APPENDIX J: APPLICATION OF NYSDEC SEDIMENT EFFECT CONCENTRATIONS (SECs) FOR EVALUATING SEDIMENT QUALITY IN ONONDAGA LAKE

Prepared For:

HONEYWELL

101 Columbia Road
P.O. Box 2105
Morristown, NJ 07962

Prepared By:

ERL Environmental

Exponent

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EXECUTIVE SUMMARY

As part of the Onondaga Lake remedial investigation and feasibility study (RI/FS), Honeywell is evaluating various remedial alternatives for addressing contaminated sediments in the lake. A key component of these evaluations is the selection of a method for assessing sediment quality throughout the lake. The New York State Department of Environmental Conservation (NYSDEC) developed a set of site-specific sediment quality guidelines (SQGs) for evaluating sediment quality in the lake as part of the baseline ecological risk assessment (BERA) for the RI/FS. These SQGs were referred to as sediment effect concentrations (SECs). The NYSDEC then calculated a probable effect concentration (PEC) for each chemical parameter of interest (CPOI) by calculating the geometric mean of the various SECs for each CPOI. The PEC for each CPOI provided a single, mid-range, consensus-based sediment quality value for that CPOI. This appendix presents an evaluation and recommendation of how the site-specific PECs can be used to identify potentially toxic sediments in the lake and to evaluate various remedial alternatives.

In calculating the site-specific PECs for Onondaga Lake, the NYSDEC used the basic approaches previously published in the scientific literature and government publications. These approaches involved statistical analyses of a database composed of matching chemical and acute toxicity data to calculate the PECs. The Onondaga Lake PECs were calculated with the results of a 10-day acute toxicity test. Data from this test were intended to be indicative of adverse toxic effects in resident infaunal biota. In addition, the NYSDEC calculated the quotients (or ratios) between the concentrations of individual chemicals in the lake sediment samples and the individual PEC values to help identify chemicals and areas of greatest concern. Because some of the specific steps in the site-specific methods differed somewhat from those that were previously published, the PEC values for some chemicals (but, not all) tended to differ by varying amounts from those that had been derived previously by others. In addition, it is possible that site-specific conditions in the lake (e.g., high concentrations of ionic constituents) may have influenced the PEC values to some degree.

Potentially toxic chemicals in most aquatic environments (including Onondaga Lake) invariably occur as complex mixtures that vary in chemical composition from place to place throughout a water body. Such spatial heterogeneity is typical of many urban lakes, bays, and harbors. Evaluation and classification of the relative toxicological risks of such heterogeneous conditions using individual guidelines, such as PECs, can be challenging, confusing, and contentious. To better understand the toxicological significance of the site-specific PECs for Onondaga Lake and to provide a single index with which to classify sediment quality throughout the lake, an approach is proposed here by which the PECs can be used effectively to evaluate sediment contamination as a result of the presence of heterogeneous toxicant mixtures.

The proposed approach for Onondaga Lake is the use of mean SQG quotients (in this case, mean PEC quotients derived for the lake), which are indices of chemical contamination that take into account both the presence and the concentrations of multiple chemicals in various sediment samples. The mean SQG quotient approach can clarify the spatial patterns and degree of chemical contamination in a spatially heterogeneous site, such as that in Onondaga Lake, using a single unitless index. The proposed approach has been used in many different regions of the U.S. and Canada by numerous federal and state agencies, monitoring programs, and ecological risk assessors. It has been accepted and published in scientific journals and government publications. All of these previous studies have shown repeatedly that toxicological risks invariably increase over ranges of increasing mean SQG quotients. The mean SQG quotient approach is founded on sound scientific methods and principles. There are a number of assumptions that analysts should be aware of when using this approach; however, none of them tend to negate its strengths and disqualify it from uses such as those intended for Onondaga Lake. Because toxicant concentrations tend to co-vary with each other, establishment of target cleanup levels with mean SQG quotients can be an effective approach to ensure reduction in risks by mixtures of chemicals.

For Onondaga Lake, mean PEC quotients were calculated using the 1992 chemical database for the lake and compared to the results of laboratory toxicity tests to provide perspective to the quotient index values and to identify the exposure/response relationships for this site. The toxicity tests included the acute, 10-day tests performed with the chironomid *Chironomus tentans* (now, *C. dilutus*) and the amphipod *Hyalella azteca*. The calculations were performed using two methods: first by including all undetected chemical concentrations as one-half their detection limits and second by omitting the undetected concentrations. For both methods, the mean of a maximum of 46 PEC quotients was calculated for each of the 79 stations sampled in the lake in 1992 to derive the mean PEC quotient for each station. The mean PEC quotients were then compared to the degree of response (i.e., mortality) in the laboratory toxicity tests that were performed on portions of the same sediment samples to determine whether an exposure/response relationship existed between the toxicity results and mean PEC quotients. The existence of such a relationship would suggest that the toxicity test organisms were responding to the degree of chemical contamination in the sediment samples, not some confounding or natural factor(s).

Although toxicity information was collected for both the 10-day amphipod and 10-day chironomid tests at all 79 stations sampled in Onondaga Lake in 1992, the NYSDEC used only the results of the 10-day chironomid test to derive the site-specific PEC values for the lake because it was found to be the more sensitive test. Therefore, the chironomid test was used in this report to gauge the toxicological risks posed by increasing mean PEC quotients. However, the data from the amphipod test were also evaluated to determine whether an exposure/response relationship existed for that test as well. In addition, mean percent mortality was calculated for samples within ranges of increasing mean PEC quotients to determine if and where there was a noticeable increase in mortality among the toxicity test organisms (i.e., an inflection point in the exposure/response relationship). Such an inflection point can be considered a potential critical value with which to evaluate the toxicity of sediments in the lake. The objective of this evaluation was to identify the point along the continuum of increasing mean PEC quotients where mortality in the test organisms first began to increase above background or reference conditions (i.e., an inflection point from a flat line to an increase or upturn). Because of the variability in these data, gaps in the database, and the lack of widely accepted, statistically derived critical values in mortality for either test, inflection points could not be derived by using scatterplots of individual data points. Rather, mortality was examined within ranges in mean PEC quotients to be consistent with methods used previously in many other regional and national data analyses of this kind.

Generally, the relationships between percent mortality in the two sediment toxicity tests and increasing mean PEC quotients were similar whether undetected chemical concentrations were excluded from the analysis or were treated as one-half the detection limits. Mortality was consistently lowest and failed to increase noticeably among the lowest ranges in mean PEC quotients. There were moderate incremental increases in mortality in the intermediate ranges in the PEC quotients, followed by marked increases in mortality in the most contaminated samples. The points along the mean PEC quotient continuum in which mortality first appeared to increase differed somewhat between the amphipod and chironomid tests and between the two data treatments (i.e., undetected values omitted and undetected values represented by one-half the detection limits). Overall, however, several iterative trials showed relatively consistent results and inflection points, suggesting that the critical values identified with these methods were robust.

Based on the chironomid test, mean mortality showed similar patterns regardless of whether undetected values were represented by one-half the detection limits or whether they were omitted from the analysis. In both cases, the exposure/response relationship did not show a noticeable increase until a mean PEC quotient range of about 2 to 5 was reached. The selection of the ranges in values (or bins of samples) made little difference in the apparent inflection point where mortality first began to increase. Average mortality among samples varied from 7 to 16 percent with no pattern of increasing mortality in samples with mean PEC quotients of 0.1 to approximately 2.0. Average mortality then increased to 18 to 23 percent in several different trials among samples with mean PEC quotients greater than 2 to approximately 5 or 6, depending upon how the non-detected data were treated. Average mortality peaked at greater than 50 percent in the most contaminated samples (quotients greater than about 7).

As described for the chironomid test above, mean mortality in the amphipod test showed similar patterns regardless of whether undetected chemical values were represented by one-half the detection limits or whether they were omitted from the analysis. In both cases, the exposure/response relationship did not show a noticeable increase until the mean PEC quotient range of approximately 1 to 2 was reached. In several different trials, using different ranges in values, average mortality ranged from about 4 to 7 percent in samples with mean PEC quotients less than about 1.0. The first substantial and consistent increase in mortality (to about 8 percent) occurred as mean PEC quotients increased above values of 1.0 to 1.4, depending upon how the data were treated.

The inflection points identified for the two toxicity tests were represented by ranges of mean PEC quotients because variability and gaps in the database and the lack of a critical value for percent mortality caused too much uncertainty in the identification of tighter inflection points with either a regression line approach or a scatterplot approach. Additional variability may have been caused by the sampling design (matching the chemistry samples with the toxicity test samples), analytical inaccuracies in determinations of the true chemical concentrations, and the tests selected to measure the true degree of toxicity. Data from long-term (20-day to 42-day) tests generally confirmed the results obtained with the 10-day tests but were available for very small numbers of samples (e.g., 15). The outcomes of the data analyses in this appendix may have differed if long-term tests or chronic (i.e., life cycle) tests had been performed on all 79 sediment samples. Such sources of variability are normal and expected in any sediment quality assessment. Previously published analyses of the relationships between ranges in mean SQG quotients and toxicity have relied on data from hundreds and even thousands of samples to limit data gaps. The Onondaga Lake database, although of high quality and highly important for this site-specific application, relied on data from only 79 samples. Combining these factors tends to cause the degree of variability apparent in these results for Onondaga Lake.

The Onondaga Lake database was analyzed with statistical methods to determine if the exposure/response relationships would be substantially improved by reducing the list of chemicals used to calculate the mean PEC quotients from 46 chemicals to those chemicals most significantly correlated with toxicity. The data were therefore evaluated using a variety of statistical and graphical methods to identify which chemicals were most correlated with the measures of toxicity. Statistical analyses showed that mortality in both tests was correlated significantly with the mean PEC quotients calculated with all 46 chemicals. The results of additional analyses showed that 23 CPOIs contributed the most to the exposure/response relationships. The correlations between mortality and the revised mean PEC quotients based on these 23 CPOIs improved over those calculated with all 46 chemicals. Revised mean PEC quotients were then examined using the data for only the 23 CPOIs to determine if the inflection points in the exposure/response relationships changed from those observed with data from all 46 CPOIs. Although the correlations between chemical concentrations and toxicity improved with the smaller list of 23 chemicals, the inflection points in mortality based on the subset of 23 CPOIs were similar to those obtained with all 46 CPOIs. That is, the inflection points where mortality first indicated a slight increase occurred when revised mean PEC quotients reached values of about 1 to 2 for the amphipod test and about 2 to 5 for the chironomid test.

The inflection points (mean PEC quotient ranges of 1 to 2 and 2 to 5) identified for the 10-day amphipod and chironomid tests, respectively, based on chemical groups were consistent with the inflection points for the two toxicity tests identified on the basis of all 46 CPOIs. These results indicate that the reduction of in the number of CPOIs did not appreciably affect the overall exposure/response relationships for the two toxicity tests. The results also underline the robustness of the inflection points for the two toxicity tests and increase confidence in the validity of the inflection points, despite the variability observed in the data.

The inflection points identified for the chironomid and amphipod tests indicated the points at which the test organisms were beginning to exhibit increased mortality in relation to the mean PEC quotients. These points were taken as the lower ends of the two ranges identified as the inflection point ranges, i.e., approximately 1.0 for the amphipods and approximately 2.0 for the chironomids. The mean PECQ values of 1 and 2 can be used in the FS to identify sediments that potentially pose risks of toxicity to benthic macroinvertebrates in Onondaga Lake.

The results of these analyses suggest that application of mean PEC quotients based on the site-specific PECs for Onondaga Lake is an appropriate and defensible method for comparing and classifying the quality of sediments throughout the lake. The use of these site-specific data to evaluate relative sediment quality in the lake is preferable to using data from other water bodies because the Onondaga Lake data reflect the site-specific conditions found in the lake. This tool condenses complicated information from numerous chemicals into one effects-based index that accounts for both the presence of chemicals in the sediments and their concentrations relative to the PECs. Thus, it can be used to interpret complicated data with a single index for estimating relative risks of effects on benthic macroinvertebrates.

SECTION J.1

INTRODUCTION

As part of the Onondaga Lake RI/FS, Honeywell is evaluating various remedial alternatives for addressing contaminated sediments in the lake. A key component of these evaluations is the selection of a method for assessing sediment quality throughout the lake. The NYSDEC developed a set of site-specific sediment SQGs for evaluating sediment quality in Onondaga Lake as part of the BERA for the RI/FS (TAMS, 2002). They were referred to as SECs. This appendix evaluates and recommends how those site-specific SECs can be used to identify potentially toxic sediments in the lake and to evaluate various remedial alternatives.

The first section of this appendix presents an overview of the site-specific SECs developed for Onondaga Lake by the NYSDEC, including their definitions and the methods by which they were derived. The next section describes the approach recommended by Honeywell for using the site-specific SECs to identify contaminated sediments in the lake and to prioritize them for various kinds of potential remedial actions. The final section presents the results of applying the recommended sediment quality assessment approach to the lake.

The quality of sediments in the lake could be classified by 1) comparing sediment chemical concentrations to existing SQGs, 2) evaluating results of sediment toxicity tests, 3) modeling dispersal of sediment-bound toxicants into the water column, or 4) modeling bioaccumulation of sediment-bound chemicals into fish and wildlife, each of which could be used to develop different remediation goals. However, the most straightforward approach to classifying sediment quality is to compare concentrations in the sediments with SQGs developed specifically for the site of interest. This report is intended to evaluate the latter approach and to introduce a method of using that approach for classifying sediment quality in Onondaga Lake.

SECTION J.2

OVERVIEW OF SITE-SPECIFIC SECs FOR ONONDAGA LAKE

As described above, the NYSDEC developed a set of site-specific SECs for Onondaga Lake. These SECs were based on the mortality results recorded in the chironomid sediment toxicity test in 1992, which was evaluated at 79 stations in Onondaga Lake and 5 stations in Otisco Lake (the NYSDEC-approved reference lake). Although information was also collected on chironomid biomass and amphipod mortality and biomass at the 79 stations, the NYSDEC used chironomid mortality to develop the SECs because it was the most sensitive endpoint (i.e., it identified the largest number of stations with significant effects). Significant effects ($P \leq 0.05$ comparison-wise) were determined by statistical comparisons in which percent mortality at each Onondaga Lake station was compared to the results from one of two stations selected from Otisco Lake (where each of the two stations had a mortality value of 2 percent). Using the 1992 chironomid mortality results, the NYSDEC developed five kinds of site-specific SECs, relying on methods that had been previously published in the scientific literature and government reports. The five SECs calculated by the NYSDEC were:

- **Effects Range-Low (ER-L):** The 10th percentile of the ascending concentration distribution associated with effects data
- **Threshold Effect Level (TEL):** Geometric mean of the 15th percentile of the ascending concentration distribution associated with the effects data and the median of the ascending concentration distribution associated with the no-effects data
- **Effects Range-Median (ER-M):** The median (50th percentile) of the ascending concentration distribution associated with the effects data
- **Probable Effect Level (PEL):** Geometric mean of the ER-M and the 85th percentile of the ascending concentration distribution associated with the no-effects data
- **Apparent Effects Threshold (AET):** The highest concentration associated with observations of no effects (i.e., the concentration above which effects were always observed).

The ER-L/ER-M method was developed by Long and Morgan (1991), the TEL/PEL method was developed by MacDonald *et al*. (1994, 1996), and the AET method was developed by Barrick *et al*. (1988). A key distinction among the various SEC methods is the manner in which effects and no-effects data are used to develop SECs for each CPOI. The ER-L/ER-M method uses only the effects data, the AET method uses only the no-effects data, and the TEL/PEL method uses both the effects and no-effects data.

A second key distinction among the various SEC methods is that only the AET method was developed explicitly for site-specific use, and it was endorsed for such use by the U.S.

Environmental Protection Agency (USEPA, 1989). By contrast, the ER-L/ER-M and TEL/PEL methods were developed primarily for generating SECs based on data from a broad range of environments. Although this "global" use does not disqualify those two methods from generating site-specific SECs, it can potentially result in some difficulties in applying the methods directly to site-specific cases (e.g., because of the lack of sufficient sample sizes). The latter two sets of SECs were intended to be used as informal, non-regulatory guidelines, whereas the AETs were eventually promulgated as regulatory tools by the state of Washington.

A third key distinction among the various SEC methods is in the kinds of data that were used to derive the guidelines. The original sets of ER-L/ER-M and TEL/PEL values were calculated with data summarized from numerous different kinds of studies, including laboratory experiments with spiked sediment bioassays, field studies of sediment toxicity and benthic community impacts, and previously published SQGs. Unlike the AET values, they were not based on data from individual samples, but, rather, summarized chemical concentrations and biological endpoints from each study. They were also based on data from numerous different kinds of biological test endpoints, whereas AETs were derived for each individual endpoint separately. In subsequent analyses of freshwater data to derive ER-L/ER-M and TEL/PEL values, data were treated on a per-sample basis (Ingersoll *et al*., 1996). In the Onondaga Lake project, therefore, the NYSDEC appropriately followed the procedures of Ingersoll *et al*. (1996) and MacDonald *et al*. (2003a) to derive freshwater guidelines (i.e., by using data from each sediment sample).

From a narrative standpoint, the various SECs can be defined as representing three different thresholds for predicting the presence of toxic effects:

- **ER-L and TEL:** Concentrations *below which* toxic effects are predicted to *rarely occur*
- **ER-M and PEL:** Concentrations *above which* toxic effects are predicted to *frequently (but not always) occur*
- **AET:** Threshold *above which* toxic effects are predicted to *always occur*.

NYSDEC assembled the five SECs for each COPC and derived a PEC for each CPOI by calculating the geometric mean of the five SECs. The PECs, therefore, were mid-range values based on five individual SECs that were derived using the same database of matching sediment chemistry and sediment toxicity information. The calculation of a consensus-based PEC is consistent with the approach initially published by Swartz (1999) to derive consensus SECs for concentrations of total polycyclic aromatic hydrocarbons (PAHs). It was used subsequently by MacDonald *et al*. (2000a) to develop consensus guidelines for total polychlorinated biphenyls (PCBs) and a set of freshwater SECs for multiple chemicals (MacDonald *et al.*, 2000b; 2003a). However, MacDonald *et al*. (2000a, 2003a) calculated PECs from SQGs that had been derived with the same narrative intent.

There is no single "correct" way to calculate sediment quality guidelines or to derive SEC values. All guidelines are viewed as method-specific and used accordingly (Wenning *et al.*, 2004). The role and importance of site-specific guidelines, such as those derived for Onondaga Lake, have been recognized and acclaimed by experts for some years (USEPA, 2000a). USEPA (2000b) recommended that the derivation and application of biocriteria be conducted on a site-specific basis. The initiative taken by the NYSDEC to derive the SECs is consistent with guidance provided by other agencies and reflects an obvious intent to provide useful and reliable tools with which to classify the sediments in the lake. Although some methods used by the NYSDEC to derive the SECs for use in Onondaga Lake differed from those previously published, these differences do not necessarily disqualify a given set of SECs from being useful.

The fundamental purpose of any set of effects-based SQGs is to use them in place of actual tests of adverse biological effects. This use is necessary because in many cases, sediment chemical concentrations are the only kind of information available for sediment samples. A variety of empirical and theoretical approaches have been used to derive such guidelines (USEPA, 1992). All have various strengths and weaknesses, and all have somewhat different assumptions and intended applications. SQGs derived with theoretical methods are intended to identify which chemicals caused an observed adverse effect, while those developed with empirical methods are intended to predict the presence or absence of such effects, but not the cause of the adverse effects. The five sets of SECs and the PECs derived by the NYSDEC for Onondaga Lake used empirical methods and, therefore, were intended to be used to predict either the presence or absence of effects.

A key consideration when interpreting predictions of toxic effects that the five SECs largely ignore is the magnitude of such effects. The degree of mortality in acute laboratory toxicity tests generally increases along with the incidence of toxicity as chemical concentrations increase in sediments (Fairey *et al*., 2001; Crane *et al*., 2000, 2002; Ingersoll *et al.*, 2004). The degree of response has also been shown to be greater in longer term, chronic and/or sublethal tests (Crane et al., 2000; Ingersoll et al., 2001; Long et al., 1998b). Evaluation of the chemistry data with the individual SECs generally can predict the presence or absence of toxic effects, but not the severity of the effects. For example, exceedance of the site-specific AET for a chemical at a station in Onondaga Lake predicts that chironomid mortality at that station is significantly higher ($P \leq 0.05$) than the value of 2 percent found at each of the two reference stations used in Otisco Lake. It does not necessarily imply that the level of mortality was severe. In numerous cases in the NYSDEC's statistical analysis, mortality levels as low as 10 to 20 percent were found to differ significantly from Otisco Lake. Statistically significant differences between mean values that do not differ greatly (i.e., reference vs. test samples) can occur when the variance of each treatment is very small (Thursby *et al.*, 1997). Use of the mean SQG quotient approach to identify exposure/response relationships helps to clarify these statistical issues.

SECTION J.3

RECOMMENDED SEDIMENT QUALITY ASSESSMENT APPROACH FOR ONONDAGA LAKE

Given that the site-specific PECs for Onondaga Lake are considered consensus values reflective of the five kinds of site-specific SECs derived for the lake, Honeywell recommends that mean PEC quotients be used to identify sediments that may pose a risk to benthic macroinvertebrate communities in the lake. This approach was developed by MacDonald *et al*. (2000b) and is a modification of the mean ER-M quotient approach developed previously by Long *et al*. (1994, 1998b, 2000).

J.3.1 DERIVATION

Mean SQG quotients are calculated in a simple, three-step process. First, each chemical concentration in the sample is divided by its SQG, resulting in an individual ratio or quotient of the concentration of that chemical in the sample to its respective SQG. Next, all the resulting SQG quotients for a particular sampling station are summed, including those that are fractions of 1.0 and those that are greater than 1.0. Finally, the sum of the individual SQG quotients is divided by the total number of SQGs for which there were data in the sample. Any set of SQGs can be used to derive the quotients, including the PECs developed by the NYSDEC for use in Onondaga Lake. Mean SQG quotients can be calculated separately for classes of different chemicals, such as the trace metals, aromatic hydrocarbons, and chlorinated hydrocarbons to help identify which classes of substances were most or least important. Both the sums of the quotients and the means can be used together as evaluation tools. However, the sums of quotients are subject to variability if different numbers of chemicals were measured in different samples, thereby precluding meaningful comparisons among the samples.

A simplified hypothetical example of the calculation of the mean PEC quotient for a sediment sample would be where only five CPOIs are present at a station and PEC quotients of 1.0, 2.0, 3.0, 4.0 and 5.0 were calculated for the five chemicals. The mean PEC quotient for the station would be the sum of the five individual PEC quotients (i.e., 1.0 + 2.0 + 3.0 + 4.0 + 5.0 = 15) divided by the total number of PEC quotients calculated at the station (i.e., 5), resulting in a mean PEC quotient of 3.0 (i.e., 15/5) for the overall station. Standard procedure is to use one-half the detection limit for a chemical concentration reported as undetected.

In the BERA, chemical concentration/PEC quotients were calculated and presented for each chemical to help identify CPOIs. The approach proposed in the present report takes those calculations one step further and provides a single index of contamination by calculating the mean of the quotients. For the present report, PEC quotients were calculated for each CPOI, and the mean PEC quotients were used as an index of the relative risk of sediment toxicity. That is, higher mean PEC quotients are considered indicative of higher degrees of risk. This information can help identify which CPOIs are most important at individual stations and rank relative risks posed by mixtures of chemicals among sampling locations.

Because the NYSDEC derived PECs for 46 CPOIs in Onondaga Lake, a maximum of 46 PEC quotients could be derived for each station, depending on how many CPOIs were actually measured at that station. For Onondaga Lake, 19 to 46 chemicals having site-specific PECs were measured at the various stations in 1992 and 2000. For a CPOI that was undetected at a station, two different methods were used in the present report to treat undetected values. In the first method, one-half the detection limit was used as an estimate of the true concentration of the CPOI at that station, and that estimated value was divided by the PEC to derive the PEC quotient for that station. In the second method, the undetected value was excluded from further consideration, and the PEC quotient was derived using only detected values. The second method of treating undetected values was found to be more effective in calculating mean PEC quotients for the lake and was therefore the method that was ultimately selected.

J.3.2 ADVANTAGES

Mean SQG quotients provide a single, easily understood, effects-based numerical index of the relative degree of chemical contamination of sediments. The numbers of chemicals in a sample that exceed their respective SQGs can be summed as an indication of the presence of these substances. It is logical to rank a site with numerous chemicals exceeding the SQGs as more contaminated than a site with few or no chemical concentrations greater than the SQGs. However, using this approach, a chemical concentration that exceeds an SQG by a small amount and one that exceeds it by many times are scored the same. The mean SQG quotient approach allows the analyst to account for not just the presence of the chemical at a concentration greater than the SQG, but also for its actual concentration in the sample. Furthermore, by comparing the mean SQG quotient in a sample to the degree and incidence of toxicity in previous surveys (e.g., probability tables from Fairey *et al*., 2001; Long *et al.*, 2000; Ingersoll *et al*., 2001), the analyst can estimate the relative degree of risk posed by the contamination in the sample.

Multiple base maps that show gradients or other patterns in concentrations for numerous individual chemicals can be important elements of a sediment quality assessment. However, they can be confusing, especially when there are multiple gradients for multiple chemicals in a complex study area. By condensing the chemical data into a single index, the analyst can display, describe, and monitor just one variable over time and space.

Because chemicals often co-vary in concentration with each other in sediments, it is logical to assume that if the concentrations of one chemical were reduced through remediation and/or source control, the risks posed by all the chemicals in the mixture also would be reduced. Therefore, in RI/FS studies of hazardous waste sites, such as Onondaga Lake, establishment of target cleanup levels with mean SQG quotients can be an effective approach to ensure reduction in contamination by mixtures of chemicals.

For Onondaga Lake, the mean PEC quotient for each station could be considered an index of the relative risk of sediment toxicity posed at that station by the full suite of CPOIs. Mean PEC quotients can therefore be used to rank various stations with respect to relative risk and thereby prioritize stations for various kinds of potential remedial actions. For example, if the mean PEC quotients at two stations are 20 and 2, the station with the higher quotient may be considered a candidate for active remediation, whereas the station with the lower quotient may be considered a candidate for natural recovery.

Most important in this project, a critical level of mean PEC quotients that is associated with elevated toxicity could be used to establish the areas of sediments most in need of remediation. That is, in addition to ranking various stations with respect to relative risk, mean PEC quotients can be used in a quantitative manner to delineate the areal extent of sediments with different degrees of risk, through contouring of the PEC quotients. This use can be valuable for estimating the size of areas that may warrant different kinds of potential remedial actions.

Classification of sediments based on mean PEC quotients generally is more useful than classifications based on individual PEC quotients for the various CPOIs that are present at the stations. Because individual CPOIs or classes of CPOIs typically have different spatial distributions, it is difficult to delineate and justify the overall contaminated areas using individual PEC quotients. The distribution of unacceptable levels of one CPOI may terminate at the point where the distribution of another CPOI starts, there may be gaps in the distributions of individual CPOIs, or there may be varying degrees of overlap among the distributions. Such spatial heterogeneity makes it difficult to classify samples based on PECs for individual CPOIs.

CPOIs typically are found as mixtures in aquatic environments, but the chemical makeup of the mixtures differs from place to place within most water bodies. Application of individual PECs to delineate the contaminated areas does not account for the presence of mixtures because each chemical is compared to its individual PEC as if it occurred alone. The challenges described above in using PECs of individual CPOIs to classify sediments can be addressed and/or overcome by using a single index of sediment quality that integrates the information provided by the individual PECs. The mean PEC quotient is such an index. A single chemical index provides a uniform, consistent basis for classifying sediment quality throughout a chemically heterogeneous area. If justified against measures of biological effects, such an index retains the toxicological relevance on which the underlying SECs were derived. However, in contrast to use of PECs of individual CPOIs, a single integrative index has the advantage of taking into account the presence of chemicals in various mixtures and their additive effects.

J.3.3 THE HISTORY OF USE OF MEAN SQG QUOTIENTS

The mean SQG quotient approach has been developed and applied for classification of sediment contamination in both saltwater and freshwater environments. Mean SQG quotients initially were calculated in evaluations of chemistry data generated in sediment quality surveys conducted in marine bays and estuaries by the National Oceanic and Atmospheric Administration (NOAA), as a part of its National Status and Trends Program. In partnerships with the states of Florida and California, surveys were conducted in Tampa Bay (Long *et al*., 1994; Carr *et al.,* 1996) and San Pedro Bay (Sapudar *et al*., 1994), respectively. In the reports from both surveys, mean ER-M quotients were calculated to compare and rank the degree of contamination among sampling sites using the ER-L and ER-M values of Long *et al*. (1995). Mean ER-M quotients were calculated in subsequent NOAA surveys of Boston Harbor (MA), Savannah River (GA), Charleston Harbor (SC), Biscayne Bay (FL), western Florida panhandle (FL), Sabine Lake (TX), San Diego Bay (CA), and Puget Sound (WA). Using the PEL values derived by MacDonald *et al*. (1996), mean PEL quotients were calculated for the report on the Biscayne Bay survey (Long *et al*., 1999).

The relationships between the degree of response and percent incidence of toxicity in laboratory toxicity tests and ranges in both the mean ER-M quotients and the mean PEL quotients were evaluated with large data sets compiled from saltwater studies completed nationwide (Long *et al.,* 1998b; 2000). In their evaluations of regional monitoring data from San Francisco Bay, Thompson *et al*. (1999) reported correlations between mean ER-M quotients and percent amphipod survival. Fairey *et al*. (2001) revised the manner in which the quotients were calculated by blending some ER-Ms, some PELs, and other SQGs to discover the best statistical relationships between mean SQG quotients and both percent incidence of toxicity and mean amphipod mortality. Results of toxicity tests were compared among ranges of increasing mean ER-M quotients in Southern California (Bay *et al.*, 2000). Using the Washington State sediment quality standards, Long *et al*. (2003) described the degree of contamination in Puget Sound using mean Sediment Quality Standards (SQS) quotients. Studies were designed and conducted in Sydney Harbor, Australia, to quantify the predictive abilities of North American SQGs, including the mean ER-M, mean PEL, and mean SEC quotients (McCready *et al*., 2003).

Using the PECs calculated by MacDonald *et al*. (2000b) for freshwater environments, exposure/response relationships were determined in several studies (Crane *et al*., 2000, 2002; Ingersoll *et al*., 2001, 2002; USEPA, 2000c). In a study of the St. Louis River area of concern (Crane *et al*., 2000, 2002), mean PEC quotients were compared to results of both the amphipod and chironomid 10-day survival tests. The exposure/response relationships observed in the study site were compared to those in data compiled from other Great Lakes sites. In the Ingersoll *et al*. (2001) paper, regional differences in these relationships were described. Exposure/response relationships were described for both amphipod and chironomid survival tests with data from up to 600 samples. Similar analyses were performed with data from a survey of the Grand Calumet River and Indiana Harbor (Ingersoll *et al*., 2002). The predictive ability of mean PEC quotients was evaluated in separate databases generated for the southeastern U.S. (MacDonald *et al.* 2003a), the Tampa Bay estuary (MacDonald *et al*., 2003b), the Calcasieu estuary of Louisiana (MacDonald *et al*. (2002a), and the Columbia River Basin (MacDonald *et al*. 2002b).

The relationships between mean SQG quotients and the incidence of impaired benthic communities were described in data analyses conducted with estuarine data sets (Hyland *et al.*, 1999, 2003). The percentages of samples in which the benthos was classified as impaired invariably increased with increasing mean ER-M and PEL quotients, often over ranges in contamination that failed to induce a significant response in laboratory toxicity tests.

The predictive abilities of most currently available SQGs were summarized in large databases compiled for a Pellston Workshop on Sediment Quality Guidelines convened by the Society of Environmental Toxicology and Chemistry (Ingersoll *et al*., 2004; Wenning *et al*., 2004; Word *et al*., 2004). In this workshop, matching chemical and toxicological data from most major regions of the U.S. were compiled and evaluated. Many of the data sets that were evaluated and compared involved the derivation of mean SQG quotients. As a part of this workshop, the attendees compared the exposure/response relationships among data sets using various mean SQG quotients and results of either toxicity tests or benthic community analyses. Data from many thousands of samples were compiled by the contributors to represent the current state of the science. In all cases tested thus far, the relative risks of toxicity and/or benthic community impairment invariably increased incrementally with increasing mean SQG quotients, regardless of which SQGs were used in the calculations.

In summary, the basic approach of evaluating sediment quality with mean SQG quotients has been applied in at least the following geographic areas:

- Tampa Bay, FL, sediment quality survey;
- Southern California bays and harbors surveys;
- San Francisco Bay estuary regional monitoring program;
- Puget Sound, WA, ambient monitoring program;
- Tampa Bay estuary monitoring program;
- Biscayne Bay, FL sediment quality survey;
- Grand Calumet River and Indiana Harbor risk assessment;
- St. Louis River area of concern risk assessment;
- Lake Calcasieu estuary, LA, risk assessment;
- Great Lakes (Great Lakes National Program Office, GLNPO) surveys of sediment quality;
- Columbia River Basin, BC, Canada, survey; and
- Southeastern U.S. estuaries sediment quality monitoring.

North American agencies and institutions that have used the basic approach include at least the following:

- National Oceanic and Atmospheric Administration;
- U.S. Geological Survey;
- U.S. Environmental Protection Agency;
- U.S. Fish and Wildlife Service;

- Washington State Department of Ecology;
- California State Water Resources Control Board;
- Florida Department of Environmental Protection;
- South Carolina Department of Natural Resources;
- Minnesota Pollution Control Agency;
- San Francisco Estuary Institute;
- Tampa Bay Estuary Program;
- Great Lakes National Program Office; and
- Southern California Coastal Water Research Project.

J.3.4 ASSUMPTIONS

A number of implicit and explicit assumptions are associated with the use of the mean SQG quotient approach. Some may not be obvious to analysts, while others might seem intuitive. None of them precludes the use of this assessment tool.

By normalizing the concentrations of different kinds of chemicals to their respective SQGs and then summing and averaging the individual quotients, there is an implicit assumption that the contributions of each chemical to toxicity are additive. That is, as the overall degree of chemical contamination increases, both the numbers of chemicals in the sediments and their concentrations increase simultaneously. Their effects are additive as summarized in a recent review (Eggen et al., 2004). The assumption is that they are not antagonistic (i.e., cancel each other) or something more than additive (i.e., synergistic). There is no experimental evidence of this additivity in nature for all chemicals for which there are SQGs. Because sets of commonly used SQGs (including the Onondaga Lake PECs) include a wide variety of chemicals, it is quite possible that the assumption of additivity is not correct. Antagonism between different chemicals has been reported in limited experiments (Oakden *et al.*, 1984). However, there is evidence of additivity in bioassays of clean sediments spiked with individual chemicals and combinations of chemicals (Sundelin, 1984; Swartz *et al*., 1988; Plesha *et al*., 1988; McLeese and Metcalfe, 1980; McLeese *et al.*, 1982). Results of these experiments, although limited in scope, included tests that involved multiple chemical classes and indicated that responses in sediment bioassays generally were additive when chemicals were first tested separately, and then tested in combinations or mixtures. There is a considerable body of evidence from analyses of field-collected sediment samples that additivity in toxicity occurs where chemical mixtures exist, because the degree and incidence of response in biological tests invariably increases with increasing concentrations of the mixtures (Ingersoll et al., 2001, 2002; Crane et al., 2002; Fairey et al., 2001; Hyland et al., 2003; Long et al., 2000; MacDonald et al., 2004; Thompson et al., 2000; US EPA 2002c; Wenning et al., 2004; Word et al., 2004). As will be described below in Section J.4, this evidence includes the data from Onondaga Lake.

By using this approach, the assumption is made that all chemicals accounted for with SQGs have the same mode (mechanism) of toxicity. Specifically, they are assumed to be acting as acute toxins. These substances have been referred to as narcotics (Di Toro and McGrath, 2000). Most of the commonly used effects-based SQGs were derived with data from acute toxicity tests or, less commonly, benthic community indices. Thus, the assumption is made that by summing and averaging the concentration/SQG quotients for multiple chemicals that these chemicals act the same way in the test samples as they did in the studies performed to derive the SQGs. That is, we must assume that these chemicals are not acting primarily, for example, as teratogens, pro-mutagens, mutagens, carcinogens, etc. If chemicals such as benzo[a]pyrene, polychlorinated biphenyls, and mercury have multiple modes of action, it is assumed that in this situation, they are acting as acute toxins. It follows, then, that the mean SQG quotients are not being used to predict bioaccumulative effects in fish, wildlife, or humans. An independent bioaccumulation-based mercury sediment target is discussed in Appendix I and utilized in the FS.

The chemicals that are accounted for in the calculations of mean SQG quotients may differ in importance in each sample with respect to causing toxic effects, but they nevertheless act together as a mixture to elicit a toxic effect. One chemical or class of chemicals at a site may be the major drivers of risk and toxicity, but they do not act alone. Some chemicals in the mixture at a site may have a relatively minor contribution to a measure of toxicity compared to other substances in the sediments there. It is assumed that the causes of toxicity are attributable to the mixtures of chemicals in the sediments, including those not measured or not accounted for with the SQGs. If this assumption is true, then it follows that site-specific SQGs derived from analyses of matching chemistry and biological data acquired in a small-scale study probably cannot be accurately applied to other sites where different mixtures of chemicals exist.

Consistent with this observation, sediment standards developed specifically for Puget Sound were not recommended for application in other regions (USEPA, 1989). That is, site-specific guidelines developed by analysis of data from a site contaminated primarily by PCBs and mercury, for example, would not be expected to accurately predict toxicity in another site in which the chemicals of concern were primarily chlorinated pesticides or aromatic hydrocarbons. It follows, then, that the site-specific PECs developed for Onondaga Lake would not be accurate predictors of toxicity elsewhere and vice versa. However, it is most likely that the site-specific PECs for the lake would be very accurate in predicting toxicity there.

The mean PEC approach assumes that samples with the same or similar numerical quotients, but with different chemical characteristics, would have the same or similar probabilities of being toxic if they were subjected to toxicity testing. That is, it is assumed that the relative toxicological risk posed by two samples with the same mean SQG quotients, but with different mixtures or proportions of chemicals, would be the same. Otherwise, the chemical index scores generated by calculating mean SQG quotients would be meaningless as estimates of relative risk. This assumption, taken to its extreme, could mean that samples with extremely different chemical characteristics, but the same mean SQG quotients would be classified the same. For example, a sample with only one chemical elevated in concentration above the SQG and no other chemicals present at detectable concentrations and a second sample with multiple chemicals only slightly elevated in concentrations could have the same mean SQG quotients and, therefore, ostensibly the same probability of causing toxicity.

There is evidence that the additivity described above is a safe assumption because both the degree of mortality and the percentages of samples classified as toxic tend to increase in a predictable pattern as mean SQG quotients increase in large data sets composed of samples with differing chemical mixtures (Fairey *et al*., 2001; Long *et al*., 2000; Ingersoll *et al*., 2001). Such exposure/response relationships have been reported in both saltwater and freshwater systems and all of them look quite similar (Word et al. 2004). As will be described below in Sections J.4 and J.5, the degree of mortality in both the chironomid and amphipod tests increased in a consistent pattern as the mean PEC quotients for Onondaga Lake increased, thereby providing additional evidence of additivity.

J.3.5 LIMITATIONS

Potential users of the mean PEC quotient approach should be aware of several important limitations in the approach. None of these limitations precludes use of this assessment tool.

First, the mean SQG quotients, by themselves, are meaningless as pass-fail criteria. Intuitively, a mean quotient of 1.0 could be viewed as mean SQG unity and indicative of an important point in a scale of contamination. A mean quotient of 1.0 could mean that, on average, the chemical concentrations were the same as the SQGs. However, the points along the chemical contamination scale where safe conditions and toxicity conditions are expected can differ considerably. They can differ as functions of the kind of SQG that was used and the sensitivity of the type of toxicity that is predicted with the mean SQG quotient. The most accurate and objective way to establish safe levels and toxic levels using this tool is to calibrate the mean SQG quotients to the results of independent tests of toxicity or benthic effects such as those previously reported (Long *et al.*, 1998b, 2000; Fairey *et al*., 2001; Ingersoll *et al*., 2001; Crane *et al.*, 2000, 2002; Hyland *et al*., 1999, 2003). Many of the PECs calculated for Onondaga Lake (with the exception of mercury) were considerably lower in concentration than other SQGs of the same kind, including the freshwater PECs of Ingersoll *et al.* (1996). The Onondaga Lake PECs were derived with the intent of identifying consensus-based, midrange risk levels. As a result, the intuitive critical value of 1.0 may not be accurate as a predictor of toxicity for the lake. Any such critical value must be determined empirically by calibrating the mean SQG quotients to measures of adverse effects, such as acute toxicity.

Second, it is important to recognize that this tool is not a panacea. It should not be used as a stand-alone assessment tool for classifying sediment quality. Use of this tool should not preclude evaluation of the raw data. Other chemicals for which there are no SQGs may be important at the site, especially for measures of effects other than acute toxicity. As is the case with any multi-parameter index, by condensing data from multiple chemicals into one index, information on individual chemicals will be masked. Analysts should use as many lines of evidence as possible to classify sediment quality.

Third, it may seem obvious, but mean SQG quotients can only be used to account for the presence and concentrations of chemicals for which the chemical concentration/SQG quotients were calculated. That is, the mean quotients cannot be assumed to be surrogates for other un-measured substances or measured substances for which there were no SQGs. They may, in fact, be accurate surrogates of the presence of other substances, but such co-variance must be determined empirically.

Fourth, because the mean SQG quotients were not initially derived as regulatory standards or criteria, their usage is not widely addressed in regulatory or risk assessment methods manuals. However, the derivation and interpretation of mean PEC quotients is included in guidance manuals issued for sediment quality assessments by the Great Lakes National Program Office of USEPA (USEPA, 2002a, 2002b, 2002c).

In summary, the primary benefits of the mean PEC quotient approach are as follows:

- It has been used in the published literature and therefore has been peer reviewed, enhancing its credibility and likelihood of success;
- It condenses information on numerous CPOIs into a single index of relative risk that can be used in a quantitative manner to rank stations and contour areas that may warrant different kinds of potential remedial actions;
- It can be used to estimate which CPOIs or classes of chemicals pose the highest and lowest degrees of risk at individual stations, and thereby help guide remedial decisions; and
- The resulting mean PEC quotients can be related to site-specific biological effects, so that their toxicological meaning can be determined empirically and used to establish critical ranges of quotients that can identify those sediments that pose little or no ecological risks and those that are most in need of remediation and/or source controls.

SECTION J.4

APPLICATION OF THE MEAN PEC QUOTIENT APPROACH TO ONONDAGA LAKE

The mean PEC quotient approach was applied to the surface sediments (i.e., 0 to 15 cm) collected in Onondaga Lake in 1992 and 2000. The approach was applied to the lake sediments to establish that there was a significant relationship between the degree of contamination as gauged by the mean PEC quotients and the degree of response in the laboratory toxicity tests. The general exposure/response relationships were examined to determine if there was an inflection point in the quotients where the degree of response (percent mortality) indicated an initial noticeable upturn or increase. The inflection point was identified as the lower end of a range in mean PEC quotient values in which mortality first showed a consistent increase. The lower end of this range in values was then proposed as the critical value below which toxicity attributable to the mixtures of chemicals in the lake sediments would not be expected, while taking into account adjustments made to account for data scatter, acute (rather than chronic) toxicity testing, relatively insensitive species, and a lack of agreement with other published values.

The calculations were completed with several methods in iterative trials to see if the same results were obtained each time. In the first method, all 46 site-specific PECs (i.e., the actual sediment quality values rather than quotients) calculated by the NYSDEC were used in the analysis. Importantly in the first method, the calculations included undetected values treated as one-half the reported method detection limit (MDL). This meant that chemical concentrations that were very low, but unknown, were treated as a real number and the number was established as one-half the MDL. In the second method, the undetected values were excluded in each sample and the denominators for calculating the quotients were adjusted accordingly. Use of hexachlorobenzene data was complicated by the presence of two sets of data for each sample, resulting from the use of two different analytical methods. Therefore, the NYSDEC approach to handling hexachlorobenzene data (either by using their calculated values, where available, or by calculating them independently by using the approach described in Appendix A of the human health risk assessment) was considered. For PCBs, only the concentrations of total PCBs were used, thereby excluding concentrations of four individual Aroclor mixtures. All calculations were based on the 0 to 2 cm data from 1992 and the 0 to 15 cm data from 2000. There were no 0 to 15 cm data available for 1992.

To biologically calibrate the mean PEC quotients calculated for Onondaga Lake, the quotients were compared with the toxicity test results (i.e., percent mortality) obtained for the 10-day chironomid and amphipod sediment toxicity tests conducted at 79 stations in 1992 (Figures J.1 and J.2). In this first trial, the database was arranged in ascending order and divided into either six or seven ranges and average percent mortality was calculated for the samples that fell into those ranges. The ranges were selected by visually examining the database to identify obvious breaks or changes in percent mortality as the mean PECQs increased. Although 40-day chironomid and amphipod tests were conducted in 2000, they were not used in this evaluation because they were only conducted at a small number of stations (i.e., 15 stations). Therefore, the data from both the 10-day chironomid and amphipod tests were used to evaluate the mean PEC quotients in these analyses.

Based on the chironomid test, average percent mortality showed similar patterns regardless of whether undetected values were represented by one-half the detection limits or whether they were omitted from the analysis (Figure J.1). In both cases, the exposure/response relationship did not show a noticeable increase until the mean PEC quotient range of 2 to 5 was reached. In the former case, mean mortality exhibited a very narrow range of 11.8 to 12.5 percent and no upward trend in the lowest three quotient categories (i.e., 0 to 0.5, 0.5 to 1, and 1 to 2), then showed a marked increase to 21.9 to 23.7 in the fourth and fifth quotient categories (i.e., 2 to 5 and 5 to 10), and finally peaked at 73.2 percent in the final category (i.e., >10). When undetected values were excluded from the analysis, mean chironomid mortality exhibited a very narrow range of 11.3 to 11.9 percent and no upward trend in the lowest three quotient categories (i.e., 0 to 0.5, 0.5 to 1, and 1 to 2), increased slightly to 14.8 percent in the fourth category (i.e., 2 to 3), showed a marked increase to 31.6 to 34.4 percent in the fifth and sixth quotient categories (i.e., 3 to 5 and 5 to 10), and peaked at 77.3 percent in the final category (i.e., >10).

As described for the chironomid test above, mean mortality in the amphipod test showed similar patterns regardless of whether undetected values were represented by one-half the detection limits or whether they were omitted from the analysis (Figure J.2). In both cases, the exposure/response relationship did not show a noticeable increase until the mean PEC quotient range of 1 to 2 was reached. In the former case, mean mortality exhibited a very narrow range of 5.0 to 5.4 percent in the lowest two quotient categories (i.e., 0 to 0.5 and 0.5 to 1), increased slightly to 8.1 percent in the third category (i.e., 1 to 2), then showed a more marked increase to 14.0 to 14.3 in the fourth and fifth quotient categories (i.e., 2 to 5 and 5 to 10), and peaked at 44.7 percent in the final category (i.e., >10). When undetected values were excluded from the analysis, mean amphipod mortality exhibited a very narrow range of 4.9 to 5.5 percent in the two lowest quotient categories (i.e., 0 to 0.5 and 0.5 to 1), increased slightly to 8.0 to 8.2 percent in the third and fourth categories (i.e., 1 to 2 and 2 to 3), showed a more marked increase to 15.4 to 16.6 percent in the fifth and sixth quotient categories (i.e., 3 to 5 and 5 to 10), and peaked at 54.0 percent in the final category (i.e., >10).

In the second trial, somewhat different ranges of data were selected to help better pin down the points in the lower end of the scale that may be the inflection points. In this trial a total of seven ranges also was identified by visually examining the database. The results are listed in Table J.1 (non-detects treated as 1/2 the MDLs) and Table J.2 (non-detects omitted). With the non-detected concentrations treated as 1/2 their respective MDLs, average percent mortality among the amphipods was 5.0 to 5.4 with mean PECQs of <0.5 and 0.5 to 1.0. Mortality increased slightly to 7.8 percent in the next range of values (1.01 to 1.34) and continued to increase incrementally as the mean PECQs increased. In the chironomid test, the average percent mortality ranged from 10 to 14 percent in the lowest four ranges in mean PECQs, then increased abruptly to 22 and 24 percent in the ranges of mean PECQs of 2 to 5 and 5 to 9.9. Therefore, this somewhat different set of ranges in data again showed that the first upturn in mortality occurred as mean PECQs reached values of either about 1 (amphipods) or about 2 (chironomids). It is noteworthy that the numbers of samples within each of these ranges differed considerably from each other, with higher numbers of samples in the lower mean PECQ ranges.

TABLE J.1

AVERAGE PERCENT MORTALITY WITHIN 7 RANGES IN MEAN PEC QUOTIENTS

Mean PECQs (1/2 MDLs)	n	Percent Amphipod Mortality	Percent Chironomid Mortality
<0.5	22	5.0	12.5
0.5 - 1.0	26	5.4	11.8
1.01 - 1.34	5	7.8	10.0
1.4 - 1.9	6	8.3	14.0
2.0 - 5.0	7	14.0	21.9
5.0 - 9.9	6	14.3	23.7
>10	6	44.7	73.2

Note: Non-detects treated as 1/2 MDLs

Table J.2 lists the results for the analyses with the non-detected concentrations omitted. Results similar to those with 1/2 the MDLs used were apparent. In this case, the points at which the first increase in mortality occurred appeared to be approximately 1.4 for the amphipod tests and, again, 2.0 for the chironomids. Therefore, these data indicated that percent mortality was consistently lowest in the initial ranges of mean PECQs, and first began to increase as values of 1.4 (amphipods) and 2.0 (chironomids) were reached. Again, as in the previous trial, the numbers of samples differed among the ranges in mean PECQs.

TABLE J.2
AVERAGE PERCENT MORTALITY WITHIN 7 RANGES IN MEAN PEC QUOTIENTS

Mean PECQs (detects only)	n	Percent Amphipod Mortality	Percent Chironomid Mortality
<0.5	12	4.9	11.3
0.5 - 1.0	31	5.5	11.9
1.01 - 1.34	8	4.8	12.8
1.4 - 1.9	7	11.7	10.9
2.0 - 5.0	10	11.8	23.2
5.0 -9.9	7	16.6	34.4
>10	4	54.0	77.3

Note: Non-detects omitted

To avoid having different numbers of samples in each range of values, a third set of trials was evaluated. In the third set of trials, the ranges in values were not established visually, but, rather, by selecting the same numbers of samples within each range. That is, the mean PECQ values were sorted in ascending order and the database divided into eight ranges, each with equal numbers of samples (i.e., 10 in each, except in the last range with 9). This approach may appear to be more objective than the others. Again, the non-detected concentrations were treated as one-half the MDLs (Table J.3) or were omitted (Table J.4). With both methods, average percent mortality among the amphipods remained consistently lowest (about 4 to 6 percent) until the ranges of 1.1-1.9 and 1.3-2.2 were reached. Average percent mortality increased slightly to 8 percent (1/2 MDLs used) and 10 percent (non-detects omitted) in those ranges and continued to increase in subsequent ranges.

The chironomid data also gave similar results in both data treatments (Tables J.3 and J.4). Average percent mortality among the chironomids was lowest (about 7 to 16 percent) and indicated no upward trend in the six least contaminated ranges. The first upturn in mortality occurred in samples within the ranges of mean PECQs of 2.6-6.4 (1/2 MDLs used) and 2.2-5.0 (non-detects omitted). Average percent mortality increased to about 18 and 20 percent in those two ranges, respectively, and continued to increase in the most contaminated samples. Note that by omitting the results for Station S48, there was a decrease in average mortality of about 4 percent within the mean PECQ range of 0.4-0.6. Station S48 exhibited high chironomid mortality (i.e., 54 percent) but had a low mean PECQ (i.e., 0.3).

TABLE J.3
AVERAGE PERCENT MORTALITY
WITHIN 8 RANGES IN MEAN PEC QUOTIENTS

Mean PECQs (1/2 MDLs)	n	Percent Amphipod Mortality	Percent Chironomid Mortality
0.1 - 0.3	10	4.2	11.0
0.3 - 0.4	10	5.9	13.8
0.4 - 0.6	10	6.5	16.4*
0.6 - 0.7	10	3.9	7.2
0.7 - 1.1	10	6.2	11.8
1.1 - 1.9	10	8.0	12.8
2.6 - 6.4	10	14.9	18.3
7.5 - 70.7	9	33.7	61.2

Notes: 1) Non-detects treated as 1/2 MDLs
2) * 12.2% without Station S48

TABLE J.4
AVERAGE PERCENT MORTALITY
WITHIN 8 RANGES IN MEAN PEC QUOTIENTS

Mean PECQs (detects only)	n	Percent Amphipod Mortality	Percent Chironomid Mortality
0.1 - 0.4	10	4.2	10.6
0.5 - 0.6	10	7.2	15.8
0.6 - 0.8	10	4.8	12.6*
0.8 - 0.9	10	4.0	9.2
1.0 - 1.2	10	5.9	11.8
1.3 - 2.2	10	10.4	13.2
2.2 - 5.0	10	13.1	20.5
7.0 - 17.8	9	33.7	58.6

Notes: 1) Non-detects omitted
2) * 8.0% without Station S48

Because variability and gaps in the database could cause too much uncertainty in the identification of tighter inflection points, the inflection points identified for the two toxicity tests were represented by ranges of mean PEC quotients. Additional variability may have been caused by the sampling design (matching the chemistry samples with the toxicity test samples), inaccuracies in determinations of the true chemical concentrations, and the tests selected to measure the true degree of toxicity. Such sources of variability are normal and expected in any sediment quality assessment. Previously published analyses of the relationships between mean SQG quotients and toxicity have relied on data from hundreds and even thousands of samples to limit data gaps. The Onondaga Lake database, although of high quality and highly important for this site-specific application, relied on data from only 79 samples. All of these factors combined would tend to cause the degree of variability that was apparent in these results for Onondaga Lake.

Exposure-response relationships published previously from freshwater and saltwater studies (Long *et al.*, 2000; Fairey *et al*., 2001; Ingersoll *et al.*, 2001; Crane *et al*., 2000, 2002) showed gradual increases in both the degree of response and the percent incidence of toxicity within ranges of increasing mean SQG quotients, much like the trend that is apparent in the data for Onondaga Lake. Therefore, the relationship between contamination as gauged with the mean PEC quotients and the response as measured with the 10-day chironomid and amphipod tests in Onondaga Lake is not unusual or unique. Instead, it is quite comparable to those previously published with other data sets. However, it is important to understand that the inflection points in the ascending chemical concentrations where toxicity begins to increase above background levels tends to differ among data sets and regional study areas because of differences in methods, the SQGs used, and the relative sensitivities of test species.

SECTION J.5

USE OF CHEMICAL GROUPS FOR EVALUATION OF MEAN PEC QUOTIENTS IN ONONDAGA LAKE FEASIBILITY STUDY

In the methods described in Section J.4, the mean PEC quotients were calculated using all chemical measurements made at each station, with the concentrations of undetected chemicals either omitted or estimated as one-half the detection limits. Although the two test species (i.e., the chironomid *Chironomus tentans* and the amphipod *Hyalella azteca*) and the two methods of treating the chemistry data provided somewhat different results, a trend of increasing response in the toxicity tests (i.e., percent mortality) with increasing mean PEC quotients was found for all iterations. The inflection points at which mortality appeared to first increase noticeably above background or baseline conditions ranged from 1.0 for the amphipod test to 2.0 for the chironomid test.

This section evaluates an alternative approach, based on the use of data for selected chemical groups, rather than all 46 individual CPOIs evaluated in the lake, to calculate mean PEC quotients. This approach was evaluated to determine if the exposure/response relationships observed in the original methods using all the chemical data could be improved substantially. The primary goal of this evaluation was to reduce the list of CPOIs to those chemicals that appeared to have the greatest influence on sediment toxicity in Onondaga Lake sediments. It was hypothesized that by reducing the list of COPCs to those that were the most significant contributors to toxicity, the exposure/response relationship would improve, and the critical range of mean PEC quotients could be better identified. This kind of revision to the basic method of calculating mean SQG quotients has been reported on a nationwide basis (Fairey *et al*., 2001). In that study of saltwater sediments, the correlation coefficients for amphipod survival and mean SQG quotients peaked at $r^2 > 0.9$, based on data for only nine chemicals (five trace metals and four organic compounds).

J.5.1 EVALUATION METHODS

As discussed in the previous section, the NYSDEC developed SQGs, including PECs, for 46 CPOIs in Onondaga Lake. For the present analysis, the 46 CPOIs were subdivided into the following six chemical groups consisting of detected COPCs having similar chemical properties:

- Benzene, toluene, ethylbenzene, and xylenes (BTEX)
- Chlorinated benzenes (monochlorobenzene, dichlorobenzenes, trichlorobenzenes, and hexachlorobenzene)
- PAHs, including 16 individual compounds
- Total PCBs (but not individual Aroclors®) and pesticides (including chlordanes and DDT and metabolites)

- Miscellaneous organic compounds (phenol and dibenzofuran)
- Metals (antimony, arsenic, cadmium, chromium, copper, lead, manganese, mercury, nickel, selenium, silver, vanadium, and zinc)

For each CPOI, only detected concentrations were used in the analysis to remove uncertainties related to undetected (i.e., unknown) values.

The potential contributions of the various chemical groups were evaluated primarily by conducting polynomial regressions of chironomid mortality on mean PEC quotients, then using the resulting r^2 values to determine the strength of various relationships. Second-order polynomials were used for each kind of mean PEC quotient, because they provided good approximations of exposure/response relationships.

To evaluate the relative importance of each chemical group, the group was removed from the database and a polynomial regression was determined. The results of that analysis were then compared to the regression based on all five groups. If the r^2 value for the remaining groups declined relative to the r^2 value for all five groups, that group was considered important in explaining the observed chironomid mortality (i.e., because its removal weakened the relationship). By contrast, if the r^2 value for the remaining groups increased or remained relatively unchanged compared to the r^2 value for all chemical groups, that group was considered unimportant in explaining the observed chironomid mortality (i.e., because its removal strengthened or had no effect on the relationship). A regression was then evaluated for chironomid mortality and for each group independently to further confirm and evaluate the strength of the relationship.

For chemical groups determined to be important in explaining the observed chironomid mortality, further evaluation of their component CPOIs generally was considered unnecessary. Such results were assumed to indicate that concentrations of the individual chemicals in each group co-varied with each other. However, for groups determined to be unimportant, regressions were evaluated for each of their component CPOIs to determine whether one or more of the CPOIs showed a strong relationship to chironomid mortality.

It should not be assumed that the chemicals most correlated with toxicity were the actual cause(s) of toxicity. Strong statistical correlations do not necessarily represent causation. Other toxic chemicals, such as ammonia, that were not measured, but co-varied in concentrations with the measured chemicals, may have actually caused the toxicity that was observed. Additional studies such as toxicity identification evaluations or exploratory laboratory experiments would be required to accurately determine the actual cause(s) of toxicity in these sediments. However, in the absence of such experimental studies, the CPOIs most correlated with toxicity were assumed to most likely have contributed to toxicity.

J.5.2 RESULTS

Prior to conducting the regression analysis, scatter plots were evaluated for seven CPOIs represented by six or fewer detected concentrations to determine qualitatively whether a potential relationship existed between concentrations of these chemicals and chironomid mortality. Those CPOIs included antimony (n=5), ethylbenzene (n=2), xylenes (n=5), trichlorobenzenes (n=6), pesticides (n=4), phenol (n=2), and dibenzofuran (n=5). The results are presented in Figures JA.1 to JA.7 of Attachment A to this appendix. Potential relationships were found for ethylbenzene, xylenes, trichlorobenzene, and pesticides; therefore, those four CPOIs were carried forward to regression analysis. By contrast, no apparent relationship with mortality was observed for antimony, phenol, or dibenzofuran. These three CPOIs were therefore not retained as CPOIs for further analysis.

The polynomial regression of chironomid mortality on mean PEC quotients for all five chemical groups retained for analysis is presented in Figure J.3. The group based on miscellaneous organic compounds was not included because its component CPOIs (i.e., phenol and dibenzofuran) were eliminated in the initial screening evaluation described above. The r^2 value of 0.41 for the five chemical groups was used as a baseline with which to evaluate the alternative mean PEC quotients based on exclusion of various chemical groups.

The results of the regression analysis for each of the five chemical groups retained for further evaluation are presented below.

BTEX

The regression for mean PEC quotients based on all groups except BTEX compounds is presented in Figure J.4. Because its r^2 value (0.51) was considerably higher than the value of 0.41 observed for all five groups combined, it was concluded that the BTEX group was not important in explaining chironomid toxicity. The regression of mortality on mean PEC quotients for the BTEX group (Figure J.5) confirmed this conclusion, because the r^2 value was low (0.12). That is, the exposure/response relationship improved over that derived with all chemicals when BTEX compounds were excluded, and the correlation between the concentrations of BTEX compounds and chironomid mortality showed high variability. Therefore, the data suggest that this group of chemicals was not among the chemicals that strongly influenced the outcome of the toxicity tests.

The r^2 values for regressions based independently on benzene (0.03), toluene (0.25), and xylenes (0.42) indicated that only xylenes showed a strong relationship with chironomid mortality (Figures JA.8 through JA.10 of Attachment A). Xylenes were therefore retained as CPOIs within the BTEX group. As discussed previously, although ethylbenzene was detected at only two stations in 1992, chironomid mortality exhibited an increasing trend in relation to increasing ethylbenzene concentrations. Although only 15 stations were evaluated for chironomid sediment toxicity in Onondaga Lake in 2000, those results were used to further evaluate whether ethylbenzene may be a contributor to sediment toxicity in the lake (Figure

JA.11 of Attachment A). Because the r^2 value for that regression was very high (0.90), ethylbenzene was retained with xylenes in the BTEX group.

Chlorinated Benzenes

The regression for mean PEC quotients based on all groups except chlorinated benzenes is presented in Figure J.6. Because its r^2 value (0.32) was lower than the value of 0.41 observed for all five groups combined, it was concluded that the chlorinated benzene group was important in explaining chironomid toxicity. The regression of mortality on mean PEC quotients for the chlorinated benzene group (Figure J.7) confirmed this conclusion, because the r^2 value was relatively high (0.33). Chlorinated benzenes were therefore initially retained as a complete group. However, because the reduction in r^2 value for groups was only moderately reduced following the exclusion of the chlorinated benzene group, the r^2 values for regressions based independently on monochlorobenzene (0.22), dichlorobenzene (0.36), trichlorobenzene (0.94), and hexachlorobenzene (0.06) were evaluated (Figures JA.12 through JA.15 of Attachment A). These results indicated that dichlorobenzene and trichlorobenzene should be retained in the group (i.e., because their r^2 values were high) and that hexachlorobenzene should be excluded from the group (i.e., because its r^2 value was very low). Because the r^2 value for monochlorobenzene was moderate in magnitude, further evaluations were conducted using the 2000 chironomid toxicity data (Figure JA.16 of Attachment A). Because the r^2 value for the 2000 data was relatively high (0.31), monochlorobenzene was retained with dichlorobenzene and trichlorobenzene in the chlorobenzene group.

PAHs

The regression for mean PEC quotients based on all groups except PAHs is presented in Figure J.8. Because its r^2 value (0.28) was considerably lower than the value of 0.41 observed for all five groups combined, it was concluded that the PAH group was important in explaining chironomid toxicity. The regression of mortality on mean PEC quotients for the PAH group (Figure J.9) confirmed this conclusion, because the r^2 value was relatively high (0.44). PAHs were therefore retained as a complete group.

PCBs/Pesticides

The regression for mean PEC quotients based on all groups except PCBs/pesticides is presented in Figure J.10. Because its r^2 value (0.42) was nearly identical to the value of 0.41 observed for all five groups combined, it was concluded that the PCBs/pesticides group was not important in explaining chironomid toxicity. The regression of mortality on mean PEC quotients for the PCBs/pesticides group (Figure J.11) confirmed this conclusion, because the r^2 value was low (0.05).

The r^2 values for regressions based independently on total PCBs (0.03) and pesticides (1.0) indicated that only pesticides showed a strong relationship with chironomid mortality (Figures JA.17 and JA.18 of Attachment A). However, because pesticides were represented by only four data points, additional evaluations were conducted using the 2000 data on chironomid toxicity (Figure JA.19 of Attachment A). In addition, because total PCBs were found to be elevated at numerous stations in the southern part of Onondaga Lake, the 2000 data were used to provide an additional evaluation of those chemicals (Figure JA.20 of Attachment A). Because the r^2 value for pesticides was very low (0.05) and the value for total PCBs was relatively high (0.29), only total PCBs were retained in the PCBs/pesticides group.

Metals

The regression for mean PEC quotients based on all groups except metals is presented in Figure J.12. Because its r^2 value (0.39) was similar to the value of 0.41 observed for all five groups combined, it was concluded that the metals group was not important in explaining chironomid toxicity. The regression of mortality on mean PEC quotients for the metals group (Figure J.13) confirmed this conclusion, because the r^2 value was low (0.14).

The r^2 values for regressions based independently on the metals that comprised the metals group (Figures JA.21 through JA.32 of Attachment A) were as follows:

- Arsenic: 0.24
- Cadmium: 0.01
- Chromium: 0.03
- Copper: 0.05
- Lead: 0.23
- Manganese: 0.01
- Mercury: 0.30
- Nickel: 0.04
- Selenium: 0.05
- Silver: 0.09
- Vanadium: 0.08
- Zinc: 0.03.

The results presented above indicate that only mercury showed a strong relationship with chironomid mortality. Mercury was therefore retained as the only CPOI within the metals group.

J.5.3 SUMMARY AND EVALUATION OF REVISED MEAN PEC QUOTIENTS

Based on the results of the evaluations discussed above, revised mean PEC quotients were calculated based on the concentrations of 23 of the original 46 CPOIs in Onondaga Lake:

- BTEX (ethylbenzene and xylenes)
- Chlorinated benzenes (monochlorobenzene, dichlorobenzenes, and trichlorobenzenes)

- PAHs (all 16 individual compounds)
- PCBs (total PCBs)
- Metals (mercury).

The polynomial regression of chironomid mortality on mean PEC quotients for the revised chemical groups is presented in Figure J.14. Its r^2 value of 0.52 is considerably higher than the r^2 value of 0.41 based on the original chemical groups (see Figure J.3), indicating that the revised groupings strengthens the relationship between chironomid mortality and mean PEC quotients. One unusual outlier point on Figure J.14 exhibited relatively high chironomid mortality (i.e., approximately 54 percent) at a very low mean PEC quotient (i.e., 0.3). That point corresponded to Station S48 in the lake and should be given special consideration when identifying potential areas of concern in the feasibility study. The correlations would improve if that data point were omitted from the calculations. Figure J.15 shows that the amphipod mortality results obtained in 1992 were related even more strongly to mean PEC quotients based on the revised groups than the results for the chironomid test, as the r^2 value was 0.62. Although long-term sediment toxicity tests were conducted at only 15 stations in Onondaga Lake in 2000, those results were also strongly related to mean PEC quotients based on the revised chemical groups (Figures J.16 through J.18). The r^2 values for 20-day chironomid mortality (0.55), 28-day amphipod mortality (0.30), and 42-day amphipod mortality (0.34) were all relatively high. Percent mortality appears to increase in these longer term tests as the revised mean PEC quotients reach a range of 3 to 5 on these graphs (Figures J.16-J.18). These additional results indicate that the revised chemical groups were useful for addressing long-term sediment toxicity in the lake.

The patterns of mean 10-day chironomid and amphipod mortality within ranges of increasing mean PEC quotients calculated with the revised list of 23 chemicals are presented in Figures J.19 and J.20, which were derived from the scatterplots presented in Figures J.14 and J.15. This information was used to determine whether inflection points in the exposure/ response relationships for the two toxicity tests could be identified, and if these inflection points differed from those identified with mean PEC quotients calculated with data for all 46 CPOIs evaluated in Onondaga Lake. The six ranges in mean PEC quotients were established by visual examination of the sorted, ascending database for each toxicity test.

For the chironomid test, mean mortality did not exhibit a marked upward trend until a quotient range of 2 to 5 was reached (Figure J.19). In the three lowest PEC quotient intervals (i.e., 0 to 0.5, 0.5 to 1, and 1 to 2), mean mortality exhibited a relatively narrow range (i.e., 8.3 to 14.9 percent) and no upward trend. However, in the fourth and fifth quotient intervals (i.e., 2 to 5 and 5 to 10), mean mortality increased markedly to 32.2 to 35.0 percent, respectively, and then peaked at 69.8 percent in the sixth quotient interval (i.e., >10). These results indicate that the degree of response in the chironomid test did not change appreciably until mean PEC quotients of 2 to 5 were reached. Therefore, it appears that the inflection point in the exposure/response relationship for the chironomid test was about 2.

For the amphipod test, mean mortality exhibited a relatively narrow range (i.e., 5.2 to 6.0 percent) and no upward trend in the two lowest mean PEC quotient intervals (i.e., 0 to 0.5 and 0.5 to 1). However, mean mortality increased slightly to 9.0 percent in the third interval (i.e., 1 to 2) and slightly again to 13.0 and 14.7 percent in the fourth and fifth intervals (i.e., 2 to 5 and 5 to 10), respectively. Mean amphipod mortality then peaked at 50.6 percent in the sixth quotient interval (i.e., >10). These results indicate that the degree of response in the amphipod test did not change appreciably until mean PEC quotients of 1 to 2 were reached, although the degree of change was much less pronounced than in the chironomid test. Therefore, it appears that the inflection point in the exposure/response relationship for the amphipod test was about 1.

The inflection points of 1 and 2 identified in the exposure/response relationships for the 10-day amphipod and chironomid tests (respectively) based on chemical groups (see Figures J.19 and J.20), were consistent with the inflection points for the two toxicity tests on the basis of all 46 CPOIs (see Section J.4; Figures J.1 and J.2; Tables J.1 to J.4). These results indicate that the reduction in the number of CPOIs did not affect the overall exposure/response relationships for the two toxicity tests. They also underline the robustness of the inflection points for the two toxicity tests, and increase confidence in the validity of the inflection points.

J.5.4 IDENTIFICATION OF CRITICAL MEAN PEC QUOTIENTS

The inflection points identified for the 10-day chironomid and amphipod tests in Sections J.4 and J.5 indicate where the test organisms began to exhibit increased mortality above background levels in relation to the mean PEC quotients. Therefore, the inflection points can potentially be used as critical values for identifying sediments that pose risks to benthic macroinvertebrates in Onondaga Lake. Based on chemical groups (i.e., the preferred method of calculating mean PEC quotients for the lake, as described earlier), mean chironomid mortality did not exhibit a marked upward trend until a quotient range of 2 to 5 was reached and mean mortality increased markedly from 13.0 to 32.2 percent (i.e., a substantial increase of 19.2 percent). Mean mortality at quotients less than 2 to 5 exhibited a relatively small range and no consistent upward trend. These results indicate that a mean PECQ of 2 (i.e., the minimum value of the mean PEC interval that defined the inflection point) may be the critical mean PEC quotient for the chironomid test in Onondaga Lake.

For the amphipod test, an upward trend in mean mortality did not occur until a mean revised PEC quotient range of 1 to 2 was reached. However, mean mortality at the inflection point increased only slightly from 5.2 to 9.0 percent (i.e., an increase of only 3.8 percent). Mean mortality at quotients less than 1 to 2 exhibited a very small range and no upward trend as the degree of contamination increased. These results indicate that a value of 1 could be considered the critical mean PEC quotient for the amphipod test, although the increase in mortality at the inflection point (i.e., 3.8 percent) was much lower than the increase in mortality at the inflection point for the chironomid test (i.e., 19.2 percent). In addition, mean amphipod survival at the inflection point was very high (i.e., 91 percent), indicating that meaningful toxic effects were not yet observed.

The analyses of the Onondaga Lake data performed with a variety of different methods in this appendix indicated that both sediment toxicity tests were stronglycorrelated with the mixtures of chemicals and several groups or classes of chemicals in the lake sediments. These observations are important because they suggest that the chemicals accounted for with the Onondaga Lake PEC values probably contributed to or may have caused the toxicity that was observed. In addition, these patterns suggest that the effects of confounding natural factors probably were minimal. The actual cause(s) of toxicity can only be determined with laboratory experiments (e.g., toxicity identification evaluations). However, in the absence of such studies, there is a reasonable body of evidence to indicate that at least some of these 46 chemicals (most likely the 23 CPOIs) contributed to the observed sediment toxicity. Therefore, based on these observations, it is reasonable to identify the degree of chemical contamination that was associated with an increase in the magnitude of toxic response as a critical value for identifying sediments that pose potential risks of sediment toxicity to benthic macroinvertebrates in the lake.

The analyses reported in this appendix identified mean PEC quotients of approximately 1 (for amphipods) and approximately 2 (for chironomids) as important points at which toxicity first appears to increase. The magnitudes of response in both toxicity tests were consistently low in less contaminated samples, indicating that the degree of contamination in those samples was not sufficient to cause elevated toxicity. It is important to note that these two values were identified as inflection points in multiple trials and analyses, each using several different methods. If different outcomes had resulted from the multiple iterative trials, the confidence in these values would be reduced. However, that was not the case. Therefore, it appears that mean PEC quotients of 1 and 2 can be used with confidence in the evaluation of contaminated sediments in Onondaga Lake.

SECTION J.6

CONCLUSIONS

The basic concept of classifying the relative degree of chemical contamination of sediments with mean SQG quotients has been used by sediment quality analysts since the mid-1990s. It has been used by several federal agencies, multiple state and provincial agencies, and many regional ambient monitoring programs in numerous states and Canadian provinces. It has also been accepted and published in the peer-reviewed scientific literature and in government publications. Analyses of large freshwater and saltwater data sets invariably show that both the degree of toxic response and incidence of toxicity increase as mean SQG quotients increase.

Using mean SQG quotients to evaluate sediment toxicity condenses complicated information from numerous chemicals into one effects-based index that accounts for both the presence of chemicals in the sediments and their concentrations relative to SQGs. Thus, mean SQG quotients can be used to interpret complicated data with a simple index for estimating relative risks of effects on benthic macroinvertebrates. However, an understanding of several important assumptions and limitations to the use of this tool is critical to using it correctly. This tool is used best in sediment quality assessments as one of multiple lines of evidence with which to classify the relative quality of sediments.

The application of this tool in the form of mean PEC quotients based on the 46 site-specific PECs for Onondaga Lake is an appropriate and defensible method for ranking and comparing the quality of sediments throughout the lake. Both the 10-day chironomid and amphipod survival tests provided useful information with which to identify the levels of chemical contamination that appeared to be important in Onondaga Lake. The degree of response (percent mortality) was lowest in the samples that were least contaminated, incrementally increased in samples with intermediate levels of contamination, and highest in the most contaminated samples. Thus, there is evidence that both test species were responsive to the increasing degree of contamination, not to some natural factors or confounding properties of the sediments.

The lake data were analyzed using several different methods in attempts to accurately identify the response patterns in the toxicity tests relative to the increasing degree of sediment contamination (i.e., the exposure/response relationships). Results were obtained by treating undetected chemical concentrations as either one-half the detection limit or by omitting those unknown concentrations; exclusion of undetected chemical concentrations was found to be the preferred method of calculating mean PEC quotients. In addition, statistical analyses identified a subset of grouped chemicals that was most correlated with toxicity and, therefore, may have most likely contributed to the observed toxicity. Revised mean PEC quotients were then calculated using a subset of five groups of 23 CPOIs that appeared to be the most important contributors to toxicity. It should not be assumed, however, that these chemicals could conclusively be identified as the actual cause(s) of the observed sediment toxicity. Additional studies using toxicity identification evaluations or exploratory experiments conducted in a laboratory would be required to accurately determine the actual cause of toxicity in these sediments.

The results of the evaluations of sediment toxicity in relation to mean PEC quotients indicate that inflection points in the exposure/response relationships appeared to occur within the mean PEC quotient range of approximately 1 to 2 for the amphipod test and approximately 2 to 5 for the chironomid test. The degree of response in the two toxicity tests consistently increased slightly over that in the least contaminated samples at these two points. These results were consistent regardless of whether mean PEC quotients were calculated using undetected chemical values or whether the quotients were calculated based on all 46 CPOIs or for the subset of five groups of 23 CPOIs. These or very similar ranges were identified as important in several iterative trials, using several different methods to establish ranges in mean PEC quotients. The consistency of these results indicates that the inflection points were robust and increases confidence in the validity of the inflection points.

Based on the inflection points identified in the exposure/response relationships for the 10-day chironomid and amphipod sediment toxicity tests, critical mean PEC quotients of 1 and 2, respectively, were identified. Both critical mean PECQ values should be used in the FS to identify sediments in Onondaga Lake that pose risks of toxicity to benthic macroinvertebrates.

SECTION J.7

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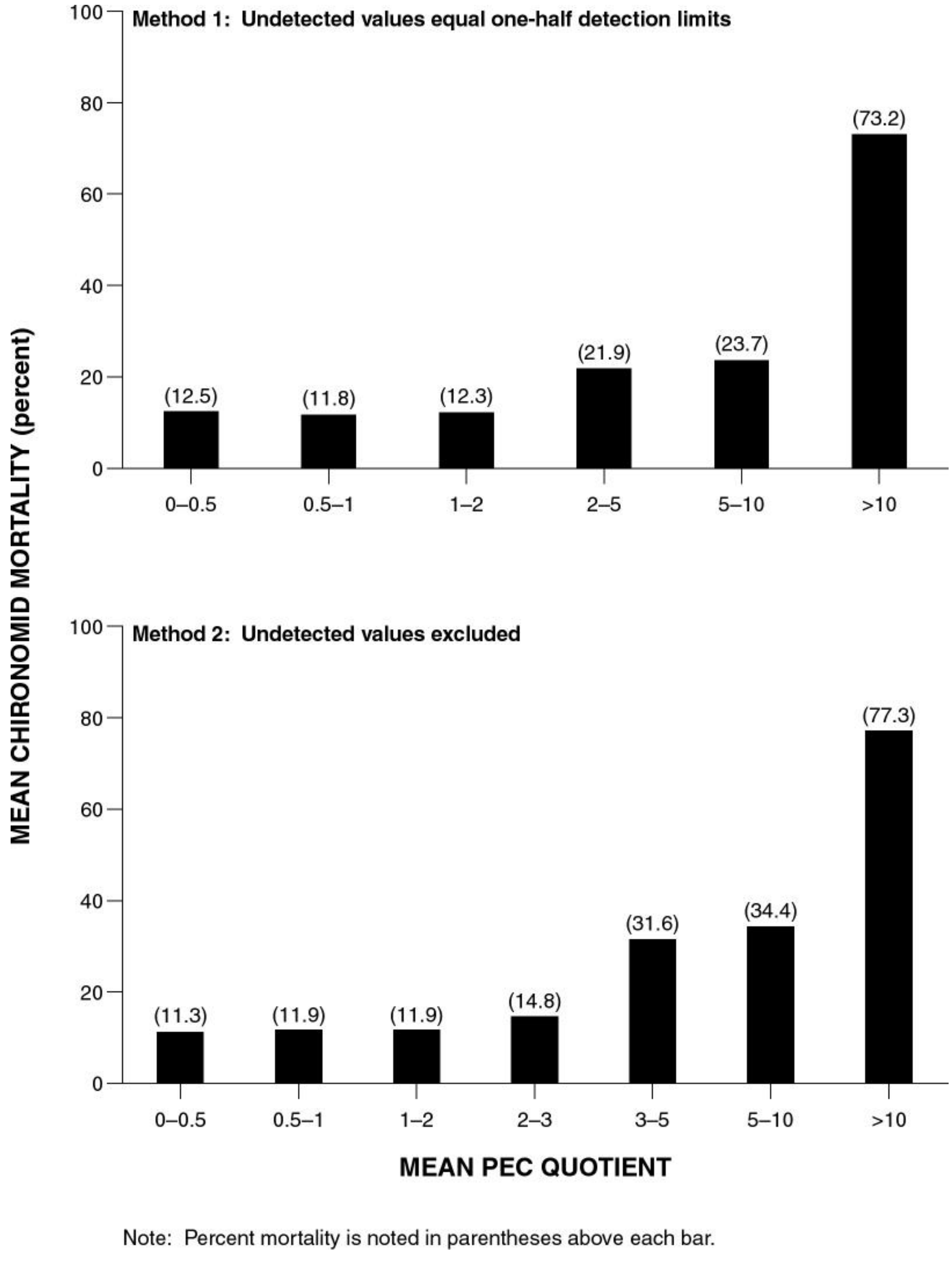
APPENDIX J

FIGURES



Note: Percent mortality is noted in parentheses above each bar.

FIGURE J.1

Onondaga Lake
Syracuse, New York

SUMMARY OF 10-DAY CHIRONOMID MORTALITY
IN RELATION TO MEAN PEC QUOTIENTS
CALCULATED USING TWO METHODS

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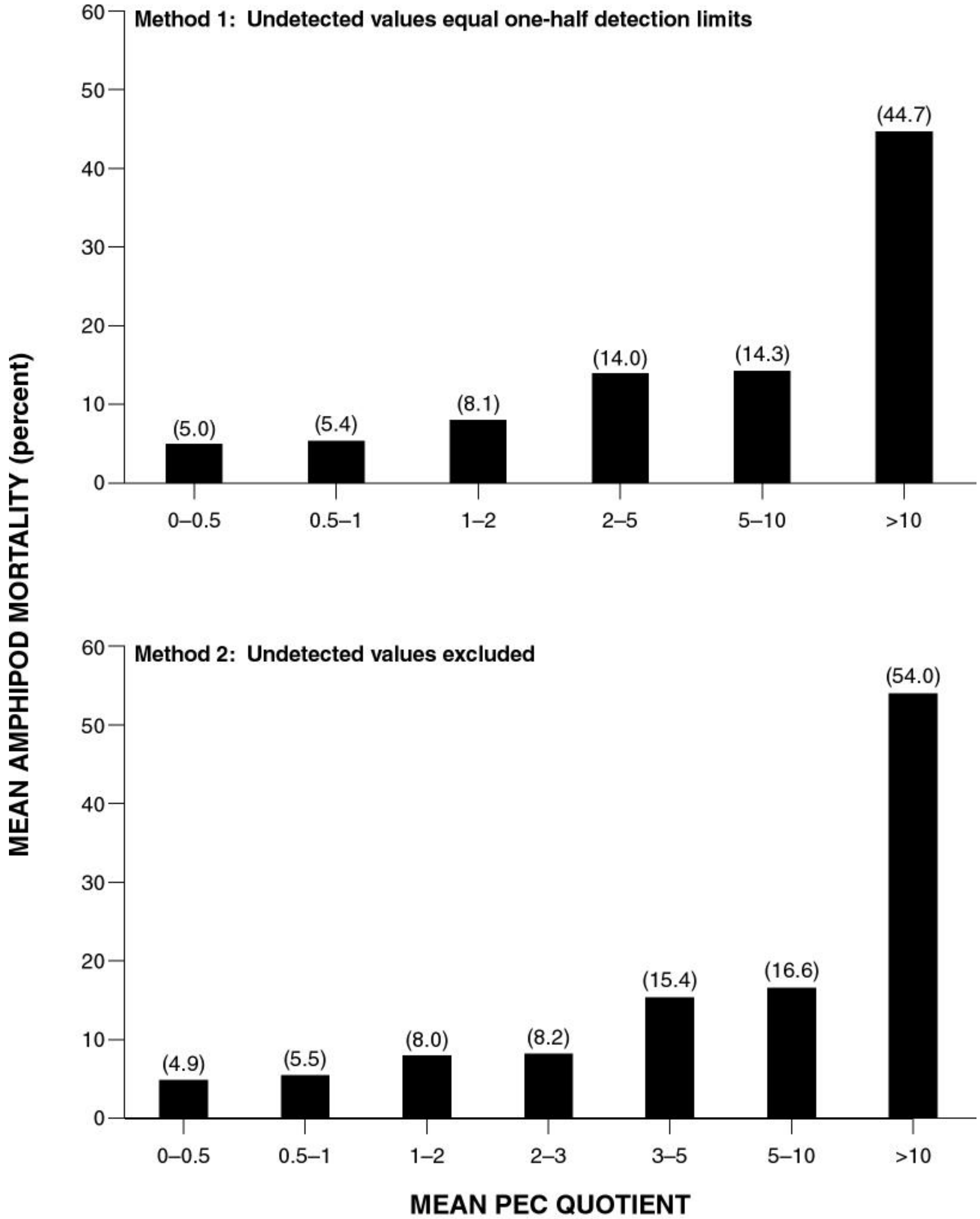

Note: Percent mortality is noted in parentheses above each bar.

FIGURE J.2

Onondaga Lake
Syracuse, New York

SUMMARY OF 10-DAY AMPHIPOD MORTALITY IN RELATION TO MEAN PEC QUOTIENTS CALCULATED USING TWO METHODS

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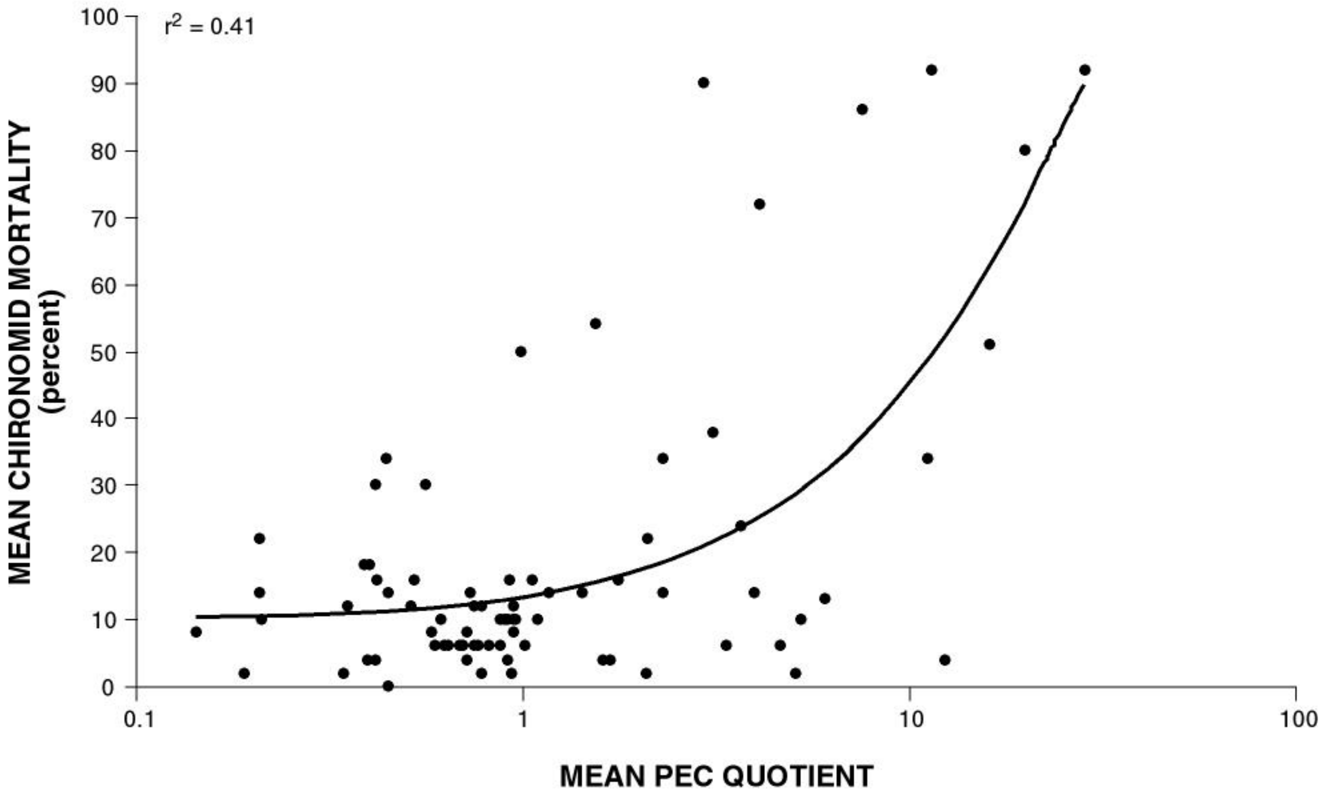

FIGURE J.3

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Using the Grouped Method Applied to Data Collected in 1992

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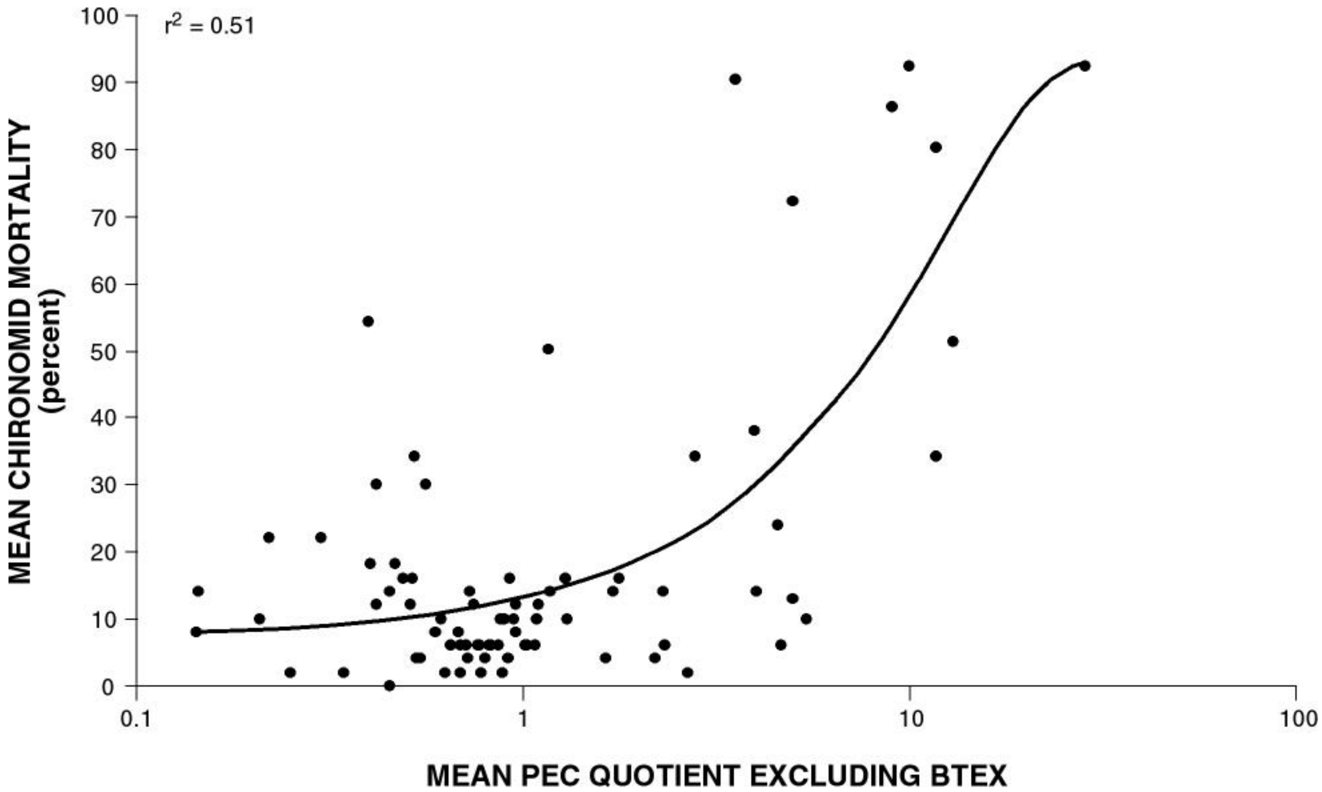

FIGURE J.4

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Chemical Groups Excluding BTEX Compounds

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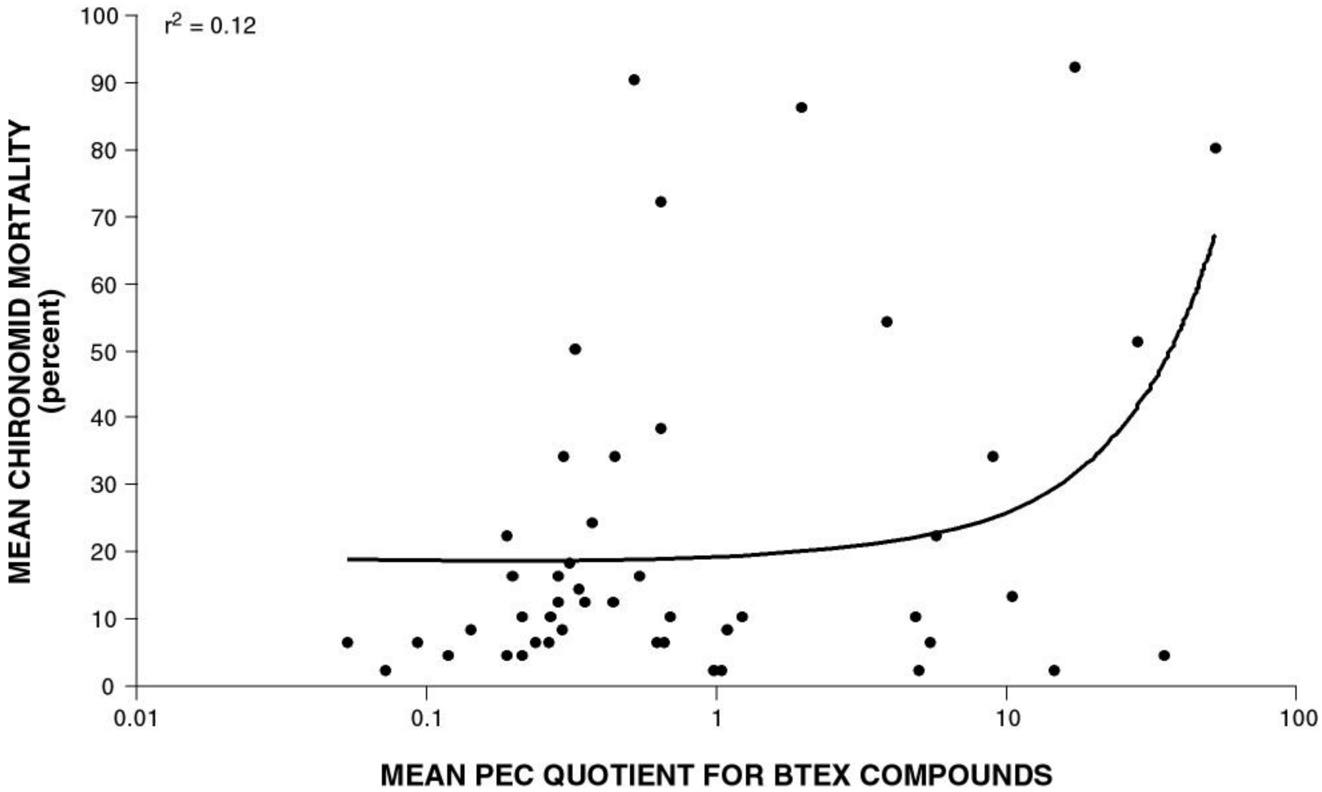

FIGURE J.5

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on BTEX Compounds

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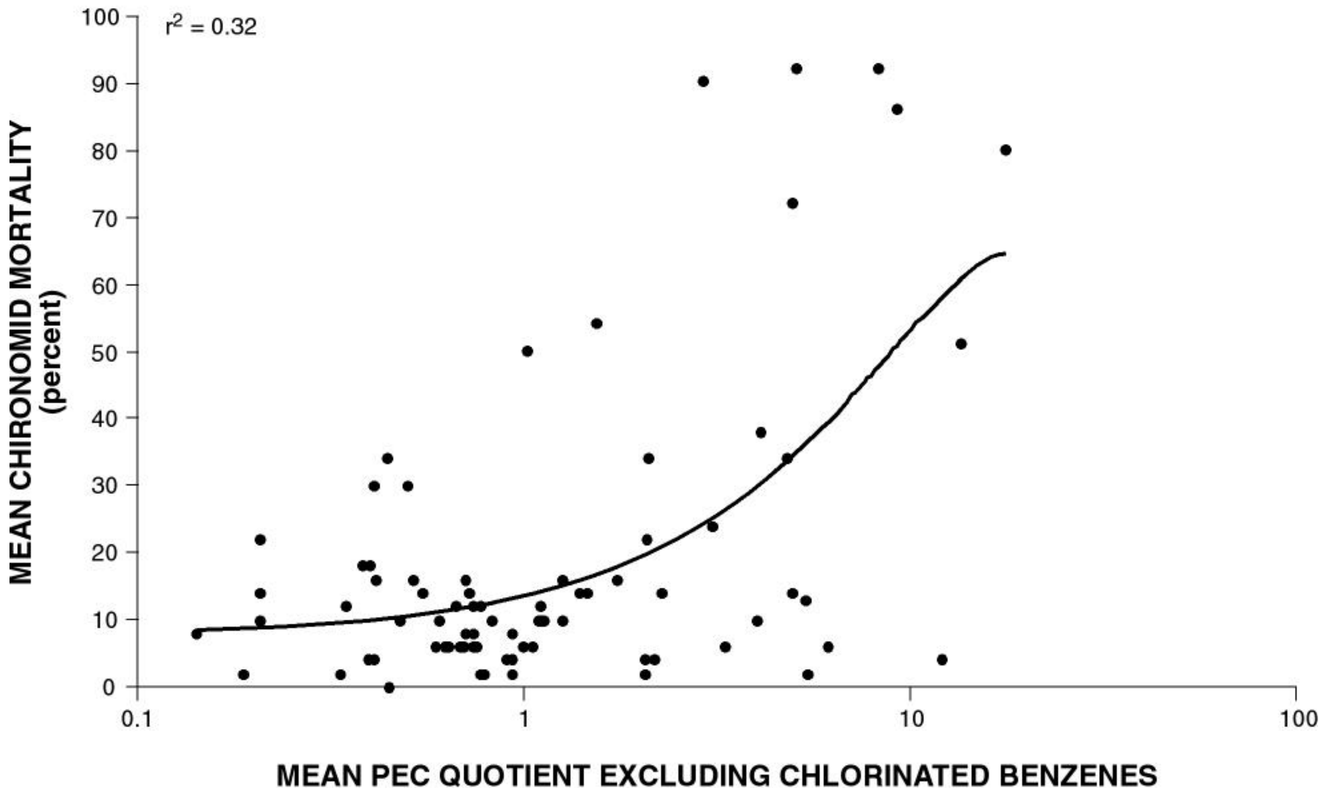

FIGURE J.6

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Chemical Groups Excluding Chlorinated Benzenes

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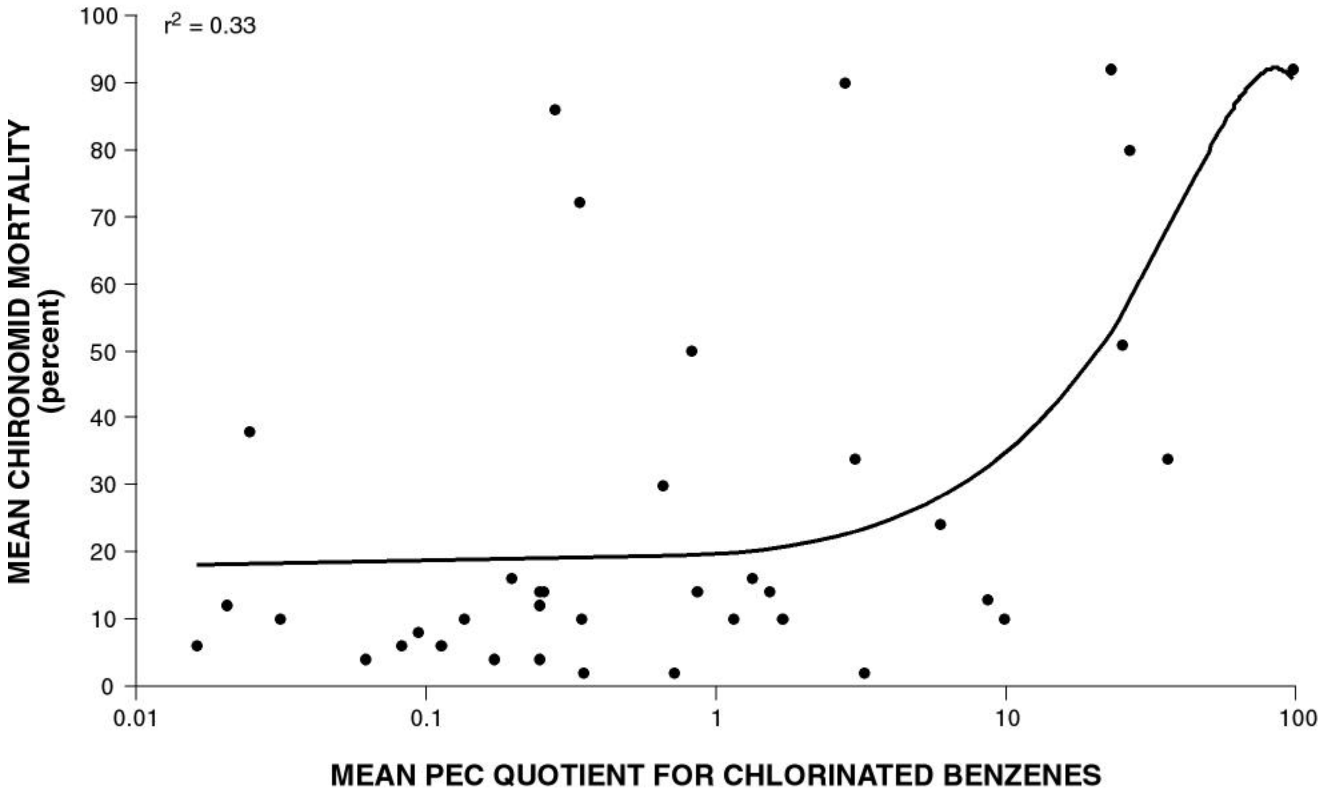

FIGURE J.7

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Chlorinated Benzenes

Exponent®

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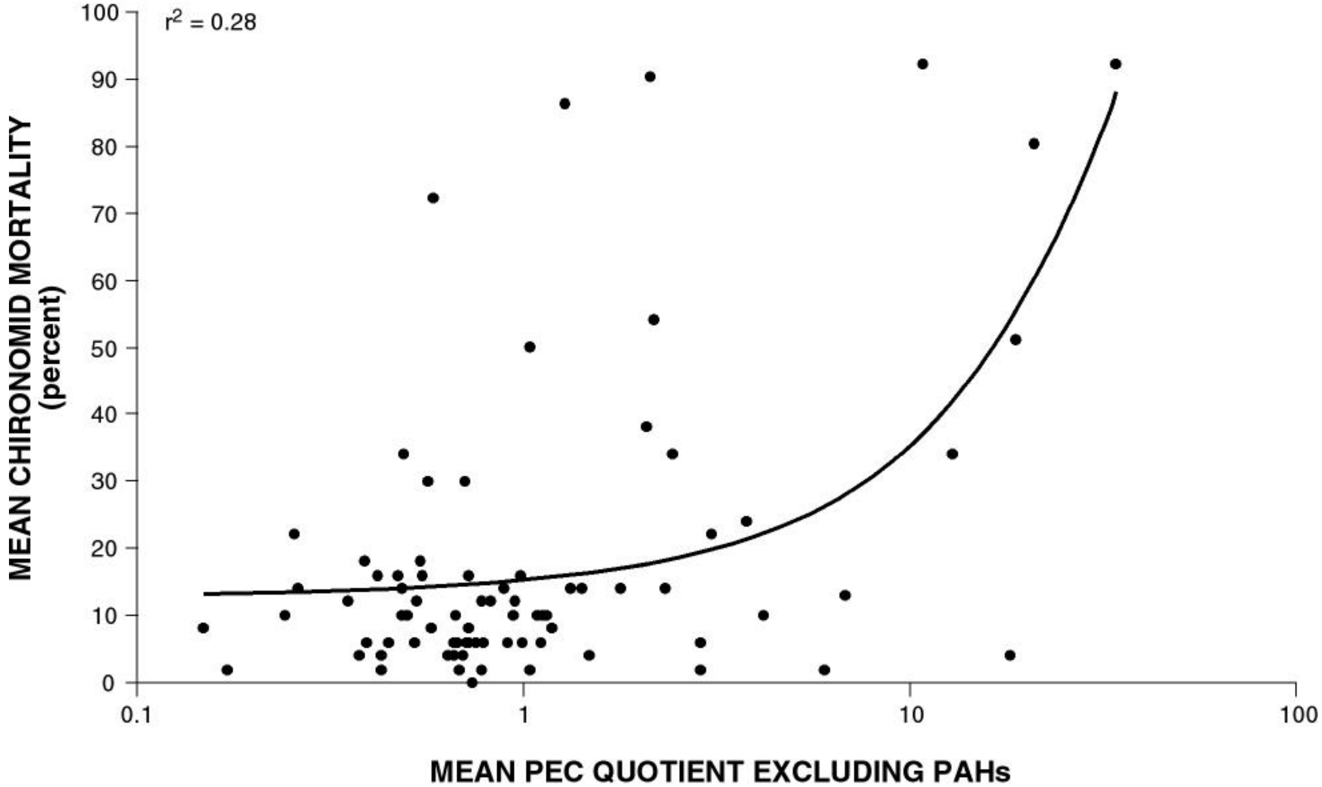

FIGURE J.8

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Chemical Groups Excluding PAHs

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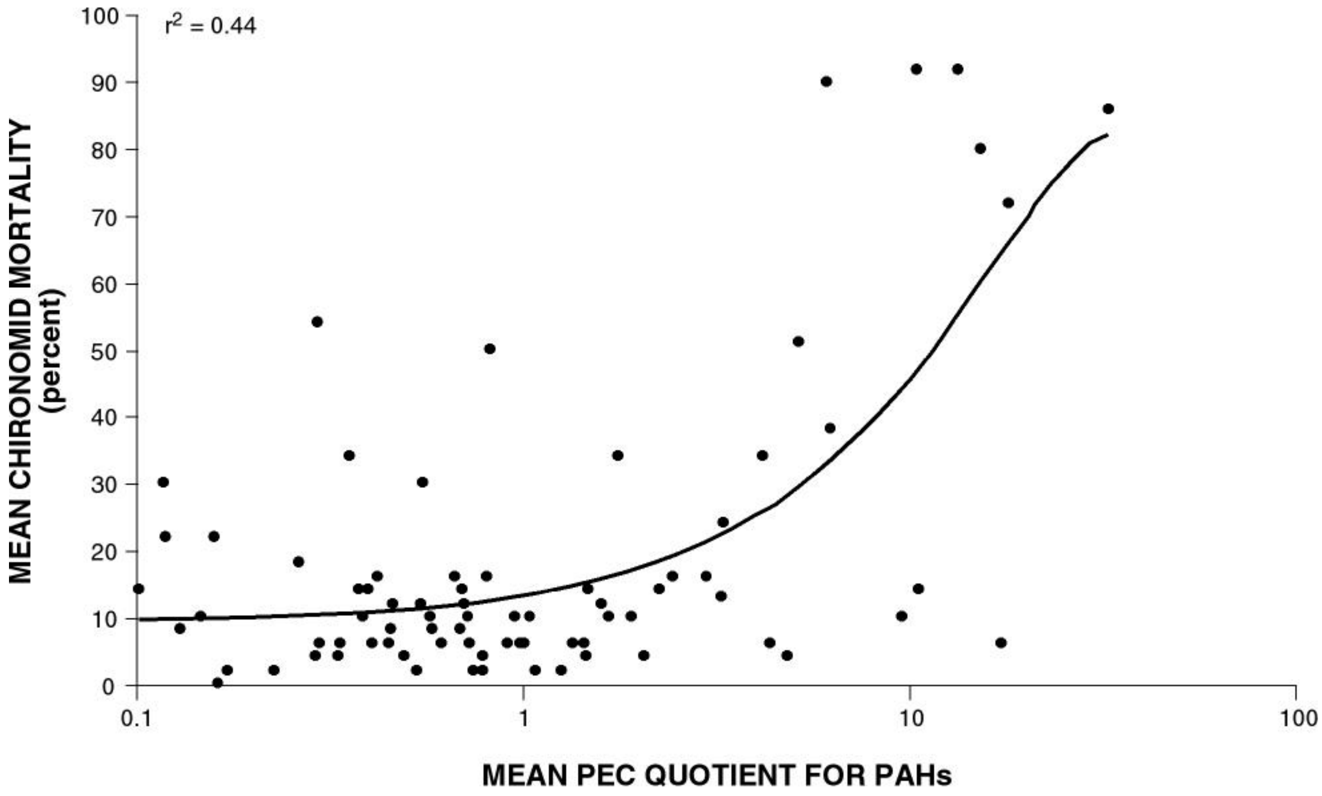

FIGURE J.9

ONONDAGA LAKE

SYRACUSE, NEW YORK

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Comparison of Chironomid Mortality and Mean PEC Quotient Based on PAHs

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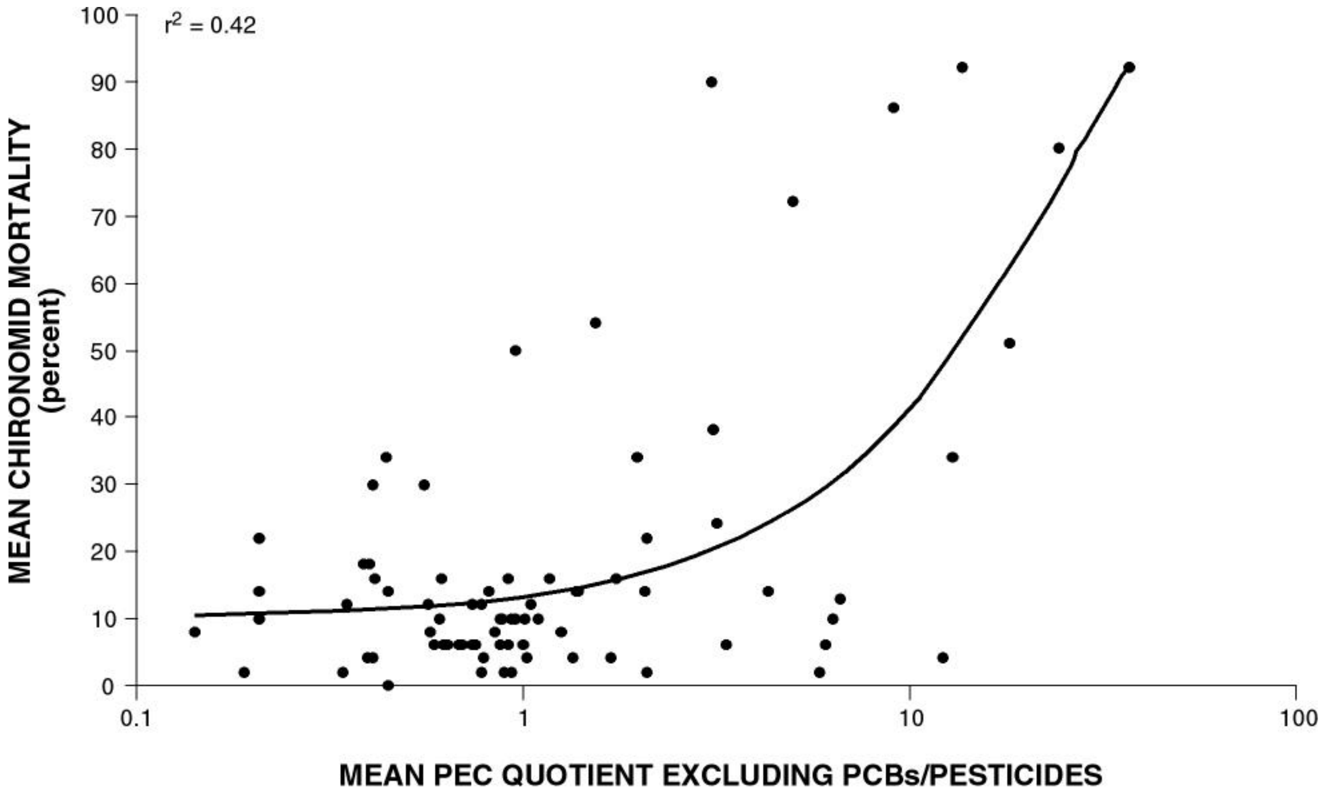

FIGURE J.10

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Chemical Groups Excluding PCBs and Pesticides

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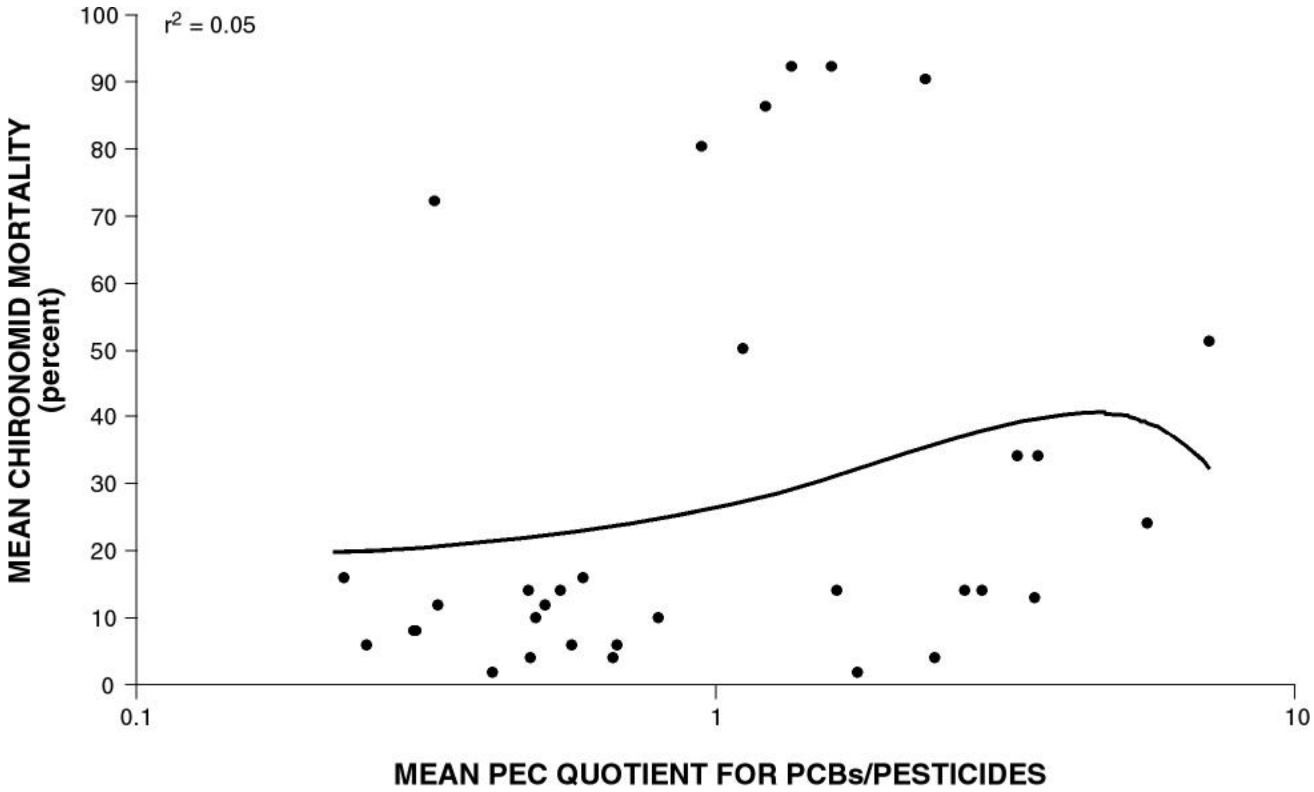

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FIGURE J.11

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on PCBs and Pesticides

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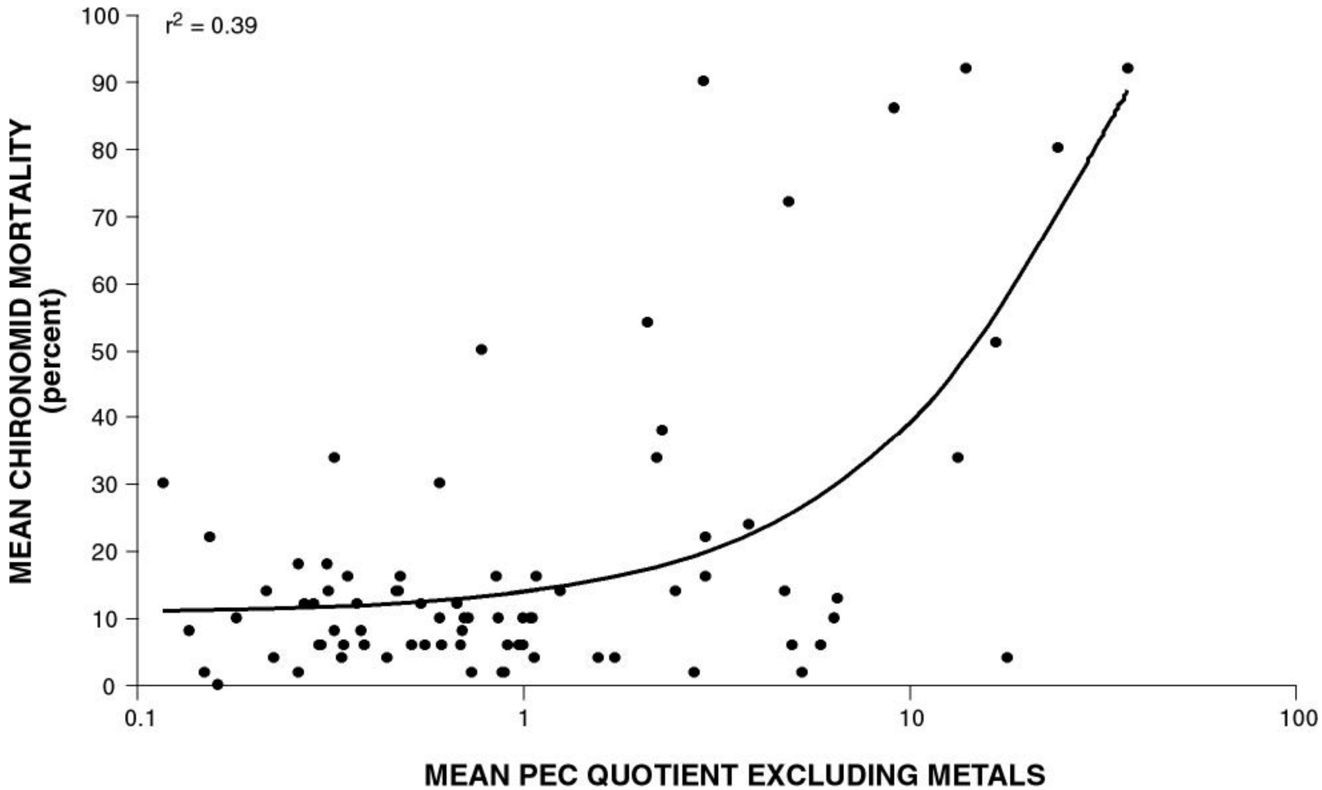

FIGURE J.12

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC quotient Based on Chemical Groups Excluding Metals

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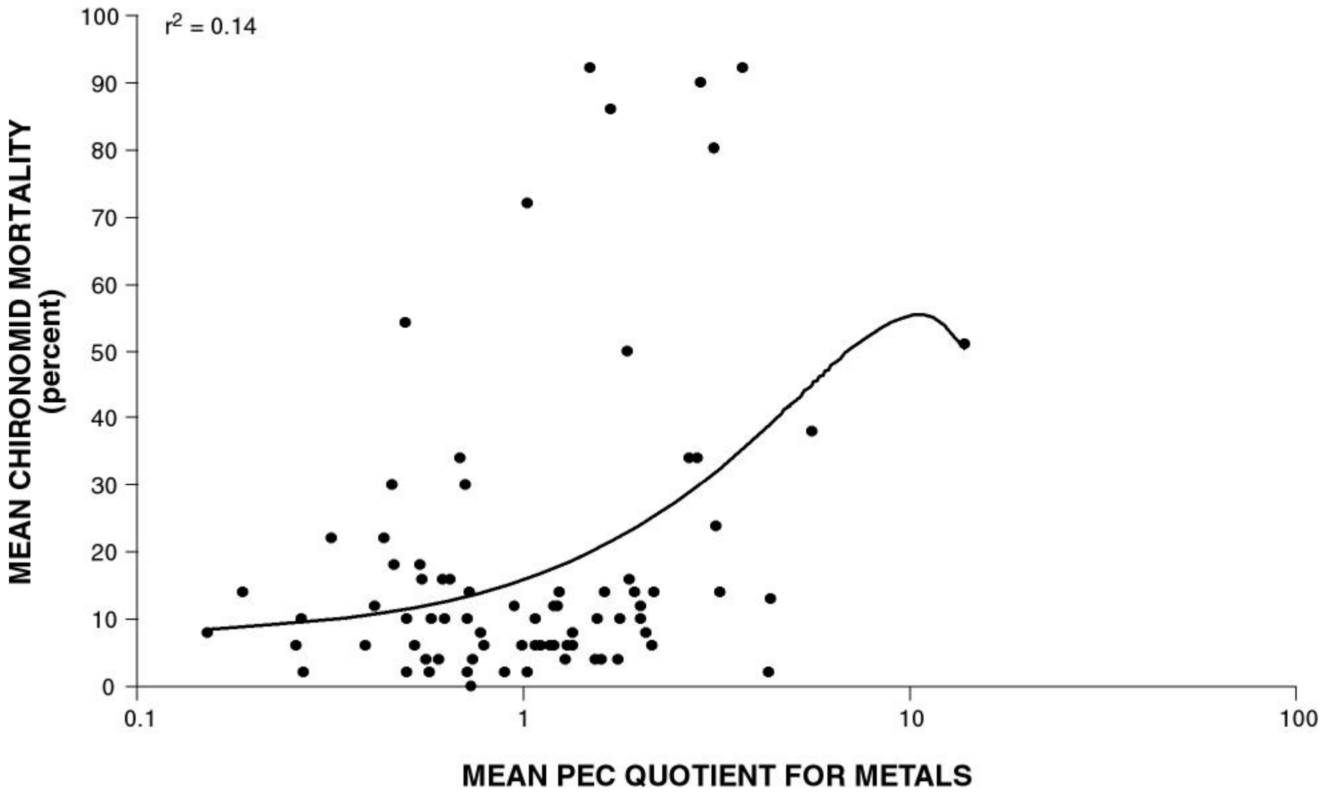

FIGURE J.13

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Metals

Exponent®

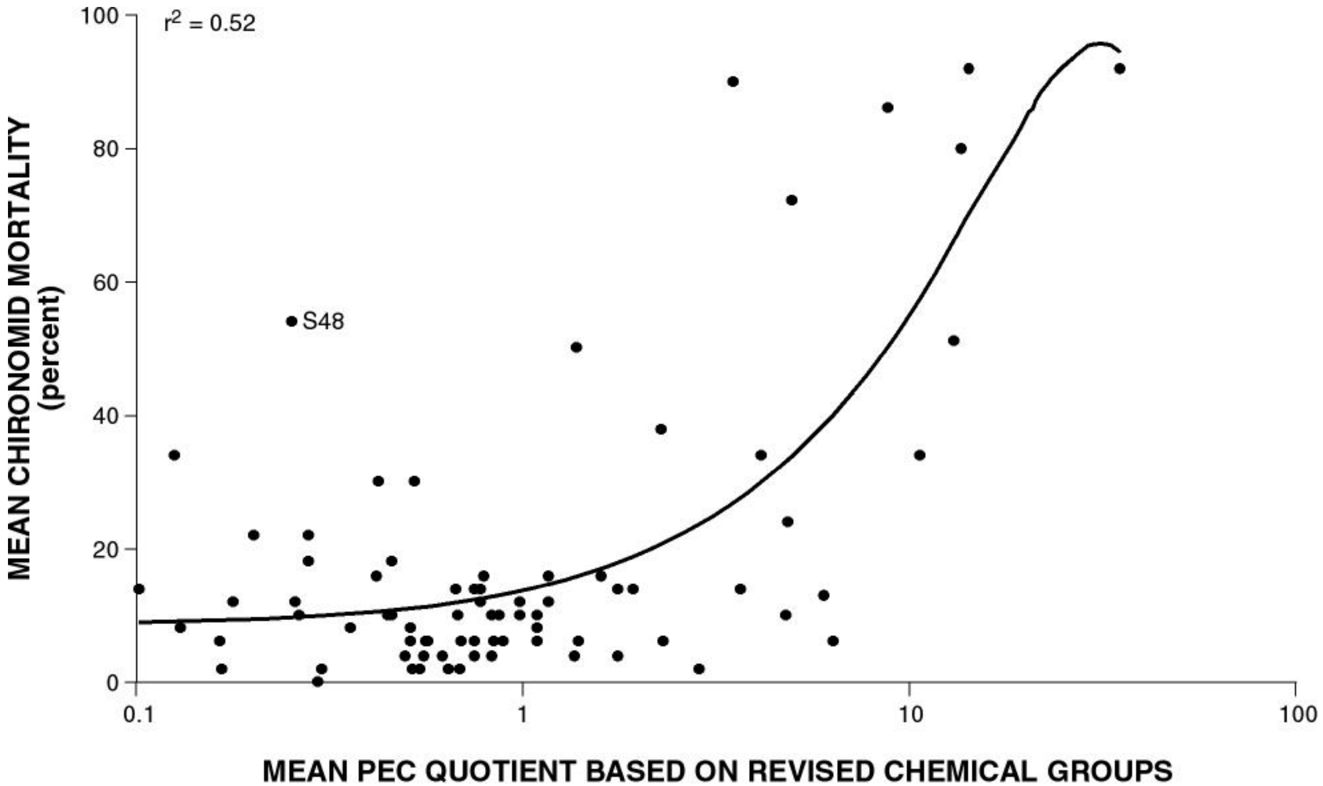

FIGURE J.14

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality and Mean PEC Quotient Based on Revised Chemical Groups

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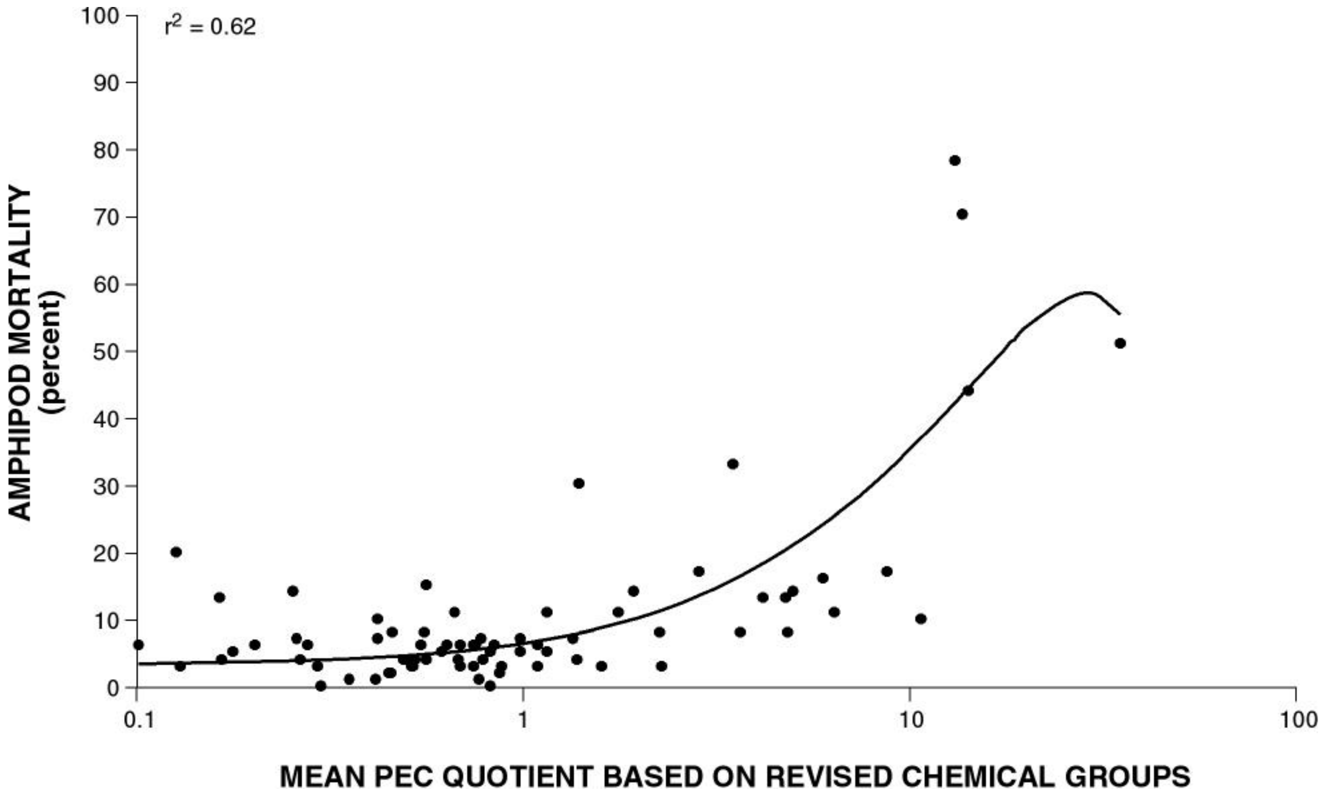

FIGURE J.15

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Amphipod Mortality and Mean PEC Quotient Based on Revised Chemical Groups

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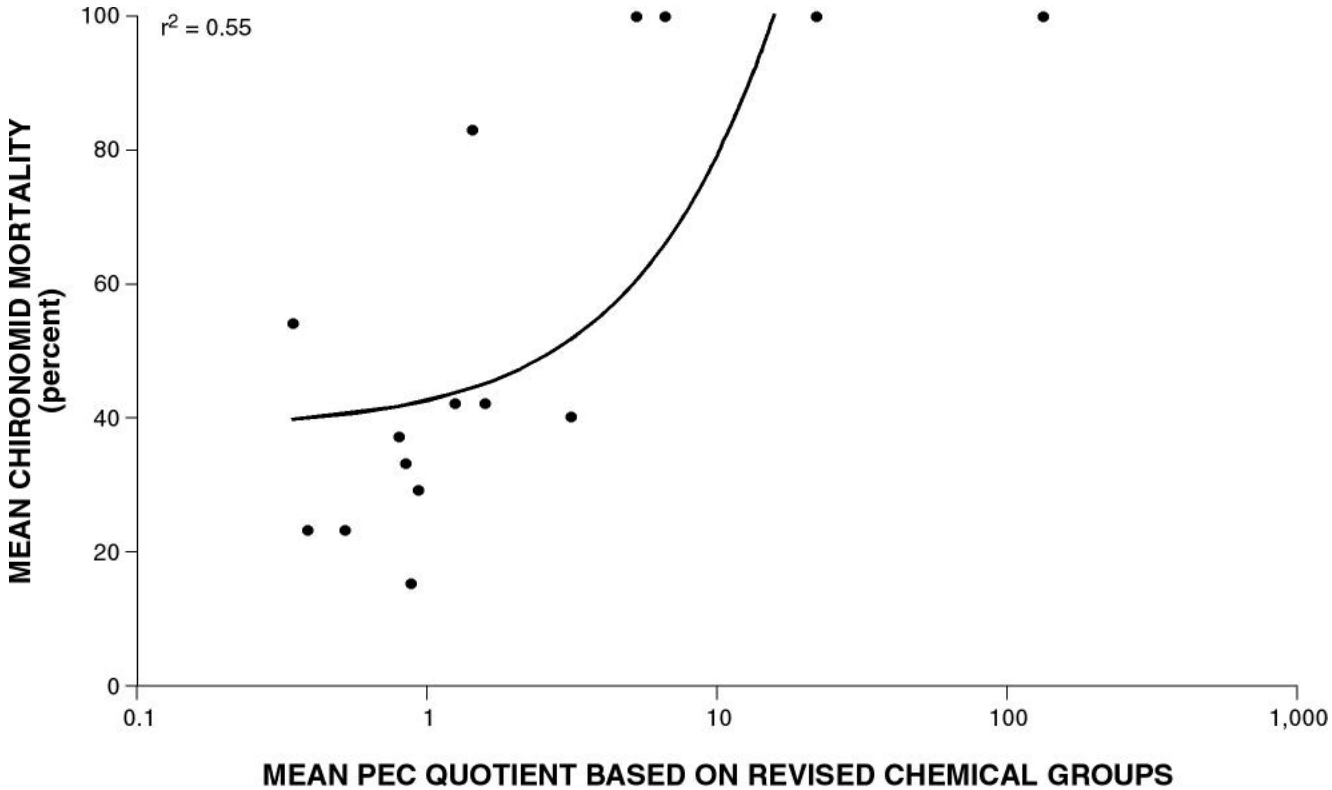

FIGURE J.16

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of 2000 Chironomid Mortality (20-day) and Mean PEC Quotient Based on Revised Chemical Groups

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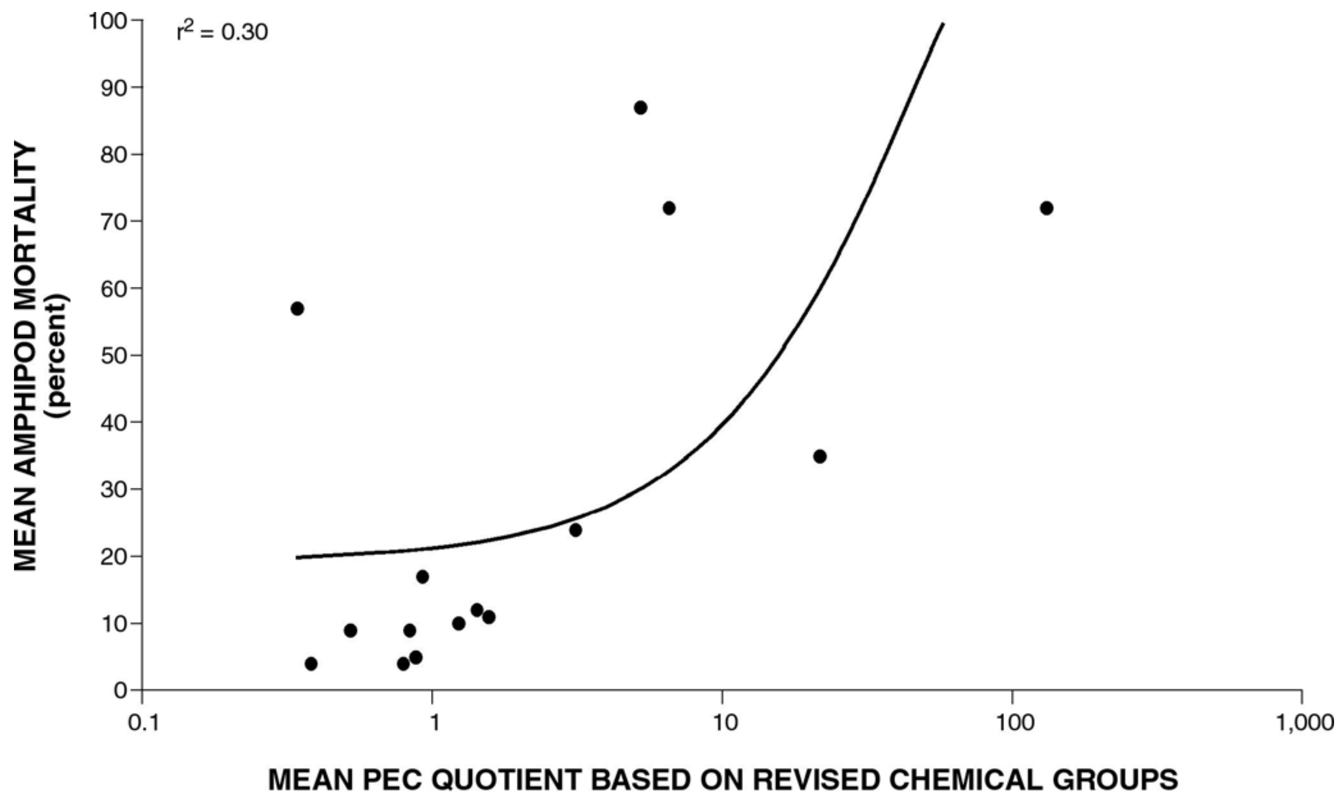

FIGURE J.17

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of 2000 Amphipod Mortality (28-day) and Mean PEC Quotient Based on Revised Chemical Groups

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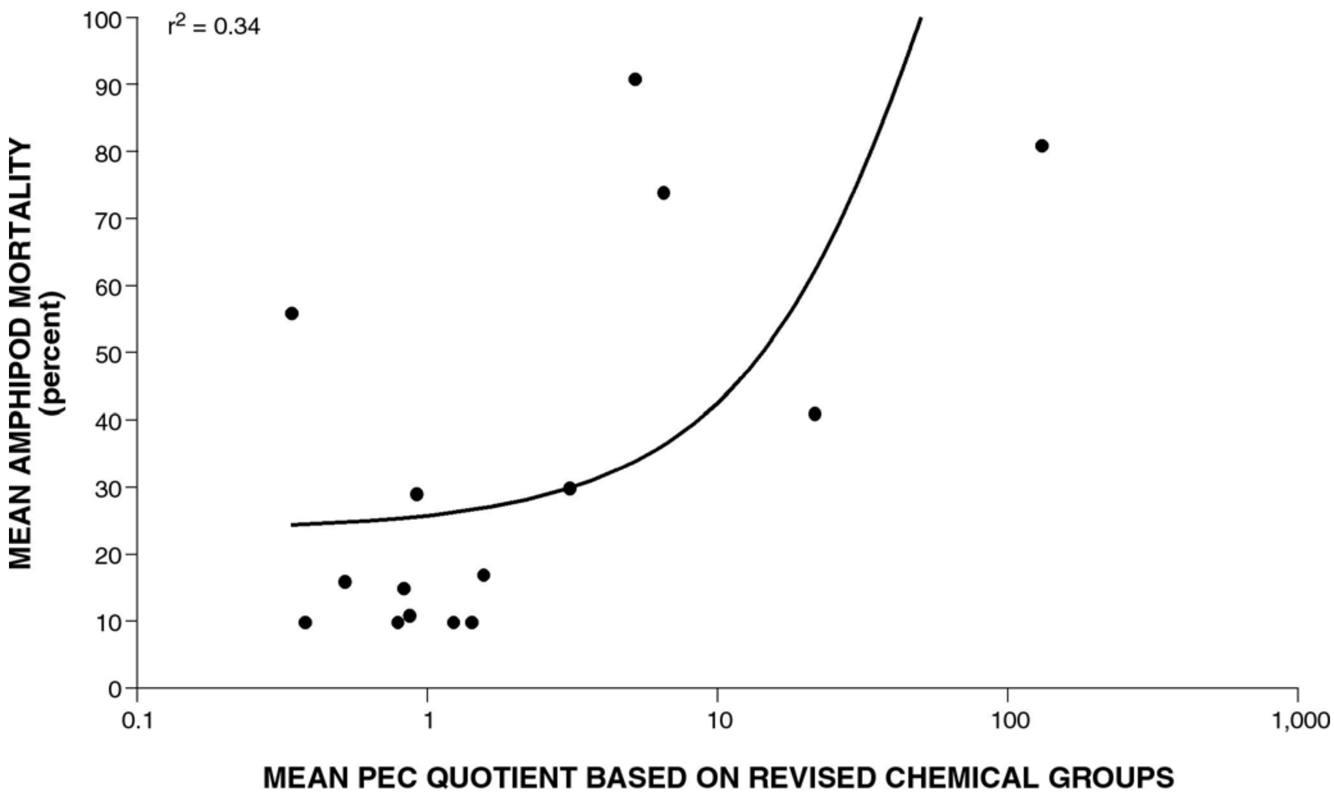

FIGURE J.18

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of 2000 Amphipod Mortality (42-day) and Mean PEC Quotient Based on Revised Chemical Groups

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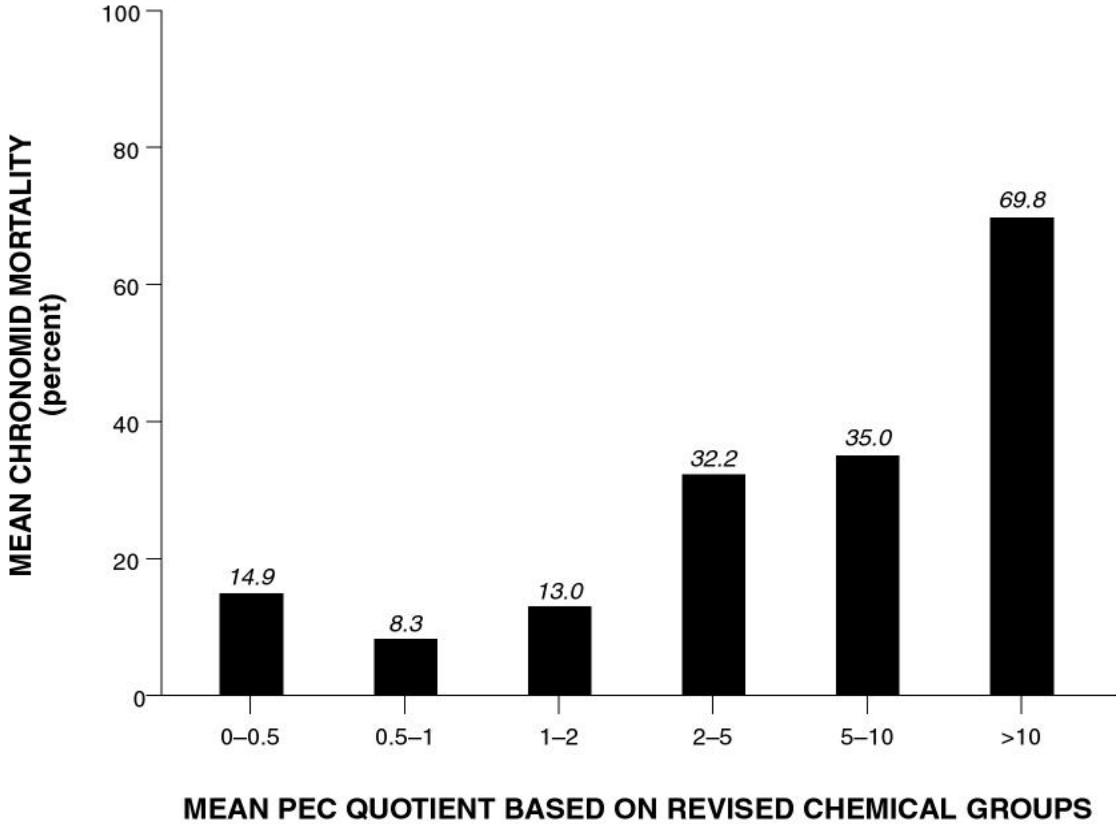

NOTE: Percent mortality is noted above each bar.

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FIGURE J.19

ONONDAGA LAKE

SYRACUSE, NEW YORK

Summary of 10-day Chironomid Mortality in Relation to Mean PEC Quotients Calculated Using Revised Chemical Groups

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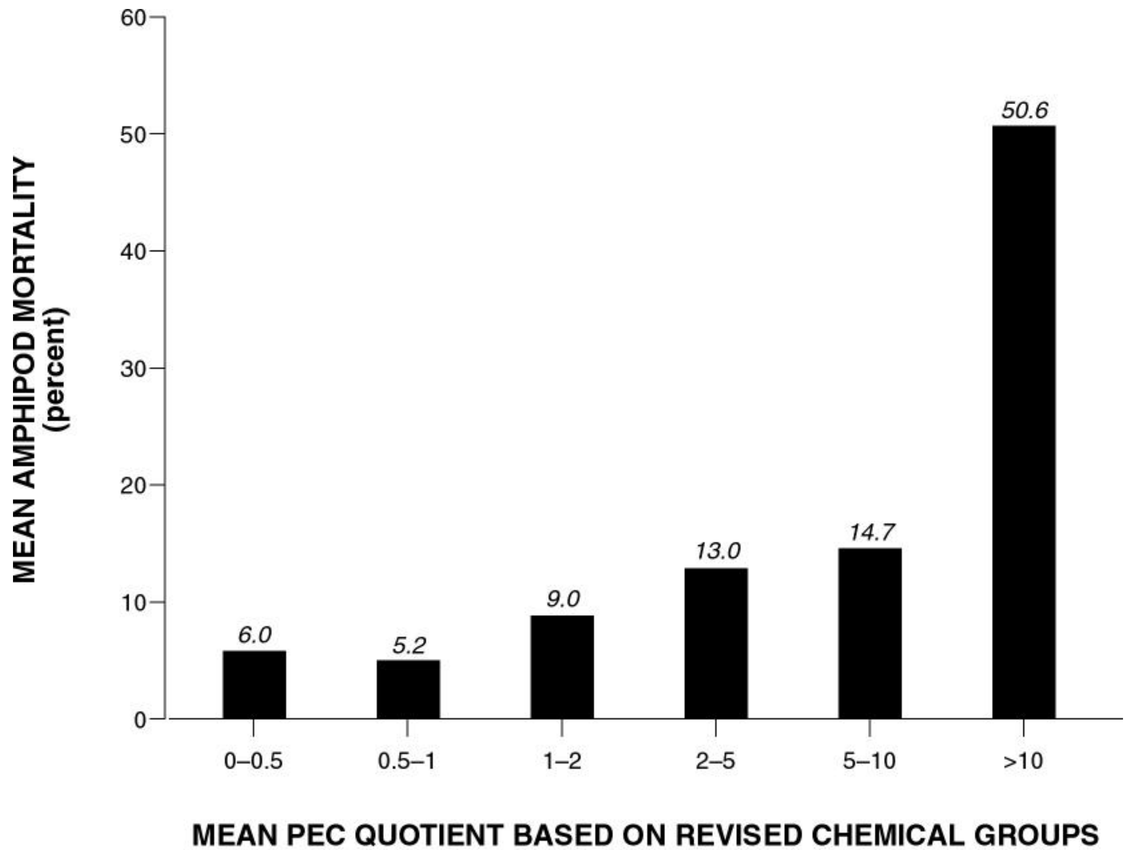

NOTE: Percent mortality is noted above each bar.

FIGURE J.20

ONONDAGA LAKE

SYRACUSE, NEW YORK

Summary of 10-day Amphipod Mortality in Relation to Mean PEC Quotients Calculated Using Revised Chemical Groups

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ATTACHMENT A

COMPARISON OF CHIRONOMID MORTALITY WITH PEC QUOTIENTS FOR SELECTED CPOIs

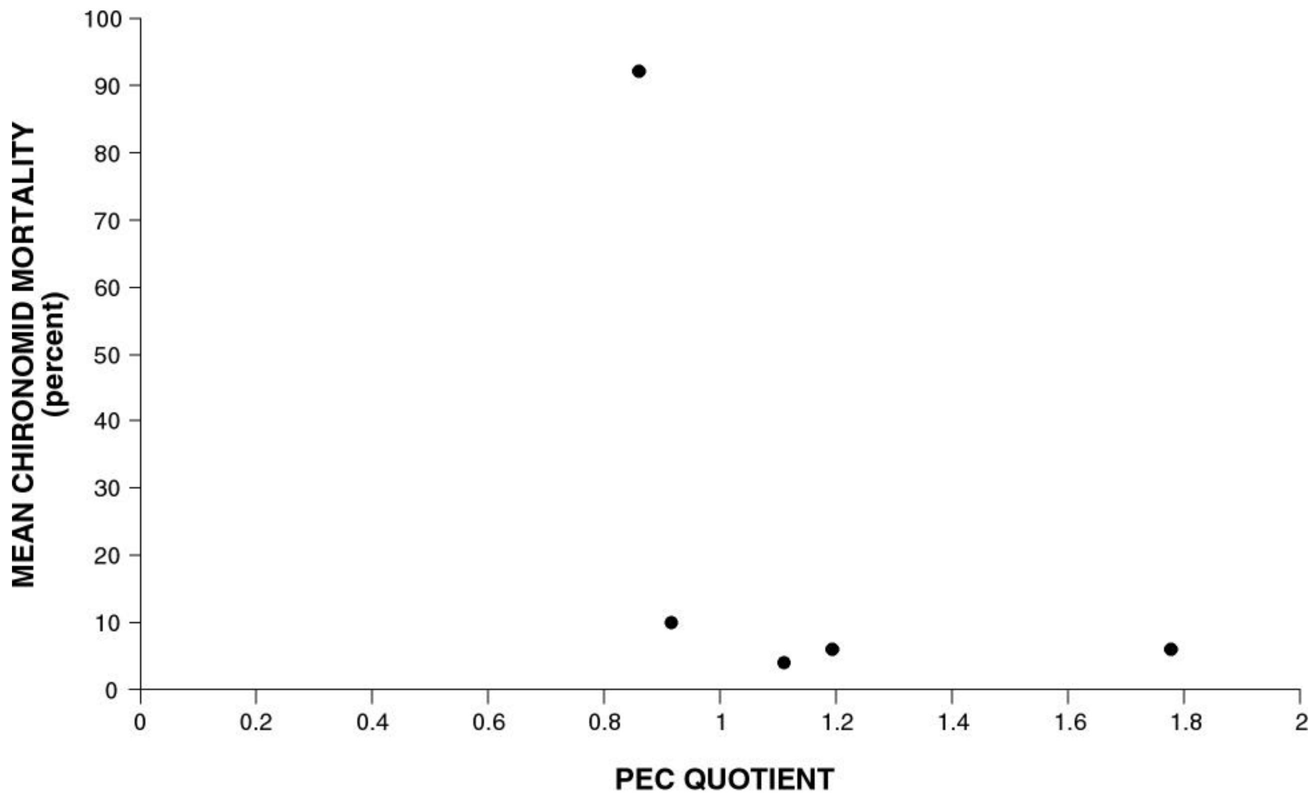

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FIGURE JA.1

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Antimony Based on Data Collected in 1992

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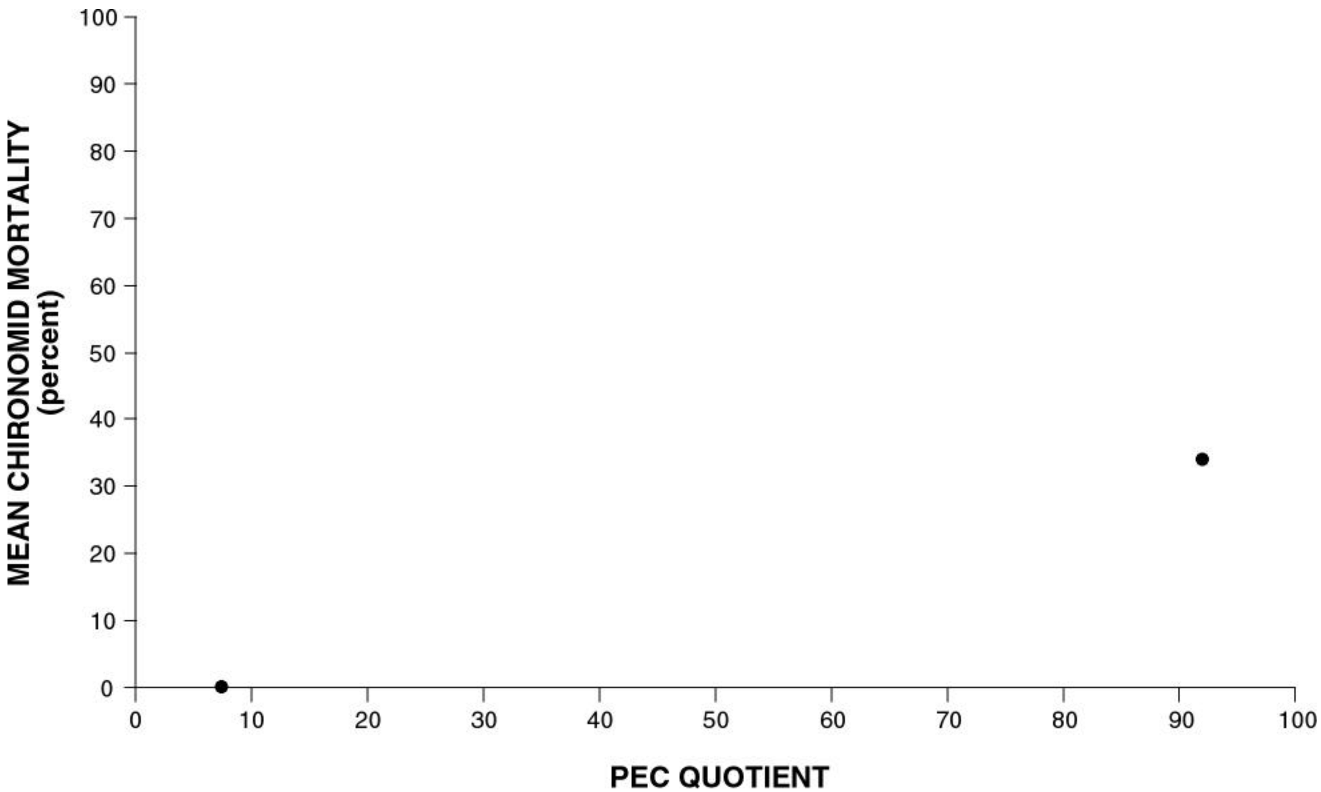

FIGURE JA.2

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Ethylbenzene Based on Data Collected in 1992

Exponent®

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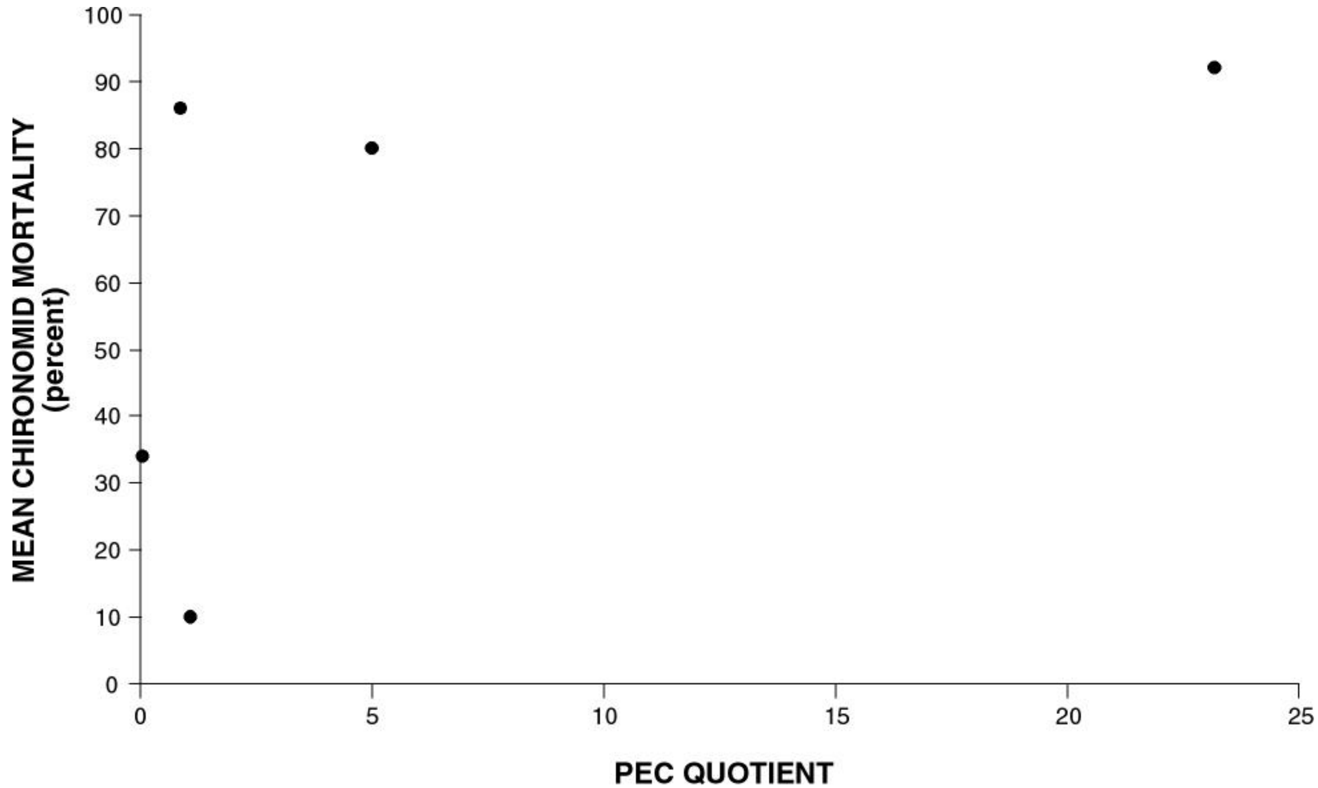

FIGURE JA.3

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Xylenes Based on Data Collected in 1992

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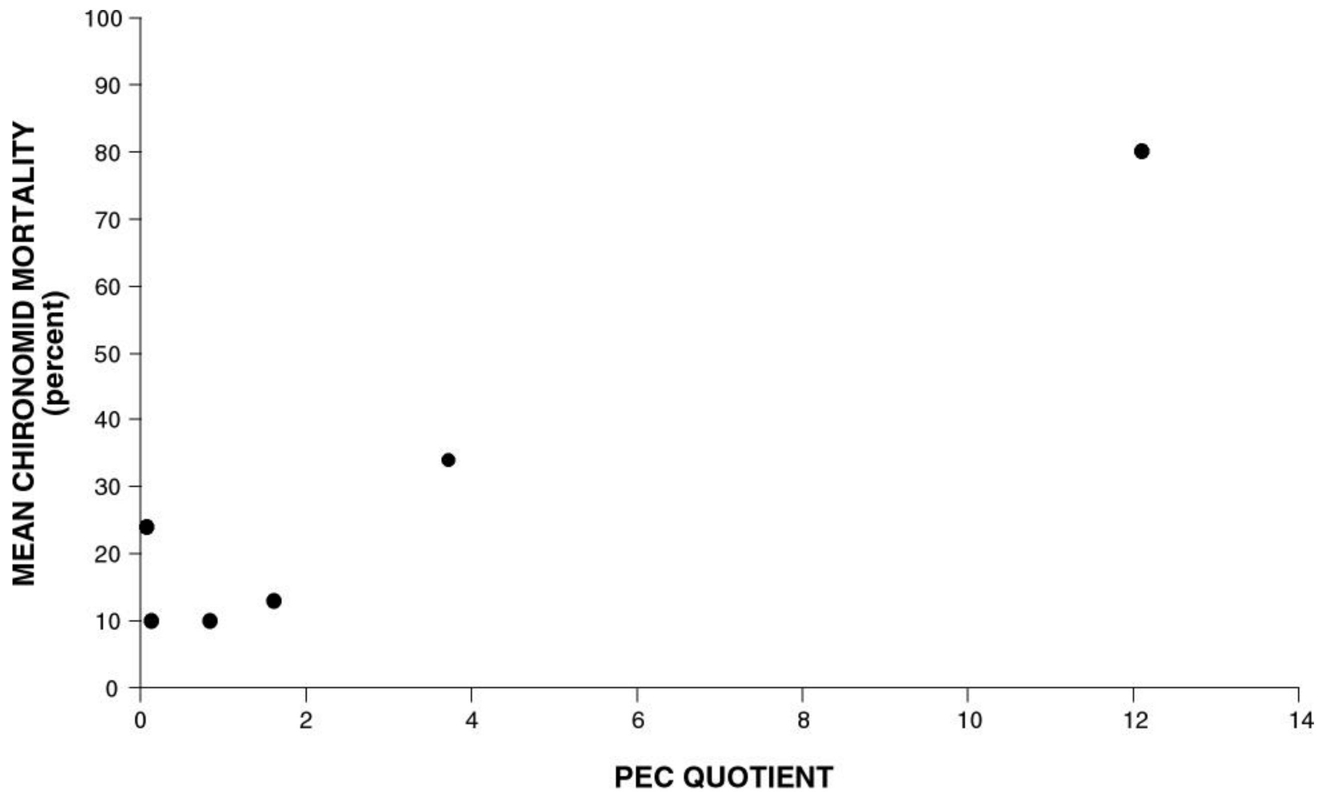

FIGURE JA.4

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Trichlorobenzenes Based on Data Collected in 1992

Exponent®

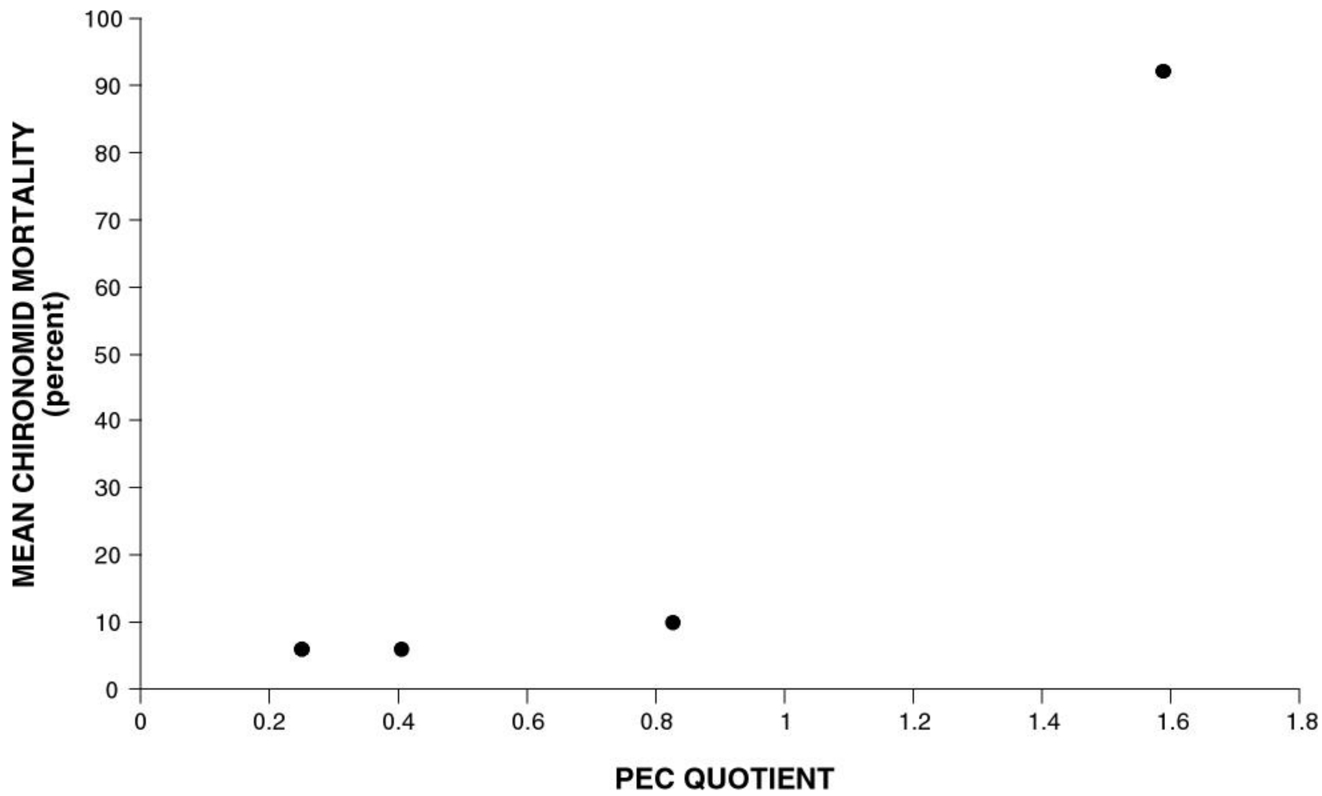

FIGURE JA.5

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Pesticides Based on Data Collected in 1992

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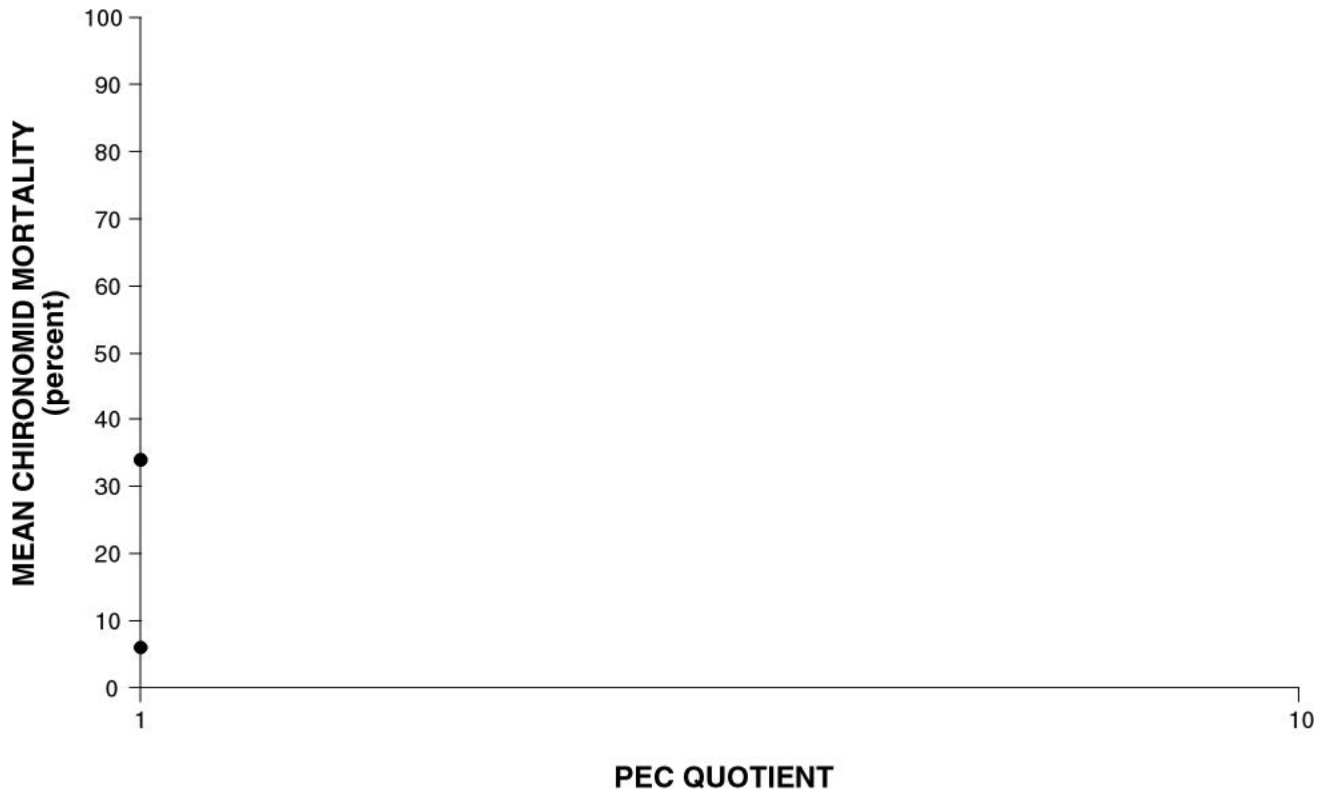

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FIGURE JA.6

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Phenol Based on Data Collected in 1992

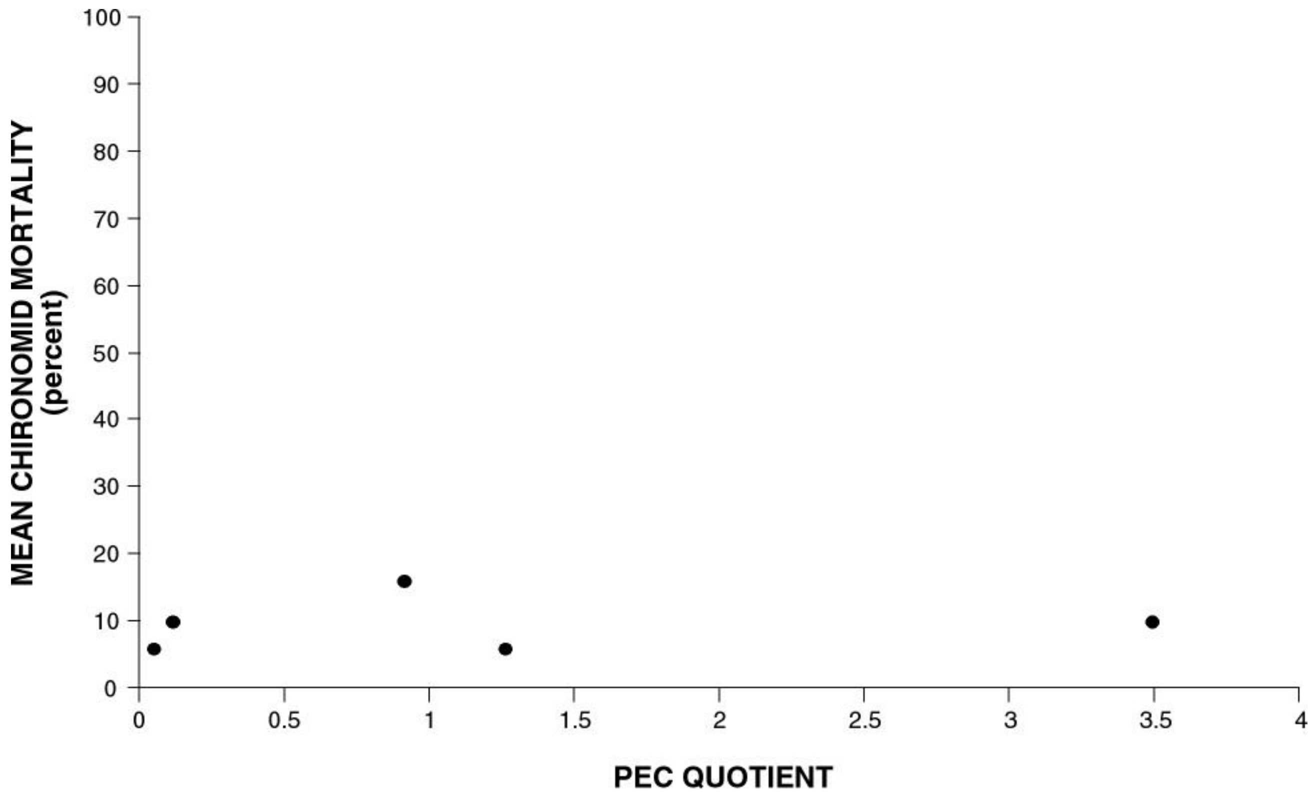

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FIGURE JA.7

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Dibenzofuran Based on Data Collected in 1992

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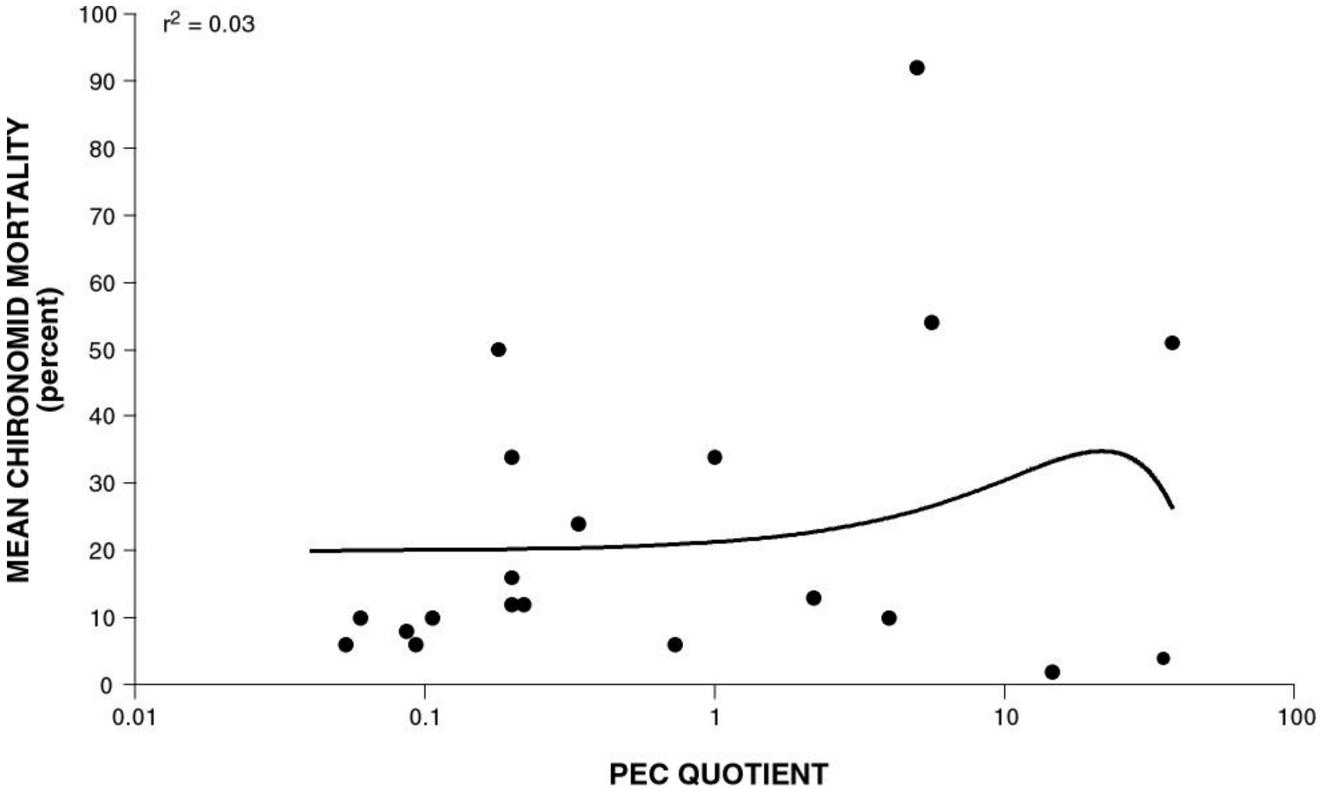

FIGURE JA.8

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Benzene Based on Data Collected in 1992

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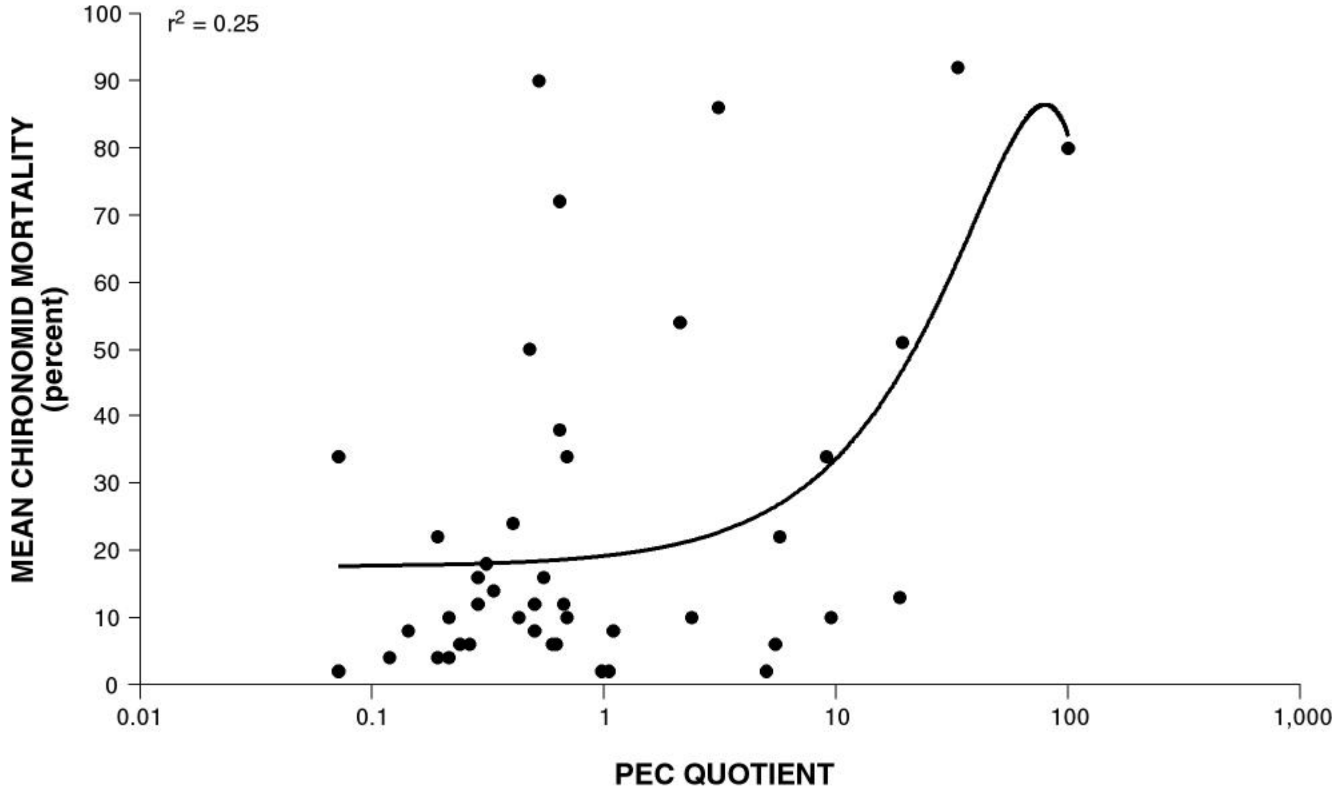

FIGURE JA.9

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Toluene Based on Data Collected in 1992

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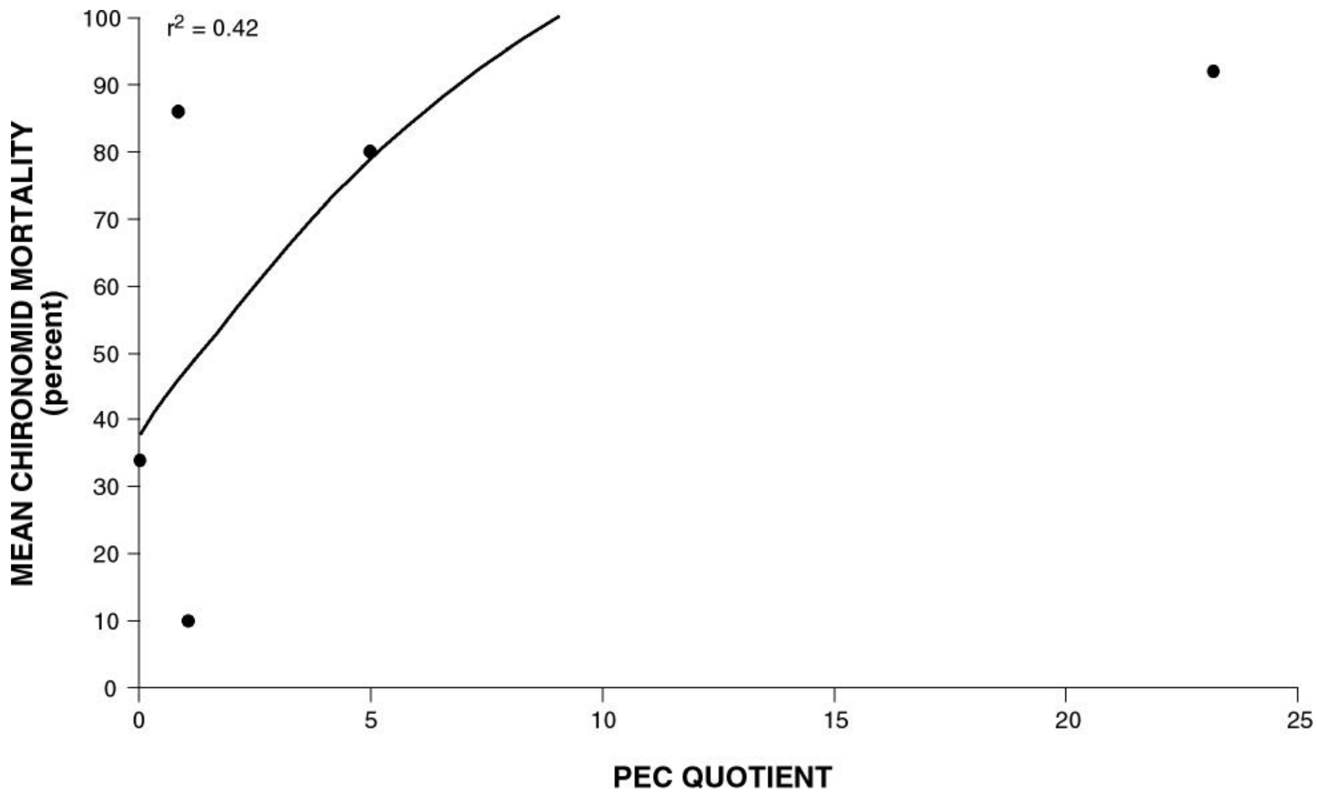

FIGURE JA.10

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Xylenes Based on Data Collected in 1992

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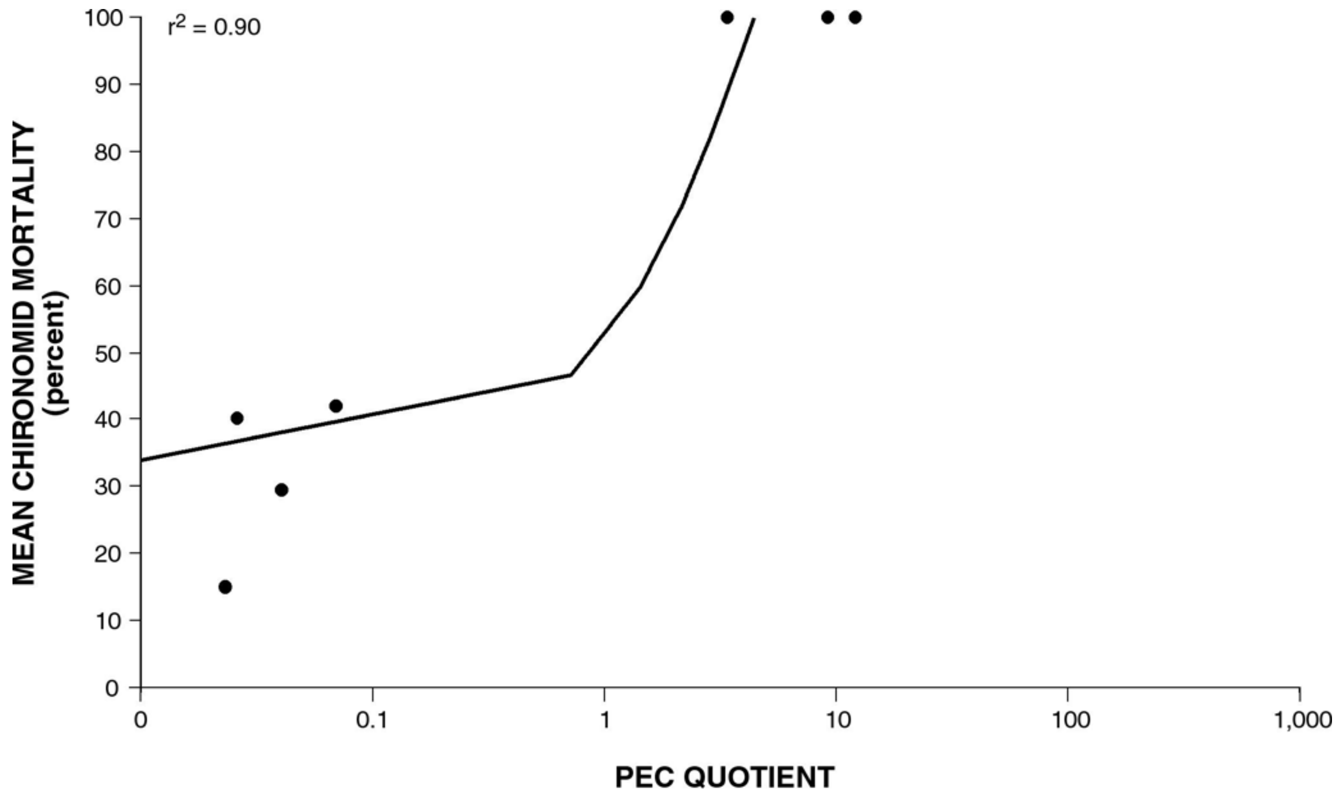

FIGURE JA.11

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Ethylbenzene Based on Data Collected in 2000

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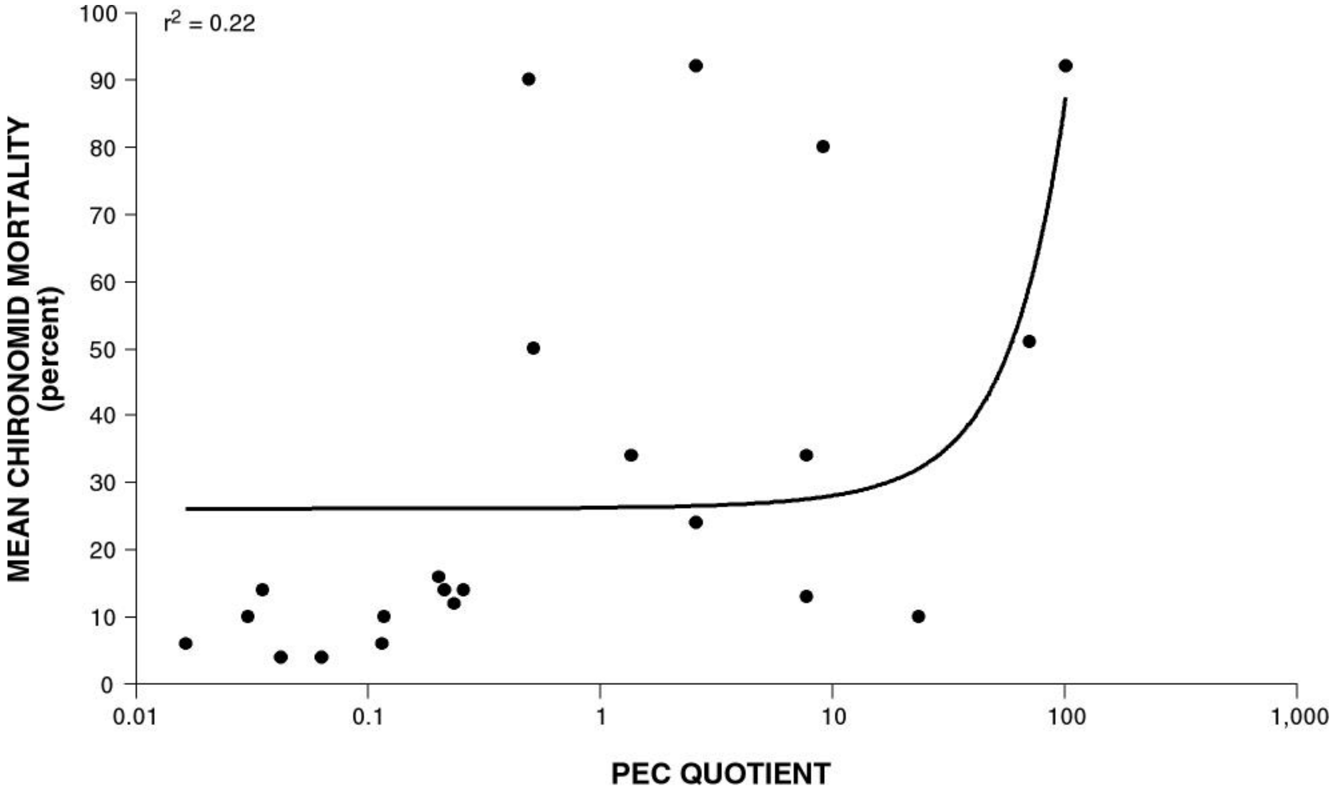

FIGURE JA.12

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Monochlorobenzene Based on Data Collected in 1992

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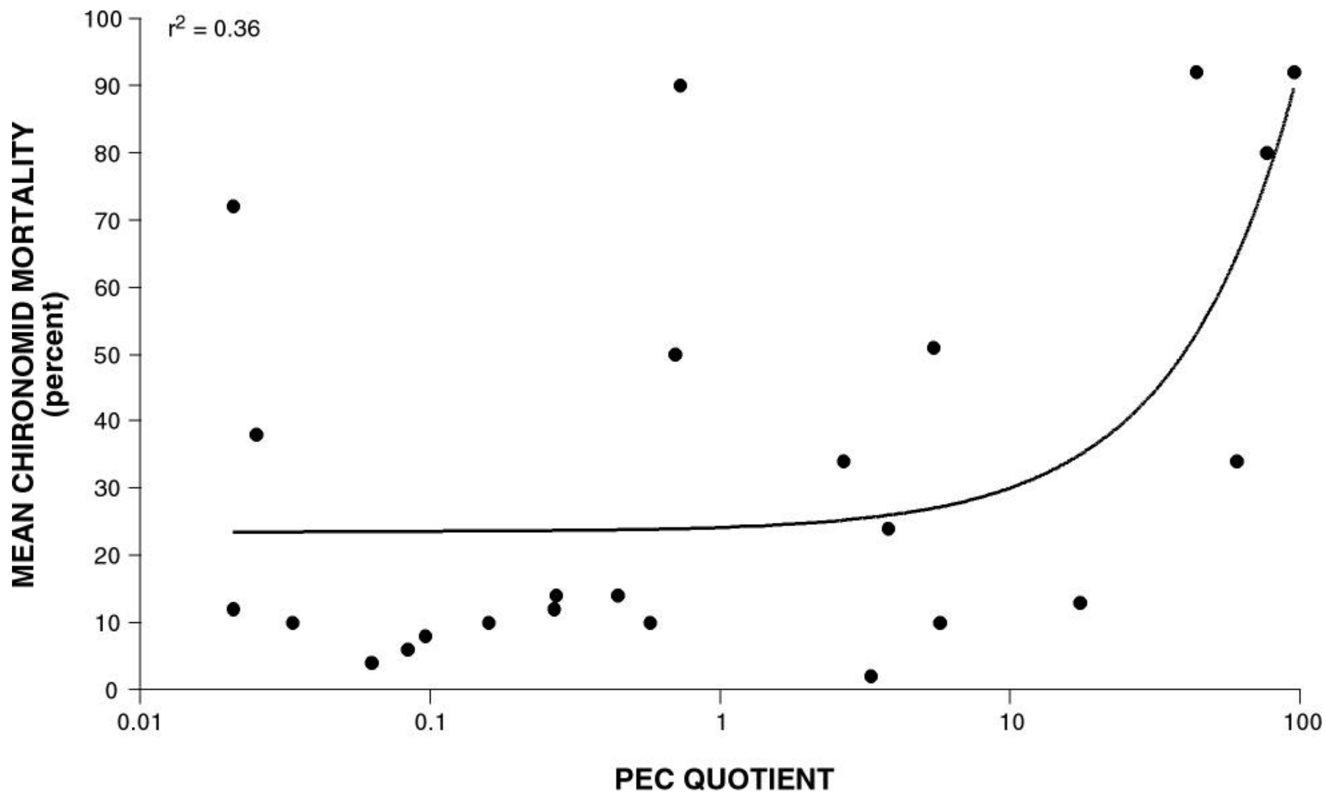

FIGURE JA.13

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Dichlorobenzenes Based on Data Collected in 1992

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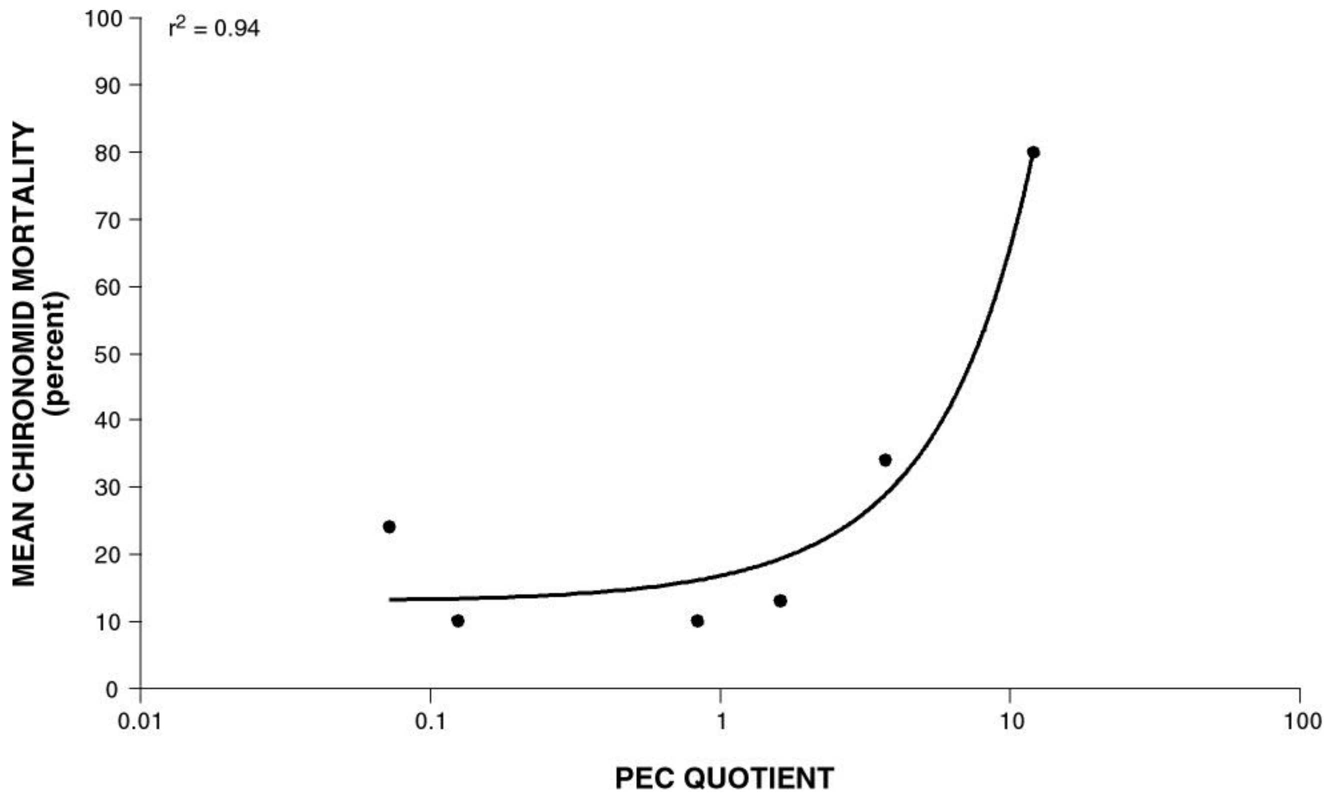

FIGURE JA.14

ONONDAGA LAKE

SYRACUSE, NEW YORK

Comparison of Chironomid Mortality with PEC Quotients for Trichlorobenzenes Based on Data Collected in 1992

Exponent®

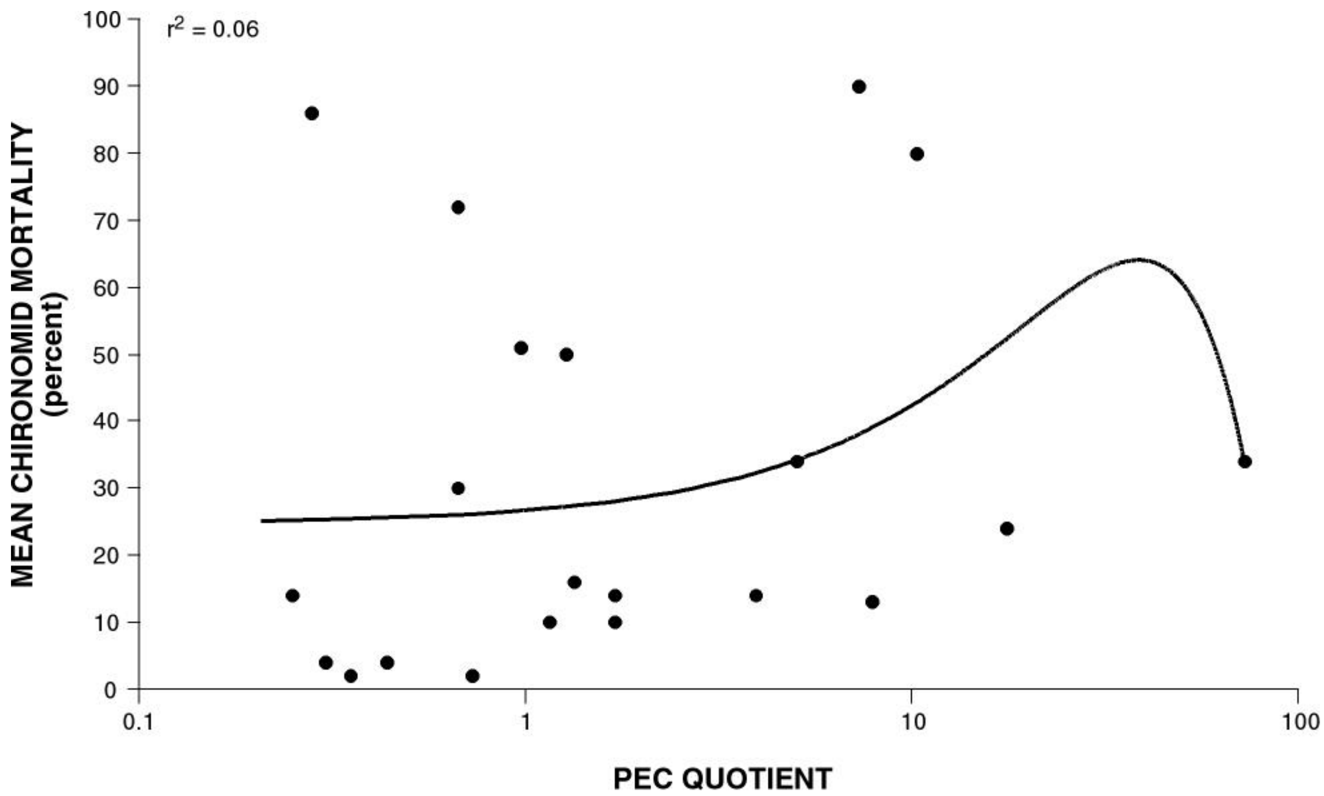

FIGURE JA.15

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Comparison of Chironomid Mortality with PEC Quotients for Hexachlorobenzene Based on Data Collected in 1992

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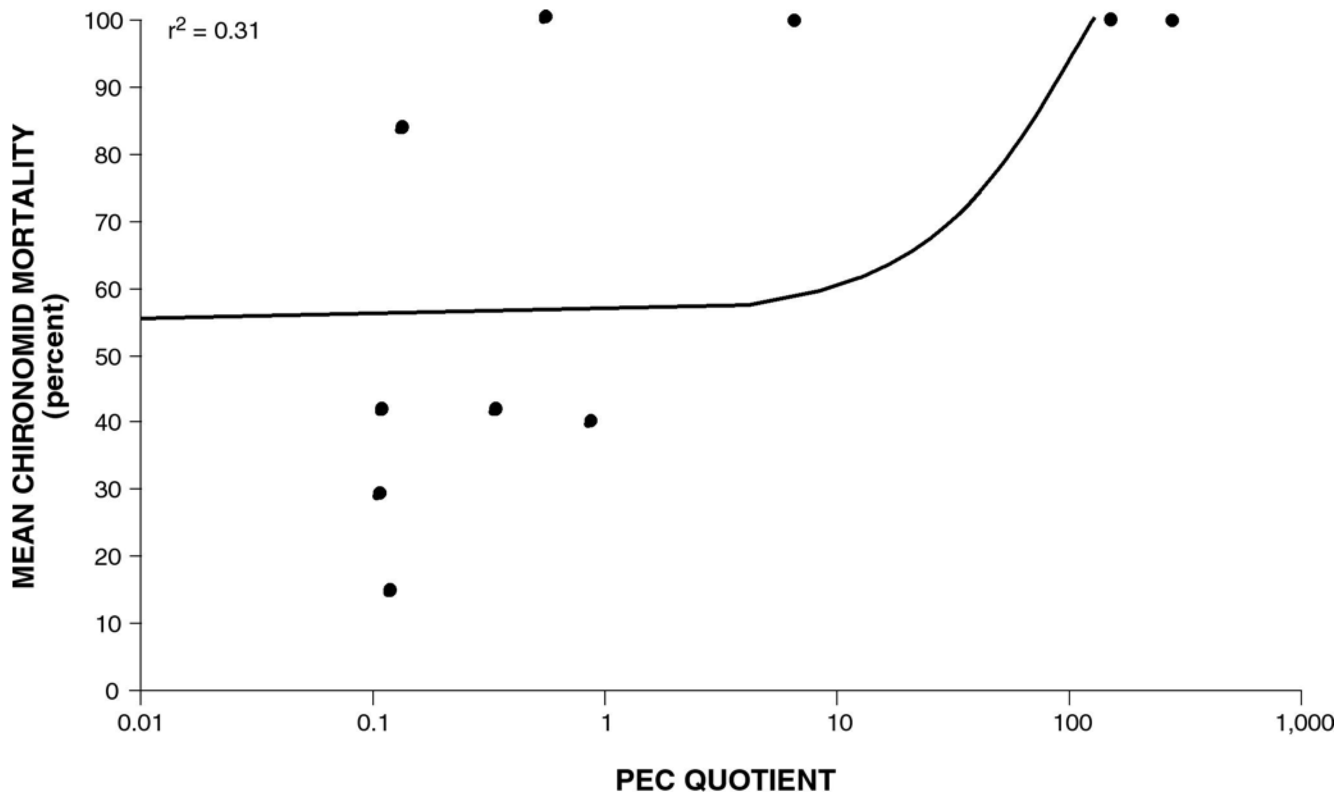

FIGURE JA.16

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Comparison of Chironomid Mortality with PEC Quotients for Monochlorobenzene Based on Data Collected in 2000

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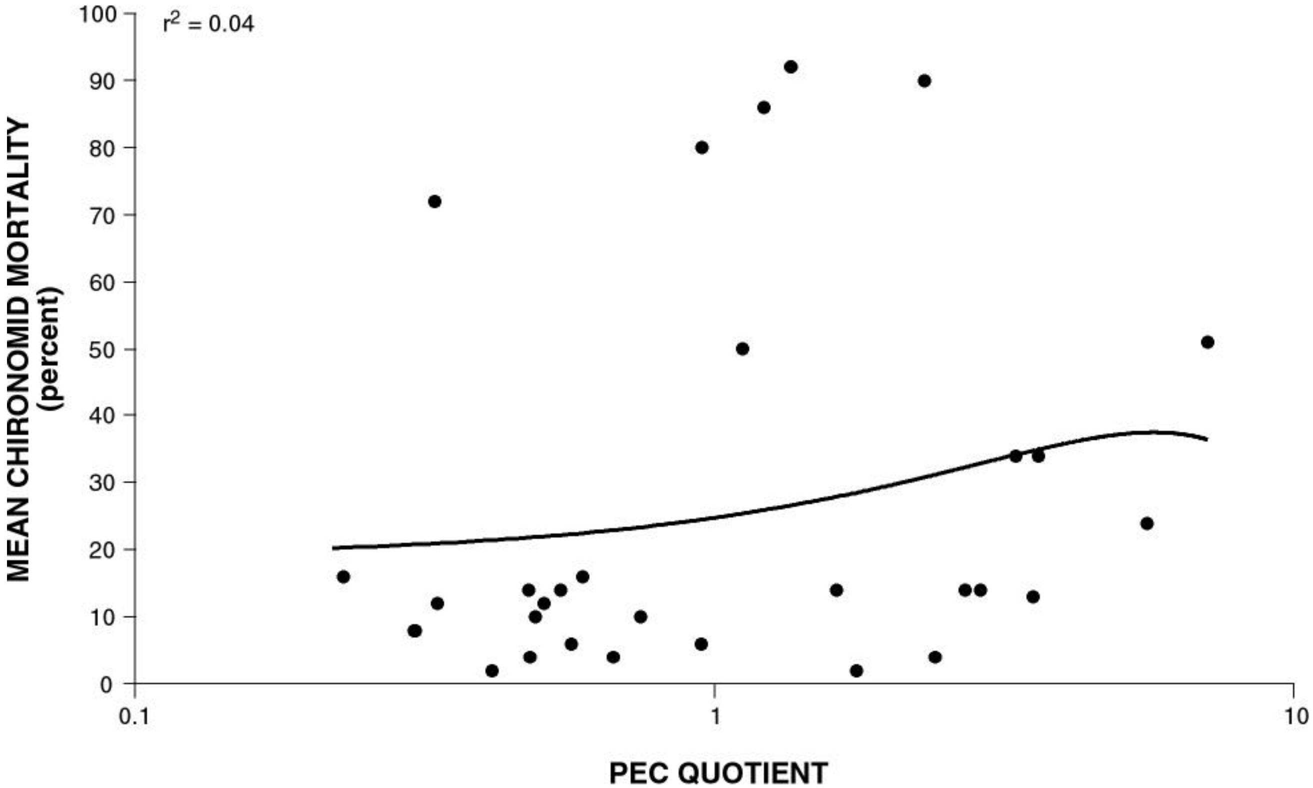

FIGURE JA.17

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Comparison of Chironomid Mortality with PEC Quotients for Total PCBs Based on Data Collected in 1992

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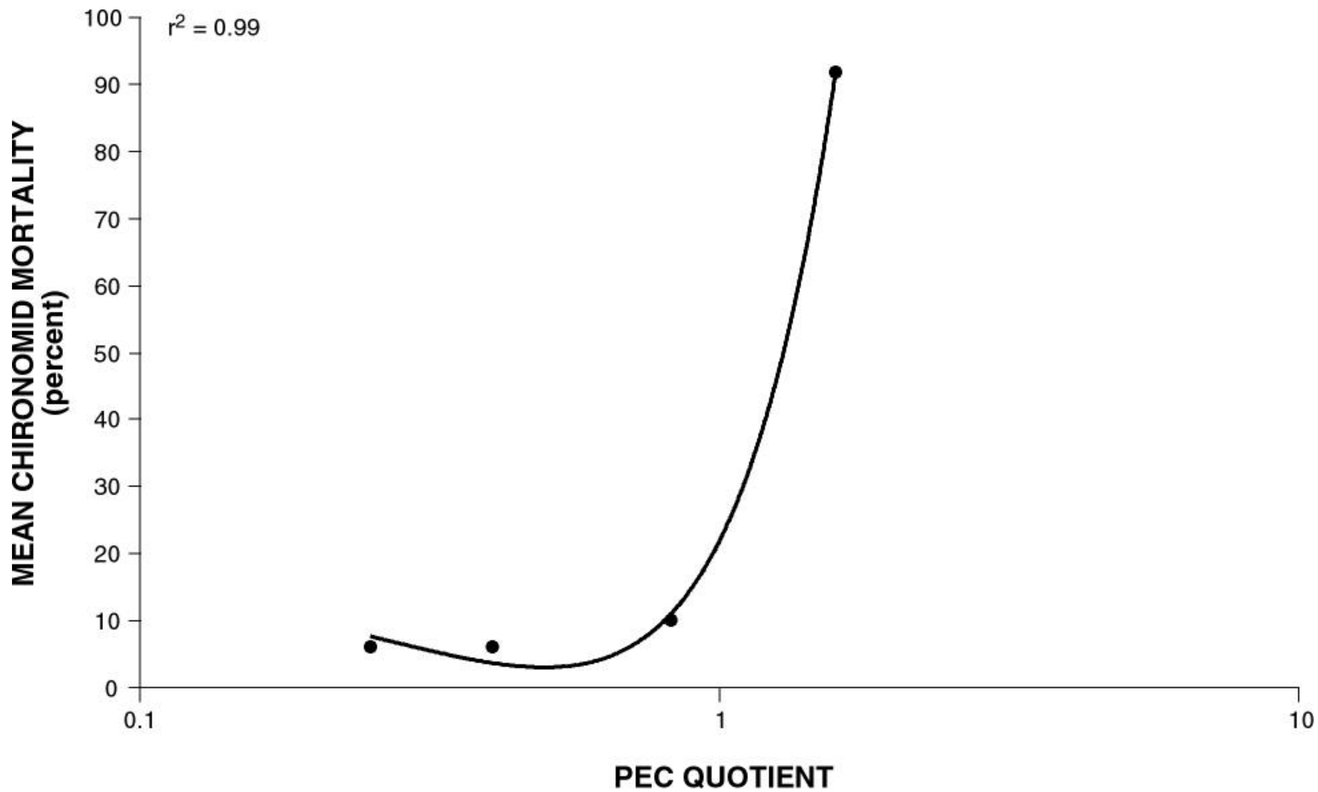

FIGURE JA.18

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Comparison of Chironomid Mortality with PEC Quotients for Pesticides Based on Data Collected in 1992

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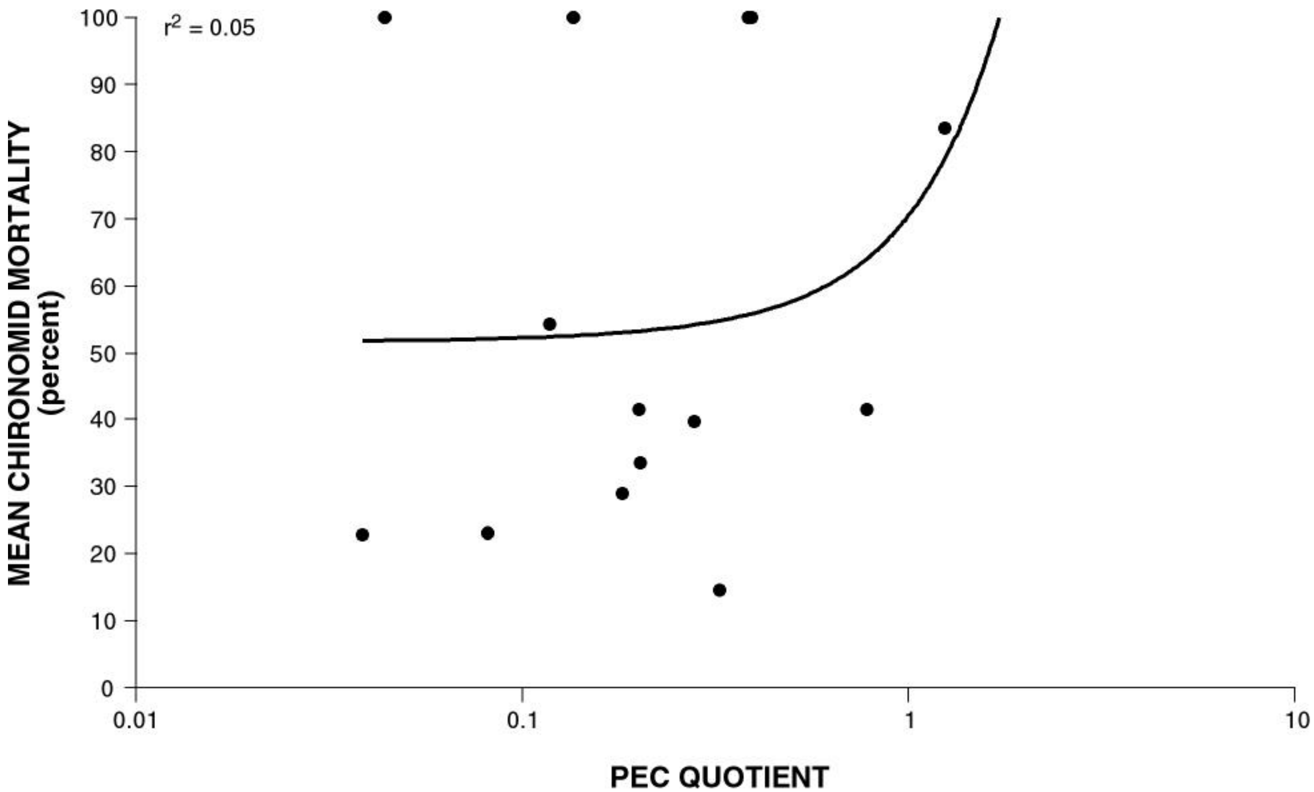

FIGURE JA.19

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Comparison of Chironomid Mortality with PEC Quotients for Pesticides Based on Data Collected in 2000

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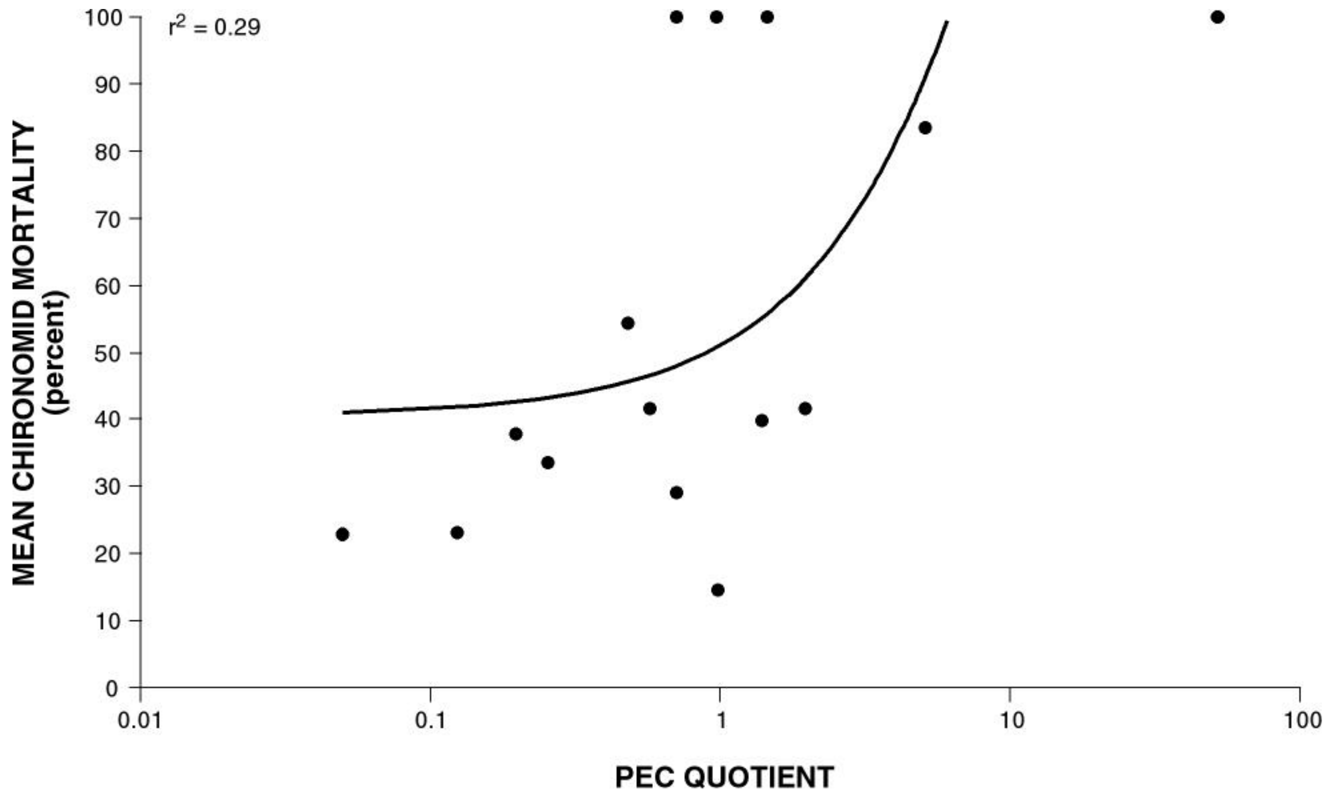

FIGURE JA.20

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Comparison of Chironomid Mortality with PEC Quotients for Total PCBs Based on Data Collected in 2000

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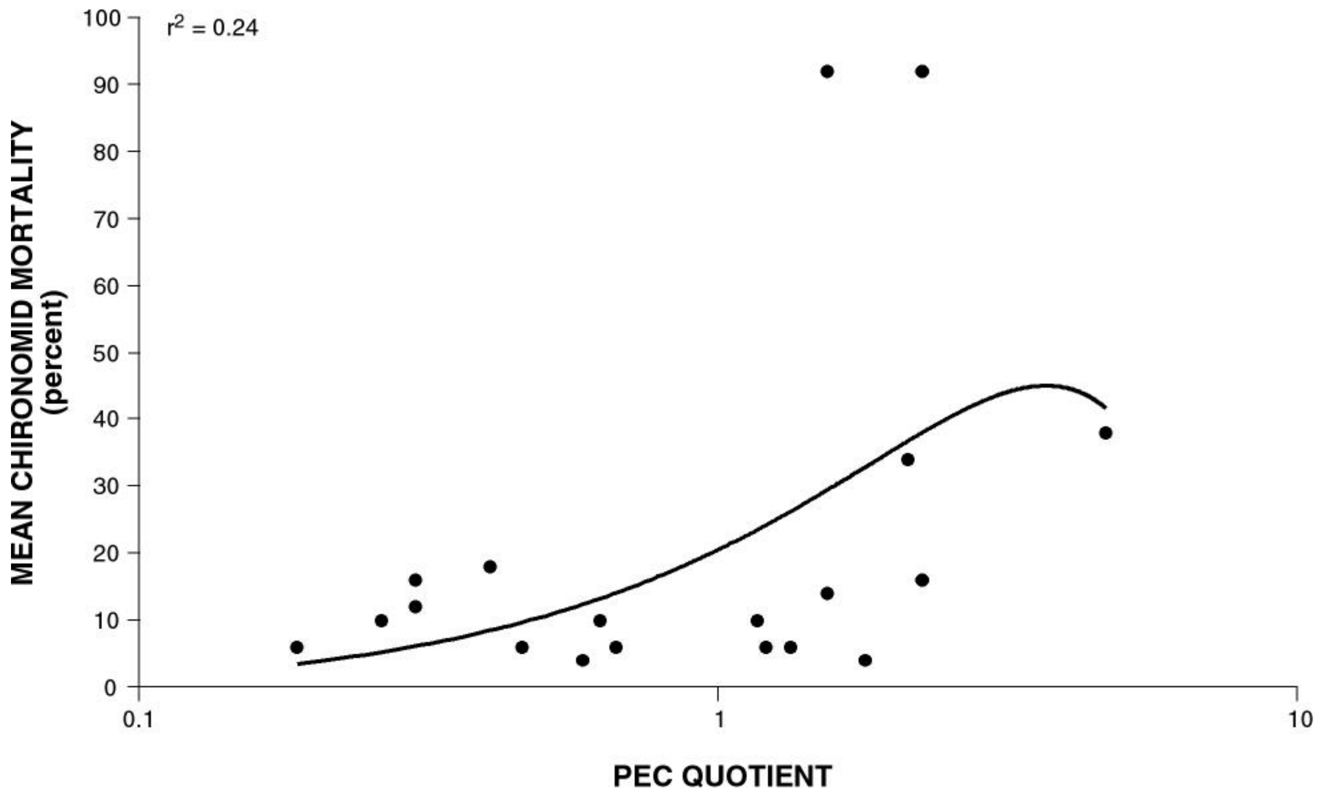

FIGURE JA.21

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Comparison of Chironomid Mortality with PEC Quotients for Arsenic Based on Data Collected in 1992

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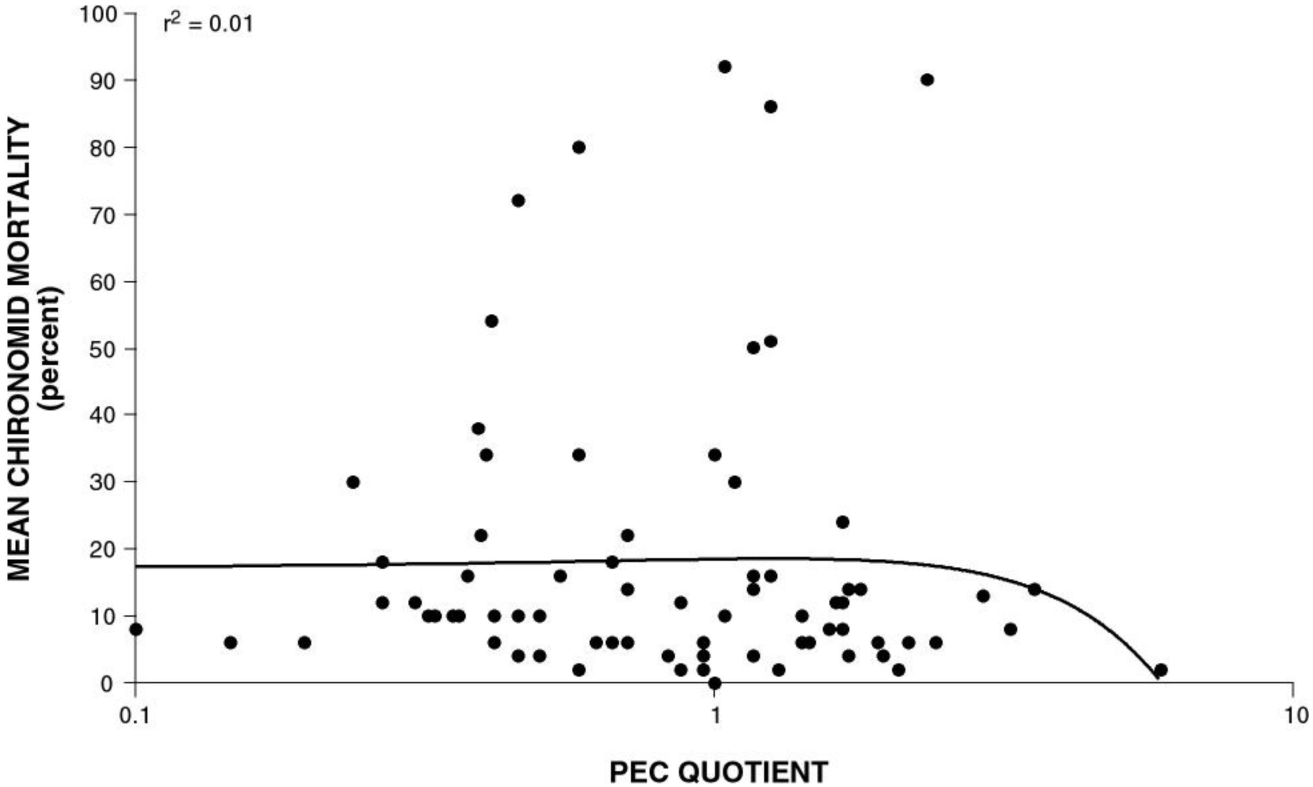

FIGURE JA.22

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Comparison of Chironomid Mortality with PEC Quotients for Cadmium Based on Data Collected in 1992

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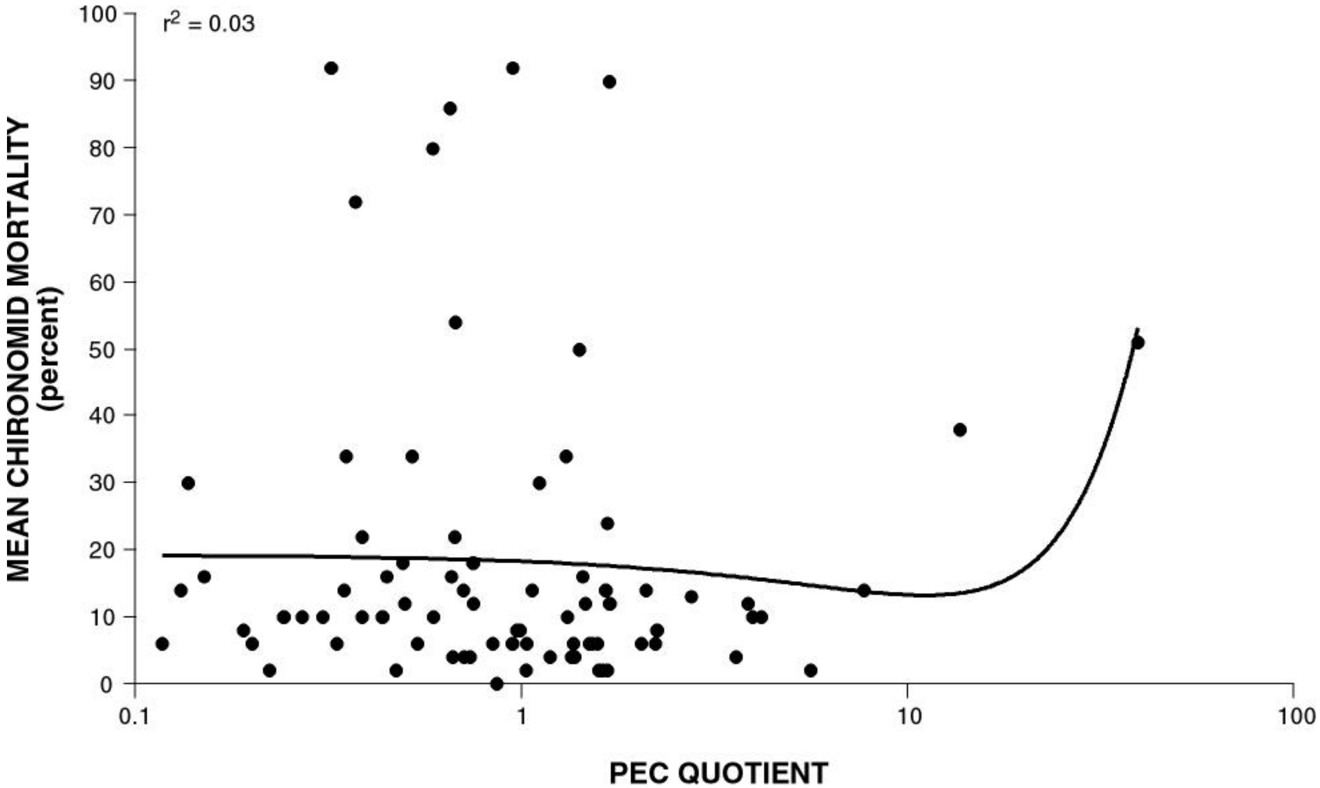

FIGURE JA.23

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Comparison of Chironomid Mortality with PEC Quotients for Chromium Based on Data Collected in 1992

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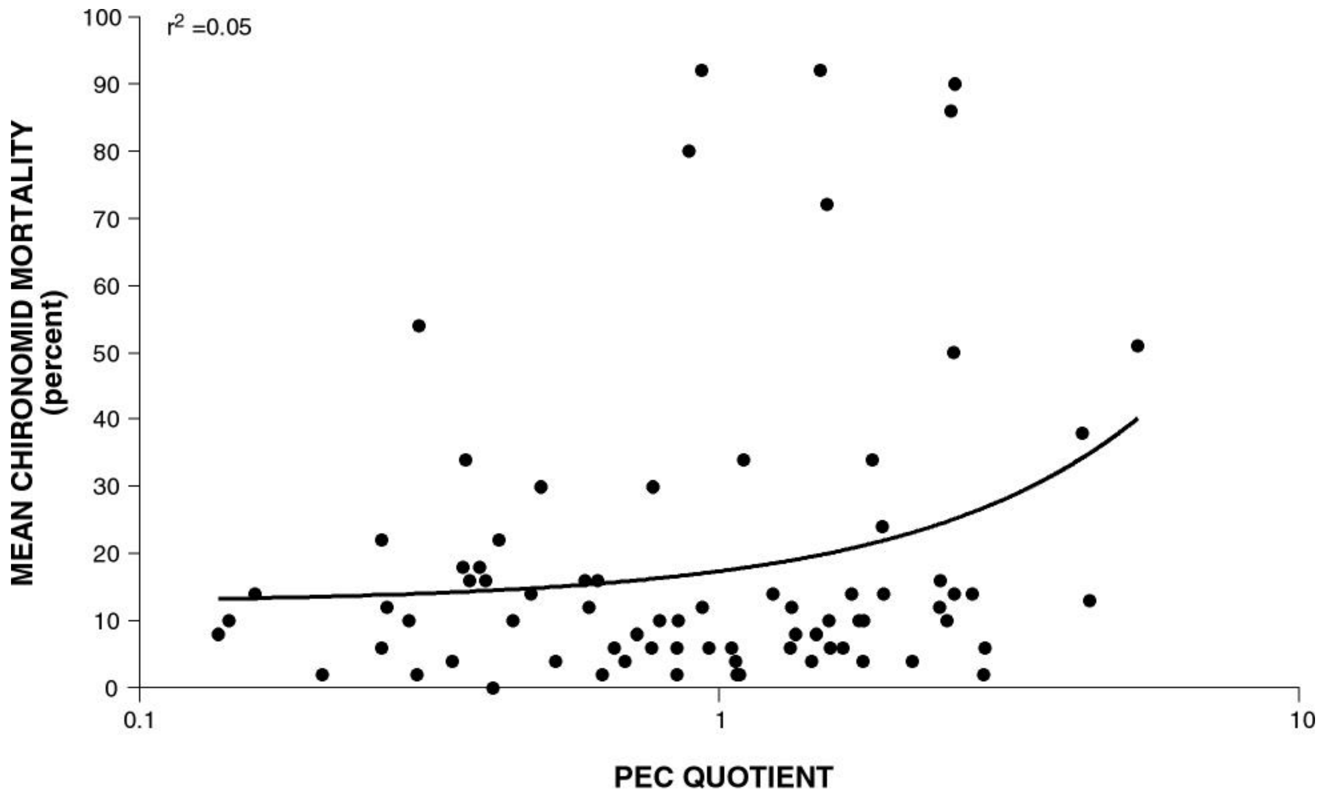

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FIGURE JA.24

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Comparison of Chironomid Mortality with PEC Quotients for Copper Based on Data Collected in 1992

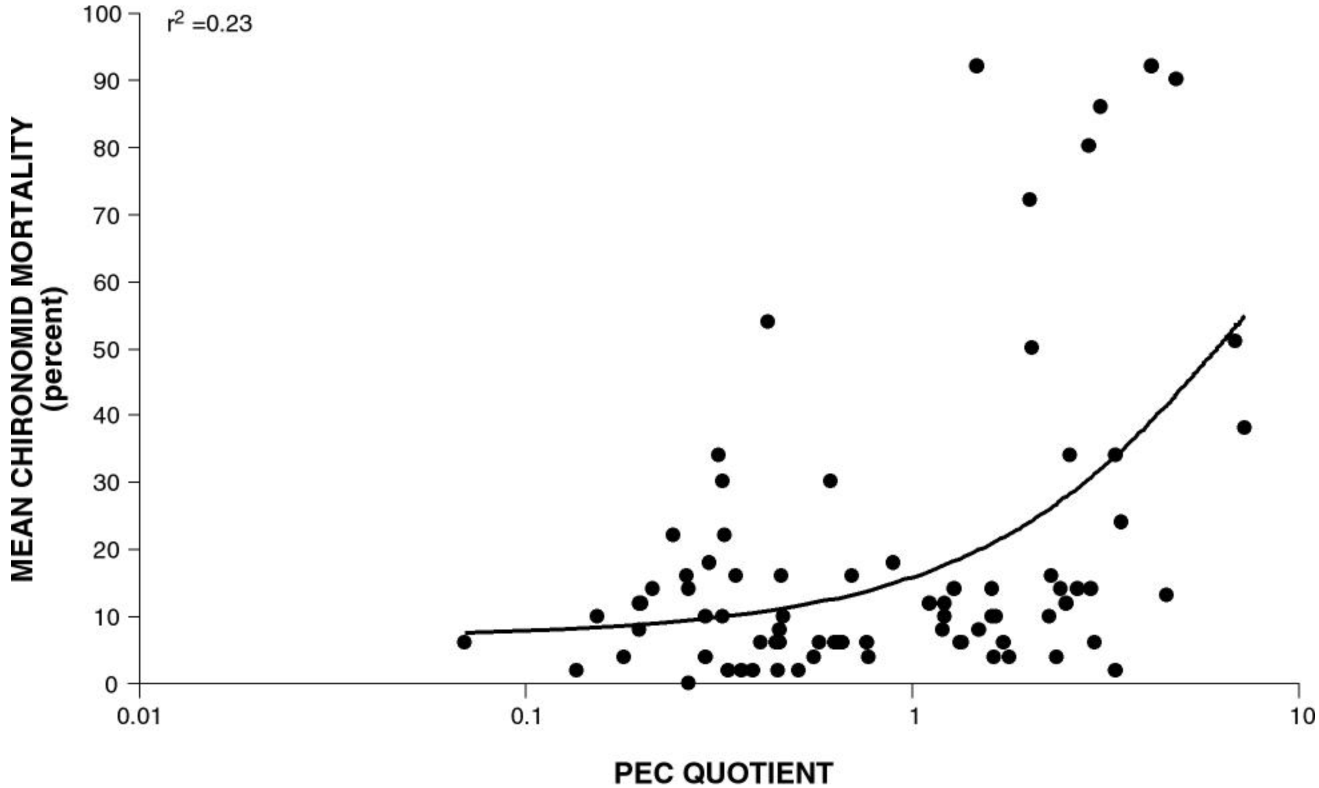

FIGURE JA.25

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Comparison of Chironomid Mortality with PEC Quotients for Lead Based on Data Collected in 1992

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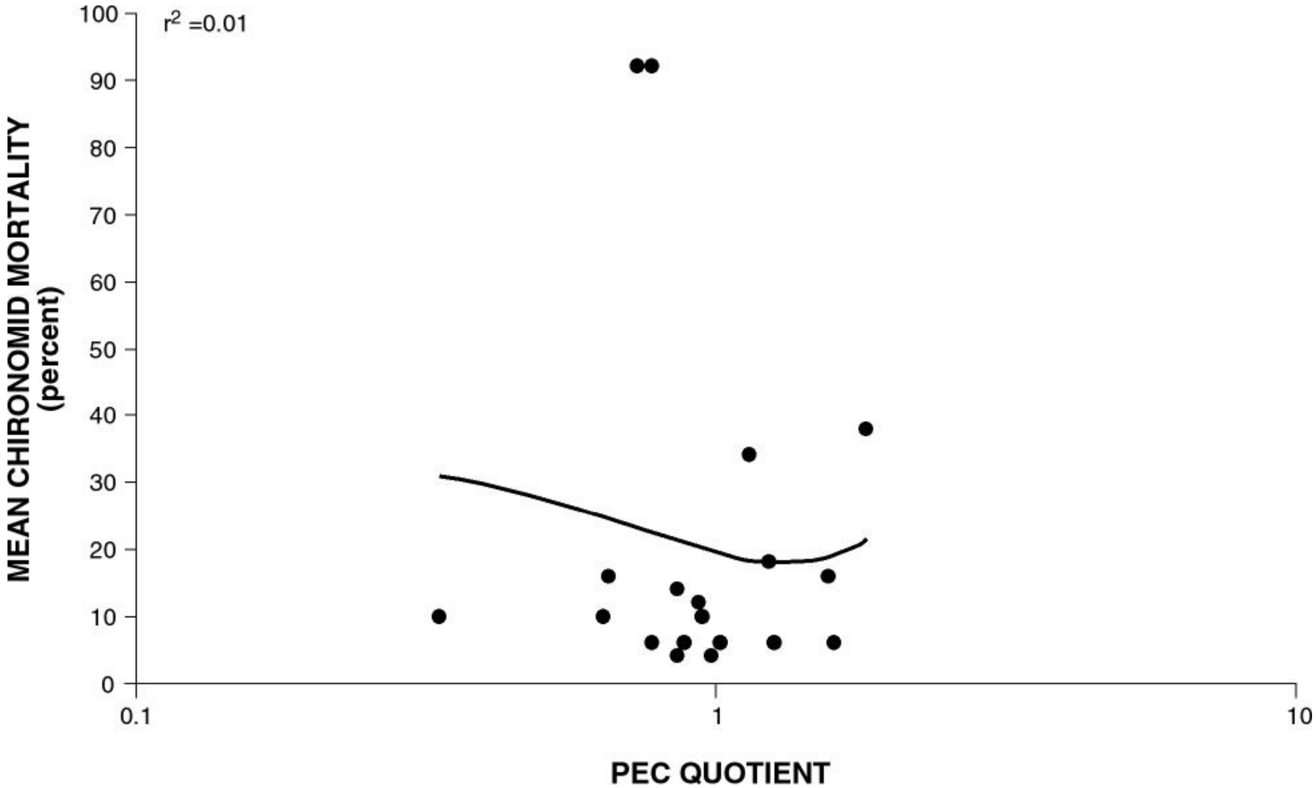

FIGURE JA.26

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Comparison of Chironomid Mortality with PEC Quotients for Manganese Based on Data Collected in 1992

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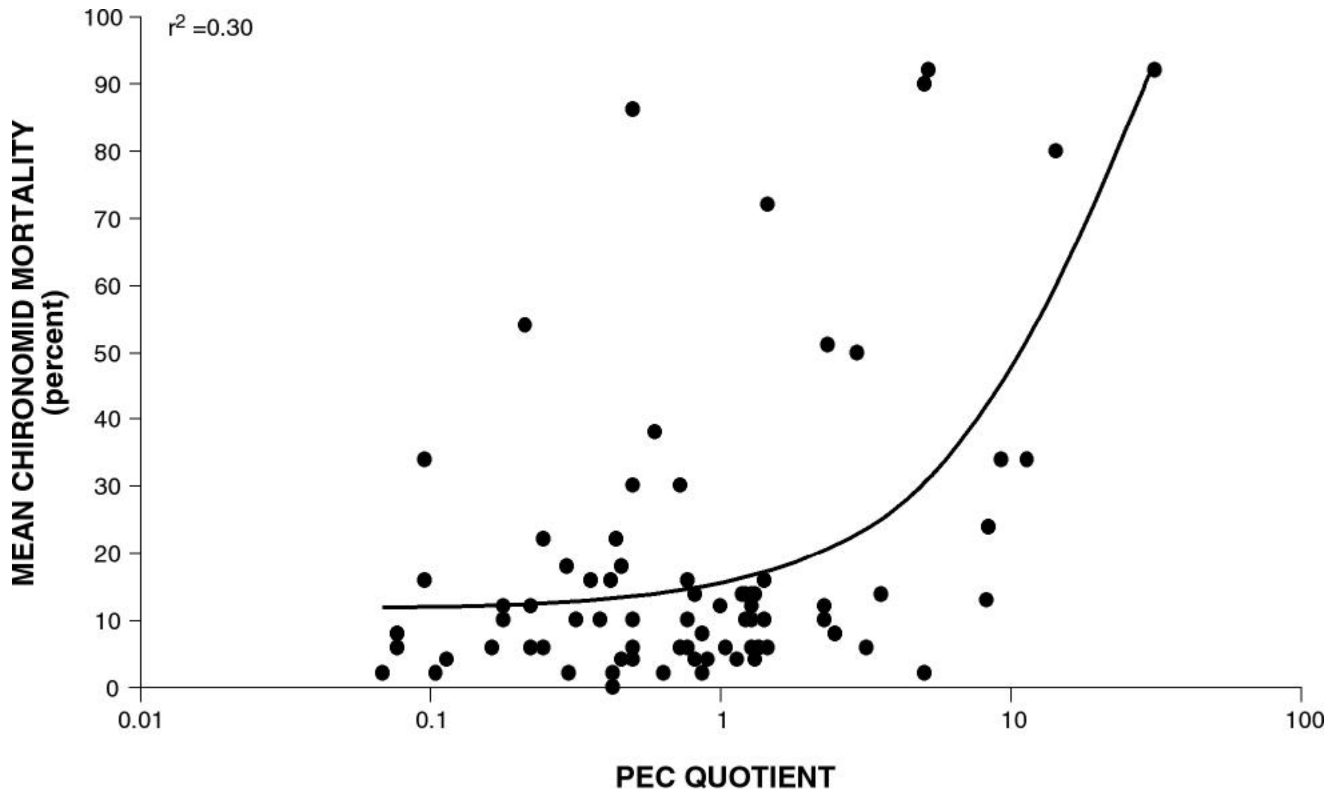

FIGURE JA.27

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Comparison of Chironomid Mortality with PEC Quotients for Mercury Based on Data Collected in 1992

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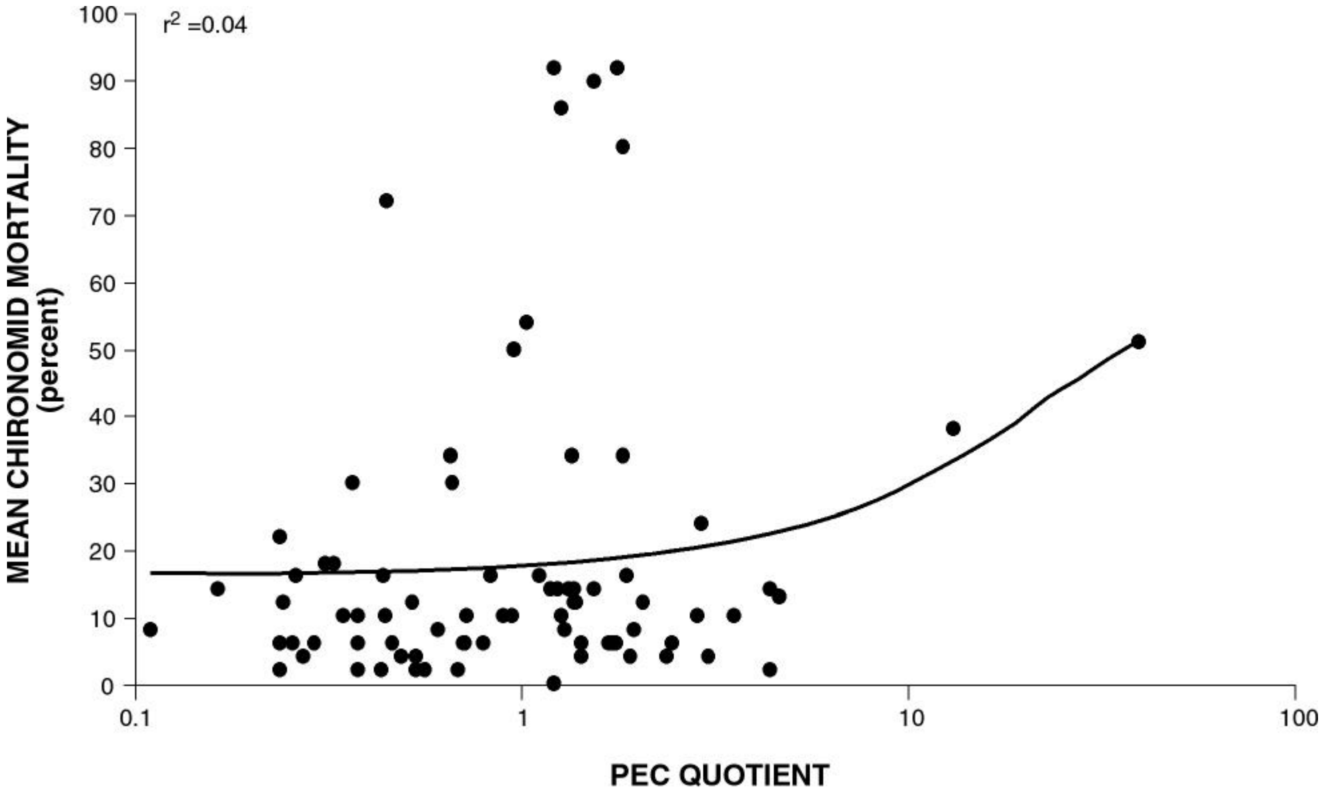

FIGURE JA.28

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Comparison of Chironomid Mortality with PEC Quotients for Nickel Based on Data Collected in 1992

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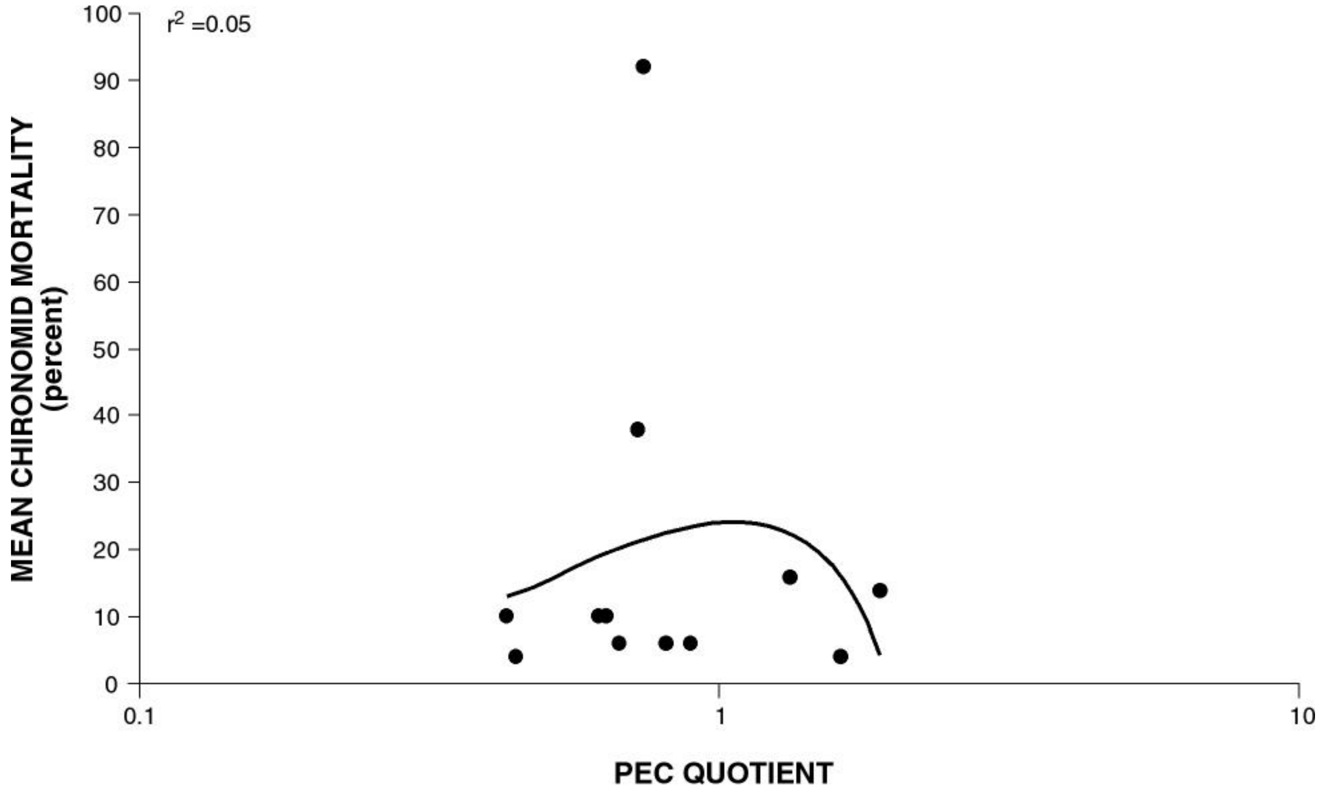

FIGURE JA.29

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Comparison of Chironomid Mortality with PEC Quotients for Selenium Based on Data Collected in 1992

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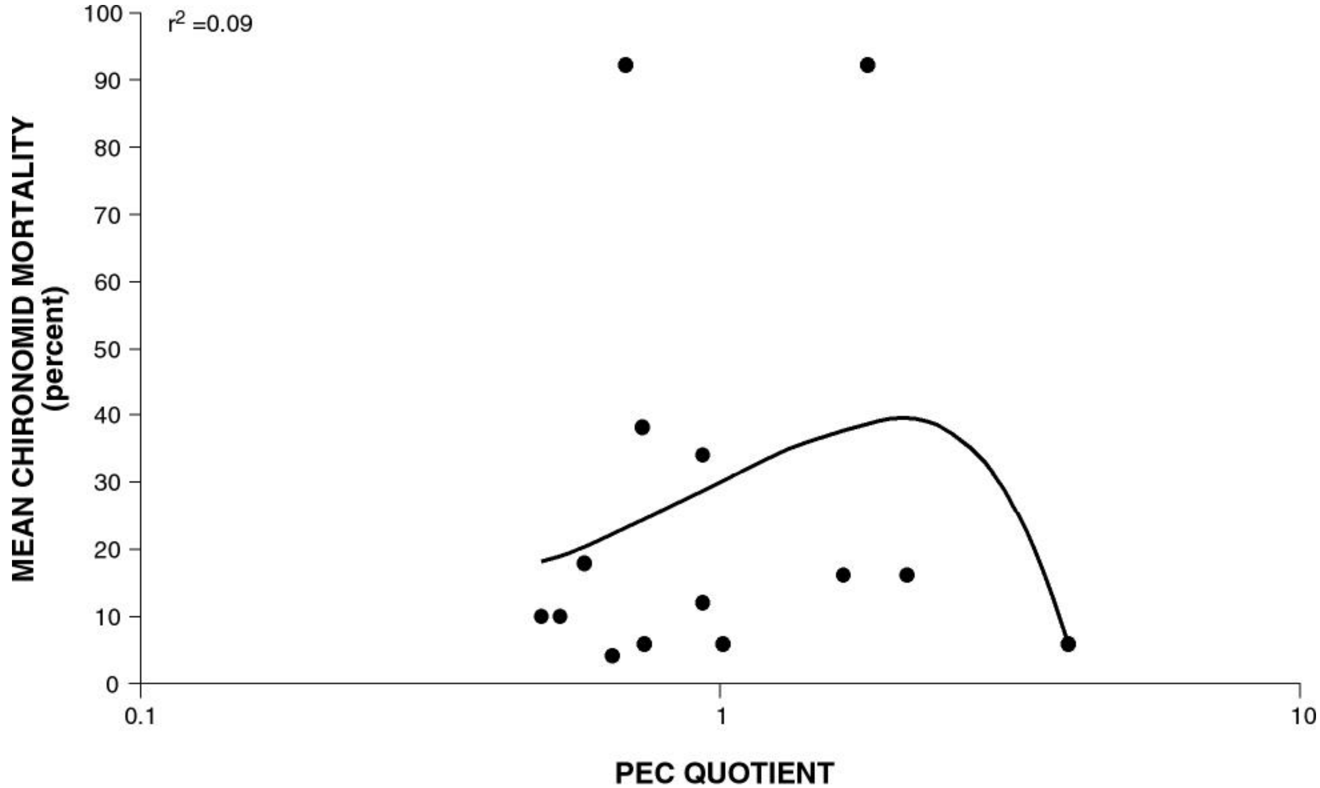

FIGURE JA.30

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Comparison of Chironomid Mortality with PEC Quotients for Silver Based on Data Collected in 1992

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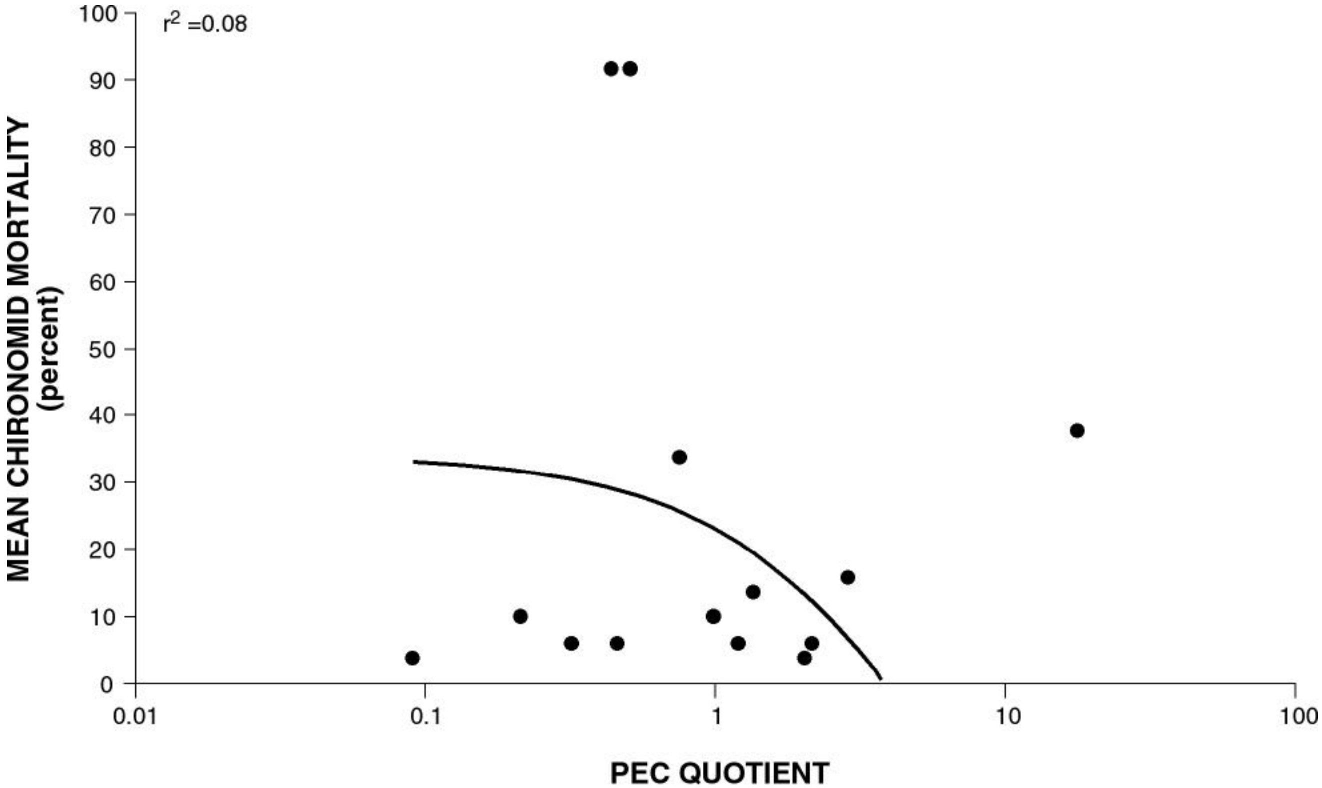

FIGURE JA.31

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Comparison of Chironomid Mortality with PEC Quotients for Vanadium Based on Data Collected in 1992

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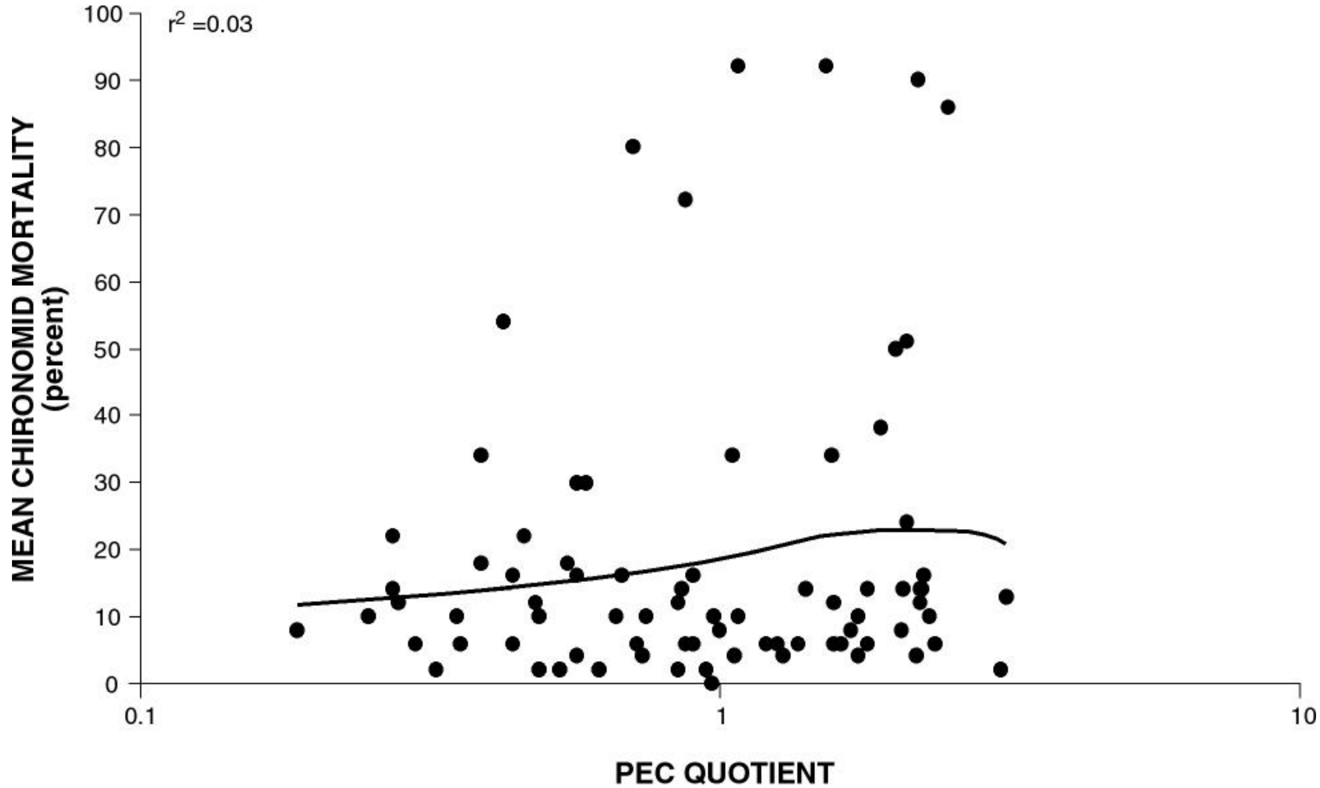

FIGURE JA.32

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Comparison of Chironomid Mortality with PEC Quotients for Zinc Based on Data Collected in 1992

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