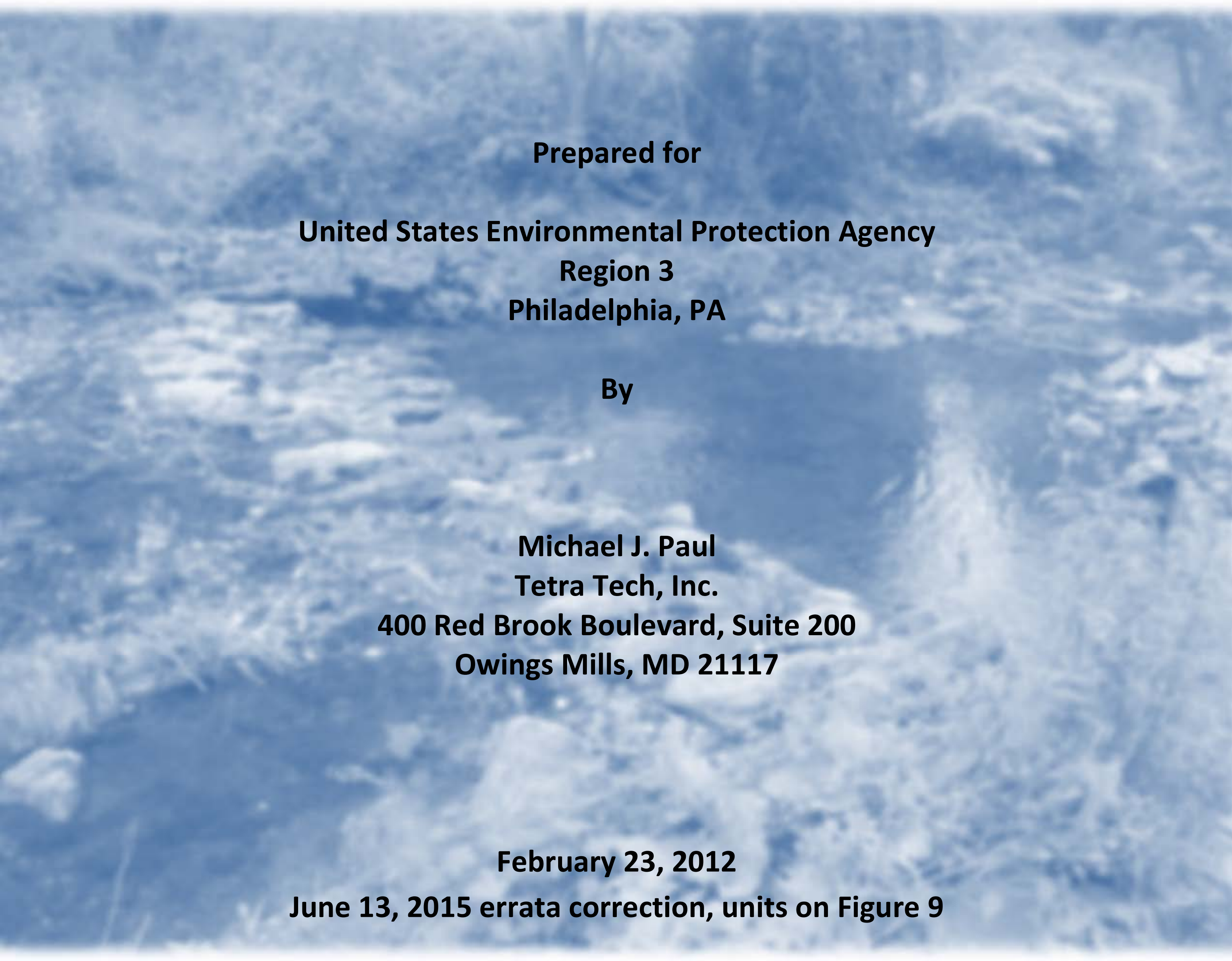
Evaluation of Nutrients as a Stressor of Aquatic Life in Wissahickon Creek, PA



Prepared for

United States Environmental Protection Agency
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By

Michael J. Paul
Tetra Tech, Inc.
400 Red Brook Boulevard, Suite 200
Owings Mills, MD 21117

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June 13, 2015 errata correction, units on Figure 9

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CONTENTS

LIST OF FIGURES

LIST OF TABLES

INTRODUCTION

The objective of this document is to evaluate support for the basis that nutrients are a stressor on the condition of aquatic life in Wissahickon Creek, Pennsylvania. This creek has been listed as impaired under the 303(d) section of the Clean Water Act for nutrients as it pertains to the aquatic life beneficial use and this analysis is focused on evaluating the strength of evidence for that conclusion. The approach adopted in this validation effort is the USEPA Stressor Identification (SI) Guidance document (USEPA 2000a). It proceeds among a very similar path to the SI process, except that instead of evaluating along several potential stressors, this analysis evaluates the strength of evidence for a single stressor. As a result, the major difference between this document and the USEPA (2000a) guidance is this starts with a single stressor, constructs a conceptual model of the causal path, generates predictions based on the model, analyzes the evidence from within the study area in support or refutation of the causal model, characterizes the evidence, and then evaluates the strength of evidence or probability for nutrients as a stressor of aquatic life using the SI scoring tables. The next section describes the conceptual model of nutrient impacts and makes a series of predictions based on that model, the following section evaluates the evidence for consistency with those predictions and puts those into the context of evidence from outside the study area (scientific literature) where appropriate, and the last section describes conclusions of the two evidentiary lines in terms of consistency of the different lines of evidence and strength of support for the conclusion.

CONCEPTUAL MODEL OF NUTRIENT IMPACTS IN WISSAHICKON CREEK AND PREDICTIONS

An important part of evaluating the basis for a causal linkage is starting with a conceptual model of how a stressor is linked to a use impairment response. In the case of the Wissahickon, the presumed basis for the impairment is an impact on aquatic life use as evaluated with invertebrate assemblage indicators. This impact was associated with notable observations of excessive algal growth in the channel, the proliferation of which was presumed due, in part, to excess nutrient concentrations which were felt to contribute to impairment of the use. The stream was, therefore, listed for nutrient impairment, among other stressors, the mitigation of which is intended to contribute to restoring aquatic life use. This effort is focused on nutrient stressors contributing to aquatic life use impairment, and therefore, the conceptual model discussed here (Figure 1) is for the effects of nutrients on the invertebrate assemblage.

The principal human activities in the Wissahickon watershed are urban/suburban/industrial and these land uses are the principal sources of nutrients (Figure 2). These nutrients enter the stream through direct discharge of treated wastewater, non-point source runoff of applied nutrient fertilizer, eroded nutrient bearing sediment, accumulated atmospheric nutrient inputs from surfaces during storms, and erosion of nutrient bearing soils from hillslopes or streambanks that occur as a result of land and channel alteration. Point and non-point source nutrient inputs result in increases in nutrient concentrations in surface water from direct runoff as well as increases in soil nutrient concentrations

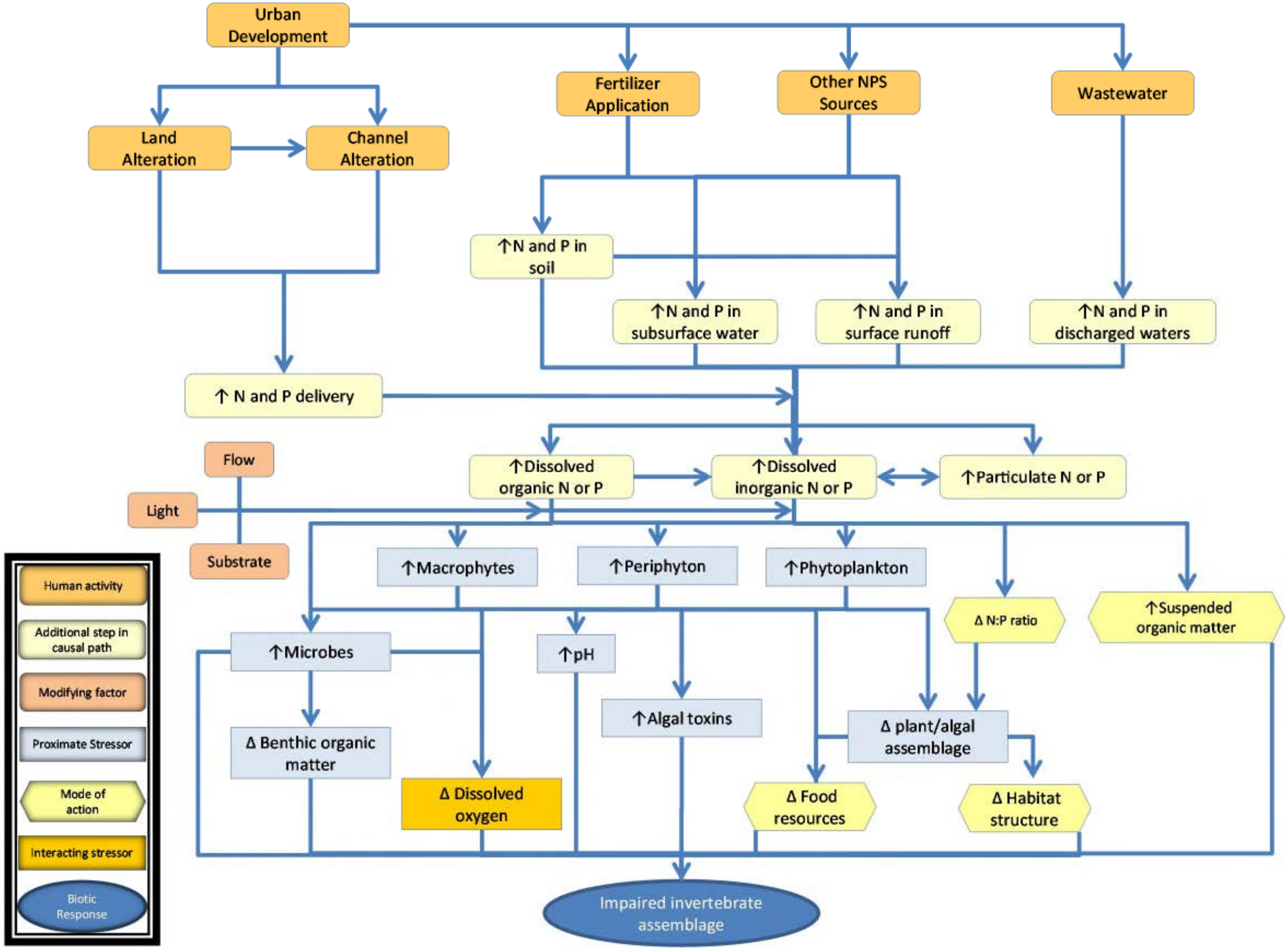


Figure 1 - Conceptual model of nutrient effects on invertebrates in Wissahickon Creek, PA

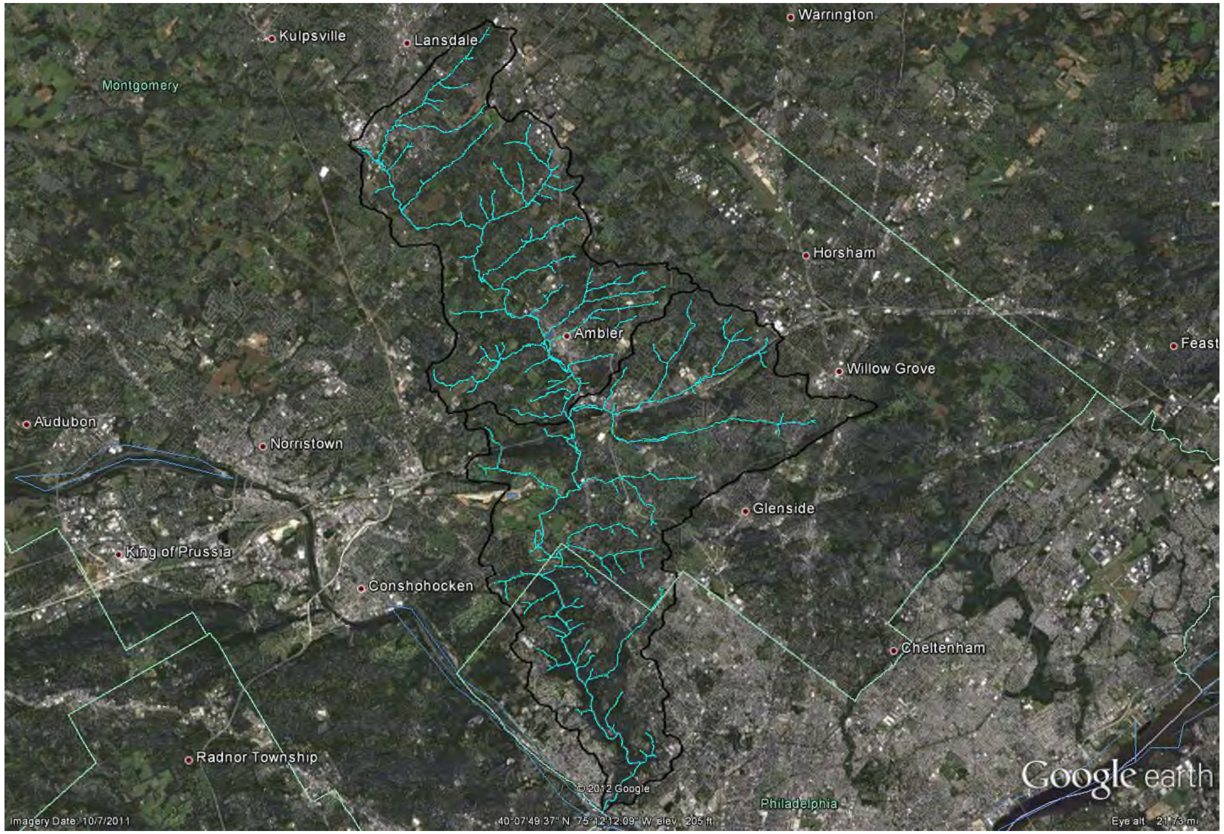


Figure 2 - Google Earth image of the suburban/urban Wissahickon watershed, indicating flow lines and HUC 12 subwatersheds.

and subsurface water concentrations that are released over time (Allan 1995, Dodds 2002) . The effects of these direct and indirect nutrient inputs is an increase in dissolved organic, inorganic and particulate nutrient concentrations under both storm and baseflow conditions (USEPA 2000b). Within the stream channel, nutrients will cycle between dissolved and organic/particulate forms as mineralization and uptake occur and nutrients move through the stream network (Stream Solute Workshop 1990).

Increased nutrient concentrations affect a number of proximate stressors to invertebrates. Increased organic matter production is a primary response which leads to increases in macrophyte, periphyton, and phytoplanktonic production (Allan 1995, Dodds 2002). While the occurrence on the latter is likely small in the Wissahickon because the flow conditions prohibit a true phytoplankton, the former two are likely important responses.

Note that primary producer response can be limited by light, flow, and substrate (Allan 1995, Dodds 2002). Where the stream is shaded from riparian canopy, primary production may be light limited and therefore show limited response to nutrient enrichment. Similarly, during periods of high flow which are more frequent in urban watersheds, shear on the bed can remove plant and algal biomass. Lastly, unstable substrates can limit the accumulation of primary producer biomass. These modifying factors are important in interpreting causal-response data (USEPA 2000a).

Increased organic matter load to the system has several effects. First, it increases the organic matter for heterotrophic decomposers (microbes) which decompose the excess organic matter, the respiration from which reduces dissolved oxygen (Suberkropp 1995, Wetzel 2001, Dodds 2002, Gulis et al. 2004). Combined with increased dissolved oxygen inputs from the increased primary productivity, one prediction is an increase in diel flux with concomitant reduced oxygen minima and increased dissolved oxygen maxima. These changes are influenced by reaeration rates in the channel, the magnitude of which will either exacerbate or mitigate oxygen responses.

Nutrients also directly stimulate heterotrophs, which increases decomposition rate, respiration, and alters the standing stock of natural organic matter (Gulis et al. 2004, Cross et al. 2007). Thus, it is important to highlight that nutrient enrichment has a dual effect on heterotrophic respiration because it indirectly stimulates respiration by increasing organic matter supply and directly stimulates heterotrophic respiration by enhancing nutrient limited microbes. Increased productivity alters the dissolved inorganic carbon dynamics as well, which affects pH, alternatively increasing (during the day) and decreasing (at night) pH as photosynthesis and respiration alter dissolved carbon concentrations (Wetzel 2001).

Increased water column and benthic production changes the amount and quality of suspended organic matter, both increasing the amounts and altering the composition of that material (Allan 1995). Nutrient enrichment is expected, therefore, to alter the nutrient composition of organic matter especially periphyton (Wetzel 2001, Dodds 2002).

Lastly, increased nutrient concentrations also alter the competitive balance among plant and algal species, which differ in their nutrient uptake kinetics. This competition determines species presence/absence and abundance (Hutchinson, 1959, Tilman 1977, 1981, 1985). This effect occurs through the alteration of nutrient concentration, as well as the ratio of nutrients. Both effects change the competitive balance and result in a change in the composition of plant and algal flora. Plant and algal species vary in their palatability and edibility, so these changes have an effect on food resources to consumers (Wetzel 2001, Dodds 2002). Moreover, changing the composition and amount of plant and algal material alters the physical habitat for invertebrate colonization, movement, feeding, and reproduction.

Each of the proximate stressors noted above, in turn, can affect aquatic life, specifically invertebrates. Decreases in dissolved oxygen have chronic and acute effects on invertebrate mortality, depending on the magnitude, frequency, and duration of the reduction, because the invertebrate stream fauna includes many oxygen sensitive species (Allan 1995). Similarly, invertebrates differ in their pH sensitivities, and prolonged exposure to pH alteration negatively impacts these species (Allan 1995).Alteration of organic matter standing stocks affect populations of shredders and the collector-gatherers that rely on specific organic matter and the timing of its availability for their production (Cross et al. 2003, 2005, 2006, 2007). Moreover, alteration of the microbial flora on decomposing organic matter alters the palatability of that resource for shredders (Suberkropp et al. 1983, Allan 1995). Both of these responses alter invertebrate fauna. Changes in the quality of food resources also affect herbivorous invertebrates and invertebrate species that filter feed because of the effect on processing and nutrition (Resh and Rosenberg 1984). Lastly, the alteration of physical habitat mediated from excess plant and algal growth will also negatively affect certain invertebrate species adapted to natural habitat conditions (Resh and Rosenberg 1984, Allan 1995).

The causal model presented was used to posit several causal paths that were further evaluated with empirical information from the system and from outside the system (Figure 1). The causal paths were used to generate a series of predictions important to testing the hypothetical nutrient-response linkage, namely:

1. Evidence of increased nutrient concentrations in the stream associated with runoff and discharges, as well as baseflow;
2. Evidence of altered ambient/water column N:P ratio associated with elevated nutrient loads;
3. Evidence of increased algal/plant biomass at locations pursuant or coincident with elevated nutrients;
4. Evidence of altered plant/algal assemblage structure pursuant or coincident with elevated nutrients;
5. Evidence of altered suspended organic matter composition and altered periphyton nutrient ratios pursuant or coincident with elevated nutrients;
6. Evidence of altered dissolved oxygen dynamics (greater diel flux, lower minima, and higher maxima) pursuant or coincident with elevated algal/plant biomass;
7. Evidence of altered pH pursuant or coincident with elevated algal/plant biomass;

8. Evidence of altered invertebrate assemblage composition pursuant or coincident with elevated alga/plant biomass, altered dissolved oxygen, altered pH, altered assemblage composition.

Relevant supporting/refuting evidence from within the case (Wissahickon) was used to test these predictions and evidence from outside the case was used to place the case-specific evidence in context.

TESTING PREDICTIONS WITH EVIDENCE FROM THE CASE

The causal models were used to generate a series of predictions. These predictions and their verification provide the evidence used to establish the strength of the causal model and score the lines of evidence using the SI process scoring procedures (USEPA 2000). The verification process evaluates data from the case itself which provides the information necessary to test co-occurrence of stressors and responses, evidence of exposure to nutrients, stressor-response relationships, and symptomology. Similarly, evidence from outside the case (scientific literature) was used to develop the causal models as well as to anchor the analysis in the context of effect levels observed elsewhere, and that information was used to evaluate the evidentiary lines relying on evidence from elsewhere. These predictions are considered in order.

PREDICTION 1 - EVIDENCE OF INCREASED NUTRIENT CONCENTRATIONS IN THE STREAM ASSOCIATED WITH RUNOFF AND DISCHARGES, AS WELL AS BASEFLOW

For the first prediction, EPA evaluated nutrient concentrations in the Wissahickon. Data from 2005 were the most consistent across the majority of sites, so these data were used to calculate arithmetic annual and growing season (July- September) average nutrient concentrations. TN concentrations varied from <2 mg/L for a few sites in the southern portion of the watershed, to above 6.6 mg/L for several hot spots within the watershed (red symbols in Figure 3).

Similarly, TP concentrations varied across the watershed, with several sites exhibiting average annual TP concentrations less than 0.11 mg/L, but several sites exhibiting concentrations in excess of 1.3 mg/L, mostly located in the upper part of the watershed.

Similarly, seasonal average nutrient concentrations exhibit large variability across the watershed, with values of TN between 1 and 6.5 mg/L in the southern part of the watershed and several elevated TN locations in the middle and upper watershed with average seasonal concentrations above 8.3 mg/L (Figure 4). TP varied from average concentrations less than 0.3 mg/L for a few locations, but increasing to values above 0.8 – 1 mg/L in the southern watershed, to greater than 1 and 2 mg/L in the middle and upper portions of the watershed (Figure 4).

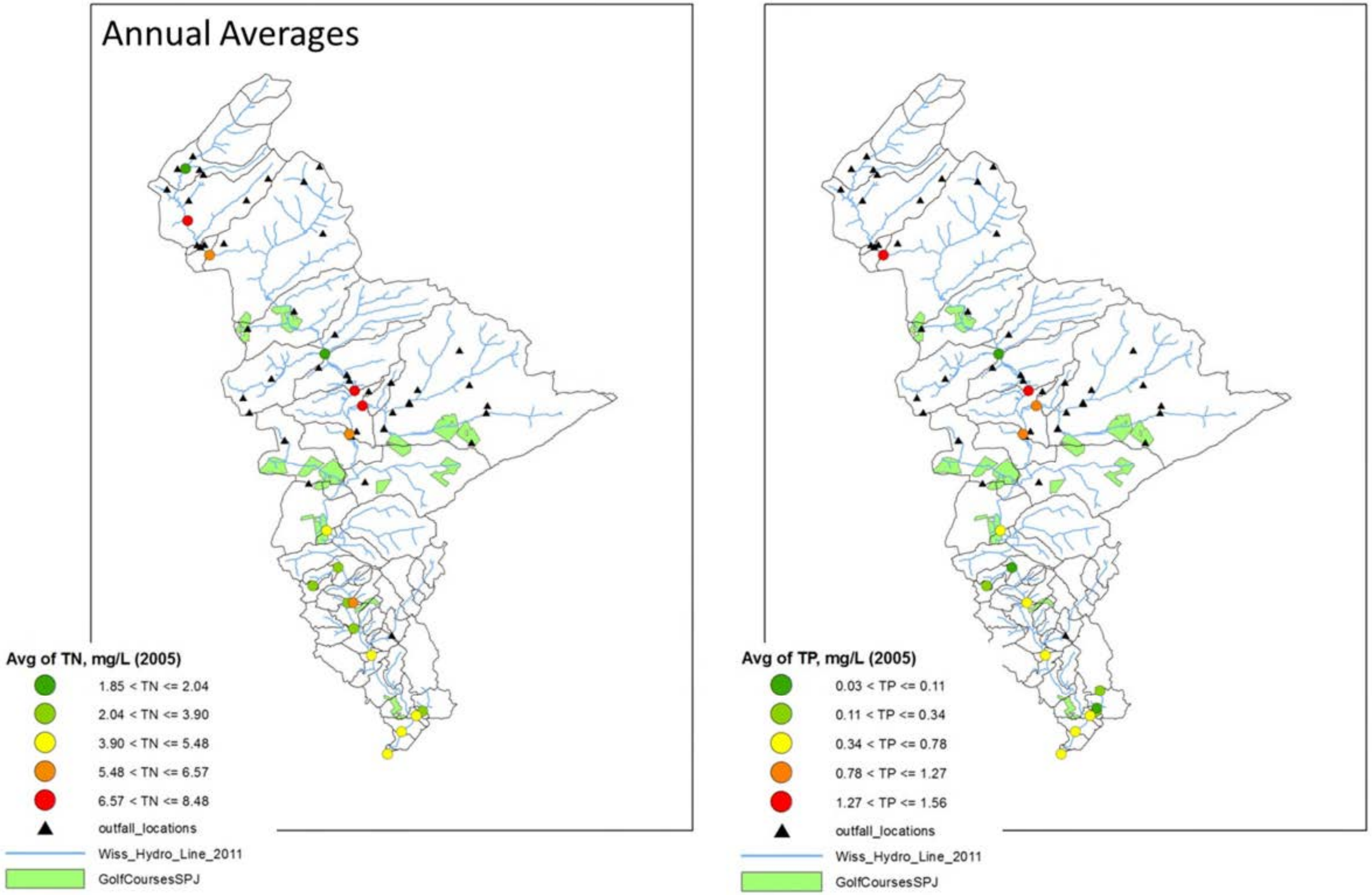


Figure 3 - Average annual TN and TP concentrations (mg/L) in the Wissahickon watershed.

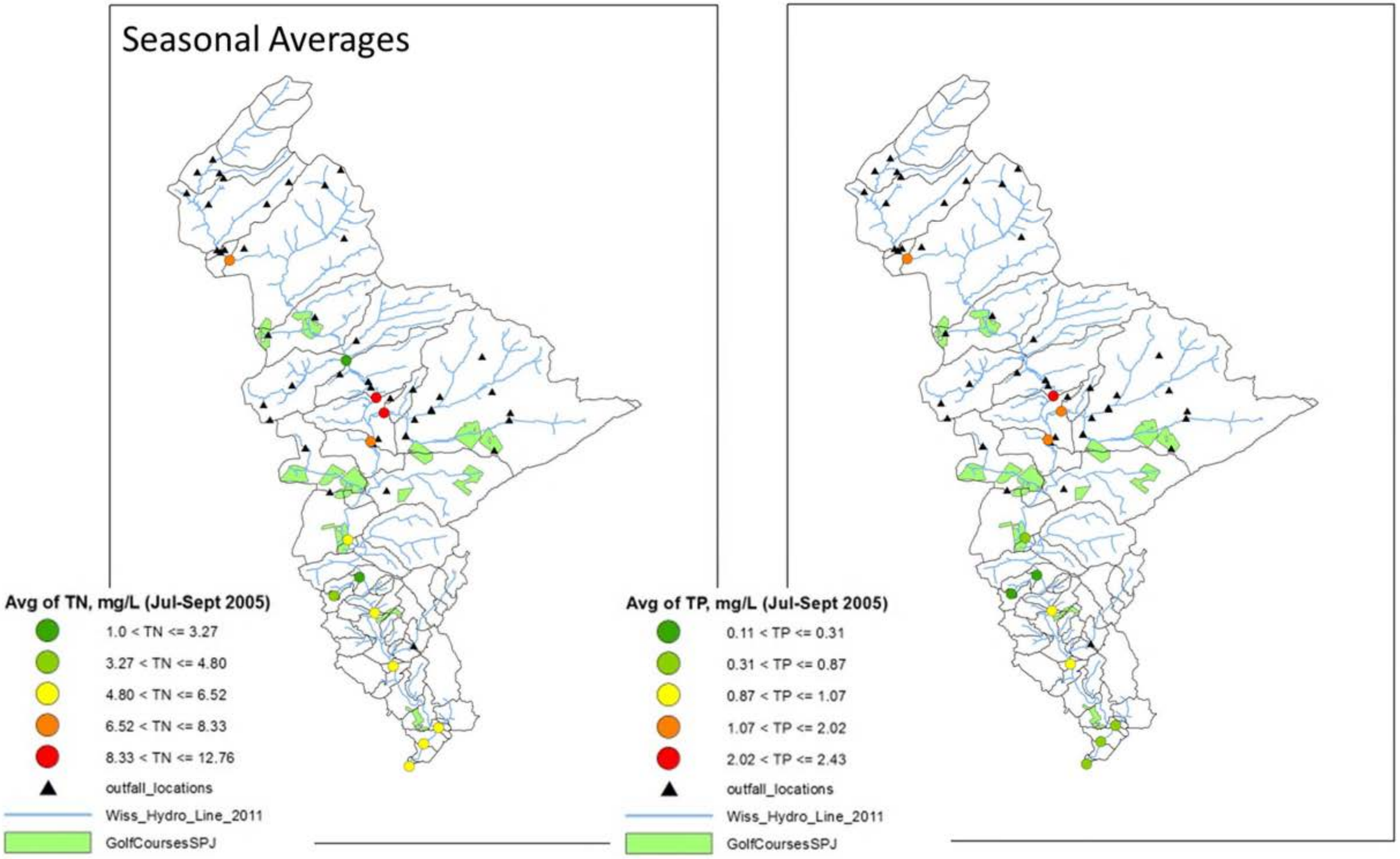


Figure 4 - Average seasonal TN and TP concentrations (mg/L) in the Wissahickon watershed.

To put these concentrations into regional context, TN concentration considered as indicative of eutrophic, elevated conditions in streams is 1.5 mg/L (Dodds et al. 1998), TN concentrations associated with excessive nuisance algal growths are below 3 mg/L , with some well below even 1 mg/L (Dodds and Welch 2000), and the EPA reference Piedmont TN concentrations were 0.70 mg/L (USEPA 2000) which was verified by reference site populations sampled as part of the large national, probabilistic Wadeable Streams Assessment (Herlihy and Sifneos 2008). Comparable values for TP include 0.075 mg/L as the eutrophic boundary (Dodds et al. 1998), values up to 0.4 mg/L but as low as 0.002 mg/L associated with nuisance conditions (Dodds and Welch 2000), and Piedmont EPA reference TP concentrations of 0.036 mg/L (USEPA 2000) up to 0.060 mg/L, identified as part of the Wadeable Streams Assessment (Herlihy and Sifneos 2008).

We also looked at groundwater sample data provided by PADEP's groundwater sampling program, and of the 14 stations sampled in the watershed, 7 exhibited NO_3 concentrations above 3 mg/L, 5 exhibited TP concentrations above 0.050 mg/L, and 6 of the stations exhibited upwards trends in at least one nitrogen parameter (NH_4, NO_2 or NO_3). Groundwater, too, appears elevated in nitrogen and phosphorus, consistent with predictions about enrichment influencing both surficial and groundwater resources.

Given this information, the first prediction is verified, concentrations of both N and P are substantially elevated in the Wissahickon; moreover, they appear enriched in several sites to concentrations that are consistently associated with eutrophic conditions, with a high likelihood of eliciting eutrophic responses including excess and nuisance algal biomass conditions.

PREDICTION 2 - EVIDENCE OF ALTERED N:P RATIO ASSOCIATED WITH ELEVATED NUTRIENT LOADS

The conceptual model also posits that enrichment in nutrients will likely also alter N:P ratios, which can alter competitive relationships among taxa that vary in their preferences for N and P concentrations and ratios. A survey of reference site N:P ratios in the Piedmont region yielded molar ratios of 184:1 (weight ratio of 83:1) which is above the traditional Redfield ratio of 16:1 (7:1 by weight) considered indicative of balanced growth and would indicate that the Piedmont is generally P limited (Paul and Zheng 2007), an observation also consistent with the general N:P ratios resulting from the Wadeable Streams Assessment (Herlihy and Sifneos 2008), which indicates reference site ratios of 25:1 (11:1 by weight), still indicative of P limitation.

N:P ratios in the Wissahickon were calculated based on paired N and P data for 2005 and seasonally during the growing season (July, August, September). These indicated a diverse range of ratios (Figure 5). Seasonally, only two sites had ratios, by weight, above 7, suggesting that most sites in this watershed were relatively N limited, in contrast to reference sites. As annual averages, four sites exhibited values above 9, but the majority still indicated more relative N limitation (greater P enrichment relative to N enrichment) when compared to reference sites. This alteration in nutrient ratios relative to reference is additional evidence for enrichment in these watersheds, consistent with the prediction. It is important to note that ratios, at such high nutrient concentrations, have reduced applicability in inferring true limitation since it is unlikely, at the nutrient concentrations observed, that either N or P are limiting primary producer growth. The relative ratios, however, inform conclusions about the degree of relative enrichment and while both nutrients in this watershed were enriched relative to reference in this watershed, the Wissahickon exhibits greater relative P enrichment.

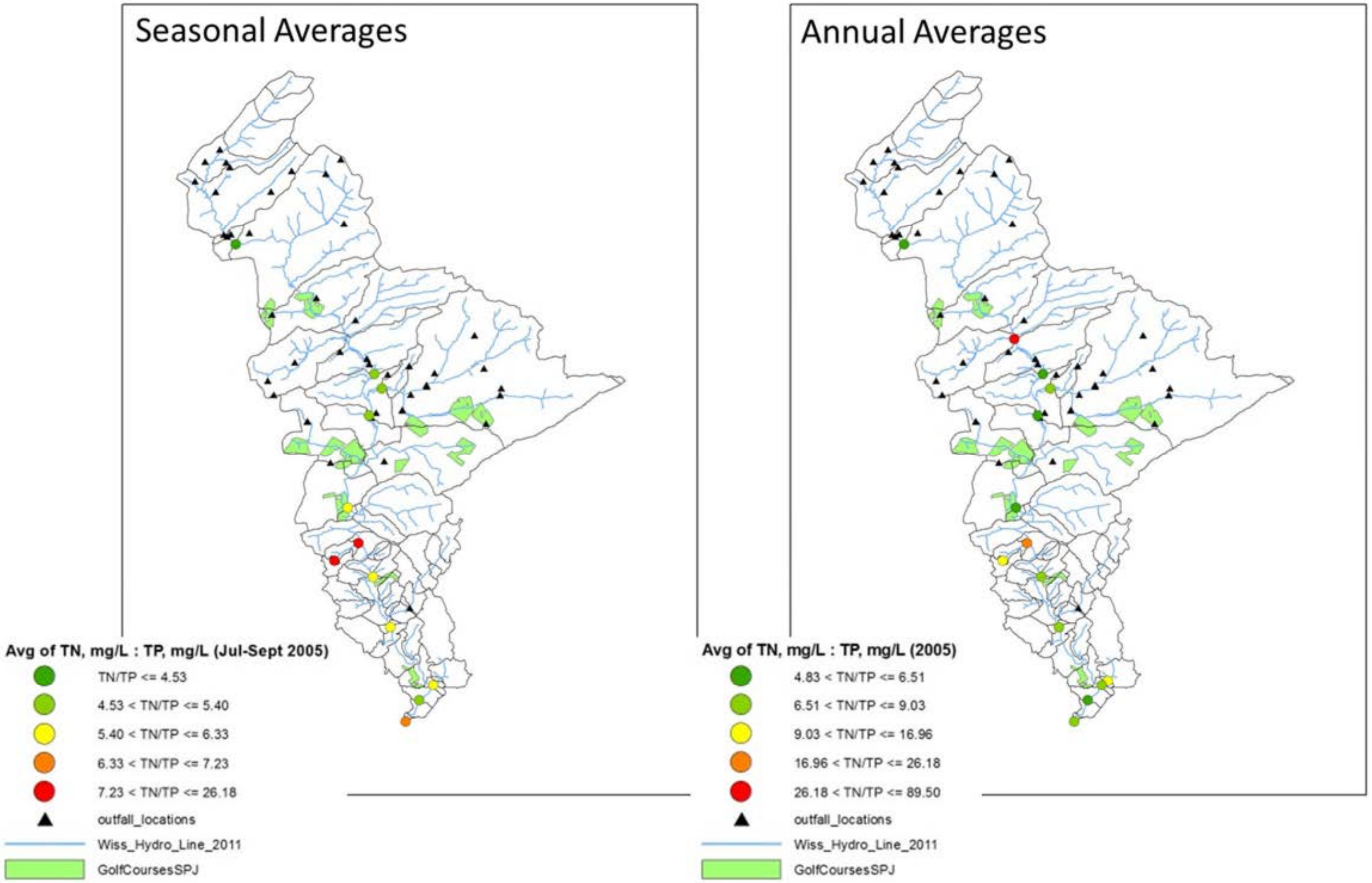


Figure 5 - Seasonal and Annual average N:P ratios in the Wissahickon.

PREDICTION 3 - EVIDENCE OF INCREASED ALGAL/PLANT BIOMASS AT LOCATIONS PURSUANT OR COINCIDENT WITH ELEVATED NUTRIENTS

The third prediction is evidence of increased algal/plant biomass. Chlorophyll a data from the stream were evaluated. There were periphyton and/or attached algal data available for 4 sites in 2005 for the data set available to EPA. Maximum chlorophyll a values in that dataset ranged from 166 to 461 mg m^{-2}, with an overall average maximum of 265 mg m^{-2}. For context, nuisance or excessive periphyton biomass is considered to occur when maximum chlorophyll exceeds 150 to 200 mg m^{-2} (Dodds et al. 1998, Suplee et al. 2009), although concentrations from 50 to 100 mg m^{-2} have also been suggested as indicative of nuisance concentrations (Horner et al. 1983, Nordin 1985, Welch et al. 1988). In addition, trophic boundaries have been estimated for streams and mean benthic chlorophyll a of 70 mg m^{-2} and maximum concentrations of 200 mg m^{-2} are considered the boundary between meso- and eutrophic streams . In the Wissahickon, 3 of the 4 site maximum values were greater than 200 mg m^{-2} indicating eutrophic nuisance conditions. A study by Carrick and Godwin (2006) characterized periphyton at nine sites in the Wissahickon. Chlorophyll a averaged 201 mg m^{-2} across the sites and ranged from 44 to 444 mg m^{-2} (Figure 6). Every site had average values greater than 50 mg m^{-2} and 6 of the 9 had average values greater than 200 mg m^{-2}, again consistent with eutrophic nuisance conditions based on the literature cited above.

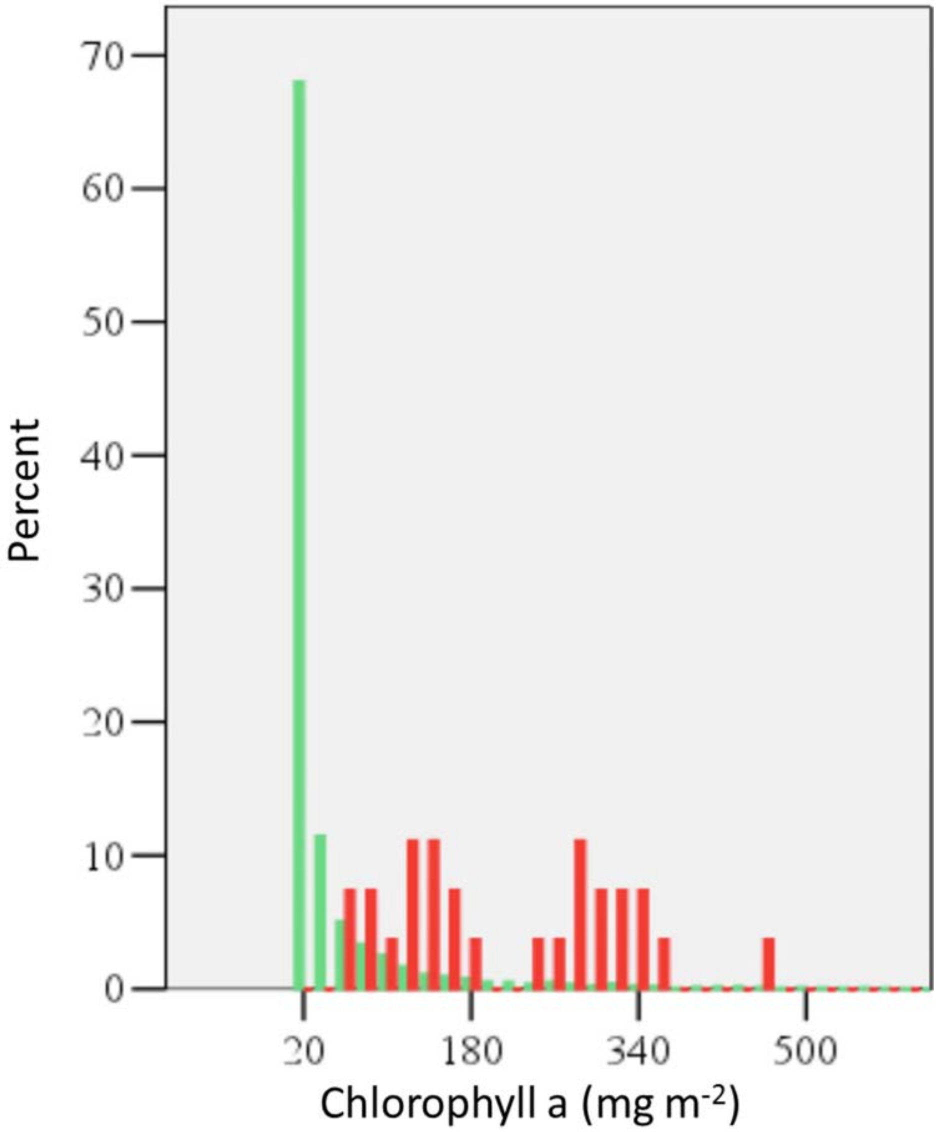


Figure 6 - Percent distribution of algal periphyton chlorophyll (mg/m2) measured from 410 streams throughout the world including North America and compared with measurements made in Wissahickon Creek. From Carrick and Godwin (2006). Green bars are worldwide data and red bars are Wissahickon observations.

Lastly, PADEP sampled algae at 10 sites in the Wissahickon in 1998 (PADEP 2002). During that sampling, average sample chlorophyll a ranged from 48 to 276 mg m^{-2}. Also, 5 of the 10 sites had average concentrations above 200 mg m^{-2}, 7 had concentrations above 100 mg m^{-2}, and all but one had concentrations above 50 mg m^{-2}.

Given this information, the third prediction was also verified, concentrations of chlorophyll a and, by association, algal biomass were substantially elevated in the Wissahickon; moreover, chlorophyll a values that are consistently considered nuisance levels were observed in each sampling effort.

PREDICTION 4 - EVIDENCE OF ALTERED PLANT/ALGAL ASSEMBLAGE STRUCTURE PURSUANT OR COINCIDENT WITH ELEVATED NUTRIENTS

The 4th prediction given the causal model developed is that nutrient enrichment should elicit a shift in algal species composition from species that are better competitors under low nutrients to those that are better under high nutrient conditions. Fortunately, quite a bit is known about nutrient preferences for diatoms species; so much so that the composition can often be used to infer the actual nutrient concentrations, and essentially diagnose whether nutrient enrichment is present or not, at a site (Ponader et al. 2005, 2008).

Diatom data collected in 1998 (West 2000) and in 2005 (Carrick and Godwin 2006) were sent to Dr. Lei Zheng, a professional phycologist with extensive experience in nutrient enrichment research. He evaluated the diatom species composition for what it indicated in terms of likely nutrient status in the Wissahickon.

With regards to the West (2000) data, Dr. Zheng commented that most stations are dominated by nutrient tolerant diatoms indicating elevated nutrient concentrations in the stream as a whole. The dominant taxa in all these sites, e.g., *Nitzschia inconspicua, Nitzschia amphibian, Navicula minima, Rhoicosphenia curvata, Melosira varians, Amphora pediculus, Synedra fasicualuata, Navicula gregaria, Navicula viriduna* var*. rostellata, Gomphonema parvulum, Cocconeis placentula* are all strong nutrient indicators. The only exception is *Navicula confervacea,* which is a high pH indicator. It was, according to Dr. Zheng, easy to conclude that these sites were nutrient enriched because of these specific taxa.

With regards to the diatom data collected at Wissahickon sites by Carrick and Godwin (2006), Dr. Zheng concluded that although the dominant species vary a little from the West (2000) report, some differences of which were due to different naming conventions for the same species, the most dominant species were, once again, pollution tolerant/nutrient preferring taxa. The dominant taxa included *Navicula lanceolata, Rhoicosphenia abbreviate* (*Rhoicosphenia curvata* in the first report), *Amphora pediculus, Melosira varians, Nitzschia inconspicua,* and even *Cocconeis placentula.* All are nutrient pollution tolerant taxa and streams heavily dominated by these taxa are considered nutrient enriched.

As a result of this analysis, the 4th prediction was also verified, the composition of flora in the stream reflected nutrient enrichment, with nutrient intolerant species being replaced by nutrient tolerant and even eutraphentic species. These results were also considered diagnostic, which is a more highly weighted element in causal analysis and SI because it is unequivocal evidence in support of the causal model.

PREDICTION 5 - EVIDENCE OF ALTERED SUSPENDED ORGANIC MATTER COMPOSITION AND ALTERED PERIPHYTON NUTRIENT RATIOS PURSUANT OR COINCIDENT WITH ELEVATED NUTRIENTS

Data on the composition of suspended organic matter composition was insufficient to test this prediction.

However, in the Carrick and Godwin (2006) report, they measured the nutrient content of periphyton. The N:P content of periphyton is a reflection of the nutrient environment and species internal cellular content will often reflect that of the surrounding water, especially as nutrient concentrations increase. The authors report the following regarding cellular nutrient ratios:

> "High concentrations of periphyton tissue nutrients (C, N, and P) were present in Wissahickon periphyton (Table 2). The average C:N:P ratio in the Wissahickon was approximately 8:1:1 (based on average concentrations), which is much lower than the Redfield ratio (106:16:1) that reflects balanced growth for algae. This low tissue ratio reflects the large stores of nutrients present in Wissahickon periphyton relative to periphyton [from] other streams (Kahlert 1998). Collectively, these results strongly suggest that periphyton were nutrient sufficient in Wissahickon Creek, and not likely growth limited by N or P. With tissue nutrient ratios so similar to water column values, the periphyton appear to have adapted to high ambient nutrients, which in this case is indicative of extreme enrichment (Cross et al. 2005)." (Carrick and Godwin 2006, p. 6)

This observation supported the prediction that nutrient enrichment alters the nutrient composition of algal and plant tissue, consistent with the causal model predictions.

PREDICTION 6 - EVIDENCE OF ALTERED DISSOLVED OXYGEN DYNAMICS (GREATER DIEL FLUX, LOWER MINIMA, AND HIGHER MAXIMA) PURSUANT OR COINCIDENT WITH ELEVATED ALGA/PLANT BIOMASS

It was fortunate in that the data available for this watershed included a number of long-term continuous dissolved oxygen (DO) deployments using recording sondes at several sites that allow insight into diel DO dynamics in the stream. For example, continuous DO was measured at two sites in 2005, one that exhibited relatively lower nutrients (WISS210_DO) and one that exhibited much higher nutrients (WISS500_DO). The data indicate a general trend consistent with the prediction, namely, that sites with higher nutrient concentrations exhibit greater diel flux (range of values), lower diel minima and higher diel maxima (Figure 7).

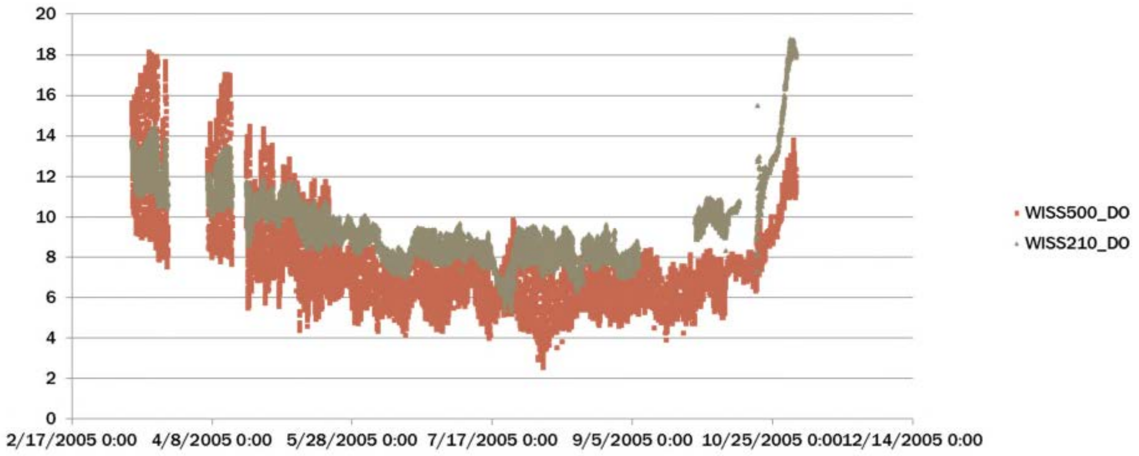


Figure 7 - Continuous dissolved oxygen (mg/L) from two sites in the Wissahickon watershed in 2005.

The DO data were analyzed and relationships between nitrogen and phosphorus and diel DO characteristics (average daily average, maximum, minimum, and range) were evaluated statistically using correlation and regression. Annual and seasonal average data were used. We had insufficient algal biomass data to relate to DO measurements directly.

Annual average nutrient concentrations were significantly negatively correlated with DO. As nutrient concentrations increased, DO responded in a significant manner – average DO decreased, daily maxima were lower, daily minima lower, but diel range was higher (Table 1). Essentially the same trends were noticed when analyzing the data as seasonal data as well (Table 2).

Table 1 - Response of seasonal average DO metrics to average seasonal nutrient concentrations in the Wissahickon. Arrows represent significant regressions (p<0.05) with red indicating negative and black positive. NS = not significant.

Seasonal Averages	Daily Avg DO	Daily Max DO	Daily Min DO	Daily DO Range
↑TKN	↓	↓	↓	NS
↑TP	↓	NS	↓	↑

The original hypothesis predicted higher DO maxima and possible higher daily average with the greater algal production observed in the Wissahickon due to nutrient enrichment. However, nutrients also stimulate heterotrophic production, which has only a net loss effect on dissolved oxygen in streams. So, any hypothesis related to DO would have to also predict the effect of nutrient enrichment on greater heterotrophic production and, therefore, respiration through decay of allochthonous and autochthonous carbon, which would be predicted to lower daily average DO, further lower daily minimum DO, and also daily maximum DO. It might also be predicted to affect diel range, but likely only shifting the range lower rather than curtailing the range entirely. When adding the effect of heterotrophic nutrient enrichment response to the prediction, the resulting data are even more consistent with the prediction of nutrient enrichment effects on the Wissahickon.

Table 2 – Response of annual average DO metrics to average annual nutrient concentrations in the Wissahickon. Arrows represent significant regressions ($p<0.05$) with red indicating negative and black positive. NS = not significant.

Annual Average	Daily Avg DO	Daily Max DO	Daily Min DO	Daily DO Range
↑TN	↓	↓	↓	↑
↑TP	↓	NS	↓	↑

Of particular interest is the extent of DO response relative to water quality standards. We analyzed the response of the 10th percentile seasonal DO observation to DO and, it too, was significantly negatively related to both TN and TP (Figure 8). Note that several values below 4 mg/L were observed.

Other evidence supports the observations noted here. Namely, the PADEP (2002) report similarly indicates that minimum DO concentrations at two sites were below the water quality standard and that one average DO concentration was below the standard. In addition, the two sites with minimum DO below the minimum DO standard also had the highest diel flux, consistent with predictions of nutrients enriching production and leading to the effects predicted in the causal model.

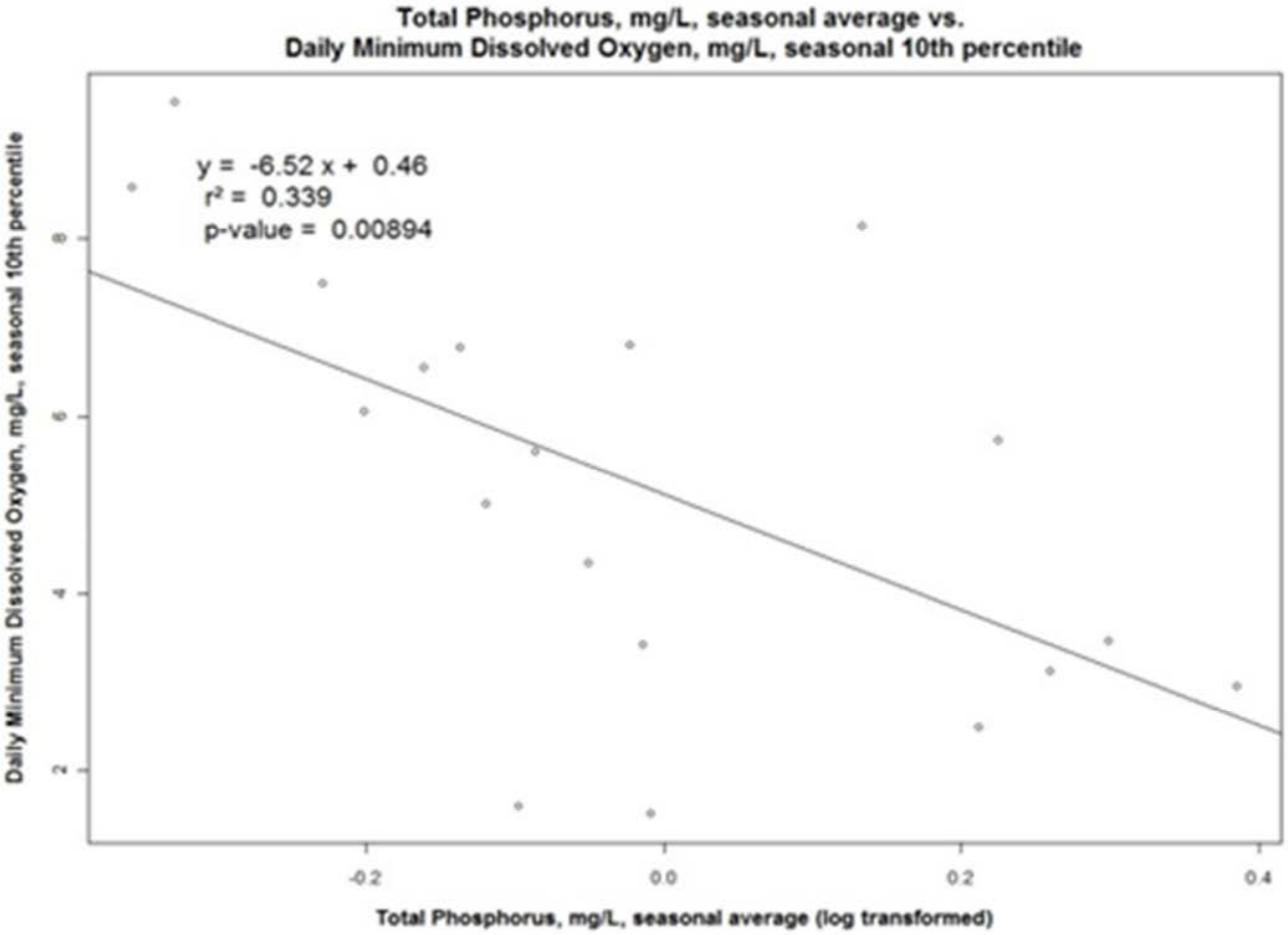
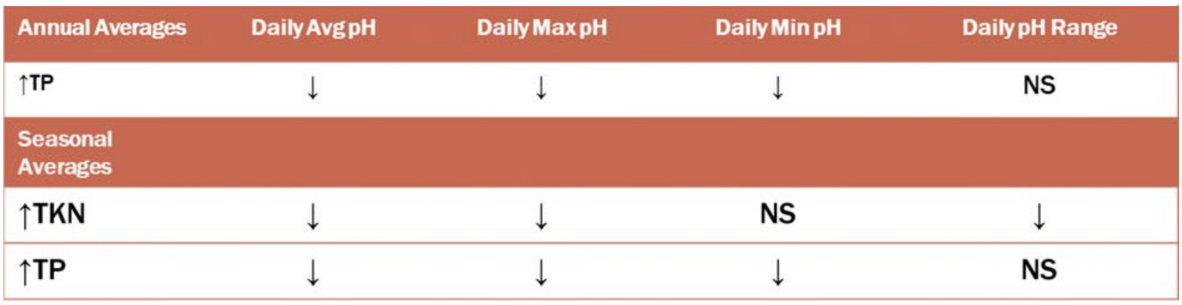


Figure 8 – Relationship of seasonal average TP (mg/L) do the 10th percentile observed diel DO concentration (mg/L) in the Wissahickon.

PREDICTION 7 - EVIDENCE OF ALTERED PH PURSUANT OR COINCIDENT WITH ELEVATED ALGAL/PLANT BIOMASS

The Wissahickon appears to be a well buffered system as pH ranges were not large (essentially all under 1.5 pH units). Once again, there was insufficient paired periphyton algal biomass and water chemistry data to relate to pH measurements directly. However, general pH metric trends were consistent with a model of nutrient enrichment of heterotrophic respiration and a moderated effect of algal primary production (Table 3). Annual and seasonal average, maximum, and daily minimum pH declined with nutrient concentrations, especially TP, consistent with increased respiration decreasing DO and increasing dissolved inorganic carbon concentrations that reduce pH overall.

Table 3 - Response of annual and seasonal pH metrics to average seasonal and annual nutrient concentrations in the Wissahickon. Arrows represent significant regressions ($p<0.05$) with red indicating negative and black positive. NS = not significant.



Annual Averages	Daily Avg pH	Daily Max pH	Daily Min pH	Daily pH Range
↑TP	↓	↓	↓	NS
Seasonal Averages				
↑TKN	↓	↓	NS	↓
↑TP	↓	↓	↓	NS

It is worth noting that pH violations (of the upper end of the PADEP pH standard of 6 to 9) were recorded in 2 and 3% of the observations at two sites in the Wissahickon, coincident with locations exhibiting excessive algal biomass and large DO fluctuations. The pH violations were attributed to nutrient enriched metabolism (PWD 2007) since other major sources of pH fluctuation were absent.

PREDICTION 8 - EVIDENCE OF ALTERED INVERTEBRATE ASSEMBLAGE COMPOSITION PURSUANT OR COINCIDENT WITH ELEVATED ALGA/PLANT BIOMASS, ALTERED DISSOLVED OXYGEN, ALTERED PH, ALTERED ASSEMBLAGE COMPOSITION

The last prediction relates to the biological condition and evidence of its relationship to the proximate stressors of algal biomass, DO, pH, and altered assemblage composition. Given the range of stressors and the nearly uniform presence of impacts across the watershed and cumulatively downstream, there was little expectation that a clear signal with these specific endpoints would manifest itself in the invertebrate assemblage. Indeed, the nearly uniform severely impacted biological conditions across the watershed meant the response signal was limited (PWD 2007). However, consistent with predictions, there was a significant decline in Total Richness and an increase in Hilsenhoff Biotic Index (HBI) scores (higher values indicate fewer sensitive species) with chlorophyll a (Figure 9). Response to DO metrics were weak, however HBI scores also showed a significant decline with average daily pH, suggesting a loss in sensitive species as pH declined (Figure 10).

Outside of the case, it is worth remembering that in the nutrient stressor-response models developed for Piedmont streams to identify nutrient endpoints, significant declines in invertebrate condition were identified with increasing nutrients across the Piedmont region (Paul and Zheng 2007) even after correcting for effects of co-occurring stressors (Paul et al. 2011). That document also noted, however, that heavily urban streams exposed to a range of stressors, showed a more traditional wedge shaped response to nutrients, where it appears nutrients constrain the upper end of biological condition, but the range of co-occurring stressors make simple relationships difficult. In addition, a report summarizing a large sampling effort of chemical and biological characteristics across the watershed also attributed biological impacts to large DO and pH impacts associated with algal biomass (PWD 2007). Also, similar negative impacts of nutrients on invertebrate assemblages were noted from similarly situated Piedmont streams in adjacent watershed associated with similar stressor sources (Rief 2002a, 2002b).

In light of the responses observed, the last prediction has some support but the data set was sparse and a variety of confounding factors limited an unequivocal demonstration. More information on macroinvertebrate assemblages across a larger condition gradient in the watershed would improve confidence in these conclusions.

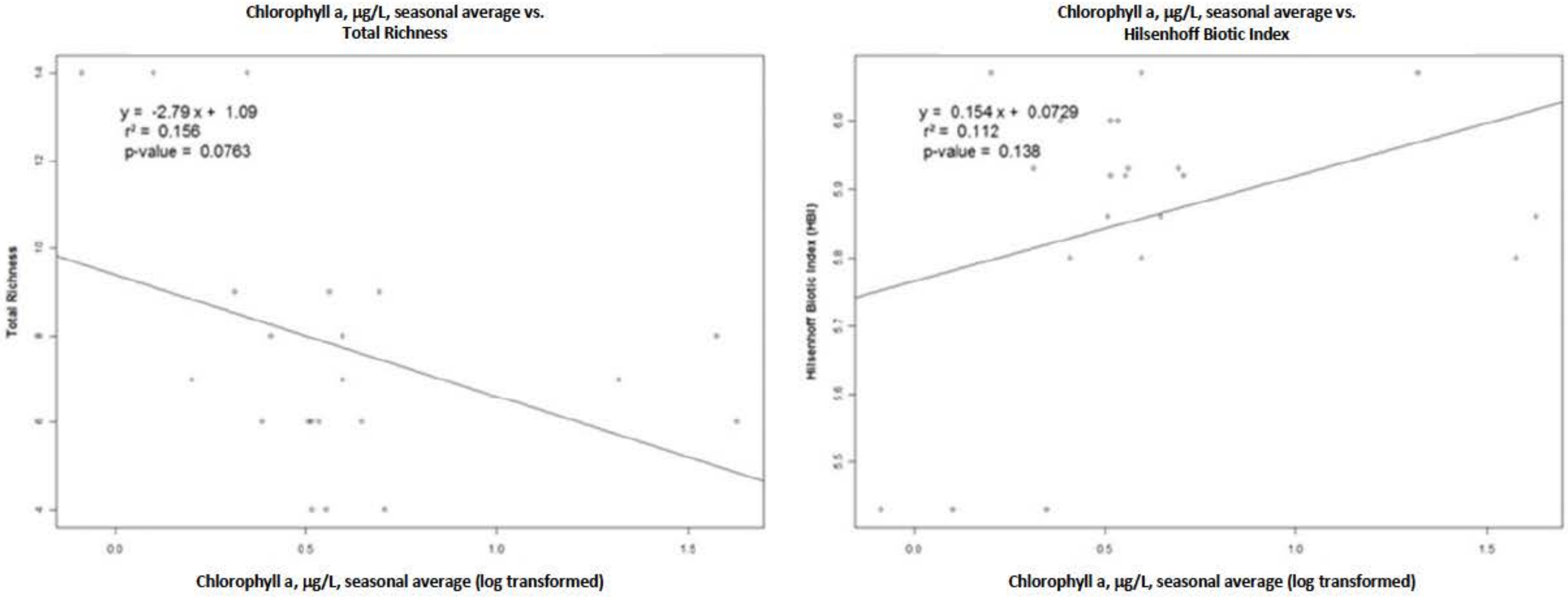


Figure 9 – Response of total invertebrate richness and Hilsenhoff Biotic Index to chlorophyll a in the Wissahickon.

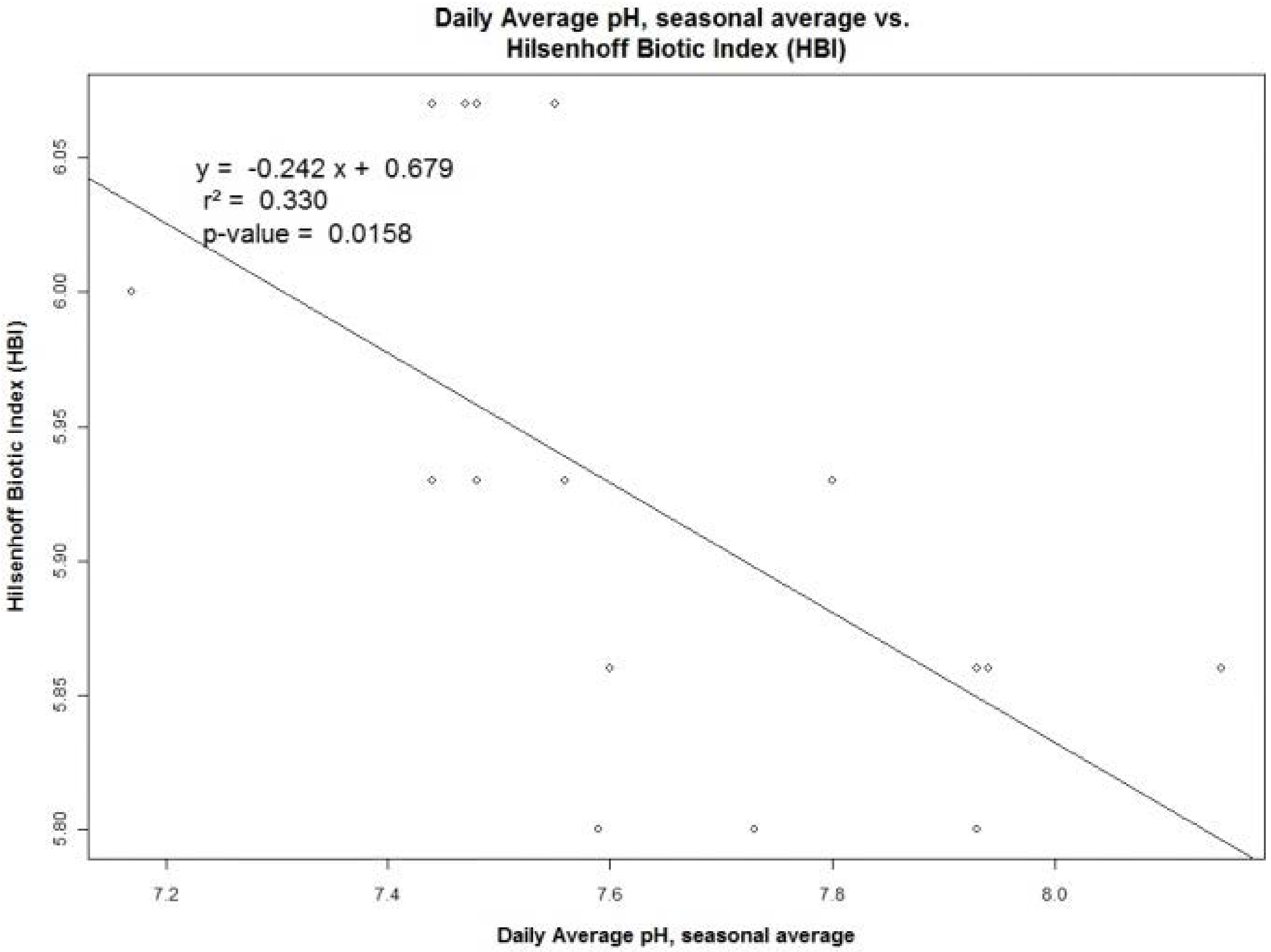


Figure 10 – Response of Hilsenhoff Biotic Index to seasonal average pH in the Wissahickon.

EVIDENTIARY SUMMARY AND CONCLUSIONS

After evaluating the existing evidence, 6 of the 8 predictions from the causal model were substantially supported (Table 4), the fifth and last one were limited by insufficient data and the last one limited by confounding co-occurring stressors to make certain conclusions, even though several relationships appeared to provide support.

Table 4 – Summary of evaluation of the conceptual model predictions

	Prediction	Evidence Supporting
1	Increased nutrient concentrations in the stream associated with runoff and discharges, as well as baseflow	Yes
2	Evidence of altered N:P ratio associated with elevated nutrient loads	Yes
3	Evidence of increased algal/plant biomass at locations pursuant or coincident with elevated nutrients	Yes
4	Evidence of altered plant/algal assemblage structure pursuant or coincident with elevated nutrients	Yes
5	Evidence of altered suspended organic matter composition pursuant or coincident with elevated nutrients	Limited
6	Evidence of altered dissolved oxygen dynamics (greater diel flux, lower minima, and higher maxima) pursuant or coincident with elevated alga/plant biomass	Yes
7	Evidence of altered pH pursuant or coincident with elevated alga/plant biomass	Yes
8	Evidence of altered invertebrate assemblage composition pursuant or coincident with elevated alga/plant biomass, altered dissolved oxygen, altered pH, altered assemblage composition	Limited

This information was then used to score the SI process tables (USEPA 2001, 2012). All the evidentiary lines from within the case itself were supported (Table 5). Nutrient responses co-occurred in space with the nutrient exposure, which was fairly homogenous across the watershed. Data described above supported the consistency of this co-occurrence for nearly every step in the pathways. There was clear evidence of exposure to nutrients as evidenced by the algal assemblage data, the consistency of the causal path predictions, and the lack of sensitive invertebrate and fish taxa in these streams. The causal model prediction exercise conducted above provided evidence that the steps in the pathway were present. This same analysis included stressor-response evidence from the case to support much of the prediction evaluation, which was consistent with predictions. The exposure manipulation and laboratory test lines were not applicable for the reasons given in the Table and the temporal sequence line was unknown because of a lack of temporal data related to the stressor. Nutrients were elevated throughout the period of data availability. Several predictions from the causal model were made and verified with data from the case. Finally, there was diagnostic evidence of nutrient exposure in the algal assemblage composition which is a strong line of evidence of the presence of nutrients in stressful levels in the stream.

Table 5 – Summary table of scores for types of evidence that use data from the case.

Type of Evidence	Description	Wissahickon
Spatial/Temporal Co-Occurrence	The biological effect must be observed where and when the cause is observed, and must not be observed where and when the cause is absent.	+ (Effects are always present where the stressor is present)
Evidence of Exposure or Biological Mechanism	Measurements of the biota show that relevant exposure to the cause has occurred, or that other biological mechanisms linking the cause to the effect have occurred.	++ (Algal diagnostic nutrient effect present; inverts only exhibit tolerant taxa)
Causal Pathway	Steps in the pathways linking sources to the cause can serve as supplementary or surrogate indicators that the cause and the biological effect are likely to have co-occurred.	+ (All steps in the causal pathway are supported; but limited data for the final step for reasons given above)
Stressor-Response Relationships from the Field	As exposure to the cause increases, intensity or frequency of the biological effect increases; as exposure to the cause decreases, intensity or frequency of the biological effect decreases.	+ (Strong linkage present for most steps except last step in causal model, which is confounded)
Manipulation of Exposure	Field experiments or management actions that increase or decrease exposure to a cause must increase or decrease the biological effect.	NA (No such manipulations with data exist for the case)
Laboratory Tests of Site Media	Controlled exposure in laboratory tests to causes (usually toxic substances) present in site media should induce biological effects consistent with the effects observed in the field.	NA (No lab tests are applicable)
Temporal Sequence	The cause must precede the biological effect.	0 (No temporal data, i.e no data preceding the cause)
Verified Predictions	Knowledge of a cause's mode of action permits prediction and subsequent confirmation of previously unobserved effects.	+++ (Several predictions of the causal model were made and verified)
Symptoms	Biological measurements (often at lower levels of biological organization than the effect) can be characteristic of one or a few specific causes.	D (Diagnostic evidence of nutrient enrichment is indicated in algal assemblage composition)

Reasoning using evidence from outside the case was also considered (Table 6). The conceptual model and verification exercise was based on scientific theory and known ecological principles, so the cause-effect models were inherently consistent with theory. The effect levels observed in the case were well above those observed to have effects in lab experiments, field experiments, and field studies, as evidenced by the literature described above. Mechanistic models of nearby Indian Creek described in other reports (Paul et al. 2011), indicate consistency with this line of evidence. The exposure manipulation line has no relevance here because of limited known data on nutrient reductions to levels considered to reduce effects in Piedmont streams. There is ample evidence that analogous stressors known to stimulate productivity in other systems (N, P, Fe, Si, etc.) have caused similar responses to those observed in response to N and P in the Wissahickon. Therefore, the line of evidence related to the effect of analogous stressors was supported.

Table 6 - Summary table of scores for types of evidence that use data from elsewhere.

Type of Evidence	Description	Wissahickon
Mechanistically probable cause	The relationship between the cause and biological effect must be consistent with known principles of biology, chemistry and physics, as well as properties of the affected organisms and the receiving environment.	+ (The cause-effect relationship proposed is consistent with theory)
Stressor-Response Relationships from lab studied	Within the case, the cause must be at levels associated with related biological effects in laboratory studies.	++ (The levels of nutrients are above those observed to elicit responses in experiments)
Stressor-Response Relationships from field studies	At the impaired sites, the cause must be at levels sufficient to cause similar biological effects in other field studies.	++ (The levels of nutrients are above those observed to elicit responses in other field studies)
Stressor-Response Relationships from ecological mechanistic models	Within the case, the cause must be at levels associated with effects in mathematical models simulating ecological processes.	+ (Responses are consistent with mechanistic simulation models of nutrients in similar and even nearby streams)
Manipulation of Exposure at other sites	At similarly impacted locations outside the case sites, field experiments or management actions that increase or decrease exposure to a cause must increase or decrease the biological effect.	NA (Limited nutrient reduction efforts have been undertaken in comparable streams)
Analagous Stressors	Agents similar to the causal agent at the impaired site should lead to similar effects at other sites.	++ (Nutrients cause very similar responses at other sites)

Putting these two previous evidentiary tables together, the lines of evidence were scored (Table 7). Each line of evidence was consistent with nutrients as a cause of stress contributing to biological impairment in the Wissahickon, as evidenced by altered algal and invertebrate assemblages. This means that there should be high confidence that nutrients are contributing to biological impacts in this stream. Invertebrate response data were limited, and substantive and defensible reasoning was given explaining why the response was not stronger.

Table 7 - Summary table of scores for evaluating multiple lines of evidence.

Type of Evidence	Description	Wissahickon
Consistency of Evidence	Confidence in the argument for or against a candidate cause is increased when many types of evidence consistently support or weaken it.	+++ (All evidentiary lines convincingly support the case for the cause)
Explanation of Evidence	Confidence in the argument for a candidate cause is increased when a post hoc mechanistic, conceptual, or mathematical model reasonably explains any inconsistent evidence.	++ (Invertebrate responses exhibited a weaker linkage, but the weaker responses can be defensibly explained based on the uniformity of impact and the confounding effect of co-occurring stressors with nutrients. Moreover, the limited amount of invertebrate data and reduced range of potential response limit the confidence with which any conclusion can be drawn with this evidence.)

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APPENDIX 1 – DATA SUMMARY

To evaluate nutrients in the Wissahickon Creek watershed, we obtained water quality data from the USGS National Water Information System (NWIS), Pennsylvania's Surface Water Quality Network (WQN), and Philadelphia Water Department's water quality data (WS_LIMS_Baseline, WS_LIMS_Post2005, WS_LIMS_PWD, and WS_Sonde). We obtained benthic macroinvertebrate index data (EPT Richness, Total Richness, and Hilsenhoff Biotic Index) for 34 sites from PWD (WS) and PADEP (WQN) as well.

Table 1 shows the number of sites with water quality data from each of the data sources above.

Table 2 presents the number of years for which each site was sampled, the year sampling started, and the most recent year when it was sampled.

Table 3 presents the types of analytical data available from each source including the analyte, the units presented in the original data, and a conversion factor, if required. Conversion factors were required when data from multiple sources were reported in different units of measure. To combine these data, standard unit conversions were applied. Additionally, nutrient species may be measured and reported as the species in its entirety (e.g., nitrate as NO_3) or as the base nutrient (nitrate as N). We converted values reported as the species to values reported as the base nutrient using molecular/atomic weight ratios. This allowed combining similar data into a common data set, as well as combining species to obtain total nutrient concentrations. For example, total nitrogen could be calculated from the component species ($TN = TKN + NO_2 + NO_3$), if all the components were reported as N, not as the species.

Table 4 presents the diatom data available from each source.

Table 5 presents the sites for which both nutrient data and benthic macroinvertebrate data were available in the same year.

Table 1. Number of Sites from Each Data Source

Data Source	Number of Sites	
USGS NWIS	6	
PA WQN	2	
WS LIMS (Baseline)	33	55 unique sites
WS LIMS (Post 2005)	22	
WS LIMS (PWD)	25	
Sonde Data	10	

Table 2. Available Data

Data Source	Site	Number of Years Sampled	First Year Sampled	Most Recent Year Sampled
USGS NWIS	WISS125	30	1959	2011
USGS NWIS	WISS210	4	1967	1970
USGS NWIS	WISS500	17	1962	2011

USGS NWIS	WS1475	4	1972	1976
USGS NWIS	WS209	1	1999	1999
USGS NWIS	WS622	9	1967	1979
PA WQN	WQN0115	10	2002	2011
PA WQN	WQN0193	10	2002	2011
WS LIMS & Sonde	BELL150	1	2001	2001
WS LIMS & Sonde	BELL400	3	2005	2007
WS LIMS & Sonde	BELL850	1	2005	2005
WS LIMS & Sonde	BELLEIN	2	2003	2004
WS LIMS & Sonde	BELLEOUT	2	2003	2004
WS LIMS & Sonde	CATH250	1	2005	2005
WS LIMS & Sonde	CRES150	1	2001	2001
WS LIMS & Sonde	CRES200	1	2006	2006
WS LIMS & Sonde	CRES250	1	2001	2001
WS LIMS & Sonde	CRES350	1	2001	2001
WS LIMS & Sonde	CRES500	1	2001	2001
WS LIMS & Sonde	CRES600	1	2001	2001
WS LIMS & Sonde	CRES700	1	2001	2001
WS LIMS & Sonde	CRES800	1	2001	2001
WS LIMS & Sonde	LORR175	1	2004	2004
WS LIMS & Sonde	MONO100	1	2008	2008
WS LIMS & Sonde	MONO200	3	2005	2007
WS LIMS & Sonde	MONO250	10	1999	2010
WS LIMS & Sonde	MONO840	5	2001	2005
WS LIMS & Sonde	PROP200	1	2005	2005
WS LIMS & Sonde	RADI750	1	2005	2005
WS LIMS & Sonde	SAND150	4	2004	2008
WS LIMS & Sonde	SAND200	2	2001	2003
WS LIMS & Sonde	WISE150	1	2001	2001
WS LIMS & Sonde	WISE250	2	2005	2006
WS LIMS & Sonde	WISS125	11	2001	2011
WS LIMS & Sonde	WISS130	5	1999	2011
WS LIMS & Sonde	WISS135	1	2009	2009
WS LIMS & Sonde	WISS140	8	2002	2010
WS LIMS & Sonde	WISS150	5	1999	2005
WS LIMS & Sonde	WISS160	3	1999	2001
WS LIMS & Sonde	WISS200	1	2001	2001
WS LIMS & Sonde	WISS210	1	2005	2005
WS LIMS & Sonde	WISS215	2	2006	2010
WS LIMS & Sonde	WISS300	2	2001	2005
WS LIMS & Sonde	WISS320	1	2006	2006
WS LIMS & Sonde	WISS375	1	2001	2001
WS LIMS & Sonde	WISS400	1	2003	2003
WS LIMS & Sonde	WISS410	6	2004	2010
WS LIMS & Sonde	WISS425	1	2006	2006
WS LIMS & Sonde	WISS430	2	2005	2007
WS LIMS & Sonde	WISS440	1	2005	2005
WS LIMS & Sonde	WISS450	3	2005	2007
WS LIMS & Sonde	WISS500	9	2001	2011
WS LIMS & Sonde	WISS550	6	2001	2008
WS LIMS & Sonde	WISS600	1	2003	2003
WS LIMS & Sonde	WISS625	1	2001	2001
WS LIMS & Sonde	WISS675	1	2001	2001
WS LIMS & Sonde	WISS700	7	2001	2010
WS LIMS & Sonde	WISS713	1	2006	2006

WS LIMS & Sonde	WISS720	2	2005	2006
WS LIMS & Sonde	WISS735	1	2005	2005
WS LIMS & Sonde	WISS750	1	2003	2003
WS LIMS & Sonde	WISS790	1	2005	2005
WS LIMS & Sonde	WISS800	1	2001	2001

Table 3. Water Quality Data Available by Data Source (analytical data may not be available for all sites from given source)

Analyte	Units	Conversion Factor	USGS	WQN	WS LIMS Baseline	WS LIMS Post-2005	WS LIMS (PWD)	WS Sonde
Physical Parameters								
Color	PCU	1	X					
Discharge	cfs	1	X					
Discharge, instantaneous	cfs	1	X					
Dissolved oxygen	mg/L	1	X	X	X	X	X	X
Dissolved oxygen (% saturation)	%	1	X					
Flow	cfs	1		X				
pH	none	1	X	X	X	X	X	X
pH test temperature	deg C	1				X		
pH, filtered	none	1	X					
Specific conductivity	uS/cm	1	X	X	X	X	X	X
Temperature, air	deg C	1	X					
Temperature, water	deg C	1	X	X	X	X	X	X
Turbidity	NTU	1	X		X	X		X
Turbidity	JTU	1	X					
Nutrients								
Ammonia, dissolved (as N)	mg/L	1	X					
Ammonia, dissolved (as NH4)	mg/L	0.775	X					
Ammonium, total	mg/kg	1			X			
Ammonium, total (as N)	mg/L	1	X	X	X	X	X	
Ammonium, total (as NH4)	mg/L	0.775	X					
Nitrate, dissolved (as N)	mg/L	1	X					
Nitrate, total	mg/kg	1			X			
Nitrate, total	mg/kg (wet wt)	1			X			
Nitrate, total (as N)	mg/L	1	X	X	X	X	X	

Nitrate, total (as NO3)	mg/L	0.226	X					
Nitrite + Nitrate, dissolved (as N)	mg/L	1	X					
Nitrite + Nitrite, total (as N)	mg/L	1	X					
Nitrite, dissolved (as N)	mg/L	1	X					
Nitrite, dissolved (as NO2)	mg/L	0.304	X					
Nitrite, dissolved (as NO3)	mg/L	0.226	X					
Nitrite, total	mg/L	1	X	X	X	X	X	
Nitrogen, dissolved (total of all forms, as N)	mg/L	1	X					
Nitrogen, total	mg/L	1	X	X				
Organic nitrogen, dissolved	mg/L	1	X					
Organic nitrogen, total	mg/L	1	X					
Total Kjeldahl Nitrogen, dissolved	mg/L	1	X					
Total Kjeldahl Nitrogen, total	mg/kg	1			X			
Total Kjeldahl Nitrogen, total	mg/kg (wet wt)	1			X			
Total Kjeldahl Nitrogen, total	mg/L	1	X		X	X	X	
ortho-Phosphate	mg/kg	1			X			
ortho-Phosphate (as P)	mg/L	1		X	X	X	X	
ortho-Phosphate, dissolved (as P)	mg/L	1	X					
ortho-Phosphate, dissolved (as PO4)	mg/L	0.327	X					
ortho-Phosphate, total	mg/L	1	X					
Phosphate, dissolved (as P)	mg/L	1	X					
Phosphate, dissolved (as P)	mg/L	0.327	X					
Phosphate, total	mg/L	1		X				
Phosphorus, total (as P)	mg/L	1	X		X	X	X	
Algal measures								
Ash-free dry mass	g/m2	1	X					
Ash-free dry mass (periphyton)	g/m2	1	X					
Chlorophyll a (area)	mg/m2	1	X				X	
Chlorophyll a (volume)	ug/L	1			X		X	
Chlorophyll, total	ug/L	1			X			

Geosmin	ng/L	1			X	X	X	
Periphyton dry mass	g/m2	1	X					
Pheophytin	mg/m2	1	X					
Other Water Quality Indicators								
Acid neutralizing capacity (field)	mg/L	1	X					
Acid neutralizing capacity (lab)	mg/L	1	X					
Biological oxygen demand, 5-day	mg/L	1	X		X		X	
Chemical oxygen demand (high)	mg/L	1	X					
Chemical oxygen demand (low)	mg/L	1	X					
Chloride	mg/L	1	X					
E. coli	# Colonies/100 mL	1	X					
Enterococci	# Colonies/100 mL	1	X					
Fecal coliform	# Colonies/100 mL	1	X					
Fluoride, dissolved	mg/L	1	X					
Fluoride, total	mg/L	1	X					
Organic carbon, dissolved	mg/L	1	X					
Organic carbon, total	mg/L	1	X					
Solids, total	mg/L	1			X	X	X	
Solids, total suspended	mg/L	1	X	X	X	X	X	
Sulfate	mg/L	1	X					
Total coliforms	# Colonies/100 mL	1	X					
Total coliforms	most probably number	1	X					
Total dissolved solids	mg/L	1		X	X	X	X	

Table 4. Diatom Data Available by Data Source (data may not be available for all sites from given source)

Analyte	**Units**	**USGS**	**WQN**	**WS LIMS Baseline**	**WS LIMS Post-2005**	**WS LIMS (PWD)**	**WS Sonde**
Ankistrodesmus	# Organisms/mL			X	X		

Asterionella	# Organisms/mL			X			
Cocconeis	# Organisms/mL			X			
Coelastrum	# Organisms/mL			X			
Cyclotella	# Organisms/mL			X			
Cymbella	# Organisms/mL			X			
Diatoma	# Organisms/mL			X			
Fragillaria	# Organisms/mL			X			
Golenkinia	# Organisms/mL			X			
Lyngbya	# Organisms/mL			X			
Melosira	# Organisms/mL			X	X		
Meridion	# Organisms/mL			X			
Navicula	# Organisms/mL			X	X		
Pandorina	# Organisms/mL			X			
Pediastrum	# Organisms/mL			X			
Phytoconis	# Organisms/mL			X			
Pleurosigma	# Organisms/mL			X			
Rhoicosphenia	# Organisms/mL			X			
Scenedesmus	# Organisms/mL			X			
Skeletonema	# Organisms/mL			X			
Surirella	# Organisms/mL			X			
Synedra	# Organisms/mL			X	X		
Synura	# Organisms/mL			X			
Tabellaria	# Organisms/mL			X			

Table 5. Sites for which water quality data and benthic data were available for the same year.

Data Source	Site
WS	BELL400
	BELL850
	CATH250
	MONO250
	PROP200
	WISS125
	WISS150
	WISS210
	WISS300
	WISS440
	WISS500
	WISS550
	WISS700
WQN	WQN0115
	WQN0193