

Comments filed to U.S. EPA draft report, "National Inventory of Sources of Emissions for Five Candidate Title III Section 112 (k) Hazardous Air Pollutants: Benzene, 1,3-Butadiene, Formaldehyde, Hexavalent Chromium, and Polycyclic Organic Matter"

Review of Residential Wood Combustion Emissions

U.S. Environmental Protection Agency External Review Draft Report:

National Inventory of Sources of Emissions for Five Candidate Title III Section 112 (k) Hazardous Air Pollutants: Benzene, 1,3-Butadiene, Formaldehyde, Hexavalent Chromium, and Polycyclic Organic Matter

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Introduction

The Hearth Products Association (HPA) has contracted AGI Technologies (AGI) as an independent consultant to review the U.S. Environmental Protection Agency's (EPA) draft report on the national inventory of sources of emissions for five candidate section 112(k) hazardous air pollutants (HAPs). The review focused on the report's implications for residential wood combustion (RWC) with emphasis on polycyclic organic matter (POM) and benzene emissions attributed to RWC. Based on the review it is concluded that: 1) The five candidate section 112(k) pollutants presented in EPA's draft report were selected based on convenience, not on a basis of threat to public health from urban area sources as required in section 112(k), and the data sets for the five pollutants are incomplete. Because of this, combustion sources, such as RWC, appear as more significant human health threats among urban area sources than they may in fact be., 2) The parameters (7-PAH and 16-PAH) used to assess POM emissions from RWC and other sources are not acceptable surrogates for total POM., 3) The number of actual measurements on which 7-PAH, 16-PAH and benzene emission factors are based are grossly inadequate to provide values representative of the entire population of RWC devices in use., 4) The quality of the relatively few actual measurements of 7-PAH, 16-PAH and benzene emissions are low, and the method used to calculate overall RWC emission factors from them is flawed., 5) An erroneously large national level activity value for RWC was used to calculate total national emissions., and, 6) There are significant errors, uncertainties, inconsistencies, and lack of data associated with the assignment of rural versus urban emissions and with the comparison between the national emission inventory and the emission inventories of the three urban areas presented in the draft report. A discussion of each of these six points follows.

Selection of HAPs and Completeness of Data Sets

Section 112 (k)(3)(B)(i) of the Clean Air Act Amendments of 1990 (CAAA) states that the national strategy shall,

"identify not less than 30 hazardous air pollutants which, as the result of emissions from area sources, present the greatest threat to public health in the largest number of urban areas and that are or will be listed pursuant to subsection (b),"

Section 1.2 (Selection of Pollutants) of the EPA draft report states:

"The five HAPs addressed in this report were selected by EPA based on prior studies, availability of inventory data, availability of health effects data, and input from state and local agencies."

Five HAPs is not at least thirty. The preliminary publication of emission inventories of five HAPs based on the convenience of available information without a detailed public health threat evaluation for urban area sources has produced misleading results. Combustion sources dominate the emissions of four out of five of the candidate HAPs which in itself may over predict the significance of combustion sources. In addition, it is noted in the EPA draft report (page 3-1) that emission estimates are not included from some categories of sources due to the lack of data. The tendency of many readers is to assess the relative contribution of various sources to the total emission inventories. With data simply not included for some sources, the relative contribution of those that are listed appear higher than they are in fact.

Specifically, of concern to the HPA is the POM (actually POM surrogates) and benzene emissions attributed to RWC. The publishing of incomplete data sets for them combined with the fact that their significance has not been put in the proper perspective with other HAPs has the effect of overemphasizing the potential health threat of RWC.

Surrogates for POM

A national emission inventory for POM was not presented in the draft report. Instead, national emission inventories for seven specific polycyclic aromatic hydrocarbons (7-PAH) and sixteen specific polycyclic aromatic hydrocarbons (16-PAH) were presented in lieu of an emission inventory for total POM. In addition, an emission inventory for extractable organic material (EOM) has been developed by the EPA (reference 1) to be used to predict POM emissions. The EOM database was discussed in the draft report but was not used because of the lack of analogous urban data bases. None of these three parameters are good surrogates for total POM.

Polycyclic aromatic hydrocarbons (PAH) as a group is a subset of POM, furthermore, the seven and 16 specific polycyclic aromatic hydrocarbon compounds (7-PAH and 16-PAH) comprise a very small subset of the total number of PAH. The emission factors for 7-PAH and 16-PAH for RWC listed in the EPA draft report are taken from the EPA "Locating and Estimating Air Emission" (L & E) series (reference 2), which in turn takes the values from EPA's AP-42 compilations (reference 3) and two other older reports (references 4 and 5). The EPA draft report states, "... *the complex mixture of POM consists of literally thousands of organic compounds.*" The L & E document states, "*Theoretically, millions of POM could be formed.*"

It is widely recognized that the concentration of specific chemical compounds that make up the POM fraction of emissions from various air pollutant source categories (e.g., vehicular exhaust versus RWC) vary widely. This can be seen in reviewing the relative concentration of the PAH compounds for different source categories tabulated in the L & E document. In fact, the documented variability of specific PAH species from source category to source category has been suggested as a tool to apportion the relative contribution of different pollutant source categories to measured ambient concentrations (references 6 and 7). For example, in reference 6, it was concluded that the relative proportion of specific PAH species vary over several orders of magnitude from different source types.

The facts that: 1) Seven and 16 specific compounds are being used as indicators of total POM levels which are made up of thousands to millions of different compounds, and, 2) The relative fraction of total POM made up of specific compounds vary widely from source category to source category, demonstrate that the relative magnitude of emissions by source for 7-PAH and 16-PAH (ranked in Tables 2-1 and 2-2 in the EPA draft report) are not representative of the relative contribution by source for POM. In light of these concerns selecting a different set of 7 (or 16) PAH as "surrogates" for total POM may have resulted in a different ranking of significant source categories. It certainly would have resulted in different estimates of the magnitude of total POM emissions from different source categories.

While EOM was not used as a surrogate for POM in this draft report, its use as a surrogate for POM by the EPA in another report (reference 1) was cited. It was not used in this draft report because of the lack of analogous urban inventories with which to compare the national emission inventory. However, in regards to EOM, the EPA draft report does state,

[EOM] *"is believed to contain the PAH and substituted-PAH compounds that predict cancer risk better than any individual PAH or any sum of PAH species (Lewtas, 1993)"* (Lewtas, 1993, is reference 7 here)

This statement can be misinterpreted by the reader as implying that EOM is a good surrogate for POM -- it is not. Putting organic material into solution with solvents is simply a mechanism to allow bioassay tests to be conducted. The magnitude of the EOM value is not a direct indicator of the magnitude of the POM content. Much of the organic material collected by emission sampling equipment will contribute to the EOM value. Emissions from a source could have a high EOM value without any POM present at all. EOM is simply a measure (an inexact measure) of organic compounds with a low vapor pressure and that can be put into solution with solvent. For this reason, RWC shows a high EOM value as compared to many other source categories. Numerous measurements of the organic carbon, the elemental carbon and the inorganic content of RWC emissions have shown that more than 80% of particulate emissions (which includes the condensable fraction) are made up of organic compounds (references 7,9 & 10). For RWC the overwhelming majority of these compounds are oxygenated aliphatic and monoaromatic compounds not POM. (references 11-13). As with 7-PAH and 16-PAH, the relative percentages of EOM by source category shown in reference 1 are not representative of the relative percentages of POM, moreover, in the case of RWC, the EOM value is high simply because RWC emissions are very high in non-POM organic compounds that will show up as EOM.

RWC Data Base Size

There were 22.9 million households in the United States which used wood for heat in 1990 (reference 14). Some of these households have more than one wood burning device. The 22.9 million household value is based on a survey that assigned it a 10.1% relative standard error (RSE). Based on the facts that some homes have more than one wood burning device (e.g., both a fireplace and a woodstove) and that the survey value has a 10.1% RSE, a reasonable estimate of the total number of wood burning devices in the United States in 1990 would be 25 million. A supplement report to the Household Energy Consumption Survey (reference 15) reported 8.4 million households burned more than one cord of wood per year and 14.5 million households burned less than one cord of wood per year. Most frequently homes which burn more than one cord of wood would primarily be using a woodstove and those burning less than one cord of wood would be using a fireplace. If wood furnaces are grouped with woodstoves and one takes into account that about 7% of the households that use wood as a primary source of heat use a fireplace (references 15 and 16), and that some homes may have more than one wood burning device, a reasonable estimate of the total number of woodstoves and fireplaces in the United States in 1990 would be nine million and 16 million, respectively. Based on wood use data (references 14-16) it can be estimated that about 21% of the total cordwood burned in the United States was burned in fireplaces and 79% in woodstoves. It also should be noted, as will be discussed later, that the HPA believes that no more than about 5% of the woodstoves in use in 1990 were new technology catalytic/non-catalytic stoves.

The EOM emission factor for all wood burning devices listed in reference 1 is based on only 14 tests (reference 17). One conventional woodstove (a Scott brand stove) was tested 12 times and one catalytic woodstove (an Earthstove brand) was tested twice. The 7-PAH and 16-PAH emission factors for conventional woodstoves listed in Appendix B of the EPA draft report are taken from an L & E document (reference 2) which took the data in turn from AP-42 (reference 3) which took the data from the same study (same stove and same 12 tests -- reference 17) as for EOM. The 7-PAH and 16-PAH emission factors for high technology catalytic/non-catalytic woodstoves were based on the two tests on the Earthstove reported in reference 17 and from data from six additional references listed in AP-42.

According to the L & E document, "*There are fewer PAH emissions test data for fireplaces as compared to woodstoves.*" Of the two references cited in the L & E document for fireplace tests, the tests listed in one of the references (reference 5) are described in detail in reference 11. The tests are comprised of sampling a single fireplace two times for PAH (two wood types). The second fireplace reference cited in the L & E document (reference 4) is a 1980 literature review of work conducted in the 1970's. The method used to develop the emission factors are not documented in the review but they appear to be a blend of the results of data from the single fireplace tested and reported in reference 4, a discussion (no quantitative data are provided) of emissions from wood chips held over a bunsen burner on a wire mesh, (reference 18) and a single benzo(a)pyrene number from a 1972 reference (reference 19) not yet obtained by the reviewer.

The benzene emission factor for all wood burning devices listed in Appendix A of the EPA draft report is from AP-42. AP-42 takes the number from the same reference (reference 17) for the mean of 11 tests (one of the twelve samples was not reported) on the single conventional Scott brand stove. The text of the EPA draft report notes that POM emission data are available from an U.S. EPA L & E document (reference 20). The L & E document cites the EPA FIRE data base (reference 21). The FIRE data base cites AP-42.

In summary, the data base for emission factors is not adequate. The EOM emission factor for all 25 million wood burning devices is based on one conventional and one catalytic stove. The benzene emission factor for all 25 million wood burning devices is based on one conventional stove. The 7-PAH and 16-PAH emission factors for the eight to nine million conventional woodstoves, which are responsible for the overwhelming majority of wood consumption and POM emissions, are based on one woodstove. The 7-PAH and 16-PAH emission factors for the 16 million fireplaces are based on little more than one fireplace. There appear to be a few more tests available for PAH for high technology catalytic/non-catalytic woodstoves; however, since they represented a relatively small fraction of the total woodstoves in use in 1990 and their emission factors are smaller than conventional stoves, a detailed accounting of the origins of the 7-PAH and 16-PAH emission factors was not conducted in this review.

Basing emission factors on a limited number of tests is a more serious problem for RWC than most other sources of POM and benzene because of the very high variability that can be expected for emissions from RWC. It has been well documented that combustion conditions such as temperature, available oxygen, and residence time will particularly influence the production of POM (reference 22). There are many hundreds of types or models of wood burning devices in use, many dozens of tree species are commonly used for wood fuel, draft characteristics vary from home to home (chimney conditions), household altitude is variable, there are variations in fuel wood seasoning and storage practices (wood moisture), and there are wide variations in home owner operation of a wood burning devices (burn rate, burn duration, damper setting, kindling approach, etc.). Each of these parameters have significant impacts on combustion conditions and will impact both POM and benzene emissions. Beyond the variability in woodstove emissions which is due primarily to the differences in combustion conditions and has been well documented for other air pollutants such as particles and carbon monoxide, the variability in the chemical makeup of wood is an additional source of variability for POM as specific POM compounds will be formed by the rearrangement, oxidation and combining of compounds contained in the wood fuel. Wood is composed of lignin, cellulose, hemicelluloses, and resins. The ratios of these major chemical groups vary from tree species to tree species, particularly in wood from deciduous versus coniferous trees. Resin content, for example, may be particularly important for both benzene and POM emissions as resins are composed of polyaromatic structures.

To provide insight into the variability of POM and benzene emissions associated with RWC, the reader is referred to reference 17 which provides the basis for the EOM and benzene emission factors for all 25 million wood burning appliances and the 7-PAH and 16-PAH emission factors for somewhere between eight and nine million conventional woodstoves. Emissions from a single conventional woodstove (a Scott brand stove) and a single catalytic woodstove (an Earth brand stove) were measured. There were twelve tests performed on the conventional woodstove. Replicate runs were performed each on low and high burn rates using pine fuel at both high and low altitude, and replicate runs were performed on low and high burn rates using oak fuel at low altitude (six sets of conditions with a replicate run on each). The mean EOM, 7-PAH, 16-PAH, and benzene emission factors for conventional stoves from these tests are 23.4 lbs/ton, 0.051 lbs/ton, 0.69 lbs/ton, and 1.95 lbs/ton, respectively. The standard deviation around the EOM, 7-PAH, 16-PAH, and benzene means are 19.5 lbs/ton, 0.052 lbs/ton, 0.42 lbs/ton, and 0.88 lbs/ton, respectively. These standard deviations represent 83%, 102%, 61%, and 45% of the means for the EOM, 7-PAH, 16-PAH and benzene values, respectively. It must be emphasized that these values are for a single stove tested 12 times with two fuels, two altitudes, and two burn rates and that one half of the test were replicate tests. The magnitude of the variation in POM and benzene emissions among the very large number of parameters encountered among the real-world use of RWC devices must be very large. Statistically using one conventional stove and one plus fireplace to represent millions of devices is fundamentally in error.

Quality of Emission Factors

The 7-PAH, 16-PAH and benzene emission factors used in the draft report are ranked in terms of quality in the L & E documents (references 2 and 20). They all have an emission factor rating of E except for the one fireplace data set for 7-PAH and 16-PAH which was derived from the 1980 literature review article. The emission factor rating for that data set is U5. An E rating is the lowest and is described as "poor", the U rating is defined as unrated or unratable. The U5 subcategory is further defined as having a "lack of supporting documentation."

The EOM emission factor developed from the 12 tests on a single conventional stove and on two tests on a catalytic stove were obtained by using a non-reference, non-standard sampling protocol. (One author of this review was also the co-author of the study on which the EOM emission factor is based C reference 17.) An aliquot of solvent extracts from filters, extracts from XAD-2 resin and probe rinses underwent gravimetric and chromatographic analyses. The EOM value is the sum of the gravimetric and chromatographic determinations on each of the three solutions. It is the authors' opinion that the propagated uncertainty of the technique and subsequent addition of six values produced a precision of no better than 30%. Also the gravimetric sample was lost for one run reducing the EOM data set to 13.

Beyond the accuracy and precision of the fundamental measurements, the HPA is concerned about how the emission factors for the various wood burning devices were weighted to produce overall emission factors that were subsequently multiplied by the total national wood use to obtain total RWC values for the national emission inventory tabulation. For example, the L & E document from which the EPA draft report took the weighted 7-PAH and 16-PAH emission factors has tabulations for conventional woodstoves (Table 4.1-1), non-catalytic woodstoves (Table 4.1-2), catalytic woodstoves (Table 4.1-3), pellet stoves (Table 4.1-4), and fireplaces (Table 4.1-5). The emission factors of 0.035 lbs/ton for 7-PAH and 0.517 lbs/ton for 16-PAH for residential wood combustion are listed in appendices A and B of the L & E document and in appendix B of the EPA draft report without any explanation of the calculations used to derive them. Apparently, a weighting factor was used to account for the relative usage of the various devices. In addition, the PAH data for fireplaces shown in Table 4.1-5 of the L & E document are missing a number of the 7-PAH and 16-PAH compounds. There is no explanation on how this lack of

data was treated in calculating the weighted emission factors. The Household Energy Consumption and Expenditures survey results (reference 14) also show that 3.5 million cords were burned in other wood burning devices (primarily wood furnaces). No weighting or emission factors have been presented for them.

While an EOM emission inventory was not presented in the EPA draft report, its efficacy as a surrogate for POM was discussed. It is important to emphasize here that EOM is not a good surrogate for POM and that the EOM emission inventory for RWC developed in EPA's section 122 (c)(6) document (reference 1) has significant errors and uncertainties associated with it. The development of the EOM emission factor for residential wood combustion is addressed in detail in section A.24 of appendix A of reference 1. In that appendix it is stated,

"Statistical data from a 1990 annual survey of residential homeowner use conducted by the EIA were used to develop the weighing factors to apply to the available emission factor data to represent the split between woodstove and fireplace use. The average nationwide percentage of wood consumption is 28 percent for fireplaces and 72 percent for woodstoves. Consumption for woodstoves can be further divided into approximately 70 percent conventional woodstoves (no control devices) and 30 percent catalytic/non-catalytic woodstoves."

There are two key issues that need to be addressed in regards to these statements. First, the data in the referenced EIA report do not provide a direct mechanism to calculate the relative wood usage between fireplaces and woodstoves. Upon reviewing that data the 28% to 72% split for wood use between fireplaces appears reasonable. However, it is not a rigorous quantitative number that can be used to calculate weighted emission factors. (It is unclear whether this split is the same as was used to calculate weighted 7-PAH and 16-PAH emission factors as discussed in the preceding paragraph.) The second issue is that the estimate of 30% wood use in catalytic/non-catalytic woodstoves in 1990 is too high. Under federal regulations conventional woodstoves could be manufactured up to July 1, 1988 and sold up to July 1, 1990. While many manufacturers started manufacturing and selling Phase I (and Phase II) certified catalytic and non-catalytic woodstoves prior to these cutoff dates, 30% wood use in them during 1990 is not a reasonable estimate based on the turnover rate of woodstoves. It is likely that less than 5% of the woodstoves in use in 1990 would have been catalytic/non-catalytic stoves. Wood use attributed to them as compared to conventional woodstoves, would be more or less at the same percentage, since they are on one hand, more efficient but on the other hand, one might argue more serious wood burners would purchase them. The low estimate is confirmed by the results of a survey conducted by the Oregon Department of Environmental Quality in Portland, Oregon for 1993 (reference 23). The results showed that 35% of the homes with woodstoves and stove-like inserts had certified devices in 1993. Certified devices in this case included both Oregon and EPA certified devices. The state of Oregon did not allow conventional stoves to be sold retail after July 1, 1986. (Oregon certification was, by in large, the model on which EPA certification was subsequently based). Consequently, in Portland, Oregon, homeowners took six and one-half years to replace 35% of their woodstoves with certified stoves. Again, nationwide (except for Oregon) conventional woodstoves could be sold up to July 1, 1990 and, of course, the base year for the EPA draft report is 1990.

It is further stated in section A.24 of Appendix A of reference 1 that,

"Table A-16 lists the emission factors for each pollutant that were used in the inventory. The EOM factor is a weighted emission factor which represents conventional and catalytic/non-catalytic woodstove use. The EOM emissions factor was developed from test results for 14 woodstoves; 12 of these were conventional stoves and the other 2 were catalytic designs. The

EOM emission factor represents an average of these test results weighted based on the percentage of conventional and catalytic/non-catalytic woodstove use described above."

There are three points that need to be addressed regarding these statements. First, the EOM emission factor was developed from 12 tests on a single conventional woodstove and on two tests with a single catalytic woodstove which will effect the weighing calculation. Secondly, as previously discussed, a 30% catalytic/non-catalytic number is too high for 1990, and, third, the EOM emission factor (18.66 lbs/ton) listed in Table A-16 of Appendix A of reference 1 was multiplied by 45.6 million tons (rounded off and reported as 46 million tons in Appendix A) to produce a total emission value for RWC of 425,448 tons/yr. No weighting was performed for the 28% of wood purported by the EPA draft report to be used by fireplaces.

The benzene emission factor of 1.938 lbs/ton reported in Section A.16 of Appendix A of the draft report is the mean of the 12 tests (actually 11 since one data set was not reported for VOC) on one conventional wood stove (reference 17). This mean value was multiplied by the 45.6 million tons to produce a total benzene emission value for RWC of 44,190 tons/yr. No weighting was performed for fireplaces or other wood burning devices.

National Level Activity Value

The total cordwood usage value of 45.6 million dry tons for 1990 was reported in appendices A and B of the draft report, in appendix B of both the POM and benzene L & E documents and in appendices A and B of reference 1 (rounded off and reported as 46 million in appendix A of reference 1). This value was multiplied by the 7-PAH, 16-PAH, EOM and benzene emission factors to obtain the total RWC values for 1990 of 800 tons/yr, 11,800 tons/yr, 425,448 tons/yr, and 44,190 tons/yr, respectively.

According to the Household Energy Consumption and Expenditure survey (references 14 and 15), 29.1 million cords of wood were burned in 1990 (actually December 1989 through November 1990). The Energy Information Administration uses a conversion factor of 1.163 tons per dry cord (reference 16) which is consistent with the mean cord weight of 1.212 tons per dry cord determined for 36 tree species (standard deviation around the mean of 0.386 tons) commonly used as fuel (reference 24). The 29.1 million cords of wood burned in 1990 multiplied by the conversion factor of 1.163 tons per dry cord yields 33.8 million tons of wood burned in 1990, a significantly lower value than the value (45.6 million tons) used by the EPA. Additional confusion regarding the correct national level activity value appears to be associated with the facts that several of the appendices previously listed cite a document that does not contain the national activity level (reference 25 here), and several erroneously state that the data is for the 1994 base year not 1990. Appendix B of the benzene L & E document does cite the correct Energy Information Administration document which states that the wood energy consumption in 1990 was 786 trillion BTU which corresponds to 45.6 million tons of wood (see Table 5, page 12 of reference 26). However, the Energy Information Administration subsequently reported that the 786 trillion BTU estimate for 1990 was revised to 581 trillion BTU (see Table 1, page 16 of reference 16). The number of tons of wood corresponding to 786 and 581 trillion BTU are 45.6 and 33.8 million, respectively.

City-Specific Inventories and Rural versus Urban Allocations

The city-specific inventories for Chicago, Houston and Phoenix are difficult to compare to each other and to the national emission inventory because they contain different source categories, are incomplete, were developed by different methods, the base years were 1993 not 1990 (except for the Chicago area source

inventory) and of course three cities represent a very small sample size when compared to the total number of urban areas in the US. These problems were, in essence, acknowledged in the concluding paragraphs of the text of the draft report. On page 7-6 of the draft report it is stated that:

"It is not possible to conclude at this time which of the inventories most accurately reflects "actual" or "representative" emissions in urban areas or the contribution of major, area, and mobile sources to total emissions", and

"It is also not possible at this time to conclude that the HAPs inventory data presented here for Chicago, Houston, and Phoenix can serve as the basis for developing a national strategy to control emissions of 112(k) HAPs."

Beyond the general limited value of the three city-specific inventories, their value for RWC assessment is small. RWC was not included in the Chicago inventory. The methods used to estimate RWC emission factors (or any of the emission factors) in Houston and Phoenix were not presented in the draft report. In regards to the city-specific inventories, it is stated in the draft report that, *"the presentation of the POM emissions data ranges from simply "POM" to a listing of a few individual compounds."* As previously discussed the use of different POM compounds or different groups of POM compounds to predict total POM can produce different results and the cross comparison of data sets comprised of different POM compounds is not a sound approach. Of course, understanding the relative contributions of sources to POM is key to assessing the significance of RWC.

As directed in the Section 112 (k) of the CAAA, the EPA must identify area sources of a least 30 HAPs that present the greatest threat to urban populations. For that reason the emissions for each source category in the national emission inventory were allocated in the draft report between urban and rural contributions. In appendix A (page A-42) of the draft report 20% of the national HAPs emissions from RWC are attributed to urban areas and 80% to rural. While these values appear reasonable, the reference cited by the draft report (reference 26 here) does not provide a direct mechanism to calculate the relative emission factors between rural and urban categories. The 20%/80% split is clearly an estimate with a large uncertainty not a rigorously derived value. Table 4-7 in the draft report shows the total 44,190 tons/yr of RWC benzene emissions split between 8,838 tons/yr for urban and 35,353 tons/yr for rural, or a 20%/80% split. However, Table 4-11 shows the total 800 tons/yr of RWC 7-PAH emissions split between 668 tons/yr urban and 132 tons/yr rural or an 83.5%/16.5% split. Similarly, Table 4-12 shows the total 11,800 tons/yr of RWC 16-PAH emissions split between 9,852 tons/yr urban and 1,949 tons/yr rural or also an 83.5%/16.5% split. Consequently the apportionment for 7-PAH and 16-PAH RWC emissions between urban and rural is almost exactly opposite than it is for benzene and not the 20% urban/80% rural discussed in appendix B.

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