MECHANISMS OF ENTRY OF LEAD-BEARING DUSTS INTO HOUSES IN PORT PIRIE.

Alex Kutlaca
BSc, MEnvSt

A dissertation submitted in fulfilment of the requirements
for the degree of Doctor of Philosophy in The Mawson
Graduate Centre for Environmental Studies
The University of Adelaide
April 1998

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HIGH
LEAD
CONTENT
PAINT
WHITE-LEAD
-Lasts!

To Punky and Penny.

After all, you started it all.

TABLE OF CONTENTS

Page Number

TITLE PAGE i

FRONTISPIECE ii

TABLE OF CONTENTS iv

LIST OF TABLES xv

LIST OF FIGURES xix

LIST OF PLATES xx

ABSTRACT xi

DECLARATION xii

ACKNOWLEDGMENTS xiii

CHAPTER ONE INTRODUCTION 1

1.1 PREAMBLE 1
1.2 THE CONTEXT OF RESEARCH 3
1.3 GENERAL RESEARCH DIRECTION 6
1.4 THESIS STRUCTURE 7
1.5 THE STUDY SITE 11
1.6 PORT PIRIE SMELTER DETAILS 16
1.7 LEAD AND HEALTH IN PORT PIRIE EARLIER THIS CENTURY 17
1.8 AIMS AND OBJECTIVES 17
1.9 GENERAL METHODOLOGY 19
1.9.1 VACANT HOUSE SUITE - SITE SELECTION AND HOUSE DETAILS 19
1.9.1.1 SITE SELECTION 19
1.9.1.2 GENERAL HOUSE DESCRIPTION 19
1.9.1.3 SOURCE OF HOUSES 22
1.9.1.4 POTENTIAL HOUSE DIFFERENCES 22
1.9.2 RESEARCH WORK TIMING IN VACANT HOUSE SUITE 22
1.9.3 SAMPLING MATERIALS AND METHODS 22

1.9.3.1 SOIL SAMPLING 23
1.9.3.2 SURFACE DUST SAMPLING 23
a) Vacuum pump 23
b) Dust-collecting device 23
1.9.3.3 DEPOSITING DUST SAMPLING 24
1.9.3.4 AIR SAMPLING 24
1.9.4 ANALYTICAL METHODOLOGY 24
1.9.4.1 DEPOSITING DUST 25
1.9.4.2 STATIC DUST 25
1.9.4.3 AIR-DERIVED SAMPLES 25
1.9.5 ADDITIONAL ELEMENT DATA 25
1.9.6 DATA INTERPRETATION 26
1.9.6.1 RAW DATA APPEARANCE 26
1.9.6.2 STATISTICAL COMPARISON OF DATA 27
1.9.7 CONTEMPORANEOUS SAMPLING - POST CLEAN-UP STUDY 27

CHAPTER TWO LEAD THROUGH HISTORY: INTEREST AND INTOXICATION - - AN HISTORICAL REVIEW 30

2.1 THE EMERGENCE OF LEAD 30
2.2 ORIGINS OF LEAD USAGE 31
2.3 LEAD'S INITIAL PRODUCTION INCREASE 33
2.4 HEALTH EFFECTS ASSOCIATED WITH LEAD INTOXICATION 34
2.4.1 GREEK AND ROMAN CULTURES 34
2.4.2 MORE RECENT CULTURES AND HISTORICAL LINKAGES 37
2.5 SKELETAL LEAD LEVELS 41
2.5.1 ANCIENT CIVILISATION SKELETAL LEVELS OF LEAD 41
2.5.2 EARLY TO PRESENT CIVILISATION SKELETAL LEAD LEVELS 42
2.5.2.1 EUROPEAN CIVILISATION 42
2.5.2.2 JAPANESE CIVILISATION 43
2.6 LEAD INTOXICATION IN MODERN TIMES: A BRIEF HISTORY 44
2.7 IN SUMMARY - LEAD TODAY AND IN THE FUTURE 48

CHAPTER THREE LEAD; THE ENVIRONMENT AND EXPOSURE PATHWAYS 50

3.1 LEAD: PHYSICO/CHEMICAL PROPERTIES 50
3.1.1 ELEMENTAL LEAD DETAILS 50
3.1.2 LEAD SALTS 51
3.1.2.1 SOLUBILITY UNDER LABORATORY CONDITIONS 51
3.1.2.2 LEAD'S SOLUBILITY IN VARIOUS WATER BODIES 52

3.1.2.3 HARVESTING FROM ORE 52
3.2 CURRENT PRODUCTION AND ENVIRONMENTAL EMISSIONS 53
3.2.1 PRODUCTION 53
3.2.1.1 END USE PATTERNS 54
3.2.1.2 LEAD IN GASOLENE 57
3.2.2 ENVIRONMENTAL DISCHARGE 57
3.2.2.1 NATURAL SOURCE LEAD EMISSIONS 57
a) Total 57
b) Atmospheric 58
3.2.2.2 ANTHROPOGENIC-SOURCED LEAD EMISSIONS 58
a) All sources 58
b) Gasolene lead 60
3.3 LEAD IN THE ENVIRONMENT 62
3.3.1 SOIL 62
3.3.1.1 CONTRIBUTORS TO SOIL LEAD 62
a) Natural sources 62
b) Anthropogenic influences 63
- i) Mineralised soil areas 63
- ii) Urban areas 64
- iii) Roadway-adjacent soil 65
- iv) Rural areas 66
- v) Smelter locations 66
3.3.1.2 SOIL LEAD, PATHWAYS TO HUMANS - ASSOCIATED HEALTH RISKS 66
3.3.2 WATER 68
3.3.2.1 PATHWAYS TO CONTAMINATION 69
3.3.2.2 GROUND WATER 69
3.3.2.3 SURFACE WATER 69
3.3.2.4 HUMAN CONTACT WITH LEAD VIA WATER 70
a) Reticulated water 70
b) Rain water storage tanks 71
3.3.3 AIR 71
3.3.3.1 AIR-SUSPENDED PARTICLE DYNAMICS 71
3.3.3.2 AIR LEAD LEVELS 72
a) Remote areas 72
b) Rural areas 73
c) Urban areas 73
3.3.3.3 LONG-TERM AIR MONITORING 75
a) All locations 75
b) Smelter-proximate locations 78
3.3.4 FOOD 79
3.3.4.1 DIETARY LEAD 79

3.3.4.2 SOURCES OF LEAD INTO FOOD 80
3.3.5 DUST 83
3.3.5.1 AN OVERVIEW OF THE ROLE OF DUST 83
3.3.5.2 LEADED DUST AS A PATHWAY TO HUMANS 84
3.3.5.3 SOIL AND HOUSE DUST 85
3.3.5.4 SOIL, STREET DUST AND HOUSE DUST 85
3.3.5.5 OTHER LEAD CONTRIBUTORS TO DUST 85
3.3.5.6 LEAD LEVELS - EXTERNAL AND INTERNAL DUST 87
a) External dust 87
- i) Remote and rural 88
- ii) Urban 88
- iii) Surrounding the home 88
- iv) Industrial regions 89
b) Indoor dust 89
3.3.5.7 URBAN VERSUS RURAL DUST LEAD SCENARIOS 91
3.3.5.8 HEAVILY-INDUSTRIALISED REGIONS 93
3.3.5.9 MINERALISED AREAS 94
3.3.5.10 DUST MOVEMENT DYNAMICS 94
a) House structural factors 94
b) House occupant influences 95
c) General comments 95
3.3.5.11 DEPOSITING LEAD AND LEAD LOADING DATA 96
a) Depositing dust lead (leadfall) 96
b) Lead in dust loading 96
3.3.5.12 HOUSE DUST SOURCE CONTRIBUTORS 97
3.4 ENVIRONMENTAL LEAD LEVELS 99
3.4.1 LEAD INTAKE BY INSPIRATION OF CONTAMINATED AIR 99
3.4.2 LEAD INTAKE BY FOOD 99
3.4.3 LEAD INTAKE BY CONSUMPTION OF SOIL AND DUST 100
3.4.3.1 SOIL 100
3.4.3.2 DUST 101
3.4.4 MAJOR ENVIRONMENTAL COMPARTMENT CONTRIBUTIONS TO BLOOD LEAD 102
3.4.4.1 ENVIRONMENTAL LEVELS: EXPOSURE/EFFECT INDICATORS 102
3.4.4.2 HUMAN EXPOSURE POTENTIAL 103
3.4.5 PATHWAYS TO INTOXICATION 103
3.5 INDICES OF A HISTORY OF LEAD USAGE 104

CHAPTER FOUR LEAD & ITS RELATIONSHIP WITH HUMAN HEALTH - GREY METAL OR GREY MATTER, THE CHOICE MAY NOT BE OURS 107

4.1 MEDICAL ASPECTS OF LEAD'S TOXICOLOGY - GENERAL 107
4.1.1 PREAMBLE 107
4.1.2 INTRODUCTION 108
4.2 LEAD'S INTERACTION WITH THE HUMAN BODY - THE DETAILED 110
TOXICOLOGY OF LEAD 110
4.2.1 THE ABSORPTION OF LEAD 110
4.2.1.1 GASTRO-INTESTINAL TRACT (GIT) ABSORPTION 111
a) Normal physiological absorption variability 112
b) Chemical species absorption variability 112
c) The influence of food on absorption 113
d) Age-related absorption variability 114
e) Occupational influences on absorption 114
f) Dietary factors influencing absorption 114
- i) Calcium and Phosphorous 115
- ii) Other trace elements 116
- iii) Vitamin D 118
- iv) Other dietary factors 119
- v) Summary comments 119
4.2.1.2 RESPIRATORY TRACT ABSORPTION 120
4.2.2 LEAD'S DISTRIBUTION IN THE BODY 121
4.2.2.1 BINDING SITES, PATHWAYS AND BODY POOLS 121
4.2.2.2 SOFT TISSUE STORAGE 122
4.2.2.3 BONE STORAGE 123
4.2.2.4 OTHER LEAD STORAGE SITES 124
4.2.3 ABSORBED LEAD AND ITS EXCRETION OR RETENTION 125
4.2.3.1 EXCRETION MECHANISMS 125
4.2.3.2 RETENTION SEQUELAE 125
4.2.4 HUMAN HEALTH EFFECTS OF LEAD - OVERVIEW 126
4.2.4.1 INTRACELLULAR AND CYTOLOGICAL EFFECTS 126
4.2.4.2 BODY SYSTEM EVIDENCE OF LEAD'S TOXIC EFFECTS 127
4.2.5 HUMAN HEALTH EFFECTS OF LEAD - SYSTEMS AND ORGANS 129
4.2.5.1 INFLUENCE ON HAEMOPOIESIS 129
a) ALA-S effects 132
b) ALA-D effects 132
c) Ferrochelatase effects 132
4.2.5.2 LEAD'S EFFECT UPON THE KIDNEY 134
4.2.5.3 EFFECT OF LEAD UPON REPRODUCTION 134
a) Lead and maternal effects 135

b) Lead and paternal effects 135
c) In utero lead contamination 135
4.2.5.4 LEAD'S TOXICITY TO THE DEVELOPING NERVOUS SYSTEM 137
a) Central nervous system effects of lead 138
b) Peripheral nervous system effects of lead 139
c) Neurobehavioural/neuropsychological effects of lead 140
d) Developmental CNS toxicity of lead 143
4.2.5.5 CARDIOVASCULAR TOXICITY OF LEAD 145
4.2.5.6 GASTROINTESTINAL TOXICITY 146
4.2.5.7 IMMUNE SYSTEM EFFECTS 146
4.2.5.8 GENOTOXIC EFFECTS 147
4.2.5.9 CARCINOGENIC EFFECT OF LEAD 147
4.2.5.10 TERATOGENIC/GAMETOTOXIC EFFECTS OF LEAD 148
4.2.5.11 SUMMARY 148

CHAPTER FIVE PRELIMINARY INVESTIGATIONS 150

5.1 INTRODUCTION 150
5.2 DUST LEAD CONTENT - ESTABLISHING THE METHODOLOGY 151
5.3 INITIAL HOME VACUUM CLEANER BAG GRAB SAMPLES 151
5.3.1 SAMPLES ACQUIRED FROM PORT PIRIE RESIDENCES 152
5.3.2 OUT-OF-TOWN SAMPLES 154
5.4 BULK DUST PARTICLE SIZE RANGE AND LEAD CONTENT 155
5.4.1 DUST PARTICLE SIZE RANGE - SIEVING TECHNIQUE 155
5.4.2 DUST PARTICLE SIZE RANGE LEAD CONTENT 159
5.5 SAMPLED DUST PARTICLE SIZE DETERMINATION - LASER TECHNIQUE 163
5.5.1 METHODOLOGY 164
5.5.2 RESULTS 164
5.5.2.1 VACUUM PUMP/VACUUM CLEANER DATA COMPARISON 164
a) Vacuum pump and filter data (P Suite data) 165
b) Vacuum cleaner data (D Suite data) 166
c) P and D suite comparisons 166
d) Particle size relationship with lead content of dust 171
5.5.2.2 District differences 172
5.5.2.3 Influence of distance from smelter on particle size 172
5.5.2.4 House aspect - facing smelter versus facing away 173

CHAPTER SIX VACANT HOUSE SOIL ASSESSMENT 174

6.1 INTRODUCTION 174

6.2 OBJECTIVES 175
6.3 INITIAL OBSERVATIONS AND SITE DESCRIPTION 175
6.4 SAMPLING PROTOCOL AND LOCATION 176
6.5 RESULTS 177
6.5.1 SOIL LEAD AND ZINC LEVELS 177
6.5.2 SOIL CONTAMINANT LEVELS: HOME BY HOME COMPARISONS 179
6.5.3 SOIL CONTAMINANT LEVELS: BY DISTANCE FROM SMELTER 181
6.5.4 SOIL CONTAMINANT HOME CONCENTRATION GRADIENTS 183
6.5.5 SOIL CONTAMINANT LEVEL BY HOME AREA 185
6.5.6 HOUSE ASPECT: FACING SMELTER VERSUS FACING AWAY 189
6.5.7 VACANT HOME SOIL LEAD/ZINC RELATIONSHIP 191
6.5.7.1 CORRELATION BETWEEN SOIL LEAD AND ZINC VALUES 191
6.5.7.2 SOIL LEAD AND ZINC RATIOS 193
6.5.8 CONCLUDING REMARKS 194

CHAPTER SEVEN WITHIN-HOUSE DEPOSITED DUST 196

7.1 INTRODUCTION 196
7.2 RESEARCH OBJECTIVES 197
7.3 METHODOLOGY 198
7.3.1 SAMPLE ACQUISITION 198
7.3.2 SAMPLE ANALYSIS 199
7.3.3 SAMPLING DETAILS 199
7.4 STATIC DUST - SUMMARY RESULTS 200
7.5 PRELIMINARY OBSERVATIONS 201
7.6 CONCERNS REGARDING OPERATOR DIFFERENCES 202
7.7 DISTRICT DIFFERENCES 203
7.8 MAIN ROOM DIFFERENCES 205
7.9 ROOMS FACING VERSUS FACING AWAY 206
7.10 HOUSE BY HOUSE COMPARISON 208
7.11 HOUSE CONTAMINATION WITH DISTANCE FROM SMELTER 210
7.12 CONTAMINATION DUE TO FIREPLACE PRESENCE 211
7.13 FLOOR SURFACE DIFFERENCES 213
7.14 CARPET TYPE AND CONDITION 216
7.14.1 CARPET PILE DIFFERENCES 216
7.14.2 CARPET WEAR LEVELS 217
7.15 BETWEEN ROOM DIFFERENCES 218
7.16 WITHIN-ROOM DIFFERENCES - LOCATION IN (MAIN) ROOMS 220
7.16.1 FIREPLACES 224
7.16.2 WINDOWS 225
7.16.3 DOORWAYS 227

7.16.4 FIREPLACE/WINDOW/DOOR COMPARISON 229
7.17 HALLWAYS AND HOUSE MAIN (FRONT) ENTRANCES 229
7.17.1 HALLWAY MAIN ENTRANCES 229
7.17.2 HALLWAY CONTAMINANT GRADIENTS 230
7.18 WINDOW SILL AND WELL 233
7.19 ROOMS FACING WALKWAY VERSUS OPPOSITE 234
7.20 SUMMARY OF CONTAMINANT DISTRIBUTION WITHIN ROOMS 234
7.21 POST-CLEANUP STATIC DUST ASSESSMENT 236

CHAPTER EIGHT WITHIN-HOUSE DEPOSITING DUST 239

8.1 INTRODUCTION 239
8.2 RESEARCH OBJECTIVES 239
8.3 METHODOLOGY 240
8.3.1 DUSTFALL-COLLECTING TECHNIQUE 242
8.3.2 ADDITIONAL DUSTFALL ASSESSMENT 242
8.3.3 SAMPLING STRATEGY 242
8.4 SAMPLE ANALYSIS 243
8.5 DUST DEPOSITION RESULTS 244
8.5.1 INTERNAL DUSTFALL COLLECTION - SUMMARY DATA 244
8.5.2 DUSTFALL COLLECTION, ROUND BY ROUND 246
8.5.3 DUST DEPOSITION: DISTRICT COMPARISON 250
8.5.3.1 ALL ROUNDS COMBINED 250
8.5.3.2 DISTRICT DIFFERENCES ACCORDING TO HOUSE STATUS 252
8.5.3.3 DISTRICT DIFFERENCES ROUND BY ROUND 258
8.5.3.4 LEADFALL/DUSTFALL DISTRICT RELATIONSHIP - INDIVIDUAL COLLECTION ROUNDS 260
8.5.4 DUSTFALL HOME BY HOME 261
8.5.5 SAMPLING LOCATION DISTANCE FROM SMELTER 264
8.5.6 DUST DEPOSITION DIFFERENCES BY ROOM 265
8.5.6.1 ROOM TYPE 266
8.5.6.2 MAIN ROOM DIFFERENCES 268
8.5.7 PRESENCE OF A FIREPLACE IN A MAIN ROOM 272
8.5.8 WINDOW VERSUS CENTRE OF ROOM LOCATION 273
8.5.9 DEPOSITING DUST GRADIENTS FROM A WINDOW INTO A ROOM 276
8.5.10 ROOMS ADJACENT TO WALKWAY VERSUS ROOMS OPPOSITE 278
8.5.11 DEPOSITING DUST CHARACTERISTICS - AWAY VERSUS FACING 279
8.5.11.1 GENERAL INVESTIGATION 279
8.5.11.2 DISTRICT DIFFERENCES FACING VERSUS FACING-AWAY 281
a) Port Pirie West 281
b) Solomontown 281

8.5.11.3 SUMMARY COMMENTS 282
8.6 POST-CLEANUP DUST DEPOSITION 283
8.6.1 INITIAL DATA INTERPRETATION 283
8.6.2 FURTHER POST-CLEANUP DATA INTERPRETATION 286
8.7 DUSTFALL COLLECTION IN EXTERNAL STRUCTURES 287

CHAPTER NINE LEAD-IN-AIR STUDY 289

9.1 INTRODUCTION 289
9.2 DUST ENTRY INTO BUILDINGS 290
9.3 INDOOR/OUTDOOR AIR LEAD CONCENTRATION RATIOS 291
9.4 RESEARCH OBJECTIVES 294
9.5 SAMPLING STRATEGY 294
9.6 METHODOLOGY 295
9.6.1 SAMPLING EQUIPMENT 295
9.6.2 OPERATING CONDITIONS 296
9.6.3 SAMPLING PROTOCOL 297
9.6.4 SAMPLE CHEMICAL ANALYSIS 298
9.7 RESULTS 298
9.7.1 PRELIMINARY EXPERIMENTATION 298
9.7.2 MAIN RESEARCH COMPONENT 298
9.7.3 TEMPERATURE TO AIR LEVEL COMPARISON 299
9.7.4 OCCUPIED VERSUS VACANT HOUSES 300
9.7.5 HOUSE ASPECT: FACING THE SMELTER OR FACING AWAY 301
9.7.6 POST-CLEANUP AIR STUDY 302
9.7.7 RELATIONSHIP BETWEEN LEAD-IN-AIR AND LEADFALL LEVELS 303
9.7.7.1 PREAMBLE 303
9.7.7.2 EXPERIMENTAL RESULTS 304

CHAPTER TEN GENERAL DISCUSSION 306

10.1 INTRODUCTION 306
10.2 PRELIMINARY RESEARCH 308
10.3 DUST SAMPLING TECHNIQUES 309
10.3.1 DEPOSITED DUST 309
10.3.2 DEPOSITING DUST 310
10.3.3 CONTAMINATION INDICATORS: STATIC DUST VERSUS DUSTFALL 310
10.4 CONTAMINANT DUST DISTRICT DIFFERENCES 310
10.5 CONTAMINANT LEVEL VARIATION WITH DISTANCE FROM SMELTER 312
10.6 HOME SPATIAL VARIABILITY OF LEAD CONTAMINATION 313

10.6.1 SOIL 313
10.6.2 STATIC DUST 314
10.6.2.1 MECHANISMS OF DUST ENTRY 315
a) Windows 315
b) Doorways 316
- i) Main external doorways 316
- ii) Hallway to room doorways 316
c) Windows versus doorways as potential entry points 317
d) Fireplaces 319
10.6.3 DEPOSITING DUST 320
10.6.3.1 MECHANISMS OF DUST ENTRY 320
a) Windows 320
b) Main entrance doorways 321
c) Fireplaces 322

CHAPTER ELEVEN CONCLUSION AND RECOMMENDATIONS 331

11.1 CONCLUSION 331
11.2 RECOMMENDATIONS 334

APPENDICES 337

APPENDIX ONE
ANALYTICAL PROCEDURES 337

APPENDIX TWO
PRELIMINARY DUST DIGEST METHOD ESTABLISHMENT 339

APPENDIX THREE
QUALITY CONTROL PROTOCOL AND RESULTS 343

APPENDIX FOUR
ROBERT KEHOE, LEAD & HEALTH RESEARCH IN THE USA 349

APPENDIX FIVE
HOME ACTIVITIES DISTRIBUTING LEAD AROUND THE HOME 350

APPENDIX SIX
STANDARDISATION OF SAMPLING AND ANALYSIS TECHNIQUES 351

APPENDIX SEVEN
AAS OPERATING CONDITIONS AND QUALITY CONTROL 352

APPENDIX EIGHT
BULK DUST SIEVING DETAILS 356

APPENDIX NINE
VACUUM PUMP DUST-COLLECTION SYSTEM 357

APPENDIX TEN
VACUUM PUMP DUST COLLECTION METHOD PROTOCOL 361

APPENDIX ELEVEN
PARTICLE-SIZING DUST PROCESSING TECHNIQUES 362

APPENDIX TWELVE
PYKNOMETRY DETAILS 364

APPENDIX THIRTEEN
INITIAL TESTING OF XRF CAPABILITIES 365

APPENDIX FOURTEEN
HANDPIECE AND TUBING DUST RETENTION 369

APPENDIX FIFTEEN
STATIC DUST COLLECTION VALIDATION STUDY 374

APPENDIX SIXTEEN
POTENTIAL DUSTFALL COLLECTION APPROACHES 377

APPENDIX SEVENTEEN
DEPOSITING DUST COLLECTION PROTOCOL 384

APPENDIX EIGHTEEN
COLLECTED DUST ANALYSIS METHODOLOGY 388

APPENDIX NINETEEN
VACUUM PUMP, ROTAMETER, AND GAS METERS 391

CHAPTER TWELVE BIBLIOGRAPHY 395

LIST OF TABLES

Page Number

Table 2.1	Concentration of elemental lead in excavated Japanese bones.	44
Table 2.2	Numbers of children potentially exposed to sources of lead in the USA, 1984.	49
Table 3.1	Solubility of various salts of lead.	51
Table 3.2	Estimate of recent world lead production.	53
Table 3.3	Lead production in the ancient world.	54
Table 3.4	Total Global Emissions of trace metals into the environment.	59
Table 3.5	Worldwide anthropogenic atmospheric lead emissions (1993) by source.	61
Table 3.6	Urban, rural and remote lead-in-air levels from various studies.	74
Table 3.7	Range of contamination for external dust from various studies.	87
Table 3.8	Indoor dust lead levels from various studies.	90
Table 3.9	Lead in house dust data derived from lead-smelter districts.	93
Table 3.10	A review of lead loading data from the literature.	97
Table 3.11	Suggested factors affecting house dust lead concentration.	98
Table 3.12	Estimated mean blood lead increase resulting from a dust lead increase.	101
Table 3.13	Environmental media contributions to blood lead levels for US children.	102
Table 4.1	Recognised body system medical conditions resulting from acute lead exposure.	128
Table 5.1	Preliminary indication of lead levels of house dust from Port Pirie.	153
Table 5.2	Preliminary indication of lead levels of house dust from outside Port Pirie.	154
Table 5.3	Dust sieving results for occupied home floor dust samples.	156
Table 5.4	Dust sieving results for vacant home floor dust samples.	156
Table 5.5	Summary of dust sieving particle size results for above-ceiling dusts.	157
Table 5.6	Summary of dust sieving particle size results for all dust samples.	158
Table 5.7	Occupied house sieved floor dust particle size versus lead concentration.	159
Table 5.8	Vacant house sieved floor dust particle size versus lead concentration.	160
Table 5.9	Vacant house sieved ceiling dust particle size versus lead concentration.	161
Table 5.10	Summary of sieved dust class mean lead concentration results.	162
Table 5.11	Variability of repeat estimate of dust particle size.	165
Table 5.12	Comparison of mean (class) particle size for P Suite and D Suite dust.	167
Table 5.13	Summary of dust particle sizing results.	169
Table 5.14	Dust-collecting method variability comparison.	170
Table 5.15	Further dust-collecting method variability comparison.	170
Table 5.16	Regression of particle size against dust lead concentration and loading.	171
Table 5.17	Mean distances between sampling sites and smelter components.	172
Table 5.18	Particle size variation with square of distance from smelter.	173

Table 6.1 Soil lead statistics, total and by district. 178
Table 6.2 Soil zinc statistics, total and by district. 178
Table 6.3 Soil contaminant concentrations home by home. 179
Table 6.4 Correlation matrix for soil elements. 181
Table 6.5 Soil contaminant variation with square of distance from smelter. 182
Table 6.6 Distance class mean soil lead concentration. 184
Table 6.7 Distance class mean soil zinc concentration. 184
Table 6.8 Mean soil lead and soil zinc statistics by home area. 186
Table 6.9 Home area soil lead concentration comparison. 186
Table 6.10 Home area soil zinc concentration comparison. 187
Table 6.11 Soil element level for home aspect facing smelter versus facing away. 189
Table 6.12 Soil element level for home aspect facing smelter versus facing away. 190
Table 6.13 Lead/zinc correlation coefficients for all data, and by district. 192
Table 6.14 Lead/zinc correlation coefficients by home area. 192
Table 6.15 Lead/zinc correlation coefficients by home location. 192
Table 6.16 Lead/zinc soil ratios for total data suite and by district. 193
Table 6.17 Lead/zinc soil ratios by home area. 193
Table 7.1 Mean dust loading, total and by district. 200
Table 7.2 Mean dust lead concentration, total and by district. 201
Table 7.3 Mean dust lead loading, total and by district. 201
Table 7.4 Operator dust collection comparison - all data. 202
Table 7.5 Operator dust collection comparison - adjacent floor areas. 203
Table 7.6 District differences in surface dust levels. 204
Table 7.7 House room contamination comparison. 205
Table 7.8 House/smelter aspect comparison. 206
Table 7.9 Further house/smelter aspect comparison. 208
Table 7.10 House by house contamination comparison. 209
Table 7.11 Contaminant level variation with square of distance from smelter. 211
Table 7.12 Influence of a fireplace upon floor contamination levels. 212
Table 7.13 Influence of floor surface on contaminant levels. 214
Table 7.14 Influence of floor surface on contaminant levels - simplified version. 215
Table 7.15 External floor (concrete) dust collection summary. 215
Table 7.16 Influence of carpet pile on degree of dust and lead contamination. 216
Table 7.17 Influence of carpet wear on degree of dust and lead contamination. 218
Table 7.18 Room to room differences in floor contaminant levels. 219
Table 7.19 Room to room differences in floor contaminant levels- simplified. 219
Table 7.20 Within-room contaminant spatial variability. 221
Table 7.21 Window-base floor and opposite-room floor sampling comparison. 225
Table 7.22 Contaminant level variation with square of distance in from windows. 226
Table 7.23 Doorway, behind door and opposite-room floor comparison. 227
Table 7.24 Hallway entrance dust levels. 230

Table 7.25 Contaminant level variation with square of distance into hallway. 230
Table 7.26 Contaminant distribution within rooms of houses - a summary. 235
Table 7.27 Influence of house cleanup on floor contaminant levels. 237
Table 7.28 Surface contamination build-up over post-cleanup test period. 237
Table 8.1 Dustfall collection rounds. 241
Table 8.2 Summary dust deposition data. 244
Table 8.3 Summary comparison of alfoil- and cup-collecting leadfall. 245
Table 8.4 Individual collection cycle summary depositing dust data. 247
Table 8.5 Inter-round dust deposition differences. 248
Table 8.6 Inter-round dust deposition differences. 250
Table 8.7 District differences in dust deposition. 251
Table 8.8 Grouped different house condition dust collection mean data. 252
Table 8.9 Depositing dust parameter ratios for all data and by district. 254
Table 8.10 Grouped semi-open and fully-open house status summary data. 256
Table 8.11 Depositing dust parameter ratios for two house status classifications. 257
Table 8.12 District differences in dust deposition round by round. 259
Table 8.13 District differences in leadfall/dustfall on a collection round basis. 260
Table 8.14 House by house comparison of depositing dust data. 262
Table 8.15 Regression of depositing dust parameters on an individual house basis. 263
Table 8.16 Depositing dust contamination level variation with distance from smelter. 265
Table 8.17 Mean room dust deposition parameter summary values. 266
Table 8.18 Main room comparisons - all sampling rounds combined. 268
Table 8.19 House closed main room dust deposition summary data. 271
Table 8.20 House open main room dust deposition summary data. 271
Table 8.21 Influence of an open fireplace in a room on depositing dust. 273
Table 8.22 Within room differences in depositing dust levels. 274
Table 8.23 Regression analysis of window versus centre depositing dust parameters. 275
Table 8.24 Variability in depositing dust variables according to location in room. 276
Table 8.25 House aspect depositing dust comparison - house closed. 279
Table 8.26 House aspect depositing dust comparison - house open. 280
Table 8.27 Pre- and post-cleanup summary depositing-dust data. 284
Table 8.28 Post-cleanup static dust collection data summary. 285
Table 8.29 Post-cleanup facing-smelter versus facing-away summary data comparison. 286
Table 8.30 External/internal depositing dust comparison. 287
Table 9.1 Indoor-Outdoor Relationships for Lead in Air. 293
Table 9.2 Air sampling results. 299
Table 9.3 Occupied versus vacant home air lead summary. 300
Table 9.4 Air sampling in test houses before and after house de-dusting. 302
Table 10.1 Summary data of Port Pirie and Arnhem studies. 306
Table 10.2 Different room (estimated) total room mean lead loading. 318
Table 10.3 Different room (estimated) total room mean leadfall. 321

Table 10.4	Correlation coefficients for a variety of test parameter comparisons	326
Table 10.5	Mean lead/zinc ratios for various sampled media.	328
Table 10.6	Mean home lead/zinc ratios for soil and static dust.	329
Table A2.1	Influence of acid digest mix used for assay on analytical result.	341
Table A2.2	Influence of test dust quantity used for analysis on analytical result.	342
Table A3.1	Standard reference dust samples for analysis.	345
Table A7.1	Multiple replicate sample analysis results.	353
Table A13.1	XRF Instrument and method detection limits.	365
Table A13.2	XRF sampling time accuracy.	366
Table A13.3	XRF Instrument surface soil penetration.	367
Table A13.4	XRF Instrument standard soil comparison.	368
Table A14.1	Handpiece dust and lead retention comparison.	369
Table A14.2	Summary handpiece dust retention.	370
Table A14.3	Various tubing type dust/lead retention.	371
Table A14.4	Summary of dust and lead retained regression relationships.	372
Table A15.1	Preliminary operator sampling efficiency comparison summary.	374
Table A15.2	Successive sampling sweep dust collection efficiency.	375
Table A15.3	Different floor surface sampling absolute dust recovery efficiency.	376

LIST OF FIGURES

Page Number

Frontispiece	High paint lead advertisement	ii
Figure 1.1	Study site location.	11
Figure 1.2	Mean distribution of wind direction at the smelter site, Port Pirie.	12
Figure 1.3	Port Pirie township, smelter, high-risk district and test house locations.	13
Figure 1.4	Soil lead levels in and around Port Pirie.	14
Figure 1.5	Locational map of Port Pirie including specific test house location.	21
Figure 2.1	World lead production over the last 6,000 years.	32
Figure 3.1	Lead demand by end-use category internationally.	55
Figure 3.2	Lead demand by end-use category in Australia.	56
Figure 3.3	Consumption of lead in the USA in 1987 by percentage category of use.	60
Figure 3.4	Bi-variate correlations between child blood lead and environmental measures.	68
Figure 3.5	US leaded gasolene consumption and relationship with airborne levels of lead.	76
Figure 3.6	Mean blood lead levels of US population, six months to 74 years.	77
Figure 3.7	Major sources of lead in the environment, and transfer pathways to humans.	104
Figure 3.8	Lead concentration profile in the snow strata of Northern Greenland.	105
Figure 4.1	Changes in US blood lead level considered as onset of adverse health effects.	109
Figure 4.2	The effects of lead upon haem biosynthesis	130
Figure 4.3	Multi-organ impact of lead's effect upon the haem pool.	131
Figure 5.1	Collected dust cumulative frequency particle size plot	168
Figure 6.1	Typical home soil sampling pattern.	176
Figure 7.1	Various room regions classification.	221
Figure A7.1	Typical AAS calibration curve.	354

LIST OF PLATES

Page Number

Plate One	Port Pirie test house: 58 Federation Road.	28
Plate Two	Port Pirie test house: 19 Lily Street.	28
Plate Three	Port Pirie test house: 27 Vera Street.	29
Plate Four	Port Pirie test house: 97 York Road.	29
Plate Five	Static dust vacuum pump sampling system.	238
Plate Six	Depositing dust sampling system.	288
Plate Seven	Air-sampling system.	305

THESIS ABSTRACT

The world's largest single-line lead smelter situated at Port Pirie in South Australia has continued its smelting operation for more than 100 years. This has resulted in a continuous distribution of lead in the form of dust into the local environment. Clearly, an environmental problem of disturbing proportion has existed for perhaps a century, this being manifest in the context of human/lead contact, with sinister health sequelae for populations at risk. Accepting the role of lead-bearing dusts as being the vehicle for carriage of the toxin to, primarily, children, research was directed toward demonstrating spatial variability in dust lead contamination patterns both statically and dynamically within the confines of vacant test houses, thus eliciting specific entrance pathways.

Contaminant levels in existing house surface dust, depositing dust and air-suspended dust both indoor and outdoor were examined as also home soils proximate to the dwelling structure. For such research, some novel sampling equipment was designed and utilised, evidence being presented to suggest the appropriateness of use of such sampling equipment. Lead levels in dust and soil are high, particularly house dust (eg >6,000$\mu g.g^{-1}$). This contamination by lead-bearing dust at household level appears widespread within the areas investigated, the source of the dust and the lead present being closely related. Test houses appear to be easily penetrated by contaminant dust, and further, appear to retain such dust, representing a focus of contamination. The bulk of the dust-contained lead within the house lies in the hand-adherent particle size range.

Within-house lead level spatial variability is apparent, windows and external doorways being demonstrated as major contaminant entry points. Rooms with open fireplaces are slightly more contaminated than those without, although fireplaces *per se* appear not to be major entry points for leaded dust. Houses exhibit almost an order of magnitude greater lead-bearing dust deposition rates under open windows and doors format than closed, hallways revealing the greatest levels of room contamination.

Surface soil contamination appears to be more a recent and on-going phenomenon rather than an historical artefact, spatial variability at home level reflecting this contemporary nature and also, as for deposited house dusts, suggesting source directionality. Lead levels in air and depositing dust appear largely independent of soil lead levels, soils being suggested as a poor indicator of inside home contamination levels. Contaminated depositing dusts within the home appear to be more newly-entering and contemporary than simply re-entrained and historical.

The bulk of surface dusts sampled in the vacant houses would appear to have deposited there during the house vacancy period, previous foot tracking being suggested as a minor pathway for home contamination in both vacant and occupied houses. Rates of leaded dust deposition in "de-dusted" houses are very similar to control uncleaned houses soon after house cleaning. Thus the value of comprehensive systematic house de-dusting as part of a community lead intervention program would appear to be questionable.

DECLARATION

I hereby declare that none of the material contained in this thesis has been accepted for the award of any other degree or diploma in any institution and that, to the best of my knowledge and belief, the thesis contains no material previously published or written by another person, except where due reference has been made in the text of the thesis. I consent to this thesis being made available for photocopying and loan, if applicable, and if the thesis is accepted for the award of the degree.

ALEX KUTLACA.

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CHAPTER ONE INTRODUCTION

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Many an Aeon moulded earth before
the highest, man, was born.
Many an Aeon too may pass when
earth is manless and forlorn.

Tennyson

Truth emerges more readily from error than from confusion.

Francis Bacon

1.1 PREAMBLE

For the last century, the lead smelter situated at Port Pirie, South Australia, the site of the world's largest single train pyro-metallurgical lead smelter (Hart et al., 1981; Luke, 1991), has maintained operations. This has resulted in the on-going and undesirable environmental distribution of lead, and its continuing build-up in the smelter surrounds (Thomas, 1978), the most likely method of dispersion being wind-blown dust (Body et al., 1988). Such fugitive and point source emissions have contributed to an environmental lead sink in Port Pirie, and the subsequent contamination of homes, typically in the form of lead-bearing dust (Body, 1986i). Analogous observations have been made at various other smelter locations internationally (Harper et al., 1987). Associated with such contamination in Port Pirie, there has been an enhanced health risk for resident children particularly, seven percent of whom in 1982 demonstrated blood lead levels greater than the then level of concern (Section 1.5).

Lead could be fairly described as the exemplar of an environmental chemical. Undoubtedly it is toxic, ubiquitous, persistent, non-biodegradable, bio-accumulative and demonstrates manifold pathways to humans (Simms, 1986; Legge, 1993). Once distributed to the environment, particulate lead cannot be recovered, remaining a potential source of contamination virtually indefinitely (Nriagu, 1990), lead being one of the most widely dispersed toxins of the twentieth century (Shy, 1990).

Literature review reveals that the most sensitive component of the population to lead intoxication is children from neonate to the age of approximately three years (SRC, 1990; Fergusson and Kim, 1991; IPCS, 1991), the same age range in which maximal lead uptake is likely to occur (McMichael et al., 1988). Further, it would appear that children are also more susceptible to a given intake of certain toxins per body weight (EPA USA, 1986), lead being one of these (SRC, 1990). This susceptibility in children relates to a number of factors which include an enhanced absorption from a given dose of lead (Farfel and Chisholm,

1990), and/or a greater sensitivity to body (eg blood) contaminant levels than adults (Mushak and Crocetti, 1990). This will be further investigated in greater detail in Chapter Four.[1]

The most significant source of contamination for young children is attributable to the mouthing of leaded dust-contaminated hands or non-food items, this dust deriving principally from the home environment (Duggan and Inskip, 1985a; Environment Protection Agency [EPA] USA, 1986; Fergusson, 1986; SRC, 1990; Fergusson and Kim, 1991).

With growing international awareness of the potential health concerns (eg neurobehavioural or developmental deficit) associated with (previously-considered) low levels of exposure to lead by, particularly, children (Needleman et al., 1979 & 1990; McMichael et al., 1988; Needleman and Gatsonis, 1990; Anon., 1995a), there has been significant downward revision of the threshold lead exposure level considered by medical authorities to demarcate the emergence of symptoms (EPA NSW, 1993).[2] Such re-evaluation of "safe" exposure levels warrants substantial investigation of the potential sources of lead contact particularly for children in Port Pirie, given its currently-acknowledged high levels of lead contamination of soil, air and dust (Body, 1988a), and the greater sensitivity of children to such environmental contamination (Chapter Four).

Given the smelter presence in the township of Port Pirie, the context of this research focuses on the contamination of the home as a result of its spatial relationship with the lead smelter (Section 1.5), acknowledging the well-demonstrated and long-recorded presence of elevated blood lead levels in people residing in such communities (Australian Government, 1925).[3]

Commencing in 1984, a ten year, $30 million lead abatement program in Port Pirie involved a very active home lead decontamination program. Despite this program and an existing current multi-million dollar State Government-funded program which commenced in 1994 (and which also involved home lead decontamination), (Arnold, 1993), little is known regarding the specific nature and source of the contaminant dust pathway to residents, and particularly children (Body et al., 1988b, & pers. comm.). This is also the case elsewhere (Raab et al., 1987), an important focus being the mechanism(s) of contaminant dust entry into homes and a child's subsequent contact with such dust. Dust is a very pervasive material

---

[1] Chronic lead intoxication from the Latin word for lead *Plumbum* (Anon., 1984), hence its chemical symbol Pb, is often termed Plumbism (Drasch, 1982), and has also been referred to as Saturnism (eg Saturnine gout [Nriagu, 1983c]), its effects eliciting a Saturnine or sluggish, gloomy, grave, phlegmatic or taciturn temperament [Anon., 1984]).

[2] Research eliciting this growing awareness of the toxic effects of lead at low exposure levels has been conducted in a variety of countries, the literature reflecting that the USA is the principal, with other countries such as Australia and (the former) Yugoslavia making significant contributions. The health-threatening effects of exposure to such previously-considered "safe" levels has been reported by a number of researchers (eg Marecek et al., 1983; Bellinger et al., 1984, 1986a & b, 1987a, 1991b & 1992; Smith, 1985; Dietrich et al., 1986, 1989, 1992 & 1993; Fulton et al., 1987; Schwartz and Otto, 1987b; ATSDR, 1988a; Mushak et al., 1989; Shy, 1990; Silbergeld, 1990; Alperstein et al., 1991; Baghurst et al., 1992; Needleman and Jackson, 1992; Rosen, 1992; Thacker et al., 1992; Wasserman et al., 1992).

[3] This has been extensively reported in the literature (eg McIntire and Angle, 1972; Nordman et al., 1973; Angle et al., 1974; Roberts et al., 1974a & b; Lagerwerff and Bower, 1975; Landrigan et al., 1975, 1976a & b; Baker et al., 1977; Muir, 1977; Rice et al., 1978; Roels et al., 1978 & 1980; Angle and McIntire, 1979; Brunekreef et al., 1981; Landrigan and Baker, 1981; Landrigan and Hayes, 1981; Swaine et al., 1981; Landrigan et al., 1982; McMichael and Johnson, 1982; Mahaffey et al., 1982a; Popovac et al., 1982; Hartwell et al., 1983; Neri et al., 1983; Angle et al., 1984; Prpic-Majic et al., 1984; Duggan and Inskip, 1985; McMichael et al., 1985; Englert et al., 1987; Harper et al., 1987; Maravelias et al., 1989; Triger et al., 1989; Graziano et al., 1990; Hertzman et al., 1990; Baghurst et al., 1992; Langlois et al., 1996; Murgueytio et al., 1996; Cikrt et al., 1997).

with which humans are continually in contact (Allott et al., 1992). This research attempts to address some of these concerns.[4]

1.2 THE CONTEXT OF RESEARCH

Lead, a non-naturally occurring element in its metal form, is a metal of antiquity, and is the first won from its ores (Gowland, 1902). It has a history of more than 7,500 years acquaintance with humans (Fergusson, 1986) and has been a known poison for thousands of years (Berney, 1993 - Chapter Two). Associated with its long history of human use, lead also reflects a very long history of well-defined symptoms associated with its systemic effects on humans subsequent to adventitious contact (Chapter Four). The element has no known useful physiological functions in humans (Lawther, 1986).

Pervasive on-going lead contamination of the environment arising as a result of many decades of (largely) uncontrolled discharge has re-emerged as a serious health threat, constituting a prime environmental health problem in Australia (EPA NSW, 1993) and the premiere environmental health threat currently facing American children (Landrigan, 1990; Landrigan and Curran, 1992; Mushak, 1992).[5]

The incidence of overt plumbism (chronic or frank lead poisoning) *per se* has been, and appears still to be falling internationally (Pocock and Ashby, 1985; SRC, 1990) and nationally (EPA NSW, 1993). The shifting paradigm of the relative toxicity of lead at ever-decreasing body levels (Dietrich et al., 1993), however, promotes the contemporary debate regarding the possible (Simms, 1986) or likely (Needleman, 1990) subtle effects of <u>low</u> body level lead on the cognitive development of children, potentially eliciting affected behaviour and mentation. Results of epidemiological studies in several countries over the last decade have heightened the community's concerns regarding (previously-considered) low or safe levels of lead intoxication (Davis and Svendsgaard, 1987; Needleman, 1988 & 1990; Needleman and Bellinger, 1988; Landrigan and Curran, 1992; Mushak, 1992; - Chapter Four). Such "low" blood lead levels are currently considered to be intimately associated with a small but significant decrement in intellectual development, particularly arising from impairment to the developing central nervous system (CNS) of children from neonate to around three years of age (Davis and Svendsgaard, 1987; Landrigan and Curran, 1992; Mushak, 1992; EPA NSW, 1993; - extensively cited in Chapter Four). These concerns have elicited a worldwide re-evaluation of current environmental lead emission controls, and a review of the

---

[4] McCunney et al. (1992) note, from a broad inspection of the literature, that over the period 1976 - 1990, within the environmental categories of publications, air pollution, especially that due to dust was the most heavily indexed topic. Greater than 70% of the world's population lives in urban areas (World Resources, 1989), such individuals are thus very likely to come into regular contact with contaminated dust, particularly children who are more likely to ingest dust, and also who are likely to be more susceptible to the actions of contained toxins (Allott et al., 1992).

[5] Seventeen percent of all US preschool children display elevated (> 15μg.dL^{-1} or μg/100ml; dL = deci-Litre or 100ml) blood lead levels corresponding to 68% of poor minority inner-city children (ATSDR, 1988; Mushak and Crocetti, 1989 & 1990; Landrigan and Curran, 1992). In Australia, the National Health and Medical Research Council (NHMRC) estimate that in NSW alone, between 14 and 40% of preschool children would display a blood lead level greater than 15μg.dL^{-1}, the current level of concern for blood leads being set at 10μg.dL^{-1} (EPA NSW, 1993).

(previously-considered) relative "safety" of many lead-contaminated environmental compartments (Anon., 1991).

Much work has been undertaken over the last two decades attempting to elucidate the guise(s) under which this "ubiquitous hazard" (Lin-Fu, 1973a; p. 23; Sayre et al., 1974; p. 170; Landrigan and Curran, 1992; p. 279; Berney, 1993; p. 30; EPA NSW, 1993; p. 5) subtly affects the health of infants, how this "silent epidemic" (Fee, 1990; p. 575) wreaks its havoc (Needleman, 1991) and mayhem (Ramazzini, 1713).

A variety of pathways for a number of sources of available lead have been suggested. These include house paint containing lead as an opacifier that is deteriorating (Lin-Fu, 1973; Clark et al., 1985; Sturges and Harrison, 1985) or that is being ablated (Hamilton, 1982), certain industrial employment (Baker et al., 1979), contact with lead-contaminated soils (Thornton, 1982), lead-contaminated food (Kolbye et al., 1974) and drinking water (Morse, 1979b), high-lead glazes on pottery (Klein et al., 1970; Miller, 1982), pewter (CEH & CAPP, 1987) or dinnerware (Wallace et al., 1985), some home-activities (Palmisano, 1969) and emissions to the environment from mining (Chiaradia et al., 1997) and industry (Mahaffey et al., 1982a) and from the combustion of leaded gasolene (Millar and Cooney, 1982; Mushak and Crocetti, 1996a & b).[6]

Of notable import and more recent detection, however, is a major source of human contamination to children in their infancy years, and typically proceeding via the normal mouthing of fingers or non-food items. This may occasion the ingestion of substantial quantities of lead-bearing dust, particularly within the confines of the domicile.[7] Over the last two decades it has been demonstrated by a number of research groups (eg Angle et al., 1974; Roberts et al., 1974b; Roels et al., 1980) that lead in air and lead in dust correlate strongly, and that proximate to an industrial area, the quantitative contribution of air lead to

---

[6] Typically such contaminant acquisition from contact with leaded gasolene arises via contact due to proximity to emitting sources. Common sources include traffic and its associated density (Butler et al., 1974; Fergusson and Stewart, 1992), employment (Goldman et al., 1987; Gulson et al., 1993) or residence proximity to motor garages (Hamilton, 1925b; Chambers et al., 1984; Davies et al., 1987a & 1990; Gulson et al., 1993), and also, on occasion, somewhat unusually, eg occupational chain-saw operators (van Netten et al., 1987).

[7] This has been extensively reported elsewhere (eg Chisholm, 1973; Lin-Fu, 1973; Needleman and Scanlon, 1973; Angle et al., 1974 & 1984; Barltrop et al., 1974 & 1975; Lepow et al., 1974 & 1975; Roberts et al., 1974a & b; Sayre et al., 1974; Ter Haar and Aronow, 1974; Vostal et al., 1974; Day et al., 1975; Lagerwerff and Brower, 1975; Landrigan et al., 1975 & 1976; Waldron, 1975; Solomon and Hartford, 1976; Baker et al., 1977; Duggan and Williams, 1977; Rice et al., 1977 & 1978; Yankel et al., 1977; CDC, 1978; Dolcourt et al., 1978; Roels et al., 1978 & 1980; Barltrop and Meek, 1979; Sayre and Katzel, 1979; Schmitt et al., 1979; Billick et al., 1980; Charney et al., 1980; Duggan, 1980, 1983a, b & 1984; Walter, 1980; Brunekreef et al., 1981 & 1983; Fergusson et al., 1981; Sayre, 1981; Charney, 1982a, b & 1983; Hammond, 1982; Johnson et al., 1982; Landrigan et al., 1982; Lin-Fu, 1982a & b; Morton et al., 1982; Popovac et al., 1982; Reeves et al., 1982; Stark et al., 1982; Aurand et al., 1983; Culbard et al., 1983a & b; Hartwell et al., 1983; Kawai et al., 1983; Mahaffey, 1983a & b; Mielke et al., 1983; RCEP, 1983; Task Force, 1983; Yaffe et al., 1983; Brunekreef, 1984 & 1985; Culbard and Johnson, 1984; Elwood et al., 1984; Gallacher et al., 1984; Hamilton et al., 1984; Baghurst et al., 1985; Bornschein et al., 1985b, 1986, 1987, 1989 & 1991; Duggan and Inskip, 1985; Duggan et al., 1985; Hammond et al., 1985; Jensen and Laxen, 1985; Manton et al., 1985; McMichael et al., 1985; Rabinowitz et al., 1985a & b; Bellinger et al., 1986a & b; Chisholm, 1986; EPA USA, 1986; Harper et al., 1987; Laxen et al., 1987b; Strehlow and Barltrop, 1987; ATSDR, 1988a & b; Calder et al., 1989; Marcus and Cohen, 1989; Reagan and Silbergeld, 1989; Roberts and Ruby, 1989; Davies et al., 1990; Farfel and Chisholm, 1990; Hertzman et al., 1990; Thornton et al., 1990; CDC, 1991; Clark et al., 1991; Cotter-Howells and Thornton, 1991; Fergusson and Kim, 1991; Lewis et al., 1991; Figley and Makohon, 1992; Sayre and Ernhart, 1992; Bernigau et al., 1993; Thornton, 1993; Ewers et al., 1994; Farfel et al., 1994; Gulson et al., 1994b; Lanphear, 1994; Anon., 1995a; Cambra and Alonso, 1995; Farfel and Lim, 1995; Marcus and Elias, 1995; Menton et al., 1995; Rabinowitz, 1995; Goulet et al., 1996). It would appear that the infant would spend most of its time within this location (Thompson et al., 1973; Lefcoe and Inulet, 1975; Binder et al., 1976). This reflects the historical trend away from the association of lead in air as a prime predictor of lead in blood for children to dust lead and (adherent) hand lead levels, suggesting a more indirect relationship for air and blood lead (Brunekreef et al., 1983; Brunekreef, 1984).

children's blood lead levels is negligible by comparison with that of (adherent) hand lead levels (Davies et al., 1990) and also home "fallout" lead (Angle et al., 1974; p. 27).

Children demonstrate an enhanced likelihood of ingesting significant quantities of such leaded dust due to:

> a) repetitive hand/finger sucking (normal developmental behaviour [Rabinowitz, 1995] which tends to peak at pre-school years [Bornschein et al., 1985]), and
>
> b) the mouthing of non-food items eg toys (Lockeretz, 1975), soil (Barltrop et al., 1975) etc, again a normal behavioural pattern (Bornschein et al., 1986), although some children may demonstrate such traits to excess (Hawley, 1985; La Goy, 1987; Calabrese, 1992).

EPA USA (1986) estimates that dust accounts for > 90% of the additive lead attributable to living in an urban environment or near a smelter. Such observations link with reports of a strong association between house dust lead and child hand lead loading (quantity of lead per hand) (eg Sayre et al., 1974; Bornschein et al., 1985; Duggan et al., 1985; Davies and Watt, 1986; Clark et al., 1987; Davies et al., 1990) and playground dust lead and hand lead loading (Roels et al., 1976 & 1980; Charney et al., 1980 & 1983; Brunekreef et al., 1983; Gallacher et al., 1984; Sherlock et al., 1985; Clark et al., 1985). Further, strong association also appears between (child) blood lead levels and indoor house dust lead and hand lead levels (Sayre et al., 1974; Charney et al., 1980; Roels et al., 1980; Brunekreef et al., 1981; Charney, 1982b; Gallacher et al., 1984; Bornschein et al., 1985b, 1986, 1987, 1988 & 1991; Clark et al., 1987 & 1991; Davies, 1990). Elwood et al. (1984) also report this relationship for children, but not adult women, further supporting the role of house dust and hand lead.

Charney et al. (1983) note that good personal hygiene and housekeeping can significantly reduce dust and lead loadings (quantity of dust and lead per unit area respectively) on surfaces in homes and blood lead levels in children. This indoor-deposited dust represents that house dust fraction that is most readily available to small children, hence it demonstrates the most substantial biological significance. It can settle on every surface in the room, and provides a ready source of lead by the hand-to-mouth route (Roels et al., 1980; Brunekreef et al., 1981; Charney, 1980 & 1982; Yaffe et al., 1983; Bornschein et al., 1985b; Elias, 1985; Watt et al., 1993), Chisholm et al. (1985) concurring and suggesting that the child's dwelling constitutes the principal environmental source of lead. Whilst workers such as Milar et al. (1980) and Dietrich et al. (1985) report a relationship between child blood lead and home descriptors such as dust lead loading and concentration, work of notable heuristic value by Bornschein et al. (1985) indicates that this relationship is mediated by hand lead. Elsewhere, this hand-to-mouth activity has been demonstrated to account for at least 50% of a child's total lead intake (RCEP, 1983; Davies et al., 1990).[8]

A number of workers describe successful dust control as a means of lead prevention management (Charney et al., 1980; Roberts et al., 1991; Mielke et al., 1992). Farfel and Chisholm (1990), however, report on a leaded paint abatement program in Baltimore, USA, where blood lead levels in resident

---

[8] In the context of sampling this house dust, collection by a domestic vacuum cleaner would appear to provide dust that would be less accessible to small children, given that the dust would be acquired from the carpet and a variety of less accessible (to a child) sources, and pulled with some force. This will be pursued at length in Chapter Five.

children actually increased after the work was completed, citing the absence of dust-suppression measures during the intervention stage as the reason.

1.3 GENERAL RESEARCH DIRECTION

Preliminary literary research examines the dimension of lead contamination of the environment by way of reference to global production and environmental discharge statistics. Such pursuit reviews a plethora of studies assessing the degree and (possible) mechanisms of environmental contamination (and associated human intoxication) of various locations internationally. Dissertation, however, will focus on a specific lead-contaminated site, namely Port Pirie in South Australia, a smelter location that has been in continuous operation for more than a century (SAHC, 1983). The presented work centres on lead's pollution of the home environment, particularly the house, in order to investigate the distribution of lead-contaminated dust and also potential mechanisms of entry of such dust into the dwelling.[9]

Such investigation is warranted by a perceived paucity of, and associated need for, related data (Al-Radday et al., 1993a). Further, such contaminant material source and pathway identification constitutes a high priority for abatement programs at many lead-contaminated sites (EPA NSW, 1993; Maynard et al., 1993; Gulson et al., 1994b).

In this pursuit, vacant houses appeared more appropriate than occupied dwellings in that occupant activity may introduce substantial between-house depositing-dust variability. This variability might occur due to occupants either contributing directly to contaminant entry, or by their modifying the pattern of dust entry, movement or deposition (Fergusson and Kim, 1991). The value of such a choice of test houses rests in the opportunity of looking at contaminant patterns as a result of sustained and undisturbed exposure. Essentially, the vacancy of a specific house controlled for anthropogenic sources of dust entry such as dust/soil track-in with footwear (Charney et al., 1983; Culbard and Johnson, 1984; Thornton and Jones, 1984; Roberts and Ruby, 1989; Budd et al., 1990; Roberts et al., 1990, 1991a & b, 1992, 1993a & b; Camann et al., 1991a & b, Lewis et al., 1991; Mielke et al., 1992), dust adherent to clothing (Crockford, 1992) or pets (Brunekreef, 1985) or as introduced by other activity (Thompson et al., 1973; CDC, 1978; von Lindern, 1984; Fergusson and Kim, 1991; Gulson et al., 1995). Additionally, simple dust particle resuspension by human activity within the house is also likely to contribute (Lefcoe and Inulet, 1971; Yocom et al., 1971; Sterling and Kobayashi, 1977; Moschandreas et al., 1979; Yocom, 1982; Fergusson and Kim, 1991; Gulson et al., 1995). Murphy and Yocom (1986) suggest that such resuspension may be comparable in dimension to outdoor-sourced dust penetration and deposition. Resuspension of settled indoor dust may be responsible for very high indoor levels of lead in air during periods of high indoor activity (eg homes with young children). Such a situation represents an enhanced health risk for children who will experience leaded dust exposure through inhalation as well as ingestion pathways (Chapter Four).

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[9] For the purposes of this discussion, the term "home" refers to the house and the block of land upon which it is sited, whereas house refers simply to the dwelling.

The process of cleaning with a vacuum cleaner in occupied homes has been noted by some workers to register a greater than normal dust deposition level, typically almost four times as great as normal levels (Ewers et al., 1994). Further, the cleaning process may in fact preferentially concentrate the smaller-sized, higher lead content particles in the room by failing to trap them in the cleaner bag (Body, 1986i).[10]

1.4 THESIS STRUCTURE

Included in Chapter One is a discussion of the general aims and objectives of this work and some descriptive detail of the Port Pirie research area. Also included is a general discussion of research methodology, incorporating test house details and location, sampling equipment used, the timing of certain components of the work, and data interpretation techniques.

Given the inter-disciplinary nature of this research, in order to adequately comprehend the dimension of the current lead problem internationally and locally, some reference to lead's physico-chemical nature/properties, and the environmental distribution/discharge of the ore and its refined progeny should be indulged (Chapter Three). Further, to effectively encapsulate the importance of lead to a modern society, it appears necessary to also provide a brief overview of the scale of the mining of the ore, the modern-day usage of the metal, the (possibly) cavalier approach to its environmental discharge which subsequently elicits frequent and intimate adventitious human contact, and its relative environmental stability subsequent to such release. Such review would lack patency if it failed to also focus on the approach, over essentially the last two to three decades, to an assessment of lead's build-up within various environmental compartments, and associated (environmental) pathways contributing to excessive human exposure.

Considering that the major concern regarding lead in our modern society is in a health context, it is imperative to investigate the interaction of the element with normal human biochemistry, *apropos* to which is presented a fairly detailed literature review of the physiological implications of lead intoxication at various levels (Chapter Four). No review of this toxin's disturbance to homeostasis could be construed to be too detailed given its oft-suggested subtle but insidious action.

The dimension of such literature review (Chapters Three and Four) reflects the prodigious quantity of published material pertaining to the subject, the very nature of the complexity of lead's multi-faceted interaction with our environment and ourselves, the calibre of the penetration of the metal into our environment and daily life, and the relative importance ascribed to the health implications associated with undue lead exposure.

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[10] Figley and Makohon (1992) report very high airborne concentrations of leaded dust resulting from the use of a portable vacuum cleaner, such an appliance, under test conditions, collecting an average of less than 60% of previously-applied leaded dust to floors. Lefcoe and Inculet (1975) note a 100 to 1,000% increase in air-suspended particles depending upon particle size range within homes as a result of the use of the portable domestic vacuum cleaner, Ewers et al. (1994) acknowledging the cleaner's ability to concentrate leaded dust at carpet surfaces, enhancing potential exposure. Even the act of making a bed contributed by a factor of between ten and fifty times. Sweeping of floors has also been suggested as a contributor to re-entrainment of dust (Chambers et al., 1984).

No comprehensive thesis pertaining to the role of lead in a modern society with its associated manifold environmental implications would be complete without due reference to the historical emergence of such a metal. In this regard, a separate section (Chapter Two) traces, albeit only briefly, a history of the mining/smelting and use of this (frequently) much-revered metal, and its oft-cited association through the millennia with lead intoxication and poisoning. Such a backdrop is presented to contextualise the current problem with lead in our modern society, particularly the dimension of environmental contamination, discussion of which is presented in Chapter Three.

A preliminary research section (Chapter Five) deals with some initial investigation in regards to the character of dust to be sampled for this study. Given the very heterogenous nature of dust (Clark et al., 1991), it is intended that such initial investigation will facilitate a better understanding of the dust involved in this study, and the dimension of home contamination by lead-bearing dust. Given that new sampling equipment and sampling procedures are employed for this work, some initial method validation appears warranted, and is discussed briefly below. Detailed discussion of sampling equipment used is presented at a later stage (Chapters Five, Six, Seven, Eight and Appendices).

Lead distribution in the surface soil proximate to the vacant homes is investigated (Chapter Six) in order to provide an environmental "context" for the vacant houses, enhancing the overall "picture" of home contamination in these test residences.[11] Further, such home soil investigation also attempts to assess the spatial variability of surface soil contamination, although this does not constitute a substantial component of the overall research project.

A subsequent chapter (Seven) focuses on deposited lead-bearing dust in the house as the principal pathway of lead to the occupants, and deals with the (existing) spatial variability in such dust lead distribution in vacant "test" homes in Port Pirie. Body notes that the "distribution of lead-contaminated dust within homes clearly needs further evaluation" (1986f; p. 4). Duggan and Inskip (1985) suggest a substantial spatial variability in dust lead concentration within a home, yet acknowledge, as Laxen et al. (1988), a relative paucity of and a demonstrated need for published data pertaining to intra-household variation as also noted by Seifert (1984; and also pers. comm. 1994). The "static" lead-in-dust contamination study conducted here comprises a systematic assessment and comparison of the dimension and spatial variability in lead concentration and dust and lead loading of, principally, floor level (multiple) dust samples both within and between the test houses.

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[11] Whilst literature reveals that there exists a modest correlation (r lies in the range of 0.1 to 0.4) between blood lead and soil lead (concentration) levels (SRC, 1990), for the purposes of the vacant home research conducted here, the contribution of such soil-derived data would be more in the context of a preliminary investigation into any relationship that might exist between (say) average floor dust lead concentration/loading level and respective home surface soil lead contamination levels. Linkage between soil contaminant level and analogous house floor dust level revealing statistically significant association has been demonstrated for a number of compounds elsewhere eg pesticide residues (Camann et al., 1990; Budd et al., 1990; Lewis et al., 1994). The relationship between soil and house dust will be discussed at greater length in Chapters Three and Ten.

In earlier work conducted in Port Pirie (Body, 1986d) only three lead-in-dust concentration figures were elicited per home.[12] Substantially greater sampling from homes here appears very desirable in order to give a more complete picture of the degree and variability of contamination of individual houses. Reflecting the work of a number of other groups, Yankel et al. (1977) detect a lack of a significant correlation between blood lead levels and simply the concentration of lead in dust deriving from domestic vacuum cleaner dust bags. This does not seem surprising since it is not the dust lead concentration *per se* but the quantity of lead per unit area of the house (the lead loading) which in fact reflects the degree of "dirtiness/dustiness" of a house (as Roels et al., 1980; von Lindern, 1984).[13]

Laxen et al. appear emphatic with respect to the need for multiple in-home sampling, "there being a clear limitation in the use of a single dust sample to characterise dust leads within a home", (1988; p. 8.). Window sills and wells will also be sampled from, these having been proposed as primary dust entry points and sites of potential child contamination (Charney, 1982b).

This sampling strategy is expected to yield valuable information regarding the degree of contamination of various parts of rooms and various rooms of selected Port Pirie houses. Such assessment should provide a valuable bench-mark to estimate the potential range of home leaded dust levels in Port Pirie, and facilitate a more precise rationale for a subsequent home contamination assessment and intervention program. Here, particular attention will be paid to such variables as the sampling location within the room, and its relationship to certain room features (eg doors, windows, fireplaces). Focus will be directed to inter-room differences, and also to (presumed) contaminant source pathways, eliciting comparison of contributions to dust lead build-up in rooms according to room feature. Such an initial procedure should constitute an integral part of any dust-characterisation study, in order to arrive at some comprehension of existing home contamination calibre and patterns, and to delineate a more precise rationale for future investigation and intervention strategies.

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[12] Whilst some pre-existing data for lead-in-dust concentrations were available, lead loading data were not available. Typically, researchers have used dust lead concentration as their test environmental parameter rather than lead loading, presumably because it appears a less variable measure than loading (Duggan and Inskip, 1985). Bornschein et al. (1986), however, suggest that lead loading is a far more representative parameter of house contamination. From the literature, lead loading (in homes) displays a substantially better correlation coefficient with (occupant) blood and hand-attached lead levels than the much more frequently sampled lead concentration of floor-level dust (Roels et al., 1980; Charney, 1982a & b; Culbard et al., 1984; Brunekreef, 1985; Duggan and Inskip, 1985; Rabinowitz et al., 1985a; Davies and Watt, 1986; Davies et al., 1987d, e & 1990; Laxen et al., 1987; Clark and Bornschein, 1990; Roberts et al., 1990 & 1991; Lanphear, 1994). Davies et al. (1990) and Thornton et al. (1990) claim a correlation between blood lead and dust lead concentration of $r = 0.21$ and between blood lead and dust lead loading of $r = 0.46$, similar to Bornschein et al.'s (1986) figure of $r = 0.53$. These data suggest that the quantity of lead present on the surface of carpet is more critical than the concentration of lead in dust in determining exposure risk. Contrasting this, Laxen et al. (1987a) detected no difference between dust lead concentration and lead loading correlations with blood lead data, although they used a home vacuum cleaner for sampling (in only two locations per house), eliciting concerns regarding non-random sampling and selective capture of larger-sized contaminant dust particles only (as Watt et al., 1983; Duggan and Inskip, 1985; Body, 1986f & i).

[13] A number of workers have used a house dirtiness index when reviewing the houses of children in test areas (eg Yankel et al., 1977; Duggan, 1980; Roels et al., 1980; Diemel et al., 1981; Charney, 1982b; Culbard et al., 1983a; Angle et al., 1984; von Lindern, 1984; McMichael et al., 1985; Raab et al., 1987; Kim and Fergusson, 1993). Many have demonstrated a reasonably good association between this rating and house dust contaminant loading (Kim and Fergusson, 1988) and (occupant) children blood lead levels. Ultimately, the critical factor for child health here is the quantity of lead ingested. This value would appear to be more accurately reflected by the total quantity of available lead than simply by its dust concentration. Dust sampled from homes using several different techniques, reflects good correlation between resident children's blood lead levels and dust sampled only when using a dust-sampling system not dissimilar to that proposed here (Lanphear, 1994).

Complementing this work in the suite of vacant houses is a study estimating the rate of contamination (by dust) of these Port Pirie (former) homes, detailing changing patterns of depositing dust (dustfall) and lead in depositing dust (leadfall) at various locations within the house, and under different seasonal and residence conditions (Chapter Eight).[14]

Lead-in-air levels within and immediately outside these test houses and also a suite of similarly-located occupied houses has been monitored, and will be discussed (Chapter Nine). Here, the intention is to provide an indication of the extent of contamination of the test houses by (initially) suspended contaminated material, and also to estimate the influence of house occupancy on the relative penetration of lead-bearing dust into houses. Since infants spend the majority of their time within the family house, it is more the indoor rather than outdoor environment which determines a child's contaminant exposure (Binder et al., 1976; Budiansky, 1980).

Linked with such a (low-volume) dust-in-air sampling program, a contemporaneous dust-deposition sampling program as described below (Chapter Eight), will also be conducted within some of the test homes. The intention of such dual sampling is to estimate the relationship between (home) ambient air and depositing dust lead levels.[15] It is acknowledged that meteorological variability has not been controlled for in this study. Such contemporaneous sampling of air lead levels and indoor dust lead re-contamination rates, however, could facilitate the genesis of a linking model which could elicit a means of making meaningful comparisons between subsequently-assessed depositing dust data derived (by necessity) from different rooms, different houses, different locations and at different times.

These current levels of investigation are complementary in that they contribute to a more substantial overall comprehension of the dimension and calibre of current lead-in-dust contamination at dwelling level. These investigations are intended to contribute to a comprehension of the dimension and nature of leaded dust distribution in homes, and unravel an understanding of the linkages between various test home environmental compartment contaminant levels.

Chapter Ten presents a general discussion of the potential linkages between the contamination of the various environmental compartments investigated in previous chapters. It suggests pathways for contaminated dust entry into houses and its distribution therein, proposing suggestions for appropriate management of dust entry into houses. Concluding remarks and recommendations are included in the penultimate chapter (Eleven), the final chapter listing consulted references.

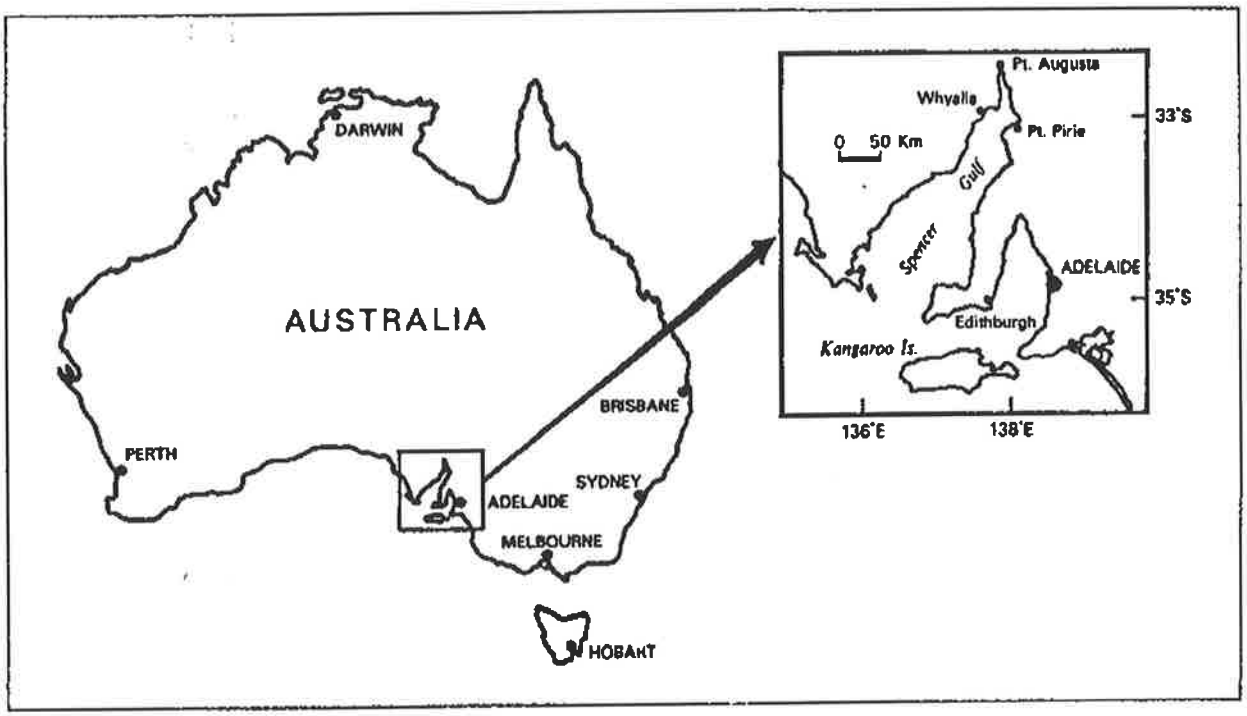
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[14] The intent of conducting lead-in-dust research in vacant houses within the township of Port Pirie in the first instance is to reveal current concentrations and loadings of lead in a variety of dust samples taken from various locations within the residences, much of which has not been conducted previously. Prior to this study, no lead loading data or for that matter lead in depositing dust data were available for any Port Pirie houses, vacant or occupied.

[15] It is assumed that the operation of (low-volume) indoor air-sampling devices would have minimal effect upon normal dustfall regimens.

1.5 THE STUDY SITE; A GENERAL DESCRIPTION OF PORT PIRIE

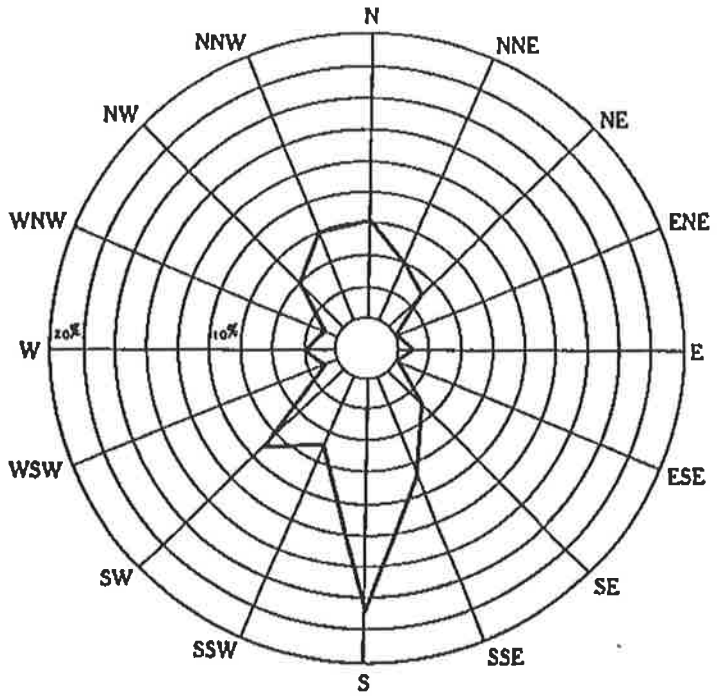
Located 227kms north of Adelaide, the township of Port Pirie covers an area of around 18km^2, and lies on a flat coastal plain of between ten and thirty kilometres width. The township is located in a rural environment, with the CBD situated on the estuary of the tidal Port Pirie River close to its mouth in Spencer Gulf. This is just south of the large smelter works which is situated on the western bank of the estuary (Anon., 1996; Figure 1.1). Much of the area surrounding the township is low-lying land mostly sparsely vegetated as a result of the dry climate, high salt content of the soil, and human intervention (Buckley, 1983).



Source: Ward, 1981; p. 88.

Figure 1.1 Study site location.

Port Pirie is a fairly dry dusty township, with an average annual precipitation of around 345mm.yr^{-1}, and a mean evaporation/precipitation ratio approaching 6.5/1 (Kinhill, 1990). Winds may be very changeable even on a daily basis (Harris, 1986), extending from NNW - SSE (Cartwright et al., 1976; Harris, 1986). Prevailing winds emanate principally from the SE in summer and from the NW in winter (Figure 1.2),with the smelter essentially lying upwind of the city (McMichael et al., 1986). Smelter operations commenced in 1889, the Broken Hill Associate Smelters Pty Ltd (BHAS) purchasing the site in 1915 (Australian Government, 1925).



Source: Cartwright et al., 1976; p. 71.

Figure 1.2 Mean distribution of wind direction at the smelter site, Port Pirie.

Demonstrating a population of 14,110, with around 20% unemployed, the township has the smelter works as its largest single employer with a staff of approximately 900 (Mayfield, 1994), this having declined from a figure closer to 1,800 twelve years previously (Buckley, 1983). Currently, this site is the world's largest single train lead smelter, producing approximately five percent of world lead smelted from lead concentrate. Since its commissioning, the smelter has produced of the order of 15 million tonnes of lead (1997 figs; A. Gilbert, pers. comm.). The ore derives from the mine site at Broken Hill, where it is concentrated. A connecting railway line approximately 400kms in length and completed in 1888 links Broken Hill to Port Pirie.[16]

As can be seen from Figure 1.3, the smelter is located close to the CBD. Marked districts such as Port Pirie West and Solomontown exhibit the most elevated soil and dust lead levels (Tiller et al., 1976; Baghurst et al., 1985; Body, 1986d, f, & 1988a), and represent enhanced health risk residential locations (T. Maynard, pers. comm.). A preliminary presentation of the location of the test houses in which research is conducted is also depicted here, further discussion of these houses being covered subsequently (Section 1.9).

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16 The Silverton Tramway Company was responsible for constructing the railway line connecting Broken Hill with Port Pirie. In 1888, upon project completion, James Smith Reid, principal of the company became an exceedingly rich businessman (Bell, 1993).

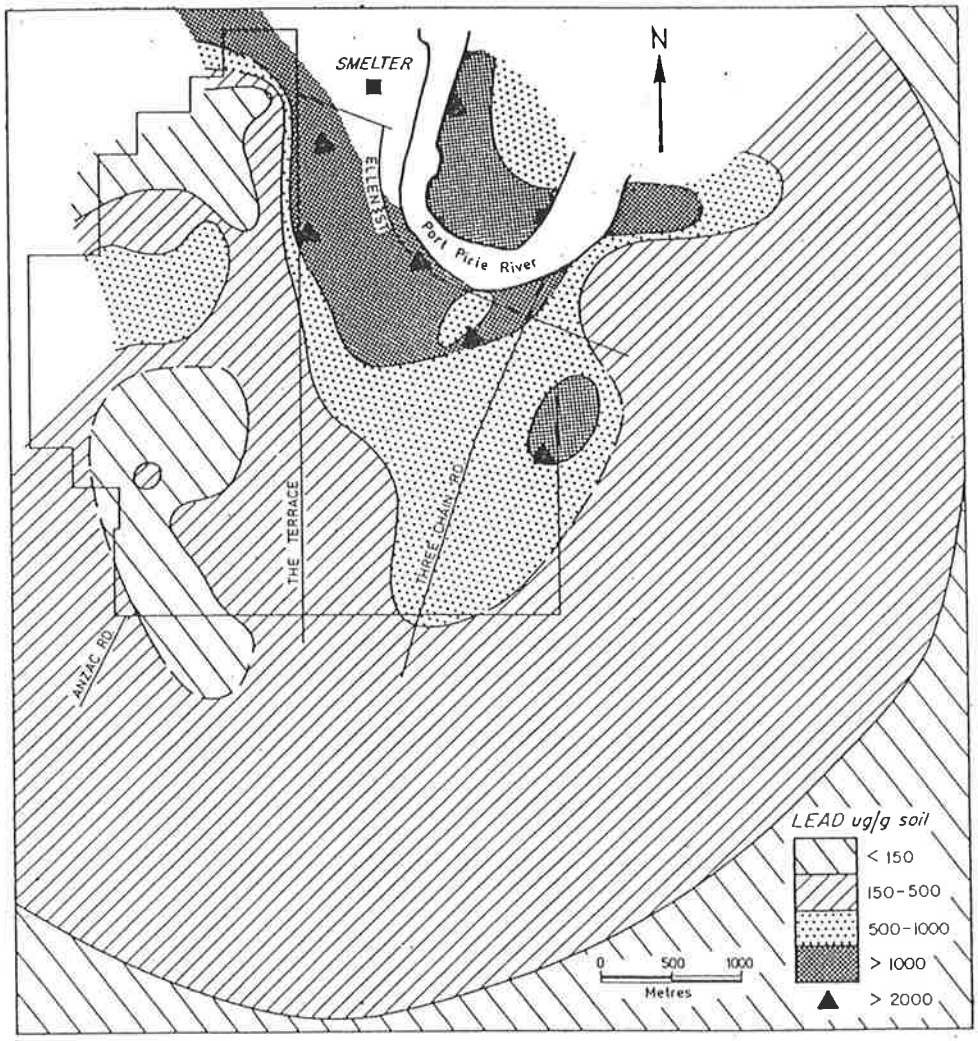


Source: DENR, 1996 ; aerial photo.

Figure 1.3 Port Pirie house, high risk area and smelter locations.

1 = 6 Agnes Street; 2 = 64 Edward Street; 3 = 58 Federation Road; 4 = 29 Fourth Street; 5 = 19 Lily Street; 6 = 3 Sabine Street; 7 = 35 Square Street; 8 = 34 The Terrace; 9 = 27 Vera Street; 10 = 136 Warnertown Road; 11 = 97 York Road.

Given that the smelter at Port Pirie has been in continuous operation for more than a century (Luke, 1991), and acknowledging a long history of environmental discharge of both concentrates and products of manufacture (Body, et al., 1988), it is not surprising that a number of researchers have noted that the surrounding environment reflects considerable contamination of soil (Tiller et al., 1975 & 1976; Cartwright et al., 1976; Tiller and de Vries, 1977; Merry and Tiller, 1978; Merry et al., 1981), crops (Merry et al., 1981) and also marine sediments, grasses, shells and fauna (Dossis and Warren, 1981; Harbison 1981 & 1984; Swaine et al., 1981; Ward and Young, 1982; Bell, 1983; Olsen, 1983; Tiller et al., 1989). This dispersed contamination in soil extends further than 40kms from the CBD (Merry and Tiller, 1978; Merry et al., 1981), certainly extending over proximate residential development, some of which has existed for virtually as long as the smelter. The highest concentrations of lead in surface soil have been detected within two kilometres of the smelter, levels decreasing with distance from the smelter. The range of soil lead concentrations extends from background levels (< 50$\mu g.g^{-1}$) to, occasionally, greater than 5,000$\mu g.g^{-1}$ (Tiller et al., 1976; Body, 1986e; Figure 1.4).



Source: Tiller et al., 1976; p. 9.

Figure 1.4 Soil lead levels in and around Port Pirie.

There exists a greater likelihood of elevated soil lead concentrations extending further in a direction SSE of the smelter (Landrigan, 1983), presumably reflecting the prevailing wind pattern. Body et al. (1988) also note that the spread of contaminant dust over the township is in a southerly and westerly direction.

Air lead levels in Port Pirie have been monitored by a variety of techniques since the mid 1960s (Sweetapple, 1984). This author notes, however, that although marked reductions have occurred, levels in the early 1980s were marginally above the NHMRC-recommended maximum level. Later work by Goh and Hope indicates that some increase in lead-in-air levels has been noted more recently (1985a & b, 1986, 1992).

Lead-health concerns have existed in Port Pirie for many years (Aust. Govt., 1925), with greater than 1,000 employees of the smelter having been suspected of having suffered lead poisoning between the years 1928 and 1959, only 241 cases having been confirmed (Body, 1988b). The health of the children of the township has also been in doubt, the mean capillary blood lead level of 1,241 tested school children (≈ 50% of the total) approximating 18.2$\mu g.dL^{-1}$ in 1982 (Sinclair and Dohnt, 1984). Seven percent of those children tested at that time demonstrated a blood lead level of > 30$\mu g.dL^{-1}$, the then NHMRC-recommended level of concern (Calder et al., 1986; Wilson et al., 1986).[17]

There exists strong evidence to suggest that the major primary source of environmental lead contamination in the township derives from the BHAS smelter. Clustering of children with elevated blood lead levels occurs for those resident in the areas of Port Pirie which are closest to the smelter constituting the higher health risk areas (Maynard et al., 1993; Figure 1.3). These areas also exhibit the most elevated levels of lead in soil and house dust (Tiller et al., 1976; Baghurst et al., 1985; Body, 1986d, f, & 1988a). There is potential for the contamination by lead of house dust via a number of sources, although apart from home activity, the principal internal source would be degrading paint. From work by Body et al. (1988), however, no correlation was demonstrated between paint contamination and the lead concentration of household dust samples, the presence of paint flakes being very infrequently detected in house dust samples. Body et al. (1988) also note that there exists little or no correlation between the presence of lead-based paints in homes and elevated child blood lead levels in Port Pirie.

A comprehensive ten year cohort study investigating the effect of lead exposure on childhood development commenced in the early 1980s, (SAHC, 1983). During 1983, extensive media publicity was given to Port Pirie's lead pollution and child health problems (Vimpani et al., 1985), such media attention reappearing occasionally until the present (pers. obs.).[18]

From a number of studies of smelter communities described in the literature, it would appear that a summary of contributing pathways to the elevated risk of (child) lead intoxication would include lead-contaminated soils, lead-bearing dust (atmosphere-entrained, house dust and street dust), atmospheric

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[17] Children of smelter workers may also be at greater risk than those whose parents are not smelter employees, Baker et al. (1977a) and Chiaradia et al. (1997) concluding that leaded dust transported home on parents' clothing may contribute to enhanced home contamination and elevated child blood lead level. Additionally, a number of investigators conclude that proximity of residence to a lead smelter is associated with elevated child blood lead levels (Roels et al. 1978a & 1980; Walter and Yankel, 1980; Brunekreef, 1981; Graziano et al., 1990).

[18] Humans are not the only species demonstrating elevated tissue lead levels, Koh and Babidge (1986) finding even greater levels for tested Port Pirie dogs. Sutton and Hariono (1987) note elevated tissue lead levels in flying foxes (*Pteropodidae var. sp.*) residing near a lead smelter in Queensland, the lead deriving from contaminated fur and the species' regular grooming habits. Marino et al. (1990) report elevated blood lead levels for the pet dog of an old home undergoing restoration incorporating (leaded) paint removal, the animal having frequently been seen licking its dusty coat.

emissions, and contaminated food and water supplies. Whilst some difference regarding the ranking of the relative significance of these pathways exists between researchers (Elwood, 1986), there appears to have been a trend to consensus in the mid 1980s with atmospheric lead being relegated a minor role as a contributor to child blood lead levels (directly; ie acting more as a conduit), contaminated dust and soil being considered to be the dominant influences (Farfel, 1985; Body, 1988b amongst others listed in Section 1.2 above).

From work conducted in Port Pirie a decade ago, the potential hazard from ores, concentrates and process fume is suggested to be substantially greater than that from smelter slags given the last-mentioned's substantially lower bio-availability (Body, 1986b). Pathways of leaded material to the environment of Port Pirie encompass plant emissions from a variety of locations eg fumer or baghouse, stack emissions and possible escape from concentrate stockpiles (A. Gilbert, pers. comm.). This has been very much a strong part of the town's history (Body et al., 1988).

1.6 PORT PIRIE SMELTER DETAILS

Current production from the smelter is of the order of 200,000 tonnes of lead per annum (A. Gilbert, pers. comm.). Body et al. (1988) estimate that fugitive emissions from the plant over the period of its operation are of the order of 160,000 tonnes of lead, typically in the form of lead-bearing dust and fume. Additionally, unquantified amounts have been distributed into the atmosphere as a result of the entrainment of dust arising from uncovered concentrate stockpiles on the wharf, uncovered slag heaps (Landrigan, 1983) and also from the railway line corridor from Broken Hill (Body, 1986a). For the life of the smelter's operations, smelter waste products (typically slag) have been deposited nearby, typically to the West and South West of the smelter works, on previously unused, low-lying land, constituting land fill for land-reclamation and flood-buffering purposes. This activity has also contributed to the limited vegetation in the area.

Until 1996, the concentrate was shipped from Broken Hill to Port Pirie in open-topped railway rolling stock. Railway cars are now covered. Again until the end of 1996, it was normal practice for the zinc-rich concentrate to be stockpiled uncovered on the wharf, awaiting marine transport to a zinc smelter elsewhere. Storage is now maintained under cover. Current practice incorporates the storage of lead concentrate in an open pit area west of the plant.

1.7 LEAD AND HEALTH IN PORT PIRIE EARLIER THIS CENTURY

Health concerns with smelter workers in Port Pirie, typically blast furnace workers of recent (European) migrant descent, emerged at the turn of the century. Such concerns escalated in frequency and calibre between 1900 and 1925, despite management-claimed improvements in plant operation and contaminant control. Further, evidence of frank lead poisoning in a number of workers became increasingly apparent immediately prior to the 1920s. Such an increase in lead intoxication led, progressively, to a large increase in the number of employees at the smelter being reported as having become incapacitated due to this malady (Australian Government, 1925), warranting the 1925 (Australian) Royal Commission into Plumbism.[19]

Despite the implementation of some of the Commission's recommendations such as the introduction of dust-suppression measures, lead intoxication of smelter employees, as defined by a patient-demonstrated blood lead level exceeding a medically-determined (population) maximum threshold level (Chapter Four), remained a problem.[20]

1.8 AIMS AND OBJECTIVES

Cognisant of a (relatively) vague comprehension of the dust transport environmental scenario (Baker, 1984; Body et al., 1988a), particularly in a situation such as Port Pirie with its smelter presence, elevated (above normal background) lead-in-dust/soil concentration levels (Tiller et al., 1976; Body, 1986e) arid climate and dusty conditions and smelter-proximate housing, a much clearer understanding of a number of aspects of the contaminated dust with domicile relationship is therefore warranted. Certainly an arid climate is conducive to dust generation (Burge and Hoyer, 1990; Crockford, 1992), and also extremely concentrated localised environmental accumulation of contaminant material, particularly where low wind velocity and frequent thermal inversion limit pollutant dispersal opportunity (Landrigan and Baker, 1981).

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[19] The Royal Commission enquiry regarding lead-related health sequelae in Port Pirie was not the first such investigation in Australia. Some 14 years earlier, a Royal Commission investigating the health of miners (Australian Government, 1911), visited Chillagoe (north Gulf country of Queensland), in order to establish the relationship between miners' occupations and their general health status. Little was concluded, however, save that alcoholism was a greater problem than lead poisoning, and little action was taken by mine management to alleviate the perceived (by some) problem other than ensuring that the works were hosed down twice daily (Bell, 1993). Persistent doubts existed in the general community, however, and a second Royal Commission visited Chillagoe in 1930. Subsequent to their investigation, the Commissioners reported that there had been a long history of corruption and inefficiency by management. Such revelations, however, subsequently assumed centre stage, with comments regarding concerns with respect to miners' health appearing to have been relegated a lower priority (Australian Government, 1930; Bell, 1993).

[20] Currently, on occasion, despite frequent blood monitoring of employees' blood lead levels, some employees exhibit blood lead levels of four to five times the currently NHMRC-recommended level of concern. Such workers are typically temporarily transferred to other duties not intimately involved with the smelting process (pers. obs.).

Specific amongst such aspects, the aim of research work here is directed toward determining the distribution of within-home contaminant dust populations and their variability in, primarily, selected vacant Port Pirie houses, and investigating potential mechanisms of house entry of such dust. Thus, this proposed research for dissertation will focus on a number of somewhat discrete domains pertaining to contaminant dust mainly within the confines of the (vacant) domicile, including settled dust, settling dust, suspended dust and surface soil. These integral parts of the current study of vacant houses are listed in greater detail below, and include:

a) the acquisition of an appropriate sample of available vacant houses within the confines of Port Pirie to be used as sampling sites for a dust-sampling study. This will need to encompass houses in two principal high health risk areas of the township, in order to fairly represent the contamination of houses by lead-bearing dust at either location.

b) the development and testing of appropriate dust-sampling equipment and techniques to be used for this study.

c) The undertaking of a vacant home soil contaminant level study conducted within the confines of the home block.

d) The completion of a comprehensive "static" (principally) lead-in-dust contamination study, comprising a systematic assessment of the spatial variability in dust loading levels within rooms and between rooms of mainly floor-level dust samples in the test house suite.

e) The comprehensive and fairly long-term assessment of indoor "longitudinal" dust deposition in these test houses, the spatial (within-room and between-room) and temporal variability of contamination of the house environment by depositing dust being the focus of this component of the research.

f) The on-going monitoring of lead-in-air levels conducted both in the house and immediately outside, in order to provide information regarding the dimension of contamination within the house, and also the degree of penetration of lead-bearing dust into houses given varying atmospheric conditions.

g) Complementing d) & e) above, a contemporaneous air-lead and depositing-dust sampling study in order to investigate whether a relationship between the levels of lead in both environmental compartments may be established. Further, this component of the work will be conducted in a number of houses that have been de-dusted and thoroughly cleaned immediately prior to investigation, in order to investigate the contribution of such an intervention cleanup process to a reduction in house re-contamination rates.

These current levels of investigation appear complementary in that they contribute to a more comprehensive overall understanding of the dimension and calibre of current lead-in-dust contamination at home level in Port Pirie. In a general sense, such an ordered program of investigation is also intended to further enhance the comprehension of a currently poorly-understood process by which the infiltration of houses by contaminant dust occurs.

1.9 GENERAL METHODOLOGY

1.9.1 VACANT HOUSE SUITE - SITE SELECTION AND HOUSE DETAILS

1.9.1.1 SITE SELECTION

Given that a very small selection of vacant houses was made available for research purposes (approximately 16), limited choice could be made as to those which were to be used for this work and those rejected. Within such constraints, houses were selected so as to be as evenly located over the two test districts as was possible at the time, with all houses that had been vandalised or damaged during the occupant exiting stage or the subsequent house vacancy period being rejected.

1.9.1.2 GENERAL HOUSE DESCRIPTION

A preliminary inspection of eleven available appropriate vacant homes involved with the study revealed quite some similarity in dimension of block and house size, although some variation was noted with respect to layout of external structures including gardens, lawns, sheds, fences and open soil areas. The dwellings, however, displayed considerable similarity with respect to construction materials, house size and form and number of rooms. All were of a similar age, set back a similar distance from the road, and none of the houses were, or appeared to have previously been, air-conditioned.

The test houses all exhibited very similar room configuration, dimension and layout; essentially four similarly-sized main rooms, each with only one window. The four main rooms were symmetrically distributed either side of a central hallway which linked the front and rear doors linearly. This was by far the most common mode of house construction in the two studied districts. The front (street-facing) two rooms were always bedrooms, the lounge room being one of the two rear rooms. A kitchen and a bathroom was usually located at the back of the dwelling, either side of the central passageway. A list of the homes involved is indicated below:

TEST HOUSE ADDRESSES

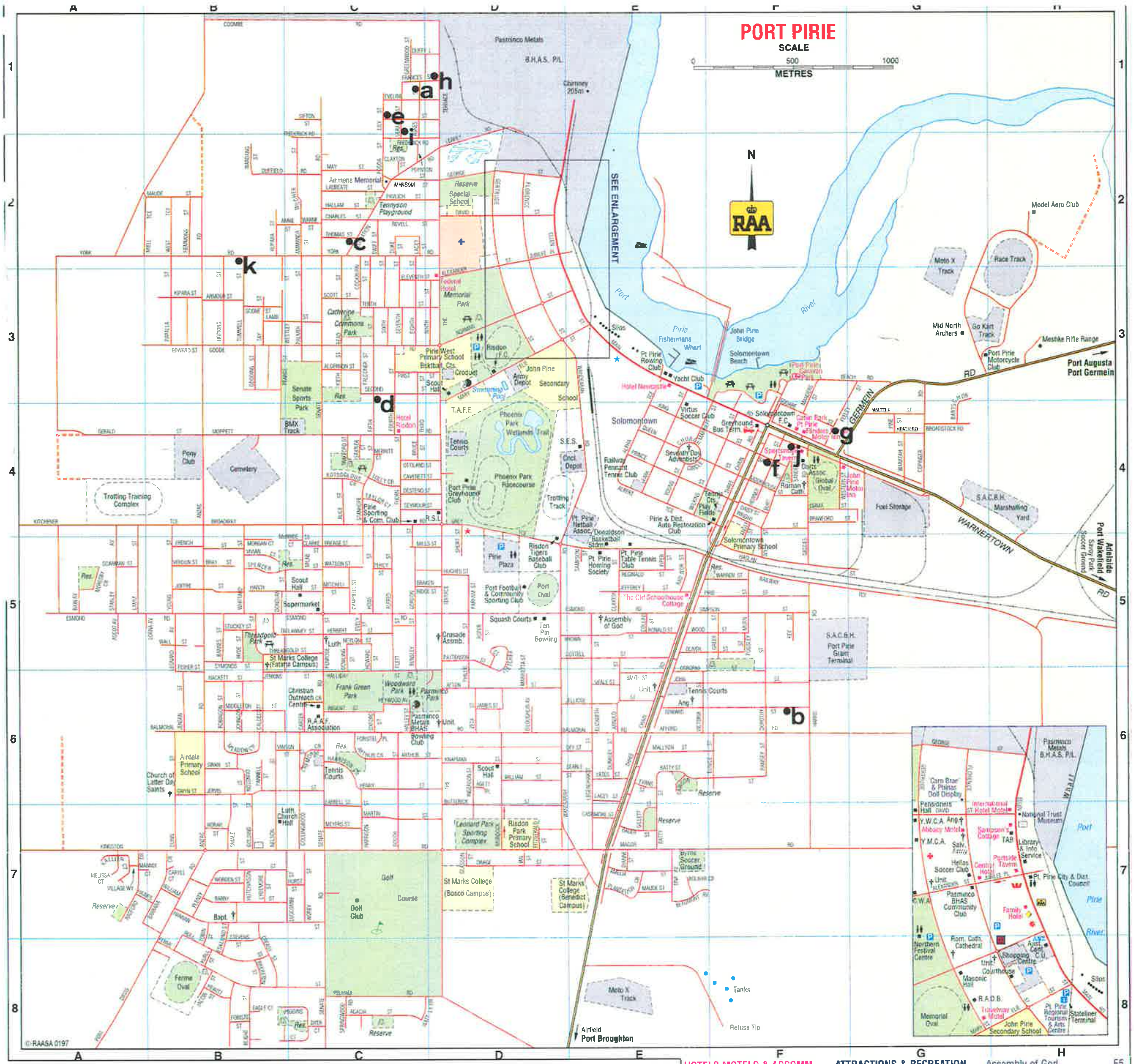
a) 6 Agnes Street
b) 64 Edward Street
c) 58 Federation Road
d) 29 Fourth Street
e) 19 Lily Street
f) 3 Sabine Street
g) 35 Square Street
h) 34 The Terrace
i) 27 Vera Street
j) 136 Warnertown Road
k) 97 York Road

Of the houses made available for this research, all but one (34 The Terrace) were of similar construction type. They were timber-framed with horizontally laid sheets of (old) galvanised iron constituting the external construction material including the roof (Plates One - Four, pages 28 & 29). Internal walls were constructed from either plaster-board or wood (panelling or boards). The dwelling at 34 The Terrace was constructed from brick and stone and was covered with a galvanised iron roof, and had an additional sun room attached to the rear, opposite the bathroom and kitchen. All dwellings reflected front entrances either directly facing or facing away from the smelter works, apart from one residence (97 York Rd), the eastern side of which directly faced the smelter.

All windows were located between 80 and 85cm above floor level, and were all similarly-sized sash-type with intact external insect screen coverings. The front and back doors were also very similarly sized, and also had undamaged screen outer doors. None of the houses reflected a separate dining room, the lounge room functioning as a dining room if a kitchen eating area was not available.

Given that there existed an initial concern regarding the contribution of degraded painted (internal) surfaces of the houses contributing to dust lead loading within the houses, all internal walls, ceilings and window surrounds of all test houses were tested for lead content. Utilising a rapid paint test (Dolan et al., 1984a; Dolan, 1986), virtually all painted (internal) surfaces demonstrated lead levels no greater than two percent, such surfaces mostly exhibiting less than detection limits. Upon inspection, there was no evidence of major peeling, flaking or chalking paint in any of the test house rooms. From dust speciation work conducted in the mid-late 1980s in a suite of Port Pirie houses, no demonstrable contamination of house dust by painted surfaces was detected (Body, 1986d).

More precisely than is indicated in Figure 1.3, the location of the suite of test houses selected for research [labelled a) - k) and corresponding to the above listing] is shown in Figure 1.5 below. All test houses are located between 0.4 and 3.3kms from the centre of the smelter's operations, south and west of the smelter.



HOTELS MOTELS & ACCOMMODATION
Abbacy Motel G7
Cabin Park Pt Pirie F4
Central Hotel G7
Family Hotel H7
Federal Hotel D3
Flinders Range Motor Inn F4
Hotel Newcastle E3
Hotel Risdon C4
International Hotel Motel G7
John Pirie Motor Inn G4
Port Pirie Caravan Park F3
Portside Tavern H7
Sampson's Cottage G7
Sportsmans Tavern F4
The Old Schoolhouse Cottage E5
Travelway Motel G8

GENERAL
Army Depot D3
C.W.A. G7
Cemetery B4
Council Depot E4
Courthouse H8
Fire Station D4
Golf Club C7
Greyhound Bus Terminal F4
Hospital D2
Library & Info. Service H7
Masonic Hall G8
Northern Festival Centre G8
Pasminco BHAS Com. Club G7
Pensioners Hall G7
Pirie Plaza D5
Police E3
Port Pirie City & District Council H7
Post Office B7,H7
R.A.A.F. Association C6
R.A.O.B. G8
R.S.L. D4
RAA District Office H7
RAA Service Depot F4
Refuse Tip F8
S.E.S. D4
Scout Hall C5,D3,D6
Shopping Centre H8
Stateliner Terminal H8
Supermarket C5
TAB H7
Toilets C6,D3,D5,F3,F4,G7,G8,H7
Y.M.C.A. G7
Y.W.C.A. G7

ATTRACTIONS & RECREATION
Carn Brae & Phinas Doll Display G6
Airmens Memorial C2
BMX Track C4
Byrne Soccer Ground E7
Croquet D3
Darts Association F4
Donaldson Basketball Stdm E5
Go Kart Track G3
Golf Course C7
Hellas Soccer Club G7
Meshke Rifle Range H3
Mid North Archers G3
Model Aero Club H2
Moto X Track E8,G2
National Trust Museum H7
Pasminco BHAS Com. Club G7
Pasminco Metals BHAS Bowling Club C6
Phoenix Park Racecourse D4
Pirie & Dist Auto Rest. Club E4
Play Fields E4
Pony Club B4
Port Football & Community Sporting Club D5
Port Pirie Greyhound Club D4
Port Pirie Motorcycle Club H3
Pt Pirie Rowing Club E3
Pt Pirie Homing Society E5
Pt Pirie Netball Assoc. E4
Pt Pirie Regional Tourism & Arts Centre H8
Pt Pirie Sporting & Community Club C4
Pt Pirie Table Tennis Club E5
Race Track H2
Railway Pennant Tennis Club E4
Risdon Football Club D3
Risdon Tigers Baseball Club D5
Senate Sports Park C3
Solomontown E4
Solomontown Beach F3
Solomontown Football Club F4
Squash Courts D5
Swimming Pool D3
Ten Pin Bowling D5
Tennis Courts C6,D4,E6,F4
Trotting Track D4
Trotting Training Complex A4
Virtus Soccer Club E4
Yacht Club E3

PLACES OF WORSHIP
Anglican Church E6,G7
Assembly of God E5
Baptist Church B7
Catholic Church F4
Catholic Cathedral G8
Christian Outreach Centre C6
Church of Latter Day Saints B6
Crusade Assembly D5
Lutheran Church C5
Lutheran Church Hall B7
Salvation Army G7
Seventh Day Adventists E4
Uniting Church D6,E6,G7,G8

FINANCIAL INSTITUTIONS
ANZ Bank H8
Aust. Central Credit Union H7
Bank SA H8
Commonwealth Bank H7
National Bank H7
Westpac Bank H7

EDUCATION
Airdale Primary School B6
John Pirie Secondary School D3
Risdon Park Primary School D7
Solomontown Primary School F5
Special School D2
St Marks College (Benedict Campus) D7
St Marks College (Bosco Campus) D7
St Marks College (Fatima Campus) B5
T.A.F.E. D4

PARKS & OVALS
Catherine Commons Park C3
Ferme Oval B8
Frank Green Park C6
Global Oval F4
Leonard Park Sporting Complex D7
Memorial Oval G8
Memorial Park D3
Pasminco Park D6
Port Oval D5
Tennyson Playground C2
Threadgold Park B5
Woodward Park C6

Source: RAA SA, 1997.

Figure 1.5 Locational map of Port Pirie including specific test house location.

a = 6 Agnes Street; b = 64 Edward Street; c = 58 Federation Road; d = 29 Fourth Street; e = 19 Lily Street; f = 3 Sabine Street; g = 35 Square Street; h = 34 The Terrace; i = 27 Vera Street; j = 136 Warnertown Road; k = 97 York Road.

1.9.1.3 SOURCE OF HOUSES

Vacant houses were kindly made available for research by the Public and Environmental Health Branch, South Australian Health Commission (SAHC). These previously-occupied houses derived from two districts in Port Pirie, namely Port Pirie West and Solomontown, both regarded as high health risk areas for children (Maynard et al., 1993; Figure 1.3).

1.9.1.4 POTENTIAL HOUSE DIFFERENCES

The soil at only one residence (27 Vera St) had been replaced to a depth of approximately 15cm as part of a lead abatement program in 1993, (P. Bartholomew, pers. comm.). At that time, intervention at the Vera St home also included roof and ceiling replacement, substantial inner wall replacement, thorough cleaning and de-dusting and fence replacement. None of the other homes were incorporated within the lead abatement program.[21]

This research work acknowledges concerns regarding a lack of controls for such variables as dwelling design, condition, construction materials and relationship and proximity of dwelling to currently-presumed major sources of contaminant dust release. Variability in dust movement resulting from weather, extent and effectiveness of former occupant cleaning protocol, and duration of house vacancy prior to commencement of research activity is also not controlled for.

1.9.2 RESEARCH WORK TIMING IN VACANT HOUSE SUITE

The first major component of investigation to be conducted in these homes was the static dust survey, commencing November, 1993. The maximum interim house vacancy period prior to study commencement approximated three months.

Subsequent investigations encompassed soil survey, dustfall study and air study. From commencement to completion of all field work, a period of two years and three months elapsed, subsequent to which, a number of the vacant homes were demolished.

1.9.3 SAMPLING MATERIALS AND METHODS

Specific detailed description of the equipment used for this research will be made in the appropriate chapters, and often, additionally, as part of an appendix. A very general description of the equipment utilised will be made here.

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[21] This approach to dealing with contaminated soil at home level first commenced in 1988 as part of the SAHC's lead-intervention program, there being no recorded indication of similar activity at any of the other test homes. Whilst home owners might have undertaken similar activity privately, given the cost of house renovations and/or old soil removal and new soil acquisition compared with property values, such action would not seem likely.

1.9.3.1 SOIL SAMPLING

Soil contaminant concentration was assessed using a portable X-Ray Fluorescence Detector (XRF) kindly made available for a limited time by Mike van Alphen, Environmental Health Centre, Port Pirie (Chapter Six).

1.9.3.2 SURFACE DUST SAMPLING

At time of writing, according to the literature, methods for representative sampling of house dust for evaluation of lead content have not been standardised, but should be (Bero et al., 1993; Farfel et al., 1994; Gulson et al., 1995; Clickner and Rogers, 1995). A number of workers use and have used different dust-collection techniques (Laxen et al., 1987a & 1988; Wixson, 1988), inferring difficulty in comparing results from different studies (Fergusson and Schroeder, 1985; Roberts et al., 1991a & b).[22]

For this study, surface dust was sampled over previously-determined surface areas with a dedicated vacuum pump and dust-sampling device (aerosol monitoring cassette with filter paper) adapted from related equipment used by other workers (eg Solomon and Hartford, 1976; Fergusson and Schroeder, 1985; Kim and Fergusson, 1993), with some modification (Chapter Seven, Appendices Six, Nine and Ten).

a) Vacuum pump

It was intended that this purpose-designed sampling pump be portable, robust, easy to use, self-contained, battery-powered, easily calibrated whilst in use, not excessively noisy, more efficient than previously used related equipment, and that it would contribute minimally to dust disturbance within the room/house in which it was being used. It is suggested that this is a more appropriate alternative to the suggested "only available (sampling) system if AC power is not available" (Gulson et al., 1995; p. 15).

b) Dust-collecting device

The (static) dust acquisition device was selected for its appropriateness for the task, convenience of use, relative durability, re-usability and low cost. The choice of specific filter used was based on extremely low residual levels of contaminants such as lead, and filter characteristics appropriate for selected chemical analytical processing of contents (Section 1.9.4 below).

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22 Sweeping sampling techniques (Harrison, 1979), wet wipe techniques (Sayre et al., 1974; Sayre and Katzel, 1979; Brunekreef et al., 1981; Diemel et al., 1981; Gallacher et al., 1984; Rabinowitz et al., 1985a & b; Farfel et al., 1994; Dewalt et al., 1995), sticky tape use (Lepow et al., 1974; Dolan, 1986b), petri dish sampling (Chiaradia et al., 1997) and brush and pan techniques (Rice et al., 1978; Angle and McIntire, 1979; Harrison, 1979; Watt et al., 1983; Duggan, 1984; Aschengrau et al., 1991) have been cited in the literature as have vacuum cleaner bag grab samples (Angle and McIntire, 1979; Thornton et al., 1980; Body et al., 1985; Sturges and Harrison, 1985; Thornton, 1988; Moffat, 1989; Jensen, 1992; Schulz et al., 1993; Gulson et al., 1994b) hand-held mini vacuum cleaners (and dedicated specialised pump and collector) samples (Aschengrau et al., 1991). Concerns arise with the reliability and/or consistency of such methods (Gulson et al., 1995). The equipment used here is proffered as a potential model for a standardised dust-collecting technique.

1.9.3.3 DEPOSITING DUST SAMPLING

As referred to in Chapter Eight and Appendix Sixteen, the literature indicates that a number of devices have been previously used for the purposes of collecting depositing dust, from a review of which, however, most were deemed unsatisfactory for the purposes of this study. Consequently, acknowledging that "more detailed testing of dustfall accumulation within a house is especially needed" (Gulson et al., 1995; p. 17), a new collecting device was designed by the researcher, considerations focusing on cost, ease of use, durability, accuracy and precision and appropriateness for this type of study.

1.9.3.4 AIR SAMPLING

Also designed by the researcher, is the air-sampling apparatus, the sampling equipment being similar to that used by some other workers (eg Davies et al., 1987b & c). The sampling pump used (240 volt AC-powered) is similar to that used for the surface dust study (12 volt DC-powered), and uses an identical dust-collecting device (Chapter Nine). Whilst in principle, air has been sampled from homes in a not dissimilar fashion elsewhere (eg Davies et al., 1987b & c), a number of different approaches were made by the researcher here in order to enhance the accuracy and precision of estimate of volume of sampled air. Specific design also aimed to minimise fluctuations in pumped air flow rate, substantially reduce construction costs and time required by the researcher to monitor equipment during operation, and also to minimise room disturbance during sampling.

1.9.4 ANALYTICAL METHODOLOGY

Reference in this text is often made to the use of ethanol, which in all cases is AR Grade pure ethanol (Univar, Ajax Chemicals, Regency Park, SA). Reference is also made to the use of water for analytical purposes, this being ultra pure "polished" water, which, unless otherwise indicated, refers to water as a chemical reagent (ie > 15meg Ohm), derived from a MilliQ reverse osmosis/super high quality water purification system (Millipore Corporation, Bedford MA). All samples analysed for metal content were initially acid digested using analytical grade reagent (Univar, Ajax Chemicals) concentrated nitric and hydrochloric acids (sequentially) according to a protocol referred to in Appendix One. Subsequent to this process, the solution containing the solubilised dust was analysed using various equipment as indicated below. It should be noted here that dust lead concentrations as elicited by acid digestion here actually refer to acid-available metal contents and not total metal contents of dust.[23]

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[23] The estimation of lead content of dust by extraction with (say) nitric acid however is certainly more appropriate than a number of other non-acid techniques (Noller and Bloom, 1975; Que Hee et al., 1986), and is much quicker than hydrofluoric acid extraction. Further, as a standard reference method (Long et al., 1979), it provides a very reasonable approximation to the total metal concentration (Clayton and Tiller, 1979), the difference between the two techniques diminishing as dust metal concentration increases, and appearing non-existent above concentrations of 500 - 1,000$\mu g.g^{-1}$ (K. Tiller, pers. comm.). Preliminary validation of the digest method comparing various digest techniques revealed minor differences (Appendix Two).

1.9.4.1 DEPOSITING DUST

Depositing dust samples were analysed for lead using a (flame) Atomic Absorption Spectrophotometer (AAS - Varian, Model Spectra 250 AA; Varian Aust., Melbourne) using an Acetylene/Air gas mix, demonstrating a minimum detection limit (for lead) of $\approx 0.06\mu g.ml^{-1}$ analysis solution (Anon, 1993a).[24]

1.9.4.2 STATIC DUST

Static dust samples were analysed for multiple elements using an Inductively Coupled Plasma Atomic Emission Spectrometer (ICP AES) employing de Galan nebulisation techniques (ARL 3580B IC; Fisons Scientific, Sydney), and demonstrating a minimum detection limit (for lead) of $\approx 0.05\mu g.ml^{-1}$ analysis solution. (J. Waters, pers. comm.).

1.9.4.3 AIR-DERIVED SAMPLES

Air derived samples and quality control (QA/QC) blanks (Appendix Three) were analysed for multiple elements using an Inductively Coupled Plasma-Mass Spectrometer (ICP MS), employing de Galan nebulisation techniques (Fisons/VG Plasmaquad Turbo Plus; Fisons Scientific, Sydney), demonstrating a minimum detection limit (for lead) of $\approx 0.005\mu g.ml^{-1}$ analysis solution (J. Wattling, pers. comm.).

1.9.5 ADDITIONAL ELEMENT DATA

The essential core of this work pertains to the role that the contaminant lead plays in a variety of home environmental compartments. Consequently, the vast bulk of statistical interpretation will be directed at this element only. In some instances, however, it will become necessary to expand or consolidate upon an observation on the basis of the use of multi-element data. Such multi-element data were available from soil, static dust and air-derived data suites for virtually all samples, and a small proportion of the dustfall-derived data suite (post-cleanup study only).

Typically the only other elemental data considered to any real extent will be zinc, another smelter discharge product, although copper and iron may also be considered on occasion.

Throughout analysis, a comprehensive quality control program was conducted, details of which are included in Appendix Three.

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[24] AAS techniques for analysis for lead were certainly the most commonly-chosen technique in much of the earlier lead studies (Noller and Bloom, 1975), and were utilised as standard reference methods (Long et al., 1979) their accuracy and precision constituting desirable analytical standards (Tinsley et al., 1983; Slavin, 1988; Soter et al., 1988). Limited multi-element facility and moderately long analysis time have, however, led to the use of other analytical equipment, as listed below, as the industry standard (J. Waters, pers. comm.).

1.9.6 DATA INTERPRETATION

1.9.6.1 RAW DATA APPEARANCE

Initial inspection (frequency histogram and box and whiskers plot format) of the ordered data for the various dust/lead parameter data suites corresponding to soil, static dust, dustfall and air revealed clearly-evident (considerably) positively "skewed" data distributions. This was also apparent for other element raw data. This non-normal distribution of data sets was also reflected by skewness of fit and kurtosis values, and also differences between arithmetic mean and median values. Normality testing utilising the normality test function of Statworks (Cricket Software, Heyden & Son Ltd, London) also revealed deviation from a normal distribution ($P \leq 0.05$).[25]

This confirmed other workers' similar observations regarding the propensity for (particularly) lead concentrations in house dust, air and soil samples to be log-normally distributed, and of the need to use log-transformation of the data to more closely resemble a normal distribution than does the raw data.[26]

Corresponding plots for (natural) log-transformed individual analyte data sets (dust loading, lead loading, dustfall, leadfall and lead concentration of soil, dust and air data) much more closely resembled normal distributions, confirmed by skewness of fit and kurtosis values and also normality testing with statistical package ($P \leq 0.05$). Further, differences between geometric mean and median values for tested elements were much less than was the case for analogous raw data, reflecting that such performed log-transformation of data returned the original data distributions to those much more effectively approximating normal distributions.

Thus, throughout, all subsequent statistical interpretation requiring normally-distributed data was conducted on (natural) log-transformed (raw) data, unless specified to the contrary.

Given the small total number of houses (eleven), and the number of variables corresponding to which data were collected, bi-variate rather than multi-variate statistical analyses were typically conducted on these data suites.

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[25] Where one discrete measurement is made upon each of a number of different specimens, "it is not uncommon to encounter log-normal distributions" (Miller and Miller, 1988; p. 38). Such log-normal distribution of data for any one set of samples, would appear typically skewed positively or towards the lower end, and is very common for environmentally-derived data (Duggan and Inskip, 1985).

[26] In a well-written review of previously-conducted home leaded dust investigations, Duggan and Inskip (1985) remark that many earlier researchers conducted statistical interpretation on raw lead in dust concentration data despite the very evident skewness of their data at the time, the potential for inappropriate conclusion being readily apparent. In fact, these authors report that since the distribution of lead in dust concentrations for any one set of samples is "typically skewed towards the lower end, there exists a strong case for using (medians or) geometric means", (1985; p. 20). Davies notes that dust lead frequency distributions are most typically positively skewed and approximate log-normal distributions, such that log-transforming the data will "result in a Gaussian or normal distribution" (1988; p. 25), the geometric mean value thus being the best estimate of central tendency (Johnson and Katz, 1970). This is consistent with the views of many researchers (eg Aitchison and Brown, 1966; Schubert et al., 1967; Archer and Barratt, 1976a; Duggan, 1981; Stark et al., 1982; Hartwell et al., 1983; Gallacher et al., 1984; Bornschein et al., 1985b & 1986; Davies et al., 1985, 1987a, b & 1990; Duggan and Inskip, 1985; Rabinowitz et al., 1985a; Elwood, 1986; Davies and Thornton, 1987; Harper et al., 1987; Laxen et al., 1987a & 1988; Raab et al., 1987; Culbard et al., 1988; Schilling and Bain, 1988; Schwar et al., 1988a; Davies and Ballinger, 1990; Farfel and Chisholm, 1990 & 1991; Cotter-Howell and Thornton, 1991; Kim and Fergusson, 1993; Farfel et al., 1994).

It is acknowledged here that the number of houses utilised for this study is not large, and does not necessarily represent a random selection of houses from Port Pirie (the selection criteria being essentially that these dwellings were the only vacant dwellings made available for research purposes). Acknowledging the remarkable similarity of most dwellings in the two study districts, and the fairly large sample size of specimens collected, however, such a suite of houses does demonstrate a very reasonable representation of houses in general in the study area, much of this work providing valuable information in the context of a pilot study.

1.9.6.2 STATISTICAL COMPARISON OF DATA

Given the normal appearance of log-transformed data sets generated here, comparison of data sets was typically conducted utilising Student's *t*-test unless otherwise stated. Whilst it could be suggested that difference registered at (say) the 95% confidence interval could constitute sufficient evidence to claim that two data populations differed statistically significantly (Gourevitch, 1965), such terminology was not strictly used here. Rather, the level at which difference was detected was presented as (say) $P \leq 0.05$, the inference being that such data set difference was more pronounced than difference at (say) $P \leq 0.10$. Similarly, no registered difference between data sets at the $P \geq 0.20$ level is more convincing an argument that data sets demonstrate no difference, than difference registered at (say) $P \geq 0.10$ level.

1.9.7 CONTEMPORANEOUS SAMPLING - POST CLEAN-UP STUDY

A component of the total research conducted here, was directed towards investigation of the re-contamination of some of the test houses by leaded dust subsequent to a very thorough cleanup of the house. For these four test houses, this involved intense vacuuming of floors, all horizontal surfaces and above-ceiling spaces using, exclusively, heavy duty high-efficiency Nilfisk vacuum cleaners (Nilfisk Aust., Concord West, Sydney, NSW) fitted with high efficiency particle collecting/filtering (HEPA) systems (as Farfel and Chisholm, 1991). Such filters purportedly remove 99% of small ($< 0.3\mu$) particles (Roberts et al., 1992). Also conducted was total home internal surface wet wiping/washing including all floor surfaces, and silicone sealing of any ceiling/wall/floor cracks between structural components. House external surfaces were also washed down. Throughout, workers wore clean clothes and maintained clean hands. Clean shoes were worn at all times, and no walking over cleaned floors occurred. Subsequent to cleanup completion, houses were closed, including both windows and doors, and no further entry was permitted until sampling commenced. The intention of this cleanup procedure was to simulate the earlier SAHC-undertaken intervention process in many (deemed contaminated) Port Pirie houses in the two high-risk districts constituting part of the earlier lead-intervention program.



Plate One Port Pirie test house: 58 Federation Road.



Plate Two Port Pirie test house: 19 Lily Street.



Plate Three Port Pirie test house: 27 Vera Street.



Plate Four Port Pirie test house: 97 York Road.

CHAPTER TWO LEAD THROUGH HISTORY: INTEREST AND INTOXICATION - AN HISTORICAL REVIEW

You will see by it that the Opinion of this mischievous Effect from Lead, is at least above Sixty Years old; and you will observe with Concern how long a useful Truth may be known, and exist, before it is generally receiv'd and practic'd on.

Benjamin Franklin

- from a letter written to Benjamin Vaughan in 1786 cited in McCord, 1953.

The Present is the living sum total of the whole Past.

Thomas Carlyle

2.1 THE EMERGENCE OF LEAD

Lead, the most abundant of the heavy metals in the Earth's crust (Aitchison, 1960), is historically reputed to be the first metal to have been beneficiated from its natural ores (Gowland, 1902; Krysko, 1979; Nriagu, 1983a). The element has variously been described as one of the seven metals of antiquity (Aitchison, 1960; Waldron, 1973). It has a recognised history of considerable human usage dating back well over five thousand years (Gowland, 1912; Patterson, 1971; Waldron, 1973; Nriagu, 1983b), with lead-containing artefacts distributed widely throughout the ancient world (Nriagu, 1983a). Recorded examples of lead beads have been noted in ancient civilisations such as the residents of Çatal Hüyük in Asia Minor ca ≈ 6,000BP, and Yarim Tepe in Iraq dating back a further thousand years (Gale and Stos-Gale, 1981).[1] The ability to refine metals from their ores by smelting would seem to suggest the watershed between the essentially agricultural society of the late Neolithic Age and that of the commencement of the Early Bronze Age as depicted in the Aegean region (Gale and Stos-Gale, 1981).

Given a relatively low melting point of 327.4^{0}C for the parent metal (and not usually found in the Earth's crust in this form - Aitchison, 1960), it would seem likely that the first casting of lead occurred by accident in the embers of a campfire or primitive hearth (Gowland, 1912; Tylecote, 1964; Davies, 1987). The more recent substitution of silver for lead particularly for jewellery, made lead ores potentially very valuable, given that lead sulphide (galena) ore could contain up to ten percent silver (Nriagu, 1983a).

---

[1] According to Cotter-Howells (1991), the oldest known piece of lead, discovered in Turkey, dates back to ≈ 6,500BP, although Krysko (1979) suggests that small beads of metallic lead have been found with other human-produced artefacts dating from 7,000 - 6,500BP in excavations on the central Anatolian Plain, again reflecting the metal's ancient historical emergence. Ancient alchemists believed lead to be the oldest metal, associating it with the planet Saturn, hence saturnine.

2.2 ORIGINS OF LEAD USAGE

In the ancient world, Egyptians used oxides of lead for cosmetics such as eye-paints and medicaments, and elemental lead for coinage, standard weights and in glass-making to increase the product's brilliance, colour, and ease of engraving, (Smith, 1986b; Davies, 1988).[2] In pre-dynastic Egypt, lead was also used as sinkers for fishing nets, ornaments, water containment and to inlay wood (Lucas and Harris, 1962; Waldron, 1973), frequent reference to lead's properties being made in hieroglyphics of around 3,500BP (Moore et al., 1977b). In fact, the use of lead for water reticulation systems stems directly from this era (Partington, 1935; Waldron, 1973). Reference in the Ptolemaic and later papyri to the "plumber" as one who "prepares and repairs water pipes and containers" also derives from this era (Partington, 1935; p. 82).

Romans used lead sheet for roof coverings and also to protect the keels of ships, and most of their cooking and eating utensils were made from pewter, which comprised, at that time, half lead and half tin. Roman production may have reached the peak figure of approximately 60,000 tonnes per year, the material also being used for water ducting, ceramic glazes, jewellery and coinage (Grandjean, 1975; Lin Fu, 1980; Nriagu, 1983a & b). Further uses of lead in this society included bronze and brass formulation, solder, figurines, vases, containers, writing tablets, burial containment of the dead, warfare projectiles, pigments for markers, paints and glazes, cosmetics and medicaments (Gilfillan, 1965).

In the early bronze era lead was often used for repairing broken pottery (Gale and Stos-Gale, 1981). Subsequent to the decline of the Roman empire, lead's use declined drastically, and did not re-emerge with any vigour until the commencement of the Industrial Revolution, around the latter half of the eighteenth century, subsequent to which, production escalated (Davies, 1988). Estimates of global lead production over time are reflected below (Figure 2.1).

Reference to the use and refining of lead in ancient literature is legion, eg in the book of Jeremiah (ca. 2,610BP), the Prophet, alludes to the process of cupellation,

> "The bellows are burned, the lead is consumed of the fire: the founder melteth in vain: for the wicked are not plucked away. Reprobate silver shall men call them, because the Lord hath rejected them"
>
> (Jeremiah, 6; 29 - 30),

and also from the Old Testament, lead is included in

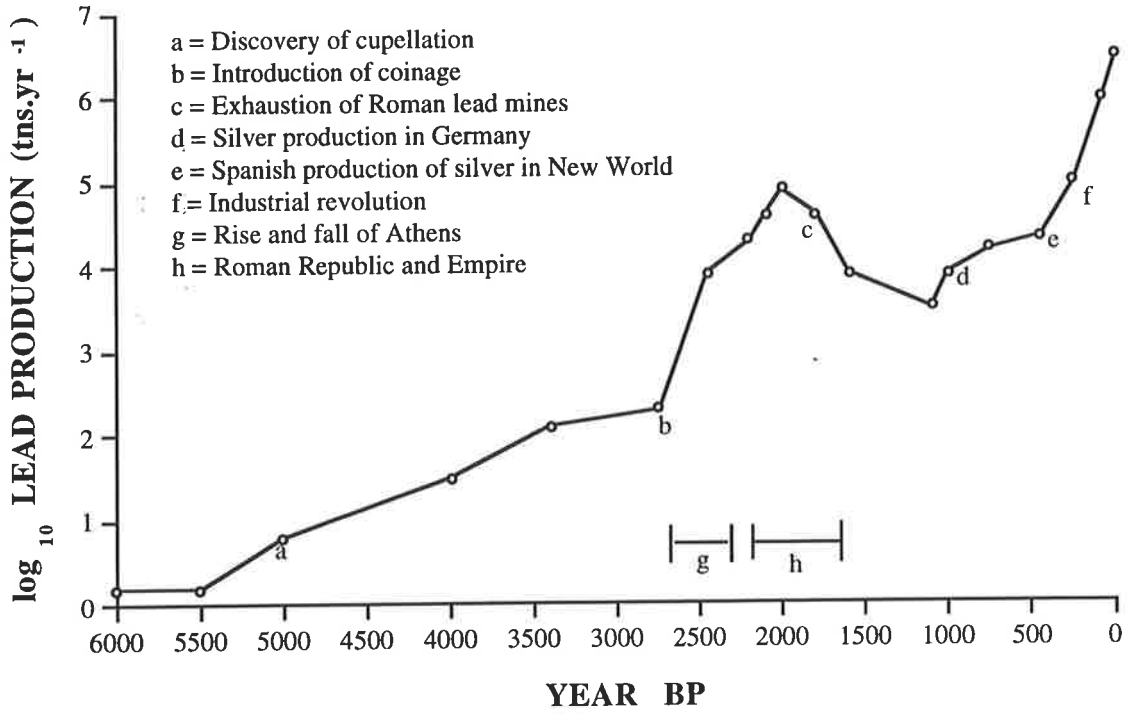
> "... the multitude of riches; with silver, iron, tin and lead which are traded in fairs"
>
> (Ezekiel, 27; 12).

---

[2] Initially, mined galena ore was used as an ornament or as an eye-paint (Partington, 1935; Aitchison, 1960), although its subsequent importance as a metal was relatively minor until the discovery of its natural association with silver (Waldron, 1973; Nriagu, 1983b). The process of silver-lead separation (cupellation) dates back again more than 5,000 years, probably to tribes living on the Black Sea coast near Pontus (Gale and Stos-Gale, 1981), following which both metals began to be used progressively more extensively for jewellery, ornaments and storage vessels. In fact, lead mines were frequently called silver mines due to the frequent mineral association with and importance of silver, and were valued highly (Aitchison, 1960; Waldron, 1973; Simms, 1984).

Further reference regarding the relative value of the metal for use as lead tablets for writing upon appears as:

> "Oh that my words .... were graven with an iron pen and lead in the rock forever!"
>
> (Job, 19; 23 - 24).



Source: Settle and Patterson, 1980.

Figure 2.1 World lead production over the last 6,000 years.

Even in such ancient times, lead appeared to have been valued as a useful material and often became eagerly sought after, literature reference to its value frequently appearing, eg in the book of Numbers at the time that Moses slew the Midianites (ca 3,450BP), it was recorded that he seized all their valuable metals,

> "... and the Lord commanded Moses (to take) only the gold and the silver, the brass, the iron, the tin and the lead"
>
> (Numbers, 31; 22).

From around 2,550BP, the Greeks mined and smelted lead intensively, strong Roman interest commencing between one and two centuries later (Drasch, 1988). Much of the wealth of ancient Athens derived from the infamous Laurion silver/lead mine, where the life expectancy of its slave workers in the smelter was significantly less than a year subsequent to commencing work (McCord, 1953a). Further, a high death rate of children of lead workers at the time was also noted (Athenaeus, second century AD).

At the time it was well known that the fumes given off by molten lead in its production were extremely toxic, but the use of slaves or prisoners for the purpose often precluded the contamination of the general population and the need to consider or adopt any occupational health and safety guidelines (Athenaeus, second century AD).

A significant proportion of the population of the Laurion region, however, probably also received unhealthy doses of lead (Hopper, 1968 and 1979). Smelting and cupellation took place on higher ground with lead dust/fume raining down on the township, mine tailings and slags being used on most roadways (Wertime, 1980). More-soluble than lead sulphide, lead oxide (litharge) was commonly employed as an aquarium cement in dams and ore-washing sites, thus contaminating large areas of the municipality (Wertime, 1980), and also the water supply of downstream communities (Hopper, 1968).[3]

Lead ornaments have been found in Cretan tombs and in the excavations of Troy (Gowland, 1912) and Tel Halaf (Aitchison, 1960) dating back, at these locations, for greater than five millennia. Ancient lead mines have been worked in Sardinia and Carthage, and also in Assyria where lead was used as a form of currency more than four millennia ago (Partington, 1935). The Sumerians of the early dynastic period of Ur (Mesopotamia; ca. 5,000BP) displayed incredibly high standards in the fine techniques of working lead, reflecting a considerable acquaintance with the metal (Aitchison, 1960; Gale and Stos-Gale, 1981). Lead tumblers and food-preparing utensils were also occasionally found in such antiquities.

2.3 LEAD'S INITIAL PRODUCTION INCREASE

Lead's use, however, reached pre-eminence with the Roman culture with its extensive use in water reticulation systems for their towns and houses (Nriagu, 1983a). In this context, lead was extensively used for roofs, gutters, downspouts, tanks, general household plumbing and repairs (Gilfillan, 1965; Waldron, 1973), and, notably, for the construction of the great aqueduct of Lyons.[4] Subsequent to the conquest of Britain by the Romans, as a result of the Romans' avidity for lead (Waldron, 1973), the native mines were extensively worked, reflecting the value of lead to the Romans (Nriagu, 1983a).

Whilst the extensive use of lead water-carrying systems (Tylecote, 1964; Nriagu, 1983b) would have represented a health hazard to both the Romans, as also to the Greeks (Drasch, 1982; Anon., 1992a; Pearce, 1992), it would appear that both cultures exposed themselves to a far greater health risk. Frequently their bronze or copper cooking vessels were coated with lead or a silver-lead alloy (stagnum), so-preventing the leaching (during cooking) of copper and copper salts such as oxides or sulphate (verdigris) from the pot, thus spoiling the taste of the food.[5]

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[3] The relative solubility of various lead salts is discussed in greater detail in the subsequent chapter.

[4] Here, almost 12,000 tonnes of lead were used for just one of the syphon units (Aitchison, 1960).

[5] It would seem that the practice derived originally from Greek culture (ca 2,150BP), perhaps two centuries prior to the Romans' affinity for such culinary endeavour (Gilfillan, 1965).

A remark from Pliny nearly two thousand years ago, commenting upon the use of such leaden pots for food preparation, embellishes this point,

> "When copper vessels are coated with stagnum, the contents have a more agreeable taste, and the formation of destructive verdigris is prevented."
>
> (150ADa; p. 48).

Such vessels were also found to be of great value to both Greek and Roman cultures in preparing wine and grape syrup, the latter being used extensively as a sweetening agent due to the virtual absence of sugar (imported from India) in such times (Gilfillan, 1965; Nriagu, 1983c). Essentially, the grape (or other fruit) juice was boiled down to perhaps half of its original volume in a lead-lined pot, the resultant *sapa* or *defrutum*, respectively, being exquisitely sweet (Waldron, 1973; Nriagu, 1983a & c; Emsley, 1987).

The sweet taste derived from the leaching of lead acetate from the vessel into the *sapa* or *defrutum*, the dissolution being enhanced by the cooking temperature and the highly acidic nature of the brew.[6] This product was subsequently added to wine or foodstuffs and acted as both a sweetener and a preservative, since like many heavy metals, lead acts as an enzyme inhibitor, thus preventing fruit souring and fermenting under storage (Waldron, 1973). Blended Roman wines utilising *sapa* and *defrutum* prepared in this fashion were abundant (Gilfillan, 1965; Nriagu, 1983a & c), and were commonly (and not inappropriately) known as *confusum* (Nriagu, 1983c).

Additionally, the use of *defrutum* for the sweetening and preserving of many dainty foods reserved particularly for the aristocracy also contributed to their enhanced lead dietary intake. Of the 450 recipes in the Apician cookbook, a collection of recipes from the first to the fifth century AD (Apicius, ca 1,400BP), 85 recipes called for the liberal addition of *sapa* or *defrutum* (Nriagu, 1983c; Emsley, 1987).

2.4 HEALTH EFFECTS ASSOCIATED WITH LEAD INTOXICATION

A detailed discussion dealing with the human health effects associated with undue lead exposure is conducted in Chapter Four. Included below is a more general, historically-oriented discussion of the health concerns evidenced by excessive lead exposure.

2.4.1 GREEK AND ROMAN CULTURES

The use of *sapa*-enhanced wines in Roman times was so prolific, particularly amongst the aristocracy (Pliny, ca 150ADb), that Pliny and others noted that non lead-rich wine was unusual,

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[6] See Chapter Three regarding solubility of lead salts.

"...genuine unadulterated wine is not to be had now, not even by the nobility"

(150ADa; p. 48).

Pliny also acknowledged that the consumption of high-lead wines was having a deleterious effect on the health of the population,

"From the excessive use of such wines arise dangling, paralytic hands"

(ca 150ADb; p. 58).

Such comments echoed Dioscorides, a Greek physician during the Hellenic period (ca 150AD), who referred to such lead-containing products as

"most hurtful to the nerves" and "lead makes the mind give way"

(Alexipharmaca V, No 9, cited in Waldron, 1973; p. 394).

Whilst it was clearly indicated at the time that the use of lead for cooking vessels was dangerous to the health with a variety of reported symptoms, the practice was extremely common (Gilfillan, 1964; Waldron, 1973; Nriagu, 1983a).[7] If the assumption at the time was that, if ignored, the associated health problems would go away, it was certainly not to be, various contemporary sources suggesting that lead poisoning was a major influence in the ruin of the Roman culture (Kobert, 1909; Gilfillan, 1964 & 1965; Mack, 1973; Nriagu, 1983a & b). It should be noted, however, that some opposition to such a view does currently exist (Waldron, 1973; Simms, 1984).

Apart from cooking vessels, lead-alloy pewter was also a commonly-used material for the production of drinking vessels and eating implements, and as a pigment for glazes typically for food preparing/storage implements, further enhancing the prospect of lead intake (Emsley, 1987). Lead was also commonly used for the manufacture of domestic paints. Due to their not insignificant cost, the use of many of these paints eg Pompeian red, was restricted mainly to the more well to do materially, and often used in association with water-reticulation systems (Gilfillan, 1965). Such "red lead" was manufactured from an oxide of lead (*minium* - Chapter Three), and was the favourite colour of the time.

Compounding lead exposure, the use of elemental lead for water reticulation systems was also extremely common and known to be harmful to the health at the time, yet the practice continued to prosper, as the Roman architect Vitruvius reported (ca 2,100BP),

> "Water is much more wholesome from earthenware pipes than from lead pipes. For it seems to be made injurious by lead because *cerusse* is produced by it; and this is harmful to the human body. Thus if what is produced by anything is injurious, it is not doubtful but that the thing is unwholesome in itself."[8]

(De Architectura VIII, No. 6; p. 4).

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[7] In fact, reports from between 2,200 and 2,100BP suggest that wine prepared and stored in lead-lined vessels led to sterility, abortion, headache and sleeplessness (Gilfillan, 1964).

[8] *Cerusse/cerussa* refers to hydrated lead carbonate; See Chapter Three.

The first of the ancients to clearly recognise lead as a cause of symptoms such as podagra (colic) and palsy (paralysis) was undoubtedly Hippocrates (ca 2,370BP; Alderson, 1852; Browning, 1961). It would appear, however, that the first unquestioned clinically-detailed account of lead poisoning due to the ingestion of lead was made by Nicander (Waldron, 1978), a Greek poet and physician (ca 2,150BP) who clearly referred to the link between lead and colic (abdominal pain), oedema (fluid retention and swelling), gout (joint inflammation), palsy, constipation and,

> "a decided pallor amongst lead workers, like the colour of lead"
>
> (Nicander, ca 2,150BP; p. 600).

Citing Nicander, Major further refers to and explains the visual disturbances and encephalopathy also associated with lead carbonate workers of the era, acknowledging the severity of such health sequelae,

> "The harmful cerussa, that most noxious thing
> Which foams like the milk in the earliest spring
> With rough force it falls and the pail beneath fills
> This fluid astringes and causes grave ills.
> The mouth it inflames and makes cold from within
> The gums dry and wrinkled, are parched like the skin
> The rough tongue feels harsher, the neck muscles grip
> He soon cannot swallow, foam runs from his lip
> A feeble cough tries, it in vain to expel
> He belches so much, and his belly does swell
> His sluggish eyes sway, then he totters to bed
> Phantastic forms flit now in front of his eyes
> While deep from his breast there soon issue sad cries
> Meanwhile there comes a stuporous chill
> His feeble limbs droop and all motion is still
> His strength is now spent and unless one soon aids
> The sick man descends to the Stygian shades."
>
> (1939; p. 11).

In the first century AD, the Greek physician Dioscorides described the symptoms associated with the ingestion of lead monoxide as follows,

> "The drinking of litharge causes oppression of the stomach, belly and intestines, with gripping pains; it suppresses the urine, while the body swells and acquires an unsightly leaden hue."
>
> (ca first cent. AD; p. 31).

Further, Dioscorides and Galen (early second century AD, [Drasch, 1982]) report details of lead colic after the cosmetic use of cerusse and litharge for dyeing hair, a common practice of the time. Not long thereafter, Pliny described red-lead and lead acetate as,

> "deadly poisons, and not to be used medicinally"
>
> (ca 150ADb; p. 54).

Further, he described the vapours associated with the manufacturing processes of the metal as,

> "being clearly dangerous to humans and all animals especially dogs"
>
> (ca 150ADc; p. 31).[9]

The ancients were quite well aware of the dangerous character of lead and the plumbism-associated symptoms such as colic, gout and palsy. However the Greeks and the Romans still continued to expose themselves to the effects of the harmful metal through their food and drink, resulting from time to time in epidemics of lead poisoning throughout (particularly) the Roman world (Gilfillan, 1965). It would seem then that, whilst acknowledging the toxic properties of lead *per se*, both cultures tried to dissociate lead's health hazard from the very necessary/desirable use of the metal in their culinary activities (Southwood, 1986).

In a general sense, the populations apparently failed to relate symptoms of sub-lethal lead intoxication with lead's ingestion through culinary activity. As an occupational hazard as encountered in, for example, lead mining, however, the effects were only too well known (McCord, 1954; Hunter, 1969).[10]

Of the Julio-Claudian emperors whose consumption of lead via their diet could best be described as *ad libitum*, it has been suggested (Nriagu, 1983b) that Tiberius (14 - 37AD), Caligula (37 - 41AD), Claudius (41 - 54AD), Nero (54 - 68AD), Galba (68 - 69AD), and Vitellius (69AD), to name a few, were significantly affected by lead, such that their behaviour and health could be linked to severe lead intoxication, pandemic of the times (Nriagu, 1984). The tradition of a lifestyle characterised by decadence and over-indulgence seemed to have been passed on from generation to generation (Emsley, 1987), many displaying classic symptoms of frank lead poisoning, with personalities and political decisions adversely influenced (Nriagu, 1983a).

On the brighter side, Elagabalus (218 - 222AD) a noted profuse drinker, squanderer, pleasure-seeker, gourmand and a purported mad person showed some sanity when he chose not to have children "in case one of them turned out to be thrifty" (Birley, 1976; p. 310).

2.4.2 MORE RECENT CULTURES AND HISTORICAL LINKAGES

As with reports from the Greek and Roman eras, through subsequent history, many references have been made to (particularly) the podagra (gout) symptoms associated with lead ingestion, yet the practice of using lead or lead-containing alloys for food-preparing equipment still continued for many centuries (Emsley, 1987). Gout, "the daughter of Bacchus and Venus" (Galen, ca first century AD, cited in Schnitker, 1936), had also long been associated with lead ingestion and considered symbolic of either occupational hazard or the libertine lifestyles of the leisured class whose members brought their grief upon themselves through their excesses (Bywaters, 1962; Ball, 1971; Nriagu, 1983b; Wedeen,

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[9] Dogs, as horses, are particularly susceptible to lead-induced pharyngeal/laryngeal spasm and paralysis.

[10] It could be argued that in a somewhat analogous yet Machiavellian and curious way, such a philosophy may even prevail today regarding the very necessary employment opportunity offered to residents of (say) a smelter town versus the well-known health-threatening effects of the metal.

1984; Nriagu, 1986). Litharge-sweetened wine *ad gustum* was often the culprit, both in ancient and more modern times.

Such sweetened wine was popular in Britain and France in the late eighteenth century, as was cider prepared similarly, with colic and gout again diseases of the more affluent. The working class, however, was not spared either. As a result of similar preparation procedures for less-costly cider, and the particularly acidic nature of that brew yielding a strongly tart taste, cider required considerable sweetening, again, often with lead monoxide, although resulting in a lower final concentration of lead in the beverage (Emsley, 1987). Consumption from a pewter tankard, common at the time, would only enhance lead intake.

The indictment of lead regarding colic and also gout was well recognised in the eighteenth century when the former was referred to in William Buchan's Domestic Medicine,

> "The nervous colic prevails among miners, smelters of lead, plumbers, and the manufacturers of white lead. It is very common in the cyder counties of England, and is supposed to be occasioned by the leaden vessels used in preparing that liquor. It is likewise a frequent disease in the West Indies where it is also termed the dry gripe."
>
> (cited in Wedeen, 1984; p. 121).

In terms of their bibulous habits, there would appear a common vein running through the gouty aristocrats of the Roman Empire and the gouty nobles of eighteenth and nineteenth century Britain. Certainly both assuaged prodigious thirsts with (often) profuse quantities of lead-containing wines. English port was very popular, and frequently very heavily-fortified by vintners with lead monoxide as a sweetener and preservative (Nriagu, 1983a). The incidence of port consumption and gout both peaked in the eighteenth and nineteenth centuries in Britain (Emsley, 1987).

Typically, according to Wedeen, the gouty patient displayed,

> "an exquisite taste with an enlarged understanding including the finest Portuguese wines"
>
> (1984; p. 101).

Lead poisoning was common in spring time in Germany, when people traditionally prepared sauerkraut in glazed pottery containers, the highly-acidic nature of the preparation leaching much of the lead from the glaze (Emsley, 1987). Curiously, however, in Germany in the seventeenth century, the addition of litharge to wines was directly connected with colic and gout and thus banned, the practice being decreed a capital offence. By contrast, some eighteenth and nineteenth century British recipes recommended litharge for the improvement of home-made wines, with lead-containing wines still being bottled in Britain earlier this century (Nriagu, 1983a).

Complicating the situation further, despite the recognition of the association of wine with gout, liquor was often recommended in the treatment of the very symptoms of gout. Punch, often similarly prepared, was a noted remedy for gout (Wedeen, 1984). Often a mixture of basic lead carbonate (white lead) and sulphur was recommended to treat tuberculosis. Lead acetate (sugar of lead - *saccharum saturni*) and opium were frequently administered to halt diarrhoea and internal haemorrhage, and to treat gonorrhoea (Emsley, 1987). A plaster containing a mixture of lead oxide, linseed oil and herbs was concocted around 30AD and was used for treating skin complaints, it still being used commercially

in Birmingham in the 1890s (Wedeen, 1984). Called diachylon plaster, it was eventually removed from sale due to the scandal which arose as a result of the removal of the cream from the plaster and its ingestion as an abortifacient (Emsley, 1987).[11]

As long ago as 1767, the Queen's physician George Baker showed lead-in-wine to frequently be the culprit in cases of gout and colic, linking such symptoms with those of occupational illnesses displayed by painters of the day and often their families also. He pondered on how these and other children might be afflicted and concluded that it occurred as a result of chewing toys covered in paint (Nriagu, 1983a).[12]

During the eighteenth and nineteenth centuries, the use of lead-alloy pewter vessels for cooking and drinking and the storage of foodstuffs and acidic beverages was common practice in England. Lead-containing crystal and lead-glazed pottery was also common, with lead water pipes also frequently found (Thackrah, 1832), as they are still today in some areas (Anon, 1992a; Pearce, 1992).[13] In Victorian England, plumbers and white lead manufacturers often succumbed to chronic lead poisoning, as Thackrah noted,

> "Plumbers are exposed to the volatilised oxide of lead which rises during the process of casting. Men who are much in this department are soon affected with colic and palsy"
>
> (1832; p. 23),

and

> "The manufacturers of white lead are subjected to its poison both by the lungs and by the skin. Men soon complain of headache, drowsiness, sickness, vomiting, griping, obstinate constipation, and to these succeed colic or inflammation of the bowels, disorders of the urinary organs, and finally, the most marked of the diseases from lead, palsy. We observed the muscles of the forearm, more frequently and sooner to suffer than other parts"
>
> (1832; p. 26).

Pottery glazers, typically women, often suffered colic, gout, miscarriage, infertility and even paralysis and blindness. Conditions did not improve until the end of the last century when lead contents of glazes were significantly reduced as with pewter, paints and plumbing in the earlier part of the twentieth century (Emsley, 1987, Landrigan, 1990). Ramazzini, often considered the parent of modern occupational medicine, refers to industrial lead poisoning in potters and portrait painters,

> "In almost all cities there are other workers who habitually incur maladies from the deadly fumes of metals. Among these are the potters. When they need roasted or calcined lead for glazing their pots, they grind the lead in marble vessels. During this process, their mouths, nostrils, and the whole body take in the lead poison. First their hands become palsied, then they become paralytic, splenetic, lethargic and toothless"
>
> (1713; p. 13).

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[11] In Roman times, *sapa* and onion made an effective abortifacient (Gilfillan, 1965), the acidic nature of the onion presumably enhancing the solubility and absorption of lead.

[12] This would appear to foreshadow comments by the Queensland medical practitioner Gibson (1892 & 1904) who has frequently been cited as the first person to connect leaded paint with child lead intoxication (Berney, 1993).

[13] In these areas, tap water is suggested to be by far the major source of child lead exposure for an estimated four million homes, particularly with soft plumbo-solvent waters (Anon, 1992a), tap water from leaded pipes constituting the main single source of population lead exposure in much of Britain today (Pearce, 1992).

Painters were frequently associated with symptoms likened to frank lead poisoning (Drasch, 1982) such as palsy, frequent headache, insomnia, blindness and mental disturbance extending to insanity, from which the expression "Crazy as a painter" no doubt derives (Gilfillan, 1965; p. 53). Again, Ramazzini notes the symptoms in painters,

> "I have observed that nearly all the painters whom I know, both in this and other cities, are sickly. For their liability to disease, there is an immediate cause. I mean the materials of the colours that they handle and smell constantly, such as red lead, cinnabar (mercuric sulphide) and white lead"
>
> (1713; p. 15).

Thus, not only were mine/smelter workers recognised as being likely to suffer from the effects of lead poisoning, but also those associated with such industries as painting, plumbing, roofing, sculpturing, pewterware-manufacture, ship-building, enamelling, and pottery-glazing (Nriagu, 1983a). Plumbism was also recorded as a disease associated with actors through the use of lead-pigmented hair dyes (Drasch, 1982). Regarding the incidence of occupational plumbism in Paris during the early nineteenth century, of the 1,213 patients admitted into the Hospital of Charity suffering from lead colic, approximately 490 were employed in the manufacture of lead compounds, 390 were painters, 61 were lead refiners, 14 were plumbers, and around 20 worked in shot- or glass-making factories (Tanquerel des Planches, 1839).

Ceteris paribus, one of the puzzling features of the British Empire in the eighteenth century appeared to be the relative vigour of the industrial revolution despite the purported idiocy of the ruling class, as suggested by some (eg Emsley, 1987). Such a situation has often been compared with the situation in Rome some one and a half millennia earlier, the consumption of lead being indicted for health decline (Nriagu, 1983b). Analogously, the contribution of lead to the health decline of ruling members of the British Empire has certainly often also been cited as a cause (Nriagu, 1983c). As an example, Nriagu (1983a) cites King George III who was well known for his predilection for very sweet port, and (highly-acidic) lemonade and sauerkraut, both prepared in heavily lead-glazed crockery, and who eventually went blind and insane.

Despite recognition of lead's association with ill-health for at least two and a half thousand years and despite its acknowledgment as a major occupational poison (Legge, 1897), its use has been, and still is, in ascendancy (Nriagu, 1988).[14]

Lead-associated colic, gout, palsy and related symptoms appeared to have been frequently acknowledged and suffered from amongst (particularly) the ruling class in various earlier-day societies. The less well-off socially or the more abstemious of the citizenry, however, were apparently spared the extreme symptoms of such maladies, yet lead's culinary or vinous association still continued into the current century.

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[14] World demand for lead has increased dramatically this century (Southwood, 1987), particularly more recently, as reflected by an increase of 25% between 1970 and 1990, reaching a record demand (OECD, 1992); See Chapter Three.

2.5 SKELETAL LEAD LEVELS

As discussed in detail later (Chapter Four), bone is the ultimate human repository for systemic lead (Landrigan, 1990), storing perhaps 90 - 95% of human-absorbed non-excreted lead (Barry and Mossman, 1970; Drasch, 1982). Thus skeletal remains provide a good historical record of the extent of environmental contamination and the levels of human exposure in specific societies (Settle and Patterson, 1980; Jaworowski et al., 1985; Emsley, 1987; Patterson et al., 1987). Hence, determination of the lead concentrations of skeletal remains offers the potential to compare average quantitative lead burdens from present day with those of inhabitants of earlier centuries or millennia. Under certain circumstances, however, lead may be leached from bones during fossilisation, therefore levels in very old bones, except perhaps for unusually well-preserved samples, cannot necessarily be regarded as representing the life-time concentration of the individual (Waldron, 1982; Jaworowski et al., 1985).[15]

Fergusson (1986) suggests that despite difference between workers regarding actual levels of lead found in ancient skeletal remains, there would appear to be reasonable agreement on a lead content difference between ancient and modern skeletal remains approaching two orders of magnitude. This is consistent with Mushak's (1992) estimate of a 300-fold increase in body lead burden with the transition from prehistoric to modern humans, the data of Shapiro et al. (1975) and Grandjean et al. (1979) concurring.

The next section (2.5.1) makes some (brief) reference to skeletal lead contents providing some insight into the degree of environmental exposure to lead through recent history.

2.5.1 ANCIENT CIVILISATION SKELETAL LEVELS OF LEAD

Studies of the skeletal remains of ancient or pre-historic (between 1,500 and 5,000 years previously) Nubians (Grandjean et al., 1979) and Peruvians prior to the industrial use of lead (Ericson et al., 1979; Settle and Patterson, 1980) have revealed levels of lead in bone and teeth 20 (Drasch, 1982), 100

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[15] Concerns must arise as to the potential for variation in skeletal lead concentration as a result of post-mortem alterations with time (Drasch, 1982; Jaworowski et al., 1985). Patterson et al. (1987) report that concentrations of metabolic lead in buried ancient bones may be obscured by the replacement of calcium in apatite by excessive amounts of soil moisture-dissolved lead. Moore (1986) acknowledges that some controversy surrounds the technique of bone lead evaluation, pointing out that the concentrations of lead in such bones are greatest at the bone surface. Barry and Connolly (1981) also note that a significant correlation between lead in skeletal bone and lead in proximate (grave) soil exists, Waldron (1982) concurring and suggesting (as the former workers) very strongly that lead may exchange between soil and bone. Skeletal remains maintained above-ground in sealed/semi-sealed crypts or vaults, however, would tend to minimise the effects of soil/skeletal lead exchange (Emsley, 1987; Kosugi, 1988). Inter-study comparisons may also be flawed to some degree by variability in skeletal lead concentrations depending upon the particular skeletal component chosen and the location of the sampling site (Jaworowski et al., 1985), and also the (biological) age of the individual(s) chosen (Drasch, 1982). It is not the theme of this dissertation, however, to investigate the significance of such changes and differences other than to accept that the distribution of skeletal lead in ancient and contemporary remains may not be uniform or fixed. This may render it somewhat difficult to make skeletal lead level comparisons obtained by some different studies of different populations and erae other than on the basis of significant differences in the measured skeletal burdens of lead (Jaworowski, 1985).

(Shapiro et al., 1973b & 1980; Grandjean et al., 1975 & 1979; Ericson et al., 1979), 500 (Patterson, 1965; Ericson et al., 1979; Settle and Patterson, 1980) to 1,000 times (Patterson et al., 1987) lower than those found in contemporary urban populations.[16]

Similarly, 5,000 year old teeth from Nubia exhibited lead levels which were very similar to 500 year old teeth from Greenland, current day Greenland resident teeth exhibiting between 10 and 100 times the aforementioned lead content (Grandjean and Jørgensen, 1990).

Since no use of lead by these first-mentioned two cultures is apparent (Grandjean et al. 1975 & 1979), skeletal lead concentrations or burdens from these times can be assumed to be taken as the "physiological zero point" (Drasch, 1982; p. 199).

2.5.2 EARLY TO PRESENT CIVILISATION SKELETAL LEAD LEVELS

2.5.2.1 EUROPEAN CIVILISATION

A comparison of (European) Roman citizen skeletal lead levels continuing for the 500 year Roman and Merovengian periods (Jaworowski et al., 1985) and contemporary urban humans has indicated a near equivalence (Patterson et al., 1987).

This appears reasonably well supported by other work suggesting a Roman civilisation level of approximately half that of today's urban residents (Drasch, 1982; Jaworowski et al., 1985).[17]

In the intervening period, however, there would appear to have been a fairly sharply-defined rise in skeletal lead burden approximately coinciding with the period late Middle Ages (ca 300 - 400BP) to Industrial Revolution (ca 300BP) in (particularly) European region dwellers (Waldron, 1973; Nriagu, 1978b; Emsley, 1987).[18]

This skeletal lead burden increase approaching one order of magnitude continued from the Middle Ages peaking approximately two and a half centuries ago, and remaining stable until the end of the nineteenth century (Jaworowski et al., 1985; Emsley, 1987). In a general sense, this increase has been associated with a significant increase in the smelting and overall use of lead by various societies, skeletal burdens reaching a peak level not dissimilar to that of Roman Britain inhabitants (Jaworowski

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[16] By comparison, Cleymaet et al. (1991), in a comparative study of the lead content of surface enamel of children's teeth conducted in Belgium (first world urban residents) and Kenya (third world rural residents), suggest that the mean lead level in the surface enamel of the Belgian children is of the order of eight to ten times that of their Kenyan counterparts, such difference reflecting Drasch's "Civilisation Poison" (1982; p. 227). In a similar sense, from another study, Patterson et al. (1991) again indicate a difference of closer to 1,000 fold comparing mean body (skeletal) burdens of lead between Pre-Colombian American Indians (ca 1,000BP) and modern urban-dwelling Americans.

[17] This is in sharp contrast to levels for Roman bones from England, however, for which almost tenfold-greater mean levels have previously been reported (Mackie et al., 1975; Ahlgren et al., 1981). These remains derive from a period of great lead extraction from native mines and associated smelting activity (Aitchison, 1960; Nriagu, 1983a; Jaworowski et al., 1985). France, also a member of the Roman Empire at the time, was spared the saturnine "blessing" of the Roman lead technology of the era due to its paucity of lead ores. Correspondingly, their residents' skeletal remains of the time exhibit significantly lower lead burdens.

[18] Middle Ages' European mean skeletal lead levels increased sharply reaching extremely high levels, due to sources of contamination in humans' immediate environment. These included pewter tableware and kitchen utensils, lead water pipes, lead-glazed pottery, contaminated food, cosmetics, paints, drugs, etc., which, in the past, caused occasional epidemic poisonings (McCord, 1954; Grandjean, 1975), most of which (although not all [Mahaffey, 1978]) having been eliminated as of the post 1960s period (Grandjean et al., 1979).

et al., 1985). Substantial skeletal bone lead evidence does suggest that lead poisoning has occurred not only in ancient times, but also more recently (Shapiro et al., 1980; Aufderheide et al., 1981). Certainly evidence from European skeletal remains exists, suggesting that chronic lead intoxication was also significantly contributing to ill-health well before the industrial revolution eg skeletal remains of Pope Clement II would seem to indicate that he died from lead poisoning nearly 1,000 years ago (Gilfillan, 1965).

This elevated lead burden in the European population reflected the unprecedented cultural, social and economic development of that time. A much slower decline to present day levels was experienced, however, reflecting (some) improvement in the control and understanding of adventitious lead intoxication (Settle and Patterson, 1980). This arose as a result of improved hygiene (Settle and Patterson, 1980; Emsley, 1987), and due to the disappearance of many of the old household and medical sources of exposure (Mahaffey, 1978; Jaworowski et al., 1985; Pearce, 1992). Drasch (1982) comments somewhat to the contrary, suggesting that current skeletal lead burdens, arising predominantly as a result of the profligate use (and environmental discharge) of tetra-ethyl lead as a gasolene additive, are twice the level of Middle Ages' and Roman era residents. As Jaworowski et al. (1985) note, such comments are, however, to be strongly criticised for the conclusions reached, such critique being based upon some researchers' use of inappropriate experimental controls.

2.5.2.2 JAPANESE CIVILISATION

In keeping with this suggestion of an observed increase in average skeletal lead levels with an increase in industrial activity including lead smelting (ca Roman Britain and Industrial Revolution Europe [Loane, 1938]), data from Japan suggest that the elemental composition of ancient human ribs through the periods from the Jomon era (5,000 - 2,300BP), to the Edo era (300 - 120BP) and to contemporary bones reflects an approximate ten-fold increase in skeletal lead levels in the Edo era (Kosugi et al., 1986 & 1988). This was a time when the town of Edo (now Tokyo) was highly polluted by environmental lead (Kosugi, 1988; Table 2.1) .

These changes in Japanese bone lead concentration over time reflect lead's use and popularity for that culture. In Japan, people first traded and used lead in limited quantities commencing in the eighth century AD, litharge being used for flint glass, jewellery, paints and glazes (Tanaka, 1978). Its use rose only very gradually until the introduction of guns in the sixteenth century, however, when lead was used for bullet manufacture.[19] In the Edo era, the demand for silver obtained from the cupellation of lead increased dramatically resulting in a further increase in consumption of lead, with lead-containing face powder a very popular and widely-used commodity (Kosugi et al., 1986).

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19 Few deposits of lead were to be found in Japan, Japanese people first trading in lead, to any great extent, no earlier than perhaps four centuries ago (Kosugi et al., 1988).

Table 2.1 Concentration of elemental lead in excavated Japanese bones.

	JAPANESE HISTORICAL ERA (Years Before Present)				
	JOMON	**YAYOI**	**KOFUN**	**KAMAKURA & MUROMACHI**	**EDO**
	5,000 - 2,300	**2,300 - 1,700**	**1,700 - 1,400**	**700 - 500**	**300 - 120**
MEAN LEAD LEVEL ($\mu g.g^{-1}$)	1.2	1.3	1.9	1.7	12.2

Source: Data from Kosugi et al., 1986.

2.6 LEAD INTOXICATION IN MODERN TIMES: A BRIEF HISTORY

This section includes a brief review of the twentieth century events leading to the emergence, over the last three decades, of the current view that lead poisoning constitutes "a major public health issue" (Rabin, 1989; p. 1,668). This echoed comments of Davis and Svendsgaard (1987), Grant and Davis (1987) and Nriagu (1988). The US Surgeon General claimed in 1991 that "lead is the most widespread environmental health hazard facing young children in modern American society" (Sullivan, 1991; p. 3). This was strongly supported by Landrigan and Curran (1992) and Mushak (1992), Urbanisation has thus been associated with an increased prevalence of elevated body lead burden (Landrigan, 1990).

Nearly a century ago, colic in early childhood was suggested to occur as a direct result of exposure to heavily-leaded sweet-tasting exterior paint used on Queensland (Aust.) house verandahs (Jacobs, 1995). This was considered to arise as a direct result of flaking-paint ingestion, such paint already having a long-recognised (informal) association with elevated blood lead levels and lead poisoning in children (Gibson, 1892 & 1904). Subsequent legislation severely limiting the use of leaded paint on verandah railings and outside surfaces within reach of children's fingers (The Lead Paint Law; [CQB BMA, 1922]), was enacted in 1922 (Rabin, 1989).[20]

[20] Lockart Gibson, a Queensland medical practitioner, first recognised the problem in 1892, which he termed "the toxicity of habitation", (Turner, 1897; p. 475). A. J. Turner, an Australian physician, reported (1897) persistent CNS impairment of severely lead-poisoned children. Other countries that either banned the use of leaded paint indoors or severely restricted children's contact with it were France (1920), Greece (1922), Tunisia (1922), Belgium (1926), Great Britain (1926), Sweden (1926), Poland (1927), Spain (1931), Yugoslavia (1931) and Cuba (1934), (Roels et al., 1978; Rabin, 1989). Whilst such legislation in these countries had been recognised by the US College of Physicians, discussion of such legislation in the US legislature was notably absent (Cushing, 1934; Fee, 1990). In an effort to reduce this exposure source for children, legislation was introduced in the US in 1971 (the Lead-based Paint Poisoning Prevention Act, [Billick, 1983]) restricting the use of high-lead paints in residential locations (Jacobs, 1995). Enacted more than 75 years after the toxicity of leaded paint had been demonstrated, and over the relentless objections of the lead industry (Reich, 1992), the Act was revised and amended several times, subsequently incorporating abatement programs. Implementation by the Federal Department of Housing and Urban Development, however, was considered "listless" (Mauss, 1994; p. 412). Farfel and Chisholm (1990) report on efforts to abate lead-based paint exposure for greater than 15 million US children as having largely failed, sadly noting that lead-based paint over 22 years old had been sighted on retail shelves in 1985.

Consequent to this, numerous cases of childhood lead poisoning were diagnosed by practitioners in the 1920s and 1930s, despite physicians' lack of familiarity with the symptoms, and also the imprecision of diagnostic tools (Rabin, 1989). Gastrointestinal or nervous system-demonstrated symptoms of lead intoxication were well recognised earlier this century (Shie, 1924). Also recognised were copious examples of occupational exposure eliciting the malady such as mining of oxides, carbonates and sulphates, rather than sulphides. This was due to the much poorer solubility of sulphides in gastric juices (Hamilton, 1925a). Less than a century ago, the rate of chronic lead poisoning in lead metallurgical plants was frequently very high (Nriagu, 1983b); the frequency of lead poisoning amongst employees of a lead smelter in Schemnitz, Hungary exceeding 70% (Hamilton, 1914). Other examples of occupational exposure included storage battery manufacture, enamelling and glazing works, leaded paint manufacture, printing, glass manufacture and the brass industry (Hamilton, 1925a). From such examples, the recognition of the pivotal role of dust in lead intoxication arose (Hamilton, 1924). As early as the 1930s it was recognised that lead intoxication was the most common single cause of poisoning in children in the USA (Ross and Brown, 1935).

Despite health concerns, high-lead paint was extensively used in the 1930s and 1940s in many countries (Moore, 1986), and was still being sold in some countries (including Australia) in the 1960s and occasionally, even the 1970s, eg the USA (M. van Alphen, pers. comm.). Gibson (1904) noted that children were exposed to lead through house paint, house dust and dirt, and that there appeared to be a seasonal variation in lead intoxication, with summer peaks. He referred to a propensity of some children to place non-food items in their mouths and also to (normal) mouthing of (presumably lead-contaminated) fingers (Lin-Fu, 1980). Such an observation, sadly, remained dormant or forgotten for the next sixty years (Rabin, 1989). The same could be said of similar comments made in 1767 by the Queen's physician George Baker (Section 2.4.2).

During the early part of this century, increased mortality amongst smelter workers due to lead-related diseases was investigated and noted internationally, nationally and locally in the early history of Pt Pirie (Australian Government, 1925; Bell, 1993). Such investigation precipitated little change to smelter operating practice and also working conditions for smelter workers for at least the first half of this century (Rabin, 1989; Landrigan and Curran, 1992; Bell, 1993).

Associated with the on-going and somewhat cavalier use of lead in paint into the 1950s in both the USA and Australia (Berney, 1993), there was a dramatic increase in the use of lead in gasolene and a whole host of consumer products (Landrigan and Curran, 1992), such that the annual consumption of lead in the USA almost doubled between 1940 and 1977 (Berney, 1993). Allied with this jump in the consumption of lead, Bradley et al. (1956) claimed that 90% of (7 - 60 month old) low-income area children presenting at baby clinics exhibited blood lead levels exceeding $30\mu g.dL^{-1}$, approximately one quarter of which displaying blood lead levels of $60\mu g.dL^{-1}$ or greater. At this time, $70\mu g.dL^{-1}$ lead in blood was reported as a level of concern for children since, at this level, the emergence of a variety of symptoms associated with lead poisoning were deemed to occur (Lin-Fu, 1982b).

Substantial research into the health effects of lead was undertaken almost solely by Robert Kehoe (representing the tetra-ethyl lead industry), who published widely, and was the United States' most vocal and influential scientist working on lead hazards for nearly 40 years (Graebner, 1988). Further details regarding the key role played by Kehoe with lead and public health is included in Appendix Four.

From the 1950s through to the 1970s, research into the hazards of lead flourished, with the results of such research demonstrating that:

a) lead in the environment arises as a result of human use of lead in industry,
b) lead accumulates in humans proportional to the amount of lead found in the environment,
c) lead is absorbed by the body from the environment,
d) such absorption as measured by blood, urine, faecal and other tissue lead levels, is an indication of exposure and poisoning, and
e) various environmental media represent exposure sources eg soil, air, dust, water and food

(Graebner, 1988; Berney, 1993).

Progressively, more attention seemed to be directed toward the lead problem particularly in the 1960s and in the USA (Needleman, 1988), focusing upon old, chalking and deteriorating (flaking or peeling) leaded house paint in, particularly, low cost housing areas (Rabin, 1989; Berney, 1993). Many researchers (eg Byers, 1959) demonstrated that prolonged and repeated exposure to lead was typically associated more frequently with significant neurological damage rather than large single exposures to lead (Berney, 1993). Community lead screening programs began to emerge in the 1960s, with often, the discovery of large numbers of individuals affected with intoxication (Berney, 1993). Further, large numbers of children exhibited blood lead levels exceeding $40\mu g.dL^{-1}$, most revealing no symptoms of overt lead poisoning (Lin-Fu, 1979). Permanent CNS impairment resulting from lead intoxication elicited heightened concerns regarding the environmental exposure of this metal in the 1970s (Berney, 1993), particularly for children who, contemporary research indicated, demonstrated more effective gut lead absorption than adults (King, 1971; Alexander, 1974). The blood lead concentrations at which level researchers were claiming that lead elicited a definite adverse effect on haem production fell progressively during the 1960s and 1970s to blood lead levels as low as $5\mu g.dL^{-1}$ (Berney, 1993).

In 1970, the US Surgeon General defined undue lead exposure for children as a condition registered by a blood lead concentration of $40\mu g.dL^{-1}$, at a time when 45% of children screened in New York City exhibited a blood lead concentration greater than this value (US DHEW, 1970). This shifted the emphasis on lead intoxication from treatment of overt lead poisoning to its prevention through mass screening and an attempt to reduce hazardous exposure by modifying the child's environment (Rabinowitz, 1995). Such measures included decontamination strategies involving the removal of lead sources such as contaminated soils, house dust and old lead-based paints, although some evidence existed suggesting that intervention would lower blood lead levels to some extent, but really only during or immediately after the period when the work was being carried out (Charney et al., 1983). From a review of a number of lead smelter studies during the 1970s (eg Yankel et al. [1977] and Landrigan et al. [1975 & 1976]), EPA USA (1977) concluded that people living in the vicinity of a major industrial lead source were exposed to abnormally high lead concentrations, and that subjects living closer to smelters exhibited greater blood lead concentrations than those living further away.

From lead isotope research (Garibaldi et al., 1975; Manton, 1977), Manton determined that from 7 - 41 percent of blood lead deriving from lead in air, arose from leaded fuel.[21] ATSDR (1988a) concluded that, overall, burnt leaded fuel was responsible for 90 - 95 percent of lead in air in the USA.

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[21] Berney (1993) claims a figure of half of children's blood lead on average derives from gasolene lead.

Blood lead levels tracked air lead levels very closely (Billick et al., 1979 & 1982; Annest et al., 1983; Figure 3.6), such results confronting Kehoe's earlier comments. Further, such data, at different times, elicited in the USA, Britain and in the (then) Federal Republic of Germany, substantially-reduced legally-enforced gasolene lead maximum levels, reductions commencing in the mid 1970s in the USA and Germany, (Jost and Sartorius, 1979; EPA USA, 1986; Rabin, 1989; Berney, 1993). Gasolene lead consumption fell in the USA by 73 percent between 1975 and 1984, with a similar reduction in lead in air levels over that same time period (ATSDR, 1988b; Figure 3.5), mean blood lead levels over the first five years falling by 37 percent (NCHS, 1984).[22] Ten years before the British reduction in motor spirit lead maximum level, an identical reduction was promulgated in Germany, eliciting an air lead level reduction of between 20 and 65 percent corresponding to rural and inner city sampling stations respectively (Jost and Sartorius, 1979). Such reductions in lead in fuel associated with variations in the isotopes of lead introduced into leaded fuel at manufacture facilitated some tracking of these isotopes in fuel, air and blood, resulting in conclusions that these isotopes in blood varied similarly to those in (leaded) fuel (Garibaldi et al., 1975; Manton, 1977; Facchetti, 1985).

Progressively, during the 1970s and 1980s, evidence began to accumulate suggesting that even low-level lead (asymptomatic) exposure eliciting a blood lead level of the order of perhaps 15 - 35$\mu g.dL^{-1}$ was sufficiently high to be associated with irreversible (Needleman et al., 1990; CDC, 1991), subtle deterioration of the developing CNS, registering as impairment of mental development (Bellinger et al., 1984; McMichael et al., 1985, 1986 & 1988; Needleman et al., 1979 & 1990; Needleman, 1986 & 1988; Needleman and Bellinger, 1988). Much of these data derived from a series of longitudinal epidemiological studies conducted in Boston, Cincinnati, Cleveland and Port Pirie. The results of these are discussed in more detail in Chapter Four. Even more recent work has suggested that neuropsychological damage occurring from early childhood exposure (pre and postnatally) may be elicited at levels closer to 10 - 15 $\mu g.dL^{-1}$ (Dietrich et al., 1989; Lyngbye et al., 1990; Needleman et al., 1990; Bellinger et al., 1991; CDC, 1991; Pocock et al., 1994; Tong et al., 1996; McMichael, 1997 - Chapter Four). Further research in the 1980s indicated a substantially greater prevalence of chronic kidney disease and anaemia with associated increased mortality amongst residents of a large lead-mining town in the USA (EPA USA, 1986), heart disease also being noted (Moore, 1986).

Whilst mean blood lead levels in the USA appear to have decreased in the mid 1980s (IPCC, 1991), commensurate with decreasing lead in air levels (Fergusson, 1986), as also in many other countries, the threat to children from lead intoxication still remains high, however, given data suggesting the detection of neurotoxic effects at ever-decreasing blood lead levels. Lead still remains widespread in the workplace today, Landrigan and Curran (1992) noting that over two million US adults are, occupationally, risking severe exposure to lead. Such people work in smelters and foundries, or are involved in construction, demolition, painting, battery-making, microelectronics, stained glass-making, shipbuilding, automobile radiator manufacture and exhaust repair (Landrigan et al., 1981; Anon, 1985; Gowland et al., 1987; Andrén et al., 1988; Landrigan, 1990; Rossol, 1992).

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[22] Australia would seem to have been slower to adopt lead in motor spirit reduction strategies than many other developed countries, a national maximum level of lead in fuel of 0.3$\mu g.L^{-1}$ being elusive, despite reference having been made to the effect that such lead had affected the health of 25% of Australia's children (EPA NSW, 1993). In 1976, W. Germany reduced its maximum fuel lead level to 0.15$\mu g.L^{-1}$, less than 25% of the current SA maximum level. The use of gasolene lead declined in many countries well before Australia (Dabelstein et al., 1987; EPA NSW, 1993).

Despite earlier, expensive lead-abatement work, today, as in previous years, lead-based paint remains the major source of childhood lead exposure and poisoning in the USA (ATSDR, 1988a; CDC, 1991). An estimated two million urban-dwelling, white American children aged from six months to five years (the largest single group of children in the USA), most of whom are in contact (residentially) with leaded paints, demonstrate a blood lead concentration greater than $10\mu g.dL^{-1}$ (Landrigan and Curran, 1992), the current US level of concern (Rosen, 1992). Interestingly, this threshold level has progressively decreased by a factor of 16 over the last 60 years (Figure 4.1).

Contributors to a net decline in mean blood lead levels, apart from leaded gasolene use, are the phasing out of lead-soldered food-containing cans, replacement of leaded water pipes/reticulation systems, more care taken regarding the removal of leaded paint, and increased legislation controlling environmental discharge of lead-bearing wastes (Fergusson, 1986). Despite such activity, lead exposure still represents the current environmental health problem that potentially affects the largest number of American children (Berney, 1993). Such comments embellish the results of earlier work by others (eg Mushak and Crocetti, 1989), these authors providing an estimate of potential health risk to children in the USA (Table 2.2). The serious threat to public health appears to continue.

2.7 IN SUMMARY - LEAD TODAY AND IN THE FUTURE

Lead poisoning may be regarded as one of the first occupational diseases contracted by humans many millennia ago. For more than 3,000 years, lead-related symptoms and mortality have been recorded, with the knowledge of the impact of lead on health evolving over this time, yet paradoxically lead would appear more commonplace today than ever before. Current, detailed awareness of the health concerns associated with the societal use of lead became increasingly apparent nearly a century ago, with extensive scientific investigation commencing by 1930 (Kinnison, 1976).

Currently, although of decreasing frequency, occasional frank lead poisoning still occurs, typically as a result of occupational exposure (SRC, 1990).

Following an attempted reduction in lead-emitting sources during the 1970s in the USA, a decline in average blood lead levels was observed, almost certainly due to the reduction in the use of leaded gasolene (EPA USA, 1986; SRC, 1990; IPCC, 1991; Berney, 1993), contemporaneous lead-paint abatement programs being largely unsuccessful (Farfel and Chisholm, 1990; Landrigan, 1990). Lead as a building, fabricating, industrially- and domestically-used metal is obviously well-entrenched in our society, its near future manufacture from raw materials being virtually guaranteed by today's demand for the metal. Thus, we will need to be vigilant in order to reduce the environmental distribution of this metal by way of waste material discharge, and to attempt to eliminate, as Chisholm, "a wholly preventable disease" (1971; p. 23).

Table 2.2 Estimated numbers of children potentially exposed to sources of lead in the USA, 1984.

SOURCE	DATE OF CONCERN	NUMBER OF CHILDREN * (millions)
Lead-based paint	Hazards reported in the literature since the 1920s. Reestablished and accepted in the USA by the end of the 1960s. Banned (@ > 0.06%) by CPSC in 1978. Still present in housing units.	12.0 ‡
Lead from gasolene	First regulated in the USA in 1973. Controversy dates to 1925, when lead additives were introduced. Analysis of New York City screening (1979) and NHANES II (1982 & 1983) data confirmed contribution of gasolene lead to blood lead.	5.6 †
Stationary & industrial sources	Nineteen Seventies	0.2
Dust/soil	Some work done in 1977. Follow-up increased concern in 1984.	5.9 - 11.7
Water/ plumbing	Ban on lead solder in water pipes and systems in 1986. Proposed reduction of maximum contaminant level from 50 to 20$\mu g.dL^{-1}$ in 1988.	10.4
Food	In 1970, 90% of food cans contained lead solder; in 1986, 20% of cans contained lead solder.	1.0

* These numbers are not additive because children are usually exposed to multiple environmental sources of lead.

‡ Number of children estimated to be living in unsound lead-painted housing in 1984.

† Number of children under seven years of age living in the largest 100 standard metropolitan statistical areas (SMSAs).

Source: Mushak and Crocetti, 1989.

Having discussed the historical context of excessive lead exposure, it appears appropriate to now discuss the physico-chemical properties of lead and its distribution within our environment (Chapter Three) and the detailed impact of lead upon human health in a contemporary setting (Chapter Four) prior to embarking upon a discussion of the field work conducted.

CHAPTER THREE LEAD; THE ENVIRONMENT AND EXPOSURE PATHWAYS

Lead is a material different from all other materials, a metal which you feel is tired, perhaps tired of transforming itself and that does not want to transform itself anymore: the ashes of who knows how many other elements full of life, which thousands upon thousands of years ago were burned in their own fire.

Primo Levi, 1975

Lead is toxic wherever it is found, and it is found everywhere.

ATSDR, 1988a

3.1 LEAD: PHYSICO/CHEMICAL PROPERTIES

3.1.1 ELEMENTAL LEAD DETAILS

Elemental lead, (Pb), a soft, easily malleable and dense metal (d = 11.342gm.cm^{-3} @ 25^{0}C [Moeller, 1952]), is highly lustrous when freshly cut, tarnishing easily and displaying a shiny blue-white silvery lustre in its pure form (Smith, 1986a). It occurs extremely rarely in this format naturally (Hammond, 1985), or not at all (Moeller, 1952). The surface of the pure form of lead is stable in dry air, however, in moist air, it is quite readily oxidised, resulting in the production of a surface coating of lead monoxide. This dull, blue-grey coloured lead coating is moderately reactive in the presence of atmospheric carbon dioxide, producing lead carbonate (Anon., 1976). For a metal, lead displays a relatively low melting point (327.4^{0}C, [Aitchison, 1960]), fuming commencing at around 500^{0}C (Muskett et al. 1979; Boiling point ≈ 1,770^{0}C, SRC [1990]), and a fairly high resistance to corrosion. These desirable characteristics have elicited a history over the last 7,000 years of its harvest and use as a pure metal and in alloys such as pewter (Chapter Two).

Industrially, lead has been employed as an opacifier in paints, typically as lead carbonate, chromate, sulphate and various oxides (Raab et al., 1987; van Alphen; 1993), in glazes and crystal glass where high refractive indices produce aesthetically-pleasing optical effects (DHSS, 1980; Ferguson, 1986; Simms, 1986). Organo-lead compounds such as tetra-methyl and tetra-ethyl lead demonstrate desirable thermodynamic qualities warranting their extensive use for more than sixty years as motor spirit octane-enhancing (anti-knock/pre-ignition) agents (Kehoe, 1961a). Lead has also been used in compounds in creams and cosmetics and folk medicines such as Azarcon and Greta (DHSS, 1980; CEH & CAPP, 1987; Mushak and Crocetti, 1990; Alperstein et al., 1991; Wixson and Davies, 1991).[1]

[1] Notable examples of this are hair-darkening formulations, (DHSS, 1980), eye cosmetics from India, eg surma having a lead content of up to 80%, and some unusual medical aids such as nipple-protecting cream for breast-feeding women. Currently in Saudi Arabia, some traditional folk remedies and cosmetics containing high concentrations of lead are still being used despite awareness of potential health effects (Anon., 1995a; Rabinowitz, 1995).

3.1.2 LEAD SALTS

3.1.2.1 SOLUBILITY UNDER LABORATORY CONDITIONS

Depending upon chemical species present and given appropriate chemical conditions, lead will quite readily form a number of compounds based upon reaction with other chemical species. Lead salts such as acetate, chlorate and nitrate are reasonably water soluble, although the chloride is only slightly soluble, and salts such as the carbonate, chromate, phosphate, sulphate and sulphide are virtually water insoluble. Lead carbonate, chromate, nitrate, phosphate and sulphide are quite acid soluble, with the chloride demonstrating only slight acid solubility (Hammond, 1985a).

Quite some variability in water solubility of lead salts under fixed conditions is apparent as reflected by Table 3.1.

Table 3.1 Solubility of various salts of lead.

LEAD SPECIES	SOLUBILITY IN PURE WATER (pH = 7)	COMMON/MINERAL NAME
Lead (elemental - Pb)	Insoluble	
Lead acetate [$Pb(CH_3COO)_2$]	443g.L^{-1} @ 20^{0}C	Sugar of lead
Lead carbonate [$(PbCO_3)_2.Pb(OH)_2$]	1.1mg.L^{-1} @ 20^{0}C	Cerrusite/White lead
Lead chloride ($PbCl_2$)	9.9g.L^{-1} @ 20^{0}C	Cotunnite/Mimetite
Lead chromate ($PbCrO_4$)	0.058mg.L^{-1} @ 25^{0}C	Crocite
Lead nitrate [$Pb(NO_3)_2$]	377 - 565g.L^{-1} @ 20^{0}C	-
Lead (mon)oxide (PbO)	17mg.L^{-1}@ 20^{0}C	Litharge/Greta
Lead (di or per)oxide (PbO_2)	0.81mg.L^{-1} @ 20^{0}C	Plattnerite
Lead tetroxide (Pb_3O_4)	Insoluble	Minium/Red lead/ Saturn red/Azarcon
Lead silicate ($PbSiO_3$)	Insoluble	-
Lead sulphate ($PbSO_4$)	42.5mg.L^{-1} @ 25^{0}C	Anglesite
Lead sulphide (PbS)	0.86mg.L^{-1} @ 20^{0}C	Galena
Sodium Chloride (NaCl) (by comparison)	354g.L^{-1} @ 20^{0}C	Common Salt

Source: data derive from Moeller, 1952; IARC, 1980; Howe, 1981; Sax, 1984; SRC, 1990.

3.1.2.2 LEAD'S SOLUBILITY IN VARIOUS WATER BODIES

The relative water solubility of various lead salts in the environment will depend upon the particular lead species, the relative "hardness" of the water in which it is to be dissolved (ie the quantity of other dissolved salts present, particularly salts such as calcium carbonate [lime]), the pH (acidity) of the water and the water's temperature (Morse et al., 1979b). In general, the softer the water, the lower the pH (more acidic), and the warmer the water, the greater the solubility of lead salts.

3.1.2.3 HARVESTING FROM ORE

Pure metal is obtained by extraction from its ores, the most common of which is lead sulphide (galena), which demonstrates a density of $\approx 7.5 gm.cm^{-3}$. Not uncommonly, lead may also be found naturally as an oxide such as lead monoxide and lead tetroxide, a carbonate and a sulphate (anglesite) (Anon, 1985a).

Given the relatively low melting point, lead is quite easily extracted from its ores by smelting, although concentration of the ore typically precedes smelting. Lead's prodigious use over this period of time (Section 3.2 below) has long been associated with its acknowledged toxicity (Chapters Two and Four).

Element number 82, a Group IV-B metal on the periodic table (Howe, 1981), it has been termed a "heavy metal" in reference to its physico-chemical attributes, other such metals being cadmium, copper, mercury and zinc (Sax, 1984; p. 2,641). Elemental lead displays an atomic weight of 207.2 (Anon., 1976), and can be found as any of four stable isotopes, namely:

a) ^{204} Pb
b) ^{206} Pb
c) ^{207} Pb
d) ^{208} Pb.

The last three arise as final products of the radioactive decay of Uranium (U) and Thorium (Th), respectively, ^{238} U, ^{235} U and ^{232} Th, although the origin of ^{204} Pb appears uncertain, as it would seem that it is not the product of any radioactive decay series (IARC, 1980).

In the known decay series, there exist four, mostly unstable, radioactive isotopes of lead, namely:

a) ^{210} Pb
b) ^{211} Pb
c) ^{212} Pb
d) ^{214} Pb.

The first of these has a half-life of 22 years, the remainder having much shorter half-lives (Anon., 1985a).

Lead demonstrates two stable oxidation states, Pb^{2+} (divalent) and Pb^{4+} (tetra-valent), the divalent state being by far the more common in the inorganic chemistry of lead, with the tetra-valent state being commonly demonstrated in organo-lead compounds, typically lead additives to gasolene (Moeller, 1952).

3.2 CURRENT PRODUCTION AND ENVIRONMENTAL EMISSIONS

3.2.1 PRODUCTION

Precise data regarding current global lead production/consumption from either the beneficiation of ores or concentrates and also from recycling processes appear difficult to attain and reflect differences between information sources. A reasonable estimate would be of the order of four to six million tonnes produced annually (Table 3.2), consumption not yet having stabilised (Settle and Patterson, 1980; OECD, 1992).

Table 3.2 Estimate of recent world lead production.

YEAR	QUANTITY ($x10^6$ tonnes.yr^{-1})	SOURCE
1978	5.2	Settle and Patterson, 1980
1980	5.2	Brown et al., 1984
1981	5.5	World Bureau of Metal Statistics, 1983
1985	4.1	Nriagu and Pacyna, 1988
1989	5.8	ILZSG, 1991a
1990	5.7	OECD, 1992

Source: data derive as indicated.

Australia produces between nine and ten percent of world lead output (ALI, 1992), approximately 40% of which derives from Port Pirie.

By comparison, lead production figures for the ancient world are included below (Table 3.3).

Table 3.3 Lead production in the ancient world.

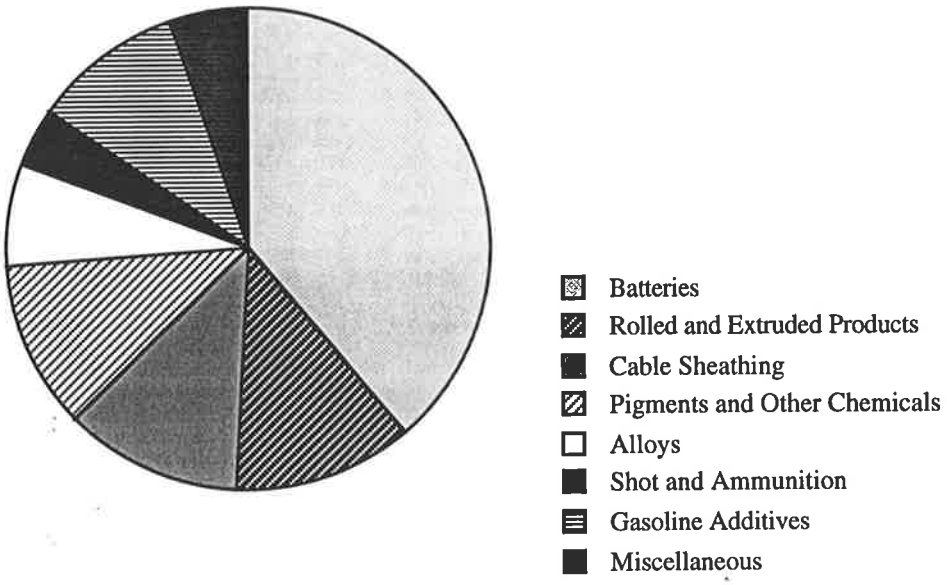
ERA	TOTAL PRODUCTION ($\times 10^3$ tonnes)	
	WORLD TOTAL	CUMULATIVE TOTAL
COPPER AGE (5,900 - 4,100BP)	3,170	5,960
BRONZE AGE (4,100 - 3,200BP)	3,580	9,540
IRON AGE (3,200 - 2,050BP)	14,310	23,850
ROMAN EMPIRE (2,050 - 1,500BP)	14,960	52,670
BARBARIC AGE (1,500 - 1,000BP)	4,250	56,920

Source: data derive from Nriagu, 1983b.

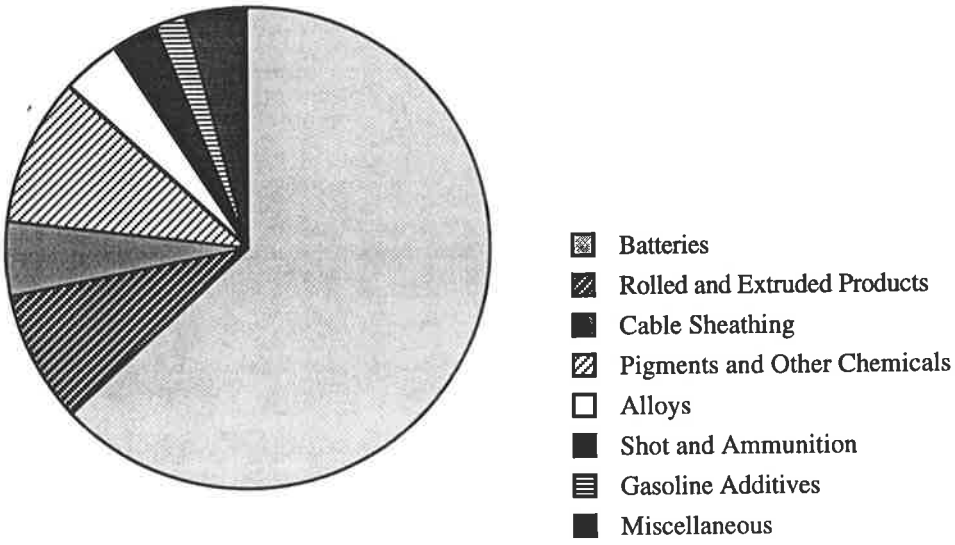
3.2.1.1 END USE PATTERNS

Reflecting demand for lead, between 1940 and 1977, the annual US production and consumption of lead almost doubled (Nriagu, 1988; Berney, 1993). Typical end-use patterns globally and locally are reflected by the following figures (Figure 3.1 and 3.2).

OECD LEAD END USE CATEGORY - 1970



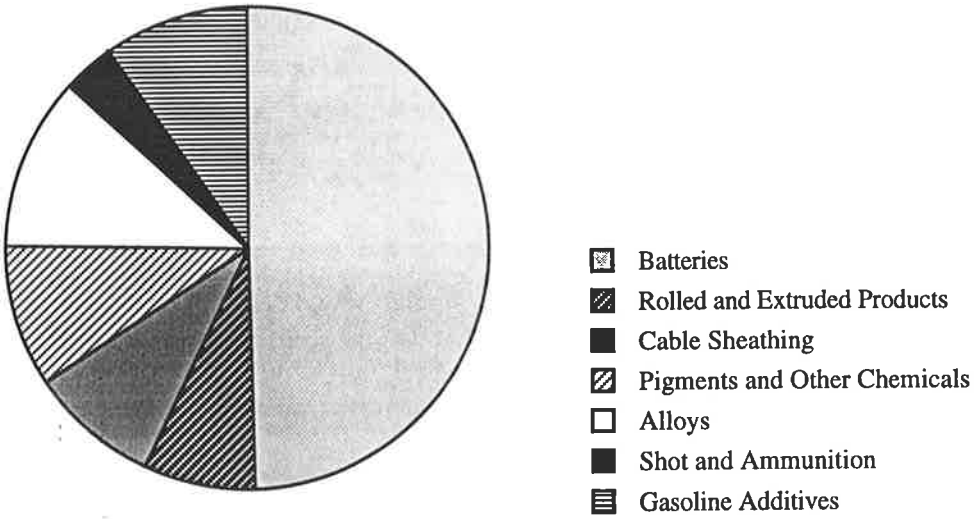
OECD LEAD END-USE CATEGORY - 1990



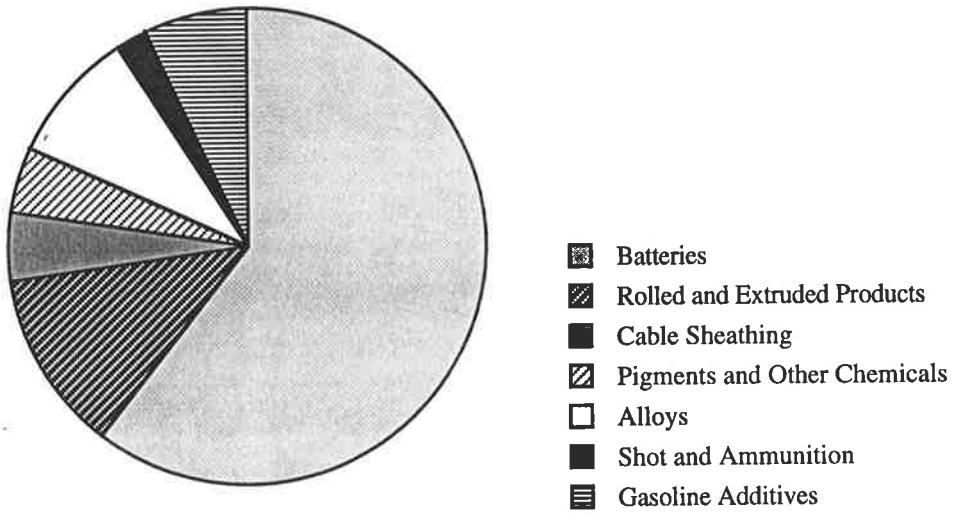
Source: OECD, 1992.

Figure 3.1 Lead demand by end-use category internationally.

AUSTRALIA LEAD END-USE CATEGORY - 1970



AUSTRALIA LEAD END-USE CATEGORY - 1990



Source: EPA NSW, 1993.

Figure 3.2 Lead demand by end-use category in Australia.

Storage batteries have always been the principal single end-use for lead (Anon., 1994), the majority of which in 1990 were for use in the automotive industry, the single largest industry user of lead internationally (Davies, 1988; OECD, 1992) and locally (EPA NSW, 1993). Lead may also be used for radiation shielding, cable sheathing, corrosive chemical storage and for providing ballast for yachting keels and tall buildings in earthquake-prone regions. It may be alloyed for electrical and electronic solder, balancing wheels and piano keys, and to seal wine bottles (Pasminco, 1992).

3.2.1.2 LEAD IN GASOLENE

This use of lead arose as a result of an initial idea and chemical reaction conducted by C. Lowig in 1853 (Shapiro and Frey, 1963; Willemsens and Van der Kerk, 1965). Such information subsequently lay dormant for more than a half-century, eventually eliciting work by Gruttner and Krause between 1915 and 1925 to develop the field of organo-lead chemistry (Nriagu, 1990). This work revolutionised the automobile industry. The demand for leaded gasolene, "the product of American research" (Nickerson, 1954, p. 570) rapidly escalated, such that by 1950, the United States consumed annually greater than 100,000 tonnes of lead for that purpose, the production of tetra-ethyl lead ranking amongst the top ten industrial chemical enterprises in the country (Nickerson, 1954). To the tune of "a tiger in the tank of almost every car",[2] this figure marched, by 1970, to 270,000 tonnes in the USA, and 375,000 worldwide (Grandjean and Nielsen, 1979). The production of plumbiferous gasolene, this "Gift of God" (US PHS, 1925; p. 42) became one of the largest organic chemicals-producing industries for that time. From the 1960s - 1980s, the largest single source of lead emitted to the environment derived from the combustion of gasolene (Lau and Wong, 1983), Shy concluding that rather than being a gift, such gasolene was "The mistake of the twentieth century" (1990; p. 168).

3.2.2 ENVIRONMENTAL DISCHARGE

3.2.2.1 NATURAL SOURCE LEAD EMISSIONS

a) Total

The major natural sources of lead emission to the atmosphere include volcanic emission, forest fires and sea-salt spray, with wind-borne dust/soil particles raised by passing winds constituting by far the largest contribution (Peirson et al., 1973; Pacyna, 1986; Nriagu, 1989). Lead (non-elemental) is relatively abundant in the earth's crust, with estimates of average earth crust lead concentration lying within the range 2 - 16$\mu g.g^{-1}$ (Moeller, 1952; Taylor, 1964; LDA, 1974; Heslop and Jones, 1976). Natural lead may be mobilised and become available as a result of natural weathering of rocks and volcanic activity (Pacyna, 1986; Zabel, 1989), Goldberg and Gross (1971) estimating that natural

[2] A well-known Esso slogan (source unknown) for their product in the 1960s, and referring to the octane-enhancing attributes of leaded fuel (pers. obs.).

mobilisation due to such causes amounts to 210,000 tonnes per year, (≈ 16% of all natural lead emissions to the environment). This compares with an (estimated) annual production (1989) of greater than five million tonnes (Table 3.2 above), and an estimated (known) annual consumption of greater than four million tonnes of the element (ILZSG, 1991b).

b) Atmospheric

Natural sources' atmospheric discharge of lead have been conservatively estimated to approximate 12,000 tonnes per annum (Nriagu, 1988 - more conservative than his earlier figure of 25,000 tonnes; Nriagu, 1979). This (chronologically) later figure equates to around 0.1% of total (natural and anthropogenic atmospheric) discharge, Galloway et al. (1982) reporting around 0.3% (Table 3.4). Settle and Patterson (1980), however, cite an even more conservative figure of around 2,000 tonnes per annum for natural atmospheric discharge, or around 0.5% of the quantity released to the atmosphere via anthropogenic sources, (cited in Nriagu, 1989). They further note that human activity has become the pre-eminent agent in the global atmospheric cycle of such toxic metals as lead.[3]

Contrasting this, however, Jaworowski et al. (1981) claim that the flow of lead into the global atmosphere from all anthropogenic sources was only seven percent of the natural flow, much less than its natural long-term fluctuation. Such a conservative estimate, by comparison with the data of the first-mentioned workers, reflects the considerable differences of opinion shared by different workers regarding levels of anthropogenic and natural discharge of lead (Lawther, 1986), and the difficulties encountered with the accurate assessment of naturally-emitted quantities of lead to the environment. This should be considered when interpreting data regarding historical skeletal lead burdens as presented in Chapter Two.

3.2.2.2 ANTHROPOGENIC-SOURCED LEAD EMISSIONS

a) All sources

Combustion of fossil fuels (coal and oil) and wood, production of iron, steel, lead, copper, zinc and cement, and waste incineration or direct environmental discharge constitute the major sources of anthropogenic lead emissions to the environment (Nriagu, 1978a & 1979; Hutton and Symon, 1986; Nriagu and Pacyna, 1988; Malkin et al., 1992). Nriagu (1988) (conservatively) estimates a total annual environmental discharge of, approximately, 1.3 million tonnes deriving from both natural and anthropogenic sources. Mushak (1992) claims up to two million tonnes, of which significantly greater than ninety percent derives from anthropogenic sources (as Nriagu, 1988), a reflection of which appears in Table 3.4.

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[3] Such data do not include direct anthropogenic discharge of lead to soil and water, a total estimate for which exceeds the estimate for total atmospheric discharge by a factor of approximately three (Nriagu and Pacyna, 1988), further enhancing the anthropogenic/natural environmental lead discharge ratio. Nriagu (1989) estimates that the ratio of anthropogenic to natural emissions of lead is approximately 27:1.

Table 3.4 Total Global Emissions of trace metals into the environment (Natural and Anthropogenic).

ELEMENT	GLOBAL EMISSIONS OF TRACE METALS ($x10^3$ tonnes.yr^{-1}) *			
	ATMOSPHERE	WATER	SOIL	TOTAL
Antimony	3.5	18.1	26.0	47.6
Arsenic	18.8	40.8	82.2	141.8
Cadmium	7.6	9.4	21.9	38.9
Chromium	30.1	142.0	896.1	1,068.2
Copper	35.0	112.1	954.0	1,101.1
Indium	0.02	-	-	0.02
LEAD	**332.1**	**138.2**	**795.7**	**1,266.0**
Manganese	38.1	262.2	1,669.7	1,970.0
Mercury	3.6	4.6	8.3	16.5
Molybdenum	3.2	11.1	88.0	102.3
Nickel	55.9	113.0	325.1	494.0
Selenium	3.8	41.2	41.2	86.2
Thallium	5.1	-	-	5.1
Tin	6.4	-	-	6.4
Vanadium	86.0	12.0	132.1	230.2
Zinc	132.0	226.0	1,372.0	1,730.0

* Most figures are mid point of estimated range.

Source: Nriagu, 1988; p. 142; additional data from Galloway et al., 1982; Nriagu and Pacyna, 1988.

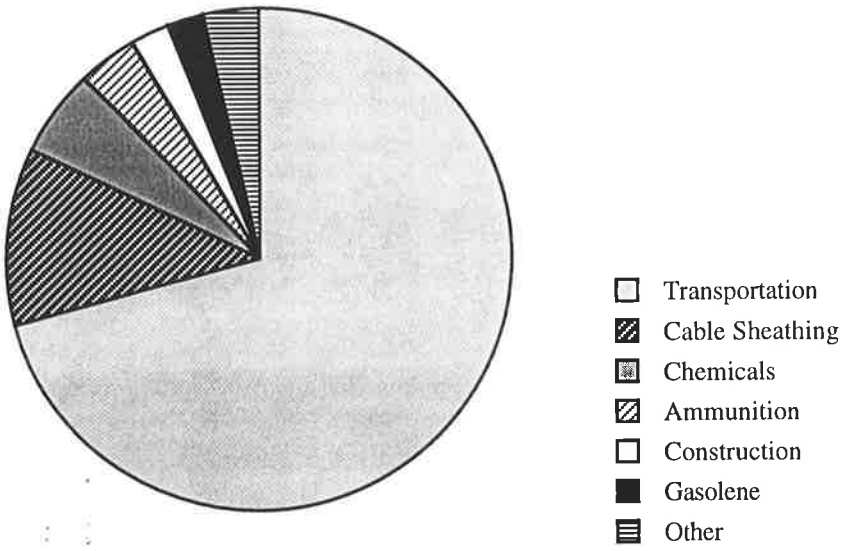
On the basis of approximately 25% of annual production being environmentally discharged (Nriagu, 1988), it seems clear that a substantial quantity of lead is being environmentally dispersed.[4] Acknowledging such estimates, given a total historical production of the order of 275 - 300 million tonnes (Nriagu and Pacyna, 1988; Flegal and Smith, 1992), then of the order of 125 - 150 million tonnes have been cumulatively dispersed to the environment over the last 5,000 years. Such quantities dwarf figures corresponding to the environmental discharge of any other inorganic toxic material (Mushak, 1992).[5]

Contemporary consumption figures for the USA, for example, reflect the proportion distributed to the environment both directly and indirectly (Figure 3.3).

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[4] NAS (1980), however, cite a figure closer to 50% dispersed to environmental compartments with associated potential for excessive human exposure over the preceding five decades.

[5] Roman production of the metal may have exceeded 60,000 tonnes annually (Grandjean, 1975; Lin Fu, 1980; Nriagu 1983a & b). Nriagu (1983a) estimates that prior to about 1500BP, at the time of the end of the Roman Empire, a cumulative total lead production of around 39 million tonnes had been cast.



Total 1987 consumption = 1.2 million tonnes.

Source: Booz et al., 1990.

Figure 3.3 Consumption of lead in the USA in 1987 by percentage category of use.

Settle and Patterson (1980) find similarly for the ratio anthropogenic/natural sources environmental lead distribution, but their figure for total anthropogenic discharge is around fifty million tonnes to date.

b) Gasolene lead

Nriagu (1979) reports that annual anthropogenic atmospheric discharge of lead has increased by an order of magnitude between the decades 1911 - 1920 and 1971 - 1980, essentially arising as a result of the introduction in 1925 (and subsequent widespread and rapidly escalating use) of tetra-ethyl and tetra-methyl lead. Between the years 1926 and 1985 in the United States alone, greater than seven million tonnes of lead was distributed to the atmosphere as a result of the combustion of leaded fuel (Nriagu, 1990). Further, data suggests that over the period 1950 - 1980, atmospheric emissions of lead exceeded previous all time total worldwide anthropogenic emissions (Nriagu and Pacyna, 1988). Estimated worldwide anthropogenic atmospheric lead emissions for 1983 appear in Table 3.5 below.

The greatest single source of lead released directly to the environment derives from the combustion of leaded gasolene; for 1968 greater than 90% of the US lead emissions derived from that source (Billick et al., 1980). Waldron (1980) cites a total figure of 210,000 tonnes of lead anthropogenically-emitted to the environment annually during the mid to late 1970s. For the year 1975, Nriagu (1979) estimated that that year's total emission approached 450,000 tonnes, a level more than a third greater than the figure for 1983, much of the decline arising due to lead-free motor fuel replacing leaded motor fuel (Nriagu and Pacyna, 1988).[6] A decline also occurred due to an international reduction in

[6] The demise of leaded fuel was a particularly acrimonious one with multi-national oil and giant motor vehicle-producing companies eventually bowing to concerns regarding public health risks associated with the product. This would appear to be one of the first environmentally unfriendly products to be forced off the market (Nriagu, 1990).

industrial emissions of particulate lead, encouraged by enhanced awareness of health concerns associated with heightened environmental lead levels.

Table 3.5 Worldwide anthropogenic atmospheric lead emissions (1993) by source.

LEAD EMISSION SOURCE	EMISSION ($x10^3$ tonnes.yr^{-1})
Cement production	0.02 - 14.2
Coal combustion	
electricity-generating installations	0.8 - 4.7
industry and domestic use	1.0 - 9.9
Oil combustion	
electricity-generating installations	0.2 - 1.7
industry and domestic use	0.7 - 2.2
Phosphate-containing fertiliser use *	0.06 - 0.3
Pyrometallurgical non-ferrous metal production	
mining	1.7 - 3.4
lead production	11.7 - 31.2
copper & nickel production	11.1 - 22.1
zinc and cadmium production	5.5 - 11.5
Refuse incineration	
municipal	1.4 - 2.8
sewage sludge	0.2 - 0.3
Secondary non-ferrous metal production	0.1 - 1.4
Steel and iron manufacturing	1.1 - 14.2
Wood combustion	1.2 - 3.0
Mobile sources	248.0
Miscellaneous	3.9 - 5.1
TOTAL (MEDIAN)	**288.7 - 376.0 (332.0)**

* Particularly if some component of fertiliser derives from sewage sludge (Harrison and Laxen, 1981; Hutton and Symon, 1986).

Source: data derive from Nriagu and Pacyna, 1988.

3.3 LEAD IN THE ENVIRONMENT

For the purposes of discussion here, as Kawai et al. (1983), the terminology ppm lead in dust, and $\mu g\ lead.g^{-1}$ dust are considered equivalent.

3.3.1 SOIL

Given lead's relative (chemical) stability in soil (Moeller, 1952; NSF, 1979; EPA USA, 1986), and acknowledging on-going production and environmental discharge for millennia, (apart from naturally-occurring soil mineralisation) soil in many areas globally has become the repository site for anthropogenic lead. Soil constitutes the major environmental reservoir for lead, (Nriagu, 1988; Mielke et al., 1992), all soil everywhere in the world containing some lead (Waldren and Stöfen, 1974). Soil lead at either background or contaminated level arises from a wide range of primary and secondary sources, travelling along a plethora of different pathways (Davies, 1988), with the accumulation of lead in most soil occurring primarily as a function of the rate of deposition from the atmosphere (Gulson et al., 1981; Thornton, 1982; SRC, 1990).[7]

3.3.1.1 CONTRIBUTORS TO SOIL LEAD

a) Natural sources

Naturally, levels of lead in soil can vary widely between different geographical regions, influenced by variations in initial parent material, its subsequent degree of weathering and soil moisture and organic material content (Davies, 1978). Human influences upon soil lead levels focus on land use practices and also proximity to potential (particularly point) pollutant sources.

Many researchers have conducted baseline surface soil lead surveys in a plethora of different geographical regions. Such workers have concluded that soil that is free from natural mineralisation and the influence of anthropogenic contaminant sources contains lead at concentrations well below $50\mu g.g^{-1}$ (eg Swaine, 1955; John, 1975; Davies, 1978; Archer, 1980; McKeague and Wolynetz, 1980; Davies, 1983a; Reaves and Berrow, 1984; Davies and Paveley, 1985; EPA USA, 1986; McGrath, 1986; Shacklette and Boerngen, 1986). Rural sites in industrialised countries would tend to reflect slightly higher values (Duggan and Inskip, 1985) eg mean soil lead levels of around $55\mu g.g^{-1}$ (CEPA, 1992), $70\mu g.g^{-1}$ (Davies, 1978) to $80\mu g.g^{-1}$ (Farmer and Lyon, 1977). The normal soil lead concentration

[7] Torr (1993) defines background concentration of an element/contaminant as that concentration which is not attributable to any single, localised event or (anthropogenic) activity.

level for US soil remote from mobile or stationary lead-emitting sources has been suggested to lie at around 8 - 25μg .g^{-1} soil (EPA USA, 1986).[8]

b) Anthropogenic influences

As for natural sources of soil contamination, the principal source of soil contamination arises from the settling of contaminated airborne dust (Davies and White, 1981; Reagan and Silbergeld, 1989), aeolian activity influencing this deposition of aerosol lead (Archer and Barratt, 1976a; Fergusson et al., 1980; Fergusson and Simmonds, 1983; Simmonds et al., 1983).

Apart from direct mine/smelter discharge to soil eg slag deposition in soil (Maskall and Thornton, 1993), other direct anthropogenic sources of lead to soil include insecticides eg lead arsenate (Fleming and Parle, 1977; EPA USA, 1984), sewage sludge (Van Loon et al., 1973), landfill sites from which the contaminant often finds its way subsequently into groundwater, (Kutlaca, 1992, 1993 & 1994), fertiliser impurities (Hutton and Symon, 1986), degraded leaded paint flakes (Landrigan et al., 1982, with a dry film lead content in older paints of up to 50% [Rundle and Duggan, 1986; Schwar et al., 1988]), lead shot/ammunition (CEH & CAPP, 1987; Zabel, 1989; PPK, 1992) and fishing sinkers (Morton et al., 1982; Rossol, 1992).[9]

Indirect sources focus mainly upon already-entrained atmospheric deposition of lead-bearing soil particles or dust (Prpic-Majic et al., 1984), and run-off of contaminated water from mining, smelting, scrap metal and recycling facilities (Lagerwerff and Brower, 1975; Thornton, 1980). Overall, atmospheric deposition of lead to soil would appear to be, by far, the greatest contributor to soil lead burdens (Hutton and Symon, 1986; Davies, 1988; Nriagu and Pacyna, 1988; EPA NSW, 1993).

- i) Mineralised soil areas

Human influences can, however, drastically alter such baseline soil lead levels as reflected by many earlier and also contemporary researchers (Davies, 1978; Landrigan et al., 1982, EPA USA, 1984; Thornton et al. 1985; Rundle and Duggan, 1986; Davies, 1988; Schwar et al., 1988; IPCC, 1991; Maskall and Thornton, 1993). Common sites for such elevation of contaminant levels may frequently be previous or current lead/zinc mining/smelting areas (Davies and White, 1981). Mean soil lead level figures as high as 6,600μg.g^{-1} for garden and field soil adjacent to a former lead/zinc-mining area were cited by Davies and Roberts (1975), with some levels as high as 48,000μg.g^{-1}. Similar levels (mean = 5,000μg.g^{-1}) for mineralised soil in a region of Derbyshire were arrived at by Barltrop et al. (1975), with some high levels approaching 30,000μg.g^{-1}. Thornton et al. (1985), conducting a more extensive survey of the same region, report similar findings (mean = 5,600μg.g^{-1}; range = 1,200 -

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[8] The definition for surface soil would appear to be variable with a number of different interpretations having been made, rendering some comparisons difficult. Whilst some workers have termed the top 15cm horizon as surface soil (McGrath, 1986), more frequently the term refers to the top two to ten centimetres (Davies, 1988), with the top two to five being the most commonly assumed (Gulson et al., 1995).

[9] From an extensive survey of Port Pirie houses in the mid 1980s, Body (1986c) demonstrated that 43% of indoor paint samples demonstrated in excess of two percent lead, and 24% in excess of ten percent lead, corresponding figures for outside surfaces being 63% and 31% respectively.

22,000μg.g^{-1}) for 100 garden soils. In an earlier study in an old zinc-mining area of Somerset, Thornton et al. (1980) note a mean surface soil lead concentration approaching 2,400μg.g^{-1} (range = 104 - 6,880μg.g^{-1}, n = 607).

- ii) Urban areas

Urban soil has been extensively studied by many workers, and virtually universally reveals contamination by heavy metals such as lead (Purves, 1968; Skogerboe et al., 1979; Khan, 1980; Ratcliffe, 1981; Duggan and Inskip, 1985; Chaney and Mielke, 1986; Davies et al., 1987a; CEPA, 1992). Mean soil lead levels of the order of 500μg.g^{-1} (Farmer and Lyon, 1977; DOE, 1978 & 1982, Davies et al., 1979) to 1,000μg.g^{-1} (Duggan and Inskip, 1985) for average urban levels have been reported for the UK, with other workers citing even more elevated levels elsewhere, eg 1,950μg.g^{-1} in Christchurch NZ (Shellshear et al., 1975), 1,550μg.g^{-1} in Vancouver (Schmitt et al., 1979) and 2,000μg.g^{-1} in Detroit (Ter Haar and Aronow, 1974). From a study of urban soil lead concentrations in a number of US cities, Nriagu and Pacyna (1988) quote a range of from 100 - 1,100μg.g^{-1} for such locations. Urban soil lead levels are typically greater in central areas of cities than in peripheral suburbs (Davies, 1978), Davies et al. (1979) claiming that a major factor influencing levels is distance from the centre of a city. They report mean (London) levels (523μg.g^{-1}) of more than an order of magnitude greater than those levels in an adjacent rural zone 30kms distant (30μg.g^{-1}). Similar observations were also noted in Birmingham (Davies and Houghton, 1984), and in various locations in the USA (Kinard et al., 1976; Brams, 1977; Mielke et al. 1983 & 1984).

Similarly, Thornton et al. (1985) have comprehensively demonstrated an elevated (geom.) mean garden soil lead level in London (647μg.g^{-1}, n= 579) compared with garden soil derived from throughout England, Scotland and Wales excluding London (230μg.g^{-1}, n = 3,550). They also note a significantly lower mean lead level in houses 30 years of age or younger compared with houses of greater than 100 years of age, an observation also reported by Davies (1978). It would appear likely that such differences would reflect different home histories of external surface contamination from fossil fuel residues and also weathering paint (Landrigan et al., 1982; Yaffe et al., 1983). Such contaminants are suggested to wash from walls and rooftops into predominantly foundation-adjacent soil, thus magnifying levels there (Roberts et al., 1991b, 1992, 1993a & b). This observation is supported by a detectable soil lead gradient (decreasing soil lead concentration with increasing distance from the house) again observed by a number of workers (Ter Haar and Aronow, 1974; Getz et al., 1979; Wheeler and Rolfe, 1979 and Preer et al., 1984). A number of researchers would claim that this house-proximate surface soil lead contamination remains indefinitely (eg Lewis, 1977; ATSDR, 1988a; Mielke et al., 1989).

The major contributor to such elevated urban soil lead levels is suggested to be automobile exhausts (Purves, 1968; Davies, 1978; Khan, 1980; Ratcliffe, 1981; Chaney and Mielke, 1986; Davies et al., 1987a; Davies, 1988). In areas where old painted dwellings display flaking paint, however, old leaded paint may be a major contributor to soil lead levels (Solomon and Hartford, 1976; Landrigan et al., 1982), Yaffe et al. (1983) reporting that such lead-contaminated soil may constitute a proximate and continuous source of lead for children. Further, substantial contributions may result from

industrial activity such as smelting of ore or concentrate (Brunekreef, 1984), metal recycling involving fusion (Dolcourt et al., 1981), electroplating (Hutton and Symon, 1986) and older style newspaper printing works (Jensen and Laxen, 1985). Other suggested industrial emission contributors include lead/acid battery production (Fergusson et al., 1981; Hutton and Symon, 1986; Fergusson and Kim, 1991; Moseholm et al., 1992), storage (Fleming and Parle, 1977) and recycling (Landrigan et al., 1975a; Dolcourt et al., 1981; Romieu et al., 1997). Also noted are a number of home-activities involving lead (Section 3.3.5.5). Mielke et al. (1983) suggest that (urban) soil lead levels are an important measure in accounting for the good correlation between the degree of urbanisation of a community and the magnitude of the lead contamination problem.

- iii) Roadway-adjacent soil

Soil adjacent to roadways may frequently be contaminated with gasolene-derived lead (Butler et al., 1975; Day et al., 1975; Waldron, 1975), typically with the most marked effect occurring within a 20 - 30 metre corridor either side of the carriageway (Butler and MacMurdo, 1974). Such contamination appears typically in the top one or two centimetres (Lau and Wong, 1983), resulting from the aerial deposition of dust (Harrison et al., 1981a; Rapsomanikis and Donard, 1985), and also as a result of water run-off of road-deposited materials (Harrison and Laxen, 1981; Revitt et al., 1990). Such roadside dust appears to be one of the most highly enriched in toxic elements of any urban dust (Hopke et al., 1980), demonstrating typically greater lead concentrations than urban soil (Harrison, 1979). The concentration of lead in soil (as with most other metallic elements) typically increases with a decrease in the mean particle size (Fergusson and Ryan, 1984). The lead that is present here derives almost exclusively from combusted motor spirit (Linton et al., 1980). Within this fringe, many researchers have estimated soil lead levels, with proffered figures for rural area topsoil adjacent to a busy road being varied and of the order of 150 - 550$\mu g.g^{-1}$ (Lagerwerff and Specht, 1970), 200 - 1,100$\mu g.g^{-1}$ (Ward et al., 1977), $\approx$1,000$\mu g.g^{-1}$ (Motto et al., 1970; Muskett and Jones, 1980; Rodriguez-Flores and Rodriguez-Castellon, 1982; Duggan and Inskip, 1985), and 30 - 2,000$\mu g.g^{-1}$ (EPA USA, 1986) above natural soil levels.

Virtually all of this contamination is resident in the top ten centimetres (EPA USA, 1986) and distributed within fifty metres of the roadway (Duggan and Inskip, 1985). Apart from distance from carriageway (Archer and Barratt, 1976a; Solomon and Hartford, 1976; Beckwith et al., 1985; Duggan and Inskip, 1985) and local weather conditions including rain, wind speed and direction (Simmonds et al., 1983; Piron-Frenet et al., 1994), traffic density and mode would appear to be the only other main determinant of associated proximate soil lead levels (Quarles et al., 1974; Little and Wiffen, 1978; Wong and Tam, 1978; Wheeler and Rolfe, 1979; Fergusson et al., 1980; Agrawal et al., 1981; Garcia-Miragaya et al., 1981; Fergusson and Simmonds, 1983; Simmonds et al., 1983; Mielke et al., 1984; Lyngbye et al., 1987; Kim and Fergusson, 1988; Fergusson and Kim, 1991; Nam et al., 1994). Chaney and Mielke (1986) note that elevated soil lead levels adjacent to main urban roads reflect uneven distribution of traffic, with the greatest quantity of lead emitted in city centres (Rapsomanikis and Donard, 1985; Boni et al., 1988).

- iv) Rural areas

Rural agricultural land can also be a recipient of adventitious lead, apart from that deriving from motor vehicle exhausts. The use of lead-based pesticides (Fleming and Parle, 1977), the addition of fertilisers which have a component deriving from sewage sludge (Andersson and Nilsson, 1972; Van Loon et al., 1973), or the direct addition of sewage sludge to soil (Richardson, 1980; Hutton and Symon, 1986), particularly over long periods of time may all contribute to the enhanced accumulation of lead by soil (Harrison and Laxen, 1981).

- v) Smelter locations

Much literature abounds relating to the influence that both primary and secondary lead smelters effect upon the immediate environment (Davies and White, 1981).[10] Although different local conditions and smelter details will influence the dimension of the environmental contamination elicited by the smelter (Adamson, 1980), soil contamination is related to a number of variables including the mean smelter emission rate, the period of time that the plant has been operational, local topography, prevailing wind force/direction and local climatic conditions (Roberts, 1974a & 1975).

Different soil horizon sampling practices by different researchers, and different local meteorological conditions and topography complicate soil lead concentration comparisons. This is particularly so with respect to estimating concentration changes with changing distance from (presumed) sources, however, clear evidence exists to suggest that soil proximate to a smelter site may be highly contaminated, Ward et al. (1977) detecting a highest soil lead value of 126,000$\mu g.g^{-1}$ (12.6%) adjacent to a New Zealand smelter. Roberts (1975) reports upper soil lead concentrations of between 15,000 and 20,000$\mu g.g^{-1}$ adjacent to a Toronto smelter, Palmer and Kucera (1980) in the vicinity of two smelters in Missouri registering a high value of 60,000$\mu g.g^{-1}$. Kerin (1975) notes a mean value of 5,000 - 9,000$\mu g.g^{-1}$ proximate to a smelter in the (former) Yugoslav republic, Ragaini et al. (1977) citing levels of around 8,000$\mu g.g^{-1}$ adjacent to a smelter in Kellog, Idaho. Reflecting contaminant travel from source, Davies (1988) notes that a number of these workers has registered soil lead contamination at distances of several kilometres from the smelter site.

3.3.1.2 SOIL LEAD, PATHWAYS TO HUMANS - ASSOCIATED HEALTH RISKS

Some conjecture exists regarding the relative potential for soil to contribute to human contact with lead. Certainly some would suggest that soil-contained lead is considered to be a significant potential contributor to human contact with lead (Barltrop, 1966; Barltrop et al., 1974; Sayre et al., 1974; Roels et al., 1976; Yankel et al., 1977; Walter and Yankel, 1980; Yaffe et al., 1983), in fact, it represents one of the primary links for the material to be transferred to humans (Davies, 1988). Rabinowitz et al.

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[10] A primary lead smelter produces elemental lead from ores or concentrates, whereas a secondary smelter is more a recycler of used metallic lead materials.

(1985) demonstrate that lead levels in outdoor soil are a strong predictor of children's blood lead levels, Weitzman et al. (1993) reporting a statistically significant decline in blood lead levels (albeit modest) amongst children who lived in homes receiving soil abatement. Bornschein et al. (1987) observe that lead from soil contributes significantly to children's blood lead levels, but not to such an extent as house dust lead and hand-attached lead, Davies et al. (1985) concluding that soil lead contributed significantly to interior house dust lead.

Thus, lead in soil may make a contribution to the lead burden of urban-residing children, the proportion of which appears poorly defined. Barltrop (1977) and Thornton (1988) claim that the lead content of house dust from individual homes increases *pari passu* with corresponding soil lead concentration values, Murphy and Yocom (1986) suggesting that the contribution of lead to indoor dust derives mainly from outdoor soil.

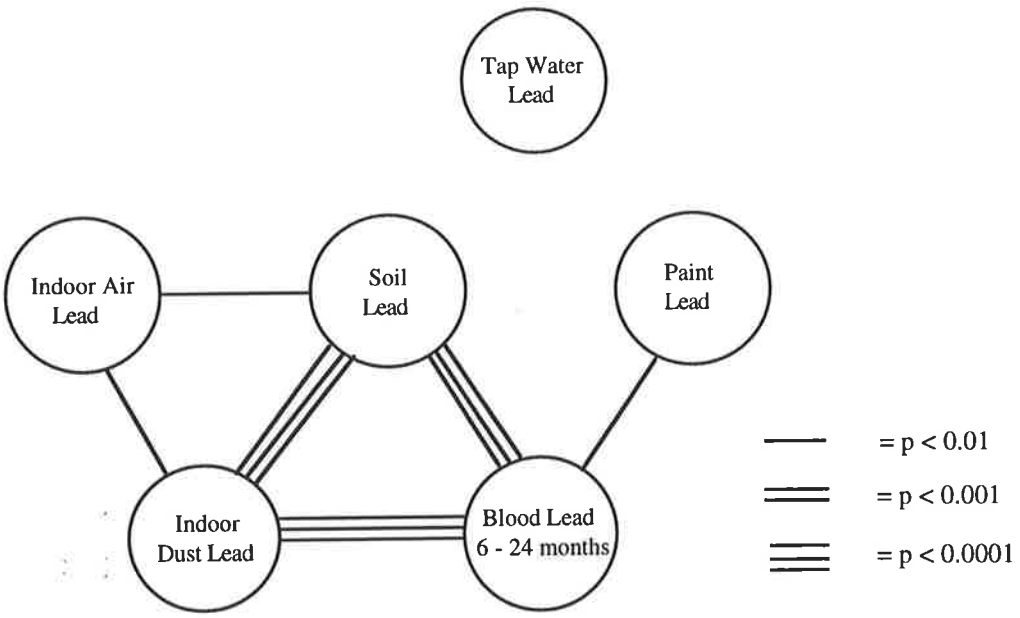
From a national survey of households from 53 representative UK towns, Thornton et al. (1985) note that house dust lead concentration is significantly and positively related to house soil lead concentration, Culbard et al. (1988) proffering a figure of $r = 0.53$ ($p = 0.001$, $n = 4,512$). In an area of an old lead mine with very heavy soil contamination, Davies et al. (1985) and Davies and Wixson (1985) noted a moderate correlation ($r = 0.50$) between outside soil lead concentration and indoor dust lead concentration. Other studies proximate to lead smelters have reported similarly, eg $r = 0.20$ (Yankel et al., 1977), 0.28 (Davies et al., 1987b & 1990; with $r = 0.40$ for dust lead concentration with dust lead loading), 0.30 (Angle and McIntire, 1979), 0.40 (Diemel et al., 1981) and 0.55 (Davies and Thornton, 1987). Other workers conclude that the source of lead in residential soil may not necessarily be clearly (Pisaniello et al., 1996) or robustly (Gulson et al., 1995) related with the source of lead in house dust. A number of researchers report that the importance of soil as a predictor of blood lead has diminished as a result of recent work (eg Aschengrau et al., 1991; Clark and Bornschein, 1993; Farrell et al., 1993; Weitzman et al., 1993; Gulson et al., 1995).

Fergusson and Kim (1991), using a multi-element assay approach, claim that soil may contribute a modest to a significant amount to house dust, perhaps 30% directly from soil. Earlier, Fergusson (1986), as Fergusson et al. (1986) report that 45 - 50% of house dust came from soil or street dust, Davies et al. (1985) and Calabrese and Stanek (1992b) observing similarly.[11] Culbard et al. (1983), Culbard and Johnson (1984) and Thornton et al. (1985) also concur, Gulson et al. (1994b) as Steele et al. (1990) concluding that soil is not a major contributor to house dust in either urban centres or a mining town. Seemingly in contrast with this, however, Gulson et al. (1994a) do link the majority of house dust with the orebody in the township of Broken Hill, the inference being that home soil is also similarly linked.

Based upon both urban and mining community studies, Clark et al. (1991) as Bornschein et al. (1991) suggest a lead pathway from soil to air to house dust to children's hands and eventually to blood. Rabinowitz et al. (1985), from their longitudinal study of urban neonates, demonstrate strong correlation between soil, dust and blood lead levels, and marginally weaker correlation between these two parameters and indoor air lead levels (Figure 3.4).

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[11] Previously, Fergusson et al. confirmed that soil "is a major contributor to house dust", (1986; p. 221).



Source: Rabinowitz et al., 1985; p. 100.

Figure 3.4 Bi-variate correlations between a child's blood lead and various environmental measures.

The extent to which soil lead can contribute to human absorption of the contaminant (ie the material's bioavailability) will vary to some extent according to lead's concentration in the soil (Barltrop et al., 1975; EPA NSW, 1993), the size of the lead-bearing particles (Davies, 1991), and also their physico-chemical species (Hemphill, 1991). It has been suggested for contained soil lead (EPA USA, 1986) that every soil-lead incremental increase of 1,000$\mu g.g^{-1}$ will elicit an increase in resident child blood lead levels of the order of 3 - 7$\mu g.dL^{-1}$, although individual child habits and local conditions may alter this relationship. As evidenced by children living in several UK areas where soil lead levels were as substantially elevated as 8,000$\mu g.g^{-1}$, yet where their demonstrated blood lead levels were only minimally elevated, it is claimed that the presence of high concentrations of lead in the soil is not necessarily always hazardous, even to children (Barltrop et al., 1974).

Contrasting this, Landrigan et al. (1975) report from Smeltertown Texas, an extremely dry and dusty environment with mean soil lead levels of around 3,500$\mu g.g^{-1}$ and mean dust lead levels reaching as high as 22,000$\mu g.g^{-1}$, greater than half of the one to nine year old children revealed abnormally high blood lead levels. Given the propensity for children to mouth, eat or play with soil and the countless number of (residential) dwellings worldwide demonstrating coverage with degraded leaded paint contributing to an enhanced soil lead burden, a substantial health hazard clearly exists.

3.3.2 WATER

As for soil, freshwater bodies may also become long-term repositories for anthropogenic lead contamination (EPA USA, 1986), particularly if pH levels are moderate to high, effectively de-solubilising salts of lead, and (potentially) metallic lead (AAS, 1981). Lead may be present in water

either as a dissolved salt, suspended/precipitated as a particulate (EPA NSW, 1993), or adsorbed onto the surfaces of some organic sediment particles (Rendell et al., 1980).[12]

3.3.2.1 PATHWAYS TO CONTAMINATION

Contamination can arise directly from industrial and sewage discharge (Lester et al., 1979), and lead shot/ammunition and sinkers use (Morton et al., 1982; CEH & CAPP, 1987). Contamination may arise indirectly via atmospheric deposition (Patterson, 1982), soil leaching and/or surface stormwater run-off from roadways (Laxen and Harrison, 1977; Hamilton et al., 1987; Hewitt and Rasheed, 1988), leaching in mineralised areas (Abdullah and Royle, 1973; Aston and Thornton, 1977) and near contaminated sites where soil lead levels are elevated (El Saadi and Langley, 1991). It may also occur via discharge from industry, contaminated groundwater redirection, leaching of fertiliser particularly if incorporating sewage sludge, and deposition of previously-suspended leaded aerosol (Harrison and Laxen, 1981; Hutton and Symon, 1986; Hamilton et al., 1987). Patterson (1982) notes that aerial deposition of lead-bearing dust/soil particles is the principal source of lead contamination in salt and fresh water bodies, such deposition occurring in either dry or wet modes. Two estimates for the concentration of lead in rainwater put the figure in the range 27$\mu g.L^{-1}$ (for the UK - Cawse, 1980) to 34$\mu g.L^{-1}$ (for the USA - Lazrus et al., 1970). This is a more substantial concentration than the average concentration of lead in surface waters, inferring that some precipitation of lead from rainwater is occurring, presumably finding its way to sediments (Dong et al., 1984). According to Davies (1988), most of this (fresh) water contamination eventually finds its way to the marine environment, with particularly estuarine and coastal sediments becoming environmental "reservoirs" for the metal (Martincic et al., 1990a & b).

3.3.2.2 GROUND WATER

Ground water bodies are most typically affected by the geochemical composition of the bedrock, although prolonged (over-) application of fertilisers prepared from sewage sludge may also contribute to enhanced lead levels given sufficient time (EPA USA, 1986; Hutton and Symon, 1986). Illicit, clandestine disposal of lead contaminated industrial wastes directly to permeable soil with aquifers close to the surface may also contribute to elevated ground water lead levels (SRC, 1990).

3.3.2.3 SURFACE WATER

Durum et al. (1971) note a mean concentration level for lead in surface water in the USA of approximately 4$\mu g.L^{-1}$. An analogous estimate for surface water arising from non-mineralised areas

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[12] At low pH, lead salts are more water soluble, although if water contains many dissolved salts, eg salts of Calcium and Magnesium (water "hardness"), lead's solubility is reduced (Moeller, 1952). For moving water, the volume of stream flow is directly related to the potential for lead contamination of the water, as is the concentration of lead in sediments, with lead moving from dissolved to suspended compartments and adsorbing onto sedimenting particles with an increase in pH and conversely so also (EPA USA, 1986).

of SW Britain reflects mean levels in the range 1 - 3μg.L^{-1} (Aston and Thornton, 1977). Several other workers cite a mean figure of 4μg.L^{-1} for such an area (Chow, 1978; NRC, 1980; Davies, 1988).

3.3.2.4 HUMAN CONTACT WITH LEAD VIA WATER

a) Reticulated water

The principal means of human contact with elevated lead levels in water is by way of the metal's use in domestic water-reticulation systems for pipes, tank linings and also solders, a number of researchers having inferred associated health concerns (eg Moore et al., 1977a; Matthew, 1981; Sharret et al., 1982; Fergusson, 1986; Hutton and Symon, 1986; RSC, 1986). This is typically more pronounced a problem in older houses in the UK (Richards and Moore, 1984; Raab et al., 1985), the practice typically having been discontinued there earlier this century (Moore et al., 1977a, Moore 1979 & 1983; Davies, 1988). Leaded water-reticulation networks with health sequelae have also been reported from Belgium (Sartor and Rondia, 1981), France (Bonnefoy et al., 1985), the former West Germany (Milgrom, 1984), Wales (Elwood et al., 1978) and the USA (Morse et al., 1979), albeit for the last example, only to a minor extent (Levin, 1989). The ability of water to abstract lead by contact (ie water's plumbo-solvency) is influenced by a number of factors, namely

- the pH of the water
- the hardness of the water
- the stasis time in pipes
- the quantity of lead on the internal pipe surface
- the temperature of the water
- the turbulence/mixing of the water
- the age of solder joints.

Acidic, soft, long-standing, turbulent and warm water demonstrates the greatest plumbo-solvency, with old soldered joints succumbing to leaching away with time (Beattie, 1975; Moore, 1980 & 1983; Sartor and Rondia, 1981; Worth et al., 1981; RCEP, 1983; Elwood et al., 1984; EPA USA, 1986; Fergusson, 1986; Hutton and Symon, 1986; Levin, 1989). Davies (1988) notes that where lead stearate has been used as a stabiliser in PVC pipes, lead may also be slowly leached out into the water.

Average UK lead in water levels assessed in the mid 1970s revealed a mean level less than 10μg.L^{-1}, with approximately ten percent of homes assayed exhibiting levels in excess of 50μg.L^{-1} (Moore et al., 1985). For Scottish homes (with softer water), however, approximately one third of homes displayed levels in excess of the recommended maximum (50μg.L^{-1}), and twenty percent in excess of 100μg.L^{-1} (DOE, 1977). At about this time, a substantial water lead abatement and lead pipe replacement initiative was undertaken by the British Government, particularly in major cities (Davies, 1988; Pearce 1992).[13] Recent research has reflected a marked decrease in lead in water concentration

[13] The addition of lime (calcium carbonate) has been a common practice to increase the hardness of the water and also the pH to approximately 8.5 - 9.0, thus reducing markedly, the plumbo-solvency of lead.

levels (approx. ninety percent of London tap water samples demonstrated lead levels below $10\mu g.L^{-1}$) as a result of such activity (Moore et al., 1985; Sherlock et al., 1985; Raab et al., 1987; Strehlow and Barltrop, 1987). In contrast to these data, reticulated water supplies in Australian urban locations, where lead has rarely been used for such systems apart from jointing solder, suggest lead levels of from 10 - $20\mu g.L^{-1}$ (AAS, 1981). More recent work by Gulson et al. (1994a) infers lower levels, with an insignificant contribution to the lead budget in humans being made by lead in water.

b) Rain water storage tanks

In drier regions, such as parts of Australia and the USA, domestic rain water tanks fed by roof catchment may be used to augment/replace (poorer quality/unavailable) reticulated water for drinking purposes. Such practices may heighten residents' exposure to lead (Body, 1986g) particularly if decaying organic material (eg leaves) is present, contributing to the water's plumbo-solvency by lowering the pH of the water. Young and Sharpe (1984), observing levels in a comprehensive US study, occasionally noted levels exceeding $50\mu g.L^{-1}$, the then World Health Organisation (WHO) maximum suggested level for lead in water (WHO, 1984).[14] Body (1986g), noted approximately 12% of tested Port Pirie rain water tanks exhibited lead levels exceeding the WHO maximum with around two percent exceeding $300\mu g.L^{-1}$. He further observed that a pH of around three increases the dissolution rate of lead considerably, and the presence of leaf litter in tanks increases lead's solubility in contained water by a factor of up to fifty, noting some correlation between highly-leaded roof paints and water lead content. A negligible contribution of lead to stored rainwater from Zinc/alume®-plated roofs, gutters or storage systems via leaching was detected (Body, 1986h).

Apart from locations where water is particularly plumbosolvent and/or where the use of leaded construction materials for water-reticulation systems is commonplace (Morse et al., 1979b), drinking water represents only a minor route of lead exposure for humans in the USA (Mahaffey et al., 1982a). Pearce (1992), however, claims that tap water is the main source of lead exposure in much of Britain. Most of this lead derives from the use of leaded materials for the household water-reticulation system (eg water pipes), such homes numbering in excess of four million (Anon., 1992a).

3.3.3 AIR

3.3.3.1 AIR-SUSPENDED PARTICLE DYNAMICS

Airborne lead levels, unlike those for soil and water, may be much more transitory in nature (Peirson et al., 1973). Often, air movements are necessary to maintain some suspended atmospheric (lead-bearing) soil/dust particles (EPA US, 1986), larger-sized particles demonstrating shorter atmospheric lifetimes than those for smaller particles (Moschandreas et al., 1979). Despite (relatively) brief atmospheric lifetimes for most suspended dust particles (typically sized around ten microns (μ -

[14] This figure dropped to $10\mu g.L^{-1}$ in September 1992 (Pearce, 1992).

one micron = 10^{-6} metre) or larger [Peirson et al., 1973]), some lead-bearing particles may be transported large distances (up to thousands of kilometres) whilst suspended (Lee, 1972; Moschendreas et al., 1979). Typically, such particles are less than one micron in diameter, (Lee, 1972; Butler et al., 1975), although air lead concentrations in remote areas are usually low (EPA, 1986; Fergusson, 1986; Davies, 1988; EPA NSW, 1993).[15] Such (air) transported lead-bearing particles may settle in areas ranging from proximate, to a considerable distance from a point source (Settle and Patterson, 1982). Typically, such particle variation in transported distance remains a function of particle size (Milford and Davidson, 1985; Zabel, 1989). Tiller (1989) indicates the potential for severe local soil pollution within two kilometres of a large smelter stack, and above-background metal levels at distances greater than 40 kilometres from the smelter source. Davies (1988) notes similarly, EPA NSW (1993) reporting a region of 500 square kilometres being contaminated surrounding a smelter in Sudbury, Canada. Ultimately, virtually all lead-bearing particles will settle (Settle and Patterson, 1982), transference to surface structures occurring as a result of either wet (rain, snow, sleet, hail) or dry deposition (Davies, 1988). Such dry deposition fluxes are influenced by wind speed (Kartal et al., 1993), temperature (Cawse, 1980; Davidson, 1980; Elias and Davidson, 1980; Davidson et al., 1982; Galloway et al., 1982; Settle and Patterson, 1982), settling surface morphology and texture (EPA USA, 1986; Nazaroff and Cass, 1989) and also the particle size distribution (Duggan and Inskip, 1985).

Principal anthropogenic contributors to atmospheric lead include the combustion of leaded motor spirit, industrial discharge, and the combustion of solid hydrocarbons such as coal, oil and wood for electricity-generation. Also indicted are aged paint removal (Farfel and Chisholm, 1990), and the combustion of municipal waste at a waste repository site (Giordano et al., 1983; Wadge and Hutton, 1987; Dennison and Silbergeld, 1988; Fernández et al., 1992; Mitchell et al., 1992, and Malkin et al., 1992 who report on incinerator workers with elevated blood lead levels), such material often containing a range of heavy metals including lead (Kutlaca, 1992). Another contributing factor is the resuspension of soil dust by wind or human activity (Kartal et al., 1993).

3.3.3.2 AIR LEAD LEVELS

a) Remote areas

Lead-in-air concentrations from sparsely-populated remote areas (eg Atlantic Ocean, undeveloped continental South American and Asian sampling sites) tend to be low to very low by comparison with more developed areas, mostly between two and three orders of magnitude lower, (Nriagu, 1979 &

[15] Metallic air pollution is generally associated with suspended solid particulate material (Harrison, 1986b), such material generally lying in the particle size range 0.6 - 10.0μ, (Davidson, 1980). Lead-bearing particles arising from the (automobile) combustion of leaded gasolene usually lie within the particle size range < 5μ, and may be transported large distances at tropospheric levels (EPA NSW, 1993) further contributing to the ubiquitous nature of the contaminant (Wiersma and Davidson, 1986). Average atmospheric residence time for particulate lead is suggested to be approximately ten (US DHHS, 1991) to thirty (NRC, 1972) days. Even in the early 1960s, researchers were observing that the air lead concentration in modern cities was continually on the rise (Einbrodt et al., 1968; Moyers et al., 1972). Such long-range transport of lead aerosol is reflected by the detection of accumulated lead in the Greenland ice cap (Murozumi et al., 1969).

1989; Ng and Patterson, 1981; Settle and Patterson, 1982; EPA USA, 1986; Davies, 1988; Nriagu and Pacyna, 1988). For such remote areas, levels fall into the concentration range of one to twenty nanograms per cubic metre of air ($ng.m^{-3}$; $1ng = 10^{-9}g$). Recorded levels from polar regions are usually less than one $ng.m^{-3}$, and occasionally 0.1 $ng.m^{-3}$ (EPA USA, 1986; Wiersma and Davidson, 1986), with a suggested level of $0.05ng.m^{-3}$ for natural lead in the atmosphere (EPA USA, 1986).

b) Rural areas

For moderately remote areas displaying no vehicular or industrial activity and virtually no development and lying only a moderate distance from reasonably-populated industrialised centres, lead-in-air levels of the order of 10 - $100ng.m^{-3}$ may be noted (Wiersma and Davidson, 1986). Such levels appear more elevated than for truly remote areas, and are most likely influenced by emissions from these proximate emission sources, essentially due to prevailing geo-meteorological conditions conducive to dust transport (EPA USA, 1986). A number of rural sites in the USA monitored over the early 1970s reflected a concentration in the range 45 - $140ng.m^{-3}$ (EPA USA, 1986). Kretzschmar et al., (1980) report slightly greater levels (mean = $174ng.m^{-3}$) for two Belgian sites, although it could be concluded that given a greater average European population density, and closer proximity to industrialised centres here than for US examples, greater influence from major atmosphere contaminating sources might have been experienced. This would seem compatible with the results of Roels et al. (1980) proffering an average lead-in-air level for rural Belgium of around $370ng.m^{-3}$. Lead-in-air concentrations for moderately developed rural locations typically fall in the range 100 - $300ng.m^{-3}$ (AAS, 1981), although this will obviously vary depending upon proximity to developed urban regions (DHSS, 1980; Chamberlain, 1983a & b; Jones and Stephens, 1983; RCEP, 1983).

c) Urban areas

Resulting from a more concentrated vehicle use and the greater likelihood of industrial emissions in urban areas (in reasonably to heavily industrialised countries), lead-in-air levels tend to be higher in urban compared with rural areas (Fergusson, 1986; El Saadi and Langley, 1991; Table 3.6). This would appear consistent with the same observation for lead in dust from rural and urban areas (Section 3.3.5 below), in that given the greater (urban) air lead concentrations, dust deposition (dustfall containing lead) would also be expected to be greater (Duggan and Inskip, 1985; EPA USA, 1986; Strehlow and Barltrop, 1987). Urban lead-in-air concentrations sampled over considerable periods of time (numbers of years) fall into the approximate range of 0.4 - 2.0 micrograms per cubic metre air ($\mu g.m^{-3}$) (var. sources - Table 3.6), AAS (1981) noting for Australian urban locations, a slightly greater figure of between 1.0 and $3.0\mu g.m^{-3}$. Shorter term monitoring, however, may reveal concentrations that are much greater, particularly proximate to busy main roads or industries such as lead smelters with peak (daily) levels often exceeding longer term averages by up to an order of magnitude (DHSS, 1980; NAS, 1980; Jones and Stephens, 1983a; RCEP, 1983; Fergusson, 1986).

Table 3.6 Urban, rural and remote lead-in-air levels from various studies.

SETTING	LOCATION	MEAN LEVEL ($\mu g.m^{-3}$)	SOURCE
URBAN	Australian urban (1970 - 1980)	1.0 - 3.0	AAS, 1981
	Australian urban - Sydney school (1975-1977)	1.1 - 2.4	Garnys et al., 1979
	Belgium (1978)	0.5	Roels et al., 1980
	Italy (1974-1979)	4.5	Facchetti and Geiss, 1982
	Major European cities (1975 - 1981)	0.4 - 8.0	Jones and Stephens, 1983
	UK (1976-1981)	0.5 - 0.9 (5 Yr. Ave = 0.65)	McInnes, 1982
	USA (1970s)	0.8 - 2.0	EPA USA, 1986
URBAN/ ROADWAY	Australian urban (1975 - 1980)	5.0 - 10.0	AAS, 1981
	UK (1973-1979)	6 - 25	Colwill and Hickman, 1981
	UK (1978)	0.4 - 2.2	Turner et al., 1980
SMELTER/ LEAD WORKS	Belgium (1974 - 1978)	2.7 - 4.1 Ave = 3.7 @ 1km Ave = 0.8 @ 2.5km	Roels et al., 1980
	El Paso - USA (1972 & 1977)	10.0 and 5.5 both @ 0.4km	Morse et al., 1979a
	(the former) Yugoslavia	24.2 - 38.4	Fugas, 1977
	" " "	21.2 - 29.2	Popovac et al., 1982
RURAL	Australia	0.1	AAS, 1981
	Belgium (1972 - 1975)	0.174	Kretzschmar et al., 1980
	Belgium (1978)	0.37	Roels et al., 1980
	" " "	0.370	Roels et al., 1980
	Italy (1976 - 1980)	0.33	Facchetti and Geiss, 1982
	UK (1974)	0.13	Cawse, 1974
	UK (1975 - 1981)	0.05 - 0.5	Jones and Stephens, 1983
	UK (1976 - 1979)	0.076	Davies, 1988 (citing Cawse, 1977, 78, 79 & 81)
	UK (1986)	0.008 - 0.077	McInnes, 1987

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	USA (1974)	0.13	Duce et al., 1975
SEMI-RURAL	UK (1957 - 1974)	0.130	Salmon et al., 1978
SEMI-REMOTE	USA (1969 - 1970)	0.008	Chow et al., 1972
	USA (1976 - 1977)	0.021	Elias and Davidson, 1980
	" " "	0.0022	Davidson et al., 1982
REMOTE	Antarctic (1974)	0.000076	Maenhaut et al., 1979
	Bermuda (1973 - 1975)	0.0041	Duce et al., 1976
	Greenland (1965)	0.0005	Murozumi et al., 1969
	" " (1978 - 1979)	0.008 & 0.018	Heidam, 1983
	" " (1979)	0.00015	Davidson et al., 1981
	Nepal (1979)	0.00086	Davidson et al., 1981
	Pacific Ocean (1979)	0.00017	Settle and Patterson, 1982

Source: data derive variously as indicated in table.

A suggested range for Australian urban environments proximate to roads bearing moderate to heavy traffic is between 5.0 and 10.0$\mu g.m^{-3}$ (AAS, 1981). Baker et al. (1979) investigating contamination by lead proximate to a lead chemicals-producing plant in Joplin, Missouri, note air lead levels consistently above the (then) recommended maximum level of 200$\mu g.m^{-3}$.

3.3.3.3 LONG-TERM AIR MONITORING

a) All locations

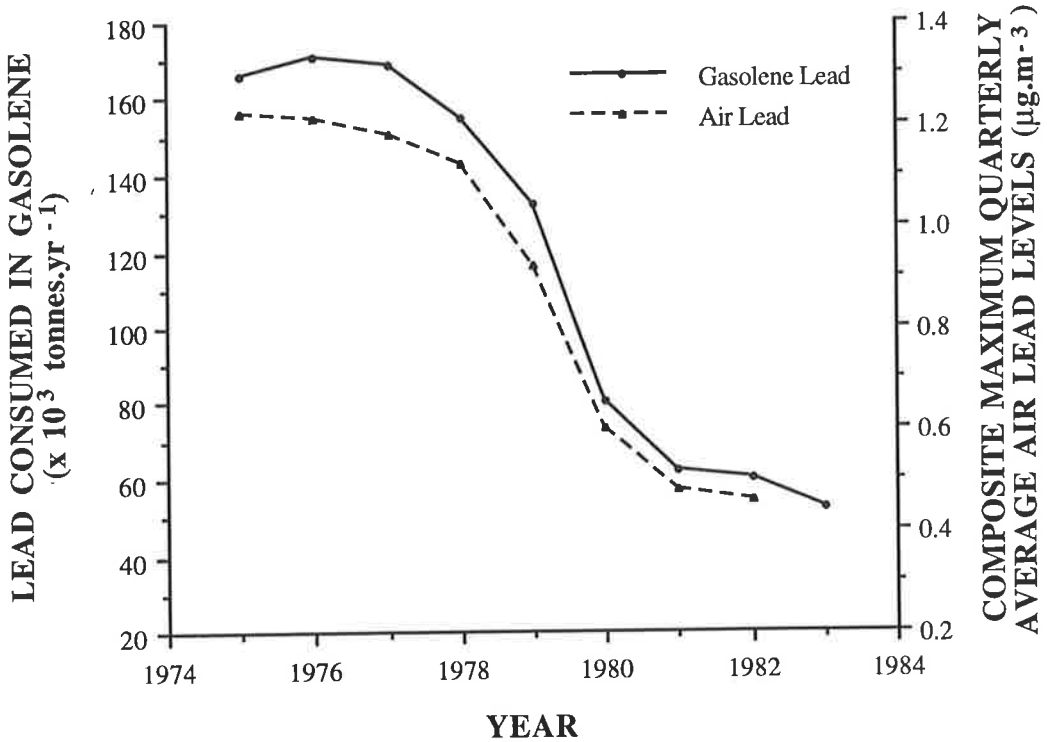
Long-term monitoring of lead-in-air levels may reveal interesting trends in concentrations, as for example the National Filter Analysis Network which boasts the most complete set of data on ambient air concentrations available (EPA USA, 1986). Over an eighteen year period of continuous monitoring, the mean lead level in air in US urban environments has reduced from 1.2$\mu g.m^{-3}$ (1970) through 0.8$\mu g.m^{-3}$ (1979), (EPA USA, 1986) to 0.1$\mu g.m^{-3}$ (1988), (EPA USA, 1989c & 1991). The greatest reductions in level occurred subsequent to 1977, such reductions being attributed almost exclusively to the success of the unleaded gasolene program. This program incorporated both a reduced (leaded) gasolene lead content and a move to the use of unleaded gasolene (Houk, 1982; Annest et al., 1983; EPA USA, 1986 & 1991; Davies, 1988; EPA NSW, 1993).

Undoubtedly in the mid to late 1980s, some reduction in lead-in-air levels in central London occurred as a result of a decreased consumption and emission of gasolene lead (Strehlow and Barltrop, 1987), the combustion of leaded gasolene constituting the principal source of particulate

lead in most urban atmospheres (van Borm et al., 1990). Much controversy abounds, however, regarding the research methodology applied here and the interpretation of the resultant data, given that a number of other environmental variables were not assessed over this study's duration (Elwood and Gallacher, 1984; Quinn, 1985).[16]

Davies (1988) citing McInnes (1987) notes that ambient air lead monitoring at 18 locations (rural, urban and kerbside) over the period 1984 to 1987 (encompassing January 1986, at which time the maximum permitted lead in gasolene level dropped from $0.40g.L^{-1}$ to $0.15g.L^{-1}$) reflected a range of reductions in level of between 24 and 74 percent, with an average all-site reduction of 53 percent. Page et al., (1988) reported similar findings (52 to 61 percent reduction) from data derived from four Welsh sampling stations. Ten years before the British reduction in gasolene lead maximum, an identical reduction was promulgated in (the then) Federal Republic of Germany, eliciting an air lead level reduction of between 20 and 65 percent corresponding to rural and inner city sampling stations respectively (Jost and Sartorius, 1979). Such observations may also relate to levels of lead in indoor dust, Jensen (1992) detecting a decrease in geometric mean lead level in Copenhagen homes from $240\mu g.g^{-1}$ in 1979 to $10\mu g.g^{-1}$ in 1990.

An interesting association between gasolene lead usage and mean air lead levels over the period of lead in petroleum products reduction in the USA becomes apparent when the data are viewed contemporaneously (Figure 3.5).

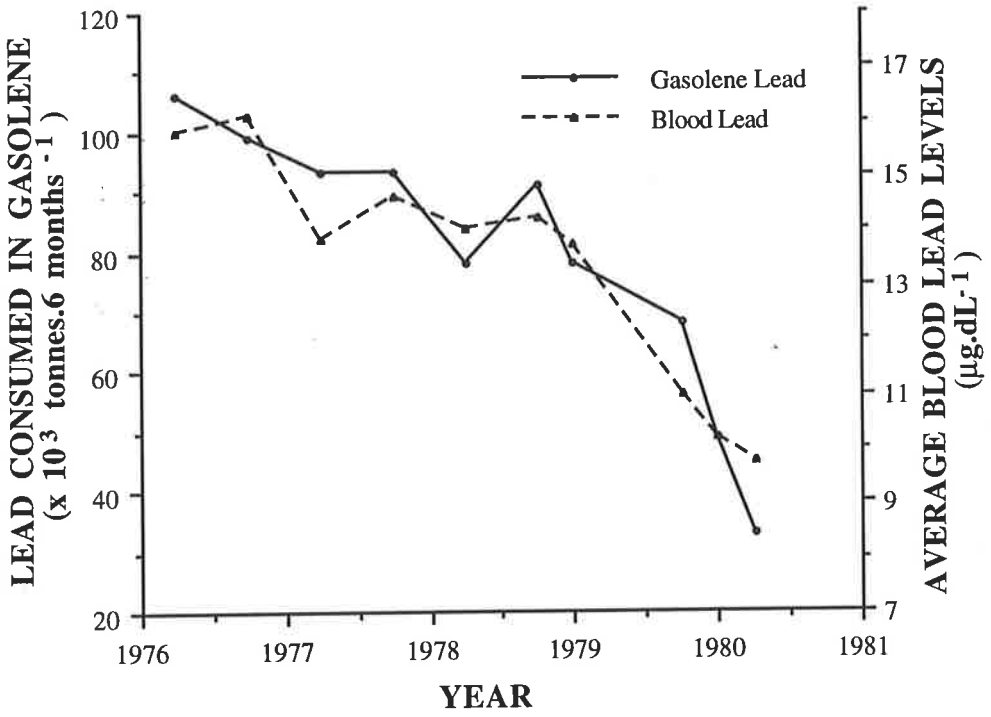


Source: data derive from Billick et al., 1980; Annest et al., 1983; EPA USA, 1984; Mielke et al., 1984; Fergusson, 1986; Shy, 1990.

Figure 3.5 US leaded gasolene consumption and relationship with airborne levels of lead.

[16] Certainly public awareness of the "perils of petrol" (unknown source) abounded, as did a move toward smaller-engined more fuel-efficient cars, some researchers suggesting that such factors would be difficult to control for and could be construed to constitute a confounding influence (Elwood, 1984 & 1986).

If mean blood lead levels are also plotted over the same period of time, a potential contributor to elevated blood lead levels on a population basis can readily be construed. Further, two exhaustive NHANES-(National Health and Nutritional Examination) Surveys conducted assessments of average blood lead levels in the USA, NHANES-1 conducted in February 1976 and NHANES-2 in February 1980. These national surveys revealed an approximate 37 percent decrease over that time (from 15.6 to 9.8$\mu g.dL^{-1}$ - Figure 3.6), a result that shocked the legislators (Berney, 1993).



Source: Mielke et al., 1984; p. 73.

Figure 3.6 Mean blood lead levels of US population, six months to 74 years.

This blood lead drop was quite obviously linked to a drop in the use of tetra-ethyl lead (53,000 to 24,000 tonnes over the time period [Maresky and Grobler, 1993]) by way of the introduction of unleaded motor spirit in 1976 (Annest, 1983). Such a relationship demonstrates high significance ($P < 0.001$), Billick et al. (1979 & 1982) claiming that blood lead levels tracked air lead levels very closely. Such time trends were similarly reflected by data from lead-poisoning screening programs (Annest et al., 1983; Schwartz et al., 1984; EPA USA, 1986).[17]

[17] Whilst some would suggest that such a downward shift in blood lead could also have arisen as a result of public awareness to the dangers of leaded paint, this would have been unlikely to explain an overall population reduction, since paint lead toxicity is predominantly associated with young children (Lin-Fu, 1982). Contrasting this, however, over the ten year period 1972 - 1982, a similar decline in average blood lead level in adult women in Wales was detected (Elwood, 1983a & b), as also for a study of both sexes between 1976 and 1984 (Elwood and Toothill, 1986). Both of these studies demonstrated very little contemporaneous change in the quantity of lead in gasolene, and an overall decrease in dietary lead. Elwood suggested thus that the results of the NHANES study were inconclusive with respect to the role of gasolene lead, and that "attempts to link a widespread fall in blood lead levels during this period to a change in lead levels in a single environmental source are naive" (1986; p. 3). A substantial decrease in the quantity of tetra-ethyl lead added to South African motor spirit over the years 1984 - 1990, is suggested by Maresky and Grobler (1993) to be associated with a significant contemporaneous decline ($P < 0.01$) in blood lead levels. Frank et al. (1990) proposed that higher teeth (dentine) lead levels were associated, in a number of different populations,

b) Smelter-proximate locations

A number of lead smelter/human health studies conducted during the 1960s, 70s and, particularly, the 1980s (although neither originally nor exclusively, eg as Oliver, 1914a & b), demonstrated that subjects living closer to smelters exhibited greater blood lead concentrations than those living farther away.[18] Further reported was that such point source (air) emissions gave rise directly to elevated lead levels in surrounding dust, soil, water and other media (Duggan and Inskip, 1985). Such levels of contamination were noted to decrease with increasing distance from the smelter, lead being absorbed by humans from air, dust and soil (Landrigan et al., 1975a, b & 1976a; Yankel et al., 1977).

Ambient air lead concentration "typically decreases with increasing vertical distance from emission source" (Schroeder, 1974; p. 144; as also Bazell, 1971; Lin-Fu, 1973a; EPA USA, 1986) and increasing horizontal distance from source, (typically exponentially - Roberts et al., 1974a; Yankel et al., 1977; Chamberlain et al., 1978; EPA USA, 1986; Fergusson and Stewart, 1992). Additionally, a number of researchers have observed that air lead levels are generally lower indoors than outdoors (Shephard et al., 1960; Jacobs et al., 1962; Biersteker et al., 1965; Yocom et al., 1971; Butler and MacMurdo, 1974; Sterling and Kobayashi, 1977; Chamberlain et al., 1978; DOE, 1978; Halpern, 1978; Alzona et al., 1979; Moschandreas et al., 1979; Cohen and Cohen, 1980; Diemel et al., 1981; Yocom 1982; Elwood, 1983 cited in Davies, 1986; EPA USA, 1986; Hamilton and Revitt, 1986; Murphy and Yocom, 1986; Davies, 1987; Davies et al., 1987b & 1990; Davies, 1990). Acknowledging these observations, comparisons of air-sampling results from different studies and for different locations may thus frequently appear inappropriate, given the number of sampling variables apparently not controlled for by researchers (Simmonds and Fergusson, 1983; Duggan, 1984; Elwood, 1986). These include sampling time, distance from presumed source and site topography, season and particular meteorological conditions eg wind speed, height above ground, location relative to other structures, sampling implement utilised, traffic density etc. (Fergusson, 1986; Fergusson and Stewart, 1992; Kim and Ferguson, 1994).[19] The mean size of airborne lead-bearing particles typically decreases with increasing distance from a point source (Daines et al., 1970).

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with greater mean gasolene levels of tetra-ethyl lead. Similar results from different countries were reported by others such as Chartsias et al. (1986), Ducoffré et al. (1990) and Gilli et al. (1990).

[18] These include Oyanguren and Perez (1966), McIntire and Angle (1972), Graovac-Leposavic et al. (1973), Nordman et al. (1973), Milham and Strong (1974), Roberts et al. (1974a & b), Clark (1975), Landrigan and Baker (1975a & b), Landrigan et al. (1975a, b, & 1976a), Baker et al. (1977a & b), Winegar et al. (1977), Yankel et al. (1977), Landrigan and Baker (1981), Popovac et al. (1982) and Berney (1993).

[19] From the literature, substantial seasonal variations in ambient air levels may occur. In cool, non-arid regions, winter air lead levels usually reach higher maxima than corresponding summer maxima (Faoro and McMullen, 1977; Strehlow and Barltrop, 1987; Moffat, 1989). This appears to be attributable to a more stable atmosphere elicited by a higher (winter) incidence of inversion conditions and reduced upward currents limiting dispersal (Cawse, 1975; Noller and Bloom, 1975; Little and Wiffen, 1978; Dorling and Sullivan, 1980; Strehlow and Barltrop, 1987; Kim and Ferguson, 1994). Port Pirie (high-volume sampler) data appear contradictory, with spring/summer air lead levels greater than winter levels (Maynard et al., 1993). Simmonds et al. (1983) report that when similar winter and summer wind direction and speed levels exist, similar air lead levels prevail. Blood lead levels may also demonstrate a seasonal variation with summer levels typically exceeding winter levels (Stark et al., 1982; Farfel, 1985; Rabinowitz et al., 1985a; Berney, 1993; Maynard et al., 1993). Moffat (1989) claims that such variation is also apparent with hand-adherent lead, yet Laxen et al. (1988) report greater floor dust and lead loading mean values for rooms of houses in winter.

Many researchers have investigated the emission patterns of lead smelters and monitored the resultant contaminant stream dispersion patterns. Lead-in-air concentrations appear to fluctuate on a daily basis, and are influenced principally by variations in rate of emission, wind direction and wind speed (Jannsens and Dams, 1975; Demuynck et al., 1976). Given that a number of these factors may change frequently during a sampling period (eg wind direction and intensity), various regulatory bodies specify long sampling times to attempt to reduce/account for some of the variability (Fergusson, 1986).

Partly for this reason, as will be seen in Chapter Nine, air sampling times in homes were chosen to be longer (approximately 15 days) rather than shorter (approximately 24 hours as for current high-volume sampler standards, where EPA USA propose continuous 24 hour sampling, once every 6 days [Fergusson, 1986]).

3.3.4 FOOD

3.3.4.1 DIETARY LEAD

Of the major sources of lead exposure for a general urban Australian population, the consumption of food and beverages constitutes the major route apart from direct contact with leaded dust deriving from automobile exhausts (EPA NSW, 1993). For children dust/soil lead and secondarily food/water lead are generally considered to be the two major sources of body lead (WHO, 1991). For young children living in a heavily-industrialised dusty environment, however, dietary lead may not be (proportionally) such a significant source, although still constituting a substantial pathway to elevating body lead burden (Duggan and Inskip, 1985; WHO, 1991). In clean, non-industrialised rural areas, dietary lead may be the predominant source for most individuals, particularly adults (EPA USA, 1986).

Whilst strictly not appropriate to this category, the contribution by alcohol consumption and cigarette smoking is noteworthy. From a variety of sources, high alcohol consumption and also cigarette smoking may contribute to a dietary lead intake similar to or even exceeding that of water (NAS, 1980; Pocock et al., 1983 & 1987; RCEP, 1983; Simms, 1986; Andrén et al., 1988). Lead may represent a component of cigarette smoke (Dockery and Spengler, 1981a; Spengler et al., 1981; Kalcher et al., 1993; Krause et al., 1993), and quite apart from being of a respirable particle size (Dockery and Spengler, 1981b), may settle on food or food-preparing surfaces or utensils, hence contaminating food (Gallacher et al., 1984).

Other miscellaneous (quasi-dietary) sources include lead in the chewed paint coating of pencils (Schaplowsky, 1971), chewing of toothpaste tubes (Lin-Fu, 1973a) and ingestion of toothpaste containing lead (Berman and McKiel, 1972; Shapiro et al., 1973a). Several workers refer to the ingestion of printing inks containing lead pigments as a potential contributor to paediatric lead poisoning (Hankin et al., 1973; Eaton et al., 1975).

3.3.4.2 SOURCES OF LEAD INTO FOOD

The entry of lead to food may arise as a direct result of crop uptake from soil (Tiller et al., 1976; Tiller and de Vries, 1977; Davies, 1978; Thornton et al., 1980; Merry et al., 1981; Davies and Crews, 1983; Thornton and Jones, 1984; Thornton, 1985; Temmerman and Vandermeiren, 1987; Li and Thornton, 1993), or stock grazing on contaminated pasture, eating both contaminated herbage and soil particles (Barltrop et al., 1975; Cartwright et al., 1976; Merry and Tiller, 1978; Thornton and Abrahams, 1983; Xu and Thornton, 1985). The significance of this may vary widely, however, depending upon the chemical nature (bioavailability) of the soil lead species (Colbourn and Thornton, 1978; Galloway et al., 1982).

Such passage to soil/vegetation may occur through the use of (lead)-contaminated water (eg sewage waste) to irrigate crops (Thornton, 1982) or as a result of aerial deposition of leaded dust onto plants or vegetables (Goodman and Roberts, 1971; Crecilius et al., 1974; Lagerwerff and Brower, 1974; Noller and Smythe, 1974; Roberts et al., 1974b; Beavington, 1975; Tiller et al., 1975; Tiller and de Vries, 1977; Yankel et al., 1977; Wong and Tam, 1978; Davies and White, 1981; MAFF, 1982; Chamberlain, 1983b; Davies, 1983a; Duggan and Burton, 1983; Ratcliffe and Beeby, 1984; Thornton and Jones, 1984; Thornton, 1985; EPA USA, 1986; Sherlock, 1987; Temmerman and Vandermeiren, 1987; Harrop et al., 1990; Moseholm et al., 1992; Onesanya and Ajewole, 1993). Some workers claim that this uptake can be a significant contributor to plant tissue lead content (EPA USA, 1986; Harrison and Johnston, 1987; Harrison and Chirgawi, 1988), other workers reporting to the contrary (MacLean et al., 1974; Zimdahl and Koeppe, 1977).[20] Kalac et al. (1991) note extensive accumulation of lead by two different mushroom species growing in soil located within six kilometres and nine species growing within one kilometre of a lead smelter in central Bohemia (former Czechoslovakia).

From work conducted by Tiller et al. (1976), CSIRO, (1977), Tiller and de Vries (1977) and Merry et al. (1981), vegetables grown on lead-contaminated soil in Port Pirie generally did not demonstrate markedly greater lead concentration values than those grown in other areas of SA. In general, Tiller et al. (1976) did not find a strong correlation between lead concentration in soil and in home-grown vegetables in Port Pirie, although this was not the case for some root crops. Compared with the lead that contaminates food during processing, packaging, and handling, the levels of lead in basic food crops and meats appears minimal (NRC, 1980). Summarising the work of a number of researchers, Davies (1988) reports a well-noted association between the distance from a road and a decline in grass/plant lead concentration for roadside plants, the source of the lead being mainly automobile exhausts. He notes, however, the difficulty in accurately apportioning the quantity of lead actually on the leaves (as dust fall-out - up to 100% of which could be effectively removed by adequate plant washing) from that contained within the tissue of the plant, concluding that, subsequent to appropriate washing, apart from lettuce, there appeared to be no measurable effect on the lead concentration of the edible portion due to (deposited) lead in air.

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[20] Lead-contaminated rainwater may be absorbed through the leaves of some plants (Davies, 1988). Thornton (1982) suggests that the lead concentration of various plant tissue may vary widely, some concentrating the contaminant in the roots, others transferring some to leaves and occasionally seed. Thornton (1985) also notes the bio-concentration of lead by radishes grown in contaminated soil.

A number of workers report that in general, by comparison with a variety of other contaminant metals, soil lead availability to plants is fairly low (Tiller, 1977; RCEP, 1983). Davies and Thornton (1988) claim that a direct relationship exists between soil and plant lead concentration. Davies and White (1981) report a retention of (formerly soil) lead, mainly in plant roots, with some translocation to edible leaves. Certainly, many factors influence lead absorption by plants, including variables such as plant type, root network, dimension, and soil pH, buffering capacity and organic material content. At normal soil pH levels, around pH ≈ 8, salts of lead are quite stable (Cartwright et al., 1976). Soil lead water solubility increases with decreasing pH such as may occur with the increase in soil sulphur as emitted from industrial process or the decaying of organic material resulting in the formation of more soluble and mobile heavy metal humic acid complexes (Bornstein and Bolter, 1991). These species' bioavailability and biological system disposition kinetics may vary markedly from the uncomplexed parent material (McKinney and Rogers, 1992). Typically for soil demonstrating a normal pH range, depositing lead becomes strongly complexed by soil organic matter, this bond being resilient and the species non-migratory (Berrow et al., 1987).

Potential exists for the contamination of foodstuffs at a local level as a result of local elevated soil contamination (Davies and Thornton, 1988), and the application of fertilisers containing elevated levels of a variety of heavy metals (Hutton and Symon, 1986). This would be expected to be the case in situations where the fertiliser derived from sewage sludges (Andersson and Nilsson, 1972; Richardson, 1980; Hutton and Symon, 1986), although Davies and Thornton (1988) note that such a form of lead has only a limited bioavailability to many plants. Such would also be the case for lead-containing pesticides (eg lead arsenate) although trends over the last decade suggest a general decrease in the use of lead for the production of chemicals used in the food-producing arena (ALDA, 1992). The availability of such materials for use at domestic level has declined substantially, although remnant material still exists in many home gardens (A. Langley, pers. comm.). Davies (1988) summarises the work of several researchers identifying substantially elevated food lead levels where food has been produced at domestic level in a variety of UK urban soil (eg Davies et al., 1979; Davies 1983a; Davies and Houghton; 1984; Denton, 1988). Such observations have also been made in the USA (Preer and Rosen, 1977; Preer et al., 1984).

Dairy products, meat and fish (marine and fresh water) have also been reported variously as potential sources of lead in food, although typically where such a contribution is considered to be significant, it occurs in areas of considerable local contamination of waters (Davies, 1988) and/or sediments (Dong et al., 1984).

Regular market basket surveys reveal that over the last two decades, average Australian food lead levels are decreasing (AAS, 1981; NHMRC, 1987 & 1990; EPA NSW, 1993), particularly subsequent to the introduction of unleaded gasolene in 1985. This appears to reflect a heightened awareness of food production and processing techniques regarding the potential for lead contamination (NHMRC, 1990; ALDA, 1992). It also represents a decrease in direct atmospheric fallout of leaded particles onto foodstuffs, this being suggested as the major pathway for lead contamination of foods currently in Australia (Anon., 1995a).[21]

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21 Food here is classed as "table-ready condition", hence incorporating the potential for contamination by (say) lead-containing dust within the home environment.

Subsequent contamination of food may also occur as a result of contaminated preparation surfaces (Gallacher et al., 1984; Moffat, 1989), food preparation/serving/storage in contaminated water (Berlin et al., 1977; Moore et al., 1977a; Moore, 1979; DHSS, 1980; Little et al., 1981; Smart et al., 1981 & 1983) or inappropriate containers such as soldered metal cans (Settle and Patterson, 1980; RCEP, 1983; Farfel, 1985; EPA USA, 1986; Sherlock et al., 1987; EPA NSW, 1993). Also indicted is the use of lead-crystal glassware (Rossol, 1992) pewter (CEH & CAPP, 1987), or inappropriately-glazed (leaded) ceramic ware (Leonard and Lynch, 1958; Klein et al., 1970; Osterud et al., 1973; Barltrop and Meek, 1975; DHSS, 1980; Bird et al., 1982; Wallace et al., 1985; CEH & CAPP, 1987; Mushak and Crocetti, 1990; Anon., 1995a). This is particularly so if the foodstuff concerned is of an acidic nature (Schaffner, 1981; Duggan, 1983b), tomatoes being a good example (Anon., 1995a).[22]

It is acknowledged that national lead-in-air levels are decreasing, particularly in urban areas, as a result of the decreasing use of leaded gasolene. Lead, however, is very soil stable (Fergusson et al., 1980; Brunekreef et al., 1981; Gulson et al., 1981) and persistent (Cartwright et al., who, commenting upon soil contamination arising from the smelter located at Port Pirie, claimed that "the existing contamination will last indefinitely" [1976; p. 80]).[23] Soil lead levels would thus not be expected to be decreasing in this region, rather, increasing, but at a decreasing rate (EPA NSW, 1993). Such a situation is suggested to represent a long-lasting risk factor for child intoxication (Silvany-Neto et al., 1989). Contrasting this with a dramatic decrease in the consumption of dietary lead as a result of decreasing food lead content both in Australia (NHMRC, 1990) and in the USA (US DHHS, 1991), it would appear likely that decreased aerial deposition of leaded dust and improved food preparation and storage techniques would be largely responsible for the decrease in food lead levels. In fact, reported food lead levels may be somewhat high, external surface leaded dust contamination biasing the actual food lead content (Kacprzak, 1982), Fergusson (1986; p. 18) stating that this further emphasises "the importance of dust as a lead source".

Both for the 1987 and the 1990 Australian food surveys, all food lead intakes were estimated to be below to well below the Provisional Tolerable Weekly Intake level. For children living in Lead Hot Spot areas such as inner city locations, or proximate to point sources of lead emission where contaminant levels are particularly high (eg smelters), however, it is suggested that the quantity of lead in the diet is likely to be substantially greater than market basket estimates (EPA NSW, 1993). Given the substantial decreases in food lead concentration over the last two decades, Mushak and Crocetti (1990) conclude that for the USA, lead in food had largely been eliminated as a potential major source of body lead for children in general. These authors, however, do acknowledge that in special cases, food lead (often arising from aerial deposition of lead onto food) may still elicit significant

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[22] Wallace et al. (1985) claim that coffee may leach substantial quantities of lead from a non-new lead-glazed cup in as little time as 15 minutes. Lead compounds such as lead oxide, carbonate and monosilicate may become quite soluble when fused as a glazing layer, particularly when in contact with moderately low pH foodstuffs (Bertic and Stahuljak, 1961). These authors report on 40 patients from a small village in (the former) Yugoslavia, all of whom suffered from the effects of lead-rich pickled foods. Lead may also be leached by some citrus fruit juices (Klein et al., 1970) and home made wines (Whithead and Prior, 1960). In 1754, John Lind warned that lemon juice or wines should not be stored in glazed earthenware jugs, Sir George Baker commenting similarly in 1772 for the production and storage of cider, for which he was pronounced from the pulpit as a faithless son of Devon (Meiklejohn, 1954).

[23] Whilst some literature suggests that lead in soil is to some extent bio-available, others refer to its stability, eg, locally, the abandoned Dean Rifle Range with its bullet-riddled "hill of lead" was to be levelled as part of a rehabilitation plan for the area, and the earth was to be harvested for lead, copper and zinc, the Draft (MFP) EIS indicating little risk of "biological assimilation" of heavy metals thus constituting a health risk (PPK, 1992; p. 168).

exposure (Mushak and Crocetti, 1992). Louekari et al. (1991) conclude that dietary lead intake is not accurately reflected by blood lead levels.

Beverages also may contain lead as a direct result of the processing and storage techniques undertaken. Anecdotal evidence suggests that illegal home "moonshine"-distilling apparatus utilising discarded motor vehicle radiators as condensers has contributed to a sizeable number of lead intoxication patients' admittance to hospital for chelation therapy (Palmisano et al., 1969; Chisholm, 1971; Lin-Fu, 1973a; Chapter Four), the acidic nature of the brew leaching the lead from the solder holding the radiator and piping together (Chisholm, 1971; SRC, 1990).

3.3.5 DUST

Included in this general review of dust is a variety of dust types including soil surface dust (essentially the top layer of soil), street dust, air-suspended dust, and principally (internal) house dust.

3.3.5.1 AN OVERVIEW OF THE ROLE OF DUST

From a review of the literature over the last three decades, it would appear that many workers have concluded that children living in lead-contaminated areas, particularly an urban/industrial environment, display elevated blood lead levels (compared with those not living in such an environment) and that the inhalation of airborne lead is not sufficiently cogent an explanation for the mechanism leading to such elevated levels (EPA USA, 1977; Duggan, 1980; Duggan and Inskip, 1985; EPA USA, 1986; SRC, 1990; IPCS, 1991).

Whilst the premise for many years was that the major contribution to most children's lead intake was from lead in food and drink and that inhalation of lead-bearing dust/soil particles was a significant route (Duggan et al., 1985), early this century it was noted by an Australian physician, whilst observing normal child behaviour, that ingestion of lead via soil or dust by mouthing dirty hands (and also painted objects) was a potential source of lead (Gibson et al., 1892; Gibson, 1904).[24]

Recent work as part of the Three Cities Study has demonstrated the less-than-expected importance of soil lead as a determinant of child blood lead levels, placing further emphasis on the role of dust as a predictor of blood lead levels (Aschengrau et al., 1991; Clark and Bornschein, 1993; Farrell et al., 1993; Weitzman et al., 1993). Humans, certainly and virtually universally, contact materials having derived from the Earth's crust such as sediments, soil, and dust, with dust constituting the most pervasive of such materials (Fergusson and Kim, 1991). Teege (1968) reports that dust is ubiquitous in the surrounding atmosphere, its continuous deposition leading to relentless soiling of surfaces.

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[24] Whilst some work relating to lead in dust concentrations has been reported from as long ago as the 1930s, (MOH, 1930; Dunn and Bloxam, 1933; Kaye and Reznickoff, 1947) it has really only been a phenomenon of the last two to three decades that researchers have concluded that dust may constitute a significant mechanism of trace metal exposure and potential contribution to body burden to residents of (particularly) an urban society (Duggan and Inskip, 1985; Fergusson and Kim, 1991). Contrasting this, however, soil lead concentrations have been interpreted much more as contaminant source indicators, and consequently have been much more extensively dealt with by researchers (Duggan and Inskip, 1985).

As is the case for soil particles, the concentration of lead in dust is also suggested to increase with decreasing particle size (Randall et al., 1975; Archer and Barratt, 1976b; Barratt, 1976a; Dorn et al., 1976; Duggan and Williams, 1977; Spitler and Feder, 1979; Biggins and Harrison, 1980; Fergusson et al., 1980; Hopke et al., 1980; Linton et al., 1980; Johnson et al., 1982; Fergusson and Ryan, 1984; Duggan and Inskip, 1985; Davies and Watt, 1986; Steele et al., 1990; Fergusson and Kim, 1991; Moseholm et al., 1992), this relationship appearing to continue to the sub ten micron particle size range (Duggan et al., 1985). Contributing to the appreciation of the role of lead-bearing dust as a major pathway to child lead intoxication has been the work of a number of researchers demonstrating modest to high levels of dust/lead on children's hands associated with observations of the frequency of hand-to-mouth activity (Lepow, 1974 & 1975; Day, 1975; Duggan and Williams, 1977). Milar et al. (1980) conclude that blood lead levels vary with the quantity of lead on children's hands, and that the extent of environmental contamination is a better predictor of blood lead levels of children than either sociological or demographic factors. Included below is a more detailed discussion of a number of the pathways of adventitious lead into humans.

3.3.5.2 LEADED DUST AS A PATHWAY TO HUMANS

Suggestions by a number of workers of intakes by children of the order of 20 - 200µg lead per day via the dust pathway compare with or exceed estimated levels of intake via food or water, and infer that uptake of lead via inhalation is small by comparison (Duggan and Inskip, 1985; Duggan et al., 1985; Ferguson, 1986). Supporting this premise, it has been noted by a number of researchers that children living in lead-contaminated environments demonstrate a peak blood lead level at two to three years of age, coinciding with an observed peak in hand-to-mouth activity at that time (reviewed by CDC, 1991). Such a peak in blood lead levels is not observed in children living in protected environments (Duggan et al., 1985), eg as addressed by work conducted by Haas et al. (1973), Zielhuis et al. (1978) and Sartor and Rondia (1981), where the blood lead level of long-term hospital child patients was monitored.

Dust both indoors and outdoors may contain significant quantities of lead (Vostal et al., 1974; Landrigan et al., 1976a; Yankel et al., 1977; Roels et al., 1978; Sayre, 1981; Stark et al., 1982; Charney et al., 1983; Duggan, 1983a; Mielke et al., 1983; Yaffe et al., 1983; Elwood and Gallagher, 1984; Duggan and Inskip, 1985; Duggan et al., 1985; Body, 1986d & f, Body et al., 1988), particularly in developed areas (Fergusson and Kim, 1991), external dust "reservoirs" constituting the major source of contamination of the indoor dust system (Allott et al., 1992). Such dust may provide quite a reasonable index of the degree of contamination of such an area (particularly house dust; Duggan and Inskip, 1985; Fergusson and Kim, 1991), with an exponential decline in contaminant levels (both dust lead concentration and dustfall) with increasing distance from the (presumed) source being a common observation (Bolter et al., 1977; Clevenger et al., 1991a). The contribution of lead to house dust from such (principal) sources as the deposition of (atmospheric) gasolene combustion and industrial-emission products, paint lead, street dust and soil lead will obviously vary between different locales, and also for different seasons and meteorological conditions (eg wind direction and speed) at the same location (Vostal et al., 1974; Lepow et al., 1975; Rabinowitz et al., 1976; Solomon and

Hartford, 1976; Harrison, 1979; Sayre and Katzel, 1979; Charney et al., 1980; Diemel et al., 1981; Sayre, 1981; Johnson et al., 1982; Moorcroft et al., 1982; Duggan, 1983a; Fergusson and Schroeder, 1985). The relative contribution of environmental lead to house dust (and its importance) has, however, been a subject for much controversy and disagreement (Vostal et al., 1974; Solomon and Hardtford, 1976; Sayre and Katzel, 1979; Charney et al., 1980; Diemel et al., 1981; Sayre 1981; Johnson et al., 1982; Moorcroft et al., 1982; Duggan, 1983b; Fergusson and Schroeder, 1985; Fergusson and Kim, 1991; Gulson et al., 1995).

3.3.5.3 SOIL AND HOUSE DUST

There exists a complicated dynamic relationship between a number of the environmental lead "pools" or compartments such as street dust, various house dust populations and soil (Duggan and Inskip, 1985). Soil-derived particles may constitute a component of all above-mentioned dust types, potential existing for the enrichment of some soil-derived elements in dust (Fergusson and Kim, 1991). Reasonable correlations have been demonstrated between soil and internal house dust lead levels (Jordan and Hogan, 1975; Diemel et al., 1981; Reeves et al., 1982 and Cameron et al., 1983; Fergusson and Kim, 1991), and also between external and internal dust lead levels (Solomon and Hartford, 1976; Duggan, 1980). Fergusson and Kim, from a multi-element assay of soil and corresponding house dust (1991), have noted a strong association in the concentrations of a range of approximately 40 elements, suggesting that soil is a significant contributor to/component of house dust. These authors have identified elements such as Arsenic, Copper, Cadmium, Lead and Zinc under various different circumstances, claiming that they arise from soil originally, this being the contention of other workers also (eg Harrison et al., 1981a; Culbard and Johnson, 1984; Duggan and Inskip, 1985; Duggan et al., 1985; Thornton et al., 1985; Brunekreef, 1986; Thornton, 1988).

3.3.5.4 SOIL, STREET DUST AND HOUSE DUST

Such a dynamic process is also suggested to exist between soil and street dust, such dust being capable of subsequently entering the house and significantly contributing to dust contaminant levels and also aerosol contaminants. This is especially so as far as elements such as lead are concerned (Diemel et al., 1981; Fergusson and Schroeder, 1985). Importantly here, a comprehension of the dynamics of indoor dust deposition and its possible re-entrainment/re-suspension merits considerable attention (as Hamilton and Revitt, 1987; Fergusson and Kim, 1991).

3.3.5.5 OTHER LEAD CONTRIBUTORS TO DUST

Whilst sources of anthropogenic dust lead contamination are manifold, in an urban setting, one of the more significant contributors, apart from previously lead-contaminated soil, would be leaded gasolene use. Much work relating to this topic appears in the literature, with evidence for the source of the lead deriving from work suggesting a direct relationship between dust contaminant levels and

traffic density (Ter Haar and Aronow, 1974; Archer and Barratt, 1976a & b; Duggan and Williams, 1977; Fergusson et al., 1980; Ellis and Revitt, 1982; Lau and Wong, 1982; Fergusson and Simmonds, 1983; Hamilton et al., 1984; Nasralla, 1984; Ali and Nasralla, 1985; Duggan et al., 1985; Brown, 1986; Brunekreef, 1986; Harrison, 1986; Day, 1977; Warren and Birch, 1987; Lee et al., 1988; Schwar et al., 1988a & b; Valerio et al., 1988; Kazemi, 1989). Also inferred, is an inverse relationship between dust contaminant levels and sampling site distance from roadways (Archer and Barratt, 1976a; Solomon and Hardtford, 1976; Duggan, 1984; Beckwith et al., 1985). Such a relationship has also been observed for the proximity of sampling site and contaminant point source such as a mining region, a smelter (Landrigan et al., 1975a & b; Muskett et al., 1979; Gallagher et al., 1984; Thornton et al., 1985) or gasolene stations (Davies et al., 1987a).

Further, other sources may include home lead activities such as the burning of fossil fuels or old car batteries at home for heating (Lin-Fu, 1973a).[25] Other home-activities include leadlighting (Anon., 1985), the lead glazing of pottery (Anon., 1985; Baxter et al., 1985; Anon., 1995a), fishing/diving weight production (Morton et al., 1982; Rossol, 1992), type-printing and quench-hardening of cutlery (Kawai et al., 1983; Jensen and Laxen, 1985), glazing of windows (Schwar et al., 1988b), working with artists' paints (Rossol, 1992), and the (domestic) use of some insecticides (Fleming and Parle, 1977; EPA USA, 1984). Most commonly suggested is the redecoration of homes involving the removal of lead-based paints by sanding (Feldman, 1978; Stark et al., 1982; McBride, 1984; Chisholm et al., 1985b; Fergusson and Schroeder, 1985; Manton, 1985; Chisholm, 1986; Rundle and Duggan, 1986; Inskip and Hutton, 1987; Laxen et al., 1987a & 1988; Raab et al., 1987; Strehlow and Barltrop, 1987; Schwar et al., 1988b; Davies et al., 1990; Farfel and Chisholm, 1990 & 1991; Marino et al., 1990), chipping (Landrigan et al., 1982) and blasting (Landrigan et al., 1982; Alperstein et al., 1991). Paint burning is also potentially dangerous, since it may give rise to lead fume, this having been noted particularly with "do-it-yourself" heat gun use (Fischbein et al., 1981; Inskip and Atterbury, 1983; CEH & CAPP, 1987; Inskip and Hutton, 1987; Laxen et al., 1987a & b; Rey-Alvarez and Menke-Hargrave, 1987; Farfel and Chisholm, 1991). Thus, methods for the safe and effective abatement of lead-based paint in houses have yet to be developed.[26]

Many such home-activities may generate and distribute lead within the house, considerably elevating dust lead levels. Such activity may consequently elicit intoxication of residents, particularly children, some specific examples being discussed in greater detail in Appendix Five. Subsequent to abrasive removal of paint, highly-contaminated dust may remain within the confines of the dwelling (Kinateder, 1995), sometimes for greater than a year despite intensive and regular cleaning (Inskip and Hutton, 1987; Roberts et al. 1991b). Such contaminants remain captured in, particularly, carpet (Rey-Alvarez and Menke-Hargrave, 1987) and cracks in floors (Rundle and Duggan, 1986).

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[25] Dolcourt et al. (1980) report on two North Carolina families that burnt discarded batteries within the house for heating and also the illicit recycling of battery lead on the kitchen stove, resulting in encephalopathy in a resident child, ultimately leading to permanent brain damage. Interestingly, this was not the first incident of such a nature, Williams et al. reporting similarly nearly fifty years previously (1933).

[26] Chisholm et al (1985) have observed that when lead-poisoned children are returned to "lead abated" structures, their blood lead levels invariably increase to unacceptable levels. Charney et al. (1983) report that repeated dust removal following lead paint abatement was associated with a modest reduction in occupant children's blood lead levels whereas lead paint abatement alone was not. Traditional practices of abating residential lead-based paint would appear to have been shown to be associated with acute increases in levels of lead (concentration and loading) in house dust and (subsequently) the blood of children and workers (Feldman, 1978; Fischbein et al., 1981; Rabinowitz et al., 1985a & b; Amitai et al., 1987; Rey-Alvarez et al., 1987; Farfel and Chisholm, 1990 & 1991).

Sources contributing lead to dust may, at times, appear controversial. From a review of the Centre for Disease Control (Atlanta) data, Stark et al. (1982) conclude that a significant number of children with undue lead absorption occupy buildings that were previously inspected for lead-based paint hazards, and found to display no such hazard. For nearly half of these confirmed cases of elevated blood lead, no possible source of lead paint could be discovered, similar to the findings of Kennedy (1978). Thus, the sourcing of contributors to (say) soil lead contamination, and therefore inferentially dust contamination, is an important public health issue, and one that warrants further investigation.

3.3.5.6 LEAD LEVELS - EXTERNAL AND INTERNAL DUST

The discussion below focuses on the relative variability in contaminant levels under different circumstances and is separated into externally- and internally-derived dust samples. No universal standards for the collection, storage, pre-analysis treatment and analysis of dust currently exist. This elicited a strong concern to be expressed more than a decade ago by a number of workers including Duggan and Inskip (1985) and Fergusson (1986). Such concerns infer problems regarding inter-study data comparison given the different sampling/analytical techniques undertaken (Appendix Six). Despite these concerns, there is still value to be derived from an exploration of such data in order to provide some insight into the dimension of the contamination and its spatial variability.

a) External dust

From a review of a number of literature-reported studies, dust lead concentrations from such sources are summarised below (Table 3.7). Most of these data derive from the 1970s and 1980s when it would appear that such investigations were most frequently conducted (pers. obs.).

Table 3.7 Range of contamination for external dust from various studies.

SETTING	EXTERNAL DUST LEAD CONCENTRATION (Range of mean values - $\mu g.g^{-1}$)	NUMBER OF SURVEYS
RURAL	35 - 190	10
URBAN	200 - 2,800	25
INDUSTRIAL	2,000 - 6,000	17

Sources: data derive from various sources (eg Dunn and Bloxam, 1933; Kaye and Reznickoff, 1947; Hunt et al., 1971; Hogger, 1973; Lepow et al., 1974 & 1975; Roberts et al., 1974b; Day et al., 1975; Archer and Barratt, 1976a; Duggan and Williams, 1977; Farmer and Lyon, 1977; Roels et al., 1978; Czuchajowska et al., 1980; Fergusson et al., 1980; Hopke et al., 1980; Roels et al., 1980; Rundle and Duggan, 1980; Brunekreef et al., 1981; Harrison and Laxen, 1981; Ellis and Revitt, 1982; Lau and Wong, 1982; RCEP, 1983; Chan and Chu, 1984; Dong et al., 1984; Fergusson and Ryan, 1984; Hamilton et al., 1984; Nasralla, 1984; Ali and Nasralla, 1985; Anagnostopoulos et al., 1985; Beckwith et al., 1985; Davies et al., 1985; Duggan and Inskip, 1985; Duggan et al., 1985; Jensen and Laxen, 1985; Brown, 1986; Fergusson et al., 1986; Davies et al., 1987a & b; Fergusson, 1987; Strehlow and Barltrop, 1987; Culbard et al., 1988; Davies, 1988; Fergusson and Kim, 1991).

- i) Remote and rural

Typically, rural areas reflect a lower dust lead concentration, with some of the surveys reviewed suggesting no real increase in concentration levels in rural regions above what is termed background (Duggan and Williams, 1977; Day et al., 1985).

- ii) Urban

Mean urban dust lead concentrations tend to be higher, typically by an order of magnitude compared with those from rural areas. Nasralla (1984) indicates a rather low mean figure for lead in street dust derived from the surrounds of Jeddah ($7.5\mu g.g^{-1}$; presumably, essentially normal background levels), with a corresponding mean value for the streets of the inner city (lying between 125 and $925\mu g.g^{-1}$) approaching two orders of magnitude greater. Three of the studies reviewed (Day et al., 1975 & 1979; Duggan and Williams, 1977; Strehlow and Barltrop, 1987) sampled consistently and analysed equivalently, and their data for rural and urban settings (incorporating geometric means) can be compared. These data confirm the results of other work (eg Day, 1977), and also the earlier-suggested order of magnitude difference between urban and rural dust lead concentrations. Also noted is a fifteen-fold greater dust lead deposition (leadfall) rate for London ($800\mu g.m^{-2}$ per day [$\mu g.m^{-2}.d^{-1}$]) compared with the level of $50\mu g.m^{-2}.d^{-1}$ for the (rural) Suffolk site.

Hunt et al. (1971), from a survey of 77 US cities, note a mean lead in external dust concentration of $1,600\mu g.g^{-1}$, Fergusson et al. (1980) citing a mean level of $2,800\mu g.g^{-1}$, not dissimilar to the figure of $2,600\mu g.g^{-1}$ for Hong Kong streets proffered by Lau and Wong (1982). London street dust reflects a mean level of around $1,500\mu g.g^{-1}$ and $1,000\mu g.g^{-1}$ for major and minor roads respectively (Duggan and Williams, 1977). Schwar et al. in a more recent study (1988) report a significantly higher (mean) street dust lead concentration in more developed inner-city areas than in the more peripheral London suburbs. A mean of $2,000\mu g.g^{-1}$ for Lancaster street dust has been reported (Harrison and Laxen, 1981), with approximately $1,000\mu g.g^{-1}$ for street dust from Glasgow (Farmer and Lyon, 1977), Manchester (Day et al., 1975) and Birmingham, (Archer and Barratt, 1976a & b). This Birmingham study's results confirmed the observation for the difference between arterial, main, and residential road sites with (median) levels of lead in road dust of 1,800, 1,300 and $950\mu g.g^{-1}$ respectively, again inferring that vehicle emissions constitute a major dust lead source at such locations.

- iii) Surrounding the home

Duggan and Inskip, reviewing many of the earlier studies, raise a concern regarding the relative paucity of samples deriving from areas "where children might be expected to play and/or spend a considerable amount of time" (1985; p. 20), as opposed to the ever-popular footpath, road, or road gutter. This criticism would seem not to have been adequately addressed in later work by other

researchers, with areas such as front and rear porches of houses, footpaths, sand pits/play areas close to the house not receiving sufficient attention.

- iv) Industrial regions

Industrial locations reflect lead in dust concentrations that are considerably greater than those for, principally, urban areas, with mean (or median) concentrations generally lying in the range of 2,000 - 6,000$\mu g.g^{-1}$ for surveys involving more than (say) ten samples being taken (Archer and Barratt, 1976a; Czuchajowska et al., 1980; Roels et al., 1980). Mean concentrations do range widely, however, (Davies, 1988) with single measurement concentrations well above 10,000$\mu g.g^{-1}$ having been recorded in the near vicinity of lead smelters (Roberts et al., 1974b; Archer and Barratt, 1976b; Elwood et al., 1977; Turner, 1980; Turner et al., 1980). Occasional values approaching 100,000$\mu g.g^{-1}$ (ten percent) have been reported, (Davies, 1988), Farfel concluding that in an industrial society/setting, "lead poisoning is endemic" (1985; p. 333). Speculation arises as to whether these high levels arise principally as a result of settled stack or low-level emissions or from tracking of lead-rich dust from the works on vehicles (Jensen and Laxen, 1985) or the wheels of vehicles (Turner, 1980).

b) Indoor dust

Duggan and Inskip (1985) and Davies (1988) both remark upon the relative paucity of lead in indoor dust data deriving from essentially rural areas; a review of subsequent literature provides only a very modest addition. Certainly the contrary is the case for urban, industrialised or mineralised soil areas. Most of the subsequent discussion pertains to urban studies with some data deriving from rural areas being used for comparative purposes. Reviewing the literature, it quickly becomes obvious that there exists a wide range of individual lead in dust concentrations reported. Also evidenced is a wide variation in reported mean/median house dust lead concentration levels deriving from a wide variety of geographic locations, a point that would not appear surprising to Fergusson and Kim, given their comment "lead is the most polluting metal, in the urban environment" (1991; p. 128).

Mean and/or median house dust lead concentration values lie in the range 300 to 1,250$\mu g.g^{-1}$ (17 studies) with two particularly high sets of results, namely a median figure of 3,500μg.g-1 (DOE, 1982) and an arithmetic mean of 11,000$\mu g.g^{-1}$ (Lepow et al., 1975). Again a concern is raised regarding the variability in applied sampling procedures and analytical technique and also data interpretation applied by the various authors thus affecting the validity of making real comparisons between all different sets of results.[27]

A review of the literature reveals the following as a summary of data pertaining to indoor dust (Table 3.8).

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[27] On occasion, no reference is made as to whether samples were fractionated on the basis of component particle size (ie sieved), although a number of the workers selected merely the sub 1,000 or sub 500 micron fraction of the dust as their test dust. As will be demonstrated in Chapter Five, a significant variation in lead content of varying particle size ranges exists for dusts.

Table 3.8 Indoor dust lead levels from various studies.

SETTING	LOCATION	MEAN DUST LEAD CONC. &/OR RANGE ($\mu g.m^{-3}$)	SOURCE
RURAL	-	-	-
URBAN			
Home-sampled	Birmingham - UK	2,780 100 - 280,000	Archer and Barratt, 1976a
	Hartford - USA	11,000 -	Lepow et al., 1974 & 1975
	Illinios - USA	617 170 - 1,440	Solomon and Hartford, 1976
	Memphis - USA	404 -	Baker et al., 1977a & b
	New York - USA	1,240 -	Rice et al., 1977
	Omaha - USA (urban)	572 ± 147	Angle et al., 1974
	Omaha - USA (suburban)	335 ± 68	Angle et al., 1974
	Omaha - USA	- 110 - 610	Angle and McIntire, 1975
	UK	1,263 39 - 30,060	RCEP, 1983
	USA	250 -	Milar and Mushak, 1982
Schools & offices	Birmingham - UK (schools and homes)	- 30 - 157,000	DOE, 1982
	Newcastle - UK (offices)	510 -	Dunn and Bloxam, 1933
URBAN/ INDUSTRIAL			
Home-sampled from smelter workers' homes	Leeds - UK	1,024 107 - 2,275	LCC, 1982
	Memphis - USA	2,687 -	Baker et al., 1977a & b
	New York - USA	3,310 -	Rice et al., 1977
	USA	3,000 -	Milar and Mushak, 1982
Near smelter 0 - 1.6km	El Paso - USA	22,190 2,800 - 103,750	Landrigan and Baker, 1981
Near smelter ≥ 6.4km	El Paso - USA	973 200 - 22,700	Landrigan and Baker, 1981
Near smelter	Canada	2,626 355 - 8,750	Roberts et al., 1974a & b
	Canada	2,388 200 - 8,500	Roberts et al., 1974a & b

TABLE CONTINUES OVER PAGE

	TABLE CONTINUED FROM PREVIOUS PAGE		
	London - UK	2,850 500 - 8,700	Turner et al., 1980
	Netherlands	1,054 463 - 4,741	Brunekreef et al., 1981
Near paint-removal works	USA	7,580 4,930 - 10,000	Landrigan et al., 1982
Elevated soil lead $\leq 1{,}000\mu g.g^{-1}$	UK	531 190 - 2,450	Barltrop et al., 1975
Elevated soil lead $1{,}000\text{-}10{,}000\mu g.g^{-1}$	UK	1,564 390 - 25,000	Barltrop et al., 1975
Elevated soil lead $\geq 10{,}000\mu g.g^{-1}$	UK	2,582 1,000 - 5,100	Barltrop et al., 1975

Source: data derive as indicated; presentation format adapted from Duggan and Inskip, 1985; p. 21.

3.3.5.7 URBAN VERSUS RURAL DUST LEAD SCENARIOS

Strehlow and Barltrop (1987) note an approximate 15 fold greater level of dust lead concentration from London streets ($1{,}881\mu g.g^{-1}$) as compared with those derived from Suffolk (rural) streets ($144\mu g.g^{-1}$), although the corresponding comparisons for house dust demonstrated only a three fold greater level of lead for London ($850\mu g.g^{-1}$) as compared to Suffolk ($\approx 300\mu g.g^{-1}$). Further, their observation that Suffolk house dust reflects more lead than does the corresponding street dust (by approximately a factor of four), would suggest, given that the area is presumed to be a non-contaminated environment, that internal sources may also give rise to elevated dust lead levels in house dust (Duggan and Inskip, 1985; Fergusson and Kim, 1991).

Contrasting this observation for rural areas, Davies and Thornton (1987), Strehlow and Barltrop (1987) and Culbard et al. (1988) report for urban areas, respectively, a sixty, fifty and thirty percent lower level of lead in indoor dust than corresponding road dust. This is quite in keeping with Duggan and Inskip's observation (1985) that concentrations of lead in house dust tend to be lower than those in outdoor dust (or soil) when the latter are unusually high (no level proffered) and *vice versa*.

Schwar et al. (1988) also claim lower levels for rural compared with urban residential dust, and distinct high dust lead concentration areas associated with areas demonstrating industrial activity, as Fergusson and Kim (1991). Archer and Barratt's investigation of Birmingham house dust (1976a) revealed a median figure of around $1{,}000\mu g.g^{-1}$. Duggan and Inskip, (1985; p. 23) claim that such a (high) value, and high upper limit individual values (as high as $157{,}000\mu g.g^{-1}$; confirmed by DOE [1982]) "strongly suggest a contribution from an internal source such as leaded paint or other non-vehicular exhaust sources". Harrison (1979), Landrigan et al. (1982), Culbard et al. (1983), Culbard and Johnson (1984), and Thornton (1988) also report internal sources as being potentially significant contributors to indoor dust lead levels. This is particularly so, they suggest, if external dust and soil

lead concentrations are not greatly elevated, and only if they (paint and non-vehicular exhaust) are the major outside contributors of trace elements to the house dust, as suggested by Fergusson and Kim (1991).

Alternatively, there is a suggestion that there exists some sort of dust lead concentrating mechanism or "lead trap" provided by a house, Sayre and Katzel (1979) claiming that, facilitated by poorly sealing windows (particularly in older-styled houses), the collection of surface dust inside homes is a surface phenomenon and "an active and dynamic process . . . . at least in part related to atmospheric air levels of lead" (1979; pp. 181 - 182).[28] This suggestion will be explored in Chapter Eight dealing with contemporaneous leadfall and lead-in-air sampling. Contrasting this internal source concept, the lead trap may in fact also arise as a direct result of the selective enrichment of some trace metals during the house-cleaning process. For example, the vacuum cleaner may selectively retain only the larger dust particle sizes by the failure to capture, particularly from carpet-covered surfaces, the finer dust particles, as has previously been reported by Body (1986f). As will be demonstrated in Chapter Five, typically the heavy metals are concentrated in the smaller-sized (denser) particles, as noted by Duggan and Inskip (1985) and Fergusson and Kim (1991).

Culbard et al.'s (1988) comprehensive baseline UK indoor dust study (4,638 randomly selected house dust samples) presented a geometric mean lead in indoor dust level of 561$\mu g.g^{-1}$, quite commensurate with the above-mentioned range of mean levels. Archer and Barratt's (median) figure of 1,000$\mu g.g^{-1}$ deriving from approximately 800 dust samples from schools, shops and offices (1976a), suggests that such commercial sites present contaminant levels slightly greater than residential sites, a finding of other workers also (eg DOE, 1982). Albeit acknowledging that concentrations vary substantially with location, Fergusson and Kim's rather comprehensive review of the literature (1991) also suggests a mean or median dust lead concentration of around 300 - 1,000$\mu g.g^{-1}$, again aligning with other workers' results.[29]

It should be noted here that the selection of houses tested in many of these above-mentioned studies was not random. It is by no means coincidental that the two studies reflecting the greatest mean/median levels for dust lead concentration in an urban setting, (Lepow et al., 1975; DOE, 1982) arrived at these (high) levels, since both were targeted at homes of children that displayed, respectively, chronically-elevated and elevated blood lead levels. Hence, the above-cited range of 300 to perhaps 1,250$\mu g.g^{-1}$ would appear a reasonable estimate for the range of mean lead concentrations for house dust.

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[28] For timber houses, a clear direct relationship exists between house age and lead content of house dust (Harper et al., 1987; Culbard et al., 1988; Kim and Fergusson, 1988; Fergusson and Kim, 1991), although this does not appear to hold for brick style houses, where, according to Fergusson and Kim (1991), the concentration of lead in dust is relatively constant at around 500$\mu g.g^{-1}$. For vacant houses, this house age/house dust lead relationship appears not to hold (Sayre and Katzel, 1979), Fergusson and Kim (1991) inferring that such an observation suggests that activity in the house helps to generate the dust.

[29] Subsequent to reviewing other workers' data, Fergusson and Kim conclude that "there is remarkable agreement between different studies" (1991; p. 128), although they acknowledge concerns regarding variability between sampling and analytical techniques, yet they make little reference to differences in data interpretation techniques eg geometric means compared with arithmetic means compared with medians.

3.3.5.8 HEAVILY-INDUSTRIALISED REGIONS

Contrasting the relative shortage of house dust data deriving from rural environments, there exists quite an extensive data base for research conducted in houses in industrialised areas, although again the choice of non-random locations for investigation is frequent. Sampling has typically been conducted in homes of lead workers eg Baker et al. (1977), Rice et al. (1977), LCC (1982), and Milar and Mushak (1982). Mean dust lead levels of between (approximately) 1,000$\mu g.g^{-1}$ and 3,000$\mu g.g^{-1}$ have been frequently noted, such a choice of test house, however, making some comparisons complicated. These data suggest dust lead levels in homes in substantially industrialised regions are greater than those experienced by dwellers of a more residential urban setting.

Recorded levels of dust lead concentration in Australian, British, European and United States' homes proximate to a smelter works have also frequently been investigated, such studies often revealing high levels of contamination (eg McIntire and Angle, 1972; Angle et al., 1974; Roberts et al., 1974a & b; Angle and McIntire, 1979; Turner et al., 1980; Brunekreef et al., 1981; Fergusson et al., 1981; Landrigan and Baker, 1981; Popovac et al., 1982; Angle et al., 1984; Prpic-Majic et al., 1984; Body, 1986d & f; Body et al., 1988a & b; Graziano et al., 1990; Langlois et al., 1996). Proximate to a large smelter in El Paso, Texas, Landrigan and Baker (1981) reported an arithmetic mean house dust lead level (n = 53) of close to 22,000$\mu g.g^{-1}$, with substantially reduced mean levels (973 $mg.g^{-1}$) at a distance of six and a half kilometres from the works. At other locations the following was noted in houses close to smelter works (Table 3.9).

Table 3.9 Lead in house dust data derived from lead-smelter districts.

LOCATION	MEAN DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	RANGE ($\mu g.g^{-1}$)	SOURCE
Arnhem - NETHERLANDS	1,050 (Arith.)	460 - 4,700	Brunekreef et al., 1981
London - UK	-	500 - 8,700	Turner et al., 1980
Toronto - CANADA	1,550 (Geom.)	470 - 8,400	Roberts et al., 1974a & b

Source: data derive as indicated above.

These authors confirm that house dust contaminant levels in industrialised regions generally lie below roadside and outdoor dust levels respectively. This is in keeping with data for indoor and outdoor dust lead levels in homes in non-industrial, urban residential areas as noted by Duggan and Inskip (1985), Culbard et al. (1987), Davies and Thornton (1987) and Strehlow and Barltrop (1987).

3.3.5.9 MINERALISED AREAS

House-acquired dust from mineralised areas has been reported by a number of workers as displaying elevated contaminant levels (Barltrop et al., 1975, Culbard et al., 1988). Both groups proffered (geometric) mean levels close to 1,200$\mu g.g^{-1}$ for areas in Derbyshire, UK. This observation is in keeping with other workers (Cotter-Howells, 1991; Cotter-Howells and Thornton, 1991; Gulson et al., 1994b), where it is apparent that house dust contaminant levels remain high, either naturally (ie from natural soil weathering), or as a result of prior human activity. Contrasting these observations are the data of Davies et al. (1985) who, referring to an historic lead-mining region of north eastern Wales, note only a moderate dust lead level elevation from expected background levels to 340$\mu g.g^{-1}$.

3.3.5.10 DUST MOVEMENT DYNAMICS

This discussion regarding lead in dust and its traffic/dynamics, at times may be complemented by a contemporaneous discussion pertaining to other trace metal movements, the premise being that the movement of dust contaminant trace metals as a population will fairly represent the movement of individual components in the dust phase. This would appear to be in keeping with concepts discussed by Fergusson and Kim (1991).

a) House structural factors

Any comprehensive discussion relating to indoor dust should focus upon dust entry into and retention by the house itself. Accepting that a house doesn't provide an absolute barrier to dust penetration (Yocom et al., 1971; Butler and MacMurdo, 1974; Butler et al., 1975; Sterling and Kobayashi, 1977; Davies et al., 1987b & c; Koutrakis et al., 1991 & 1992), trace metals in dust or aerosols may penetrate houses to varying degrees, and eventually settle given sufficiently stable conditions within the house (Settle and Patterson, 1982). Given sufficiently strong within-home air movements, however, dust may be subsequently re-entrained (Alzona et al., 1979; Crockford, 1992), such a transfer process warranting further investigation. Thus, trends in concentrations of trace metals from outside to inside and also within houses can isolate sources (Fergusson and Kim, 1991). This is particularly so with respect to material that is either externally or internally distinctively sourced (Archer and Barratt, 1976a & b; Sayre and Katzel, 1979; Diemel et al., 1981; Duggan and Inskip, 1985; Bornschein et al., 1986; Davies et al., 1987a). As an example, (internal) house dust lead concentrations have been demonstrated to be maximal on the inside of window wells or sills (independently of whether they have been painted or not), or close to windows, with decreasing concentration gradients radiating away from the window (Archer and Barratt, 1976a; Sayre and Katzel, 1979; Sayre, 1981; Bornschein et al., 1986; Davies et al., 1987a). This would suggest that a proportion of the lead present in the house derives from external sources, a presumed entry point for such dust being a window (Archer and Barratt, 1976a; Sayre and Katzel, 1979; Fergusson, 1986).

There exists some earlier research noting that the concentration of lead in house dust tends to be lower than that in outdoor dust (or soil) when the latter is unusually high (Sayre and Katzel, 1979). These authors suggest that poorly-fitting windows allow the entry of airborne dust from outside by facilitating "an active dynamic process to occur at or near windows" (p. 181). From previous work conducted in Port Pirie, there appeared little evidence inferring lower internal dust lead concentrations than home soil lead levels (Body, 1986i; Body et al., 1988).

b) House occupant influences

Another potentially-significant mechanism of entry for contaminant dust arises as a result of employees of (say) a smelter demonstrating poor personal hygiene and carrying home dust in/on their clothes, on footwear or on their bodies (Butler and MacMurdo, 1974; Archer and Barratt, 1976b; Ouw and Bisby, 1976; Solomon and Hartford, 1976; Baker et al., 1977a & b; Rice et al., 1977 & 1978; CDC, 1978; Chisholm, 1978; DOE, 1978; Dolcourt et al., 1978; Fielding and Russo, 1978; Millar, 1978; Fergusson et al., 1981; Landrigan and Hayes, 1981; Milar and Mushak, 1982; Morton et al., 1982; Kawai et al., 1983; Landrigan and Bridbord, 1984; von Lindern, 1984; Duggan and Inskip, 1985; Jensen and Laxen, 1985; Ryu et al., 1985; Brunekreef, 1986; Moore, 1986; Crockford, 1992; Ewers et al., 1994). The socks of lead workers have been noted to carry very high lead levels, particularly those most involved with lead contact (Archer and Barratt, 1976a), although these authors acknowledge that such levels could be associated with lead excretion via sweat. Lead workers' motor vehicles may also constitute a source of home lead contamination (Turner, 1980; Landrigan and Hayes, 1981; Jensen and Laxen, 1985; Clark and Bornschein, 1990; Chiaradia et al., 1997).

Lead-smelting employees with elevated blood lead levels, typically exhibit high house dust lead levels which correlate well with their blood lead levels (Duggan, 1983). The homes of such smelter employees may reveal house dust contaminant concentrations enriched for a number of elements by a factor of between three and twenty six times (Rice et al., 1977; Fergusson et al., 1981). Other suggested potential sources of contaminant entry into the house are the feet/shoes of children or adults contacting external contaminant hot spots (Archer and Barratt, 1976a; Harrison, 1979; Schwar et al., 1988a; Farago et al., 1994), and also pet (especially dog) entry (Brunekreef et al., 1983). Unfortunately, scant lead loading data are available from any of these studies.

c) General comments

The results of a number of studies, appear to reflect clearly-apparent seasonal variability in environmental dust lead concentrations (Cawse, 1974 - 1981; Noller and Bloom, 1975; Little and Wiffen, 1978; Dorling and Sullivan, 1980; Strehlow and Barltrop, 1987; Kim and Ferguson, 1994). Further, good correlation has been demonstrated between the level of home contamination externally and internally (Diemel et al., 1981; Davies et al., 1985; Budd et al., 1990; Camann et al., 1990; Lewis et al., 1994). It is acknowledging that the variability in sampling, sample preparation, analysis and data-interpretation techniques applied will influence the presented values. From the obvious variability

in indoor dust lead levels resultant from a review of a number of the published works, it would still seem likely, however, that at least some variability in internal dust lead levels arises as a result of local climatic/seasonal conditions (Laxen et al., 1988).

3.3.5.11 DEPOSITING LEAD AND LEAD LOADING DATA

a) Depositing dust lead (leadfall)

The literature reveals that little work appears to have been conducted in this area apart from a few, limited studies eg Angle et al. (1974 & 1984), Roberts et al. (1974a &b), Barltrop (1975), Landrigan et al. (1975), Turner et al. (1980), Diemel et al. (1981), RCEP (1983), Clark et al. (1985), Strehlow and Barltrop (1987), Davies (1988). The findings of some of these are referred to below.

In rural areas in the UK, the mean annual deposition rate of lead (in dust) in outdoor locations is of the order of 55$\mu g.m^{-2}.d^{-1}$ (RCEP, 1983), levels in a (previously) heavily lead-contaminated rural region approximating 250$\mu g.m^{-2}.d^{-1}$ (Barltrop et al., 1975). Urban locations in the USA and the UK have been reported as reflecting leadfall levels of the order of 375$\mu g.m^{-2}.d^{-1}$, (Angle et al., 1974) to 800$\mu g.m^{-2}.d^{-1}$ (Roberts et al. 1974a; RCEP, 1983; Strehlow and Barltrop, 1987) with a figure of around 965$\mu g.m^{-2}.d^{-1}$ for central London (RCEP, 1983). For areas in the general vicinity of point sources such as a lead smelter, detected levels appear to be nearly an order of magnitude greater (3,000 - 8,220$\mu g.m^{-2}.d^{-1}$- eg Barltrop, 1975a; Landrigan et al., 1975; Turner et al., 1980; Davies 1988). Close to a lead smelter, reported mean leadfall levels include 3,460$\mu g.m^{-2}.d^{-1}$ (Dorn et al., 1975), 5,270$\mu g.m^{-2}.d^{-1}$ (Roberts et al., 1974a), 6,800$\mu g.m^{-2}.d^{-1}$ (Landrigan and Baker, 1981) and 8,065$\mu g.m^{-2}.d^{-1}$ (Barltrop, 1975a). Certainly the availability of earlier-derived data pertaining to leadfall rates within dwellings appears to be limited, a mean figure of approximately 7$\mu g.m^{-2}.d^{-1}$ (range 0.3 - 1,900) having been referred to in two urban studies (Diemel et al., 1981 and Clarke et al., 1985). More recently, Chiaradia et al. (1997) report a range for homes of lead miners in a rural setting of 0.3 - 28.3$\mu g.m^{2}.d^{-1}$, compared to mean figures for urban environments such as Sydney and Brisbane of 2.9 and 1.9$\mu g.m^{2}.d^{-1}$ respectively.

b) Lead in dust loading

Similarly, despite a substantial amount of literature-sourced work relating to dust lead concentrations having been conducted, surprisingly scant attention has been directed towards the assessment of within-home lead loading data. This will be referred to again in Chapter Seven. A review of some studies is summarised in Table 3.10 below. When assessing the lead loading of an area, the resultant value will obviously depend upon the technique utilised and the sampling intensity (as Fergusson and Schroeder, 1985), with inter-study comparison of results frequently lacking some validity due to the different techniques utilised in the various studies.

Table 3.10 A review of lead loading data from the literature.

LOCATION	MEAN DUST LEAD LOADING ($\mu g.m^{-2}$)	RANGE ($\mu g.m^{-2}$)	SURFACE	SOURCE
Arnhem - Nthrlnds.	165	20 - 880	Floors	Diemel et al., 1981
Auckland - NZ	50 - 225	-	Floors/window sills	Reeves et al., 1982
Birmingham - UK	60	4 - 600	Floors	Davies et al., 1990
Christchurch - NZ	100	5 - 4,040	Floors	Kim and Fergusson, 1993
Cincinnati - USA	810	40 - 70,000	Floors	Bornschein et al., 1986
Rochester	2,060 - 6,980	100 - 87,310	Floors/window sills	Sayre and Katzel, 1979
and	12,748	-	Floors below windows	" " "
Buffalo - USA	2,230	-	Floors 3m in from windows	" " "
Rochester - USA	-	22- 5,210	Floors	Vostal et al., 1974
Rochester - USA	1,320 - 2,840	-	Floors/window sills	Charney et al., 1980
Various - USA	105 - 310	-	Floors	Rice et al., 1977
Various - USA	850 - 1,200	-	Floors	Rice et al., 1978

Source: data derive variously as indicated.

From a study by Kim and Fergusson (1988), the metal loading for lead, copper, zinc and cadmium in house dust correlates strongly with the degree of "dustiness" of the house. Further, dustiness itself correlates strongly with the amount of carpet wear, with a ten-fold increase in dustiness from new to threadbare carpets corresponding to a seven-fold increase in the loading of cadmium, copper and lead, with only a two fold increase in zinc. Fergusson (1986) notes that the dust loading may vary from a few to several thousand milligrams per square metre, such a dust loading not necessarily correlating with the concentration of lead in the dust (Fergusson and Schroeder, 1985).

3.3.5.12 HOUSE DUST SOURCE CONTRIBUTORS

Potential exists for a number of external (to the house) sources to contribute to house dust (as for street dust), although given the potential for indoor sources to also contribute trace metals to indoor dust under some circumstances, such dust appears more complex with respect to source (Fergusson and Kim, 1991). Johnson et al. (1982) claim that possibly half of house dust is internally-generated, with Culbard et al. (1983) reporting that 20 - 25% derives from soil. This is similar to Fergusson (1986) who suggests that 40% of house dust is organic, and 30% derives from soil, the remainder deriving from a variety of other sources (Fergusson and Schroeder, 1985). Given the smelter presence in Port Pirie, and acknowledging substantially elevated (above background) lead levels in soil (Tiller et al., 1975 and 1976; Tiller and de Vries, 1977; Body, 1986e), dust (Body 1986d), tank water (Body,

1986g) and air (Goh and Hope, 1992), it would appear likely that the majority of lead contamination indoors would arise as a result of the entry of externally-derived contaminant material. This will be pursued in further detail in Chapters Seven and Eight. A number of factors would thus appear to affect the concentration of lead in house dust, some of which are listed in Table 3.11 below.

Table 3.11 Suggested factors affecting house dust lead concentration.

FACTOR	SOURCE
Area of exposed soil	Culbard et al., 1988
Distance from:	
commercial garages	Chambers et al., 1984; Davies et al., 1987a & 1990
mining/smelting activity	Landrigan et al., 1975a &b; Muskett et al., 1979; Thornton et al., 1985
roads	Harper et al., 1987; Culbard et al., 1988
Dustiness/carpet wear	Kim and Fergusson, 1988 & 1993
Dustfall/suspended particles inside	Diemel et al., 1981
Home decoration activity involving paint removal by sanding [30]	CDC, 1978; Fielding and Russo, 1978; Landrigan et al., 1982; Manton, 1985; Chisholm, 1986; Rundle and Duggan, 1986; Davies et al., 1987b & 1990; Inskip and Hutton, 1987; Laxen et al., 1987a & 1988; Raab et al., 1987; Strehlow and Barltrop, 1987; Culbard et al., 1988; Schwar et al., 1988b; Farfel and Chisholm, 1990 & 1991; Marino et al., 1990
House age	Harrison, 1979; Culbard et al., 1983a & 1988; Davies et al., 1985; Thornton et al., 1985; Harper et al., 1987; Kim and Fergusson, 1988 & 1993
Presence of:	
a fireplace	Kim and Fergusson, 1990 (unpubl. data); Fergusson and Kim, 1991
old paint	Sayre et al., 1974; Sturges and Harrison, 1985; Bornschein et al., 1986; Culbard et al., 1988
Street dust contaminant level	Diemel et al., 1981; Fergusson and Schroeder, 1985; Culbard et al., 1988; Kim and Fergusson, 1988 & 1993
Traffic density	Quarles et al., 1974; Little and Wiffen, 1978; Wong and Tam, 1978; Wheeler and Rolfe, 1979; Fergusson et al., 1980; Agrawal et al., 1981; Garcia-Miragaya et al., 1981; Fergusson and Simmonds, 1983; Simmonds et al., 1983; Mielke et al., 1984; Lyngbye et al., 1987; Kim and Fergusson, 1988; Fergusson and Kim, 1991; Nam et al., 1994

Source: Adapted from the original in Fergusson and Kim (1991), supplemented by Kim and Fergusson (1993) and various others as indicated.

[30] A recently-sanded, lead-painted house may display dust containing several percent lead (Fergusson, 1986). Levels may remain high for up to a year subsequently even with frequent cleaning (Farfel and Chisholm, 1990).

3.4 ENVIRONMENTAL LEAD LEVELS; EXPOSURE RISKS AND HUMAN CONTACT

The opportunity for human contact with lead via a number of pathways clearly exists. The following section deals with an appraisal of the work conducted by a number of researchers detailing estimates of the potential intake of body lead as a result of contact with different potential sources of (environmental) lead.

3.4.1 LEAD INTAKE BY INSPIRATION OF CONTAMINATED AIR

Certainly there have been many studies attempting to establish a relationship between ambient air levels and associated blood levels of lead. Under conditions of high lead-in-air levels, the relationship appears to become non-linear (EPA USA, 1986; SRC, 1990), although at air lead levels less than approximately 3.5$\mu g.m^{-3}$, the relationship would appear to be approximately linear (SRC, 1990).[31] For children, blood lead/air lead slopes range from 1.52$\mu g.dL^{-1}$ per $\mu g.m^{-3}$ (Yankel et al., 1977; Walter et al., 1980) through 1.92 (Angle and McIntire, 1979) to 2.46 (Roels et al., 1980), with a median blood lead/inhalation slope of approximately 1.9$\mu g.dL^{-1}$ blood per $\mu g.m^{-3}$ air inspired (SRC, 1990). A corresponding figure for adults is of the order of 1.25$\mu g.dL^{-1}$ (Gross, 1979), 1.32$\mu g.dL^{-1}$ (Azar et al., 1975) through 1.75$\mu g.dL^{-1}$ (Griffin et al., 1975) to 2.14$\mu g.dL^{-1}$ (Rabinowitz et al., 1973, 1976 & 1977), a weighted average slope estimate being approximately 1.6$\mu g.dL^{-1}$ blood per $\mu g.m^{-3}$ air inspired (SRC, 1990).

3.4.2 LEAD INTAKE BY FOOD

Reflecting the decrease in the use of leaded fuel and soldered food cans in the USA over the last two decades, it has been suggested that average food lead concentrations and also estimated average dietary intake of lead have decreased from around 210μg per week ($\mu g.wk^{-1}$), for an average two year old child, in 1982 to 90$\mu g.wk^{-1}$ in 1985 to 35$\mu g.wk^{-1}$ in 1988 (CDC, 1991).[32] Australian data infer a level of 100$\mu g.wk^{-1}$ of dietary lead for an average nine month old infant and 115 - 205$\mu g.wk^{-1}$ for an average two year old child (NHMRC, 1987). The contribution to blood lead levels from the consumption of water containing lead at a level < (approximately) 100$\mu g.L^{-1}$ is reported to be a blood lead increase of the order of 0.06$\mu g.dL^{-1}$ (blood) per each μg(lead).L^{-1} of water (Pocock et al.,

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[31] Given that normal ambient air lead levels (for other than areas proximate to high traffic density or industrial activity) reside at levels typically substantially less than the above-mentioned figure, the relationship can be assumed to be linear, although for air lead levels > 3.5$\mu g.m^{-3}$, composite figures have been arrived at for the impact upon blood lead levels of people residing in such areas (SRC, 1990).

[32] This estimate incorporates daily intake of fluids, suggested to be of the order of 0.03litres.kg^{-1} body weight for an adult and 0.05litres.kg^{-1} body weight for a child (McKone and Daniels, 1991). Settle and Patterson (1980) suggest food intake lead for Americans at the time was approximately 0.2$\mu g.gm^{-1}$ food, compared with dietary lead intake for prehistoric peoples to be of the order of 0.002$\mu g.gm^{-1}$. The intake for formula-fed infants may be substantially higher in areas where tap water demonstrates elevated lead levels (WHO, 1991).

1983). Typically though, the contribution to body lead burden from drinking water is small (IPCS, 1991), except when substantial quantities of lead have been used for water-reticulation systems (Raab et al., 1985) and where water is particularly plumbo-solvent (Moore et al., 1985). This is reported to be the case in certain parts of the UK (Moffat, 1989), with Elwood et al. (1984) suggesting, for Welsh women, that water lead provides a greater contribution to blood lead than air lead.

Given such estimates for dietary lead intake, combined with reported dietary lead absorption coefficients (Chapter Four), a blood lead concentration increase of 0.04$\mu g.dL^{-1}$ [population study] - 0.2$\mu g.dL^{-1}$ [infant study] per microgram of ingested lead per day total intake would seem appropriate (Ryu et al., 1983; EPA USA, 1986; EPA USA, 1989b). This would suggest, for Australian children, an average (dietary) increase (above background) in blood lead levels of approximately 2.3 - 3.6$\mu g.dL^{-1}$ for infants and 2.8 - 4.1$\mu g.dL^{-1}$ for a two year old child (EPA NSW, 1993).

3.4.3 LEAD INTAKE BY CONSUMPTION OF SOIL AND DUST

3.4.3.1 SOIL

Many workers have proffered estimates of mean child soil ingestion rates, EPA USA (1989b) citing a figure of around 100mg soil per day. Hoffnagle and DeCesar (1987) reviewed the literature, and, for a smelter community, cited a figure of 60$mg.d^{-1}$ for the quantity of dietary soil lead contributing to mean blood lead level, other workers suggesting similarly (eg Hawley, 1985). For the purposes of quantifying the contribution of ingested soil containing lead to an individual's blood lead level, El Saadi and Langley (1991) consider that there is no useful purpose distinguishing between contribution to blood lead level made by the consumption of contaminated dust or soil, there being a reasonable correlation between lead levels in either environmental compartment. Given the variability in soil type, sampling regime, depth of soil tested, and age of children involved in various studies relating soil lead to blood lead levels, direct comparison of results becomes difficult. EPA USA (1986) estimates a blood lead concentration increase of 0.6 - 6.8$\mu g.dL^{-1}$ for each increase of 1,000$\mu g.g^{-1}$ in soil lead for towns with operating smelters. This figure was based upon research conducted by Roberts et al. (1974a & b), Yankel et al. (1977), Angle and McIntire (1979) and Walter (1980). Stark et al. (1982) cite a median figure of around 2$\mu g.dL^{-1}$ per each 1,000$\mu g.g^{-1}$ soil lead, a significantly smaller figure than that proffered by El Saadi and Langley (1991).

A number of studies relating to the soil lead/blood lead relationship have revealed conflicting results (Elwood, 1986; Wixson and Davies, 1991), differences being demonstrated between the effect of soil lead deriving from mining or smelting waste (Hemphill et al., 1991; Wixson and Davies, 1991). Steele (1990) claims that the former is less bioavailable due to larger particle size and different particle chemical form.[33] Further, Bornschein et al. (1986 & 1988) infer a decreasing dose-response

[33] Hemphill et al. (1991) claim that mean particle size emitted from ore mining or milling sources is of the order of 100 microns (μ) (range = 10 - 1,000μ), whilst that for smelting sites is of the order of two microns (range = 1 - 5μ). Gulson et al. (1993) report that global conclusions (as Steele et al. [1990]) regarding the lower impact on blood lead levels of occupants of a town demonstrating soil and dust contaminated by mine-waste are not necessarily valid, each mining community needing to be assessed individually.

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relationship over increasing soil lead levels. A number of workers, however, claim that soil's role as a contributor to blood lead levels is more in an indirect sense (eg Bornschein et al., 1988), ie soil surface scrapings do not correlate directly with blood lead levels, but indirectly via house dust (Steele et al., 1990). Wixson and Davies (1991) claim, however, that the major influence on the relationship between soil and blood lead relates more to child activity than any particular characteristics of soil.

3.4.3.2 DUST

Under most circumstances, the individual most exposed to contaminated soil will be a child aged between approximately two and three years (McMichael et al., 1988; ANZECC, 1992). For dust ingestion, Duggan and Inskip (1985) proffer a similar age range. EPA USA (1986) suggest that every dust-lead incremental increase of 1,000$\mu g.g^{-1}$ will elicit an increase in resident child blood lead levels of the order of 3 - 7$\mu g.dL^{-1}$. It is acknowledged, however, that individual child habits, dust bioavailability and absorption/retention factors and local conditions may alter this relationship considerably (El Saadi and Langley, 1991). Reviewing the literature, provides data that would appear consistent with EPA USA's (1986) estimate, as indicated below (Table 3.12).

Table 3.12 Estimated mean blood lead increase resulting from a dust lead increase.

MEAN BLOOD LEAD INCREASE ($\mu g.dL^{-1}$) PER EACH 1,000$\mu g.g^{-1}$ INCREASE IN DUST LEAD *	SOURCE
1.5	Yankel et al., 1977
2.3	Barltrop et al., 1974
2.5	Galke et al., 1978
3.5	Roels et al., 1980
3.9	Shellshear et al., 1975
4.0	Angle and McIntire, 1979
5.0	Reeves et al., 1982
5.2	Schmitt et al., 1979
6.0	Roberts et al., 1974a & b
6.8	Watson et al., 1978
8.6	Baker et al., 1977a
9.0	Milar and Mushak, 1982
MEAN (MEDIAN) 4.9 (4.5)	

* Some of these values are averages from presented ranges.

Source: data derive from indicated references.

3.4.4 MAJOR ENVIRONMENTAL COMPARTMENT CONTRIBUTIONS TO BLOOD LEAD

3.4.4.1 ENVIRONMENTAL LEVELS: EXPOSURE/EFFECT INDICATORS

A (conservative) summary or aggregate estimate of the slope for ambient air lead's contribution to blood lead levels for (particularly) urban-dwelling children incorporating both direct (respiration) and indirect (via soil and dust) pathways has been proposed by EPA USA (1986), and appears in Table 3.13. Typically, such aggregate values have been reported as 3 - 5$\mu g.dL^{-1}$ per $\mu g.m^{-3}$ lead in air (Brunekreef, 1984; EPA USA, 1986). Angle et al. (1984) infer that the indirect soil/dust contribution to increases in blood lead is approximately 4 - 5$\mu g.dL^{-1}$ above the direct respiration contribution. Testing the estimate revealed that predicted blood lead values demonstrated reasonable agreement with measured values for a variety of children living at different locations in the USA in areas having comparable ambient air concentrations, although actual blood lead concentrations were slightly greater than predicted (SRC, 1990).

Table 3.13 Environmental media contributions to blood lead levels for US two year old children, incorporating background levels and incremental air contributions, (ie slope of curve).

	OUTSIDE AIR LEAD CONCENTRATION ($\mu g.m^{-3}$)						
	0.0	**0.25**	**0.50**	**0.75**	**1.00**	**1.25**	**1.50**
LEAD SOURCE							
Background (non-air)							
Food, water & beverages	2.37	2.37	2.37	2.37	2.37	2.37	2.37
Dust	0.30	0.30	0.30	0.30	0.30	0.30	0.30
Subtotal	2.67	2.67	2.67	2.67	2.67	2.67	2.67
Background (air)							
Food, water & beverages	1.65	1.65	1.65	1.65	1.65	1.65	1.65
Indirect - Ingested Dust							
(with lead deposited from air)	0.00	1.57	3.09	4.70	6.27	7.84	9.40
Direct							
Inhaled air	0.00	0.50	1.00	1.50	2.00	2.50	3.00
TOTAL	**4.32**	**6.39**	**8.41**	**10.52**	**12.59**	**14.66**	**16.72**

Source: SRC, 1990; p. 35

3.4.4.2 HUMAN EXPOSURE POTENTIAL

The most significant industrial contributors of lead to air derive from sources such as smelters (primary and secondary) and battery manufacturing plants (Hutton and Symon, 1986), such emissions ultimately enhancing soil and dust lead levels proximate to the source via dust deposition (SRC, 1990). Comparing analogous soil and dust contaminant levels reveals reasonable positive correlations for lead, as also for blood lead (Fergusson, 1986), soil and dust both contributing significantly to blood lead variance (Angle and McIntire, 1975 & 1979; Chisholm, 1976; Landrigan et al., 1976a & b; Yankel et al., 1977; Chisholm and Barltrop, 1979; Schmitt et al., 1979; Charney et al., 1980 & 1983; Brunekreef, 1981 & 1983; Sayre, 1981; DOE, 1982; Gallacher et al., 1984). Both adults and children are susceptible to intoxication from such sources, children particularly so, and particularly from dust ingestion via the hand mouth route (EPA USA, 1986). The major source of drinking water lead arises as a result of the corrosion of lead-containing plumbing materials used for water-reticulation systems.

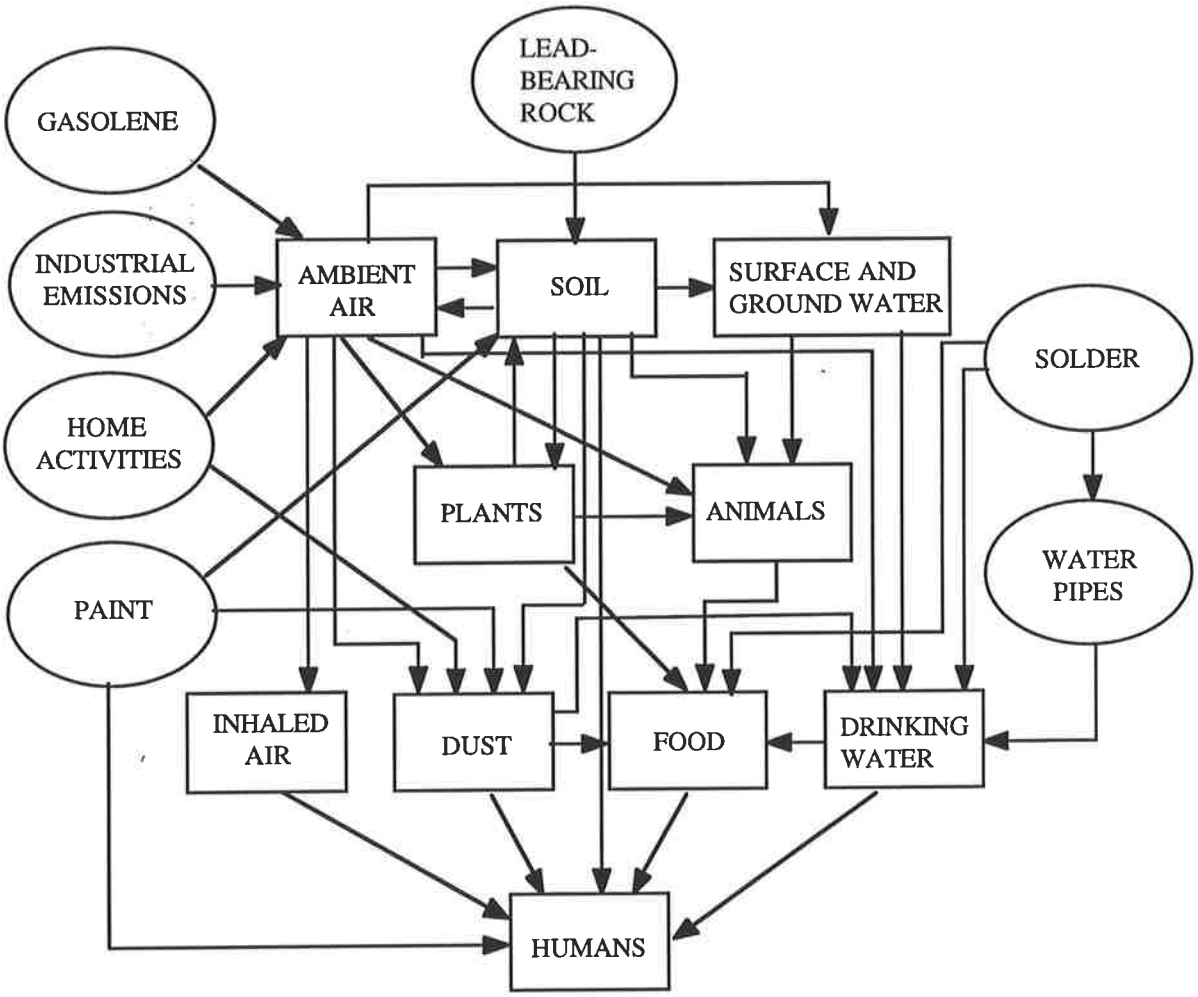
3.4.5 PATHWAYS TO INTOXICATION

Human exposure to environmental lead occurs along a plethora of pathways, with (initial) sources to the environment including combusted gasolene, leaded solder, lead-containing paint, lead-bearing rock, the transportation of lead ore or concentrate and industrial processes such as smelter operations (Duggan, 1983a, b & c; RCEP, 1983; Brunekreef, 1984; Elwood, 1984 & 1986; Body, 1986a; EPA USA, 1986; Fergusson, 1986; Davies, 1988). Environmental media to which this lead is liberated eg soil, dust, water and air act as intermediaries in a myriad pathways from original source to human receptor (Figure 3.7).

The relative significance of each main pathway has been the subject of much conjecture in the literature (Elwood, 1986), substantial exposure variability being suggested between adults and children (EPA USA, 1986). Early work by Barltrop et al. (1974 & 1975) in lead-mineralised areas of Derbyshire diminished the significance of lead-contaminated dust and soil as pathways for child lead contact. Focusing upon children in urban areas, Day et al. (1975) and Duggan and Williams (1985) concluded to the contrary, stressing the importance of the contaminated hand/finger-to-mouth mode of lead intoxication in children particularly. Subsequent work incorporating estimates of the quantity of lead on hands of children further supported the significance of the hand-to-mouth route of contaminant ingestion (Sayre et al., 1974; Vostal et al., 1974, Lepow et al., 1975), the last-mentioned group claiming that the hand-to-mouth exposure route was the principal cause of the excessive lead accumulation in the observed children.

Further work by Charney et al. (1980), Brunekreef et al., (1981) and Gallagher et al. (1984) demonstrated that child hand lead content was an important predictor of blood lead level. Roels et al. (1980) concluded from a study of children attending schools proximate to a lead smelter, that the

contribution to blood lead made by the ingestion of hand lead was far greater than that made by the inhalation of airborne lead, Davies (1988) concluding similarly. For children from neonate to toddler, given that most time is spent within the house (Chapin, 1974; Dockery and Spengler, 1981a), the dust pathway, particularly in a contaminated environment, has become accepted as the major source of intoxicant lead (Duggan, 1983a, b & c; Brunekreef, 1984; Elwood, 1986 amongst others cited in Chapter One).



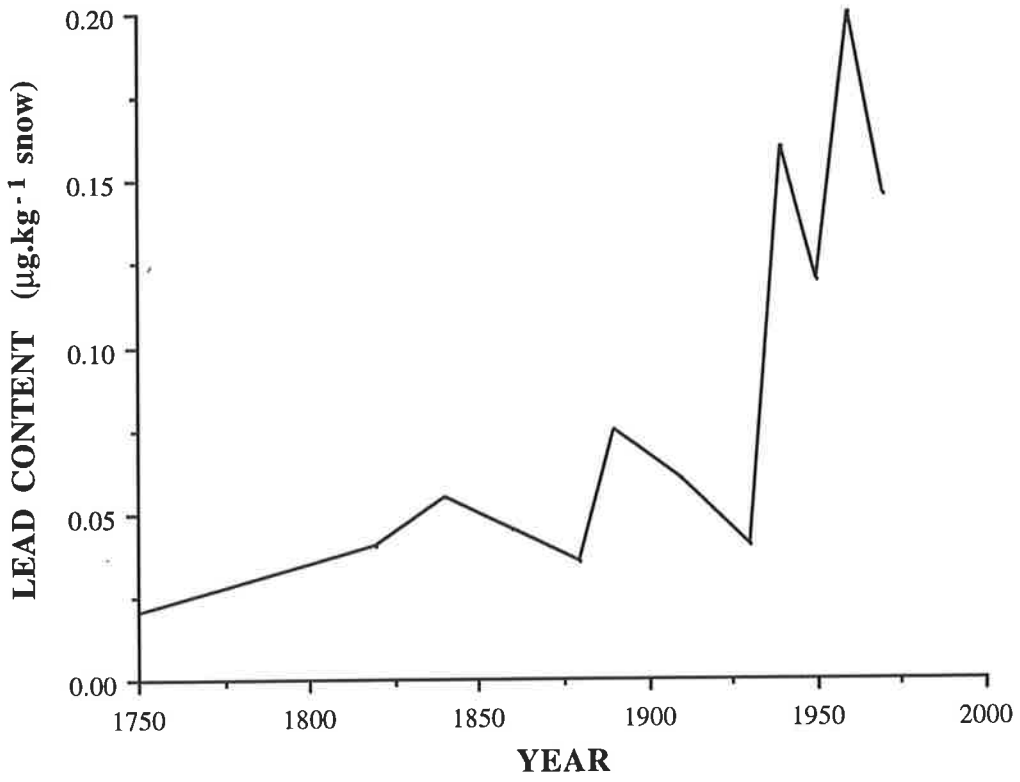
Source: Adapted from a figure in Mushak, 1992; p. 293

Figure 3.7 Major sources of lead in the environment, and transfer pathways to humans.

3.5 INDICES OF A HISTORY OF LEAD USAGE

Herron et al. (1977) note that lead and other metals are currently being deposited in the Greenland Ice Sheet at two to three times the natural rates as detected by assays of layers deriving from 10,000 years ago. Lead in snow and ice in Greenland increased threefold and linearly with time from the Industrial Revolution until approximately 1950. In the next fifteen years, levels tripled again

(Murozumi et al., 1969). These workers report lead levels in recent snow in Greenland to have exhibited lead concentrations 200 times greater (1,000 times greater according to Patterson et al. [1991]) than those of ancient ice drillings immediately below (ca 3,000BP), with lead/silicates ratios in detected dust increasing by a factor of over 100 times during this period of time. Hansen et al. (1983) report that average blood lead levels in residents of Greenland are comparable to those found in European industrialised areas. From such observations, these authors conclude that, in earlier times, most of the detectable lead in the troposphere derived from soil dust entrained by wind, although in more modern times, in Northern Hemisphere tropospheric regions, they estimate that greater than 99 per cent of the lead derives from human activity. Working independently, Ng and Patterson (1981) and Boutron and Patterson (1986) arrived at very similar estimates. Further reflecting the temporal increase of lead discharge to the environment over the metal's history of human use, fossil bi-valve shells deriving from ca 8,200BP and harvested from the eastern Canadian Arctic, have been demonstrated to display one fifth the level of contained lead that modern day shells reflect (Bourgoin and Risk, 1987).[34] Some researchers claim that much of the change in environmental levels has taken place within much more recent history (Ng and Patterson, 1981; Boutron and Patterson, 1986; Bourgoin and Risk, 1987; Shy, 1990 - Figure 3.8).



Source: Shy, 1990.

Figure 3.8 Lead concentration profile in the snow strata of Northern Greenland.

[34] The carbonate phase has been suggested to be a receptacle for certain chemical species including lead (Bertine and Goldberg, 1972).

Elevated environmental lead levels are detectable almost everywhere on the earth's surface (ATSDR, 1988a), even in very remote areas (Poole and Smythe, 1980); contamination by industrial lead has occurred extensively and globally (Davies, 1988) as with DDT (Carson, 1963) and PCBs (D. Suzuki, pers. comm.). Given lead's very long soil residence time due to its inherent chemical stability, a long history of lead production and associated environmental discharge by, particularly, developed countries, constitutes the background for one of the most widely-dispersed and persistent environmental pollutants (ATSDR, 1988a; Davies, 1988). Tyler (1978) reports that 90% of total soil lead present today will still remain there for more than 200 years. This has ultimately resulted in the potential for elevated body lead burdens (and potential adverse health effects) in humans, particularly those living in more highly-contaminated environments.

Whilst Fergusson (1986) would suggest that (population mean) blood lead levels, as an index of overall lead exposure of humans, have been decreasing for possibly the past thirty years in many countries, world lead production has not stabilised (IPCS, 1991; OECD, 1992), and copious environmental discharge still continues (CDC, 1991). The potential for exposure to lead from air, soil and dust may have fallen from the mid-1970s in many countries as a result of the declining use of leaded fuel (Sitwell and Wood, 1989), Simms, however, claiming at an international conference on lead and its environmental distribution and health impact, that "everyone living today is contaminated with lead" (1988; p. 488), a point to which no person present objected.

CHAPTER FOUR LEAD AND ITS RELATIONSHIP WITH HUMAN HEALTH - GREY METAL OR GREY MATTER, THE CHOICE MAY NOT BE OURS

The appointment of the Commission was due to the unions representing the employees at the Broken Hill Associated Smelters having brought under the notice of the Government that there was a large increase in the number of employees at the smelters reported as becoming incapacitated by lead poisoning, a position which was stated by the management of the smelters to be inexplicable.

Dr K Moore,

Chair, Royal Commission into the cause or causes of Plumbism amongst the employees of the Broken Hill Associated Smelters Pty Ltd at Port Pirie, March 1925.

The only person who is a worse liar than a faith healer is his patient.

Abraham Lincoln

4.1 MEDICAL ASPECTS OF LEAD'S TOXICOLOGY - GENERAL

4.1.1 PREAMBLE

Health: a state of complete physical, social, and psychological well-being and not merely the absence of (apparent) disease or infirmity; health therefore is an holistic concept (WHO definition; Calman, 1991).

Whilst this dissertation centres upon lead-bearing dust and its relationship to select dwellings in the township of Port Pirie, it is nevertheless appropriate to consider some aspects of the extremely complicated physiological/biochemical interaction of lead with the human body, in order to provide some perspective of the ubiquitous and pervasive nature of the material once in this realm. Given the numbers of children estimated to suffer from the effects of lead toxicity internationally, it constitutes one of the most common childhood diseases (Falk and Ing, 1989).

4.1.2 INTRODUCTION

Naturally-occurring levels of lead in various environmental compartments (soil, air, water) may influence human tissue lead levels (Nriagu, 1988). This is also the case for anthropogenically environmentally-distributed lead (EPA USA, 1986).[1] Acknowledging regional variability in (environmental) lead levels, human blood lead levels may vary accordingly, ranging from certain remote populations demonstrating little or no detectable body lead (Goldwater and Hoover, 1967; Hecker et al., 1974; Piomelli et al., 1980; Poole and Smythe, 1980; Rabinowitz and Needleman, 1982) to other areas (eg urban or industrial location) where tissue lead levels may be dangerously elevated (CDC, 1991).[2] Lead is not a human body-required nutrient (Piomelli et al., 1980), and there would appear to be no useful biological role played by this element (Archer and Barratt, 1976a; Culbard et al., 1984; SRC, 1990). Thus, under ideal circumstances, only an extremely small amount of lead, representing normal background environmental levels, is likely to be present in any body tissue. Its greater presence within the bodies of many people demonstrates environmental contamination arising essentially from anthropogenic activity (Nriagu and Pacyna, 1988).

As of May, 1993, the NHMRC has suggested a level of $10\mu g.dL^{-1}$ (down from an earlier figure of $25\mu g.dL^{-1}$) as an overall goal to set for a maximum blood lead level for the population (EPA NSW, 1993), with the issue of lead contamination affecting community health continuing to grow in profile in Australia (Calvert, 1995). Over the last six decades, based upon the results of on-going medical research internationally, opinions have progressively changed regarding the dimension of the threshold blood lead level eliciting detectable symptoms. On a number of occasions, it has been vigorously claimed by a number of researchers that the recognised blood lead level considered, at the time, to constitute the threshold level eliciting symptoms, should be reduced substantially, as, for example, the case for the USA (Figure 4.1).[3]

With the development of more sensitive techniques for the assessment of lead-induced human health decline, the literature indicates that the child lead health problem is of greater dimension than originally thought. Adverse health effects are currently being revealed in children demonstrating a blood lead level as low as $10\mu g.dL^{-1}$, with no evidence of a threshold for many health effects (Reagan and Silbergeld, 1989).[4]

Whilst the literature abounds with community, city or national blood lead assays, for the purposes of a reference here, from a large sample of children living in an urban environment (Toronto) in the

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[1] Whilst an enhanced body burden of lead may be detected in a number of body tissues, one of the most convenient and reproducible tissues to choose to measure the lead level in is blood.

[2] Blood lead levels typically reflect only fairly recent lead exposure (McEvoy, 1980; Smith, 1985; Simms, 1986), lead's blood half-life being of the order of 28 days (Rabinowitz et al., 1976; Harrison and Laxen, 1981; Pocock and Ashby, 1985). Thus while blood lead levels are quite sensitive to changes in exposure to lead (Chisholm et al., 1985b), they really only provide limited information for assessment of dose-response relationships in relation to chronic toxic effects such as lead nephropathy, and also estimates of the dimension of long-term exposure to lead (Griffin et al., 1975; Rabinowitz et al., 1976; Chamberlain, 1978; EPA USA, 1986). Skeletal lead, with a half-life of 20 - 30 years (Rabinowitz et al., 1976; Beckman, 1989), and tooth dentine lead have been suggested as more (effectively) indicative of former history of lead exposure (Moore, 1986).

[3] In 1970, the US Surgeon General defined undue exposure to lead as a blood lead level $\geq 40\mu g.dl^{-1}$, at a time when > 45% of children screened in New York city demonstrated levels greater than this (Berney, 1993).

[4] A 1993 NHMRC report suggests that as many as 44% of Australia's children may have blood lead levels in excess of $10\mu g.dL^{-1}$ (Cribb, 1993), inferring a considerable community health problem.

mid 1980s, Wang et al. (1988) noted a mean blood lead level of 10.4μg.dL^{-1}, with a slightly earlier review for the US population reflecting a mean value of around 9.5μg.dL^{-1}. This compares with a 1982 mean blood lead level for 1,241 Port Pirie children (≈ 50% of total) of 18.2μg.dL^{-1}, (Sinclair and Dohnt, 1984) with seven percent of those tested demonstrating a blood lead level of > 30μg.dL^{-1}, the then NHMRC-recommended level of concern (Calder et al., 1986; Wilson et al., 1986).

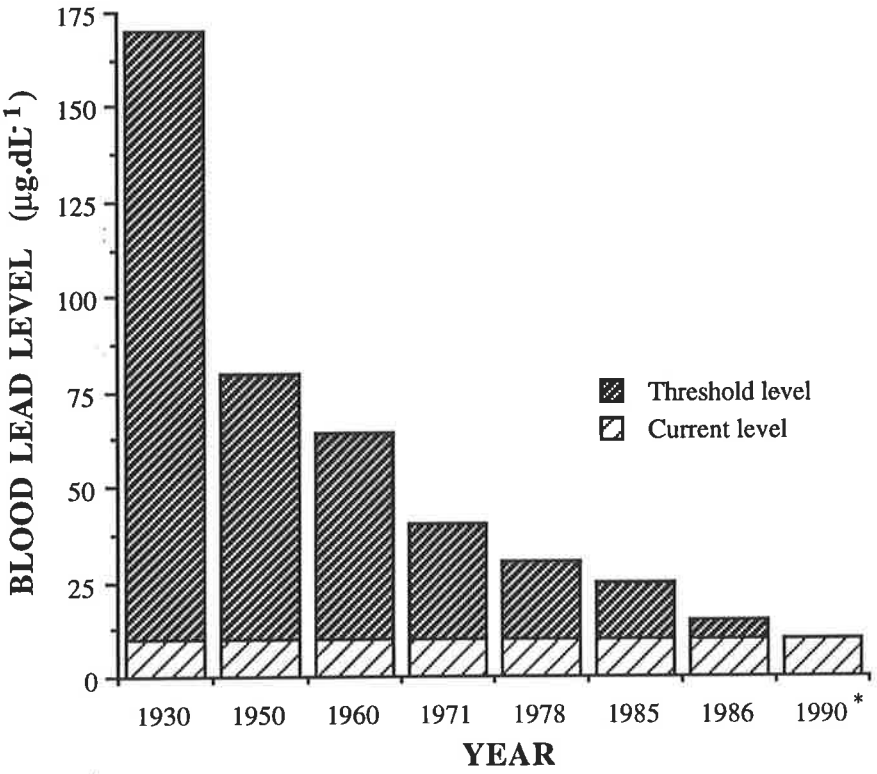


Figure 4.1 Changes in US blood lead level considered as the threshold for onset of adverse health effects.

Sources of data: US DHEW, 1970; Lin-Fu, 1972; Anon., 1982; Mahaffey et al., 1982a; Bornschein et al., 1985; Farfel, 1985; AAP, 1987; Davis and Svendsgaard, 1987; Needleman, 1988; Rabin, 1989; Silbergeld, 1990; Wixson and Davies, 1991; Mushak, 1992; Rosen, 1992; Rossol, 1992; Sibbison, 1992; Berney, 1993; Lamphear, 1994; Mauss, 1994.

In a similar vein, there has been a dramatic decrease in the number of deaths due to frank lead poisoning through this century (Charney et al., 1980; Moore, 1986; Alperstein et al., 1991; Sayre and Ernhart, 1992), paralleled by a decrease in the number of notifications of industrial lead poisoning (Waldron and Stofen, 1974), yet it would appear that until the 1970s the quantity of lead that humans were being exposed to (particularly in an urban environment) was increasing (NAS, 1980). This was consistent with the ever-increasing use of lead-alkyl automotive fuels (Moore, 1986). Subsequent to the decline in production of such fuels in the USA and Western Europe in the early-mid 1970s (ILZDG, 1991a & b), mean blood lead concentrations have been declining in the USA and particularly Germany (Jost and Sartorius, 1979; Mahaffey et al., 1979 & 1982; Sinn, 1980 & 1981;

Annest et al., 1982 & 1983; CDC, 1982; Needleman et al., 1982 & 1984; Rabinowitz and Needleman 1982).

Given this progressive blood lead concentration decline since the early 1960s and especially into the 1980s (Wixson and Davies, 1991), plus a heightened awareness of the deleterious to health nature of lead, the focus of attention, and that of greatest concern, turned to the toxic effects of lead occurring at low-level exposure. Such lead intoxication frequently related to asymptomatic and/or sub-clinical effects (Landrigan, 1990), including neurobehavioural effects such as disturbed Mental Development Index (MDI).[5]

Also noted were IQ deficits (Needleman and Gastonis, 1990; Needleman et al., 1990; Pocock et al., 1994; Tong et al., 1996), and elevated hearing thresholds (Schwartz and Otto, 1987a & b). Further lead-induced effects include growth retardation in infants exposed both prenatally (Davis and Svendsgaard, 1987) and postnatally (Schwartz et al., 1985), and often, asymptomatic effects such as hypertension (McMichael and Johnson, 1982; Lippman, 1990) in (typically) middle-aged occupationally-exposed men (Moore, 1986). Dose-response relationships for all these effects demonstrate no real evidence of a minimum level threshold down to the lowest levels of internal exposure, even at a figure of $< 10\mu g.dL^{-1}$ lead in blood (SRC, 1990). Further, principal health effects of body lead (typically at symptomatic level) are experienced by a number of body systems including cardio-vascular, urinary, haem-producing, both central and peripheral nervous (CNS and PNS), musculo-skeletal, gastro-intestinal tract (GIT) and reproductive. These will be discussed in greater detail below.

4.2 LEAD'S INTERACTION WITH THE HUMAN BODY - THE DETAILED TOXICOLOGY OF LEAD

4.2.1 THE ABSORPTION OF LEAD

Lead's entry to the body and absorption into the bloodstream may be either via absorption from the gastrointestinal tract or, subsequent to its inhalation, absorption from the respiratory tract (Knelson, 1974; RCEP, 1983; Duggan and Inskip, 1985; ATSDR, 1988b; SRC, 1990; EPA NSW, 1993). A third, less-significant alternative for the absorption of certain lead compounds is through the skin. This, however, is considered by some to be, quantitatively, only a minor pathway (RCEP, 1983), and applying virtually only to organo-lead compounds, eg as arising from leaded fuel (tetra-ethyl lead), (Laug and Kunze, 1948; RCEP, 1983; EPA USA, 1986; SRC, 1990). Contrasting this, however, Lioy (1990) reports that lead acetate is not normally readily absorbed by skin, but if it is presented contained in an oil, it becomes extremely absorbable. Further, other researchers claim that inorganic lead may also be absorbed via the integument, diffusing through and into the blood quite rapidly

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[5] MDI, eg Bayley MDI and Bayley Psychomotor Development Index (PDI), are "neurobehavioural tests assessing child learning skills" (SRC, 1990; p. 98)

(Stauber and Florence, 1988). This may occur in not insignificant quantities, particularly under elevated moisture (eg sweat) conditions (Florence et al., 1988; Lilley et al., 1988; Beckman, 1989). Such a phenomenon obviously appears most significant for individuals employed in a hot and contaminated environment such as a smelter (Lilley et al., 1988).

For occupationally-exposed individuals, or (adult) residents of a heavily-industrialised area including lead smelting plant, the inhalation of airborne lead (typically aerosol or entrained small particle size dust) would appear to constitute by far the most important pathway for an elevated blood lead level (Roels et al., 1980; Triger et al., 1989). Air lead levels represent, for children, typically a more indirect route of contamination (Farfel, 1985; Rabinowitz et al., 1985a), many researchers claiming that its direct contribution to (child) body burden is minor (eg Snee, 1981 & 1982; Laxen et al., 1985). Murphy and Yocom (1986) suggest for non occupationally-exposed adults, that lead intake via ingestion and inhalation pathways would not be greatly different, but for 30 month old children, the former pathway could be 1,000 times more significant than the latter.

For children, the principal absorption pathway is by ingestion of leaded materials such as soil and dust (David et al., 1972; CEH & CAPP, 1987; SRC, 1990), and paint where the lead content of the paint is proportional to its relative toxicity (Schwartz and Levin, 1991). Such non-food ingestion typically arises as a result of normal mouthing behaviour of contaminated surfaces (Rabinowitz, 1995).[6]

4.2.1.1 GASTRO-INTESTINAL TRACT (GIT) ABSORPTION

This pathway for absorption of lead from typically food, beverages and adventitious sources represents the major entry point for lead into the body for the general population (Hamilton and Minski, 1972; Ragan, 1983; EPA USA, 1986; SRC, 1990), and particularly for children demonstrating normal mouthing behaviour (Sayre et al., 1974), and those demonstrating pica.[7] In the rat model, there appears to be no feedback mechanism limiting the GIT absorption of lead, certainly the body burden of lead does not influence the element's gastric absorption (Conrad and Barton, 1978).

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[6] Rabinowitz et al. (1985b) note a positive association between blood lead levels and the presence (and content) of lead paint, especially with home refinishing activity in an urban centre. Davies et al. (1990) claim that 97% of a child's daily intake of lead occurs via ingestion of dust, food and water, and only three percent via an inhalation pathway.

[7] Mouthing behaviour is normal for children to perhaps three years of age, and is a major way in which children consume non-food material (Barnes, 1990). Pica, an extreme of normal mouthing behaviour, may be defined as an unusual propensity or craving to mouth/eat non food items (NAS, 1980). Of particular significance here are soil, dust, and also paint in the form of either painted objects such as toys and/or flaking paint from previously painted surfaces (Sayre et al., 1974; Vostal et al., 1974; Charney et al., 1980), some lead salts demonstrating a "sweet" taste (Gibson et al., 1892). Normal mouthing behaviour commences in the first year of life, peaking early in the second year, and subsequently subsiding (Lin-Fu, 1973b). Persistence beyond 18 - 24 months is considered abnormal mouthing behaviour (Lourie et al., 1963). Soil pica is not uncommonly observed in children of two to three years of age, estimates of daily soil ingestion being of the order of 10 - 1,000mg (Lepow et al., 1974; Day et al., 1975; Solomon and Hartford, 1976; NRC, 1980; Kimborough et al., 1984; Binder et al., 1986), La Goy (1987) proffering a maximum figure of 5,000mg. The dust to hand to mouth contaminant pathway was first suggested by Gibson (1904), although its significance with respect to dust was overshadowed by the leaded paint problem for many years. Sayre et al. (1974) linked overlapping peaks in mouthing behaviour and blood lead levels in young children and also house dust lead and hand-adherent lead levels, claiming such evidence strongly supported the hand-to-mouth pathway as a prime source, as others subsequently eg Vostal et al. (1974), Landrigan et al., (1975), Lepow et al. (1975) and McMichael et al. (1985). In a study of four separate communities, Gallacher et al. (1984) noted that hand lead levels were an important contributor to blood lead levels, and a substantially better indicator of blood lead levels than was the level of pica (as determined by questionnaire).

Suggested percentage absorption for adults is of the order of five percent (Sherlock, 1987) through ten percent (Kehoe, 1961a & c; DHSS, 1980; Duggan, 1983b; RCEP, 1983) to as high as 20% (EPA USA, 1986; SRC, 1990).[8] For children, this figure appears to be closer to 50% (Alexander et al., 1973; Alexander, 1974b; Chamberlain, 1978; Ziegler, 1978; Mahaffey et al., 1979; DHSS, 1980; Budiansky, 1981; Duggan, 1983c; RCEP, 1983; EPA USA, 1986; Moore, 1986; SRC, 1990), Barltrop and Strehlow (1978) proffering a figure of approximately 40% for long-term hospitalised children.

In both adults and children, a damaged epithelial mucosal layer arising as a result of (say) gastrointestinal ulceration may elicit a massive increase in absorption (Bercovitz and Laufer, 1991). Non gut-absorbed dietary lead is eliminated in the faeces (Vahter et al., 1991), Ziegler et al. (1978) commenting that young children excrete lead more rapidly than adults.

a) Normal physiological absorption variability

Large individual to individual variation in absorption of lead may occur in the gut, and relates to individual physiological difference and also contemporaneous presence of a variety of other chemical species interacting upon absorption mechanisms (Waldron and Stofen, 1974). As an example, the presence of phosphate ions in the gut enhances the insolubility of lead (Luke, 1991). An interesting defence mechanism in humans is postulated by Moore (1988) based upon data of Sherlock et al. (1982) and Lacey et al. (1985), namely that as blood lead concentration increases, absorption rates of lead decrease and excretion rates increase.

b) Chemical species absorption variability

The solubility of lead is a major determinant of gut absorption, and hence bioavailability and toxicity (Clevenger et al., 1991b). This will depend to some extent upon the solubility of the particular chemical species (Waldron and Stofen, 1974; Duggan and Inskip, 1985; Moore, 1986; SRC, 1990; Steele et al., 1990) (Table 3.1) and also other physico-chemical factors such as particle size and form (Hamilton and Revitt, 1987; ATSDR, 1988b). Larger particle-sized metal is less easily absorbed than smaller-sized (< 6μ) particles (Barltrop and Meek, 1979; Healy et al., 1982; Moore, 1986). Curiously, Davies (1988), citing several researchers (including Harrison et al., 1969; Karhausen, 1973; Barltrop and Meek, 1975; Rabinowitz et al., 1980; Heard and Chamberlain, 1983; Mahaffey, 1983b), claims that the particular chemical form in which lead is administered to the gut has only minimal influence on its uptake. This would appear to differ from more recent work suggesting to the contrary, the relative solubility (and hence bio-availability) of the chemical species being an important determinant of the lead salt's relative toxicity (Foster and Lott, 1980; Harrison et al., 1981a; Sax, 1984; Duggan and Inskip, 1985; EPA USA, 1986; SRC, 1990; Clevenger et al., 1991b; IPCC, 1991). Early work by

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[8] Kehoe's work appeared to demonstrate some inadequacies when reviewed by Chamberlain et al., (1978) who revised the estimate to approximately 15%. For 40 years in the USA, Robert Kehoe was a key player in the program and politics of lead and health research and also the development and use of leaded fuels. A summary of his involvement in the overall lead "picture" and its possible intrigue is included in Appendix Four.

Sax (1963) and Waldron and Stofen (1974) also closely linked lead salt species' water solubility with their relative toxicity.

Resulting from the work of several researchers, it has been demonstrated that individuals resident in a community experiencing elevated lead levels in various environmental compartments may not necessarily reflect expected (elevated) body lead levels. The inference here is that the relative bioavailability of the particular lead species is not sufficiently high to elicit elevated absorption. This may include some types of mining waste (Cotter-Howells, 1991; Cotter-Howells and Thornton, 1991; Freeman et al., 1992; Ruby et al., 1992 & 1993; Thornton, 1993), typically lead silicate from smelter slag which demonstrates poor solubility under most laboratory conditions (Body, 1986b; Clevenger et al., 1991b; Hemphill et al., 1991; Davis et al., 1992). Davis et al. (1992) and Ruby et al. (1992 & 1993) report that mining waste demonstrates lower bioavailability than either urban- or smelter-derived waste. Steele et al. (1990) claim that a larger mean particle size for mine waste than for smelter waste infers poorer aeolian transport, with lower house penetration and hand retention, and, ultimately, poorer gastric absorption. Thus, soil (lead) species' concentration will not necessarily represent a determinant of absorption of the compound of lead given various salts' varying bioavailability.

Whilst particle size and chemical form of lead are influential factors in determining GIT uptake (Duggan and Inskip, 1985; Steele et al., 1990; Davis et al., 1992), pH at GIT site and gut transit time are also relevant. For example, a pH of around seven, as experienced in post-duodenal small intestine, will severely limit dissolution compared with stomach pH of around one or two (Ruby et al., 1992). In the domain of specific bio-availability of various chemical species of lead, clearly, more work needs to be conducted.[9]

c) The influence of food on absorption

Substantial enhancement of the rate and extent of lead's absorption may relate to the absence of food in the gut (Rabinowitz et al., 1980), as well as a dietary insufficiency of calcium, iron, other trace elements and Vitamin D (Heard et al., 1983; SRC, 1990). From Chamberlain et al.'s (1978) experiments, a number of individuals in a test group revealed 25% absorption of nominally (water) insoluble lead sulphide taken in the absence of food. Further, the influence of age upon the uptake of lead from the gut appears to vary with the quantity and nature of food in the stomach (Chamberlain et al., 1978; Johnson and Tenuta, 1979; Heard and Chamberlain, 1982). Under fasting conditions, the absorption of lead subsequent to ingestion appears to be substantially greater than that for lead ingested with food (Garber and Wei, 1974; Aungst and Fung, 1981). Blake (1976) proffers data suggesting mean uptakes of forty five and six percent respectively, with Rabinowitz et al., (1980) arriving at a figure of 35% absorption subsequent to 16 hours fasting. Working similarly to Blake (1976), Heard and Chamberlain (1983) claim a figure of approximately 63% absorption after fasting.

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[9] Regarding the collection of environmental samples with reference to the degree of lead contamination, and the potential for childhood intoxication, given individual variability in absorption with respect to chemical species and particle size etc, and children's propensity to demonstrate a larger percentage uptake from the gut than adults, it would appear prudent, in a precautionary context, as Duggan and Inskip, to measure "total lead" in the available environment rather than attempt to measure biologically available lead (1985; p. 17). This is the foundation of adopted analytical methods included herein (Chapters Five - Nine).

Davies (1988) cites two studies reporting that beverage lead is more readily absorbed than lead in food. This has been confirmed by other workers (Garber and Wei, 1974; Barltrop, 1975b; MAFF, 1989), and also Rabinowitz et al. (1980) in which case, beverages were consumed between meals when gut-contained food levels were low. If such adult subject studies are representative of children, such results would suggest that lead consumed between meals (eg drinking water, paint chips, dust/soil) contributes more to lead intoxication than lead ingested with solid food.

d) Age-related absorption variability

Age is an important determinant of the rate of absorption of lead from the gut (Moore, 1986), with young human and animal study infants all demonstrating enhanced absorption and retention of lead compared with analogous adult studies (Alexander, 1974; Alexander et al., 1974; Kostial et al., 1974; Quaterman and Morrison, 1978; Ziegler et al., 1978; Mykkanen and Wasserman, 1981; Duggan and Inskip, 1985).

e) Occupational influences on absorption

Lead poisoning tends to be more prevalent in mines where the fine-grained and more soluble lead oxide and lead carbonate ores are exploited (Hunter, 1969; Nriagu, 1983b). Hamilton (1925a) concludes that the low incidence of plumbism in miners of galena ore (the principal lead ore of the ancient world) was related to the poorer solubility of lead sulphide in lung and gastric regions. McCord (1953a & 1954) in his history of lead mining in the USA notes that there were few cases of lead poisoning from the early workings when galena was mined, however, when lead carbonate was mined in Utah after 1870, it became an all too common hazard. Further, Hamilton notes that some Americans and Australians still mine oxidised ores of lead (chiefly carbonate and sulphate), "compounds which are much more soluble in gastric juice than the sulphide" (1924a; p. 584).

f) Dietary factors influencing absorption

The dietary minerals most likely to be present in the average diet and demonstrating a detectable influence on the absorption of dietary and adventitious gut-contained lead are reviewed here. Much of the research referred to below has been conducted in the animal model (typically the rat), although that work which has been conducted with humans would appear commensurate with animal model-derived data (Luke, 1991). The inference of much of the reviewed literature in this regard (discussed below) is that a deficiency in the essential elements may result in enhanced absorption of lead and its greater toxicity. Relevant dietary elements include the following:

- i) Calcium and Phosphorous

The retention of lead passaging the gastrointestinal system varies inversely with dietary calcium content (Ziegler et al., 1978; Brunekreef et al., 1983; Moore, 1986). Decreased dietary calcium in the presence of, particularly, enhanced acidity reflects a substantially increased absorption of lead (Coleman et al., 1978; Blair et al., 1979). Sorrell et al. (1977) report that blood lead levels and dietary calcium intake for one to four year old children are inversely correlated, such work being confirmed by others (Johnson and Tenuta, 1979; Mahaffey, 1980). Mahaffey et al. (1976) demonstrate, with one to four year old children with blood lead levels exceeding $40\mu g.dL^{-1}$, significantly lower intakes of calcium and phosphorous but comparable iron intake to a low blood lead control group. Further, Johnson and Tenuta (1979) claim for children aged one to six years that blood lead levels correlated negatively with dietary calcium intake, although the elevated blood lead group also demonstrated less dietary zinc intake than the control group - ii) below.

Such results support research work dating from early this century noting that lead toxicity in animals was more substantial for animals maintained on a low dietary calcium intake as a consequence of increased retention and decreased (renal) excretion of lead (Goyer, 1986; Luke, 1991). Such observations were responsible, historically, for attempts to reduce lead's toxicity and prevent lead poisoning in occupationally-exposed individuals by the provision of a high source of calcium, notably milk (Mahaffey and Goyer, 1973; Mahaffey, 1980).[10] Fischbein (1983) confirmed this function of milk, suggesting that its high concentrations of zinc, phosphorous and even protein were also making a contribution. Such results are not without critique, Bushnell and DeLuca, (1981) suggesting that the presence of lactose in the gut actually enhances the absorption of lead in juvenile rats. In this context, the net effect of milk might either represent an enhancement or a diminution of lead absorption, depending upon the nutritional status of the individual (Stephens and Waldron, 1975; Bell and Spickett, 1981).

Work by Graziano et al. (1990) in Serbian, Albanian and Gypsy communities in a lead-smelting city Titova Mitrovica and a control, non lead-exposed town Pristina in (the former) republic of Yugoslavia, failed to document an association between milk consumption and blood lead level. In an animal model, Fullmer (1991) demonstrated essentially a bifid and opposite interaction of calcium and lead absorption, revealing intestinal lead absorption was initially enhanced by a calcium deficiency. This response was subsequently inhibited by a prolonged dietary lead intake. Further work, particularly in the human model is warranted here, given milk's (presumed) protective role in situations where elevated lead exposure is predicted, especially for children.

Genetic damage arising as a result of a lead enhanced diet in test animals appears to be greater for animals fed a low calcium diet (Deknudt and Gerber, 1979). Dedknudt et al. (1977) claim that the frequency of chromosomal abnormality in monkeys administered lead in conjunction with a low calcium diet is significantly increased compared with a lead diet/normal calcium control group. Poirier

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[10] As a high lead-based paint sprayer in the 1970s, resulting from a Miscellaneous Workers' Union regulation, my employer was required to provide me with 1.5 litres of milk daily. Observations made more than a century ago suggested that milk consumption prevented the occurrence of plumbism in workers in the white-lead industry (Tanquerel des Planches, 1839).

et al. (1984) reveal that calcium and magnesium prevent an increase in pulmonary adenoma formation in mice administered lead subacetate.

The data suggest that two mechanisms are acting in the transport of lead from the lumen of the gastrointestinal tract across the gut wall, only one of which is influenced by the presence of calcium (Meredith et al., 1977). Additionally, Snowdon (1977) claim a (dietary) mineral insufficiency as a basis for the development of lead pica during infancy. Systemic calcium concentrations do not appear to influence retention of lead however (Quarterman and Morrison, 1975; Meredith et al., 1977), Barton et al. (1978a) noting as a cause for this observation, that both lead and calcium share binding sites and absorptive proteins, gut-contained iron acting very similarly (Barton et al., 1978b). Supporting this, Grunden et al. (1974) have shown, *in vitro,* that lead inhibits the transport of calcium across rat duodenal walls. Peng et al. (1979) observe that lead further interferes with calcium metabolism through increases in serum calcium without the expected associated equivalent rises in calcitonin level. Moore (1986) notes that the mechanism for this effect attributed to dietary calcium is not well understood.

Heard and Chamberlain (1982) observe for adult subjects that both calcium and phosphorous dietary intake reduces lead uptake by the gut, coincident with others' observations that the presence of calcium and phosphorous in the diet precipitates insoluble calcium phosphate, this species co-precipitating much of the lead present EPA USA (1986). This conclusion sits well with Mahaffey et al.'s observation (1976) that children aged one to four years displaying blood lead levels greater than $40\mu g.dL^{-1}$ demonstrated a significantly lower dietary intake of calcium and phosphorous ($P \leq 0.01$) than that of a control group displaying similar iron intake - ii) below. Reflecting relatively high dietary calcium levels, the regular consumption of substantial quantities of milk has been demonstrated to be inversely related to blood lead concentrations (Johnson and Tenuta, 1979; Blake et al., 1983; Brunekreef et al., 1983).

Tissue phosphate interacts with lead, limiting the availability of lead to the cellular transport pool elsewhere in the body (Moore, 1986). Dietary phosphate deficiency may significantly increase body lead burden by decreasing intraluminal lead precipitation thus enhancing its absorption and also its retention (Lindh et al., 1980; Moore, 1986). Subsequently, however, an enhanced dietary phosphate intake will rapidly lead to a reduction in primary lead absorption from the gut (Barton and Conrad, 1981). Plant phosphate is stored mainly in phytate (inositol-hexaphosphate) form (Moore, 1986). Calcium phytate reflects a great affinity for lead, such that plant-sourced dietary calcium phosphate will reduce absorption of dietary lead thus providing a natural protective factor against lead toxicity (Wise, 1981). Heard and Chamberlain (1982) report that a combination of reasonable levels of calcium and phosphorous in the diet may reduce absorption of lead to less than ten percent of what it might otherwise be in their absence.

- ii) Other trace elements

Lead absorption from the gastro-intestinal tract may be reduced by the presence of some other dietary minerals such as iron (Six and Goyer, 1972; Watson et al., 1980), zinc (Cerklewski and Forbes, 1976; Conrad and Barton, 1978; El-Gazzar et al., 1978), copper (Klauder and Petering, 1975),

cadmium (Salangina et al., 1982), magnesium (Fine et al., 1976; Cerklewski, 1983), selenium and chromium (Chowdhury and Chandra (1987).

Dietarily iron-insufficient adults may demonstrate two to three times greater lead absorption than subjects who have sufficient iron in their diets (Watson et al., 1980). Correspondingly, children with elevated blood lead concentrations (> $30\mu g.dL^{-1}$) frequently demonstrate an increasing prevalence of iron deficiency with increasing blood lead concentrations (Yip et al., 1981). In a group of 66 children, Chisolm (1981) reports an inverse relationship between chelatable iron and chelatable body lead levels utilising urinary levels of delta-amino laevulinic acid (ALA) as an index.[11] Occupationally lead-exposed (adult) workers treated with a daily intake of dietary iron and thiamine demonstrated negative correlation between dosage of dietary iron and blood lead levels (Ito et al., 1987). Waxman and Rabinowitz (1966) and Dhir et al. (1985) note that ferrous iron (Fe^{++}) offers some protection against the inhibition of haemoglobin synthesis and cell metabolism by lead. This observation appears well aligned with the observations of Bota et al. (1982) who report that iron appears to reduce the toxic effect of lead on blood enzyme and liver catalase activity.

Markowitz and Rosen (1981) studying a group of children with plumbism, observed that this group's mean zinc levels were significantly lower ($P \leq 0.05$) than the mean values for a normal control group. Additionally, the mean zinc levels in children with elevated blood lead levels was reduced even further by chelation treatment. Similarly, Chisholm's (1981) research with a group of children all of whom displayed blood lead concentrations in the range 45 - $60\mu g.dL^{-1}$, elicited a reported inverse relationship between body lead levels and the quantity of systemically active zinc. Further, all subjects were subsequently challenged with EDTA chelation therapy, Chisholm noting an inverse relationship between urine levels of ALA and systemically active (or chelatable) zinc.[12] For an occupationally-exposed group, lower blood lead levels were reported for workers exposed to a combination of lead and zinc compared with a control group with little or no zinc exposure (Dutkiewicz, 1979). Subsequent to EDTA chelation with zinc sulphate (body zinc replacement) in a group of occupationally-exposed lead-intoxicated patients, Boscolo et al. (1983a & b) reported that such patients displayed lowered blood pressure, elevated erythrocyte amino laevulinic acid dehydratase (ALA-D) activity, decreased urinary excretion of ALA, ameliorated haem synthesis and reduced zinc depletion, results which suggest that zinc plays a protective role in lead toxicity.[13]

Zinc added to the diet of horses grazing on lead-contaminated pastures, protects the animals from clinical signs of lead toxicity (Chisholm, 1981; Brewer et al., 1985; Goyer, 1986). The (relatively high) frequency of pharyngeal and laryngeal paralysis normally evidenced by lead-contaminated foals (Schmitt et al., 1971; Willoughby et al., 1972; Buchauer, 1973) is substantially reduced by

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[11] ALA is an indicator of lead's toxicity, the urine concentration of which is directly related to blood lead levels and its human toxicity within the haem system.

[12] Chelation treatment is often the first choice for medical intervention in episodal lead intoxication. It may be life-saving for childhood lead poisoning, decreasing the body burden of lead, particularly the most toxic readily-mobile fraction, far more rapidly than normal excretory processes (Piomelli et al., 1984). Such therapy in the treatment of lead poisoning, typically using calcium sodium EDTA (ethylene diaminotetra-acetic acid), is a pharmacological process arising as a result of pharmacokinetic intervention in order to strip excess available lead from body tissues, and excrete lead as lead's chelated complex, a reasonably soluble urinary compound. In the process, other di-valent cations such as zinc (Zn^{++}) or calcium (Ca^{++}) may also be removed.

[13] Haem is an iron-rich porphyrin which is a precursor of haemoglobin. ALA-D is a normal cytosolic enzyme that catalyses the condensation of two units of ALA to form porphobilinogen, a normal part of the haem cycle, and an enzyme that is directly inhibited by the presence of lead (SRC, 1990).

dietary zinc augmentation, the zinc acting to almost eliminate the inhibition of ALA-D by lead (Haeger-Aronsen et al., 1976). Such a protective effect of zinc against lead toxicity has also been demonstrated in the chick embryo (Srivastava and Tandon, 1984).[14]

Another element influencing the relative toxicity of lead is cadmium. Enhanced effects such as increased body weight loss and increased brain, liver and adrenal weights have been suggested by Anca et al. (1982) for rats given lead and cadmium compared with those given either lead or cadmium only. Salangina et al. (1982) report enhanced toxic effects on kidney, liver and haemopoietic systems as a result of a combination of both elements as compared to either alone, inferring a synergistic effect of these two metals (Fahim and Khare, 1980). The presence of cadmium appears to inhibit the retention of lead and/or reduce its absorption and increase its excretion, although lead appears not to affect the accumulation of cadmium (Suzuki, 1981), Thawley et al. (1977) suggesting that in combination, the toxic effect upon the haemopoietic system was less than for lead alone.

- iii) Vitamin D

Some reference to the role of Vitamin D in the absorption of lead from the gastro-intestinal tract is made in the literature, although it would appear to be somewhat contradictory. This confusion reflects the difficulties experienced in selecting a single dietary component and examining it in isolation (as Moore, 1986).

Vitamin D, associated with the uptake of calcium (Moore, 1986), also influences lead absorption, Smith et al. (1978) reporting enhanced lead absorption with elevated concentrations of Vitamin D. Contrasting this, however, elevated blood lead levels in lead-poisoned children have been associated with a Vitamin D deficiency (Sorell et al., 1977; Rosen et al., 1980; Box et al., 1981; DOE, 1982). Rosen et al. (1980) and SRC (1990) conclude that lead impaired the hepatic (and subsequently renal [Mahaffey et al., 1982b; Rosen and Chesney, 1983]) conversion of Vitamin D to its active hormonal form 1,25-dihydroxy-vitamin D, or alternatively, enhanced its conversion into an inactive product.

More recent work by Sherlock et al. (1985), however, suggests that no significant population correlation between blood lead and serum Vitamin D levels exists, such a relationship being detected in only some individuals with substantially elevated blood lead concentrations. From Hart and Smith's (1981) animal model studies, dietary cholecalciferol augmentation resulted in enhanced intestinal absorption of lead, and also changes in the distribution of lead within tissue (Barton et al., 1980). Further, from this work, it was demonstrated that a deficiency of Vitamin D resulted in reduced lead excretion, a subsequent single dose of the vitamin accelerating body clearance of lead even where lead was deposited largely in bone (Moore, 1986).

Lead-exposed children exhibiting blood lead concentrations of from 33 to 120$\mu g.dL^{-1}$ have been observed to demonstrate marked reductions in serum levels of 1,25-dihydroxy-vitamin D, with the most striking decrease being observed in children with blood lead levels exceeding 62$\mu g.dL^{-1}$ (Rosen

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[14] This function of zinc would seem commensurate with a personal observation of a colony of sugar gliders (*Petaurus breviceps*) that had become contaminated with lead from painted aviary walls, demonstrating detectable blood lead levels. For some time after the leaded paint was removed, these animals were observed to vigorously lick the zinc wire coating of the aviary .

et al., 1980 & 1981). These authors also report for child blood lead concentrations of 33 to 55μg.dL^{-1} that significant depression of circulating 1,25-dihydroxy-vitamin D had been detected.[15] The results of this study proffered a strong inverse correlation between 1,25-dihydroxy-vitamin D levels and blood lead over the entire range of blood lead levels measured in the study (12 to 120μg.dL^{-1}), with little change in correlation slope at blood lead concentrations exceeding 30μg.dL^{-1}.

Apart from being involved in normal bone remodelling and intestinal absorption of lead, at the cellular level, Vitamin D is also intimately involved with normal intracellular calcium homeostasis and associated calcium-mediated processes that are essential for virtually all mammalian cellular integrity and function, and fundamental body processes (EPA USA, 1986; SRC, 1990; IPCS, 1991). Further, Vitamin D is involved in normal cellular differentiation (EPA USA, 1986) and is essential for normal immunoregulatory function (IPCS, 1991), thus lead-elicited impaired production of Vitamin D may demonstrate profound and pervasive effects on tissues and cells of widely varying body type and function (EPA USA, 1986; SRC, 1990).

The increase in absorption of lead under conditions of low levels of Vitamin D arises as a result of a complex array of biochemical factors, although a major contributor would be expected to be the overall poor state of nutritional health, the low level of Vitamin D representing, more appropriately, an indicator of nutritional status.

- iv) Other dietary factors

Many other substances may influence the absorption of lead, with compounds affecting the intestinal barrier (such as ethanol) eliciting increased lead absorption (Mahaffey et al., 1974; Flora and Tandon, 1987). Further, dietary substances which enhance the solubility of gut-contained lead will increase its absorption, eg acid solubilisation such as ascorbic or citric acid, and in combination with amino acids (Conrad and Barton, 1978). The influence of amino acids on solubility varies with subject age and amino acid type (Quarterman et al., 1980). Lead may also increase the toxic effects of other materials, mercury being an example. In the rat model, dietary administration of lead nitrate results in increased mercury deposition in the kidney associated with an enhanced lethality (Congiu et al., 1979).

- v) Summary comments

In summary, lead's absorption from the gut, and also its relative toxicity is influenced by nutritional status, specifically the presence of a number of essential and trace elements both in the diet and in body tissues. A deficiency of such essential and trace elements is most frequently associated with enhanced (dietary) lead absorption and subsequent toxicity of lead (Mahaffey, 1980; Fischbein, 1983) a surplus providing some protective function against lead (Chowdhury and Chandra, 1987).

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[15] For lead-exposed children displaying blood lead levels in the range 33 to 55μg.dL^{-1}, 1,25-dihydroxy-vitamin D levels appeared to be reduced to levels comparable to those observed in children demonstrating severe renal insufficiency (SRC, 1990), consistent with an effect of lead on renal 1,25-dihydroxy-vitamin D production, supporting the renal production component suggested by Mahaffey et al. (1982) and Rosen and Chesney (1983).

Acknowledging normal childhood mouthing behaviour and a more rapid rate of gastro-intestinal passage, a greater proportion of the intake and absorption of lead by children will occur as a result of an absence of food in the gut than would be expected to occur for an adult (Mahaffey, 1985). Further, Mahaffey also claims that the protective effect of any dietary essential/trace element that is dietarily insufficient would thus necessarily be less pronounced than that for adults.

4.2.1.2 RESPIRATORY TRACT ABSORPTION

This pathway, for the general population, is considered to be the less significant contributor to body lead than that absorbed via the gastro-intestinal system (EPA USA, 1986; Moore, 1986; ATSDR, 1988a; SRC, 1990; CRC, 1991; IPCS, 1991; EPA NSW, 1993). The proportional uptake from the total respiratory tract (ie around 40%, [EPA USA, 1986]) is suggested to be more substantial, for adults, than that for the gastro-intestinal tract (Moore, 1986). SRC notes (1990) that virtually all of the material deposited in the lower respiratory tract (perhaps 30 - 50% of lead in ambient air depending upon particle size and ventilation rates) is absorbed (as also Kehoe, 1961a, b & c; ICRP, 1966; Nozaki, 1966; Rabinowitz, 1977; Chamberlain et al., 1978; Morrow et al., 1980; Gross, 1981).[16] Confirming this, from post mortem-derived tissues, it appears that lead does not accumulate in the lungs of either occupationally-exposed subjects (Barry, 1975) or non-occupationally exposed individuals (Gross et al., 1975). Much of these data pertaining to percentage uptake of lead-bearing particles in ambient air, however, derive from measurements made with adults, little data being available for children.

Different airway architecture in children, particularly smaller passage diameters, elicits enhanced particle deposition rates of the order of 1.6 - 2.7 times those of adults (James, 1978). Additionally, enhanced respiration rates have been noted for children (Davies, 1988). Children typically demonstrate ≈ 16% of the mean body weight of adults, yet approximately 40% the ventilation (Barltrop, 1972). DHSS (1980) claim a figure of 70% retention of respired ambient air lead (and 100% absorption of lung-deposited lead) for children of approximately two years of age, this figure being subsequently endorsed by others (eg RCEP, 1983).

Certainly some inhaled lead is deposited on the moistened surfaces of air passages before the lungs (eg the trachea), where muco-ciliary action returns this material to the oro-pharynx and then oesophagus. Respiratory tract absorption of lead is very rapid, lead passaging directly into the general circulation rather than passing through hepatic circulation where some clearance may occur as for gut-absorbed lead (IPCS, 1991).

Typically, occupationally-exposed populations receive a greater proportion of their exposure through inhalation, the converse being the case for the general population, for whom the effects of

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[16] Lung deposition rates of lead-bearing particles introduced as a result of normal respiration, varies with particle size (Waldron, 1975). Increasing particle size reflects a decreasing alveolar deposition rate to a minimum over the range where Brownian diffusion predominates (sub micron particle size range). At a particle size range larger than this level (≈ > 0.5MMAD - mass median aerodynamic diameter), an increase in particle deposition likelihood with increasing particle size occurs due to direct impaction and sedimentation of particles (Davies, 1988). Particles of a size around five microns or less will usually penetrate the respiratory passages (USSC, 1973), particle sizes of two to four microns will typically reach the deeper lung structures where no protective mucosa is apparent (Hwang, 1972; Lee, 1972; Natusch et al., 1974).

exposure are independent of the route of entry (SRC, 1990; IPCS, 1991).[17] Between-individual biological difference with respect to absorption will, as for gastric absorption, lead to different blood lead levels for different individuals given similar air lead exposure (Snee, 1982a).

Angle et al.'s comprehensive environmental study of school-age children revealed significant correlation between blood lead level and "fallout" lead (1974; p. 27), but virtually none with air lead levels, demonstrating the importance of the gastro-intestinal absorption pathway for children, and also the hand-to-mouth pathway.

For children, particularly those from neonate to around three to four years of age, the propensity toward elevated body lead levels under certain environmental conditions would thus appear to present a substantial environmental health problem. This is particularly so given that they constitute the population group most sensitive to irreversible damage to their developing central nervous system, this process continuing up to around three years of age (EPA USA, 1986; Moore, 1986; CEH & CAPP, 1987; ATSDR, 1988a, Needleman et al., 1990; SRC, 1990; CDC, 1991; Dietrich et al., 1993; Lanphear, 1994).

4.2.2 LEAD'S DISTRIBUTION IN THE BODY

4.2.2.1 BINDING SITES, PATHWAYS AND BODY POOLS

Subsequent to absorption of lead into the blood stream, the bulk of the blood lead is transported in the erythrocyte (red blood cell) membranes (Manton and Cook, 1979; Everson and Patterson, 1980; De Silva, 1981). The lead is ligand-bound to, principally, carboxyl groups, as well as dibasic phosphate or glycero phosphate in a non diffusible form (Moore, 1986 & 1988; Luke, 1991). This lead is primarily associated with the haemoglobin component of the erythrocyte (Moore, 1988), and remains fairly strongly bound within the cell (Raghaven et al., 1981; Moore, 1986), with lead demonstrating a particular affinity for foetal haemoglobin (Ong and Lee, 1980a).[18] Once attached to the erythrocyte, lead cannot be dialysed from the cell, suggesting an insoluble lead-haemoglobin complex (Ong and Lee, 1980a). This "bound" component of lead in blood, as also the strength of that bond, remain an important determinant of lead's potential biological effect, since only freely-available blood lead, capable of being transported to other organs or systems, may demonstrate toxic effects elsewhere in the body.

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[17] Jaworowski et al. (1985) proffer an interesting observation purportedly confirming the findings of Chamberlain (1983b) regarding the prominence of the ingestion over the inhalation route for lead intoxication of humans. These authors note that cows and humans living in a rural population exposed to leaded fuel breathe the same air yet (obviously) have a different diet. Thus the lack of detectable skeletal lead in French cows from the same setting as humans with readily detectable skeletal lead levels suggests that inhalation is of minor importance in the contamination of contemporary humans with lead.

[18] Approximately 80% of lead is bound in this fashion (Moore, 1986). The remainder is bound to a variety of blood proteins and, with very limited quantities, associated with blood lipids. The uptake of lead by haemoglobin can be inhibited by the presence of iron *in vitro*, suggesting common binding sites. Calcium has been demonstrated to bind to the same fraction of erythrocyte membrane proteins as lead, inferring competition between the cation species for binding sites (Ong and Lee, 1980a).

The remainder of the lead in blood which is responsible for distributing lead to various body organs (Baloh, 1974), resides in a relatively labile form in plasma, bound to microligans, in albumin or (in trace amounts) in high molecular weight globulins (Ong and Lee, 1980b). Given its easily diffusible character, and short biological half-life of perhaps less than five days (Moore, 1986), this lead may readily cross cell membranes (Finkel, 1983), passaging in and out of capillaries and entering parenchymal cells of many tissues including central nervous system, kidney and liver. Such bound lead passes virtually unimpeded across the placenta (Barltrop, 1969; Carpenter, 1974; Fahim, 1976; Kaul et al., 1983; Borella et al., 1986; Mayer-Popken et al., 1986; Landrigan, 1990), and is readily distributed to the foetus (SRC, 1990). This process commences well within the first trimester, tissue lead levels increasing progressively throughout pregnancy (Barltrop, 1969).[19] This plasma lead exists in equilibrium with erythrocyte lead, Cavalleri et al. (1978), Everson and Patterson (1980) and De Silva (1981) noting that plasma lead levels rise in accordance with erythrocyte lead levels.

Lead's mean half-life in blood is of the order of 25 - 28 days for both inhalation and ingestion pathways (Griffin et al., 1975; Rabinowitz et al., 1976; Chamberlain, 1978; EPA USA, 1986). Blood lead concentration adjustments responding to abruptly changing exposure patterns may express over different time periods. An increased lead exposure (and implied uptake) expresses typically within 40 - 60 days (Tola et al., 1973; Griffin et al., 1975). A net exposure decrease normally requires a longer time period for blood lead levels to reflect this, blood levels being somewhat dependent upon the calibre of the past exposure (Rabinowitz et al., 1977; Gross, 1981; O'Flaherty et al., 1982). Given reasonably stable environmental exposure to lead (a normal situation for urban adult dwellers), little change in blood lead concentration is to be expected with age (Davies, 1988). Children however display a concentration peak at around two to three years of age, followed by a slow decline (De Silva and Donnen, 1980; DOE, 1981 & 1983; Annest et al., 1982; McMichael et al., 1985). No statistically significant gender-related blood lead concentration difference exists for young children at (presumed) comparable exposure levels, although (exposure-related) difference does exist for adults, adult males demonstrating significantly greater levels ($P \leq 0.05$), (DOE, 1981, 1982, 1983, 1986, & 1987; Mahaffey et al., 1982a).

4.2.2.2 SOFT TISSUE STORAGE

Any blood lead which is not excreted via the kidney or is not subject to biliary clearance into the gastrointestinal tract is distributed to soft interstitial tissues where levels tend to stabilise in early adult life (Davies, 1988). For adults, lead's distribution to most body soft tissues is of a much smaller proportion than that distributed to bone (Section 4.2.2.3), with the exception of two soft tissue sites where elevated levels subsequent to protracted exposure are to be anticipated (SRC, 1990). These two sites are the renal cortex (Cramer et al., 1974) and in the aorta, possibly as arteriosclerotic deposits (Cantarow and Trumper, 1944; Radosevic et al., 1961; Barry, 1975 & 1981; Gross et al., 1975;

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[19] Carpenter (1974) concludes for rats that blood lead crosses the placental membrane rapidly and in significant amounts even at relatively low maternal blood lead levels. Congenital lead poisoning is not a recent observation. It was first noted in the children of occupationally-exposed workers during the Industrial Revolution, albeit workers of this era typically demonstrating substantially greater blood lead concentrations than workers currently (Lin-Fu, 1982b).

McMichael and Johnson, 1982). In heavily occupationally-exposed individuals, Barry (1975) has also detected soft tissue lead accumulation in the liver, this organ demonstrating the greatest relative tissue uptake after kidney (Sherlock, 1987), brain (typically selectively the hippocampus [Okazaki et al., 1963; Grandjean, 1978]), pancreas and skin. Soft tissue lead levels demonstrate a half-life of the order of around two months (Moore, 1986). This pool remains fairly rapidly exchangeable between blood and soft tissues, with a reasonably high turnover rate (SRC, 1990). From this pool, transfer may proceed to a less labile, intermediate pool in muscle (SRC, 1990) and skin (Luke, 1991).

4.2.2.3 BONE STORAGE

Subsequent to this intermediate pool storage site in muscle, failing transfer back to blood and excretion, the lead is deposited in calcified tissue in the ultimate long-term, high-volume storage body repository, the long bones. For adults, this represents approximately 80 - 95% of their total body burden of lead (Samuels et al., 1989; Saltzman et al., 1990; SRC, 1990), the analogous figure for children lying closer to 70% (Boeckx, 1986).[20] Here there exist two physiologically distinct pools of lead. One such pool houses lead in the form of relatively inert, insoluble lead phosphate in dense bone, with a half-life of the order of 20 years (Holtzman, 1978; Moore, 1986) to 30 years (Rabinowitz et al., 1977). Reversion to unpolluted bone levels is thus unlikely for most urban dwellers (Bryce-Smith and Stephens, 1981). There also exists another much more labile component in trabecular bone and also marrow where the lead may readily exchange between bone, blood and soft tissue (Rabinowitz et al., 1977; SRC, 1990).

For aged individuals in situations of reasonably steady-state exposure to environmental lead, such as might be commensurate with a long-term moderately busy, urban environment dweller, the "final" most inert repository for body lead may harbour in excess of 200mg of lead (Barry and Mossman, 1970). A total body loading, however, may reach a level as high as one gram for long-term occupationally-exposed aged individuals (EPA USA, 1986; Davies, 1988). The lower figure is easily sufficient a quantity to elicit fatal lead poisoning if released systemically at any one time (Emsley, 1987).[21] Such bone deposition of lead locates typically at the growing ends of bones in children particularly, although it appears not to interfere with bone growth or integrity (Moore, 1986).

This very stable bone lead repository may not necessarily remain inert (Marcus, 1985), and may, however, become a much more rapid-release systemic source of lead under circumstances of relatively rapid bone remodelling, such as might occur in growing children and also subsequent to a fracture

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[20] Although the most biologically labile and available pool of body lead is in the bloodstream, this pool constitutes perhaps only around one to two percent of the total adult body burden (EPA USA, 1986). Given the reasonably rapid clearance of lead from this compartment, blood lead concentrations can really only constitute an index of reasonably recent exposure to and subsequent absorption of lead (Saltzman, 1990). Bone lead stores on the other hand are much more stable a storage form, with accumulation commencing during foetal development (Barltrop, 1969). The majority of soft tissue lead levels change little with age, and bear little or no relationship to the concentration of lead in bones (Samuels et al., 1989).

[21] Both bone lead storage sites are in dynamic exchange (SRC, 1990) and trabecular bone-resident lead may exchange with blood lead within only days (EPA USA, 1986). Subsequent to the removal of exposure to lead for occupationally-exposed individuals, (and in fact all individuals with dense bone-stored lead), the earlier storage of substantial quantities of lead in dense bone, may elicit persistently elevated blood lead levels, even well after the cessation of exposure (Goyer, 1984). Patterson et al. (1991) cite a more modest mean body burden of 40mg of lead per 70kg adult, approximately 1,000 times the figure for Pre-Colombian, Southwest American Indians.

(SRC, 1990). Skeletal lead release may occur subsequent to bone resorption with ageing (Bercovitz and Laufer, 1992) or as also experienced by osteoporitic, post-menopausal women (Silbergeld et al., 1988; SRC, 1990). Pregnancy-related physiological/metabolic stress with increased calcium demands particularly in trimester three eliciting elevated foetal exposure (Manton, 1985; EPA USA, 1986; Silbergeld, 1990; Rabinowitz, 1995), post partum lactation with neonate transfer (Silbergeld, 1987; Silbergeld et al., 1987) and generalised stress (Waldron and Stofen, 1974), may all elicit bone resorption and in-situ lead mobilisation, which, ultimately, may mean elevated foetal exposure (SRC, 1990). Neonate exposure may also be elevated due to the presence of lead in breast milk (Rabinowitz et al., 1976; Huat et al., 1983; Wolff, 1983). Mobilisation of bone-stored lead may also become a concern subsequent to bone demineralisation as may occur with senescence-induced remodelling, kidney dialysis patients and other conditions such as dietary-elicited osteopenia and osteomalacia (Wittmers et al., 1988; Silbergeld, 1990).

4.2.2.4 OTHER LEAD STORAGE SITES

Another tissue where lead may accumulate is teeth, both the dentine and the enamel (Cleymaet et al., 1991), but particularly the former, which is the most responsive component of teeth to lead exposure (Needleman and Shapiro, 1974; Proud, 1976; Khandekar et al., 1978; Needleman et al., 1979, 1982, 1984 & 1990; Purchase and Fergusson, 1986; Paterson et al., 1988; Bercovitz et al., 1993). As for dense bone, teeth are a long-term storage repository where lead levels have been demonstrated to be permanent within the tooth (Strehlow, 1972), and increasing with age (Needleman et al., 1974; Moore, 1986) at a rate proportional to exposure (Steenhaut and Pourtois, 1981; SRC, 1990). Human scalp hair, a tissue that has been widely utilised as an indicator of environmental exposure (Doi et al., 1988), may also demonstrate the presence of systemically-deposited lead (Moon et al., 1966 & 1988; Barltrop et al., 1974; Rabinowitz et al., 1974; Roberts et al., 1974a & b; Barltrop, 1975b; Dale, 1976; Hammer et al., 1976; Baker et al., 1977a & b; Chittleborough, 1980; Fergusson et al., 1981; Heyworth et al., 1981; Matsubara and Machida, 1985; Paschal et al., 1989). Other tissues where lead may be deposited include finger nails (Rabinowitz et al., 1976; Kanabrocki et al., 1979; Moore, 1986), pubic hair (Wilhelm et al., 1990) and skin (Moore, 1986).

Hartwell et al. (1983) in a survey of environmental and human contamination in four smelter communities, determined that scalp hair was more useful than blood or urine in determining relationships between environmental metal levels, distance of residence from the smelter and body toxin burden.[22] Tooth lead, measured as a result of abscission of primary dentition or caries-related extraction, has become an important indicator of storage lead reflecting previous long-term integrated environmental exposure, particularly given the difficulty in acquiring long bone biopsy under normal

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[22] Scalp hair is a conveniently-acquired biopsy material (Raghupathy et al., 1988), the trace element concentration of which is likely to correspond with body stores, especially bone, thus reflecting metabolic changes in stored lead over significant periods of time (Eltayeb and van Grieken, 1990). Thus it is a reasonable indicator of environmental pollution (Carvalho et al., 1984; Wibowo et al., 1986). In healthy persons, scalp hair lead levels may be from ten to fifty times higher than in blood (Schumacher et al., 1991) facilitating easier analysis. Measures of scalp hair lead may be questionable in terms of accurately reflecting internal body (lead) burdens (SRC, 1990). Ideally, such data should not be used to evaluate internal dose-response relationships if adequate steps have not been taken to ensure that the contained trace metal has not arisen as a result of direct atmospheric contamination of the hair (EPA USA, 1986; SRC, 1990).

circumstances (Moore, 1986; Needleman and Bellinger, 1988). Such data have been collected for over two decades in (particularly) the USA, clearly revealing that the lead content of (secondary dentition) children's teeth is greater for urban dwellers than for rural dwellers (Needleman et al., 1972; Stewart, 1974; Shapiro et al., 1975; Fosse et al., 1978). This evidence complements similar conclusions elicited by another study of urban vs rural blood lead levels (O'Heany et al., 1988).

4.2.3 ABSORBED LEAD AND ITS EXCRETION OR RETENTION

4.2.3.1 EXCRETION MECHANISMS

Subsequent to absorption into the bloodstream, dietary lead may be excreted via the kidney or the gut having been hepatically cleared. Of recently-absorbed lead, perhaps 50 or 60 percent is eliminated within only a few days of absorption (EPA USA, 1986). Much of the remainder will eventually be transported to calcified tissue (Rabinowitz et al., 1976) although not all will be stored, some of which (≈ 50% [Kehoe, 1961a & c]) will be excreted eventually (Chamberlain et al., 1978; EPA USA, 1986). Of all excreted lead, Rabinowitz et al., (1973) note that approximately 76 and 16 percent respectively is found in urine and faeces, the balance finding its way into hair (Moon et al., 1968 & 1988; Rabinowitz et al., 1974 & 1976), nails (Suzuki, 1988; Wilhelm et al., 1991), sweat (Florence et al., 1988; Lilley et al., 1988; Stauber and Florence, 1988; Beckman, 1989; Barnes, 1990; Omokhodion and Crockford, 1991; Omokhodion and Howard, 1991) and skin (Batschelet et al., 1979; Moore, 1986). Moore (1986) reports that even normal skin exfoliation and the shedding of hair may be means by which lead is naturally excreted by the body.

EPA USA (1986) reports that infants have a lower total excretion rate than adults for exogenous lead, although contemporaneously claiming the converse for endogenous lead. Further, work conducted on children in the context of retention of recently-absorbed lead suggests that children retain more of this lead than adults (Alexander et al., 1973; Ziegler et al., 1978).

4.2.3.2 RETENTION SEQUELAE

Poisoning of individuals by lead typically reflects a chronic condition resulting from the progressive accumulation of lead within the body, although such poisoning is much less common today, given society's general awareness of the toxicity of lead. This body accumulation of lead often reflects months to years of continuous or intermittent exposure, with clinical symptoms possibly not being displayed (or detected) for much of the exposure time. The toxic effects of lead poisoning, however, would be expected to be demonstrated within days or weeks of a rather high exposure and substantial intake of adventitious lead (ATSDR, 1988).[23]

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[23] Although developed-world examples of frank lead poisoning are typically very unusual at a community or population level, in third world countries and former Eastern bloc countries, typically where few environmental contaminant discharge controls exist, the disease is more frequently noted.

The most sinister scenario with respect to lead poisoning arises as a result of long term exposure (of the order of perhaps months to more than a year or two) of children in the first two or three years of life, at exposure levels eliciting only moderately (continually) elevated blood lead levels (CDC, 1991). Victims typically display very few if any readily-detectable symptoms at the time (Landrigan, 1990). Such a (pre-school) child is at a heightened exposure/toxicity risk since the intake of lead on a unit body basis is typically much greater than for adults (ATSDR, 1988), absorption rates also being greater (Alexander et al., 1973; Ziegler et al., 1978). Certainly mouthing activity (and also pica) is maximal at this time (Moore, 1986; Mushak et al., 1989). The child at this time is undergoing rapid development, associated metabolic and physiological activity occurring permitting lead to induce enhanced toxicological effects at a number of locations, particularly the developing CNS (Mushak, 1992).

4.2.4 HUMAN HEALTH EFFECTS OF LEAD - OVERVIEW

4.2.4.1 INTRACELLULAR AND CYTOLOGICAL EFFECTS

At an intracellular level, lead has particular affinity for the carboxyl groups of glutamic and aspartic acid, the sulphydryl groups of cystein, and the phenoxy groups of tyrosine (Luke, 1991). Such activity elicits adverse influence on (particularly) mitochondrial membranes and enzymes involved with energy metabolism and ion transport, interfering with the normal metabolism of cations, eg calcium (Moore, 1986). In fact, the principal target organelle for this toxin's activity for a range of cell and tissue types is the mitochondrion (SRC, 1990). Here, its effect is represented by structural changes to the mitochondrion (eg swelling and distortion of internal membrane surfaces), and marked disturbances in mitochondrial function. Such effects are associated with observable intra-mitochondrial lead accumulation (Silbergeld et al., 1977; EPA USA, 1986). Mitochondrial changes appear intimately intertwined with cellular biochemical effects such as altered energy metabolism and cellular respiration pathways, with modification of the kinetics of a whole plethora of intracellular reactions involving calcium (Bull, 1980).

At a cytological level, lead exerts its toxic effect by eliciting disturbance to normal cell function in a number of body organs and systems. Absorbed lead is quite rapidly distributed by the bloodstream according to rate of blood delivery to various organs. It is then redistributed to various tissues proportional to their respective affinities for the toxin, constant exposure eliciting approximate steady state intercompartmental (body) distribution (Davies, 1988). Whilst lead's distribution is ubiquitous within the environment (Nriagu, 1988; Landrigan and Curran, 1992; Berney, 1993) it is also so within the body (Moore, 1986).

4.2.4.2 BODY SYSTEM EVIDENCE OF LEAD'S TOXIC EFFECTS

The principal organs/systems impaired by lead's toxicity are the haem synthesis and recycling systems, lead eliciting a host of haematological effects here, (Piomelli et al., 1982; Hammond et al., 1985). Other reflections of lead toxicity include renal effects (Goyer and Rhyne, 1973; Goyer, 1986; Goyer et al., 1989), effects on the normal reproductive function of both males and females at medium to high levels (Rom, 1980) and disturbances to hormonal function, particularly disturbed Vitamin D synthesis (Rosen et al., 1980; Mahaffey et al., 1982b). Toxic effects also reflect impaired perinatal indicators of subsequent development such as birth weight and gestational age (Bornschein et al., 1989; Bellinger et al., 1991a & b), disturbance to postnatal indicators of physical development eg stature (Lauwers et al., 1986; Schwartz et al., 1986; Shukla et al., 1987 and Huseman et al., 1987 - who suggest that this is mediated via a lead-impaired release of thyroid-stimulating hormone), and neurological effects particularly in the developing CNS of children.

Typically, exposure to lead in this early period in life may result in irreversible damage to the CNS, indexed as cognitive and other neurobehavioural decrements (Needleman et al., 1979 & 1990; Schroeder et al., 1985; Bellinger et al., 1987b, 1988 & 1991b; Fulton et al., 1987; Hatzakis et al., 1987; McMichael et al., 1988; Dietrich et al., 1989, 1990 & 1993; Hansen et al., 1989; Winneke et al., 1990; CDC, 1991). Such damage may also register as neurophysiological decrements indexed by such decline as hearing acuity. This registers as an elevation of hearing threshold with increasing blood lead level, with no apparent blood lead threshold (Otto et al., 1982 & 1985; Robinson et al., 1985 & 1987; Schwartz and Otto, 1987a & b), much of the CNS still undergoing development within the first two to three years of life (Moore, 1986).

Such significant changes at the intracellular level manifest at system level particularly at the high end of the exposure spectrum as functional aberration to haem synthesis and its role for normal erythropoiesis. This is evidenced as anaemia, the "fundamental" (Moore, 1988; p. 426) and "classical" (Baker et al., 1979; p. 62) manifestation of lead intoxication. Also evident is renal deterioration, and even failure, (Landrigan, 1990), wrist drop (Shie, 1924), and complex CNS disturbance. More covert (possibly) sub-clinical correlates include slowed (typically peripheral) nerve conduction passage (Catton, 1970; Feldman et al., 1973; Roberts et al., 1974; Landrigan et al., 1976; Araki and Honma, 1982; Jeyaratnam et al., 1985; Schwartz et al., 1988) and disturbances to normal excretion of uric acid (Mahaffey, 1985).

Another literature-noted clinically-obvious non-threshold effect of lead's toxicity is hypertension (Lippman, 1990) which is dose-related and potentially confounded by many other factors, (EPA USA, 1986) although, currently, not as frequently detected as earlier this century (Landrigan, 1990). Also included here is gout (Landrigan, 1990), a variety of gastrointestinal disorders including colic and constipation (Aub et al., 1924; Shie, 1924; EPA USA, 1986; Moore, 1986; SRC, 1990; IPCS, 1991), pallor (Shie, 1924) and occasionally palsy (paralysis) (Lin-Fu, 1980).[24] Lead-elicited carcinogenesis

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[24] For modern medicine, saturnine gout has, for more than two centuries, been associated with substantial lead exposure, typically as a disease faced by the middle-upper class wealthy elite who, as Wedeen (1984) suggests, consumed copious quantities of lead-containing wine and foodstuffs, as was also described by Nriagu (1983b) to be very much the norm in Roman days (Chapter Two).

has been inferred, although, currently, evidence for the carcinogenic nature of lead for humans appears inadequate (Maynard et al., 1993 - Section 4.2.5.9).

Acute intoxication in children particularly may be evidenced by subtle behavioural changes including reduced attention span, increased crying and general fearfulness and a frequently-reported general regression (Byers and Lord, 1943; White and Fowler, 1960; Weiner, 1970; Walter, 1980). A characteristic feature of lead poisoning particularly under conditions of poor oral hygiene is the appearance at the gingival margin of a bright blue band referred to as Burtonian blue line/gum (Moore, 1986), reflecting both extra- and intracellular deposition of (insoluble) lead-containing materials, particularly lead-sulphur compounds (Honigsman et al., 1974).

A number of these conditions are discussed in more detail below, summary details of which for acute exposure are included in Table 4.1.

Table 4.1 Recognised body system medical conditions resulting from acute lead exposure.

BODY SYSTEM	CONDITION ARISING FROM ACUTE LEAD INTOXICATION
URINARY	Nephropathy/Nephritis leading to fluid retention, hypertension and cerebrovascular accident, porphyria, ultimately leading to renal failure.
CVS	Electro-cardiographic abnormalities, anaemia, hypertension, heart disease.
MUSCULO-SKELETAL	Gout (particularly long-term exposure), wrist-drop due to palsy of extensor muscles
GIT	Nausea, constipation, anorexia, severe colic, gastric distress and malaise, diarrhoea, persistent vomiting
PNS & CNS	Various encephalopathy: neuritis of (particularly) optic nerve leading to aberration and blindness, hearing aberration, peripheral tremor and numbness, distal ataxia (loss of muscular co-ordination resulting in disturbed gait, poor motor control), confusion, disturbed speech, sleeplessness, fatigue/malaise, irritability, behaviour change, dementia, madness, convulsions, epilepsy, paralysis, coma, death (mainly due to cerebral oedema and haemorrhage).
REPRODUCTIVE	decreased fertility in men particularly. Spontaneous abortion.

Source: EPA USA, 1986; Shy, 1990; SRC, 1990; Best, 1991; Mushak, 1992; OECD, 1992; Berney, 1993.

4.2.5 HUMAN HEALTH EFFECTS OF LEAD - SYSTEMS AND ORGANS

From the literature, a vast selection of tomes pertaining to the specific human toxic effects of lead exists, including many excellent reviews (See eg EPA USA 1986; ATSDR, 1988b; SRC, 1990; CDC, 1991; IPCS, 1991). This section provides a summary of the findings regarding specific target systems and organs from the literature consulted.

Certainly lead-associated symptoms have been noted for millennia (Landrigan, 1990), some authors suggesting that the Roman use of (sweet-tasting) lead acetate in food and wine production (Chapter Two) contributed to the decline of the Empire (Mack, 1973).

Of greater concern currently, however, given the apparent decline in frank lead poisoning over the last two to three decades (Moore, 1986; SRC, 1990; Luke, 1991), is the more subtle effect of, generally, low(er) body levels of the toxin particularly upon infants. Over the last two decades, however, from a review of the literature, particular attention appears to have been directed toward epidemiological research focussing on potential neurotoxic effects of lead on the developing child (particularly at sub-clinical level). Such effects include, most importantly, delayed or impaired neurobehavioural development and performance (Section 4.2.5.4).

4.2.5.1 INFLUENCE ON HAEMOPOIESIS

Haem, an iron (Fe^{++})-rich porphyrin is a precursor of haemoglobin, a conjugated protein consisting of a haem group bound to four polypeptide chains arranged in a tetrahedryl configuration. This chemical species demonstrates the ability to combine reversibly with oxygen, and is the principal binding site for the transport of oxygen by erythrocytes in the blood (Harper, 1971). Analogues of haem are also incorporated into a number of other high molecular weight proteins, eg myoglobin, the P-450 (cytochrome) component of the mixed function oxygenase system (EPA USA, 1986), and a number of cytochromes involved in cellular respiratory, energetics and electron transport chain pathways. Of the myriad adverse biochemical body interactions of lead, the impact of lead on haem synthesis and thus erythropoiesis is most sinister biochemically, the toxin effecting its influence via disturbance to a number of enzymes and enzyme-catalysed reactions (Moore, 1986).[25]

Given lead's potential to compete at haem binding sites with oxygen (IPCS, 1991), and also lead's ability to disturb the synthesis of haem (EPA USA, 1986), it has the potential to affect fundamental metabolic and energy-transfer processes (Moore, 1986; Davies, 1988; SRC, 1990). Such a dramatic effect infers toxicity to a variety of different organ and tissue types.

Consequent to changes to the erythrocyte membrane elicited as a result of lead binding to the membrane (Moore, 1988), considerable shortening of the normal 120 day red blood cell life span

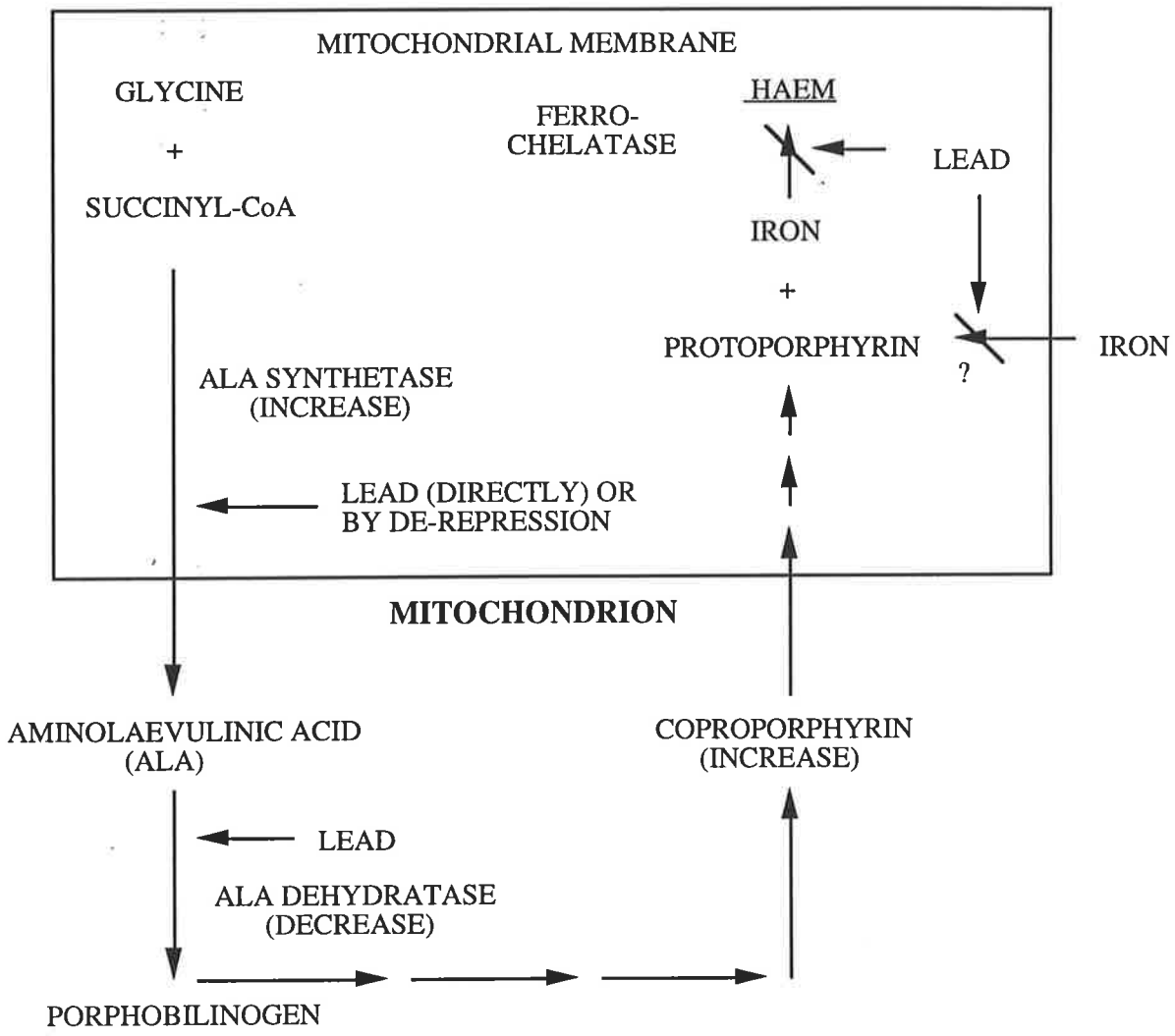
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[25] Acknowledging the heterogeneity of leucocytes, lead alters a number of functions of these cells also. These include disturbances to phagocytosis and the patency of the immune response (Gainer, 1974; Blakely and Archer, 1981; Blakely et al., 1982), reflecting as a deterioration of the response to infection (Hemphill et al., 1971).

occurs (EPA USA, 1986), inferring a direct toxic effect to the membrane (Waldron, 1966; Moore and Goldberg, 1983).

As a result of a combination of shortened erythrocyte life with an associated direct inhibition to the oxygen-binding capacity of haem (in blood) due to lead binding competitively (EPA USA, 1986), frank anaemia is a common symptom of lead intoxication (SRC, 1990). The toxin induces this malady in children at very much lower levels of exposure than for adult men (Moore, 1988). Since children and women have smaller iron stores than men on average (Moore, 1988), and iron deficiency can influence lead's toxicity (EPA USA, 1986), women and children are more susceptible to lead's toxicity (Hemphill et al., 1971).

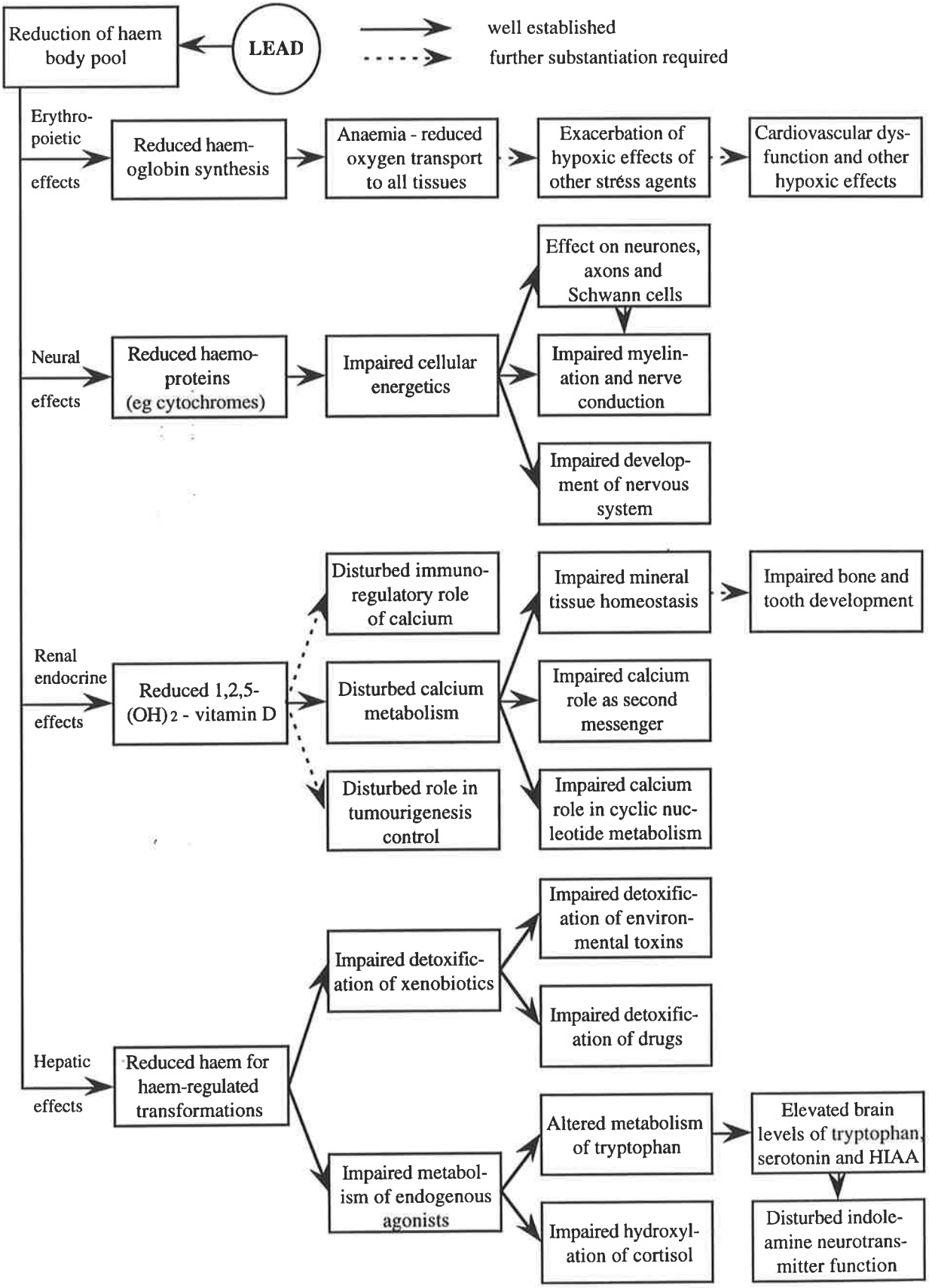
Lead's interference with haem biosynthesis is reflected by Figure 4.2.



Source: SRC, 1990.

Figure 4.2 The effects of lead upon haem biosynthesis

An indication of the significance of lead's interference with haem biosynthesis resulting in the disrupting effect upon a wide variety of important physiological processes is reflected in Figure 4.3.



Source: SRC, 1990; p. 72

Figure 4.3 Multi-organ impact of lead's effect upon the haem pool.

Lead exerts its influence here by principally altering the activity of three enzymes (EPA USA, 1986), namely delta-amino laevulinic acid synthetase (ALA-S), delta-amino laevulinic acid dehydratase (ALA-D) and ferrochelatase. These are discussed separately below.

a) ALA-S effects

The presence of lead indirectly stimulates, through feedback derepression, the mitochondrial enzyme ALA-S an enzyme which is responsible for catalysing the condensation of glycine in the presence of succinyl-coenzyme A to form delta-amino laevulinic acid. This enzyme's activity is the rate-determining step in the biosynthesis of haem (SRC, 1990). It has been documented by a number of researchers that occupationally-exposed workers display elevated ALA-S activity (Takaku et al., 1973; Campbell et al., 1977; Meredith et al., 1978).

b) ALA-D effects

Lead also directly inhibits the cytosolic enzyme ALA-D (Haeger-Aronsen et al., 1971; Valentine et al., 1982), an enzyme responsible for the condensation of two units of ALA to form porphobilinogen. This enzyme's activity correlates inversely with blood lead levels of, both occupationally-exposed (Roels et al., 1976; EPA USA, 1986) and non occupationally-exposed (urban-residing) individuals (Hernberg and Nikkanen, 1970; Chisholm et al., 1985a; ATSDR, 1988b). Additionally, the activity of this enzyme appears to be inhibited at even very low blood lead levels of $\approx 3\mu g.dL^{-1}$ (Hernberg and Nikkanen, 1970), with no evidence for a minimum blood lead concentration threshold for such enzyme inhibition being yet apparent (EPA USA, 1986). More recent work has revealed that such lead-induced enzyme inhibition occurs at trace levels, ie picomolar rather than previously-postulated micromolar levels (Markovac and Goldstein, 1988), again with no apparent threshold effect (CDC, 1991; Flegal and Smith, 1992).

c) Ferrochelatase effects

The activity of the mitochondrial enzyme ferrochelatase which catalyses the insertion of iron (Fe^{++}) into the protoporphyrin ring to form haem, is also decreased by lead (Goyer and Rhyne, 1973). This inhibition occurs either by direct inhibition of the enzyme's activity, or by alteration of the intramitochondrial transport of iron (Moore and Goldberg, 1985; EPA, 1986), contributing to the anaemia occasioned by lead poisoning. The inhibition by lead of ferrochelatase activity results in a rise in protoporphyrinogen synthesis, an accumulation of protoporphyrin IX (present as zinc protoporphyrin [ZPP] in circulating erythrocytes, resulting from the substitution of zinc rather than iron in the porphyrin moiety [EPA USA, 1986]), and an increased excretion of urinary coproporphyrin. This substituted species (ZPP) remains bound in the haem component of haemoglobin for the life of the erythrocyte.

The combined effect of lead's stimulation of ALA-S and inhibition of ALA-D is to elicit an increased blood, plasma and urine ALA level (SRC, 1990), Meredith et al. (1978) observing for a group of occupationally-exposed individuals and controls, an exponential relationship between blood lead and blood ALA. EPA USA (1986) cites a number of similar category individuals demonstrating a direct relationship between blood lead concentration and the logarithm of urinary ALA concentration. ATSDR (1988b) notes a similar relationship in children, although principally at blood lead levels > $40\mu g.dL^{-1}$. For adults, this correlation is detectable at blood lead concentrations below $40\mu g.dL^{-1}$ (Selander and Cramer, 1970) albeit with a less steep slope than that applicable for values greater than $40\mu g.dL^{-1}$ (SRC, 1990).

Normal globin synthesis is inhibited by the presence of lead in blood, typically at levels below $70\mu g.dL^{-1}$, and arising independently of the changes to the biosynthesis of haem (Ali and Quinlan, 1977; Moore, 1988). Thus, eliciting a marked depression of haem synthesis, lead intoxication results in an overall reduction of the haemoglobin concentration in blood, with a decreased net haemoglobin production. Coupled with this is an increased rate of erythrocyte destruction typically reflected by enhanced red cell haemolysis resulting from increased erythrocyte fragility (Moore, 1986).[26] Clinically, this manifests as hypochromic, normocytic anaemia with associated reticulocytosis (SRC, 1990), such decreased haemoglobin levels and anaemia, long having been detected in lead workers (SRC, 1990).

It has also long been recognised for workers displaying lead intoxication or individuals with heightened exposure to lead with anaemia sequelae, that an increase in urinary coproporphyrin is a good indicator of such a condition. This is a detectable phenomenon at blood lead levels of approximately $40\mu g.dL^{-1}$ (EPA USA, 1986). Campbell et al. (1977) observe that lead inhibits the mitochondrial enzyme coproporphyrinogen oxidase which catalyses the conversion of coproporphyrinogen to protoporphyrinogen, thus eliciting an elevated urinary coproporphyrin level.

Similarly, for occupationally-exposed individuals, EPA USA (1986) notes a threshold for the detection of a decrease in haemoglobin levels at blood lead levels in excess of approximately $50\mu g.dL^{-1}$, frank anaemia in adults arising at a blood lead level of approximately $80\mu g.dL^{-1}$ (Tola et al., 1973; Baker et al., 1979; Grandjean, 1979).

Exposure to lead inhibits the formation of the (haem-containing) protein cytochrome P-450 in children, reflecting decreased hepatic mixed function oxygenases activity, (Alvares et al., 1975; Saenger et al., 1984). The former authors suggest for children suffering acute lead poisoning, that they did not metabolise the test drug antipyrine as rapidly as did controls, a reaction mediated by hepatic mixed-function oxygenases.

Clearly, lead's influence on the biosynthesis and metabolism of the haem molecule is complicated and multi-faceted. Haem's role in various body systems and organs is essential, a full description being beyond the purview of this text.

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[26] This may be partly due to lead's inhibition of pyrimidine-5'-nucleotidase, resulting in an accumulation of pyrimidine nucleotides (cytidine and uridine phosphates) in the erythrocyte or reticulocyte. The enzyme inhibition and nucleotide accumulation of pyrimidine nucleotides affects erythrocyte membrane stability and survival by substantial alteration to cellular energetics (Angle et al., 1982; EPA USA, 1986).

4.2.5.2 LEAD'S EFFECT UPON THE KIDNEY

For more than a century, it has been recognised that lead is toxic to the kidney and is often deposited there (Moore, 1986). Severe kidney disease ranging from chronic nephropathy to renal failure frequently results (EPA USA, 1986). The identification of contributing causes and mechanisms however is somewhat incomplete, given the complexities and often long-term nature of human exposure to lead and other nephrotoxic agents (SRC, 1990).

Typically, the evolution of lead nephropathy is (clinically) silent, the major focus being the progressive destruction and subsequent fibrosis of proximal tubular cells (Vacca et al., 1986; Goyer et al., 1989).

For occupationally-exposed workers chronic lead-induced nephropathy has been evidenced at blood lead levels as low as $30\mu g.dL^{-1}$. This figure for children is substantially higher, perhaps exceeding $100\mu g.dL^{-1}$ (EPA USA, 1986), although this may reflect the much longer term exposures to lead that are (presumably) necessary to induce nephropathy. The cells lining the proximal tubules appear to be the renal tissue most highly sensitive to lead (Goyer and Rhyne, 1973). Such tissue damage as may occur manifests as tubular necrosis or interstitial fibrosis leading to atrophy. This arises particularly subsequent to chronic, reasonably high doses of lead, with disturbance to the renin-angiotensin balance and ensuing hypertension (Goyer and Rhyne, 1973; Campbell et al., 1985) possibly leading to cerebrovascular accident (Dingwall-Fordyce and Lane, 1963).[27] Lead absorption even at relatively low levels has been associated with significant blood pressure elevation in humans (Pirkle et al., 1985; Harlan, 1988), and also in the rat model (Victery et al., 1982).

At blood lead levels in the range 40 - $80\mu g.dL^{-1}$, lead induces the formation of dense intranuclear lead-protein complex inclusion bodies in the cellular lining of the proximal tubules, with associated increased reabsorption of uric acid by these cells. This is clinically evidenced by hyperuraemic gout (Goyer and Rhyne, 1973), and proteinuria (Vacca et al., 1986). Such malfunction arises essentially due to lead-induced damage to renal tubular function, resulting in retention of uric acid and guanine in muscles and joints of particularly the lower limbs. This may lead to a strikingly oedematous condition, and can be very painful (Best, 1991).

Renal disease-elicited mortality in lead workers has been frequently observed (Cooper and Gaffey, 1975; Malcolm and Barnett, 1982; McMichael and Johnson, 1982; Selevan et al., 1985), Landrigan (1990) reporting, from a review of these studies, an average two to three fold increase in chronic nephritis and fatality resulting. In their study, Selevan et al. (1985) reported a positive association between duration of employment in a lead smelter and mortality from nephritis.

4.2.5.3 EFFECT OF LEAD UPON REPRODUCTION

Deriving from epidemiological data collected over the last century and a half, it has recognised over most of this time that (high) lead exposure for pregnant women is associated with an elevated

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[27] Typically, lead inhibits the metabolic activation of Vitamin D in the cells lining the proximal tubules at blood lead levels less than $25\mu g.dL^{-1}$ (Rosen et al., 1980).

likelihood of spontaneous abortion (SRC, 1990). Further, much data reflects reproductive toxicity for (both sex) lead workers with high to very high exposure levels of lead, increased spontaneous abortion being also reported for wives of exposed male workers (Oliver, 1914b; Hamilton and Hardy, 1974). Such observations have been confirmed by laboratory studies employing high doses of lead in test animals (EPA USA, 1986), although these authors do stress a need for more critical research evaluating the more subtle toxic effects of lead on the developing foetus.

a) Lead and maternal effects

Foetal development is endangered by the presence of maternal lead since the toxin may passage the placenta in a relatively unimpeded fashion (Barltrop, 1969; Carpenter, 1974; Fahim, 1976; Kaul et al., 1983; Borella et al., 1986; Mayer-Popken et al., 1986; Landrigan, 1990). Importantly here, this occurs at a time during gestation before the embryonic nervous system has being laid down (Barltrop, 1969; Carpenter, 1974; Borella et al., 1986; Mayer-Popken et al., 1986), maternal and cord blood lead concentrations correlating highly (Alexander and Delves, 1981; Rabinowitz and Needleman, 1982).

b) Lead and paternal effects

EPA USA reports that "there are currently no reliable data pointing to adverse effects in offspring following lead exposure of fathers *per se*" (1986; p. 120). Luke (1991), however, referring to a recent (uncited) review, notes a clear association between an individual's blood lead level, when in the range 40 - 70$\mu g.dL^{-1}$, and abnormalities such as low sperm counts (also noted by Cullen et al., 1984). Such changes accompany increases in the numbers of abnormal sperm and alterations in some hormonal parameters (noted by Assennato et al., 1986), with the inference that the impact of such lead-elicited disturbances may be visited upon future offspring (EPA USA, 1986). This accords with observations of adverse impacts by lead on sperm (depressed sperm count) and testes (peritubular fibrosis) occurring in chronically exposed men displaying blood lead levels of the order of 40 - 50$\mu g.dL^{-1}$ (ATSDR, 1988b). Other noted effects upon the male reproductive system include erectile dysfunction, asthenospermia, hypospermia, teratospermia, and changes in semen cytology such as reduced sperm motility and marked morphological changes (Lancranjan et al., 1975; Bell and Thomas, 1980; Cullen et al., 1984).

Some researchers report that, mediated by a lead-induced depression of paternal testosterone levels, lead influences the sex ratio of offspring, resulting in the birth of more girls than boys (James, 1992; Dickinson and Parker, 1994).

c) In utero lead contamination

Early research has observed lead-elicited decreased length of gestation and decreased child birth weight (Fahim et al., 1976; Nordstrom et al., 1979), although these results have been questioned on the basis of both inappropriate analytical aspects and improper controlling for confounding variables

(EPA USA, 1986). Other workers of that era, however, have found no significant association between prenatal lead exposure and effects of that ilk (Clark, 1977; Roels et al., 1978b; Alexander and Delves, 1981). Davis and Svendsgaard (1987), from a review of a number of more recent studies, conclude that the duration of gestation is affected by exposure to lead during pregnancy, and that these effects can occur even when blood-lead levels are below $15\mu g.dL^{-1}$. Needleman et al. (1984) and Bellinger et al. (1984), however, did not find such a significant relationship.

The maternal (in utero) environment and associated child effects has been investigated locally (Port Pirie) within the last decade, with results confirming a lead-elicited decreased length of gestation, decreased child birth weight and a significantly greater incidence of miscarriage and stillbirth for mothers residing proximate to the potential lead source compared with a control group (Vimpani et al., 1985; McMichael et al., 1986). Fischbein (1983) reports that the incidence of miscarriage and stillbirth is greater in the wives of men who worked in industries eliciting heightened (personal) occupational lead exposure. Ward et al. (1987) found statistically significant inverse correlations between placental lead levels and birth weight, head circumference and placental weight, although not with gestational age. Moore et al. (1982) as Dietrich et al. (1986) reported significant negative correlations between maternal (or cord) blood lead levels and gestational age.

According to SRC (1990), having reviewed a large number of human studies, the US EPA has noted that developmental consequences of low-level prenatal exposure to lead do occur, even at cord or maternal blood lead levels of approximately $15\mu g.dL^{-1}$. These include reduced birth weight and gestational age, and also neurobehavioural deficits or delays (Section 4.2.5.4), although, as Ernhart et al. (1985 & 1986) and McMichael et al. (1986), no clear evidence of an association between prenatal lead exposure and (human) congenital anomalies/malformations has been demonstrated.

Contrasting these results, however, Needleman et al. (1984), in a study of 4,354 infants born in Boston, found a statistically significant association between cord blood lead levels and the collective occurrence of minor anomalies. The most common of these anomalies were lymphangiomas, haemangiomas, minor skin anomalies, and undescended testicles, although no individual anomaly was significantly associated with blood lead level. From these data, there appeared to be no association between blood lead level and major malformations, birth weight (curiously), or gestational age, the last-mentioned concurring with data of Bellinger et al. (1984).

Data from the Cincinnati (Ohio) study conducted by Bornschein et al. (1987) indicated that for each natural log increase in blood lead level, a decrease in birth weight ranged from 58 - 601g, depending upon the mother's age, this being a statistically significant inverse correlation. Further, follow-up of 260 infants from this cohort demonstrated an inverse correlation between postnatal growth rates from three to fifteen months of age and postnatal increases in blood lead (Schwartz et al., 1985; Shukla et a., 1987). This appears to confirm a strong association between (postnatal) lead exposure and (child) growth retardation, blood lead level being a statistically significant predictor of (less than seven-year-old) children's height, weight and also chest circumference (Schwartz et al., 1986).[28] Here, the strongest relationship with blood lead level was with child height. No threshold was

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[28] Not inconsistent with this observation and the observation that lead interferes with neuropsychological development (Section 4.2.5.4), data from the US National Health Examination Survey demonstrate a positive relationship between height and intelligence in children up to eight years of age (Wilson, 1986).

evident down to the lowest observed concentration of $4\mu g.dL^{-1}$, although the mechanism for this phenomenon remains unclear (ATSDR, 1988b).

The frequency of miscarriages and stillbirths has been associated with maternal blood lead levels (Fischbein, 1983), although from McMichael et al.'s study (1986), whilst the maternal blood lead concentrations at birth for women residing in Port Pirie was greater than those residing elsewhere, and the frequency of stillbirths and miscarriages for women residing in Port Pirie was also higher, the average maternal blood lead level at delivery was significantly lower for stillbirths than for live births (McMichael et al., 1986). This observation appears somewhat "puzzling" (SRC, 1990; p. 104). Puzzling also was the observation that the proportion of Port Pirie pregnancies resulting in low birth-weight infants was more than twice that for outside (Port Pirie) pregnancies, yet the maternal and cord blood lead levels were somewhat lower in low birth-weight pregnancies than in pregnancies with birth weights greater than 2,500gms. Such (apparently) anomalous findings from this study have prompted Davis and Svendsgaard to suggest that there exists a mechanism for the increased transfer of maternal lead to the foetus, which acts as a "sink" (1987, p. 299) for the mother's body burden.[29] Such a suggestion is supported by the work of others demonstrating an inverse correlation between placental lead level and birth weight, head circumference and placental weight (Ward et al., 1987), and, for cases of stillbirth and neonatal death, increased levels of lead in the placenta (Wibberley et al., 1977).

4.2.5.4 LEAD'S TOXICITY TO THE DEVELOPING NERVOUS SYSTEM

From the wealth of literature available in this regard, lead's influence on the nervous system would appear to be the most thoroughly researched body system interaction with the toxin. This is due to the likelihood of lead-elicited CNS effects occurring at quite low body levels of lead, levels which were previously held to be harmless (Needleman and Bellinger, 1988). Such a well-researched body system interaction with lead also accords to the presumed irreversibility of such effects, particularly in children in infancy (EPA USA, 1986; CEH & CAPP, 1987; Mushak, 1992).

Lead is potentially harmful to both the central and peripheral nervous systems, and has been known for more than a century to be a neurotoxin capable (at reasonably high body levels) of eliciting CNS encephalopathy. This represents the most severe and sinister neurobehavioural effect of lead in adults (SRC, 1990), even progressing to fatality in some individuals (Gloag, 1980).[30] Fischbein (1983) claims that the toxin elicits its influence on the CNS by interfering with some neurotransmitters, IPCS stating that lead does not appear to "specifically alter any particular neurotransmitter system" (1991; p. 119). They support this claim by reporting that lead interferes with pathways utilising dopamine, noradrenaline, serotonin, gamma-aminobutyric acid and acetylcholine, all having been shown to be affected in the animal model (Shallenburger, 1984).

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[29] This was previously noted as a result of observations of women employed in British white lead factories around the turn of the century, some of whom claimed that pregnancy was a means by which they elicited a transfer of their body lead burden to their unborn child, thus protecting themselves (Oliver, 1914b).

[30] Such CNS intoxication was recognised and associated symptoms clearly described as a disease of occupationally-exposed individuals more than 150 years ago, being termed saturnine encephalopathy (Tanquerel des Planches, 1839; Westphal, 1888; Oliver, 1891, 1914a & b; Aub et al., 1926).

a) Central nervous system effects of lead

A number of workers have observed overt neurological impairment signs at blood lead levels ≥ 40μg.dL^{-1} (Baker et al., 1979 & 1983; Haenninen et al., 1979; Zimmermann-Tansella et al., 1983; Campara et al., 1984). Several researchers have observed occulomotor function disturbances in adult lead workers at mean blood lead concentrations of approximately 60μg.dL^{-1} (Baloh et al., 1979; Spivey et al., 1980; Glickman, 1984), and deficits in reaction time and hand-eye coordination of lead workers with similar mean blood lead levels (Morgan and Repko, 1974).

At a more sinister level, however, early encephalopathic symptoms that may develop within weeks of initial exposure include dullness and dizziness, personality changes such as irritability to hyperirritability, poor attention span, and headache. Continuing exposure may elicit muscular tremor, short-term memory loss, sleep disturbance, and hallucinations (EPA USA, 1986; Moore, 1986; SRC, 1990; CDC, 1991; IPCS, 1991), a condition that may worsen given continued or increased exposure to the toxin. Such deterioration may occur abruptly (Moore, 1986), characteristic symptoms including ataxia, convulsions, delirium, stupor, paralysis, coma and eventually death (Cumings, 1959; NAS, 1972; ATSDR, 1988b). Post-mortem, fatal encephalopathy brain histopathology findings reveal, similarly for adults and children (Davies, 1988). Observed pathology includes cerebral oedema, arteriole and capillary dilation, and proliferation and swelling of endothelial cells. Also noted are variable anoxia-elicited neuronal degeneration (commonly, lesions detected reside in the nuclei of cranial nerves such as optic, auditory, olefactory and glossopharyngeal), proliferation of perivascular glial cells (as a direct toxic reaction to lead rather than as a result of neuronal degeneration [Pentschew, 1965]), focal necrosis and evidence of elevated cerebro-spinal fluid pressure (Goyer and Rhyne, 1973; Moore, 1986; Davies, 1988; SRC, 1990).

If the episode is near fatal, recovery may be very slow with potential for permanent mental retardation, partial or total blindness and motor paralysis (Luke, 1991).[31] Such "pseudoparalysis" appears clinically indistinguishable from progressive paralysis of the insane, subsequent permanent epilepsy being not an uncommon association (Oliver, 1914b). More typically, following a more prolonged but less severe exposure to lead, encephalopathy progresses slowly, with an observed mental deterioration resembling general paralysis with noted demonstrable arteriosclerosis (Luke, 1991).

Severe lead encephalopathy is usually not apparent for adults until blood lead concentrations exceed ≈ 100μg.dL^{-1} (Smith et al., 1938) to 120μg.dL^{-1} (Kehoe 1961a, b & c). Overt neurotoxicity, however, may be evidenced at lower blood lead levels (of the order of 50 - 70μg.dL^{-1} [Baker et al., 1979; Haenninen et al., 1979], perhaps even as low as 40 - 60μg.dL^{-1} [SRC, 1990]). Both groups of workers reinforce that first-detected symptoms are gastrointestinal disorder, anaemia and extensor muscle weakness. Parkinson et al. (1986) detected these and a number of other symptoms such as differences in neurobehavioural or psychological indices between occupationally-exposed workers (with time-weighted mean blood lead levels of ≈ 50μg.dL^{-1}) and a control group.

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[31] Blindness as a result of lead encephalopathy has frequently been reported historically (Gibson, 1904). Its pathology relates to atrophy of the optic nerve following degenerative neuritis and often is associated with astrocyte and microglial cell proliferation (Pentschew, 1965).

A study of workers at a storage battery plant by Campara et al. (1984) revealed consistent and significant dose-response trends in symptoms such as loss of appetite, lower-limb paresthesis, upper-limb weakness and dropping of objects as for other more characteristic encephalopathic symptoms. Here, from a group of 20 men with a mean blood lead level of 53$\mu g.dL^{-1}$, respondents exhibited significantly poorer skills when participating in neurobehavioural tests with general performance for cognitive, visual-motor coordination tasks and verbal reasoning ability most markedly impaired. Lead-elicited neurobehavioural disorders are discussed in more detail in Section 4.2.5.4 c) below.

High-level lead exposure produces encephalopathy in children, however, the presence of any signs of acute lead poisoning in some children may not appear evident at mean blood lead levels less than 105$\mu g.dL^{-1}$ (range 60 - 300$\mu g.dL^{-1}$) (NAS, 1972 - summarising paediatric data from Chisholm [1962 & 1965] and Chisholm and Harrison [1956]). This review further noted that acute lead intoxication symptoms other than encephalopathy were observed at a mean blood lead concentration of 178$\mu g.dL^{-1}$ (range 60 - 450$\mu g.dL^{-1}$) and encephalopathic signs such as ataxia, stupor, hyperirritability, convulsions and coma were associated with a mean blood lead level of 330$\mu g.dL^{-1}$ (range 90 - 800$\mu g.dL^{-1}$).

The distribution of blood lead levels associated with death (mean = 327$\mu g.dL^{-1}$) would appear virtually the same as for levels associated with encephalopathy (SRC, 1990). Acute encephalopathy in the most susceptible of a group of children studied has been demonstrated at blood lead levels of approximately 100$\mu g.dL^{-1}$ (Cumings, 1959; Rummo et al., 1979; EPA USA, 1986).

Symptomatic, acute lead intoxication of children may or may not involve encephalopathy, however, numerous studies report that in both situations such poisoning is very likely to result in permanent neurological, behavioural and/or cognitive impairment (EPA USA, 1986). This conclusion appears at variance with pre-1940's views to the effect that unless severe symptoms of brain dysfunction were elicited (eg coma or convulsions - childhood lead-elicited encephalopathy being recognised in the 1930s and 1940s, [Fergusson, 1986]), no permanent damage as a result of the lead exposure would occur. Researchers in the early 1940s, however, demonstrated (apparently) permanent neuropsychological impairment (Section 4.2.5.4) in children in the absence of severe encephalopathy (Byers and Lord, 1943; the first group to report the neuropsychological effects of lead, [Fergusson, 1986]).

b) Peripheral nervous system effects of lead

As with lead-elicited CNS damage, the PNS-centred results of lead intoxication (particularly under circumstances of high exposure levels) have also been recorded for more than a century and a half (Tanquerel des Planches, 1839), with motor neuron neuropathy occasioning extensor muscle palsy, and leading to conditions such as wrist or ankle drop, these being the classical clinical manifestation (ATSDR, 1988b; Landrigan, 1990).[32] Histopathological investigation of the PNS subsequent to fatal encephalopathy typically reveals axonal degeneration and also segmental demyelination of motor

[32] Such observations were also noted more than two thousand years ago by the Greek poet and physician Nicander (Major, 1939).

neurones (Fullerton, 1966; Windebank and Dyck, 1984). Motor axons represent the principal target of lead in the PNS (Landrigan, 1990).

More typically, however, lead-elicited damage to the PNS arises as a result of lower exposure to the toxin, at which level of exposure, individuals appear asymptomatic on presentation (SRC, 1990). Such level of intoxication may result in more subtle effects such as the potential for impairment of sensory nerve function (Luke, 1991), and a slowing of motor nerve conduction velocity (NCV) (Araki and Honma, 1982). Schwartz et al. (1988) report a negative correlation between motor NCV in children and blood lead levels (down to ≈ 30 - 40$\mu g.dL^{-1}$), although they conclude that maximal motor NCV is an insensitive measure of low-level lead toxicity.

Seppäläinen et al. (1979 & 1983) observe decreased motor NCV in occupationally-exposed individuals exhibiting blood lead levels of between 30 and 48$\mu g.dL^{-1}$. This concurred with the results of Araki et al. (1980), Rosen et al. (1983) and Triebig et al. (1984), yet contrasted the work of Spivey et al. (1980) who found no significant NCV difference between lead workers with lead levels of 60 - 80$\mu g.dL^{-1}$, and non-exposed controls. EPA USA (1986), from an evaluation of the results of 28 studies, notes that NCV effects occur in adults at blood lead levels below 70$\mu g.dL^{-1}$ and possibly as low as 30$\mu g.dL^{-1}$, Ehle's review of the literature (1986), proffering a higher minimum blood lead level of 60$\mu g.dL^{-1}$.

Frank peripheral neuropathy has been observed in children at blood lead levels of 60 - 80$\mu g.dL^{-1}$ (Erenberg et al., 1974). Work by Landrigan et al. (1976) with children living near a lead smelter situated at Kellogg, Idaho, has shown an inverse correlation between peroneal NCV and blood lead levels over a blood lead range of 13 - 97$\mu g.dL^{-1}$, Schwartz et al. (1988) detecting a threshold for NCV impairment at blood lead levels of 20 - 30$\mu g.dL^{-1}$.

c) Neurobehavioural/neuropsychological effects of lead

At blood lead concentrations of around 30$\mu g.dL^{-1}$ or greater, neurobehavioural testing has revealed effects in adults (SRC, 1990). For occupationally-exposed groups demonstrating mean blood lead levels of ≈ 50 - 80$\mu g.dL^{-1}$, disturbances to reaction time, visual motor performance and hand dexterity, as well as changes to IQ test and cognitive performance, memory skills, learning ability, mood, coping ability and general nervousness level have been observed by a number of researchers (eg Grandjean et al., 1978; Haenninen et al., 1978; Valcuikas et al., 1978a & b; Arnvig et al., 1980; Mantere et al., 1982; Hogstedt et al., 1983; Campara et al., 1984; Pocock et al., 1994; Tong et al., 1996). Baker et al. (1979 & 1983) report impaired verbal concept formation, disturbed visual/motor performance, (short-term) memory loss, and increased rates of depression, confusion, anger, fatigue and tension amongst lead workers with blood lead levels ≥ 40$\mu g.dL^{-1}$.

Hyperactivity has been observed in lead-intoxicated children (Byers and Lord, 1943; Wesolowski et al., 1979; Needleman, 1980; Hamilton, 1984), and in (previously) encephalopathic children whose mean blood lead level at the time of episode was 88$\mu g.dL^{-1}$ (Rummo, 1974; Rummo et al., 1979). Schwartz and Otto (1987) analysing the NHANES II data set for 4,519 children aged four to nineteen years, suggest that the likelihood of child hyperactivity increases significantly with increasing blood lead levels (as blood lead percentile rank within age group), although the probability of speech

impairment is not related to blood lead level. These authors conclude that blood lead level (as blood lead percentile rank within age group) is significantly associated with delays in three (child) developmental milestones, namely the age at which the child first sat up, walked and talked.

Hyperactivity in children with low-level lead exposure has also been reported (David et al., 1972; David, 1974; Baloh et al., 1975; de la Burdé and Choate, 1975). David et al.'s findings (1983) clearly demonstrate that certain hyperactive children with elevated lead levels are significantly improved behaviourally after a trial of chelation medication. The DHSS (1980), however, from its review of the health effects of lead, finds no evidence confirming an association between high body lead levels and hyperactivity in children, although these authors do acknowledge definitional differences between countries and health workers with respect to hyperactivity. Silbergeld and Goldberg (1974) report that lead produces an animal model of hyperactivity having potential (human) clinical relevance, and proffer this as an explanation for some cases of hyperactivity in children.

At approximately five years subsequent to acute lead intoxication in a group of children, Smith et al. (1963) detected intellectual impairment independently of whether frank encephalopathy had been demonstrated or not. This theme of the potential for permanent neurotoxic effects associated with childhood lead exposure at, particularly, low levels continued through the late 1950s (Byers, 1959) and into the 1960s (Patterson, 1965; Lin-Fu, 1967). Further, from a review of a number of studies dealing with (apparently) asymptomatic children exhibiting relatively high body lead burdens (as identified through lead screening programs) (eg de la Burde and Choate, 1972 & 1975; Kotok, 1972; Perino and Ernhardt, 1974; Rummo, 1974; Kotok et al., 1977; Rummo et al., 1979; Ernhardt et al., 1981), EPA USA (1986) concludes that, in general, high-risk lead exposure groups perform less well in IQ or other psychometric tests than do referent, lower exposure control groups. EPA USA (1986) further concludes that a consistent pattern of lower IQ ratings and other neuropsychological deficits among the higher lead exposure children reflected cognitive impairment in (apparently) asymptomatic children with blood lead levels of the order of $50\mu g.dL^{-1}$ and above.

As a result of sophisticated research conducted primarily in the 1980s and early 1990s, such lasting impairment would appear to be a more subtle hallmark of fairly low level (apparently asymptomatic) exposure and (infant) body lead loading for a surprisingly sizeable sector of an urban society (Needleman et al., 1982, 1984 & 1990; Needleman, 1986 & 1988; Needleman and Bellinger, 1988; Bellinger et al., 1991a & 1992; Baghurst et al., 1992; Dietrich et al., 1993).

Neurobehavioural impairment including altered behavioural profiles, fine motor dysfunction and IQ deficits of $\approx$ five points in asymptomatic lead-intoxicated children have been associated with blood lead levels of 50 - $70\mu g.dL^{-1}$ (de la Burde and Choate, 1972; Rummo et al., 1979). On the basis of EPA USA (1986) estimates of dentine lead data generated by Needleman et al. (1979 - and subsequently reanalysed by Needleman et al. [1982 & 1985], Bellinger and Needleman [1983] and EPA USA [1986]), four point IQ deficits have been associated with child blood lead levels of 30 - $50\mu g.dL^{-1}$. Such blood lead levels and associated (relatively low) body lead burdens are considered to be "more representative of general paediatric populations" than lead intoxicated children (SRC, 1990; p. 77). Here, elevated dentine lead children demonstrated significantly lower Full-Scale Weschler Intelligence Scale of Children - Revised (WISC-R) scores, significantly poorer performance in tests of

auditory and verbal processing, significantly lower outcomes in tests for attentional performance, and rated more poorly in a teachers' behavioural rating test (as initially noted by Needleman et al., 1979).

Hawk et al. (1986) and Schroeder and Hawk (1987) demonstrated a highly significant inverse linear relationship between IQ and blood lead level for low Socio-Economic Status (SES) African-American children over the range 6 - 46$\mu g.dL^{-1}$, reflecting lead-induced IQ decrement without any evident threshold down to very low blood lead levels. Such work was replicated by Fulton et al. (1987) in children in higher and less uniform SES in Edinburgh Scotland, the data confirming a significant inverse dose-effect relationship between blood lead level and cognitive ability, with no blood lead threshold evident down to a level of 5.6$\mu g.dL^{-1}$.[33] McMichael et al. (1988) also link cumulative environmental lead exposure with early childhood mental retardation with no readily-apparent threshold.

ATSDR (1988b), as Fulton et al. (1987), concludes, from a review of data of such ilk, that at blood lead levels of 25$\mu g.dL^{-1}$, cognitive ability impairment and IQ deficit is likely to occur in the general population. In fact, evidence associating neurobehavioural deficits with blood lead levels of the order of 10 - 15$\mu g.dL^{-1}$ (or possibly even lower) is demonstrated in some studies focusing upon the prenatal exposure to the toxin (SRC, 1990; - Section 4.2.5.4).

Consistently lower IQ values, and noted neuropsychological deficits among higher lead exposure children have encouraged EPA USA (1986) to conclude that cognitive deficits occur primarily in (apparently) asymptomatic children with markedly elevated blood lead levels (in the range 50 - 80$\mu g.dL^{-1}$), such intoxication eliciting an IQ decrement of the order of five points.

Some opposition to these general conclusions regarding lead's influence upon (say) IQ and neurobehavioural scores of exposed children, has been voiced however. These general population studies have been described both as "studies of reasonably adequate or good quality" (SRC, 1990; p. 79), and as failing to demonstrate any statistically significant effects of lead exposure on IQ or other neurobehavioural measures (Harvey et al., 1983 & 1984; Smith et al., 1983; Lansdown et al., 1986; Pocock et al., 1987).

The (seemingly) overwhelming evidence for such an effect of lead upon IQ and other neurobehavioural scores would, however, appear to be convincing (Pocock et al., 1994; Tong et al., 1996; McMichael, 1997), particularly at the higher lead exposure level. If accepted, this represents major implications for public health when considered on a general population basis (Davis and Svendsgaard, 1987; Grant and Davis, 1987; Nriagu, 1988; Rabin, 1989; Landrigan and Curran, 1992; Mushak, 1992).

Further, Rosen remains convinced by the results of "consistent and conclusive studies" that such a deleterious effect of lead, namely "a constellation of neurotoxic and other adverse effects of lead" (1992, p. 1,278) is also visited upon its unsuspecting victims at low(er) exposure levels. From a thorough review of literature-derived investigation results including Needleman et al. (1979), Yule et al. (1981), Schroeder et al. (1985), Hawk et al. (1986), Lansdown et al. (1986), Bellinger et al. (1987a, 1991a & 1992), Fulton et al. (1987), McMichael et al. (1988), Dietrich et al. (1989 & 1993), Lyngbye et al. (1990), Needleman et al. (1990), Winneke et al. (1990) and Baghurst et al. (1992),

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[33] Here, a major source of lead exposure is drinking water reticulated through lead-containing vessels (Fulton et al., 1987).

lead's neurotoxicity would appear to have been detected at blood lead levels as low as $10\mu g.dL^{-1}$, or perhaps even lower as noted by Rosen (1992), with CDC (1991) claiming no apparent threshold for the lead-IQ relationship having been established.

Rosen (1992) reports, from a review of the literature, that there is an average decrease of 2.5 IQ points for each $10.0\mu g.dL^{-1}$ increase in blood lead levels. He further claims that the results of these prospective and cross-sectional studies indicate causality between remarkably low levels of lead exposure and neurobehavioural/cognitive/IQ deficits in young children. This estimate appears to be slightly lower than the figure of a four to seven IQ point deficit for each $10.0\mu g.dL^{-1}$ increase in blood lead levels as proposed by Bellinger et al. (1992) and Dietrich et al. (1993). Based upon the strength and consistency of the evidence cited above, the CDC in 1991, lowered the threshold level defining childhood lead intoxication to $10\mu g.dL^{-1}$ (CDC, 1991).

The concept of reverse causality (ie mental deficit causing enhanced pica which causes elevated lead exposure), as expounded by De Silva and Christophers (1997) as their explanation for the association between elevated blood lead levels and IQ deficit, would appear to attract little support from workers such as McMichael (1997). Here, McMichael directs strong criticism toward the pair for their confusion between the relative merit of cross-sectional vs longitudinal study interpretation, and their failure to acknowledge the Boston cohort study result demonstrating an inverse relationship between cord blood lead levels and early postnatal intellectual functioning (Bellinger et al., 1987a). Further, McMichael (1997) also criticises De Silva and Christophers' work for the authors' unwillingness to extrapolate to humans, results from animal studies which clearly demonstrate that lead exposure impairs problem-solving, attention span, task perseverance and behaviour, as Hastings et. al. (1979), Cory-Schlecta et al. (1981), Rice (1985) and Gilbert and Rice (1987).

Due to the neurological and behavioural dysfunction resulting from lead intoxication of the child, Needleman suggests that that mental impairment places the child at a greater risk of subsequent criminal activity (Sibbison, 1992), other workers remarking similarly (eg Masters, 1997).

d) Developmental CNS toxicity of lead

Neurobehavioural testing of 592 children from a Port Pirie cohort study and associated blood lead concentrations have been reported by several workers (Baghurst et al., 1985 & 1987; Vimpani et al., 1985, 1987 & 1989; Wigg et al., 1988). For these children, mean blood lead levels were $14.3\mu g.dL^{-1}$ at six months, increasing to $20.8\mu g.dL^{-1}$ and $21.2\mu g.dL^{-1}$ at 15 and 24 months respectively. Approximately 20% of the children exhibited blood lead concentrations greater than $30\mu g.dL^{-1}$ at 24 months, at which time Bayley Mental and also Psychomotor Development Indices (MDI and PDI respectively) were conducted. Reduced MDI scores were significantly associated with higher (integrated) postnatal blood lead levels ($P < 0.05$) as with six month blood lead concentrations ($P < 0.01$), suggesting nearly a two point MDI deficit for every $10\mu g.dL^{-1}$ blood lead level increase, and with no threshold effect (Wigg et al., 1988). This was not the case for prenatal, delivery or cord blood levels. From these findings, elevated blood lead levels at six months of age have a greater negative effect upon infant mental development than do elevations at other times antenatally, as also for the first two years postnatally.

Ongoing blood lead and neurobehavioural testing of this group (McCarthy Scales of Children's Abilities) at three and four years of age also took place (McMichael et al., 1988). Children's scores using McCarthy General Cognitive Index (GCI) were significantly and inversely correlated with log blood lead levels at six months, two years and also three years. Scores also correlated inversely with an integrated blood lead average from birth to four years, an estimated integrated average blood lead increase of from 10 - 30$\mu g.dL^{-1}$ eliciting an approximate 7.2 points decrease in GCI score.

Bellinger et al. (1984, 1985a & b, 1986a & b and 1987a & b) in a similar prospective study with 249 Boston children (excluding children of less than 34 weeks gestational age), determined delivery cord blood lead concentrations and categorised these as either low (mean = 1.8$\mu g.dL^{-1}$), medium (mean = 6.5$\mu g.dL^{-1}$) or high (mean = 14.6$\mu g.dL^{-1}$) exposure groups, subsequently gauging blood lead and MDI and PDI levels every six months. Slight but not significant correlation between cord blood lead category and gestation age was observed, although within gestational age categories, no correlation between cord blood lead category and gestation age was apparent (Bellinger et al., 1984). Further, an inverse correlation between cord blood lead levels and MDI scores at age six, 12, 18 and 24 months was noted, the MDI score difference between the low and the high lead groups being 4.8 points (Bellinger et al., 1985, 1986a & b, 1987a). MDI did not correlate with postnatal blood lead level, and PDI did not correlate with either cord or postnatal blood lead levels.

At approximately five years of age, MDI deficits correlated significantly with blood lead levels at two years of age, but not with concurrent or prenatal blood lead levels (Bellinger et al., 1987b). Winneke et al. (1985a & b) demonstrated that maternal blood lead levels at birth, and cord blood lead levels were very highly correlated, and that maternal blood lead levels almost equally accounted for the variance in neurobehavioural test scores at six to seven years of age as did (child) contemporary blood lead levels.

Dietrich et al. (1986 & 1987a) in a prospective study of inner-city Cincinnati-born children, noted that the effect of prenatal lead exposure on birth weight and gestational age partly mediated lead's effects upon MDI score. Here, higher prenatal blood lead levels were associated with reduced birth weight and reduced gestational age, each of which was significantly associated with reduced MDI scores (Dietrich et al., 1987b). Postnatal growth rates between three and fifteen months from this Cincinnati group were inversely correlated with postnatal increases in blood lead levels over this same time period (Shukla et al., 1987).

From this work, Dietrich (1987b) reported a dose-related deficit in MDI score of 0.84 MDI points for each single $\mu g.dL^{-1}$ increase in prenatal blood lead for male infants at six months of age. This contrasts Baghurst et al. (1985 & 1987) and Vimpani et al. (1985 & 1987) who observe that MDI score deficits are more closely related to postnatal (rather than prenatal) exposure. These authors ascribe an average two point MDI score drop (at 24 months) to a mean blood lead concentration increase of 10$\mu g.dL^{-1}$ (at six months of age). This postnatal observation of the last-mentioned two groups is claimed to arise as a result of a significant increase in exposure between six and fifteen months of age, this increase overwhelming the more subtle effects of lower prenatal lead exposure (EPA USA, 1986; Davis and Svendgaard, 1987).[34] McMichael et al. (1988), following this cohort

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[34] This period of the child's development is marked by crawling and walking and suggested peaks in hand-to-mouth activity and pica (Gibson, 1904; Snowdon, 1977).

through four years of age, note an inverse correlation between (McCarthy) GCI scores and integrated postnatal blood lead levels, reporting that a blood lead concentration increase from 10 to 30$\mu g.dL^{-1}$ corresponds to a 7.2 point GCI score drop.

From a number of studies including the above-mentioned, there would appear clear evidence of the deleterious effect on neurobehavioural development of exposure to lead during early development, especially prenatally. Such resultant neurobehavioural deficit appears associated with prenatal (internal) exposure levels (maternal or cord) of as little as 10 - 15$\mu g.dL^{-1}$ or possibly even lower (EPA USA, 1986; Davis and Svendsgaard, 1987; Grant and Davis, 1987; ATSDR, 1988; SRC, 1990). In review of several prospective (longitudinal) epidemiological studies, Davis and Svendsgaard (1987) conclude the magnitude of these neurobehavioural performance deficits ranges from two to eight (Bayley) MDI points for every 10$\mu g.dL^{-1}$ increment in blood lead level. These authors further conclude that such exposure to lead (even at levels as low as 10 - 15$\mu g.dL^{-1}$ and possibly lower), is also linked with reduced gestational age and lowered birth weight. Schwartz et al. (1985) claim, for the year 1980, nearly one million children may have been born in the USA alone, with blood lead levels sufficiently high to put them at risk of development perturbations, although these authors do acknowledge that that figure has been reducing with time due to reducing ambient exposure levels.

4.2.5.5 CARDIOVASCULAR TOXICITY OF LEAD

Lead has been known to demonstrate cardiotoxic properties for more than a century (Nriagu, 1988), contributing to cardiomyopathy, and affecting cardiovascular metabolism by acting upon the renin-angiotensin, killikrein-kinin, adrenergic and prostaglandin systems (Carmignani et al., 1983). The metal has been reported to be responsible for eliciting increases in blood pressure via renal pathways in (particularly) middle-aged men having experienced some fairly low level extended exposure, no blood lead threshold having been inferred (Pocock et al., 1984 & 1985; Harlan et al., 1985; ATSDR, 1988b). The contribution to this group's hypertension by lead alone however would appear to be small by comparison with other contributing factors (EPA USA, 1986).

Chronic lead poisoning of occupational origin in adult males ≥ 46 years of age has been associated with cardiac lesions and electrocardiographic abnormalities (Kosmider and Petelenz, 1962). Kirkby and Gyntelberg (1985) confirm a significantly greater incidence of ischaemic electrocardiographic changes in lead workers than controls, with an associated slight increase in diastolic (but not systolic) blood pressure. Contrasting this, Pocock et al. (1984), EPA USA (1986), Weiss et al. (1986) and de Kort et al. (1987) all report a positive correlation between systolic pressure and blood lead level, Weiss et al. (1986), however, proffering no relationship between blood lead level and diastolic pressure. In five fatal cases of lead poisoning in young children, Kline (1960) notes that degenerative changes in cardiac muscle have been reported as the proximate cause of death. Voors et al. (1982), from a study of 75 (adult) autopsies, found a positive correlation, after adjustment for age, between aortic lead level and death from heart-related disease. In separate studies of long term employees (> 25 years) of typesetting or newspaper-printing industries experiencing low-level lead contact, Rossol (1992) reports a significant elevation in frequency of stroke as a cause of death for workers.

4.2.5.6 GASTROINTESTINAL TOXICITY

Typically, gastrointestinal symptoms occur at high to very high blood lead levels associated with encephalopathy (> 60 - 80$\mu g.dL^{-1}$; EPA USA, 1986). Such symptoms have, however, also occasionally been noted in occupationally-exposed individuals at blood lead levels as low as 40 - 60$\mu g.dL^{-1}$ (Baker et al., 1979; Haenninen et al., 1979). Colic has consistently been evidenced as an early symptom of this group's lead intoxication (SRC, 1990). For children, colic is a common symptom of lead intoxication, IPCC (1991) reporting a lowest-observed-effect level of the order of 60 - 100$\mu g.dL^{-1}$. Such colic, however, is also typically associated with other signs of acute lead poisoning such as severe constipation (although diarrhoea is also sometimes reported [Davies, 1988]), epigastric pain, anorexia, nausea and/or intermittent vomiting (NAS, 1972 - based upon the data of Chisholm and Harrison [1956] and Chisholm [1962 & 1965]; Hernberg [1980]; ATSDR, [1988b]). Such gastrointestinal symptoms have long been recognised as being characteristic of lead poisoning. Scant attention has been paid to defining appropriate dose-effect relationships at ambient exposure levels, greater concern being directed to neurotoxic effects (SRC, 1990). Typically, gastrointestinal symptoms are displayed by occupationally-exposed individuals, (primary exposure being via inhalation), but also by children whose primary exposure to lead is orally (SRC, 1990).

4.2.5.7 IMMUNE SYSTEM EFFECTS

Little work appears to have been conducted in this realm, most of the data being collected from (adult) lead workers who acquire the lead via inhalation. Occupationally-exposed individuals exhibiting blood lead levels in the range 22 - 89$\mu g.dL^{-1}$ (median 55$\mu g.dL^{-1}$) demonstrate more colds and influenza infections annually with significant suppression of secretory IgA levels, although no significant alteration to serum immunoglobulin levels appears evident (Ewers et al., 1982). These data are contrasted by results of Kimber et al. (1986), who also examined immune function in lead workers occupationally exposed for between four and thirty years. This test group's blood lead level at the time of testing ranged from 25 - 53$\mu g.dL^{-1}$ (mean = 38$\mu g.dL^{-1}$). The researchers' data suggest no changes to serum concentrations of IgG, IgA or IgM with elevated blood lead levels, albeit acknowledging no significant alteration to serum immunoglobulin levels. No indication of immune system effects has been observed in children exhibiting blood lead levels $\geq$ 40$\mu g.dL^{-1}$ (Reigart and Graber, 1976). Gainer (1974) working on the mouse model detected lead's role in aggravating viral disease through Interferon induction repression in different cell types. Lutz et al. (1992) note, from animal models, that lead is an immunosuppressive agent at levels far below those required for overt toxicity, and from preliminary human work they find no age-related relationship for the observed influence of lead on human immunosuppressive function.

In a related sense, anecdotal evidence suggests an association between elevated frequency of SIDS (Sudden Infant Death Syndrome) and child skeletal lead level (Drasch and Ott, 1988) and also foetal lead exposure (Boeckx, 1986).

4.2.5.8 GENOTOXIC EFFECTS

Acknowledging contradictory *in vivo* test results, SRC (1990) concludes that lead does have some effect on human chromosomes. Babich et al. (1985) report effects including gene mutations, chromosomal aberrations and DNA lesions. Sirover and Loeb (1976) and Jenette (1981) observe an increased frequency of sister chromatid exchange, impairment of spindle formation and alterations to the fidelity of DNA synthesis, replication and repair. Results from the work of Grandjean et al. (1983) suggest increased sister chromatid exchange frequency for lead-exposed workers, and also an increased frequency with increasing duration of occupational exposure independent of blood lead level. No increase in exchange frequency was noted by Dalpra et al. (1983) for environmentally-exposed children exhibiting blood lead concentrations in the range 30 to 63$\mu g.dL^{-1}$.

Other data reveal that lead exposure occupationally or by voluntary ingestion eliciting blood lead concentrations in the range 20 - 90$\mu g.dL^{-1}$, is associated with increased frequency of abnormal mitosis (Forni et al., 1976 & 1980), increased mitotic activity for peripheral lymphocytes (Bulsma and DeFrance, 1976), and an increased incidence of chromosomal aberrations (Forni et al., 1976 & 1980; Nordenson et al., 1978; Al-Hakkak et al., 1986). Other workers, however, have found little evidence of chromosomal aberration (Schmid et al., 1972; Maki-Paakkanen et al., 1981). From cultured human lymphocyte studies, Grandjean and Andersen (1982) and Niebuhr and Wulf (1984) observe an organo-lead-associated increased frequency of sister chromatid exchange rate with altered chromosome length.

From a review of the literature and acknowledging conflicting results, EPA USA (1988) concludes that there is evidence to suggest a clastogenic effect of lead, further noting that calcium deficiency may enhance the genotoxicity of lead as it does other manifestations of lead's toxicity.

4.2.5.9 CARCINOGENIC EFFECT OF LEAD

Lead, or compounds of lead, have been reported to display some characteristics of low-potency tumour initiators (Jenette, 1981) and promoters (Kazantzis, 1989) in animals, although these effects have been demonstrated at high exposure levels (Babich et al., 1985). For these studies, however, the potential for a contributory role by other chemicals present exists (Blumer and Reich, 1980). Oral and parenteral treatment of rats with soluble lead compounds reflect a significantly increased frequency of renal carcinoma (SRC, 1990).

A weak association between occupationally lead-exposed individuals with elevated risk of carcinogenesis of, particularly, respiratory and digestive tracts and to a lesser extent the urinary tract has been noted (Cooper, 1976; Kang et al., 1980; Neuberger and Hollowell, 1982; Cooper et al., 1985; Gerhardsson et al., 1985; Selevan et al., 1985; EPA USA, 1986), although poor experimental control for exposure to other chemicals (eg arsenic), and confounding by smoking (Cooper, 1976; IARC, 1987; EPA USA, 1988) detracts from the integrity of the results (SRC, 1990). Neuberger et al. (1990) report, regarding long-term residents of Galena Kansas, a lead and cadmium mining

Superfund site reflecting substantially elevated levels of lead in mine waste, soil, dust and water, that residents experienced a significantly enhanced rate of skin cancer, associating this with contact with the heavy metals.

McMichael and Johnson (1982) and EPA USA (1988 & 1989a) conclude that current human data are inadequate to refute or demonstrate the potential carcinogenicity of lead exposure for humans. IARC (1987) concurs, and classifies lead as a possible human carcinogen. Magos (1991) claims that no evidence for an association between exposure and neoplasm exists, although acknowledging for those studies where some evidence of an association had been detected (eg Kazantzis, 1989), that confounding factors had not been controlled for, and could explain any perceived relationship.

4.2.5.10 TERATOGENIC/GAMETOTOXIC EFFECTS OF LEAD

Whilst toxic and teratogenic effects upon the conceptus have been reported for animal studies, no teratogenic abnormalities resulting from lead contact in utero have been reported for humans (EPA USA, 1989a). Fischbein proffers lead as a "suspected" teratogen (1983; p. 443), but cites no significant evidence. Needleman et al. claim that lead has been considered a "potential" human teratogen since the early twentieth century (1984; p. 2,956). Heavy metals such as lead have been reported to demonstrate a gametotoxic effect (particularly to sperm), simply by absorption through the skin, (Mortimer, 1994).

4.2.5.11 SUMMARY

Young children, from neonate to around three years of age, experiencing maximum mouthing behaviour (Quah et al., 1982; SRC, 1990) are at the greatest risk (medically) regarding the effects of lead intoxication (Silbergeld, 1990; IPCS, 1991). Essentially, behavioural and physiological characteristics of children of that age range are associated with an enhanced intake and retention of environmental lead at a time when they are most susceptible to the (irreversible) damage caused by the toxin (as Ernhardt et al., 1988). Adults may also suffer harmful effects as a result of contact with the contaminant at modest levels (Wedeen, 1984; Goyer, 1986; Silbergeld, 1990). Children, however, due to their particular body biochemistry's greater propensity to absorb and retain ingested lead, suffer more severe and sinister manifestations of toxicity at lower blood lead levels (Farfel, 1985). The likely pathway of lead to children is via house dust. There exists the propensity for infants to ingest disproportionately greater amounts of dust than older individuals, particularly at the time that they are most likely to contact lead and least able to effectively deal with it. Clearly, this group of the population is the sector where effective prophylactic action most needs to be applied.

Children particularly have been used as "bio-indicators" (Reagan and Silbergeld, 1989; p. 200) of lead exposure for centuries, despite the preventability of exposure, and the irreversible effects of this toxin. Despite some researcher differences regarding the presumed dimension of the risk of lead to the health of individuals accorded by a certain exposure, erring on the side of caution would appear highly desirable for the benefit of the children. It would appear clear to many workers (eg Mushak

and Crocetti, 1990), that, in order to achieve a reduced level of intoxication of children in particular, the only solution would be to elicit a reduction or removal of their manifold sources of environmental lead exposure. In this context, the following chapters deal with a detailed scientific study of some of the potential sources of exposure to lead-bearing dust within the house, its spatial distribution therein, and its mechanisms of entry.

CHAPTER FIVE PRELIMINARY INVESTIGATIONS

Industrialism is the systematic exploitation of wasting assets . . . progress is merely an acceleration in the rate of that exploitation. Such prosperity as we have known up to the present is the consequence of rapidly spending the planet's irreplaceable capital.

Aldous Huxley

Magic begins in superstition and ends in science.

Will Durant

5.1 INTRODUCTION

Given that much of the material of research interest was to be (vacuum-acquired) surface dust samples from houses in Port Pirie, some preliminary analytical work with dust samples deriving from a number of both occupied and vacant houses was undertaken.

Some pre-existing home dust lead data based upon variable sampling techniques (eg vacuum cleaner-collected and brush-collected) and also analytical techniques for home dust lead concentrations is available for Port Pirie (eg Body, 1986d). It was felt, however, that some preliminary vacuum cleaner bag dust grab sampling was necessary, with the intention of indicating in broad terms the degree of home contamination (by leaded dust) and its variability. Further, given the potential for change that may have occurred in Port Pirie with the passage of time and the (possible) effects of nearly ten years of a very active lead-intervention program (Maynard et al., 1993), the acquisition of some more current domestic floor dust/lead data was undertaken.[1] In this respect, an initial sample of 32 occupied (available) homes from the local environs was selected for dust sampling, some of which would be used for particle-sizing work (Section 5.5) and some for supplying internal dust standards for dust analysis (Appendix Three).[2]

All occupied-home samples derived from residences where occupiers were provided with a clean new bag to suit their type of domestic vacuum cleaner, and asked to use such bags for cleaning only internal floor surfaces of their homes according to their normal cleaning practices.[3] When full, bags containing

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[1] Changes here refer to both changed operations of the smelter (A. Gilbert, pers. comm.), around town dust mobility-reducing activities eg as a result of the Greening Program (G. Norton, pers. comm.), and also domestic dust-stabilising activities and better house-cleaning practices as a result of the enhanced local resident awareness of the role of mobile dust in the lead intoxication of residents of Port Pirie (C. Phipps, pers. comm.).

[2] It is acknowledged that such a group is not randomly selected. House owners selected derived from staff and clients of the Environmental Health Centre (EHC) and also general members of the public who were willing to participate. Some culling of potential participants was necessary in order to provide a reasonable geographical spread of sampling locations around Port Pirie.

[3] In all situations, it was requested that the vacuum cleaner would be used for the purposes of cleaning floors only, although from discussions with a number of EHC case-workers and also residents of Port Pirie, given the very dry and dusty nature of the local environment, it is not at all uncommon for residents of the township to use their vacuum cleaners for a whole host of different cleaning purposes including house exteriors, automobile interiors, ashtrays, clothes and even domestic pets (var. pers. comm.).

between 230 and 460gms of material (Mean = 377gms) were collected for analysis. Apart from a preliminary sieving through a mesh size of one millimetre in order to remove large balls of carpet fibre and other (typically) non-dust items, no sample preparation occurred. Samples were transferred to white 500ml sealable screw-top plastic containers (Polarcup, Bankstown NSW Australia), were thoroughly mixed, and a sample analysed for lead content. This sample derived from a composite of ten approximately equal sub-samples taken from the body of the original bulk dust with a stainless steel spatula whilst mixing of the bulk dust took place. Results of the lead analyses of these test dust samples appear in section 5.3.1 below.

5.2 DUST LEAD CONTENT - ESTABLISHING THE METHODOLOGY

Prior to the determination of lead concentration of these dust samples, however, the technique to be routinely used for analysis had to be determined/validated. To this end, the influence of various acid digest or solubilising mixes and test quantities of dust regarding measured lead in dust concentration was investigated. Acid digestion of dust-contained lead was performed to solubilise the material. This process, using selected strong oxidising acids, serves a second function of oxidising any organic material, hence freeing the lead for dissolution and removing probable analytical interferences (Harrison and Laxen, 1977; Oxley, 1982). Subsequent to this concentrated acid digestion of dust using an appropriate acid digest (as established below), dust-lead concentration assessment was performed using AAS techniques. Conditions of operation of the AAS, calibration, standard reference solution preparation, specific quality assurance measures adopted, accuracy and precision results are as indicated in Appendix Seven. QA/QC details are referred to in Appendix Three.

A description of this preliminary digest investigation is contained in Appendix Two. From this work it was decided to adopt the digest mix two parts concentrated Nitric acid (≈ 15 Molar [M]) to three parts concentrated Hydrochloric acid (≈ 13M) (added consecutively) as the standard acid digest mix for all digests constituting this work, and to routinely use an aliquot of 100mg of dust for all subsequent (bulk sample) dust lead analysis experiments.[4]

5.3 INITIAL HOME VACUUM CLEANER BAG GRAB SAMPLES

As an initial indicator of home floor (dust) contaminant levels, individual dust samples were collected from 27 occupied houses within the township of Port Pirie and ten from outside. Results of analysis of such dust for lead appear in Sections 5.3.1 and 5.3.2 below.

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[4] As will be evidenced in Chapter Seven, the mean quantity of dust collected from test house floors with the vacuum pump method was 131mg (n = 528).

5.3.1 SAMPLES ACQUIRED FROM PORT PIRIE RESIDENCES

Many researchers have noted that the concentration of lead in dust varies inversely with the particle size (eg Randall et al., 1975; Archer and Barratt, 1976b; Barratt, 1976a; Dorn et al., 1976; Duggan and Williams, 1977; Spitler and Feder, 1979; Biggins and Harrison, 1980; Fergusson et al., 1980; Hopke et al., 1980; Linton et al., 1980; Johnson et al., 1982; Fergusson and Ryan, 1984; Duggan and Inskip, 1985; Davies and Watt, 1986; Steele et al., 1990; Fergusson and Kim, 1991; Moseholm et al., 1992). Given the potential range in dust sample particle sizes present in vacuum cleaner bag dust grab samples (Fergusson and Ryan, 1984), it is thus considered that analysis of such bulk test dust for lead concentration in the same fashion as for (size) sieved dust would be likely to return results with greater variability. This is confirmed by the results of the occupied home bulk floor dust sample analyses. Such mean values reflect notably greater Relative Standard Deviation (RSD) values than those for sieved dust samples (eg Appendices Two and Seven), as also for experiments determining the lead content of the various size fractions of sieved dust (Section 5.4.2). A list of the mean results of triple replicate lead analyses of these Port Pirie bulk floor test dust samples appears in Table 5.1 below.

Table 5.1 Preliminary indication of lead levels of house dust from Port Pirie.

	VACUUM CLEANER-DERIVED DUST SAMPLES		
DUST NUMBER	**DUST LEAD CONC.** ($\mu g.g^{-1}$)	**SD**	**RSD (n = 3)** (%)
1	500	44.9	9.0
2	520	65.6	12.6
3	610	40.9	6.7
4	615	99.9	16.2
5	640	40.9	6.4
6	800	45.8	5.7
7	835	51.9	6.2
8	900	49.3	5.5
9	980	260.1	26.5
10	980	51.6	5.3
11	1,080	44.0	4.1
12	1,080	51.5	4.8
13	1,120	137.3	12.3
14	1,130	48.3	4.3
15	1,250	39.1	3.1
16	1,280	185.2	14.5
17	1,390	79.4	5.7
18	1,400	145.3	10.4
19	1,440	56.7	3.9
20	1,920	215.5	11.2
21	1,945	205.9	10.6
22	2,210	183.1	8.3
23	2,405	344.6	14.3
24	3,550	68.7	1.9
25	4,305	233.8	5.4
26	4,530	493.5	10.9
27	4,795	323.3	6.7
MEAN	**1,637**	**133.6**	**8.6**

5.3.2 OUT-OF-TOWN SAMPLES

Dust samples were similarly acquired from surrounding townships, and also analysed for lead in triplicate (Table 5.2). Mean distance from Port Pirie CBD to sampling sites here is 13.4kms.

Table 5.2 Preliminary indication of lead levels of vacuum cleaner-derived house dust collected from outside the township of Port Pirie.

	VACUUM CLEANER-DERIVED DUST SAMPLES		
DUST NUMBER	**DUST LEAD CONC.** ($\mu g.g^{-1}$)	**SD**	**RSD (n = 3)** (%)
28	106	23.2	21.9
29	130	20.0	15.4
30	138	18.6	13.5
31	152	41.1	27.0
32	190	50.0	26.3
33	231	23.5	10.2
34	256	64.9	25.4
35	260	18.3	7.0
36	330	120.0	36.4
37	420	98.1	23.3
MEAN	**221**	**47.8**	**20.6**

Whilst sample sizes were not large, it appeared from the frequency histogram and box and whisker plots for the Port Pirie-derived data that the data were slightly positively skewed, although testing revealed no notable variation from normality ($P \geq 0.05$). Acknowledging moderate sample size for out of town samples, difference existed between the two data sets for both raw and log-transformed modes ($P < 0.01$). Thus the source of the contaminant lead in house dust would appear to derive from a source in or near to Port Pirie.

5.4 BULK DUST PARTICLE SIZE RANGE AND LEAD CONTENT

As the literature indicates, house dust reflects a very heterogeneous nature (Hunt et al., 1992), demonstrating substantial variability in dust lead concentration with particle size (cited above). Thus in order to make meaningful comparisons between different house dust samples, variation in lead content of house dust would need to incorporate an evaluation of the variability of collected-dust particle size (or size range).

For this purpose, seven separate bulk floor dust samples from different (occupied) available homes located within the two high risk areas of Port Pirie were acquired and subjected to further analysis.[5] These samples were acquired as composite floor samples from all rooms of these houses, using Nilfisk vacuum cleaners with HEPA-standard collectors. Such bulk Nilfisk-derived dust samples were also similarly acquired from the suite of vacant test houses that form the basis of most of the research work included below. These samples were not requested from the respective occupiers of these test homes before premises were vacated. Rather, they were acquired subsequent to the completion of all studies in the vacant houses prior to any cleaning of these houses, again as composite floor samples from all rooms of test houses. Such sampling incorporated a mean house vacancy period of approximately 23 months (range = 15 - 27m).

5.4.1 DUST PARTICLE SIZE RANGE - SIEVING TECHNIQUE

These dust samples were individually separated on the basis of particle size using a sieve shaker (Fritsch Analysette 3E, LABTECHNICS Aust., Kilkenny SA) and a stack of seven tight-fitting stainless steel (200mm) sieves (Endecott Brand - Mining Supplies Australia, Edwardstown SA) according to the procedure described in Appendix Eight. The mesh sizes utilised were as follows:

APERTURE SIZE (μ)	500	250	150	75	45	32

Gravimetric results of such particle size separation are as indicated in Table 5.3. Similarly, for eleven composite floor dust samples, one acquired from each house in the suite of vacant test houses, sieving revealed the following particle size mass distribution (Table 5.4).

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[5] Here also, it is acknowledged that these houses were not randomly selected. They derived from a list of available houses, and were selected to "geographically" reasonably fairly represent houses located within the two test districts.

Table 5.3 Dust sieving results for occupied home floor dust samples.

	PROPORTION (MASS %) OF TOTAL DUST SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
DUST							
A	11.2	29.2	22.5	17.3	9.2	10.7	0.1
B	11.4	22.6	20.3	18.0	10.4	16.9	0.1
C	9.1	24.9	19.6	14.9	11.8	19.0	0.4
D	8.6	19.7	18.1	15.9	11.1	23.3	3.4
E	21.2	33.5	16.6	9.0	11.1	7.7	0.2
F	11.6	19.3	13.7	13.0	14.1	25.6	2.9
G	9.6	9.4	9.7	11.7	14.7	24.7	20.1

Table 5.4 Dust sieving results for vacant home floor dust samples.

	PROPORTION (MASS %) OF TOTAL DUST SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
HOUSE DUST							
6 Agnes St	27.8	5.7	3.0	3.6	7.1	34.9	18.0
64 Edward St	10.6	3.9	3.6	8.0	18.5	49.7	5.4
58 Federatn. Rd	18.2	11.2	5.9	4.9	12.9	7.3	38.9
29 Fourth St	8.6	4.2	3.4	5.8	17.0	25.1	35.6
19 Lily St	12.9	3.9	1.9	3.0	8.8	40.0	29.2
3 Sabine St	16.4	3.1	1.6	3.2	10.3	21.2	43.8
35 Square St	16.7	17.4	30.4	16.2	8.9	1.3	9.0
34 The Terrace	38.4	29.2	6.9	4.9	5.5	1.2	13.8
27 Vera St	23.6	7.4	4.9	5.3	10.9	28.2	19.4
136 Warner. Rd	13.8	3.2	1.8	5.1	11.9	20.8	43.0
97 York Rd	21.7	8.2	3.9	3.1	9.7	17.4	35.5

For comparative purposes, similar sampling of above-ceiling space dust as a composite of the entire above-ceiling space (horizontal) surface area in each of the test houses (apart from the house located at 97 York Rd, where no sample was available) was also undertaken, the following particle size mass distribution resulting (Table 5.5).

Table 5.5 Dust sieving particle size mass results for vacant home ceiling upper surface dust samples.

	PROPORTION (MASS %) OF TOTAL DUST SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
HOUSE DUST							
6 Agnes St	19.9	11.4	7.9	11.7	16.8	3.9	28.5
64 Edward St	2.0	1.9	2.5	8.2	19.8	51.9	13.3
58 Federatn. Rd	0.8	3.1	4.6	11.6	19.5	15.3	44.9
29 Fourth St	1.4	1.8	2.4	7.6	26.1	12.8	47.6
19 Lily St	15.0	5.6	4.4	7.8	17.5	40.2	9.2
3 Sabine St	0.3	0.5	1.8	15.3	51.3	14.0	16.7
35 Square St	5.1	4.8	4.4	7.0	14.7	40.6	23.0
34 The Terrace	27.8	17.9	8.7	9.9	10.4	8.3	16.5
27 Vera St	22.4	16.7	11.8	14.8	12.0	8.6	13.4
136 Warner. Rd	17.6	11.7	9.4	15.8	17.6	3.4	24.5
97 York Rd	-	-	-	-	-	-	-

A summary of these data appears in Table 5.6 below.

Table 5.6 Summary of dust sieving particle size results for all dust samples.

	MEAN PROPORTION (MASS %) OF TOTAL DUST SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
DUST							
Occupied Home - Floors	11.8	22.6	17.2	14.2	11.8	18.3	3.9
Vacant Home - Floors	19.0	8.8	6.1	5.7	11.1	22.5	26.5
Vacant Home - Ceilings	11.2	7.5	5.8	11.0	20.6	19.9	23.8

Strong similarity exists between the two sample types derived from the suite of vacant houses particularly in the smaller particle size range, ie that dust which is most likely to attach to a child's hands; both demonstrating between 60 and 64% of total dust collected lying in the < 75μ particle size range.[6]

There exists (slightly) greater representation in the larger particle size class for occupied-home floor dust than for vacant houses. This suggests the possible contribution of larger-sized dust particles by tracking or other human domestic activity. Differences between above-ceiling and floor dust may reflect the differences in both source and cleaning regime, (tracking in ceiling is assumed to be a negligible contribution, with cleaning not being practised there).

Above-ceiling space dust is assumed to be constantly collecting by aerial deposition, and reflects a smaller mean particle size. Such a similar (vacant home) floor dust profile suggests that continuing collection is also more the case for these dust samples than for occupied home floor dust samples. A smaller proportion of the sub-75μ particle size dust for occupied house floor dust (34%) particles suggests

[6] Coarse particles - a few hundred microns in diameter are easily brushed off hands and clothing whereas smaller-sized particles adhere more readily and are more likely to be ingested (Duggan and Inskip, 1985). Work by these authors has indicated that only particles below 100μ are likely to be significantly retained on the hands. Chaney (1988), as Steele (1990), also notes that particles of a maximum size of around 100μ are likely to be hand-adherent. Duggan et al. (1985) claim, for a child, that the mean hand-adherent particle diameter is around 4.5μ with 90% of particles being < 10μ, and for adults, ≈ 2μ, and 98% respectively, with a maximum particle size of around 180μ. Que Hee et al. (1985), on the other hand, report that particles of size < 246μ adhered equally well to hands, although theirs appears not to be an exhaustive study. Thus sampling dust of only a particle size range below (at most) 200μ is appropriate for contaminant studies within the home environment. A number of workers have demonstrated a hand-adherent lead range of 10 - 30μg lead per pair of hands (Sayre et al., 1974; Charney et al., 1980; Brunekreef et al., 1983; Gallacher et al., 1984), however Roels et al. (1980) in their study of children attending a school located one kilometre from a lead smelter demonstrated levels approaching 700μg of (hand) lead per child. This infers either differing hand-adherent particle size spectra observations, or simply, different dust contaminant profiles.

the influence of human activity in the home, possibly involving normal cleaning regimes. It may also be possible that human activity within the house may result in the on-going resuspension of such dust, as also their deposition upon non-floor surfaces, Murphy and Yocom (1986) claiming that such resuspension is comparable in dimension to outdoor-sourced dust penetration and deposition.

5.4.2 DUST PARTICLE SIZE RANGE LEAD CONTENT

Dust lead analysis of these seven different particle size ranges for each of the 28 test dust samples was subsequently conducted. Mean results of these analyses are as indicated in Tables 5.7 - 5.10.

Table 5.7 Occupied house sieved floor dust particle size versus lead concentration distribution.

	LEAD CONCENTRATION ($\mu g.g^{-1}$) AND PROPORTION (%) OF TOTAL FLOOR DUST LEAD SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
DUST							
A	80 (2.1)	86 (5.9)	152 (8.1)	270 (11.1)	542 (11.8)	2,360 (59.9)	4,275 (1.2)
B	118 (2.1)	151 (5.4)	172 (5.5)	290 (8.2)	690 (11.2)	2,533 (67.0)	3,014 (0.6)
C	1,150 (9.3)	546 (12.1)	607 (10.6)	940 (12.5)	1,066 (11.2)	2,558 (43.2)	3,428 (1.2)
D	280 (2.4)	590 (11.7)	1,098 (20.0)	920 (14.8)	998 (11.1)	1,198 (28.2)	3,445 (11.8)
E	250 (7.5)	475 (22.5)	750 (17.6)	912 (11.7)	1,055 (16.6)	2,148 (23.5)	2,133 (0.6)
F	875 (8.4)	758 (12.1)	657 (7.4)	1,035 (11.1)	950 (11.0)	2,060 (43.5)	2,787 (6.6)
G	7,367 (6.7)	9,558 (8.4)	8,045 (7.3)	7,225 (7.9)	6,905 (9.6)	14,545 (33.7)	14,000 (26.4)

Table 5.8 Vacant house sieved floor dust particle size versus lead concentration distribution.

	LEAD CONCENTRATION ($\mu g.g^{-1}$) AND PROPORTION (%) OF TOTAL FLOOR DUST LEAD SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
HOUSE DUST							
6 Agnes St	787 (7.0)	925 (1.7)	846 (0.8)	1,323 (1.5)	2,151 (4.9)	4,557 (51.1)	5,714 (33.0)
64 Edward St	4,654 (5.4)	4,332 (1.8)	3,051 (1.2)	2,327 (2.0)	2,747 (5.6)	13,414 (72.9)	19,143 (11.2)
58 Federatn. Rd	2,777 (6.8)	5,700 (8.7)	9,718 (7.8)	8,677 (5.7)	6,432 (11.3)	6,366 (6.3)	10,113 (53.4)
29 Fourth St	2,529 (5.8)	4,188 (4.7)	4,656 (4.2)	2,723 (4.2)	2,472 (11.2)	3,795 (25.4)	4,673 (44.5)
19 Lily St	2,726 (5.4)	3,115 (1.9)	4,390 (1.3)	4,923 (2.3)	3,746 (5.1)	7,209 (44.7)	8,664 (39.3)
3 Sabine St	5,818 (12.0)	7,689 (3.0)	9,479 (1.9)	5,131 (2.1)	3,408 (4.4)	6,479 (17.4)	10,722 (59.4)
35 Square St	1,523 (16.1)	1,720 (19.1)	527 (10.2)	629 (6.5)	1,540 (8.7)	2,683 (2.2)	6,504 (37.3)
34 The Terrace	469 (10.9)	377 (6.7)	1,052 (4.4)	1,339 (4.0)	2,000 (6.7)	3,558 (2.5)	7,738 (64.8)
27 Vera St	2,409 (10.5)	3,183 (4.4)	3,789 (3.5)	4,296 (4.2)	3,575 (7.3)	7,207 (37.8)	8,956 (32.3)
136 Warner. Rd	8,205 (14.2)	2,420 (1.0)	3,912 (0.9)	2,940 (1.9)	2,719 (4.1)	7,706 (20.3)	10,579 (57.6)
97 York Rd	8,915 (33.3)	9,213 (13.1)	11,829 (8.0)	5,142 (2.8)	2,726 (4.6)	3,474 (10.5)	4,543 (27.9)

Table 5.9 Vacant house sieved ceiling dust particle size versus lead concentration distribution.

	LEAD CONCENTRATION ($\mu g.g^{-1}$) AND PROPORTION (%) OF TOTAL ABOVE-CEILING SPACE DUST LEAD SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
HOUSE DUST							
6 Agnes St	1,670 (4.3)	3,495 (5.2)	5,311 (5.4)	5,749 (8.7)	7,606 (16.6)	11,069 (5.5)	14,686 (54.3)
64 Edward St	1,943 (0.6)	4,614 (1.3)	3,587 (1.3)	3,062 (3.7)	3,629 (10.5)	8,504 (64.5)	9,376 (18.3)
58 Federatn. Rd	3,565 (0.3)	4,446 (1.2)	5,366 (2.1)	5,430 (5.3)	7,069 (11.6)	11,454 (14.7)	17,270 (65.0)
29 Fourth St	1,420 (0.6)	2,590 (1.3)	3,084 (2.2)	3,005 (6.8)	2,832 (21.8)	3,493 (13.1)	3,873 (54.2)
19 Lily St	1,569 (3.2)	4,049 (3.1)	8,611 (5.2)	9,486 (10.2)	9,604 (23.0)	8,137 (44.8)	8,377 (10.6)
3 Sabine St	1,354 (0.2)	3,145 (0.7)	2,283 (1.2)	1,676 (11.8)	1,939 (45.7)	2,576 (16.5)	3,008 (23.2)
35 Square St	2,965 (5.4)	3,983 (6.9)	3,531 (5.5)	3,141 (7.8)	2,365 (12.4)	2,694 (39.1)	2,789 (22.9)
34 The Terrace	1,025 (7.0)	1,873 (8.3)	3,566 (7.7)	4,918 (12.0)	6,477 (16.7)	7,636 (15.6)	8,044 (32.8)
27 Vera St	2,186 (5.9)	4,401 (8.8)	5,912 (8.4)	6,840 (12.2)	9,989 (14.4)	16,528 (17.0)	20,668 (33.4)
136 Warner. Rd	1,267 (5.0)	1,883 (5.0)	3,012 (6.4)	2,537 (9.1)	3,289 (13.1)	5,602 (4.3)	10,265 (57.1)
97 York Rd	-	-	-	-	-	-	-

Table 5.10 Summary of sieved dust class mean lead concentration results for all dust samples.

	MEAN DUST LEAD CONCENTRATION ($\mu g.g^{-1}$) AND MEAN PROPORTION (%) OF TOTAL DUST LEAD SIEVED INTO VARIOUS PARTICLE SIZE RANGES						
SIZE (μ)	**> 500**	**> 250 - 500**	**> 150-250**	**> 75-150**	**> 45-75**	**> 32-45**	**≤ 32**
DUST							
Occupied Home - Floors	1,446 (5.5)	1,738 (11.2)	1,640 (10.9)	1,656 (11.0)	1,744 (11.8)	3,915 (42.7)	4,726 (6.9)
Vacant Home - Floors	3,710 (11.6)	3,897 (6.0)	4,841 (4.0)	3,586 (3.4)	3,047 (6.7)	6,041 (26.5)	8,850 (41.9)
Vacant Home - Ceilings	1,896 (3.2)	3,448 (4.2)	4,426 (4.6)	4,584 (8.8)	5,480 (18.6)	7,769 (23.5)	9,836 (37.2)

Quite noticeably, decreasing particle size reflects increasing dust lead concentration, particularly for above-ceiling dust (which can be assumed to be a largely undisturbed historical record of normal aerial deposition of contaminant dust), and also, to a lesser extent, vacant house floor dust. This pattern is less apparent for occupied house floor dust, suggesting human contributions and/or a pronounced different source dust mixing effect. Regressing (class) mean dust lead concentration against an assigned middle-of-class value for each particle size range reveals no significant relationship for any of the dust types ($P \geq 0.25$). Que Hee et al. (1985) note a generally increasing lead concentration with decreasing particle size, although claiming, however, that the concentration of lead in their sieved test dust samples was generally independent of particle size.

Decreasing particle size range classes were generally associated with an increasing proportion of the total (unfractionated) dust lead (as Que Hee et al. [1985]), although regressing (class) mean proportion of lead contained against the assigned mid value for each particle size range again demonstrated no relationship for both floor dust samples ($P \geq 0.35$). For above-ceiling dust, however, a moderate relationship was demonstrated with $r = 0.69$ ($P < 0.085$). Confirming the observation that 77% of the lead present in unfractionated floor dust was contained in the < 150μ fraction (Que Hee et al., 1985); the value from this work is 76%.[7]

[7] Such results for Port Pirie do not necessarily contrast the findings of Harris (1986) who reported that most of the lead-bearing particles found in Port Pirie houses or ceilings were less than 20μ in size, these having been transported by prevailing winds. Essentially, whilst most lead-bearing particles may be this size, the proportion of lead that they contain may be disproportionately less as related to their volume which varies according to the cube of the radius of the particle.

Similarly, Nriagu, (1978) reported for street dust that 90% of the total lead was associated with particles of a size less than 250μ, this also holding for roadside dust. For this size range, all floor samples here constituted 83% of the total lead contained in unfractionated floor dust, but 93% of unfractionated above-ceiling dust-contained lead.

Floor level particle size and lead distribution patterns in vacant homes more closely resemble those of above-ceiling space regions than floor level contamination of occupied homes, where human activity would appear to contribute substantially to the contamination or its within-house mixing/selective removal. Given the similarity between above-ceiling space dust and vacant home floor dust and acknowledging the period of house closure and vacancy, the floor-contaminating dust present in vacant houses would seem to infer a more recent depositional profile (ie during the vacancy period) than a more longer term one. This will be pursued in Chapter Eight.

5.5 SAMPLED DUST PARTICLE SIZE DETERMINATION - LASER TECHNIQUE

One reason for choosing to use the specialised surface dust-sampling equipment employed herein (Chapter Seven) was in response to the need to attempt to more accurately sample that population of surface dust that would be representative of the "available" dust that a child could reasonably be expected to ingest by the hand mouth route. The bulk of the dust sampled with this equipment should lie in the (particle size) range sub-micron to perhaps 100μ (cited above). This is more appropriate than that dust population which would be expected to be collected if (say) domestic vacuum cleaner bag grab samples were utilised as primary sources of dust samples, a not uncommon research practice.[8]

To assess the range of dust particles sampled by the proposed vacuum pump cassette technique (Chapter Seven), 27 cassette-contained floor dust samples (Suite P) each deriving from a separate room from nine of the test houses were submitted for (dust) particle sizing utilising laser particle sizing equipment (Mastersizer X, Malvern Industries, Worchester, UK). Further, for comparison, a second suite of 25 floor dust samples (Suite D) were acquired as (domestic) vacuum cleaner grab samples from a variety of occupied homes in the two test districts of Port Pirie.[9] Both sets of samples were processed similarly and assessed for particle size range. Dust samples acquired by the vacuum pump cassette technique derived from mid-floor carpeted surfaces only.

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[8] Thus when collecting dust in the context of that dust which constitutes the exposure threat to a child, it is critically important to sample dust for which the particle size range approximates that which is most likely to adhere to a child's hands (Que Hee et al., 1985; Rinehart et al., 1993).

[9] Whilst it was requested that the vacuum cleaner be used for house internal cleaning purposes only, no guarantee can be proffered here that the cleaner was not used for other cleaning purposes additionally, eg car, clothes or footwear, external structures, pets etc. All such houses had almost entirely carpeted floor surfaces. Vacuum cleaner bag grab samples were not available from the suite of test houses, although it is assumed here that the vacuum cleaner dust samples that were acquired from the occupied houses would be representative in a general sense of other houses in the two districts.

5.5.1 METHODOLOGY

A description of the vacuum pump and cassette system and operating technique for collection of dust is described in Appendices Nine and Ten respectively. Processing of cassette-contained and vacuum cleaner bag dust samples and instrument set-up details and operating conditions are described in Appendices Eleven and Twelve.

5.5.2 RESULTS

Results of the particle sizing work are presented as frequency histograms; particle diameter on a horizontal logarithmic scale vs frequency on a linear scale. Particle diameter for each dust sample is classified into 32 instrument-selected (equivalent) logarithmic particle size ranges for each of two lenses, although these ranges are obviously different for the 100mm and 300mm lens systems. Statistics derived from these data included median, and ten and ninety percent threshold levels (tenth and ninetieth percentiles), corresponding to that particle size, below which, respectively, fifty, ten and ninety percent of particles in the sample are distributed. Also provided is an overall distribution mean value for particle size for each dust.

Although sample sizes for the three parameters median, and 10^{th} and 90^{th} percentiles were not large (n = 27 & 25 respectively for P & D Suites), normality testing and inspection of their distributions was not conclusive in suggesting a significant departure from a normal distribution. Natural log transformation of data revealed no more normally-distributed a data set, such that subsequent comparison of data was conducted utilising raw data unless otherwise indicated.

Initially, some data were log-transformed and comparisons made statistically using matched pair *t*-test in both raw and log-transformed modes.

5.5.2.1 VACUUM PUMP/VACUUM CLEANER DATA COMPARISON

All particle size results are means of at least three replicates of each dust, from which average RSD values for P and D dust samples are as shown in Table 5.11 below.

Table 5.11 Variability of repeat estimate of dust particle size.

	COLLECTED DUST PARTICLE SIZE DISTRIBUTION			
	CASSETTE DUST (P Suite)		VACUUM CLEANER DUST (D Suite)	
	AVE. RSD (%)	SIZE RANGE (μ)	AVE. RSD (%)	SIZE RANGE (μ)
10th percentile	3.8	0.1 - 14.6	3.8	0.2 - 13.3
50th percentile	4.5	0.1 - 15.2	4.0	0.9 - 20.0
90th percentile	10.2	0.2 - 29.4	5.6	0.3 - 28.5

a) Vacuum pump and filter data (P Suite data)

Dust particle sizing for the P suite of dust samples was conducted twice, once using each lens system. Inspecting both output histograms for each dust revealed that the distributions appeared normal, with no evidence of skewed curves.

Comparisons were made between the corresponding (10, 50 & 90%) threshold values for (separately) the 100 and 300 millimetre lens systems in both raw data and natural log-transformed modes. Matched pair *t*-test comparison between analogues (for both raw and transformed data) revealed no difference for the median and 90% threshold levels ($P \geq 0.20$), and for the 10% threshold level ($P \geq 0.08$), with, for the 100mm and 300mm lenses respectively, means/SDs of 4.1/1.0 and 4.3/0.9. There existed no difference between distribution mean data for dust as provided by the two lens systems ($P \geq 0.20$).

The two lens systems display no detection differences at the tenth percentile, median and ninetieth percentile levels with the given particle size ranges. From an inspection of the particle class size data corresponding to less than the tenth percentile, some detection differences between the two lens systems would seem likely to exist. Logically, this is to be expected given the different particle size cut-off minima corresponding to the two lenses; 0.2 and 0.5μ for the 100 and 300mm lenses respectively. For the purposes of this work, however, it is not critical that such (minor) detection difference between lenses might exist at the < 10% threshold level (the smallest-sized particle range) but not the median or 90% threshold level. It is acknowledged that there would be sub-micron sized particles collected differently by the vacuum pump cassette system (0.8μ filter), domestic vacuum cleaners and also children's hands (Duggan et al., 1985). Further, the detection of these smaller-sized particles is not any more important an extreme of the collecting particle size spectrum for the purposes of this part of the discussion. The large particle size end of the spectrum is that end where it would be expected that differences between different dust-collecting

systems would be likely to be detected. Physical differences between collecting systems, (ie between vacuum pump/cassette-collecting systems and vacuum cleaner bag grab samples), as reflected by collected particle-size spectra, reflect where major differences are detected. This will also be seen below (Section 5.5.2.1 c).

As mentioned earlier, some detection difference between lenses may exist at the lower end of the particle size range (ie < 10^{th} percentile), although no difference exists at the 10^{th}, 50^{th} and 90^{th} percentile values for the dust as detected with either lens. Noting that from all P samples (as measured by the 300mm lens) a mean of 99.61% (SD = 0.88) of particles lie in the < 183μ range, (the higher cut off level for the 100mm lens [Anon., 1992c]), it was decided to use the data deriving from the 300mm lens for the purposes of comparison with dust from the D suite (Section 5.5.2.1 b).

Recalling the slightly lower particle size detection limit of the 100mm lens, it was felt that a better representation of the very small particle sizes might be arrived at using this lens alternative, and data from this source was used to estimate mean/median particle size.

b) Vacuum cleaner data (D Suite data)

Inspection of the output curves and size class data for the 300mm lens for samples from this suite suggests that most dust samples demonstrated between five and ten percent (Mean = 9.1%) of their particles with sizes larger than 180μ, (the upper threshold level for the 100mm lens). Further, there exists a substantive difference between the distribution means and medians for dust samples assessed by the two lens formats ($P \leq 0.01$), the 100mm lens returning the lower values. Thus it was decided to utilise data for this suite provided by only the 300mm lens.

c) P and D suite comparisons

An initial simple side by side comparison of the proportion of dust particles falling into the 32 particle size categories for each of the two dust samples (both at 300mm) (Table 5.12) using matched pair *t*-test reveals no difference between the two dust suites ($P > 0.20$).

Table 5.12 Comparison of mean (class) particle size for P Suite and D Suite dust.

		DISTRIBUTION OF DUST PARTICLE SIZES (% by mass)			
		MEAN PROPORTION INCLUDED WITHIN SIZE RANGE		MEAN PROPORTION BELOW UPPER THRESHOLD LEVEL (Cumulative Frequency)	
PARTICLE SIZE CLASS NUMBER	PARTICLE SIZE RANGE (μ)	CASSETTE DUST (n = 27)	VAC. CLNR. DUST (n = 25)	CASSETTE DUST	VAC. CLNR. DUST
1	> 0.50 - 1.32	1.2	0.4	1.2	0.4
2	> 1.32 - 1.60	0.9	0.3	2.2	0.7
3	> 1.60 - 1.95	0.9	0.3	3.1	1.0
4	> 1.95 - 2.38	1.3	0.4	4.4	1.3
5	> 2.38 - 2.90	1.7	0.6	6.1	1.9
6	> 2.90 - 3.53	2.2	1.0	8.3	2.9
7	> 3.53 - 4.30	2.8	1.2	11.1	4.2
8	> 4.30 - 5.24	3.6	1.5	14.7	5.7
9	> 5.24 - 6.39	4.7	1.8	19.4	7.5
10	> 6.39 - 7.78	5.8	0.7	25.2	8.2
11	> 7.78 - 9.48	6.9	2.2	32.1	10.4
12	> 9.48 - 11.55	7.8	2.7	40.0	13.1
13	> 11.55 - 14.08	8.5	3.4	48.5	16.4
14	> 14.08 - 17.15	8.8	4.2	57.3	20.6
15	> 17.15 - 20.90	8.5	5.1	65.9	25.8
16	> 20.90 - 25.46	7.8	6.1	73.7	31.8
17	> 25.46 - 31.01	6.8	6.8	80.4	38.6
18	> 31.01 - 37.79	5.6	7.3	86.0	45.9
19	> 37.79 - 46.03	4.4	7.4	90.3	53.3
20	> 46.03 - 56.09	3.2	7.3	93.6	60.6
21	> 56.09 - 68.33	2.3	6.6	95.8	67.2
22	> 68.33 - 83.26	1.5	6.0	97.3	73.3
23	> 83.26 - 101.44	1.0	5.2	98.3	78.4
24	> 101.44 - 123.59	0.7	4.8	98.9	83.3
25	> 123.59 - 150.57	0.4	4.1	99.4	87.4
26	> 150.57 - 183.44	0.3	3.5	99.6	90.9
27	> 183.44 - 223.51	0.2	2.6	99.8	93.5
28	> 223.51 - 272.31	0.1	2.1	99.9	95.6
29	> 272.31 - 331.77	0.05	1.5	99.9	97.2
30	> 331.77 - 404.21	0.03	1.2	99.9	98.4
31	> 404.21 - 492.47	0.03	0.9	99.9	99.3
32	> 492.47 - 600.00	0.05	0.8	100.0	100.0

Presenting the cumulative frequency data graphically, however, reveals that there is considerable difference between the cassette and vacuum cleaner-derived data, with distribution median values falling in classes 13 and 18 respectively, ie 11.55 - 14.08μ and 31.01 - 37.79μ particle size ranges (Figure 5.1).

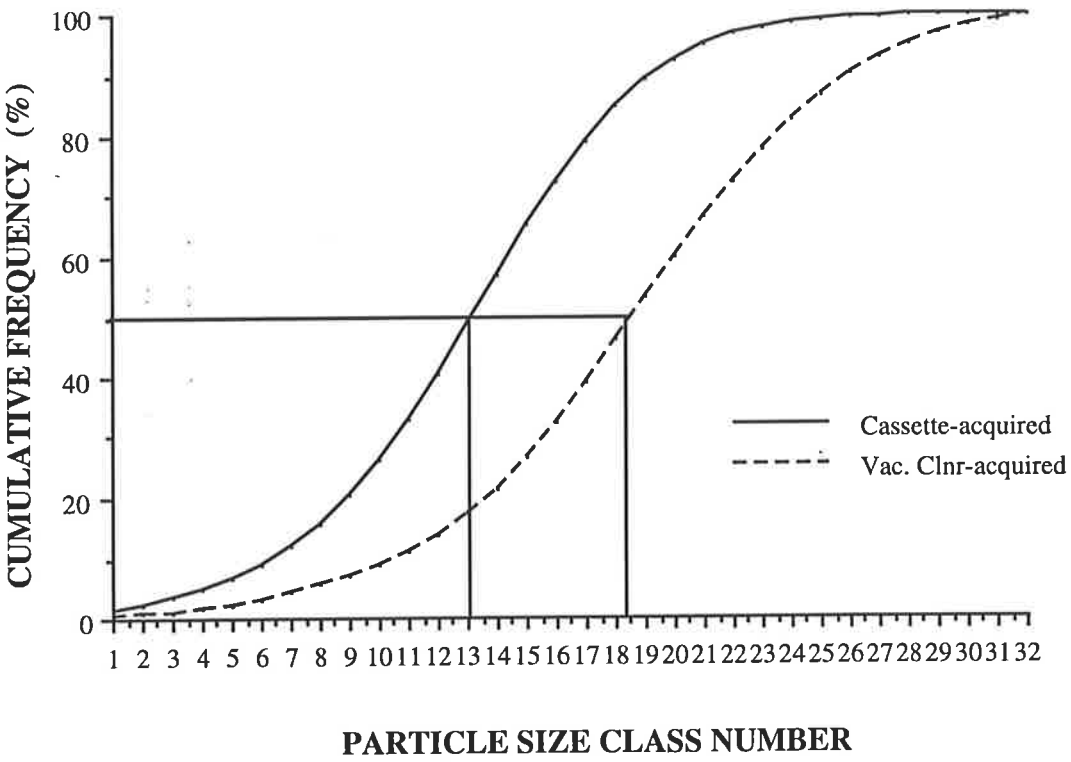


Figure 5.1. Collected dust cumulative frequency particle size plot.

Comparison of the 10^{th}, 50^{th} and 90^{th} percentiles for both suites (for either lens) using matched pair *t*-test reveals difference for all categories in both raw and log-transformed modes ($P \leq 0.01$), class mean values for the vacuum cleaner-derived dust samples always being larger than for their cassette-collected counterparts. Such difference was also demonstrated for particle size distribution means for the two suites of dust ($P < 0.01$). Table 5.13 below summarises these data.

The cassette system provides a narrower particle size collection range and demonstrates a smaller mean collected particle size than the corresponding vacuum cleaner system. It is acknowledged, however, that operator differences and model (efficiency) variability of domestic vacuum cleaners may be a significant contributor to such differences.

Table 5.13 Summary of dust particle sizing results.

	DUST PARTICLE MEAN SIZE			
	CASSETTE-DERIVED DUST (Means and SD)		VACUUM CLEANER-DERIVED DUST (Means and SD)	
	100mm lens (n = 26)	300mm lens (n = 26)	100mm lens (n = 8)	300mm lens (n = 25)
10th percentile	4.1 (1.01)	4.3 (0.87)	8.9 (1.3)	11.9 (8.7)
50th percentile	15.4 (3.5)	15.4 (3.6)	36.8 (4.6)	49.7 (31.9)
90th percentile	47.1 (13.2)	47.8 (15.3)	137.2 (22.3)	169.9 (96.3)
Distribution Mean	22.0 (5.3)	22.0 (6.1)	55.5 (7.1)	73.4 (40.8)

The effective particle size range adhering to a child's hand is from sub-micron to perhaps 100μ with a mean hand-adherent particle diameter for a child of around 4.5μ with 90% of particles being < 10μ (Duggan and Inskip, 1985; Duggan et al., 1985; Steele et al., 1990). Thus, it is concluded that this specialised cassette, floor dust-collecting system is more representative (than vacuum cleaner-collected dust samples) of that population of the dust on floor surfaces potentially likely to adhere to a child's hands and thus contribute to a child's subsequent intake. Hence, this sampling system has been chosen as the dust-collecting technique utilised throughout this work when sampling from surfaces. Such a technique is sufficiently sensitive to be capable of being used to detect spatial contamination differences between various (small) areas within and between rooms of the test house suite. Concerns are expressed here, however, regarding some other alternative sampling techniques being used for such a purpose, particularly if collected particle size data is not available.

It is assumed that no difference between the floor dust from the homes from Suite P and Suite D existed prior to sampling (ie with respect to the particle size ranges of dust present). From a comparison of the RSDs corresponding to the mean proportion of dust particles of a size less than (say) 180μ, 101.4μ, and 68.3μ for both dust-collecting systems, substantially greater variability in proportion of particles with a size less than the chosen size existed with vacuum cleaner bag grab samples than that arising from the other sampling technique (Table 5.14).

Table 5.14 Dust-collecting method variability comparison.

	MEAN PROPORTION OF DUST INCLUDED WITHIN PARTICLE SIZE CLASS (%)					
	CASSETTE-SAMPLED DUST			VACUUM CLEANER-SAMPLED DUST		
PARTICLE SIZE CRITERION (μ)	MEAN	SD	RSD (%)	MEAN	SD	RSD (%)
< 183.4	99.6	0.9	0.9	90.9	11.5	12.6
< 101.4	98.3	2.1	2.2	78.4	16.3	20.8
< 68.3	95.8	3.6	3.7	67.2	16.9	25.2

This is expressed with respect to mean particle size at the 10^{th}, 50^{th} and 90^{th} percentiles in Table 5.15.

Table 5.15 Further dust-collecting method variability comparison.

	MEAN PARTICLE SIZE OF DUST (300mm lens data)					
	CASSETTE-SAMPLED DUST			VACUUM CLEANER-SAMPLED DUST		
PARTICLE SIZE CRITERION	MEAN	SD	RSD (%)	MEAN	SD	RSD (%)
90^{th} percentile	47.8	15.3	32.1	169.9	96.4	56.7
50^{th} percentile	15.4	3.6	23.6	49.7	32.0	64.4
10^{th} percentile	4.3	0.9	20.5	11.9	8.7	72.9

Such greater RSD values for vacuum cleaner-collected dust are as would be expected given the different vacuum cleaner models, bags and operators employed in this regard, and their relative efficiency at abstracting dust particles. No comment may be made regarding the relative variability of any one vacuum cleaner when used exclusively as a dust sampler in all tested houses. Results elicited here, however, warrant the need for a standardised technique for the acquisition of dust from surfaces for subsequent contaminant studies.

d) Particle size relationship with lead content of dust

Cognisant of a literature-suggested inverse relationship between particle size and lead concentration (cited above), the potential for such a relationship was investigated here. Concerns do exist, however, that the collected dust particle size range as acquired by this system may not be sufficiently broad to detect such a relationship. Further, it will be recalled (Section 5.4.2), that for bulk Nilfisk-collected dust samples, no real relationship between mean (class) particle size and lead content could be demonstrated, not dissimilar to the results of Que Hee et al. (1985). Regressing dust lead concentration and (cassette-collected) floor lead loading (dependent variables) with particle size, provided the following (Table 5.16).

Table 5.16 Regression of particle size against dust lead concentration and loading.

	CORRELATION COEFFICIENTS (r =) DEPENDENT VARIABLE			
	CASSETTE-SAMPLED DUST		VACUUM CLEANER-SAMPLED DUST	
INDEPENDENT VARIABLE (particle size)	**LEAD CON-CENTRATION**	**FLOOR LEAD LOADING**	**LEAD CON-CENTRATION**	**FLOOR LEAD LOADING**
10th percentile	- 0.59	- 0.41	- 0.17	-
50th percentile	- 0.44	- 0.45	- 0.01	-
90th percentile	- 0.66	- 0.30	- 0.07	-
Distribution Mean	- 0.64	- 0.30	- 0.03	-

All correlations were in the negative realm confirming a general increase in lead concentration with decreasing particle size as suggested above. For the dust acquired by the cassette system, it is evident that moderate correlation exists between dust lead concentration and particle size ($P \leq 0.001$), and to a lesser extent cassette lead loading ($P \leq 0.10$). A much weaker correlation exists between particle size and dust lead concentration for the vacuum cleaner-acquired dust samples, these not being notable for the 10th percentile ($P > 0.20$) or the remainder ($P \geq 0.40$). Estimates of floor loading were not available with the vacuum cleaner system. Acknowledging a considerable difference between the correlation coefficients corresponding to the two dust-collection systems, and noting the narrower particle size range collected by the cassette system, it is further evident that the cassette system is more representative for the collection of (floor-located) "available" dust (introducing less variability) for the purposes of estimating the potential to contaminate a child's hands.

5.5.2.2 District differences

The cassette dust samples derived essentially from two districts of Port Pirie, namely Port Pirie West and Solomontown. Thus, the potential for district differences in particle size range was investigated. In all situations, Port Pirie West particle size data reflect a larger mean value than that deriving from Solomontown, where the mean house-smelter distance is greater than that corresponding to Port Pirie West (Table 5.17). District comparison reflects notable difference with respect to particle size for both the 100 and 300mm lenses at median, 90th percentile and distribution mean levels ($P \leq 0.05$). At the tenth percentile such difference appears modest ($P < 0.10$).[10]

5.5.2.3 Influence of distance from smelter on particle size

The mean distance between location from which samples were taken and various locations in the smelting works appears in Table 5.17.

Table 5.17 Mean distances between sampling sites and smelter components.

	MEAN DISTANCE FROM SAMPLING HOUSE TO PLANT LOCATION (kilometres)			
DISTRICT	**WHARF**	**CENTRE**	**STACK**	**OVERALL MEAN**
Port Pirie West	1.3	0.8	1.2	1.1
Solomontown	2.0	2.4	2.2	2.2

The test houses located in Solomontown are sited at a greater mean distance from the smelting plant than those from the other district, commensurate with the smaller mean dust particle size collected there. This supports comments by other workers regarding an inverse relationship between sampling distance from (presumed) contaminant source and deposited particle size (cited above).

Regressing individual sample particle size parameters with the corresponding square of the distance from various parts of the plant reveals the following correlation coefficients (Table 5.18).

[10] This observation at the tenth percentile is not unexpected given the small proportion of the total dust pool being examined, and may also reflect the sampling instrument's insufficient sensitivity when approaching the sub-5μ range (Mean tenth percentile = 4.25μ). It may however also suggest that dust particles in this (small) size range are transported more easily, and further, atmospherically, than larger-sized particles (as Lee, 1972; Peirson et al., 1973; Butler et al., 1975; Moschandreas et al., 1979; Settle and Patterson, 1982; Milford and Davidson, 1985; Harris, 1986; Zabel, 1989; Moseholm et al., 1992) and are thus more uniformly distributed over the two districts. Daines et al. (1970) clearly demonstrate a decreasing particle size deposition profile with increasing distance from a point source.

Table 5.18 Particle size variation with square of distance from smelter.

INDEPENDENT VARIABLE SQUARE OF DISTANCE FROM SAMPLING LOCATION TO:	PARTICLE SIZE vs DISTANCE CORRELATION COEFFICIENTS (r =) DEPENDENT VARIABLE - PARTICLE SIZE 10^{th} Percentile	50^{th} Percentile	90^{th} Percentile	DISTRIBUTION MEAN
Nearest smelter boundary	- 0.20	- 0.33	- 0.35	- 0.33
Wharf - concentrate unloading point	- 0.09	- 0.31	- 0.27	- 0.18
Smelter works central point	- 0.20	- 0.39	- 0.37	- 0.34
Smelter stack	- 0.11	- 0.33	- 0.30	- 0.22

Correlation coefficients reflect only a moderate (inverse) relationship here. In all cases apart from the tenth percentile, however, coefficients corresponding to particle size against the square of the distance from the plant centre as independent variable are notable ($P \leq 0.06$), and more impressive than those corresponding to either the wharf or stack, although similar to those corresponding to the boundary. A larger data base with a greater range of distances from the smelter would appear helpful here.

5.5.2.4 House aspect - facing smelter versus facing away

Acknowledging that contaminated dust movement around the township of Port Pirie by particle suspension and wind action is a likely mechanism for contaminant spread (Harris, 1986), the potential for such dust collection within homes to vary according to aspect of the home with respect to the smelter works was investigated, ie rooms facing the smelter as opposed to those facing away from the smelter. Such comparison reveals lower values for mean particle size for smelter-facing rooms as compared with facing away rooms. Acknowledging small sample size, this difference is notable for data deriving from the 300mm lens for the 10^{th}, 50^{th} and 90^{th} percentiles and the distribution mean ($P \leq 0.05$). Such difference appears less apparent for the 100mm lens system ($P \leq 0.10$). Clearly some difference in particle size distribution from facing (smelter) versus facing away house aspects reflects spatial differences in patterns of (within-house) dust deposition/accumulation. This will be further investigated with respect to various dust lead parameters subsequently (Chapters Seven and Eight).

CHAPTER SIX VACANT HOUSE SOIL ASSESSMENT

The Earth does not belong to us; we belong to the Earth. All things are connected, like blood that unites one family. Mankind did not weave the web of life. We are but one strand within it. Whatever we do to the web, we do to ourselves. All things are bound together.

Chief Seattle to President Franklin Pierce in 1844 about the sale of land now the State of Washington.

All matter is composed of water, earth, air and fire. Love and strife are the two forces that bring together and separate these four elements.

Empedocles 2,490 - 2,430BP

6.1 INTRODUCTION

The dimension and variability of soil contamination by lead within the confines of Port Pirie has previously been investigated, to some extent, approximately a decade ago (Body, 1986e & f). Whilst the assessment of dust contamination of vacant test houses is the major focus of this dissertation, some assessment of the calibre of soil contamination at the test house sites will also be undertaken. Here, soil lead contamination was investigated, in order to contextualise the environmental "setting" of these test homes, complementing other home contamination investigations described in subsequent chapters.

Cognisant of the long-term stability of soil-lead contamination levels (Lewis, 1977; Roberts et al., 1990, who suggest "indefinite", p. 436; Davidson and Elias, 1986; ATSDR, 1988), and the potential for (former) occupant-elicited modification of the soil environment in these homes, a limited (non-random) spatial survey of soil lead contamination proximate to test houses was performed.[1] Residential soil from the two districts in which the test houses are located, previously demonstrated mean soil lead levels in the range 500 - 3,000$\mu g.g^{-1}$ predominantly, although some evidence of mean soil lead levels between 3,000 and 10,000$\mu g.g^{-1}$ was also demonstrated (Body et al., 1985).

For this purpose, a portable Energy Dispersive X-Ray Fluorescence (XRF) Detector, (Lead Analyser, Model 9292 - 30mCi Cd^{109} X-ray source; Texas Nuclear Spectrace Technologies, Texas USA) was utilised. Such a technique is simple, quite rapid and accurate (Bero et al., 1993).[2]

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[1] Given Port Pirie's low-lying nature and preponderance to flooding, it became a very common practice for home-owners to acquire "fill" in order to elevate the block and stabilise the covering topsoil (G. Norton, pers. comm.). Historically, this material was frequently acquired from the smelting works and would typically have contained by-products of the concentrate preparation and smelting process, crusher dust, gravel and slag being common randomly-distributed examples, such practice ceasing by the early 1980s (Body et al., 1985). Typically, ground disturbance associated with development would bring such contaminant materials to the surface, this having been considered to be a potential source of lead for children (I. Calder, pers. comm.).

[2] This chosen technique "can yield fully quantitative analyses for soil samples in environmental studies" (Goldstein et al., 1996; p. 2,318), demonstrating a very comparable and linear relationship with acid digestion techniques ($r = 0.98$). This is particularly so in locations of contaminated soil (Bernick et al., 1995; Pyle et al., 1996). Vincent and Boyer (1995) report that instrument variance is minor compared with variance attributable to sample differences.

6.2 OBJECTIVES

Whilst the literature reveals that there exists a reasonable correlation (r lying in the range of 0.2 to 0.4) between occupant blood lead and home soil lead (concentration) levels (Angle et al., 1974; Barltrop et al., 1974 & 1975; Roberts et al., 1974b; Schilling and Bain, 1988; Bornschein et al., 1989; Calder et al., 1989; Falk and Ing, 1989; Madhavan et al., 1989; Marcus and Cohen, 1989; Reagan and Silbergeld, 1989; Thornton et al., 1990; Cotter-Howells, 1991; Weitzman et al., 1993; Farfel and Lim, 1995; Marcus and Elias, 1995; Rabi, 1995), its relative contribution to a child's lead burden would appear to be somewhat poorly defined (Lanphear, 1994).

Soil lead/house dust lead concentration comparisons have already been conducted for some houses in Port Pirie (Body et al., 1988), but in a more limited form (per home) than is being proposed here, and with analysis techniques which did not appear to conform to the then currently-accepted international practice (Tinsley et al., 1983; Body et al., 1985; Que Hee, et al., 1985; Slavin, 1988; Soter et al., 1988). The literature clearly reflects numerous examples where both soil and dust lead concentration levels in homes are appraised for their potential statistical relationship, and also for their relationship with occupant blood-lead levels.

Work undertaken here is intended to map soil lead concentrations at various locations around and proximate to the suite of vacant test houses, in order to establish some assessment of the spatial variability of soil contamination levels for these homes. It is also intended to investigate soil contaminant differences between the two test districts and also individual houses acknowledging different smelter-house location distances. Essentially, soil lead levels for these test homes will be assessed as a home contamination descriptor and also to establish whether such soil contaminant levels may act as a potential indicator for house dust contamination levels.

6.3 INITIAL OBSERVATIONS AND SITE DESCRIPTION

In the context of a general description of the soil surrounding the house, XRF analysis was conducted (where possible) in front and back yards, and both sides of houses, with a (sampling) focus on proximity to potential house entry points for soil particles/dust.[3] Such (presumed) potential entry points included front and rear doors and also windows (as Charney, 1982b and Al-Radday et al., 1993a), sampling being carried out in line with these features and perpendicular to them. In all cases, there existed concrete footpaths and

[3] Typically, one side of the house was located between 800 mm and 1.5 metres from the side boundary fence and was utilised, principally, as a walkway, with little/no surface vegetative cover. Here, surfaces were mostly fairly open soil or soil with a small quantity of gravel. This side of the house was classed as walkway.

porches, essentially at front and rear entrance locations. These were not sampled from as part of the soil study.

Due to individual block/house configuration variation, it was not always possible to sample in equivalent locations for each home, this being reflected by differing numbers of samples deriving from the individual homes. A typical home soil sampling pattern is indicated in Figure 6.1.

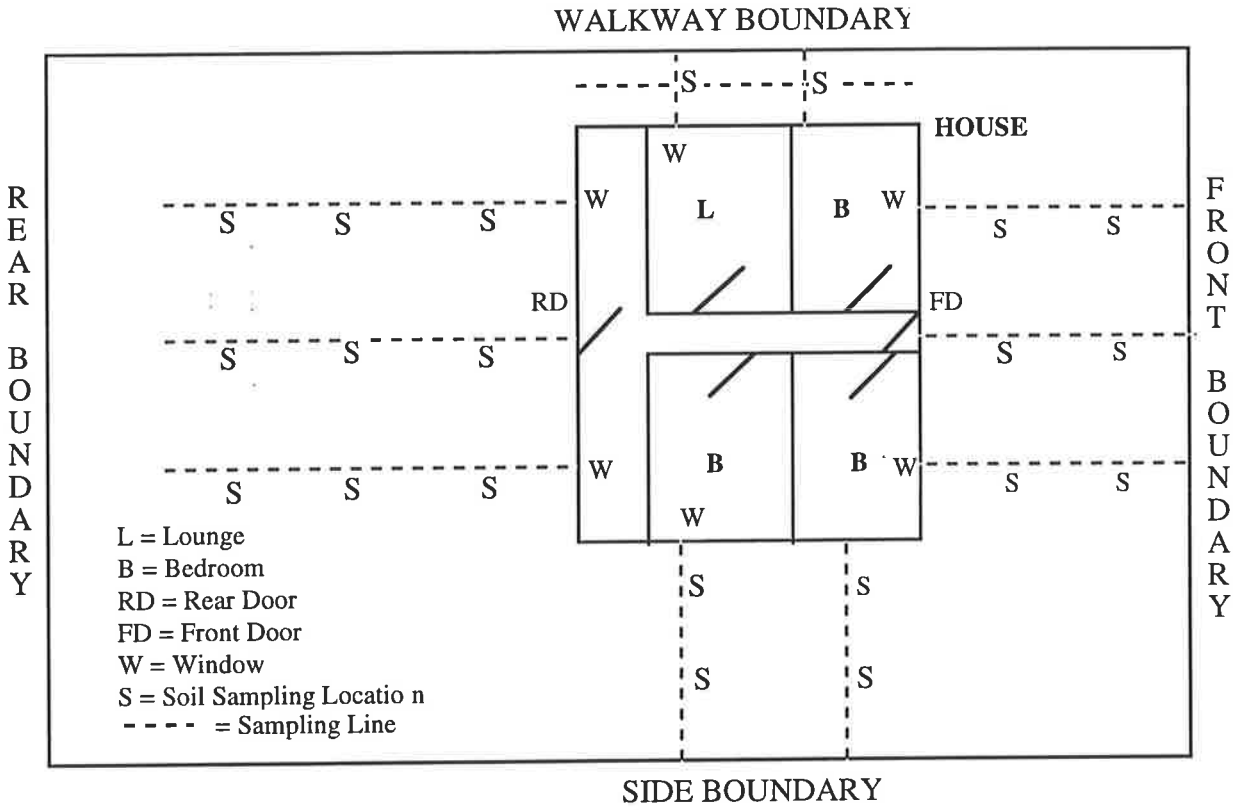


Figure 6.1 Typical home soil sampling pattern.

6.4 SAMPLING PROTOCOL AND LOCATION

Sample surface preparation was minimal typically, although on occasion, some careful removal of loose grass/leaf litter with minimal surface soil disturbance was necessary (as Aschengrau et al., 1991) to facilitate appropriate contact of instrument with the soil surface. Soil surfaces were slightly loosened (to a depth of approximately one centimetre) with a stainless steel spoon, gently mixed in order to homogenise the sample, and levelled. Subsequent to this, the soil surface was slightly tamped such that the probe could be placed vertically upon the soil surface. No soil or decomposed litter material was removed or transferred to an alternative site. Careful inspection was made of surfaces to be examined to ensure that no paint flakes were present, although removal of visible paint flakes with stainless steel forceps was only necessary in one location. No surface dust readings were acquired from concrete porches or footpaths, and

all XRF assessment was made at least 400 mm away from the walls of structures, concrete edges, gutter overflow pipes and other human-constructed objects.[4]

Any area which appeared to have been used for waste material collection, burial, incineration, or storage for foreign material (soil, fill etc) or any recently-exposed soil was avoided.

A total of 173 soil surface locations from ten homes were selected and assessed in situ using a one minute sampling time, with the sampling head resting vertically erect on the soil surface, according to operator instructions (TN Technologies, 1994). Surface soil sampling varies in definition between some workers, but typically refers to the top 5 cm (Thornton, 1990; Gulson et al., 1995). This component of the soil horizon is suggested to be the most relevant for sampling, "especially with regard to the potential (home and child) intake of lead" (Duggan and Inskip, 1985; p. 11). No account was taken of the potential range of soil particle sizes at varying sites, although all chosen sites displayed soil cover, and no chosen sampling sites displayed a predominance of small stones/pebbles at or near surface level.

Prior to analysis commencing, the instrument was calibrated and the standard running protocol checked. Sampling incorporated a standard blank (crushed quartz SiO_2) and a standard reference soil material for a number of elements (13.5 grams Standard Reference Material 2710 - Montana Soil, National Institute of Standards and Technology [NIST], Department of Commerce, Maryland, USA; certified elements and their concentrations are as indicated in Appendix Thirteen) once each for each home assessed. Both elemental standard and standard blank materials were contained in identical polyethylene sample cups approximately 20mm high and 25mm in diameter, at least three quarters full and with a covering Mylar (6 micron) film (both Coherent Scientific, Hilton SA). Instrument output produced soil concentration readings for a suite of six elements contemporaneously.

6.5 RESULTS

Clearly-evident, non-normal, skewed initial data distributions were subsequently (natural) log-transformed prior to data interpretation, as discussed in Chapter One.

6.5.1 SOIL LEAD AND ZINC LEVELS; TOTAL AND BY DISTRICT

Summary soil lead and soil zinc data and also mean soil analyte data corresponding to the two districts from which the houses were selected, namely Port Pirie West (n = 7 houses, 130 samples) and Solomontown (n = 3 houses, 43 samples) are indicated in the Tables 6.1 and 6.2 below.

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[4] The closest distance to any structure was chosen to be one metre, however, in the walkway between house and fence, this was not always possible as the distance between the two may have been as little as 80cm, in which case, the midway point was chosen.

Table 6.1 Soil lead statistics, total and by district.

	SOIL LEAD CONCENTRATION ($\mu g.g^{-1}$)		
	TOTAL DATA (10 houses; n = 173)	**PT PIRIE WEST DATA (7 houses; n = 130)**	**SOLOMONTOWN DATA (3 houses; n = 43) ***
Mean	1,184	1,254	995
Median	1,234	1,329	830
SD	2.04	1.98	2.18
Skewness	0.96	1.05	0.93
Kurtosis	0.94	1.04	0.55

* A small sample size for Solomontown exists, and this may constitute a limitation to this investigation.

Table 6.2 Soil zinc statistics, total and by district.

	SOIL ZINC CONCENTRATION ($\mu g.g^{-1}$)		
	TOTAL DATA (10 houses; n = 173)	**PT PIRIE WEST DATA (7 houses; n = 130)**	**SOLOMONTOWN DATA (3 houses; n = 43)**
Mean	2,203	2,318	1,889
Median	2,367	2,416	1,905
SD	1.87	1.77	2.10
Skewness	0.79	0.99	0.73
Kurtosis	0.98	0.88	0.50

Soil lead and zinc levels are significantly elevated above what would be considered to be background. Such an observation for Port Pirie soils in general has been made previously (eg Cartwright et al., 1976; Tiller, 1989). Additionally, mean soil lead data for the total data suite substantially exceeds $500\mu g.g^{-1}$, a level of 500 - $1,000\mu g.g^{-1}$ having been proffered as a threshold for contaminated soil locally (Calder et al., 1989; El Saadi and Langley, 1991), as also internationally (Yankel et al., 1977). EPA USA (1986) cites threshold levels of $1,000\mu g.g^{-1}$ for soil and $500\mu g.g^{-1}$ for dust (Chapter Seven).[5]

[5] The criterion for the identification of lead-contaminated materials, as set by the South Australian Minister of Health in 1984 was $500\mu g.g^{-1}$ (Body et al., 1985). Yankel et al. report that soil lead levels exceeding $1,000\mu g.g^{-1}$ are "unacceptable", and will contribute to elevated blood lead levels (1977; p. 763). Analogously, Duggan and Williams (1977), assessed the risks from contaminant lead in street dust, estimating that the highest tolerable concentration should be set at a maximum of $300\mu g.g^{-1}$, others citing a level of $\approx 600\mu g.g^{-1}$ as warranting investigation (GLC, 1980; Madhavan et al., 1989). Gulson is cited as claiming a level of $300\mu g.g^{-1}$ lead in soil is safe (Legge, 1993).

Statistically comparing (separately) soil lead data and soil zinc data from the two districts using a student's *t* -test revealed no difference between the analyte data sets corresponding to Solomontown and Port Pirie West for both elements ($P \geq 0.05$). Thus for lead and zinc, individual district soil analyte data were not considered to be representative of different soil contaminant populations, and were thus grouped.

6.5.2 SOIL CONTAMINANT LEVELS: HOME BY HOME COMPARISONS

A comparison of soil lead and zinc concentrations for individual homes was conducted in order to assess the potential for variation in soil contaminant levels between homes. Small sample sizes per home (n = 10 - 22; Ave = 17.3), were not helpful for optimal statistical interpretation, although statistical comparisons between homes were still undertaken using these data.

Summary soil contaminant data on a home by home basis are indicated in Table 6.3 below.

Table 6.3 Soil contaminant concentrations home by home.

		MEAN HOME SOIL CONTAMINANT LEVEL		
HOME	**n**	**LEAD ($\mu g.g^{-1}$)**	**ZINC ($\mu g.g^{-1}$)**	**LEAD/ZINC RATIO [6]**
6 Agnes St	**20**	652	1,400	0.47
64 Edward St	**20**	707	1,386	0.51
58 Federatn. Rd	**18**	1,570	2,805	0.56
29 Fourth St	**15**	1,423	3,038	0.47
19 Lily St	**19**	1,605	2,700	0.60
3 Sabine St	**13**	1,815	3,432	0.53
34 The Terrace	**22**	1,666	2,589	0.64
27 Vera St	**20**	1,638	2,628	0.62
136 Warner. Rd	**10**	903	1,615	0.56
97 York Rd	**16**	709	1,676	0.42
Data set Mean	**17.3**	**1,184**	**2,204**	**0.54**

As can be seen from these data, all mean home soil lead data are greater than $500\mu g.g^{-1}$, six examples demonstrating mean soil lead values substantially greater than $1{,}000\mu g.g^{-1}$, lead, a level considered to constitute contaminated land (El Saadi and Langley, 1991). Such results are in keeping with data derived

[6] Raw lead/zinc ratio data also appear to be log-normally distributed, data interpretation here utilising the (natural) log-transformed ratios of original lead and zinc soil data.

from earlier work in both these districts, demonstrating mean levels in the range 100 - 10,000$\mu g.g^{-1}$, , with a median value of $\approx$ 1,000$\mu g.g^{-1}$, (Body et al., 1985).

Applying ANOVA for (separately) soil lead and zinc values by home reveals that there exists a range of mean values for both soil lead and soil zinc, difference being noted between low and high mean home soil analyte values ($P \leq 0.001$). Removal of data corresponding to the highest and lowest mean home soil lead concentration homes (Sabine St and Agnes St respectively) prior to conducting ANOVA analysis demonstrates similar difference to earlier comparison ($P \leq 0.001$), this also being the case for soil zinc values. Repeating this exercise with the removal of home data corresponding to the homes reflecting the two largest (Sabine St and The Terrace) and two smallest (Agnes St and Edward St) mean home soil analyte concentration values also demonstrated similarly ($P \leq 0.001$). Thus concerns regarding the possibility of any one particular house strongly weighting the mean house soil analyte results appear unfounded.

Such home to home soil lead level variability as has been noted might reflect former occupant activity such as acquisition of additional soil material of varying degree of contamination by lead and/or zinc, or the earlier practice of the use of a variety of materials derived from the smelter for solid fill (Body et al., 1985). Some evidence of recent soil replacement was apparent at one of the locations (Vera St). This was confirmed as having been conducted in 1993, some fifteen months prior to soil assay, as part of a previous soil remediation/replacement program. None of the other vacant homes received similar treatment as part of this program. At this site, all soil was removed to a depth of approximately 150mm, and replaced by clean, uncontaminated soil (< 50$\mu g.g^{-1}$; P. Bartholemew, pers. comm.).

Presuming that soil contaminant levels arise substantially as a result of historical (resident) home soil profile modification utilising, typically, smelter slag as fill (Body et al., 1985), it would be expected that soil lead/zinc ratios might reflect this. Mean lead/zinc ratio for several different types of smelter slag was 0.1 (0.1 - 0.2), substantially smaller ($P < 0.0001$) than the mean home soil lead/zinc ratio of 0.5 (0.3 - 1.3). Smelter-emitted contaminants on the other hand would, by the very nature of the smelter's process, have generally contained more lead than zinc, (A. Gilbert, pers. comm.), potential smelter-derived (non-slag) contaminant sources demonstrating a mean lead/zinc ratio value of 6.7 (0.2 - 14.3), small values for lead/zinc ratio deriving typically from zinc concentrates. Such data do not support the hypothesis that the soil-contained contaminants detected arose as a result of the earlier occupant use of smelter-derived fill material. Moreover, given the current (elevated) levels of soil contamination by lead and zinc at the residence located at Vera St, some source of contaminant material must have influenced soil lead/zinc levels here subsequent to the soil's (recent) replacement as part of the lead intervention program. Further, that these soil contaminant levels here are not dissimilar to those at other sampled domestic locations suggests that the contamination process is contemporary and does not appear to differ markedly between sites.

Presuming that the current degree of soil contamination at the surface level arises as a result of more recent contaminant deposition, then it could be concluded that the slightly smaller soil lead concentration variability reflects either that the source of lead is less variable than that for zinc, or that the soil's retention ability for zinc might be marginally lower than that for lead. This would appear not unreasonable given lead's rather high soil stability (Cartwright et al., 1976; Fergusson et al., 1980; Brunekreef et al., 1981; Gulson et al., 1981).

The Sabine St house reveals the largest mean soil contaminant level for both elements, with the A St house demonstrating the lowest mean value for soil lead and the second lowest for soil zinc. Exce..... correlation between mean home soil lead and zinc (pair) values (for the total data suite, r = 0.87) reflects good agreement between the data sets, and similar origins for these two soil elements. By comparison, correlation coefficients (r) of 0.27 and 0.44 for lead (dependent variable) with copper and iron respectively and 0.23 and 0.46 for zinc (dependent variable) with copper and iron respectively, and 0.62 for copper (dependent variable) with iron, suggest that mean home soil copper and iron levels reflect poorer agreement with lead and zinc than between themselves (Table 6.4)

Table 6.4 Correlation matrix for soil elements.

	CORRELATION TEST VARIABLE (r =)			
DEPENDENT VARIABLE	**LEAD**	**ZINC**	**COPPER**	**IRON**
Lead	*	0.87	0.27	0.44
Zinc	-	*	0.23	0.46
Copper	-	-	*	0.62
Iron	-	-	-	*

Such regression underscores the close relationship between lead and zinc as smelter products, and also their presence in the soil around homes as contaminant materials, the other two analytes being reasonably normal soil elements (for Port Pirie). Acknowledging such a good correlation between home soil lead and zinc concentration values, the further inference is that the number of different sources of home soil contamination would not appear to be large. This will be pursued in the context of sampled house dust and indoor/outdoor air subsequently (Chapter Ten). Concerns regarding the potential for unequal weighting of lead/zinc ratio data by home due to unequal home sample numbers appeared unnecessary, as no relationship between these two parameters was apparent ($P \geq 0.35$).

6.5.3 SOIL CONTAMINANT LEVELS: BY DISTANCE FROM SMELTER

Given that the use of smelter-derived waste materials for home fill purposes was (historically) not infrequent and apparently random, it would seem reasonable to assume that if the major contributor to home soil lead levels was such material, then it would be unlikely to be able to detect a soil concentration gradient radiating out from the smelter.

Such a (crude) concentration gradient for lead, however, was revealed in soil by Body et al. (1985) close to the smelter, and also, by other workers, radiating further afield (> 40kms) from Port Pirie (Tiller et al., 1975; Cartwright et al., 1976; Merry and Tiller, 1978). In Body's study, soil samples derived from a larger number of locations (35 open space areas, 24 schools/public buildings and 192 houses), were sampled at four different soil horizons and were distributed over a wider range of distances from the smelter than is the case here. It seems appropriate, thus, to assess whether any such soil concentration gradient for lead or zinc with distance exists here. All soil element data were treated separately, and soil concentration regressed against the squares of the following distances for each sampling location:[7]

a) distance from home to nearest smelter boundary
b) distance from home to central point of smelter works
c) distance from home to smelter stack, and
d) distance from home to concentrate-unloading point on wharf.[8]

The following table (Table 6.5) lists correlation coefficients for these alternatives. Also included, are analogous correlation coefficients for these distance alternatives with sampled copper and iron values for each location, essentially intended for comparison with the data for lead and zinc.

Table 6.5 Soil contaminant variation with square of distance from smelter.

	CONTAMINANT LEVEL vs DISTANCE CORRELATION COEFFICIENT (r =)			
	DEPENDENT VARIABLE			
INDEPENDENT VARIABLE	**LEAD CONC.**	**ZINC CONC.**	**COPPER CONC.**	**IRON CONC.**
SQUARE OF THE DISTANCE FROM THE SAMPLING LOCATION TO:				
Nearest smelter boundary	- 0.26	- 0.24	0.06	- 0.08
Wharf - concentrate unloading point	- 0.29	- 0.25	- 0.08	- 0.06
Central point of smelter works	- 0.25	- 0.22	- 0.08	- 0.06
Smelter stack	- 0.28	- 0.24	- 0.08	-0.05

[7] Soil element concentration was also regressed against distance with only very marginal differences in correlation coefficient values (as compared with the square of the distance), this alternative being subsequently disregarded.

[8] Discussing wind erosion of coal stockpiles, Smit (1980) reports that they may constitute major sites for the aerial resuspension and wind transport of particulate material over substantial distances if they are actively being worked as are the zinc concentrate stockpiles on the wharf at Port Pirie, Davies and White (1981) commenting similarly regarding wind-blown lead mine waste in Wales. Particles with an equivalent spherical particle diameter of ≤10μ, can be carried in suspension by air currents. For larger-sized particles, 100μ up to a maximum of 500 - 1,000μ, saltation is the likely carriage mechanism. For a more detailed discussion of wind carriage of particles see Bagnold (1941).

All correlations for lead and zinc reflect decreasing mean home soil element values with increasing distance for the four distance alternatives, although there does not appear to be a particularly strong relationship between the various distance categories and mean soil lead or zinc levels. This is not really surprising in that the sample size of test homes is not large, and that the range of distances from the homes to (say) the centre of the works is only from ≈ 0.5kms (Vera St, Agnes St and The Terrace) to 3.3kms (Edward St), the average distance being 1.4kms.[9] Again there would appear to be reasonable agreement between the lead and zinc data, although only a moderate correlation with distance. All relationships between soil level and distance from smelter (for lead and zinc specifically) demonstrate significance ($P \leq 0.003$).

From the correlation coefficient data corresponding to soil copper and especially iron with distance, however, a much poorer relationship would appear to be suggested, all regression confirming that no distance/soil element concentration relationship is apparent ($P \geq 0.20$).

In summary, such data would appear to support the claim that both soil lead and zinc derive from similar sources and, given the location of the two test districts (Figures 1.1 & 1.3), point to the smelter area as the source. Soil copper and iron data reflect a lower likelihood that these elements derive from a common source, and, little likelihood that either element derives similarly to lead and/or zinc. The inference here is that these detected (particularly iron) levels represent normal soil levels.

The presumption that the movement around the township of lead-contaminated novel dust particles deriving from the smelter occurring as a result of (say) wind activity is neither supported nor negated by the lack of an observed strong soil contaminant gradient radiating out from the smelter works. The concept that a historic, contaminated dust pool deriving from more than a half century ago and relentlessly moving around the township at the caprice of aeolian activity would seem, however, somewhat unlikely, given the soil lead/zinc gradients demonstrated, and the expectation of a more even spread of contamination due to mixing and redistribution.

6.5.4 SOIL CONTAMINANT HOME CONCENTRATION GRADIENTS

It is important to establish whether an overall soil contaminant concentration with distance from house gradient exists amidst the surface soil of test homes. If so, occupant activity could be modified accordingly with site rehabilitation practices possibly implemented. Further, soil contaminant concentration gradients in close proximity to potential house entry-points for soil-derived materials/dust may have significant implications for house contamination.

The data for such a concentration "gradient" effect were examined separately for both lead and zinc, with each element's data set being distributed initially into one of seven classes corresponding to in-line distance away from the house.[10] These distance classes were as follows:

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[9] Earlier work by Body et al. (1985) covered a distance range of from 0.5 to 5.5kms, and sampled from 192 homes.

[10] Since all test houses demonstrated the presence of concrete paths adjacent to door entry points, distance from the doorway was actually taken to be the distance from the concrete edge directly in-line with the door.

Distance from sample point to house:

a)	One Half Metre	CLASS ONE SOIL
b)	One Metre	CLASS TWO SOIL
c)	Two Metres	CLASS THREE SOIL
d)	Three Metres	CLASS FOUR SOIL
e)	Four Metres	CLASS FIVE SOIL
f)	Five Metres	CLASS SIX SOIL
g)	> Five Metres	CLASS SEVEN SOIL

Corresponding distance class mean soil lead concentrations are as follows (Table 6.6):

Table 6.6 Distance class mean soil lead concentration.

MEAN SOIL LEAD CONCENTRATION ($\mu g.g^{-1}$)						
DISTANCE CLASS						
ONE (n = 9)	**TWO (n = 60)**	**THREE (n = 37)**	**FOUR (n = 17)**	**FIVE (n = 21)**	**SIX (n = 11)**	**SEVEN (n = 18)**
2,522	1,135	1,266	1,027	1,221	962	1,013

T-test comparison for other than Class One permutations revealed no difference between any two class combinations ($P \geq 0.25$). All Class One comparison permutations revealed concentration levels for Class One to be greater than all other class permutations ($P \leq 0.025$). Given such a result, investigation for soil zinc was also undertaken, distance class mean soil zinc concentrations appearing as follows (Table 6.7):

Table 6.7 Distance class mean soil zinc concentration.

MEAN SOIL ZINC CONCENTRATION ($\mu g.g^{-1}$)						
DISTANCE CLASS						
ONE (n = 9)	**TWO (n = 60)**	**THREE (n = 37)**	**FOUR (n = 17)**	**FIVE (n = 21)**	**SIX (n = 11)**	**SEVEN (n = 18)**
4,040	2,267	2,165	1,965	2,099	2,067	1,878

Again, all Class One permutations demonstrate greater soil zinc concentrations than all others ($P \leq 0.025$). For all other comparisons of class permutations, no real difference is apparent ($P \geq 0.30$). This is identical to the results for lead. These data suggest that a general soil lead/zinc concentration decrease occurs with increasing distance from the house structure. Regressing soil element concentration (dependent variable) against the square of the sampling distance from the house, revealed, for lead and zinc, poorly-defined inverse relationships ($P \leq 0.25$). Analogous regression for copper and iron demonstrated a moderate, but direct relationship ($P \geq 0.10$). Whilst no clear contamination level with sampling distance from house relationship is apparent, for lead and zinc, some increase in soil contaminant level occurs proximate to the dwelling. This is not evident for the other two elements.

Data pertaining to soil in close proximity to the dwelling (Class One) reflect markedly greater contaminant levels, contributing to modest gradients for both lead and zinc, and inferring a focus of soil contaminant level proximate to the dwelling. This is not the case for iron or copper.[11] Such increase in soil contaminant concentration adjacent to the main dwelling has been reported previously for other study locations (Culbard et al. 1983a & 1988; Duggan and Inskip, 1985; Thornton, 1985; Davies and Thornton, 1987; Camman et al., 1990; Roberts et al., 1991b; Watt et al., 1993), but not for Port Pirie. Such an observation for Port Pirie would appear to be in keeping with the concept of a "sea" of mobile contaminant dust depositing disproportionately larger quantities of dust proximate to houses, where, presumably, wind speeds are reduced by the dwelling structure.[12] In a later chapter (Chapter Ten), the potential relationship between soil and dust contaminant level within the home will be investigated.

6.5.5 SOIL CONTAMINANT LEVEL BY HOME AREA

Upon initial inspection of the test homes, given their basic similarity in lay-out, home (outside) areas were divided into one of four different sub-areas based upon area usage patterns, relationship to various house entry points, and to a limited degree, vegetation patterns.[13] The four different home locations were:

Location:

a) Back of property
b) Front of property
c) Side of property
d) Walkway

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[11] This would seem to further suggest that detected lead and zinc levels represent above normal soil background level contamination, detected copper and particularly iron levels representing more normal soil elemental levels. Schwar et al. (1988b) investigating school playground contamination demonstrated similarly, ie a sharp decrease in surface soil concentration with increasing distance from the wall, levels typically peaking within a metre or so from the wall.

[12] It is noted elsewhere (Duggan and Inskip, 1985; Thornton, 1985; Culbard et al. 1988; Camman et al., 1990; Roberts et al., 1991b; Watt et al., 1993) that this concentration results from the walls/roof acting as a "collecting" surface for contaminated material, a reasonably plausible explanation for Port Pirie also. It should be noted, however, that typically these higher soil contaminant level areas are usually less-vegetated, lower-activity and, to some degree, wind-protected regions (at ground or near-ground level) of the homes, thus presumably, facilitating the (potential) accumulation of mobile airborne contaminant materials.

[13] Essentially, all premises were similar in that grass coverage was by far the most common vegetative feature on a surface area basis. The next most common was in fact the absence of any vegetation, or simply, bare soil cover, typically the case in the walkway and some parts of the back yard.

Investigation of soil contaminant levels in each of these areas was conducted. Typically, areas such as the front, and side of the home reflected grass cover of varying calibre albeit absent in some locations, these areas demonstrating packed soil or occasionally a mix of soil with a small amount of gravel. All residences revealed the presence of at least a few shrubs/trees. No side area for Vera St was present, there being two walkways. Backs of houses typically displayed grass cover, various shrubs/trees and also, again, a number of bare soil areas. Walkways were typically just soil, or a mix of soil with a small quantity of gravel. Concrete paths (≈ 500 - 600 mm wide) were common to all dwelling back and front boundaries.

Soil lead and zinc data for the different home areas is summarised in Table 6.8.

Table 6.8 Mean soil lead and soil zinc statistics by home area.

		MEAN SOIL LEVELS ($\mu g.g^{-1}$)	
HOUSE AREA	**n**	**LEAD**	**ZINC**
Back	**53**	1,010	1,892
Front	**60**	1,199	2,198
Side	**43**	1,217	2,303
Walkway	**17**	1,734	3,191

Again, a firm relationship between the two elements exists. Walkways reveal the greatest mean home levels for both lead and zinc, and backyards the lowest, although it is acknowledged that for walkways, sample size is not large. A comparison (*t*-test) of the soil lead and zinc data corresponding to the different home areas is indicated below (Tables 6.9 & 6.10).

Table 6.9 Home area soil lead concentration comparison.
(**x** denotes no difference; ✓ denotes a difference at $P \leq 0.05$)

	BACK	**FRONT**	**SIDE**	**WALKWAY**
BACK	*	x	x	✓
FRONT	-	*	x	✓
SIDE	-	-	*	✓
WALKWAY	-	-	-	*

Table 6.10 Home area soil zinc concentration comparison.
(x denotes no difference; ✓ denotes a difference at $P \leq 0.05$)

	BACK	FRONT	SIDE	WALKWAY
BACK	*	x	x	✓
FRONT	-	*	x	✓
SIDE	-	-	*	✓
WALKWAY	-	-	-	*

In all comparison permutations for both these soil elements consistent results occur, suggesting, again, commonality with respect to source. Concerns arise regarding the potential for weighting of mean soil analyte concentration of various home areas as a result of unequal mean sampling distance from the house, given earlier observations that soil lead concentration appears most elevated proximate to foundations (cited above). These differences in sampling distance from house mean values occur due to the non-central location of the house on the block and also some associated limitations to sampling location, with for example, in all cases, a smaller home front yard than rear yard. Mean sampling distance from house for front yards ($n = 60$) was 1.71m, side yards ($n = 43$) 2.28m, and back yards ($n = 53$) 3.01m. No major weighting of soil analyte results for these three home areas due to grossly unequal distance from house sampling appears likely, given the data presented in Table 6.7 above. Mean walkway sampling distance from houses was 0.82 metres ($n = 17$), however, this home area reflecting greater mean soil analyte values ($P \leq 0.05$) than other home areas (Tables 6.9 and 6.10).

Such lower mean sampling distance from house values for walkway-located sampling points compared with other home areas, elicits concerns regarding the potential for leverage of walkway-sampled data values. Comparison of all samples collected within a distance of 1.5m of house foundations, on the basis of they having been either walkway-derived or not, revealed a substantially greater mean value for walkway-derived samples ($P < 0.005$). Walkways, thus, represent a more contaminated home area than other areas.

Separately regressing soil lead and zinc concentration with the square of the sampling distance from the house for each of the four home areas, reflected no clear relationship for side, front and back areas ($P \geq 0.10$). Walkways, however, demonstrated a strongly inverse relationship with, for lead and zinc, $r = 0.55$ and 0.611 respectively ($P \leq 0.025$). Such results infer a reasonable soil contaminant gradienting effect close to the house in walkways. Given the observation that walkways were more (wind) protected areas (pers. obs.), and not as vegetated or heavily trafficked/disturbed as the other three areas, it would seem likely that these areas are collection points for contamination, less post-depositional redistribution of contaminant dust occurring there than elsewhere. Selecting all non-walkway sampling locations at a maximum distance of 1.5m from the house, and regressing soil contaminant concentration with the square of their distance from the house, revealed a poor association for all elements ($P \geq 0.25$).

In all homes, both the back and the front areas are the zones in which most occupant activity would be expected to occur given that these areas provided the most convenient access to the dwellings. Further, such areas also appeared to be the areas most frequently reflecting some attention having been paid to grass levels, path access and a general state of "tidiness". For both lead and zinc there appears to be no difference between mean soil levels in front and back areas in these test houses ($P \geq 0.20$), suggesting that contamination rates vary little between these locations, and/or that human activity may contribute to effective mixing of soil surface contaminants in these areas. This of course presumes that the distribution of contaminants to home area soil is an on-going situation, rather than such contamination resulting from the acquisition by residents of contaminated surface fill previously. Given the close association between soil lead and zinc levels, this not being the case for any other soil element combination comparison, the former would appear more likely.

The vegetative cover of the side of the home tended to be more well-developed, and less disturbed than front or back, with less evidence of human thoroughfare. On some occasions, however, there appeared to be side areas of homes revealing (variously-located) diminished grass density, with evidence of a compacted sand or sand/gravel base. No recent evidence of the use of such areas as driveways existed, in fact several of the residences permitted no vehicular access to these locations or the property in general.

Although sample size for these less well vegetated areas was small ($n = 11$), comparison with remaining side area data revealed that lead and zinc levels in these more open areas were not observed to be as high as the other side area data ($P \leq 0.025$). This open-ness and diminished surface cover may reflect the inability to retain (say) wind-borne contaminant material due to the absence of any ground cover, and the less-protected character of these areas. Rather, such areas may facilitate the collection of contaminant materials in adjacent areas where some protection from wind is offered such as by enhanced grass cover.[14]

By comparison, given the degree of protection from wind at ground level and the likely infrequent passage through the walkway, as evidenced by seized gates, stored items and partial obstruction/blockage by trees, debris or other structures, and the typically bare soil areas, it would seem very likely that such an area is a collection point (trap) for (say) airborne contaminant material. Such accumulation would presumably occur in reduced air-flow locations, unlike other areas of the home.

Paint flakes were very rarely sighted here even under close inspection, and generally speaking, paint condition on facing walls appeared not to be degraded (flaking or chalking), painted surfaces frequently demonstrating recent recoating (pers. obs.). Given the sizeable surface area of the sides of houses facing walkways, and their narrow width, it seems quite possible that such an area is also a potential collection point for (difficult to see) paint flakes, possibly also contributing to an elevated soil contaminant level. It would seem very appropriate for future research to be directed to assessing (say) lead-bearing dust particles derived from/depositing in this location in order to indicate their source of origin as deriving from paint, smelter or even gasolene. Lead loading and dust lead depositional studies here would also be insightful.

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[14] These areas did not demonstrate any evidence of recent human activity as regards soil disturbance (as noted at Vera St). Acknowledging levels of soil contamination at Vera St (comparable with levels at other homes despite the soil's fairly recent replacement), it would seem unlikely that occupant activity such as (say) turning of soil was responsible for the lower contaminant levels in home side areas demonstrating minimal grass cover.

6.5.6 HOUSE ASPECT: FACING SMELTER VERSUS FACING AWAY

Given that for both lead and zinc soil concentrations, there appears no difference between front, back and side areas of homes ($P \geq 0.20$), the potential for influence of mobile contaminant dust on that aspect of the home that faced the smelter and that which faced away from the smelter was investigated.

In all situations, except for the house located at 97 York Rd, that boundary of the block that directly faced the smelter was either the back of the property (n = 5) or the front (n = 4). The eastern side of the dwelling at 97 York Rd directly faced the smelter, there being no walkaway present. Hence, in the first instance, data corresponding to 97 York Rd were excluded from the comparison of home aspect facing the smelter vs facing away.

Summary lead and zinc soil data for the two aspect alternatives are included in Table 6.11 below.

Table 6.11 Soil element level for home aspect facing smelter versus facing away.

MEAN SOIL CONTAMINANT CONCENTRATIONS - HOME ASPECT TO SMELTER SPATIAL RELATIONSHIP ($\mu g.g^{-1}$)				
	EXCLUDING 97 YORK Rd		INCLUDING 97 YORK Rd	
ELEMENT	FACING SMELTER (n = 64)	FACING AWAY (n = 39)	FACING SMELTER (n = 68)	FACING AWAY (n = 43)
Lead	1,394	863	1,350	875
Zinc	2,412	1,624	2,350	1,641

Excluding the data for 97 York Rd, *t*-test revealed a greater mean soil concentration for lead and zinc for that aspect of the home facing the smelter compared with that aspect facing away ($P \leq 0.002$). Including the data for 97 York Rd in the total data suite made little difference to overall mean lead and zinc concentration levels, difference still existing between facing and facing-away home aspects ($P \leq 0.003$).

Since lead and zinc are the principal metals produced at the smelter, it is useful to compare this apparent distribution difference for these elements over a home area to the home area distribution of other (presumed) non-smelter derived elements contained in the surface soil. Such comparison may reveal soil contamination processes.

The two selected elements were iron and copper, summary data for which appear in Table 6.12 below.

Table 6.12 Soil element level for home aspect facing smelter versus facing away.

MEAN SOIL CONTAMINANT CONCENTRATIONS - HOME ASPECT TO SMELTER SPATIAL RELATIONSHIP ($\mu g.g^{-1}$)				
	EXCLUDING 97 YORK Rd		INCLUDING 97 YORK Rd	
ELEMENT	FACING SMELTER (n = 64)	FACING AWAY (n = 39)	FACING SMELTER (n = 68)	FACING AWAY (n = 43)
Copper	222	200	222	204
Iron	13,125	12,984	13,127	13,754

T-test comparison of facing and facing-away categories for the two elements revealed no difference for either element ($P \geq 0.50$), although it should be noted that copper levels do not depart substantially from instrument detection limits. Such results would seem to strongly confirm that contamination of soil by lead and zinc is different for different aspects of the home soil. These data also suggest that the contaminant material is mobile (arising as a result of wind activity) and current given the directionality of home aspect soil contamination differences. The smelter area would appear to be the source, the facing-away aspect of homes reflecting a "lee" or wind-protected character. This would appear to contrast earlier suggestions that contamination of Port Pirie homes arises as a result of the ceaseless movement of much earlier-distributed (fugitive) smelter emissions back and forth around the township (Body et al., 1985 & 1988a).

This work's results derive from summary data for both Port Pirie West and Solomontown (Figure 1.3), with the test houses from these two districts lying in two separate "islands". Given that a line drawn from the centre of each house cluster to the centre of the smelter subtends an angle of greater than seventy degrees with the other, such home aspect differences appear to convincingly demonstrate the on-going nature of the emissions from the general direction of the smelter works.

Acknowledging the different locations of these test homes, in order to establish contaminant movement patterns around the township, attention should be paid to the local prevailing winds at each home location. The presumption here is that at least a proportion of the contamination of the home environment arises as a result of the home deposition of wind-borne contaminant material. Since this research is directed at the degree and variability of home environment contamination, specifically patterns of house (internal) contamination, and not to any great degree the mechanisms by which the home (external) areas have experienced increased (above background) levels of contaminants, no investigation of prevailing wind patterns will be entertained here. Further, future work should also direct greater emphasis toward the location of any test dwelling with respect to the smelter and which areas of the home are:

a) potentially more likely to be more contaminated, and

b) potentially more likely to be frequented by children.

This should complement investigations directed toward establishing the prevailing wind direction in relation to the dwelling and the smelter if homes lie within the general distance range (from the smelter) of these test homes.

Body et al. (1985) as Harris (1986) conclude that wind at close to ground level is the major transport mechanism of lead-contaminated materials within Port Pirie. Acknowledging this, it could be presumed that an overall soil contaminant concentration gradient would be expected to arise, with decreasing levels radiating out from the smelter area, which in fact, was reported by Body et al. (1985). They report that a (crude) concentration gradient for lead does exist in soil in Port Pirie, soil contaminant levels decreasing with increasing distance from the smelter, and at four different soil horizons, including the top (approximately) two centimetres. At home level, however, they claim that given the potential for modification to home area soil environment by occupant activity, this overall surface soil contaminant gradient effect may be somewhat veiled. Further, it would be expected that such a gradient may be difficult to detect over such a short distance as (say) the length of a home block or several home blocks anyway. Such comments only serve to enhance the significance of the observed facing vs facing-away contaminant observations noted here.

It would be of interest to test whether there exists a decreasing contaminant soil concentration gradient along the walkway of homes from proximal to smelter to distal to smelter ends, insufficient sample numbers here precluding such a study.

6.5.7 VACANT HOME SOIL LEAD/ZINC RELATIONSHIP

Both lead and zinc are primary concentrates for the smelter operation at Port Pirie, and both elements appear distributed in the smelter's waste-stream. It is thus appropriate to establish whether there exists any relationship between these two elements in the home (surface) soil matrix. If, for example, correlation coefficients between soil lead and soil zinc concentrations and/or lead/zinc concentration ratios varied little between districts, homes and/or home areas, inference could be drawn as to the potential similarity of source of these two elements as they are detected in the soil of the home environment.

6.5.7.1 CORRELATION BETWEEN SOIL LEAD AND ZINC VALUES

Regressing soil lead concentrations (dependent variable) against soil zinc concentration demonstrates correlation coefficients for district, home area and individual homes, as indicated in Tables 6.13 - 6.15 below:

Table 6.13 Lead/zinc correlation coefficients for all data, and by district.

SAMPLE SOURCE	CORRELATION COEFFICIENT (r =)
All samples	0.93
Port Pirie west	0.92
Solomontown	0.97

Such regression reveals the following relationship between soil lead and zinc for the total data set:

ln soil lead = - 1.1 + (1.07 x ln soil zinc).

Table 6.14 Lead/zinc correlation coefficients by home area.

HOME AREA	CORRELATION COEFFICIENT (r =)
Back	0.94
Front	0.89
Side	0.97
Walkway	0.90

Table 6.15 Lead/zinc correlation coefficients by home location.

HOME LOCATION	CORRELATION COEFFICIENT (r =)
6 Agnes St	0.83
64 Edward St	0.92
58 Federatn. Rd	0.86
29 Fourth St	0.85
19 Lily St	0.89
3 Sabine St	0.93
34 The Terrace	0.95
27 Vera St	0.95
136 Warner. Rd	0.99
97 York Rd	0.89

Despite smaller sample size on a home by home basis, there still appears to be little variation in the relationship between soil lead and zinc concentration. Further, for both lead and zinc, the SD of soil element data for the individual homes is lowest for Fourth St and highest for Warnertown Rd, with the corresponding lead and zinc SD figures appearing to be very similar for all homes. This infers that the potential for variability in both lead and zinc home soil levels is clearly related. Consistency of elemental association in soil when data is separated according to district, home area or by home reflects that home soil detected lead and zinc are intimately related at home level, and also in the context of contaminant source. This concept of commonality of origin for these two contaminant materials further supports the theory of a pool of contaminants emanating from a source(s) and spreading throughout the township on an ongoing basis. Although these data confirm a close relationship between home soil detected lead and zinc with respect to possible source, they don't actually contribute to a closer understanding of the actual source(s) of the contaminant material, this providing avenue for future research.

6.5.7.2 SOIL LEAD AND ZINC RATIOS

Comparisons of bulk surface soil lead/zinc ratios by district and home area appear in Tables 6.16 and 6.17 below. Soil lead/zinc ratio home by home has been previously referred to (Table 6.2).

Table 6.16 Lead/zinc soil ratios for total data suite and by district.

LOCATION	MEAN SOIL LEAD/ZINC RATIO
Total	0.54
Pirie West	0.54
Solomontown	0.53

Table 6.17 Lead/zinc soil ratios by home area.

HOME AREA	MEAN SOIL LEAD/ZINC RATIO
Driveway	0.50
Back	0.53
Front	0.55
Side	0.51
Walkway	0.54

Certainly different houses would have had different soil treatments by owners over the preceding years, with new, uncontaminated top soil cover to a depth of at least 15cm having been provided at one of the houses as part of the recent lead intervention program. The penetrating depth of the x-ray source is estimated to lie between one and two centimetres only (Appendix Thirteen), yet there would appear to be only slight variability in the (surface) lead/zinc ratios from different areas around the homes, as also from the two different districts, despite moderately large variability in home area soil lead and zinc concentration.

Comparison of lead/zinc ratios for home soil in the Port Pirie West district vs the Solomontown district (*t*-test), reveals no difference ($P \geq 0.85$), implying common sources for home soil contaminants in the two districts. This is as also reflected by the above regression comparisons of soil lead and soil zinc concentration by district. Similarly, comparing the distributions for soil lead/zinc ratios by home area revealed no difference for any home area permutation comparison ($P \geq 0.20$).

Comparing the distributions for soil lead/zinc ratio data (by home) revealed of the 45 home/home comparison permutations, 16 revealing difference ($P \leq 0.05$). This variation in lead zinc ratios on a home by home basis is not unexpected, since it reflects modestly differing contamination deposition patterns, and/or potential individual home-owner recent activity in modifying, to some extent, the soil environment of the home. It may also reflect modest sample size.[15] Acknowledging the earlier, common practice of soil augmentation by use of smelter-derived material (typically slag), however, the lead content of which would be expected to be substantially lower than the zinc content (Section 6.5.2), it seems unlikely that the source of these currently-detected surface soil contaminants is from that earlier era. Subsequent to the earlier practice of fill importing in homes, changes to home soil resulting from occupant activity, differential weathering and chemical species mobility would suggest greater ratio variability from site to site than is indicated here. The currency of surface soil contamination at home level is again implied.

6.5.8 CONCLUDING REMARKS

Very similar lead/zinc ratios exist for all areas of the home despite differing degrees of soil contamination, and the absence of any soil contaminant gradient effect apart from immediately adjacent to structures. Acknowledging this and facing vs facing-away contamination differences, it seems quite reasonable to deduce that soil contamination is more an on-going process rather than an historical pattern, since it would be presumed that given the latter alternative, more lead/zinc variability would be anticipated as a result of occupant activity. Further, these data would also tend to suggest that such soil contamination would not be substantially related to home leaded-paint attrition, since greater variability in lead/zinc ratio would be expected with the use of different paints in different homes (M. van Alphen, pers. comm.). Clearly, soil lead and zinc are intimately related in the contaminating source. The similarity of facing vs facing-away home soil lead concentration differences for the two different districts further supports the concept of the on-going nature of contaminant emissions, and suggests that the source is proximate to the smelting works.

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15 Certainly future research in this direction should focus on a larger number of samples per home, and also a larger number of homes with a greater range of distance from the smelter.

Contaminated material such as dust would appear to be most likely to collect in somewhat protected areas around the home, eg the walkway. In all homes studied, windows opening onto such walkways were from bedrooms, and also occasionally from kitchens and lounge rooms, all rooms where most of a child's indoor time would be anticipated to be spent.

Whilst only a preliminary study, this thesis component appears strategic in eliciting the need for further research pertaining to home soil contamination and its relationship to (say) home dust contamination. This is particularly so for rooms that display potential entry points such as doors and windows proximate to (presumed) more highly-contaminated home areas such as walkways. This is pursued later (Chapters Seven and Eight).

CHAPTER SEVEN WITHIN-HOUSE DEPOSITED DUST

"Dust is the great poison of the factory worker, whether flour, animal, plant fibres or coming from minerals, stone, lime, coal or metal."

Charles Thackrah, 1832

So they rode on above the dust, in which they were writing their own legend.

Patrick White

7.1 INTRODUCTION

For lead-smelting communities where inhabitants reside proximate to smelting works, the risk of contamination is reported to be greater than for those residing further away (Harper et al., 1987). For children particularly, this health risk derives principally from lead-containing house dust originally deriving from smelter-produced environmental emissions (cited extensively above).[1] Considering the ready availability of house dust under most situations, infants' mouthing habits and their propensity to spend most of their time indoors on floors (Thompson et al., 1973; Barltrop et al., 1975; Lefcoe and Inulet, 1975; Binder et al., 1976; Solomon and Hartford, 1976; Budiansky, 1980; Harrison et al., 1981; Goyer, 1986; La Goy, 1987, Needleman et al., 1990a; Calabrese and Stanek, 1992a), a study of the dimension of contamination of house dust and its spatial variability is warranted.[2]

Port Pirie presents particular concerns here, given its smelter presence, open concentrate-receiving and storage locations until late 1997, history of smelting lead for greater than a century, current works' practices, and climate conducive to the generation and transportation of dust. From preliminary work undertaken a decade ago, it became clear that the concentration of lead in the majority of house dust samples exceeded a level of 500$\mu g.g^{-1}$ (Body, 1986d; Dolan, 1986a), a level indicative of contamination (EPA USA, 1986). Currently, transport mechanisms for such lead-bearing dust into the domicile are not well understood (Maynard et al., 1993), hence warranting the work undertaken below.

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[1] Over the last 20 years, as reflected by the literature, there appears to have been much antipodean opinion and published material relating to the relative contribution of the various sources of human body lead burden. There does appear to have been, however, a significant degree of consensual arrival during the mid 1980s, with lead-contaminated house dust in an urban environment being accepted as constituting a significant contribution via the hand mouth route (Chapter One).

[2] From observational studies conducted in Port Pirie, children spend approximately 75% of their time indoors in the domicile, 5% outdoors, and the remainder away from their home (L. Weih, pers. comm.). From a review of other studies, Mulik and Lewis (1988) note that an average (adult) person spends an estimated 50 - 90 percent of time indoors, and a one to two year old child even more (Mulik and Lewis, 1988).

7.2 RESEARCH OBJECTIVES

Sampling from various locations within vacant residences in Port Pirie was conducted in order to examine existing, home-contamination patterns, incorporating domestic dust and lead mass loadings, the dust samples' lead concentration values and their spatial variability.

Little is known regarding the spatial variation of dust lead contamination within houses and rooms of houses (Duggan and Inskip, 1985; P. Body, pers. comm.; B. Seifert, pers. comm.). Thus, in order to facilitate a comprehensive domestic residence dust contaminant study, a sampling strategy should incorporate multiple within-room sampling from a number of rooms. Further, such a study should also pay particular attention to such variables as the location of a sampling area within a room, its relationship to room features such as doors and windows, and the spatial relationship of a room to the lead-smelting works, typically much of which has not been previously undertaken.[3]

Significant spatial variability of lead-in-dust concentration and lead mass loading levels from dust deriving from various dwellings, both within houses and also between houses is presumed to occur (Duggan, 1984; Duggan and Inskip, 1985; Duggan et al., 1985; Body 1986d). No previous examples of multiple ($n > 50$) dust sampling from houses in Port Pirie (or elsewhere) have been noted, as has the near absence of multiple house dust lead mass-loading data provided by any previously-conducted research.

Earlier work conducted in Port Pirie (Body, 1986d; Body et al., 1988) acknowledges an absence of data relating to lead loadings and a paucity of data pertaining to indoor levels of dust-lead contamination near house features such as external doorways to complement data mainly collected from window-sills.[4] Body et al.'s assumption (var. refs) that little or negligible lead-bearing material enters houses with humans or by mechanical action (as opposed to wind or air movements) is not strongly supported by any of his work, and in fact is seriously contested by other workers eg Graziano et al., (1990). Conducting research in vacant houses will preclude the need to account for the component of contaminant entry into homes that arises from this pathway. Body et al. (1985), however, clearly indicate the pressing need to evaluate the spatial distribution of lead-contaminated dust within homes.

It would thus appear desirable to acquire a substantially larger number of dust samples from each studied house compared with earlier-conducted studies incorporating perhaps only three to five lead-in-dust concentration figures per house. Such a strategy should provide a more complete "picture" of the extent and distribution of residential contamination, and in combination with further work (Chapter Eight), an indication of the pattern of contamination of individual houses.

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[3] Duggan (1984) muses over the potential spatial variability of dust lead concentration within rooms of houses, suggesting multiple sampling locations to compensate for this, such an approach thus not precluding the obtaining of representative data.

[4] Of 10,810 samples taken for analysis, 6,544 (61%) derived from flaking/chalking paint, 3,071 (28.4%) were constituted by soil/aggregate and only 1,055 samples (9.8%) were dust samples (lead in dust concentrations only). Of these, 699 (6% of total) were collected from within 223 houses, the remainder deriving from school and public buildings. Of these house-derived samples, 24% derived from above ceiling space dust, 41% from window-sill dust and only 6% from floor locations. Acknowledging the need to evaluate dust lead loadings in homes additional to dust lead concentration estimation, Body approaches this in later work (Body, 1986f), although here again, far too few samples are collected. He also concludes that dustfall/leadfall in rooms are factors having been hitherto ignored largely, and areas where subsequent focus should be directed (Chapter Eight).

7.3 METHODOLOGY

Such multiple in-house dust sampling, as has been outlined below, incorporates mainly floors (including different floor coverings) given that such areas would provide the most likely avenue for child leaded dust acquisition (Que Hee et al., 1985; Bornschein et al., 1987). Elevated horizontal and vertical surfaces such as (internal) window-sills and window wells, and walls were also sampled from in an identical fashion (Appendix Fourteen). This was conducted in order to provide data corresponding to suggested potential entry points for contaminated dust (Thompson, 1993) and also to house locations where contamination would be less likely to arise as a result of (earlier) foot tracking.

Sampled surface area for walls was typically the same as for floors, 0.093 square metres, although for window sills and window wells, it was the entire surface area available.

Given (presumed) sampling constraints in occupied houses resulting from positioning of furniture, reluctance of owner/occupiers to provide access to all areas of the domicile, and concerns regarding potential for sampling bias, it was decided to assess the spatial variability of lead-contaminated dust in vacant homes only.[5] Such studies were conducted within approximately three months of occupant departure from their respective residence, and with all houses studied within one twelve month period of time. From time of occupant departure until investigation, all house windows and external doors were kept closed, virtually no entry into the residence occurring.

7.3.1 SAMPLE ACQUISITION

All dust samples were acquired using a specially designed vacuum pump drawing air (and dust) through a stainless steel sampling wand and a 37mm closed face aerosol monitoring cassette with enclosed 0.8μ cellulose ester filter, where the dust was retained (Plate 5, page 238). Such a filter type is a (US) National Institute for Occupational Safety and Health (NIOSH)-suggested standard for sampling lead in dust in air (Ashton and Gill, 1992; Gelman Sciences, 1993).

Analogous sampling using a similar technique has been occasionally used by other workers previously (eg Angle and McIntire, 1975; Solomon and Hartford, 1976; Caffo et al., 1980; Turner et al., 1980; Brunekreef et al., 1981; Diemel et al., 1981; Milar and Mushak, 1982; Que Hee et al., 1985; Aschengrau et al., 1991). For this work, however, a number of modifications have been made by the researcher, a more detailed description of the instrument used appears in Appendix Nine.

Such a technique provides a relatively convenient and speedy means for acquiring multiple house dust samples (incorporating lead loading data), and contrasts earlier views suggesting that establishing lead

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[5] Further, given that future work would be directed toward determining rates of (dust) contamination of various areas within houses, work which would entail the positioning of dustfall sampling devices in various areas within the domicile for considerable periods of time (Chapter Eight) and assuming no interference with sampling devices, it was again considered appropriate to conduct this research in a suite of vacant houses. Such a choice of sampling site complements a planned subsequent contemporaneous static dust/dustfall study conducted in houses post lead clean-up.

loadings on carpets is too difficult to achieve (eg Body et al., 1985). Further, multiple room sampling with such a technique will avoid non-random or inconsistent sampling problems associated with soft brushing of dust into containers or vacuum cleaner bag grab samples (Roberts and Ruby, 1989; Gulson et al., 1993), and also be more sensitive to detecting spatial variability of room contamination (Que Hee et al., 1985).[6]

From the work of others utilising dust sampling systems not dissimilar to that used for this work, much better correlation between resident child blood lead levels and house dust contaminant levels have been demonstrated for dust lead loading rather than dust lead concentration (Lanphear, 1994).

7.3.2 SAMPLE ANALYSIS

All analytical work was conducted according to a predetermined analytical protocol incorporating a strict QA/QC regime. Features of these are discussed in detail in Appendices One and Three.

Briefly, cassette-contained dust was removed and processed with a two acid digest, namely Hydrochloric and Nitric. Such solubilised dust samples and controls were analysed for lead as one of a suite of nineteen elements using Inductively Coupled (argon) Plasma Atomic Emission Spectroscopy (ICP AES).[7]

7.3.3 SAMPLING DETAILS

A total of 528 cassette-collected dust samples deriving from ten houses in two different districts of Port Pirie (seven located in Port Pirie West and three in Solomontown) constituted the main body of this work, with a further approximately 50 cassettes constituting a house post-clean-up study.[8] Quite apart from more effectively elucidating (within-home) spatial contamination variability than by simply determining a handful of dust lead concentration values, multiple sampling was also selected in order to acquire lead loading data. The literature suggests that the variability in floor dust lead loading appears to be "much greater than the observed variability of lead in dust concentrations", (Bornschein et al., 1986; p. 324). Thus, a lead loading estimate from only one or two house locations would not be sufficiently representative of the contamination patterns within that house.

Dust samples were acquired in various locations from a total of 58 discrete rooms from these ten houses, comprising the (typically) four main rooms, hallway, kitchen and (back) porch. Samples were not collected from bathrooms and/or laundries since some houses had these rooms incorporated in discrete

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[6] Gulson et al. (1993) remain unconvinced as to the efficacy of using dust samples derived from domestic vacuum cleaner bag grab samples, given the unknown sampling history of the implement's use and its dust particle capture efficiency.

[7] For the purposes of this work, virtually all focus is directed towards lead, although as will be seen below, for comparison purposes in one section of the work, corresponding analyte in dust values for three other elements will be investigated.

[8] A further, approximately, 170 cassettes were incorporated into a preliminary validation program (Appendices Fourteen and Fifteen). Apart from the work by Que Hee et al. (1985), as Roberts and Ruby (1989) very little work has been undertaken to validate dust-collection techniques employed for contamination studies (pers. obs.).

structures attached to or separate from the main dwelling, whilst others had such rooms as (lean-to) additions to the main structure, there existing too much variability in such room detail from a sample of only ten houses for reliable comparisons to be made.

Numerical data collected included dust loading (gm dust per square metre of floor area), lead loading (μg lead per square metre of floor area) and dust lead concentration (μg lead per gram dust [$\mu g.g^{-1}$], equivalent to parts per million [ppm]). Information was also collected regarding house district location, floor surface material and condition, room type, presence/absence of a fireplace in the room, spatial relationship to smelter works, location in the room from where the sample derived and proximity to features such as doorway, window, fireplace etc.

From an initial visual inspection of the dust loading, lead concentration and lead loading data graphically, there appeared to be a clearly-evident, considerably non-normal positive skewness of the distribution, log-transformation of the (raw) data suite being undertaken in order to more closely resemble a normal distribution. All subsequent statistical interpretation was thus conducted upon log-transformed data.

7.4 STATIC DUST - SUMMARY RESULTS

A summary of the static dust data appear in Tables 7.1 - 7.3 below.

Table 7.1 Mean dust loading, total and by district.

	DUST LOADING ($g.m^{-2}$)		
STATISTIC	**TOTAL DATA (10 houses; n = 528)**	**PT PIRIE WEST DATA (7 houses; n = 352)**	**SOLOMONTOWN DATA (3 houses; n = 176)**
Mean	1.08	0.92	1.48
Median	1.24	1.05	1.59
SD	3.41	3.57	2.88
Skewness	-0.33	-0.22	-0.41
Kurtosis	1.70	1.18	3.94

Table 7.2 Mean dust lead concentration, total and by district.

STATISTIC	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$) TOTAL DATA (10 houses; n = 528)	PT PIRIE WEST DATA (7 houses; n = 352)	SOLOMONTOWN DATA (3 houses; n = 176)
Mean	4,249 ‡	4,004	4,789
Median	4,040	3,556	4,570
SD *	2.12	2.35	1.52
Skewness	0.38	0.51	-0.35
Kurtosis	2.70	1.90	1.87

* SD for lead in dust levels is usually not small (Hamilton et al., 1984).

‡ The criterion for the identification of lead-contaminated dust as set by the South Australian Minister of Health in 1984 is 500$\mu g.g^{-1}$ (Body et al., 1985).

Table 7.3 Mean dust lead loading, total and by district.

STATISTIC	LEAD LOADING ($\mu g.m^{-2}$) TOTAL DATA (10 houses; n = 528)	PT PIRIE WEST DATA (7 houses; n = 352)	SOLOMONTOWN DATA (3 houses; n = 176)
Mean	4,580	3,685	7,076
Median	4,933	3,786	7,083
SD	4.35	4.78	3.14
Skewness	-0.21	-0.06	-0.08
Kurtosis	2.01	1.72	3.08

7.5 PRELIMINARY OBSERVATIONS

For the suite of house dust samples, regressing cassette-contained lead (dependent variable) against cassette-contained dust, demonstrated a correlation coefficient (r) of 0.87 ($P \leq 0.001$), with an identical

figure for dust and lead loading (quantity per unit area) values. The latter is in good agreement with a figure of (r =) 0.873 demonstrated for Christchurch house dust (Kim and Fergusson, 1993). These authors demonstrate a correlation coefficient between the dust lead loading and dust lead concentration of (r =) 0.51 ($P < 0.001$), again similar to a figure from this work of (r =) 0.56 ($P \leq 0.001$), both data sets demonstrating no correlation between lead concentration and dust loading.

Such excellent correlation between these parameters confirms that the dust sampled in this work is a very good indicator of the degree of contamination (loading) of a location, the source of the dust and the source of the lead being closely related here.

Regressing cassette-contained lead (dependent variable) against cassette-contained dust for the entire data suite produces the following regression relationships:

ln contained lead = 1.2 + (1.05 x ln contained dust), and for loading values

ln lead loading = 8.4 + (1.04 x ln dust loading).

7.6 CONCERNS REGARDING OPERATOR DIFFERENCES

For the entire suite of cassette-collected dust samples, two workers were involved. Each used only one sampling pump set-up throughout, and each operator collected samples from all homes involved in this study. Collected sample numbers from each individual home were split approximately equally between the two workers. Typically, each worker would collect samples exclusively from one room apart from one or two samples which were collected in that same room by the other operator for validation purposes.

For the main data suite, mean dust loading, lead concentration and lead loading values for each operator are as follows (Table 7.4).

Table 7.4 Operator dust collection comparison - all data.

		GROUPED MEAN DATA		
OPERATOR	**n**	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
A	**299**	1.1	4,323	4,833
B	**229**	1.0	4,155	4,269

There was no apparent difference between the results for the two operators for any of the above dust parameters ($P \geq 0.30$), hence the data suite was considered as one set. Throughout the study, paired adjacent samples were acquired from the centre of various room floors from the test house suite, each operator collecting one of these samples.[9] Statistical comparison of these results appear below (Table 7.5).

Table 7.5 Operator dust collection comparison - adjacent floor areas.

		GROUPED MEAN DATA		
OPERATOR	**n**	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
A	**22**	1.4	4,193	5,866
B	**22**	1.5	4,064	6,112

Using matched pair *t*-test comparison, no real difference between operators was detected for any dust parameter ($P \geq 0.30$). Regressing the data for the individual operators for dust loading, dust lead concentration and lead loading revealed correlation coefficients (r) of 0.84, 0.93 and 0.95 ($P \leq 0.001$) respectively, suggesting that good agreement existed between the data sets corresponding to the two operators. Such a result is in keeping with the work reported by Lanphear (1994), who noted that dust lead measurement variation resulting from technician differences was negligible ($P \geq 0.18$).

7.7 DISTRICT DIFFERENCES

Data derived from houses located in one of two districts, namely Port Pirie West (n = 7 houses) and Solomontown (n = 3 houses).

Mean district dust loading, lead concentration and lead loading values are as indicated in Table 7.6.

[9] It is acknowledged that there may be differences between adjacent centrally-located squares of carpet with respect to dust and lead loading. Such adjacent-square spatial variability is presumed to be small by comparison with within-room or between-room variability. Preliminary validation investigations prior to the commencement of the main study revealed no apparent differences between operators with respect to dust and lead collected and concentration of lead in collected dust when side by side comparisons were made with test dust-contaminated pieces of new carpet ($P \geq 0.15$) in a laboratory setting (Appendix Fifteen).

Table 7.6 District differences in surface dust levels.

DISTRICT	n	GROUPED MEAN DATA: DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Port Pirie West	**352**	0.92	4,004	3,685
Solomontown	**176**	1.48	4,789	7,076

Comparison of the various categories reveals no difference between the two districts for dust lead concentration ($P \geq 0.075$), but district differences for dust and lead loading are evident, Solomontown demonstrating substantially greater dust and lead loading mean values than applies for Port Pirie West ($P \leq 0.05$). From these data, Solomontown is a dustier locale than the other district, although it appears not to be affected by a substantially different dust population.[10] Correlation coefficients for lead loading (dependent variable) with dust loading were respectively for Port Pirie West and Solomontown, $r = 0.84$ and 0.93, regression relationships for the two districts respectively, being

$$\ln \text{ lead loading} = 8.3 + (1.03 \times \ln \text{ dust loading}), \text{ and}$$

$$\ln \text{ lead loading} = 8.4 + (0.97 \times \ln \text{ dust loading}).$$

Applying analysis of variance (ANOVA), controlling for dust loading reveals no difference between lead loading levels for Port Pirie West and Solomontown ($P \geq 0.75$).

Whilst the number of samples deriving from Solomontown approximates half that deriving from the other district, it should be noted that the number of residences from which these samples derive is also small by comparison. Thus, it may be that the contribution to overall mean (dust and lead) loading values made by (say) one of these three houses may be responsible for the statistically-significant differences between the two districts. This will be investigated below (Section 7.10).

[10] Essentially, differences in mean lead loading levels between the two districts are explained by differences in dust loading levels between the two districts. Clearly district-based dust contamination level differences appear not to be substantial.

7.8 MAIN ROOM DIFFERENCES

The four main rooms of the test houses were categorised as either front or back (ie facing the street address of that house - called front rooms) or those rooms behind the front rooms (back rooms). These pairs of rooms were further divided into Left or Right, corresponding to their spatial location (left or right of the central passage as viewed from the street frontage). All rooms displayed one sash type window. A summary of all data corresponding to each of the four main rooms appears in the table below (Table 7.7).

Table 7.7 House room contamination comparison.

		GROUPED MEAN DATA		
ROOM LOCATION	**n**	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
Left Front	**96**	1.26	4,110	5,183
Right Front	**91**	1.0	4,687	4,668
Left Rear	**86**	0.64	3,342	2,149
Right Rear	**91**	1.10	3,686	4,069

Such summary data reveal that the two front rooms appear more contaminated than the two rear rooms, and that for all three dust parameters, mean values are lowest for the left rear room. This will be further investigated in a subsequent chapter (Chapter Eight). Comparison of these data reveal for grouped left side of house rooms no difference from right side of house rooms ($P \geq 0.10$). Between grouped front rooms and rear rooms, however, difference does exist for dust loading, dust lead concentration and lead loading ($P \leq 0.025$), front rooms demonstrating the greater mean dust parameter values.

Further comparison of these data reflects no difference for any dust parameters between the two individual front rooms ($P \geq 0.10$), modest difference existing, however, between the two rear rooms for dust and lead loading ($P \leq 0.05$). Comparison of right front room with right rear room reveals no difference for any parameter ($P \geq 0.05$), although comparison of left front and rear rooms displays difference for dust and lead loading, the rear room exhibiting lower levels ($P \leq 0.002$). Similarly, comparison of left front and right rear rooms reveals no difference ($P \geq 0.20$), although for left rear and right front rooms, difference is noted for dust loading ($P \leq 0.05$), and dust lead concentration and lead loading ($P \leq 0.002$). Such room location contamination comparison for only floor-derived samples reveals virtually identical results with respect to difference between categories.

7.9 ROOMS FACING VERSUS FACING AWAY

Dust samples were collected where practicable from all rooms of test houses, all houses except 97 York Rd having front yards directly facing the smelter works, or facing away, the classification terms facing and facing away, respectively, being applied.[11] The eastern side of the York Rd house directly faced the smelter, and was termed as facing. Thus, main rooms of all test houses were described as either facing or facing away (from the smelter). Of the ten test houses, five exhibited front rooms facing the smelter and four had back rooms facing the smelter.

Mean dust loading, lead concentration and lead loading values for each house aspect are as indicated in Table 7.8 below.

Table 7.8 House/smelter aspect comparison.

		GROUPED MEAN DATA		
HOUSE ASPECT	**n**	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
Facing Smelter	**271**	1.14	4,924	5,592
Facing Away	**248**	1.01	3,567	3,594

Compared statistically (*t*-test) there exists no difference in dust loading between either aspect ($P > 0.20$), although a very substantial difference exists between the two house aspects for both lead in dust concentration and also lead loading, the facing aspect reflecting the greater mean level ($P \leq 0.001$). Given these differences and noting that dust loading has been well-correlated with lead loading for the data suite, it appears unusual that no difference in dust loading between either aspect can be demonstrated.

The facing aspect of test houses reveals, similarly, a greater mean value for dust zinc concentration and zinc loading ($P \leq 0.05$). Such a difference has also been observed for lead and zinc levels in home soils ($P \leq 0.002$; Chapter Six). It will be recalled that for copper and iron (presumed non-smelter products), no difference in soil element concentrations for facing versus facing-away aspects of the home soil was apparent, albeit soil copper levels lying close to (soil-analysis) instrument quantitation limits ($P \geq 0.20$; Chapter Six). For static house dust data, difference for dust copper concentration and dust iron

[11] Although this work focuses upon the metal lead, this section will also discuss metal in dust levels for three other elements, Zinc, Copper and Iron, for reasons that will be explained below. All details regarding data sets for these analytes are as stands for lead ie concentration and loading levels appear log-normally distributed, consequently log-transformed data is used throughout and there exist no operator differences for metal loading per sample or its dust concentration for any of these other three elements mentioned ($P \geq 0.25$).

concentration exists ($P \leq 0.05$), the facing aspect demonstrating the greater mean level for such dust parameters.[12]

Splitting the dust data for facing versus facing-away rooms of the house into floors and non floor subsets reveals for floor-derived samples, no difference for any of the dust parameters for any of the elements ($P \geq 0.20$) with the exception of dust lead, zinc and copper concentration, these being greater for facing locations ($P \leq 0.05$). Further, for other than floor-derived samples, difference exists for dust contaminant concentration and loading for both lead and zinc ($P \leq 0.05$). Mean values for all dust parameters are greater for facing rooms than for facing-away rooms. By contrast, dust copper and iron data revealed for facing versus facing-away (non-floor) room data, no difference for dust copper and dust iron concentration ($P \geq 0.05$), and no difference for dust iron loading ($P \geq 0.05$) although for facing rooms' dust copper loading a greater mean value was noted ($P \leq 0.05$).

Rooms facing the smelter are more contaminated than those facing away. These floor/non-floor differences here suggest that entry of the contaminants would be by other than foot traffic means. It is possible, however, that such foot traffic constitutes, subsequent to dust entry, an effective house floor dust mixing mechanism. Such results would seem to be reasonably consistent with the facing versus facing-away aspects of soil-derived data (Chapter Six), with the possible exception of the non-floor (dust) copper loading data. These may be artefactual, or may possibly relate to an internal copper source such as deriving from the use of a vacuum cleaner (Koutrakis et al., 1991) or a kerosene heater (Koutrakis et al., 1992).

The data regarding smelter-facing main rooms demonstrating greater levels of contamination than those facing away (for dust lead concentration and lead loading at $P \leq 0.001$) appear somewhat at variance with the data suggesting moderately greater levels of contamination for front rooms as compared with rear rooms (Section 7.8). The latter data, however, may simply be masquerading as a (modest) surrogate for the former, in that of the houses either facing the smelter or facing away, more than half their number directly face the smelter. Hence the front versus rear room results reflecting greater lead loading than rear rooms may have actually only been leveraged by the disproportionate balance between facing and facing-away houses. To pursue this, front main rooms of these test houses were split into those facing and those facing away, this also being conducted for hallways and also for grouped rear porches and kitchens, eliciting the following results (Table 7.9).

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[12] It was noted by Yocom (1982) that wind produces pressure differences between the windward and leeward side of the house, and, depending upon the location of the windows and doors, can have a profound effect upon air exchanges within the structure.

Table 7.9 Further house/smelter aspect comparison.

FRONT MAIN ROOM ASPECT	n	GROUPED MEAN DATA: DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Facing Smelter	**96**	1.05	5,230	5,472
Facing Away	**136**	1.28	3,190	4,081

Dust loading appears to be greater in facing-away rooms, yet in smelter-facing rooms, lead loading is greater, although neither difference is particularly impressive ($P \geq 0.25$). Dust lead concentration, however, is greater in facing smelter main rooms ($P \leq 0.001$), this also being the case for zinc ($P \leq 0.01$). Analogous comparison for hall contaminant levels (facing versus facing away) appear somewhat inconclusive, demonstrating greater mean dust loading again for facing-away hallways ($P < 0.01$), with dust lead (and zinc) concentration registering only slightly greater mean levels in facing hallways ($P \geq 0.10$), although sample size is not large. Comparison of grouped kitchen and rear porch samples provided similar results with slightly greater dust loading levels in facing-away locations. Only marginally greater mean lead loading levels were detected in facing applications ($P \leq 0.125$), but substantially greater dust lead and zinc concentration mean values were noted in facing locations ($P \leq 0.05$).

What these data are suggesting is that, in general, the smelter-facing aspects of houses reflect a greater effect of lead contamination than analogous facing-away aspects, this effect being a potential contributor to the observed difference between front and rear room mean contaminant levels as noted above.

7.10 HOUSE BY HOUSE COMPARISON

Acknowledging that the test houses are located in various sections of two districts in Port Pirie, differences in levels of dust/lead loading and lead concentration on a house by house basis was examined.

One of the ten houses (29 Fourth St) appears to be substantially dustier than the others, this also translating to a higher lead loading. This may be artefactual given the small sample size, although it may also reflect previous occupant cleaning regimes; this particular house appeared to be the least well-maintained of the test house suite. Differences in dust and lead loading figures between the different houses may reflect normal variability and/or former occupant cleaning efficacy. A table of mean values for dust loading, lead loading and dust-lead concentration by house appears below (Table 7.10).

Table 7.10 House by house contamination comparison.

HOUSE	n	DISTRICT	GROUPED MEAN DATA		
			DUST LOADING ($g.m^{-2}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
6 Agnes St	**74**	Pirie West	1.21	2,513	3,029
64 Edward St	**38**	Solomontown	0.54	5,211	2,796
58 Federation Rd	**32**	Pirie West	1.33	9,431	12,567
29 Fourth St	**7**	Pirie West	6.73	3,584	24,103
19 Lily St	**61**	Pirie West	1.50	4,477	6,724
3 Sabine St	**56**	Solomontown	2.49	3,844	9,563
34 The Terrace	**71**	Pirie West	0.35	5,216	1,821
27 Vera St	**72**	Pirie West	0.64	3,235	2,054
136 Warnertown Rd	**70**	Solomontown	1.46	4,919	7,175
97 York Rd	**33**	Pirie West	1.73	3,675	6,350

Despite there being no controlling for time between former occupant departure and static study sampling, however, it may also reflect different rates of contaminant penetration into the house as may be indicated by (internal) dustfall variation between houses (Chapter Eight). Comparing such house by house (static) dust data (both all samples and, separately, floors only) with dustfall data (presented in Chapter Eight) for these same houses reveals mean correlation coefficients for all dust parameters (r) of less than 0.10 ($P \geq 0.15$). This infers little relationship between mean estimates of dustfall under varying conditions for these individual houses and the extent of their individual (static) contamination when first vacuum-sampled. Although such comparison failed to control for seasonal variation, the poor correlation suggests that the extent of existing house (static) contamination (a reflection of the preceding composite "picture" of contamination arising as a result of former occupant residence plus depositing dust addition subsequent to occupant departure) is not a particularly impressive indicator of the subsequent deposition of dust occurring during the depositing dust investigations. Further, substantially greater (order of magnitude) levels of dust/lead and zinc mean loading in these vacant houses as compared with similarly-located, identically-sampled occupied houses (L. Weih, pers. comm.) reflects static dust build-up during house vacancy periods. Such difference may allude to differences in contamination rate between vacant and

occupied, (presumably) reasonably-frequently cleaned, houses. In a later chapter (Eight), this concept will be pursued in some of these residences subsequent to a rigorous house-cleaning program, with contemporaneous sampling of depositing and static dust contaminant material.

Given the district differences regarding dust and lead loading referred to earlier (Section 7.7), and the possibility that the contribution to mean dust and lead loading values made by only one of the three Solomontown houses being responsible for the statistically-significant differences between the two districts, district comparisons were again made with the removal (separately) of the data corresponding to each of the three Solomontown houses. Removal (separately) of data for 64 Edward St (furthest house from the smelter works) or 136 Warnertown Rd (adjacent to a main road) does not affect the above-mentioned significant district differences for dust and lead loading ($P \leq 0.05$ level), although subsequent to (solely) removal of the data for 3 Sabine St, district differences exist for lead loading only. Such district differences in dust and lead loading as have been observed above (Section 7.7) may be real. Alternatively, such difference may simply be attributable to the small number of residences situated in Solomontown compared with Port Pirie West given the above findings which demonstrate the potential for the data to be "leveraged" by the addition/removal of the data corresponding to only one Solomontown house. A larger sample size (of houses), particularly with more residences deriving from Solomontown, would assist in answering this definitively.

7.11 HOUSE CONTAMINATION WITH DISTANCE FROM SMELTER

Noting the range of mean values for the various parameters corresponding to the ten individual houses, it seemed appropriate to consider whether distance from (say) smelter works centre, or smelter boundary etc correlated with the level of dust contamination. Regressing (separately) dust loading, lead loading and also dust lead concentration (as the dependent variable) against the square of the distance of the sampling site from the various locations to the smelter works elicited the following coefficients (Table 7.11). Increasing distance from smelter correlates with increasing dust and lead loading as well as dust lead concentration, albeit rather poorly, ($P \geq 0.10$). Regressing the various dust parameters (dependent variable) against the distance from one of the smelter features to the sampling site, exhibited no less poor a correlation coefficient, and thus was not considered further.

Essentially, little inference can be drawn regarding the relationship between distance from smelter and degree of contamination of homes in the context of static dust and its potential source on the basis of these data. A substantially increased sample size of houses with a greater range of distances from the smelter may, however, be more helpful in this regard.[13]

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[13] Whilst the range of distances from the various sampling sites to the smelter centre ranged from 0.5 - 3.3kms (mean = 1.4kms), this may be insufficiently large to detect any contamination distance effects.

Table 7.11 Contaminant level variation with square of distance from smelter.

	CONTAMINANT LEVEL VERSUS DISTANCE CORRELATION COEFFICIENT (r =)		
	DEPENDENT VARIABLE		
INDEPENDENT VARIABLE	**DUST LOADING**	**DUST LEAD CONC.**	**LEAD LOADING**
SQUARE OF THE DISTANCE FROM THE SAMPLING LOCATION TO THE:			
Nearest smelter boundary	0.03	0.06	0.05
Wharf - concentrate unloading point	0.03	0.08	0.06
Smelter works central point	0.06	0.08	0.09
Smelter stack	0.05	0.09	0.08

Acknowledging that there exist differences in degree of contamination of homes from the test house suite, and that this would not appear to be related simply to distance from a presumed source, it must be accepted here that the house by house contamination differences are explained by normal variability, encompassing differences in effectiveness of contaminant penetration, vacancy period, climatic variability and former occupant cleaning activity. These test houses will thus be assumed to be representative of the houses located in the two test areas, although it is stressed here that they do not necessarily represent a random sample or representative sub-population.

7.12 CONTAMINATION DUE TO FIREPLACE PRESENCE

At least one room in nearly all of the houses included in this research contained a fireplace, the openings of some of which were closed over with boarding, whilst for others they were still patent. Fireplaces were typically located in one or sometimes two of the four main rooms of the dwelling, very occasionally the kitchen. Only floor-sampled dust deriving from one of the four main rooms or the kitchen

was included for this comparison, thus excluding porch and hall-derived samples, since fireplaces were never found in these locations.[14]

The data were initially split into four categories, corresponding to rooms containing no fireplace, a closed fireplace, a partially-closed fireplace or an open fireplace, however, small sample numbers for the partially-closed fireplace (ten samples from one room of one house) and the closed fireplace (33 samples from three rooms from three houses) categories suggested that they be classified as rooms with a modified fireplace.[15]

Mean values for dust and lead loadings and dust lead concentration are included in Table 7.12 below.

Table 7.12 Influence of a fireplace upon floor contamination levels.

	GROUPED MEAN DATA		
FIREPLACE STATUS	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
No fireplace (n = 265 samples from 30 rooms from 9 houses)	0.91	3,623	3,300
Modified fireplace (n = 43 samples from 4 rooms from 4 houses)	0.57	3,364	1,919
Open fireplace (n = 101 samples from 12 rooms from 7 houses)	1.51	4,500	6,811

For all three parameters, the mean value for rooms with an open fireplace is greater than for rooms lacking a fireplace; in effect, the existence of a fireplace in a room is an indicator for more substantially contaminated floor surfaces. In fact, statistical comparison between these two categories for the parameters dust and lead loading and dust lead concentration supports this, revealing that there exists clear difference for all three parameters between no fireplace rooms and rooms with an open fireplace ($P \leq 0.002$).

The data corresponding to the modified fireplace category appear contradictory to those corresponding to the absence or presence of an open fireplace, in that (modified fireplace-equipped) rooms display the lowest mean levels for all three parameters. This may pertain to the overall small sample size, a non random sub-sample, and/or also the fact that all rooms demonstrating a closed fireplace derive from a

[14] Dust sampled from locations such as window sills and window wells was not included in this group since it was presumed that windows were *per se* a potential source of contaminant entry into the room (as Charney, 1982b; Al-Radday et al., 1993a).

[15] The partially-closed fireplace had a poorly-fitting board placed over the central opening of the fireplace, a closed fireplace being (again) an ill-fitting board replacing/covering the cast iron fireplace surround, although probably offering only marginally better sealing (if any) than the former.

facing-away category, such (facing-away) rooms reflecting lower mean values for dust lead concentration and lead loading ($P \leq 0.001$) than for rooms facing the smelter works (Section 7.9).[16]

Given this apparent contradictory nature of the data corresponding to modified fireplaces and also given the small sample size for modified fireplaces, this feature class was not included in statistical interpretation, subsequent subset comparison being made between the absence of fireplaces and the presence of open fireplaces only.

From such comparisons for (separately) Port Pirie West and Solomontown-derived data, again all parameters demonstrate greater mean values for rooms with (open) fireplaces than those without. Such difference for Port Pirie West data is readily apparent for all dust parameters ($P \leq 0.05$), although for Solomontown, this is so only for dust lead concentration at this level ($P < 0.05$).[17]

Given above-noted differences in room contamination levels depending upon whether the particular room faces toward the smelter or away (Section 7.9), differences between rooms with open fireplaces versus those without fireplaces for rooms either all facing or all facing away from the smelter were investigated. For facing smelter rooms, mean values for the three dust parameters in rooms with an open fireplace were larger than for corresponding rooms with no fireplace ($P \leq 0.05$). For opposite-facing rooms, dust parameter mean values for rooms with an open fireplace were, again, numerically larger, substantially so for both dust lead concentration and lead loading ($P \leq 0.05$). Dust loading, however, demonstrated no such difference.

Thus, the presence of an open fireplace in a room is an indicator of a more contaminated room than a room devoid of a fireplace.

7.13 FLOOR SURFACE DIFFERENCES

Three different house floor surface materials were encountered in the sample suite. These included carpet, linoleum (lino) and wooden boards. All rooms studied exhibited the presence of only one floor surface covering type. A summary of mean data is included in Table 7.13 below.

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[16] It is also possible that this may relate to the presumption/observation by former home owners that the presence of a fireplace in a room is an indicator of a more dusty/contaminated room. This may have led to their attempting to close the fireplace's opening (particularly if the boarded up fireplace was in the child's bedroom), and possibly practising more rigorous routine cleaning of such rooms. It is possible that such action, they might presume, would result in reduced levels of contamination within the room.

[17] Caution need be applied when interpreting this result since sample size for Solomontown homes with fireplaces and those without are not large, and the data available (regarding fireplaces) may not be truly representative of apparent differences between different rooms. Acknowledging that it has not been demonstrated that the two districts are influenced by different dust populations (Section 7.7), choosing to split and analyse the data in this fashion was partially based upon the need to investigate whether small sample size in Solomontown would reveal some limitations with respect to conclusion-drawing in that district separately.

Table 7.13 Influence of floor surface on contaminant levels.

		GROUPED MEAN DATA		
FLOOR SURFACE	**n**	**DUST LOADING** ($g.m^{-2}$)	**DUST LEAD CONCENTRATION** ($\mu g.g^{-1}$)	**LEAD LOADING** ($\mu g.m^{-2}$)
Carpet	**361**	1.39	3,897	5,433
Linoleum	**97**	0.58	4,524	2,609
Wood	**23**	0.80	3,735	2,976

As Solomon and Hartford reported (1977), the quantity of dust and lead collected from carpeted surfaces here was greater than that acquired from wooden (or linoleum-covered) floors, the former trapping the dust and demonstrating more effective "reservoirs" of contaminated dust (Rey-Alvarez and Menke-Hargrave, 1987; Figley and Manohan, 1992; Bero et al., 1993; Kim and Fergusson, 1993).[18]

For all dust parameters, no difference between linoleum and wood surfaces is apparent ($P \geq 0.10$). Comparing carpet and linoleum, however, mean (carpet) levels are greater for both dust and lead loading ($P \leq 0.05$), but not for dust lead concentration (lower for carpets). Sample numbers for wooden floor sampling locations were not large, however.

Mean dust lead concentration for linoleum exceeds that for both carpet and wood ($P \leq 0.05$), thus it seems likely, for the comparison with carpet, that given linoleum's smooth surface, it was more effectively (or regularly/easily) cleaned previously, and hence retained less contaminated material than carpeted surfaces. Thus the state of contamination would appear to be more recent than (say) carpet, this being commensurate with linoleum-covered surfaces' substantially lower lead loading than carpeted surfaces.[19] Wooden surfaces may respond similarly, although typically wooden surfaces revealed more surface contour imperfections as compared with linoleum, and thus may have retained (some) more contaminant material, some of which might be expected to be of a larger particle size than that being deposited by (say) dustfall.

If data for the two categories wood and linoleum are grouped and termed as hard surfaces, mean values for the discussed parameters are as indicated in Table 7.14 below.

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[18] Further, Cannell et al. (1987) noted an approximately linear relationship between shoe contact area and contaminant transport capacity and that the rougher, more fibrous nature of a carpeted surface removes proportionately more of shoe-attached material than a smooth, impervious surface.

[19] Essentially, the suggestion here is that linoleum surfaces sampled here reflect more of a contribution to their overall contaminated state from (post occupant departure) depositing dust than (say) tracking, due to more frequent and/or effective home floor surface cleaning. This infers that the smaller-sized dust particles, as would be expected to arise from dustfall, demonstrated greater dust lead concentration than their larger-sized tracked-in particle analogues, the latter being, typically, of a lower mean lead concentration than the former (Chapter Five), and being more regularly/effectively collected by normal home cleaning regimes (eg sweeping) than would be the case for carpet.

Table 7.14 Influence of floor surface on contaminant levels - simplified version.

FLOOR SURFACE	n	GROUPED MEAN DATA		
		DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Carpet	**361**	1.39	3,897	5,433
Hard surfaces	**120**	0.61	4,361	2,676

The combined hard (floor) surfaces class demonstrates a lower mean dust and lead loading value ($P \leq 0.001$). The mean dust lead concentration for non-carpeted surfaces was, however, larger than for carpeted surfaces, albeit only moderately ($P \geq 0.10$), as Solomon and Hartford also reported (1977). Whilst little may be inferred regarding differences in the type of dust being collected by these surface types (ie with respect to dust lead concentration), it is clearly evident that carpeted surfaces are reservoirs for dust and lead (much more so than surfaces such as linoleum or wood). This contaminant material is, not unreasonably, thus termed "available", given the sampling technique used.

Interestingly, of eight externally-sampled concrete surfaces (typically front or rear doorway entrance locations immediately outside the door), the data reveal the lowest dust loading levels, yet the greatest dust lead concentration and lead loading levels of any surface ($P \leq 0.001$). In this situation, this is not so much saying something about the collection surface *per se* as it may be about the dust depositing and being collected from such an outside-house source. Mean levels are indicated below (Table 7.15).

Table 7.15 External floor (concrete) dust collection summary.

SURFACE	n	GROUPED MEAN DATA		
		DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Concrete	**8**	0.55	16,766	9,146

Whilst sample numbers are small, it may not be unreasonable to suggest that such dust represents one of the potential house-invading contaminating agents given its relatively high lead concentration. Dilution of such dust occurs with other differently-sourced dust within the house, retention of which appears likely on carpeted surfaces particularly (as Farago et al., 1994).

7.14 CARPET TYPE AND CONDITION

Carpeted surfaces from which dust samples were collected were not uniform in their appearance. Noted differences in carpet type pertained to carpet pile (being described as either short, medium or long) and also differences in the degree of wear demonstrated by carpeted surfaces, although, in general, this wear variability was rather small.[20] Carpet wear was described as ranging from Rating one (no detectable wear) to Rating four (patchy damaged carpet), although there were no examples of Rating four carpets in those areas of carpet sampled for this work, virtually all carpet sampled being classed as either Rating one or Rating two (evenly distributed compressed pile). Rating three was described as detectable differences in carpet condition between the best spots and that spot being sampled, although no holes were apparent and the fabric was not worn through to the matrix.

7.14.1 CARPET PILE DIFFERENCES

Mean values for dust parameters corresponding to different carpet pile lengths are included in Table 7.16 below.

Table 7.16 Influence of carpet pile on degree of dust and lead contamination.

	GROUPED MEAN DATA		
CARPET PILE	**DUST LOADING** ($g.m^{-2}$)	**DUST LEAD CONCENTRATION** ($\mu g.g^{-1}$)	**LEAD LOADING** ($\mu g.m^{-2}$)
Short (n = 84 samples from 10 rooms from 6 homes)	1.69	3,430	5,785
Medium (n = 169 samples from 16 rooms from 5 homes)	1.20	4,852	5,826
Long (n = 108 samples from 11 rooms from 6 homes)	1.52	3,053	4,637

[20] On only two occasions, substantial localised wear was detected in a carpeted surface at one particular spot displaying wear through to the matrix of the carpet (both close to a doorway entrance). Such areas were not sampled, an adjacent area being chosen instead. In general, the carpeted surfaces from the rooms of the houses appeared surprisingly evenly and slightly worn.

These data do not appear to suggest any trends in any of the variables with differing pile depth. Of the nine test houses that had more than one room carpeted, all had more than one different type of carpet present, several houses having three different carpet types present. Examining the data for (separately) short, medium and long pile carpet split into district differences also does not suggest any relationship between carpet pile and any of the three dust parameters. Further, dust loading values are substantially greater in Port Pirie West for short pile carpet, yet greater for Solomontown medium and long pile carpet ($P \leq 0.05$).

Mean dust lead concentration was greater for the three carpet pile categories in Solomontown, notable only though for short and medium pile ($P \leq 0.05$). Lead loading mean values were greater in Solomontown for medium and long pile carpet, but lower for short pile carpet ($P \leq 0.05$).

Similarly, examination of the (carpeted floor) data on a facing versus facing-away basis reveals mean levels for all three parameters that are consistently greater for facing floors as compared to facing-away floors, but notable only for short pile dust and lead loading and also medium pile lead concentration mean values ($P \leq 0.05$). It will be recalled that for all surfaces, facing versus facing-away floors exhibit greater mean values for dust parameters. That this effect appears less detectable progressing from short to long pile carpets would suggest that sampling from longer-piled carpets would be more difficult/less consistent using the vacuum pump technique, this also being apparent from the results of Que Hee et al. (1985).

Controlling for both district and facing (smelter) aspect, there is again no consistent difference between short and medium pile carpet for any of dust loading, lead loading or lead concentration in either Port Pirie West or Solomontown. Further, for short compared with long pile carpet comparison, Port Pirie West reflects a consistently greater mean value for all three parameters for both facing and facing-away aspects for short-pile carpeted floor surfaces, all of which are substantially greater apart from facing-away dust lead concentration ($P \leq 0.05$). No analogous comparison can be made for Solomontown-deriving (short versus long pile) samples, there being no long pile carpet sampled from this district.

Acknowledging the small individual sample sizes when controlling for all variables, and the lack of any substantial evidence consistently reflecting a relationship between dust/lead levels and carpet pile dimension, these data reveal no real trends. This reflects a need for further investigation regarding the relationship between carpet pile and dust retained. Further, this work also cannot control for former occupant cleaning protocol and its relative efficacy, or the variability in dust-retaining potential between different carpet brands; again, a larger sample size would be helpful.

7.14.2 CARPET WEAR LEVELS

An increase in carpet dust loading (or the vacuum pump's ability to collect dust) from carpets displaying an increasingly worn appearance was noted, there existing substantial difference between Ratings one and two, two and three and one and three ($P \leq 0.05$). This is also the case for carpet lead loading ($P \leq 0.05$), although no such difference is apparent for analogous dust lead concentration data. A summary of contamination data corresponding to the various carpet wear rating indices is included in the table below (Table 7.17).

Table 7.17 Influence of carpet wear on degree of dust and lead contamination.

WEAR INDEX	GROUPED MEAN DATA DUST LOADING ($g.m^{-2}$ & RSD)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$ & RSD)	LEAD LOADING ($\mu g.m^{-2}$ & RSD)
Rating one (n = 205 samples from 26 rooms from 8 homes)	1.19 (509.9)	3,680 (7.0)	4,398 (11.6)
Rating two (n = 144 samples from 24 rooms from 8 homes)	1.52 (257.9)	4,220 (6.2)	6,423 (14.0)
Rating three (n = 12 samples from 6 rooms from 4 homes)	4.20 (41.9)	3,636 (5.3)	15,286 (7.9)

The sample size for wear Rating three carpets is not particularly high, consequently there is some potential here for an unrepresentative estimate for this category. Noting the large category differences in mean dust and lead loading figures particularly (and also the relatively low RSD values for the means of wear-rating three carpets), it would not appear unreasonable to conclude that available dust/lead increases directly with carpet wear. Insufficient data were available to examine such a phenomenon for different pile carpets, these conclusions applying to carpeted surfaces as a single group.

7.15 BETWEEN ROOM DIFFERENCES

All room to room comparisons below are conducted on the basis of floor samples only. Such floor-sampled data reveal high dust and lead loadings and also dust lead concentrations for halls, porches and (for comparison) also outside-door locations. *T*-test comparison between halls and porches reveals no difference for any of the three parameters ($P \geq 0.20$), which does not seem surprising given that both rooms represent main domicile entrance avenues for residents. Further, comparison reveals difference between halls and bedrooms and also halls and lounge rooms for all three parameters, hall mean values being the larger ($P \leq 0.005$). Not dissimilarly, porch mean parameter values are also high, and comparisons with bedrooms and lounge rooms reveal difference for dust lead concentration and lead loading ($P \leq 0.01$), but not as dramatically for dust loading ($P \leq 0.10$).

Summary values for dust and lead loading and dust lead concentration for all room types are included below (Table 7.18).

Table 7.18 Room to room differences in floor contaminant levels.

ROOM	n	GROUPED MEAN DATA: DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Bedroom	**286**	0.96	3,842	3,699
Hall	**47**	2.20	5,084	10,744
Kitchen	**14**	0.50	3,301	1,636
Lounge	**101**	1.16	3,497	4,053
Porch	**7**	1.80	8,420	15,190
Step outside house (by comparison)	**8**	0.55	16,766	9,146

No difference between bedrooms and lounge rooms exists for any dust parameter ($P \geq 0.10$). In one sense, this is not surprising given that these two room types typically constitute the four main rooms of each of the dwellings, and are very similar in dimension, shape, construction and condition (in individual houses) to each other.

Grouping data for bedrooms and lounge rooms as for halls and porches, mean values are as indicated below (Table 7.19).

Table 7.19 Room to room differences in floor contaminant levels - simplified.

ROOM	GROUPED MEAN DATA: DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Bedrooms and lounge rooms	1.01	3,749	3,788
Halls and porches	2.07	5,427	11,237

For (grouped) bedrooms and lounge rooms, there appear notably lower mean values than for halls, encompassing all three dust parameters ($P \leq 0.001$). When compared with porches, again this is the case for dust lead concentration and lead loading ($P \leq 0.001$), but for dust loading more marginally ($P < 0.10$). In both situations, mean levels for all three parameters are greater for porches and also halls than for (grouped) bedrooms and lounge rooms.

Comparison of grouped bedrooms and lounge rooms with grouped halls and porches reflects greater mean values for all dust parameters for the latter ($P \leq 0.001$). Comparison of halls with individual main rooms by position in the house (ie left front, right rear etc) and also grouped front main rooms and grouped rear main rooms reveals, for all permutations, numerically greater mean values for hall dust and lead loading ($P \leq 0.002$) and also dust lead concentration ($P \leq 0.04$).

Kitchens exhibit the lowest mean levels for all three dust parameters, although this may relate to small sample size (14) and/or all kitchens but two demonstrating the presence of linoleum or wooden floors. Contamination levels for such floor coverings have been reported to be notably lower than for carpeted surfaces ($P \leq 0.05$; Section 7.13).[21]

Such kitchen contaminant levels appear much lower (all three parameters) than for grouped hall and porch data ($P \leq 0.05$), and similarly also, exhibit smaller mean dust and lead loadings than for grouped bedrooms/lounge rooms ($P \leq 0.05$). This is not apparent, however, for a comparison between kitchens and grouped bedrooms/lounge rooms for dust lead concentration ($P > 0.15$).

Thus, certain rooms of the house, such as halls and porches, provide significantly greater opportunity for child contact with available dust and lead than do rooms such as bedrooms and lounge rooms, the levels of contamination in halls and porches inferring primary entry (and perhaps collection) points for contaminant dust.

7.16 WITHIN-ROOM DIFFERENCES - LOCATION IN (MAIN) ROOMS

An infant spends a significant amount of time within the house, this reflecting the potential for that child to make contact with leaded dust. Given the rooms in which it would be most likely for the child to spend considerable periods of time, ie one of the four main rooms (bedrooms or lounge rooms), it is appropriate to assess whether any foci of contamination may be detected, or whether a (dust) lead "gradient" actually exists in such rooms.[22]

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21 It may also be that such non-carpeted floors as are usually found in kitchens, are (presumably) more easily/effectively cleaned than carpeted floors. Lower contaminant levels in kitchens may also relate to these rooms having been perceived to be more deserving of a rigorous cleaning regime (by former occupiers) given that they are food-preparing areas.

22 Given that these rooms were typically considerably larger than kitchens, porches or halls, more cassette-collected dust samples were actually acquired from these rooms. This is also appropriate given that a child is most likely to spend most of its time in one of the four main rooms (L. Weih, pers. comm.).

Initially, sampled floor locations from all main rooms were assigned as deriving from three (floor) regions of the room, the centre of the room, the periphery of the room or from the mid-way position (Figure 7.1), mean data corresponding to which are presented below (Table 7.20).

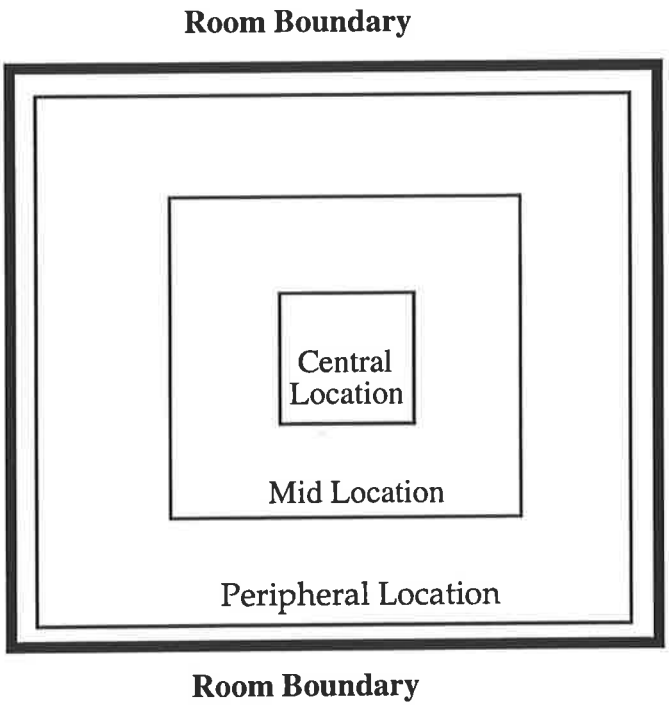


Figure 7.1 Various room regions classification.

Table 7.20 Within-room contaminant spatial variability.

POSITION SAMPLED IN ROOM	n	GROUPED MEAN DATA		
		DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Centre	**52**	1.08	3,573	3,843
Mid-way	**121**	1.06	3,612	3,394
Periphery	**239**	0.93	4,119	3,836

Statistical comparison of these data revealed no real difference between centre of room and mid-way, mid-way and periphery, and centre and periphery location categories for any of the three parameters ($P \geq 0.125$). Analysis of the same room region location categories when controlling for district revealed (for Port Pirie West) no real difference for dust and lead loading for all permutations ($P \geq 0.15$). This was also the case for all dust lead concentration comparison permutations ($P \geq 0.15$), except between centre and

periphery of room locations ($P > 0.075$). For Solomontown data, no demonstrable difference was detected between centre and mid, mid and periphery and centre and periphery location permutations for any of the three dust parameters ($P \geq 0.15$).

Controlling for facing versus facing-away aspects of the house revealed no major difference between mean values corresponding to any combination of centre, mid and peripheral sampling location for all three dust parameters for away-facing rooms ($P \geq 0.075$). Similarly for facing aspect rooms for all sampling location combinations and all dust parameters, no real difference was detected ($P \geq 0.075$), with only one exception, this being a numerically larger mean dust lead concentration value for peripheral as compared to central sampling location ($P < 0.05$).

Excluding rooms with an open fireplace from the full data set revealed no real difference between any of the dust parameters for centre and mid, and centre and peripheral room regions ($P \geq 0.10$). Difference, however, was detected between the mid and peripheral regions for dust lead concentration, the mean for mid-located samples being lower than for the peripherally-located samples ($P < 0.05$). Excluding (open) fireplace and doorway floor samples firstly, and then subsequently fireplace, doorway and window-base floor samples, however, all room region categories (Cent, Mid, Per) exhibited no difference for all dust parameters for any category permutation ($P \geq 0.05$).[23]

This result would appear to contrast work by Bornschein et al. (1986), who note much greater lead loading levels on floors proximate to walls and corners. These authors' research, however, pertained to work conducted in houses where the source of the lead in dust was related (in substantial part) to the paint on the walls of the residence. This is not the case in these test houses.

In rooms with an open fireplace, no substantial difference is apparent between centre and mid room locations for all dust parameters, and between mid and peripheral and central and peripheral locations ($P \geq 0.05$). Difference, however, was detected between central and peripheral locations for dust lead concentration, the mean value being greater peripherally ($P < 0.05$).

Thus it appears that there is no evidence to suggest that any of the three room regions are, consistently, any more contaminated than any other, although it may be possible that certain specific floor locations (eg proximate to doorways, window-base floors or open fireplaces) demonstrate elevated contamination levels.[24]

Floor surfaces adjacent to doorways, (open) fireplaces and windows are all classified as belonging to the peripheral sampling point location category. Presuming that these room features are all contributors to dust entry into a room, it would be expected that the peripheral sampling category might reflect substantially greater levels for some/all of the dust parameters. It is possible, however, that as there is minimal difference between peripheral and (say) central room region mean contaminant values, there must be some effective (floor) dust mixing mechanism operating. Alternatively, it could be postulated that such a room periphery-to-centre contamination gradient may in fact exist but be masked by lower levels for dust parameters in other peripheral locations such as (say) extremities of rooms (corners) most distant from these presumed entry points. Conflicting with this, however, comparison of corner of room locations

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[23] Doorway sampling area (0.0930 sq. meters) was selected as a (doorway) centrally-located floor area immediately inside the room, with the centre of the sampling area located 300mm into the room in order to avoid any edges or joins in floor surface material.

[24] Whilst it has been reported that major potential entry points for contaminated dust into a room are windows and/or doorways (Charney, 1982b; Al-Radday et al., 1993a), evidence above infers that fireplaces should also be included, at least insofar as to suggest that such features are indicators of more heavily-contaminated rooms (Section 7.12).

with other peripheral locations including and excluding (independently) fireplaces, window floors and doorways revealed no real difference for any of these categories for any of the dust parameters ($P \geq 0.10$).

No data are presented here to suggest that corners are disproportionately greater collection points for contaminated dust than any other particular room location, hence it cannot be concluded that dust collection at such locations masks what might otherwise be a definable contaminant gradient within the room. [25]

With no apparent within-room, large-scale contamination gradient yet demonstrated, it would appear that the within-room distribution of contaminated dust results from a complicated mechanism (possibly) involving deposition of airborne dust with potential entry points at windows, fireplaces, doorways and possibly even above-ceiling space dust. Further, also implicated is the operation of some (floor) dust mixing mechanism, perhaps through normal air movement patterns and/or a foot -racking effect.

Alternatively, given the (apparently) fairly uniform distribution of floor dust in rooms in these test houses, it could be postulated that suspended dust deposition constitutes the principal contributor to floor contamination. Additionally, deposition occurs in a fairly evenly distributed fashion across floors in that it would seem unlikely that foot tracking would be evenly distributed within a room, particularly in corners. This contrasts the concept of a contamination gradient detected in rooms (eg halls), with a decrease in the concentration of an artificially-introduced dye being noted on a step by step basis as a room was entered from outside, the inference being that foot trafficking was the major contributor to room contamination (eg Lane, 1986). This is further discussed below (Section 7.17).

An analogous comparison was made for data corresponding to sampling locations on walls (where foot traffic would be expected to make minor contributions to contaminant levels) immediately under windows and sampling sites symmetrically located on walls opposite windows. Here, although sample sizes were not large, no difference was noted between these two locations for any of the dust parameters ($P \geq 0.25$). Comparison of wall sampling areas to floors, however, demonstrated approximately an order of magnitude larger mean dust and lead loading mean values for floors ($P \leq 0.001$), with minimal difference in dust lead concentration ($P > 0.50$). Al Radday et al. (1993) also noted greater dust and lead loading mean values for floors as compared to walls.

Regarding the potential for room contamination to arise as a result of the entry and deposition of above-ceiling space dust, for grouped room data, as mentioned above, no contamination gradient appeared detectable from periphery to centre; there exists minimal difference between peripheral and central or peripheral and mid locations for all floor data as also for (singly or collectively) floor data excluding fireplace, window-base and doorway-proximate samples ($P \geq 0.05$). If such a mechanism of dust entry did arise, the expectation would be that the major point of entry of above-ceiling space dust would be adjacent to damaged ceiling cornices, the most commonly personally-observed ceiling imperfection. Acknowledging that available house sample size was not large, comparison was made between rooms demonstrating (presumed) good dust sealing (no visible cornice damage) and those that exhibited less satisfactory ceiling sealing. No difference existed for any dust parameter between peripheral, mid and

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[25] If such collection in corners did occur, it may arise as a result of reduced air movement patterns and subsequent deposition, or alternatively, less effective (normal) cleaning regimes. Corner sampling locations were not actually selected directly touching the corner (wall/skirting board), in order to avoid the possibility of non random sampling of dust resulting from the concentration of dust collection at the junction of floor surface covering with wall/skirting board, or any surface irregularities located there. In most cases, corner sampling locations lay $\approx$ 2.5cm from the floor wall/skirting board junction.

central (floor) sampling locations for rooms with ceiling defects as compared to those reflecting intact ceilings and cornices ($P \geq 0.10$). Further, no major difference existed between mean dust parameter values corresponding to peripheral and mid and peripheral and central locations in rooms with damaged ceilings ($P \geq 0.075$), and no detectable contaminant gradient was evident. Finally, no notable difference in room mean (floor) dust parameter levels was detected between rooms displaying intact ceilings and those that did not ($P \geq 0.075$).

Thus, in order to detect contaminant locational foci within rooms, a more rigorous approach to the categorisation of floor sampling locations appears warranted. To further investigate as to whether a contamination gradient exists within main rooms, floor samples falling into the categories of samples derived from directly under windows and those (symmetrically-located) from the opposite side of the room were compared. Such comparison was also subsequently undertaken for doorway-located, doorway-opposite, fireplace-proximate and opposite samples.[26]

7.16.1 FIREPLACES

Floors sampled in front of all open fireplaces were compared with symmetrically-located analogues opposite such fireplaces (ie on the other side of the room). This revealed larger mean values for all dust parameters for (floor) locations opposite fireplaces, but not of any substantive import for any dust parameter ($P \geq 0.25$). In a number of situations, however, fireplaces were noted to be located opposite windows. Limiting comparison to rooms where windows were adjacent to and not opposite walls with fireplaces reduced sample size somewhat ($n = 9$), but again revealed no notable difference between the two categories for any of the dust parameters ($P \geq 0.20$), dust and lead loading being numerically larger adjacent to such fireplace locations compared with across the room.

Comparison of (separately) room centre and peripheral contamination in rooms with fireplaces to rooms without, reveals for rooms with a fireplace, greater mean values for all dust parameters (both centrally and peripherally), with notable difference for dust and lead loading ($P \leq 0.03$). Such a result supports a suggestion by others that within-house lead in dust concentration is influenced by the presence of a fireplace (eg Fergusson and Kim, 1991).

Fireplace floor sample contaminant mean levels were compared with those corresponding to centre of room samples revealing minor difference for all dust parameters ($P \leq 0.20$). Regressing lead loading (dependent variable) with the square of the distance (in metres) from the fireplace toward the centre of the room, (incorporating Mid-located samples also) reveals a correlation coefficient of $r = 0.16$ ($P > 0.50$), such poor relationship appearing as follows:

$$\ln \text{ lead loading} = 8.8 - (0.05 \times \text{square of the distance in from fireplace}).$$ [27]

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[26] In virtually all situations, rooms were approximately square, with the sole window in the middle of (the horizontal axis of) one wall, and the door, again in (or close to) the (horizontal) middle of an adjacent wall. Samples such as floor-opposite-window and floor-opposite-doorway were selected as floor samples located symmetrically opposite respectively window and doorway-sampling sites. Behind-door samples were also assessed.

[27] Such lead loading was also regressed against the distance in from the fireplace, and resulted in an approximately equivalent regression relationship.

Given this slight (decreasing) lead loading gradient from the fireplace floor to the centre of the room, it could be inferred that some (slight) floor dust contamination may appear to arise as a result of dust entry through the chimney, possibly resulting from certain specific wind conditions.

7.16.2 WINDOWS

Whilst all mean dust parameter values are greater for floor samples deriving from the base of windows, this is noteworthy only for dust lead concentration ($P < 0.05$), (Table 7.21).

Table 7.21 Window-base floor and opposite-room floor sampling comparison.

		GROUPED MEAN DATA		
POSITION SAMPLED IN ROOM	**n**	**DUST LOADING ($g.m^{-2}$)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD LOADING ($\mu g.m^{-2}$)**
Floor at base of window	**41**	1.14	5,074	5,766
Floor opposite window	**14**	1.39	2,939	4,078

Such data infer that window-base floors are closer to a dust entry point than floor locations on the opposite side of the room where less lead-contaminated dust is detected. Essentially, windows are the likely dust entry-point, in keeping with comments by others (eg Lepow et al., 1974 & 1975; Archer and Barratt, 1976a; Sayre and Katzel, 1979; Satyr, 1981; Bornschein et al., 1986; Fergusson, 1986; Davies et al., 1987a; Farfel and Chisholm, 1990 & 1991; Al-Radday et al., 1993c).[28]

Excluding rooms with fireplaces, mean data were again greater at window-base locations for all dust parameters, difference again only appearing substantial for dust lead concentration ($P < 0.05$). Comparing only rooms with an open fireplace, mean values for floors at the base of windows were substantially greater for dust lead concentration plus lead loading than for floors opposite windows ($P \leq 0.02$). This suggests that a more pronounced dust gradient (window floor to opposite window floor) results in such rooms due to greater draught through the window and the room and out through the chimney.

Following these results, comparison of contaminant levels for window floor versus centre of room locations was undertaken for the following:

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[28] Fergusson (1986), reviewing work by Butler and MacMurdo (1974), Butler et al. (1975), Diemel et al. (1981), Gloag (1981), Facchetti et al. (1982) and Elwood (1984), concludes that aerosol lead penetrates houses, with detected levels there varying between 45 and 95 percent of outside levels. Essentially, houses do not offer substantial protection from aerosol lead. This is discussed further in Chapter Nine.

a) all rooms

b) only rooms without fireplaces, and for

c) only rooms with an open fireplace.

Such investigation depicts numerically greater mean levels for all three dust parameters for window floor locations for all three alternatives. Statistical comparison reveals notable difference for lead loading for category c) as well as greater dust lead concentrations for all three categories (all at $P \leq 0.05$). Regressing mean dust parameter value (dependent variable) against the square of the (perpendicular) distance in from the room's window (at floor level and measured along window central line) elicits the following (Table 7.22).[29]

Table 7.22 Contaminant level variation with square of distance in from windows.

	CONTAMINANT LEVEL VERSUS DISTANCE SQUARED CORRELATION COEFFICIENT (r =)		
ROOM TYPE	**DUST LOADING**	**DUST LEAD CONCENTRATION**	**LEAD LOADING**
All Main Rooms	0.14	0.31	0.37
Main Rooms without a Fireplace *	0.08	0.28	0.24
Main Rooms with an Open Fireplace ‡	0.26	0.50	0.44

* For dust lead concentration and lead loading, at significance, $P \leq 0.085$

‡ For dust lead concentration and lead loading, at significance, $P \leq 0.025$

All dust parameters reflect decreasing levels with increasing distance (from the window) into the room. Little relationship between dust loading and distance was evident ($P \geq 0.15$). The floor contamination gradient radiating from a window is more clearly-defined for rooms demonstrating the presence of an open fireplace. Regressing lead loading (dependent variable) against the square of the distance in from the window for each alternative reveals, respectively, the following relationships:

a) All main rooms

$$\ln \text{ lead loading} = 8.7 - (0.16 \times \text{square of the distance in from window})$$

b) Main rooms without a fireplace

29 As for 7.16.1 above, dust and lead loading was also regressed against the distance in from the window, but this reflected poorer correlation. Such was also the case for dust lead concentration.

ln lead loading = 8.4 - (0.11 x square of the distance in from window).

c) Main rooms with an open fireplace

ln lead loading = 9.2 - (0.23 x square of the distance in from window).

Comparison of mean data for window-base floors in rooms with no fireplace and rooms with an open fireplace reveals numerically larger mean values for rooms with an open fireplace, for both dust and lead loading ($P \leq 0.01$). Similarly, mean values for all dust parameters are greater for floor centre locations in rooms with an open fireplace compared with those with no fireplace, notably so for dust and lead loading ($P \leq 0.03$). Further, from these results and the three relationships immediately above, floor dust appears more contaminated at window-base level than centre of room. Additionally, for particularly lead loading, the presence of an open fireplace in a room contributes notably to a more pronounced and well-defined (lead loading) gradient arising from the window into the room, although of n = 36 (all rooms sampled at window-base floor) here, only ten samples derive from rooms with an open fireplace.

7.16.3 DOORWAYS

In a similar context, comparison of floors sampled in a (main room) doorway to floor areas sampled on the opposite side of the room and floors sampled behind (open) doors revealed the following.[30]

Table 7.23 Doorway, behind door and opposite-room floor comparison.

	GROUPED MEAN DATA		
POSITION SAMPLED IN ROOM	**DUST LOADING** ($g.m^{-2}$)	**DUST LEAD CONCENTRATION** ($\mu g.g^{-1}$)	**LEAD LOADING** ($\mu g.m^{-2}$)
Floor in doorway	1.08	4,520	4,888
Floor opposite doorway	1.00	3,269	1,930
Floor behind door	0.87	3,159	2,749

[30] Whilst it cannot be clearly stated as to what position the door remained in before test or whether during occupancy it was regularly opened and closed, when sampling, all doors were found to be wide open, the average angle between the door and the wall immediately behind the door approximating 30 degrees. At least for the period of time that the houses were vacant, they were undisturbed, no doors being moved during this time.

Whilst mean values for the three dust parameters for main room doorway-derived samples were numerically larger than room-opposite samples and also behind-door samples, this was insubstantial ($P \geq 0.10$) for all category permutations other than lead loading for doorway versus floor opposite doorway ($P < 0.05$). Excluding fireplace-containing room data, however, mean doorway-sampled dust lead concentration is larger than its room-opposite analogue ($P < 0.05$). Further, from such rooms, doorways also demonstrated greater mean dust parameter values than behind doorway-sampled areas, this being notable for dust lead concentration and loading ($P \leq 0.05$). For rooms with open fireplaces, the sample size of floor opposite doorway samples may be too small to make comparison with doorway samples.

Doorway to centre of room comparison for all rooms revealed numerically greater mean dust parameter values for doorway-derived samples. Such difference is, however, marginal for dust lead concentration and loading ($P \leq 0.10$). Excluding data derived from rooms without open fireplaces, however, reflected, for both dust lead concentration and lead loading, a more substantial difference ($P \leq 0.05$). No real difference was detected between centres of rooms and floors opposite doorways for all rooms and also for rooms without a fireplace for all dust parameters ($P \geq 0.10$).

Thus, the presence of an open fireplace in a room would again appear to be an indicator of a more contaminated room, its presence tending to mask differences that arise due to other (presumed) dust into room contributing pathways. Regressing lead loading (dependent variable) against the square of the distance into the room from the doorway for: a) all main rooms, b) only those without a fireplace and c) only those with an open fireplace demonstrated correlation coefficients of $r = 0.18$, 0.18 and 0.31 ($P \leq 0.10$) respectively, and revealed the following:

a) All rooms

ln lead loading = 8.4 - (0.06 x square of the distance in from doorway).

b) Rooms without a fireplace

ln lead loading = 8.3 - (0.05 x square of the distance in from doorway).

c) Rooms with an open fireplace

ln lead loading = 8.6 - (0.03 x square of the distance in from doorway).

Regressing dust lead concentration (dependent variable) against the square of the distance into the room from the doorway for: a) all main rooms, b) only those without a fireplace and c) only those with an open fireplace demonstrated correlation coefficients of $r = 0.32$, 0.40 and 0.59 ($P \leq 0.015$) respectively.

Thus, there exists a detectable contamination gradient emanating into the centre of rooms from their doorway, the presence of an open fireplace in a room demonstrating a more pronounced and well-defined floor gradient. Doorways thus appear to be another entry point for dust into rooms, such determination being in keeping with similar work by others (eg Wixson and Davies, 1991).

7.16.4 FIREPLACE/WINDOW/DOOR COMPARISON

As has been previously demonstrated, rooms with open fireplaces reveal greater levels of contamination, both centrally and peripherally ($P < 0.03$). Further, data also suggest more pronounced and well-defined (inverse) gradient values for dust lead concentration and lead loading with distance in from the window base and doorway for rooms with an open fireplace (Table 7.22 and regression relationships immediately below it). Clearly thus, the presence of an open fireplace in a room is associated with an enhanced level of contamination in that room. Essentially, this infers that the source of contamination from windows becomes more predominant in rooms with an open fireplace, windows, however, contributing more to the room's contaminant level than fireplaces.

This scenario compares to the analogous situation for distance in from doorways, where the presence of an open fireplace in a room does contribute to a slightly better correlation between lead loading and distance in from the doorway. Essentially, the doorway becomes a more important source of room contamination if a fireplace is present in the room. A (relative) flattening of lead loading gradient from doorways into the centre of the room exists for fireplace-containing rooms as compared to the case for window-base floor gradients. This suggests that the presence of an open fireplace in a room facilitates a more substantial increase in room contamination arising from windows rather than doorways. Again, the predominant dust entry point into rooms is from windows. It is acknowledged here, however, that this observation may be due somewhat to this study having been undertaken subsequent to a period of house vacancy and closure.

7.17 HALLWAYS AND HOUSE MAIN (FRONT) ENTRANCES

7.17.1 HALLWAY MAIN ENTRANCES

Hall entrance mean data (Table 7.24) are substantially greater for all three dust parameters than for any of the other room location category permutations (Tables 7. 21 - 7.23), ($P \leq 0.05$). Excluding rooms with fireplaces, again hall entrance mean data are substantially greater for all three dust parameters ($P \leq 0.01$ for all permutations, except hall entrance versus main room doorway where $P < 0.05$).[31] These data suggest a gradient in contaminated dust from (front door) hall entrance to inside main rooms attached to hallways. Also inferred is the contribution to main room contaminant levels due to the presence of a fireplace in that room facilitating air movement into the home through the front doorway into the hall and

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[31] Hall entrance was the location designation for the floor area immediately inside the front door, typically, 50cm from the door line to the centre of the sampled area.

providing an exit through the chimney. Small sample size of houses with no open fireplaces precluded comparison of hallway contamination levels with houses containing open fireplaces.

Table 7.24 Hallway entrance dust levels.

SAMPLING LOCATION	GROUPED MEAN DATA		
	DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Hall entrance (n = 10)	2.74	7,634	20,902

7.17.2 HALLWAY CONTAMINANT GRADIENTS

The above-detected contamination gradient concept was investigated with respect to entrance halls of these homes. A number of evenly-spaced samples (typically four) were taken from the front hallway of each of the homes, arranged linearly and centrally from the (centre of the) front doorway inwards for the length of the hall, the first sample centring on a point 50cms inwards of the front door. Mean hallway length was 4.5 metres (range 3.8 to 8.2). Appropriate correlation coefficients are indicated below (Table 7.25).[32]

Table 7.25 Contaminant level variation with square of distance into hallway.

SAMPLING LOCATION	CONTAMINANT LEVEL VERSUS DISTANCE INTO HALLWAY CORRELATION COEFFICIENT (r =)		
	DUST LOADING	DUST LEAD CONCENTRATION	LEAD LOADING
Square of distance in from front door	0.43	0.11	0.77

[32] Nine of the ten hallway floor surfaces were carpet, with one linoleum surface. Removal of the linoleum-derived data made very little difference to mean dust and lead loading and dust lead concentration data, and also subsequent correlation coefficients.

A moderate correlation exists between dust loading and square of distance into the hallways in these houses, the coefficient for lead loading being notably better (for both, $P \leq 0.0025$). The relationship between dust lead concentration and the square of the distance is substantially poorer ($P > 0.475$).

These data imply that it is not so much the concentration of the lead in dust which is the determining factor regarding the degree of contamination at a particular (hallway) floor location, but more, the quantity of dust (and hence lead) depositing at any point. This is commensurate with a good correlation existing between dust and lead loading for hallways. Regressing lead loading against the distance from the front door into the hallway elicits the following:

$$\ln \text{ lead loading} = 9.7 - (0.04 \times \text{square of distance into hallway}).$$

Such a decreasing lead loading gradient from the entrance to these houses reveals that the front door is an entry point for dust into the hallway. Whether such contamination arises as a result of (foot) tracking and/or deposition of suspended dust entering through an open door, or under the door, appears somewhat unclear from these data.

As will be recalled, mean data for all dust parameters were greater for hallways than for main rooms ($P \leq 0.01$). Further, no difference was apparent between peripheral and central room regions for any of the dust parameters when controlling for fireplaces and doorways ($P \geq 0.10$), although a statistically greater mean value for all dust parameters did exist for doorway entrances in main rooms as compared to centres of rooms. Apart from a doorway to centre of room lead loading gradient, and a more substantial window-base floor into room lead loading gradient, a very moderate lead loading gradient exists for fireplaces into centres of rooms. No other dust lead concentration or lead loading gradient from room periphery to centre is apparent for the main rooms of the dwelling.

Separate comparison of grouped hall-sampled floor areas with all other floor areas referred to above indicated greater mean values for all dust parameters for hall-sampled floor areas than those for any other alternative for all permutations excluding doorway versus window (base) floor and versus fireplace floor ($P \leq 0.05$). Acknowledging this result, and also that the rate of decrease in lead loading levels with distance into halls is substantially greater (slope of regression curve) than for any lead loading gradients in main rooms, (recall greater mean lead loading levels in hallways than main rooms), it seems likely that the doorway is the more substantial entry point for contaminant dust into houses.

There exists no substantial difference in lead loading between the corner of (main) room locations and centres of rooms, and also other (non-corner) peripheral locations. Thus, the foot trafficking pathway seems less likely to be the major contaminant contributor, given that it is unlikely that foot trafficking would be as frequent in the corner of (main) rooms or other peripheral locations as in the centre of rooms.

No demonstrable difference for any of the dust parameters (paired *t*-test) is indicated for paired samples corresponding to (main room) doorway-derived samples and adjacent hall samples ($P \geq 0.25$). Presuming that there would be substantially more foot traffic along the hall than into either of the front rooms (typically both bedrooms), there would again appear to be a greater contribution to floor contamination levels arising as a result of dustfall than from foot-deposited dust.

One of the test houses (19 Lily St) exhibited a substantially wider hallway than any of the other houses, thereby facilitating the sampling of three hallway gradient dust samples. These gradient pathways were

located centrally (one) and peripherally (two). Each gradient was parallel to the long axis of the hallway, equally spaced across the width of the hallway, and with samples acquired at equally-spaced intervals (longitudinally) from the doorway into the room.

Regressing lead loading (dependent variable) with the square of the distance in from the front door for the central pathway and the two peripheral pathways elicited modest correlation coefficients of r = 0.22, 0.70 and 0.58 respectively, resulting in the following relationships:

a) Central path

ln lead loading = 9.2 + (0.007 x distance in from front door).

b) Peripheral path 1

ln lead loading = 9.9 - (0.04 x distance in from front door).

c) Peripheral path 2

ln lead loading = 9.7 - (0.06 x distance in from front door).

Here, the lead loading gradient along the (longitudinal) centre of the hallway reveals virtually no relationship with distance ($P \geq 0.75$), yet the peripheral hallway gradients parallel to and either side of the central path demonstrate more definable contaminant gradients, albeit with small sample size ($P \leq 0.30$). In this house alone, unavoidably, there had been some person entry through this front hallway in the house vacancy period immediately prior to static dust assessment. In all other houses, entry was made through the rear of the dwelling and at all times absolutely no walking over areas to be sampled occurred. The suggestion here is that when these houses were occupied, normal human home activity (including cleaning) contributed to dust re-suspension and mixing within homes, (as Fergusson and Kim, 1991), but a period of (undisturbed) vacancy subsequent to occupant departure allowed for the build-up of novel dust deposition.

Virtually all of this (recent) foot traffic had been along a fairly centrally-located path through the hallway. Assuming that most of the former foot traffic had been similarly directed, the central path would thus appear to reflect the greater influence of foot traffic. The peripheral hallway paths, however, seemingly register less of such an effect, there being a much more pronounced, declining lead loading gradient into the hallway from the front door. Whilst the sample size here is small, these data imply that in this instance dust entry into a hallway is by deposition from outside dust penetration, the effect of foot traffic being more one of mixing of already-existing dust. Further work needs to be directed toward this observation, given the scant attention paid to this area by previous research, and current belief regarding the significance of contribution to home floor-level contamination by occupant activity, such belief having constituted the cornerstone of some home decontamination/intervention programs.

7.18 WINDOW SILL AND WELL

Sample size for (inner) window sill (14) and window well (10) samples was not large due to a marked variability in window style and condition, limiting sampling in many instances.[33] No real difference in dust lead concentration between window sills and wells was indicated ($P \geq 0.10$). There was, however, a greater mean dust and lead loading level for window wells rather than sills ($P \leq 0.05$). This is not unexpected given the (dust-trap) nature of the window well, and also from inspection, the reduced likelihood, compared with the window sill, of former occupant cleaning activity at that location.

Despite (separately) removing from the data set, data corresponding to a number of dwellings, including that deriving from the house located at 34 The Terrace (the house exhibiting, visually, very little deposition of dust on window sills), difference in mean dust and lead loading between sills and wells remained ($P \leq 0.05$). No difference between sills and wells existed, however, for dust lead concentration for any house group permutation, suggesting that the dust collected on both window sills and wells derived from the same source.

Given these small sample sizes and the noted similarity (dust lead concentration) of window sill with window well dust, it was decided to group window sill and well dust. Further confirming the decision to group window data corresponding to sills and wells, individual comparison for dust lead concentrations and lead loadings of sills, wells and lastly sills plus wells with the three room regions (centre, mid and peripheral categories) revealed substantially greater mean values for (individually) sills, wells and also sills plus wells categories for grouped data from all main rooms ($P \leq 0.005$). Dust loadings for sills, wells and sills plus wells did not differ to any great extent to dust loading for any of the room region data ($P \geq 0.075$).

Excluding rooms with fireplaces elicited no change to these above results. Window sill/well/sill + well comparison with doorway similarly revealed notably lower mean values for doorways for dust lead concentration and lead loading for all permutations ($P \leq 0.001$). Dust loading for doorways is substantially lower than for window wells and window sills plus wells ($P \leq 0.01$), but not lower than window sills individually ($P > 0.10$). Excluding rooms with fireplaces essentially made no change to these category differences.

It will be recalled that hall entrance comparison with window-base floors for all rooms and rooms without an open fireplace revealed greater mean values for hall entrances for dust and lead loading ($P \leq .002$) and dust lead concentration ($P < 0.05$). Comparison of hall entrances with grouped window samples indicates that whilst mean hall values for all dust parameters are larger, this is not notably so for any parameter ($P \geq 0.10$).

Thus, window-derived samples, as doorway-derived samples, reflect primary entrance points for contaminant material. In rooms with open fireplaces, mean values for window-derived samples are greater for all dust parameters than all other room locations examined above ($P \leq 0.05$), although window-derived sample size is not large ($n = 5$).

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[33] In a number of situations, no window well or sill was present, or damage to the window well precluded sampling from that surface.

7.19 ROOMS FACING WALKWAY VERSUS OPPOSITE

In Chapter Six it was noted that soil lead levels were greatest in (narrow) walkways between fences and house sides, significantly greater than other regions of the home such as back yard, front yard and other side of the house. Hence, it appeared appropriate to compare static dust data similarly.

Comparison between all rooms adjacent to a walkway and those opposite that walkway demonstrated larger mean values for dust and lead loading on the side of the house opposite the walkway ($P \leq 0.05$). If comparison is made between only rooms with a window facing the walkway and rooms on the other side of the house with windows facing that side, again mean dust parameter values for opposite-facing rooms with windows are slightly greater, no substantial difference being apparent, however ($P \geq 0.10$).

Separate comparison of front walkway-adjacent versus opposite-side rooms and rear walkway-adjacent versus opposite-side rooms also reflected no real difference in mean dust parameter values ($P \leq 0.15$). Thus, no conclusion can be drawn regarding the dimension of contamination of rooms adjacent to walkways compared with other main rooms in the home as might be suggested by soil lead contamination values nearby. This observation, however, does not preclude the possibility that these walkway-proximate windows are preferential, contaminant entry points. Rather, it would seem possible that under window open conditions, such dust entry that does take place may be effectively mixed throughout the house by wind or subsequent occupant activity. Dynamic dust contamination patterns will be discussed in detail in the following chapter (Chapter Eight).

Interestingly, if comparison of such walkway-adjacent rooms with a window opening onto the walkways is made with (separately) hallways, porches and lastly combined hallways/porches, mean dust and lead loading and dust lead concentration data are always numerically lower for walkway-adjacent rooms ($P \leq 0.025$). Thus, hallways and porches, despite being proximate to soil with lower contaminant levels than those outside walkway-facing rooms, demonstrate greater levels of contamination than all main rooms. This implies that either walkway dust is less mobile and adjacent rooms with windows opening onto walkways less likely to be infiltrated by contaminant dust via windows, or front and rear doors are more likely to be entry points for contaminated material, not discordant with observed contaminant gradient patterns found in hallways (Section 7.17.2 above).

7.20 SUMMARY OF CONTAMINANT DISTRIBUTION WITHIN ROOMS

Window wells/sills and hall entrances reveal substantially more elevated levels for all dust parameters than any of the other locations ($P \leq 0.05$). Presuming that (former) cleaning protocols such as vacuuming for hallways did not differ from those for any other room, this implies that these two sites are potential entry points for house contamination. This trend also continues, although to a lesser degree, for window (base) floors and doorway floors where the progression of entry of contaminant material decreases with

increasing distance into the residence. Within the main rooms, where a child would be expected to spend most of its time, there exists little discrimination regarding room location and degree of contamination apart from window-base and doorway locations.[34] A summary of mean room location dust parameter values appears below (Table 7.26).

Table 7.26 Contaminant distribution within rooms of houses - a summary.

	GROUPED MEAN DATA					
	ALL ROOMS			ROOMS WITHOUT A FIREPLACE		
LOCATION	DUST LOADING ($g.m^{-2}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)	DUST LOADING ($g.m^{-2}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Window wells and sills	3.44	11,541	39,676	-	-	-
Window-base floors	1.14	5,074	5,766	0.95	4,950	4,679
Floors opposite windows	1.39	2,939	4,078	0.90	2,780	2,508
Fireplace floors	1.12	2,913	3,255	-	-	-
Floors opposite fireplaces	1.11	3,694	4,082	-	-	-
Doorway floors	1.08	4,520	4,888	1.33	4,307	5,726
Floors opposite doors	1.00	3,269	1,930	1.04	2,931	3,053
Floors behind doors	0.87	3,159	2,749	0.82	2,641	2,155
Main room centres	1.08	3,574	3,843	0.87	3,214	2,791
Hall entrances	2.74	7,634	20,902	-	-	-

It is thus suggested that a reasonable in-house, normal dust-mixing process is active, although doors of main rooms appear to provide some degree of protection from contaminant levels, behind-door samples being considerably lower than those for the doorway immediately adjacent. Fireplaces do not appear to be

[34] It will be recalled that greater mean values were indicated for dust lead concentration and lead loading for window-base floors and also doorway locations for all comparison permutations ($P \leq 0.05$). This, however, is not the case for any fireplace floor dust parameter.

major contributors to room contaminant levels, although sample size is not large and there may be confounding effects of other adjacent features in the rooms (eg doorways or windows). The lowest lead loading levels are located at floors opposite doors. This location is typically the furthest distance (into main rooms) from the front door, yet also only a half room width away from the window. This is presumably part of a contamination gradient radiating from the hall entrance.

The contamination pattern within a room is obviously a complicated pattern of superimposed different pathway mechanisms including doorways, windows and possibly fireplaces. Complementing this static data suite, Chapter Eight deals with depositing dust patterns. Here, rates of contamination at varying locations in these vacant houses will be estimated independently of any occupant influence upon depositing dust rates.

7.21 POST-CLEANUP STATIC DUST ASSESSMENT

As indicated in Chapter One, investigation of deposited dust was conducted in four of the test houses subsequent to a very thorough cleanup of the house. This cleanup procedure was intended to simulate the earlier SAHC-conducted intervention process in (deemed contaminated) Port Pirie houses constituting part of the previous lead intervention program.

Approximately seven to ten days subsequent to test house cleanup, floor sampling was conducted in each of these four test houses, two houses in each district. Sampling was conducted under extremely clean conditions, maintaining clean clothes and hands throughout. Shoes were removed prior to entry, all necessary equipment being placed on clean paper on the floor prior to sampling. Each tested house was entered only once, and all sampling completed with minimal disturbance before exiting in as short a time as possible. All doors and windows were kept closed during sampling as also the post-cleaning period prior to dustfall sampling (Chapter Eight).

Data corresponding to this collection phase were compared with total floor-collected data from the same four houses prior to cleanup, and also data corresponding to collection of samples from the same test sites as used for the post-cleanup sampling, a summary of which appears below (Table 7.27).

This house-cleaning procedure has removed a substantial quantity of the existing dust, notable dust and lead loading difference being apparent ($P \leq 0.001$). The opposite was the case for dust lead concentration, there being no demonstrated difference ($P > 0.325$). Importantly, these houses remained vacant for several months after the main sampling round before cleanup commenced. Thus it would seem likely that the test houses involved would have become more contaminated than indicated above due to the ingress into the house of further contaminant material which would (presumably) subsequently settle on floor surfaces. Thus whilst at least two thirds of the (vacuum pump cassette system-detectable) existing dust appears to have been removed by the cleanup process, contaminated material still remains.

Table 7.27 Influence of house cleanup on floor contaminant levels.

HOUSE CLEANING STATUS	n	GROUPED MEAN DATA		
		DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Pre Cleaning - All samples from the four houses	**226**	1.43	3,967	5,381
Pre Cleaning - Matched locations to postcleaning	**16**	1.83	3,424	6,287
Postcleaning	**16**	0.41	4,245	1,752

Subsequent to this cleanup step, dustfall was also estimated in these test houses adjacent to the vacuum-sampled floor areas for a period of 47 days (Section 8.6). Extreme care was taken to preclude any human contact/contamination of these 16 vacuum-sampled locations. These sampling locations were then vacuum-sampled again after the 47 day depositing-dust collection phase. Over the period of dustfall collection, these 16 floor surfaces (adjacent to dustfall collectors) have become further contaminated with deposited dust as indicated by the following summary data (Table 7.28).

Table 7.28 Surface contamination build-up over post-cleanup test period.

FLOOR STATUS	n	GROUPED MEAN DATA		
		DUST LOADING ($g.m^{-2}$)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	LEAD LOADING ($\mu g.m^{-2}$)
Pre Exposure	**16**	0.41	4,245	1,752
Post Exposure	**16**	0.68	4,635	3,137

A small increase in dust loading is apparent ($P < 0.10$), although this is more pronounced for lead loading ($P < 0.05$), with no appreciable change to dust lead concentration level. Such data imply that dust accumulation (as detected at floor level) within homes is an ongoing process, and that dust being detected by the technique employed here, originates as newly-entering (room/house) dust. This will be further examined in subsequent chapters.

Acknowledging moderate data suite size, data indicate greater mean values for all static dust parameters at window-base (floor) than room centre locations, although this is only notable at a modest level for lead loading ($P < 0.10$). Such difference was far less apparent from the results of the sampling regime immediately subsequent to house cleanup. These observations support the concept of the ongoing entry of novel dust into vacant closed homes, a likely entry point in main rooms being windows. Equally, mean values for all dust parameters were larger for facing smelter as opposed to facing-away categories, although, again, only at a modest level for dust loading and dust lead concentration, but more notably so for lead loading ($P < 0.01$).



Plate Five Static dust vacuum pump sampling system.

CHAPTER EIGHT WITHIN-HOUSE DEPOSITING DUST

Oh unseeing Lead, would that thou hadst never appeared in the earth or in the sea or on the land, but that thou didst have thy habitation in Tartarus and Acheron, for out of thee arise many things pernicious to mankind.

Timocrean of Rhodes

The invisible and nonexistent look much alike.

Delos B. McKown

8.1 INTRODUCTION

The preceding chapter dealt with the contamination of homes by existing (static) surface dust contained within the domicile. Complementing this, this chapter deals more specifically with the collection of depositing dust or dustfall in the suite of vacant houses.[1] This work addresses on-going (dynamic) dust sampling in the context of a pilot study for subsequent work in occupied houses. When extrapolated to the wider community, techniques employed here would appear to be of value in assessing the rate of (re-) contamination of occupied houses, with particular merit for assessing those subsequent to intervention clean-up procedures as undertaken by the SA Health Commission. Previous researchers have referred to a need to acquire depositing dust data within houses under conditions of home closure (Body et al., 1986f). No indication has been proffered, however, as to how measured dustfall/leadfall would accurately represent the contamination process in houses, and whether such collected dust would vary quantitatively from room to room and under different house conditions eg doors or windows open or closed.

8.2 RESEARCH OBJECTIVES

The work described below is intended to reflect potential temporal and spatial variability in dust deposition within the test homes under varying home conditions, focussing upon the rate of contamination by lead of nine selected, vacant homes in certain areas of Port Pirie. Such investigation attempts to define

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[1] Dustfall is the term applied for the quantity of collected dust depositing onto a particular surface over a certain time frame. The quantity of lead contained in this collected dust is termed the leadfall. In the literature, dustfall is frequently used as a generic term referring to both the quantity of collected dust and also contaminant material such as lead. Units for both are micrograms per square metre surface area per day ($\mu g.m^{-2}.d^{-1}$). Dust deposition on floor surfaces arises principally as a result of gravitational forces and Brownian diffusion, influencing a wide variety of particle sizes, a far smaller quantity of finer-sized dust particles settling upon ceilings and walls (Sinclair et al. 1988; Nazaroff and Cass, 1989; Nazaroff et al., 1990; Figley and Makohon, 1992).

certain areas in the home where patterns of (dynamic) contaminated dust deposition may be likely to represent a potential health concern. This acknowledges that rates of house contamination are an important predictor of the potential of a dwelling to contribute to an occupant's lead uptake (Body et al., 1988; Davies, 1988).

8.3 METHODOLOGY

Acknowledging the potential for large variability in diurnal dust deposition rates and possible levels of contamination due to varying atmospheric conditions (Duggan, 1981; Duggan and Inskip, 1985), dust deposition was assessed over more protracted periods of time. Further, the health hazard associated with lead intoxication is a medium to long-term problem, and so is best assessed by estimating rates of contamination over weeks or months rather than days (as Diemel et al., 1981; Duggan, 1981 & 1984; Laxen et al., 1985).

Due to the absence of any pre-existing data indicating dust deposition rates within homes in Port Pirie (albeit a very dusty environment), the chosen technique for assessing such rates (Appendix Sixteen) was initially presumed to be insufficiently sensitive to be capable of accurately determining daily or possibly even weekly dustfall/leadfall rates with reasonable precision.[2] Additionally, given the distance of the field site from Adelaide, and time and access constraints, it was decided to assess dustfall/leadfall within the test houses under varying conditions over periods of six to eight weeks. Further contributing to the limitations of this study, there was an ever-present concern regarding the continuing availability of the test houses, and the potential for vandal entry to these vacant houses with subsequent experiment loss, a problem that Turner (1980) also notes.[3]

All work conducted here was undertaken in a suite of vacant houses in Port Pirie. Thus, this thesis does not investigate that component of house contamination by depositing dust directly resulting from occupant activity eg re-entrainment, re-suspension and re-deposition (Fergusson and Kim, 1991), clothing transport (Archer and Barratt, 1976; Rice et al., 1977; Sayre and Katzel, 1979; Fergusson et al., 1981; Landrigan and Bridbord, 1984; Duggan, 1985; Brunekreef, 1986) or foot tracking of dust (Lane, 1986; Camann et al., 1990; Roberts et al., 1990, 1991b, 1992 & 1993b).

Variations in dust settlement resulting from weather, dimension and construction variability of test houses, house vacancy prior to study and extent and effectiveness of former occupant cleaning protocol are not controlled for in this body of work.

It was thus decided to proceed according to the following schedule:

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[2] Here it was anticipated that with such short duration sampling, experimental error associated with the chosen sampling technique might be reasonably high, and minimum dustfall/leadfall detection limits might not be exceeded.

[3] Initial concerns regarding the labour-intensive nature of such research were soon realised. Whilst fewer dust deposition samplers per cycle might have been employed, and perhaps a greater number of cycles tested (perhaps each for a shorter period of time), given the relative paucity of pre-existing dustfall data and uncertainties regarding the efficiency and precision of the technique, dust deposition variability within the home, and test house availability, the details of the procedure evolved as such. Certainly concerns regarding the security of vacant houses over protracted periods of time were ever-present, however, vandalism of homes/sample loss was not substantial.

a) To establish dust deposition rates in closed vacant test houses, all windows and external doors in all test homes were kept closed for the duration of the cycle. Careful entry was made to the houses only at the commencement and completion of the study period to set up and retrieve dust collectors respectively. Three such "closed" collection rounds were performed, two of which were approximately six months apart.[4]

b) To attempt to reflect house dust deposition rates under "moderately" open conditions simulating a home under more "normal" (for Port Pirie) residency conditions, two separate "semi-open" dust collection cycles approximately six months apart were conducted with all (opening) sash-type windows in the residences opened 15cm for the duration of the collection period. All windows were fitted with external insect screens.

c) Reflecting common Port Pirie home conditions particularly during the warmer months, two further "fully-open" dust deposition assessment cycles (again approximately six months apart) were performed with, as with b) above, all windows open 15cm, and with both front and back main doors being left wide open for five hours per day, five days per week. All houses had screen front and back doors fitted.[5]

Final dustfall collection rounds are as indicated below (Table 8.1).

Table 8.1 Dustfall collection rounds.

ROUND NUMBER	PERIOD OPEN	SEASON	DURATION (days)	HOUSE STATUS
One	Dec 8 1993 - Feb 2 1994	Summer	56	All closed
Two	Feb 2 1994 - Mar 22 1994	Summer/Autumn	48	All closed
Three	Mar 22 1994 - May 2 1994	Autumn	41	Windows open
Four	May 2 1994 - June 8 1994	Autumn/Winter	37 *	Doors/windows open
Five	June 8 1994 - Aug 16 1994	Winter	69 ‡	All closed
Six	Sep 22 1994 - Nov 4 1994	Spring	43	Windows open
Seven	Nov 4 1994 - Dec 22 1994	Spring/Summer	48	Doors/windows open

* Collection period closed prematurely as a result of some vandalism to one house, and fears of more serious interference with experimentation.

‡ For various technical reasons, this collection period was allowed to continue beyond the anticipated eight week duration.

[4] In order to reflect potentially differing dust-deposition rates with season in Port Pirie, all alternatives were tested on at least two occasions approximately six months apart, one over Summer, the other over Winter.

[5] These "open" conditions were based upon an investigation of houses in the area, and observed typical occupancy patterns. Such a chosen open-house conditions regime is only intended to represent an estimate of average house opening patterns. It is acknowledged that substantial variation in such patterns is likely.

8.3.1 DUSTFALL-COLLECTING TECHNIQUE

In order to sample dustfall/leadfall indoors, despite there existing no standard method, a number of potential (pre-existing) alternative techniques were reviewed, and their advantages/disadvantages considered (Appendix Sixteen).[6] Whatever dust-collecting method was to be employed, it would have to be able to provide a relatively simple, cheap, reliable, accurate and precise means of estimating dustfall and leadfall. Further, the selected technique would need to facilitate the consistent and relatively simple harvesting of the entire dust sample from all collectors per cycle, in addition to the transport of collectors (without any loss of sample) back to Adelaide for appropriate laboratory analysis.

The technique finally devised to estimate dustfall in vacant houses in Port Pirie incorporated the use of multiple dustfall collectors, constructed from shaped pieces of virtually equivalent length pre-weighed heavy-duty aluminium catering foil (Sudden Paper & Plastics, Lonsdale SA; Plate Six, p. 288). Further details are included in Appendix Sixteen.

8.3.2 ADDITIONAL DUSTFALL ASSESSMENT

Since this proposed technique was novel, and had previously been untried, it was considered necessary to contemporaneously assess leadfall by an alternative (previously-tried) technique in matched locations in order to make comparison between different dust-collection techniques. It was concluded that if significant differences in the rates of lead collection existed between the two collection techniques, then caution would be needed in interpreting the data provided by the alfoil tray technique.

The other technique chosen utilised wax-coated paper cups, a system similar to the poly-styrene cups used by Aurand et al. (1983), Seifert et al. (1984), Dolan et al. (1986) and Bernigau et al. (1993). At time of trial commencement, concerns existed regarding the substantial depth differences (5.0cm versus 17.5cm), and the (presumed) associated potential differential in dust-retaining properties between the two collectors. As will be seen below, this would appear to be a needless concern.

8.3.3 SAMPLING STRATEGY

This work is focused upon the collection of depositing dust within the home environment subsequent to the former occupants having vacated their homes. In the main, dust was collected by alfoil collecting tray and paper cup at various locations from within the test houses. This constituted by far the bulk of the dustfall/leadfall sampling conducted. For comparison purposes, however, some dust was collected by the same techniques within structures external to the house such as laundries and external bathrooms, in order

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[6] Literature reflects the concern over the absence of any standardised procedure for the collection of dustfall within homes.

to estimate approximate depositing dust rates external to the home (Section 8.7). The latter alternative constituted a minor component of this work. Additionally, part of the work deals with depositing dust collection in some of these homes subsequent to a very thorough cleaning (Section 8.6). These will be discussed separately below.

Alfoil dust collectors were placed in various floor-level locations in the test house suite including main rooms, hallways and kitchens over seven separate collection rounds, (each collection round comprising approximately fifty five collectors). The total time of dust collection was 342 days. The placement of dust collectors in a variety of locations in a house was intended to assess potential spatial differences in dust deposition within the room and the house. In a main room, collectors were usually placed in the centre of a room and at the base of a window with the centre of the collector being located 30cm inwards from the window toward the centre of the room. The longer edge of all alfoil trays used for this work was located parallel to the plane of the window in the room.

Depositing dust was collected in the hallway at a level approximately the same distance from the front door (into the hallway) as the centre dust-collecting position of the two front rooms was located from the front wall, this being the (longitudinal and transverse) centre of the hallway. Dustfall collectors were also placed in the hallway with the centre of the collector being located 30cm into the hallway from the plane of the door. Again, the longer edge of the tray was located parallel to the plane of the doorway. All entry into test houses was subsequently made via the back door in order to minimise disturbance to collectors located in the hallway.

Details of the procedure for laying out and collecting depositing dust collectors is included in Appendix Seventeen. Details of the procedure for harvesting the sample from the dust collectors is indicated in Appendix Eighteen.

Data provided by the alfoil tray technique were total dust and lead collected per individual alfoil collecting tray, and also dust lead concentration, whereas data provided by the cup collectors were simply the quantity of lead collected over the collection period. Various other data corresponding to house and sampling locations were also recorded including house district location, room type, facing aspect of room, and siting of collectors in the room.

8.4 SAMPLE ANALYSIS

Lead in dust was assessed using flame AAS subsequent to acid digest of the entire contents of the dust collectors, details of which as for the comprehensive QC protocol are referred to in Appendix Three.

8.5 DUST DEPOSITION RESULTS

Again, it appeared appropriate for all data to be treated as log-normally distributed, and all statistical interpretation to be based upon the (natural) log-transformed data sets unless otherwise indicated.

8.5.1 INTERNAL DUSTFALL COLLECTION - SUMMARY DATA

Depositing dust/lead data corresponding to a total of 359 alfoil collectors and 237 paper cup collectors were employed to represent all collection rounds for this piece of work. Mean collected dust and lead per alfoil collector was of the order of 442.1mg and 2.81mg respectively. The minimum collected quantity of lead in a collector over the test period was 128.2µg, mean field blank-contained lead being 24.3µg. All subsequent dustfall and leadfall values are expressed on a per day basis. A summary of the results derived from alfoil collecting trays appears below (Table 8.2).

Table 8.2 Summary dust deposition data.

	ALFOIL TRAY-COLLECTED DEPOSITING DUST (n = 359)		
STATISTIC	**DUSTFALL** ($\mu g.m^{-2}.d^{-1}$)	**DUST LEAD CONCENTRATION** ($\mu g.g^{-1}$ dust)	**LEADFALL** ($\mu g.m^{-2}.d^{-1}$)
Mean	9,323	6,350	59
Median	11,996	6,579	69
Std Dev	3.5	2.1	4.0
Skewness	0.7	0.9	0.8
Kurtosis	0.4	0.7	0.4

This rate of dustfall appears similar to an indoor rate of 20,000$\mu g.m^{-2}.d^{-1}$ as measured by a different technique in more than 100 homes in five different and widely-separated USA cities (Schaefer, 1971). Leadfall rates compare with a mean figure of 7.3$\mu g.m^{-2}.d^{-1}$ from the Arnhem (Netherlands) study conducted in houses located proximate to a medium-sized lead smelter (Diemel et al., 1981). For houses proximate to a lead smelter, Aurand et al (1983) proffer a very variable figure with a geometric mean value of approximately 50$\mu g.m^{-2}.d^{-1}$. Clark et al. (1985) report, for houses of varying condition located in

an urban environment, a range of from 0.8 - 7.3μg.m^{-2}.d^{-1}. Clearly, demonstrated test house leadfall levels are not inconsiderable.

A comparison of the two dust-collecting techniques when employed side by side, contemporaneously over the first five collection rounds of the dust deposition study, reveals similar results for the quantity of collected lead as indicated below (Table 8.3).

Table 8.3 Summary comparison of alfoil- and cup-collecting leadfall.

	COLLECTED LEADFALL (μg.m^{-2}.d^{-1})	
STATISTIC	**ALFOIL TRAY (n = 237)**	**PAPER CUP (n = 237)**
Mean	37	41
Median	32	37
Std Dev	4.0	3.8
Skewness	1.2	1.5
Kurtosis	0.4	0.5

Comparing leadfall collected (contemporaneously) in the alfoil trays, and that collected in the paper cups reveals no difference between either collecting technique ($P > 0.40$), the correlation coefficient for leadfall collected by both techniques demonstrated $r = 0.97$, ($P < 0.001$) suggesting a close relationship between (leadfall) data elicited by each technique. Cup-collected deposited dust was only continued with for the first five rounds of overall dustfall collection. Comparison of all leadfall results for either collection technique again suggest no substantial difference between the two data sets ($P > 0.30$). Separate, individual collection-round comparison of alfoil-collected versus cup-collected leadfall reveals no real difference ($P \geq 0.20$).

Regressing alfoil-collected lead (dependent variable) against cup-collected lead revealed the following relationship:

$$\ln \text{alfoil-collected lead} = -0.04 + (0.99 \times \ln \text{cup-collected lead}).$$

From these results, there exists minimal difference between the two collecting systems as they have been utilised for this study. Such a conclusion acknowledges that the (leadfall) detection sensitivity of the two techniques may not be sufficiently great to discriminate between them if any difference actually does exist.

As part of a more limited, method-comparative study, dust was also collected by another researcher in one metre square, fabricated aluminium trays (0.15cm wall thickness, 5.0cm edge height) located adjacent

to analogous aluminium foil containers in the centre of rooms in vacant houses and exposed for 47 days. Regressing leadfall collected by each technique elicited a correlation coefficient of (r =) 0.99 ($P > 0.10$; n = 4), (aluminium tray data courtesy of M. van Alphen, SAHC). Given this consistency between methods, the data elicited by the alfoil study appears to provide a reasonable estimate of the dustfall occurring in houses in Port Pirie.

Regressing alfoil-collected leadfall (dependent variable) against dustfall (n = 359) reveals a correlation coefficient of (r =) 0.86 ($P \leq 0.001$), and elicits the following relationship:

$$\ln \text{alfoil-collected leadfall} = -4.7 + (0.96 \times \ln \text{alfoil-collected dustfall}).$$

The total time of dust-deposition collection over the seven separate cycles was 342 days. This is sufficiently long a period of time to be considered to be fairly representative of the dustfall affecting the nine separate test houses. The value of r infers that there exists a good, linear relationship between the quantity of dust falling inside the test houses, and the quantity of lead contaminating this dust. By comparison, regressing leadfall against dust lead concentration demonstrates a coefficient of (r =) 0.44 ($P \leq 0.01$). Variability in dust lead concentration thus explains only around 20% of the variability in leadfall, compared to dustfall's 74%. Such results confirm that the rate of dust collection within a room of a house (as assessed by this technique) would appear a very reasonable index of the rate of (lead) contamination of that room, and that the sources of the dust and the lead are closely related.

Recalling from Chapter Seven, the regression relationship between static dust and lead, viz:

$$\ln \text{collected lead} = 1.2 + (1.05 \times \ln \text{collected dust}).$$

Such similarity in data for static dust and dustfall determination for these houses would suggest that in both contexts, a close relationship exists between the quantity of dust and lead collected, and also that the sources of dust and lead in both situations, are quite closely and similarly related.

8.5.2 DUSTFALL COLLECTION, ROUND BY ROUND

Dustfall collection took place over seven separate, independent collection periods, data now being presented on a (grouped) cycle by cycle basis (Table 8.4). Correlation coefficients corresponding to alfoil-collected leadfall against cup-collected leadfall for the individual collection rounds reflect the good agreement between the two collection techniques. From alfoil-collected data, there is considerable inter-collection cycle variability in mean cycle dustfall and leadfall, the ratio of the highest to lowest collection-round mean reading for each parameter being, respectively, 19.3 and 15.7. Such variability infers differences in dust collection arising from differences in house status at the time of collection (eg all closed versus windows open etc), and also possibly, different seasonal effects (eg wind direction, duration and speed).

Table 8.4 Individual collection cycle summary depositing dust data.

	GROUPED MEAN DATA						
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED		Correl. Coeff. Alfoil- versus Cup-Collected Leadfall * (r =)
COLLECTN. ROUND.	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	
One	**45**	3,527	4,761	17	**45**	18	0.87
Two	**42**	2,765	3,178	9	**44**	12	0.93
Three	**52**	13,411	8,129	109	**56**	94	0.94
Four	**52**	33,880	4,068	138	**56**	167	0.90
Five	**46**	1,754	10,799	19	**51**	20	0.98
Six	**57**	19,533	8,973	175	-	-	-
Seven	**65**	18,198	7,203	131	-	-	-

* For all, $P \leq 0.001$.

The concentration of lead in collected dust per cycle varies by a factor of 3.4. Assuming that the sensitivity of the system to gravimetrically determine dustfall and chemically determine leadfall are not limiting factors, it may be concluded from these data that moderately different dust populations (ie dust samples varying only slightly in lead concentration) are being collected by the dust collectors over the test periods. Such a conclusion, however, may be somewhat confounded by the need to consider the different districts from which the test houses were selected, and also potential seasonal variation in dustfall rates for the different districts. Seasonal variability in dust deposition rates have previously been noted by a number of workers (eg Jeffries and Snyder, 1978; Duggan and Burton, 1983). These factors are considered below. This between-collection cycle range in dustfall and leadfall mean value is more substantial than the range for dust lead concentration, suggesting that the quantity of dust/lead depositing on an internal (floor) surface is a more appropriate determinant of the degree of contamination of that surface than is the dust lead concentration.[7]

Comparing Rounds One, Two and Five, (all closed house, Summer, Summer and Winter seasons respectively), dust deposition in the suite of houses is greater in the (late) Spring - Summer - early Autumn months. When comparing Rounds Three and Six (partly-open, Autumn and Spring respectively) the Spring season again reflects greater dustfall. Contrasting this, however, comparing Rounds Four and Seven (windows and doors open, Winter and Summer respectively) dustfall is greater in winter months,

[7] Archer and Barratt (1976a) claim that deposited dust is not as informative as dustfall for sampling, given, typically, a lack of knowledge of the collected (settled dust) sample's history. Fergusson et al. (1980) note, however, that over a limited area, individual samples may be (collectively) fairly representative of that area.

although Round Four was the shortest of all collection rounds. Alfoil-collected leadfall contrasts the seasonal trend for dustfall, with greater levels being detected in the Autumn - Winter - (early) Spring period. The use of cup-collectors was not continued through Rounds Six and Seven, which limits possible conclusion regarding seasonal variation in leadfall (as detected by this technique), although, again, the Winter months demonstrate greater levels of leadfall in the test houses. Dust lead concentration thus fairly clearly reflects greater levels during the Winter/Spring months, although detected (seasonal) variation in dustfall and particularly leadfall is more substantive.

Such seasonal summary data reflects that leaded dust constitutes a more substantial proportion of the total depositing dust pool in the winter months than the summer months. For the summer months, given the rather dry/dusty nature of the environment, other, more general (less leaded) dust arising from (say) much of the un-vegetated vacant land nearby predominates. From these data, whilst some seasonal trends appear eg such as enhanced leadfall in winter months, a larger sample size here might more conclusively demonstrate real seasonal differences in dust deposition rates within these houses.

The presence of apparently confounding mean cycle data such as Round Four dustfall data may simply reflect an atypical weather event. Clearly there are large differences in dustfall/leadfall values between house closed and semi-open/open categories, these differences appearing to be more substantial and consistent than any demonstrated seasonal differences.

Statistical comparison of different collection-round dustfall and leadfall is reflected in Table 8.5.

Table 8.5 Inter-round dust deposition differences.
[**x** indicates no difference, ✓ indicates a difference ($P \leq 0.05$)]

LEADFALL* Round	DUSTFALL One	Two	Three	Four	Five	Six	Seven	DUSTFALL Round
One	*	x	✓	✓	✓	✓	✓	One
Two	✓	*	✓	✓	✓	✓	✓	Two
Three	✓	✓	*	✓	✓	✓	✓	Three
Four	✓	✓	x	*	✓	✓	✓	Four
Five	x	✓	✓	✓	*	✓	✓	Five
Six	✓	✓	✓	x	✓	*	x	Six
Seven	✓	✓	x	x	✓	x	*	Seven
	One	Two	Three	Four	Five	Six	Seven	

LEADFALL*

* Alfoil-collected

From Table 8.4, there appeared quite some variation in between-cycle collected dust levels. This is reflected by Table 8.5 demonstrating that difference in dustfall exists at virtually all inter-round comparison levels, with the exceptions of Rounds One and Two (both house closed and consecutive tests) and Rounds Six and Seven (respectively, semi- and fully-open house consecutive tests). This is not surprising given the similar season and conditions for the first example. The second example indicates that there is not a great deal of difference between semi- and fully-open house conditions, although this is not mirrored by the dustfall difference between Rounds Three and Four.[8] Again, seasonal, weather event and/or possible district differences may enhance variability and confound any true comparisons.

Some similarity in leadfall data for comparisons between Rounds Three and Four and Rounds Six and Seven (both examples demonstrating no difference at $P \geq 0.15$) also implies that opening the doors for 25 hours a week in addition to leaving the windows open for the duration of the test period does not make a substantial difference to alfoil-collected leadfall in the test house suite. This is confirmed by Round Three and Seven comparisons, Round Four and Six comparisons and also Round Four and Seven comparisons, all of which register no real difference between the leadfall recorded for semi- and fully-open houses ($P \geq 0.05$). Such a result infers that seasonal differences in collected leadfall (substantial or otherwise) are not sufficiently great to reflect differences in leadfall between house semi and fully-open categories.

These data further infer that seasonal effects do not contribute to a substantial and consistent difference between the three sets of (house status) alternatives for dustfall or (alfoil-collected) leadfall, acknowledging the fact that all three comparisons comprise collection rounds which are approximately six months apart. Such a conclusion, however, may be confounded by possible district differences or individual house differences, further investigation being necessary. Below, the data will be separated into two districts for a closer inspection.

An analogue of Table 8.5, the following examines (separately) round by round dust lead concentration and cup-collected leadfall differences (Table 8.6). Regarding dust lead concentration, little inference can be drawn other than to note that the effect of seasonal differences on dust lead concentration cannot really be clearly defined by the data as presented here (ie Rounds Three and Six and Rounds Four and Seven). To some extent, however, the above observation regarding the lack of detectable seasonal influence upon leadfall is supported by dust lead concentration data demonstrating no difference between collection Rounds Three and Six ($P \geq 0.40$) and also Three and Seven ($P \geq 0.30$), although this contrasts the relationship between collection Rounds Four and Seven. Round Four highlights some dust deposition peculiarities which may arise, as mentioned earlier, as a result of a particular weather event. That Rounds Three and Seven demonstrate no difference for dust lead concentration as well as leadfall ($P \geq 0.20$) again suggests that differences between semi-open and open-house conditions are not substantial; certainly not by comparison with differences between either category and house closed conditions.

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[8] Despite this substantial difference in dustfall between Round Three and Four, as will be seen below, mean leadfall for Round Four is not greater than for Round Three ($P > 0.15$). This may suggest that during this collection period, a storm event occurred depositing dust in collectors, such dust exhibiting a relatively low dust lead concentration. This dust might originate as soil particles from proximate farming land that would have been freshly harrowed at that time prior to seeding and awaiting the first winter rains (pers. obs.).

Table 8.6 Inter-round dust deposition differences.
[x indicates no difference, ✓ indicates a difference ($P \leq 0.05$)]

LEADFALL* Round	One	Two	Three	Four	Five	Six	Seven	DUST LEAD CONCENTRATION Round
One	*	✓	✓	x	✓	✓	✓	One
Two	x	*	✓	✓	✓	✓	✓	Two
Three	✓	✓	*	✓	✓	x	x	Three
Four	✓	✓	✓	*	✓	✓	✓	Four
Five	x	✓	✓	✓	*	x	✓	Five
Six	-	-	-	-	-	*	x	Six
Seven	-	-	-	-	-	-	*	Seven
	One	Two	Three	Four	Five	Six	Seven	

LEADFALL*

* Cup-collected

Seasonal cup-collected leadfall data comparisons are limited and provide seemingly conflicting results. Rounds One and Five reflect no difference ($P \geq 0.50$), yet Rounds Two and Five reveal a greater winter months mean value ($P \leq 0.001$). Such results infer that seasonal effects upon leadfall may not be really significant as has also been indicated by alfoil-collected leadfall data above, although, again, possible district and individual house differences have not been controlled for. For an ideal study of this kind, a larger sample size of test houses with some closed-house contemporaneous controls would be more desirable, and with sampling continuing for at least two consecutive years.

8.5.3 DUST DEPOSITION: DISTRICT COMPARISON

8.5.3.1 ALL ROUNDS COMBINED

Initially, data were split into two sub-groups, corresponding to each of the two districts from which the test houses were selected, namely Port Pirie West (n = 5 houses) and Solomontown (n = 4 houses), summary data details of which are indicated below (Table 8.7).

Table 8.7 District differences in dust deposition.

	GROUPED MEAN DATA						
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED		Correl. Coeff. Alfoil- versus Cup-Collected Leadfall * (r =)
COLLECTN. LOCATN.	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	
All houses	**359**	9,323	6,350	59	**252**	41	0.97
Port Pirie West (5 houses)	**230**	8,513	5,390	46	**169**	34	0.96
Slmntwn. (4 houses)	**129**	10,961	8,507	93	**83**	57	0.98

* For all, $P \leq 0.001$.

No real difference ($P > 0.05$) between the dustfall data for the two districts is apparent. Comparing cup-collected leadfall for the two districts reveals difference ($P \leq 0.01$), as also does comparison of both dust lead concentration and alfoil-collected leadfall for the two districts ($P \leq 0.001$). Correlation coefficients (r =) for leadfall (dependent variable) with dustfall are, respectively for Port Pirie West and Solomontown, 0.86 and 0.88, regression relationships for the two districts respectively, being:

$$\ln \text{leadfall} = -4.0 + (0.87 \times \ln \text{dustfall}), \text{ and,}$$

$$\ln \text{leadfall} = -5.6 + (1.09 \times \ln \text{dustfall}).$$

Applying ANOVA analysis and controlling for dustfall reveals difference between leadfall levels for Port Pirie West and Solomontown, the latter demonstrating greater leadfall ($P < 0.01$). Essentially, district differences in mean leadfall levels cannot be explained simply by district differences in dustfall levels, the dust being a more significantly lead-contaminated population in Solomontown test houses.

From these data, it can be concluded that the type of dust falling inside houses in the two districts is different. Solomontown houses reflect slightly greater levels of dustiness but substantially greater levels of contamination. Further, both these techniques for depositing dust collection are sufficiently sensitive to detect such differences as may be present. As can again be seen, good agreement between the two dust-collecting systems exists, varying little between the two districts. It is also apparent that the alfoil technique is more sensitive than the paper cup technique, given the levels at which difference between the

two districts is detected when dust deposition sampled by each technique is compared. Such greater sensitivity of alfoil collectors is commensurate with their substantially larger surface area.

8.5.3.2 DISTRICT DIFFERENCES ACCORDING TO HOUSE STATUS

Such seasonal influences on dust deposition as may exist in Port Pirie have not been demonstrated by this data set (Section 8.5.2); in fact, some conflicting results have been presented. Thus for the purposes of district comparisons, seasonal influences will not be considered, and results will be categorised simply under house open/closed format. A comparison of grouped, same collection cycle-type dustfall collection for total, Port Pirie West and Solomontown data appears in Table 8.8.

Table 8.8 Grouped different house condition dust collection mean data.

		GROUPED MEAN DATA					
		ALFOIL TRAY-COLLECTED			CUP-COLLECTED	
COMBINED COLLECTION ROUNDS		n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
One, Two & Five (All closed)	All Data	133	2,565	5,562	14	140	17
	Pirie West	89	2,300	5,341	12	94	14
	Slmntwn.	44	3,223	6,058	20	46	24
Three & Six * (Open windows)	All Data	112	16,324	8,560	140	56	94
	Pirie West	68	16,007	7,274	116	39	87
	Slmntwn.	44	16,865	11,212	189	17	115
Four & Seven ‡ (Open doors & windows)	All Data	117	23,985	5,589	134	56	167
	Pirie West	73	23,346	4,106	96	38	126
	Slmntwn.	44	25,084	9,319	234	18	300

* Round Three only for paper cups.

‡ Round Four only for paper cups.

From these data, there is only modest difference in dustfall between the two districts with Solomontown slightly dustier under semi- and fully-open conditions. This, however, is not the case for dust lead concentration and also leadfall (both alfoil- and cup-collected) where substantial district differences are revealed. For all collection-cycle categories, the leadfall detected in the Solomontown test houses is numerically greater than that for Port Pirie West, this also being the case for the concentration of lead in the collected dust. Comparison of these data between the two districts reveal for combined Rounds One, Two and Five, greater mean values for Solomontown houses for all dustfall parameters except dust lead concentration ($P \leq 0.02$). For combined Rounds Three and Six, however, difference between the two districts exists for alfoil-collected leadfall and dust lead concentration ($P \leq 0.01$), and cup-collected leadfall ($P \leq 0.05$), but not for dustfall ($P > 0.10$). Combined Rounds Four and Seven data demonstrate equivalently, ie difference for alfoil-collected leadfall and dust lead concentration and also cup-collected leadfall ($P \leq 0.001$), but with no real district difference for dustfall ($P > 0.10$).[9]

Good agreement between the two leadfall-detecting systems exists, with correlation coefficients (r =) corresponding to alfoil-collected versus cup-collected leadfall for the three categories closed, semi-open and fully-open being, respectively, 0.93, 0.94 and 0.90. Such good agreement between the two leadfall-collecting techniques for each collection round category implies that the source(s) of the lead being collected by either leadfall-collecting technique is/are closely related, this relationship changing little if at all for any house status category. Further, analogous correlation coefficients for (alfoil-collected) leadfall versus dustfall for the three house collection categories are (r =) 0.59, 0.69 and 0.62 respectively. The inference here is that the sources of (collected) lead and dust are fairly closely related, and importantly, do not appear to vary greatly with varying house status.

The alfoil dust-collecting technique is clearly more sensitive to detecting district (leadfall) differences than the cup-collecting technique. Such district differences appear most pronounced when doors and windows remain open during part or all of the collection period.[10] Acknowledging the dramatic increase in leadfall levels from closed to open houses, it seems likely that the source of this in-house deposited contaminant dust is externally-derived.

The assumption that sampling house dust re-contamination rates in one area would be representative of those occurring in other areas of Port Pirie (Body et al., 1985 & 1988; Body 1986f & 1986i) is inappropriate since it is unsubstantiated. Further, such a comment also appears incorrect given these above-presented data for the two districts sampled. In addition, comparing combined-cycle, combined-district data corresponding to a closed house with firstly semi-open and then fully-open house alternatives reveals notably greater levels for all depositing dust parameters under semi-open house conditions and all parameters apart from dust lead concentration for fully-open house conditions ($P \leq 0.001$). No real difference in leadfall is detected between semi-open and fully-open house conditions ($P \geq 0.30$).

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[9] Dustfall data *per se* would seem to provide little information re district differences other than suggesting under semi/fully-open house conditions, no district differences in house dustiness exist ($P \geq 0.10$). Under closed-house conditions, Solomontown houses reflect the contrary, ie a notably greater mean dustfall value ($P \leq 0.02$). Section 8.5.3.3 & 4 below dealing with district differences on a round by round basis further support this.

[10] Given the substantial differences between dustfall/leadfall in closed versus open houses, and the district dust lead concentration differences under open window and window plus door conditions (such district differences not being substantial under closed window and door conditions), it would seem logical to conclude that such dust that was being collected in dustfall collectors deposited there as either novel dust entering the room with air movements or alternatively as a result of enhanced re-entrainment of existing dust in the room. This will be addressed in detail in a later section (8.5.4) dealing with dustfall assessment in cleaned houses.

Difference, however, is apparent between dustfall data (fully-open demonstrating greater mean value) and dust lead concentration data (semi-open house demonstrating greater mean value), both at $P \leq 0.01$.[11] Thus, controlling for house (open/closed) condition needs to be considered by investigators when making inter-house comparisons.

A simple comparison of the three dust-collecting cycles (closed, semi- and fully-open house), eliciting a simple ratio of the different categories for semi-open and fully-open to closed alternatives is presented below (Table 8.9).

Table 8.9 Depositing dust parameter ratios for all data and by district.

		RATIO OF GROUPED MEAN DATA			
		ALFOIL TRAY-COLLECTED			CUP-COLLECTED
COMBINED COLLECTION ROUND RATIO		DUSTFALL	DUST LEAD CONC.	LEADFALL	LEADFALL *
Three & Six / One, Two & Five	All Data	6.4	1.5	9.8	5.7
	Pirie West	7.0	1.4	9.5	6.3
	Slmntwn.	5.2	1.9	9.7	4.8
Four & Seven / One, Two & Five	All Data	9.4	1.0	9.4	10.1
	Pirie West	10.2	0.8	7.8	9.1
	Slmntwn.	7.8	1.5	12.0	12.6

* The ratios for cup-collected leadfall are actually:
- Round Three divided by combined Rounds One, Two and Five, and
- Round Four divided by combined Rounds One, Two and Five respectively,

since cups were not used to collect dust for Rounds Six and Seven.

There exists a large difference in dustfall between house closed and semi-open and fully-open situations, such ratios for the two districts not appearing to differ greatly. This is also the case for the leadfall collected by the alfoil and cup techniques. Since n is not large here (17 and 18 respectively), it may contribute to the variation between alfoil- and cup-collected ratios for particularly Rounds Three and

[11] Given that Round Four data demonstrate some quite low dust lead concentration values, comparison of just Round Six (semi-open) and Seven (fully-open) data reveals no major difference for any of these dust deposition parameters ($P \geq 0.05$). Analogous comparison of Round Three and Four data reflects no difference again for alfoil-collected leadfall ($P \geq 0.15$), but difference for dustfall and dust lead concentration ($P \leq 0.01$). Thus care must be exercised when making comparisons between semi-open and fully-open categories based upon dustfall data and dust lead concentration data, in that a specific weather event could have influence on dustfall levels and thus also dust lead concentration, but not necessarily leadfall levels. This again reinforces the value of utilising leadfall rather than dust lead concentration data as an index of room contamination.

Six. Despite the large increase in dust (and lead) collected from closed to fully- and semi-open alternatives, the concentration of the lead in dust does not appear to vary to the same degree. Here, the ratios corresponding to house fully-open/house closed and house semi-open/house closed lie close to unity, implying that the source(s) of lead-contaminated dust under these varying house conditions doesn't vary greatly. Further, as an indicator of the dimension of room contamination, dust lead concentration is insufficiently specific, and thus is not the best depositing dust parameter choice.

It also seems likely that such dust is (principally) externally-derived, novel dust entering the room from via (say) the doors/windows given the substantial differences in dust deposition under varying house conditions illustrated above. Alternatively, it could be argued that under open-house conditions, with air/wind streaming through the house, a type of Bernoulli effect could be produced within the rooms as a result of which, existing dust is re-entrained and deposited or dust is abstracted from the above-ceiling space. Reference will be made to this in a later section.

By comparison with the differences in dustfall/leadfall levels between closed and semi- or fully-open house status categories, the differences between the last-mentioned two dust-collection categories (semi-open and fully-open) appear rather small and mostly not significant (as first mentioned in 8.5.2 above). Such an observation implies that the extra contribution to dustfall/leadfall inside houses arising by doors being open for five hours a day five days a week as well as windows is not substantial.[12] On an individual house by house basis, no real dustfall or leadfall differences exist between semi-open and fully-open house categories ($P \geq 0.05$) although sample size is typically not large. Similarly, dust lead concentration would not appear to differ between either house condition apart from two individual house situations ($P \leq 0.05$). Such individual house by house comparison also demonstrates that differences clearly exist between house closed and either semi-open or fully-open conditions for particularly dustfall and leadfall for all houses ($P \leq 0.05$), but again, sample size is not large.

On a grouped house basis for separate districts, comparison of combined Rounds Three and Six data with combined Rounds Four and Seven data reveals differences for dustfall and dust lead concentration for each district, the fully-open house category exhibiting greater mean levels ($P \leq 0.01$). For leadfall, however, no difference is demonstrated ($P \geq 0.15$).

It is acknowledged that some difference exists between dust deposition in semi- and fully-open house states as has been demonstrated by this data suite. These, however, may arise as a result of seasonal/weather events, particularly given that leadfall has not been shown to differ between these two house categories for any individual house, for either district group of houses or for all the houses grouped. Whilst dustfall is greater when doors are left open, dust lead concentration is generally lower, thus seemingly "diluting" the dust entry stream into the house. Given the larger dimension of the doorway as compared to a 15cm window opening, and its location at ground level, it could be construed that doorways facilitate the entry of some dust types not entering through windows. For example, larger-sized, lower lead concentration dust particles, may travel externally by a process of saltation (as Bagnold, [1954] and Raudkivi, [1967]), with a path amplitude that may not be sufficiently great to reach the window opening and enter. Smaller-sized, higher lead concentration particles, however, may penetrate the residence more

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[12] For example, it will be recalled that no difference for dustfall exists between Rounds Six and Seven, and none for alfoil-collected leadfall between Rounds Three and Four, Six and Seven, Three and Seven and Four and Six ($P \geq 0.05$). Some (minor) seasonal influence is apparent as demonstrated by dust lead concentration data, however, although there is also some inconsistency here, the results for collection Round Four appearing unusual.

easily (Yocom, 1982). Again, it is also possible that with doors open, greater wind speeds result within homes, which would elicit greater re-suspension and subsequent deposition of existing dust which would then be collected.

There appears only minor difference in detected leadfall arising as a result of doors being left open for 25 hours a week, particularly as compared to the differences between house closed and either of the other two alternatives. Thus, data for the semi and fully-open conditions are grouped here, a summary of data appearing below (Table 8.10).

Table 8.10 Grouped semi-open and fully-open house status summary data.

		GROUPED MEAN DATA				
		ALFOIL TRAY-COLLECTED			CUP-COLLECTED	
COMBINED ROUNDS		n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
One, Two & Five (All closed)	All Data		2,565	5,562	14		17
	P. West	90	2,300	5,341	12	94	14
	Slmntwn.	43	3,223	6,058	20	46	24
Three, Four Six & Seven (Combined Open)	All Data		19,922	6,864	137		125 *
	P. West	141	19,462	5,411	105	76	105 *
	Slmntwn.	85	20,713	10,187	211	36	182 *

* Rounds Three and Four only.

As above, reducing this to a series of house status depositing-dust parameter ratios again facilitates interpretation of the data, eg $^{\text{combined house open}}/_{\text{combined house closed}}$ (Table 8.11).

Again, good correspondence between alfoil- and cup-collected lead is quite apparent, correlation coefficients (r =) for all closed and also for combined open being, respectively, 0.93 and 0.89, again revealing that the close relationship between alfoil- and cup-collected leadfall changes little if at all from house closed to (combined) open condition. Further, regressing (alfoil-collected) leadfall (dependent variable) against dustfall for both house closed and (combined) open status category yields correlation coefficients of 0.59 and 0.62 respectively, suggesting that the (source) relationship between collected lead and dust changes little with varying house status category.

Table 8.11 Depositing dust parameter ratios for two house status classifications.

COMBINED COLLECTION ROUND RATIO		RATIO OF GROUPED MEAN DATA			
		ALFOIL TRAY-COLLECTED			CUP-COLLECTED
		DUSTFALL	DUST LEAD CONC.	LEADFALL	LEADFALL
Three,Four,Six,Seven / One, Two & Five	All Data	7.8	1.2	9.6	7.6
	Pirie West	8.5	1.0	8.6	7.6
	Slmntwn.	6.4	1.7	10.8	7.6

Opening windows and/or doors contributes significantly to an increase in dust deposition inside rooms in (grouped) test houses, district-grouped test houses and also in individual houses. Such comparison shows that Solomontown residences appear to acquire disproportionately more contamination than Pirie West houses, with the contaminating dust affecting Solomontown residences revealing substantially greater lead levels. For combined district houses, all dustfall parameter mean values are greater under combined open-house conditions than closed ($P \leq 0.05$). Solomontown residences demonstrate equivalently ($P \leq 0.001$), with Pirie West also at this level apart from dust lead concentration which is not notably greater under such combined open-house conditions ($P \geq 0.05$). This infers a different source of dust affecting Solomontown residence dust collectors under open-house conditions, although a larger Solomontown sample size may not necessarily support this. Given the difference in (alfoil-collected) leadfall in such houses under these open conditions, it seems logical to again suggest that for these houses, the dust being collected under open-house conditions is externally-derived novel dust, whilst under closed-house conditions, substantially less dust is being collected. Contrasting this, however, it could possibly be construed that a greater proportion of dust collected under closed-house conditions than under open-house conditions derives from internal sources.

An inverse relationship exists between mean dust particle size and lead concentration (Chapter Five), and a direct relationship exists between average air and suspended particle speed within a room and the suspended particle size spectrum (Sinclair et al., 1985; Crockford, 1992). Acknowledging this, one would expect that if the collected dust derived predominantly from internal sources, then under higher average (in room) air velocities (ie open-house conditions), the average lead in deposited dust concentration would be expected to be lower than for closed-house conditions since a greater mean particle size would be collected, reflecting a lower mean lead in dust concentration. This has not been evidenced, Solomontown data demonstrating to the contrary. From Chapter Seven, the most substantial contaminant dust levels were detected in window sill/well locations, and hallway entrances, with pronounced gradients being detected in hallways, further suggesting external dust sources penetrating houses.

8.5.3.3 DISTRICT DIFFERENCES ROUND BY ROUND

Although summation of data in the above fashion provides insight into dustfall differences between the two districts and also the influence of the opening of doors/windows in the houses during the collection periods, interpretation of data in this fashion, however, may fail to decipher the potential for seasonal variations to influence dustfall and leadfall, hence the following interpretation (Table 8.12).

Some difference in mean collection round dustfall exists between the two districts, however, this is occasional and not the general rule. Such difference that does exist may be a local effect, or may simply reflect the small sample number of houses tested. Conversely, there is a much more consistent difference round by round between the two districts with regard to the quantity of lead falling onto collectors, with Solomontown in all cases but one (Round One - closed) displaying greater quantities of leadfall and a higher mean dust lead concentration than Port Pirie West. These Solomontown mean leadfall and dust lead concentration values are not always substantially greater. Notable difference, however, is demonstrated for Rounds Four, Five, Six and Seven, for dust lead concentration and for cup- and alfoil-collected lead ($P \leq 0.01$). Interestingly, in collection Round One, despite the dustfall being greater, both cup and alfoil-collected leadfall are lower, as also the dust lead concentration. This may simply be artefactual, or if not, suggests that, for a part of the year at least, there exists more than one major source of dust affecting home dust deposition. In all cases where difference is not detected at the $P \leq 0.01$ level, this remains so even at more relaxed confidence levels ($P \geq 0.25$), clearly reflecting either strong similarity or difference.

Removing the leadfall data (separately) corresponding to 136 Warnertown Rd. (proximate to a main road) or 64 Edward St. (furthest house from the smelter works - Figure 1.3) from the data base for Solomontown, does not alter those examples displaying substantial difference from their other district analogue. Solomontown is still the more contaminated location on the basis of the leadfall recorded in the (other) test houses. Equally, removing 136 Warnertown Rd dustfall data or 64 Edward St dustfall data changes the relationship of the two districts' dustfall data little. Given this rather small sample size of test houses in Solomontown, however, the possibility of data leverage existing as a result of the removal/inclusion of one subset of data must be assumed to be very real. Again reasonable agreement exists between alfoil and cup-collected leadfall.

Table 8.12 District differences in dust deposition round by round.

Difference refers to *t*-test difference; **x** = no difference; ✓ = difference ($P \leq 0.05$).

		GROUPED MEAN DATA					
		ALFOIL TRAY-COLLECTED			CUP-COLLECTED	
ROUND NO. & SEASON	LOCATION	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
ONE	Pirie West	31	3,454	5,677	20	31	20
(Summer)	Slmntwn.	14	3,696	3,225	12	14	15
	Difference		x	✓	x		x
TWO	Pirie West	28	2,627	3,158	8	29	12
(Sum/Aut)	Slmntwn.	14	3,063	3,218	10	15	12
	Difference		x	x	x		x
THREE	Pirie West	36	13,035	7,978	104	38	88
(Autumn)	Slmntwn.	16	14,295	8,479	121	18	111
	Difference		x	x	x		x
FOUR	Pirie West	36	34,829	2,976	104	38	126
(Winter)	Slmntwn.	16	31,844	8,223	262	18	300
	Difference		x	✓	✓		✓
FIVE	Pirie West	30	1,330	8,260	11	33	11
(Winter)	Slmntwn.	16	2,947	17,849	53	18	58
	Difference		✓	✓	✓		✓
SIX	Pirie West	32	20,167	6,557	132		-
(Spring)	Slmntwn.	25	18,747	13,405	251		-
	Difference		x	✓	✓		-
SEVEN	Pirie West	37	15,821	5,616	89		-
(Spr/Sum)	Slmntwn.	28	21,888	10,009	219		-
	Difference		✓	✓	✓		-

8.5.3.4 LEADFALL/DUSTFALL DISTRICT RELATIONSHIP - INDIVIDUAL COLLECTION ROUNDS

The relationship between depositing dust and the lead contained in that dust for the two districts is reflected below, incorporating regression of alfoil-collected leadfall (dependent variable) against dustfall (Table 8.13).

Table 8.13 District differences in leadfall/dustfall on a collection round basis.

	CORRELATION COEFFICIENT FOR ALFOIL-COLLECTED LEADFALL VERSUS DUSTFALL (r =)	
DUSTFALL COLLECT-ION ROUND	**PORT PIRIE WEST**	**SOLOMONTOWN**
One	0.62	0.89
Two	0.73	0.80
Three	0.59	0.87
Four	0.86	0.89
Five	0.68	0.87
Six	0.67	0.85
Seven	0.48	0.84
Average	**0.66** (RSD = 17.8%)	**0.86** (RSD = 3.9%)

Consistently throughout all collection rounds, the correlation coefficients corresponding to Solomontown dustfall/leadfall data are higher than those for Port Pirie West. More consistently thus, for Solomontown, the source of dust and the source of lead collected in the collectors are more closely-related than is the case for Port Pirie West. It was previously noted that quite large dustfall and leadfall differences existed between windows closed and windows open and also between windows closed and windows and doors open-house states. Given that there is little variation in leadfall/dustfall correlation coefficients for Solomontown for the different collection rounds (as compared to Pirie West data), the (lead-bearing) dust being collected in the test houses in Solomontown throughout this study likely derive principally from fewer external sources than is the case for Port Pirie West.

Under closed-house conditions, the correlation coefficients (r) for Pirie West dustfall and leadfall are very similar despite season, yet these coefficients reflect the most pronounced difference for window/door

open winter (0.86) and summer (0.48) alternatives. This may simply be artefactual; it is not supported by other summer-derived data. Alternatively, it may again suggest that in Port Pirie West there may be a greater number of sources of dust (of varying degrees of contamination with lead) being collected by the collectors than is the case for Solomontown, this being particularly evident in the summer season. Given the proximity of the Port Pirie West test houses to substantial, undeveloped areas of vacant land immediately to the west of the district boundary, more so than is the case for Solomontown test houses, this may reflect the enhanced penetration into test houses of (lower lead level) soil-derived dust resulting from a more pronounced vegetation decline over the Summer period.

The data presented above reflect some district and seasonal variability. On the basis of these data, however, the Solomontown test houses, essentially lying South of the smelter works, display a greater level of lead deposition than those in the other locale. This difference appears most pronounced in the Autumn/Winter period, a time when dustfall and leadfall would be higher in both districts (comparing Rounds One and Two with Round Five, and also Round Three with Six and Round Four with Seven). It is interesting to note that the highest concentrations of lead in dust are recorded for both districts in winter in Round Five, ie with the windows closed. Further, the Solomontown district reflects a disproportionately greater elevation in dust lead concentration over the Winter Spring months than has been evidenced for the Port Pirie West district. This may be related to seasonal prevailing wind direction and speed.[13]

The potential for a seasonal effect relating to the concentration of lead in indoor-deposited dust cannot thus be ignored, although as acknowledged previously, some data inconsistencies with (presumed) seasonal influences exist. This conclusion that lead deposition appears to be greater closer to winter than summer is interesting, given that during winter, the correlation between dustfall and leadfall is also more substantial for both districts. Further, quite apart from again suggesting that lead falling in the test homes is intimately related to dust entering these homes, (ie dust is the vehicle for lead entry into homes), this conclusion suggests that there exists one/few principal dust source(s) of contaminant material entry as collected in the test houses.

8.5.4 DUSTFALL HOME BY HOME

Given that the houses involved in this project are located in various sections of two districts in Port Pirie, it is of advantage to examine differences in levels of dustfall and leadfall on a house by house basis. A table of mean values for dustfall, leadfall, and dust-lead concentration on a house by house basis appears below (Table 8.14). Dustfall variability from home to home is substantially less than home to home dust lead concentration variability (RSD values 33.0 and 49.3% respectively), inter-home mean dust lead concentration reflecting a 4.7 times range. Inter home mean leadfall variability is the largest of the three alfoil-determined dust deposition parameters, exhibiting a coefficient of variation of 57.4%. Again, there is good agreement between the two collection methods.[14]

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[13] Cartwright (1976) demonstrated that the greatest soil contamination by smelter-emitted metals lay in the prevailing wind direction (NNW - SSE at near ground level, and predominantly from the west at high-level), although concluding that most of this contamination occurred before 1940.

[14] Recalling from above, *t*-test comparison of means for (pooled) alfoil-collected versus cup-collected leadfall yields no apparent difference ($P \geq 0.40$), and similarly on an individual (collection) round by round basis ($P \geq 0.20$).

Table 8.14 House by house comparison of depositing dust data.

HOME LOCATION	GROUPED MEAN DATA						Correl. Coeff. Alfoil- versus Cup-collected Leadfall * (r =)
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED		
	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	
64 Edward St	**21**	9,846	14,865	146	**6**	48	0.98
58 Federatn. Rd	**40**	6,167	5,931	37	**28**	25	0.94
29 Fourth St	**26**	11,373	3,229	37	**23**	27	0.97
19 Lily St	**60**	6,963	6,159	43	**46**	36	0.98
3 Sabine St	**45**	12,430	7,494	93	**28**	57	0.97
35 Square St	**21**	7,532	6,620	50	**19**	39	0.99
27 Vera St	**71**	7,994	7,030	56	**50**	38	0.97
136 Warnrtn. Rd	**42**	12,198	8,355	102	**31**	79	0.97
97 York Rd	**33**	16,549	3,182	53	**22**	51	0.93

* Based only upon Rounds One to Five for which both alfoil trays and paper cups were used.

Simple regression between these depositing dust parameters for the various houses revealed the following (Table 8.15).

The most consistent correlation, and also the one at the highest mean level is that of dustfall against (alfoil tray-collected) leadfall. Such good correlation implies that the source of the lead and the dust are closely related, and that the dust depositing within individual test houses derives from (principally) perhaps only a small number of different dust sources. Given that all such individual house correlation coefficients indicate a high likelihood of a linear relationship between the quantities of (collected) dust and lead, and that given that there is a reasonable spread in location of these test homes around the two districts, (themselves spread around the confines of Port Pirie) it seems likely that the nature of the dust (lead) affecting the test homes collectively is common to all/most of these individual houses.

On a home by home basis, no real difference between cup- and alfoil-collected leadfall exists for all houses ($P \geq 0.15$) apart from 58 Edward St, although here cup-collected sample size is quite small. Further reflecting the good linear relationship between the two techniques over a range of lead concentrations (3.5 times leadfall range from lowest to highest), are the indicated correlation coefficients (Table 8.15).

Table 8.15 Regression of depositing dust parameters on an individual house basis.

HOME LOCATION	CORRELATION COEFFICIENT (r =)	
	DUSTFALL VERSUS LEADFALL *	LEADFALL VERSUS DUST LEAD CONC.
64 Edward Street	0.91	0.25
58 Federation Road	0.87	0.39
29 Fourth Street	0.88	0.34
19 Lily Street	0.89	0.31
3 Sabine Street	0.92	0.83
35 Square Street	0.93	0.48
27 Vera Street	0.88	0.38
136 Warnertown Road	0.89	0.53
97 York Road	0.89	0.51
Average	**0.90** (RSD = 2.3%)	**0.45** (RSD = 38.4%)

* Alfoil tray-collected data only.

Additionally, for these test houses, a principal determinant of the degree of contamination affecting a certain surface area is the quantity of lead-containing dust falling onto that area. This precept is certainly not challenged by the much less consistent and very average relationship between leadfall and the dust lead concentration as indicated above. Simply put, there do exist differences in leadfall from house to house as would be anticipated due to location, dust source(s) and perhaps house construction. These differences, however, are not related principally to just the lead-in-dust concentration of collected in-home dust as would be implied by many earlier, separate research investigations where dust lead concentration was the sole dust parameter assessed in order to determine the dimension of site contamination; leadfall data must also be presented.

There exists a large difference between the quantity of dust and lead collected under house open as opposed to closed conditions (Tables 8.8 & 8.9 above). Further, there is a (relatively) small variation in dust lead concentration determined from fully-open to closed-house test periods (ratios of 0.8 and 1.5 for Port Pirie West and Solomontown respectively) as compared to differences in dustfall (10.2 and 7.8 respectively) and leadfall (7.8 and 12.0 respectively). Under such circumstances, it is apparent, again, that collected dust lead concentration may not be the most appropriate indicator of the dynamic contamination pattern of a room.

Differences, however, do exist between the test houses in the context of differences in dustfall, leadfall and lead-in-dust concentration. This is obviously reflected by Table 8.14 above depicting mean levels for

these variables, and is to be expected, reflecting differences in house location, aspect, construction type and possibly also degree of dustiness at time of dust deposition study.

There exist only five instances where notable difference is registered between any of the home-home permutations for cup-collected leadfall, which compares with 13 permutations for alfoil-collected leadfall ($P \leq 0.05$). It is acknowledged that there were fewer collections made with cup collectors, yet on a house by house basis, there has not been demonstrated any difference between the two dust-collection techniques for any house apart from 64 Edward St, where sample size for cup collectors was not large ($P > 0.05$). Whilst this is by no means an exhaustive comparison between the two techniques, it is appropriate to conclude that the cup-collection technique has demonstrated here that it is a somewhat less sensitive technique for leadfall collection compared with the alfoil technique. Cups (or similar devices) have been used by a number of workers to collect dustfall within homes (eg Aurand et al., 1983; Seifert et al., 1984; Bernigau et al., 1993), but typically though with an effective dust-collecting surface area substantially smaller than that which was utilised here.

Of all the test houses reviewed, only the house located at 27 Vera St had been involved with the home decontamination program underway in Port Pirie at time of study.[15] This house had been thoroughly de-dusted and cleaned, new fences, ceilings, inner walls, cornices, door architraves and roof installed, and top-soil (to a depth of 15cm) replaced (P. Bartholemew, pers. comm.) all within the preceding 15 months of the commencement of study (C. Phipps, pers. comm.). It is noteworthy that the mean level of leadfall for this house is the highest of all Port Pirie West houses.

8.5.5 SAMPLING LOCATION DISTANCE FROM SMELTER

Cognisant of house to house dust deposition variability, the level of contamination of the deposited dust sample (dependent variable) was regressed against four distance alternatives between sampling site and various smelter features (Table 8.16).

Increasing distance from smelter correlates poorly with increasing dustfall, leadfall and dust lead concentration level, several regression permutations for which demonstrate no relationship between depositing dust parameter (dependent variable) and distance ($P \geq 0.10$). Correlating the three dust parameter values (dependent variable) with the square of the distance from one of the smelter features to the houses, reflects no less poor a correlation coefficient also, the absence of any firm relationship being apparent ($P \geq 0.15$). Thus, on the basis of these data, little inference can be drawn regarding the relationship between distance from smelter and degree of contamination of homes in the context of depositing dust and its potential source.

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[15] In this regard, initial consideration was apparently given to a suggestion by Landrigan (1983) regarding relocating children at Port Pirie who lived near the smelter. This option was not pursued. As an alternative, it was considered appropriate, given "the very low level to which emission losses had been reduced" (Body et al., 1988; p. 17), that a process of home decontamination and dust suppression would be equally effective at reducing child blood levels (P. Body, pers. comm.). This option would not appear to have been as effective as initially presumed (Luke, 1991), blood lead levels in children initially being reduced upon relocation, but then subsequently increasing again upon return home. The only environmental factor found to be a significant predictor of a lowering of blood lead levels in the specific cohort of children assessed was permanent relocation (Maynard et al., 1993).

Table 8.16 Depositing dust contamination level variation with square of distance from smelter.

	CONTAMINANT LEVEL VERSUS DISTANCE CORRELATION COEFFICIENT (r =)			
	DEPENDENT VARIABLE			
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED
INDEPENDENT VARIABLE	DUSTFALL	DUST LEAD LEAD CONCN.	LEADFALL	LEADFALL
SQUARE OF THE DISTANCE FROM THE SAMPLING LOCATION TO:				
Nearest smelter boundary	0.14	0.20	0.23	0.15
Wharf - concentrate unloading point	0.14	0.14	0.20	0.12
Smelter works central point	0.14	0.18	0.21	0.14
Smelter stack	0.14	0.13	0.19	0.11

A substantially increased sample size of houses with a greater range of distances from the various smelter features may, however, be more helpful in this regard. Supporting this, as part of a dustfall collecting method validation study (Appendix Sixteen), depositing dust was collected in an identical fashion from within a building within the grounds of the smelter.[16] From 16 collectors, mean data for dustfall and leadfall were respectively 9,454 and 488.7$\mu g.m^{-2}.d^{-1}$, with a mean dust lead concentration of 51,694$\mu g.g^{-1}$. By comparison with the above data, this implies that proximity to the smelter does relate (inversely) to level of contamination to some degree, there being insufficient data points to examine this relationship more fully.

8.5.6 DUST DEPOSITION DIFFERENCES BY ROOM

As mentioned above, the layout of these homes is very similar, with essentially four main rooms symmetrically distributed either side of a central hall/passage extending from front to back doors, with bathroom and kitchen at the rear of the dwelling. Room to room depositing dust differences were

16 The construction of this former store room, was not dissimilar to that of the test houses. It was located approximately 250 metres from the smelter centre and stack, and 350 metres from the wharf.

investigated by comparison between bedrooms, halls, lounge rooms and kitchens over the entire test collection period.

8.5.6.1 ROOM TYPE

Summary mean depositing-dust parameters room by room are indicated below (Table 8.17).

Table 8.17 Mean room dust deposition parameter summary values.

GROUPED MEAN DATA							
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED		Correl. Coeff. Alfoil- versus Cup-collected Lead (r =)
ROOM	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	
Bedroom	**198**	9,023	6,582	59	**134**	36	0.97
Hall	**52**	11,997	6,275	75	**37**	66	0.98
Kitchen	**34**	8,500	5,845	50	**32**	40	0.96
Lounge	**71**	7,706	5,902	46	**49**	33	0.95

Again, it is evident that both dust-collecting techniques display good agreement, there appearing no difference between the two for collected leadfall ($P \geq 0.60$). As may be readily seen, hall-collected dustfall, and leadfall (both alfoil- and cup-collected) reveal the greatest mean levels. Comparison of these room data sets reflects no difference for any dustfall parameter for any permutation of lounge, bedroom and kitchen, acknowledging a small sample size for kitchens ($P \geq 0.20$). For lounge and bedroom comparisons this does not seem surprising given the very similar physical nature of these (one of four) main rooms.

Grouping the data for lounge and bedrooms, and comparing to kitchens, again reveals no difference for any dustfall parameter ($P \geq 0.50$). Comparison of hall-collected dust with that collected from other rooms paints a similar picture. No notable difference was apparent between hall-collected dust parameters and any of kitchen, bedroom or grouped lounge and bedroom-collected samples ($P \geq 0.05$). Contrasting this, however, comparison of hall with lounge room samples reveals, for halls, demonstrably greater levels of dustfall and leadfall as detected by both techniques ($P \leq 0.05$). No comparison of any single room class with halls illuminated difference for dust lead concentration. This was also the case for combined lounge and bedroom class comparison with halls ($P \geq 0.15$).

Given that these room data span two dust-collecting cycles with collection continuing for nearly twelve months, it is reasonable to assume that these data provide a fair representation of average dust deposition patterns within rooms of Port Pirie houses. Contrasting the results from the static dust study (Chapter Seven) where halls demonstrated greater deposited contaminant levels, halls here do not seem to represent more contaminated locations. In fact, also reflected by depositing dust data, are much less significant dust-depositional differences between any permutations of other rooms than analogous comparison of static dust levels has revealed.

Acknowledging that such apparently featureless results may be influenced by the open/closed status of the houses at the time of dust deposition, it appeared appropriate to make such room comparisons for either house closed or (combined) open conditions.[17]

For combined Rounds Three, Four, Six and Seven (house open), statistical comparison of all bedroom, kitchen and lounge room permutations (including grouped bedroom and lounge) for all dust deposition parameters reflects no difference ($P \geq 0.15$). For all comparisons of halls with other rooms, this is also the case apart from bedrooms versus halls and bedrooms/lounge rooms compared to halls where recorded hall leadfall is lower ($P \leq 0.05$).

For combined Rounds One, Two and Five (house closed), comparison of room permutations for bedrooms, lounge rooms and kitchens (including grouped bedroom and lounge) for all dust deposition parameters again demonstrates no difference ($P \geq 0.15$). Contrasting this, however, under such closed-house conditions, for all room with hallway comparison permutations (including grouped bedroom and lounge) for all dustfall parameters apart from dust lead concentration, all mean hall dust deposition parameter values are from moderately to substantially greater ($P \leq 0.05$).

Such data suggest that under closed-house conditions, when within-house air movements are (presumably) less intense, clear differences in inter room dust deposition become apparent, hallways reflecting that they are areas where more pronounced dust deposition occurs. From previously, (Chapter Seven) hallways demonstrated substantially greater static dust and lead levels than other rooms, after essentially a period of up to three months of house vacancy subsequent to former occupant departure, during which time the house remained closed. Such observations infer that static dust patterns as assessed, essentially reflect (predominantly) a house-vacancy period of deposition of newly-entering house dust.

Under such closed home conditions with less-disturbed air (internally) it is suggested that dust penetration into houses is less effective, as also the degree of mixing of within-home air-suspended material. Further, depositional differences between rooms clearly indicate greater levels in hallways, indicating that hallways act as primary house contamination sites/conduits.

Under open-house conditions, however, when within-home air currents are greater, such analogous room to room differences in dust deposition rates do not appear to exist. In fact, under open-house conditions, leadfall is greater in bedrooms and grouped bedrooms/lounge rooms than halls.[18] It thus seems likely that enhanced distribution of potentially-depositable material within the house occurs during this

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[17] Given modest sample size for semi-open and fully-open house categories, and that for individual houses, no difference exists for any dust parameter for semi-open versus open category, these two data sets will be grouped here, resulting in essentially a comparison of closed houses and non-closed houses.

[18] Under open-house conditions on windy days, homes demonstrated detectably enhanced within-home air movements, particularly in hallways (pers. obs.).

time, presumably as a result of greater air movements of suspended contaminated dust streaming into other rooms.

Regarding the concern of whether the dustfall registered in these collectors arises as a result of the re-entrainment of existing surface dust within the room or, alternatively, as a result of the deposition of novel dust entering the home, these data suggest that the latter is more likely. Further reference to this concern will be made later in this chapter (Section 8.6).

From these room to room comparisons, no dust lead concentration differences have been revealed. This implies that the contaminating dust pool demonstrates some uniformity, although this is hard to reconcile with the noted occasions where, for example, leadfall may be very different for certain room to room permutations, but with difference not also being registered at dustfall level. Given the good correlations between dustfall and leadfall as revealed previously, this concern remains unanswered at this stage.

8.5.6.2 MAIN ROOM DIFFERENCES

Summary depositing dust data corresponding to each of the four main rooms appears below (Table 8.18).

Table 8.18 Main room comparisons - all sampling rounds combined.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED	
ROOM LOCATION	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Left Front	**71**	8,921	6,010	53	**48**	37
Right Front	**78**	12,953	6,456	84	**49**	51
Left Rear	**47**	5,407	6,926	43	**37**	26
Right Rear	**73**	7,394	5,858	43	**49**	29

For all data categories, the mean category values corresponding to front rooms appear numerically greater than for the corresponding (same-side) rear room, apart from left rear room dust lead concentration. Comparison of (combined) front with (combined) rear room data revealed greater mean values for all dustfall parameters except dust lead concentration for front rooms ($P \leq 0.05$). Further, comparison of dust deposition data corresponding with each of the two front rooms of test houses revealed no apparent difference for any of the four test variables, and, similarly, comparison between the two rear

rooms also revealed no difference for any of the dust deposition parameters ($P \geq 0.05$). Grouped left-side versus right-side room data displayed no real difference for any dustfall parameter ($P \geq 0.05$).

Same-side front room back room comparisons for either left or right sides of the test houses revealed that for all dust deposition variables there was no difference between (same-side) front and rear rooms ($P \geq 0.10$). Analogously, front left versus rear right room data comparison reveals no difference for any dust deposition parameter ($P \geq 0.20$). An interesting result however was elicited by comparison of variables corresponding to front right (most dustfall, most leadfall) and rear left (least dustfall, least leadfall) rooms. Such comparison exhibits, for front right rooms, greater mean values for all variables except lead-in-dust concentration ($P \leq 0.02$ level).[19] This is a noteworthy observation particularly given the level of significance at which the difference was registered. It is also so given the room differences as suggested by the static floor dust loading study (Chapter Seven) where front rooms were, again, demonstrably more contaminated than rear rooms ($P \leq 0.025$), and the rear left room was typically the least contaminated of rooms. Given these room similarities in static and dynamic dust levels, at least some contribution to contamination levels in main rooms must be due to entry of novel contaminant dust. Additionally, dust penetration into the rear main rooms from (say) all room windows and also hallway and kitchen/back porch entry ways is less than front rooms probably due to the larger mean distance from external doorways to these rear main rooms. Essentially, (reduced) dust penetration into such rooms is perhaps due to a "buffering" effect provided by front rooms and also kitchen/back porches.[20]

Comparison of dust deposition data for halls with combined front room data reveals that there exists no difference for all dust deposition parameters ($P \geq 0.40$) apart from cup-collected leadfall ($P \geq 0.10$). Difference does exist however between halls and combined rear rooms for all dust deposition parameters, mean hall values being greater ($P \leq 0.002$) apart from dust lead concentration where no demonstrable difference exists ($P \geq 0.15$). Analogously, there was no real difference detected between hall data and combined front rooms centre of floor only data ($P \geq 0.05$) or hall and front room window-base floor data ($P \geq 0.15$). This infers that both halls and windows are sources of substantial dust entry into the front rooms assuming the dust collected is novel dust. Such comparison for rear main rooms reveals no difference between hall-derived data and window-base floor data ($P \geq 0.20$). For hall and rear room centre of floor data comparison, however, hall-derived dustfall and leadfall are notably larger ($P \leq 0.001$), but not dust lead concentration ($P > 0.075$).

These results strongly suggest that novel depositing dust contributes significantly to contaminant dust that is detected statically (ie with the vacuum pump apparatus) in vacant homes. Further, the data imply that the pathways of dust entry into main rooms are both through the front door/hall and window in these rooms, a contribution (presumably) also deriving from the rear entryway. It is further suggested that the contribution from the hall would be more noticeable in the front main rooms than is the case for the rear main rooms, and that there exists a decrease in dust deposition levels with distance into the house (as Al-Radday et al., 1993a). In a sense, front room/rear room differences and hall/rear room differences (in the

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[19] Such room to room comparison of main room dust deposition levels subsequently conducted using centre of room data only, revealed virtually identically to above comparisons involving centre of room plus window floor-sampled data. The only exception to this was the right rear/right front room comparison, where mean dustfall and leadfall were shown to be greater for the latter room ($P \leq 0.02$). Such observations are consistent with results obtained by Dietz and Cote (1982) indicating that the relative penetration of a perfluorocarbon tracer material into rooms of a house varied directly with the outside-wall exposure of that room.

[20] Similarity between static and dynamic dust deposition patterns implies that the elicited static dust data reflect novel deposition as would be anticipated in a vacant closed house.

absence of hall front room differences), simply underscore the importance of the contribution of additional contaminant entry into rooms of such houses via (initially) the hallway. Such a conclusion is in keeping with observations of main room (static dust) lead loading inverse gradients into the room with the square of the distance in from the window, and is also commensurate with data corresponding to hallways, (typically extending the depth of the front rooms), demonstrating substantial (inverse) gradient effects with the square of the distance in from the front door, eg for lead loading $r = 0.77$ ($P \leq 0.0025$), (Chapter Seven).

These houses were unoccupied during these test periods. Thus it is likely that normal air movements within the structure would contribute to the movement of dust particles within the domicile. Human activity (including cleaning) within the home when occupied, apart from constituting a potential dust input source (as Fergusson and Kim, 1991), could only be expected to contribute to a more effective mixing of such contaminant material within the home. Further, a period of (undisturbed) vacancy subsequent to occupant departure enhances the prospect of a dramatic build-up of novel, depositing dust; recall a virtual order of magnitude greater (static) dust/lead loading level in vacant compared to occupied houses (Chapter Seven). These concepts promote the theme that such depositing dust as is being collected here is not simply re-entrained (pre-existing) dust, but more an example of novel dust entry into the home.

If it is assumed that dust deposition arises simply as a result of re-entrainment and subsequent deposition of existing (within-room) dust, there would appear to be no strong reason for re-entrainment/re-deposition rates to vary between front and rear main rooms. Such logic appears at variance with the similarity between noted static dust/lead loading levels (which have been demonstrated to be greater in front main rooms, adjacent to presumed entrance-ways), and dustfall/leadfall rates which also reflect such static dust room differences.

This dust-collection system is sufficiently sensitive a technique to illuminate between-room dust deposition differences (eg between front and rear main rooms). Noting that such difference is small by comparison with rear room versus hall difference, it is not likely that the absence of a detected dustfall/leadfall difference between grouped left and grouped right main rooms arises simply as a result of the poor sensitivity of this dust-collecting technique.

Acknowledging these above observations, it seemed appropriate to investigate whether any room to room differences were observable under conditions of either house fully closed (combined Rounds One, Two and Three - Table 8.19) or house open (combined Rounds Three, Four, Six and Seven - Table 8.20) conditions, a summary appearing below.

As with the (combined round) summary data, these above data for house closed and (combined) house open alternatives reveal that the rear left room displays the lowest mean value for dustfall and leadfall (cup and alfoil-collected), and the right front room the greatest mean value.

Comparison of combined front main rooms with combined rear main rooms (separately for house closed and combined open alternatives) demonstrates, for front rooms, greater mean values for all dust deposition parameters except dust lead concentration, for both house status categories ($P \leq 0.02$). This further emphasises that the front rooms of these test houses represent areas of greater contaminant deposition rates.

Table 8.19 House closed main room dust deposition summary data.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED	
ROOM LOCATION	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Left Front	**28**	2,021	5,143	10	**28**	12
Right Front	**29**	3,401	5,630	19	**29**	22
Left Rear	**27**	1,325	6,833	9	**21**	12
Right Rear	**29**	2,038	4,860	10	**29**	12

Table 8.20 House open main room dust deposition summary data.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED	
ROOM LOCATION	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Left Front	**43**	23,463	6,652	156	**20**	173
Right Front	**49**	28,584	7,000	200	**20**	174
Left Rear	**30**	11,996	8,621	103	**17**	77
Right Rear	**44**	17,287	6,626	115	**20**	113

Left rear versus right rear room comparison reveals no pronounced difference for any dust deposition parameter except dustfall for either house status alternative ($P \geq 0.05$). The situation for left and right front rooms, however, is not so clear cut, in that under (combined) house open conditions, no difference exists for any of the dust parameters ($P \geq 0.10$), yet under house closed conditions, right front rooms display greater dustfall and leadfall (measured by both techniques) ($P \leq 0.01$). This situation is somewhat mimicked by combined left room versus right room comparison, where again under (combined) open-

house conditions, no difference is detected between the two alternatives for any deposition parameter ($P \geq 0.10$), yet when houses are closed, all parameters except dust lead concentration are greater in combined right main rooms ($P \leq 0.05$). This supports the concept that mixing of dust under house open conditions is more pronounced, together with significantly greater dustfall and leadfall levels than under house closed conditions, this "swamping" the more subtle deposition room differences detected under closed-house conditions.

Left front versus right rear main room comparisons for both house status alternatives reflects no difference for any deposition parameter, ($P \geq 0.05$), whilst right front and left rear main room comparisons for both house (combined) open and closed alternatives reveals difference for all variables except lead-in-dust concentration ($P \leq 0.01$). Albeit small sample size, the rear left room is typically less dusty and reflects less marked leadfall than other rooms, particularly the front right.

Under closed-house conditions, hallway and combined front room comparisons display, for all dustfall parameters apart from dust lead concentration for which there appears no difference ($P \geq 0.25$), greater mean levels for halls. This is also the case for hallway and rear room comparisons ($P \leq 0.002$). Under (combined) open-house conditions, however, very much the opposite is the case, with no difference between combined rear room with hallway-sampled dustfall, alfoil- and cup-collected leadfall ($P \geq 0.25$) and dust lead concentration ($P \geq 0.15$). Analogous combined front room with hallway comparison reveals difference for only alfoil-collected lead, the front room mean level being the larger ($P \leq 0.05$). The remaining parameters reveal minimal difference ($P \geq 0.20$). Again, these differences according to house open or closed status are substantial and reflect the effective mixing of contaminants within the house under open conditions and their much more even spatial distribution (than is the case under house closed conditions).

8.5.7 PRESENCE OF A FIREPLACE IN A MAIN ROOM

Comparison of main rooms with a fireplace with those devoid of a fireplace was conducted in order to determine whether differences in dust deposition rates existed. From Chapter Seven, it will be recalled that rooms were described according to whether there was a fireplace present, some rooms revealing the presence of a modified fireplace. Data corresponding to this last-mentioned category will not be presented here given its small sample size. Comparison of the dust collection data corresponding to rooms with an open fireplace and those without reveals no detectable difference between the two categories for all dust-collection parameters ($P \geq 0.40$). Subsequent separate comparison of these two categories into house closed and house (combined) open modes reveals similarly for all dust-collection parameters at respectively $P \geq 0.10$ and $P \geq 0.50$. Despite there being no apparent contribution to room dust deposition by fireplaces, from the differences in significance level illustrated immediately above, it would seem that under closed-house conditions the potential for such difference would appear more likely than under open-house conditions. This underscores earlier room to room depositing dust differences observed under closed but not open-house conditions, further suggesting the between-room mixing of suspended dust, particularly under open-house conditions.

A summary of dust collection details appears in Table 8.21.

Table 8.21 Influence of an open fireplace in a room on depositing dust.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED		
FIREPLACE STATUS	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
None	190	8,064	6,306	51	134	33
Open	90	8,982	5,987	54	66	40

During dustfall collection, alfoil collectors were located centrally and peripherally (window-base floor) in each room. Thus it may be possible that the influence of dust deposition levels at window-base floor locations, may "swamp" the more subtle nature of fireplace influences upon dust deposition patterns within a room. Hence a comparison of open fireplace versus no fireplace rooms was made using (room) centrally-collected data points only. Such comparison displays no apparent difference ($P \geq 0.25$) for all dust-collection parameters for:

a) Rounds One, Two and Five,
b) Rounds Three, Four, Six and Seven, and also for
c) All sampling rounds (Rounds One to Seven).

As reflected by such comparison above, these data indicate that the presence of a fireplace in a room has not been demonstrated to be an indicator of enhanced contaminant dust deposition onto (floor) surfaces, either grouped or (room) centrally-located only.

8.5.8 WINDOW VERSUS CENTRE OF ROOM LOCATION

Comparison of data suites corresponding to the two room locations for all collection rounds reflect larger dustfall and (alfoil-collected) leadfall for window-base floor locations ($P \leq 0.05$). Separate comparison of data corresponding to dust-collection under (combined) open-house and closed-house conditions, demonstrates, again, numerically greater mean values for all window base-derived dustfall parameters. Under (combined) open-house conditions dustfall and leadfall are larger ($P \leq 0.025$) similar to results obtained by Al-Radday et al. (1993a), although under closed-house conditions less substantial difference was detected for these two dust parameters ($P \leq 0.075$). Greater sample size here may be

helpful. Dust lead concentration reveals virtually no difference between either location under either house condition ($P \geq 0.20$), although this may simply reflect the proximity of the two sampling locations and/or insufficient sensitivity of collection apparatus. A comparison of window-located and centre of room-located depositing dust collectors provided the following summary data (Table 8.22).

Table 8.22 Within room differences in depositing dust levels.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED	
COLLECTOR LOCATION	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Window base	**64**	12,895	7,432	91	**34**	42
Room centre	**252**	8,073	5,981	48	**190**	33

Further pursuing window-base floor and centre of room depositing-dust investigation, paired window-base floor to room centre sample comparison was conducted. Such comparison reveals for combined (all) rounds, very slightly greater mean values for dustfall and alfoil-collected leadfall for window-base locations ($P \leq 0.075$), and similarly for closed-house rounds ($P \leq 0.065$). Under house open (combined) conditions, however, these location differences become substantially more pronounced, with window-base locations demonstrating greater mean values for dustfall and leadfall ($P \leq 0.025$), dust lead concentration not really differing between locations for either house condition ($P \geq 0.10$).

This observation may appear to conflict with earlier-stated observations regarding detectable dustfall parameter differences between hallway and front and rear main rooms under closed but not open-house conditions, reflecting more effective suspended particle mixing under the latter situation. This apparent paradox remains unanswered at this stage. Clearly, however, rates of dustfall/leadfall are heavily influenced by proximity to the (open or otherwise) window in the room.[21]

These differences in dustfall and leadfall between the two floor locations under (combined) open-house conditions suggest either:

a) that (subsequently-deposited) dust entry is through the (partly open) window, its room penetration declining with distance into the room, or

[21] Quite obviously this is highly significant in a town such as Port Pirie where windows are frequently left open for extended periods of time (given the hot, arid climate), and also acknowledging the deteriorated and frequently derelict nature of many houses in these two "high risk" districts, cracked/broken windows not being an uncommon sight.

b) greater re-entrainment and deposition of existing dust occurs proximate to the window presumably as a result of greater air movement at that location,

or a combination of both.

Given observations regarding more pronounced static dust levels at window-base floors as compared to room centre sampling locations, alternative b) does not seem as likely, ie again the concept of novel dust entry seems more probable.

The concentration of lead in collected dust does not differ significantly between the two alternatives despite there being large differences between mean dustfall/leadfall values under varying house conditions; essentially, the contaminant dust collected in the two collection locations derives largely from similar sources, or from the same source. Simple regression analysis of appropriate depositing dust variables from such a comparison elicited correlation coefficients as indicated below (Table 8.23).

Table 8.23 Regression analysis of window versus centre depositing dust parameters.

	CORRELATION COEFFICIENT (r =)			
		ALFOIL TRAY-COLLECTED		ALFOIL- VERSUS CUP-COLLECTED
COLLECTOR LOCATION	DUSTFALL VERSUS LEADFALL	DUSTFALL VERSUS DUST LEAD CONC.	LEADFALL VERSUS DUST LEAD CONC.	LEADFALL
Window base	0.92	0.02	0.41	0.98
Room centre	0.86	0.11	0.41	0.96

As has been noted previously, on an individual home by home sampling basis and for grouped data above, dustfall and leadfall are also clearly linearly-related. Similarly here this would also appear to apply for both sampling locations. Given the variety in test house location around Port Pirie and also varying dustfall and leadfall levels, the strong relationship between these two parameters again implies that collected dust generally appears to derive from a few similar or one predominant lead-bearing dust source. From these data, dustfall explains approximately four times the variability in leadfall that dust lead concentration does.

Not inconsistent with this concept, acknowledging the large variation in dustfall and leadfall levels measured at the various test locations over the entire dust-collection series, no firm relationship between either dustfall or leadfall and dust lead concentration for either test location is apparent ($P \geq 0.15$).

Further supporting this concept that the strong correlation between dustfall and leadfall reflects a single lead-in-dust source ("pool"), comparing the relative variability in dustfall, leadfall and dust lead concentration for the two locations, substantial differences are evidenced. Whilst for dust lead concentration there exists a substantial range of between 2,000$\mu g.g^{-1}$ and 36,000$\mu g.g^{-1}$ (for both windows and centres combined), expressed as a coefficient of variance, the variability in dust lead concentration appears markedly less than that for (say) dustfall or leadfall (Table 8.24).

Table 8.24 Variability in depositing dust variables according to location in room.

PARAMETER	COEFFICIENT OF VARIATION DATA (%)	
	WINDOW FLOORS	ROOM CENTRES
Dustfall	15.1	14.1
Alfoil-collected leadfall	32.0	34.0
Cup-collected leadfall	39.2	36.6
Dust lead concentration	6.5	8.0

Little can be concluded regarding differences between the two room sampling locations, both sampling locations being fairly similarly influenced by contaminating dust. Such small variability in dust lead concentration by comparison with leadfall variability supports a common dust lead pool concept, and suggests that the sources of lead collected in the dust collectors are fewer in number than the sources of dust.

8.5.9 DEPOSITING DUST GRADIENTS FROM A WINDOW INTO A ROOM

Regressing dust deposition levels (dependent variable) with the square of the distance into a room perpendicularly from the window, elicited the following results.

For combined data for all bedrooms, kitchens and lounge rooms, and incorporating all dust-collection rounds, regressing the various dustfall parameters (dependent variable) against the square of the distance into the room perpendicularly from the window, exhibits moderately poor correlation coefficients. Here, r lies in the range 0.13 - 0.25, the latter corresponding to alfoil-collected leadfall, such relationships reflecting reasonable significance ($P \leq 0.05$).[22] Markedly poorer relationships appear under house closed conditions for all parameters ($r < 0.075$; $P \geq 0.30$) apart from alfoil-collected leadfall ($r = 0.10$; $P < 0.04$).

[22] Initially regression was carried out against the distance into the room, and subsequently the square of the distance into the room, correlation coefficients for the latter being consistently better, albeit only marginally.

Under (combined) open-house conditions, a modest correlation for dustfall ($r = 0.42$; $P < 0.001$) and alfoil-collected leadfall ($r = 0.492$; $P < 0.001$) results, the latter being similar to the analogous figure for cup-collected leadfall ($r = 0.487$; $P < 0.005$). Dust lead concentration indicates substantially poorer correlation ($r = 0.19$; $P < 0.04$).

Regression relationships for either house closed or house (combined) open are as follows:

a) House closed:

ln dustfall = 7.8 - (0.01 x square of distance into room) ,
Correl. coefficient (r) = 0.03

ln leadfall (alfoil tray) = 2.6 - (0.04 x square of distance into room),
Correl. coefficient (r) = 0.10

ln dust lead concentration = 8.7 - (0.03 x square of distance into room), and
Correl. coefficient (r) = 0.07

ln leadfall (cup) = 2.7 - (0.02 x square of distance into room).
Correl. coefficient (r) = 0.04

b) House open:

ln dustfall = 10.3 - (0.1 x square of distance into room),
Correl. coefficient (r) = 0.42

ln leadfall (alfoil tray) = 5.6 - (0.2 x square of distance into room),
Correl. coefficient (r) = 0.49

ln dust lead concentration = 9.1 - (0.07 x square of distance into room), and
Correl. coefficient (r) = 0.19

ln leadfall (cup) = 5.5 - (0.2 x square of distance into room)
Correl. coefficient (r) = 0.49

The correlation coefficients relating distance and dustfall parameters for all the collection rounds combined are a composite of the analogous data corresponding to either house open or closed conditions and as such are not particularly insightful as regards dust deposition in rooms of houses. Clearly, corresponding correlation coefficients for closed versus (combined) open-house conditions, reflect, for the latter, substantially larger values, implicating the window (particularly when open) as a source of internal contamination, given the spatial relationship of the dust collectors in the room.

For all dust-collection rounds (separately), the (-ve) slope of the regression relationship for leadfall is always the most pronounced of the dustfall parameters, corresponding dustfall slopes being moderately ($\approx$ 20%) less steep. Given that the slope for dust lead concentration is typically from moderately to substantially poorer (and the relationship less significant), this would suggest that the majority of the collected dust contains contaminant material, a significant pathway for contaminant entry into the room being through the window.

Further, under open-house conditions, such sharper dust deposition gradients into a room accompanied by significantly heightened dust/lead deposition rates (Table 8.4) together with substantially larger static contaminant levels at window-base floors than room centre locations (Chapter Seven) would seem to

clearly indict windows as a major entry point for contaminant dust into such homes. That a greater proportion of the spatial leadfall variability is not explained simply by distance from a window into the room, suggests that a degree of within-room mixing of suspended material occurs under normal room conditions.

8.5.10 ROOMS ADJACENT TO WALKWAY VERSUS ROOMS OPPOSITE

Comparison of dust deposition data corresponding to rooms adjacent to the walkway (with a window facing on to it) and rooms on the other side of the house revealed no difference for any of the dust deposition parameters ($P \geq 0.50$). Equally, testing for difference between just the two back main rooms or the two front main rooms of test houses similarly categorised, again revealed no difference for any of the variables ($P \geq 0.20$). Comparison (separately) of window-base and centrally-acquired floor depositing dust samples adjacent versus opposite the walkway also demonstrated no difference for any dust deposition parameters ($P \geq 0.20$).

No evidence suggesting a relationship between the elevated walkway soil lead levels noted earlier (Chapter Six) and associated enhanced leadfall/dust lead concentration in walkway-adjacent rooms is apparent. Such a situation may arise as a result of relatively equal room penetration of contaminant dust or may be due to effective mixing of suspended material due to within-home air currents. Alternatively, the walkway side of the house may provide some degree of protection from air movements, thus limiting dust entry to rooms via windows.

Given that dust deposition is substantially lower during closed-house conditions (Table 8.4), it seems appropriate to investigate any adjacent versus opposite walkway differences under both house conditions. Further, this is important since, as has been referred to above, some of the more subtle effects of differential between- and within-room dust deposition patterns may be detected by this collection system under either closed-house (Section 8.5.6) or open-house (Section 8.5.9) conditions.

For main rooms separated into either house open or closed categories, no difference between walkway-adjacent and opposite rooms for all dust deposition parameters for both house categories is displayed ($P \geq 0.10$). Given that such a conclusion also derives from the static dust work (Chapter Seven), it seems likely that dust contamination of rooms such as walkway-adjacent may in fact not be intimately influenced by the proximate soil environment. Essentially, such soil contaminant material could remain moderately stable, the predominant stream of contaminant material entering houses, deriving more from general district air-suspended rather than locally re-entrained contaminant material.

These data suggest no enhanced deposition in rooms adjacent to walkways, although given inter-room differences and window gradient effects discussed above, it would seem an appropriate avenue to pursue with further research.

8.5.11 DEPOSITING DUST CHARACTERISTICS - AWAY VERSUS FACING

As for soils and also static dust previously, an investigation relating to possible contamination differences between facing-smelter aspects of houses versus the facing-away counterpart was conducted.

8.5.11.1 GENERAL INVESTIGATION

Summary, all-collection round, (main room) depositing-dust data for smelter-facing versus facing-away categories is not particularly insightful. No real facing/away difference in mean dustfall value is noted ($P > 0.25$), albeit a numerically larger mean value for dustfall exists for facing-smelter room locations. Mean dust lead loading is numerically larger in facing-smelter rooms, albeit such difference being marginal ($P < 0.175$). For dust lead concentration, however, the mean facing-room value is comparatively greater than its counterpart ($P < 0.001$).

Such a result fits comfortably with previously-elicited similar observations, in an analogous context, for soil lead (and zinc) concentration ($P \leq 0.002$) as also for static dust lead concentration and loading ($P \leq 0.001$).

Given such depositing dust differences, and notably in this case, this is one of the few occasions where difference has been detected in dust lead concentration between two test categories, it would seem appropriate to conduct such a comparison separately under house closed and (combined) open conditions. Summary data for these categories appears in, respectively, Tables 8.25 and 8.26 below.

Table 8.25 House aspect depositing dust comparison - house closed.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED				CUP-COLLECTED	
HOUSE ASPECT	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Facing Smelter	**49**	2,000	6,355	13	**51**	15
Facing Away	**54**	2,373	4,727	11	**54**	12

For closed houses, minimal difference is apparent between facing versus facing-away mean dustfall or leadfall values ($P \leq 0.20$), dust lead concentration however demonstrating a noticeably greater mean value for facing smelter main rooms ($P < 0.04$).

Table 8.26 House aspect depositing dust comparison - house open.

	GROUPED MEAN DATA					
	ALFOIL TRAY-COLLECTED			CUP-COLLECTED		
HOUSE ASPECT	n	DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	DUST LEAD CONC. ($\mu g.g^{-1}$)	LEADFALL ($\mu g.m^{-2}.d^{-1}$)	n	LEADFALL ($\mu g.m^{-2}.d^{-1}$)
Facing Smelter	**78**	19,272	8,596	166	**36**	143
Facing Away	**88**	21,292	5,944	127	**38**	123

For such open-house conditions, mean leadfall as collected by both techniques appears larger for facing smelter aspects of houses, this being particularly so for alfoil-collected leadfall ($P \leq 0.04$) as also for dust lead concentration ($P \leq 0.001$).[23]

This investigation thus strongly links soil lead data with the internal static dust and dustfall data in the context of the general source from which the contamination of both these environmental compartments derives. The implication here is that the source of the contamination appears to be a pool of lead-contaminated dust affecting homes as it travels from the general direction of the smelter. Further, those aspects of houses facing away from the smelter are less significantly affected, the facing aspects of houses presumably providing a sheltering effect from "waves" of such contamination for the facing-away or "lee" aspect. This conclusion is reinforced somewhat by the noted slightly greater dustfall levels detected in facing-away aspects of houses (under both house closed and open conditions).

Such results contrast the conclusions of earlier work conducted in Port Pirie (Body, 1986f), where it was reported that a long-standing pool of lead-contaminated dust (deriving from former smelter discharge practices in the earlier part of this century, [Body, 1986i]), is randomly distributed throughout the city, ceaselessly moving at the caprice of aeolian activity. Body's conclusion would imply that this current work's observation of the directionality of current contamination movements would not likely occur. If Body's conclusion was, in fact, the case, the (earlier-discharged) contaminated material would have been sufficiently well mixed and disseminated around the township by winds of varying direction over the many years for it not to differentially and more significantly affect facing rooms than facing-away rooms as is currently observed.

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[23] Cup-collection leadfall results for all collection rounds and house-closed rounds align with those for the alfoil-collected parameter for all facing versus facing-away alternatives, with no difference being detected for such comparison ($P \geq 0.30$). Difference however may exist between the two techniques, eg under open-house conditions where alfoil-collected leadfall demonstrates house aspect difference ($P \leq 0.04$), cup-collected data reflects no such difference ($P \geq 0.30$). This technique difference could be related to a smaller sample size with the cup technique, or alternatively, such data could simply reflect the lack of sensitivity of the cup-collecting method.

8.5.11.2 DISTRICT DIFFERENCES FACING VERSUS FACING-AWAY

Comparing Port Pirie West and Solomontown data sets (separately) for facing and facing-away room categories elicited results consistent with those for the total data set above.

a) Port Pirie West

For Port Pirie West for all collection rounds, substantial difference was detected only for dust lead concentration, facing rooms displaying the greater mean level ($P \leq 0.001$).

Despite a moderately larger dustfall mean level for facing-away rooms ($P < 0.10$), mean leadfall is only very slightly greater for facing rooms. Most of these Pirie West houses are distributed to the west of the township and smelter, some of them having their facing-away aspect adjacent to undeveloped land that remains open, flat and generally devoid of vegetation. Typically, this land has been used for the disposal of smelter slag materials, and/or that has been reclaimed from low-lying salt flats by the use of smelter-derived (typically slag) material. Despite this potential source of contaminant dust, and again, slightly greater dustfall levels, these facing-away aspects of the home demonstrate lower levels of leadfall and dust lead concentration than those rooms facing the smelter. This may also reflect the relative stability of smelter slag material.

Further comparison under (separately) house closed and open categories reveals greater dustfall for facing-away main rooms under both house closed and open conditions ($P \leq 0.02$). Despite greater dust lead concentration mean values for facing aspect rooms for both house status categories however ($P \leq 0.05$), only minor difference in leadfall is detected between facing and facing-away rooms for either house status category ($P \leq 0.25$).

b) Solomontown

Solomontown data for all collection rounds reveal no difference between facing and facing-away rooms for dustfall or leadfall ($P \geq 0.15$), mean data for both of which are larger for the facing-smelter room aspect. Dust lead concentration data are not quite so clear cut, however, the mean facing-smelter value being marginally larger than its facing-away analogue ($P < 0.06$).

Separate comparison for house closed and open formats demonstrates no house facing/away aspect dustfall difference for either house status category ($P \geq 0.40$), dust lead concentration being substantially greater for facing rooms only under open-house conditions ($P < 0.015$). This is reflected by Solomontown facing room alfoil-collected leadfall under open-house conditions, the mean value for which is notably larger ($P < 0.015$). No real leadfall facing/away difference is apparent under house closed conditions ($P < 0.75$).

Despite a modest sample size for Solomontown, the two districts demonstrate consistency with respect to this facing/facing-away observation. Clearly further investigation would be appropriate.

8.5.11.3 SUMMARY COMMENTS

Greater dustfall in facing-away house aspects of Port Pirie West houses ($P \leq 0.02$) suggests common wind direction and an area of the town (undeveloped open soil) potentially likely to provide dust for wind entrainment, such dust being less contaminated (or less mobile) than dust deriving from proximate to the smelter.

This is opposite to the situation in Solomontown houses where dustfall is larger (albeit only marginally) in facing smelter rooms as is leadfall, again possibly reflecting prevailing winds. It also suggests that less dust is available to the facing-away house aspects, due to residential development to the South and South West of Solomontown limiting the generation of dust.

In Solomontown houses leadfall is only moderately larger in facing smelter rooms, Pirie West homes demonstrating slight difference, in contrast, however, to lower dustfall in these (facing) rooms. Thus dust being collected by dustfall collectors in this study appears to derive from more than one source, that affecting facing smelter rooms more significantly than facing-away rooms being more substantially contaminated.

By inference thus, occupation of rooms facing the smelter would imply a greater (potential) exposure to a component of (lead-bearing) dust deriving from the direction of the smelter. This is the case for both districts. Given that this is also the case for soil and static dust contaminant patterns, such observations suggest that the source of the contaminants derives from the general vicinity of the smelter works, and that home contamination would appear to be a current phenomenon.

As referred to previously (Section 7.9), front/rear main room static dust differences seemed to be representative more of facing versus facing-away differences than simply room differences. Here, front versus rear main room differences in dust deposition (Section 8.5.6.2) reveal greater mean values for all deposition parameters apart from dust lead concentration for combined rounds, house closed, and also (combined) house open.

Such data do not contrast the results of the facing versus facing-away investigations above. Here, as was the case for many of the comparisons made between various categories in this dustfall chapter, dust lead concentration rarely appeared to demonstrate significant difference. That the various facing versus facing-away category comparisons so often demonstrated to the contrary confirms that a principal factor in contaminant deposition is the spatial relationship between collection location and smelter. Essentially, front to rear main-room differences rather than being a surrogate for facing versus facing-away from the smelter differences, may more accurately be described as a composite of at least two depositional influences. These are:

a) deposition <u>rate</u> differences resulting from potential buffering of rear main rooms by front main rooms and hallways, and

b) dust lead <u>concentration</u> differences arising as a result of the relative alignment of the sampling location with the general direction of the smelter.

The relative veracity of this latter depositional influence alternative appears somewhat embellished by comparison of depositional data for front main rooms in either facing versus facing-away mode. No dustfall differences between the two alternatives exist ($P \geq 0.90$), yet substantial difference in dust lead concentration does exist, with facing smelter front main rooms exhibiting a greater mean value ($P \leq 0.001$), no real difference in leadfall being apparent ($P \geq 0.15$). Analogously, combined halls and kitchens data reveal greater facing room dust lead concentration ($P \leq 0.02$), but no differences between any other dustfall parameter ($P \geq 0.30$), although sample size is not particularly large.

That these two presumed influences on room contamination (rate and concentration) could not necessarily be so clearly defined from the static dust suite of data reflects that such data illuminate contamination that arises as a composite of various sources (existing dust plus newly-entering) plus the mixing of existing dust during prior house occupancy. This implies that as an indicator of the potential for a child to acquire contaminant dust from (say) a room floor, the static dust/lead loading level would appear satisfactory, but, as an indicator of the mechanism of room contamination, depositional dust collection eliciting particularly leadfall and dust lead concentration data would appear more appropriate.

8.6 POST-CLEANUP DUST DEPOSITION

8.6.1 INITIAL DATA INTERPRETATION

From a substantial quantity of work conducted in Port Pirie a decade ago, Body (1986f) concluded that it was essential that the assessment of the rate of re-contamination (by leaded dust) of houses be conducted in order to evaluate the possible re-introduction of lead sources into the child's environment. As was referred to in the introductory chapter, a component of the total research conducted here was directed towards an investigation of dust lead deposition in four of the test houses subsequent to a very thorough cleanup of each house. This cleanup was intended to simulate current SAHC house intervention/decontamination strategies.

Subsequent to test house cleanup, another depositing-dust collection round was conducted under closed-house conditions (Round Eight), incorporating 64 aluminium alfoil collectors located at floor level. Dust deposition sampling was conducted in each of these four test houses in closed-house format, two houses in each district. Four remaining test houses which were not cleaned were also sampled from (again two from each district), and used as control houses.[24]

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[24] Set-up and collection of dust collectors was conducted under extremely clean conditions, with the operator wearing clean clothes and having clean hands. At all times, shoes were removed, and minimal contact made with floor surfaces. Each tested house was entered only once, and all sampling completed with minimal disturbance before exiting in as short a time as possible. All doors and windows were kept closed after clean-up and prior to experiment set-up. During dustfall-collection phase, external doors and windows were kept closed, although internal doors were left in the same position as was the case for all previous dustfall-collection rounds, ie open. This was also the case during collector retrieval.

Summary depositing-dust data deriving from these eight houses in closed-house format prior to cleanup (as presented in Table 8.4 above), was compared to analogous data deriving from these houses post clean-up, a summary of which appears in table 8.27 below.

Table 8.27 Pre- and post-cleanup summary depositing-dust data.

		GROUPED MEAN DATA		
DUST COLLECTION ROUND	**n**	**DUSTFALL** ($\mu g.m^{-2}.d^{-1}$)	**DUST LEAD CONC.** ($\mu g.g^{-1}$)	**LEADFALL** ($\mu g.m^{-2}.d^{-1}$)
BEFORE CLEAN-UP				
One	**45**	3,527	4,761	17
Two	**42**	2,765	3,178	9
Five	**46**	1,754	10,799	19
Combined		2,565	5,562	14
AFTER CLEAN-UP				
Eight				
- All houses	**64**	4,731	4,864	23
- Cleaned houses only	**42**	4,707	4,998	24
- Control houses	**22**	4,779	4,626	22

Clearly, neither dustfall nor leadfall are any lower during the post-cleanup assessment period than for any of Rounds One, Two or Five. Comparing this latest dust deposition period-derived data for cleaned houses with combined Rounds One, Two and Five displays a larger mean value for dustfall and leadfall for the post-cleanup round ($P \leq 0.01$). The post-cleanup collection period was conducted between December 21, 1995 and February 2, 1996, ie over the summer period, which corresponds to previous collection Rounds One and Two. Comparing the data corresponding to Round One with the post-cleanup data for the four cleaned houses reveals lower mean values for dustfall and leadfall for Round One ($P \leq 0.02$). Round Two data reflect smaller mean values for all dust deposition parameters however ($P \leq 0.01$). Comparison with Round Five (winter collection) dust deposition characteristics reveal greater dustfall in cleaned houses for the post-cleanup round ($P < 0.01$), no real difference for leadfall ($P > 0.20$), but a substantially lower dust lead concentration than for Round Five ($P < 0.01$). Acknowledging modest sample size, comparison of data derived contemporaneously from those cleaned houses and the non-cleaned control houses reveals no apparent difference for all dust deposition parameters ($P \geq 0.60$). Such a house cleanup appears to contribute minimally to a reduction in house re-contamination rates. By comparison, if

the difference in static dust levels detected by vacuum pump technique at floor locations adjacent to dustfall collectors (Section 7.20) between the commencement and completion of this 47 day (post clean-up) depositing-dust collection round are enumerated as a rate of increase per day (ie expressed as a function of exposure time), they appear as below (Table 8.28).

Table 8.28 Post-cleanup static dust collection data summary.

GROUPED MEAN DATA		
DUSTFALL ($\mu g.m^{-2}.d^{-1}$)	**DUST LEAD CONC.** ($\mu g.g^{-1}$)	**LEADFALL** ($\mu g.m^{-2}.d^{-1}$)
5,744	5,131	30

The similarity between these analogues is notable, although the dustfall/leadfall rate suggested by the vacuum pump/cassette technique is slightly greater. This may, however, be attributable to the fact that the carpet at time of commencement of the 47 day test period was not completely clean, in effect, residue dust remained subsequent to the house cleanup procedure. Some of this dust was harvested with the first vacuum pump sampling, and some obviously remained to be sampled at the end of the 47 day test period together with the newly-deposited dust. Given that these dustfall/leadfall values as derived by either technique are really quite similar, it would seem not unreasonable to claim that the depositing dust-collection technique utilised throughout this work (alfoil tray) would provide a good estimate of the quantity of dust and lead actually depositing upon (floor) surface areas. Additionally, the vacuum pump/cassette sampling system is an effective means by which the quantity of dust depositing on a certain surface over a certain period of time can be estimated, such an estimate constituting a reasonable surrogate for depositing dust rates of deposition. From the above comparisons, subsequent to a thorough internal house cleanup procedure simulating that which was earlier-conducted in houses in Port Pirie as part of the SAHC lead-intervention program, there exists no evidence to suggest any likelihood of a consequent reduction in subsequent dust/lead deposition rates. Essentially, that dust being collected in aluminium dustfall collectors as part of the dust deposition study, would not appear to be the result of re-entrained existing (within home) dust, but more, a reflection of novel dust penetrating these test houses, sedimenting, and being entrapped by the dust collectors. It is inferred here that that is also the case for settling/settled dust on room surfaces such as floors.

Certainly caution should be applied when interpreting such short term (two to three month duration) sampling procedures in order to reflect normal house dust acquisition, since re-contamination subsequent to decontamination over such a short time may not accurately reflect long-term normal house lead-bearing dust accumulation. Given that dust deposition appeared not to be larger for any of the three (pre-cleaning) collection periods, however, it would seem reasonable to suggest that the dust being collected then was

newly-entering (the house), and that seasonal variation influenced this little, if any. Once again, longer term longitudinal studies in a variety of locations would be helpful.

8.6.2 FURTHER POST-CLEANUP DATA INTERPRETATION

Since the data pool for the post-cleanup dust deposition study was not particularly large, only a few category comparisons will be made here. As was the case for the larger dust deposition study immediately above, no apparent district difference appears to exist for any of the dust parameters for either the cleaned or the control house group ($P \geq 0.35$). Although hallways revealed numerically larger mean values for dustfall and leadfall than all other rooms, no between-room depositing-dust parameter differences were registered in either cleaned or control house category ($P \geq 0.10$). Similarly, whilst window-base floor collection points displayed numerically larger mean dustfall and leadfall values than centrally-located analogues in cleaned as for control houses, matched pair comparison demonstrated no difference ($P \geq 0.25$). This was also the case for the combined cleaned/not cleaned data suite. For the full data suite deriving from this last sampling period, facing-smelter located room-sampling points indicated slightly less dustfall, but substantially greater leadfall and dust lead concentration than facing-away analogues ($P \leq 0.02$). Such summary data for the post-cleanup period are indicated below (Table 8.29).

Table 8.29 Post-cleanup facing-smelter versus facing-away summary data comparison.

		GROUPED MEAN DATA		
HOUSE ASPECT	**n**	**DUSTFALL** ($\mu g.m^{-2}.d^{-1}$)	**DUST LEAD CONC.** ($\mu g.g^{-1}$)	**LEADFALL** ($\mu g.m^{-2}.d^{-1}$)
Facing smelter	**32**	4,662	5,820	27
Facing away	**32**	4,802	4,066	20

These results are consistent with those from the main dust deposition data suite above (Table 8.25).

Despite modest sample size, facing smelter versus away comparison was made (separately) for both cleaned houses and control houses. Whilst mean values for all dust deposition parameters for the facing smelter category were numerically larger for both cleaned and control houses, no facing/away dustfall difference was apparent for either house group ($P \geq 0.50$). For leadfall and dust lead concentration, both house categories demonstrated modestly larger mean values for facing-smelter house aspects ($P \leq 0.10$). Given that the full (post-cleanup) data suite revealed more pronounced facing/away difference for leadfall and dust lead concentration, it could be concluded that the smaller sample size for cleaned/control house categories contributed to the less pronounced house-aspect difference in these houses.

8.7 DUSTFALL COLLECTION IN EXTERNAL STRUCTURES

As a part of the dustfall collection component of this thesis work, some external dustfall collection was made, typically within available outdoor structures such as outdoor bathrooms, laundries, sheds and play-houses throughout the duration of the dust-collection study. These very basic structures tended to be built with an external wall (typically constructed of galvanised iron) mounted upon a wooden frame. No internal walls were present, windows typically being of a louvre construction, and doors ill-fitting or absent completely. For this collection work, both alfoil and cup collectors were used, typically located at ground level. This was only a very limited study, with insufficient collectors used to provide any more than an indication of external dustfall and leadfall levels. In some locations, concerns were raised regarding the ability to stably locate an alfoil tray and still be able to recover it after the exposure period, and also that the external surface of the tray might have become dusty as well, thus confounding the gravimetric determination of collected dust within the confines of the collector. Hence it was more convenient to use cup collectors to estimate simply leadfall values, there being only a small number of alfoil collectors that survived either weather extreme or vandal attack.[25] Combined collector data for both districts for grouped collection rounds are summarised below (Table 8.30).

Table 8.30 External/internal depositing dust comparison.

	GROUPED MEAN DATA				
		DUST DEPOSITION COLLECTION (ALL COLLECTORS)			
	WITHIN EXTERNAL STRUCTURES	WITHIN THE HOUSE			
		ALL COLLECTION ROUNDS COMBINED	HOUSE CLOSED	WINDOWS OPEN ONLY	DOORS AND WINDOWS OPEN
Dustfall ($\mu g.m^{-2}.d^{-1}$)	16,341	9,323	2,565	16,324	23,985
Leadfall ($\mu g.m^{-2}.d^{-1}$)	294	59	16	140	134
Dust lead conc. ($\mu g.g^{-1}$)	17,991	6,350	5,562	8,560	5,589

[25] The use of clean wooden bases wrapped in clean paper to which the alfoil collector was taped with masking tape would address collector stability problems, no anti-vandalism solutions being proffered.

For this work, no dustfall/leadfall readings were acquired from sites located directly outdoors. It can only be presumed that dustfall/leadfall levels in the open would be greater than those registered here. Earlier work elsewhere has suggested a five times greater dustfall rate externally compared to indoors (Whitby et al., 1957). This would seem to be in accord with limited work conducted in Port Pirie by Sweetapple, van Alphen (unpubl.), and work conducted elsewhere.[26] As a general comment, such external structures as have been tested here appear to provide less protection from the contaminant dust stream than main dwelling structures, particularly if windows and doors remain closed. This would be of significance in cases of (say) external play rooms or cubby houses for children, and also external bathrooms and laundries. Clearly, however, the main house structure does provide some resistance to penetration of contaminant dust, the substantially lower dust lead concentration observed within the house implying that a certain component of the (external) leaded dust population fails to enter the house as easily as outdoor structures. The determination of home modifications to further limit the penetration of contaminant dust into the house would certainly appear warranted as part of a future research program.



Plate Six Depositing dust sampling system.

[26] Lead present as a suspension in air consists of predominantly sub-micron particles, the deposition velocity of which will be determined by Brownian and eddy diffusion, with a greater deposition velocity being expected in the more turbulent outdoor atmosphere (Hamilton and Revitt, 1987). Relevant data are presented in Chapter three.

CHAPTER NINE LEAD-IN-AIR STUDY

This most excellent canopy, the air, look you, this brave o'erhanging firmament, this majestic roof fretted with golden fire, why, it appears no other thing to me than a foul and pestilent congregation of vapours.

William Shakespeare

[London's] Inhabitants breathe nothing but an impure and thick mist, accompanied with a fuliginous and filthy vapour, . . . corrupting the Lungs and disordering the entire habit of their bodies; . . . .

John Evelyn, 1661

9.1 INTRODUCTION

Monitoring of airborne lead contamination as arising from a point source and assessed proximally, may be helpful to industry and environmental watchdogs. Subsequent legislative control resulting in a reduction or removal of the lead from the emitting source(s) may not, however, necessarily reduce home contamination problems significantly, since (presumed) secondary contaminating streams such as contaminated soil or dust may elude pre-existing monitoring networks which were principally employed to measure stack discharge (EPA NSW, 1993).

Current air quality standards for particulate matter are based upon the assessment of outdoor, ambient atmosphere, total suspended particulate (TSP) matter utilising high-volume air samplers (EPA USA, 1986). Lepow et al. (1975) and Yocom (1982), however, indicate that outdoor air quality standards based upon TSP concentration are not meaningful as a means of assessing indoor air lead levels as well as total personal exposure to particulate matter, hence there exists a need to assess indoor air. Laxen et al. (1985) contend that urban air lead levels to which a resident population is exposed have been overestimated in the past, largely due to inappropriate siting of (outdoor) samplers. The role of such samplers is best contextualised as being part of an emission control strategy.[1] In Port Pirie, there exist a number of locations where ambient (outdoor) air lead levels are currently assessed regularly by way of high-volume samplers

[1] It would be expected that differences in air movements (intensity and direction) between the location site of a high-volume sampler and a room in a home would clearly infer different air contaminant levels for the different locations. District mean lead-in-air levels as detected by high-volume samplers cannot be assumed to be representative of indoor air levels (Dowd, 1984), and hence personal exposure (Tosteson et al., 1981 & 1982; Yocom, 1982; Spengler and Soczek, 1984; Spengler et al., 1985). Cohen and Cohen (1980) as Brunekreef (1984) report that larger-sized lead-containing particles may be co-sampled by high-volume samplers, these particles penetrating less well into houses than smaller-sized particles (Iwatsuki et al., 1984). Rowe et al. (1984) concur, suggesting a smaller (albeit slightly) mean particle size sampled indoors than outdoors.

operated according to a recommended (EPA USA) protocol. Such samplers are typically located at sites which are mainly non-residential and centrally-located. [2]

A number of studies has demonstrated that different pollutant levels may exist indoors and outdoors (Anon., 1979), several attempting to relate blood lead levels to outdoor air concentrations (eg Chamberlain, 1983; RCEP, 1983). Such external air lead measurements, however, may have little relevance to levels within a building/house (Davies, 1987) given that the outdoor sampler-indicated contaminant levels may have been derived at some considerable distance from homes. Further, dust deposition within houses may result from the slowing of air movements within the structure, the predominant pathway for child lead intoxication being by way of hand mouth contact with deposited materials and not inhaled suspended material.

Personal exposure to (internal) air lead will vary between individual children, particularly dependent upon the time spent indoors (Mulik and Lewis, 1988). Given that children spend a substantial component of their time indoors (Chapin, 1974; Dockery and Spengler, 1981a), ambient (external) air data does not accurately reflect a home-dwelling child's actual exposure to airborne lead (Binder et al., 1976; Alzona et al., 1979; Moschandreas et al., 1979; EPA USA, 1986). In fact, personal exposure demonstrates much better correlation with indoor rather than outdoor air lead levels (Snee, 1982b; Spengler and Soczek, 1984; Lioy, 1991). Wallace (1996) reports poor correlation between personal exposure and outdoor air levels.[3]

9.2 DUST ENTRY INTO BUILDINGS

Most buildings have gaps in their structures such as around doors and windows, roof to wall and ceiling borders, masonry and wall material cracks etc (pers. obs.). Assuming natural ventilation of such buildings, there exists some degree of air exchange internally and externally, cracks, gaps, closed and (particularly) open windows/doors contributing substantially to contaminant entry (Sterling and Kobayashi, 1977; Cohen and Cohen, 1980; Crockford, 1992). Certainly dwellings in the two selected districts of Port Pirie, given their frequent basic/cheap nature, would appear to offer little resistance to penetration of air and suspended material. Such observations have also been made with respect to similarly-constructed houses elsewhere (Yocom et al., 1971; Butler et al., 1975), and even for closed "tight" houses (Butler and MacMurdo, 1974). Sterling and Kobayashi (1977) observe that house structures provide relatively little protection from invading contamination.

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[2] EPA USA propose a three monthly mean of $< 1.5 \mu g.m^{-3}$ lead in air, with sampling occurring for at least 24 hours continually, once every six days (Fergusson, 1986).

[3] Tosteson et al. (1982a & b) conducting work in occupied houses elsewhere, compared personal air monitoring (exposure) devices with fixed devices for lead in air, and revealed (respectively) levels of $0.16 \mu g.m^{-3}$ and 0.092 - $0.12 \mu g.m^{-3}$, inferring that using fixed monitors (either indoors or outdoors) as indicators of personal exposure was not necessarily the most appropriate technique. They did, however, demonstrate better correlation between personal and indoor lead than personal and outdoor lead levels. In this study, the assessment of indoor and outdoor lead-in-air concentration levels is intended more to define the effectiveness of exclusion from homes of lead-bearing dust by the domicile structure. Andersen (1972) notes that given that most people spend more time in the house than elsewhere, any attempt to reduce the penetration of contaminants into the house may be more effective a health risk-reducing strategy in long-term discharge locations than technical pollution discharge control at source.

From a comparison of dust counts and particle size ranges for indoor versus outdoor collected dust in residences, Jacobs et al. (1962) reported no difference between the two populations for either category. Schaefer et al. reported a very good correlation between indoor and outdoor dustfall rates for urban homes, the indoor rate being "dramatically higher when windows were kept open" (1971; p. 174). Butler and MacMurdo (1974) have noted a fairly rapid equilibration of indoor air with outdoor air, Harrison (1979) concurring. Alzona et al. (1979) claim that this is complete within several hours in reasonably well-sealed rooms, and more promptly in less well-sealed rooms.

Aerosols that do enter buildings assisted by external air movements will tend to settle indoors (Crockford, 1992) due to the absence of or limitation to air movements within the structure (Sinclair et al., 1985), and subsequently, a certain amount of mechanical energy will be required to re-entrain them (Crockford, 1992).

9.3 INDOOR/OUTDOOR AIR LEAD CONCENTRATION RATIOS

Virtually all of the results of other workers presented here apply to occupied buildings, there being nothing apparent from the literature for analogous investigation into vacant structures. Early work by Shephard et al. (1960) indicated that indoor air levels of suspended particulates in a smoky (industrial) area displayed a reasonable correspondence with outdoor values when windows were kept open, levels being approximately even. When the windows were closed, however, the indoor concentration values dropped to around half the outdoor values, particularly at night. Freed et al. (1983) claim, for particles $\leq 1.0\mu$ in size, a level of 40% for penetration under closed window regime and 100% for open window regime. Other workers indicate an internal level of around 80% of that of external total suspended particulates in occupied dwellings (Biersteker et al., 1965; Sterling and Kobayashi, 1977).

Alzona et al. (1979) report a static ratio of approximately 0.3:1 for (respectively) indoor to outdoor airborne particulate matter, with equilibration occurring within several hours. Halpern (1978), however, notes significant variability between sites depending upon whether sites were air-conditioned or not, and also associated with location of dwelling and construction style, Yocom et al. (1971) finding to the contrary. Alzona et al.'s (1979) "rule of thumb" ratio of 0.3:1, as also noted by Diemel et al. (1981), assumes that re-suspension of material that enters the room other than by airborne means is excluded, typically the case for vacant houses. Diemel et al. (1981) infer a more substantial value for resuspension of pre-existing deposited dust for occupied houses, reporting an input/output ratio (I/O R) approaching 0.6:1.

The few studies conducted in the United Kingdom prior to the work of (D. J. A.) Davies et al. (1987b, e & 1990) have revealed mean I/O Rs for lead in air of the order of 0.6 - 0.8:1 (Andersen, 1972; Chamberlain et al., 1978; DOE, 1978; [B. E.] Davies, 1987), (D. J. A.) Davies' group (1987b & 1990) proffering a figure of around 0.61:1. These figures appear to be roughly in accord with the data summarised by Yocom (1982) and also Hamilton and Revitt (1986), although these last-mentioned authors reveal a greater range figure of around 0.4 - 0.8:1. Whilst some European/US studies reflect lower I/O Rs during winter when windows are more commonly kept closed tightly (EPA USA, 1986), this may not be the case in Port Pirie, given the

climate, the condition of many of the windows generally and the observed practice of keeping windows open almost throughout the year.[4]

Overall, mean ratios lie in the range of 0.6 - 0.8 for non-air conditioned houses (Sterling and Kobayashi, 1977; Yocom, 1982; Davies, 1990), and 0.3 - 0.5 for air-conditioned houses (Tosteson et al., 1982a & b; Yocom, 1982). Halpern (1978) reports significant difference between these two alternatives, but not between I/O Rs corresponding to commercial and residential buildings. Alzona et al. (1979) observe that the contribution to suspended particulate material arising from the resuspension of already-deposited surface dust is not an important factor in suspended particulate levels in an unoccupied room, Moschandreas et al. (1979) and Diemel et al. (1981) concurring. These authors cite substantially reduced resuspension in closed undisturbed structures due to decreased air movement and turbulence (as Sinclair et al., 1985; Roed and Cannell, 1987), although they note that mean outdoor levels are very much a determinant of mean indoor levels.

Increased indoor activity, however, has been reported to contribute to an enhanced resuspension effect (Moschandreas et al., 1979; Hamilton and Revitt, 1987) thus increasing the magnitude of the I/O ratio. Diemel et al. (1981) note a substantially more important role than does Alzona et al. (1979) for such activity-induced resuspension compared to simple dust penetration alone. Moderate consistency of mean I/O Rs would suggest that in the absence of major indoor sources of lead and with minimal resuspension of pre-existing indoor-deposited dust lead, outdoor air sampled proximate to the dwelling would be a reasonable predictor of indoor air lead concentration.

The moderately wide spread in these recorded values may be explained by a number of variables such as changeable atmospheric conditions which would appear to have a large impact upon the outdoor level (Alzona et al., 1979), but a much smaller effect (negligible according to Hamilton and Revitt [1987]) upon indoor air lead levels. This effect purportedly varies according to house design and construction (Hamilton and Revitt, 1987), eg wind-induced wall vibration inducing air-stirring currents indoors (Alzona et al., 1979).[5] Reflecting potential ratio variability is the substantial temporal variability of external air-lead levels as noted by Butler and McMurdo (1974), Halpern (1978) and Moschandreas et al. (1979).

Davies' group reports that the external air lead level demonstrates a substantial influence on indoor levels, with the physical structure of the house (walls, doors, windows) preventing some of the suspended particulate matter penetrating into the indoor air (1987a, b & c). They do note, however, that internal air lead concentration closely follows external levels (Davies et al., 1987b), the latter explaining (r^2 =) 72% of the variability in internal levels (1987c). Roberts et al. (1974) note that outdoor air lead concentration accounts for 85% of the variance of indoor air lead concentration.

A summary of various workers' results for mean lead in air concentration (µg lead/cubic metre air - $\mu g.m^{-3}$) are indicated in Table 9.1.

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[4] From the literature, it would appear that seasonal variations in ambient air lead levels occur. Most frequently, winter air lead levels reach higher mean maxima than corresponding summer maxima (Faoro and McMullen, 1977; Duggan, 1981; Rabinowitz and Needleman, 1982; Yocom, 1982; Strehlow and Barltrop, 1987; Kim and Fergusson, 1994). This appears to be attributable to a more stable atmosphere elicited by a higher (winter) incidence of inversion conditions (Strehlow and Barltrop, 1987). Yocom (1982) reconciles higher I/O Rs during summer and lower I/O Rs during winter with less influence of outdoor levels on indoor levels in winter when the house is more likely to be closed.

[5] Most of these data derive from a colder climate than is experienced in Port Pirie, and it might be expected that sealing of dwellings would thus be less effective locally. Certainly in some of the test houses, wall movement and (closed window) curtain displacement during strong wind activity was easily detectable.

Table 9.1 Indoor-Outdoor Relationships for Lead in Air.

	GROUPED MEAN DATA			
	AIR LEAD CONCN. ($\mu g.m^{-3}$)		INDOORS/OUTDOORS RATIO	SOURCE *
LOCATION	INDOORS	OUTDOORS		
Birmingham	0.87	0.89	0.98	1
Toronto	0.92	4.87	0.20	2
New York	0.21	0.27	0.82	3
" "	0.38	0.60	0.63	"
" "	0.63	0.65	0.81	"
" "	0.36	0.42	0.87	"
Pittsburg	-	-	0.42	4
New York	0.09	0.13	0.66	5
Pittsburg	-	-	0.53	6 & 8
Arnhem (Netherlands)	0.26	0.41	0.62	7
Wales (various)	-	-	0.66	9
Birmingham	0.44	0.63	0.70	10
Birmingham	0.26	0.43	0.61	11
Bradford and Manchester	0.33	0.39	0.86	12

* 1. Butler and MacMurdo (1974) 2. Roberts et al. (1974b) 3. Halpern (1978) 4. Alzona et al. (1979) 5. Moschandreas et al. (1979) 6. Cohen and Cohen (1980) 7. Diemel et al. (1981) 8. Yocom (1982) 9. Elwood 1983 (cited in Hamilton and Revitt [1987]) 10. Davies and Watt (1986) 11. Davies et al. (1987b & 1990) 12. Al-Radday et al. (1993a.)

Source: Adapted from Hamilton and Revitt, 1987, p. 99; additional data from Davies (1987), Davies et al. (1987b and 1990), Murphy and Yocom (1986), and cited authors.

Davies' group also detected that concurrent sampling for different houses elicited very similar results, despite the fact that the houses were quite widely scattered (1987a, b & c). Explaining this, they claim that the same air mass was influencing the differently-located houses, and that meteorological factors (rainfall, wind speed and direction) constituted potential influence upon the detected air lead concentration (Davies et al., 1987c). These authors also noted the lack of any literature-detected data describing the proportion of house (static) dust lead which derives from air lead deposition.[6] Such activity will be investigated below.

[6] Hamilton and Revitt (1987) note a virtual absence in the literature of indoor : outdoor ratios for deposited dust/lead, suggesting as a reason, that dust lead levels show substantial spatial variation. This is not entirely correct. Between-home dust concentrations and loadings may certainly vary substantially spatially, but given that no studies are evidenced demonstrating multiple house (static) dust sampling ($n \approx 60$ per house or more) utilising appropriate sampling equipment such as here, within-home (static) dust variability could not really have been commented upon.

Indoor air pollutant concentration levels may lag behind those of external air during peaks and troughs in pollutant levels in external air (Davies, 1988), although, such adjustment occurs fairly rapidly. Sampling for periods of days rather than hours will reduce such an effect to insignificance (Davies et al., 1987c).

9.4 RESEARCH OBJECTIVES

Acknowledging the penetration of air-suspended dust into residences, the focus of this work is directed toward establishing an index of the relative penetration of suspended contaminants into (closed) vacant houses, and its potential spatial variability. Differences in indoor/outdoor air lead concentration ratios between vacant and occupied houses will also be investigated in order to determine the influence of human activity here.[7] Indoor/outdoor sampling is also conducted in some of the test house suite subsequent to a thorough cleaning, in order to assess whether the presence of existing settled contaminated dust in rooms is influential upon indoor/outdoor suspended contaminant ratios. This will further suggest the relative contamination significance of novel versus re-entrained dust within rooms of homes as detected by its collection on surfaces in these rooms (Chapter Eight). Such sampling should also, given contemporaneous dustfall collection, elicit an (internal) lead-in-air and associated leadfall relationship.

Closed, vacant houses were selected for the majority of the work conducted here, in that, under such conditions, it is assumed that such a situation would represent the minimum (external) dust penetrability of homes. Normal occupancy demonstrates, typically, (some) open windows and doors for at least part of each day (pers. obs.). It was planned that this preliminary pilot study would subsequently be expanded upon, and constitute part of a larger on-going SAHC-conducted air-sampling strategy for both occupied and vacant houses in Port Pirie. Such work appears necessary, given the absence of any indoor air lead concentration data from houses in Port Pirie.

9.5 SAMPLING STRATEGY

For this work, it is assumed that outdoor/indoor air exchange is a continual phenomenon (as Crockford, 1992). Further, it is also assumed that the contribution from indoor sources such as dusting/flaking paint, home activities (other than home renovation) and cigarette smoking is small by comparison with sources such as closed or open windows/doorways and occupational activity including passage through home entry points (Alzona et al., 1979). Given the long-term smelter presence in the township, substantially elevated soil and dust lead levels (var. refs.), it is assumed that indoor sources of lead would be minor for occupied houses, and irrelevant for vacant houses, as for comparable work conducted elsewhere (Yocom, 1982). In

[7] Whilst the main thrust of this work is directed toward air sampling in vacant homes, sampling was also conducted in occupied homes in order to provide a framework for valid interpretation of the elicited data. The literature demonstrates the relative paucity of available such data corresponding to Port Pirie.

the Port Pirie region, this has been reflected by earlier work conducted by Body et al. (1988), who noted that the proportion of surface dust lead particles composed of by paint lead was small. This is consistent with general literature reference to dwellings constructed in areas of elevated soil-lead and (external) air-lead levels, such references claiming minor contributions to lead-in-dust indoors deriving from internal sources other than home renovations (Brunekreef, 1984; Fergusson and Schroeder, 1985; Rowe et al., 1985). None of the houses studied exhibited active fireplaces or combustion heaters during investigation.

It has been reported that indoor suspended particulate levels decline with increasing distance into the room from external doors and windows of occupied houses (Lebowitz, [undated] - cited in Ju and Spengler, [1981]). Others, however, have concluded subsequent to multiple point (occupied) house air sampling, that levels of respirable, suspended particles are similar in various rooms of the houses of non-smokers (Myers, 1978), Ju and Spengler, (1981) concluding that between-home differences were generally much greater (and much more important) than between-room differences in any test home.[8]

Given that the test houses in this study were closed and vacant at the time of test, and given limitations to sampling equipment availability, sampling was only conducted at one location in each of the rooms tested.

9.6 METHODOLOGY

For a number of occupied houses located in both Port Pirie West and Solomontown, contemporaneous indoor/outdoor assessment of lead-in-air levels was conducted as part of an overall two year study, in order to provide an estimate of mean indoor and outdoor air lead levels. Identical air monitoring was also conducted during this period in the suite of vacant houses.

9.6.1 SAMPLING EQUIPMENT

Whilst an example of the use of a high-volume sampler for indoor air assessment appears in the literature (Al-Radday et al., 1993a), it would seem that such an equipment choice may be inappropriate. Quite apart from the noise factor associated with the use of a high-volume sampler indoors, the use of such an instrument would also seem inappropriate due to the potential for such a device to alter/influence the indoor/outdoor air exchange dynamics and rate, when operating at an air flow rate of between 300 litres per minute ($l.min^{-1}$) (Kretzschmar and Pauwels, 1982), through $1,400l.min^{-1}$ (Pluss and Strauss, 1973; Dolan, 1987), to $2,000l.min^{-1}$ (Cohen, 1973). This would impact upon the internal air quality, a point referred to subsequently by Al-Radday et al. (1993b), echoing several earlier workers including Lepow et al. (1975) and Yocom et al. (1982; p. 505). Thus, it seems appropriate to design air sampling equipment that will minimally disturb the environment within which it is located, and also occupants of the dwelling.

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[8] Such between-home fluctuation was substantially greater than the external air suspended particle fluctuation, Ju and Spengler (1981) concluding that such an observation reflected the influence of occupant activity.

There exists no clearly-established overall standard for such air monitoring. Noting the variation in sampling technique/equipment that has been revealed by the quite limited (literature-reviewed) studies of this type that have been conducted to date (eg Yocom et al., 1982; Davies et al. 1987b & e), it was decided to adopt the use of a number of robust, diaphragm-type vacuum pumps for air sampling. Each pump aspirated air samples through a PVC manifold and two subsequent samplers, not dissimilarly to Davies et al. (1988 & 1990), who also conducted multiple-house internal air sampling.[9] Sampled air passaged 37mm closed face aerosol monitoring cassettes (Gelman Sciences, Ann Arbor, Michigan) with their sampling faces pointing downwards (as USSC, 1973; Alzona et al., 1979). The cassettes contained a paper filter and backing pad (0.8 micron poly cellulose ester; Gelman Sciences - NIOSH standard for lead in dust in air; [Gelman Sciences, 1993]), other workers having used similar filter materials (eg Caffo et al., 1980; Turner, 1980; Davies et al., 1987b, e, & 1990).[10] Attached directly to the pump was an adjustable, variable orifice, moving ball rotameter (0 - 10l.min^{-1}; Series VF Visi-Float; Dwyer Instruments Inc, Michigan City, Indiana) to reflect flow rate. Additionally, two multi-purpose gas meters (Toyo Gas Meters; Davies Shephard, Caringbah, NSW) were used to more accurately indicate total air sampled by each cassette over the sampling period. The sampling apparatus is described in more detail in Appendix Nineteen.

9.6.2 OPERATING CONDITIONS

Air flow rates were set at around one to two litres per minute (mean = 1.5, SD = 0.18), similar to rates used by some other workers such as 0.8l.min^{-1}, (Moschandreas et al., 1978), 1.0 - 1.5l.min^{-1} (Diemel et al., 1981), 1.7l.min^{-1} (Alzona et al., 1979; Turner et al., 1979; Watt et al., 1983; Que Hee et al., 1985), 2.0 - 2.5l.min^{-1}, (Davies et al., 1987b, c & 1990; Gulson et al., 1993 & 1994b), 3.8l.min^{-1}, (Lewis et al. 1991), 4.0l.min^{-1}, (Koutrakis et al., 1992) and 7.0l.min^{-1} (Caffo et al., 1980). The sampling location of the cassettes inside rooms was set at the (suggested) standard sampling height of approximately 1.2 - 1.5 metres above floor level (Roberts et al., 1974b; Harrison et al., 1981b; Spengler et al., 1981; Durando and Aragón, 1982; Davies, 1988). This corresponds to a level slightly above window sill height. External cassettes were also located with sampling face pointing vertically downward, and on the same horizontal plane as their indoor analogue. Both internal and external cassettes were located no more than ten centimetres from the window pane.[11]

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[9] One pump simultaneously connected to two aerosol monitors (indoor and outdoor) thus controlled for fluctuations in flow rate between the separate samplers if, for example, they were each connected to a separate pump.

[10] Sampling at a rate of 2l.min^{-1}, it would take perhaps 11 days to sample one complete roomful of air based upon a room of dimension 4m x 3.2m x 2.8m. Over that period of time it is estimated that many room air exchanges would have taken place (as Crockford, 1992). Considering that for such a new filter, the maximum air flow rate would be of the order of 645l.min^{-1} (Gelman Sciences, 1993; p. 56), some degree of contamination of filter would not suggest a substantial change in filter air flow rate. This was in fact evidenced by casual observation of the flow rates indicated by attached rotameter (see below), virtually no variability being demonstrated throughout the duration of the use of a cassette. Alzona et al. (1979) claim that pointing cassettes vertically downwards eliminates the (very minor) contribution to lead-in-air levels from settling larger-sized particles. For particles in the size range one to ten microns, with downward-facing samplers, Davies (1968) reports that an underestimate of perhaps four percent of air lead concentration may occur due to not sampling sedimenting larger particles.

[11] Such a height for cassettes was similar to that chosen by Thomas et al. (1985) and Davies et al. (1987c & 1990), although their selected height of 1.5 - 2.0m was chosen essentially to minimise child interference, the latter group preferring to have used a figure of ≈ 1.0m to simulate a child's contact zone.

9.6.3 SAMPLING PROTOCOL

A variety of environmental factors such as wind speed and direction may change a number of times in as short a space of time as a single day (Duggan and Inskip, 1985), this applying notably in Port Pirie (G. Norton, pers. comm.). Further, concerns arose regarding the contribution to lead-in-air levels that might occur as a direct result of domicile entry under closed vacant house sampling conditions whilst setting up/removing sampling equipment during short sampling-time experiments (as Duggan, 1984). It was thus decided to sample air continuously from a house for a period of approximately 14 days (mean = 14, SD = 2.19), similar to Yocom et al.'s (1971) and Krause et al.'s (1993) chosen sampling duration.[12] Duggan (1984) clearly appears emphatic about the need to sample continuously over longer periods of time (eg weeks to months) rather than shorter time periods (eg 24 hours), suggesting that the dimension of the sampling error varies inversely with sampling period duration.[13]

Sampling external air at low flow rates, Laxen et al. (1985) and Koutrakis et al. (1992) sampled for seven days continuously. The former group, however, acknowledge the considerable variation in air lead levels that may occur from week to week and also season to season, and conclude that Elwood et al.'s (1984) indoor air study sampling time of seven days was short, possibly leading to a misleading indication of the average exposure. Contemporaneous lead-in-air and dust-deposition sampling for an identical duration constituted part of this study. It was necessary to conduct dustfall collection for at least 10 - 15 days in order to acquire sufficient material to virtually guarantee comfortably exceeding the detection limits of the sampling system. Lead-in-air sampling was thus conducted continuously for an identical period. Such a decision was also commensurate with the limitations regarding available air sampling equipment and available time for access to the field site.

Calibration details for measuring equipment are referred to in detail in Appendix Nineteen. Given the approximate quantity of air to be sampled over the sampling period, and (preliminary) external air TSP data derived from high-volume sampling data (M. van Alphen, pers. comm.), it was not considered appropriate to attempt to gravimetrically determine the quantity of material intercepted by the cassette filter. Air pumps, which were specified as continuous operation, were extremely reliable, not excessively noisy, and very stable with respect to air flow rate and air-temperature changes. They were located external to the room being sampled, in order to contribute minimally to within-room air movement changes resulting from sampling.[14] When repeat sampling in a home was conducted, at all times, the pump and filter apparatus was set up in an identical fashion and location (Plate Seven, page 305). Flow rates at all sampling locations were kept as similar as was possible. Flow rates throughout sampling at each site were monitored on a daily basis (typically twice) to also ensure constancy of flow and sampled particle size range (Cohen, 1973; USSC, 1973).

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[12] Even longer sampling times would be more desirable eg sampling for perhaps 28 - 30 days as Diemel et al (1981), but practical limitations and time constraints indicated a shorter time period would be more appropriate.

[13] Duggan (1981) claims that since the hazard from exposure to environmental lead is typically a long-term one, it is best judged by a knowledge of airborne lead averaged over a lengthy period - essentially weeks or months rather than days, Kowalczyk et al. (1982) and Godbeer et al. (1984) also reporting that short-term (24 hr) sampling may elicit results that are too variable due to daily wind direction changes.

[14] The estimated electricity costs of operation of pumps was around 75 cents per week of operation. Where householders were willing to participate in home air sampling on their premises, they were reimbursed for this cost.

9.6.4 SAMPLE CHEMICAL ANALYSIS

All cassette-contained material was digested by the same process (sequential Nitric and Hydrochloric acid solution) as that for all other cassette-contained material eg the static dust suite (Appendix One). Subsequent to digestion and dissolution, all solutions were analysed for multiple elements using Inductively Coupled Plasma-Mass Spectrometry (Fisons/VG Plasmaquad Turbo Plus). A strict QA/QC program was applied, details of which appear in Appendix Three.

9.7 RESULTS

9.7.1 PRELIMINARY EXPERIMENTATION

A preliminary study of 22 available mixed-occupancy premises distributed around Port Pirie was conducted in order to test the equipment, refine the operating process, if necessary, and elicit an initial estimate of outside lead-in-air levels adjacent to the house. All samples were externally-derived, sampling cassettes being located within ten centimetres of the building. Mean sampling time was 13.7 days, mean sampling volume was 29 cubic metres of air, and the mean lead-in-air concentration was $0.17\mu g.m^{-3}$. Resulting from this preliminary work, the following major component of the air sampling section followed.

9.7.2 MAIN RESEARCH COMPONENT

From a total of 83 paired indoor/outdoor sampling locations, 36 were collected from 32 occupied dwellings and the remainder from the ten vacant dwellings. All sampling locations derived from either Port Pirie West or Solomontown. All sampling from vacant dwellings was conducted in closed-house format, ie all windows and external doors remained closed for the duration of the study. From the (normally-distributed) data corresponding to the volume of air drawn through cassettes located internally and those located external to the test sites, there was no demonstrated difference between these two house sampling locations ($P \geq 0.80$). Mean quantity of air drawn through cassettes was for inside-located cassettes 29.78 cubic metres (SD = 7.42), and for outside-located cassettes 29.78 cubic metres (SD = 7.64). Data reveal that lead-in-air concentration distributions (internally and externally-derived), and also I/O R distributions are positively skewed. Subsequent natural log-transformation of the data produced more normally-appearing distributions, and was used for analysis throughout unless otherwise specified. Ayer notes that air sampling for suspended material produces results "that may reasonably be described by lognormal distributions" (1989; p. 24). Summary air lead concentration data are indicated in Table 9.2 below.

Table 9.2 Air sampling results.

		GROUPED MEAN DATA		
		AIR LEAD CONCENTRATION ($\mu g.m^{-3}$)		I/O RATIO
SAMPLING LOCATION	n	INDOOR AIR	OUTDOOR AIR	
All samples	**83**	0.10	0.19 [15]	0.52
Port Pirie West	**44**	0.09	0.18	0.45
Solomontown	**39**	0.11	0.20	0.54

Regressing indoor air values (dependent variable) against outdoor air values reveals a correlation coefficient of (r =) 0.67 ($P \leq 0.001$), suggesting a good relationship between the data collected for these two sampling location types. This relationship is described by the following:

ln indoor air lead conc. = - 0.9 + (0.89 x ln outdoor air lead conc.).

Measured indoor air lead concentration levels are clearly linked with those detected outdoors. No difference between different district-derived indoor air concentration, outdoor air concentration and I/O R data was detected ($P \geq 0.185$). Regressing I/O R against external air lead level reflects a correlation coefficient of (r =) 0.12, indicating a lack of any real relationship between these two variables ($P \geq 0.30$). This independence implies that it is not the external air lead concentration which "drives" the variability in I/O R, but more, the relative penetrability of the dwelling by suspended leaded dust. Indoor air concentrations for the three sampling location categories above, all appear substantially lower than analogous outdoor air concentrations ($P \leq 0.001$). Given the duration of each in-home sampling study, and also the overall time period over which the study was undertaken, it is difficult to comment upon the potential for district difference. If such difference in fact exists, it would logically seem to be related to specific weather event. A larger number of shorter duration studies would clearly be helpful here.

9.7.3 TEMPERATURE TO AIR LEVEL COMPARISON

Regressing I/O R (dependent variable) against mean daily maximum temperature for the duration of the collection period demonstrates a poor inverse relationship with a correlation coefficient (r =) of - 0.10 ($P \geq 0.36$), suggesting little influence on ratio by seasonal temperature variability.

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[15] Although not strictly comparable, air-suspended lead concentration deriving from particles of a mean of ten microns or smaller, as sampled with a high-volume sampler over the years 1988 - 1991 inclusive, revealed a mean lead level of $1.3\mu g.m^{-3}$ (Goh and Hope, 1992), clearly substantially greater than that recorded immediately outside the residence. This would suggest that basing (say) exposure models on the data provided by a few high-volume samplers located in the general vicinity of the test area might not be the most appropriate approach.

Regressing (separately) indoor and outdoor air lead levels (dependent variable) with mean daily maximum temperature for the duration of the collection period again reveals inverse relationships, and correlation coefficients of (r =) - 0.24 and - 0.21 respectively, although such regression suggests that a minor relationship exists between these variables ($P \leq 0.065$). These modest lead-in-air trends with temperature, however, concur with the results of other workers who report, for other locations, a small increase in lead-in-air values during the colder months (Faoro and McMullen, 1977; Duggan, 1981; Rabinowitz and Needleman, 1982; Yocom, 1982; Strehlow and Barltrop, 1987; Kim and Fergusson, 1994).

9.7.4 OCCUPIED VERSUS VACANT HOUSES

Given concerns regarding the influence of human activity upon indoor (and possibly also outdoor) air lead levels, it seemed appropriate to investigate possible differences between occupied and vacant homes at the time of the study, a summary of which appears in Table 9.3 below.

Table 9.3 Occupied versus vacant home air lead summary.

		GROUPED MEAN DATA		
		AIR LEAD CONCENTRATION ($\mu g.m^{-3}$)		I/O RATIO
SAMPLING LOCATION	n	INDOOR AIR	OUTDOOR AIR	
Occupied premises	**72**	0.13	0.22	0.60
Vacant premises	**94**	0.08	0.16	0.46

Investigation in (closed) vacant houses displays a correlation coefficient of (r =) 0.63 between indoor air lead concentration and its outdoor analogue, with a relationship between indoor air lead concentration (dependent variable) and outdoor air lead concentration as described by the following:

ln indoor air lead concn. = - 0.99 + (0.69 x ln outdoor air lead concn.).

Performing similar analysis for occupied homes reveals a correlation coefficient of (r =) 0.66, and

ln indoor air lead concn. = - 0.96 + (0.89 x ln outdoor air lead concn.).

Whilst r is not dissimilar for either category, the slope of the curves suggests either that the penetration of dust into occupied houses exceeds that of vacant houses, (this being commensurate with Alzona et al.'s observations [1979]), or that the difference reflects the effect of human activity on re-entrainment of already-settled (indoor) leaded dust. Comparing external lead-in-air levels between occupied and vacant houses reveals a greater mean value for the occupied category, this also being the case for indoor air lead concentrations ($P \leq 0.0075$). Comparing I/O Rs between occupied and vacant houses shows a moderately greater mean value for occupied than for vacant dwellings ($P < 0.02$). These data suggest that levels of lead in air within occupied houses are greater than for vacant homes, reflecting the influence of occupant activity within the residence (including opening windows and doors). Importantly also here, externally-derived air lead levels are greater, although given the greater I/O Rs for occupied houses, the indoor component is the more substantially increased value in occupied houses. Modest sample size notwithstanding, such greater external lead-in-air levels adjacent to occupied dwellings reflects the influence of human activity on, presumably, the re-suspension of leaded dust proximate to the residence. Sampling was generally conducted as distantly as was practicable from normal house entry points, concern arising regarding the potential for non-random lead-in-air concentration estimate enhancement due to the passage of house occupants through entry points.

Given the observation regarding house external air lead concentration values, house air lead concentration was contemporaneously assessed both indoors and outdoors in ten vacant homes incorporating ten adjacent occupied (neighbour) homes with samplers very similarly located at all premises. Acknowledging small sample size, such matched pair vacant versus occupied house comparison also exhibits greater levels of indoor air lead in occupied homes, as also a greater indoor/outdoor air ratio ($P \leq 0.05$). Contrasting this observation and also the earlier observation of heightened external air lead levels adjacent to occupied homes ($P \leq 0.0075$), the outdoor air lead levels assessed here reveals no difference between (paired) vacant and occupied homes ($P > 0.20$) which may reflect a fairly small sample size ($n = 10$ for each category), no other explanation for this contrast appearing appropriate.

Given these occupied versus vacant house I/O R differences, and acknowledging Davies et al.'s claim that outdoor air is a reasonable predictor of indoor air lead level (1987b), inter-house comparison of contaminant level thus suggest the need for some attempt at control for occupancy status.

9.7.5 HOUSE ASPECT: FACING THE SMELTER OR FACING AWAY

Comparison of facing with facing-away from smelter air-sampling locations for the entire data suite demonstrates, for facing aspects, numerically larger mean values for indoor air lead levels, outdoor air lead levels and I/O R. Such difference, however, was modest for both indoor and outdoor air ($P \leq 0.20$), outdoor air reflecting the greater difference. The far less substantial I/O R difference ($P > 0.90$) infers that the relative penetrability of either house aspect would not appear to differ greatly. Contemporaneous facing and facing-away comparison for grouped occupied and vacant homes displays similarly, as did comparison for (separately) occupied and vacant homes. Thus, little difference between facing and facing-away aspects of home-sampling locations has been illuminated by these results, although mean air lead levels both indoors and outdoors are numerically greater for facing-smelter house aspect.

This result is different to but does not necessarily contrast the results of Elwood (1983) who notes, for the movement of petroleum-derived lead in an urban environment, that the street-facing room demonstrates greater I/O Rs than the non street-facing room ($P \leq 0.05$). The Port Pirie result possibly suggests better within-house mixing of aerosol. This is presumably related to houses located there being less well-sealed to air movements (doors and windows frequently being left open for extended periods of time) compared to houses located in colder climates, the latter often being multi-storey. Whilst not indicated in the Elwood study (1983a & b), these houses may in fact be part of a terrace continuum of houses, presenting a substantial barrier to the travel of contaminant suspended material away from the street. Further, also not indicated is whether or not the sampling was contemporaneous. Given that the sample size of contemporaneous sampling in Port Pirie is not particularly large, it would seem appropriate to further investigate this aspect subsequently in both vacant and occupied (more frequently open) homes.

9.7.6 POST-CLEANUP AIR STUDY

Subsequent to a very thorough house cleanup, three consecutive sampling periods were conducted in each of four such test houses utilising two pumps and two sets of indoor/outdoor paired cassettes, one pair sited at opposite ends of the house (one facing the smelter and the other facing the opposite direction). These data were compared with data collected by multiple sampling from the same four houses at various times prior to cleanup, the results appearing below (Table 9.4). Comparison of outdoor air lead levels for the two sampling phases (pre- and post-cleanup) exhibits a substantially lower mean value for the post-cleanup sampling phase, as also for mean indoor air levels ($P \leq 0.01$). This is presumably weather/season related. I/O R comparison, however, reflects no such difference ($P > 0.60$). This observation supports the notion that the dust being collected is not just re-entrained pre-existing room dust, but newly-entering dust.

Table 9.4 Air sampling in test houses before and after house de-dusting.

	AIR LEAD CONCENTRATION ($\mu g.m^{-3}$)				I/O RATIO	
	INDOOR AIR		OUTDOOR AIR			
TEST HOUSE	PRE-CLEANUP	POST-CLEANUP	PRE-CLEANUP	POST-CLEANUP	PRE-CLEANUP	POST-CLEANUP
One	0.24	0.08	0.36	0.13	0.67	0.65
Two	0.14	0.07	0.21	0.14	0.66	0.50
Three	0.06	0.05	0.16	0.14	0.38	0.34
Four	0.12	0.07	0.21	0.12	0.54	0.55

Acknowledging small sample size for post home-cleaning sampling (n = 6), no difference in external home air lead levels is apparent between any of the four houses ($P \geq 0.50$). This is also the case for internal house air lead levels ($P \geq 0.10$). Regression of indoor air lead levels against outdoor levels provides a figure for r of 0.62, and the following relationship

$$\ln \text{indoor air lead conc.} = -1.0 + (0.84 \times \ln \text{outdoor air lead conc.}).$$

The effect of de-dusting of houses as part of an intervention program clearly appears minimal as reflected by the I/O R; certainly, there is not a dramatically decreased I/O R subsequent to intervention. It could be suggested that, as a result of the sampling of unsettled cleanup-disturbed or re-entrained existing internal dust due to the chronological proximity of sampling following cleanup (Figley and Makohon, 1992), some contribution to indoor air lead levels could have resulted. In this work, however, in order to avoid just such a possibility, all houses that were cleaned were subsequently closed (windows, doors etc) for from two to four weeks after cleanup and prior to sampling. During this time, houses were entered once for static dust sampling seven to ten days following the cleanup, all sampling being conducted in a clean fashion, with doors and windows closed. Absolutely no other worker entry was permitted apart from this.[16]

The sample size here is not large, and reflects the limited opportunity for sampling in these test houses. Given the results, however, these data imply that the effect of house cleanup upon subsequent indoor air lead levels is minimal. Such a conclusion infers that that air lead sampled in the test houses derives more from novel dust entry into the test homes rather than from the re-suspension of existing dust. To consolidate upon these findings, a more substantial investigation into home air lead levels than has been presented here is clearly warranted. Comparison of facing (smelter) versus facing away for the post-cleanup sample again reveals no difference between these two alternatives for indoor or outdoor air lead level, and also I/O R ($P \geq 0.50$), although, again, sample size is not large.

9.7.7 RELATIONSHIP BETWEEN LEAD-IN-AIR AND LEADFALL LEVELS

9.7.7.1 PREAMBLE

Contemporaneous lead-in-air and dust-deposition assessment was made under closed house conditions in the four test homes used for post-cleanup monitoring. The intention was to estimate the relationship between ambient air and depositing dust lead levels, Peirson et al. (1973) claiming that (dust) deposition rates typically follow those of airborne contaminant concentrations. Butler and MacMurdo (1974), Alzona et al. (1979) and Al-Radday et al. (1993a) report rapid equilibration between indoor and outdoor air, and the settling of dust internally as a result of the still air inside rooms.

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16 In order to hasten dust settlement, apart from ensuring the closure of all windows and doors, external chimney stacks were sealed over with commercial grade plastic sheeting to minimise air draughts. These were removed with extreme care prior to the commencement of sampling. Prior to house cleanup, none of these test houses had been occupied or cleaned for nearly two years, facilitating the on-going entry of dust under varying house conditions ie open/closed windows/doors etc. During the house-cleaning process, all workers wore clean shoes/clothing, and minimised house exit and re-entry. Ceilings were cleaned first, followed by walls, windows, doors and lastly floors.

From work conducted in vacant homes, Sayre and Katzel note that settled surface dust represents a depositional phenomenon, such an "active and dynamic process being at least in part related to atmospheric air levels of suspended lead" (1979; p. 181). Angle et al. report the same rank order relationship for a series of dustfall and air lead values derived from various collection points, inferring an "obligatory intercorrelation of air lead and atmospheric fallout" (1984; p. 169). They further observe that such a model deserves investigation in a variety of community studies. Analogously, Budd et al. (1990) claim that a significant relationship exists between the concentration of some pesticides in house floor dust and in the surrounding air, Lewis et al. (1991) demonstrating similarly. Camann et al. note a "strong and statistically significant correlation of the indoor air and dust concentrations for all (studied) pesticide compounds" (1991; p. 1,113). This group also detected a strong correlation for these compounds between their air levels and their loading as detected by a surface sampling technique not dissimilar to that used for this research.

From a study involving the characterisation of airborne heavy metals proximate to a smelting plant utilising X-ray powder diffraction spectrometry techniques, Harrison et al. (1981) conclude that the phases identified in samples of floor dust are also present in the atmospheric samples. Further, they claim confirmation that sedimented floor-level dust is typical of deposited airborne material. Essentially, the I/O R reflects the penetration efficiency of the suspended aerosol, and also the decay of the material in indoor air, this depositing by Brownian diffusion and sedimentation (Koutrakis et al., 1991 & 1992).[17]

9.7.7.2 EXPERIMENTAL RESULTS

Dust deposition was collected for a period of 47 days, over which time three separate sets of indoor and outdoor air monitoring cassettes situated in two different locations in each house were used consecutively, with minimal time required to change over the cassettes and no entry to the houses being necessary. Thus, from each test house, a total of six (indoor) lead-in-air values were averaged and regressed against the mean of approximately ten dustfall collectors located throughout the house. As an index of the relative precision of I/O R values from these examples of multiple same-location samplings, the coefficient of variation of I/O Rs for these test houses ranged between 0.5 and 2.0%. Regressing leadfall (dependent variable) against same house (indoor) lead-in-air mean values reveals a correlation coefficient of (r =) 0.999, and the following relationship:

$$\ln \text{leadfall } (\mu g.m^{-2}.d^{-1}) = 5.2 + (0.75 \times \ln \text{lead in air } [\mu g.m^{-3}]) .$$

Clearly a strong relationship between these two variables emerges, the corresponding correlation coefficient for leadfall (dependent variable) with external air lead concentration appears as r = 0.60. Thus, from these data, as a surrogate leadfall parameter, indoor air lead concentration would appear to represent a satisfactory substitute.

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[17] It is acknowledged that the concentration of air-suspended contaminants may vary throughout a room (USSC, 1973). Under closed house conditions, however, with substantially reduced air movement levels as compared to open house conditions, this difference is not likely to be significant within a moderately-sized room (Thompson et al., 1973), equilibration of suspended particulate material being suggested to be rapid (Koutrakis and Briggs, 1992).

This conclusion, however, is less convincingly so for external air lead concentration. Certainly given the ease of set up, sample analysis and also the reduced time exposure period required for lead-in-air assessment as an index of contamination within a room, such sampling would be preferable to dust deposition collection. This of course would need to be further tested in both open vacant houses and also occupied houses.



Plate Seven Air-sampling system demonstrating placement of external sampling cassette and gas meters. Note consistency of meter-indicated total gas flow.

CHAPTER TEN GENERAL DISCUSSION

It is not just a mistake for public health agencies to cooperate and collaborate with industries in investigating and deciding whether public health is endangered - it is a direct abrogation and violation of the duties and responsibilities of those public health organisations.

Statement by Clair Patterson to the 1966 US Senate Investigating Committee hearing regarding the role of lead in combusted gasolene, air pollution and public health.

Imagination is more important than knowledge.

Albert Einstein

10.1 INTRODUCTION

Acknowledging a threshold level of concern for lead in residential dust of 500$\mu g.g^{-1}$, and 500 - 1,000$\mu g.g^{-1}$ for soil, mean lead levels noted in this study for a variety of environmental compartments are reasonably high, and are indicated below (Table 10.1). Comparable parameter mean values from the Arnhem (Netherlands) lead study where health concerns regarding lead were also expressed (Brunekreef et al., 1981) are also included. Research there focused upon houses located between 450 and 1,000 metres from a secondary lead smelter, smaller than that located at Port Pirie (Diemel et al., 1981).

Table 10.1 Summary data of Port Pirie and Arnhem studies.

	VARIOUS PARAMETER SUMMARY DATA	
PARAMETER	**PT PIRIE STUDY**	**ARNHEM STUDY**
DUST LEAD CONC. ($\mu g.g^{-1}$)		
Static Dust	4,250	957
Depositing Dust	6,350	1,007
Soil	1,184	240
Indoor Air	0.10	0.26
Outdoor Air	0.19	0.41
LEAD LOADING ($\mu g.m^{-2}$)	4,580	120
LEADFALL ($\mu g.m^{-2}.d^{-1}$)	59	7.34

Source: Arnhem Study summary data derive from Diemel et al., 1981.

Sections of the township of Port Pirie, as represented by sampled dust and to a lesser extent surface soil from a number of test houses, reflect considerable contamination by lead. The source of this lead has clearly been demonstrated to be intimately associated with collected indoor dust. Such a conclusion justifies work such as that undertaken here, and validates the need for further investigation.

As also noted by Fergusson and Ryan (1984), lead levels in dustfall appear greater than for deposited dust, these authors claiming that this pattern arises as a result of the smaller particle size range of depositing dust. Archer and Barratt (1976a) report that these larger particles in settled dust dilute settling dust and may arise from a different pathway.

Observed lead-in-air levels inside and outside appear low by other study standards (Tables 3.6 and 9.1), although in keeping with (external) lead-in-air concentrations for moderately developed rural Australian locations which typically lie in the range 0.1 - 0.3$\mu g.m^{-3}$ (AAS, 1981). This reflects the small country town character of Port Pirie, and also the residential nature of these premises, all but one of which are located abutting low traffic density roads. Given the smelter presence and the generally contaminated nature of the environment, it may also reflect the effective distribution of airborne contaminants by aeolian activity. Further, such levels also reflect the average distance between sampling site and presumed point source, lead-in-air levels typically decreasing exponentially with increasing horizontal distance from a point source (Fergusson and Stewart, 1992). Thus, basing resident exposure models on data deriving from a few existing high-volume samplers within the township may be inappropriate, given the larger mean lead-in-air levels recorded by these devices, and their purpose-selected location proximate to the smelter. The substantially greater static and depositing dust lead concentration, static dust lead loading and leadfall levels, infer greater penetrability of Port Pirie houses by lead-bearing dust, compared to those from the Arnhem study. Similarity of I/O R, however, appears to contest this, given lower air-suspended lead levels for Port Pirie. It may be that the Arnhem study, using different sampling techniques, underestimated dust lead levels, or that the particle size range of the collected dust varied from that in the Port Pirie study. This may have been compounded by the occupied nature of the Arnhem houses.

The observed substantially lower surface soil lead concentration values compared to house dust values infers that some sort of dust lead trapping process (as Fergusson and Schroeder, 1985) and a concentration mechanism for lead in dust is occurring as dust enters homes. From results reported by a number of researchers at a variety of locations (eg Sayre and Katzel, 1979; Fergusson and Schroeder, 1985; Body, 1986f; Fergusson et al., 1986) there would appear to be a process of enrichment of contaminant materials within the home as compared to residential soils. Kim and Fergusson (1993) acknowledge this concentration mechanism consistency amongst other workers' results, and contemplate (as others, eg Sayre and Katzel, 1979; Thornton et al., 1985) that it arises as a result of the selective enrichment within houses of finer, more contaminant-rich particles and the selective removal, by cleaning, of the larger, lower contaminant level particles.[1]

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[1] Thornton et al. (1990) in a study of urban/rural households, arrive at a value of 2.2 for the ratio indoor dust mean lead concentration compared to mean soil lead concentration, Kim and Fergusson (1993) derive a figure of 2.6, a figure similar to that demonstrated by this work, namely 3.6, although house vacancy and closure prior to assessment here may have contributed to elevation of this ratio somewhat. This is further discussed below.

10.2 PRELIMINARY RESEARCH

Comparing the mean lead concentration of floor dust acquired from houses located within the township of Port Pirie (1,637$\mu g.g^{-1}$) with those deriving from houses immediately outside the township (221$\mu g.g^{-1}$), it is evident that the source of the leaded dust derives locally. Further, the mean concentration of lead in dust collected by domestic vacuum cleaner from all selected houses in Port Pirie (1,637$\mu g.g^{-1}$) is substantially lower and more variable than the corresponding value for dust acquired from the test houses by the dust-collecting instrument developed specifically for the research conducted here (4,249$\mu g.g^{-1}$). It is acknowledged that test houses are not identical, although all derive from within four kilometres from the centre of the smelter, and are characteristic of houses located in the two districts. Further, they all lie within a ninety degree radial sector of the township, with the smelter located at the centre hub.

Special high-efficiency vacuum cleaners sampling from test house floors indicated a mean lead in floor dust concentration of 5,454$\mu g.g^{-1}$, this figure being substantially larger than the corresponding figure for similarly-sampled occupied home floors in the same districts in Port Pirie (2,248$\mu g.g^{-1}$; L. Weih, pers comm.). Occupancy thus would appear to be associated with a lower mean house floor dust lead level, tracking of larger-sized dust particles (of a lower dust lead concentration) presumably contributing to this lower mean level. It would seem that such dust particles would be unlikely to remain suspended in air due to their size, and would be precluded entry by simple air exchange alone. Occupant cleaning activity also would elicit some influence here.

The majority of the dust-contained lead lies in the less than 100 micron particle-size range, this being of the particle size order of hand-adherent dust, a decreasing (dust) particle size corresponding to an increasing lead content. Clear implications for (child) occupant health thus necessarily follow.

Particle size spectrum similarities between above-ceiling space and floor-collected dust in vacant houses infers similarity of pattern of contamination, particularly given the size differences between these dust samples and those deriving from the floors of similarly-located occupied houses. In this context, for occupied house floor surfaces, however, it is likely that occupant activity constitutes a potential source of material entry. This appears to involve larger-sized particles (particularly in the > 75μ particle size range), which are less likely to enter the house via deposition of previously suspended material. Representation deficit of the < 75μ particle size range dust for occupied home floors implies resuspension due to human activity, possible collection by routine cleaning protocol and/or some subsequent deposition upon non-floor surfaces. The suggestion is made here that the contamination of both vacant house location categories, as reflected by particle size, is both contemporary and on-going. Further, it is also likely that the collected-dust particle size spectrum for above-ceiling space dust appears not to have changed substantially, historically.

10.3 DUST SAMPLING TECHNIQUES

10.3.1 DEPOSITED DUST

Deposited dust particle-sizing work reveals a lower mean particle size and a smaller overall particle size range for cassette-collected compared to domestic vacuum cleaner-collected floor dust samples. Smaller-sized particles are thus less likely to be retained by domestic vacuum cleaners, hence supporting the premise of non-random sampling by this method, as presumed by Body (1986f). The use of domestic vacuum-cleaner bag grab samples for sample provision thus elicits the need for caution, given concerns regarding how truly representative of the house contaminant floor dust population such a sampling system really is.

Conversely, the chosen dust-collecting system utilised for this study retains smaller-sized particles more readily than domestic vacuum cleaners, providing a dust sample of particle size range more representative of the expected (child) hand-adherent dust lead population. Thus, the chosen dust-collecting system is appropriate for collecting dust representative of contaminant house dust populations, and also that available to a child. Further, acknowledging dust as the vehicle for lead entry into homes, the substantially better association of dust loading with lead loading rather than with dust lead concentration as noted here, necessitates the choice of a sampling system which provides lead loading data. A further limitation of vacuum cleaner bag grab samples lies with such a sampling technique providing an estimate of the concentration of lead in collected dust, but failing to provide an estimate of the total quantity of leaded dust present on floors of houses. Given various researchers' comments regarding the better association between occupant blood lead levels and floor lead loading rather than dust lead concentration (eg Davies, 1988), the choice of a sampling system providing lead loading data is thus obvious. Further, from work conducted here, within-house floor lead loading data demonstrate a greater range of values and also a greater sensitivity to spatial difference than does dust lead concentration.

From such results, it is clear that there exists a need for a standardised (deposited) dust-sampling technique, some concerns arising regarding the relative comparability of results of much of the earlier-published work. It is thus appropriate for some collected dust particle-sizing and instrument efficiency-validation for researcher-selected different sampling systems to be conducted prior to their use as representative sampling systems. Additionally, validation of different operator dust sampling efficiencies is also warranted.

From the static dust investigations, little evidence of substantial differences in dust lead concentration between the various comparative house categories has been presented, ie mostly, comparison reveals minor category difference for this parameter. This however is frequently not the case for lead loading, which infers that, as an investigative parameter for contaminant variability (particularly as an indicator spatially), lead loading really represents a more sensitive and appropriate choice. This is despite much previous work where dust lead concentration has been used solely as just such a barometer.

10.3.2 DEPOSITING DUST

Whilst a number of alternatives for depositing dust collection have been considered, the system developed specifically for this work seems to be the most desirable. The facility of providing leadfall and (gravimetrically) dustfall data and hence dust lead concentration is valuable, the results of such a collection system illuminating, again, the close association between depositing dust and lead. Analogously to (deposited) dust lead loading versus dust lead concentration data, leadfall data represents a more sensitive indicator of spatial contaminant variability than simply dust lead concentration.

Whilst good depositing dust-collection similarity for leadfall exists between alfoil trays and sample cups, dustfall and hence dust lead concentration cannot be estimated by the latter method. Thus, no relationship between depositing lead and dust may be revealed. This is a limitation for any technique where dustfall is not able to be determined, particularly if work is being directed to sourcing of depositing dust. Additionally, the larger surface area of the sampling system chosen here as compared to the cups, facilitates greater detection sensitivity, and the potential for a smaller sampling period duration.

10.3.3 CONTAMINATION INDICATORS: STATIC DUST VERSUS DUSTFALL

Two significant influences upon room contamination, namely the rate of depositing dust and also its lead concentration, are reflected by dustfall collection. These may not necessarily be so clearly defined from the static dust suite of data. Here, such data potentially display contamination that arises as a composite of various sources (existing dust plus newly-entering), plus the mixing of existing dust during prior/current house occupancy. As an indicator of the potential for a child to acquire contaminant dust from a room floor, the static dust/lead loading level represents an appropriate parameter to choose for current risk. As an indicator of the mechanism of room contamination, however, depositional dust collection, eliciting current dust lead concentration and, particularly, leadfall data is more appropriate.

10.4 CONTAMINANT DUST DISTRICT DIFFERENCES

From the soil sampling conducted in this study, no apparent difference exists between either district with respect to the calibre of the soil lead contamination, reflecting a marginally greater mean lead concentration value for Port Pirie West. This is commensurate with the results of the static dust survey indicating no real district difference for lead loading, controlling for dust loading ($P > 0.75$). Essentially, whilst Solomontown may have been a dustier locale, the degree of contamination of that dust was no different, with respect to mean lead level, to that from the other district. Mean dust particle size, however, was smaller for Solomontown than Pirie West residences, commensurate with a decreasing particle size of

depositing dust with an increasing distance from a point source (Moseholm et al., 1992), mean Solomontown test house-to-smelter distance being larger than for Port Pirie West.

The air-sampling data again revealed that district differences in degree of contamination by lead were not apparent. Unlike district mean soil lead concentration figures, however, district mean air lead levels (indoor and outdoor) as for mean static dust lead concentration levels appeared to be greater for Solomontown than for the other district, albeit only marginally.

Depositing dust data, however, are somewhat more informative. Solomontown test homes reveal a different (more contaminated) depositing-dust population ($P \leq 0.001$) in addition to exhibiting a slightly greater depositing dust rate. This translates as a greater rate of leadfall than is the case for Port Pirie West ($P < 0.001$).

Such apparently anomalous results can, however, be reconciled. It may be argued that simply on the basis of small sample size for Solomontown alone, district differences may not be readily elucidated for some of the test variables. Alternatively, it could be argued that surface soil and internal dust characteristics represent a degree of mixing of dust type materials of different sources, both cases being related to former occupant activity. Given varying contributions from a variety of different, human-elicited sources, real district differences may be somewhat masked. An alternative suggestion for the absence of district differences in static dust sampling may relate to a lack of sensitivity of the sampling technique to such differences, although, given highlighted (subtle) differences relating to house aspect facing versus facing away from the smelter and within-room spatial differences (discussed below), this would not seem likely.

Given Davies' comments (1987a, b & c) regarding the contaminant-level similarity of concurrently sampled different house air masses despite the fact that his test houses were quite widely scattered (certainly substantially more so than those sampled from here; D. Davies, pers. comm.), it appears not totally surprising that district differences in mean lead-in-air level may not have been apparent here. On each occasion, sampling was conducted for approximately a two-week period, wind speed and directional variability presumably contributing to effective mixing of suspended material in the air mass influencing each district during the sampling period.

From the post-cleanup depositing-dust work, Solomontown reveals slightly larger mean levels for all depositing dust parameters, no real difference being apparent, however ($P \geq 0.20$). Similarly, for static (floor) dust collected by vacuum pump/cassette from these houses over the post-cleanup depositing-dust exposure period, Solomontown houses demonstrate greater mean levels for all depositing-dust parameters, albeit acknowledging a modest sample size ($n = 32$). This difference is, however, more pronounced for leadfall and dust lead concentration ($P \leq 0.035$). This is not simply a result of post-cleanup pre-existing differences, these not being readily apparent immediately subsequent to house cleanup ($P \geq 0.10$).

Surface soil and air, selected as sampling media representative of current contamination, reflect some degree of mixing of lead sources, and possibly also a lack of sensitivity as a technique for discerning difference between district. Such a "mixing" conclusion is also likely for static dust. Doubtless, depositing-dust district difference patterns are complicated, although Solomontown is a more contaminated district on the basis of unoccupied-house, depositing-dust patterns principally.

This is commensurate with:

a) a greater mean house-to-smelter distance for Solomontown residences than for Port Pirie West residences,

b) an observed smaller (static) dust particle size profile detected in Solomontown houses, and

c) an inverse particle size and lead content relationship, as also reported elsewhere (eg Fergusson and Ryan, 1984; Duggan and Inskip, 1985).

Such a decreasing particle size range with increasing distance from a point source also appears commensurate with the observations of others (eg Moseholm et al., 1992).

Noted observations here are, on occasion, associated with small sample size, reflecting some limitations of parts of this study. This clearly warrants further work, particularly aided by a larger number of available houses in both districts.

10.5 CONTAMINANT LEVEL VARIATION WITH DISTANCE FROM SMELTER

Soil element data suggest a modest decreasing smelter-emitted contaminant gradient radiating from the lead works, this not being the case for other non smelter-derived soil elements which demonstrate no such gradient.[2] The potential for home occupant influence upon surface soil contaminant levels by the earlier-adopted practice of using smelter-derived material for fill purposes is acknowledged. Nonetheless, even though soil has been replaced with clean fill at the Vera St home, the mean surface soil lead level there is close to the highest level of any of the test homes.

Increasing (static) dust lead loading and dust lead concentration with increasing distance from the smelter is commensurate with a converse relationship between particle size and distance from the smelter, and a noted inverse relationship between particle size and lead content. It is likely, however, that the range of test home-to-smelter distances is not sufficiently large to reflect an inverse contaminant level with distance from the smelter relationship as demonstrated by the comparison of vacuum cleaner-derived samples from Port Pirie with those deriving from outside of the township. Again, a larger suite of houses with a greater range of house-to-smelter distances is warranted. Depositing lead and dust lead concentration are positively related to the distance from the smelter, but only very marginally so, as for static dust. This absence of a firm relationship may also be related to a small overall range in distance from sampling site to smelter.

The contrast between a generally decreasing soil lead concentration with increasing sampling distance from the smelter, versus a very modest increasing static and depositing dust lead concentration with

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[2] Again, support is garnered for the concept of the contemporary nature of home contamination by currently-emitted contaminated dust. The alternative of a continually settling and re-entraining pool of (historical) mobile contaminated surface dust would seem unlikely given the noted distance/contaminant level gradient for lead and zinc, this being absent for other surface soil elements. Excellent correlation between the surface soil concentration of lead and zinc but not between lead or zinc and copper and iron further supports this premise.

distance from the smelter may reflect the result of a mixing of a variety of sources of external dust, as revealed by the surface soil study. Such mixing of various sources of dust, some of which may be largely uncontaminated (eg the larger-sized particles) may contribute to the diluting of leaded surface-soil dust externally, these dust samples exhibiting particle sizes too large to be effectively air-suspended. Thus, such particles are unlikely to enter houses by this pathway. Consequently, only smaller-sized particles, demonstrating greater lead concentration levels, enter houses, the smaller-sized of which travel somewhat further from the source, eliciting observed house dust/smelter distance gradients.

Notably larger correlation coefficients for all depositing dust parameters with distance from the smelter as compared to analogous static dust parameters were observed. This further implies that depositing dust is a preferable sampling choice for indicating (locational) variability in home contamination potential.

In a general sense, at a macroscopic level, proximity to the smelter would appear to be a reasonable indicator of the degree of contamination of a residence, although given the "clustering" of most of the test houses in the two districts with small distance range, no firm contamination/distance relationship can be argued. Further, there is a good correlation between (static) lead loading and dust loading, seemingly independently of house distance from the smelter, as also evidenced by the Trail (USA) Lead Study (Clark and Bornschein, 1990; Hertzman et al., 1990). Both studies note the widespread nature of lead contamination amongst homes within the test area, and the potential for most houses in the townships to attain potentially high lead levels indoors. Such data appear not inconsistent with earlier comments by a number of researchers to the effect that the closer a child's home is to a smelter site, the greater is the child's exposure to and likely absorption of substances emitted by the smelter (eg Milham and Strong, 1974; Landrigan et al., 1975 & 1976; Baker et al., 1977; Yankel et al., 1977).

These observations are consistent with the observations of other workers (eg Handy et al., 1980) who report a curvilinear relationship between environmental contaminant levels in soil and dust. These authors further note that contaminant levels decrease up to a distance of around two kilometres from the point source, at which point environmental levels then no longer decrease, but remain at a relatively low level with increasing distance for some distance from the point source.

10.6 HOME SPATIAL VARIABILITY OF LEAD CONTAMINATION

10.6.1 SOIL

Concentration foci of contamination of surface soil by lead occur proximate to dwelling perimeters in test houses, commensurate with suggestions by a number of other workers (eg Watt et al., 1993). This is also the case for contamination by zinc. These observations confirm that such concentration occurs predominantly by the collection of passaging dust rather than by the deterioration of painted surfaces, particularly for the test houses here, given the substantially elevated (above background) soil lead and zinc values adjacent to the house perimeter. This concept of the impeding and (external) entrapment of

progressively moving leaded dust by the dwelling structure is consistent with the observation of a focus of contamination in the more protected walkway region of homes. Perambulating contaminated dust appears to be ensnared by this somewhat closed region, and it is suggested here that such deposited dust largely remains there as a moderately permanent addition to surface soil.

Additionally, the noted decreasing soil lead concentration with increasing distance from the home implies that this soil lead arises more from contemporary deposition and less from lead-containing fill that may or may not have been used as soil augmentation previously. At least for the house located at 27 Vera St, this is the case for the soil in general, given its recent replacement and current generally elevated lead level. Static dust parameter values at this dwelling are lower than the mean for other Pirie West houses ($P \leq 0.05$), yet depositing dust lead concentration and leadfall are larger ($P \leq 0.05$). This contamination at this location infers the relative novelty of the dust-deposition process, and also its currency, and that the dust that is currently depositing is not clearly related to that which has already deposited. Given facing-smelter versus facing-away contamination difference observations (discussed in detail below), the existing contamination infers directionality of contaminating source (ie the smelter), and also a novelty of contaminant source rather than suggested historical antecedence.[3]

Excellent correlation between soil lead and zinc concentrations at all house locations individually (r in the range of 0.83 - 0.99) including 27 Vera St, despite varying degrees of soil contamination, further detracts from the concept of the surface contamination arising from (previously) imported soil fill. This is particularly so given the proportions of lead to zinc in such material (mean $\approx$ 0.13), observed current home soil lead/zinc ratios being more substantial (mean $\approx$ 0.54).

10.6.2 STATIC DUST

Given the excellent correlation between (static) dust loading and lead loading, the source of the deposited dust is closely associated with the lead present. A mean dust lead concentration for indoor dust of 4,250$\mu g.g^{-1}$ represents substantial contamination. Carpet-covered surfaces convincingly demonstrated greater mean dust and lead loadings than either wooden or linoleum surfaces, worn carpet reflecting greater available dust and lead than carpet exhibiting little or no apparent wear. No evidence for the collection/concentration of leaded dust towards the periphery of a room has been presented here, further suggesting that the source of leaded floor dust is unlikely to derive as a result of the degrading of painted

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[3] Body (1986f) notes that 165,000 tonnes of lead have been lost as fugitive emissions around Port Pirie, mostly before 1930, claiming "most of this lead being still available today, and being detected circulating, at present, in Port Pirie" (1986f; p.6). Pasminco, coincidentally, claims that the exposure in towns such as Port Pirie "arises from past industrial practices at times when environmental standards were not as high as they are today" (1990; p. 2). Acknowledging the work of others who observe that lead is very soil stable (Fergusson et al., 1980; Brunekreef et al., 1981; Gulson et al., 1981) and persistent (Cartwright et al., 1976), if this sink of lead in Pirie is mobile and available, it must obviously be distributed above the soil, rather than within it. Assuming the contamination is spread out evenly over a circular area of 30kms radius (over such a distance, it would actually be decreasing exponentially with increasing distance from the source [Roberts et al., 1974a; Yankel et al., 1977; Chamberlain et al., 1978; EPA USA, 1986], this would equal a surface area of around 2,830 square kilometres. Further, assuming that the average concentration is 2,500$\mu g.g^{-1}$ (0.25%, and twice the mean home soil lead concentration figure elicited here), then there must be of the order of 66,000,000 tonnes of leaded dust spread over the township, or 23,320 tonnes of leaded dust per square kilometre. Assuming a specific gravity of 0.5, this equates to 11,760$m^3.km^{-2}$, or a cover everywhere of around twelve millimetres. This clearly cannot be the case.

(inner) surfaces such as walls. Corners of rooms also appear not to represent collection points for leaded dust.

A fairly uniform dust lead loading distribution on floors of test house rooms implies deposition of suspended dust as a more pronounced means of contamination than deposition of shoe-adherent dust via tracking, since it would seem unlikely for foot tracking to be evenly distributed on floor areas, particularly room corners. Specific floor regions, however, do reveal heightened concentration foci of deposited contaminant dust. These areas lie proximate to principal room entry points for lead-bearing dust, and are examined in this context.

10.6.2.1 MECHANISMS OF DUST ENTRY

Principal sources of such dust entry into test house main rooms include windows, doorways and (possibly) fireplaces, these being discussed separately below.

a) Windows

The dust lead concentration at window-base floor demonstrates a greater mean value than the corresponding centre of room location ($P \leq 0.05$). This is the case for all room categories, although only for dust lead concentration compared to floor locations on the other side of the room ($P < 0.05$). Lead loading again reflects greater mean levels at window-base floors than room centres, this difference being most notable for rooms with an open fireplace ($P < 0.05$). Comparison of window-base floor locations with sampling locations either side and the same distance from the wall (with the window) as the window-base floor location, revealed greater mean lead loading and dust lead concentration values for the window-base floor (central) location ($P \leq 0.05$). Post-cleanup static dust data display greater mean values for window-base floors compared to centre of room data for all parameters, albeit at only modest level for lead loading ($P < 0.10$). Again, this implies the novelty of such dust, with a potential entry-point being via the window.

Window-base floors in rooms with an open fireplace exhibit greater mean lead loading and dust lead concentration than analogous locations in rooms devoid of a fireplace ($P \leq 0.01$). Further, such rooms also present with substantially larger lead loading and dust lead concentration mean values at window-base floors than floors opposite windows ($P \leq 0.01$), this being less pronounced for rooms devoid of a fireplace. Given that the test houses remained closed during the vacancy period, it would appear that the potential for more enhanced air movement through rooms with open fireplaces reflects as greater dust deposition in those rooms.[4]

Windows represent such dust entry locations for main rooms, air flowing through the room with one possible outlet via the fireplace and chimney. It may even be that, as a result of the design of the chimney, air may be positively extracted from the room through the chimney, creating a (slightly) lowered pressure

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[4] Essentially, air-suspended particulate matter levels indoors vary (directly) largely upon the velocity of air movement. Areas experiencing low ventilation rates (eg as for house closed, or where foot traffic is low in occupied houses) will display reduced levels of suspended particulate material and lower rates of depositing particulate (Thompson et al., 1973; Al-Radday et al., 1993a).

region in the room, additional air being drawn into the room through potential entry points such as windows. Such results support earlier comments that poorly-fitting windows allow the entry of airborne dust from outside by facilitating "an active dynamic process to occur at or near the window" (Sayre and Katzel, 1979; p. 181). These authors claim that such deposition occurs in the presence or absence of house occupants, Cohen and Cohen (1980) loosely associating levels of indoor air-suspended lead with the number of windows in a room.

No association between walkway-adjacent rooms (with a window opening onto the walkway) and enhanced mean lead loading/dust lead concentration values in such rooms exists. This infers either that heightened leaded dust house entry from the walkway via facing windows does not occur, or, if it does, that effective, within-house dust mixing reduces the detection of such difference.

b) Doorways

- i) Main external doorways

Minimal difference in mean dust and lead loading levels exists between bedrooms and lounge rooms, as also applies between halls and porches. House entrance-ways such as halls and back porches demonstrate substantially greater dust and lead loading levels than other main rooms. Pooled halls and porches' mean dust and lead loading levels exceed analogous parameters for pooled other rooms by 2.11 and 2.65 times, respectively, this difference being highly significant ($P \leq 0.001$). Combined with a well-defined dust and lead loading (decreasing) gradient with increasing distance from the door into the hallway, the clear suggestion is that the external doorway is an obvious choice for a potential contaminant dust entry point. Only one test house failed to present with a (centrally-located) declining lead loading gradient into the hallway from the front door. This was the only house where foot traffic occurred in the hallway subsequent to house vacancy. Given the period of house vacancy for the test house suite and the substantially greater dust and lead loading levels for these vacant houses as compared to a suite of test occupied houses, it seems likely that the contamination process occurs principally as a result of the penetration of suspended lead-bearing dust into the house.

- ii) Hallway to room doorways

Similarly to window-base floor with centre of room comparison, (main room) doorway sampling reflects greater mean contaminant levels than centre of room ($P \leq 0.05$). This is also notable for mean lead loading level when compared to floor locations on the other side of the room ($P < 0.05$). Such observations are more pronounced for rooms containing an open fireplace.

Given the decreasing lead loading gradient from room doorways to the centre of the room, it is likely that the contamination in the hallway occurs principally as a result of dust entering from within the hallway such as from the entrance door, rather than from main rooms into the hallway. The noted lack of difference in lead loading and dust lead concentration mean value between hallway sampling areas

(transversely centrally-located in the hallway) and immediately adjacent to within-room doorway sampling locations also supports this conclusion.

c) Windows versus doorways as potential entry points

Correlation between distance into a (main) room from window-base floors or doorways (as also into hallways from external doors) and lead loading reflects more pronounced coefficients than is the case for distance versus dust lead concentration. This infers that the determining factor for the calibre of contamination at a certain location is more closely related to the quantity of dust depositing at that location than its lead concentration. Again, dust lead concentration values alone appear insufficiently sensitive an index of room/house contaminant level, particularly if spatial variability is of interest.

The observed, more-pronounced contaminant levels at window-base floors than main room doorway floors, imply the predominance of windows as dust-entry points into main rooms. Again, for rooms with an open fireplace, such difference is more pronounced, with lead loading gradients to the centre of the room more well-defined. This conclusion regarding the predominance of windows as dust entry-points derives from investigations in houses having remained unoccupied and closed for some time prior to sample acquisition, the proportion of total floor leaded (static) dust having entered the house subsequent to (former) occupant departure being unclear. From substantially enhanced dust deposition in rooms with windows open 15cm compared to closed, and given the propensity of Port Pirie residents to leave windows at least partially open for much of the day (pers. obs.), it would seem likely that a greater proportion of static floor dust would have entered via the window in occupied homes.[5]

Window well/sill comparison with room region floor data revealed substantially larger mean values for well/sills compared to central, mid and peripheral sampling locations, as also for grouped floor samples ($P \leq 0.005$), the last-mentioned also being noted by Clickner and Rogers (1995). Such comparison appears analogous to hallway contaminant gradients from the entrance-way inward. Similarity of dust and lead loading and dust lead concentration mean value between window well/sill and hallway entrance sampling locations infers that both are primary sources of contaminant material entry into houses *per se*.[6] In fact, similarity between mean leadfall levels detected in external structures ($294 \mu g.m^{-2}.d^{-1}$) and also rooms of houses under window and door open format ($137 \mu g.m^{-2}.d^{-1}$) implies that these are both major house entry points, only modest protection being offered by houses under open conditions as compared to closed-house status ($14 \mu g.m^{-2}.d^{-1}$).

From estimates of room total floor surface area and mean room lead loading levels, total room lead loading values may be estimated, and main room comparisons conducted. Such data are summarised in Table 10.2 below.

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[5] Given the virtual order of magnitude greater dustfall/leadfall rates with windows open only 15cm, it remains to be investigated what influence leaving the window wide open fully would have. Schaeffer et al. (1972) note, from a study of more than 100 homes, that dustfall was around twice as great with open windows, although no description of "open" was provided, and concerns arise with a suspected lack of sensitivity of their chosen sampling technique.

[6] Downwind of a discharge source, high pollutant concentrations may occur on a building surface proximate to where air intakes (eg windows and doors) are located, such depositional locale enhancing the potential for subsequent house penetration (Thompson, 1993).

Table 10.2 Different room (estimated) total room mean lead loading.

	FLOOR SURFACE		ROOM FLOOR LEAD LOADING		
			MEAN	TOTAL	
ROOM	MEAN AREA (sq. metres)	PROPORTION (%)	($\mu g.m^{-2}$)	MASS (μg)	PROPORTION (%)
Bedroom	14.7	17.3	3,699	54,375	13.3
Hall	5.9	6.9	10,744	63,390	15.6
Kitchen	10.7	12.6	1,636	17,505	4.3
Lounge	18.5	21.8	4,053	74,980	18.4
Porch	5.8	6.8	15,190	88,102	21.6
TOTAL *	**85**			**407,102**	

* This incorporates the lead loading for three bedrooms per house.

Front and rear entrance-ways reflect high total floor dust loading levels, but more importantly, these two rooms reflect the greatest mean floor lead loading of all rooms. Comparison of such room mean values with those for other rooms reveals striking difference for lead loading for all room permutations for (individually) front and rear entrances, this also being the case for grouped halls and porches versus all other rooms grouped ($P \leq 0.001$). Such rooms clearly represent the more predominant entry points for contaminant static dust into the house. Combined, these two rooms' floors constitute more than one third of the total floor lead loading, with only approximately one eighth of the total house floor surface area. Given that entrance-ways do not display the presence of opening windows, it must be concluded that the predominant dust entry point is via the doorway, either when open (during former home occupancy), or possibly due to its poor sealing.[7] This conclusion must be considered in relation to the average dimension of the door aperture (0.91 x 2.10m) when open compared to that of an open window (0.76 x 0.15m), and the number and duration of opening of both. On the basis of two doors open for five hours per day five days per week (eg 22 days), and five windows open for the duration of the open study (eg 32 days), doorways represent approximately 18.8 open square metre days, and windows 18.2.

The result for rear entrances of houses is not surprising given the generally deteriorated condition of rear doors, their ill-fitting status, and the presence of substantial door to floor gaps, this being the case for many such houses in Port Pirie (pers. obs.). Again, these results derive after a period of house vacancy and closure, the inference being that during normal house occupancy, entrance doors would be more likely to be left open for considerable periods of time during the day, enhancing the potential for greater dust entry.

[7] Some of the houses had small, non-opening panes of glass approximately 15cm wide x 60cm high located either side of the front door.

Given the substantially larger (≈ order of magnitude) mean dust and lead loading levels detected in these vacant houses as compared to a suite of similarly-located occupied houses (L. Weih, pers. comm.), it would seem that leaded dust entry into these homes during the period of vacancy (under closed-house format) had contributed more significantly to detected floor contamination levels than that acquired during house occupancy. This suggests that the (prior) contribution to floor lead contamination levels by occupant activity eg tracking of shoe-adherent dust, is small by comparison with the contribution made by on-going house penetration of air-suspended contaminant dust.[8] Further, similarities between deposited and depositing contaminant dust spatial patterns, also suggest the significance of dust deposition as compared to tracking as a substantive source of house contaminant entry. This contrasts observations by some workers (eg Harrison, 1979; Cannell et al., 1987), that tracking is the major contaminant dust entry pathway. For some researchers, this is reflected by observed decreasing contaminant concentration gradients from entry point to house interior, (eg Allott et al., 1992), From these authors' work, however, it is noted that the expectation is that within-house deposition of dust would result in a fairly uniform loading of floor-level contaminant material. This was not noted for Port Pirie. Additionally, pronounced hallway contaminant gradient patterns have been detected in essentially vacant houses, this representing thus, very much a depositional phenomenon.

d) Fireplaces

The presence of an open fireplace in a room is an indicator of a more contaminated room than one devoid of such. Whilst such rooms are more contaminated, it has not been clearly demonstrated that open fireplaces *per se* are substantial thoroughfares of leaded dust entry to rooms of houses, when present in a room. No difference between mean fireplace-proximate floor contamination levels and those for floor locations on the other side of the room are apparent ($P > 0.25$), although there exists a very slight (decreasing) lead loading gradient from the fireplace to the centre of the room. Further, there are more pronounced and well-defined floor-level contaminant gradients emanating from the window and doorway extending into the centre of the room in rooms with fireplaces. These observations again infer the enhanced penetration of lead-bearing dust into such rooms in the test house suite, a fireplace presumably acting principally as an air-venting mechanism for a room. Small sample size precluded further examination as to whether this phenomenon was substantially more pronounced under (house) open or closed conditions, this representing an important area for further research.

Little relationship was noted between external air lead concentration and the indoor/outdoor air ratio, inferring that it is predominantly the relative penetrability of the dwelling to suspended leaded dust which drives I/O R variability. Although sample size was relatively small, rooms with fireplaces reflected moderately larger mean I/O R values.

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[8] From the air-sampling study, mean indoor air lead levels appeared greater for occupied versus vacant houses, as did the consequent I/O R mean value. This suggests greater influence upon mean indoor air lead levels by occupant activity within the residence, including opening windows/doors, thus enhancing the relative penetrability of leaded dust. Graziano's study of different communities in a (former) Yugoslav smelter town observed the lowest levels of blood lead in Albanian women, and concluded that that arose as a result of their habit of removing shoes prior to entering the home. Alternatively, it is also possible that this ethnic group, constituted by a majority of Moslems, demonstrated a particularly pronounced degree of personal hygiene.

10.6.3 DEPOSITING DUST

Very good correlation between depositing dust and lead connotes, again, as for static dust contamination, that the source of the dust and lead are closely related. This relationship varies only to a minor degree seasonally, winter reflecting marginally greater leadfall than summer and summer marginally greater dustfall than winter. Substantial differences in mean dustfall and leadfall rates exist between house closed and house open conditions, implying that care needs to be exercised when making inter-house comparisons, such that house status is controlled for. From earlier-work, this, as for house occupancy status, has quite obviously not been the case.

Hallways reveal numerically larger dustfall/leadfall levels than other rooms, this difference being more demonstrable under closed-house conditions ($P \leq 0.05$). Such an observation suggests better mixing of suspended materials within-house under open conditions. Analogously, from static dust contaminant patterns, hallways reflect greater contaminant levels than other rooms ($P \leq 0.005$). Thus, it seems very likely that since the houses remained closed during the vacancy period, this period was the time of acquisition of much of the deposited dust. Such dust is predominantly newly-entering dust. This conclusion is supported by the observed order of magnitude greater mean dust/lead loading levels recorded in these test houses as compared to similarly-located occupied houses (L. Weih, pers. comm.).

10.6.3.1 MECHANISMS OF DUST ENTRY

a) Windows

Window-base floor to centre of room depositing dust differences reveal greater mean values for the former for all depositing dust parameters, the former being larger and more noteworthy for dustfall/leadfall under open-house conditions ($P \leq 0.025$) than under closed ($P \leq 0.075$). This observation appears contrary to observations of more effective dust mixing in rooms under open-house conditions, for which explanation appears difficult. It may be possible that a certain proportion of the contaminant entry into rooms via windows is deposited onto the floor immediately upon room entry, it remaining there and not being re-suspended and thus mixed due to its (large) particle size. Such material may not be able to enter via a closed window, but when the window is open, it may literally fall into the collector and remain there permanently.

No association between walkway-adjacent rooms and heightened mean leadfall/dust lead concentration values in either house open or closed status was detected. Again, enhanced leaded dust house entry from the walkway through adjacent windows does not occur, or, alternatively, effective, within-house dust mixing reduces the detection of such difference with opposite-facing rooms. Given the observation of depositing dust difference between window-base floor and centre of (same) room location, (a considerably shorter distance than inter-room facing versus facing-away distance), it is unlikely that the inability to

detect a walkway-adjacent room enhanced deposition phenomenon arises as a result of good dust mixing within homes or a lack of instrument sensitivity. Rather, it would suggest that heightened surface soil lead levels in walkways is unrelated to adjacent room contaminant levels, possibly because once the contaminant dust is deposited in the walkway, it remains soil stable and bound. Consistency here with static dust patterns suggests that contaminant dust is newly-depositing subsequent to room entry, and not simply re-entrained existing dust. Additionally, such dust derives more from previously air-suspended particles rather than soil-contained surface material.

b) Main entrance doorways

From estimates of room total floor surface area and mean room leadfall levels, total room leadfall values may be estimated, and main room comparisons conducted. Such data are summarised in Table 10.3 below.[9]

Table 10.3 Different room (estimated) total room mean leadfall.

	FLOOR SURFACE		TOTAL ROOM FLOOR-COLLECTED LEADFALL	
ROOM	**MEAN AREA (sq. metres)**	**PROPORTION (%) ***	**MASS ($\mu g.d^{-1}$)**	**PROPORTION (%)**
Bedroom	14.7	21.8	855	19.3
Hall	5.9	7.4	574	12.9
Kitchen	10.7	13.5	459	10.3
Lounge	18.5	23.4	836	18.9
Porch	5.8	-	-	-
TOTAL	**79 ‡**		**4,435**	

* Porch floor surface area is not included in these data.

‡ This incorporates the leadfall for three bedrooms per house, but doesn't include leadfall in rear porches, these data not being available.

Individually, bedrooms and lounge rooms both account for approximately one fifth of the depositing lead, and constitute a similar figure for the relative house floor surface area. Hallways, however, account for around 13% of the depositing lead, yet reflect only around 7.5% of the floor surface area. Clearly the rate of deposition of lead in halls is substantially greater than for bedrooms and lounge rooms, commensurate with earlier comments regarding hallways' role as a principal dust-entry site into the test

9 For this purpose, it is assumed that the depositing dust collected in alfoil collectors in a room is representative of the overall level in that room.

houses. These results appear to complement similar observations for (static) mean room lead loading levels, although it is acknowledged that halls reveal greater difference (ie greater mean values than for other rooms) for static dust levels than for depositing dust levels. The explanation for this seems not immediately apparent, although hallways are a more likely and/or secure final resting location for deposited dust than some other rooms.

As referred to above (Section 8.5.6.1) hallways more effectively reflect inter-room dust depositional differences (greater dustfall/leadfall; $P \leq 0.05$) when houses are closed, thus (presumably) reducing the effectiveness of the within-house mixing of air-suspended dust. When doors and windows were left open during depositing dust collection rounds, it was noted that there existed only a modestly greater mean level as compared to collection rounds when only windows remained open, using the closed-house mean depositing dust level as a reference point. It would thus seem appropriate to have conducted depositing dust collection under door open/window closed format in order to assess the relative contribution to total deposition and also floor (static) lead loadings by doors and windows, and also to investigate differences between halls and main rooms in depositing dust and static dust mean levels.

In this context, however, kitchens constitute some discordance between mean room (static) lead loading levels (3.1%; proportionally small by comparison with other rooms), and mean leadfall levels (10.3%; not greatly different from other rooms). Given such dust deposition rates (Table 10.3 above), it would seem thus, that kitchens may not retain recently-deposited dust, such material being re-entrained and re-deposited elsewhere.[10]

c) Fireplaces

No strong evidence for the contribution to depositing dust levels appears to be apparent for the presence of open fireplaces in rooms. Certainly under closed-house conditions, when between-room dust mixing would appear to be less effective, fireplaces constitute a potential indicator of more contaminated rooms, this not being demonstrated to be the case under open-house conditions.

This result appears somewhat at variance with the static dust results which strongly indict fireplaces as indicators of heightened room lead loading levels. Such difference may be reconciled acknowledging an insufficiency of collector numbers in rooms, and/or possibly a collector lack of sensitivity to such (suggested) subtle influence of the presence of an open fireplace in a room. Given the substantial window-base floor to centre of room depositing-dust differences detected by this system under open-house conditions, it may in fact be that a larger number of collectors per room may be necessary to detect an open fireplace's contribution to room contamination by depositing dust in order to preclude the swamping by window-entering contaminant material. It would seem appropriate to test for such difference under closed-house conditions when within-house dust mixing is less effective.

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[10] This is presumably related to the non-carpeted nature of floors there, carpeted surfaces elsewhere tending to more securely entrap deposited dust. It would seem unlikely that the relatively low static lead loading levels in kitchens result from a poor instrument sampling efficiency, dust sampling efficiency not being demonstrated to be substantially poorer for hard surfaces as compared to soft floor coverings (Appendix Fifteen). It could be argued, however, that such low static dust/lead loading levels in kitchens may have been attributable to more effective (former) occupant cleaning regimes, although the bulk of the dust present on kitchen floors was deposited there during the house vacancy period subsequent to (former) occupant departure.

10.7 FACING-SMELTER VERSUS FACING-AWAY RELATIONSHIPS

From the work conducted here, on a number of occasions it has become apparent that certain differences in dust-associated variables exist between smelter-facing as opposed to away-facing house aspects. For example, soil lead concentrations for facing aspects of home yards return larger mean values than their away-facing analogues, this being the case also for mean soil zinc levels ($P \leq 0.003$). This is not the case for non smelter-derived soil elements such as copper and iron ($P \geq 0.50$).

Depositing dust results, however, are not quite as clear, in that whilst the mean dust lead concentration is greater in smelter-facing rooms for all collection rounds combined ($P < 0.001$; and consistently so for soil and static dust lead), the smelter-facing rooms' overall leadfall mean value is only marginally larger ($P < 0.175$). For mean dust lead concentration, facing rooms again demonstrated greater mean levels for, separately, house closed and also open conditions ($P < 0.001$). Additionally, with mean dustfall and leadfall nearly an order of magnitude larger under open-house conditions compared to closed, mean leadfall in smelter-facing rooms under open-house conditions was notably larger than in facing-away analogous rooms ($P < 0.04$).

Such observations reflect smelter-facing spatial differences, and also the reduced penetration of exogenous lead-bearing dust into houses under closed conditions, with a possible lack of sensitivity of the utilised dust-collecting system to discern spatial difference between certain rooms under lower depositing dust conditions such as occurring in a closed house. Air-sampling data reflect no apparent penetrability difference (ie I/O R difference) between house aspect ($P > 0.90$), with modestly larger mean air lead levels (both indoors and outdoors) for facing-smelter aspects ($P \leq 0.20$).

That leadfall differences between facing versus facing-away rooms may be detectable, yet this not being the case for lead in indoor air levels, implies a fairly good equilibration of air-suspended contaminant material between rooms within a house (as Leaderer and Hammond, 1991; Koutrakis and Briggs, 1992; Al-Radday et al., 1993a). Such between-room, homogeneity of air-suspended material contrasting between-room depositing dust differences as noted here and also in preliminary work conducted by others (eg Al-Radday et al., 1993a), implies that depositing dust collection as conducted here may be the more sensitive technique for discriminating between different-room contamination levels or determining spatial contamination variability.

Given greater dustfall in facing-away house aspects under closed and open conditions, with greater leadfall in facing rooms (combined and for the two districts separately), the directionality of the contaminant stream points to the smelter area as a source. The similarity of contaminant patterns for soil and static dust infers a general relationship between the two media that reflects more a similarity of contaminating source rather than necessarily implying that soil directly influences static dust. Whilst this latter alternative may, in fact, be partially the case, the absence of demonstrated walkway-adjacent room greater contaminant levels as compared to other-side-of-house rooms would tend to detract from the suggestion of a more intimate (causality) relationship between house soil area and adjacent room(s) (dust) contaminant levels. Further, as will also be seen below (Sections 10.9 & 10.10), modest association

between these two media would also tend to detract from the conclusion that home soil is directly responsible for proximate room contaminant levels.

Such observations further embellish the notion of the contemporary nature of dust deposition, particularly given the complementary observations of a smaller mean deposited-dust particle size in smelter-facing as opposed to facing-away rooms ($P < 0.05$) and greater mean values for (static) dust lead concentration and lead loading in such rooms ($P \leq 0.001$). A post-cleanup, static-dust, lead-loading mean value greater than its facing-away analogue ($P < 0.01$) for cleaned as for control houses, further supports this, and again infers that the dust collected as part of the depositing dust collection study arose from newly-entering house dust rather than re-entrained existing deposited dust.

It would thus seem unlikely that a historical pool of dust circulating around the township would produce such results, particularly given the distribution of test houses in two clusters, located at virtual right angles radially from each other, and centring on the smelter works. This conclusion would appear difficult to reconcile with earlier comments by other workers (eg Body et al., 1988b) to the effect that accumulated lead in the city dust had been found to be a significant contributor to blood lead levels but not so for present-day source emissions. In fact, arguments supporting such claims are, at best, tenuous.

Observations deriving from this dissertation do not contrast results of the work of others whose research detected no difference between various home aspects (eg Ter Haar and Aronow, 1974), their results reflecting a more diffuse source of lead, eg as deriving from deteriorating house paint.

10.8 POST-CLEANUP OBSERVATIONS

Reasonable similarity between depositing dust rates as extrapolated from the static dust level increase over the depositing dust-collection period (Round Eight - Post cleanup) and contemporaneously-collected alfoil tray-elicited data would appear to commend the relative accuracy of the depositing-dust collection technique.

There exists no evidence of difference between contemporaneously-collected cleaned house versus non-cleaned control house dustfall, leadfall and dust lead concentration mean values ($P \geq 0.60$). The cleanup process has clearly contributed little to a reduction in the rate of contamination of (tested) floor surfaces, cleaned houses actually demonstrating a greater mean dustfall and leadfall value than when investigated prior to cleanup ($P \leq 0.01$). This may, however, reflect a seasonal influence, control (non-cleaned) houses revealing similarly ($P \leq 0.025$). This difference was noted after the houses had been maintained closed, vacant and sealed for some time after the cleaning process had been completed. Such results also reveal that previously-collected depositing dust (in non-cleaned houses) derives more from newly-entering house dust than re-entrained existing dust.

The absence of any pre-cleanup post-cleanup I/O R difference further supports the claim that the suspended dust being collected from within the dwelling is newly-entering and not simply resuspended pre-existing dust. Lower mean dust variable values for depositing dust collection are observed for Round One as compared to the post-cleanup study ($P \leq 0.025$), both being house-closed, and Summer collection periods. A period of two years elapsed between the two collection periods, revealing little decline in

dustfall/leadfall over this time. Again this reflects the contemporary nature of such contaminating dust, and the continuous output from the smelter as also the contamination of homes. A larger data base here would be of considerable assistance in this regard.

House intervention may provide some immediate benefit by removing existing dust, although given that the depositing dust is that population of dust currently coating surfaces, it is presumably this dust which constitutes the potential target for a child's hands.[11] Thus depositing dust represents a threat that intervention cannot eradicate, particularly given that there appears to be little suggestion that currently depositing dust is simply precipitating, re-entrained, pre-existing deposited dust. A further inference from this is that earlier research work based upon (deposited) dust lead concentration values rather than lead loading and, more importantly, leadfall parameter values, may not have directed appropriate focus to the endeavour. Quite clearly thus, de-dusting of the houses as part of an intervention program would appear to make only a minor contribution to a reduction in the longer term risk of future occupant leaded dust exposure. This appears in accord with observations of Charney et al. (1983) regarding house intervention incorporating de-dusting. These authors report that blood lead levels decreased somewhat during the period when the "teams" were working, but subsequent to the cessation of intervention, blood lead levels then rose.

The strong relationship between mean indoor air lead level and associated (contemporaneously- and proximately-acquired) leadfall level post-cleanup ($r = 0.999$), infers a convincing relationship between these two environmental compartments with respect to source of contaminant. Additionally, it further reinforces the suggestion of novelty of in-house collected depositing dust, and the continuing nature of its house entry.

10.9 RELATIONSHIP BETWEEN VARIOUS CONTAMINANT PARAMETERS

Conducting regression of a number of house test parameter mean value permutations provided the following data (Table 10.4). Only slight association exists between static lead loading/dust lead concentration and associated mean house soil lead concentration. An inverse relationship is apparent for depositing dust lead concentration/leadfall and soil lead concentration, inferring limited association between soil and depositing dust. Such inverse relationship with soil lead concentration also exists for outdoor and indoor air lead concentration, albeit, again modestly to poorly.

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[11] It would seem that decontamination of a house would not provide any assurance that contaminant levels would not build up again quickly, given the potential for recontamination by suspended outdoor dust etc. Allott et al. (1990) working in a UK urban centre, investigated the level of radioactive caesium resulting from a single event, namely the Chernobyl incident. The group investigated levels of caesium in street dust and in house dust, and examined residence times for contaminated dust, deeming the primary source a single primary input. Given the rate of entry of contaminant dust into (occupied) houses, they concluded that any decontamination process for houses would have to be conducted throughout the entire town and on a number of occasions, in order to reduce the persistence of the pollutant in house dust. Clearly for a continuous source, the problem becomes more complicated.

Table 10.4 Correlation coefficients for a variety of test parameter comparisons.

MEAN HOUSE TEST PARAMETER REGRESSION VARIABLES		CORRELATION COEFFICIENT (r =)
DEPENDENT VARIABLE	**INDEPENDENT VARIABLE**	
Dustfall	Static Dust Loading	0.35
Static Dust Lead Concentration	Soil Lead Concentration	0.29
Static Dust Lead Loading	Soil Lead Concentration	0.21
Leadfall (Alfoil-collected)	Soil Lead Concentration	- 0.50
Leadfall (Cup-collected)	Soil Lead Concentration	- 0.49
Depositing Dust Lead Concentration	Soil Lead Concentration	- 0.15
Outdoor Air Lead Concentration	Soil Lead Concentration	- 0.07
Indoor Air Lead Concentration	Soil Lead Concentration	- 0.33
Indoor Air Lead Concentration	Static Dust Lead Loading	- 0.47
Outdoor Air Lead Concentration	Indoor Air Lead Concentration	0.63
Leadfall (Alfoil-collected)	Outdoor Air Lead Concentration	0.16
Depositing Dust Lead Concentration	Outdoor Air Lead Concentration	0.35
Leadfall (Alfoil-collected)	Indoor Air Lead Concentration	0.88
Depositing Dust Lead Concentration	Indoor Air Lead Concentration	0.89

The inference here is that air-suspended and depositing contaminant dust arise largely independently of the surrounding soil. Further, as a potential source of indoor contamination, surface soil does not really appear all that relevant, within-house depositing dust deriving from a more contemporary source. Supporting this, moderate inverse relationships are also evidenced for post-cleanup leadfall and dust lead concentration with soil lead concentration. It is acknowledged here, however, that depositing dust sample size is not large ($n \approx 10$), and also, soil and dustfall sampling was not conducted contemporaneously.

Such results contrast work of Roberts et al. (1990) who report that soil lead concentration explains 45% of the variation in dust lead loading within the house, soil track-in being suggested to be a major source of lead present in house dust. This Port Pirie research also contrasts work by Hertzman et al. (1990) as part of the Trail Lead Study, where soil was claimed to constitute an important determinant of house dust lead level. For these (occupied) houses, correlations between floor and window contaminant levels were weak, such correlation from this Port Pirie work reflecting a much more notable association ($r = 0.79$). Davies et al. (1985), however, working in a former lead mining area, observed that the greater part of dust lead was not derived from exterior soil, but from alternative sources, soil lead explaining approximately 27% of dust lead variability. This figure for the Port Pirie study is around nine percent.[12]

[12] Given the period of vacancy of the test house suite, and noted depositing-dust rates during that period, such vacancy is suggested to influence the relationship between soil lead concentration and (static) lead loading levels for occupied houses, where occupant activity may contribute to modification of such a relationship.

Contaminant entry pathway differences are obviously inferred by these various studies. It should be recognised, however, that no other point/diffuse sources of contaminant distribution to various environmental compartments was noted by the work of Roberts et al. (1990, 1991a & b), such a contaminant deriving virtually exclusively from pre-existing soil-contained material. Somewhat analogously, lower mean (external) air lead levels detected as part of the Trail study (White and Hilts, 1995) compared to Port Pirie (var. Goh and Hope) may explain the more substantive relationship between soil lead and house dust lead level for the former case. Roberts et al.'s demonstrated good correlation between soil lead and carpet lead loading and strong negative correlation with shoe removal implies that tracking is a substantial contributor to indoor floor lead levels in their study (1991a & b). Harrison (1979) also reports that tracking represents a major contributor to house dust entry.

It is likely, however, that in areas where atmospheric deposition is a major concern (eg as reflected by elevated air contaminant and leadfall levels as experienced proximate to a smelter), various different mechanisms of contaminant entry into homes are at work, tracking presumably assuming a more minor role. Given the unoccupied nature of the Port Pirie test house suite, such a conclusion for these houses is not surprising. Further, this could be construed to support the concept that the bulk of the vacuum pump-acquired (static) dust deposited on floor surfaces collected there during the period of house vacancy. Given rates of contamination of cleaned, closed houses by depositing dust, and open-house/closed-house depositional differences, depositing dust predominates as a within-house contamination mechanism. A study investigating dust deposition rates conducted contemporaneously in vacant and occupied houses would appear clearly warranted.

These data convincingly support the concept of the contemporary nature of the depositing dust within rooms of homes. The inverse relationship between indoor air lead concentration and static dust lead loading would argue against suspended leaded dust particles arising simply from re-entrained static dust material. This fits well with the results of the post-cleanup study, reflecting greater mean depositing dust/leadfall rates in post-cleaning houses compared to their pre-cleaned condition.

Outdoor air lead levels remain a reasonable predictor of indoor air lead levels, in vacant houses, explaining 40% of the variability, a figure of 39% applying for houses subsequent to house cleanup. This cleanup of the houses also involved repairs to cracks in walls, ceilings, cornices, architraves and window frames. Clearly, this had minimal influence on the relative penetrability of lead-bearing dust into the domicile, and further, pre-existing deposited floor dust in vacant houses has little influence on indoor air lead levels. This is in accord with comments by Koutrakis et al. (1991) who note, in work pertaining principally to occupied homes, that only a small proportion (perhaps ten percent) of indoor air-suspended material arises by re-suspension of pre-existing settled dust. Davies et al. (1987c & 1990) observe that around 70% of the variability in indoor air lead level is explained by outdoor air lead levels, these data also deriving from a suite of occupied houses. Their inference is that such better correlation arises presumably as a result of more frequently open windows/doors in occupied houses, and more consistent penetration of contaminant dust into homes. Quite obviously indoor air lead levels are a very good determinant for both depositing dust lead concentration and leadfall, such a result being commensurate with the results of the post-cleanup depositing dust study (where r for leadfall against internal air lead concentration = 0.999).

Poor correlation between existing individual home contaminant levels for static dust and depositing dust would suggest very little relationship between the two. Essentially, this strengthens the argument favouring the source of depositing dust at floor level as deriving from novel sources and not arising from the re-suspension of pre-existing deposited dust. Substantial evidence has been presented supporting the claim that the contaminant material entering test houses is predominantly air-suspended prior to house entry, subsequent deposition occurring onto horizontal surfaces. This includes:

a) noted very good correlation between outdoor air and indoor air-suspended contaminant material level,

b) little variability in the relative penetrability of air-suspended leaded material into houses (pre and post cleanup), and

c) excellent correlation between indoor air-suspended leaded material and internal leadfall rates.

Under closed-house conditions, the mixing of house internal air masses would appear less effective, depositional patterns demonstrating more subtle spatial difference than under open-house conditions with more substantial house (internal) air movement.

10.10 LEAD/ZINC RATIO COMMENTS

From Table 10.4 above it is apparent that surface soil contaminant levels are not particularly significantly related to static dust contaminant levels. This is also the case for soil and air-suspended contaminant levels. In the context of similarity of source for these dust samples, lead/zinc ratios are examined. A summary of mean lead/zinc ratios for soil, static dust and air-suspended material is indicated in the table below (Table 10.5).

Table 10.5 Mean lead/zinc ratios for various sampled media.

GEOMETRIC MEAN LEAD/ZINC RATIO		
SOIL	**STATIC DUST**	**AIR**
0.54	0.62	Inside = 1.17 Outside = 1.12

Air-suspended material reveals a greater proportion of lead than either of the other two media, both of which suggest commonality of contaminant source. Comparison of sampled media is conducted below.

10.10.1 SOIL AND STATIC DUST

Mean home lead/zinc ratios for soil and static dust are reflected in the following table (Table 10.6).

Table 10.6 Mean home lead/zinc ratios for soil and static dust.

HOME	GEOMETRIC MEAN LEAD/ZINC RATIO	
	SOIL	STATIC DUST
6 Agnes St	0.47	0.56
64 Edward St	0.51	0.66
58 Federation Rd	0.56	0.64
29 Fourth St	0.47	0.57
19 Lily St	0.59	0.69
3 Sabine St	0.53	0.59
34 The Terrace	0.64	0.62
27 Vera St	0.62	0.72
136 Warnertown Rd	0.56	0.61
97 York Rd	0.42	0.39

The moderate variability in these mean home ratios reflects the widespread and moderately homogeneous nature of penetration of these contaminants throughout the two districts, there existing no apparent difference between them for soil lead/zinc ratio ($P > 0.50$) and dust lead/zinc ratio ($P > 0.70$). Regressing the mean home lead/zinc ratio for static dust against that for (home) surface soils reveals a correlation coefficient (r) of 0.78 ($P < 0.0085$). It could be concluded from these data that surface soil is a reasonable contributor to static dust in that soil explains around 60% of the variability in static dust lead/zinc ratio. Other data from this study, however, indicate that there exists only a slight relationship between surface soil and house indoor floor dust, soil explaining less than nine percent of the variability in static dust lead concentration, and less than five percent of the variability in dust lead loading.

Such results infer that similar contaminant dust deposits onto soil surface and also house (inside) floors, they deriving from, presumably, similar sources. The soil surface, however, would not appear to be the main, or even a substantial contributor to house floor dust. This conclusion reinforces the concept of the contemporary nature of depositing dust in Port Pirie. Essentially, contaminant dusts arise from current emissions rather than an existing historic pool of contaminant dust located at ground level, and, once deposited on surface soil, dust is fairly stably located there.

10.10.2 INDOOR AND OUTDOOR AIR

Comparison of lead/zinc concentration ratios for sampled air-suspended material implies strong similarity for indoor and outdoor alternatives ($P > 0.65$). This is also the case for (separate) pre- and post-cleanup comparison ($P \geq 0.60$). Such similarity is unaffected by house occupancy, no difference being apparent for either vacant or occupied category, this also being the case pre- or post-cleanup ($P \geq 0.50$). Comparison of (separately) indoor and outdoor air-suspended material for occupied versus vacant homes again reveals little difference ($P \geq 0.35$). Separate comparison of lead/zinc ratio corresponding to indoor and outdoor air-suspended material for homes both prior to and post house cleanup also demonstrates similarly ($P \geq 0.25$). House occupancy exerts little influence on the constitution of air-suspended, outdoor-sourced contaminant material, similarity between external and internal air-suspended material implying commonality of source.

Further, indoor to outdoor air-suspended dust lead/zinc ratio does not differ ($P > 0.70$), regressing the lead/zinc ratio for inside air-suspended material (dependent variable) against outside air elicited a correlation coefficient (r) of 0.91, ($P < 0.001$). Similarly regressing the lead/zinc ratio for inside air-suspended material (dependent variable) against the corresponding ratio for static floor dust reveals a correlation coefficient of 0.25 ($P < 0.035$). The dimension of these two coefficients reinforces the premise that the source of indoor air-suspended particulate material relates strongly to outdoor air-suspended material, internal sources (eg re-suspension of already-deposited dust) making little contribution.

Occupied homes exhibit greater indoor air lead levels compared to vacant houses ($P < 0.0075$). It thus seems likely that occupant activity is responsible for enhancing the air-suspended indoor contaminant levels in houses. This would arise by way of their allowing greater quantities of such material to enter the domicile from outside (perhaps by way of maintaining doors/windows open) rather than simply being responsible for re-suspending more already-deposited material such as floor-settled dust as a result of their activity, and/or simply tracking in new material. This is in keeping with the inverse, moderate correlation between lead concentration of indoor air and static dust ($r = 0.47$).

CHAPTER ELEVEN CONCLUSION AND RECOMMENDATIONS

One of the proofs of the immortality of the soul is that myriads have believed it - they also believed the world was flat.

Mark Twain

They laughed at Columbus, they laughed at Fulton, they laughed at the Wright Brothers. But they also laughed at Bozo the Clown.

Carl Sagan

11.1 CONCLUSION

Despite a lead abatement program operating in Port Pirie for more than ten years, and absorbing more than thirty million dollars (Aust.), some areas of the township still exhibit substantial and unacceptably high levels of environmental lead contamination of surface soil, dust, and air. Represented by the test house suite, such areas reflect lead levels in, particularly, house dust that are considerably above threshold levels for the demarcation of contaminated locations. Surface soil and dust lead levels in the home environment in Port Pirie West and Solomontown reveal that, given a young child's normal mouthing activity, there exists the potential for children resident there to succumb to lead intoxication and suffer irreversible CNS damage during their developing years, principally due to contact with lead-bearing dust within the home. Lead-in-air levels within tested Port Pirie homes appear more moderate relative to some other contaminated area study values, particularly when compared to the substantially greater than background dust and surface soil lead levels recorded for these Port Pirie houses. This reflects the considerable dimension of the lead "sink" within homes in Port Pirie.

A simple and effective method for the estimation of the level of child-available lead-containing dust on surfaces has been demonstrated and validated. Additionally, a novel system for the collection of depositing dust has been developed and successfully compared to an existing technique. The new technique is simple, consistent, and sufficiently sensitive and effective for depositing dust collection purposes. Further, it provides leadfall and dust lead concentration data, with consequent advantage for future use, in either vacant or occupied houses. Analogously, improvements to an existing air-sampling technique have also been proffered, providing enhanced accuracy and precision over existing techniques.

The standardisation of dust-collecting techniques at residence level is essential for meaningful comparison to be made between different locations from different studies. The choice of surface dust sampling equipment utilised for this work appears well-suited to the task, given the particle size spectrum of collected dust, and its similarity to the expected child hand-adherent dust particle size spectrum ($< 100\mu$). The majority of the dust particle-contained lead collected by the sampling system also lies in the

less than 100 micron particle-size range, with a decreasing dust particle size corresponding to an increasing lead content. This specialised system is certainly more representative of the collection of that sub-sample of house dust most likely to contribute to a child's subsequent intake than are domestic vacuum cleaner-derived samples. Further, the facility of producing lead loading data from the sampling technique is of advantage, in that dust lead concentration data alone are insufficiently sensitive an index of room contamination variability, particularly if spatial variability is of concern. Contaminant dust populations collected by the techniques developed for this study thus effectively represent that population of surface dust available to a child via the hand-to-mouth route, and contributing to body lead burden.

Extensive investigation has revealed that the source of lead in the home environment is closely related to the source of accumulating dust. The source of leaded within-house dust does not derive to any great extent as a result of the degrading of painted inner surfaces of the house. Externally, the calibre of lead-contaminated surface soil is most pronounced close to the outer walls of the dwelling, this presumably arising as a result of the accumulation of depositing contaminant dust. Reasonably-protected areas of the residence such as narrow walkways adjacent to side boundary fences, represent the most contaminated home area.

Preferential accumulation of contaminant dust in aspects of the house facing the smelter, as opposed to aspects facing away, reflects the continuous deposition of lead-bearing dust and points to the general smelter area as the source. Further, such observations infer both the contemporary and on-going nature of emissions from the source and their influence on homes. This confronts previous suggestions of an historical pool of contaminant dust deriving from earlier smelter practice relentlessly circulating around the township and contaminating houses. In an analogous sense, observed mean air and blood lead levels in the USA over the period mid 1970s to early 1980s clearly reflect the decline in the use, and emission as aerosol, of lead deriving from leaded motor spirit (Figures 3.5 & 3.6). Essentially, the currency of lead emissions and air and blood lead levels appear intimately related; *pari passu* once a reduction in emissions occurs, this is reflected by environmental contaminant level decreases. Given relatively unchanged lead-in-air levels assessed in the township of Port Pirie over a number of years, the on-going nature of emissions warrants attention. This also reflects upon the relative stability of precipitated leaded aerosol at ground level.

Limited direct association between external surface soil lead contamination and internal house depositing leaded dust has been demonstrated here. This investigation, however, strongly links surface soil lead data with the internal static dust and dustfall data in the context of the general source from which the contamination of both these environmental compartments derives. The suggestion here, thus, is that the contamination of surface soil occurs, in a large part, similarly to internal house dust contamination. Essentially, this similarity of contamination pattern arises as a result of the current movement of contaminant particle suspension emanating from recent source emissions assisted by (presumably) wind activity. This accumulation of lead-bearing dust is a current deposition phenomenon, given the directionality of home contamination patterns.

It is not likely that surface soil contamination arises due to former occupant home soil modification activity such as soil replacement or augmentation with smelter by-products as has previously been suggested by other researchers. Such a conclusion supports the concept of the relative (physical) stability of leaded dust in/on surface soils, and the on-going nature of contaminant dispersion to the environment

from the source. This is slightly different to the conclusions of several workers (eg Clark et al. (1991) and Bornschein et al. (1991), who suggest a more direct lead pathway from soil to air to house dust and thence to children's hands and eventually their blood, ie as driven by soil lead levels.

Dust penetration into the dwelling appears to be principally via windows and outer doorways in unoccupied houses, the latter representing the more substantial entry pathway. Reflecting this, hallways and rear entrance ways are the most heavily-contaminated rooms of the house. A virtual order of magnitude greater dust/lead deposition rate in open houses, as compared to closed houses, reflects enhanced health risks for resident children, and also the on-going dust entry from outside a house inside via doors and windows. Thus, controlling for house open/closed status when making between-house depositing dust data comparison is imperative. This has not been previously addressed in earlier studies.

Penetration of leaded dust into occupied homes is suggested to also occur predominantly via windows and doorways. Even for such houses, within-house dust-lead contamination via foot-tracking is not expected to be a major contributor compared to windows and doors. This is particularly so given substantially greater depositing dust levels under open-house conditions, and typical, occupied Port Pirie houses' normally "open" format.

No real evidence is apparent that above-ceiling space-contained dust is a substantial contributor to within-room dust contamination, although particle size spectrum similarities between above-ceiling space and floor-collected dust in houses subsequent to a period of vacancy suggest similarity of contamination process and pattern. Whilst rooms containing an open fireplace reflect greater mean contaminant levels than those without, no evidence has been presented suggesting that the fireplace/chimney is the means by which such extra lead-bearing dust enters the room. Rather, it would seem more likely that the fireplace facilitates greater aerosol penetration into the room via doorways and windows, and more enhanced air movement through the room with greater associated deposition of suspended dust on horizontal surfaces.

A close relationship exists between lead and zinc in surface soil and settled dust, implying that the number of different initial sources of home contamination by these elements is not large. Such consistency of elemental association in surface soil with district, home area or by individual home reflects the intimate relationship between these two elements at home location, and also, presumably, in the context of contaminant source(s). Their association further supports the theory of a pool of contaminants currently emanating from a source(s) and spreading throughout the township on an ongoing basis.

These data do confirm a close relationship between (home) surface soil-detected lead and zinc with respect to possible source. They do not, however, contribute to a closer understanding of the actual source(s) of the contaminant material. This clearly provides avenue for substantial future research, given demonstrated levels of house dust-lead contamination, and dust-lead re-contamination rates in cleaned houses. Constituting work to be presented elsewhere, and as part of an on-going research project by the author, this has already been undertaken by examining house dust using electron microscopy and individual dust particle-characterising techniques. Here, various house dust samples were classified according to a classification scheme based upon smelter-derived source materials such as ores/concentrates, and various intermediate and final products of the smelter's operation.

Rates of deposition of contaminant dust on floor surfaces subsequent to a substantial house de-dusting process are unaffected by the house clean-up. Control house comparison, and pre-cleanup same-house comparison reveal very similar rates of leaded dust deposition, and also relative house dust penetrability.

The outdoor air lead level represents a reasonable predictor of indoor air level in vacant houses explaining forty percent of the variability, this also being the case for vacant houses subsequent to house cleanup. The dust depositing on horizontal surfaces within houses is thus newly-entering, and not related to the precipitation of re-entrained pre-existing dust. This complements noted differences in within-house dust-lead contamination between house aspects facing the smelter and those facing away.

Given demonstrated, cleaned-house re-contamination rates, the relative merit of a house de-dusting style intervention strategy in eradicating the on-going threat that depositing contaminated dust represents, would thus appear to be questionable in the context of providing a reduction in lead exposure risk.

Some seasonal trends in depositing dust appear, winter months representing a period of greater leadfall than for summer months. This may reflect a greater dilution by other non lead-bearing dusts during the drier and dustier summer months. A larger sample size over perhaps a shorter time period, might more conclusively demonstrate real seasonal differences in dust-lead deposition rates in houses in Port Pirie.

Differences are noted between recorded levels for lead-in-air within/immediately outside of houses and existing high-volume air sampler results deriving from Port Pirie. High-volume sampling in a limited form, as is conducted in the township of Port Pirie currently, is thus not sufficiently representative of contaminated aerosol at home level. This reflects the need to further conduct lead-in-air assessment at home level in Port Pirie as part of an overall home contamination investigation.

11.2 RECOMMENDATIONS

The movement of contaminant dust toward Port Pirie houses appears to be driven by aeolian activity, and is an on-going process. At the home level, this manifests as lead contamination gradients radiating from the direction of the smelter for surface soil and within-house dust. In order to reduce such dust entry into the house, some form of barrier to dust movement within the home environment appears desirable. Trees and shrubs act as efficient screens for lead aerosol, which collects on the leaves and bark, the amount being collected being a function of the surface area of the leaves (as Fergusson et al., 1980). A tree-planting strategy already exists as part of current lead-intervention program for Port Pirie. Strong encouragement should also be directed to the planting of trees and shrubs at the domestic level. Whilst it has not been demonstrated that there is a direct linkage between contaminant levels in bare areas of soil at home level and the penetration of lead-bearing dusts into residences via proximate entrance ways, attempts to reduce the potential mobility of surface soil material by covering with surface grasses would appear appropriately provident.

Given that the walkway area typically exhibited limited vegetation or was devoid of vegetation, it seems likely that such an area could be particularly sinister re the potential for child lead contact, since contaminant levels were higher here than elsewhere. Additionally, the availability of contaminant material from this area would appear to be enhanced given the increased likelihood of contact with loose surface soil/dust particles here due to this absence of vegetation. In a precautionary sense, steps could be taken to reduce the risk of lead uptake by children by reducing children's activity in areas such as walkways.

Further, attempts could also be made to re-vegetate such an area, particularly acknowledging the proximity of the walkway to windows of rooms such as the lounge room, kitchen and child's bedroom, and thus the potential for (say) lead-bearing dusts to enter the domicile. Heightened surface soil lead levels in walkways appear unrelated to adjacent room contaminated dust levels, possibly because once the contaminant dust is deposited in the walkway, it remains bound and reasonably soil stable. Alternatively, if it did exist, such a phenomenon may be somewhat masked by good within-house dust mixing. Either way, reducing the potential for contaminant entry from particularly walkways to the house via proximate windows would seem an appropriate house contamination-management implication.

In the context of some degree of reduction in potential soil contaminant level being afforded by that aspect of the home facing away from the smelter, the closer boundary could be re-vegetated with elevated closely-sited trees to act as a wind-break. Further, attention could be paid to ensure that little, if any, bare soil remains nearby. Again, carefully selecting the site for children's activity would be presumed to be beneficial re minimising direct contaminated soil uptake.

Further, given substantially greater within-house dust deposition under open-house conditions compared to closed, consideration should be given toward reducing the frequency and duration of such open-house conditions. As part of management strategy for houses within the two high-risk districts, the air-conditioning of houses would reduce the likelihood of dust entry by encouraging windows/doors to be kept closed more frequently, and also slightly positively pressurising rooms, thus reducing the likely penetration of contaminant dust (as Yocom, 1982). This would also provide some filtering of air entering the residence, force-filtered room air ventilation exhibiting an 80% reduction in suspended particle matter (Sterling and Kobayashi, 1977).

Given demonstrated surface dust lead levels, rates of lead-bearing dust deposition, and rates of recontamination of cleaned homes implying the on-going and current stream of contamination radiating from the smelter, re-location of children displaying continuously and substantially-elevated blood lead levels must be considered. This is supported by the less than impressive blood lead level improvements exhibited by children living in "decontaminated" Port Pirie houses. Some analogy may exist between this situation and that for child residents of a number of home intervention programs in the USA where domestic paint de-leading programs were undertaken with only modest result.

Acknowledging that window-base floors, main-room doorways and hallways are areas of high leaded dust level, consideration should be given to restricting child activity in these regions. A child's room should provide the facility of being able to be effectively closed off from the outside via well-sealing, closable windows. Air-conditioning of the room would be of assistance here. It would seem desirable to select such a room to be devoid of a fireplace, and to restrict air movement in the room by keeping doors and windows closed for as substantial a period of time as is possible. Floor surfaces would ideally be wooden or linoleum, thus reducing the potential sink of contaminant dust within that room. Floors should be regularly cleaned.

Further studies conducted in both vacant and occupied houses with a larger sample size, particularly for Solomontown, are clearly warranted. Such endeavour should also be directed to investigating differences between levels and rates of contamination in homes of differing construction type and materials, and in varying degrees of house openness. Further research should also focus on the estimation of re-

contamination rates in cleaned houses in both open and closed formats, both closer to and further away from the smelter.

Clearly, Port Pirie remains a site of heavy contamination by lead at domestic level, despite more than two decades' awareness of the health concerns associated with undue lead exposure, and some comprehension of the degree of contamination of some homes. The process of home lead contamination is a current phenomenon, and cannot be ascribed to the relative vagaries of previous smelter practice. Much more comprehensive home lead level investigation is warranted than is currently being undertaken, including within-house air lead levels.

If the goal of protecting human health is paramount, it will not be adequately served by the application of outdoor air quality standards applying to samplers located in a few commercial locations. Equally, purportedly-proffered maximum environmental compartment contaminant levels which, however, are not necessarily adhered to or policed will also not benefit the public good. Further, the ignoring of published health-risk threshold levels for lead contamination of certain environmental compartments at particularly domestic level, will also not reduce the prospects of further permanent damage to the health of that component of the population that least deserves it, that is most susceptible to it, and that can do the least to avoid it.

For the sake of the children, the investigation of the health threat of residence in Port Pirie must deserve even more attention.

The unfortunate one is he who averts his head from this door. For he will not find another door.

Saadi of Shiraz (1193 - 1291)

APPENDICES

APPENDIX ONE ANALYTICAL PROCEDURES

A1.1 PREPARATION

Prior to commencement of analysis, benches were washed/wiped with clean moist cloth, followed by wet paper towel, the wetting agent being 0.5M nitric acid (HNO_3). Subsequent to this wash down of benches, clean bench coat was placed upon working surfaces. Prior to use, all glassware was soaked in 1.5M HNO_3 for a minimum of 48 hours, as Laxen and Harrison (1981). Analysis of cassette-contained material was conducted within (at most) three months of collection. Que Hee et al. (1985) note that such cassette-contained dust is very stable, and may be stored, unchanged, for many months prior to analysis.

Prior to the commencement of sample analysis, all glassware to be used was initially soaked in 1.5M HNO_3 for 48 hours, and subsequently washed three times with pure water.

A1.2 ANALYSIS

Typically, preliminary lead in dust concentrations (ie multiple replicate analyses for standard reference materials) were initially assessed using (flame) AAS prior to ICP AES, these being the two most commonly used and appropriate technologies for such purposes (Binstock et al., 1995; Pella et al., 1995).

A comprehensive QA/QC procedure was instigated throughout analysis incorporating procedural (reagent) blanks, field blanks, standard reference dust samples, an internationally-certified test dust and standard reference solutions, details of which are included in Appendix Three.

A1.2.1 CASSETTES

In a clean, dust-free, draught-free environment, cassettes were carefully opened and the entire contents of the cassette including the filter (but not the filter backing) emptied into a 40ml Pyrex test tube digestion vessel. Care was exercised to ensure minimal dust loss upon cassette opening and contents transferral to glassware. Stainless steel forceps were used to facilitate filter removal without excessive contamination or sample loss due to inappropriate handling. Care was needed here in order to ensure all dust including that

adherent to the cassette (internal) walls was transferred to the digestion vessel. To this end, the entire inside surface area of both halves of the cassette (including the inlet "port") was carefully and copiously rinsed using a fine-tipped plastic "squeezy" bottle containing dilute (5% or ≈ 0.75M) HNO_3 in pure water, and the contents carefully deposited into the digest vessel. This was important in order to avoid the potential for loss of fugitive dust adherent to the cassette (inner) surface. The (inner) neck of the digest tube was washed with the contents of the squeezy bottle to ensure that all dust adherent to the inner walls of the vessel was washed down to the base of the tube.

The digest vessel and contents were placed on a constant temperature heating block, and heated to a temperature of approximately 90^0C and maintained at that temperature until the volume of acid/dust mix in the tube had reduced to approximately one to two millilitres (mls), at which time the tube was removed from the heating block, the contents allowed to cool, following which, three mls conc. Nitric acid was added.[1] The sample was then carefully vortexed to ensure thorough mixing, and the contents brought to 130 - 140^0C for five to ten minutes and then allowed to cool to room temperature, at which time five mls conc. Hydrochloric acid was added, and the mix again gently vortexed.[2] The temperature was then carefully brought to 130 - 140^0C for an hour, subsequent to which time the temperature of the vessel was increased to 170 - 180^0C for approximately five minutes. Subsequent to this, tubes were removed from the heating block and allowed to cool. Internal "spike" standard (AR Grade Boric Acid/Spec. Pure Ruthenium Chloride) was then added to each tube, the internal walls and neck of the beaker were again washed to the base of the tube using dilute HNO_3 from a squeezy bottle, and the volume in the tube then being brought to 20mls with dilute HNO_3 in order to set the lead present in solution. After allowing solutions to stand for approximately an hour, ICP AES was performed.

A1.2.2 STANDARD DUST SAMPLES

On all occasions where standard reference dust samples were employed, 100mg of the material was gravimetrically determined to an accuracy of 0.1mg directly into the digest vessel, typically a clean dry ten millilitre Pyrex test tube. The addition by stainless steel spatula of 100mg of the dust standard to the digest vessel was by a number of separate grab sample aliquot additions of dust from different regions of the body (initially 50gm) of the (carefully-mixed) bulk dust in order to maximise aliquot dust homogeneity thus minimising the potential for unrepresentative (standard) dust sub-sampling.

From this point, the subsequent digest procedure for standard dust is essentially analogous to that conducted for cassette-contained dust.

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[1] In order to avoid excessive effervescing of the acid when added, the temperature of the contents of the digest vessel were allowed to cool prior to the addition of the digesting acid.

[2] In order to ensure that no reaction between the two acids occurs initially if they are added contemporaneously, separate addition of acids was practised, thereby enhancing the likelihood of optimising the solubilising of all the materials present in the contents of the beaker.

APPENDIX TWO PRELIMINARY DUST DIGEST METHOD ESTABLISHMENT

A2.1 INFLUENCE OF CHOSEN ACID DIGEST MIX

The literature indicates a considerable number of different concentrated acid digest mixes utilised for house dust digestion (Que Hee et al., 1985). It thus appeared appropriate to test a number of acid digest alternatives common to the literature (Tinsley et al., 1983), and ascertain if any efficiency differences exist between them for the anticipated house dust samples to be provided for this research work.

In most situations, the literature indicates (concentrated) nitric acid as the digest acid of choice singly (eg Anon., 1974; Solomon and Hartford, 1976; Anon., 1977; Sayre, 1977; Yaffe et al., 1983; Davies et al., 1985; Sturges and Harrison, 1985). Alternatively, concentrated hydrochloric acid may be used (as Rice et al., 1978; Sayre and Katzel, 1979; Diemel et al., 1981; Tinsley et al., 1983) or both acids used in combination (as Duggan and Williams, 1977; Harrison, 1979, although essentially dealing with street dust only; Tinsley et al., 1983). Harrison (1979) reports that the combination of nitric and hydrochloric acids is highly efficient, eliciting results close to that of nitric and hydrofluoric acid extraction without the associated technical problems.[3]

From a comprehensive review of the literature, the following acid digest alternatives were tested for differences in resultant dust lead concentration levels for three sieved sub-32μ test dust standards.[4] In all cases, 100mg of dust was digested with five mls of the appropriate acid mix in ten millilitre pyrex test tubes. When two acids were used, the nitric acid was routinely added first, and when effervescence had subsided, the aliquot of hydrochloric acid was subsequently added.

Selected alternative digest media

a) Hydrochloric acid
b) 1:4 Nitric Hydrochloric Mix
c) 2:3 Nitric Hydrochloric Mix
d) 3:2 Nitric Hydrochloric Mix
e) 4:1 Nitric Hydrochloric Mix
f) Nitric acid

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[3] From preliminary analytical work conducted by an outside laboratory on the standard (sieved) floor dust samples used for QA/QC here, and according to the defined protocol indicated below (ie as an initial inter-laboratory comparison), no difference in dust lead concentration results from multiple analyses between nitric, nitric/sulphuric, nitric/hydrochloric and nitric/hydrofluoric acid extractant alternatives was demonstrated ($P \leq 0.025$). Supporting this, Tinsley et al. (1983) note that they have detected no difference ($P \leq 0.05$) in elicited (sieved) dust lead concentrations between nitric acid, hydrochloric acid and various combinations thereof. No such work had previously been conducted for dust derived from Port Pirie houses.

[4] Such a sieved sample of dust was chosen in order to reduce dust heterogeneity on the basis of particle size, and (presumed) separate sub-sample lead content variability.

Three different (sub-32μ) sieved dust samples were digested with each of these acid digest alternatives, and twenty replicates of each were conducted.

A2.1.1 METHOD

One hundred milligrams of the appropriate dust was added to a clean dry ten millilitre glass test tube, and five mls of the appropriate acid added. The performed digest procedure was conducted in these tubes, and is referred to in Appendix One. AAS analysis of solutions followed.

Statistical comparison (*t*-test) of results of the six different acid digests for the three different dust samples (samples one, two and three - each batches of twenty replicates) for average lead-in-dust concentrations of 1,050μg.g^{-1}, 2,910μg.g^{-1} and 9,580μg.g^{-1} revealed no statistical difference between the acid digest results for any of the acids used ($P \leq 0.05$), in keeping with the results of Long et al. (1979) and Que Hee et al. (1985).

A2.1.2 RESULTS

A summary of the results appears in Table A2.1. An inspection of the RSDs here would suggest that there was not any one digest choice that introduced substantially more variability than any other method, mean RSDs being quite low (J. Waters pers. comm.). Such low and consistent RSD values are not typical of all previously-published work, and reflect the relative homogeneity of the dust samples, few previous workers having chosen particle size-fractionated dust samples as test dust samples as has been the case here.

Here, as Tinsley et al. (1983), very low procedural blank values were demonstrated for all tested digest solutions. From these data, it was decided to adopt the digest mix two parts concentrated Nitric acid to three parts concentrated Hydrochloric acid as the standard acid digest mix for all digests constituting this entire piece of research, such an approach having been exampled by a number of workers (Tinsley et al. 1983; Que Hee et al., 1985; Laxen et al., 1987a; J. Wattling, pers. comm.).

Table A2.1 Influence of acid digest mix used for assay on analytical result.

ACID MIX USED	MEAN LEAD IN TEST DUST CONCENTRATION VALUE ($\mu g.g^{-1}$ and %)						
	TEST DUST ONE		TEST DUST TWO		TEST DUST THREE		
	Mean lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	Mean lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	Mean lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	MEAN RSD (%)
HCl	1,038	2.4	2,903	1.7	9,453	2.8	2.3
	(24.4)		(49.7)		(260.3)		
1:4 Nitric/HCl	1,041	2.1	2,903	1.9	9,460	2.8	2.3
	(21.8)		(55.9)		(265.4)		
2:3 Nitric/HCl	1,047	2.2	2,915	2.2	9,451	2.8	2.4
	(22.6)		(63.5)		(265.1)		
3:2 Nitric/HCl	1,044	3.0	2,908	1.7	9,480	2.9	2.5
	(30.8)		(49.0)		(276.5)		
4:1 Nitric/HCl	1,043	2.5	2,908	1.8	9,516	2.7	2.3
	(26.2)		(51.6)		(255.8)		
Nitric	1,040	2.3	2,908	1.3	9,408	3.0	2.3
	(24.3)		(37.4)		(282.1)		
MEAN	**1,042**	**2.4** *	**2,472**	**1.8**	**9,461**	**2.8**	**2.3**

* These RSD values compare favourably with other RSD values where the mean value approximates four percent, eg as elicited by the Quality Control procedures demonstrated below (Appendix Three).

A2.2 INFLUENCE OF QUANTITY OF DUST DIGESTED

Having selected this acid mix for digestion, the influence of varying the quantity of dust to be digested in five mls of the chosen acid digest mix on AAS-estimated dust-lead concentration levels was investigated. Dust quantity sets consisting of ten replicate samples of 50mg, 100mg, 200mg, 300mg, and 500mg of the three standard test dust samples as shown above (all previously sieved ≤ 32μ size fractions) were utilised. Samples were digested in a manner identical to those above (A2.1.1), using the standard digest mix, with lead-in-dust concentration analyses performed subsequently.

Again, comparing the results of these analyses, for each dust tested, no difference was demonstrated between any of the five dust quantity sets analysed for each of the three dust samples tested ($P \geq 0.05$). A summary of the results appears in Table A2.2 below.

Table A2.2 Influence of test dust quantity used for analysis on analytical result.

	MEAN LEAD IN TEST DUST CONCENTRATION VALUE ($\mu g.g^{-1}$ and %)						
	TEST DUST ONE		TEST DUST TWO		TEST DUST THREE		
QUANTITY OF DUST	Mean Lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	Mean Lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	Mean Lead conc. and SD ($\mu g.g^{-1}$)	RSD (%)	MEAN RSD (%)
50mg	1,052	2.3	2,915	1.4	9,446	3.0	2.2
	(24.5)		(39.4)		(284.4)		
100mg	1,048	3.1	2,918	1.6	9,426	2.7	2.5
	(32.7)		(45.7)		(258.4)		
200mg	1,060	2.9	2,921	1.2	9,441	2.7	2.3
	(30.9)		(34.2)		(258.9)		
300mg	1,041	3.4	2,935	1.3	9,296	3.7	2.8
	(35.6)		(37.1)		(342.2)		
500mg	1,048	3.8	2,924	1.6	9,272	3.5	3.0
	(40.0)		(46.7)		(326.0)		
MEAN	**1,041**	**3.1**	**2,923**	**1.4**	**9,376**	**3.1**	**2.6**

Further, when comparing the dust lead concentration results for the three test dust samples arising from the above two (separate) sets of experiments (A2.1 and A2.2), no difference was detected between any of the concentration data sets (30 permutations) corresponding to individual (same) dust sets ($P \geq 0.05$). When all concentration readings elicited by the digest mix experiments (A2.1) for each individual dust were grouped, and compared to grouped experiment data for the respective test dust (A2.2), again there appeared to be no apparent difference ($P \geq 0.05$).

Given such results it was decided to routinely use an aliquot of 100mg of dust for all subsequent (bulk sample) dust lead analysis experiments. Such apparent consistency of results implies initially homogenous (sieved) test dust populations and consistency of analytical procedure.

APPENDIX THREE QUALITY CONTROL PROTOCOL AND RESULTS

A3.1 GENERAL DETAILS

A comprehensive quality control (QC) procedure was instigated throughout analysis incorporating calibration blanks, reagent blanks, procedural blanks, field blanks, a variety of standard reference dust samples derived from domestic floor bulk dust samples, an internationally-certified test dust (NIST 2710 - Montana sieved soil, ground in a ball mill in order to pass a 74 micron screen), standard reference solutions, and an internal spike recovery reference standard, summary details of which are described below. Given the unique nature of sampling for this work, no identical duplicate samples (apart from standard reference dust) could be provided to assist QC protocols for any of the sampling conducted. All analytical work was conducted blindly, the QC procedures following those of Dewalt et al. (1995).

From an extensive literature review, there appeared to be no international household dust standard reference sample(s) available for lead.[5] Consequently, such a set of standard dust samples of varying lead concentration were prepared. Essentially, 32 domestic vacuum cleaner-collected composite (house) floor dust samples were separately sieved (Chapter Five) to produce sub-32μ fractions.[6] This particle size range was selected as a source of reference material for a number of reasons, namely:

a) to attempt to minimise the potential for sample variability as a result of sample non-homogeneity,

b) to resemble the particle size range most likely to be adherent to children's hands and representative of that material contributing to their contamination (Duggan and Inskip, 1985), and

c) to approximate the particle size range collected by the vacuum pump cassette technique.

A3.2 STANDARD REFERENCE DUST SAMPLES

House dust was chosen as a source of reference dust standards in that it was presumed that such material would be most capable of effectively matching the matrix of the test house cassette-collected dust provided for analysis.

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[5] Pella et al. (1995) reflect a need for such an, as yet unavailable, material, and report preliminary work toward the preparation of such a material. Whilst they comment that from SEM analysis of sieved dust particles, the mean diameter of the lead-bearing particles is of the order of one micron, they chose a sub-100μ dust fraction as the basis for their dust standard, as "representing a good compromise between having the necessary homogeneity of the material and adequate yield of the dust material" (Pella et al., 1995; p. 351). Binstock et al. (1995) chose a No. 60 sieve (250μ) for their standard. Given the RSD differences between eg lead in dust values for sub-100 and sub-32 micron dust fractions reported in this dissertation (Chapter Five), it could be construed that a sub-32μ fraction (or even smaller) might be a more precise lead in dust standard.

[6] Such dust samples were selected in the belief that they would provide iso-osmolarity of matrix to sample dust and provide (as Binstock et al., 1995) similar analytical challenges in the laboratory.

Four different standard reference dust samples were selected and used throughout analysis. These were chosen since their lead content values spanned the range of concentrations that were expected to be realised by AES analysis of (cassette-collected) house dust as a result of preliminary investigations in Port Pirie. One internationally-certified standard reference material was also employed.

A3.3 STANDARD REFERENCE SOLUTIONS AND PROCEDURAL BLANKS

A standard reference solution (1,000$\mu g.g^{-1}$) was prepared, from which five diluted reference solutions were also prepared (Appendix Seven). These were all prepared in a 5% HNO_3 medium, and typically spanned the operating range of the appropriate analytical instrument. Such dilution reference solutions were prepared from the standard reference solution immediately prior to analytical work. Standard reference solutions, as also reference dust, were regularly included in the array of analysed samples, a standard being introduced approximately every ten cassette samples. For the alfoil dust collectors, the four reference solutions were assayed equally frequently every twenty dust samples, ie one quality control standard solution was assayed (in duplicate) after every five test analyses were performed. All solution standards were assayed (in duplicate) at the beginning and end of every run on the various instruments used for this work. Procedural blanks were also introduced typically at the beginning and end of each run, and also after every approximate tenth dust sample analysed. This is described in more detail below.

A3.4 QC PROTOCOL - STATIC DUST CASSETTE SAMPLES

For the analysis of the entire suite of static dust samples, reference dust samples were incorporated at better than once per every ten cassette-derived samples. Equally, procedural blanks were also incorporated at approximately every tenth sample, field blanks being incorporated at better than once per every twenty cassettes analysed.

All field and procedural blanks were initially analysed on ICP AES subsequent to which the solutions were immediately analysed utilising ICP MS, the sensitivity of the latter instrument exceeding that of the former ($\approx 0.05\mu g.g^{-1}$) by at least an order of magnitude (J. Wattling, J. Waters, pers. comm.). This was appropriate for such controls, since, typically, the AES-registered analyte value was below the detection limit of that instrument.

A3.5 QC PROTOCOL - AIR SAMPLING CASSETTE SAMPLES

Standard reference dust controls were included in the analytical regime at better than once per every ten air-derived samples. For the air lead data suite, as for static dust cassette analyses, reference dust controls were incorporated at better than once per every ten cassette-derived samples. Equally, procedural and field blanks were also incorporated at better than every tenth sample. ICP MS was again used for analysis of such blanks.

A3.6 QC RESULTS

A3.6.1 STANDARD REFERENCE DUST SAMPLES

A summary of the results of multiple analysis of the four standard dust samples together with the one certified reference dust utilised for this work is referred to below (Table A3.1).

Table A3.1 Standard reference dust samples for analysis.

		GEOMETRIC MEAN VALUE			
		PRELIMINARY REPLICATE STUDY (n ≈ 50)		ANALYTICAL PROCESS (n ≈ 25)	
DUST	CERTIFIED VALUE ($\mu g.g^{-1}$)	LEAD VALUE ($\mu g.g^{-1}$)	RSD	LEAD VALUE ($\mu g.g^{-1}$)	RSD
A1 - 5	-	553	1.6	529	5.5
G1 - 6	-	2,245	1.5	2,348	5.2
MIX - 1	-	7,938	2.0	8,326	3.8
K1 - 7	-	14,800	1.9	15,592	4.6
NIST 2710	**5,532 ± 80**	**-**	**-**	**5,176**	**4.3**

The four reference dust standards employed for analytical work had been previously multiple replicate (AAS) analysed to provide accurate estimates of their dust lead concentration levels. Individual RSD values at this stage typically remained below two percent ($n \approx 50$); weighted mean RSD = 1.9%.

The mean lead concentration value for these dust samples, as arising from their use as quality control standards in the static dust analysis work, expressed with respect to these previously-acquired mean lead concentration values for these dust standards (replicate study, where n was substantially larger), never varied by more than six percent. The (weighted) mean recovery for these test dust standards from the total analytical procedure for all static dust analytical work was 101.5%. RSD values for multiple replicate test standard dust analyses generally remained well below ten percent on a batch by batch basis. Comparison of replicate study with analytical study dust lead concentration values for each of the four test dust standards indicated no difference between data suites ($P \geq 0.075$).

A3.6.2 PROCEDURAL AND FIELD BLANKS

A3.6.2.1 STATIC DUST SUITE

For procedural and field blanks, the mean quantity of lead present was of the order of 0.027µg (n = 44) and 0.28µg (n = 25) respectively, which both appear relatively small by comparison to the mean cassette-contained lead for the entire suite of static dust samples of 556.35µg lead (n = 528). The difference between mean procedural and field blank figures would, presumably, be due to contamination of the filters/cassettes whilst handling (both packing and unpacking), the lead content of new filters directly from supplier's container being below the detection limits of analytical equipment used.

A3.6.2.2 AIR SAMPLING SUITE

For procedural and field blanks, the mean quantity of lead present was of the order of 0.019µg (n = 24) and 0.17µg (n = 24) respectively. These both appear quite small by comparison to the mean cassette-contained lead for the entire suite of air-derived samples (3.82µg lead; n = 166).

A3.6.2.3 NIST LEAD RECOVERY - STANDARD SOLUTIONS

Batch-to-batch recovery of NIST 2710 standard from this analytical work ranged from 91% to 98% for lead (weighted mean = 94.7%), and all finally adopted sample dust lead values for subsequent interpretation were based upon the calculated data corresponding to 100% lead recovery for NIST standard 2710 for the particular analytical run.

Mean batch values for the concentration of lead in the standard reference solutions as indicated during analysis never varied by greater than five percent from their presumed value.

A3.7 INTER-LABORATORY QC PROCEDURE

An inter-laboratory system of QC checking of standard materials used for this work took place with the incorporation (blindly) of all known standards. Three other laboratories participated in this work. Summary mean values for standards subsequent to multiple replicate assay elsewhere, never varied by greater than ten percent of the presumed value of standards when assayed by this researcher according to the specified analytical protocol as described above.

A3.8 STANDARDS FOR OTHER ELEMENT ANALYSIS

The predominant focus of this overall work is directed toward the analysis of dust samples for contained lead. In some aspects of this work, however, analysis for other elements was also conducted, eg zinc. Standards used for QC procedures for lead were also used in an identical fashion as controls for other elements, other than for standard solution preparation, where stock solutions for other elements of interest were prepared specifically for the element of concern. In general, the results of QC procedures for these elements aligned well with those for the principal element of concern for this work, lead.

A3.9 BALANCE PRECISION

A3.9.1 CASSETTE WEIGHING

All gravimetric determination for cassette-contained (static suite) dust was conducted to an average precision of 0.0013gm (≈ 1,200 measurements) using an electronic precision balance with a rated precision of 0.0001gm (Sartorius 1207 MP 2, Sartorius, Goettingen, Germany - Selby Scientific Ltd, Notting Hill, VIC). Fifteen different weighing standards were used equally frequently for this purpose, eight of which consisted of certified weight standards in the range 0.5gm - 50gm (Microid Metric Weights, Griffin and Tatlock, London). The remainder included a variety of combination blank cassettes in the weight range 4.0gm - 40.0gm. The balance was (internally) calibrated between every twentieth weighing, with one of two test weights being used.

The plastic reference standards were incorporated in order to compare variation in precision with regards to known standard weights and plastic sampling cassette standards for which it was presumed that some variability in weight might occur as a result of a variety of factors such as variation in temperature,

relative humidity, degassing of plastic etc. This presumption was confirmed by demonstrated precision values, which for (metal) weight standards was 0.0002gm, and for cassette weight standards was 0.0022gm.

A3.9.2 ALFOIL COLLECTING TRAY WEIGHING

All alfoil tray weighing was conducted on the same balance in a temperature- and humidity-controlled room at a bench temperature of between approximately 20 and 23 degrees Celsius. All samples both prior to use and also subsequent to return to the laboratory were left adjacent to the balance for a minimum of 48 hours for temperature equilibration. The balance was tared between every alfoil tray weighing (although this was rarely necessary), and when weighing, five minutes was allowed for the instrument to reach stability. The balance was (internally) calibrated between every tenth weighing, with one of two test weights. Further, calibration was also reviewed with the use of five control (alfoil) samples and five test weights, one of these being used every fifth tray weighing. Repeat weighing of each of these five standard pieces of folded alfoil (control samples) in the weight range five to fifteen gms was conducted equally frequently throughout the weighing process.

The demonstrated instrument precision for alfoil weight standards was 0.03mg, and similarly so for test weight standards. Mean contained dust per collector was of the order of 0.442gm, clearly well above the sensitivity threshold of the selected balance.

APPENDIX FOUR ROBERT KEHOE, LEAD AND HEALTH RESEARCH IN THE USA

Subsequent to the introduction of tetra-ethyl lead as a motor spirit octane-enhancing agent in 1924, US lead research was "dominated by the lead industry, particularly by Robert Kehoe" from the 1920s through to the 1960s (Berney, 1993; p. 5). Over that time period, Professor Kehoe (Univ of Cincinnati) was "the nation's most vocal and influential scientist working on lead hazards" (Graebner, 1988; p. 4). From 1924 to 1958, he was also the medical director and biomedical consultant of Ethyl Gasoline Corporation (Mushak, 1992), virtually the sole US manufacturer of tetra-ethyl lead. Over this time period, he was also director of the Kettering Institute laboratory, receiving funding from the lead and automobile industry in order to conduct research focussing on the health effects of lead in the environment (Graebner, 1988).

Thus Kehoe, who published widely, remained in control of the direction of research into the health effects of leaded petroleum for over 40 years (Gross, 1981; Rosner and Markowitz, 1985; Mushak, 1992). Child health concerns relating to the use of leaded petroleum were not readily accepted by Kehoe, who claimed that the manufacturer (of tetra-ethyl lead) "knew much about the product and should therefore be entrusted with seeing to health risk evaluations associated with the product's production and public use" (Rosner and Markowitz, 1985; p. 345).

Subject to a request in 1958 by the Ethyl Gasoline Corporation to increase the concentration of lead in gasolene, the US Surgeon General's appointed committee reported back that "the proposed increase in lead would pose no health hazard", but requested additional research relating to atmospheric lead (Graebner, 1988; p. 11). Such research commissioned by the Surgeon General, but conducted with the cooperation of the Kettering Institute, the automobile industry and petroleum manufacturers, concluded that the levels of airborne lead were lower in 1961/1962 than 25 years earlier (Berney, 1993).

Five years later (1966), at the US Senate hearings into air pollution, Clair Patterson accused Kehoe of conducting a whitewash in his analysis of the data (Berney, 1993), Patterson pointing out that lead levels in US cities were more than 100 times greater than they had been 30 years earlier (Patterson, 1965).

Kehoe, displaying, it would appear, a rather oleaginous personality, concluded that lead occurred naturally in the environment, including in human tissues and excreta, and that exposure to lead in air would not necessarily lead to either increased absorption or a greater body lead burden (Berney, 1993). He further argued that no relationship existed between either lead absorption or body tissue lead levels and lead poisoning (Graebner, 1988).

Thus over a period of unprecedented escalation in the use and environmental discharge of lead in the USA and unprecedented growth in the US population (the Baby Boom of the post war years, [Mushak, 1992]), such use of lead in gasolene and its consequent environmental discharge remained virtually restriction free (Graebner, 1988), as advocated by its manufacturer and endorsed by the public health watchdog.

APPENDIX FIVE HOME ACTIVITIES DISTRIBUTING LEAD AROUND THE HOME

Home-activity involving lead may include practices such as quench-hardening of cutlery in a bath of molten lead at a temperature of between 800 and 1,100 degrees Celsius (Kawai et al., 1983), at which level, it is likely that some lead fuming would occur (Moeller, 1952). Other related activities include lead type-printing (Kawai et al., 1983; Jensen and Laxen, 1985), recycling of batteries (Dolcourt, 1981), lead-based pottery glazing (Simms, 1986), lead-lighting (Anon., 1985) etc, and also the corrosion of leaded strips in decorative lead-light windows due to window condensate corrosion activity (Al-Raday et al., 1993c). Investigations revealed dust lead concentrations for two separate houses where cutlery-quenching and type-setting was practised, ranged from 260$\mu g.g^{-1}$ (for a control room) to 750,480$\mu g.g^{-1}$ (75%) for a window frame (presumed window sill) from a living room of a house containing the cutlery-tempering home workshop. Whilst sample size was rather small (13 samples) the geometric mean level from this very skewed distribution was calculated to be 2,738$\mu g.g^{-1}$, (SD = 8.3).

Lead-in-air levels at a point in front of the lead furnace in the house practising cutlery-quenching was recorded at 50$\mu g.m^{-3}$, with a figure of 6$\mu g.m^{-3}$ at a point three metres from the furnace, and 2$\mu g.m^{-3}$ in an adjacent kitchen (Kawai et al., 1983). These figures greatly exceed the EPA (USA) recommended outdoor air maximum lead level of 1.5$\mu g.m^{-3}$ (Fergusson, 1986). Whilst some volatilisation of lead would be occurring as a result of the activity of the furnace, Kawai et al. (1983) concluded that the dispersal of air-lead originating from the home workshops was likely to be only a minor contributor to surface dust lead contamination in rooms other than the home workshops. Noteworthy here is that these authors observed that those individuals who had intimate contact with the operation of the furnace displayed greater blood lead levels than those who did not. Further, the children living in these "homes" exhibited greater blood lead levels than control children of a similar age. The authors thus concluded that contaminated household dust is a major source of excessive lead exposure among children of families living in these houses. Additionally, these authors also report that their study appeared to be the first to point out that lead-containing house dust present in these homes "where lead work is being undertaken provides a significant source of lead exposure for children" (Kawai, 1983 p. 45).

Children older than fourteen, and not involved in the lead work conducted at home, presented no evidence of elevated lead absorption as opposed to younger children, where elevated blood lead levels were interpreted as a reflection of the contribution to such levels by hand-to-mouth behaviour. Given the awareness of health concerns associated with working with lead at the time of the study, such domestic activity seems rather surprising, a point not entirely lost upon Davies also (1988). It has been proposed by many workers that hand-adherent dust is regularly orally sampled by children, and that the smaller-sized particles exhibiting the greatest lead concentrations (Duggan and Inskip, 1985) are most readily retained on the skin (Raab et al., 1987). Duggan (1983) notes from a review of a number of studies, that a child blood lead level peak typically lies at around two to three years of age, coinciding with a peak in mouthing activity at that time, McMichael et al. (1985) concurring. By contrast, no peak blood lead value at two to three years of age was found in long-term patients in an Amsterdam hospital (Zielhuis et al., 1978). Thus, home activity may contribute substantially to elevated body lead burdens, particularly for young children.

APPENDIX SIX STANDARDISATION OF TECHNIQUES FOR SAMPLING, ANALYSIS AND ANALYSIS OF DATA

From an appraisal of the literature, it should be noted that no universal standards for the collection, storage, pre-analysis treatment and analysis of dust currently exists, a concern expressed more than a decade ago by Duggan and Inskip, (1985) and Fergusson, (1986). Further, given that many workers report arithmetic means whilst others geometric means or median values, comparisons of data other than in general terms becomes fraught with difficulty.

Given the environmental nature of such collected data, their distribution is likely to be skewed to the lower end of the distribution, and hence there would appear a need to use medians or geometric means as the most appropriate measure of central tendency (Duggan and Inskip, 1985). Davies reports that dust lead frequency distributions are most typically positively skewed and approximate log-normal distributions, such that log-transforming the data will "result in a Gaussian or normal distribution" (1988; p. 25).

Davies (1988) certainly raises the issue of inconsistency of sample analysis between different studies reducing the potential value for inter-study result comparison echoing Elwood (1984 & 1986). As Jones and McDonald (1983) point out from their evaluation of the efficiency differences between four of the analytical techniques adopted by several of the workers, there could be as much as a two-fold difference in measured dust lead concentration arising purely as a result of differences in simply dust analytical technique. Sampling and sample preparation technique, and analytical method thus clearly need to be standardised for future work. In this regard, some attempts toward, particularly, sampling technique standardisation are appearing in the literature (eg ASTM 1986), and a number of workers have conducted comparisons between different analytical techniques, and suggested preferred techniques (Que Hee et al., 1985 - Appendix Two). Different sampling techniques include sweeping with a brush and tray, large or small vacuuming apparatus, single/double-sided adhesive tape, wet or dry wiping towlettes, and with respect to the collection of falling dust, a number of different devices including glass/plastic deposit gauges, greased trays or glass slides adhesive tape etc (Dolan, 1986 - See Appendix Sixteen). Several of the dust-sampling techniques utilised for this research are proffered as potential models for future standardisation of technique (Chapters Seven and Eight).

Other concerns relating to the comparison of data deriving from different studies focus upon differences in external surfaces of objects (eg furniture) from which dust is collected (Ellis and Revitt, 1982; Beckwith et al., 1985; Davies et al., 1987b) ie how effectively (preferentially) will dust particles adhere to certain surfaces? Further, there also exists the need to be cognisant of the potential role of spatial and temporal effects (Fergusson and Simmonds, 1983; Duggan, 1984; Davies et al., 1987a; Fergusson, 1987). Whilst little has been investigated in this regard in the context of indoor settled dust, two groups of US' workers have investigated the spatial distribution of soil lead peripheral to houses. Ter Haar and Aronow (1974) conclude that lead from degrading paintwork and not automotive exhaust is the major contributor, yet Solomon and Hartford (1976), however, suggest to the contrary.

APPENDIX SEVEN ATOMIC ABSORPTION SPECTROPHOTOMETRY (AAS) - OPERATING CONDITIONS AND QUALITY CONTROL

A7.1 AAS OPERATING CONDITIONS

The AAS used throughout this body of work was a Varian Spectra 250 AA (Fisons Scientific, Sydney) operating with an Air-Acetylene flame.

Optimal lamp wavelength for lead was specified at 217 nanometres (nm; 1 nm = 10^{-9} metre), and this was used throughout. As specified by the operator's manual , background correction using the inclusive deuterium lamp was switched on.[7] Lamp current was set at 5mA, and slit width 1.0nm. Air flow was set at 13.5l.m^{-1}, and Acetylene at 2.0l.m^{-1}. Delay time was set at two seconds, and default time at five seconds affording a very reasonable instrument precision of one percent RSD. All readouts were conducted in manual mode.

All sample analysis readouts were made in triplicate. All sample introduction to the aspirating tube was performed manually, given concerns regarding the potential for inlet port orifice blockage by particulate material suspended within the digested solution. Such blockage could result in inaccuracies in reported lead-in-dust values subsequent to blockage. If blockage (partial or complete) occurred, the operator could easily detect this by inconsistent triplicate readouts, and also an audible difference from normal operation. In such cases the port was cleared, and standards run again. If the values for standards varied by more than ten percent from their rated concentrations, the machine was re-calibrated. Following clearing of the port, the operator would return to the penultimate sample, and run it through the spectrophotometer again, continuing with further analysis.

A7.2 AAS PRECISION

In order to assess the consistency of the instrument for multiple sampling, six separate batches, each of 25 glass tubes containing two mls of an identical lead standard of 10μg.ml^{-1} (Appendix Two) were sequentially analysed for lead concentration by the instrument.[8] Comparing both the absorbance and concentration values for each batch of analyses revealed no significant difference between any of the batches of 25 tubes tested ($P \leq 0.05$). A summary of the results appears below (Table A7.1).

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[7] Slavin (1988) reports that virtually all samples should be run with background correction on.

[8] For the purposes of discussion here, as Kawai et al. (1983), the terminology ppm lead in solution, and μg Pb.g^{-1} solution or μg Pb.ml^{-1} are considered equivalent.

Table A7.1 Multiple replicate sample analysis results.

	MULTIPLE REPLICATE, STANDARD SOLUTION SAMPLING RESULTS		
BATCH NUMBER	**MEAN ($\mu g.ml^{-1}$)**	**SD**	**RSD (%)**
One	10.46	0.03	0.3
Two	10.43	0.08	0.8
Three	10.39	0.12	1.1
Four	10.39	0.08	0.8
Five	10.39	0.11	1.1
Six	11.00	0.09	0.9

Such results appear consistent with suggestions of a relative precision of one percent or better for AAS methods (Skoog and Leary, 1992).

A7.3 ACCURACY

Utilising one standard reference solution of 1,000$\mu g.ml^{-1}$ lead in water for calibration purposes (Appendix Two) throughout the time of use of the AAS, contributed to the consistency of the instrument's analytical values. A fresh batch of such a solution was prepared every month that analytical work was undertaken, and its AAS-estimated concentration compared to all previous batches of standard reference solution which were retained throughout the continuum of this study. At no stage did the instrument-estimated concentration of a reference solution vary by greater than three percent from its presumed true value of 1,000$\mu g.ml^{-1}$ (with the AAS calibrated using the immediately preceding batch of calibration standard), or by greater than four percent when compared to all other (previous) reference solutions. This reflects the stability of the reference solutions, the consistency of their production and also the accuracy of the instrument's performance.

All standard solutions were stored in tightly-sealing glass containers at room temperature, concerns existing regarding precipitation of lead salts at temperatures between zero and four degrees Celsius. Inter-laboratory comparison of these internal lead in water standard reference solutions (also by flame AAS) was regularly undertaken as part of that laboratory's QA/QC procedure. Results here again reflect the accuracy of the instrument, the instrument-estimated concentration of the reference solution never varying by greater than four percent from its presumed true value of 1,000$\mu g.ml^{-1}$.

A7.4 CALIBRATION TECHNIQUE

A stock lead solution of 1,000µg.ml^{-1} lead in water was prepared (Appendix Two). From this stock solution, dilutions were prepared, resulting in the following calibration standards:

2, 6, 10, 16, 25µg.ml^{-1}.

These standards and a sample blank (water) were routinely used for the calibration of the instrument which was performed at the beginning of each test run, and again after every 30 test solutions, for which all were assayed in triplicate.

Further, test standards for quality assurance purposes were also prepared from the same stock solution, and with the following lead in solution values:

4, 8, 12, 20µg.ml^{-1}

These same calibration standards were also used as quality control standards throughout the operation of the instrument. Thus ten calibration standards were employed as unknowns throughout the use of the instrument for lead in solution analyses. The same batches of these quality control standards were used throughout the period of time that the instrument was used, although equivalents were prepared at the same time that new batches of stock solution (1,000µg.ml^{-1} lead in solution) were prepared, and these freshly-prepared quality control standards compared to the existing standards. At no stage was there greater than a five percent difference between corresponding standards when replicate batches of 50 samples were compared from respective standards.

A typical calibration curve for this particular instrument using the above-mentioned calibration standards is given below (Figure A7.1).

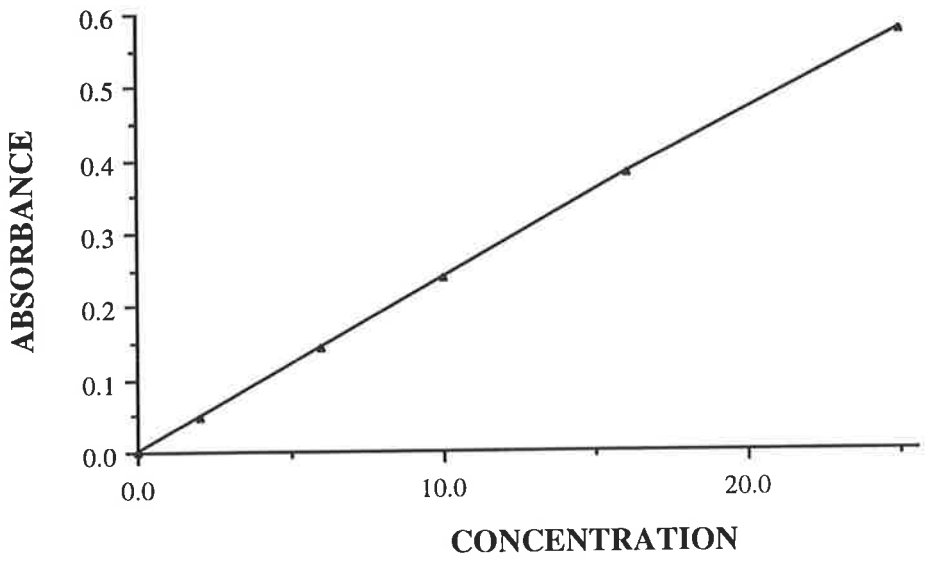


Figure A7.1 Typical AAS calibration curve.

As can be seen the relationship between absorbance and concentration remains linear over virtually the entire range of calibration standards. All readings lay at an absorbance figure greater than that corresponding to $2\mu g.ml^{-1}$, and all absorbance values corresponding to a figure greater than $\approx 15\mu g.ml^{-1}$ were diluted variously between one in two and one in two hundred in order for the final analysed value to lie within the linear component of the calibration curve.

A7.5 STANDARD LEAD SOLUTION PREPARATION

The $1,000\mu g.ml^{-1}$ lead standard described above was prepared as follows:

1 gram (.005 Mole) of 0.5 mm thick lead foil (99.999% purity; Aldrich Chemical Co., Castle Hill, NSW) was dissolved in 20mls of > 6M (1:1 conc. HNO_3 - ≈ 15M) Nitric Acid, and slowly brought to approximately 120 degrees Celsius. Whilst held at this temperature, the solution was brought close to dryness, and the heating removed allowing the solution to reach virtual dryness as it cooled. Subsequent to cooling, the final solution was made up to 1 litre with water, agitated well and the lead content compared to previously prepared batches using the AAS.

A7.6 INTER LABORATORY COMPARISON OF STANDARD DUST.

Inter-laboratory comparison of selected sieved ≤ 32μ standard dust replicates also contributed to analytical accuracy. A comparison of the mean dust lead concentration values for various dust samples as (replicate) assayed in different laboratories never indicated variation by more than ten percent from mean values elicited by this work. The concentration range of such standard dust samples spanned that which could reasonably be expected to be encountered in the conduct of the body of this research, ie 500 - $15,000\mu g.g^{-1}$.

APPENDIX EIGHT BULK DUST SIEVING DETAILS

Bulk dust was physically separated with a sieve shaker in a dry dust format. A wet technique for dust separation was not undertaken given concerns with the unknown nature of the sample, and the potential for material loss in solvent phase (J. Davis, pers. comm.)

Prior to separation, all samples were stored in glass containers at 100 degrees Celsius for 4 hours (as Fergusson and Ryan, 1984). The selected test sieves were all tightly-fitting, and were stacked with the largest mesh size sieve at the top, immediately underneath the (tight-fitting) stainless steel lid, and the smallest at the bottom of the stack, immediately above a stainless steel collection tray. Masking tape was used to seal the (annular) junctions between successive sieves, although losses of fugitive dust appeared to be very minor, typically much less than two percent of dust sieved. The mesh sizes utilised were as follows:

APERTURE SIZE (μ)	500	250	150	75	45	32

A total of 350gm of dust per sample was separated, with a maximum of 35gm of dust being separated at any one time, this being carefully introduced into the top sieve. Subsequently, the lid was applied, the stack of test sieves introduced into the shaking apparatus and secured, and the apparatus set into operation at maximum agitation level. Periodically during the sieving process, the sieve shaker was stopped, and the test sieves were gently tapped with a rubber mallet at various locations around the periphery of the sieves to discourage particle aggregation and sieve blockage.

Three separate periods of agitation for fifteen minutes were undertaken with a pause between them for the stack of sieves to again be gently tapped with a rubber mallet around the periphery of the sieves.

In a draft-free environment, the contents of individual test sieves were carefully emptied onto separate clean sheets of shiny brown paper and the sieves brushed over with a clean camel-hair brush such that dislodged material also fell onto the appropriate piece of collecting brown paper, and was not lost. Such dust was transferred to separate (tared) white 500ml sealable plastic screw-top containers (Polarcup, Bankstown NSW Australia), the quantity of dust having been collected being gravimetrically determined to an accuracy of 0.1mg, and the sample stored.

For the separation of further dust aliquots from the same source, the sieves were assembled, the junctions taped again, and the sieving procedure repeated until all remaining dust had been separated.

Between different dust samples, all sieves were scrupulously cleaned with a clean, soft, camel-hair brush followed by a rinse with a squeezy bottle containing ethanol, and finally by a burst of filtered low-pressure air (10 - 15 psi). New masking tape was used to seal junctions between adjacent sieves.

To assess the lead content of the 28 test dust samples, exactly 100mg of each dust was weighed into the digest vessel, typically a ten millilitre pyrex test tube. Digest mix and protocol was as indicated in Appendix One.

APPENDIX NINE VACUUM PUMP DUST-COLLECTION SYSTEM

A9.1 PUMP DESCRIPTION

Dust sampling was conducted using a (12 volt) battery-operated vacuum pump (Dynavac Model No. OD 2 - 12; Thomas Industries Inc, Sheboygan, Wisconsin), demonstrating a maximum capacity of > 50 litres per minute ($l.min^{-1}$) @ -75kPa (Dynavac, 1992).[9] The pump was run at a flow rate of approximately $2.0l.min^{-1}$. Que Hee et al. (1985) report, for a similar sampling system, a desirable flow rate of between 1.2 and $2.5l.min^{-1}$, at which level, the dust-collecting system's efficiency (their estimate ≈ 75% for the collection of loose [available] surface house dust) and reproducibility is optimal. These authors also claim no dependence of sampling efficiency and precision upon specific floor surface materials, or dust particle sizes below 177μ.[10] Que Hee at al. (1985) also report that this size fraction constitutes greater than 80% of the total weight of house dust. Preliminary work conducted here reveals a mean mass balance figure of approximately 72% of total dust collected in the size fraction < 150μ (Chapter Five). Thus, if a sampling method reflects the same sampling efficiency over all surfaces encountered, subsequent correlation analyses should not be affected differentially providing that the minimum detection threshold level is exceeded.

Air flow rate was indicated by an attached, adjustable, moving ball (variable orifice) rotameter (0 - $5l.min^{-1}$; Series VF Visi-Float; Dwyer Instruments Inc, Michigan City, Indiana), with regular calibration of the pump/rotameter being undertaken using a portable, multi-purpose gas meter (Toyo Model ML2500; Davies Shephard, Caringbah, NSW). To this end, the sampling system was connected to a "dummy" cassette and wand to simulate operating conditions in the field, and run for a period of at least sixty minutes.

Only two pump systems were constructed and employed to collect the entire suite of dust samples corresponding to the work described below (528), each pump system being used exclusively by one of the two workers. This purpose-chosen dust-sampling apparatus was intended to be used as a mini vacuum cleaner in order to sample according to a strictly-observed sampling regimen (Appendices Nine and Ten). Between the pump inlet end and the rotameter outlet end, a closed leak-proof plastic "expansion" chamber 75mm diameter and 100mm long was connected by 6.25mm gas thread leak-proof brass fittings (Rasch Pty Ltd, Adelaide).

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[9] The pump was chosen to be battery-powered such that even in premises where the electricity supply had been disconnected, sampling could continue, and all houses could be sampled in an identical fashion.

[10] This work undertaken by Que Hee's group (1985) was conducted with floor dust samples which could be construed to be a different population of dust to that which was collected from Port Pirie floors. From the particle-sizing work referred to in Chapter Five, however, strong similarity in the disposition of contained lead with respect to (Port Pirie floor dust) particle size range compared to results suggested by the above authors has been noted.

This chamber, containing large-hole poly-urethane foam, was utilised in order to smooth out the vacuum "pulses" produced by the diaphragm vacuum pump, and reduce consequent rotameter "flutter".[11]

As part of the apparatus chosen for the collection of surface dust, attached to the (inlet end of the) rotameter via exactly one meter of 6.25mm internal and 11mm external diameter Tygon® tubing (Air-Met Scientific, Keswick SA) was a 37mm two-piece closed-face aerosol monitoring cassette. This contained, as the sampling medium, a 37mm 0.8µ cellulose ester plain white filter and backing pad (Gelman Sciences, Ann Arbor, Michigan).[12]

The inlet orifice of the cassette was connected by a 25mm long piece of Tygon® tubing to a 150mm long, annealed, stainless steel tubing "sampling wand" (10.3mm internal diameter, 12.7mm external diameter; Atlas Steel Supplies, Richmond SA). The sampling wand was bent to approximately 45 degrees, 50mm from one end. This was designated the distal or outlet side of the sampling wand. The other end of the steel tube was cut off at an angle of 45 degrees, this being the face of the sampling wand contacting the sampled surface. This "angled" or "sampling" end was held parallel to and just touching sampled surfaces, similarly to the sampling method of Que Hee et al. (1985). Both internal and external surfaces of the sampling tube were smoothed with 1200 grade sanding paper to prevent sharp edges and thus potential locations for dust retention. The prepared sampling tube was soaked in 0.5M Nitric acid for 24 hours and washed thoroughly in clean water subsequently, to minimise preliminary contamination.

The plastic tubing was pushed a distance of approximately five to seven millimetres over both the inlet end of the cassette and also the outlet end of the sampling wand, leaving, (between these two), between approximately seven and sixteen millimetres (mean = 10.5, n = 50) of tubing, the inner bore of which, as with the sampling wand, was exposed to the air/dust flow.

Thus air (and dust when sampling) was drawn through the sampling wand, the short piece of connecting Tygon® tubing, the cassette (where the sampled dust was contained), the air continuing through the long piece of Tygon® tubing, the rotameter, the expansion chamber and finally the vacuum pump itself.[13] Both vacuum pumps were mounted on wooden boards, the underside of which were lined with 20mm thick poly-urethane foam to reduce vibration "creep" of pumps and also vibration effects on rotameters and (particularly wooden) surfaces.

It was felt that this system for sampling dust incorporating a robust diaphragm pump, expansion chamber and attached rotameter would result in an instrument demonstrating constant, consistent and (if necessary) adjustable air flow rates, this being essential for repeatable, representative sampling (Duggan and Inskip, 1985). Researchers using such a cassette/filter sample-collection system type, but not

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[11] Ashton and Gill (1992) refer to pulsating flow with a diaphragm-type pump, and its undesirability with respect to dust sampling, but make no suggestion as to how this phenomenon may be avoided. Certainly when the pump was attached to a manometer the surface of the water was substantially less disturbed after attachment of the expansion chamber (pers. obs.).

[12] The literature reports that dust adherence losses may be minimised by utilising Tygon® tubing (Que Hee et al., 1985), a tubing type possessing an extremely smooth, dense, abrasion-resistant bore, reducing interior build-up of materials (Anon., 1992b). Comparison between such tubing and clear plastic PVC tubing was undertaken to estimate average tubing dust retention and differences between tubing types (Appendix Fourteen). Both types of tubing fitted quite snugly on both the cassette and sampling wand, and throughout, all pieces were examined for any signs of damage or imperfection before use. Despite the ready availability of PVC tubing at many locations, its relative cheapness, and the determination of no significant dust retention difference between the two tubing types, all sampling for the purposes of the research work conducted herein was undertaken with Tygon® tubing.

[13] A preliminary investigation of the dust passing through the cassette and filter, and being retained by the (one metre long) connecting tube between the cassette and the rotameter revealed after a series of ten separate cassettes had been sampled, insufficient lead to reach the flame AAS detection limits (≈ 1 - 2µg lead). Que Hee et al. (1985) note that dust penetration through such a filter paper type is less than 0.1%.

practising pump flow rate assessment and regulation in the field (eg Dewalt et al., 1995), may find a consequent and substantial loss of sampling precision.[14]

A9.2 OPERATION OF PUMP

This vacuum pump/cassette dust collection system was used to collect contaminant surface dust from previously demarcated (with masking tape and a wooden stencil) square areas, such areas being typically 0.0930 square metres. Extreme care was exercised to not touch or disturb the area to be sampled with hands, feet or implements before sampling.[15] A clean (pre-weighed) cassette was used for each separate sampling site, hands being cleaned with pre-moistened towlettes (Wet Ones - Reckitt and Colman, West Ryde, NSW) every five samples taken, and also when changing from one room to another. Care was also exercised to ensure that hands (particularly fingers) made no contact with floors or other contaminated surfaces as also the entry port of cassettes, thus minimising the potential for contamination. A detailed description of the sampling procedure is described in Appendix Ten.

The chosen operation flow rate was selected not so much as to be capable of picking up as much contaminated dust as possible from the sampled area, rather in order to collect a certain component of the surface-resident dust that fairly represented that size fraction of dust that would be most likely to be picked up by a child touching a floor, and to remain adherent to that child's hands. Essentially, this is the "available" dust, (as Brunekreef et al.'s "biologically significant" [1981; p. 447] or Ewers et al.'s "available to a child" [1994; p. 8]). Duggan and Inskip (1985) acknowledge an inverse relationship between particle size and lead concentration for street dust, pavement dust and soil particles (all potential contributors to indoor dust [Brunekreef, 1984]), Raab et al. (1987) thus highlighting the difficulty in deciding upon appropriately-representative particle size ranges for dust that is to be sampled.

The particle size range of (child) hand-adherent dust particles has been suggested to be less than 150μ (Driver et al., 1989). Further, Duggan et al. (1985) report that emphasis should be directed toward smaller particle sizes when sampling. Thus, sampling of environmental dust should reflect the size range of the dust particles likely to be ingested by a child.

From initial experimentation (Chapter Five) the (distribution) mean dust particle size collected by this system was of the order of 22μ (10^{th} and 90^{th} percentile values of 4.3 and 47.8μ respectively). This appears comparable with estimates of dust particle size ranges attached to a child's hands (typically below 20μ, [Inskip and Hutton, 1987]), and distribution mean dust particle size values acquired by domestic vacuum cleaners of the order of 73.4μ (10^{th} and 90^{th} percentile values of 11.9 and 169.9μ respectively).

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[14] Acknowledging generic rotameter inaccuracies (+/- 5% - 10% when supplied with a calibration curve by the manufacturer; Anon., 1989; Appendix Nineteen), the rotameter was regularly calibrated for volume of air flow (both in the field and the laboratory) against a more accurate secondary standard, a multi-purpose (dry) gas meter (Toyo Model ML2500; Davies Shephard, Caringbah, NSW). An accuracy of one percent can readily be expected from such an instrument when calibrated against a primary standard - Appendix Nineteen). The rated accuracy of this meter is reported to be well within +/- 1% for flow rates of from 0.01 to 2.0$m^3.hr^{-1}$ (0.17 - 33.33 $l.min^{-1}$) (Anon., 1995).

[15] Window-sill and window-well surfaces sampled were not square obviously. Some sampled floor surfaces were 0.186 square metre surface area rectangles.

Further, mean dust lead concentration for all samples collected with this system is 4,249$\mu g.g^{-1}$ (Chapter Seven), as compared to a mean value for vacuum cleaner bag dust grab samples of 1,637$\mu g.g^{-1}$ (Chapter Five). Whilst comparison of these data is not strictly appropriate given the different sampling techniques used and houses from which they derive, such a comparison is still helpful, as it reflects an inverse relationship between particle size and lead concentration (as Randall et al., 1975; Dorn et al., 1976; Duggan and Williams, 1977; Spitler and Feder, 1979; Johnson et al., 1982; Duggan and Inskip, 1985; Davies and Watt, 1986; Steele et al., 1990; Fergusson and Kim, 1991). Dust lead concentration difference between the two techniques for dust sampling is clearly demonstrated ($P < 0.001$).

Concerns arose regarding the potential for both the wand and attached tubing to retain some of the dust passing through to the cassette as a result of possible irregularities of the inside surfaces of the tubes associated with slow particle movement rates, or possibly as a result of electrostatic forces generated by the plastic tubing, or perhaps both. Such dust retention is estimated in some preliminary experimentation (Appendix Fourteen). In summary, mean total losses to both sampling wand and tubing amount to 0.86mg dust and 0.00213mg lead (independently of the number of cassettes harvested with the wand and associated tubing). Compared to the average contents of all cassettes sampled for the static dust survey (248.1mg dust and 1.599mg lead; $n = 578$), this indicates that such losses (both $< 0.5\%$) are a minor source of error with this sampling technique.

All subsequent sampling for the main body of the research was performed with a new piece of Tygon® tubing being used for each room. The tube was left uncleaned between different cassette locations in that room, and with wands being cleaned with pure ethanol between different houses.

Subsequent to use, cassettes were sealed with the original sealing plugs which were removed immediately prior to sampling. These plugs were stored inside a clean sealable plastic bag whilst sampling took place. Cassettes were then stored carefully in sealable plastic bags and returned to the laboratory for re-weighing and subsequent contents analysis.[16]

At all times, cassettes were handled gently, care being exercised to ensure that no damage to the surface of the cassette occurred, with potential change to the weight of the cassette being thus elicited.

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16 All cassette weighing was conducted to 0.1mg on the same balance in a temperature- and humidity-controlled room at a bench temperature of between 20 and 23 degrees Celsius. All samples subsequent to return to the laboratory were left adjacent to the balance for a minimum of 48 hours for temperature/humidity equilibration (analogous to Spengler et al., 1981 and Ashton and Gill, 1992). The balance was tared between every cassette weighing (although this was rarely necessary), and when weighing, one minute was allowed for the instrument to reach stability. The balance was (internally) calibrated between every twentieth weighing, at which time a number of control samples and test weights were weighed. The (pre-use) weight of the cassettes (with filters) lay between 14.8070 and 15.9352gm, the mean weight being 15.032gm ($n > 2,000$).

APPENDIX TEN VACUUM PUMP DUST COLLECTION METHOD PROTOCOL

Once a new cassette had been connected to the vacuum pump, the operator commenced dust collection. Beginning from the top left hand corner of the sampling area, with the sampling wand just touching the surface material, and the bevelled orifice of the sampling wand held parallel to the surface, the wand was slowly moved parallel to the top boundary edge of the sampling area (with the top of the wand just touching the inner edge of the masking tape), to the top right corner of the selected area, and then back again over the same path. The leading edge of the sampling wand (ie in the direction of travel of the handpiece) remained slightly elevated to accommodate air flow and dust entry into the sampling wand.

On completion of the first return passage of the handpiece, the wand was lowered a distance equivalent to the handpiece orifice height below the first path, and a repeat return passage conducted immediately below and parallel to the first path. Sampling continued as such, with ensuing traverses commencing immediately below the previous traverse path in a raster-like fashion until the entire surface area selected for sampling had been covered.

A subsequent sweep was conducted over the same surface area, commencing from the bottom left of the selected area and travelling toward the top left corner of the chosen area, the surface being passaged in a manner identical to the first sampling pass (but with sweeps at right angles to the first sweep routine). During such sampling, care was taken to ensure moderate and even pressure was exerted on the sampling surface with the sampling handpiece.

Preliminary method investigation was conducted in order to evaluate the quantity of dust abducted from floor surfaces with each sweep of the wand. Summary results of this work is reported in Appendix Fifteen.

Throughout sampling, both before and during each separate sample acquisition, the rotameter setting was checked. Invariably, the setting varied only slightly or not at all during dust sample acquisition, any necessary re-adjustments being made prior to subsequent sample acquisition.

Considering that for a new filter contained in a sampling cassette, the maximum air flow rate for the cassette would be of the order of $6451.min^{-1}$ (Gelman Sciences, 1993; p. 56), some degree of contamination of filter would not suggest a significant change in filter air flow rate and hence sampling efficiency. This was evidenced with preliminary investigations in the field with cassettes reflecting no (or virtually no) reduction in flow rate as indicated by the rotameter when cassettes became virtually filled with dust (weighing of the order of five to ten grams; significantly more than that contained in any of the acquired samples; mean = 248.1mg, max = 4.987gm).

APPENDIX ELEVEN PRE-SIZING DUST PROCESSING TECHNIQUES AND INSTRUMENT SET-UP DETAILS AND OPERATING CONDITIONS

A11.1 PRE-SIZING DUST PROCESSING TECHNIQUES

Two sets of dust samples were acquired. One set (n = 27) derived from the vacuum pump and cassette sampling system (P Suite) the other (n = 25) derived from composite (domestic) vacuum cleaner bag grab samples (D Suite). To sample the latter, ten aliquots of dust from a variety of locations within the vacuum cleaner bag contents were made, and pooled.

Processing techniques for these dust samples are as follows.

A11.1.1 SUITE P SAMPLES.

Cassettes were opened in a clean draught-free environment and contents thoroughly washed with ethanol from a squeezy bottle through a simple 37mm diameter 400μ sieving cloth screen mesh apparatus - (Nylon Milling Mesh, Swiss Screens, Aust., Pty Ltd, Brooklyn Park SA) essentially to remove the (presumed) carpet fibres or other large non-dust particles. Such large-sized particles might possibly confound the final particle size spectrum. The literature would indicate that particle sizes of 400μ would be expected to be too large to remain adherent to a child's hands (Duggan et al., 1985; Que Hee et al., 1985; Steele et al., 1990).

For greater than half of the cassettes, no, or virtually no, residue remained in the sieving apparatus subsequent to washing. Such sieve-contained material was inspected macroscopically and microscopically, and revealed little or no apparent dust or soil particles, the vast bulk essentially being carpet dust fibres and some vegetative matter as expected (P. Body, pers. comm.).

Sieved material was transferred directly to approximately 100mls of ethanol in an appropriate test vessel (MSX14, Flow Cell System, Malvern Industries, Worchester, UK) with several (squeezy bottle) ethanol washings of the cassette and sieving apparatus, and then submitted to the particle sizing equipment.[17]

Mean (geometric) dust contents of (27) P suite cassettes was 145.0mg. Mean contents of the total suite of vacuum pump-sampled cassettes not including P suite samples (n = 578) was 116.5mg, no difference being apparent between these two data sets ($P > 0.375$). Again comparing these two sets of cassette-contained contaminant dust samples, there existed no difference between the cassette-contained mean dust lead concentrations (Mean = 3,634$\mu g.g^{-1}$, and 4,032$\mu g.g^{-1}$ respectively) ($P > 0.50$). Similarly, no difference was demonstrated between the P Suite and the main (Static Dust) suite for dust loading ($P > 0.10$) and lead loading ($P > 0.25$).

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[17] Ethanol was chosen as the washing and "dispersant" liquid since concerns arose regarding the solubility of contained material if (say) water was used.

A11.1.2 SUITE D SAMPLES.

Domestic vacuum cleaner bag grab samples were acquired by multiple sampling at different locations from within a cleaner bag with a large stainless steel spatula whilst mixing thoroughly. Again, as for P suite dust samples, D suite samples were sieved through 400μ sieve material, and transferred directly to ethanol in the test vessel. In this instance, approximately 200mg of dust was used for this purpose.

A11.2 INSTRUMENT SET-UP DETAILS AND OPERATING CONDITIONS

The particle-sizing device was calibrated with standard particle reference materials (Duke Scientific polystyrene latex spheres) of the following sizes, 0.5, 0.8, 20.5, 49.5 and 98.5μ, and regularly calibrated against a reference reticule. Active beam length was set at 2.4mm, sample presentation was in 2 OHD mode, analysis model was very polydispersed, and two lens systems were employed, the 100mm lens operating in the range 0.2 - 180μ and the 300mm lens operating in the 0.5 - 600μ range (Anon., 1992c). Ultrasonic stirring and continuous pump cycling of test vessel contents was employed. A refractive index of 1.3590 was chosen for the dispersant, and a sample standard refractive index of 1.5400 was chosen for the dust samples (Anon., 1992c). The SG of all dust samples was estimated utilising pyknometric techniques (Appendix Twelve).

APPENDIX TWELVE PYKNOMETRY DETAILS

Particle size determination by automated sizing equipment (Chapter Five) required that the density of individual dust samples be available. These density data were estimated by pyknometry, using a 10 ml pyknometer bottle (Laboratory Supply [SA], Edwardstown, SA) and absolute ethanol (AnalAR; BDH, Melbourne, Australia; SG = 0.789) as a wetting/displacing agent.

Briefly, the weight of the test vessel filled with ethanol was determined to an accuracy of 0.1mg. Subsequently the ethanol was decanted and the vessel allowed to dry completely and return to its tare weight. A known quantity of dust was added to the pyknometer (in the range 0.2 - 0.6gm), and the vessel carefully filled with ethanol ensuring that the dust was completely wet by the ethanol by very gently swirling the contents of the pyknometer when partly filled with ethanol. The dust was allowed to settle and the vessel completely filled with ethanol, before the lid was inserted into the test vessel and the outer surface wiped of any overflowing ethanol. The pyknometer containing dust and ethanol was then weighed, again to 0.1mg. From the difference in weight between the vessel filled with ethanol and filled with dust plus ethanol, the mass of dust-displaced ethanol can be arrived at, and hence the volume of ethanol given the density of ethanol. Since the volume of the displaced ethanol is equivalent to the volume of the dust added, and the mass of dust added is known, the density of the dust can thus be determined.

APPENDIX THIRTEEN INITIAL TESTING OF XRF CAPABILITIES

A13.1 DETECTION LIMITS

The Minimum Detection Limit (MDL) for a soil element using the XRF is defined as the analyte concentration equal to three times the SD of the analytical result on a "blank" sample (TN Technologies, 1994). Further, since soil element concentration levels less than but approximating the MDL may only indicate the presence of an element, they cannot effectively be utilised to provide accurate quantitative analysis. The Quantitation Limit, however, (that concentration threshold for analytical application of the instrument), which is defined as ten blank material SDs from a series of analyses carried out under identical sampling conditions and procedures (TN Technologies, 1994) appears a better choice for detection limits.[18] SD values calculated by the instrument reflect only instrument precision and obviously do not include procedural variables and do not reflect operator skill. Instrument and method limits are presented in Table A13.1 below.

Table A13.1 XRF Instrument and method detection limits.

	INSTRUMENT LIMITS (200 SEC TEST) *		METHOD LIMITS (60 SEC TEST)	
ELEMENT	**Detection Limits** ($\mu g.g^{-1}$)	**Quantitation Limits** ($\mu g.g^{-1}$)	**Detection Limits** ($\mu g.g^{-1}$)	**Quantitation Limits** ($\mu g.g^{-1}$)
Arsenic	71	237	59	197
Copper	125	417	313	1,043
Iron	317	1,057	684	2,279
Lead	40	133	84	280
Manganese	580	1,933	1,569	5,231
Zinc	100	333	158	526

* Data courtesy of M van Alphen.

All of the above-listed elements were included in the output data provided by the XRF. From the results of the vacant house soil study, all measured iron, and all but one lead and one zinc measurement exceeded the respective method quantitation limits indicated in the above table. A considerably greater

[18] In the range between these two sets of detection limits, the instrument may still provide valuable quantitative data.

proportion of Arsenic, Manganese and Copper indicated values lie below the quantitation limit, but for Cu, most (87%) lie above the detection limit. Arsenic and Manganese soil element data were not considered for this work. For lead, zinc and iron assessment, this technique is clearly sufficiently sensitive to be appropriate for the purpose of soil element level estimation in the test houses in Port Pirie. Further, despite a slightly lower sensitivity for elemental copper, this technique still provides valuable data, particularly in a comparative sense to lead and zinc data.

A13.2 SAMPLING-TIME PRECISION

The relative precision of XRF analysis which is limited by the statistics of counting random electrical pulse events, is influenced by the number of pulses detected. Hence increasing the exposure time will increase the relative precision in inverse proportion to the square root of the time factor (TN Technologies, 1994).

Given that with a longer exposure time, a greater precision would be elicited, the number of samples taken per unit time would, however, be limited.

A13.2.1 INSTRUMENT ACCURACY WITH DIFFERENT SAMPLING TIMES

Comparison of 60 versus 200 second sampling time utilising the NIST 2710 soil standard (lead = 5,532 $\pm$ 80$\mu g.g^{-1}$) revealed the following for instrument lead analyses (Table A13.2).

Table A13.2 XRF sampling time accuracy.

		XRF-DETECTED SAMPLE LEAD CONCENTRATION *		
ANALYSIS TIME ALTERNATIVES	**n**	**MEAN LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**SAMPLE STD DEVIATION**	**RSD (%)**
60 second analysis time	**24**	5,844	198	3.4
200 second analysis time	**24**	5,768	90	1.6

* Data courtesy of M van Alphen.

Given the advantages of a greater number of different instrument soil analyses with the shorter time period, and the (slight) associated increase in registered analyte concentration variability, it was decided to

utilise the 60 second sampling time for soil samples, as suggested in the operator's handbook (TN Technologies, 1994). This was also commensurate with the limited time availability of the instrument.

A13.3 DEPTH OF PENETRATION/SURFACE READINGS

An initial trial with a standard Pirie soil of concentration ≈ 1,200μg.g^{-1} lead with various quantities of soil in the sample holding cup reflected the following values (Table A13.3).

Table A13.3 XRF Instrument surface soil penetration.

		XRF-DETECTED SAMPLE LEAD CONCENTRATION			
SAMPLE SIZE	**n**	**SAMPLE DEPTH (mm)**	**MEAN LEAD CONCENTRATION (μg.g^{-1})**	**SAMPLE STD DEVIATION**	**RSD (%)**
One gram	**31**	5.0	1,281	42	3.3
Two grams	**29**	10.0	1,248	46	3.7
Three grams	**35**	15.0	1,341	43	3.2

This would suggest that the range of the x-ray beam's penetration was of the order of one to two centimetres (as also M. van Alphen, pers. comm.), reflecting, in the field, a "surface soil layer" as Body et al. (1985). This would acquaint well with a typical one to two centimetres surface soil scraping (TN Technologies, 1994) and is interpreted, in the first instance, as a reasonable estimate of the "available" component of surface soil cover, ie that component that is capable of entering homes as wind blown dust (Body, 1986d; Davies, 1988).

A13.4 INSTRUMENT PRECISION AND ACCURACY

From a series of multiple replicate sampling ($n \geq 30$) of known standard soil samples utilising a sampling time of one minute, instrument precision was estimated, resulting in a mean RSD value of 3.4% for lead, 2.8% for zinc, 2.3% for iron, and 18.3% for copper. Analogous, in the field replicate analyses ($n \geq 14$) reflected a mean RSD value of 3.2% for lead, 3.8% for zinc, 5.3% for iron, and 7.5% for copper.

Both the specified and the XRF-tested (arithmetic) mean for various elements in the above-mentioned NIST standard are indicated in Table A13.4 below.

Table A13.4 XRF Instrument standard soil comparison.

ELEMENT	TEST FOR INSTRUMENT ACCURACY USING NIST STANDARD (All 60 second readings) SPECIFIED CONCENTRATION ($\mu g.g^{-1} \pm 1$ SD) and RSD (%)		TESTED CONCENTRATION ($\mu g.g^{-1} \pm 1$ SD) and RSD (%)	
Arsenic	626 ± 38	(6.1)	688 ± 133	(19.3)
Copper	2,950 ± 130	(4.4)	2,989 ± 225	(7.6)
Iron	33,800 ± 1,000	(3.0)	34,430 ± 1,822	(5.3)
Lead	5,532 ± 80	(1.4)	5,506 ± 174	(3.2)
Manganese	10,100 ± 400	(4.0)	10,529 ± 1,098	(10.4)
Zinc	6,952 ± 91	(1.3)	6,889 ± 265	(3.8)

Comparison of the standard's specified values with XRF-tested results, reflects good agreement, particularly for elements such as lead and zinc. Overall, the coefficient of correlation between specified and tested is (r =) 0.996. A Wilcoxon signed rank test revealed no statistical difference between distributions ($P \geq 0.05$), although it should be noted that the number of paired samples (six) was low. Given that lead and zinc are the principal concentrates smelted at Port Pirie, and noting their low RSDs above, the data corresponding to, principally, these two elements would appear to be the most appropriate to be interpreted for soil contaminant studies. Despite somewhat larger coefficient of variation values for iron and copper, these data are still valuable, and are used particularly in a comparative capacity.

APPENDIX FOURTEEN HANDPIECE AND TUBING DUST RETENTION

A14.1 ESTIMATE OF HANDPIECE DUST RETENTION

In order to estimate the quantity of dust adhering to the internal bore of the smooth-bore stainless steel tubing sampling handpiece, some preliminary investigation was undertaken. Two handpieces (attached to a vacuum pump and cassette apparatus) were used to collect a number of samples, subsequent estimates of the quantity of dust and lead retained by the handpieces and any handpiece dust-retaining differences were arrived at. Each handpiece was used in a normal dust-sampling mode with one vacuum pump as described above, in matched and varying locations (deriving from carpet, linoleum and wood surfaces) in five of the available vacant houses.

To assess handpiece differences, each handpiece was tested by sampling one cassette in paired adjacent locations before (handpiece) cleaning, this being conducted for ten separate house/room locations. Subsequently, each handpiece was used to sample five consecutively-sampled cassettes from sets of paired adjacent locations with no handpiece cleaning between individual cassettes. This was conducted for five separate locations, with handpieces cleaned only after the five cassettes had been sampled. Thus a total of 35 cassettes per handpiece were sampled, all by one operator, a summary of the results being presented below (Table A14.1).

Table A14.1 Handpiece dust and lead retention comparison.

		GEOMETRIC MEAN LEVEL		
HANDPIECE	n	RETAINED DUST (mg)	DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)	RETAINED LEAD (μg)
A	15	Mean = 0.27 Range = 0.1 - 0.6	Mean = 1,663 Range = 400 - 14,000	Mean = 0.46 Range = 0.12 - 1.51
B	15	Mean = 0.29 Range = 0.1 - 0.6	Mean = 1,765 Range = 550 - 12,401	Mean = 0.50 Range = 0.11 - 1.24

Comparison (matched pair *t*-test) of the resulting handpiece dust (gravimetrically determined to 0.01mg; Mettler Balance, Model AE 200/5, Mettler, Switzerland) and lead retained (ICP AES), and also handpiece-retained dust lead concentration revealed no difference for any of these parameters between the

two handpieces ($P \geq 0.20$). Further, quantities of dust and lead retained and dust lead concentration expressed with respect to the quantity of dust and lead retained by the associated cassette(s) were also compared and found not to be different ($P > 0.15$).[19]

Acknowledging minimal difference between the two handsets regarding dust/lead retention, data for the two handpieces were thus subsequently grouped in order to increase sample size for estimation of overall handpiece dust/lead retention, a summary appearing below (Table A14.2).

Table A14.2 Summary handpiece dust retention.

		GEOMETRIC MEAN LEVEL		
RETAINED DUST	**n**	**RETAINED DUST (mg)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**RETAINED LEAD (μg)**
In handpiece	**30**	Mean = 0.28 Range = 0.1 - 0.6	Mean = 1,713 Range = 400 - 14,000	Mean = 0.48 Range = 0.1 - 1.5
In cassette *	**70**	Mean = 339.1 Range = 3.4 - 2,019.1	Mean = 1,696 Range = 395 - 18,450	Mean = 575.1 Range = 62.7 - 3,029
Handpiece/Cassette (%)		Mean = 0.19 Range = 0.01 - 2.9	-	Mean = 0.17 Range = 0.02 - 2.2

* No difference between the data corresponding to dust retained by these (test) cassettes and the entire suite of (subsequently) sampled cassettes from the vacant houses (n = 578) exists for any of the dust parameters ($P \geq 0.25$).

Dust and lead retention by handpieces is small by comparison to that which is collected (contemporaneously) in cassettes. Further, regressing handpiece against cassette for collected dust lead concentration produced a coefficient (r) of 0.82, implying that the dust collected in the handpiece is similar to that collected in the cassette.

No difference was apparent for any of handpiece-retained dust or lead, or also handpiece dust lead concentration between occasions when a single test cassette was sampled through a handpiece followed by handpiece cleaning, and when multiple cassettes (five) were sampled through the handpiece with no cleaning of the handpiece between separate cassettes ($P > 0.20$). The inference from this is that the inner bore of the handpiece becomes saturated with adherent dust fairly quickly (or not substantially covered at all), and that washing the handpiece between sampled cassettes would appear unnecessary.

[19] An initial inspection of the data (frequency histograms and box and whiskers plots) corresponding to dust retained and lead retained by handpiece and cassette, revealed skewed distributions, log transformation of the data producing distributions which more resembled normal distributions. Hence data were natural log-transformed before interpretation and again tested for normality of distribution. Data corresponding to the length of tubing exposed to dust when sampling appeared normally distributed and were not transformed.

A14.2 ESTIMATE OF PLASTIC CONNECTING TUBING DUST RETENTION (PVC versus Tygon® tubing)

The pieces of tubing used here were initially attached to both filter and sampling wand, and exposed to air and dust flow by sampling dust in a normal fashion from a number of areas of floor, again incorporating carpet, linoleum and wood surfaces. Tubing was exposed to sampling conditions equivalent to that appropriate to sampling two and one half cassettes (actually five consecutive sweeps of one area, each perpendicular to the former), five and also ten cassettes.[20]

Subsequent to sampling completion, the tubing was carefully removed from attachments, and pre-cut pieces of circular masking tape attached at either end. Tubes were then placed in clean five millilitre screw-top containers (Disposable Products, Technology Park, The Levels SA), and taken back to the laboratory, where they were carefully cleared of the masking tape and weighed to 0.01mg. Tubing bores were then carefully washed internally with a jet of ethanol from a fine-tipped squeezy bottle, the contents (≈ five mls) being collected in an identical sealable container, and subsequently analysed for lead content by ICP AES. Tubing pieces were then allowed to dry, and were subsequently weighed again, in order to facilitate a gravimetric estimate of (previously) contained dust. Whilst care was taken at all stages, it is acknowledged that the potential for some losses of dust exists, although it would appear from microscopic examination, that quite some force was required to dislodge particles of dust that were adherent to the tubing, ethanol washing, however, being very effective.

Summary data corresponding to dust and lead retained by tubing are as follows (Table A14.3).

Table A14.3 Various tubing type dust/lead retention.

		GEOMETRIC MEAN LEVEL		
TUBING TYPE	**n**	**RETAINED DUST (mg)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**RETAINED LEAD (μg)**
PVC	**27**	Mean = 0.37 Range = 0.05 - 2.1	Mean = 3,249 Range = 200 - 15,600	Mean = 1.21 Range = 0.14 - 7.8
TYGON®	**27**	Mean = 0.37 Range = 0.10 - 1.3	Mean = 2,610 Range = 400 - 42,000	Mean = 0.96 Range = 0.21 - 4.5

There would appear to be little retained dust difference between the tubing types, although retained lead appears noticeably lower for the Tygon® tubing. Retained dust lead concentration is also lower for

[20] The choice to test 2.5, 5 and 10 cassettes per plastic tubing piece is based upon the presumed range of numbers of cassettes to be sampled per room, and hence before a tubing change was to be made.

the Tygon® tubing, although, from these data, no indication exists to suggest that the quantity of dust or lead building up in the bore of the tubing varies between tubing types ($P \geq 0.20$). Additionally, comparison between the PVC and the Tygon® mean values for dust and lead retained/unit length of tubing also reveals no real difference ($P \geq 0.10$).

For both plastic and Tygon® tubing (separately and collectively), the quantity of tubing-adherent dust and lead is also not related to the number of cassettes sampled with that piece of tubing ($P \geq 0.05$). For example, comparing tubing that has been used to sample the equivalent of 2.5 cassettes with that which has been used to sample ten cassettes, no difference exists for any of dust or lead retained, retained per piece of tubing, or per piece of exposed tubing ($P \geq 0.05$). This is also the case for the comparison between five and ten cassettes being sampled per piece of tubing.

From such analysis, there would appear to be no difference between either the PVC or the Tygon® tubing with regards to differences in ability to retain either dust or lead, when used with this sampling technique. Again, in this instance, this may reflect that the variability in dust collected by this technique from various locations may mask dust adherence differences between the tubing types, although such differences would appear negligible given the mean quantities of dust and lead retained in the tubing compared to that which is actually collected in the sampling cassette.

It would seem likely that the bore of the tubing would become coated with dust particles at an early stage of dust sampling (probably during sampling for the first cassette, possibly within a few sweeps of the wand - certainly visually this appeared to be the case) and would (soon) reach a stage of "saturation", little further extra dust adhering.

Regressing a number of parameters for each tubing type against others is summarised below (Table A14.4).

Table A14.4 Summary of dust and lead retained regression relationships.

DEPENDENT VERSUS INDEPENDENT VARIABLE	CORRELATION COEFFICIENT (r =) PVC Tubing (n = 27)	CORRELATION COEFFICIENT (r =) Tygon® Tubing (n = 27)
Dust retained versus lead retained	0.45	0.32
Dust retained versus tubing length exposed	0.21	- 0.16
Lead retained versus tubing length exposed	0.02	- 0.20
Dust retained versus number of cassettes	- 0.12	- 0.13
Lead retained versus number of cassettes	- 0.24	- 0.18

Only retained dust versus lead correlation coefficients were notable, the remainder being moderately to poorly defined ($P \geq 0.10$). A moderate correlation exists between the quantity of lead and the quantity of dust retained in pieces of PVC tubing ($P < 0.025$) but not to such an extent for Tygon® tubing ($P < 0.10$). The corresponding correlation figure (r) for the quantity of lead and dust contained in all the cassettes sampled as part of the Static Dust Survey (Chapter Seven; n = 528) is 0.88 ($P \leq 0.001$), and that for depositing dust collected in alfoil collectors (Chapter Eight; n = 359) is 0.86 ($P \leq 0.001$).

There exists no real relationship between the quantity of dust/lead retained with (separately) the length of tubing exposed and the number of cassettes sampled. These data imply that there exists quite some variability with respect to the composition of the dust adherent to the inner bore of the connecting tubing, presumably reflecting the variability in dust being sampled from floor areas.

Given that the sampling for this preliminary testing took place in the suite of vacant houses, it could be inferred that that population of dust adhering to the inner bore of the tube is not necessarily a random sub-population (and thus not representative) of the entire contents of cassette-collected dust. This could be concluded given that the larger-sized particles passing through the tube would probably be less likely to become adherent, although no differences (matched pair *t*-test) between dust lead concentration of plastic tubing and associated cassette contents as also Tygon® tubing-adherent dust and associated cassette contents was apparent ($P \geq 0.20$). It is also possible that in the process of assessing the retention of dust in the two types of tubing, losses of dust could be occurring.

Acknowledging the work of others in this area (eg Que Hee et al., 1985, who conclude to the contrary), the results here imply that there is no detectable difference regarding dust/lead losses to the connecting tube between PVC and Tygon®. Comparison of the dust lead concentration of retained dust between the two tubing types also revealed no difference here ($P \geq 0.50$).

The mean loss of (respectively) dust and lead to all tubing tested is 0.37mg and 1.08µg, and that to sampling handpieces is 0.28mg and 0.48µg; mean total losses to both amount to 0.65mg dust and 1.56µg lead respectively. Compared to the mean contents of all cassettes sampled for this validation study (339.1mg dust and 575.1µg lead), losses to handpiece and tubing lie at less than 0.3%.

Compared to the static dust survey (mean collected dust and lead = 130.93mg dust and 556.35µg lead respectively; n = 528) again it would appear that such collecting implement-elicited losses as investigated here are a minor source of error with this sampling technique (both < 0.5% of static dust cassette suite dust and lead contents). Que Hee et al. (1985) report an (arithmetic) mean of five mg dust adherent to the walls of the Tygon® tubing. This constituted less than 20% of the cassette-collected dust, and suggested a greater than 80% sampling efficiency. From the work conducted above, greater sampling efficiencies have been arrived at, although from the work of Que Hee et al., it remains unclear as to what length of tubing was exposed to the air and dust stream. Effort was directed by this researcher in order to keep that length at a (practicable) minimum at all times.

Subsequent to this preliminary research, for the main body of research, new pieces of Tygon® tubing were used for each room of a house with a maximum of approximately ten cassettes sampled per piece of tubing.

Future work would seem to suggest (similarly) sampling from new carpet which has been previously contaminated under control conditions with a known sample of homogenous (sieved) dust, in order to determine if subtle differences between tubing types can be detected under such circumstances.

APPENDIX FIFTEEN STATIC DUST COLLECTION VALIDATION STUDY

A15.1 PRELIMINARY OPERATOR COMPARISON

Initial inter-operator comparison was conducted by controlled (vacuum pump and cassette) sampling of freshly-contaminated new carpet material. Two pieces of new carpet (one short and the other medium pile) were laid out flat on a clean surface, as if the carpet was laid on a floor. The upper surfaces of both pieces of test carpet were subdivided with masking tape into ten standard sampling areas (0.093 square meters), demarcated into two rows of adjacent strips of five sampling areas. Each test area was contaminated with 4.7gm of the < 32μ fraction of one of the dust standards, demonstrating a mean lead in dust concentration of 4,388 $\mu g.g^{-1}$ (n = 25 replicates). Such a dust loading was chosen in accordance with a level proffered by Roberts and Ruby (1989) as a representative carpet dust loading for an average household, there being no available analogous figure for Port Pirie homes. The sub-32μ size fraction was chosen in order to enhance dust homogeneity, the lead in dust concentration being chosen to lie proximate to those of some preliminary dust samples acquired from two condemned vacant Port Pirie houses.

Dust was evenly sprinkled over the test area, with remaining areas protected from contamination. Dust was artificially embedded into the carpet test area using a 150 x 25mm polished wooden roller and handle (resembling a painting roller) similar to the American Society for Testing and Materials (ASTM) procedure F608-86 (ASTM, 1986), in order to simulate normal carpet dust embedding. Subsequent sampling of this test area by one operator was followed by the contamination of a second test area, and its sampling by the other operator. Each operator sampled five separate test areas of each carpet type, sampling from one of the linear arrays of five adjacent test areas. From the results of such sampling as presented below (Table A15.1), there would appear to be little contribution to sampling variability due to operator sampling efficiency differences.

Table A15.1 Preliminary operator sampling efficiency comparison summary.

	MEAN COLLECTED DUST PARAMETER		
OPERATOR	**DUST COLLECTED (gm)**	**DUST LEAD CONCENTRATION ($\mu g.g^{-1}$)**	**LEAD COLLECTED (μg)**
A	1.54	6,624	4,336
B	1.42	6,260	4,397
***t*-test comparison**	No difference (P > 0.225)	No difference (P > 0.35)	No difference (P > 0.15)

A15.2 VACUUM PUMP SWEEP DUST COLLECTION EFFICIENCY

A15.2.1 SAMPLE SWEEP RELATIVE EFFICIENCY

In order to estimate the proportion of dust and lead taken up from floor surfaces by the vacuum pump/cassette system, (essentially determining the sampling efficiency of repeat passages of the sampling wand), multiple sweep samples were acquired from various (undisturbed) approximately central floor locations in test houses.

Given that greater than 70% of main rooms of the test houses were carpeted, and that the remainder were either linoleum-covered or wooden, seven carpeted locations (various pile length and condition) and three bare floor locations (wood & linoleum) were chosen for repeat sampling.

Sampling was conducted under exactly the same conditions as indicated in Appendices Nine and Ten, other than that each of the ten selected areas was sampled with five consecutive single sweeps of the sampling wand (each perpendicular to the former), a separate cassette being used for each sweep.

From the total of 50 cassettes, the following summary data is presented (Table A15.2).

Table A15.2 Successive sampling sweep dust collection efficiency.

	REPEAT PASSAGE DUST AND LEAD COLLECTED (Cumulative Proportion - %)					
	ALL FLOOR SAMPLES		CARPETED FLOORS		WOOD/LINO FLOORS	
SWEEP NO.	DUST	LEAD	DUST	LEAD	DUST	LEAD
One	47.5	61.5	51.3	68.4	38.5	46.4
Two	66.5	78.6	69.9	84.8	58.2	64.8
Three	81.2	90.5	81.0	91.7	81.2	88.6
Four	91.1	95.4	91.3	96.2	90.5	94.0
Five	100	100	100	100	100	100

The bulk of the dust and lead available by repeat passaging of the sampling wand over a single floor surface is acquired within the first two to three sweeps with the sampling wand. Given that most surfaces to be sampled are carpet, and from the table above, almost three quarters of the dust and close to 85% of the lead acquired by five sweeps is collected by the first two sweeps, it was decided to employ two sweeps per sampled floor area as the standard sampling procedure.

A15.2.2 SAMPLE SWEEP ABSOLUTE EFFICIENCY (RECOVERY)

The absolute sampling efficiency of the test equipment (ie the equipment recovery efficiency) was estimated by sampling dust from surfaces (0.093 square metres) subsequent to a known quantity of test dust having been applied to such a surface. Surfaces included carpet, wood and linoleum, the carpet being new (uncontaminated) medium-pile carpet (two different types), the wooden and linoleum surfaces deriving from test house floor areas that had been vigorously cleaned before sampling commenced.

Prior to the addition of contaminant dust, surfaces were sampled with two sweeps of the sampling equipment, all demonstrating less than five mg dust per swept area, and less than 25µg lead. Each test area was contaminated with 4.7gm of a < 32µ size fraction from a test dust, the concentration of which was 4,388µg.g^{-1}. Dust was again artificially embedded into/onto the test surface by using the test roller referred to above (Section A15.1). Collection efficiency was estimated after two sweeps of the sampling wand over the surface, three sweeps, and also after five sweeps, by changing sampling cassettes.

A total of twenty surfaces were sampled, ten carpet pieces (five of each type), five wooden and five linoleum-covered floors. The results of such sampling are presented in Table A15.3 below.

Table A15.3 Different floor surface sampling absolute dust recovery efficiency.

	REPEAT PASSAGE DUST AND LEAD RECOVERED (Cumulative Proportion - %)					
	ALL FLOOR SAMPLES		CARPET PIECES (n = 10)		WOOD/LINO FLOORS (n = 10)	
SWEEP NO.	DUST	LEAD	DUST	LEAD	DUST	LEAD
Two	50.8	55.1	46.7	50.2	54.8	59.9
Three	66.5	66.9	56.3	53.1	76.8	80.6
Five	77.0	72.7	67.7	62.8	86.3	82.6

Sampling from hard surfaces is more efficient than is the case for carpet, the latter presumably retaining more of the applied dust due to more pronounced surface irregularities. As a means of estimating the quantity of carpet-contained dust on floor surfaces, the sampling system chosen for this work clearly produces meaningful results, this also being the case for hard floor surface types.

APPENDIX SIXTEEN POTENTIAL DUSTFALL COLLECTION APPROACHES

From the literature, dust deposition assessment appears to have been conducted in a number of ways, reflecting the absence of an internationally-standardised method. There exists a British Standard Gauge (funnel and flask) for external dust collection (Al-Radday et al., 1993b), a discussion of its relative appropriateness for use is entertained below. Potential alternative techniques for such assessment are also briefly reviewed below, the finally-devised alternative being also included.

A16.1 ALTERNATIVE DUSTFALL-COLLECTING METHODS

A16.1.1 PLASTIC TRAYS

As indicated in the literature (Angle and McIntire, 1979; Aschengrau et al., 1991), plastic, sealable Tupperware-type containers of rectilinear profile or cylindrical plastic petri dishes (Gulson et al., 1995) are placed in a variety of locations to collect depositing dust in homes. At cessation of collection period, sealing lids are used, and contents transported to the laboratory.

A16.1.2 GREASED GLASS PLATES

As Brunekreef et al. (1981) and Diemel et al. (1981), glass plates approximately 25.0cm square are lightly coated with grease, typically silicone-based. After exposure, plates are placed inside a sealable plastic container, and transported to the laboratory. Similarly, greased petri dishes have also been utilised (Little and Heard, 1978; Diemel et al., 1981).

A16.1.3 DOUBLE-SIDED TAPE

As used by a number of researchers (eg Lepow et al., 1974), pieces of accurately-cut double-sided tape are exposed for a certain time period and are subsequently removed and placed into a sealable plastic container for subsequent laboratory analysis.

A16.1.4 PLASTIC/METAL TRAY AND VACUUM TECHNIQUE

Whilst not appearing in the literature, an initial proposition was to utilise a rectilinear plastic/metal tray exposed for a certain period of time to collect dust, and subsequently the sample could be "vacuumed" using the vacuum pump and cassette/filter apparatus used in the previous section dealing with static dust collection.

A16.1.5 FUNNEL AND FLASK SAMPLING (STANDARD DEPOSIT GAUGE)

A commonly-used technique as used by Brunekreef et al. (1981), Sweetapple (1984) and West (1986) for (typically) external dustfall, is a glass/plastic funnel in either a measuring cylinder or glass/plastic sealable bottle (BS 1747 standard deposit gauge). From the literature, such a technique has been used at a variety of locations.

A16.1.6 CUP OR JAR SAMPLING

As Aurand et al. (1983), Seifert et al. (1984), Dolan et al. (1986) and Bernigau et al. (1993) this method employs poly-styrene (coffee) cup collectors, and has been used by a number of researchers, essentially representative of a more standard deposit gauge (Dolan, 1987). Sedimentation jars have not been frequently used for dustfall purposes (Dolan, 1987), only one example of their use in a Cincinnati study having been reported (Cholak et al., 1968).

A16.1.7 MOSS BAG SAMPLING

As initially proposed by Goodman and Roberts (1971) using *Hypnum cupressiforme*, and more latterly by Al-Raday et al. (1993), this alternative incorporates the use of pieces of moss suspended in air in rooms of houses.

A16.1.8 ALUMINIUM FOIL SHEET

A small piece of adhesive-coated aluminium foil attached to a secure base is exposed for a certain time and then placed in a sealable plastic container and transported to the laboratory (as Hogan, 1971; Schaefer et al., 1972).

A16.2 ADVANTAGES/DISADVANTAGES OF THE VARIOUS TECHNIQUES

Clearly, all techniques would not provide the facility of accurately gravimetrically determining the quantity of dust collected. The tray and vacuum pump technique may provide some indication of the quantity of dust collected, but it is expected that error would be reasonably high.

Problems with the sealable plastic tray technique would appear to be related to the fidelity of the collected-sample recovery procedure and uncertainty regarding the bias of the collection system due to electrostatic forces generated by handling and location on (say) carpeted surfaces (Teege, 1968). The potential for loss of sample upon opening of the tray would be difficult to estimate.

Concerns with the time-consuming requirements (preparation and recovery) for the greased glass plates, and sample losses during harvesting and transport are apparent. Further concerns focus upon the possible falsely high levels of dust collection in that dust which has settled on the surface may be trapped by the grease layer, thus precluding subsequent re-entrainment which may be a normal component of (within-house) dust deposition. Coated surfaces providing a "trapping" function have been reported by other researchers (eg Lawson, 1980).

Double-sided tape was discounted on the basis of doubts regarding the method's potential lack of sensitivity given the small practical surface area to be exposed, and concerns regarding accurate gravimetric quantification of the collected dust load.

The time-consuming nature of the tray and vacuuming technique together with the associated cost for filters and pumps etc would need to be considered, as well as the technique's accuracy of estimation of sample quantity, given the anticipated losses of sample in the vacuuming process.

The funnel method would require some on-site post-collection sample preparation (viz funnel flushing and eluting agent collection), and would be labour-intensive and time-consuming particularly for multiple sampling locations in a house. Further, manipulating a number of these could become somewhat clumsy in confined spaces and would have a cost disincentive. Given the bulky nature of the collection equipment, concerns with the fidelity of this process for internal use with expected low dustfall rates arose. Concerns also arose regarding the small surface area available for dust capture (Dolan, 1987). Turner (1980) cites concerns with these devices for selective re-entrainment of certain particle size ranges.

Although plastic/foam cup collectors have been used elsewhere, concerns arose regarding the small surface area, the relative sensitivity of the technique compared to larger surface area collectors and collectors' propensity to demonstrate bias toward dust attraction/repulsion due to electrostatic forces generated by handling of the cup material as also their location on carpeted surfaces (Teege, 1968). These concerns associated with this dust-collecting technique appeared sufficiently large so as to contra-indicate the use of these devices. Jars appear to be also afflicted with small surface area, and uncertainty arises regarding the bulky nature of the samplers, and the potential for subsequent analytical error.

Concerns with the use of moss bags related to their potential for drying out and losing collection efficiency, (particularly given the sampling location), accuracy, and also potential for error upon analysis.

The final alternative appeared to be close to suiting the requisites of the research, but the need to utilise a six-digit balance (under appropriate operating conditions) for gravimetric determination and concerns

regarding transport of exposed collectors for weighing precluded its choice. It also appeared that the sensitivity of the technique might be somewhat limiting given the small sampling size of the collectors, thus warranting excessively long exposure times indoors. Further, as with the pre-treated glass plates, concerns arose with the use of a surface treated with an adhesive in that such a technique might positively bias the collected sample, since, subsequent to "capture", no possibility for (normally-occurring) collected dust re-entrainment in the room and deposition elsewhere would be presumed to be likely.[21]

Regarding the gravimetric determination of collected dust, in a general sense, any technique which involved the collection of dust with either water or a solvent prior to gravimetric determination would appear to elicit concerns regarding the loss of some soluble material, and complicate the determination of collected dust mass.

A16.3 THE CHOSEN TECHNIQUE

The technique finally devised to estimate dustfall in vacant houses in Port Pirie incorporated the use of pre-weighed pieces of heavy-duty aluminium foil.

This technique was selected in order to provide a sufficiently large collection surface area to facilitate reasonably frequent and (presumably) reliable gravimetric estimates of dustfall in a manner where concerns arising as a result of having to transport samples to and from the field were minimised. If the foil trays were any larger, it appeared that handling of the trays in the laboratory to remove and harvest the dust present (Appendix Nineteen) would become somewhat more difficult due to the unwieldy nature of larger trays. Significantly smaller trays might have not provided sufficient surface area to facilitate dustfall collection over the time periods chosen, and if exposure time had been too small, concerns might arise regarding the technique's ability to accurately assess the mass of fallen dust. Essentially such concerns regarding poor collector sensitivity appeared relevant when considering the use of any small surface-area collectors. Unlike Schaefer et al. (1972), trays were not coated with any dust adhesive, since this practice elicited concerns regarding the interference of the coating with subsequent chemical analysis (Lawson, 1980), and also since such a collector was chosen to mimic dust deposition patterns on surfaces in houses, none of which would normally display attached adhesive.

The selected technique would appear to be effective and probably the most expeditious of all alternatively-mentioned methods. Further, it would appear to most conveniently provide a reasonably large surface area and to be the simplest and cheapest alternative. Additional advantage was apparent, in that this technique most-accurately reflected the total dust mass loading per unit area, since, once pre-cut, prepared and pre-weighed, the pieces of aluminium foil could be laid out in the appropriate location very

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[21] Schaefer et al. (1972) report substantially greater mean values for dustfall in occupied homes in both window-open and closed formats, respectively, these data being approximately eight and two times greater than data presented here. This may represent increased dust retention by adhesive once the dust has deposited, and also/or alternatively the added contribution by the occupants of the homes by way of re-suspension/introduction of dust. Clearly, given Schaefer et al.'s (1972) smaller dustfall difference between window closed and open alternatives than this work's results, this implies that the contribution to dustfall by occupants is not insignificant.

easily, and the total dustfall sample could theoretically be easily and (virtually) completely collected.[22] As Hogan (1971) notes, the long-term stability of alfoil is helpful with respect to the gravimetric determination of the quantity of dust collected.

Throughout handling of the alfoil material, hands were kept scrupulously clean, and the collection surface handled minimally. In the field, before each tray set-up/collection, hands were extensively wiped with non oil-based hand-wipe towlettes (Wet Ones - Reckitt and Colman, West Ryde, NSW), and allowed to dry. Again, minimal hand contact with the collection surface area of the foil took place on set-up, and (scrupulously) none at tray collection time or subsequently. At no time during set-up and collection of dust-collection devices was hand contact made with any room surfaces.

A16.4 ALFOIL TRAY DETAILS

The dimension of these foil pieces was ≈ 38.2cm x 56.0cm, with the same face of the foil always being used as the collection surface.[23] All pieces were (loosely) pre-folded identically, and then weighed on a balance demonstrating a (rated) readability of 0.00001gm (Sartorius RC 210S, Sartorius, Goettingen, Germany - Selby Scientific Ltd, Notting Hill, VIC). Three separate rolls of alfoil were used throughout this work, the (pre-use) weight of the pre-folded alfoil trays lying between 9.2 and 10.1gm, the mean weight being 9.5g (n = 399).

Such collectors were subsequently transported to Port Pirie in a clean dust-free environment, and when appropriate, opened and folded into a clean wooden former producing a raised-edge alfoil tray measuring 30.5cm x 45.8cm, with five cm raised edges, displaying an effective dust-collecting surface area of 0.139 square meters. Immediately following shaping and removal from the wooden former, the tray was placed onto clean paper taped to the floor just prior to alfoil shaping, the collector being lightly attached to the paper with four small pieces of masking tape. Subsequent to exposure, the tray was carefully folded together in the same pattern as originally folded, the tape was carefully removed and the trays stored in a clean dust-free environment ready for transport back to the laboratory.[24] On return to the laboratory, the collected "package" was removed from the plastic bag, and again weighed, still containing the sample within the foil. A description of the conditions associated with the gravimetric determination of dust collectors is referred to in Appendix Three. Throughout preparation of the alfoil material, work was conducted in a dust and draught-free area, hands were kept clean, and the dust-collection surface was not touched or contaminated.

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[22] It is acknowledged that a problem with this aluminium foil sampling technique may occur when floor sampling locations are chosen in high traffic areas within occupied house situations, due to the possibility of sampling device disturbance by normal house occupant activity. Certainly this is also potentially likely with other sampling devices, but given the fairly large size of these alfoil collectors, they may be more likely to be interfered with.

[23] The width of the alfoil pieces was the width of the roll utilised which varied slightly, however, the length also varied slightly according to the cut of the sheet. Typically though, the length varied by not more than one cm.

[24] Clean pre-weighed, re-sealable plastic bags batch-tested for their initial lead content were used as transfer containers for samples. Identical bags were used to transport the alfoil collectors back to the laboratory after the collectors had been exposed to depositing dust. Subsequent to return to the laboratory, the collectors were removed and the bags were again weighed to assess for dust leakage from alfoil containers. None was apparent ($P > 0.25$). Acid washing of the inside of the bags revealed undetectable quantities of lead present.

A16.5 ALTERNATIVE (PAPER CUP) TECHNIQUE

The paper cups also used for depositing dust collection stood 17.5cm tall, and had a base diameter of 6.2cm tapering outwards to a diameter at the top of nine cm (Polar Cup, Netley, SA). One of these cups was placed adjacent to an alfoil tray at all selected alfoil tray sampling locations, cups being exposed to the atmosphere for the same period of time as the alfoil trays. The effective collection surface area of these cups was assumed to be the surface area of their opening, namely, 63.6cm^2, or 4.55% ($^1/_{22}$) of the surface area of the alfoil trays, this again being assumed to be the surface area of their opening. It was decided that gravimetric determination of dust collected in paper cups would be fraught with opportunities for error, and elicit a number of technical problems, such that such determination was not performed, the dust collected being analysed for total lead content only.

A16.6 INITIAL METHOD VALIDATION

Initial experiments were conducted to examine different folding regimes for the alfoil material, and also the positioning of the folded alfoil collector on the balance tray with respect to variations in the recorded weights of the alfoil collectors. Despite results reflecting that there existed no difference statistically between weights of differently-folded or differently-arranged (on the balance) alfoil trays ($P \geq 0.20$), all trays were folded identically, and placed on the balance.

Prior to the commencement of the first dust collection round, control alfoil dust-collection trays were prepared identically, and then treated as follows:

a) Initially, ten prepared alfoil trays were transported to Port Pirie, opened, shaped in the wooden former, placed on and attached to paper with masking tape, and then folded together, the tape removed and the trays returned to Adelaide. Subsequent to weighing, no difference in weights before and after was recorded ($P \geq 0.10$). This allayed concerns regarding the potential for handling of the trays and residual adhesive tape mass confounding the second weighing of the trays.

b) For the duration of the exposure period, 20 control trays were prepared identically to test trays, and exposed similarly, but completely encased within a large clear plastic bag with small holes punched in the bag peripheral to the exposure surface of the alfoil tray. Masking tape was used to attach ten trays to a paper base, the other ten trays not being taped to the paper base. These holes facilitated atmospheric exchange.

Subsequent to the completion of the exposure period for the trays, the bags were carefully opened, and the trays folded together still inside the plastic bag. For those trays where masking tape was used, it was

removed, all bags then being returned to the laboratory for weighing, at which time no difference in tray weights before and after the exposure period was recorded for either test group ($P \geq 0.10$). These results indicate that over the period of tray exposure, no significant residue of adhesive tape occurred, and also no detectable corrosion of the trays took place.

Trays were also analysed for lead contained on their collecting surfaces, such tray mean levels being less than 20μg lead. Thus, as a field control procedure for test trays subsequently, encasing opened trays in plastic bags was the chosen technique.

c) Ten alfoil trays were set up in test houses in a conventional fashion, and exposed for depositing-dust collection. At time of collection, trays were carefully folded in half along their longitudinal axis, and then their external (non-collecting) surfaces were carefully washed utilising a filled 500ml plastic fine-tipped squeezy bottle containing one percent (≈ 0.15M) HNO_3. The washing solvent and tray contents (≈ 100ml) were collected in a clean 250ml Pyrex glass bottle with plastic screw lid (Schott bottle) located immediately below the alfoil collector. During this procedure, care was taken to minimally disturb the (inner surface) contents of the alfoil collector. Subsequently, the foil tray was allowed to dry, folded in the normal fashion, and returned to the laboratory for dust harvesting and analysis (analogous to the normal analytical procedure as detailed in Appendix Eighteen), as were the contents of the glass bottles.

The quantity of lead attached to the outside of the tray expressed with respect to the contents of the inside collecting surface demonstrated a mean of less than three percent. Assuming that the dust attaching to the outside of the tray is of the same character as that being collected by the alfoil tray, then as with the quantity of lead collected, the quantity of dust attaching to the outside of the tray will have a minimal influence on gravimetric determination of alfoil tray contents.

d) Collecting trays were set up in the normal fashion, and placed approximately 2.5cm apart in two clusters of eight collectors in the centres of two adjacent identical small (store) rooms within a building located on the grounds of the smelting works. This location was chosen specifically given its proximity to the smelter operations, its known dusty conditions (G. Norton, pers. comm.), and also its availability (courtesy A. Gilbert, BHAS - Pasminco, Pty Ltd).

Each room had one closed sash window and was left undisturbed for the entire collection period. The common (dividing) wall between the two rooms was 2.0 metres high, the ceiling 3.0 metres high. All other walls joined the ceiling. Identical doors were fitted to each room and in the same relative location.

Depositing dust was collected in alfoil trays, and later harvested and analysed. RSD values for each set of eight alfoil collectors registered less than ten percent for dustfall, leadfall and dust lead concentration. Such reasonably low variability between collectors reflects the consistency of the sampling system, and would suggest that the precision of this depositing dust collecting system is not a major limiting factor.[25]

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[25] The presumption here is that there exists little variation in dustfall/leadfall spatially within these small rooms, the floor surface area of which was only slightly larger than the combined surface area of the eight alfoil dust collecting trays.

APPENDIX SEVENTEEN DEPOSITING DUST COLLECTION PROTOCOL

As part of on-going investigations into the contamination by lead-bearing dust of homes in Port Pirie, the sampling protocol described below addresses the regular collection of dustfall in vacant homes initially, although the technique could also be directed to dustfall collection in occupied homes.

A17.1 ATTENTION TO CLEANLINESS

Dustfall collection was initially conducted utilising both alfoil and paper cup collectors. All dustfall collection for all dust-collection rounds was conducted utilising the same type of alfoil material and also waxed paper cup container. The siting for all collector locations chosen for sample acquisition was consistently maintained for all collection rounds. At the time of retrieval of dust-collecting device the researcher ensured that the following did not occur:

a) any leaning over the collector,

b) any touching of the inner surfaces or cup sealing lip,

c) any breathing onto the collector,

d) the making of any unnecessary or rapid movements that might create air currents that might re-entrain dust that could be collected by the collector (or remove dust from them) whilst closing them, or

e) any touching or brushing of clothing or hair, touching of hands on floors or other surfaces and contaminating them, or in any way any stirring up of dust that may be situated on internal room surfaces or personal clothing.

When entering houses to retrieve dust collectors, researchers entered slowly avoiding generating air currents by any sudden movement. Immediately prior to collector retrieval time, hands were wiped with moist, clean hand-wiping towlettes and hands allowed to dry before handling collectors to minimise any accumulated dust from fingers contaminating collectors, although no direct contact between hands and collecting surfaces occurred.

A17.2 PREPARATION AND COLLECTOR SETTING-UP PROTOCOL

The following details collector preparation and setting-up procedures.

A17.2.1 ALFOIL TRAY COLLECTORS

A17.2.1.1 ALFOIL TRAY PREPARATION

Alfoil sheets were cut to length from a roll in a clean dust-free environment and were carefully, slowly and very lightly folded together according to the following pattern .

With the long axis of the piece of alfoil lying transversely in front of the researcher (ie running from left to right) with the collecting surface facing upward, the folding procedure was as follows:

- fold the top third down onto the middle third
- fold the bottom third upwards aligning with the new upper edge of the alfoil
- fold the left third over to the right and then the right third over the top
- fold the top half down onto the bottom half
- fold the left half over onto the right half
- label in the same relative location on the tray's (outer) surface, and store in a clean sealable plastic bag

A17.2.1.2 SITE PREPARATION AND TRAY LAYOUT

Collection sites consisted of a fresh piece of clean paper (≈ 0.65 sq metre) securely fastened with masking tape to the floor of the room immediately prior to tray placement, to which the tray was attached with four small pieces of masking tape lightly applied. This was done in order to stabilise the tray and preclude contact between the base of the aluminium foil and existing contaminated dust on the floor surface. Immediately prior to handling trays, hands were cleaned thoroughly with wet paper towlettes.

The trays were "shaped" using a timber frame which was kept clean at all times, during which time, minimum contact by fingers/hands was made with the collecting surface. Care needed to be exercised such that the aluminium foil was handled very gently, and also so that no damage to collectors took place. When crouching on the floor during set-up and retrieval, no unnecessary draughts were created, and no hand contact with the floor was made attempting to steady oneself.

A17.2.1.3 EXPOSED COLLECTING TRAY RETRIEVAL

When collecting the alfoil, utilising "clean" procedures as outlined above, the folding technique is as follows:

- fold the ends inwards and downwards
- fold the top and bottom edges inwards and downwards
- fold the top and bottom edges inwards and downwards to meet at the central longitudinal axis of the tray
- fold the top edge of the folded tray along the middle of the longitudinal axis of the tray to line up with the bottom edge
- fold the end quarter over from both ends, and then again to meet in the middle
- store in this condition in a clean sealable plastic bag inside a cardboard box, and avoid disturbance to the container

At time of collection, collector-locating masking tape was very carefully removed from the tray without damaging the alfoil.

A17.2.2 PAPER CUP COLLECTORS

Paper cups were removed singly from their protecting plastic sleeve only immediately prior to their placement in collecting locations. Cups were removed from the bottom of the stack in the plastic sleeve, and the bag re-sealed immediately using the bag tie provided. The top cup from the stack of cups contained in the plastic sleeve (ie the cup with its opening unprotected other than by the plastic sleeve) was routinely discarded. Lids were stored in the clean plastic sleeve in which they were purchased, and only removed immediately prior to their placement onto the collecting cup. Lids were removed from the top of the stack, the last lid in the stack (ie with its inner surface exposed within the plastic sleeve) was routinely discarded. No finger/hand contact with the inside surface of the cup or lid occurred, thus precluding the possibility of hand-attached leaded dust contamination of the inner surfaces, or the removal of cup-collected dust adhering to the fingers of the researcher when retrieving cups. At all times prior to use, cups and lids were stored in their original protective plastic sleeve inside a new sealed protecting plastic bag in a safe and clean location.

Cups were placed vertically upright for dustfall collection, securely taped to wooden bases, and placed at appropriate locations on floor surfaces. When taping the cup to its wooden base, the cup was still inside the plastic sleeve in which it was supplied with only its base exposed, until the taping procedure was completed. It was subsequently removed from the sleeve. Both at the time of placement and retrieval of cups, hands were kept clean, and great care was exercised to prevent the contamination of hands and subsequently collecting cups by dust.

When cups were collected after their period of exposure, the lids were carefully and securely attached, with a minimum of compression of either lid or cup. Transport and storage of sealed cups was always in a vertical orientation, opening upright at all times, and attention paid to their stability in the storage container such that they were never subject to falling over or compression. Cups were never stored in direct sunlight or in a hot vehicle as it was considered that excessive heat might contribute to the deterioration of the wax coating.

APPENDIX EIGHTEEN COLLECTED DUST ANALYSIS METHODOLOGY

A18.1 ANALYTICAL PROCEDURES

Analysis of contents of depositing dust collectors is as follows:

A18.1.1 ALFOIL TRAY DUSTFALL COLLECTORS

Alfoil collectors were carefully placed on a fresh clean piece of paper on the bench, and gently unfolded to full length stage with the two width folds still "closed". Utilising a filled 500ml plastic fine-tipped squeezy bottle containing one percent (≈ 0.15M) HNO_3, a clean 250ml Pyrex beaker and broad-tipped stainless steel forceps, the contained dust was carefully harvested from the collector according to the following procedure.

Holding the collector at one end and horizontally, the opposite end of the foil was carefully and progressively opened (above the opening of the beaker) using the forceps and simultaneously, the collection surface being opened was lowered slightly, and gently washed down and into the beaker using the squeezy bottle contents. Progressively, the collector was further unfolded and the collection surface washed. Every attempt was made to avoid draughts, this work being undertaken in a draught-free environment. When fully opened, the collector, held at an angle of approximately 45 degrees to horizontal with the lower end of the collector overhanging the collecting beaker, was rinsed thoroughly with the squeezy bottle contents, the contents of the dustfall collector collecting in the beaker. On the very few occasions that any adventitiously-acquired (collection-period) materials (insects, small stones, visible paint flakes etc if present) appeared, they were carefully removed with the forceps, collected, subsequently weighed, and the net foil-collected weight of material over the collection period for that particular foil collector adjusted accordingly. Typically, this washing would be continued until the beaker contained approximately 150ml of washing medium.

Subsequently, the beaker was placed on a constant temperature heating block at ≈ 90^0C for around one hour with a covering watch-glass. The temperature was then increased to around 130 - 140^0C, the watch glass removed, and the contents brought to a volume of around ten millilitres. Following this, the temperature was reduced to around 90^0C, and the contents brought to virtual dryness, with occasional washes of the walls of the beaker with the contents of the beaker using a glass pasteur pipette.[26] Also, during the higher-temperature phase, a watch glass was placed onto the beaker on two occasions for around an hour each, in order to reduce the quantity of residue adhering to the walls of the beaker. At near-dryness stage, four millilitres of conc. HNO_3 was added, and then, subsequently, six millilitres of conc. HCl, the sample vortexed, and again set at 90^0C for one hour with a covering watch-glass.

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[26] For the purposes of this analytical protocol, virtual or near dryness refers to the bottom of the beaker being allowed to remain wet, with a meniscus just visible, thus reducing the likelihood of any charring of contents (J. Waters pers. comm.). This volume would correspond to perhaps one to two millilitres.

Following this, the temperature was set at between 130 and 140^{0}C for two hours with covering watch glass, and then back to 90^{0}C until virtual dryness was reached. Subsequent to virtual dryness, the beaker was removed from the heating block, allowed to cool, and two mls of conc. HNO_3 were added to set the residue in solution. The solution volume was then made to 20ml with water.[27]

Using this 20ml re-suspension solution, the walls of the beaker were then carefully and copiously washed using a five millilitre micro-pipette and disposable glass tip. Any areas of contained "undissolved" material adherent to the floor of the beaker were vigorously repeat-pipetted in order to maximise the salts of lead entering solution.[28] Allowing the beakers to stand with a covering of Parafilm for approximately an hour at room temperature prior to analysis, was considered necessary, in order to assist with the optimal dissolution of lead salts present. AAS analysis was performed subsequently, with calibration standards of 2, 6, 10, 16, and 25$\mu g.ml^{-1}$ lead deriving from a prepared standard of 1,000$\mu g.ml^{-1}$ lead. If sample concentrations lay in excess of $\approx$ 15$\mu g.ml^{-1}$, a 1:2 - 1:5 dilution of the sample was performed (in ten percent nitric acid), and that dilution analysed, and results compared with the pre-dilution figure. If greater than (approx) 28$\mu g.ml^{-1}$ (out of range), a 1:10 dilution (in ten percent nitric acid) was performed, and the sample analysed. All AAS readings were made in triplicate.

A18.1.2 PAPER CUP DUSTFALL COLLECTORS

The paper cups used for dustfall collection were treated according to the following protocol:

Again, clean bench conditions were practised. The collection containers utilised were 125ml Pyrex beakers. Lids were carefully removed from the paper cups and washed into the beaker again using a 500ml squeezy bottle containing one percent HNO_3. The cup was then rinsed whilst held with the longitudinal axis just below horizontal with the open mouth of the cup overhanging the collection beaker. The cup was slowly rotated whilst the washing process continued, the stream of fluid being directed at the (inner) base of the collecting device initially, and then subsequently at the walls of the cup, the run-off falling into the beaker. A final wash (around 20ml) with a 1:1 water/ethanol solution from a fine-tipped squeezy bottle was performed to assist with the removal of any dust material adherent to the walls of the paper cup. The final volume of the cup wash in the beaker was around 100ml.

Beakers with watch-glasses were placed on a constant temperature heating block at around 90^{0}C for one hour with a covering watch-glass, followed by a temperature of around 130 - 140^{0}C alternating between watch-glass on (around one hour) and removed. The contents were then brought to around ten mls, after which the temperature was reduced to around 90^{0}C, the containers then being brought to virtual dryness. Subsequently, a digest mix of five mls of conc. acid digest mix (2:3 conc. HNO_3/HCl, acids

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[27] Initially, ten samples were quartered subsequent to the above-mentioned alfoil washing step, digested as above and then after heating to virtual dryness, re-suspended in, respectively, 20ml of 2.5%, 5%, 10% and 20% HNO_3, and assessed for lead concentration. Results revealed no difference in lead concentration of these test solutions ($P \geq 0.10$).

[28] Typically, this is a rather slow and tedious process, but it would appear that a potential for error could lie in the ineffective re-suspension of all contained material. Additionally, the possibility of blockage of the AAS input orifice was reduced by breaking up any clumps of material that might have been present.

added separately) was added, the sample vortexed, and again set at 90^0C for one hour with a covering watch-glass. This was followed by a temperature setting of 120 - 140^0C, the contents being brought to around ten mls, after which the temperature was reduced to around 90^0C, until virtual dryness was reached.

Again, during the higher-temperature phase, a watch-glass was used to ensure that a minimum of material was adherent to the walls of the beaker. Subsequent to virtual dryness, the beaker was removed from the heating block, allowed to cool, and one ml of conc. HNO_3 acid was added to set the lead present in solution. The solution was then made to ten mls with clean water.

Using this solution the walls of the beaker were then copiously washed using a five ml pipette and disposable glass tip. Any areas of contained undissolved material adherent to the floor of the beaker were vigorously repeat-pipetted in order to maximise the salts of lead entering solution. Allowing the beakers to stand with a covering of Parafilm for approximately an hour subsequently, was intended to optimise the dissolution of lead salts present. AAS analysis in triplicate was performed subsequently, with calibration standards as above. Again, if necessary, an appropriate dilution of the test solution with 10% HNO_3 was performed if the sample's concentration lay outside the accurate operating range of the AAS.

APPENDIX NINETEEN VACUUM PUMP, ROTAMETER, AND GAS METERS - DETAILS, TESTING AND CALIBRATION

A19.1 PUMP DESCRIPTION, OPERATION AND LOCATIONAL LAYOUT

The vacuum pumps used for this study were Dynavac Brand Model No. OD 1 (Thomas Industries Inc, Sheboygan, Wisconsin), demonstrating a maximum capacity of > 50l.min^{-1} @ -75kPa Hg (Dynavac, 1992).[29] Pumps were run at a flow rate of approximately 4.0l.min^{-1}, as indicated by an attached adjustable (variable orifice) rotameter (0 - 10l.min^{-1}; Series VF Visi-Float; Dwyer Instruments Inc, Michigan City, Indiana).

The instruments used to accurately assess total volume of air flow through each cassette throughout sampling were multi-purpose gas meters (Toyo Model ML2500; Davies Shephard, Caringbah, NSW). A total of eight vacuum pumps and sixteen such gas meters were used for this study. The rated accuracy of the gas meters is well within +/- 1% from 0.01 to 2.0m^3.hr^{-1} (0.17 to 33.33l.min^{-1}) (Anon., 1995b). Bernstein et al. (1983) claim that an accuracy of 1.0% can readily be obtained with such an instrument when calibrated against, for example, a spirometer. Units were supplier-calibrated at purchase for five different flow rates over the specified operating range (0.01 - 1.0m^3.hr^{-1}) immediately prior to delivery (P. Perry, pers. comm.).

Typically, for the work undertaken for this project, the pump unit was operated at a fairly constant flow rate of approximately 0.18 - 0.24m^3.hr^{-1} (3.0 - 4.0l.min^{-1}) abducting air through two separate cassettes in parallel via a common "tee" piece.

Pumps were monitored for flow rate maintenance as also for total flow through sampling cassette daily within the sampling period. Any fine adjustments to flow rates through paired indoor and outdoor cassettes (as assessed by rotameter and dry gas meter - see below) were to be effected (externally to the house) by the use of a hose clamp on the plastic tube between the cassette and the pump, immediately adjacent to the cassette. In practice, however, this was largely unnecessary, as flow rates through analogous cassettes varied only very slightly (mean difference = 0.22%). Dry gas meters corresponding to inside and outside cassettes were located next to each other inside the dwelling, typically suspended at about mid-window height adjacent to the window with their meter facing outwards, facilitating daily meter reading without house entry.

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[29] This type of pump was identical in all respects to that chosen for the static dust sampling apart from this (air-sampling) pump was chosen to be 240 volt-powered, necessitating the provision of power at test houses. Alternatively, power was accessed from neighbours' supply.

A19.2 CALIBRATION OF GAS FLOW DEVICES

All instruments used for the purposes of air flow assessment for this work were regularly calibrated (in a temperature-controlled environment) and at flow rates of from one to ten litres per minute against a primary standard.[30] Such standards included:

a) a wet gas test meter (Elster Handel, Mainz, Germany; courtesy D. Lapin; AMDEL Laboratories Ltd),
b) a two litre bubble tube, and
c) a ten litre water manometer.

Further, conveniently, newly-acquired additional dry gas meters over the period of time that this research took place were compared with existing meters. Between instrument variability (tested on each occasion over 168 operating hours at 24 hour intervals at varying flow rates) was never greater than one percent, and tested against a primary standard was always within two percent of the true gas flow, tests being conducted at multiple flow rates of between one half and eight litres per minute. Such precision over the entire time of experimentation reflected the consistency of the instruments, no further re-calibration of gas meters subsequent to initial supply being required.

Whilst an adjustable (variable orifice) rotameter was used to estimate flow rates in conjunction with the vacuum pump, acknowledging the limited accuracy associated with rotameters (+/- 5% - 10% when supplied with a calibration curve by the manufacturer [Bernstein et al., 1983]) it was decided that dry gas meters would be a more accurate means by which total gas volumes passing through filters could be estimated.[31] Despite the above-claimed accuracy of the test rotameters, when compared to primary standards, the rotameters displayed consistently high readings of between 1.6 and 1.9 times the wet test (and dry gas) meter readings.

Thus, basing total flow estimates upon mean rotameter readings over the duration of the sampling period would clearly introduce the potential for considerable sampling error. This was the case even when rotameter-indicated flow rates also incorporated daily/half-daily checks through the sampling period combined with total duration time of study (as suggested by Wedberg et al., 1974; Alzona et al., 1979; Turner et al., 1979; Caffo et al., 1980; Ju and Spengler, 1981, and Davies, 1987 & 1990). Such previous

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[30] Air sampling in the field was obviously conducted at a range of temperatures and barometric pressures. Given that some temperature buffering capacity was provided by the dwelling, and with all pumps, dry gas meters and indoor cassettes being kept inside and shaded, and external cassettes being kept shaded, it is suggested that the influence of varying temperature over the sampling period on indicated total gas flow would not be substantial. The temperature range experienced in the field was of the order of 15 - 25 degrees Celsius.

[31] Rotameters are typically calibrated against a primary or more accurate secondary standard at standard "room" temperature and pressure. For rotameters with linear flow rates, the actual sampling flow will approximately equal the indicated flow rate multiplied by the square roots of the ratios of the absolute temperatures and pressures of the calibration and field conditions (Leidel et al., 1977). Given the potential for moderate temperature and pressure variation at sampling locations in Port Pirie, such a measurement error would be less than desirable.

work would appear to have been better served by the use of an instrument for more accurately and precisely determining total gas flow such as has been used here.[32]

Square-edge critical orifices for gas flow limiting were considered as an alternative to dry gas meters, but whilst simple and convenient, were rejected on the basis of cost and also concerns regarding accurately accounting for all factors which would affect the flow rate through such a system (Bernstein et al., 1983; Lippman, 1989). Of such potential factors, one would appear to be variability in critical flow rates through such a device due to variability in vacuum pump performance. Such pump variability might possibly arise as a result of variability in pump operating temperature or general pump wear. Caffo et al. (1980), however, report that critical orifices are inappropriate for use with low-volume ($\approx 2l.min^{-1}$) air-sampling pumps.

Such concerns appear not to be considered as serious for some other workers where critical orifices were employed together with daily checks of flow rates through aerosol monitors using a rotameter in the field (eg Watt et al., 1983; Davies et al., 1987b & c & 1990). These authors' choice to use critical orifices and their suggestion that dry gas meters were rejected since they were too "expensive and heavy" (Watt et al., 1983; p. 233) would thus appear somewhat inappropriate.

Thus it was decided that a vacuum pump would be used in conjunction with a rotameter plus a separate dry gas meter for each aerosol monitoring cassette. Both meters were kept shaded and adjacent to each other in the house to avoid temperature differences between them, and all clear plastic PVC hoses between cassettes and dry gas meters were cut to the same length of one metre. Equally, all hoses between tee pieces and dry gas meters were also cut to equal length (again one metre; this was as short as was considered practicable). Rotameters were mounted to the vacuum pump, and were connected to the tee piece by two metres of PVC tubing. Vacuum pumps were mounted on wooden boards, the underside of which was lined with 20mm thick poly-urethane foam to reduce vibration "creep" of pumps and also vibration effects on rotameters. A plastic chamber 75 mm diameter and 100mm long containing large-hole poly-urethane foam was connected in between the pump and the rotameter in order to smooth out the vacuum "pulses" as produced by the diaphragm vacuum pump, reference to potential pulse effects on sampling efficiency being made by Turner et al. (1979).

A19.3 DRY GAS METERS COMPARISON

All dry gas meters were labelled and for all sampling, each pump was consistently used with the same two gas meters each of which consistently corresponded to either an indoor- or outdoor-sampling cassette. Comparison of such paired gas meters was conducted regularly, both in the laboratory and also in the field over the duration of the sampling study. Such comparison elicited the following results.

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[32] Turner et al.'s group (1979) running battery-powered pumps acknowledge flow rate decline during the sampling period as a result of decreasing battery voltage. This may suggest the introduction of a changing population of dust being sampled during the sampling period.

A19.3.1 IN THE LABORATORY

Initial dry gas meter comparison in the laboratory involved the use of four separate pumps, each with two sets of two in-line dry gas meters running from each side of a common tee piece (ie four pumps drawing air through a total of eight pairs of in-line dry gas meters). Such gas meter comparison was made at air flow rates through gas meters ranging from 0.5 - 4.0l.m^{-1}. Each of the eight sets of two dry gas meters in series was labelled with the pump number, and either Meter A or Meter B.

All pump runs extended over at least 24 hours. From the mean of all A meter readings flow rate through all meters labelled A was 2.83l.min^{-1} with the corresponding mean rate for all meters labelled B being 2.82l.min^{-1}. The (averaged) differences between paired meters over the entire preliminary testing period expressed as a percentage (ie [Mean A- Mean B]/Mean A) was 0.46%.

A19.3.2 IN THE FIELD

Analogously, regular in-line matched pair (indoor/outdoor pairs ie using two sets of two dry gas meters in series in indoor and outdoor locations) comparison of dry gas meters was also practised in the field at normal operating flow rates throughout the period of experimentation. From the mean of all A meter readings, flow rate through all meters labelled A was 1.323l.min^{-1} (SD = 0.086), with the corresponding mean for all meters labelled B being 1.318l.min^{-1} (SD = 0.092). The (averaged) differences between paired meters over the entire testing period expressed as a percentage (ie [Mean A- Mean B]/Mean A) was 0.39%.

Between-meter variability in sampling format and location is clearly not of significant proportion. Further, given concerns regarding variability in flow rate as a result of temperature differences experienced by differently-located equipment (eg sampling cassettes), this seems not to be problematic.

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