toxic especially in its methylmercury form through which it enters the food chain. This persistence means that, even if action were taken today to stop its use, its negative impacts will continue to be felt for many decades to come. The fact that an estimated minimum of 80 tons of mercury is used each year by miners across the Guianas, and that most of that is lost to the environment, should therefore be of critical concern to decision makers.

Globally, the negative human health and environmental impacts of mercury are well known and documented. This review seeks to highlight how this translates to the situation within the Guianas. The report collates the very wide body of research that has been done with regard to mercury use and contamination of the soil, water and air; of presence in the food chain; impact on human health; and exposure risk of both mining and non-mining communities

Through summarizing these studies, mainly carried out over the past 20 years, this review unequivocally shows that mercury is a significant and widespread issue that, thanks to its longevity, will already leave a long-term legacy for Guyana, Suriname and French Guiana. Critically it also shows that high mercury levels can be found throughout the entire region, including in 'pristine' areas, and are not restricted just to mining areas. Undoubtedly there are some specific research needs to better understand the inter-relationship between the many factors that influence mercury risk or the impact in specific environments (e.g. marine environments). However, these gaps do not detract from the conclusion that immediate action needs to be taken to phase out mercury from the gold-mining sector as quickly as possible.

## 1.0 Introduction

### 1.1 Small-scale gold mining in the Guianas

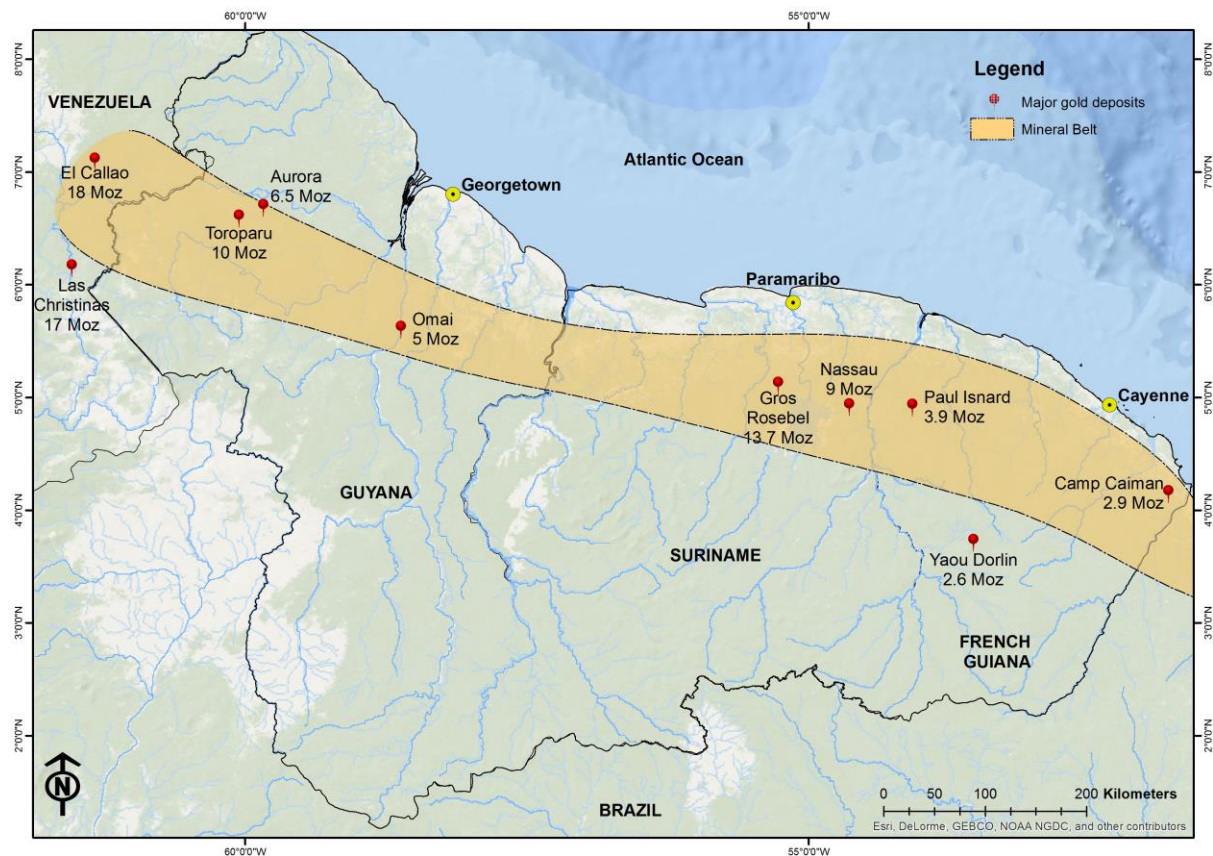
For thousands of years the most desired and enigmatic of metals, gold and mercury, have been linked both chemically and in their use by humanity through their propensity to form an amalgam. Mercury is an elemental pollutant which has been recognised both for its toxicity and usefulness in gold recovery since Roman times. Today the use of mercury in small-scale gold mining (SSGM) is a major source of pollution. In 2012, 4,477 tonnes of gold were circulating globally, approximately two thirds of which was newly mined with the remainder coming from recycling. SSGM is estimated to account for 12% of the annual supply of newly mined gold (330 tonnes/year) (PWC, 2013). In Latin American gold-producing countries, SSGM is thought to make up between 20 and 60% of gold production (Hammond et al., 2013). Conservative estimates suggest SSGM employs over 5 million individuals globally. Along with its geographical distribution this makes it a key economic pillar in many developing countries (PWC, 2013). However it is associated with numerous health, social and environmental concerns (PWC, 2013; Heemskerk Consultants in Social Sciences, 2011). In recognition of these concerns a number of global organisations (Association for Responsible Mining [ARM], United Nations Environment Program [UNEP], United Nations Industrial Development Organization [UNIDO] Global Mercury Project, World Gold Council [WGC]) are involved in initiatives to institute regulation and support the industry in becoming economically and environmentally sustainable (Echavarría, 2007; WGC, 2012; UNEP, 2012, UNIDO, 2002).

SSGM's impact on the environment is widespread and long lasting. Gold deposits are found within solid rock (primary/lode deposits) and as eroded sedimentary deposits (secondary/placer deposits: eluvial sediments formed by wind and rainfall and alluvial sediments formed in rivers and streams). Therefore, miners are attracted to various environments including upland, riverine and aquatic habitats (Hammond et al., 2007). Habitats may take many years to recover from the disruption caused by mining but the principal driver of long term damage associated with SSGM is mercury. Pollution caused by mercury leaves a legacy matched by few other pollutants due to its persistence within the environment (UNEP, 2013; Nriagu, 1994).

In Latin America evidence of gold working by indigenous peoples exists from 1200 BC and historical documents suggest mercury was employed in precolonial times to recover both gold and silver from ore (Cremers and de Theije, 2013; Nriagu, 1994). The arrival of the Spanish Conquistadors, driven in large part by stories of the precious metal resources to be won, led to the industrialisation of silver mining and the concomitant use of industrial quantities of mercury (Cremers and de Theije, 2013). Mercury was lost at numerous points in the mining process resulting in atmospheric, soil and water contamination. A staggering 196,000 tonnes of mercury may have been released in the precious metal mines of South America between the years of 1500–1900, likely enough to have altered global background levels (Nriagu, 1994).

Since the arrival of the Conquistadors there have been a number of successive gold rush events in South America. The latest series of which, beginning at the end of the 19<sup>th</sup> century, focused on the three countries of the Guianas and locations in the wider Amazon Basin (Cremers and de Theije, 2013). The rock formation of the Guiana Shield region, upon which the Guianas lie, is a continuation of the African Gold Coast and is rich in gold (**Figure 1.1**) and other minerals including bauxite, diamonds and iron (Hammond et al., 2007). However, while the extraction of most minerals has reduced since the late 1980s gold production has experienced a significant boom (Hammond et al., 2007). It is worth noting that SSGM is not restricted to areas of gold-bearing rocks as the eroded sediments from these areas can be transported great distances by river systems.

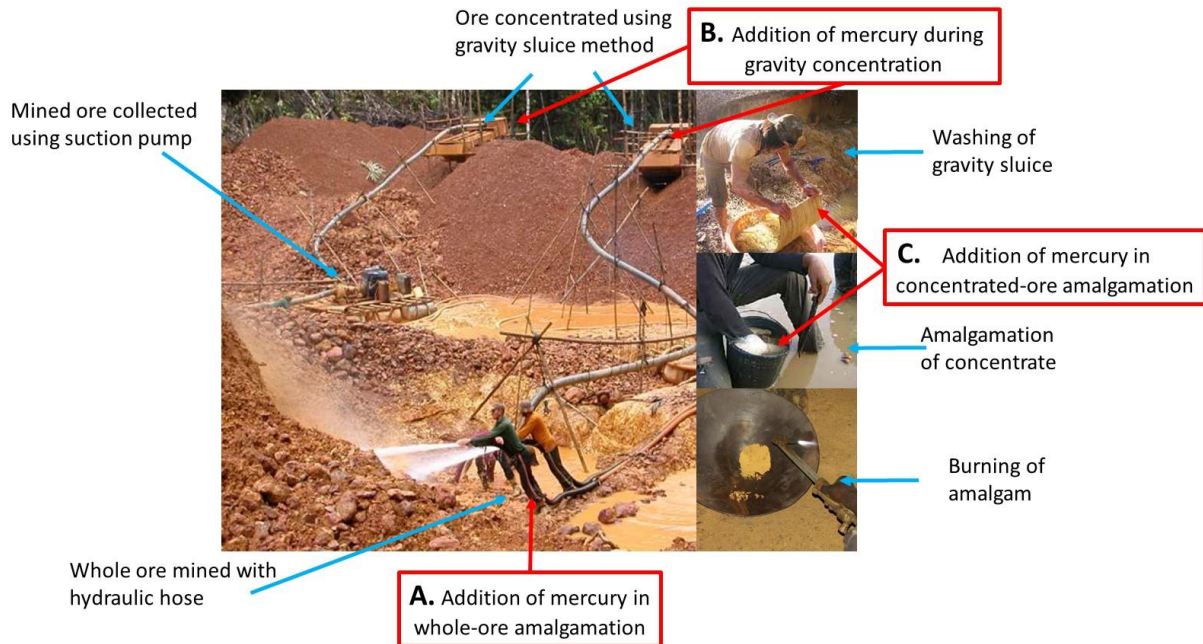
**Figure 1.1:** The principle mineral belt of the Guiana Shield



The gold rush in the Guianas has principally been driven by the long-term consequences of increased prices driven by the market floatation in the 1970s. Between 1979 and 2004 world-wide production increased 60-fold despite a dampening in global demand. Since the financial crash of 2008, the subsequent increase in gold prices has further fuelled SSGM (Hammond et al., 2007). The main socio-economic motivators for individual participation in SSGM worldwide are: poverty (individuals effectively have no other option available), crisis (economic crisis or societal disruption such as war), closure of industrial scale mines, the attractiveness of incomes (perceived or actual); supplemental income (usually subsistence farmers and indigenous people working part-time); and for profit (a more organised investment by entrepreneurs or investors) (Lowe, 2006). The porosity of the global gold market remains a key driver of informal SSGM, as illegally produced gold can be easily laundered to enter official trade chains (Taravella, 2009).

The term 'small scale' may be misleading both in terms of the importance of the industry to the economies in the Guianas, the sophistication of techniques and the environmental impact caused by modern day small-scale gold mining (**Figure 1.2**). Within SSGM there exists a sliding scale from unmechanised itinerant lone miners to large groups using heavy equipment (Cremers and de Theije, 2013). The stereotypical image of a miner with a gold pan (batea) is one that is now restricted almost exclusively to the prospecting and final processing stages (Cremers and de Theije, 2013). In addition, miners previously restricted to secondary eluvial or alluvial deposits may now employ crushers to tackle primary deposits in some areas (Cremers and de Theije, 2013).

**Figure 1.2:** Mining process and mercury addition (clockwise from bottom left) in a typical terrestrial hydraulic small-scale mining operation. *Adapted from Heemskerk Consultants in Social Sciences, 2011; UNEP 2012; Healy and Heemskerk, 2005*



Different mining practices profoundly affect the amount of mercury which is released into the environment and whether the release is principally to the atmosphere or soil and water. Practices range from the use of mercury free gravity techniques at the less environmentally damaging end of the scale to the use of whole-ore amalgamation and indoor mercury burning, where there is potential for the maximum damage to the environment and human health.

In whole-ore amalgamation mercury is added to the complete unprocessed ore before any form of concentration, often simply involving pouring mercury onto the heaps of excavated soil (**Figure 1.2 A**). This method is considered poor practice as it is inefficient with between 4 and 20 parts of mercury used for each part of gold recovered. The gold recovery rate is rarely more than 30% and there is significant wastage and release of mercury to the environment (**Table 1.1**) (UNEP, 2012). The proportions of mercury lost to different phases of the mercury cycle (**see section 3.0**) during whole ore amalgamation are estimated at 25% to the atmosphere and 75% to the soil and water (**Table 1.1**) (AMAP/UNEP, 2013). Another method generally viewed as damaging and inefficient in terms of mercury contamination and gold recovery is the spreading of mercury onto the surface of the sluice box during the process of concentrating ore to aid capture of the gold particles (**Figure 1.2 B**) (Hays and Vieira, 2005; Woltruba, 1998).

Concentrated-ore amalgamation is the most efficient and ‘cleanest’ of the mercury-reliant techniques (**Figure 1.2 C**). In this method mercury is added to the ore following completion of concentration in the sluice box or similar gravity based concentration system (UNEP, 2012). Less mercury is used and excess can be more easily recovered, hence, this method has the potential to be significantly more efficient than the above, approaching a 1:1 ratio of mercury used to gold recovered (**Table 1.1**). However, recovery methods are often primitive, such as squeezing the

mercury/gold mixture through a cloth leading to significant potential for physical contact and exposure to toxicity (UNEP, 2012). The proportions of mercury lost to different phases of the mercury cycle (**see section 3.0**) through concentrated-ore amalgamation are estimated at 75% to the atmosphere and 25% to the soil and water (**Table 1.1**) (AMAP/UNEP, 2013). The absolute amount of mercury released is also lower in concentrated ore amalgamation. However, when considering the release of mercury in different phases of the mercury cycle it is important to understand that miners may use a combination of methods of amalgamation depending on previous experience.

**Table 1.1:** Comparison of whole and concentrated ore amalgamation, global figures (UNEP, 2012; AMAP/UNEP 2013)

| Amalgamation Technique | Mercury:Gold Ratio | Gold Recovery | Release to Soil/Water | Release to Atmosphere |
|------------------------|--------------------|---------------|-----------------------|-----------------------|
| Whole Ore              | 4–20:1             | ≤ 30%         | 75%                   | 25%                   |
| Concentrated Ore       | 1–1.3:1            | 75–100%       | 25%                   | 75%                   |

All mercury based recovery methods require burning to separate the mercury:gold amalgam, resulting in mercury release to the atmosphere (UNEP, 2012). This is the stage of maximum risk for exposure to toxicity for those working in the mining industry (**see section 5.2**). Burning may be conducted using a wood fire or an acetylene torch and carried out both indoors or outdoors. Indoor burning carries an additional risk for toxicity due to the concentration of mercury vapour. The use of a retort to recover mercury at this stage can both significantly reduce the amount of mercury lost to the environment (0.05%) and greatly improve safety for miners burning the amalgam but uptake of these technologies has been poor (Heemskerk Consultants in Social Sciences, 2011).

The historical and continuing reliance of many SSGM operations on mercury based technologies has the potential to affect the region for many years to come. SSGM in the three Guianas is now one of the most pressing concerns in terms of environmental damage and human health.

## 1.2 SSGM in French Guiana:

Gold extraction in French Guiana began in the mid-1850s with a gold rush continuing until the turn of the century before production reduced reaching its lowest point in the middle of the 20<sup>th</sup> century (Richard et al., 2000; Heemskerk Consultants in Social Sciences, 2011; Horth 2011). Since the beginning of exploitation it is estimated that between 175 and 250 tonnes of gold have been produced with a proven resource of around 500 tonnes in total (Richard et al., 2000; Horth, 2011).

Gold production was reignited by the increase in gold prices in the 1970s and further fuelled by migration from Brazil (Heemskerk Consultants in Social Sciences, 2011). Declared production remained consistently above 2 tonnes per year from 1992–2000 but has fallen since the turn of the 21<sup>st</sup> century (3,469 kg in 2000; 2,576 kg in 2005; 1,300 kg in 2012 and 2013) (DRIRE, 2010; USGS, 2012, IEDOM, 2014). In 2010, declared production of around 1.2 tonnes was dwarfed by an estimated 10 tonnes of illegal production (Horth, 2011).

In 2006 the government banned the use of mercury in gold mining and elaborated more ambitious environmental procedures towards this highly impacting sector (Lefebvre, 2009). Between 2004 and

2011 the number of mining permits decreased by 50% (Heemskerk Consultants in Social Sciences, 2011), however, from 2010 onwards legal mining has undergone a slow recovery (Horth, 2011).

The 550 people directly employed in the legal mining industry as a whole in 2012 was dwarfed by the 10,000 illegal miners thought to be at work during this period (Heemskerk Consultants in Social Sciences, 2011). The majority of illegal miners were immigrants suggesting that income is the main socio-economic incentive (Lowe, 2006; Heemskerk Consultants in Social Sciences, 2011).

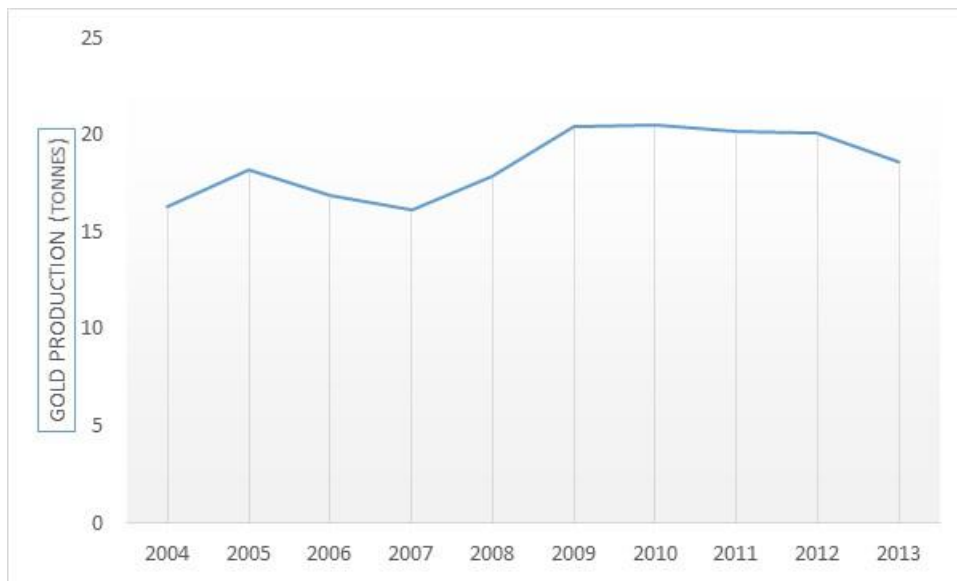
In 2005, the year before the mercury ban, data from 4 legal mining sites showed that 75% of gold came from alluvial sources with a quarter from primary rock deposits and all the sites surveyed were using mercury at this point. The high levels in illegal alluvial mining would suggest that illegal mercury use in French Guiana remains high (Laperche et al., 2014; Heemskerk Consultants in Social Sciences, 2011).

### **1.3 SSGM in Suriname:**

Gold exploration in Suriname began in the 18<sup>th</sup> century with numerous unsuccessful exploratory attempts (Heemskerk, 2000; Heemskerk and Duijves, 2013). Success came in the late 19<sup>th</sup> century, with an 1874 expedition producing the country's first concessions on the Marowijne/Maroni River. Within 3 years of the establishment of the first mines production had increased 10 fold (Healy and Heemskerk 2005; Heemskerk and Duijves, 2013). As in French Guiana, gold production reached its nadir in the 1950s and 1960s and a new stage of growth driven by the rising gold price beginning in the 1970s was disrupted by the Interior War (1986–1992) (Heemskerk 2000; Heemskerk and Duijves, 2013). Gold was the sole currency of the Surinamese interior during the war and the social problems caused by the conflict have likely continued to be a major driver of SSGM due to increased poverty and the disruption of educational opportunities which persist in affecting sections of the population (Heemskerk and Duijves, 2013). Thus, conflict and poverty have been the major local drivers of the current Surinamese gold rush for native populations (Heemskerk Consultants in Social Sciences, 2011; Heemskerk and Duijves, 2013). Since the end of the war the income incentive has increased in importance as the now safer gold fields attract immigrants, foreign prospecting companies, and urban Surinamese to the interior of the country (Mol et al., 2001; Heemskerk Consultants in Social Sciences, 2011).

Figures from Thomson and Reuters suggested approximately 20 tonnes of production in 2012 (**Figure 1.3**), however, local industry figures suggest double this amount may have been produced (Thomson and Reuters, 2014; Surimep, 2015). At this time SSGM made up approximately 60% of exports or around US\$1 billion (Surimep, 2015). There is a single large-scale operation in Suriname, the IAMGOLD concession at Rosebel, which produced 325,000 Oz in 2014 (IAMGOLD, 2014). Estimates suggest that between 20,000 and 35,000 people are directly employed in the SSGM industry (Mol et al., 2001; Heemskerk Consultants in Social Sciences, 2011; Ouboter et al., 2012). In Suriname, 65 to 75% of the miners and mining service providers are migrants, principally coming from Brazil (Heemskerk and Duijves, 2013).

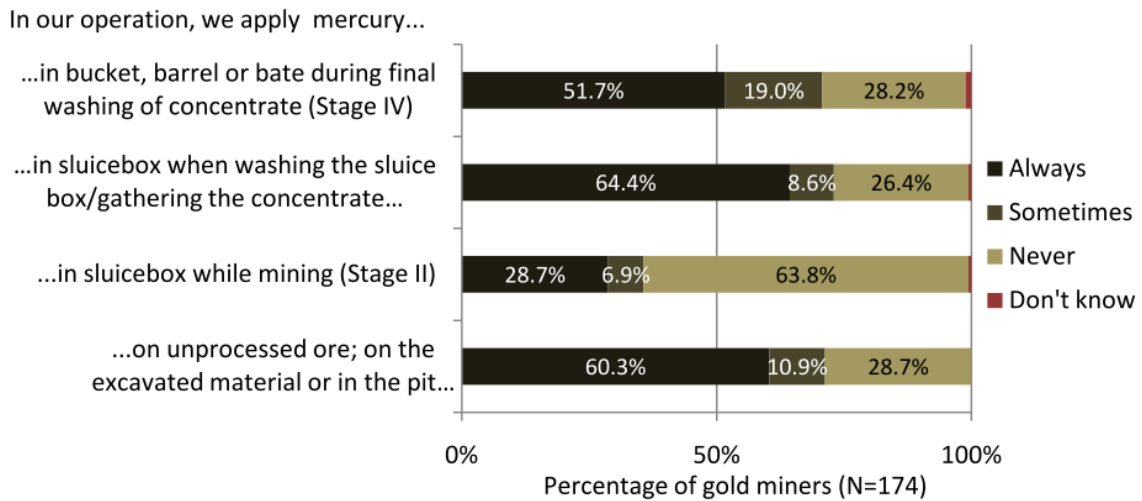
**Figure 1.3:** Annual gold production in Suriname (Thomson and Reuters, 2014).



The vast majority of small-scale gold miners in Suriname use land based hydraulic methods (Heemskerk Consultants in Social Sciences, 2011), with the remaining mining consisting of river based dredges. Duijves and Heemskerk carried out a study of small-scale gold miners' (N = 346) attitudes and practices with regards to mercury in Suriname (**Figure 1.4**). Most miners used better practice, adding mercury to concentrated ore, either while washing the sluice contents (73% always/sometimes) into the receptacle used for the final stage of processing or adding it directly to the receptacle during the final wash (71% always/sometimes). However, the majority of miners also carried out whole-ore amalgamation always (60%) or sometimes (11%). A significant minority also carried out the wasteful practice of adding mercury to the sluice during the mining process (35%) (Duijves and Heemskerk, 2014). These results are somewhat in contradiction with results from a previous survey of miners in the Benzdorp region, to the south of this study area, which reported only small amounts of mercury were used in whole-ore amalgamation. Most miners were found to only amalgamate gravity concentrates and only a little mercury was used during clean-up of the sluice box (Healy and Heemskerk, 2005).

In the attitudes and practices survey few miners reported always using a retort when burning amalgam (8.8%) (Duijves and Heemskerk, 2014), these data are in line with previous studies (Healy and Heemskerk, 2005; Heemskerk Consultants in Social Sciences, 2011; Heemskerk and Duijves, 2013). However, the great majority of miners did not burn amalgam indoors (Duijves and Heemskerk, 2014). Furthermore, most miners (82%) had personally handled mercury and in most cases did not use gloves.

**Figure 1.4:** Amalgamation practices used by small-scale gold miners in Suriname (Duijves and Heemskerk, 2014)



#### 1.4 SSGM in Guyana:

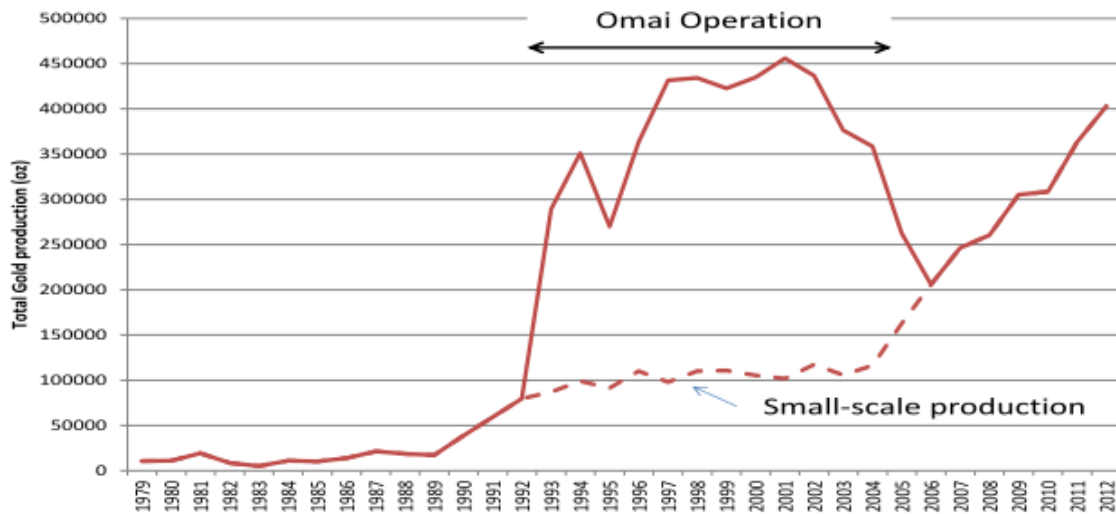
As in Suriname, the gold industry in Guyana suffered a stuttering start with numerous unsuccessful explorations of now productive areas, including the Berbice and Mazaruni districts, during the 18<sup>th</sup> and 19<sup>th</sup> centuries (Harrison, 1908; Thomas; 2009). Gold was successfully found by the British Guyana Gold Company in 1863 but production was prevented by a border dispute with Venezuela (Harrison, 1908; Thomas; 2009). The first documented commercial gold mining in Guyana started in the 1880s in the Potaro Region, with the first international commercial efforts beginning in the early 20<sup>th</sup> century (Harrison, 1908). In shortly over 10 years from the founding of the Potaro mines, Guyana was amongst the 10 most productive gold producers in the world (Harrison, 1908; Thomas, 2009). The first official record of production was 250 Oz in 1884 rising to 132,995 Oz in 1894. Pork-knockers, the local name for small scale gold miners, have been the backbone of Guyanese gold-mining since its inception and are responsible for the bulk of production with the notable exception of the large scale Omai mine<sup>1</sup> that operated from the early 1990s to the mid 2000s (Figure 1.5) (Thomas, 2009; Singh et al., 2013).

SSGM has experienced significant growth in Guyana since the late 1980s, with close to a 50% increase from 2008–2013 (Singh et al., 2013). This increase correlated with increased global gold prices contributing to a doubling in the importance of gold production to the Guyanese economy between 2007 (7% of GDP) and 2011 (15.5% of GDP) (Howard, 2010; Singh et al., 2013). In 2011, gold and bauxite export accounted for over half (56%) of all exports from the country (Veening and de Ville, 2013). As of 2013, the SSGM mining sector was thought to employ around 35,000 people (McRae, 2014). The major socio-economic drivers of growth in the sector in recent times have been the increased global prices of gold, leading to an increase in profit motive and perceived/actual

<sup>1</sup> Like many large-scale operations Omai used a cyanide based process, rather than mercury, for recovering gold. The Omai Company was responsible for a major man-made natural disaster when 400 million gallons of cyanide effluent was spilled from a tailing pond into the Omai River after a breach on August 9<sup>th</sup> 1995 (Ramraj, 2001).

income (Lowe, 2006). However, supplemental income and poverty are still important drivers in some communities (Singh, 2000; Lowe, 2006).

**Figure 1.5:** Annual gold production in Guyana (Singh et al., 2013)



All three main deposit types are present and exploited by SSGM operations in Guyana (Lowe, 2006). Alluvial deposits from the riverbeds and floodplains of both current and dried-up river systems, terrestrial eluvial deposits and gold-bearing quartz veins are mined. Up until the 1990s alluvial river deposits were the principal target of SSGM operations using river dredges (Lowe, 2006). Lack of accessibility of rich river deposits saw ‘land dredging’ (hydraulic) operations increase significantly in the 1990s and early 2000s (Lowe, 2006). However, river mining has seen a resurgence since 2005 and a tripling in the number of dredges between 2007 and 2013 (10,029–12,029 units in use 2013) (Singh et al., 2013; McRae, 2014). The breakdown of deposits through the use of hammer milling machines is the preferred method for quartz-vein deposits in Guyana (Lowe, 2006).

Howard reported that as of 2010 there was no use of mercury free gravity methods in Guyana; this despite 10 years of encouragement through the Canadian International Development Agency (CIDA) which funded the Guyana Environmental Capacity Development (GENCAPD) mining assistance programme (Howard, 2010). Data from four surveys carried out at the turn of the 21<sup>st</sup> century and presented by Lowe in his 2006 assessment of the state of the Guyanese mining industry showed that no miners were achieving good practice in terms of mercury use (**Table 1.2**). Practice varied over time and between operations with the majority being ‘moderately satisfactory’. However, unsatisfactory practice, including the adding of mercury to sluice boxes, whole-ore amalgamation and the burning of mercury indoors, were reported. Evidence from the Mahdia area suggested possible improvements in practice over time. However, there was little or no evidence of retort use at any location or time point and mercury was principally handled without gloves (Lowe, 2006).

McRae (2014) carried out a survey into mercury-use practices amongst a small group (N = 57) of miners principally working on terrestrial hydraulic operations. The results suggest some encouraging changes in mining practice in Guyana. Safe storage of mercury was consistently reported and the majority of miners used recommended concentrated-ore amalgamation methods. However, use of whole-ore amalgamation remained common. The use of gloves and retorts now appeared to be the norm in this small survey group but the recovery rate from retorts was low (50–80%), likely due to poor practice, and gloves were often inadequate being too short or of too thin a material. The

mercury use practices ranged from very careful frugal addition to very free and wasteful (McRae, 2014).

**Table 1.2:** Mining practices from four surveys carried out in Guyana (Lowe, 2006)

|  | AREA OF STUDY/ YEAR/ NUMBER OF OPERATIONS VISITED |                                 |  |                                  |
|--|---|---------------------------------|--|----------------------------------|
|  | Mahdia, 1998<br>(11 operations)                   | Mahdia, 1999<br>(21 operations) | Tiger Creek,<br>1999<br>(4 operations) | Arakaka, 2000<br>(22 operations) |
| Good (adding mercury placed in jig-box, bucket or drum with covered hands; “pumping and “spin-out” in jig box only; using a retort and gas mask when burning amalgam.)             | -   | -                               | -                                      | -                                |
| Satisfactory   | -   | -                               | -                                      | 27.3%                            |
| Moderately satisfactory  | 18.2%   | 62%                             | 50%                                    | 27.3%                            |
| Marginally satisfactory  | -   | -                               | 50%                                    | 27.3%                            |
| Unsatisfactory (massaging concentrate with mercury in the sluice box; no jig-box; “pumping” and spin-out” of concentrate in pond, creek or river; burning amalgam in the kitchen). | 72.7%   | 14%                             | -                                      | -                                |
| Very unsatisfactory (using mercury on the ground; no jig-box; burning amalgam in the kitchen).   | 9.1%  | -                               | -                                      | -                                |
| Unknown  | -   | 24%                             | -                                      | 18.1%                            |

There are some encouraging signs in the current mining practices in Guyana but the still universal use of mercury indicates it continues to pose a significant threat to the environment. Larger surveys assessing whether these less polluting methods are widespread in Guyana would be useful.

## 2.0 Sources and volume of mercury used in small-scale gold mining

### 2.1 Movement and sources of mercury in small-scale gold mining

UN Comtrade figures suggest more than 1,800 tonnes of mercury were traded globally in 2014 (UN Comtrade, 2014). The total value of reported exports between 2002 and 2006 was US\$113,587,000 (US\$22,717,000/year) while imports totalled US\$132,593,000 (US\$26,519,000/year) (Telmer and Veiga, 2009) showing a serious discrepancy. However, reporting on the mercury trade is on a voluntary basis, resulting in significant gaps in data regarding both import and export. The extent of these gaps in knowledge is illustrated by the 70 countries that do not report any trade in mercury

despite the presence of dental services where the use of mercury is essential (Telmer and Veiga, 2009).

Historically, major global producers of cinnabar (mercury sulphide, HgS), the principal mercury containing ore, included the Almaden in Spain, which ceased in the year 2000 following over two millennia of production, as well as Kyrgyzstan, and China (Veiga et al., 2006; Howard, 2010; European Commission, 2013). The sources and movement of mercury have recently been impacted by export bans in both the European Union and the USA. As part of its commitment to the UNEP Global Partnership on Mercury (GPM), the European Union banned the export of mercury and certain mercury containing products in 2011. This included certain mercury compounds and mixtures and alloys of mercury with a concentration of at least 95% mercury by weight (UNEP, 2009; Singh et al., 2013, European Commission, 2013). The EPA Mercury Export Ban Act (MEBA) of 2008 prohibited the sale and export of elemental mercury from the USA as of 2013 (Balistreri and Worley, 2009; Singh et al., 2013). Despite the ban on export and sale, the use of mercury in certain products (i.e. thermometers, electrical switches/relays, mercury measuring devices, and novelty products excluding light bulbs and button cells), has generated a significant quantity of legacy waste and by-products in both the United States and Europe. Data from 2014 indicates that the major global mercury exporters now include Canada, China/Hong Kong and Japan, perhaps as a result of the above regulatory changes (UN Comtrade, 2014).

The uncertainties regarding the global mercury trade are compounded locally within the Guianas and data are particularly sparse for French Guiana and Suriname. Data for French Guiana, where the use of mercury in gold mining is illegal, are currently lacking. However, the presence of around 10,000 illegal small-scale gold miners would suggest significant continued use of mercury (Heemskerk Consultants in Social Sciences, 2011). As an illustration in 2013, the French Gendarmerie seized 82kg of mercury during their enforcement operations on illegal mining sites. In Suriname, mercury has been placed on a negative list of substances meaning that all imports and exports require licensing and dispensation from the Ministry of Trade and Industry (Heemskerk Consultants in Social Sciences, 2011). However, there has been no official import of mercury under this 'ASYCUDA system', despite 19 tonnes of gold export from SSGM in 2011 alone and indications that the vast majority (97%) of miners involved in SSGM use mercury (NIMOS, 2013; NIMOS, 2014). Mercury is highly portable and easy to move across borders (Telmer and Veiga, 2009) making it likely that the mercury used in the SSGM industry is smuggled into Suriname without the required documentation (Heemskerk Consultants in Social Sciences, 2011).

In the case of Guyana the picture is somewhat clearer, a recent study indicates that over the period 2008–2013 mercury was sourced from 10 countries. Previous major suppliers, including Spain and the USA, were superseded in 2013 by China who provided almost 60 tons of mercury in 2013 (approximately 75% of the total imported), likely due to the above noted export bans (McRae, 2014). McRae's study indicates that there are significant discrepancies in terms of the recording of imports. Data from two of the three governmental bodies overseeing the import of mercury, the Guyana Geology and Mines Commission (GGMC) and the Guyana Revenue Authority (GRA), were compared (McRae, 2014). Data from the Pesticide and Toxic Chemical Control Board (PTCCB) who control the final import of mercury were not available. Comparison of the two available datasets showed a lack of both internal and external consistency. Notably, 63 persons were granted permits by the GGMC, but of the 30 actual importers recorded by the GRA only 9 were present on the former authority's list of permit holders (McRae, 2014).

In terms of direct supply to miners and mine operators within Suriname and Guyana mercury is principally traded within the respective capitals of Paramaribo and Georgetown. In Guyana, the

majority of importers are directly connected with the mining industry and either operate mines or sell mining equipment (McRae, 2014; Duijves and Heemskerk, 2014). Around 20% of the mercury purchased in Georgetown is resold in the mining districts, at sites including Port Kaituma, Bartica, and Mahdia. Sales are principally conducted through small retailers such as grocery stores or at local markets (McRae, 2014; Howard, 2010). In a recent survey carried out in Suriname a slight majority of miners (51.7%) had not personally bought mercury, stating that equipment owners were responsible for its provision. Of those who had purchased mercury themselves, two thirds had done so in Paramaribo. The remainder had bought in the mining areas and a very few from Guyana (Duijves and Heemskerk, 2014).

In summary, better reporting of the mercury trade is clearly necessary on both a national and international scale. This would significantly improve knowledge regarding flows of mercury within and between the Guianas and around the world (Telmer and Veiga, 2009).

## 2.2 Volumes of mercury used in small-scale gold mining

Despite its small size Guyana was 22<sup>nd</sup> in the world for mercury imports between 2002 and 2006 (Telmer and Veiga, 2009). The UN estimates mean mercury use in SSMG, based on 2008 figures, at 15 tonnes/year (range 7.5–22.5 tonnes/year) in Guyana and 7.5 tonnes/year (range 3.8–11.3 tonnes/year) in both Suriname and French Guiana. These figures have shortcomings, as the authors themselves acknowledge, and are based on outdated figures from 2008 (AMAP/UNEP, 2013).

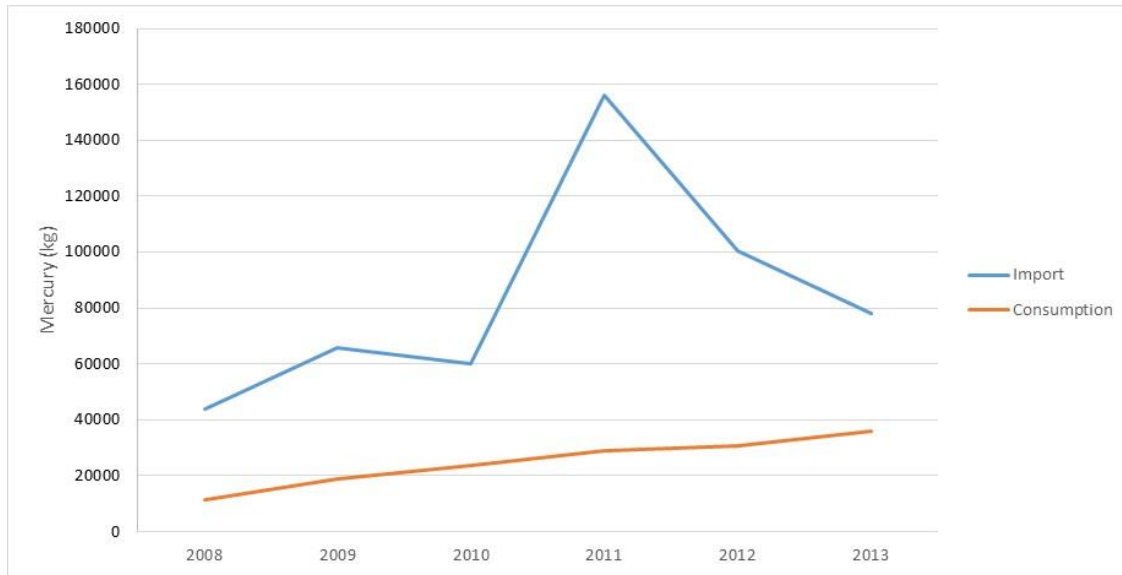
In the recent study into mercury flows in Guyana (McRae, 2014), an estimation of total use was calculated using proxy indicators. A random sample of both terrestrial and river based hydraulic dredges was taken and based on knowledge of the number and size of dredges in use plus typical mercury use per dredge, a total figure of mercury used was extrapolated. Using this method, mercury use in 2008 was estimated to be 11.37 tonnes (within the range of the UNEP figures based on data from that year (7.5–22.5 tonnes). This grew to an estimated 35.82 tonnes in 2013.

Volumes of mercury import and predicted consumption between 2008 and 2013 are illustrated in **Figure 2.2** (McRae, 2014). Official import has outstripped mining demand in Guyana by at least a factor of 2 every year from 2008–2013, peaking at 5.4 times the national requirement in 2011. Over this period total imports were 504 tonnes with an excess of approximately 355 tonnes, over 10 years' worth of demand at 2013 levels. Given that there is no large scale storage of mercury, these data suggest a large alternative market for Guyana's mercury. It is also worth noting that the ratio of mercury used in gold production has risen from 1.5:1 to 3:1 in the same period, these ratios are in line with previous use estimates (Telmer and Veiga, 2009; UNEP, 2013). Likely explanations for the excess mercury import and increase in mercury:gold ratio include clandestine production of gold, which may be smuggled out of the country, for example to Suriname where export duties are lower (Stabroek News, 2012); an increase in use of whole ore amalgamation, a practice currently admitted by a small but significant minority of miners in Guyana (McRae, 2014); and smuggling of mercury to Suriname and French Guiana, neither of which have official imports of mercury.

The most recent results on gold export from SSGM in Suriname are from 2011 and indicate that 19 tonnes of gold were exported (Heemskerk and Duijves, 2013). Extrapolating from the Guyana results, which are in agreement with international estimates of mercury:gold production ratios, this would suggest mercury consumption in Suriname of between 28.5–57 tonnes in 2011. It is notable that despite a recent five-fold price increase in Suriname, there are no indications that gold miners are making any efforts to conserve mercury (Duijves and Heemskerk, 2014). The most up to date

estimate from French Guiana of 10 tonnes of illegal gold production in 2010 would suggest mercury use of between 15 and 30 tonnes (Horth, 2011).

**Figure 2.2:** Guyanese mercury imports between 2008 and 2013, based on GRA data. *Adapted from McRae, 2014.*



Given the above information it would seem that few statements regarding mercury use in the Guianas can be made with a great degree of certainty beyond stating that it is highly likely that current official figures on mercury use represent a significant underestimate. The gathering of data on mercury use should be a high priority. Implementation of the UNEP Toolkit for Identification and Quantification of Mercury Releases (UNEP, 2013b), a standardized methodology for assessing national mercury throughout the three Guianas would be a logical first step in this process.

Given the current estimates available, it is likely that a minimum annual estimate of 80-122 tonnes of mercury is used in the Guianas region by the SSGM sector.

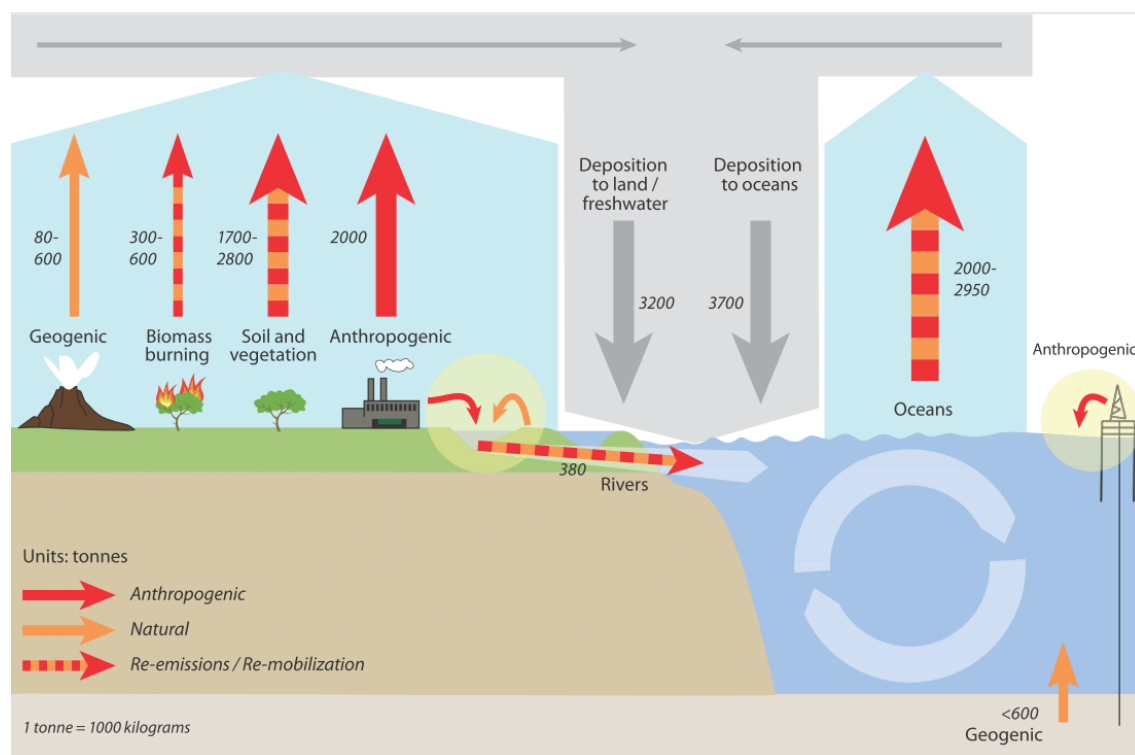
### 3.0 Mercury contamination in the Guianas:

#### 3.1 The mercury cycle:

Globally, mercury is released into the atmosphere by both natural and anthropogenic sources. The principle global sources are the natural release of mercury stored in the ocean and from the soil/vegetation, and anthropogenic release through SSGM and coal-fired power stations. Further significant natural sources include forest fires and release from volcanoes. Both anthropogenic and natural sources of mercury are likely to be important to the total mercury levels found in the Guianas, however, data illustrating the relative importance of these alternative sources are currently sparse within the region. Following release, mercury continues to cycle between the air, land and ocean/freshwater (**Figure 3.1**) (UNEP, 2013). Mercury is extremely persistent, continuing to cycle until bound to a stable compound or buried deep in ocean or lake sediment.

In the following chapter aquatic, terrestrial and atmospheric contamination in the Guianas will be summarised followed by a brief discussion of the data.

**Figure 3.1:** Volumes of release and deposition from the principal contributors to the global mercury cycle (UNEP, 2013)



### 3.2 Freshwater:

The Guiana Shield is a site of freshwater production of worldwide significance (Rosales, 2003). Such freshwater systems are not only vitally important for ecology and human health in the Guianas but may also represent a major economic resource as climate pressures further impact the already water-scarce island nations of the Caribbean (Saenz, 2014). Within the three Guianas, SSGM has already significantly affected water catchment zones of regional importance including the Essequibo Basin in Guyana and the Marowijne/Maroni Basin shared by Suriname and French Guiana (Hammond et al., 2013).

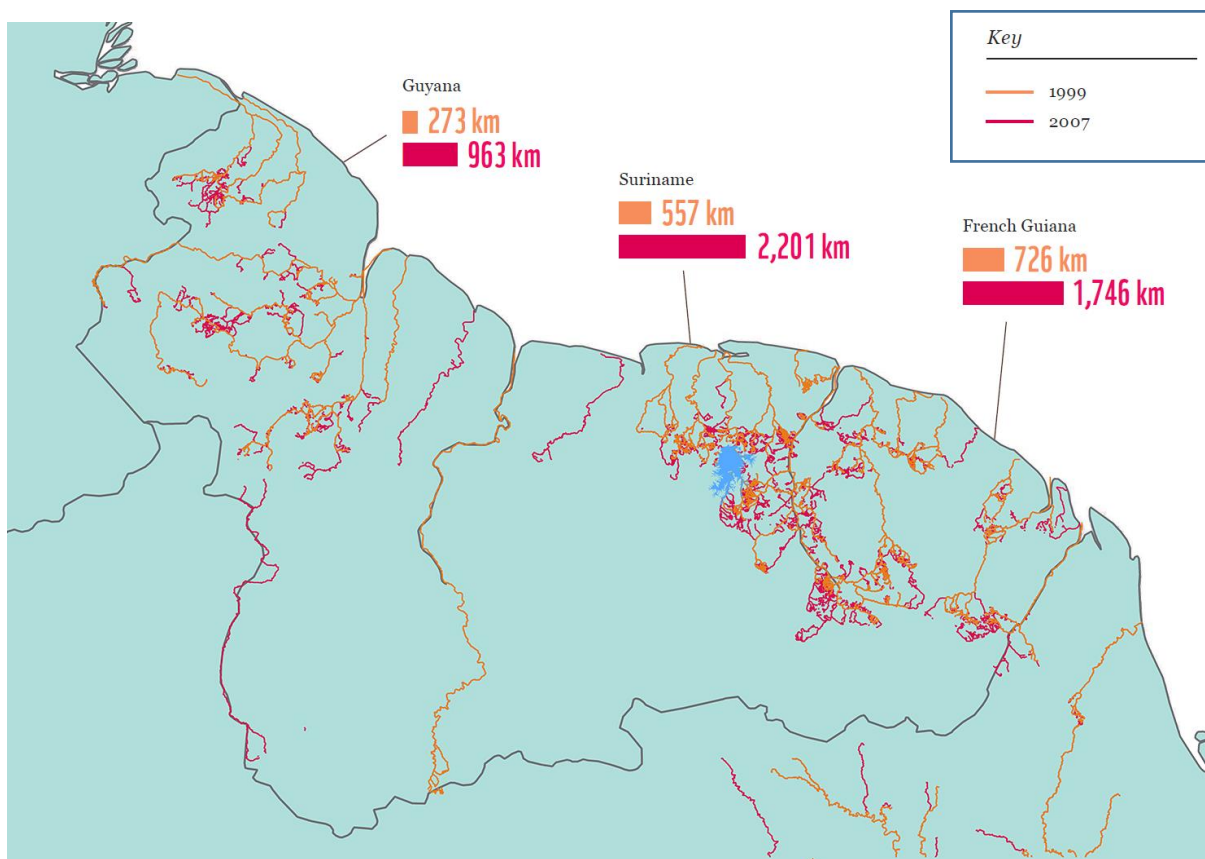
As mentioned previously, the proportion of SSGM-related mercury released to freshwater/soil systems, versus the atmosphere varies depending on the amalgamation techniques used by miners (AMAP/UNEP, 2013). Nevertheless, from the point of view of the health of humans and other organisms in the Guianas, contamination of freshwater is likely the most important point in the mercury cycle as it acts as a significant route for the transport of freshly released mercury away from mining sites (Paktunc et al., 2004; Arets et al., 2006) as well as being the major site of mercury methylation and biomagnification. Following methylation, mercury enters the food chain so rapidly that the concentration of methylmercury in sediments and water is very low (D'Itri, 1990; Veiga, 1997). As a result, and due to its key role in the food chain and human health, methylmercury will be discussed separately in section 4.1.

Mining not only affects freshwater systems in terms of mercury pollution. Sediments released by gold mining, particularly using hydraulic means, increase water turbidity and natural watercourses

are often altered affecting supply. Disruption of water supply and increased turbidity have been cited as major concerns for indigenous peoples in the Guianas (Gray et al., 2002; Heemskerk and Oliveira, 2004; Heemskerk Consultants in Social Sciences, 2011). Increased turbidity has major impacts on the ecology of the Guianas' freshwater systems, which have one of the lowest natural sediment loads in the world (Hammond et al., 2007), reducing available habitats, species diversity and reproduction (Mol and Ouboter, 2004). Turbidity is also associated with a higher mercury load, and with increased transport to areas distant to SSGM operations due to the association of inorganic mercury with organic molecules within the sediment (Gray et al., 2002; Paktunc et al., 2004; Arets et al., 2006). Additionally, disruption of watercourses may also create areas of stagnant water with high organic matter content which encourages the production of methylmercury (Gray, 2002).

Comparison of satellite data from 1999 and 2007 suggest a more than 200% increase in the length of Guiana Shield watercourses potentially affected by SSGM over this period. Within the Guianas, 4,910 km of rivers and streams were potentially affected in 2007 (**Figure 3.2**) (WWF, 2012).

**Figure 3.2:** Change in watercourses of the Guianas potentially affected by SSGM between 1999 (orange) and 2007 (red) (Adapted from WWF, 2012)



Inorganic mercury is not easily absorbed by the body after ingestion meaning that drinking water is not the major route by which mercury affects human health. Risks arise when mercury methylation occurs in aquatic systems and enters the food chain in this form. Crucially, levels of mercury exhibit an increase in concentration and toxicity with increased trophic level (biomagnification) (see section

**4.1).** The Canadian Council of Ministers of the Environment (CCME) guidelines have a safe limit of 26 ng/L of mercury in water but clarify that animals which consume aquatic wildlife may still not be protected at this level (CCME, 2003).

Neither can freshwater ecosystems be considered in isolation from their associated sediments. Levels in sediments may be many times higher than those in water due to rapid absorption of mercury. As a result sediments give a more accurate picture of the extent of mercury contamination. (Stein et al. 1996; Mol and Ouboter, 2004). This is particularly true in sediments with high levels of organic material, the ratio of mercury stored in water versus sediment ranges from 1:1,000 for coarse sand to 1:5,000–1:50,000 for organic sediments (Stanford, 1971). Furthermore, sediment may act as both a long term store for mercury and as a major substrate for methylation (**see section 4.0**) (Trimble 1981; Mol and Ouboter, 2004).

CCME guidelines have a safety cut off for freshwater sediments of 0.17 µg/g, above which adverse biological effects (lethality, reduced fertility and impaired development) begin to occur in the aquatic biome. At mercury levels above 0.486 µg/g adverse events are frequent (CCME, 1999). Although the EPA and WHO provide maximum concentrations for mercury in water the CCME is the only agency, to the best of the authors' knowledge, that provide limits for freshwater sediment. For the above reasons and due to difficulties in accurately detecting low levels of mercury in freshwater, sediment samples are the principle method for assessing mercury contamination in freshwater systems and the CCME limits are those most commonly used in literature from the Guianas.

The following sections summarise the data on mercury contamination in freshwater systems in French Guiana, Suriname and Guyana.

### *3.2.1 Freshwater contamination in French Guiana:*

A gold-bearing greenstone belt covers a large proportion of the southern half of French Guiana with extensive areas with mining potential also present in the northern half of the country (**Figure 3.4**). Laperche et al. (2014) recently published the results of an extensive 2007 study focusing on 6 rivers in French Guiana. This work covered the full length of some rivers, and involved the collection of more than 1000 sediment samples. Rivers were selected to provide a heterogeneous picture with both pristine and heavily affected sites sampled.

Background mercury concentrations in sediment were assessed using multiple samples ( $n = 51$ ) from the Upper Oyapock River, an area known to be free of mining activity, with a mean ( $\pm$  standard deviation) mercury concentration in the muddy sediment of  $0.1 \pm 0.03$  µg/g. Using this mean value and standard deviation, the authors estimated the likely maximum background level of mercury to be 0.15 µg/g, close to the guideline CCME sediment value for occurrence of adverse events (0.17 µg/g) (Laperche et al., 2014). The background concentration was in agreement with previous studies from French Guiana (Richard et al., 2000; Charlet, 2003). Notably, the estimated background in estuaries was significantly lower (around 0.05 µg/g) likely due to mixing with sedimentary deposits carried from the Amazon (Laperche et al., 2014).

The mean concentration found in non-gold-mined areas was  $0.108 \pm 0.042$  µg/g, similar to the background value, with significantly higher and more variable values associated with gold-mined areas ( $0.19 \pm 0.137$  µg/g) (Laperche et al., 2014). Values in mined areas are in accordance with results found in previous studies (maximum 0.4 µg/g in gold-mined areas) (Charlet, 2003). Looking at the data for each river sector, rather than the mined vs non-mined areas as a whole, showed that this pattern persisted, with significantly higher mercury concentrations in the sediments from gold-mined streams compared to non-gold-mined streams. The variability in mined areas can partially be

explained by patterns in the intensity of mining, with heavily mined areas in some rivers showing peaks in mercury concentration (Laperche et al., 2014).

The highest values found in the study were from the sediments in the Impératrice, Mataroni and Haut-Approuague sectors of the Approuague River (0.81–11.2 µg/g), which included two aberrantly high values of 10.05 and 11.2 µg/g (**Figure 3.3**) (Laperche et al., 2014). High concentrations (0.56–1.25 µg/g) have previously been reported in the heavily mined Saül region of the Sinnamary basin (Richard et al., 2000), and at sites close to active mining (Spadini and Charlet, 2003). The lowest values (0.01–0.03 µg/g) were seen in samples from estuarine regions of the Comté and Oyapock rivers (**Figure 3.3**). In other rivers, samples taken from sites in or down-stream of granitic areas had the lowest values, likely due to sandy sediments which are not favourable for absorption of mercury. In total, 70% of all the sediment samples had mercury concentrations below maximal background levels of 1.5 µg/g (Laperche et al., 2014).

Using the extensive dataset collected, Laperche et al. (2014) characterised five types of river site and associated mercury contamination in French Guiana:

- 1) Current or previously mined areas with unusually high mercury concentrations (0.5–10.0 µg/g);
- 2) Rivers and streams close to mined sites with high mercury concentrations (> 0.2 µg/g) characterised by ochre yellow sediments;
- 3) Rivers and streams with low mercury concentrations (< 0.15 µg/g) characterised by brown to grey sediment;
- 4) Estuarine areas with very low mercury concentrations (< 0.06 µg/g)
- 5) Granitic sandy-sediment areas with very low mercury concentrations (0.01–0.05 µg/g).

The above relationship between sediment type and mercury levels has been reported elsewhere in French Guiana, with muddy sediments and forest soils in the Sinnamary basin having higher levels than lateritic soils, likely due to the relationship between mercury absorption and organic content together with the presence of a large hydroelectric dam on the same river basin (see below) (Richard et al., 2000).

Results from this current review suggest that SSGM-related contamination of sediments is widespread with levels over 65 times the CCME guideline levels at some sites. Although the extent of contamination correlates relatively well with areas of mining it is worth noting that background levels in non-mined areas (0.108 µg/g) were higher than those found in an extensive study carried out in the United States (0.088 µg/g) (Scudder et al., 2009). This is notable given the extensive nature of other industrialised processes, not present in French Guiana, which may contribute to mercury deposition in North America. Furthermore, sediment levels may not be showing the full picture regarding freshwater contamination (**see section 4.2.2**).



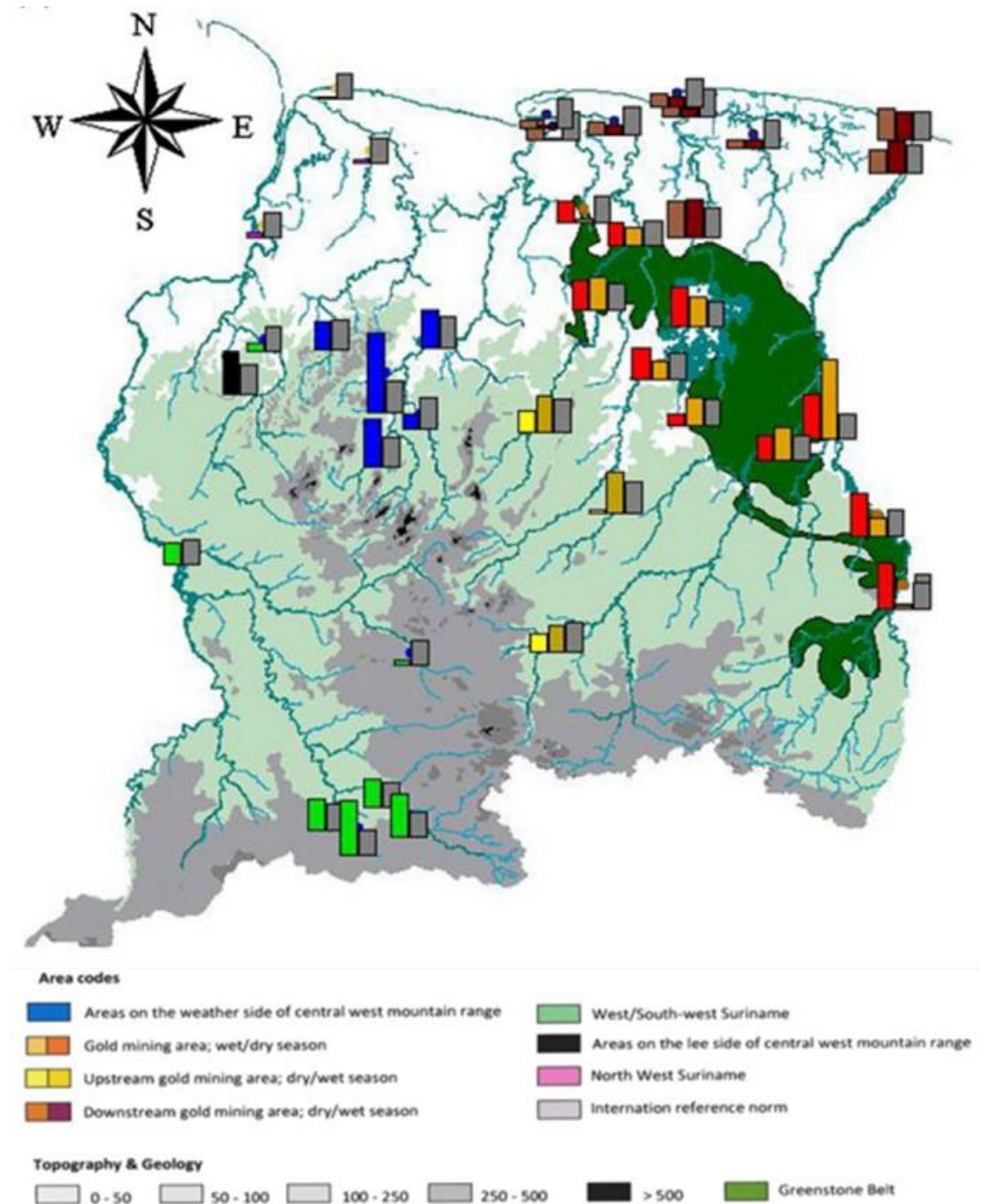
In 2012, Ouboter et al., carried out a comprehensive review of data on sediment mercury loads across Suriname, synthesising data from studies covering 53 sites across the country amassed between 2002 and 2010 (**Figure 3.4**). Data included samples from active mining areas with sites both up- and down-stream of mines as well as pristine areas with no history of gold mining. With the exception of the Brokopondo reservoir (**see section 4.1**), sample sites represented streams of similar size allowing comparison of levels of contamination between studies. Sampling concentrated on recently deposited fine sediments which readily form complexes with mercury.

Mean mercury levels in the majority of sampled areas were above the CCME standards for sediment. One of the two areas which did not show high levels of mercury was downstream from the gold mining area, however, it is likely that this is due to an artefact caused by a small number of sample sites and a lack of up to date figures. Mean levels in the gold mining area (0.22 µg/g) were matched by those in pristine areas of Central, Western, and Southern Suriname (mean concentration 0.20 µg/g, maximum 0.28 µg/g).

Core sampling was carried out on the floodplains of three rivers, two downstream of gold mining on the Saramacca River close to Kwakoebron and Pikin Saron, and one from the pristine Kabalebo River. Detected mercury levels were 0.25 µg/g and 0.22 µg/g at the sediment surface of the two respective mined rivers, reducing to 0.13 µg/g and 0.10 µg/g at a depth of 0.5 m. A similar pattern was observed on the Kabalebo, with mercury concentrations of 0.22 µg/g at the sediment surface and 0.19 µg/g at the deepest sample point (0.3 m). More recently an additional 12 floodplain core samples were taken and analysed. Nine sites showed the same pattern of decreasing mercury levels with depth. Interestingly, 10 of the 12 sites were on the Nickerie and Coppename Rivers downstream of pristine drainage areas (Ouboter, in prep.). This vertical concentration gradient is indicative of recent mercury deposition rather than high natural background levels suggesting that mining is most likely responsible for the recent mercury contamination (**see below**).

Data from Suriname shows widespread, almost country wide, mercury contamination with mean sediment loads consistently above CCME guidelines in all areas except the northwest of the country, with examples of contamination in pristine areas which appears to be derived from recently deposited mercury.

**Figure 3.4:** Mean mercury levels found in sediments in different river systems in Suriname 2002–2010; Grey bars represent the CCME Guideline for Protection of Aquatic Life of 0.17 µg/L (Ouboter et al., 2012)

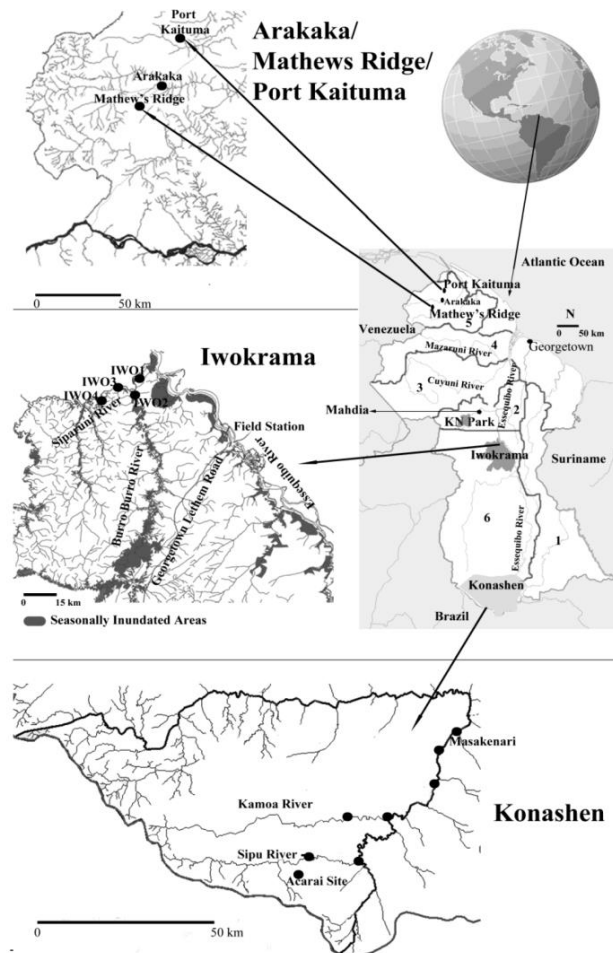


### 3.2.3 Freshwater contamination in Guyana:

Gold bearing rocks and rivers with potential for alluvial mining are present over a large area of central and Northern Guyana as well as pockets within the south, creating the potential for localised contamination over large stretches of the country (Thomas, 2009). In a study carried out between 2005 and 2009, sediment samples from four areas, covering northern, central and southern Guyana (Figure 3.5) were assessed for levels of mercury contamination (Howard, 2010). Sampling was

conducted at a number of points within the Essequibo River basin, which covers a large proportion of Guyana.

**Figure 3.5:** Map of sediment sample sites in Guyana (Howard, 2010)



Mahdia (central Guyana) is located in a large mining area and acts as a hub for the industry and, during the study period, both deforestation and soil erosion, likely linked to gold mining, were an issue in the area. Arakaka, Mathew's Ridge and Port Kaituma are in the northwest of the country, and have experienced extensive mining, both historically and at the time of study, with gold mining continuing to be the main economic activity in the area. These areas were compared with the non-mined areas of Iwokrama and the Konashen Community Owned Conservation Area (COCA). Iwokrama is the largest of four currently established protected areas in Guyana, and small-scale illegal mining was thought to be an issue during the study, whereas the Konashen COCA is considered a pristine area containing the headwaters of the Essequibo River, and although the Essequibo River area is heavily mined in its lower reaches, the closest mining sites are around 200 km east of the area studied.

Samples taken from active and historically mined areas had a mean ( $\pm$  standard deviation) mercury concentration of  $0.229 \pm 0.223 \mu\text{g/g}$ , which is above CCME quality guidelines, with a range from

0.029 to 1.2 µg/g (Howard, 2010). An earlier study focused on the smaller Potaro River (close to Mahdia within the Essequibo River basin), found mercury concentrations ranging from 0.068 to 0.321 µg/g within the tributaries affected by mining activities. These are somewhat lower than those seen by Howard but still markedly above CCME guidelines (Paktunc et al., 2004).

Similar results were obtained from a study that undertook extensive sampling of sediments from the Mazaruni River, a major tributary of the Essequibo that is also significantly affected by mining, as well as a stretch of the Essequibo itself. Sediment mercury concentrations were in the range of 0.005–0.707 µg/g and 0.004–0.225 µg/g for the Mazaruni and Essequibo, respectively (Miller et al., 2003). Miller and colleagues took core samples which comprised deep floodplain deposits, below 1m, characterized by highly weathered (ferralitic) soils thought to predate mercury use in mining operations overlain by sediment formed by flooding during recent decades. The authors note that a number of features of their results are indicative of deposition from recent human activity as evidenced by mercury concentrations within the channel bed, the sand bars, and surface floodplain sediments above local background values; the decreasing mercury concentration with core sample depth suggesting recent deposition; and local downstream increases in mercury concentrations attributable nearby mining operations. However, both addition of mercury during the mining process or mobilisation of naturally occurring mercury in the soil are equally plausible explanations for the above findings (Miller et al., 2003).

In the conservation areas of Iwokrama and Konashen mean mercury concentrations were  $0.187 \pm 0.077$  µg/g and the range was 0.053 to 0.301 µg/g (Howard, 2010). The author notes that a number of the samples taken from the Iwokrama Reserve may have been affected by illegal mining in the past (as suggested by a mean mercury concentration for Iwokrama of 0.174 µg/g) (Howard, 2010). Beyond some local speculation regarding historical mining, the Konashen area is still regarded as one of the most pristine areas of forest in the Northern Amazon with the closest confirmed mining some 200 km distant (Howard, 2010). Despite this, mean mercury loadings were also above CCME guidelines (0.19 µg/g). Evidence from other areas in the Amazon basin suggesting that naturally occurring mercury is responsible for high loadings in pristine areas, however, data from Suriname and French Guiana indicate that atmospheric deposition is more likely to be responsible in the Guianas (see below). The presence of contamination in pristine areas in Guyana mirrors the situation in Suriname, although in Suriname the evidence for the pristine nature of these sites was stronger (Ouboter et al., 2012).

These results indicate that mercury contamination is widespread in Guyana with mean concentrations consistently above safe guidelines for aquatic life. Worryingly, mercury concentrations are of concern not only in areas affected by historical or active mining in Guyana, but also in seemingly pristine areas.

### **3.3 Mercury in other aquatic environments in the Guianas:**

The majority of the data summarised above comes from river and stream environments. Mercury contamination within reservoirs has also been extensively studied, however, due to the importance of these environments in the methylation process these data will be discussed in **section 4.1**.

Research on estuary and marine environments is currently sparse suggesting that these environments also deserve further research attention. As mentioned above, the lowest values (mean 0.06 µg/g) in the French Guiana study by Laperche et al (2014), were seen in estuarine river areas with the majority of river mouths sampled having low levels. For example, the heavily mined

Ouanary River had very low mercury concentrations ( $0.05 \pm 0.22 \mu\text{g/g}$ ) likely due to the influence of Amazonian sediment deposition from the marine environment. Relatively high mercury levels ( $0.11 \mu\text{g/g}$ ) were seen in the Mana estuary, which were suggested to be caused by the orientation of the estuary, which makes sediment deposition from the marine environment less substantial. However, high sediment levels ( $0.41 \mu\text{g/g}$ ) have also been found in the estuarine sediment at the mouth of the Marowijne/Maroni river (on the border of French Guiana and Suriname), which shares the same estuary as the Mana. It is important to note, however, that the mouth of Marowijne/Maroni is more favourably oriented to allow sediment deposition from the Amazon, which would be thought to reduce mercury concentration (Ouboter et al., 2012; Laperche et al., 2014). Further studies are necessary to clarify the situation regarding patterns of estuarine contamination. Studies have also shown that conditions at some river estuaries result in environments conducive to extremely high levels of methylmercury production (see section 4.1) (Muresan et al., 2008b).

There is little data on marine mercury off the coast of the Guianas. However, dilution of mercury entering the marine environment has been demonstrated at the mouth of the Sinnamary (Muresan, 2006), and levels in mangrove sediments on the coast of French Guiana have been noted as relatively low ( $0.08 \mu\text{g/g}$ ) (Marchand et al., 2006). Mercury levels in 40 samples from 6 mud flats of the coast of Suriname were low (range  $0.03\text{--}0.08 \mu\text{g/g}$ ) (Ouboter in press). Conversely, a currently unpublished study on coastal bird contamination conducted in 2013 off French Guiana showed significant pollution rates with mercury contamination up to 3 times higher than those known to disrupt reproductive hormones in birds sampled in the Arctic (Guyaweb, 2015).

### 3.4 Levels of Mercury in the Soils of the Guianas:

Data on the distribution and contamination of soil in the Guianas is sparse compared to data on freshwater sediments, although soil concentrations have been determined in and around mining sites in the Brownsberg Nature Park, Suriname (Arets et al., 2006). This study indicated levels were similarly high in both primary and secondary forest soils, and in areas between mining pits ( $0.2 \mu\text{g/g}$ ), and that concentrations at the entrance, exit, and within a mining pit were further elevated but not as significantly as expected ( $0.31, 0.30, 0.25 \mu\text{g/g}$ ). Comparison with the concentration within the sluice box itself ( $0.78 \mu\text{g/g}$ ) suggests that the majority of mercury is transported to sites somewhat removed from the mining area, as indicated by the reduced levels of organic matter and clay within the mining pit, with which mercury tends to associate (Arets et al., 2006). However, the level of contamination close to mining sites is not uniform. In French Guiana, formerly gold-mined soils had high levels of contamination in some samples but also showed very broad variability from  $0.09\text{--}9.22 \mu\text{g/g}$  (Guedron et al., 2009).

Richard et al. (2000), took samples from forest soils rich in organic matter ( $n = 15$ ) and sandy lateritic soils ( $n = 5$ ) within the heavily mined Sinnamary basin in French Guiana. The authors took core samples and carried out vertical sub-sampling where possible, detecting a mean mercury concentration of  $0.32 \mu\text{g/g} \pm 0.18$  in forest soils, and  $0.10 \mu\text{g/g} \pm 0.05$  in lateritic soils (Richard et al., 2000). Similar levels were seen by Guedron in pristine soils in French Guiana ( $0.01\text{--}0.49 \mu\text{g/g}$ ). Overall, mercury levels were found to correlate with the levels of organic material within the soil samples, with forest soils showing peak maximum levels and a large amount of variability ( $0.05\text{--}0.83 \mu\text{g/g}$ ) (Richard et al., 2000). These results are in agreement with other studies, some of which were carried out in pristine areas, in terms of mercury concentrations and the association of high mercury levels with soil rich in organic material (Roulet and Lucotte, 1995; Guedron et al., 2006; Grimaldi et al., 2008; Guedron et al., 2009).

A single study quantifying natural lithogenic mercury by comparison with concentrations of other elements which are resistant to weathering (Nb, U, Zn, Fe) natural lithogenic mercury was found at relatively steady concentration and always below 0.04 µg/g in the 4 soil profiles (3 profiles on the same toposequence: ferralsol, Acrisol, hydromorphic soil and a single Acrisol from close to a mine using mercury). In contrast exogenic mercury, deposited from the atmosphere, varied with the highest concentrations, up to 0.5 µg/g were found in the samples close to the mine and also the ferralsol (Guedron et al., 2006).

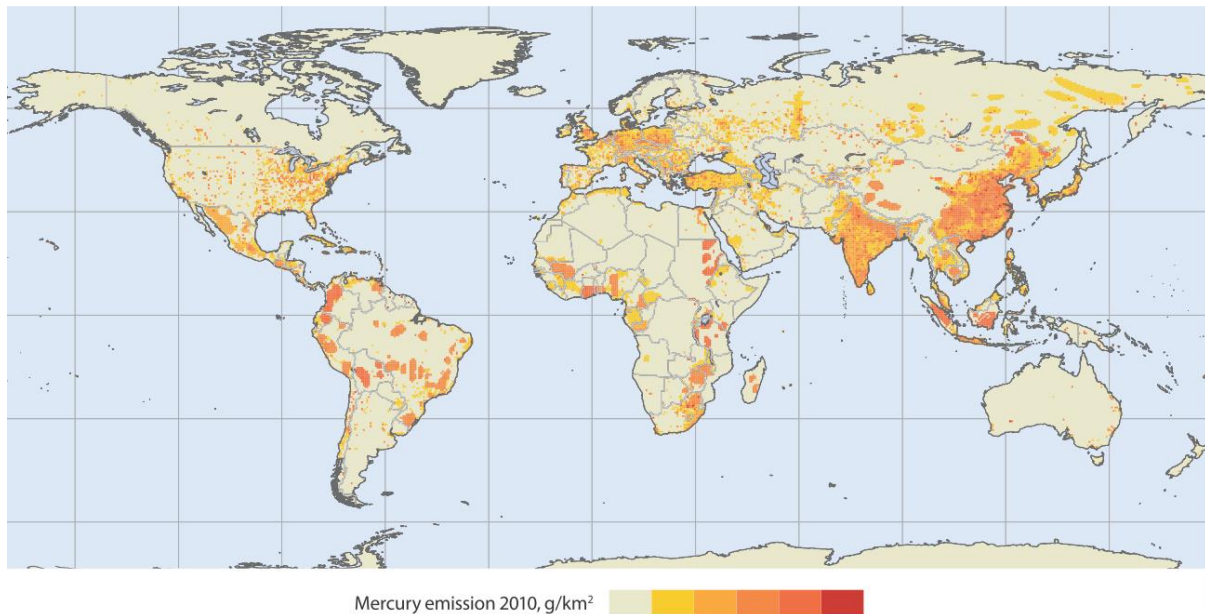
Not all soil samples show signs suggested of atmospheric deposition, however, when Richard and colleagues examined core samples from the Sinnamary basin they did not reveal significant vertical variation or patterns between different sample sites (Richard et al., 2000). Conflicting results have been found from pristine areas in French Guiana with one study showing no vertical variation in mercury concentrations (Roulet and Lucotte, 1995). However, in another study core samples did show a decreasing mercury gradient with depth and absorption of mercury contained in rain water by soil was demonstrated, giving direct evidence of atmospheric deposition (Guedron et al., 2009).

### **3.5 Levels of Mercury in the Atmosphere of the Guianas:**

Atmospheric mercury is a worldwide issue, as evidenced by the UNEP's Global Mercury Assessment Report. Atmospheric transport is the main mechanism by which mercury travels round the world and there is strong evidence for deposition of anthropogenic mercury in pristine areas around the globe (Fitzgerald et al., 1998; UNEP, 2013; Sprovieri et al., 2010). Atmospheric levels tend to be viewed over a large scale, with three distinct systems identified as covering the northern and southern hemispheres, and the tropics (Sprovieri et al., 2010; Müller et al., 2012).

A recent review indicated that levels of atmospheric mercury in South America were greater than the expected background level for the southern hemisphere (Sprovieri et al., 2010). Historic precious metal processing in South and Central America is thought to have resulted in the release of 118,000 tonnes of mercury to the atmosphere between the years 1587 and 1900 (Nriagu, 1994). Remobilisation of only 1% of this historical release would match current global annual anthropogenic release of 1960 tonnes (**Figure 3.1**) (UNEP, 2013). Currently, there are significant gaps in knowledge of emissions in South America, and across the Guianas in particular (**Figure 3.6**).

**Figure 3.6:** Map of global mercury emissions (UNEP, 2013)



A site at Nieuw Nickerie, on the coast of northwest Suriname, provides the only year-round recording of mercury levels within the tropics. Data from this site suggest that significant release of mercury may be occurring within the tropics (Müller et al, 2012). The position of the Guianas, which are passed over by the intertropical convergence zone twice each year, means that measurement of deposition of mercury from both the northern and southern hemispheric air masses is possible. Data from this site in Nickerie shows that the 2007 background levels of atmospheric mercury, 300m from the Atlantic coast and remote from sites of mining or burning of amalgam in gold retailers, are 1.40 ng/m<sup>3</sup>, which is in line with global averages (Müller et al., 2012). Similar results for background levels (1.45–1.52 ng/m<sup>3</sup>) were detected in a study by Wip et al. (2013) which sampled the atmosphere in and around Paramaribo.

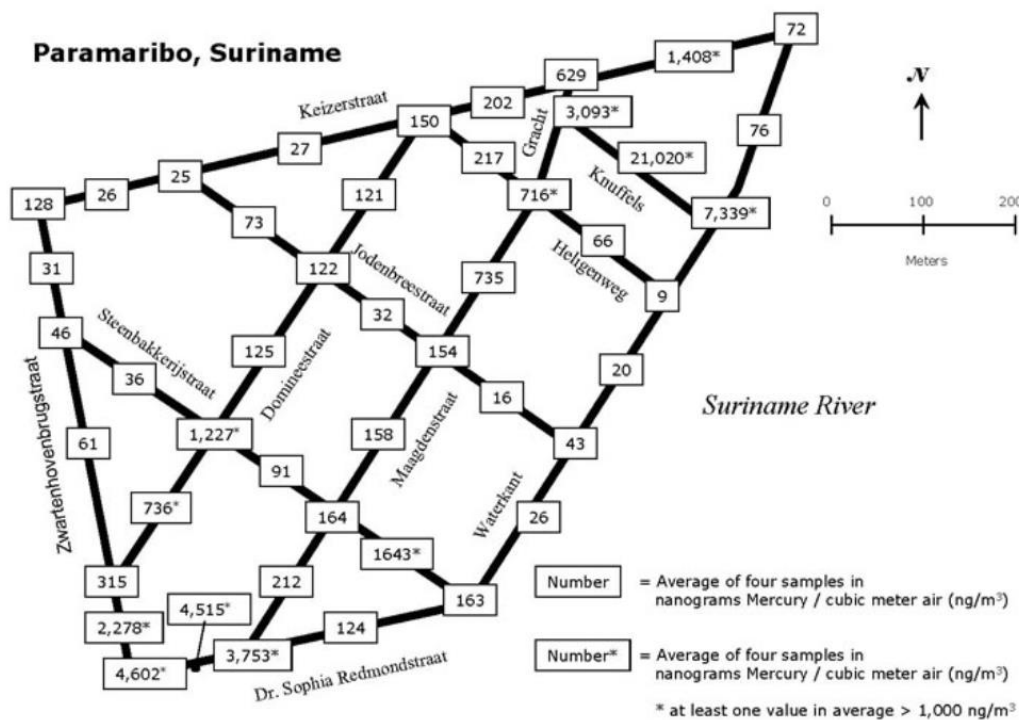
The majority of mercury entering the atmosphere within the Amazon Basin comes from SSGM through burning amalgam (Artaxo et al., 2000) (see section 5). However, release of mercury from polluted water bodies also occurs, as illustrated by the Petit-Saut reservoir in French Guiana, which releases mercury at an approximate rate of 18kg/year (Muresan et al., 2007). Evidence from French Guiana also indicates that atmospheric mercury cycling appears to be more rapid and dynamic in the tropical rain forest environments than in temperate environments (Amouroux et al., 1999).

Further data from French Guiana showed higher levels of atmospheric mercury in the gold mining area of Petit Inini River basin (15.0 ng/m<sup>3</sup>) compared to the Petit Saut Lake (2.8 ng/m<sup>3</sup>), suggesting that gold mining is affecting mercury mobilisation (Amouroux et al., 1999). Additionally, a study looking at atmospheric mercury absorbed by leaves indicated that concentrations may be higher in parts of French Guiana than in other areas of the Amazon basin (Roulet et al., 1999) although consistently high concentrations have not been found in all studies (Mélières et al., 2003).

In the city of Paramaribo, high maximum concentrations of 109.4 ng/m<sup>3</sup> were found close to gold retailers compared with a mean of 5.6 ng/m<sup>3</sup> (Wip et al., 2013). Authors note that the per capita release in Paramaribo was 3.5–6.5 times greater than that seen in areas associated with high levels of mercury due to large populations, coal burning and other industrial activity such as the automotive industry hub of Changchun City in China (Fang et al., 2004). Spikes in concentration over

5 times higher, and of longer duration than those seen in New York City were also found in Paramaribo (Carpi et al., 2002). An earlier study of central Paramaribo showed very high atmospheric mercury levels (**Figure 3.7**) leading to an estimated 1.2 tonnes of mercury being deposited in Paramaribo each year (SIH Fund, 2005).

**Figure 3.7:** Atmospheric contamination in central Paramaribo, Suriname (SIH Fund, 2005)



Beyond local health risks (see section 5), the importance of atmospheric mercury across the Guianas is principally as a putative transport mechanism from gold mining to non-mining areas. As previously mentioned, pristine areas in central Suriname are affected by mercury contamination (Ouboter et al., 2012), with deposition from the air one of the principal means through which contamination may be occurring. Evidence for atmospheric deposition includes the fact that contamination in pristine areas of Suriname correlates with areas that received high rainfall and is in the path of trade winds from mining areas (Ouboter et al., 2012). As previously noted, studies of soil in some pristine areas of French Guiana have shown a decreasing vertical gradient of mercury contamination, indicative of atmospheric deposition (Guedron et al., 2009) and contamination of pristine areas has also been demonstrated during rainfall events (Tessier et al., 2003; Guedron et al., 2011). Ouboter et al. (2012) note that there are no mercury-bearing rocks in the contaminated pristine areas, and neither is there significant deforestation, factors which have been blamed for mobilisation of soils with high natural (lithogenic) mercury content in other areas of the Amazon basin (Ouboter et al., 2012; Roulet, 1998).

### **3.6 Summary of Mercury Contamination in the Guianas:**

Some patterns appear to be consistent when looking at mercury presence and contamination in the three Guianas. Contamination of freshwater sediments is widespread affecting large areas of all three countries. There is great heterogeneity in mercury concentrations between individual samples, particularly at gold-mined sites, suggesting that extensive sampling is required to create a complete picture at both a local and countrywide level. Little is known about local atmospheric release directly from SSGM and the mercury content of terrestrial soils in any of the territories. Studies to sample air in and around SSGM operations should be carried out in order to better estimate levels of release to the atmosphere and the impact on miners' health (**see section 5**). More widespread and up to date studies on terrestrial soil are required, particularly in view of the evidence for a marked increase in the use of mercury in the past 10 years.

In French Guiana, the most in-depth survey of freshwater sediments indicates that contamination is relatively well correlated with mining activity, although a number of studies have also shown high mercury concentrations in pristine areas and the mean background level in non-mined areas is relatively high. In Suriname, contamination is high and relatively consistent throughout much of the country with pristine areas showing some of the highest mercury concentrations. In Guyana, the picture is currently less clear, and there is a definite need for an update of current data, in addition to further country-wide longitudinal studies (multiple samples taken over time) to match those which have been carried out in Suriname and French Guiana.

In other areas of the Amazon Basin, the presence of high levels of mercury in pristine areas has been blamed on natural concentration of mercury over time in association with high organic matter content (AMAP/UNEP, 2013). However, the balance of evidence, including a lack of any significant mercury containing rocks in the region; low levels of lithogenic (natural) mercury and high levels of exogenic mercury in the single quantitative study from French Guianan samples; evidence of mercury contamination and direct absorption by soil following rainfall events; and the association of high levels of contamination in Suriname with areas of high rainfall suggests that atmospheric deposition of contamination from SSGM is a critical driver of mercury pollution in the Guianas.

## **4.0 Mercury and the food chain**

### **4.1 Mercury methylation:**

The impact of mercury pollution on humans and wildlife is dependent on the form of mercury present. Methylmercury is more toxic than inorganic mercury and is absorbed by organisms from the environment at a much higher rate (Stanford, 1971). The production of methylmercury from mercury is a constant, ongoing process, occurring principally through the action of microorganisms, particularly sulphur reducing bacteria (Stanford, 1971; Coquery et al., 2003; Muresan et al., 2008). The production of methylmercury is likely to be enhanced in the tropics, due to environmental conditions. These favourable conditions include high temperature; acidic waters; plentiful organic matter favouring oxidation and subsequent binding with organic compounds; and increased biological activity in general (Lacerda and Salomons, 1998; WHO, 1990; Richard et al., 2000; Gray et al., 2002).

The process of mercury methylation primarily occurs in aquatic environments, where it is readily accumulated by living organisms, binding to both lipids (fats) and proteins within the body. The concentration of methylmercury increases in line with the trophic position in a process termed

biomagnification (Gray et al., 2002). The increasing concentration of methylmercury in the food chain, occurs independently of increasing concentrations of methylmercury in the environment (Stanford, 1971).

As would be expected, sites close to areas of mining appear to have higher methylmercury concentrations when compared with uncontaminated sites. A good example is seen in both water (0.05–3.8 ng/L vs 0.08–0.28 ng/L) and sediment (< 0.02–1.4 ng/g vs 0.03–0.08ng/g) samples from Gros Rosebell, Suriname (Gray et al., 2002). However, due to the rapid absorption by organisms, these methylmercury concentrations in the environment may be misleadingly low (D'Itri, 1990; Veiga, 1997). To address this, ratios called bioaccumulation factors (BAF) are used to calculate the relationship between concentrations of mercury in fish and the water they inhabit. As noted, these concentrations increase with trophic level and they also vary depending on conditions. For example, the standardised BAF in still freshwater for herbivorous fish (trophic level 3) is 1,115,000:1 and in predatory fish (trophic level 4) is 5,740,000:1. In fast-flowing fresh water the BAFs for herbivorous and carnivorous fish are 517,000:1 and 1,240,000:1, respectively (Sanborn and Brodberg, 2006). However, significantly higher BAF from 16–28,000,000:1 have been recorded in the piscivorous, and highly prized for eating, species *Hoplias aimara* (anjumara) in French Guiana (Boudou et al., 2005). Put another way, the concentrations of mercury in this fish, much-prized for human consumption, were up to 28 million times the levels in the surrounding water.

The potential for methyl mercury production varies across the aquatic systems of the Guianas, as illustrated by comparing mined and non-mined rivers upstream of the Petit Saut reservoir, in French Guiana, with a site of water outflow from the reservoir dam into the Sinnamary River. The mined river (LeBlonde) showed the highest mercury concentrations (25.4–34.9 ng/L), with lower levels seen in both the unmined river (Courcibo; 2.1–5.4 ng/L), and dam water outflow (2.4–3.4 ng/L). Mercury levels in predatory *H. aimara* were likely to be above guideline concentrations at all the sites tested (dry weight mercury concentration was not calculated so this cannot be confirmed). However, the difference in aquatic mercury content between the LeBlonde and Courcibo was not reflected in fish caught from these two rivers. Furthermore, the mercury concentrations in fish at the site below the reservoir outflow were up to 8 times higher in some species than that seen at the river sites upstream of the reservoir. This is despite the downstream site showing the lowest maximum aquatic mercury level of the three sites tested. Investigation into the methylmercury levels at the three sites indicated similarities between the two rivers (0.03 - 0.06 ng/L), compared with a ten-fold increase in the outflow of the Petit Saut reservoir hydroelectric dam at Sinnamary (maximum of 0.56 ng/L) (Boudou et al., 2005). These data highlight the importance of methylmercury over inorganic mercury in bioaccumulation, and point to a role played by dammed or stagnant-water areas in methylmercury production.

The large Petit Saut hydroelectric reservoir was created by flooding 350 km<sup>2</sup> of forest in the extensively mined Sinnamary Basin. Both mercury pollution in general and the methylation process have been extensively studied at the site (Coquery et al., 2003; Peretyazhko et al., 2005; Muresan et al., 2006; Muresan et al., 2008). A study looking at differences in mercury within the reservoir found that in the upper more oxygen rich layers up to 40% of the mercury is in its inorganic form, however, the deeper anoxic (oxygen depleted) layers contained less than 10% inorganic mercury (Peretyazhko et al., 2005). Typically, methylmercury makes up around 1% of the mercury found in freshwater sites in the Guianas (Coquery et al., 2003) because it is so readily absorbed into the food chain. However, below the oxycline (the point of transition between oxygenated and anoxic waters) in the Petit Saut reservoir, methylmercury was found to comprise 15% of mercury detected (Peretyazhko et al., 2005) and deep in the anoxic layer close to the dam floor the proportion of methylmercury was

consistently high (20–35%) (Coquery et al., 2003). In some samples from Petit Saut, methylmercury represents up to 40% of the mercury detected (Muresan et al., 2008).

The estimated methylation rate for the reservoir was 0.06%/day in 2004, (ie the proportion of inorganic mercury being converted to methylmercury), resulting in methylmercury production of 0.04 mg/m<sup>2</sup> of sediment surface per year (Peretyazhko et al., 2005; Muresan et al., 2008). Looking at the reservoir as a whole, net production in 2003/04 was 8.1 moles/year (1.62 kg), resulting in output to the Sinnamary estuary of 13.5 moles/year (2.7 kg) (inputs to the reservoir were 5.4 moles/year) (Muresan et al., 2006; Muresan et al., 2008). Whilst 2.7 kg may not sound like a significant amount if it were concentrated purely in fish, this yearly output would be sufficient to put 5,414 tonnes of fish at the limit of the WHO guideline of 0.5 µg/g for edible fish (WHO, 1990). Hence, the Petit Saut, and similar reservoirs such as the Brokopondo in Suriname, have been described as ‘man-made reactors’ creating perfect conditions for methylation, and subsequently releasing high levels of the toxic compound to the food chain (Mol et al., 2001; Muresan et al., 2008).

When planning new hydroelectric dams in the Guianas, the risk of methylmercury production should be a serious consideration, particularly given the relatively high mercury levels in even pristine soils and the potential for up to 20% of soil reserves to be quickly released to the water column following flooding (Roulet and Lucotte, 1995). However, these areas should not be seen as the only points of risk, as anoxic conditions may exist in many aquatic environments including river sediments, tailing ponds of SSGM operations, and at some oceanic/estuarine boundaries creating engines for the production of high levels of methylmercury (Yu et al., 2001; Guedron et al., 2011). In the upper Sinnamary estuary, for example, the presence of large amounts of organic materials, sulphides and a freshwater oxycline create conditions that promote methylation in a similar fashion to those seen in deep water areas of reservoirs. As a result, the area exports up to 80% more methylmercury than it receives from upstream sources (60 ± 20 %), leading to production of up to 5.5 ± 0.7 kg of methylmercury each year, enough to contaminate 11,000 tonnes of fish flesh. (Muresan et al., 2008b).

#### **4.2 Level of Fish Contamination in the Guianas**

Diet is the chief route of exposure to mercury in the non-mining population of the Guianas, principally via eating contaminated fish. Piscivorous (fish eating) wildlife are also at significant risk from mercury toxicity (Sanborn and Brodberg, 2006). Due to preferential absorption the vast majority of mercury found in fish flesh (97–100%) is in the form of highly toxic methylmercury (Laperche et al., 2014). The relationship between the extent of environmental contamination and mercury concentrations in fish is complex. Multiple factors including trophic level, river-current speed, extent of mercury methylation and fish age all play a part in determining levels of toxicity. For example, piscivorous or carnivorous fish are likely to have higher mercury levels due to their higher trophic position than herbivorous fish. Similarly, older fish, generally identifiable by larger size, are likely to have higher mercury concentrations than smaller younger fish of the same species in the same area.

In the following section, the extent of contamination in the freshwater fisheries of the Guianas is summarised. As there is currently little knowledge regarding levels of contamination in marine and estuarine waters the data available will be summarised at the end of this section.

#### 4.2.1 Mercury contamination in the freshwater fish of French Guiana:

In French Guiana two large studies have assessed mercury contamination in freshwater fish stocks (Richard et al., 2000; Laperche et al., 2014). Richard et al. (2000), surveyed mercury levels in fish, alongside their investigations into sediment contamination in the Sinnamary basin. The authors chose regularly fished sites on 9 rivers: Maroni (Marowijne), Mana, Iracoubo, Counamama, Sinnamary, Koori, Montsinéry, Comté and Approuague. A total of 43.7% of the samples taken were from carnivorous fish (23 species), with the remainder (56.3%) being non-carnivorous (13 species).

The mean mercury concentration across all samples was  $0.24 \pm 0.29 \mu\text{g/g}$  ( $n = 696$ ), with carnivorous species showing levels of  $0.48 \pm 0.28 \mu\text{g/g}$ , compared with  $0.05 \pm 0.07 \mu\text{g/g}$  in non-carnivorous species. Similarly, when looking at which species showed the highest and lowest levels of methylmercury toxicity, the lowest concentration was seen in a non-carnivorous species (*Myleus sp.*;  $0.01 \pm 0.01 \mu\text{g/g}$ ), with the highest in the carnivorous Tiger Catfish (*Pseudoplatistoma fasciatum*;  $1.11 \pm 0.40 \mu\text{g/g}$ ). Of the 23 carnivorous species surveyed, five showed a mean methylmercury concentration at or above the WHO guideline ( $0.5 \mu\text{g/g}$ ), representing 43.1% of the samples taken from carnivorous fish. *H. aimara*, showed mercury levels of  $0.61 \pm 0.22 \mu\text{g/g}$ . Further investigation revealed that in carnivorous species, size was a predictor for mercury toxicity levels. No non-carnivorous species showed mercury levels above the WHO limit (Richard et al., 2000).

Looking at sample site related differences, the highest mean concentration of mercury for all fish species combined was seen in the Maroni River with  $0.48 \mu\text{g/g}$ . Carnivorous species in the Sinnamary river showed the highest mean concentrations ( $0.57 \mu\text{g/g}$ ), with the Counamama River featuring the highest mercury levels in non-carnivorous species ( $0.18 \mu\text{g/g}$ ).

Laperche et al., (2007) also undertook extensive sampling of fish in their report for the French Geological Survey (BRGM), which included the sediment results summarised in **section 3.2.2** (Laperche et al., 2014). Sampling was conducted at 45 sites across the country with mercury contamination assessed in close to one thousand fish ( $n = 974$ ) (Laperche et al., 2007). As expected, there was a gradual rise in tissue mercury concentrations in line with trophic position, with piscivores exhibiting a mean concentration above the  $0.5 \mu\text{g/g}$  WHO limit.

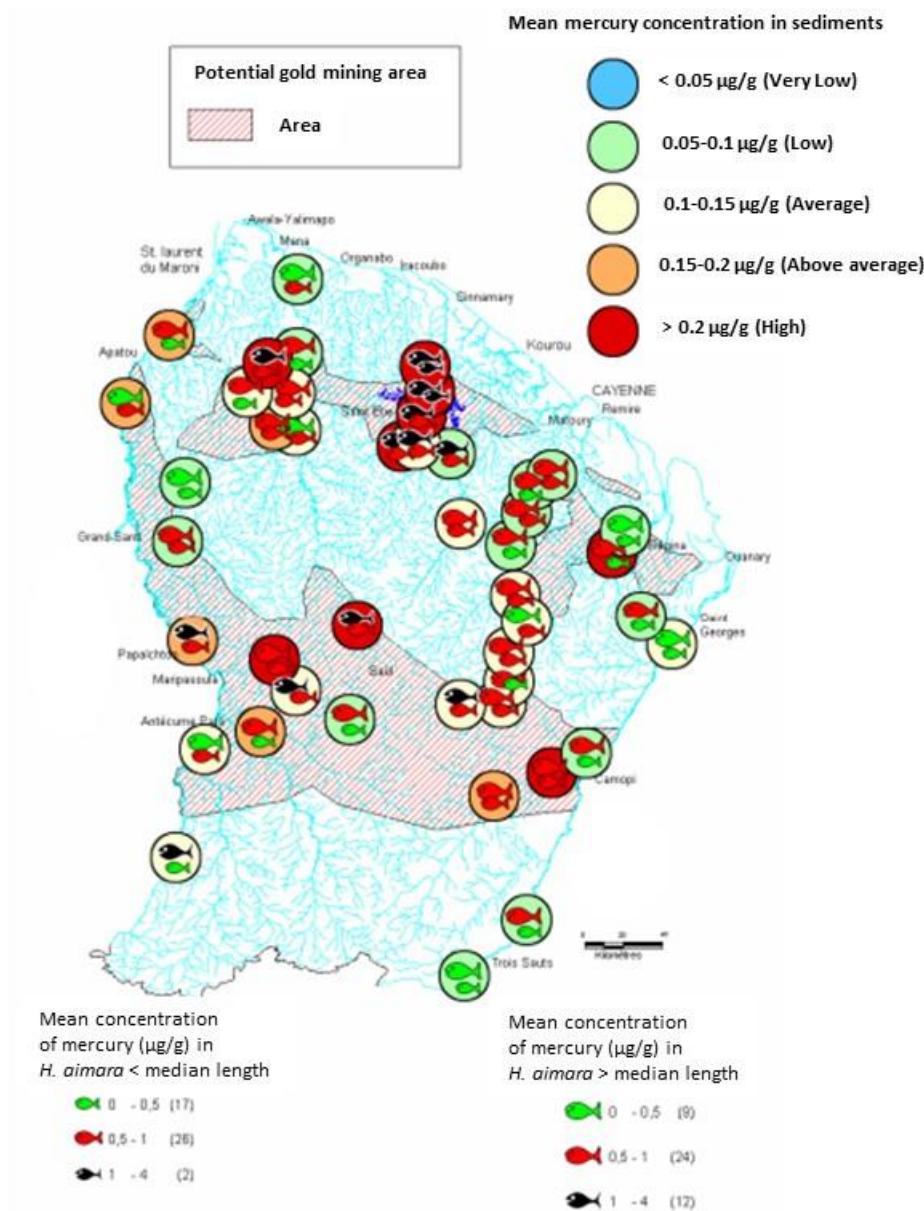
The authors focused on *H. aimara*, a highly prized eating fish thought to be a good indicator species for assessing biomagnification (Durrieu et al., 2005). The fish were split by size class, into those measuring 38–58 cm and 58–89 cm (**Figure 4.1**). Controlling for size is particularly important in this species as alongside the above noted relationship between fish age/size and mercury levels, *H. aimara* is omnivorous when small before moving to a strictly piscivorous diet thus altering exposure of individual fish to dietary mercury (Richard et al., 2000). At 62.3% of sites sampled, fish in the smaller size class had mean mercury concentrations above the WHO  $0.5 \mu\text{g/g}$  limit. At two sites (4.4%), concentrations were more than double the safe limit. For larger fish, mean mercury concentrations were above the safe limit at 80% of the sample sites. In over a quarter of sites (26.6%), and particularly around the heavily mined Sinnamary River and downstream of the Petit Saut reservoir, methylmercury concentrations double the safe limits were observed.

Interestingly, mercury concentrations in *H. aimara* did not follow the pattern of correlation with areas impacted by mining seen in the sediment samples. Although concentrations in fish mercury were consistently high in areas with high sediment mercury loads, concentrations above the WHO limit were also seen at a number of sites with sediment contamination well below the estimated maximum background levels for French Guiana ( $0.15 \mu\text{g/g}$ ). Indeed, there are other examples from French Guiana where differences in inorganic mercury levels are not reflected in contamination of

fish as discussed above (**see section 4.1**) (Boudou et al., 2005). Whether this phenomenon is a consequence of biomagnification or due to issues in detecting mercury contamination within sediments remains to be elucidated. However, the end result is that WHO safe dietary limits are exceeded at sites across the country.

Comparison of the above data from 2000 with that from 2007 (Laperche et al., 2007; Richard et al., 2000) suggest a worsening picture over this period. This is also backed up by data from a doctoral thesis indicating that following a steady decline from 1993–2000, the level of mercury contamination in the upper Sinnamary began to rise (Dominique, 2006). Given the long-term nature of mercury pollution year to year fluctuations in contamination may appear counterintuitive, however, rapid changes in mercury concentrations in fish have also been reported in the US. Globally, the data are complex and heterogeneous depending on location with a number of studies suggesting a long-term legacy of mercury pollution with little short term fluctuation (Carrie et al., 2010; Tang et al., 2013; AMAP/UNEP, 2013).

**Figure 4.1:** Comparison of the average concentration of mercury in sediments and in the *H. aimara* in French Guiana. Adapted from Laperche et al, 2007, BRGM/RP-55965-FR – Rapport final



#### 4.2.2 Mercury contamination in the freshwater fish of Suriname:

The most up-to-date and extensive data on contamination levels in freshwater fish (N = 855) in Suriname comes from the same pooled analysis of data by Ouboter et al. (2012) summarised in **section 3.2.3**. Analysis revealed 41% of piscivorous fish had mercury concentrations above the recommended WHO value of 0.5 µg/g. An earlier study on the Commewijne River, conducted in 2000, found mercury levels above 0.5 µg/g in 33% of predatory fish sampled (Quik and Ouboter, 2000), while another study (N = 318) found 57% of predatory fish in a river affected by nearby gold mining were above the WHO limit (Mol et al., 2001).

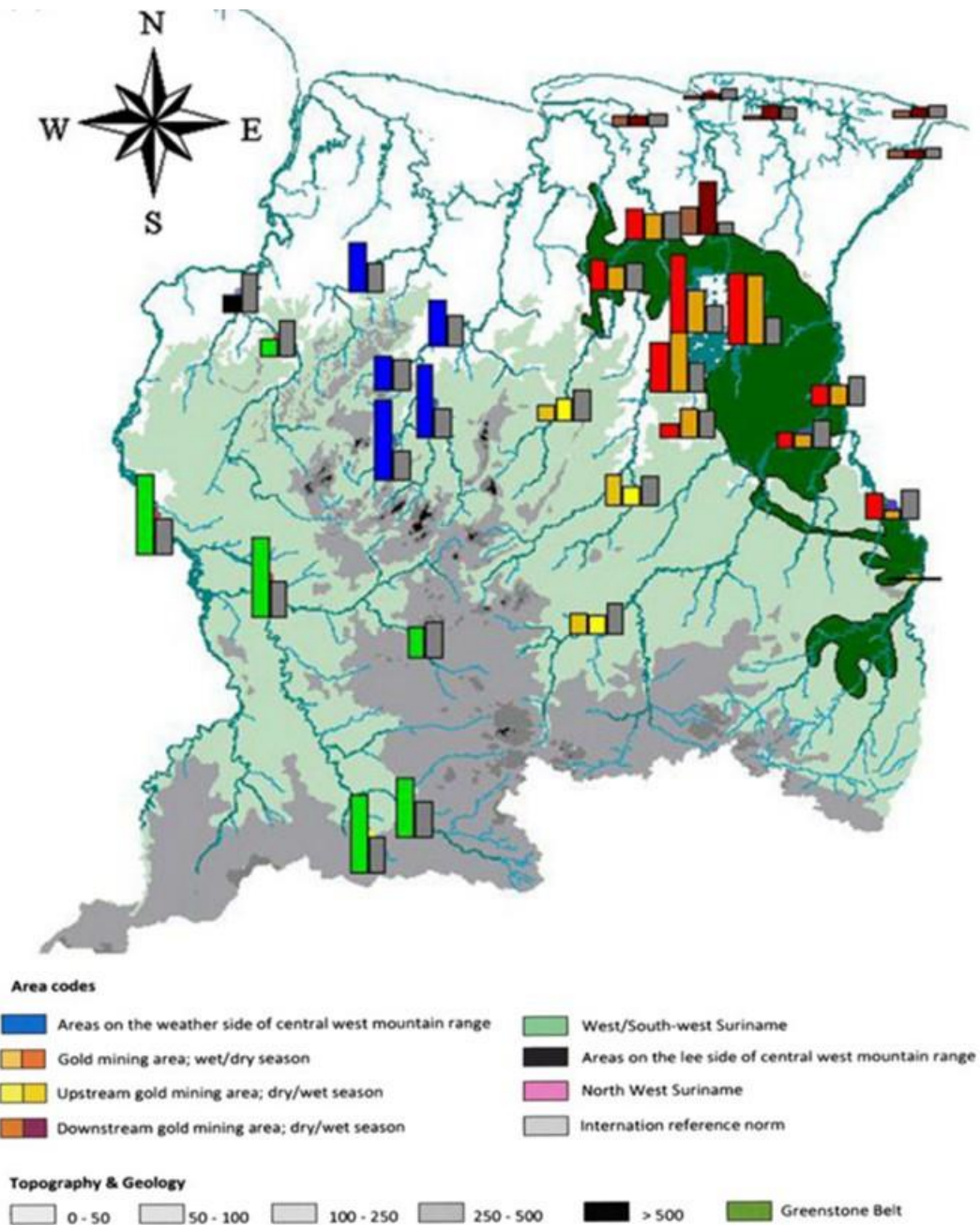
Overall, the mean mercury concentrations found in piscivorous fish in gold mining areas were consistently higher than the WHO guideline threshold (**Figure 4.2**) (Ouboter et al., 2012). Extremely

high levels, of up to 4.26 µg/g (over 8.5 times WHO guidelines), were found in a previous study on the Brokopondo reservoir (Mol et al., 2001; Mol et al., 2007). Oddly however, in the bioindicator species *H. aimara*, mean mercury levels were below the WHO limit in both gold mining areas ( $0.46 \pm 0.23$  µg/g), and also in the Brokopondo reservoir ( $0.43 \pm 0.29$  µg/g), this is likely an artefact due to the low number of fish sampled (Ouboter et al., 2012).

High concentrations of mercury were also present in fish from non-gold mining areas (**Figure 4.2**). Despite similar concentrations of mercury in river sediment from both gold mining and pristine areas (approximately 0.2 µg/g), higher fish mercury concentrations were observed in pristine areas both in *Serrasalmus rhombeus* and *H. aimara*. The pattern of high concentrations in pristine areas held true even in south-western Suriname, where sediment levels of mercury were below those found in gold mining areas (1.4 and 2.2 µg/g, respectively) but the mean mercury concentration in both *S. rhombeus* (0.75 and 0.42 µg/g, respectively) and *H. aimara* (0.65 and 0.46 µg/g, respectively) were higher. In line with previous results, mercury concentration increased with increasing size in *S. rhombeus*. The possibility that older, larger fish were being caught in pristine areas was considered. However, even when fish size was controlled for, the high concentrations of mercury found in pristine areas remained consistent (Ouboter et al., 2012).

The northwest of Suriname is considered to have the least contaminated sediment in the country. In 2001 Mol et al., took fish samples from this area (Bigi Pan Lagoon) revealing the lowest mercury concentrations obtained during their study. Mean mercury levels in the piscivorous snook (*Centropomus undecimalis*) and tarpon (*Megalops atlantics*), were found to be 0.04 and 0.03 µg/g, respectively (Mol et al., 2001). It is worth noting that these fish were from brackish lagoon waters and more up to date data from freshwater systems in these areas should be obtained to assess the contamination of fish stocks. However, beyond this work, insufficient species specific data on fish contamination has been done.

**Figure 4.2:** Mean mercury concentration in piscivorous fishes in Suriname. Grey bars represent the WHO standard for human consumption of 0.5 µg/g (Ouboter et al., 2012).



#### 4.2.3 Mercury contamination in the freshwater fish of Guyana:

There is a dearth of information on the contamination of fish stocks in Guyana, with the majority of the available data out of date and/or spatially isolated.

In 2001, the GGMC performed a survey of the Potaro River, and found that 57% of carnivorous fish sampled had mercury levels above the maximum WHO guideline concentration (0.5 µg/g). No non-carnivorous fish were above the WHO limit (GGMC, 2001; Hay and Vieira, 2005).

Singh et al. (2000), carried out a study of two rivers in mining areas in the northwest of Guyana, the Isseneru and the Kurupung. A total of 32 species were sampled across both rivers, 18 in the Kurupung and 28 in the Isseneru with some occurring at both sites, with 39% (Kurupung) and 25% (Isseneru) of species surveyed showing a mean mercury concentration above the WHO guideline limit. Methylmercury concentrations ranging from 0–1.81 µg/g were detected in fish from the Isseneru, with values of 0.24–0.928 µg/g seen in the Kurupung.

Looking at individual species at each site, the minimum concentration in the Kurupung was seen in *Chalceus macrolepidotus*, whereas the highest was from *Cynodon gibbus*, both of which are carnivorous. In the Isseneru, the lowest concentration was detected in *Satanoperca jurupari* (omnivorous) while the highest was in *Platynemichthys notatus* (carnivorous). Mean levels of methylmercury in the bioindicator species *H. aimara* were above the WHO guideline limit in the Kurupung (20 samples) but below in the Isseneru (5 samples). Caution must be exercised when interpreting these data, due to the small number of samples for some species (often only one or two), but also due to the fact that the length of fish, as a proxy for age, was not recorded.

#### 4.2.4 Summary of mercury contamination in the freshwater fish of the Guianas:

In Guyana, as with data on mercury contamination in general, there is a need for large-scale studies to improve on the completeness of the current data. However, given that there is no evidence of reduced mercury use, the sparse and out-dated information currently available suggest that high levels of contamination are likely present. Indeed data from Suriname and French Guiana, where mercury use has been similarly pervasive, reveals widespread and high levels of mercury contamination in fish. Furthermore, contamination is not restricted to areas affected by SSGM and levels of contamination in fish are sometimes greater than suggested by levels of contamination in river sediments at certain sites in both countries.

Neither, *S. rhombeus* or *H. aimara* demonstrate extensive migratory behaviour, indicating that movement of fish from areas of high contamination to less contaminated areas is unlikely to be responsible for the mismatch between mercury levels in river sediment and those found in fish (Mol et al., 2001; Ouboter et al., 2012). There are several hypotheses to explain these apparently anomalous findings, all as yet unproven. One explanation may be that in undisturbed rivers mercury is more freely available for methylation and bioaccumulation. The binding of mercury to the sediment released during mining may reduce methylation and bioavailability for uptake to the food chain by, for example, keeping mercury suspended in oxygenated portions of the water column rather than in anoxic sediment layers on the river bed (Morel et al., 1998). Conversely, in pristine areas mercury is deposited from the atmosphere without the increase in sediment load caused by mining and may therefore be more available for methylation and bioaccumulation (Ouboter et al., 2012). Another possible explanation is that mining related disruption of the aquatic biome in terms of species diversity and reproduction, demonstrated at higher trophic levels (Mol and Ouboter, 2004), is also evident at the level of microorganisms leading to less efficient methylmercury production. In such circumstances pristine systems with an undisturbed microbiome may result in more efficient production of methylmercury.

Regardless of the reason, the presence of contaminated fish stocks in pristine areas alongside mismatches between levels of inorganic mercury and fish contamination indicate that the avoidance of fishing in mining areas and testing for environmental contamination are insufficient means of safeguarding communities and wildlife from mercury toxicity.

### 4.3 Contamination of estuarine and marine fish in the Guianas

As with data on marine and estuarine environments in general, little is currently known regarding contamination of fish in these ecosystems. A single study by Mol et al. (2001), investigated fish from three different habitats: freshwater fish from both contaminated rivers and the Brokopondo reservoir ( $n = 318$ ); estuarine fish from both pristine and contaminated rivers ( $n = 109$ ) and marine species ( $n = 110$ ). They found mean mercury concentration in piscivores was highest in freshwater species ( $0.71 \pm 0.59 \mu\text{g/g}$ ), and lowest in estuarine species ( $0.22 \pm 0.16 \mu\text{g/g}$ ), with marine species occupying a middle ground with significantly higher levels of contamination ( $0.34 \pm 0.31 \mu\text{g/g}$ ) than estuarine fish. However, fish with mercury levels above the WHO guideline limit were found in both estuarine and marine waters. Although mercury levels in estuarine fish were generally lower than at other sites, key commercial species such as catfish ( $0.03\text{--}0.69 \mu\text{g/g}$ ) and sciaenids ( $0.03\text{--}0.63 \mu\text{g/g}$ ) exhibited mercury levels which peaked above the acceptable limits despite significant distance from upstream SSGM ( $> 100 \text{ km}$  distant). Mean mercury concentrations well above the WHO limits were also detected in another commercially important marine species, the crevalle jack (*Caranx hippos*;  $1.17 \mu\text{g/g}$ ) (Mol et al., 2001).

It is unclear why marine species showed higher mercury levels but the authors suggest that contamination of marine species may be due to transport of contaminated sediment from the Amazon plume (Mol et al., 2001). However this does not fit with data which indicated that similar plume sediments are actually linked to reductions in mercury contamination in estuarine sediments (Laperche et al., 2014) or low levels of bioavailable mercury off the northern Brazilian coast (de Moura et al., 2012b). An alternative theory suggests marine fish are being affected by the output of estuarine methylmercury production created by marine/freshwater oxyclines (see section 4.1) (Muresan et al., 2008<sup>b</sup>), although further investigation into relevant site-specific conditions would be required to confirm this.

Given the levels of contamination, and their importance for commercial fishing, more research into contamination of marine and estuarine fish species is required. Brazilian studies have investigated mercury contamination in the Guiana dolphin (*Sotalia guianensis*), with the data suggesting it could be an effective bioindicator species (mean mercury concentrations:  $0.4 \pm 0.16 \mu\text{g/g}$ , Amapa state;  $1.07 \mu\text{g/g}$ , Rio de Janeiro state) (de Moura et al., 2012; de Moura et al., 2012b). Therefore, as bycatch monitoring of *S. guianensis* is already carried out in the coastal waters of the Guianas, this could offer a cost effective way to monitor marine mercury levels (WWF, 2014).

## 5.0 Human health:

### 5.1 Mercury exposure and human health

Both elemental and methylmercury are neurotoxic in humans causing damage to the peripheral and central nervous system, leading to neurological and behavioural disorders. Exposure to mercury may occur through inhalation, ingestion or skin contact (WHO, 2007). Typical symptoms include tremors, motor dysfunction, memory loss, headaches and cognitive deficits. The extent of effects are dependent on age, health, dose and route of exposure but long lasting neurological dysfunction and even death can result (US EPA, 2015; Debes et al., 2006). Different societal groups are at risk from different sources of mercury. Small-scale gold miners, their families and those involved in processing gold are typically most at risk from inhalation of elemental mercury (UNEP, 2012; Veiga, 1997). The general population is typically most at risk from ingestion of fish contaminated with methylmercury,

with children and pregnant women being at particularly high risk due to increased sensitivity to exposure and the risk of foetal neurodevelopmental issues, respectively (WHO, 2007).

Exposure to mercury can be assessed by blood, urine and hair sampling (Table 5.1). There is a rapid rise in mercury levels in the blood following exposure to vapour with the peak blood concentration occurring on the day of exposure for a single event and a steady state being reached in cases of continual exposure. Levels of mercury in the blood are also affected by dietary exposure (de Kom et al., 1998; Barregård et al., 1993). Urine is the best indicator of average long-term exposure to mercury vapour, particularly if the exposure is intermittent and fluctuates over time (Barregård et al., 1993), such as would be the case with exposure through burning amalgam in SSGM. Hair samples are the least invasive method of assessing mercury exposure and offer the best indicator of dietary intake (Peplow and Augustine, 2012).

**Table 5.1:** Acceptable safety levels in humans

| Sample type | Most useful for:                                    | Safe Limit   | Authority   |
|-------------|---|--|-------------|
| Blood       | Short term vapour exposure. (also affected by diet) | < 10 ng/mL (normal)<br>> 50 ng/mL (significant toxicity) | Dobbs, 2009 |
| Urine       | Long-term vapour exposure (less affected by diet)   | 30–50 µg/g creatinine                                    | WHO, 1990   |
| Hair        | Dietary mercury                                     | 10–14 µg/g   | WHO, 1990   |

## 5.2 Direct SSGM-related mercury exposure in the communities of the Guianas:

For individuals and communities directly involved in SSGM the major source of mercury exposure is through inhalation of mercury vapours (UNEP, 2012; Veiga, 1997). Around 80% of inhaled vapour is absorbed by the lungs and enters the bloodstream, with the remainder being exhaled (WHO, 1990). Time-weighted air concentrations are the generally agreed upon means of assessing exposure.

Currently, there are very little data on local concentrations of mercury vapour released during amalgam burning in the Guianas but levels are known to consistently exceed the WHO limit for annual average public exposure (UNEP, 2012; WHO, 2007; WHO, 2000). Retorts are proven to reduce the amount of mercury loss during amalgam burning, however, the use of retorts in the Guianas is known to be very low (**see section 1.1 and 1.2**) In a US National Institute for Occupational Safety and Health survey in Venezuela, which involved full-shift samples from the breathing zones of both large-scale and small-scale gold miners, 20% of the samples taken were above the limit for occupational exposure. The miners surveyed also displayed elevated levels of mercury in their urine (Drake et al., 2001).

A single survey on mercury release from gold burning was conducted in Paramaribo. Mean mercury concentrations in gold buying shops were above the National Institute for Occupational Safety and Health recommended exposure level of 50 µg/m<sup>3</sup>, posing a serious health risk to employees (Wip et al., 2013). In 1998 de Kom et al. (1998), investigated blood and urine mercury concentrations in a small group (n = 28) of Maroon small-scale gold miners and a similar control population (n = 17). Mean urine mercury concentrations were significantly higher in the gold mining group (27.5 ± 21.1

µg/g creatinine [creatinine is used to correct for the concentration of the urine]), compared with controls ( $5.2 \pm 2.9$  µg/g creatinine). Conversely, levels in blood were significantly higher in controls (mean  $18.1 \pm 11.0$  µg/L for miners and  $26.8 \pm 14.6$  µg /L for controls). As mentioned, mercury levels in blood may have been affected by diet. However, as fish consumption was not recorded and the proportion of organic vs inorganic mercury was not analysed, the contribution of dietary intake could not be assessed. However, the higher urinary mercury concentration in miners, which is not significantly affected by dietary methylmercury (Berlin, 1986; WHO, 1990; de Kom et al., 1998), suggests that these Surinamese miners were being exposed to mercury through inhalation. The mean urinary mercury concentrations found were approaching those at which subclinical symptoms including neurotoxicity and objectively detectable tremor begin to occur ( $30\text{--}100$  µg/g creatinine) (WHO, 1990).

Singh et al., (2000) carried out a series of studies in two village communities from the Mazaruni Basin in Guyana, taking hair samples in 1997 and later at two time points in 2000. Kurupung which is located in an area where the principal economic activity is diamond mining was compared with Isseneru where gold mining is dominant. The majority of families in Isseneru own their own SSGM claim but subsist on farming activity, therefore the population is a mix of those directly, if intermittently, involved in mining and their families. All the homes surveyed had a small supply of mercury.

Between 89–96% of those sampled in Isseneru were above the 14 µg/g no effect limit for maternal hair (WHO, 1990; EnvHealth.org) while between 12–14% of the population in Kurupung were above the limit. Some interesting individual results were also obtained from this study. Of those residents of Isseneru who had significant reductions in mercury content, either between the two survey years or between the two sample points in the 2000 survey (February and September) one subject reported that she had not stored mercury at home since 1996 and had stopped burning amalgam in 1998 and another subject's father had stopped mining at Isseneru that year, although her mother was still involved in burning of amalgam. These data underline the relatively rapid effects of behaviour change on levels of mercury contamination measured in hair.

### **5.3 Indirect SSGM-related mercury exposure in the communities of the Guianas**

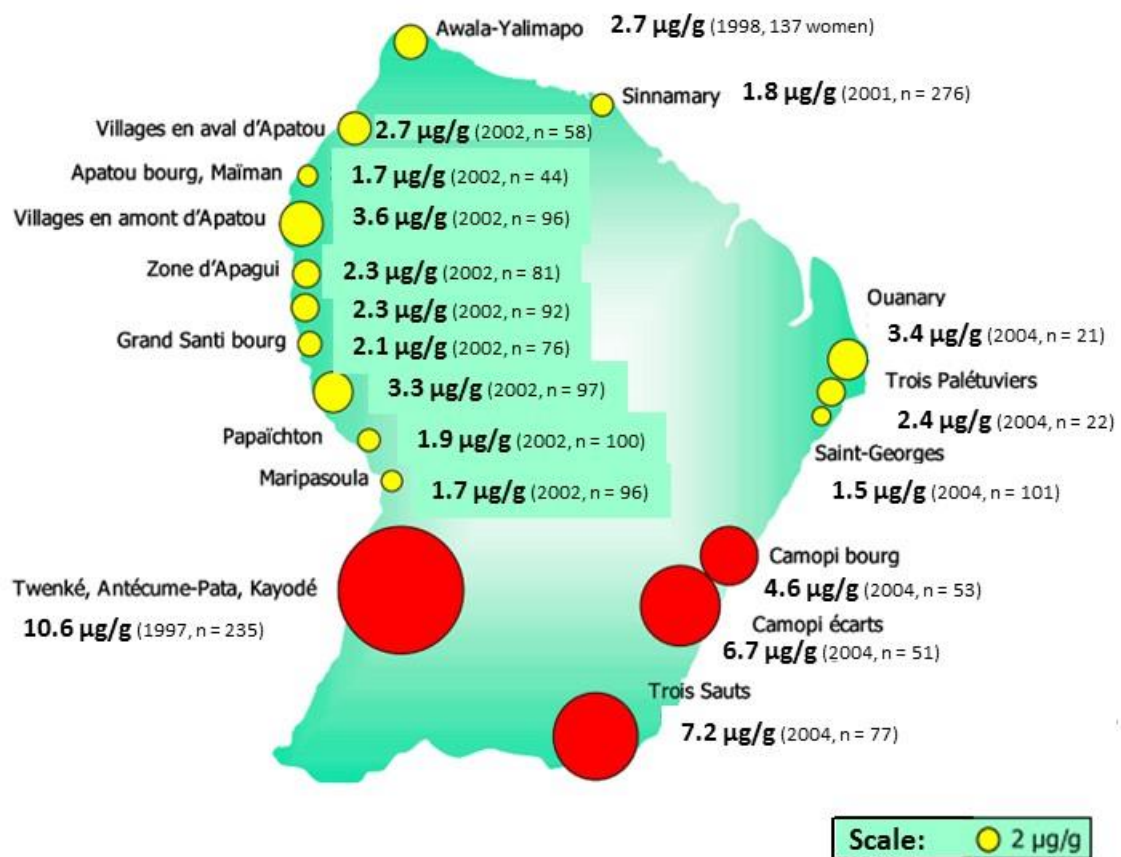
Most communities in the Guianas are likely to be exposed to the mining industry in some way, for example through mining activities carried out close to their homes. The below data are extracted from studies with no mention of direct involvement in the industry. Hair sampling is the preferred measure for assessing mercury exposure in non-mining communities both because diet is the main route of exposure for these communities and due to its non-invasive nature. WHO background levels in hair are 2 µg/g (Peplow and Augustine, 2007; WHO, 1990). There are a range of suggested safe limits for mercury in hair including the WHO 'no effect' limit of 14 µg/g for maternal hair which is based on data from 2 studies (WHO, 1990; EnvHealth.org). However, other data suggest that the safe no effect limit is 10 µg/g as there is a 5% risk of an adverse outcome with a peak mercury concentration of 10–20 µg/g (WHO, 1990).

#### *5.3.1 Indirect mercury exposure in the communities of French Guiana:*

A relatively large number of studies have been carried out in French Guiana investigating exposure to mercury, particularly in indigenous communities. Data from freshwater fish indicate that the dietary risk of mercury contamination is not equal across the country (**see section, 4.2.2**). A summary from 2007, mapped the concentration of mercury found in various studies (Laperche et al.,

2007; Quénel et al., 2007) (**Figure 5.1**). Populations in the Upper Maroni (Twenke, Antecume-Pata and Cayode) which is highly exposed to SSGM activity showed high concentrations of mercury (Quénel et al., 2007; Laperche et al., 2007). However, notably high levels of contamination were also found in the community of Trois Sauts where no SSGM occurs, but where high mercury concentrations were found in some fish species (**Section 4.2, Figure 4.1**) (Laperche et al., 2007). In a country wide survey of 500 individuals, risk of exposure was found to be closely linked to frequency and type of fish consumption (Cordier et al., 1998; Cordier et al., 2002).

**Figure 5.1:** Map of mercury concentrations in hair in French Guianan communities. *Adapted from Laperche et al., 2007, BRGM/RP-55965-FR – Rapport final*



The relationship between frequency and type of fish consumption and level of exposure was also investigated in high risk community subgroups in French Guiana. As previously mentioned mercury exposure during gestation is of particular concern due to the risk of adverse effects on foetal development (WHO, 2007). In samples taken from health centres throughout French Guiana mercury concentrations in pregnant women's hair (n = 109) was relatively low 1.6 µg/g (Cordier et al., 1998). However, in a study focused on indigenous communities in Brazil and French Guiana maternal hair concentrations were considerably higher (mean 10.3 µg/g) with a maximum of 41.7 µg/g (Chevrier et al., 2009). Cordier et al. (2002), showed that maternal concentration within indigenous populations also varied from one community to another (geometric mean [typical value, calculated as the nth root of the product of n numbers] 2.8-12.7 µg/g), with the concentration of mercury rising along with increased fish consumption in the community (Cordier et al., 2002).

Children also represent a high risk group for mercury toxicity. In the above health centre study, overall mean levels in children were relatively low at 2.5 µg/g, however, in the Wayana community of the Upper Maroni up to 79% of children had mercury concentrations above the WHO limit of 10 µg/g (Cordier et al., 1998). Children from this community (n = 97) also had markedly higher mercury concentrations in their hair (mean, 10.5 µg/g; range, 3.5–20.8 µg/g) compared with children (n = 82) from the indigenous Galipi community from Awala on the Atlantic coast (mean, 2.1 µg/g; range, 0.5–5.7 µg/g) (Chevrier et al., 2009). These data are supported by a 2002 study where a similar pattern was detected with children from the Upper Maroni having the highest geometric mean concentration of mercury in hair (10.2 µg/g) followed by those from Camopi on the Oyapock river (6.5 µg/g) and the children of Awala on the Atlantic coast (1.4 µg/g) (Cordier et al., 2002). These data followed the authors expectations based on the extent to which each of the communities were reliant on a diet of freshwater fish (Cordier et al., 1998; Cordier et al., 2002).

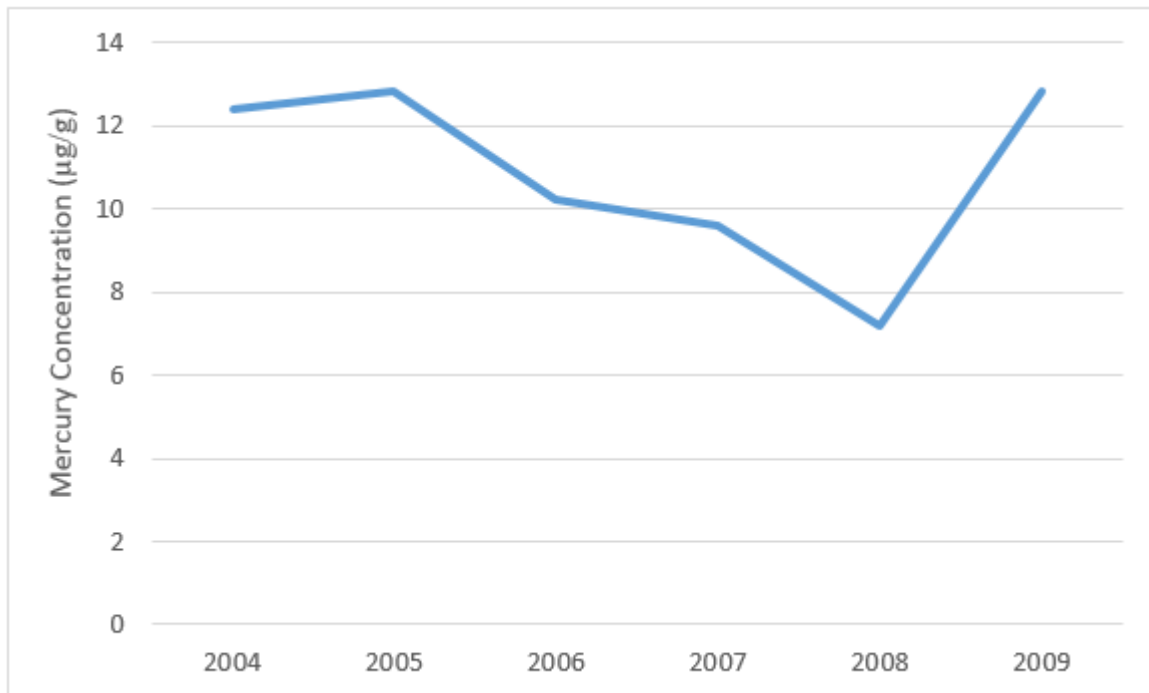
In a rare example of a longitudinal study into mercury contamination in the Guianas, Fujimura et al., (2012) collected data between 2004 and 2009 in the indigenous Wayana communities of the Upper Maroni. Hair samples (n = 387 from the villages of Cayode, Twenke/Taluwen, Antecume Para, and Elahe) and dietary habits (n = 37 from Cayode and Twenke/Taluwen villages) were assessed over the period and an additional sample of fish was also taken during a single time point (March 2009).

The mean mercury concentration from hair samples was 9.4 µg/g in men (n = 153) and 9.9 µg/g in women (n = 234). These data were in good agreement with previous studies conducted between 1997 and 2007 which found mean concentrations of 10.5–13.1 µg/g of hair (Fréry et al., 2001; Cordier et al., 2002; Quénel et al., 2007; Chevrier et al., 2009; Fujimura et al., 2012). The longitudinal data showed 30% of the subjects had mercury levels above the safety limit determined by the WHO (10 µg/g) the range was 2.8–26.6 µg/g. In a previous study where samples were taken at two time points at the beginning and end of the year, 57% of the participants from the Wayana communities of the Upper Maroni (n = 521) had mercury levels above the WHO limit (Fréry et al., 2001).

Data from Fujimura et al. (2012) further strengthens the putative relationship between fish consumption and mercury contamination. Methylmercury made up the vast majority (94.5%) of mercury found in 52 highly exposed (> 10 µg/g) individuals, indicating contamination via diet (Fujimura et al., 2012). There was a significant correlation between mercury in hair and volume of fish consumption in the 37 individuals who completed surveys on their diet. The individuals surveyed showed a preference for fish species most susceptible to mercury-contamination, especially piscivorous species such as *P. fasciatum* (0.33 µg/g) and *H. aimara* (mercury levels extrapolated from Fréry et al., 2001 due to low number of *H. aimara* caught) (Fujimura et al., 2012). In an earlier study participants also showed a preference for piscivorous fish that tend to be more highly contaminated, some of which had a mean mercury concentration of 1.62 µg/g (Fréry et al., 2001; Fujimura et al., 2012) which is more than three times the WHO safe limit. In this study all participants older than 1 year of age had a mercury intake greater than the WHO weekly safety limit of 200 µg/week for a 60-kg male, maximum intake was 450 µg/week (Fréry et al., 2001; WHO, 1990).

The longitudinal aspect of the Fujimura et al. (2012), study revealed a reduction in mercury concentration between 2005 and 2006, when mercury use in French Guiana was banned (**Figure 5.2**). The decline continued for two years before a sharp increase in 2009 back to 2004–2005 levels. Cross-sectional studies (taken at a single time point) from two sites in the Upper Maroni suggest an increase in mercury concentration in hair between 1997 and 2005 (11.4 to 13.1 µg/g) and a decrease between 2002 and 2007 (12.6–10.5 µg/g) (Quénel et al. 2007; Cordier et al. 2002; Chevrier et al. 2009; Fujimura et al., 2012). However, Fujimura et al (2012) are the first to follow a community longitudinally providing much stronger data on changes in mercury exposure over time.

**Figure 5.2:** Mercury concentration in hair samples from Cayode over time. *Adapted from Fujimura et al., 2012*



It is clear that sections of the community in French Guiana are at risk of mercury contamination, a series of studies have assessed the putative effect of elevated mercury levels on human health. As previously mentioned mercury is highly neurotoxic, in their 2009 study Chevrier et al. set out to assess levels of exposure and the presence of neurological deficits in indigenous children from the Upper Maroni and the Brazilian Amazon basin. Looking at the group as a whole (Brazilian and French Guianan children), mercury concentrations in hair were found to correlate with deficits in drawing tasks designed to measure cognitive function, this deficit was found to be worse in younger children (Chevrier et al., 2009). A similar correlation was found in a study comparing children from high exposure (Upper Maroni), medium exposure (Camopi on the Oyapock River) and low exposure (Awala on the Atlantic coast) areas (exposure assessed by level of fish consumption). In line with the more recent study younger children (5–6 years) were the worst affected, however, there was also a sex difference with boys showing a greater deficit (Cordier et al., 2002).

Cordier et al. (2002), showed a correlation between maternal exposure to mercury and neurological deficits in children suggesting gestational neurotoxicity. It should be noted that this was a retrospective study with maternal hair being sampled up to 12 years after childbirth so levels in hair may be very different to that seen during gestation. Overall, children from French Guiana and Brazil aged 7–12 with a mercury concentration of 10 µg/g of hair had a developmental delay of at least 2 years compared to those with a level of 1 µg/g (Chevrier et al., 2009).

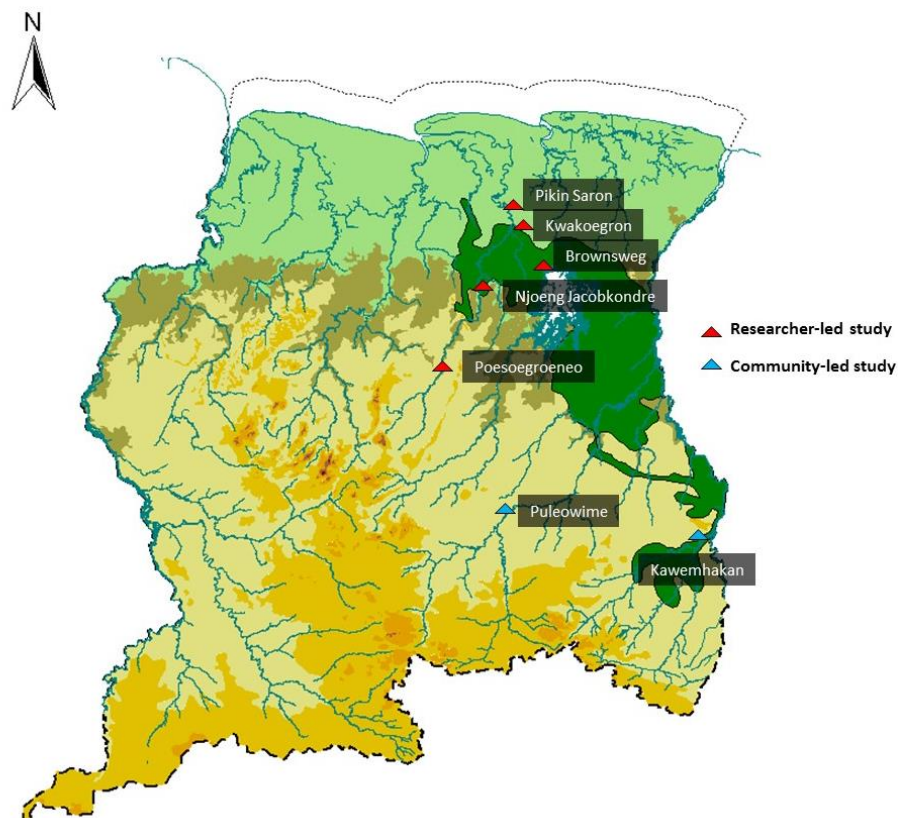
Studies in French Guiana demonstrate significant exposure of indigenous communities to mercury contamination, principally via the consumption of contaminated fish. The presence of significant neurological deficits, including developmental delay, in some children and a possible link to maternal exposure is of great concern and suggests that action to mitigate exposure is long overdue in some communities.

### 5.3.2 Indirect mercury exposure in the communities of Suriname:

Studies conducted into mercury exposure in Surinamese communities can be split into those directed by the community and those led by researchers (**Figure 5.3**). Researcher led studies have been conducted in maroon and indigenous communities while the community-led assessments of mercury exposure have been conducted by the Wayana community of southeast Suriname, in conjunction with scientific and medical professionals.

Results from small studies in the maroon community of Kwakoe Gron, 45 km south of Paramaribo and relatively close to gold mining areas, showed elevated mean mercury levels but results were below the WHO guideline concentration of 10 µg/g (mean 4.1 µg/g of hair in both studies) (Peplow and Augustine, 2007; Barnes et al., 2000). The maroon community of Kwakoe Gron was recently revisited in a survey conducted in conjunction with a study on the indigenous population of Pikin Saron. The study focused on women and children due to the increased risk of toxicity in these groups. In a 6-day survey in Kwakoe Gron, 50% of families ate fish less than twice per week. The locally caught fish consumed by the surveyed households were a mix of herbivorous, omnivorous and carnivorous species. In Pikin Saron around 2/3 of families ate fish more than 3 times a week. As in Kwakoe Gron a mix of herbivorous, omnivorous and carnivorous fish were consumed. Sampling revealed the very high levels of contamination in the aquatic ecosystems relied on by both communities (75% of fish had a mercury concentration > 0.5 µg/g) (Hawkins, 2012).

**Figure 5.3:** Map of Suriname showing location of communities involved in community-led and researcher-led mercury exposure studies. *Adapted from Ouboter, in press*

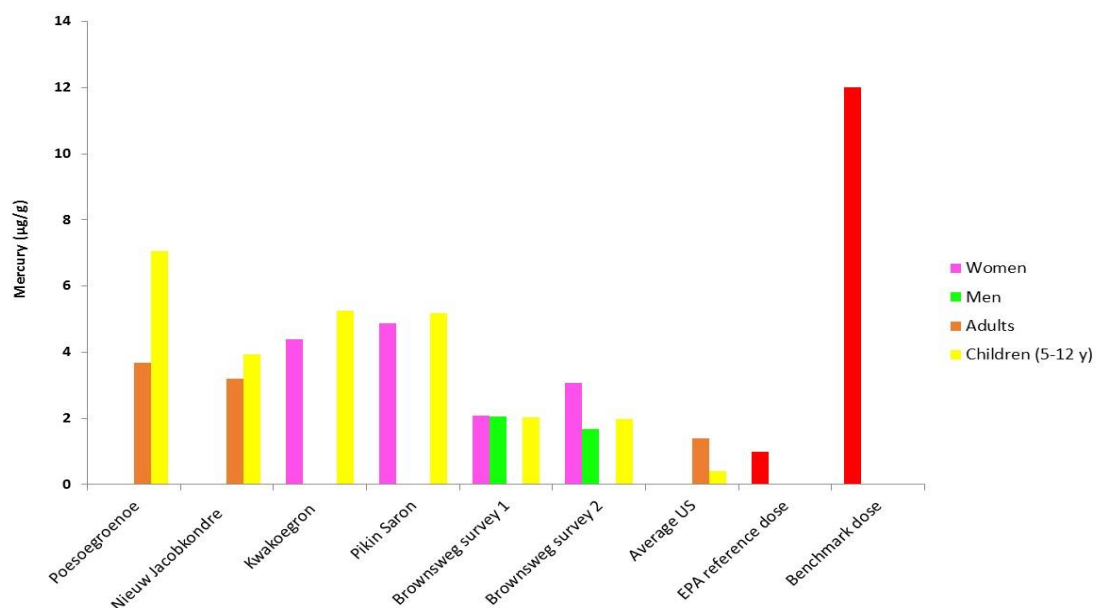


In Kwakoebron, 33 people (13 women and 20 children) provided samples of their hair while in Pikin Saron 22 people took part (9 women and 13 children). Mean mercury levels were above the EPA reference levels (1 µg) and WHO background levels (2 µg) for both children (mean 5.2 µg/g, in both communities) and adults (4.3 and 4.9 µg/g in Kwakoebron and Pikin Saron, respectively) (**Figure 5.4**). The concentration range showed individuals close to harmful levels in women (1.1 - 9.1 µg/g) and above the harmful level in the children, who are the most vulnerable to mercury toxicity (1.0 - 14.1 µg/g) (Hawkins, 2012; Hawkins et al., 2014).

Another study was conducted at the Brownsweg villages in the heavily mined district of Brokopondo (**Figure 5.3**). Piscivorous fish caught in the Brokopondo reservoir showed high mean concentrations of mercury (*S. rhombeus* mean 1.61 µg/g; *Plagioscion sp.* 0.69 µg/g and *Acestrorhynchus guianensis* 1.71 µg/g). However, mercury concentration in hair was low during surveys carried out in both the wet (women 2.08 µg/g; men 2.05 µg/g and children 2.02 µg/g) and dry season (women 3.08 µg/g; men 1.68 µg/g and children 1.98 µg/g) (**Figure 5.4**). Surveys on diet indicated that consumption of locally caught piscivorous fish was low (4.8% and 5.1% of the diet in the wet and dry season, respectively). These data further underline the importance of diet and specifically the role of fish consumption for exposure to mercury even in highly mined regions (Ouboter and Landburg, 2010).

The importance of diet over location was further emphasised in a small study conducted in two Maroon villages, Poesoegroenoe and Njoeng Jacobkondre (**Figure 5.3**) mercury levels were found to be higher in children (n = 30) than in adults (n = 13) (Ouboter, 2007). Notably, the mean mercury concentrations were higher in participants from Poesoegroenoe  $6.28 \pm 2.85$  µg/g, which was upstream of SSGM, than in Njoeng Jacobkondre ( $3.44 \pm 2.02$  µg/g) which was in the mining area (**Figure 5.4**) (Ouboter, 2007). The higher mercury concentrations in Poesoegroenoe are likely driven by the higher fish consumption in this community, where there is a preference for predatory fish. In addition this area showed high levels of mercury contamination, likely driven by atmospheric deposition (see section 3.2.3) (Ouboter et al., 2012).

**Figure 5.4:** Summary of mercury concentrations from researcher-led studies in Suriname (Ouboter, in press)



In the community-led studies, data are available from two communities from a study undertaken in 2008. The study was conducted in the communities of Puleowime (n = 158) and Kawemhakan (n = 106), close to the Wayana communities studied in French Guiana (**Figure 5.3**) with samples from new-borns, children and adults (ages 1-80 years) (Peplow and Augustine, 2012).

Exposure to mercury was high in both communities with 58% of participants having concentrations in hair above the 10 µg/g WHO limit. The mean concentration in Puleowime (16 µg/g) was significantly higher than Kawemhakan (9 µg/g) (Peplow and Augustine, 2012). There appeared to be a relationship between mercury contamination and fish consumption. All participants in Puleowime reported eating fish at least 3 times per day while in Kawemhakan, although fish consumption was also high, 25% of participants reported consuming fish less than 3 times per day. The two most common fish species eaten in both communities were piscivorous, *Hoplias spp.* (e.g. *H. aimara*) and *Cichla sp.* (Peplow and Augustine, 2012).

Neurological signs of mercury toxicity were inconclusive, however, a small number of self-selected participants who accessed medical examination showed neurological deficits (n = 3) (Peplow and Augustine, 2012). A second study focused on possible neurological problems in a subpopulation of participants from Puleowime with mercury concentrations above 20 µg/g of hair in 2008 (n = 22) (Peplow and Augustine, 2014). There was a marked reduction in the mean mercury concentration in the subgroup between 2008 (23 ± 6 µg/g) and 2012 (13 ± 4 µg/g), however, the mean was still above the WHO safety limit. Effects consistent with low level methylmercury toxicity were detected in 8 participants (33%), 10 participants (42%) had symptoms suggesting moderate toxicity, 4 had symptoms of high toxicity (17%) and 2 (8%) had symptoms consistent with very high exposure. In this study there was only a low correlation between neurological dysfunction and mercury concentration in hair. The medical team applied a 'tentative diagnosis' of Minamata disease (neurological dysfunction caused by severe mercury toxicity) in 6 participants with mercury concentrations of 9–17 µg/g, aged 20–70. Both the small sample size and lack of control for confounding factors such as illness, age and drug/alcohol abuse should be taken into account when interpreting these results. (Peplow and Augustine, 2014).

Two small preliminary studies stand out from the rest of the current work from Suriname. Research was undertaken to assess the risk of gestational neurotoxicity in the capital city of Paramaribo. Hair samples were taken from a mixed population of mothers (n = 39) of various ethnicities and 14 (36%) of the participants had elevated mercury concentrations in their hair. Worryingly, 31 of the new-borns (80%) had a higher mercury concentration in their hair than their mothers. There was also a positive correlation between mercury levels in mothers and their children (Mohan et al., 2005).

The data available in Suriname suggest that a similar picture exists to that in French Guiana with indigenous communities which are reliant on fish being at high risk of mercury toxicity. Data from the urban centre of Paramaribo on maternal exposure are worrying and should be followed up and extended to urban centres in both Guyana and French Guiana.

### 5.3.3 Indirect mercury exposure in the communities of Guyana:

There is a lack of up to date or geographically extensive data on mercury contamination in non-mining populations in Guyana. A recent study (2008–2010) summarised in a report by Singh et al. (2013), revealed mercury concentrations of up to 70.8 µg/g (at least 5 times over the WHO limit) in the hair of pregnant and nursing women from indigenous populations living close to SSGM activities. Some of those sampled were from the Isseneru community mentioned above (**see section 5.2**) in addition to individuals from the two other villages, Micobie and Masekenari.

Another study also took samples from the Micobie village which is located close to intense gold mining activities. Mercury concentrations in hair from residents were above WHO limits,  $15.4 \pm 0.9 \mu\text{g/g}$  ( $n=47$ ) (Mangal, 2001). Individuals from another village, Moraikobai, which is remote from gold-mining activity had mercury concentrations of  $5.6 \pm 0.4 \mu\text{g/g}$  of hair ( $n = 44$ ).

In 2000, M. Reinders carried out a study for Tropenbos in Region I, northwest Guyana. Mercury concentrations in the hair of individuals in the Carib community living along the Barama river were up to twice the WHO safe limit ( $2\text{--}22 \mu\text{g/g}$ ) (reported in Colchester et al., 2002).

Mercury concentrations in the hair of individuals from non-mining communities situated in areas of Guyana affected by mining appear to be elevated to concerning levels. However, there is a clear need for more extensive studies in Guyana that cover larger areas of the country and take into account factors such as diet which may affect exposure to mercury. Wider publication of currently available data would also be a helpful measure in assessing the current risk to the population.

#### **5.4 Summary of Community Mercury Exposure in the Guianas:**

Despite the small amount of data available, particularly in Guyana and Suriname, it is clear that certain communities in all three Guianas are exposed to dangerous levels of mercury. Exposure is principally through diet, although both miners and the extended communities around them are also at risk. Examples of neurological dysfunction have been demonstrated in both French Guiana and Suriname. As is the case with contamination of fish and sediments there is a disconnect between areas of gold mining and cases of high mercury exposure in some communities. In the case of human health the main driver of toxicity appears to be a reliance on predatory fish as a major food source. The developmental delay seen in children affected by mercury in French Guiana is of grave concern, and a specific public programme was launched in 2012 to mitigate and decrease contamination of populations at risk (ARS Guyane, 2015). Data from urban centres showing high levels of localised air pollution as well as elevated mercury concentrations in the hair of mothers and babies underline that this is an issue which is not restricted to indigenous communities in remote areas of the Guianas.

#### **6.0 Conclusion:**

The results of this extensive narrative review reveal a stark picture of mercury contamination in the Guianas. The SSGM industries in Guyana and Suriname, along with the large illegal industry in French Guiana, remain dependent on mercury. Excessive volumes are currently being imported into Guyana with no clear end user. The movement of mercury across borders appears likely both between the three Guianas and from larger regional neighbours. Contamination of the environment is widespread and is not restricted to mining areas. Similarly, widespread contamination of the aquatic biome is evident with many high value food fish species displaying mercury levels above safe limits for consumption. Communities with high levels of freshwater fish in their diets are at risk of mercury toxicity with some already showing symptoms including mental retardation of children. Emerging evidence also points towards urban populations remote from mining being exposed to mercury pollution.

Given that widespread mercury-reliant mining is ongoing in Suriname and Guyana and that mercury use has only been banned for a decade in French Guiana, with illegal use thought to be common (WWF, 2013), the legacy of such a persistent pollutant is hard to assess. Studies of historically mined

areas of the Sierra Nevada show that a significant proportion of fish have concentrations above safe guidelines more than 100 years after cessation of mercury use (May et al., 1999). Mercury easily forms complexes with fine sediment particles which settle in areas with low flow, such as reservoirs. Such sediments can create thick layers of mercury rich sludge under low oxygen conditions perfect for methylation. What little data there are from the Guianas suggests a long term legacy of mercury pollution is likely. Recent data from a site in French Guiana mined in the 1950s revealed high concentrations of mercury in the soil (up to 5.47 µg/g) and evidence of ongoing release to nearby aquatic systems (Guedron et al., 2011). Data suggest that contamination of catchment areas around water bodies, as well as mercury in sediments, may retard their recovery even with reduced mercury input with the potential for decades- to centuries-long recovery rates (Guedron et al., 2011; Harris et al., 2007; Sherman and Blum, 2013).

The legacy of mercury in terms of human health is a complex issue. The likelihood is that contamination of fish stocks will continue to be an issue. Alteration of deeply ingrained cultural practices may be necessary to protect indigenous communities who are heavily reliant on fish. The long term impact for those children already showing developmental delay is also unclear (Chevrier et al., 2009). Long-term studies on individuals affected by the heavy metal lead during childhood have demonstrated life-long impacts including poorer cognitive performance and reduced employment status (Chevrier et al., 2009).

Simple unilateral solutions to reduce mercury use in SSGM are unlikely to be found. The Minamata Convention on Mercury, recently signed by Guyana and under discussion in Suriname, may offer a framework through which the international and regional cooperation required to achieve a meaningful reduction in mercury use can be implemented. The prospect of such a widespread and long term legacy of pollution emphasises the need to arrest further contamination with the greatest possible speed.

## **7.0 Recommendations**

The primary objective of this review of mercury contamination in the Guianas was to collate the large body of existing information into a single document for easy reference. It is also a strong indication of the state of knowledge on mercury across the region.

In a region such as the Guianas where large parts of the territory are difficult to access, it is impractical to expect detailed information on all of the impacts of mercury in all situations in all places. However, what is clear from this review is that sufficient information exists that strongly suggests that mercury is already a significant environmental and increasingly a health issue across the entire region. There is clear evidence that small-scale miners continue to use mercury in large quantities and with little regard for safety and so levels of mercury in the environment will only increase. Given the long-lasting persistence of mercury in the environment and its known cumulative impacts, there is a need for immediate action by relevant authorities to address this issue. Listed here are two series of recommendations; the first are the policy responses required and secondly research gaps for a fuller understanding of the scale and scope of mercury impacts. It should be stressed that the existence of research gaps are not an invitation or excuse for inaction on the policy front.

## **Policy recommendations**

1. Commit to the banning of mercury in the mining sector. One mechanism for this is through the Minamata Convention (to which Guyana and France are already signatories) and implementation of a National Action Plan for the phasing out of mercury.
2. Recognise the multi-sectoral impacts of mercury and establish national cross sectoral working groups to coordinate programmes committed to phasing out mercury. These should include, as a minimum, representatives of the mining, health, environment, fisheries, trade and education sectors.
3. Strengthen and properly implement import and export laws to ensure mercury trade can only be conducted under license and that the compulsory recording of all imports and exports of mercury is carried out.
4. Increase monitoring of mining activity and movement of mercury, especially in the border areas of the Guianas with explicit coordination and intelligence sharing between authorities on each side of the national boundary.
5. Remove mining from Protected Areas and other areas of high conservation value or environmental sensitivity and illegal miners from indigenous lands.
6. Implement an awareness campaign targeting the most vulnerable groups to mercury exposure indicating precautionary measures, including fish consumption thresholds, to minimise their risk of contamination.
7. Establish mechanisms to demonstrate best practice and mercury free techniques to small scale miners including the engagement of the private sector gold mining companies to share their technical expertise.
8. Where gold refining shops exist to legally oblige them to use best technologies to prevent release of mercury into the atmosphere.

## **Research recommendations**

1. Undertake systematic reviews of the mercury status of representative mining and non-mining communities, including urban areas, throughout the Guianas starting with those perceived to be at highest risk. This will formalise a comprehensive baseline and allow change to be monitored.
2. Initiate a systematic and regular programme to determine mercury status of fish caught for sale and consumption within the Guianas.
3. Better identify bioindicator species for estuarine and marine environments and determine their mercury loading, along with the mercury loading of the freshwater bioindicator species *H. aimara* and/or *S. rhombeus*, at representative sites across the Guianas.
4. Undertake research to better understand and predict the dynamics of mercury contamination over time.
5. Increase research efforts to assess mercury levels in the marine environment.

6. Investigate the relationship between compounding factors that impact levels of mercury contamination such as the correlation between water chemistry and mercury in the food chain and the links between mercury levels in pristine areas and atmospheric mercury.

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