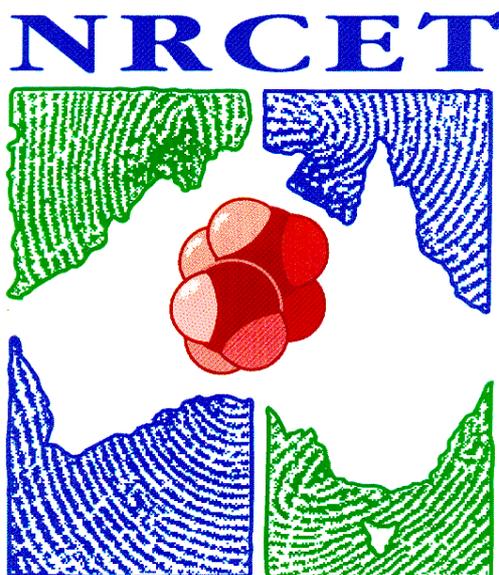


**EXAMINATION OF THE POTENTIAL EXPOSURE OF ROYAL
AUSTRALIAN NAVY (RAN) PERSONNEL TO POLYCHLORINATED
DIBENZODIOXINS AND POLYCHLORINATED DIBENZOFURANS VIA
DRINKING WATER**



A REPORT TO THE DEPARTMENT OF VETERAN AFFAIRS, AUSTRALIA

**THE NATIONAL RESEARCH CENTRE FOR ENVIRONMENTAL
TOXICOLOGY (ENTOX)**

QUEENSLAND HEALTH SCIENTIFIC SERVICES (QHSS)

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TO POLYCHLORINATED DIBENZODIOXINS AND POLYCHLORINATED
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THE DEPARTMENT OF VETERAN AFFAIRS, AUSTRALIA

This report is dedicated to

Ralph Hayden Spooner

EX Ran Warrant Officer

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COMPOUND ABBREVIATIONS

| | |
|-------|---|
| I-TEq | 2,3,7,8-Tetrachlorodibenzodioxin toxicity equivalents |
| PCDDs | Polychlorinated dibenzodioxins |
| TCDD | Tetrachlorodibenzodioxin |
| PeCDD | Pentachlorodibenzodioxin |
| HxCDD | Hexachlorodibenzodioxin |
| HpCDD | Heptachlorodibenzodioxin |
| OCDD | Octachlorodibenzodioxin |
| PCDFs | Polychlorinated dibenzofurans |
| TCDF | Tetrachlorodibenzodioxin |
| PeCDF | Pentachlorodibenzodioxin |
| HxCDF | Hexachlorodibenzodioxin |
| HpCDF | Heptachlorodibenzodioxin |
| OCDF | Octachlorodibenzodioxin |
| HCB | Hexachlorobenzene |
| DDE | Dichlorodiphenyl ethane |
| DDT | Dichlorodiphenyl trichloroethane |
| DDD | Dichlorodiphenyl dichloroethane |
| DMA | Dimethyl arsenic acid |

EXECUTIVE SUMMARY

Studies of Australian Vietnam veterans have revealed greater than expected mortality, with the highest overall levels of mortality occurring among the Royal Australian Navy (RAN).

During the Vietnam War, large quantities of phenoxy herbicides (Agent Orange) contaminated with 2,3,7,8-tetrachlorodibenzodioxin (TCDD), arsenical herbicides (Agent Blue) and organochlorine pesticides were used. There has been concern that exposure to these chemicals may have long-term adverse health effects. TCDD for example is now known to have many toxic effects in humans, including carcinogenesis.

In RAN veterans, exposure to chemicals such as the TCDD is unlikely to be related to overhead spraying or other forms of direct contact.

The aim of this study was to investigate the potential for exposure of sailors to contaminants via drinking water. On Navy ships and Army small ships, potable water was produced from evaporative distillation of surrounding estuarine water. This water would have had variable salinity and amounts of suspended solids. It may have also contained contaminants in solution.

The study was carried out in two phases. First, the co-distillation of organic pollutants such as dioxins along with water in ship's distillation units was examined. Phase One results of this study demonstrated that:

- Co-distillation of organochlorine pesticides and dioxins was observable in all experiments conducted;**
- In pure or saline water, between 75% and 95% of 2,3,7,8-TCDD was co-distilled with the first 10% of water distilled. Thus, distillation results in an increase in the contaminant concentration in the distillate;**

- The tendency of several other organochlorines to co-distill was greater than for TCDD. For dioxins a tendency of decrease in co-distillation with increasing molecular mass was apparent. Hepta- and octachlorinated dioxins showed little tendency to enrich in the distilled water;
- A compounds' co-distillation decreased with increasing levels of suspended solids in the water. This can be attributed to the increase in sorption (fugacity) capacity in the source water. At a highest level of 1.44 g total suspended solids in the water about 38% of 2,3,7,8-TCDD co-distilled in the first 10% of water distilled. Nevertheless, even at these relatively high levels of suspended solids, TCDD was enriched by almost a factor of 4 in the distillate (assuming only 10% of the water is distilled);
- Co-distillation of dioxins and organochlorines from water collected from the Brisbane River (water was added to known amount of chemicals of interest) demonstrated that the process is reproducible using estuarine water. In these samples 48 – 60% of the TCDD co-distilled within the first 10% of distilled water.

Overall, Phase One of the study clearly demonstrated that if source water is contaminated, co-distillation is a process which can result in the contamination of ships water supplies with chemicals such as dioxins.

In Phase Two of the study the investigations included the potential co-distillation of the Agent Blue component dimethylarsenic acid, which is now known to be a potent carcinogen.

In addition, experiments were carried out in which the capacity for de-novo synthesis of dioxins from the main components of Agent Orange was evaluated. Evaporative distillation entails heating of the source water using copper elements. Combustion of the components of Agent Orange has great potential to produce dioxins. Moreover, copper (which formed part of the distillation unit) is a known catalyst for dioxin formation.

Finally exposure calculations were carried out for personnel on board ships. These calculations were based on some of the first analytical results from fish samples that were caught during the early 1970's in contaminated waters from Vietnam and analysed in the 1970's for TCDD.

Phase Two results of this study were:

- **Dimethylarsenic acid does not co-distill at significant levels during evaporation and thus the drinking water on board of RAN ships was unlikely to be contaminated with dimethylarsenic acid;**
- **No de-novo synthesis of TCDD or any other dioxins from the other components of Agent Orange was detected under the experimental conditions. However, the copper element on board ships was probably significantly hotter than in the simulation experiments selected in the laboratory, and thus these results should not be used as absolute evidence that such a formation did not occur in the distillation units of the RAN ships;**
- **TCDD exposure via drinking water may have been substantial, and it is likely that solely the consumption of drinking water resulted in exposure levels that exceeded the recommended Total Monthly Intake (TMI) values for TCDD of 70 pg / kg bw / month significantly. A TMI of 70 pg/kg bw / month is a level set by many European authorities; it is also the level proposed by the draft recommendation of the National Health and Medical Research Council in Australia.**

Overall the findings of this study demonstrate that evaporative distillation of water does not remove but rather enriches certain contaminants such as dioxins in drinking water. The study provides some evidence that use in the distillation process of water contaminated with TCDD would result in contamination of potable water. Subsequent ingestion by sailors on board ships (as well as

soldiers and airmen, who were passengers) is thus a vector for exposure to these chemicals.

While it is unlikely that accurate exposure of the personnel on board ships can be estimated, the study findings suggest that the personnel on board ships were exposed to biologically significant quantities of dioxins. This may explain some of the epidemiological findings in this study group.

INTRODUCTION

Studies undertaken by the Australian Government have indicated that Australian Vietnam veterans experience greater than expected mortality (Crane et al., 1997a) and that when mortality of two cohorts of conscripted veterans are compared, greater relative mortality (Crane et al., 1997b). Subsequent studies have revealed validated elevation in certain types of cancer in the veterans, and in their children, small increases in some birth defects and the rate of deaths, particularly suicide (AIHW, 1999).

The highest elevation in mortality was among veterans of the Royal Australian Navy, rather than the land and air forces (Crane et al., 1997a). Uncertainty remains as to whether this increase in mortality is related to the use of “Agent Orange” contaminated with polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs). In addition to “Agent Orange” and various other Agents that contained dioxin impurities, “Agent Blue” was the third most commonly used herbicide. It consisted of an aqueous solution of dimethylarsenic acid (DMA), or more commonly cacodylic acid. DMA was sprayed primarily in crop destruction missions (50%) or was used in the control of grasses around base perimeters. Recent research has demonstrated that DMA is a carcinogen itself (for details see Ng 2002, attached in Appendix II) and hence in Phase 2 this project was expanded to include preliminary evaluation of exposure to DMA.

A starting point of the study was that elevated levels of mortality were found in sailors. These sailors were never present in the areas where “Agent Orange” or any other Agents were directly employed for defoliation. Hence, prior to this study it was assumed that the significantly higher incidences of mortality observed in this cohort could not be related to the use of defoliant.

However, marine vessels such as the troop carrier and supply vessel HMAS Sydney served for substantial periods in estuarine waters in Vietnam and relied on collection of potentially contaminated estuarine water, which was then distilled for drinking. According to personnel on these ships a common procedure was to produce and store

drinking water from the relatively turbid estuaries, while the distilled water that was produced in open sea was primarily used for the boilers of the vessel's steam engines.

In the general population, exposure of humans to PCDD/Fs is attributed primarily to the consumption of contaminated food (e.g. Beck et al., 1989, Liem et al., 2000). This is due to the physico-chemical properties of these chemicals including their hydrophobicity and persistence and the resulting accumulation of these chemicals in lipophilic compartments of plants (Müller et al., 1997) and animals (McLachlan, 1996 1997).

These physico-chemical properties and in particular the exceptionally high sorption coefficients and very low solubility of PCDD/Fs in water are factors which reduce the risk of contamination of surface and ground water. Environmental fate models, as well as experimental evidence, suggest that consumption of contaminated water is a pathway which can safely be neglected in the calculation of exposure of humans to these compounds (Hattemer-Frey & Travis, 1989; Fürst, 1998, Liem et al., 2000).

However, in supply ships and other vessels which regularly visited the conflict areas in Vietnam, the water consumed by the crew has an unusual history. It was often collected from near-shore marine waters that received runoff from areas which had been sprayed with Agent Orange and Agent Blue. To make this water suitable for drinking and other purposes aboard ship the water was distilled aboard.

Evaporative distillation is a process that is suitable for obtaining water which is relatively free from salts and other high boiling components with a high water solubility. However, in contrast to their high sorption coefficient, both experimental data and models have indicated that the Henry's Law constant of chemicals such as dioxins is sufficient to allow desorption from natural water surfaces (i.e. Lyman et al., 1990). It seems feasible that co-distillation could occur during the distillation process on the marine vessels.

The aim of the study was to undertake laboratory experiments which provide information to assess whether PCDD/Fs and also DMA can co-distill in significant quantities in the distillation units of ships. Further, the study aimed to evaluate

potential exposure of PCDD/Fs to personnel aboard ships. The results are useful for an assessment of exposure pathways of PCDD/Fs to crew and troops aboard ships which regularly loaded sea water for distillation and subsequent consumption.

MATERIALS AND METHODS

Distillation Experiments with Dioxins and other Organochlorines

The project's goal was to identify whether significant quantities of potentially harmful chemicals may have co-distilled into drinking water in the ships which transported Australian Troops during the Vietnam conflict. The distillation plants used on the various ships at the time of the conflict all operated using the same principles. In general, sea water was fed into an evaporator where the water was boiled by a combination of heating and reduced pressure (vacuum) and the vapour was condensed in the condenser from where it was pumped into feed tanks (Figure 1). A detailed description of the operation and function of the distilling plants of ships is given in Naval Marine Engineering Practice Vol 1 (1959). The aim of this project was to reproduce the distillation plants principal processes in the laboratory and to assess the potential for co-distillation of chemicals in the distillation unit. Ultimately, this has provided information that allows us to evaluate the potential for contamination of drinking water from distillation of contaminated sea water.

Chemicals Tested

Agent Orange, the key defoliant used during the Vietnam conflict was contaminated with up to ~ 45 ppm of 2,3,7,8-TCDD and traces of 1,2,3,7,8-PeCDD (Young et al. 1978 quoted in IoM, 1999). It has been estimated that a total of 368 pounds of dioxin were sprayed in Vietnam over a six-year period (Gough, 1986 quoted in IoM, 1999). Although the defoliants did not usually contain relevant levels of any of the other 2,3,7,8-chlorine substituted PCDD/Fs, the study was extended to include a range of other 2,3,7,8-chlorine substituted PCDD/F congeners as well as the relatively nontoxic 1,2,3,4-TCDD and a range of organochlorine pesticides including DDT, HCB, lindane and dieldrin. This extension of the compound group allowed us to assess physico-chemical properties which govern the water volatility of lipophilic

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organic chemicals and thus to predict water volatility for compounds which have not been studied here. In addition to the chemicals that were tested in Phase 1 of the project, we undertook further studies in Phase 2 of the project using DMA. A list of the chemicals used, including physico-chemical properties, is provided in Table 1.

Laboratory Distillation Plant

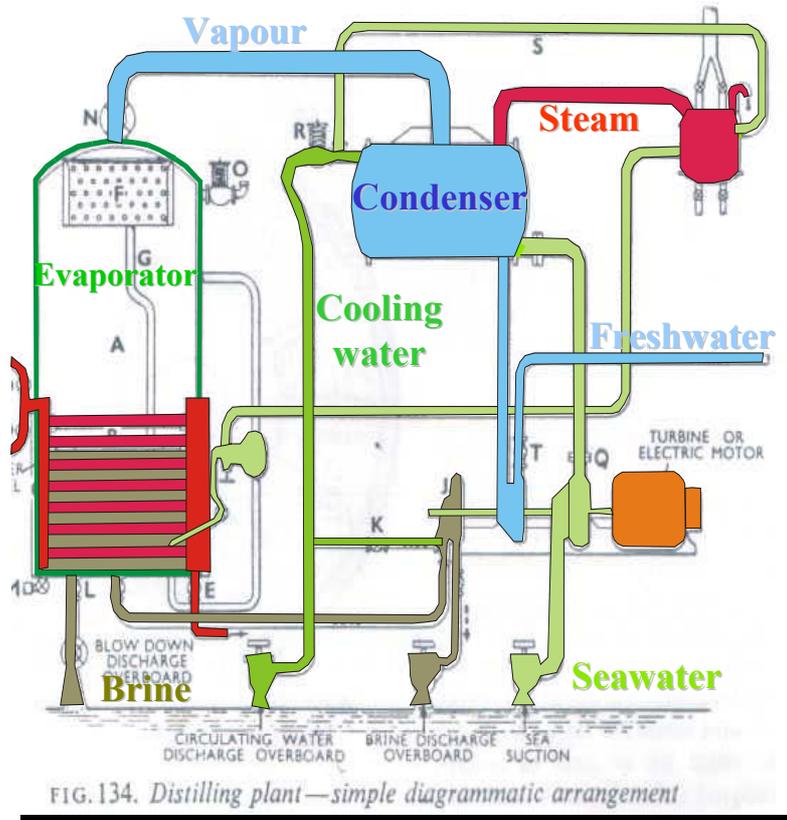
For the purpose of this study a commercially available rotary evaporator (Büchi, Switzerland) was used. Discussions with seamen and mechanics who served on RAN vessels during the conflict made certain that the principles by which solvents are evaporated in rotary evaporators were essentially the same as those used in the Naval Vessel Distillation Plants. Rotary evaporators such as the one used in this study essentially function as a batch evaporator. The water to be distilled is contained in a round bottomed flask with a seal which fits to the steam duct which leads the water vapour into the condensing chamber (Figure 2). The flask is lowered into a water bath which is maintained at the temperature of interest (in this study we used 58° C which is similar to that used in the distillation plants of the ships). In the condensing chamber, chilled water runs through a condenser coil and the water which condenses on the coils is collected in a solvent collection flask. The rotary evaporators are equipped with a pump, which is controlled through a vacuum control unit that allows accurate control of the vacuum during evaporation of the solvent. For the purpose of this study, the vacuum in the unit was set initially to 14 kPa (about 14 % of atmospheric pressure) and then slowly decreased until boiling of the water was observable. Water was then carefully evaporated since it was important to avoid non-vapour water containing the chemicals transferring through the condenser to the collection chambers ('bumping').

Distillation Experiments - Variations of Parameters

We studied the process through a series of experiments in which experimental parameters or compounds were altered. The key parameters which were altered were salinity of the water and quantity of suspended solids in the water.

For the experiments one litre round bottom flasks were cleaned with toluene and acetone and once they were dry, were spiked with a solution of the chemicals of interest. The round bottom flask was slowly swirled to coat the interior surface and to allow the solvent to evaporate. Once the solvent had evaporated, 1 L of reversed osmosis water (RO-water) was added and, depending on the variant of the experiment, known quantities of NaCl and/or sediment were added to the sample.

The round bottom flasks were then sealed, wrapped in aluminum foil and put on a shaker for at least one week so that the chemicals could equilibrate between the surface of the round bottom flask and the water.



Naval Marine Engineering Practice Volume I, 1959

Figure 1: Schematic diagram of the distillation unit on board RAN vessels

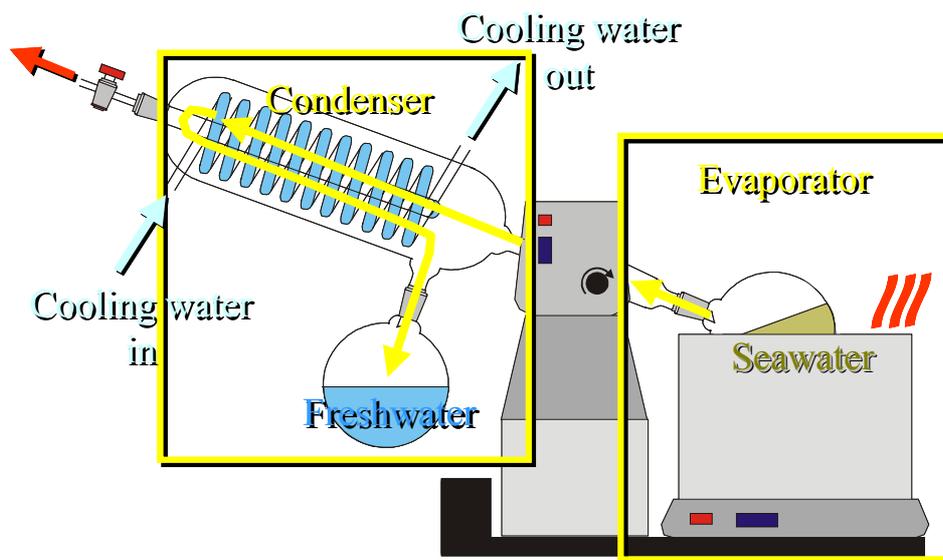


Figure 2: Schematic representation of the laboratory distillation unit

Table 1: List of chemicals used in the experiment, Molar mass, Henry's Law constant and water solubility data compiled from Mackay et al. 1992, Windholz, 1983, Paasivirta et al., 1999 and reviews by Huelster, 1994, Mueller, 1997, Cavanagh 2000 and IARC, 1987.

| Compound | Molar mass (g) | Vapour Pressure* (Pa) | Aqueous solubility | H (Pa m ³ mol ⁻¹) | Spiked amount ng/L | RRT |
|---------------------|----------------|-----------------------|-----------------------|--|--------------------|------|
| HCB | 284.8 | 0.0023 | 5 ug/L | 131 | 100 | 0.36 |
| Lindane | 290.8 | 7-213 E-4 | 2.2 – 10 mg/L | 0.005-1.5 | 100 | 0.50 |
| Heptachlor | 373.3 | 0.2 – 0.5 | 6 – 200 ug/L | 18-233 | 100 | 0.54 |
| Heptachlorepoxyde | - | 3.5-450 E-4 | 20-200 ug/L | 2-4.3 | 100 | 0.59 |
| Aldrin | 364.9 | 0.0008-0.75 | 10-200 ug/L | 1.4-91 | 100 | 0.73 |
| Dieldrin | 380.9 | 0.2-9 E-4 | 20-2000 ug/L | 0.02-5.8 | 100 | 0.82 |
| DDT | 354.5 | 0.2 – 20 E-4 | 1-460 ug/L | 0.86-7.3 | 200 | 0.86 |
| DDE | 318.1 | 1.7 – 10 E-4 | 1-55 ug/L | 0.8-124 | 200 | 0.98 |
| DDD | 320.1 | 1-9 E-4 | 2-160 ug/L | 0.27-9 | 200 | 1.02 |
| 2,3,7,8-TCDD (D4) | 322.0 | 1.2 – 6.2E-4 | 8-200 ng/L | 3.347 | 40 | 0.99 |
| 1,2,3,4-TCDD | 322.0 | 6.38E-6 | 640 ng/L | 3.8 | 8.6 | 1.14 |
| 1,2,3,7,8-PeCDD | 356.4 | 4.23E-6 | 120 ng/L ^a | 0.266 | 40 | 1.18 |
| 1,2,3,4,7,8-HxCDD | 391.0 | 1.45E-6 | 4.4 ng/L | 1.084 | 40 | 1.36 |
| 1,2,3,4,6,7,8-HpCDD | 425.2 | 1.77E-7 | 2.4 ng/L | 1.273 | 40 | 1.61 |
| OCDD | 460.0 | 1E-10 -9E-7 | 0.074-0.4 ng/L | 0.684 | 60 | 1.96 |
| 2,3,7,8-TCDF | 306.0 | 1.2 – 2E-4 | 419 ng/L | 1.461 | 40 | 1.00 |
| 1,2,3,7,8-PeCDF | 340.4 | 1.72E-5 | 236 ng/L ^b | 0.505 | 40 | 1.14 |
| 1,2,3,6,7,8-HxCDF | 374.9 | 3.08E-6 | 17.7 ng/L | 1.454 | 37 | 1.31 |
| OCDF | 443.8 | 5.0 E-10 | 1.4 ng/L | 0.191 | 61 | 1.95 |
| DMA | 138.0 | n.a. but low | 2 kg/kg | n.a. but very low | 1000 | n.a. |

RRT was calculated from retention times on a DB1 column; *vapour pressure data represent subcooled liquid vapour pressure's; ^a for 1,2,3,4,7-PeCDD; ^b for 2,3,4,7,8-PeCDF

For the volatilization study, the rotary evaporator was disassembled and all sections which could come into contact with the chemicals were thoroughly cleaned to avoid contamination of the samples. In order to determine the quantity of water which had been distilled, the mass of the collection flask was determined before the start of the experiment. The temperature was controlled through a water bath which was set to 58⁰ C as described in the manual for the ships distillation unit. In all experiments the goal was to slowly distill a fraction of the water and evaluate the amount of dioxins and organochlorines which co-distilled. Although in the initial proposal it was only proposed to analyze one distillate it was decided to distill two fractions, a first fraction of about 10 % of the water and a second fraction with about a further 30 % of the water. For the distillation process the round bottom flask was attached to the rotary evaporator and the vacuum in the system was increased until about 13 - 14 % of atmospheric pressure (14 kPa) was reached. The flask was rotated to increase the surface area of the water to be distilled and to avoid 'bumping'. Chilled water (10⁰C) was circulated through the condenser unit. The experiment was carefully observed over the first few minutes until the water temperature in the flask had increased to the assigned temperature to avoid 'bumping'. Once the system temperature had equilibrated the vacuum was carefully decreased to a pressure which resulted in a slow and steady distillation of the water. Markings on the collection flask allowed a rough assessment of the quantity which had been distilled and in the distilled fraction 1 (F1) about 10 % of the water and in the distilled fraction 2 (F2) a further 30 % of the water was collected (For details of the collected fraction see Appendix I Table 3.). Following the distillation of the first fraction F1, the inside of the condenser unit of the rotary evaporator was rinsed with RO water which was added to the F1 fraction. Once the second fraction, F2, was distilled the inside of the rotary flask was rinsed with about 10 mL of acetone followed by dichloromethane, both of which were added to F2.

Following the distillation the various fractions F1, F2 as well as the remaining non-distilled water (R) were transferred into separating funnels and subjected to liquid-liquid partitioning using dichloromethane and hexane. The nonpolar fractions were combined and concentrated to a small volume (< 500 µl). In the preliminary

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experiment and Expt.1 the samples were quantitatively transferred into 50 μl microvial inserts, concentrated under a gentle stream of nitrogen to almost dryness and filled with 15 μl of toluene.

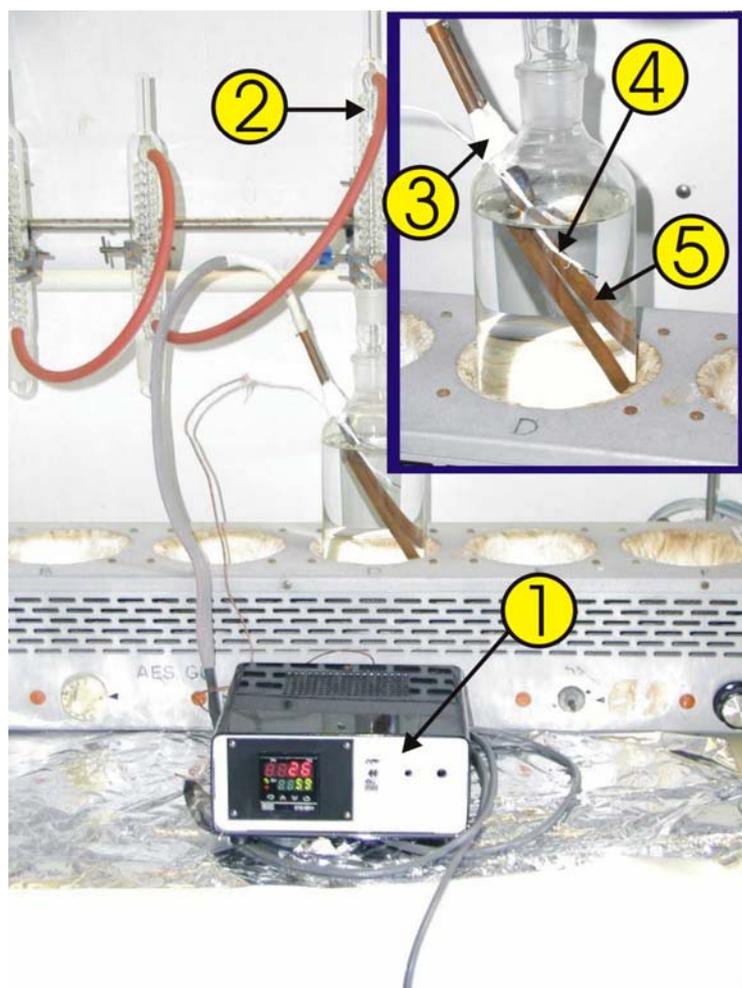
For Experiment 2 it was decided to include a clean-up step using H_2SO_4 and KOH impregnated silica gel in series in a Pasteur pipette. Samples were eluted using hexane, the hexane was evaporated and the samples were transferred into vials, concentrated and filled with toluene as described above.

Since it was of great relevance to this study to detect congeners in all three fractions (F1, F2 and R) to undertake the mass balance type approach we decided to use the most sensitive tool available for this study. The fact that known quantities of standards were added allowed the use of gas chromatography coupled to electron capture detection where the sample was injected onto columns of varying polarity. The results from the experiment indicated that mass spectrometric quantification was sufficiently sensitive for this study. Hence further analysis of PCDD/Fs and organochlorines were performed on a Gas Chromatograph (DB-5 fused silica column, 30 m, 0.25 mm i.d., 0.24 μm film thickness) interfaced to a quadropole mass spectrometer operating in selective ion monitoring mode. Organochlorines and PCDD/Fs were identified using retention times in the standard solution and evaluation of correct isotope ratios M^+ and M^{2+} . Quantification was undertaken by external calibration against some standard used to spike the samples. (Note that the study did not require absolute quantification of the concentrations since the aim was to evaluate the relative proportions of the chemicals of interest in various fractions of the distillate.)

Formation Experiments

In order to evaluate “de-novo” formation of dioxins in the distillation unit itself from precursors, a system was developed in which an electrical heating element was inserted into copper tubing to represent the heating element in the distillation units aboard ships which also consisted of copper tubing (Figure 3). The element was operated in connection to a thermocouple so it could operate at a water temperature of 55°C , which was similar to that in the ship’s distillation unit.

The copper coated element was then mounted at an angle into glass jars that were specifically designed to allow the element to be sealed inside while a condenser was mounted onto the top to make certain that the distilled water was reused in the flask. The formation experiments were conducted using the Agent Orange components 2,4,5-T and/or 2,4-D, which could act as a precursor for formation of TCDD and PeCDD. In the formation experiment, empty flasks were spiked with precursors, then RO water and additionally 30 g NaCl was added to obtain a salinity level similar to that of an outer estuary. The water was then equilibrated for 5 days or more. The copper coated heating element and thermocouple were then inserted into the solution and the opening sealed with Teflon tape. Finally, the condenser was inserted into the top of the flasks and the thermocouple was set at 55°C in the outer periphery of the flask. The formation experiments were carried out for 12 hours. In addition to the test samples, blank samples containing 2,4,5-T and/or 2,4-D were added to water but not heated, and a blank consisting only of water were also included.



1. Temperature Controller
2. Condenser
3. Teflon Seal
4. Thermometer
5. Copper header

Figure 3: Experimental set-up of the formation experiments. Agent Orange components were spiked into the glass vessel and heated up using the copper coated heating element.

RESULTS AND DISCUSSION

Blanks – Distillation Experiments

The series of blanks included in the study were: undistilled RO water, RO water with suspended solids, NaCl and water from the Brisbane River. These were analysed together with fractions F1, F2 and R. None of the compounds of interest were detected in any of the “blank” samples which were not spiked with the SPIKE standards used.

Co-distillation of Dioxins and Organochlorines from Pure Water

In Experiment 1 the emphasis was on evaluating the distillation method of RAN vessels and to determine whether co-distillation is observable. This was initially tested in the simplest experimental configuration where organochlorine pesticides and dioxins were added to a flask, equilibrated with RO water and then the sample was distilled. In the first experiment 14.1 % of the water was distilled and the distilled water was analysed for the compounds initially added. For this initial experiment the remaining water was analysed but either due to loss of the compounds, during the extraction, interferences and/or absence of the compounds no significant quantities could be detected in the second distillate and the remaining fraction. The results are thus comparable with a water sample which was spiked and equilibrated but not distilled and where the compounds were re-extracted from the initial water sample.

The results of the first distillation experiment are shown in Figure 4. These results clearly demonstrated that organochlorines and dioxins have the potential to co-distill relatively rapidly. Overall, the relative quantity of the chemicals of interest in the initially distilled fraction, compared with the concentration in the non-distilled water, ranged from 113 % to about 13 % (Appendix I, Table 1). Co-distillation was greatest for the organochlorine pesticides and lower chlorinated PCDD/Fs. These initial results suggested that almost the entire quantity of chemicals such as hexachlorobenzene or gamma hexachlorohexane and about 73- 85 % of the DDTs co-distilled with the first 14 % of water. For the polychlorinated dibenzodioxins and dibenzofurans there was a

trend to decrease co-distillation with increase in chlorine number. Interestingly in this first experiment 2,3,7,8-tetrachlorodibenzodioxin, the most toxic of all PCDD/Fs and the main contaminant in Agent Orange, was found at about 85 % of the quantity observed in the nondistilled samples and thus co-distilled the most. In contrast only about 10 – 20 % of the hepta- and octachlorinated PCDD/Fs were found in the distilled fractions in the first experiment.

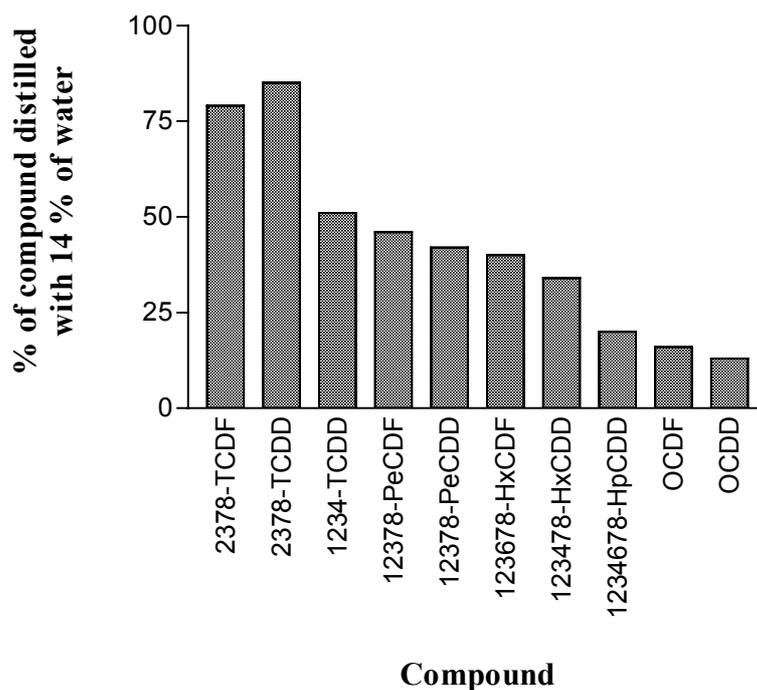


Figure 4: Co-distillation of organochlorines from pure RO water when 14 % (141 mL of 1000 mL) was distilled. The values represent the quantity in the samples distillate relative to the total quantity in a control of spiked but non-distilled water.

Effect of Salinity on Co-distillation of Organochlorines

The effect of salinity on co-distillation was assessed in Experiment 1 by evaluating the co-distillation of dioxins and organochlorines at varying NaCl concentrations. In addition to the control (with no NaCl) 15 and 30 mg/L NaCl treatments were distilled after the equilibration period and the different fractions analysed. As observed in DVA1-1, DVA1-2 and DVA1-3 organochlorines distilled very quickly and were largely associated with the initial fraction of the distilled water (Appendix I, Table 1). This was also the case for 2,3,7,8-TCDD of which ~ 95 and 87 % were found in F1 of DVA1-2 and DVA1-3 respectively. In order to compare the results, the relative fraction of the chemicals co-distilled in the first fraction were plotted against the relative retention times (RRT) of the chemicals on a DB5 capillary GC column where the reference value of 1 was attributed to 2,3,7,8-tetrachlordibenzofuran (See Table 1) (Note that the RRT of chemicals were determined in Experiment 1 and can be related to physico-chemical properties of the compounds of interest such as water solubility or vapour pressure, both of which are likely to influence the water volatility of hydrophobic organic chemicals significantly). Due to uncertainties in the accurate determination of the most relevant physico-chemical properties which govern water volatility (i.e. water solubility, vapour pressure or in combination as Henry's Law Constants) the authors have not plotted co-distillation against these properties although it is clear that these govern the water volatility. To avoid the process of selecting the "best" value from the literature the authors have decided to use the RRT which is an easily obtainable measure and is also related to the relevant physico-chemical properties of compounds. Overall the organochlorines have a RRT less than 1, and the RRT of PCDD/Fs increases with increasing molar mass).

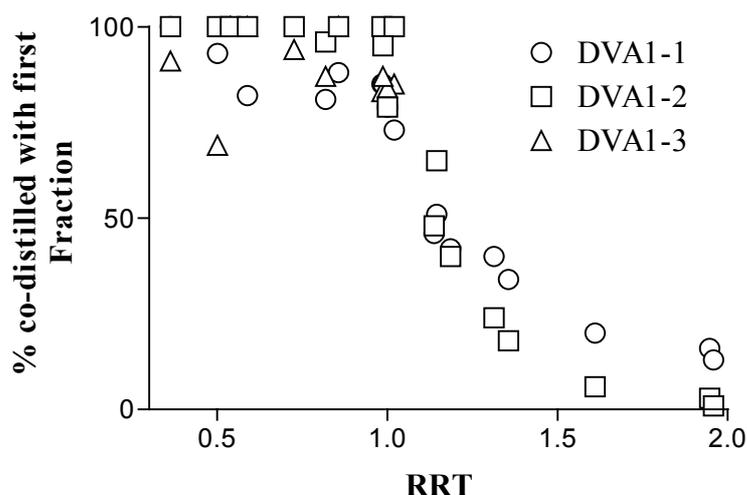


Figure 5: Plot of the relative retention times of organochlorines and PCDD/Fs versus the % co-distilled in F1 at various NaCl concentrations (DVA1-1: 0g NaCl L⁻¹; DVA1-2: 15g NaCl L⁻¹; DVA1-3: 30 g NaCl L⁻¹)

The data obtained in Experiment 1 were not suitable to quantify the effect of salinity on the co-distillation of the compounds of interest (Figure 5). The differences would be observed best with the compounds which did not rapidly co-distill but overall showed a good tendency for co-distillation. Unfortunately, interferences in the chromatogram from DVA1-3 (30g NaCl per litre) did not allow us to interpret the data from those compounds. Further uncertainties arose from differences in the amount of water which was distilled at the three NaCl levels. The interpretation was complicated by the fact that some chemicals such as the organochlorines of interest or lower chlorinated PCDD/Fs volatilise relatively quickly and are thus depleted in the remaining water phase (R). Therefore, the quantity which co-distills decreases with longer distillation times. In contrast, chemicals such as the heptachlorinated dibenzodioxins and octachlorinated dibenzodioxins and dibenzofurans concentrate in the R phase with ongoing distillation and a slight increase of co-distillation with increasing distillation can be expected. In order to evaluate this, a second fraction F2 was distilled in addition to the first fraction F1. The ratio of F1 to F2, normalised to the amount of water distilled, provides an indication of the extent to which co-distillation is decreasing/increasing with increasing distillation of the water. Chemicals for which co-distillation is more rapid than the distillation of water and

which are thus depleted should be relatively enriched in F1 compared to F2. A plot of the RRT versus the ratio of F1/F2 demonstrates that with increasing RRT the ratio decreases from > 40 for chemicals such as HCB and approaches 1 for the higher chlorinated PCDD/Fs (Figure 6). This suggests that most chemicals investigated were already depleted during the distillation of F2 and only the higher chlorinated PCDD/Fs have lower concentrations in the distilled water than in the source water if the sorption compartment in the water (i.e. organic matter and suspended solids) is very low.

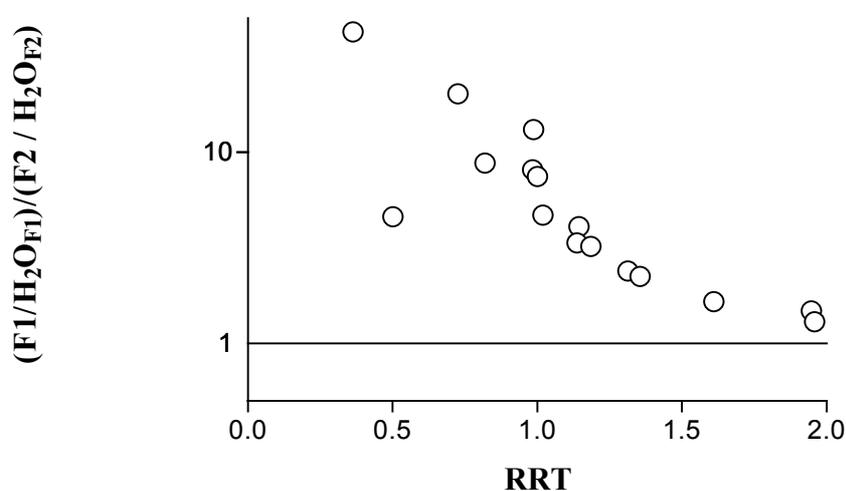


Figure 6: Plot of the relative retention times of organochlorines and PCDD/Fs versus the ratio of chemical co-distilled in F1 and F2 normalised to the respective water amount distilled (log 10 scale). Data from DVA2-9 with 30 g NaCl L⁻¹ and no TSS added.

An alternative way to analyse the results from the co-distillation study is to compare the fraction of chemical, as % of the total, that was distilled in F1 ($F1/(F1+F2+R)$) to the co-distilled fraction of the chemical in F2 (i.e. $F2/(F2+R)$). The data from DVA2-9, for example, indicate that co-distillation of those chemicals which do deplete relatively quickly is enhanced in F1 when the results are expressed on a volume basis (Figure 7). The reason for this behaviour is unclear at this stage but may suggest that some of the chemical are sorbed to the surface of the glass and the

lower relative co-distillation percentage in the second fraction is the result of kinetic limitation of desorption from the surface.

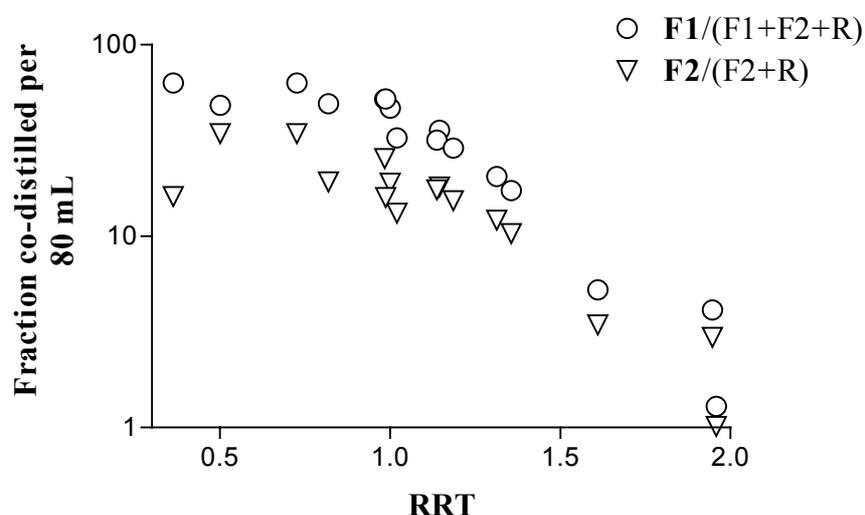


Figure 7: Plot of the relative retention times of organochlorines and PCDD/Fs versus the fraction of chemical co-distilled in F1 and F2 respectively – both normalised to a water volume of 80 mL (10 % of total water used in Expt 2). Data from DVA2-9 with 30 g NaCl L⁻¹ and no TSS added.

Despite these uncertainties it is noteworthy that an increase rather than a decrease in the co-distillation of chemicals with higher RRT was observable with an increase in salt concentration which is in agreement with the thermodynamics of mixtures and their colligative properties associated with increase in the boiling point of water.

Effect of Sorption Compartments in the Water on Co-distillation of Chemicals

To study the effect of extra sorption compartments in the water on the co-distillation of the chemicals of interest, a sample (DVA1-4) was spiked and equilibrated with a known concentration of total suspended sediments (TSS) (0.2 g L⁻¹) and 20 g NaCl L⁻¹. Inclusion of a new sorption compartment in the system resulted in a decrease in co-distillation compared with samples DVA1-1 – DVA-3. However the results from DVA1-4 with a 0.2 g TSS L⁻¹ sorption compartment showed that significant quantities of the individual compounds can co-distill (see Figure 8).

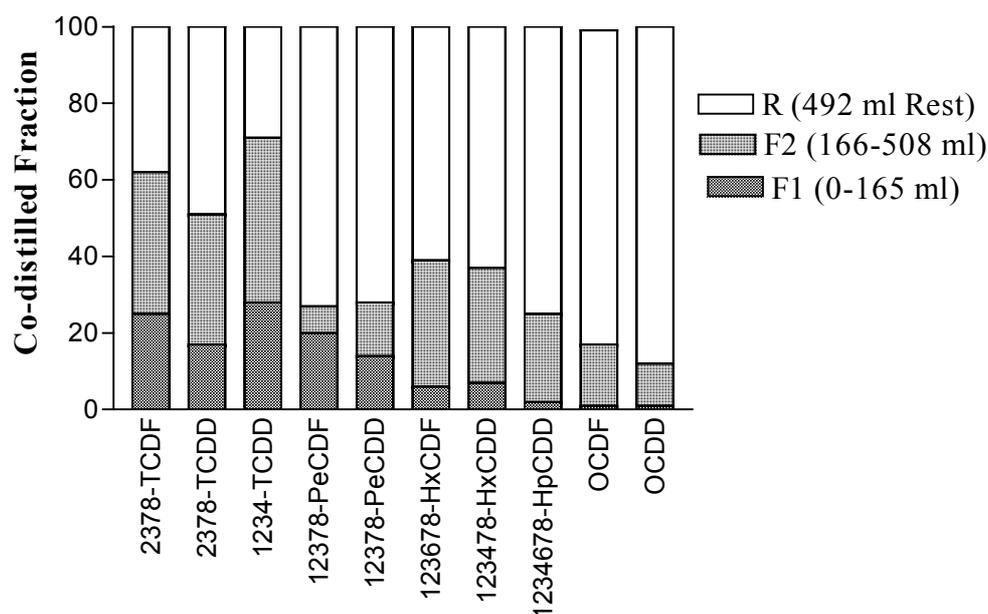


Figure 8: Quantity of PCDD/F congeners co-distilled from water containing 0.2 g total suspended sediment.

The results of the first experiment provided the first indication that suspended solids in the water will affect co-distillation. The main focus of Experiment 2 was to investigate the effect of TSS in the water on the co-distillation of hydrophobic organic chemicals. For the purpose of Experiment 2 water samples were spiked with total suspended sediment to give 0, 0.16, 0.48 and 1.44 g TSS L⁻¹. Then, in accordance with Experiment 1, samples were equilibrated for at least 2 weeks and about 10 % (F1) and an additional 30 % (F2) fractions of the water were distilled from the samples as described above.

Again, in all treatments, significant co-distillation was observed. However, the data indicated a trend of decreasing co-distillation with an increase in the TSS (Figures 9). For example, the co-distillation of 2,3,7,8-TCDD with F1 ranged from about 90 % in one of the control treatments to which no TSS was add to about 38 % in the sample with 1.44 g TSS L⁻¹. Furthermore the results suggest that the effect of TSS on co-distillation is compound-specific with a trend of decreasing effect of TSS on co-distillation efficiency (i.e. the slope of the curve decreases) with increasing compound

RRT (Figure 10). However, the linear regression models using the data set in Appendix I, Table 2, which have been obtained from the linear regression lines demonstrate that if any difference in effect between compounds is observable, the effect of TSS on a decrease in co-distillation is greatest for the higher molar mass chemicals such as OCDD or TCDD (i.e. the linear regression models in Table 2, Appendix I, predict that for DDTs or tetra- and pentachlorinated PCDD/Fs an increase in TSS concentration from 0.5 to 1.5 g L⁻¹ will result in a decrease in co-distillation in F1 by about 30-50 % compared to 55 – 70 % decrease for hepta- and octachlorinated PCDD/Fs).

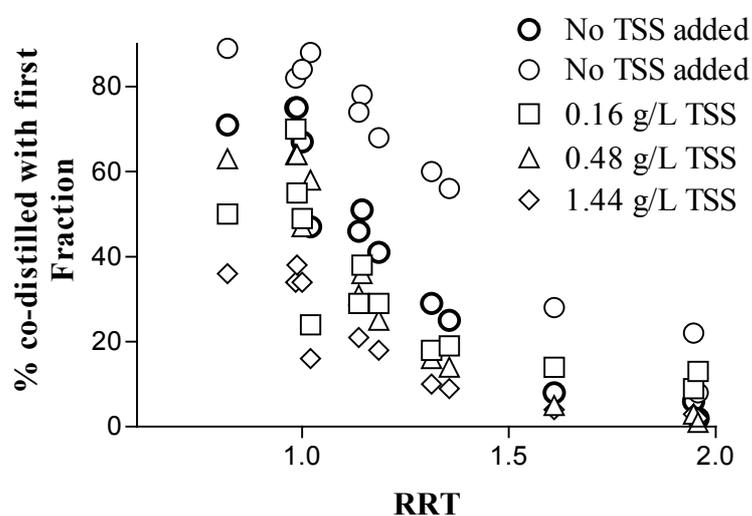


Figure 9: Plot of the relative retention times of organochlorines and PCDD/Fs versus the fraction of chemical co-distilled in F1 at different amounts of total suspended sediments (TSS). Data from DVA2-5 - DVA2-9

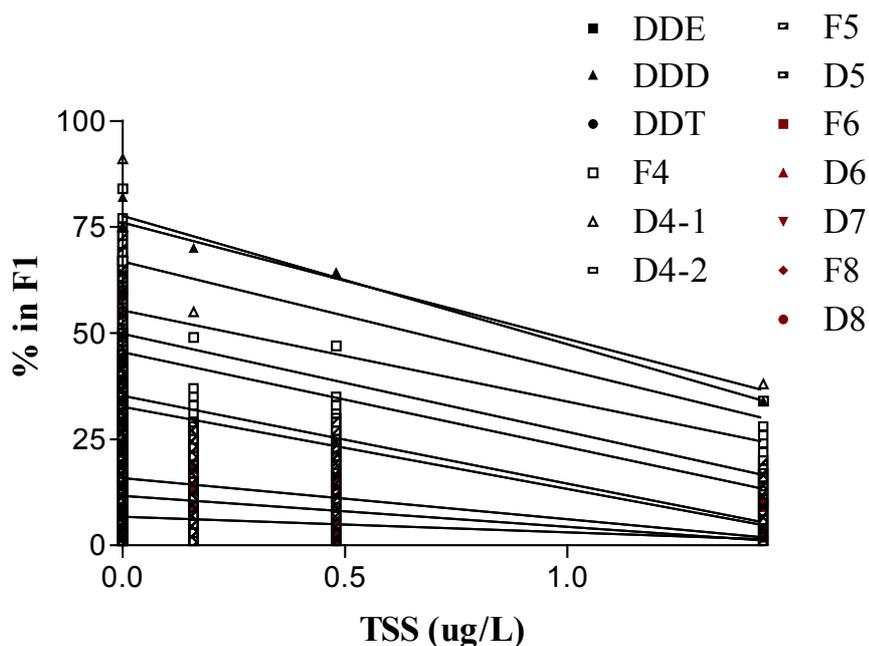


Figure 10: Co-distillation of DDTs and PCDD/Fs from natural estuarine water as a function of total suspended sediments. Data from DVA2-5 - DVA2-9.

At this point co-distillation of organochlorines and PCDD/Fs was investigated in samples which were prepared in the laboratory from pure water and chemicals. To finally confirm these effects, a comparison of these results was made with estuarine water samples. Water was collected from the Brisbane River and a known volume of this water was added to a known quantity of organochlorines and PCDD/Fs, mixed and equilibrated for 4 weeks on a shaker in the laboratory (Note that a blank of the river water was also analysed, but no chemicals of interest were detectable). Again co-distillation of organochlorines and PCDD/Fs was clearly observable in both samples. Between 48 and 62 % of the 2,3,7,8-TCDD was co-distilled with the first 10 % of the distilled water (Figure 11).

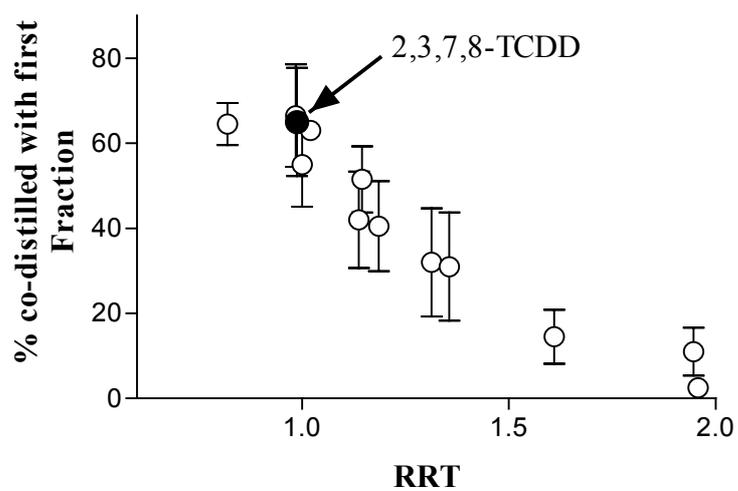


Figure 11: Plot of the relative retention times of organochlorines and PCDD/Fs versus the mean percentage of co-distilled chemicals observed in F1 using water samples collected from the Brisbane River Estuary.

Phase II – Evaluation of Co-distillation of Dimethylarsenic Acid – Agent Blue.

Background to Agent Blue – dimethyl arsenic acid (DMA)

Properties and environmental fate of DMA: In phase 2 of the project the focus was to evaluate co-distillation of Agent Blue, better known as dimethylarsenic acid (DMA) or cacodylic acid. DMA has been and is still used as a contact herbicide. It is still used in some countries such as the U.S.A. as a defoliant in cotton cultivation but also for weed control in areas which are not used for crop production (i.e. around buildings, fences, and in forest management).

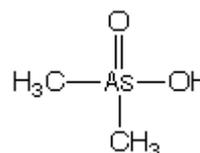
Compound Properties

DMA - C₂ H₇ AS O₂ (62) MW: 138.0

Colorless crystals (pure compound). The technical grade is 65% pure, sodium chloride being one impurity (62).

MELTING POINT: 192-198°C (pure compound) (62); 200°C (pure chemical)

SOLUBILITY: 2 kg/kg water at 25°C (pure compound) (62).



Dimethyl arsenic acid

DMA is a polar compound, highly soluble in water but practically insoluble in organic solvents. It is a clear yellowish-tan liquid. In Vietnam about 4.4 million liters (1.12 Mil.Gallons) of Agent Blue were used between 1962-1971. Interestingly, Agent Blue was used for different purposes than Agent Orange. About 50 % of the Agent Blue was used for crop destruction missions where it was the agent of choice for the destruction of rice crops. The remaining 50 % of Agent Blue was used around base perimeters and was applied by helicopters or ground vehicles (Young et al. 1978 quoted in IoM, 1999). Agent Blue is most effective against grassy plants and acts by absorbing moisture from leaves (desiccation). The application of Agent Blue is usually in the range of 4.5 – 9 litres per acre; for tall grasses, much higher doses were used because of the large foliage masses. According to the pesticide manual DMA is quickly adsorbed on soil particles and ion exchange sites and thus inactivated upon soil contact. Microbial degradation in the soil is considered to be the key process for loss of DMA from the soil with 15-80% of DMA being lost in 8 months. On the other

hand, DMA is practically non-volatile and degradation through UV radiation is not a relevant loss process.

Toxicology of DMA:

The authors of this study are not experts on the toxicity of DMA and therefore asked Dr Jack Ng, the leader of the Metals and Metalloids Toxicity program at ENTOX and an expert on arsenic toxicity, to provide a review of DMA and arsenic toxicity to this report. This review is beyond the scope of this report and therefore provided as an appendix (Appendix II). The report is provided here but remains the intellectual property of Dr Ng.

Result of the DMA Distillation Experiment

In this study we investigated whether in addition to Agent Orange contaminants, Agent Blue may also co-distil and thus drinking water contaminated with DMA may be a relevant exposure pathway. For the purpose of this study DMA (approximately 1 µg) was placed in a flask and RO water added. The solution was equilibrated as previously. Three distilled fractions (10 %, 10-30% and 30-90%) as well as the non-distilled fractions were analysed using ICPMS.

No DMA or only trace levels of DMA, detectable as arsenic, was found in any of the distilled fractions. In contrast the non-distilled fractions required dilution prior to analysis since they contained the now concentrated DMA. Since DMA is a very polar compound with extremely low vapour pressure and high water solubility this result was in agreement with the working hypothesis that DMA should not co-distill. This result suggests that DMA is unlikely to have been consumed in substantial quantities through drinking water that was produced by distillation.

A further issue is that DMA, unlike Agent Orange, was apparently used primarily for crop destruction and around army bases and not for the defoliation of mangrove areas. Hence at this point it seems less likely that DMA from the spraying of Agent Blue contributed significantly to the direct exposure of RAN members on board ships. An exception, however, may be the potential contamination of food (and in particular fish) and fresh water that was obtained from Vietnam. At this point some uncertainty

persists to whether food was obtained in significant quantities from Vietnam for supply on board navy ships.

Formation Study

Formation of PCDD/Fs from Agent Orange components could not be measured in the water with the present experimental set-up. This aspect of the experimental studies was not fully satisfactory. The 2,4,5-T used for the experiment was contaminated with PCDDs which complicates the interpretation of the results (i.e. we would have to evaluate an increase of TCDD concentration which is much more difficult than the appearance of a practically non-existent compound). Secondly, there were problems with determining suitable spiking volumes of Agent Orange components as well as with the standard solutions for elution. Finally, a key problem of this section of the study was that the temperature of the surface of the copper in the distillation unit on board ship was probably about 110 °C while the copper element in the laboratory experiments were set to heat the water to 60 °C (which is equivalent to the water temperature in the distillation units on ships). While we can draw few conclusions from the formation experiments to date we aim to set-up new experiments in near future in which we hope to resolve most of the problems that disallow any valid conclusion from this part of the study.

Preliminary Estimation of Dioxin Exposure Uptake by Sailors from Water

A key goal of this study was to attempt an evaluation of exposure of the RAN personnel aboard ship on duty in Vietnam by consumption of contaminated water as well as from other potential pathways. The following assessment should only be seen as a preliminary attempt as still many of the input factors remain uncertain. Firstly, we have attempted to estimate exposure directly through the consumption of distilled water. Due to the lack of information on water concentrations of PSDD/Fs from the period and at sites of interest, we attempted to calculate water concentrations from concentrations in fish from Vietnam waters taken during the relevant period (Baughman and Meselson, 1973).

The TCDD concentration in fish described by Baughman and Meselson (1973), gives an estimate of the fish lipid concentration (2.5 %) and experimentally determined fish-water bioconcentration factors are available (Loonen et al. 1994). A best estimate suggests that the TCDD water concentration in the Vietnam waters where Baughman and Meselson 's fish were exposed were in the range of 0.04 – 0.7 ng/L.

The distillation experiments carried out in the present study show that, if we assume that about 50 % of the water is distilled and the rest is discharged, distillation would result in an enrichment of TCDD by about a factor of 2 (i.e. 0.08 – 1.4 ng/L). Recent information suggests that possibly only 5-10 % of the water was distilled which would increase this enrichment factor to approximately 10-20, depending on the TSP of the water.

We assumed that on average sailors consumed on average 5 L of this water per day. The direct consumption of this water would lead to a daily body burden of about 0.4 – 7 ng/day. The water was also used to prepare food and as a result of the hydrophobic character of the dioxins the TCDD would also accumulate in the food. Hence we may estimate that the total exposure due to water contaminating food was similar to that of the direct consumption of drinking water – another 0.4 – 7 ng/day.

Table 2: Exposure Model Input Parameter including uncertainty.

| | | Literature values used | Estimated Uncertainty | Reference |
|---|--|---------------------------|-----------------------|----------------------------|
| A | Concentration of TCDD in fish | 49 – 1020 ng/kg FW | Large > 10 | Baughman & Meselson, 1973 |
| B | Estimated fish lipid content to calculate fish lipid based concentration | 2.5 % | 2-4 | Mueller et al. 1997 |
| C | Estimated fish lipid based concentration $C=A*100/B$ | 2.8 – 44 ug/kg lipid | See col. A & B | |
| D | Bioconcentration factor (fish lipid- water) | 81000-170000 | Medium – less 5 – 10 | Loonen et al. 1994 |
| E | Calculated water concentration upper estimate using $E=C/D^s$ | 0.043 – 0.69 ng / L water | Large > 10 | |
| F | Enrichment factor from distillation assuming 50 % water is distilled | 2 [#] | Low (<2) | This study |
| G | Period for which water was distilled during period in Vietnam Waters | 14 days | Low (<5) | Personal Communication RAN |
| H | Daily portion of water directly consumed by RAN members | 5 L | Low (<2) | Personal Communication RAN |

^sNote that the density of the fish lipid was assumed to be 1 kg/L thus the values in C are directly transformed to a per L lipid basis.

[#]Likely to be substantially underestimated according to recently acquired information. Probably 10 is more appropriate.

Now it is important to estimate the overall exposure period. According to reports from RAN personnel the rule on board ship was to produce drinking water primarily during the periods when the ship was in the turbid estuarine water since the water was less pure and could have caused potential damage to the engines if used in boilers. While in the pristine water offshore the distillation units produced water primarily for the ships engine. Hence the drinking water that was produced during the periods in contaminated waters lasted for a significant portion of the return trip. If we assume that a 14 days reserve was produced during such a trip, the total body burden of dioxin through direct consumption of water that originated from distilling in Vietnamese waters are estimated to be between about 10 ng to 190 ng per journey. For a 70 kg person the body burden would be in the range of 12 - 200 pg / kg bw per day for a 14 day period in addition to the background exposure through TCDD and other dioxin-like chemicals in the food and the environment (It should be noted that the late 1960s and the early 1970s were probably the peak exposure periods for these chemicals and the body burden of the population was thus already substantially higher than at present).

The uncertainty of these exposure assessment figures is relatively large. The key uncertainties are related to the concentration in the fish and the calculation of the water concentration from these fish data. Since the anchorage of the Australian RAN ships near Van Tau were in the vicinity of a large mangrove region that was regularly sprayed with Agent Orange, it is likely that the water concentrations at the anchorage were higher than those predicted from the fish data.

Comparison of Exposure Data with Effects

While it is beyond the scope of this study to evaluate effects of dioxin exposure in RAN members it is of interest to put the exposure data into some perspective relative to current guidelines and observable effect values. At present (2002) a series of intake criteria are applied by WHO and various committees in Europe and the key difference in the guidelines are the exposure period over which the intake value is calculated. In principle the WHO has targeted a daily intake of 2 pg TEq/kg bw per day whereas other European committees have expressed their values on weekly exposure (14 pg kg bw per week) or a monthly value (70 pg kg bw per month). In

terms of risk assessment it is reassuring that various committees developed very similar values despite following different methods for establishing these intake values. On the other hand these guidelines are all based on a similar range of toxicological studies where animal studies and biochemical markers were used to obtain low observable effect levels for the various endpoints (van Leuwen 2001, oral presentation at NRCET) (Table 3). Australia's NHMRC has recently distributed a document in which the value of 70 pg per kg bw per month was recommended for adoption in Australia.

A notable exception to these agreements in international guidelines for dioxin exposure in humans is the approach taken by the US-EPA who established a recommended guideline which is more than 2 orders of magnitude lower than the values set by the WHO/ICPS and various European committees. The US-EPA have concluded in their recent dioxin-reassessment document that the dioxin background contamination which is in the range of 0.5 – 2 pg per kg bw per day at the present may pose a significant cancer risk between 10^{-2} – 10^{-3} .

In comparison, the exposure of the RAN members from water related sources may have been 12-200 pg /kg bw /day or 360 – 6000 pg / kg bw / month. This indicates that in addition to normal background exposure, RAN members may have received exposure which is one to two orders of magnitude above the acceptable intake values and at a level above the observed effect levels in experimental animals (Table 3).

Table 3: Animal body burdens and related estimated human daily intake (EHDI)
(Table from van Leeuwen, presentation at NRCET, August 2001)

| Response | Maternal body burden (ng/kg bw) | Human EDI (pg/kg bw 1 day) |
|--------------------------|---------------------------------|----------------------------|
| Rats | | |
| Decreased sperm counts | 28 | 14 |
| Immune suppression | 50 | 25 |
| Genital malformations | 73 | 37 |
| Monkeys | | |
| Neurobehavioural effects | 42 | 21 |
| Endometriosis | 69 | 35 |

Table 4: Acceptable intake guidelines of dioxin-like chemicals. Comparison between the standards for human intake of dioxins, furans and dioxin-like PCBs agreed by WHO (1998), EC-SCF (2001) and FAO/WHO (2001).

| Standard | pg / kg bw / day | pg / kg bw / week | pg / kg bw / month |
|-------------------------|------------------|-------------------|--------------------|
| WHO consultation (1998) | 1-4 | (7-28) | (30- 120) |
| EC-SCF (2001) | (2) | 14 | (60) |
| FAO/WHO (2001) | (2.3) | (16) | 70 |

Exposure values in brackets were calculated for the other periods for easier comparison.

CONCLUSION

This study evaluated the potential for co-distillation of organic pollutants by ship distillation units.

This study clearly demonstrated that distillation of water contaminated with hydrophobic chemicals such as dioxin-like chemicals of organochlorine pesticides results in contamination of the distilled water. The extent of co-distillation largely depends on the physico-chemical properties of the compounds of interest. 2,3,7,8-Tetrachlorodibenzodioxin, the primary contaminant in Agent Orange, the defoliant used in the Vietnam conflict, co-distills relatively rapidly and thus will be enriched in the distilled water. Other PCDD/Fs with up to six chlorine substitutions as well as all organochlorine chemicals investigated also have the tendency to enrich in the distilled water when compared to the source water.

A relationship between physico-chemical properties and co-distillation is apparent since with increasing relative retention time the tendency for co-distillation decreased. The effect of salinity on co-distillation could not be determined, but the studies indicated that suspended particles or a sorption compartment in the water reduced the tendency for co-distillation. However, enrichment of TCDD in the distilled water was still high at 1.4 g TSS L⁻¹. The same was found for water which originated from the Brisbane River Estuary which suggests that co-distillation of such chemicals is highly likely.

During this study, it was noted that evaporative distillation is widely used through the world. Based on the results of this study, the authors strongly recommend that for a prevention of contamination of all who consume water produced by evaporative distillation it should be made certain that the source water used to be distilled does not contain volatile or semivolatile hydrophobic pollutants. If this cannot be guaranteed we recommend the use of a post-distillation purification step such as a sorption phase that removes such contaminants. (Activated charcoal or carbon filters would be suitable for this purpose.)

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APPENDIX I

Watervolatility of PCDD/Fs

Appendix Table 1: Relative quantity of organochlorines and PCDD/Fs. Blue shaded areas mark uncertainty with peaks due to interferences and broad peaks, yellow shaded areas – sample not quantifiable and result based on average of total in other samples – (F2+R)

| | Blank | DVA1-1 0 NaCl; 0 TSS | | DVA1-2 15 g/L NaCl; 0 TSS | | DVA1-3 30 g/L NaCl; 0 TSS | | | DVA1-4 30 g/L NaCl; 0.2 g/L TSS | | |
|---------------------|-------|-------------------------|----------|------------------------------|----------|------------------------------|----------|----------|------------------------------------|-----------|-----------|
| | | F1 | R | F1 | R | F1 | F2 | R | F1 | F2 | R |
| Distilled amount | | 141 | 629 | 92 | 620 | 161 | 343 | 496 | 165 | 343 | 492 |
| HCB | ND | 114 | ND | 100 | 0 | 91 | 9 | 0 | NA | NA | NA |
| Lindane | ND | 93 | ND | 100 | 0 | 69 | 31 | 0 | NA | NA | NA |
| Heptachlor | ND | 101 | ND | 100 | 0 | 100 | 0 | 0 | NA | NA | NA |
| Aldrin | ND | 82 | 3 | 100 | 0 | 100 | 0 | 0 | NA | NA | NA |
| Heptachlorepo. | ND | 102 | 1 | 100 | 0 | 94 | 6 | 0 | NA | NA | NA |
| DDE | ND | 81 | ND | 96 | 4 | 87 | 11 | 2 | NA | NA | NA |
| dieldrin | ND | 88 | ND | 100 | 0 | 100 | 0 | 0 | NA | NA | NA |
| DDD | ND | 85 | ND | 100 | 0 | 83 | 16 | 1 | NA | NA | NA |
| DDT | ND | 73 | ND | 100 | 0 | 85 | 14 | 1 | NA | NA | NA |
| 2,3,7,8-TCDF | ND | 79 | 5 | 79 | 21 | 84 | 11 | 5 | 25 | 37 | 38 |
| 2,3,7,8-TCDD | ND | 85 | 3 | 95 | 5 | 87 | 9 | 3 | 17 | 34 | 49 |
| 1,2,3,4-TCDD | ND | 51 | 10 | 65 | 35 | 82 | 11 | 7 | 28 | 43 | 29 |
| 1,2,3,7,8-PeCDF* | ND | 46 | 15 | 48 | 52 | 80 | 0 | 20 | 20 | 7 | 73 |
| 1,2,3,7,8-PeCDD* | ND | 42 | NQ | 40 | 60 | 84 | 0 | 16 | 14 | 14 | 72 |
| 1,2,3,6,7,8-HxCDF | ND | 40 | NQ | 24 | 76 | 65 | 14 | 21 | 6 | 33 | 61 |
| 1,2,3,4,7,8-HxCDD | ND | 34 | NQ | 18 | 82 | 57 | 17 | 26 | 7 | 30 | 63 |
| 1,2,3,4,6,7,8-HpCDD | ND | 20 | NQ | 6 | 94 | 45 | 18 | 37 | 2 | 23 | 75 |
| OCDF | ND | 16 | NQ | 3 | 97 | 42 | 12 | 47 | 1 | 16 | 82 |
| OCDD | ND | 13 | NQ | 1 | 99 | 40 | 6 | 54 | 1 | 11 | 88 |

Watervolatility of PCDD/Fs

Experiment 2

| | DVA2-3 30 g/L NaCl; 0 TSS | | | DVA2-4 Brisbane River | | | DVA2-5 Brisbane River | | | DVA2-6 0.2 g TSS L ⁻¹ ; 30 g NaCl | | |
|---------------------|------------------------------|----------|----------|--------------------------|-----------|-----------|--------------------------|-----------|-----------|---|-----------|-----------|
| | F1 | F2 | R | F1 | F2 | R | F1 | F2 | R | F1 | F2 | R |
| Distilled amount | 78 | 231 | 491 | 64 | 233 | 503 | 84 | 239 | 509 | 84 | | 492 |
| HCB | | | | | | | | | | | | |
| Lindane | | | | | | | | | | | | |
| Heptachlor | | | | | | | | | | | | |
| Aldrin | | | | | | | | | | | | |
| Heptachlorepo. | | | | 46 | 54 | 0 | | | | | | |
| DDE | 89 | 9 | 2 | 61 | 29 | 11 | 68 | 13 | 19 | 50 | 27 | 22 |
| dieldrin | | | | | | | | | | | | |
| DDD | 82 | 17 | 1 | 58 | 38 | 4 | 75 | 21 | 4 | 70 | 30 | 0 |
| DDT | 88 | 11 | 2 | 62 | 38 | 0 | 64 | 21 | 15 | 24 | 35 | 41 |
| 2,3,7,8-TCDF | 84 | 9 | 7 | 48 | 36 | 16 | 62 | 18 | 20 | 49 | 29 | 22 |
| 2,3,7,8-TCDD | 91 | 7 | 3 | 56 | 34 | 10 | 74 | 14 | 12 | 55 | 32 | 13 |
| 1,2,3,4-TCDD | 78 | 10 | 12 | 46 | 36 | 18 | 57 | 15 | 27 | 38 | 28 | 34 |
| 1,2,3,7,8-PeCDF* | 74 | 13 | 13 | 34 | 40 | 26 | 50 | 18 | 33 | 29 | 33 | 38 |
| 1,2,3,7,8-PeCDD* | 68 | 15 | 17 | 33 | 40 | 27 | 48 | 16 | 35 | 29 | 30 | 41 |
| 1,2,3,6,7,8-HxCDF | 60 | 17 | 22 | 23 | 36 | 41 | 41 | 13 | 46 | 18 | 26 | 56 |
| 1,2,3,4,7,8-HxCDD | 56 | 18 | 26 | 22 | 35 | 43 | 40 | 12 | 48 | 19 | 25 | 56 |
| 1,2,3,4,6,7,8-HpCDD | 28 | 14 | 58 | 10 | 19 | 70 | 19 | 4 | 77 | 14 | 11 | 75 |
| OCDF | 22 | 14 | 64 | 7 | 11 | 82 | 15 | 3 | 82 | 9 | 6 | 84 |
| OCDD | 8 | 7 | 85 | 3 | 5 | 92 | 2 | 2 | 96 | 13 | 4 | 84 |

Shaded columns – due to interferences no quantification could be performed in this sample – the % are calculated assuming the mean recovery from DVA2-3 and DVA-4.

Watervolatility of PCDD/Fs

| | DVA2-7 30 g/L NaCl; 0.60 g TSS L ⁻¹ | | | DVA2-8 30 g/L NaCl; 1.8 g TSS L ⁻¹ | | | DVA2-9 30 g/L NaCl; 0 g TSS L ⁻¹ | | |
|---------------------|---|-----------|-----------|--|-----------|-----------|--|-----------|-----------|
| | F1 | F2 | R | F1 | F2 | R | F1 | F2 | R |
| Distilled amount | 86 | 219 | 495 | 75 | 238 | 487 | 115 | 233 | 452 |
| HCB | | | | | | | | | |
| Lindane | | | | | | | | | |
| Heptachlor | | | | | | | | | |
| Aldrin | | | | | | | | | |
| Heptachlorepoxy. | | | | | | | | | |
| DDE | 63 | 26 | 11 | 36 | 36 | 29 | 71 | 16 | 13 |
| dieldrin | | | | | | | | | |
| DDD | 64 | 29 | 7 | 34 | 42 | 24 | 75 | 19 | 7 |
| DDT | 58 | 31 | 12 | 16 | 46 | 38 | 47 | 20 | 33 |
| 2,3,7,8-TCDF | 47 | 33 | 20 | 34 | 31 | 35 | 67 | 18 | 15 |
| 2,3,7,8-TCDD | 64 | 24 | 12 | 38 | 36 | 26 | 75 | 12 | 13 |
| 1,2,3,4-TCDD | 36 | 33 | 31 | 29 | 29 | 43 | 51 | 26 | 23 |
| 1,2,3,7,8-PeCDF* | 31 | 33 | 36 | 21 | 26 | 54 | 46 | 28 | 27 |
| 1,2,3,7,8-PeCDD* | 25 | 34 | 41 | 18 | 24 | 58 | 41 | 26 | 32 |
| 1,2,3,6,7,8-HxCDF | 16 | 26 | 58 | 10 | 14 | 76 | 29 | 25 | 46 |
| 1,2,3,4,7,8-HxCDD | 14 | 28 | 58 | 9 | 16 | 74 | 25 | 23 | 53 |
| 1,2,3,4,6,7,8-HpCDD | 5 | 13 | 82 | 4 | 9 | 87 | 8 | 9 | 83 |
| OCDF | 3 | 7 | 90 | 3 | 7 | 91 | 6 | 8 | 86 |
| OCDD | 1 | 5 | 94 | 2 | 5 | 93 | 2 | 3 | 95 |

Appendix I Table 2: Parameters from linear regression between the relative fraction of chemicals codistilled in F1 (%) and the amount of total suspended sediments in the treatment. In all cases $n = 5$ and besides two controls treatments included 0.16; 0.48 and 1.44 g TSS L⁻¹.

| Compound | slope | intercept | r ² | p |
|---------------------|-------|-----------|----------------|--------|
| DDE | -21 | 73 | 0.62 | 0.11 |
| DDD* | -24 | 76 | 0.97 | 0.0017 |
| DDT | -23 | 59 | 0.37 | 0.28 |
| | | | | |
| 2,3,7,8-TCDD | -22 | 76 | 0.69 | 0.084 |
| 1,2,3,4-TCDD | -17 | 55 | 0.45 | 0.22 |
| 1,2,3,4,7-PeCDD | -18 | 46 | 0.47 | 0.20 |
| 1,2,3,4,7,8-HxCDD | -16 | 33 | 0.40 | 0.25 |
| 1,2,3,4,6,7,8-HpCDD | -7.8 | 16 | 0.35 | 0.29 |
| OCDD | -2.9 | 6.7 | 0.18 | 0.47 |
| 2,3,7,8-TCDF | -21 | 67 | 0.63 | 0.11 |
| 2,3,4,7,8-PeCDF | -19 | 50 | 0.45 | 0.22 |
| 1,2,3,4,7,8-HxCDF | -17 | 35 | 0.40 | 0.25 |
| OCDF | -5.8 | 12 | 0.31 | 0.33 |

Appendix I Table 3: Experimental set-up for distillation experiments including quantity of water distilled with each fraction (in mL)

| | F1 | F2 | Rest |
|--|-----------------------|----------------|------|
| Experiment 1 – Total Volume 1000 mL | | | |
| Expt 1: (DVA-1 0- B0) RO Blank | Combined | | |
| Expt 1: (DVA-1-B11) RO – Blank | 107 | 893 (Combined) | |
| Expt 1: (DVA-1-B12) RO + TSS Blank | Combined | | |
| Expt 1: (DVA-1-V1) RO + SPIKE | 141 | 230 | 629 |
| Expt 1: (DVA-1-V2) RO + 15 g NaCl + Spike | 92 | 288 | 620 |
| Expt 1: (DVA-1-V3) RO + 30 g NaCl + Spike | 161 | 343 | 496 |
| Expt 1: (DVA-1-V4) RO + 30 g NaCl + 0.2 g TSS +SPIKE | 165 | 259 | 575 |
| Expt 1: (DVA-1-R) RO + 30 g NaCl + 0.2 g TSS +SPIKE | Combined for recovery | | |
| Experiment 2 – Total Volume 800 mL | | | |
| Expt.2 (DVA-2-1) RO + 24 g NaCl – BLANK | 70 | 280 | 450 |
| Expt.2 (DVA-2-2) Brisbane River – BLANK | 65 | 230 | 505 |
| Expt.2 (DVA-2-3) (RO+24 g NaCl+SPIKE) | 78 | 231 | 491 |
| Expt.2 (DVA-2-4) (Brisbane River + SPIKE) | 64 | 233 | 503 |
| Expt.2 (DVA-2-5) (Brisbane River + SPIKE) | 84 | 239 | 477 |
| Expt.2 (DVA-2-6) (RO+24 g NaCl+ 0.16 g TSS + SPIKE) | 86 | 225 | 489 |
| Expt.2 (DVA-2-7) (RO+24 g NaCl+ 0.48 g TSS + SPIKE) | 86 | 219 | 495 |
| Expt.2 (DVA-2-8) (RO+24 g NaCl+ 1.44 g TSS + SPIKE) | 75 | 238 | 487 |
| Expt.2 (DVA-2-9) (RO+24 g NaCl+ 0 g TSS + SPIKE) | 115 | 233 | 348 |

Appendix I Table 4: Summary of Experimental variations

| | Preliminary Expt | Experiment 1 | Experiment 2 |
|------------------------|--------------------------|--|---|
| Water used | RO-Water | RO-Water | RO-Water Brisbane River Water |
| Salinity | 0 g NaCl L ⁻¹ | 0 g NaCl L ⁻¹ 15 g NaCl L ⁻¹ 30 g NaCl L ⁻¹ | 0 g NaCl L ⁻¹ 30 g NaCl L ⁻¹ |
| Total suspended solids | 0 g L ⁻¹ | 0 mg L ⁻¹ 200 mg L ⁻¹ | 0 mg L ⁻¹ 200 mg L ⁻¹ 600 mg L ⁻¹ 1800 mg L ⁻¹ |
| Compounds used | OC Pesticides | Dioxins and OC pesticides | Dioxins and OC pesticides |
| Clean-up | None | None | Minicolumn |
| Instrumental Analysis | GC-ECD | GC-ECD and GC-MS | GC-MS |

Appendix I Table 5: Variations of the experimental set-up for the formation study.

| Variants | 1-liter dest. Water with 30g NaCl | 1-liter Brisbane-River water |
|----------------------------------|--|--|
| Blank | + 20µl Spike Mix* + 15µl Recovery Standard + Warm up to 60°C with copper header (8h) | + 20µl Spike Mix* + 15µl Recovery Standard + Warm up to 60°C with copper header (8h) |
| 2,4-D | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + Warm up to 60°C with copper header (8h) | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + Warm up to 60°C with copper header (8h) |
| 2,4-D Control | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + Stay for 8h without heading | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + Stay for 8h without heading |
| 2,4,5-T | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4,5-T + Warm up to 60°C with copper header (8h) | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4,5-T + Warm up to 60°C with copper header (8h) |
| 2,4,5-T Control | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4,5-T + Stay for 8h without heading | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4,5-T + Stay for 8h without heading |
| 2,4-D and 2,4,5-T | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + 1µg 2,4,5-T + Warm up to 60°C with copper header (8h) | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + 1µg 2,4,5-T + Warm up to 60°C with copper header (8h) |
| 2,4-D and 2,4,5-T Control | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + 1µg 2,4,5-T + Stay for 8h without heading | + 20µl Spike Mix* + 15µl Recovery Standard + 1µg 2,4-D + 1µg 2,4,5-T + Stay for 8h without heading |

*Spike Mix = 50% Internal Standard Solution and
50 %Supplement Internal Standard/Solution

APPENDIX II:

Appendix II A REVIEW OF ARSENIC TOXICITY

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Sources of arsenic in the environment

Arsenic is an element that occurs widely in rocks, soil, water, air, plants, and animals. Arsenic ranks 20th in abundance in the earth's crust (Cullen et al., 1989). Arsenic is a ubiquitous element. The crust of the earth contains about 2mg/kg of arsenic and is highest at 620mg/kg in its deep inside core (Zheng, 1994). Arsenic is a metalloid exhibiting chemical and physical properties of both metallic and non-metallic nature. There are three kinds of arsenic derivatives with the color of grey, yellow and black respectively. Grey arsenic is also called metallic arsenic and is both a fine thermo-conductor and electro-conductor and relatively stable at normal temperatures (Wang, 1997). Arsenic can show either electro-positive or electro-negative valence in its compounds. The valencies of arsenic are 0, -3, 3 and 5. Although arsenic is found in nature to a small extent in its elemental form, it occurs mostly as inorganic and organic compounds in either the As^{(III)(+3)} or As^{(V)(+5)} valency state, as in AsF₃ and AsF₅ (Welch et al., 1988). Arsenic gains three electrons and forms AsH₃ (Arsine) when reacted with hydrogen, but loses three or five electrons in forming As₂O₃ or As₂O₅. These two forms of arsenic are inorganic species and are the forms most prevalent in water (Irgolic, 1994; Clifford et al., 1994; USEPA, 2000).

There are numerous natural sources as well as anthropogenic activities that may introduce arsenic into food and drinking water. The primary natural sources of arsenic include geological formations such as rocks, soil, and sedimentary deposits, geothermal activity, and volcanic activity. These contribute to the regional background base level of arsenic and to abnormal geochemical arsenic conditions in some local areas. The major present and past sources of arsenic distribution influenced by human activities include the use of arsenic in wood preservatives, agricultural uses (including the use as pesticides), industrial uses, mining and smelting

activities. Arsenic contamination in water depends on the level of human activity, the distance from the sources of pollution and the dispersion and fate of the arsenic that is released (USEPA, 2000). A recent estimation has placed the ratio of natural to anthropogenic atmospheric inputs of arsenic at 60:40 (Chilver et al., 1987).

Marine and freshwater organisms usually have much higher concentrations of arsenic in their structure than is present in the background of the surrounding water, but such “fish arsenic”(organic arsenic) is chemically and physiologically different from inorganic arsenate and arsenite. Arsenic occurs in seawater mainly as inorganic arsenate, at levels of approximately 2µg/L. In marine animals, levels of arsenic can be as high as 100 mg/kg (Shibata, 1992; Maher, 1988; Tamaki et al., 1992). The presence of organic arsenicals in marine organisms is commonly assumed to be due to the accumulation of compounds that have been synthesised from arsenate. This accumulation results in a much higher concentration of arsenic in the organism than that in the environmental water. Sometimes the level of this accumulation is astonishingly high. For example, in marine algae the mean arsenic concentration is 51.2±14.92 mg/kg, which is about 10000 times higher than that in the seawater. The fact that most of the arsenic in marine organisms is in organic forms indicates that arsenic is not simply entering the organism, but is transformed into organic forms within the organism.

Arsenic pollution in the aqueous environment mainly comes from arsenic containing waste-water produced from industrial processes. These wastes can pollute the soil and water sources. Serious pollution can cause acute arsenic poisoning incidents. In 1961, a mine in Guizhuo province in China produced arsenic-containing waste-water which contaminated the drinking water source and caused acute arsenic poisoning of residents (Zhao et al., 1963).

Acute toxicity

From human acute poisoning incidents, the LD₅₀ of arsenic has been estimated to range from 1 to 4 mg/kg (Vallee et al., 1960; Winship, 1984). This dose would correspond to a lethal dose range of 70 to 280 mg for 50% of adults weighing 70kg.

At non-lethal, but high acute doses, inorganic arsenic can cause gastroenterological effects, shock, neuritis (continuous pain) and vascular effects in humans (Buchanan, 1962).

Although certain arsenic compounds are toxic to humans, others are not. The toxicity of arsenic varies greatly with its chemical species, and its toxicity has a strong relationship with the status of arsenic in the component: inorganic arsenic is believed to be more toxic than organic arsenicals (Kaise et al., 1992). The LD50 values in rats for some arsenic species being (in mg/kg): arsine 3, arsenite 14, arsenate 20, monomethylarsonic acid (MMAA) 700-1800, dimethylarsinic acid (DMAA) 700-2600, arsenobetaine >10000 and arsenocholine 6500 (Mellan et al., 1956). The sequence of the toxicity potency of common arsenicals is:



Chronic toxicity

Long term exposure to arsenic results in chronic arsenic poisoning (arsenicosis). This has been reported in people who live in endemic areas with higher arsenic concentrations in drinking water or in burning coal. Arsenicosis has also been reported in people exposed to arsenic in the workplace. Skin lesions are the typical symptom of chronic arsenic poisoning. Among these changes the keratosis of the extremities of the limbs is very typical of arsenic poisoning. Melanosis and lesions are common changes. Chronic arsenic poisoning also causes damages including the respiratory system, digestive system, circulatory system, neural system and renal system. The most significant consequence is that inorganic arsenic causes cancers in various organs especially the skin, lung and bladder (ATSDR, 2000; IPCS, 2001). Episodes of chronic arsenic poisonings are reported around the world from America to Asia and Europe (Smedley et al., 2001).

A large number of adverse effects have been reported in humans after chronically exposure to drinking water highly contaminated with inorganic arsenic. The most

common symptoms of inorganic arsenic exposure appear on the skin and occur after 5-15 years of exposure equivalent to 700 µg/day for 70kg adult. They include alterations in pigmentation and the development of keratoses which are localized primarily on the palms of the hands, the soles of the feet and the torso. The presence of hyperpigmentation and keratoses on parts of the body not exposed to the sun is characteristic of arsenic exposure (Yeh, 1973; Tseng, 1977). Chronic exposure to inorganic arsenic is often associated with alterations in gastroenterological function (Morris et al., 1974; Nevens et al., 1990) and development of peripheral vascular disease (hardening of the arteries to the arms and legs, that can cause pain, numbness, tingling, infection, gangrene and clots). In Taiwan, black foot disease has been the most severe manifestation of this effect. Peripheral neuropathy may be present after long term exposure to inorganic arsenic in drinking water (Hindmarch et al., 1977; Southwick et al., 1983). People consuming arsenic contaminated water have an increased risk of diabetes (Lai et al., 1994; Rahman et al., 1998; Tseng et al., 2002). An increased risk of mortality from diabetes was reported in those occupationally exposed to arsenic in Sweden (Rahman and Axelson, 1995; Rahman et al., 1998).

Inorganic arsenic entering body by drinking water can cause multi-site cancer in human body. Asian, Mexican and South American populations with exposures to arsenic in drinking water generally at or above several hundred micrograms per liter are reported to have increased risks of skin, bladder, and lung cancer in Taiwan (Chen et al., 1985), in Argentina (Hopenhayn-Rich et al., 1996) and in Chile (Smith et al., 1998). The current evidence also suggests that the risks of liver and kidney cancer may also be increased following exposure to inorganic forms of arsenic. The first studies of dose-dependent effects of arsenic associated with skin cancer were observed in Taiwan (Tseng et al., 1968; 1977). Tseng's reports and other corroborating studies such as those by Albores et al. (1979) and Cebrian et al. (1983) on drinking water exposure and exposure to inorganic arsenic in medicines (Cuzick et al., 1982) and in pesticides (Roth, 1956) led the USEPA to use skin cancer as the endpoint to classifying inorganic arsenic as a human carcinogen (group A) by the oral route (USEPA, 1984). It has been estimated that more than 50 million people worldwide are at risk exposing to elevated levels of arsenic from the environment (Table 1).

Table 1. Episodes of arsenic caused poisonings and areas of potential arsenic contamination around the world

| Country or Area | Population at risk | Sources of As exposure | Discover date | References |
|------------------------|---------------------------|--|----------------------|---------------------------------|
| Bangladesh | 50,000,000 | Drinking water: artisan well | 1980s | Saha (1984) |
| West Bengal, India | 1,000,000 | Drinking water: artisan well | 1980s | Chowdhury et al., (1999) |
| Xinjiang, China | 100,000 | Drinking water: artisan well | 1980s | Wang (1994); Cheng (1998) |
| Inner Mongolia, China | 600,000 | Drinking water: bores | 1990s | Cheng (1998) |
| Guizhuo, China | 20,000 | Burning-coal | 1950s | Cheng (1998) |
| Taiwan, China | 200,000 | Artisan well | 1950s | Tseng and Chiou (1958) |
| Ronpibool, Thailand, | 1,000 | Water contaminated by tin mining waste | 1980s | Choprapwon et al., (1995; 2001) |
| Mexica | 400.000 | Drinking water | 1983 | Cebrian et al., (1983) |
| Chile | 437.000 | Drinking water | 1971 | Sancha et al., (2001) |
| Argentina | 2,000,000 | Drinking water | 1981 | Sancha et al., (2001) |
| Bolivia | 20,000 | Drinking water | 1997 | Sancha et al., (2001) |
| Peru | 250,000 | Drinking water | 1984 | Sancha et al., (2001) |
| Vietnam | Millions | Drinking water | 2001 | Berg et al., (2001) |
| Western U S A | - | Drinking water | 1988 | Welch et al., (1988) |
| Hungary | 220,000 | Drinking water | 1974 | Sancha et al., (2001) |
| Romania | 36.000 | Drinking water (bores) | 2001 | Gurzau et al., (2001) |

Recently Berg et al (2001) reported cases of high levels of arsenic contamination in an alluvial tract of the Red River in Vietnam as it passes through the city of Hanoi and the surrounding rural districts. The average arsenic concentration in the ground waters in this area was found to be 159µg/L, and varied from 1 to 3050 µg/L in rural ground

water samples from private small scale bores. In a highly affected rural area, the groundwater used directly for drinking water had an average concentration of 430 µg/L. The high arsenic concentrations found in the bores (48% above 50 µg/L and 20% above 150 µg/L) indicate that several million people consuming untreated groundwater might be at a considerable risk from chronic arsenic poisoning. [The WHO recommended drinking water guideline

Toxicity of organic arsenicals

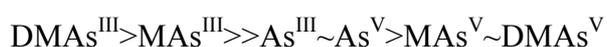
Arsenic exists in several forms that vary in toxicity and occurrence. It is important to consider those forms that can exert toxic effects and to which people may be exposed. Inorganic arsenic can have adverse effects; arsenite (As^{+3}) and arsenate (As^{+5}) are the most prevalent toxic forms found in drinking water. Volatile compounds, such as arsine (AsH_3) are very toxic; however arsine is not found in drinking water or food. Moreover, the primary organic forms (arsenobetaine and arsenocholine) found in fish and shellfish seem to have little or no toxicity.

Previous studies indicated that the organic arsenicals MMA and DMA that are the methylation metabolites of the inorganic arsenic were hundreds of times less likely to produce genetic changes in animal cells than inorganic arsenic. Organic arsenicals are less reactive in tissues, kill less cells and are more easily excreted in urine. But recent observations by researchers have challenged the idea that methylation transformation of inorganic arsenic is a detoxification mechanism (Petrick et al., 2000). These authors found that the organic arsenic - Monomethylarsonous acid (MMA^{III}) is more toxic than arsenite in Chang Human Hepatocytes. The mean LC_{50} based on LDH (leakage of lactate dehydrogenase) assay in phosphate media was 6µM for MMA^{III} and 68µM for arsenite. Using the assay for K^+ leakage in phosphate media, the mean LC_{50} was 6.3 µM for MMA^{III} and 19.8 µM for arsenite. The mean LC_{50} based on the XTT [the tetrazolium salt 2,3-bis(2-methoxy-4-nitro-5-sulfophenyl)-2H-tetrazolium-5-carboxanilide inner salt] assay in phosphate media was 13.6µM for MMA^{III} and 164 µM for arsenite. The results of these three cytotoxicity assays reveal the following order of toxicity in Chang Human Hepatocytes:



These data demonstrate that MMA^{III} , an intermediate in inorganic arsenic methylation, is highly toxic and again raise the question as to whether methylation of inorganic arsenic is a detoxication process. MMA^{III} (Monomethylarsonous acid) has been detected for the first time in the urine of some humans exposed to inorganic arsenic in their drinking water (Aposhian et al., 2000). A speciation technique for eight arsenic compounds (special emphasis on identification and quantification of DMA^{III} and MMA^{III}) present in biological samples was developed (Le and Ma, 1997). The observations (Styblo et al., 1998; Petrick et al., 2000) that organic arsenic MMA^{III} is more toxic than inorganic arsenite indicate that (1) future studies concerning urinary arsenic profiles of arsenic-exposed humans must determine MMA^{III} concentrations, (2) previous studies of urinary profiles dealing with humans exposed to arsenic need to be re-examined and re-evaluated, and (3) since MMA^{III} is more toxic than inorganic arsenite, a re-examination is needed of the two hypotheses which hold that methylation is a detoxication process for inorganic arsenic and that inorganic arsenite is the major cause of the toxicity and carcinogenicity of inorganic arsenic.

Mass et al. (2001) reported that DMA^{III} is 386 fold more potent than As^{III} and summarised the genotoxicity potencies of methylated arsenicals *in vitro* (single cell gel assay) as follows:



The authors implied that trivalent methylated arsenic may be the toxic intermediate metabolites. However, the role of these methylated arsenicals in arsenic carcinogenicity *in vivo* remains unknown. The processes of metabolism of arsenicals *in vivo* are very complicated. There are several organs and systems involved in these processes.

Dimethylarsinic acid (DMA^V) - oxidative damage

Yamanaka et al. (1990) first presented a detailed metabolic pathway for arsenic in oxidative stress, as shown in Figure 1; dimethylarsine (a trivalent arsenic form) is a minor *in vivo* metabolite of DMA^V produced by a process of reduction *in vivo*. Dimethylarsine can react with molecular oxygen forming a (CH₃)₂As• radical and superoxide anion. This (CH₃)₂As• radical can add another molecule of molecular oxygen and form the (CH₃)₂AsOO• radical. A hydroxyl radical may be produced via cellular iron and other transition metals. Exposure to these free radicals can cause DNA damage such as DNA single-strand breaks (See Figure 1).

Matsui et al. (1999) reported the association of 8-OHdG with arsenic exposure in humans. They detected the role of oxidative DNA damage in human arsenic carcinogenesis by studying cases of arsenic-related skin neoplasms and arsenic keratosis as well as in 11 cases of arsenic unrelated Bowen's disease. The levels of 8-Hydroxy-2'-deoxyguanosine were significantly higher in arsenic-related skin neoplasms than in arsenic-unrelated Bowen's disease. 8-Hydroxy-2'-deoxyguanosine was also detected in normal tissue adjacent to arsenic-related Bowen's diseases lesions. Yamanaka et al. (2001) reported the urinary 8-OHdG levels (obtained by sampling from the urinary bladder) in mice gavaged with 720 mg/kg of DMA were increased to about 3, 3, and 8 times control levels at 3, 6, and 9h after treatment, respectively (Yamanaka et al., 2001). In a long-term rat carcinogenesis study, hepatic 8-OHdG levels were increased in DMA-treated rats, suggesting an elevated rate of free radical attack on DNA (Wanibuchi et al., 1997).

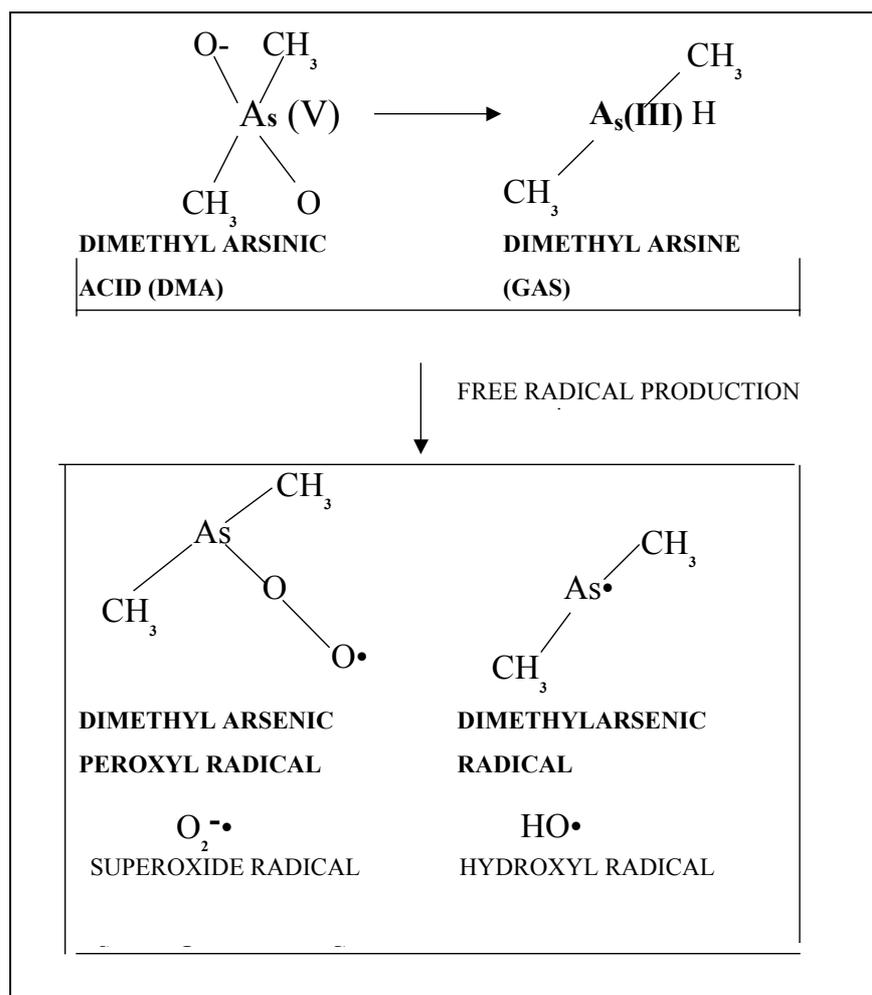


Figure 1. Free radicals from DMA^V. DMA^V may be reduced to a trivalent arsenic form, dimethylarsine. This minor pathway of arsenic metabolism may generate free radicals such as dimethylarsenic radical, dimethylarsenic peroxy radical, or indirectly generate superoxide and hydroxy radical (from Yamanaka et al., 1990).

Arsenic carcinogenicity

Evidence of inorganic arsenic being carcinogenic to humans is sufficient based on strong epidemiological studies (IARC, 1987; IPCS, 2001). However, the evidence to demonstrate arsenic is a complete carcinogen to animals is lacking until recently Ng (1998,1999) successfully demonstrated for the first time the formation of tumours in laboratory mice chronically exposed for over two years to inorganic arsenic in drinking water. This animal model is the most successful model for arsenic carcinogenesis effect and provides the possibility for further *in-vivo* study of arsenic carcinogenicity.

Cancer promoter

Much of the previous evidence has suggested that arsenic is a cancer promoter rather than a carcinogen in animal studies.

The urinary bladder has long been described as a major organ of chemical carcinogenesis, mostly because the excreted proximate metabolites of ingested chemicals being kept in the urinary bladder for relatively long duration with continuous exposure to the urothelium. DMA is the major excreted metabolite of ingested arsenic in numerous reports about human exposure to inorganic arsenic. Most of the absorbed DMA is excreted as unchanged DMA in the urine of various animal species. DMA itself acts as a urinary bladder carcinogen or particularly as a potent cancer promoter.

The indication that rat bladder was responsive to DMA-induced carcinogenesis first reported by Yamamoto et al. (1995; 1997). After a long time and many negative reports of arsenic carcinogenicity *in vivo* studies, their results are very significant, and showed that bladder cancer resulted from all four DMA-exposed F344/DuCrj rat groups after pre-exposed to cancer initiators. Multiple initiators were used through the carcinogenesis processes in several organs. In the initiated groups, DMA significantly enhanced the tumour induction in the urinary bladder, kidney, liver, and thyroid gland. Induction of preneoplastic lesions (glutathione S-transferase placental form-positive foci in the liver and atypical tubules in the kidneys) was also significantly increased in DMA-treated groups (see Table 2).

Urinary bladder tumours are so far the most frequently induced tumour type in the animal carcinogenesis studies. Following the work of Yamamoto et al (1995), at least 3 other research groups achieved the same result. For example, Wanibuchi et al. (1996) obtained the positive result of rat bladder tumours when only one initiator, N-butyl-N-(4-hydroxybutyl)nitrosamine was given followed by DMA exposure in 6-week-old male F344 rats. The development of preneoplastic lesions and tumours (papillomas and carcinomas) in the urinary bladder was enhanced by the treatment with DMA in a dose-dependent manner. A significant increase in multiplicity of

tumours (papillomas and carcinomas) was observed even at a relatively low concentration of DMA (10ppm). On the other hand, no preneoplastic lesions and tumours were observed in the rats treated with DMA alone. Wanibuchi and Fukushima (2000) also reported other results in relation to the structure-activity. In this work, rats were given control, arsenite (17.3ppm), MMA (187 ppm), DMA (184ppm), or TMAO (182ppm) respectively in their drinking water after initiated with N-butyl-N-(4-hydroxybutyl)nitrosamine developed 0.4, 0.3, 1.3, 3.8, and 0.9 bladder tumours per rat. The rat bladder tumour incidence was elevated at the $p < 0.05$ level for all three methylated arsenicals, with DMA clearly the most active chemical (See Table 2 and Table 3).

Recent studies have shown that tumours in various organs observed in humans in arsenic-endemic area can all be induced by arsenicals in animals as a cancer promoter. The kidney is the major route for the elimination of ingested arsenic from the human or animal body. Most of the metabolites are methylated, such as DMA, MMA and TMA. Most reports of the metabolism of ingested arsenic show accumulation of DMA in the kidney and liver. Studies also reported relatively high accumulation of ingested DMA in the lung and skin. Besides the results reported by Yamamoto et al (1995) and Wanibuchi et al (1996; 2000), several more studies report tumours involving many other organs. Yamanaka et al. (2001) used hairless mice receiving a topical application of dimethylbenz(a)anthracene (DMBA) with or without the exposure to ultraviolet B (UVB), a tumour promoter and the administration of DMA by the oral route promoted not only the formation of papillomas induced by DMBA alone but also the formation of malignant tumours induced by way of the formation of atypical keratoses by treatment with DMBA and UVB. In their previous report (Yamanaka et al., 2000), hairless mice (Hos: HR-1) were treated for 25 weeks with UVB light at 2KJ/m^2 twice per week with or without exposure to 400 or 1000 ppm of DMA in the drinking water. All three treated groups developed skin cancer, but the combination of UVB light and 1000 ppm of DMA in the drinking water induced more tumours per mouse during weeks 13 to 19 than did UVB light exposure alone. No statistically significant elevations in the percentage of tumour-bearing mice or tumours per mice occurred with the combination of UVB and 400 ppm DMA treatment versus UVB treatment alone.

In another study (Yamanaka et al., 1996), male ddY mice initiated with 4-nitroquinoline 1-oxide (4NQO) known as an initiator of lung tumours and subsequently exposed to 200 or 400 ppm DMA for 25 weeks in their drinking water developed lung tumours that were described as adenocarcinomas and adenosquamous carcinomas. The percentage of mice bearing tumour nodules was increased with the administration of both 4NQO and DMA than those mice given 4NQO alone, or given both 4NQO and glycerol. Furthermore, the average number of tumours nodules per mouse was significantly higher in mice administered DMA than those given glycerol (known to have potent promoting activity in lung tumourigenesis initiated by administration of 4NQO in ddY mice).

Wanibuchi et al. (1997) reported that in rats initiated with diethylnitrosamine DMA (25, 50, or 100 ppm) was active as a promoter in this experimental system for liver carcinogenesis based on the area stained for glutathione S-transferase placental form-positive foci. Rat hepatic ODC (Ornithine decarboxylase; it's activity is often interpreted as a biomarker for cell proliferation) activity was increased after administration of both 10 and 50ppm DMA in the drinking water. In a recent structure-activity study (Wanibuchi and Fukushima, 2000), rats initiated with diethylnitrosamine developed about 1.5, 2.5, 2.4, and 2.5 glutathione S-transferase placental form-positive foci/cm² following exposure to promotional regimens of control, MMA, DMA, or TMAO, respectively. The number of the liver foci per area was statistically elevated at the $p < 0.05$ for all three methylated arsenicals (Table 2).

Transgenic animals have been used for developing arsenic carcinogenesis model. Since the mechanisms of arsenic carcinogenesis are not clear, different transgenic animals have been tested in developing the manifestation of the arsenic carcinogenic effects. Genetically altered rodents were used in anticipation for the rapid identification of carcinogens. The use of these models holds great promise for shortening the length of the bioassay and reducing its cost. The enhancement of the sensitivity of the models could allow agents to be tested at lower doses more relevant to human exposure.

The p53 gene is a tumour suppression gene, it is a very important gene in preventing tumour occurrence. Salim et al. (1999) reported the use of p53^{+/-} mice, to which DMA was given to p53 heterozygous knockout and C57BL/6T wild-type male mice

in their drinking water for 80 weeks. Exposure to 50 or 200 ppm of DMA caused an increase in the total numbers of spontaneous tumours in wild-type mice and earlier induction of tumours in the p53 knockout mice.

Ornithine decarboxylase (ODC), an important enzyme in the polyamine biosynthetic pathway, is aberrantly regulated in epithelial tumours of rodents and humans. In murine skin, it has been shown that ODC overexpression provides a sufficient condition for tumour promotion. K6/ODC transgenic mice in which ODC overexpression was targeted to hair follicle keratinocytes might provide a sensitive model for identifying genotoxic carcinogens (Chen et al., 2000). A two-stage mouse skin carcinogenesis experiment in K6/ODC mice was performed by using the initiator 7,12-dimethylbenz[a]anthracene (DMBA, 50 µg, single application), 12-O-tetradecanoylphorbol-13-acetate (TPA, 5 µg twice per week) and DMA (3.6 mg twice per week). Twenty weeks after initiation, the average number of tumours per mouse was 20.7, 19.4, 9.7, and 0.0 in the four treatment groups of DMBA → TPA, DMBA → DMA, DMBA → control cream, and acetone vehicle → DMA, respectively. The DMBA → TPA treatment group responded with an increased tumour incidence about 2 weeks faster than the DMBA → DMA group. Microscopically, most tumours in the three responding treatment groups were squamous papillomas, and some squamous carcinomas with disorderly arrangement and atypical nuclei (Morikawa et al., 2000).

Table 2. Organic Arsenicals in Various Animal Carcinogenicity Models

| Arsenical Type | Animal Species & Strains | Dose & Treatment/Regime | Exposure Duration | Exposure Pathway | Results | Role of As | Reference |
|----------------|--------------------------------------|--|-------------------|--------------------------------------|---|------------|------------------------|
| DMA | Hos:HR-1 female hairless mice | 400 & 1000ppm DMA+UVB | 25 weeks | Drinking water | Skin Cancer | Promotor | Yamanaka et al., 2000 |
| DMA | Hos:HR-1 female hairless mice | DMA+ DMBA±UVB | 25 weeks | topical DMA drinking water | Skin tumour | Promotor | Yamanaka et al., 2001 |
| DMA | F344/DuCrj male rats | 50,100,200,400ppm +DEN,MNU,DMH BBN,DHPN | 30 weeks | Drinking Water & i.p(Initiator) | Bladder,Kidney Liver &Thyroid Tumour | Promotor | Yamamoto et al., 1995 |
| DMA | F344 male rats | DMA(0, 2, 10, 25, 50 and 100 ppm)±BBN | 32 weeks | Dinking water | Bladder tumour | Promotor | Wanibuchi et al., 1996 |
| DMA, MMA,TMAO | F344 male rats | MMA(187ppm) DMA(184) TMAO (182ppm)+BBN | | Drinking water | Bladder tumour | Promotor | Wanibuchi et al.,2000 |
| DMA | ddY male mice | 200 or 400 ppm DMA+4NQO | 25 weeks | Drinking water | Lung tumour | Promotor | Yamanaka et al., 1996 |
| DMA | F344 male rats | 25, 50, or 100ppm DMA +DEN | 8 weeks | Food and Drinking water . Den by i.p | Positive in Ito test | Promotor | Wanibuchi et al., 1997 |
| MMA, TMAO | F344male rats | MMA,TMAO + DEN | | | Positive in Ito test | Promotor | Wanibuchi et al., 2000 |
| DMA | P53 ⁺ &C57BL/6T male mice | 50 or 200 ppm | 80 weeks | Drinking water | Tumour increase & early induction of tumour | Promotor | Salim et al., 1999 |
| DMA | K6/ODC male mice | DMA 3.6 mg twice/week +DMBA or TPA | 20 weeks | Topical application | Skin tumour accelerated | Promotor | Morikawa et al., 2000 |
| DMA | A/J male mice | 50, 200 or 400 ppm | 25-50 weeks | Drinking water | Lung tumours | Carcinogen | Hayashi et al.,1998 |
| DMA | F344/DuCrj male rats | 0, 12.5, 50, 200 ppm (negative in 0 & 12.5ppm) | 104 weeks | Drinking water | Bladder tumour | Carcinogen | Wei et al.,1999 |
| DMA | F344 female rats | 100ppm | 24 months | Food | Bladder tumours | Carcinogen | Kitchin 2001 |
| DMA | B6C3F1 female mice | 500ppm | 24 months | Drinking water | Fibrosarcomas in multiple organs | Carcinogen | Kitchin 2001 |
| DMA | K6/ODC male mice | 10, 100ppm DMA | 5 months | Drinking water | Skin tumour | Carcinogen | Chen et al.,2000 |

DMA=Dimethylarsinic acid; MMA=Monomethylarsonic acide; TMAO=Trimethylarsine oxide; As(III)=Arsenite; As(V)= Arsenate; BBN= N-butyl-N-(4-hydroxybutyl)nitrosamine; DEN=diethylnitrosamine; MNU=N-methyl-N-nitrosourea; DMH=1,2-dimethylhydrazine; DHPN= N-bis(2-hydroxypropyl)nitrosamine; 4NQO=4-nitroquinoline-1-oxide; DMBA=7,12-dimethylbenz[a]anthracene; TPA=12 -O-tetradecanoylphorbol-13-acetate; ODC=ornithine decarboxylase; UVB= Ultraviolet B

Complete carcinogen

Hayashi et al. (1998) used A/J mice given 50, 200, or 400 ppm DMA in drinking water for either 25 or 50 weeks. At 25 weeks there were no DMA-related pulmonary tumours. Alveolar adenomas were not increased by DMA administration at 50 weeks. However, mice fed with the highest dosage (400 ppm) of DMAA for 50 weeks produced more pulmonary tumours than untreated mice (mean number per animal 1.36 versus 0.50; p<0.05). Increased numbers of lung hyperplasia, papillary adenomas, and adenocarcinomas were observed after treatment with 50, 200, or 400

ppm of DMA for 50 weeks. In this experiment, DMA is acting as a complete carcinogen in A/J mice which known to be susceptible to developing pulmonary tumours that are normally seen in humans exposed to arsenic.

Wei et al. (1998) also tested DMA as a complete carcinogen and found it to be carcinogenic to rat urinary bladder at 50 and 200 ppm. Total of 144 F344 rats (10 weeks old at the start) were used to determine the carcinogenicity of dimethylarsinic acid (DMA). DMA (purity 100%) concentrations of 200, 50, 12.5 and 0 ppm were given to 4 groups of rats in the drinking water respectively for 104 weeks. From weeks 97 to 104, urinary bladder tumours were observed in 12 of 31, 8 of 31 and none of 33 in groups 1-3, respectively. No bladder tumours were observed in the negative control group 4.

An unpublished study work (cited by Kitchin, 2001) was performed by Life Sciences Research of Israel in support of the U.S. EPA pesticide registration for cacodylic acid (USEPA, 1994). In this study, female F344 rats exposed for 2 years to up to 100 ppm DMA in their food developed dose-related transitional cell bladder tumours with bladder hyperplasia observed in both sexes. Control female rats had 0/59 incidence of bladder papillomas and carcinomas, while female rats exposed to 100 ppm DMA had 10/58 incidence (17%, $p < 0.01$). Of the female rats that responded to DMA exposure with bladder tumours, six were classified as carcinomas and four were classified as papillomas. Female B6C3F1 mice exposed to up to 500ppm DMA in their drinking water showed only an increase in fibrosarcomas in multiple organs.

ODC (ornithine decarboxylase) activity is induced in epithelial tumours in both rodents and humans. ODC has been extensively studied in mouse skin and rat liver carcinogenesis systems and ODC induction is considered as a biomarker for cancer promotion. Chen et al. (2000) tested ten known carcinogens or noncarcinogens using the K6/ODC transgenic mouse model. In K6/ODC mice given either 10 ppm DMA, 100 ppm DMA, or 10 ppm arsenite in their drinking water for 5 months, 8, 22, or 15% of the mice developed skin squamous papillomas, respectively. No tumours were observed in the control group of K6/ODC transgenic mice not exposed to arsenic and in nontransgenic littermates given 100 ppm DMA.

Table 3. Inorganic arsenicals in various animal carcinogenicity models

| Arsenical Type | Animal Species& Strains | Dose & Treatment/ Regime | Exposure Duration | Exposure Pathway | Results | Role of As | Reference |
|--------------------------------|------------------------------------|---|--|--|---------------------------------------|--------------|----------------------------|
| As(III) | Rats | 250ppm | 24 months | Diet | Negative | - | Byron et al 1967 |
| As(V) | Rats | 400ppm | 24 months | Diet | Negative | - | Byron et al 1967 |
| As(III) | Rats | 416ppm | 29 months | Diet | Negative | - | Kroes et al 1974 |
| As(III)/ As(V) | Beagles | 5 - 125ppm | 24 months | Diet | Negative | - | Byron et al 1967 |
| As(V) | Cynomologus Monkeys | 0.1mg/kg/day 5 days/week | 15 years | Diet | Negative | - | Thorgeirsson et al 1994 |
| As(III) | Rats | 5 mg/L | lifetime | Drinking water | Negative | - | Schroeder et al 1968 |
| As(V) | Rats | 5 mg/L | lifetime | Drinking water | Negative | - | Kanisawa et al 1969 |
| As(III) | Inbred C ₃ H Mice | 10 mg/ L | 16 weeks | Drinking water | Lower tumour rate than controls | - | Schrauzer et al 1974 |
| As(III) | F344 rats | As(III)(17.3ppm) | | Drinking water | Bladder tumour | Promotor | Wanibuchi et al., 2000 |
| As(III) | Tg.AC female mice | 200 ppm As(III) + TPA | 22 weeks As ⁺³ pretreated 4 weeks | Drinking water | Skin tumour increased | Promotor | Germolec et al.,1998 |
| As ₂ O ₃ | mice | 1.2mgAs/kg (mother) 5 ug As/pup | | subcutaneous | Lung adenoma in off-spring mice | Carcinogen | Rudnay et al., 1981 |
| As ₂ O ₃ | Hamsters | 3mg As/kg Weekly | 15 weeks | Intratracheal instillation | Respiratory tract tumours | Carcinogen | Pershagen et al., 1984 |
| As(III) | Rats | 8 mg/capsule | | Surgical implantation in stomach | Stomach adenocarcinomas | Carcinogen | Katsnelson et al.,1986 |
| As(III) | Rats | 160 ppm As/L+ i.p 30mg/kg of DEN | 175 days | Drinking water | Renal tumours | Carcinogen | Shirachi et al.,1983 |
| As(III) | p53 ^{+/-} male mice | 50 ppm +Choline- Deficient Diet + expose to p-cresidine | 26 weeks | Diet | Increase bladder tumour | Cocarcinogen | Popovicova et al.,2000 |
| As(V) | C57BL/6J and MT- female mice | 500ug/L | up to 26 months | Drinking water | Multiple tumours | Carcinogen | NG et al., 1999 |
| As(III) | K6/ODC male mice | 10ppm As(III) | 5 months | Drinking water | Skin tumour | Carcinogen | Chen et al.,2000 |
| As(III) | Hairless Skh1 mice | 10mg/L As(III)+1.7KJ/m ² UVR 3 times weekly | 26 weeks | Drinking water | Skin tumour | Cocarcinogen | Rossmann., 2001 |

As(III)= Arsenite; As(V)= Arsenate; DEN=diethylnitrosamine; TPA=12-O-tetradecanoylphorbol-13-acetate; ODC=ornithine decarboxylase; UVB= Ultraviolet B; UVR=Ultraviolet radiation; MT=Metallothionien knock-out.

Mode of action for arsenic carcinogenicity

The mode of action for arsenic carcinogenicity (DMA included) is still unclear. Although earlier studies have failed to come up with a successful arsenic tumour model, the effort for the development of arsenic tumours of animal models has provided a lot of information on arsenic genotoxic effects. An *in-vitro* study has shown that arsenic induces chromosome aberrations and DNA-protein cross-links (IARC, 1980). Reactive oxygen species (ROS) produced by dimethylarsenic acid, one

of the major metabolites of inorganic arsenics, were suggested to be involved in the generation of DNA damage (Yamanaka et al., 1989). Yamanaka and Okada (1994) also assumed that of the two radicals, dimethylarsenic peroxy radical and the superoxide anion radical produced from the reaction between molecular oxygen and dimethylarsine, the dimethylarsine peroxy radical plays a dominant role(s) in causing DNA damage, and at least causing DNA single-strand breaks. Dimethylarsenic acid has the potential to promote rat liver carcinogenesis, possibly via a mechanism involving stimulation of cell proliferation and DNA damage caused by ROS (Wanibuchi et al., 1997). Sodium arsenite produced DNA-protein crosslinks and arsenic mediated DNA-protein interactions may play a major role in arsenic carcinogenesis (Dong and Luo, 1993). Arsenic was suggested to enhance the development of skin neoplasias via chronic stimulation of keratinocyte-derived growth factors and may be a rare example of a chemical carcinogen that acts as a co-promoter (Germolec et al., 1998).

Unlike some heavy metals such as cadmium and nickel that bind directly to the DNA, arsenic showed no direct interaction with DNA. The mechanism of the action of arsenic as a carcinogen is not well understood but the affecting process can be inferred from data that suggest arsenic causes large deletion (Hei et al., 1998) or methylation changes of DNA (Mass and Wang, 1997; Zhao et al., 1997) that could affect gene expression. The inhibition of the DNA repair function has also been suggested as one of the mechanisms of inorganic arsenic carcinogenicity (Hartwig, 1998; Tran et al., 2002). The oxidative DNA damage in skin neoplasms related to arsenic attack is suggested by the *in-vivo* evidence in arsenic-induced Bowen's disease and the cases of bladder tumours among Taiwanese who lived in areas of high arsenic-contaminated artisan well drinking water (Matsui et al., 1999).

In conclusion, sufficient evidence to establish that both inorganic arsenic and its methylated forms (DMA and MMA) are either complete carcinogens or cancer promoters or both although the mode of action for their carcinogenic effects require further research. The synergistic effects of DMA / MMA and others carcinogenic

Watervolatility of PCDD/Fs

compounds have been reported extensively. The potential synergistic effect between DMA and dioxins remains unknown.

Appendix II: **References**

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