For individual water bodies, the data set could represent changes in the water body over time, or it could represent differences in individual opinions of the same water body. Again, the final data set should capture the range of variability in both the response and predictor variables as described above.

The multinomial models described above are used in Step 8 (below) to select an optimal value or range of Chl *a* and associated N and P values that minimizes adverse impacts to all competing designated uses. The multinomial model framework allows decision makers to view competing uses graphically. This framework also overcomes issues in interpretation between multiple use models because the response is on the same scale (e.g., 1 to 5). If each designated use has a different response variable (e.g., fish species CPUE, benthic macroinvertebrate community score, recreation, treatment cost, etc.) then relating the relative importance of these metrics becomes an issue. For these models, some sort of normalization scheme must be developed if competing uses are to be discussed and debated. These types of multinomial analyses can be done using most commonly available statistical software packages, such as SAS or Minitab.

If significant relationships are not found in Step 6 analyses, then either (1) such relationships do not exist, or (2) the existing data were inadequate to identify the relationships. In this case, proceed to Step 9 to attempt to determine the factors that are confounding relationships between nutrients and response variables, or collect new data and then re-evaluate the relationships.

6.7 Step 7: Evaluate P and N vs. Response Variables

We strongly recommend the level of use suitability models discussed in Step 6. After these analyses are completed, the direct evaluation of relationships among total and dissolved P and N and primary and secondary response variables (e.g., D.O., pH) generally will be required, for example, in helping to select appropriate P and/or N criteria that will achieve the desired Chl *a* levels or secondary response levels. If significant relationships are found, they may be used to support the decisions made using the designated use models. In this case proceed to Step 8.

Primary response variables include Chl *a* (phytoplankton or periphyton) and SAV. Secondary response variables might include DO, pH, and various measures of the structure of the fish, algal (phytoplankton or periphyton), SAV, and benthic macroinvertebrate communities. Data for the other variables that affect primary production are necessary for developing relationships between nutrients and Chl *a* concentrations, as well as for evaluating designated uses. Examples of direct assessments based on field measurements of P, N, and Chl *a* and response variables are explored in this section.

Figure 6 summarizes trends in TN, TP, Chl *a* and TSS in the Neuse River near Kinston, NC from 1982-1995. From 1982-1995, TN concentrations remained fairly consistent. Beginning in 1988; however, TP concentrations declined when detergents containing phosphorus were banned from use in the watershed. Chl *a* levels also decreased during this period, suggesting that it may have responded to the decline in TP. In the Neuse River at Kinston, analysis of the data compiled from 1982–1995 indicates that TP averaged 136 µg/L, TN averaged 1481 µg/L, and the N:P ratio averaged about 9.0, suggesting that the river was P-limited.

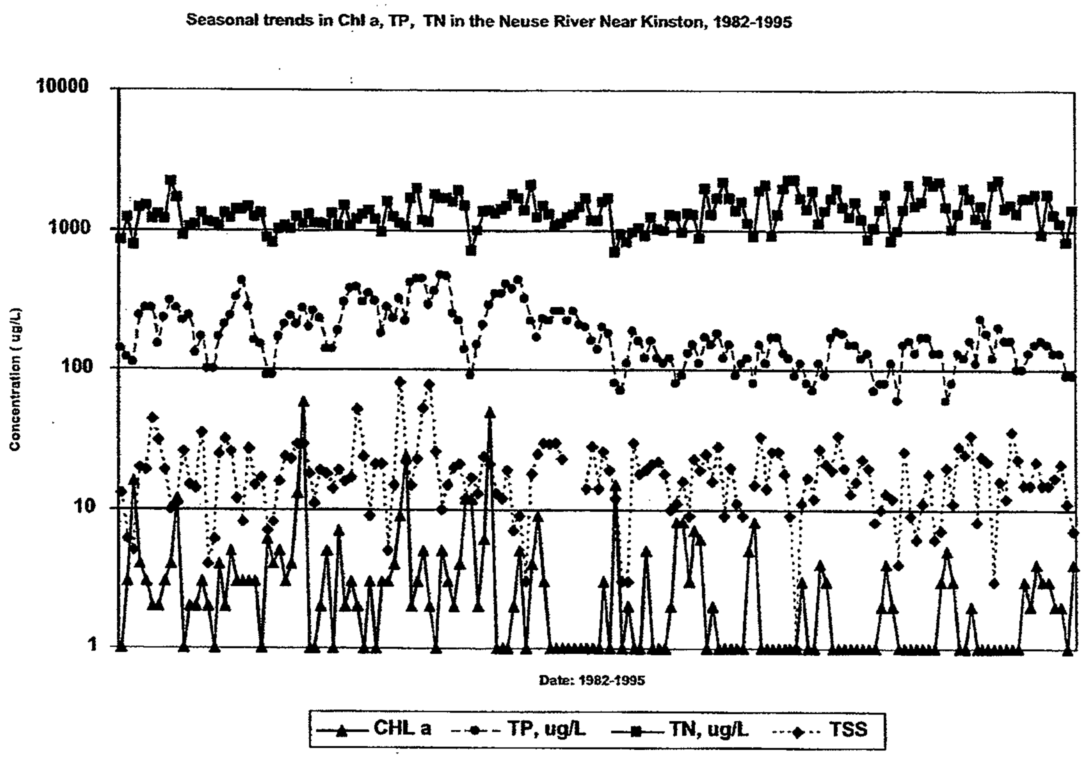

Figure 6. Seasonal Trends in Chl a, TP, TN, and TSS in the Neuse River Near Kinston. North Carolina. 1982-1995.

Box-and-whisker plots (Figure 7) are useful tools for examining shifts in the nutrient concentrations over growing seasons. From the box-and-whisker plot, investigators can visualize shifts in the mean, and median over time. Also, the presence of both low and high values in various years can be identified. Scatter plots examining relationships among response and effects variables are useful for visually assessing potential relationships among variables that can be incorporated into a predictive model. Generally, these plots are done on a base 10 logarithmic scale or on log10-transformed data. The following comparisons are useful for defining relationships between nutrients and algal biomass:

- measurements of Chl a, N, and P among seasons;
- growing season comparisons of Chl *a*, N, and P;

- summer mean total limiting nutrient concentration with summer mean and maximum Chl *a*;
- pre-maximum growth period (i.e., spring, pre-runoff) mean dissolved limiting nutrient concentration with maximum algal biomass;
- mean annual dissolved nutrient concentration with the 75th percentile mean algal biomass;
- cellular concentrations of the limiting nutrient with maximum algal biomass.

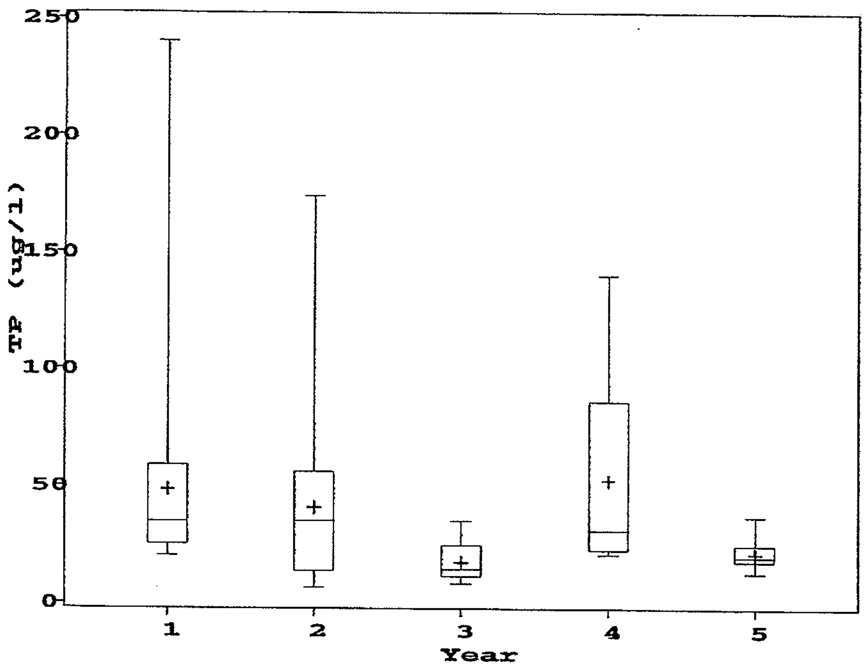

Figure 7. Comparison of TP (ug/l) in Summer Growing Seasons During Five Consecutive Years in the Neuse River.

Effects-based models are typically derived using ordinary least squares (OLS) regression techniques. These types of analyses can be done in Excel and most commonly available statistical software packages. Models of the form:

Response variable = f(effects variables) are typically used to develop the predictive relationships. The New England Interstate Water Pollution Control Commission commissioned a literature review of OLS regression models applicable to rivers and streams (2001). Regression models for different stream and river types with various substrates are presented in the document.

In most cases, a large amount of variability is seen in bivariate relationships. For example, a case study of Hinkson Creek, MO, was presented in Warren-Hicks et al. (2005). A significant relationship between daily TN and sestonic Chl *a* was observed (Figure 8) throughout the year, but no such relationships were apparent for data from the growing season. The R^2 for the relationship for the entire year was 0.43, indicating that TN accounted for about 43% of the variability in Chl *a*.

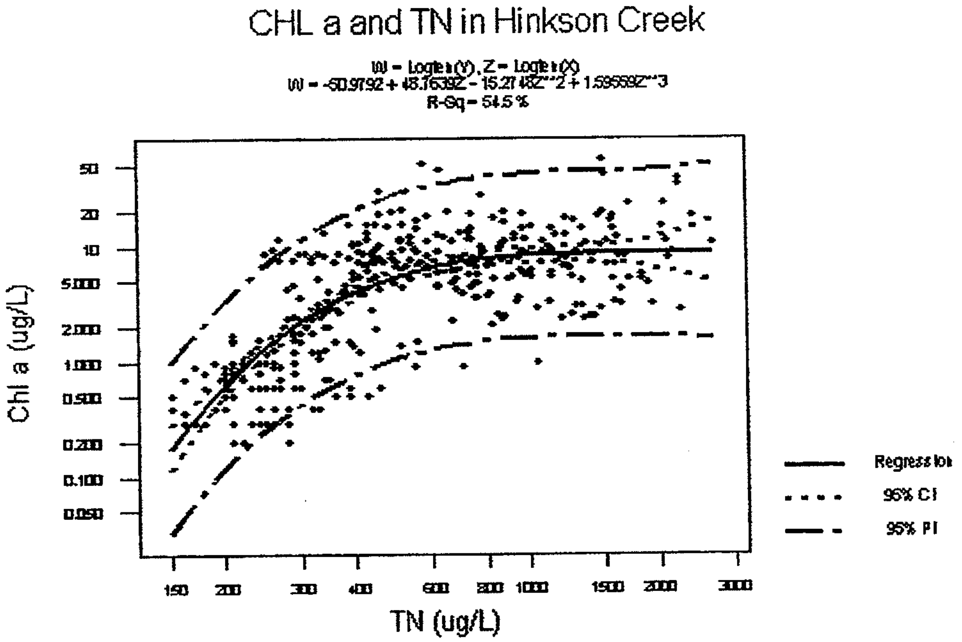

Figure 8. Linear Regression Model of Sestonic Chl a and TN in Hinkson Creek Near Columbia. Missouri. Februarv 1995–Januarv 1996.

By fitting a cubic, polynomial model to the data (Figure 9), the R^2 increased to about 0.55. The R^2 for the relationship between TP and Chl *a* was only 24% (Figure 10). The R^2 for the multiple regression between TP, TN and Chl *a* also was 43%, but in this equation the coefficient for TP was negative, which indicates that the variability in TN is dominating the relationship.

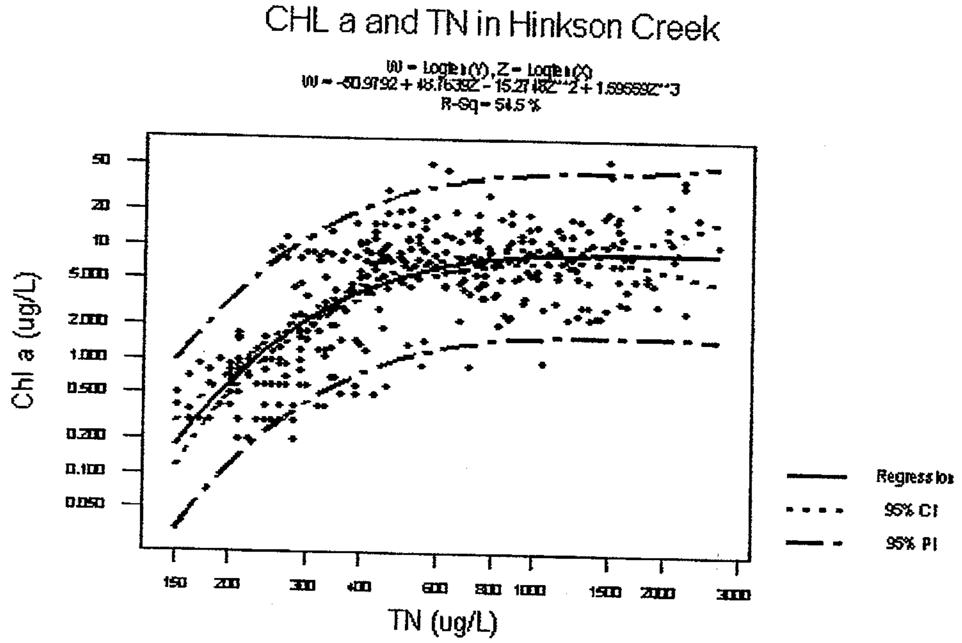

Figure 9. Cubic Model of Sestonic Chl a and TN in Hinkson Creek Near Columbia, Missouri. February 1995–January 1996.

The largest issue in the development of effects-based models is dealing with the variability in the response and predictor measurements. In many cases, linear relationships between the response and predictors variables are weak or non-existent (see Figure 10). Models developed from these data will have a large error in model prediction. Therefore, using the models to select nutrient criteria may result in selection of criteria that are over-protective or under-protective of designated uses. Investigators must factor the model prediction error into any management decisions.

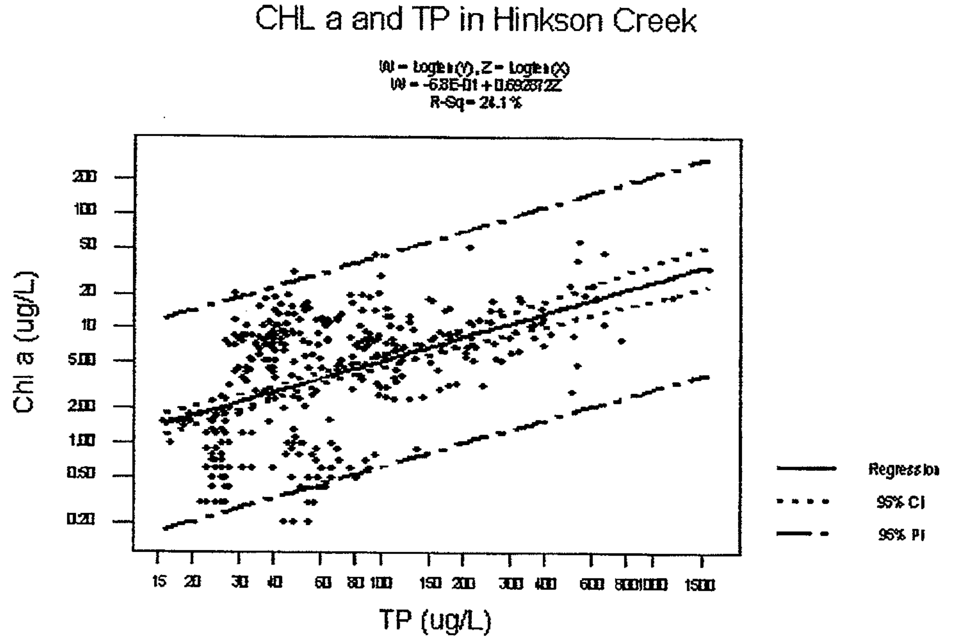

Figure 10. Linear Regression Model of Chl a and TP in Hinkson Creek Near Columbia. Missouri. February 1995–January 1996.

6.8 Step 8: Use Effects Data to Select Trial Criteria

We recommend that the selection of trial nutrient criteria be a based on the results of discussions among a group of stakeholders selected to represent all parties that have an interest in nutrient criteria for the water body or class of water bodies under evaluation. A net benefits approach to selecting trial criteria, such as discussed in Jensen et al. (2004), is recommended for reaching consensus or agreement. Jensen et al. (2004) studied the relationship between designated uses and nutrient data in nine Texas reservoirs. All of the study reservoirs had four uses that were evaluated: aquatic life diversity, sport fishing, recreation and water supply. The study determined that Chl *a* was the parameter most directly related to all uses, and that it should be the parameter selected for numerical criteria development.

Each designated use was evaluated in relation to Chl *a* concentrations. In no case were precise quantitative relationships available, but the general patterns and directions were clearly established. Jensen et al. (2004) proposed a conceptual model of the general relationship between Chl *a* and each designated use (Figures 2-4, from Jensen et al. 2004). These types of plots can be developed using the multinominal analyses described in Step 6.

The role of the stakeholders is to select the Chl *a* concentration or range of concentrations that maximizes the net benefit among competing designated uses. Figure 5 (from Jensen et al. 2004) presents a graphical illustration for selecting an optimal concentration or range of concentrations for competing designated uses (for more on this topic, see Jensen et al. 2004). Plots such as Figure 5, can be generated using the analyses described in Step 6. Based on the perceived value of each of the uses, each use could be given a different weight and then the relationships re-evaluated. Once the optimum range for Chl *a* is selected, it would become the trial Chl *a* criterion. Next, the models of P and/or N vs. response variables developed in Step 7 would used to set trial P and/or N criteria. Criteria for P and/or N would only be necessary if they were shown to be significantly related to designated uses and response variables. If no significant relationships are identified, then it would be inappropriate to set P and N criteria, because of the lack of cause and effect relationships. In such cases, it would be appropriate to only have a Chl *a* criterion.

In waters with a clearly defined limiting nutrient (i.e., atomic N:P ratio >16, or >7 by weight), nutrient criteria development should focus on that nutrient (i.e., P). In most freshwater bodies, derivation of phosphorus criteria should be the primary focus for nutrient criteria development (VA WRRC 2004), while in estuaries and coastal marine waters derivation of nitrogen criteria should be the primary focus for nutrient criteria development (US EPA 2001).

Several mathematical methods are available for optimizing the choice of Chl *a,* N, or P criteria. For example, Figure 5 shows an average of optimized curves with the individual use curves in the background. Bayesian statistical methods provide a means of finding an average curve (Gelmen et. al 1998). Multi-attribute decision analysis may provide a means for optimizing the curves for choice of Chl *a* (or N or P) without loss of information (Clemen 1996). These methods are beyond the scope of this document.

6.9 Step 9: Determine Confounding Factors

For water bodies in which significant relationships among nutrients, designated uses, and response variable cannot be identified, these relationships may not exist because of confounding factors. Variables that may confound the relationships between nutrient concentrations and algal (phytoplankton and periphyton) and submerged aquatic vegetation (SAV) production include nutrient bioavailability, suspended sediments, turbidity, shading, transparency, stream scouring, stream velocity, invertebrate grazing, and stream substrate. For example, if the waterbody of interest has decreased transparency because of naturally high concentrations of dissolved organic carbon or suspended sediments, photosynthesis may be inhibited and algal biomass reduced. Such relationships could justify higher nutrient criteria provided downstream conditions would not be adversely affected, or indicate no linkage between designated uses and nutrients. The same types of procedures described in Step 7 should be used to evaluate the significance of these potentially confounding factors.

6.10 Step 10: Use Non-Effects-Based Data to Select Trial Criteria

For water bodies or classes of water bodies for which relationships among Chl *a*, levels of use support, and response variables cannot be adequately defined, either because of high variability or confounding factors, criteria development could be based solely on (1) the results of user perception surveys, or (2) ambient Chl *a* levels. User perception surveys compare user perceptions of water quality and designated uses of the water body with measurements of P, N, Chl *a*, water clarity and/or other nutrient and response variables. Tables 3 and 4 present results of an example survey for a hypothetical lake. With these kinds of qualitative data, relationships between user perception and response variables could be developed using the methods in Step 6. If significant relationships are identified, then Figures, such as 2-5 (Jensen et al. 2004), can be generated using multinominal analyses. The results are then used by the stakeholder group to select the Chl *a* concentration or range of concentrations that maximizes the net benefit among competing designated uses using the same process as in Step 8. If no significant relationships between P and/or N vs. response variables are identified, which is likely if none were present for Chl *a* and designated use support, then it would be inappropriate to set P and N criteria, because of the lack of cause and effect relationships. In such cases, it would be appropriate to only have Chl *a* criteria. For such waters, the Chl *a* criterion could be set it as an upper estimate of long-term average levels, such as the mean plus 99% confidence level or the upper 99th percentile, which cannot be exceeded for some fixed percent of the time, (e.g., 10%).

The literature shows that user perception differs regionally as well as on a site-specific basis. One good example is taken from Minnesota, where large regional differences occur in the concentration of TP, an indicator of lake trophic state, which is considered acceptable for drinking water supply and primary contact recreation (Table 6). Perception regarding the level of algal biomass that constitutes a nuisance condition (Table 7) also varies across the region. In Oregon, Chl *a* levels >15 µg/L are considered to impair the beneficial uses of natural lakes, reservoirs, rivers, and estuaries. However, in North Carolina, the Chl *a* criterion is 40 µg/L. Chl *a* criteria for Ontario lakes are quite restrictive relative to other regions in North America (Table 8). These differences reflect regional water quality objectives and related uses that are locally attainable.

As stated previously, the recommended approach is to use quantitative data to determine relationships between the level of use support and Chl *a*. By linking the qualitative perception survey results with quantitative nutrient and response variable measurements, nutrient criteria that are scientifically sound, yet meet the needs of users can be obtained. If states or other