# A Before-After-Control-Impact Analysis for Cornell University's Lake Source Cooling Facility

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## 1. Objective

The primary objective of this report is to determine if levels of three water quality parameters (chlorophyll **a**, total phosphorus, turbidity) have shown statistically significant changes in the southern portion of Cayuga Lake coincident in time with start-up of Cornell's Lake Source Cooling (LSC) facility. Statistical determinations are made based on a Before-After-Control-Impact (BACI) design applied to in-lake monitoring data collected over the 1998 – 2005 interval.

## 2. Cayuga Lake and the Lake Source Cooling Facility

Cayuga Lake is the fourth easternmost of the New York Finger Lakes (Figure 1); it has the largest surface area (172 $km^2$) and the second largest volume (9.38 x $10^9$ $m^3$) of this system of lakes (Schaffner and Oglesby 1978). The lake is long (61.4 km along its major axis) and narrow, extending along a north/south axis (Figure 1). Its stratification regime is warm monomictic, stratifying strongly in summer, but complete ice cover has only rarely occurred (Oglesby 1978). The lake's large hypolimnion remains cold (e.g., < 5 ºC) through the summer. The watershed area of this alkaline hardwater lake is ~ 1150 $km^2$ (Oglesby 1978). Water exits the basin through a single outlet at the northern end of the lake. The long-term average flushing rate of the lake is slow, about 0.08 $y^{-1}$ (Oglesby 1978, Effler et al. 1989). The City of Ithaca and Cornell University are located at the southeastern end of Cayuga Lake.

Cayuga Lake is generally considered to be mesotrophic (e.g., Oglesby 1978). This position is supported by available long-term measurements of the trophic state indicators of total phosphorus (TP) and chlorophyll **a** (Chl) for the epilimnion in deep-water locations (UFI 2007). Phytoplankton growth in the lake is phosphorus-limited (Oglesby 1978). Conditions in the southern end of Cayuga Lake, particularly the southernmost 2 km with depths < 6 m (Figure 1; designated the "shelf"), have generally been considered degraded relative to the deep-water portions of the lake (Oglesby 1978). The occurrence of higher turbidity levels is a prominent feature of this perceived degradation (e.g., Oglesby 1978). Total phosphorus concentrations have routinely exceeded 20 $\mu g \cdot L^{-1}$ on the shelf (UFI 2003), the "guidance" value (i.e., open to some regulatory discretion) for New York [New York State Department of Environmental Conservation (NYSDEC) 1993] to protect recreational uses of lakes. Recently, NYSDEC added this portion of the lake to the state's list of water quality limited systems (as per section 303d of the Federal Clean Water Act), which may be followed by a "total maximum daily load" (TMDL) analysis.

The shelf receives a number of external inputs, including effluents from two domestic waste treatment facilities (Ithaca WWTP and Cayuga Heights WWTP), spent cooling water from Cornell's Lake Source Cooling (LSC) facility, and inflows from the two largest tributaries of the lake (Cayuga Inlet and Fall Creek; Figure 1). Average effluent flows for the two treatment facilities are 0.3 and 0.07 $m^3 \cdot s^{-1}$; the TP limit for these discharges is 1 $mg \cdot L^{-1}$ (Great Lakes basin standard). The annual average flow rate from the LSC facility has been about 0.7 $m^3 \cdot s^{-1}$, which is 8% of the total flow to the southern portion of Cayuga Lake. During the growing season (June – September) tributary flow rates decrease and the average flow from LSC increases to 1.1 $m^3 \cdot s^{-1}$. As a result, LSC contributes about 30% of the total inflow during June – September. The two tributaries represent ~ 35% of the total inflow to the lake (Oglesby 1978). The shelf is flushed rapidly; approximately once per day during the May – October interval (UFI 2008).

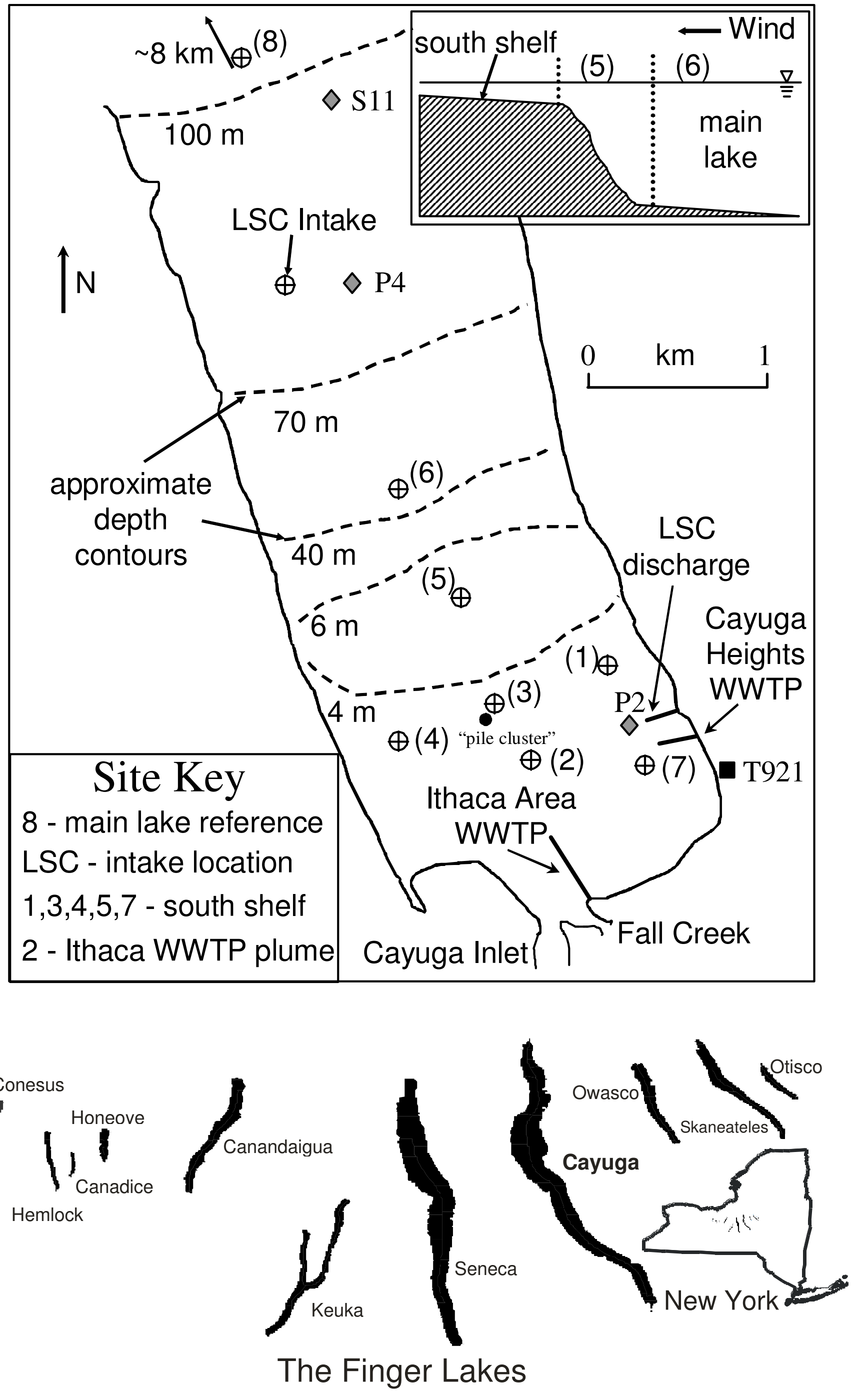


Figure 1. Sampling sites, setting, approximate bathymetry, for LSC monitoring program, southern end of Cayuga Lake; Cayuga Lake's position within the Finger Lakes of New York. Locations of sampling sites and point source discharges are approximate. Sites sampled during the 1994 – 1996 environmental impact study (P2, P4 and S11; Stearns and Wheler 1997) are included for reference.

Cornell University began operating a lake source cooling (LSC) facility in July 2000 that utilizes the cold waters of the hypolimnion of Cayuga Lake to meet its campus cooling needs. Water is drawn from a depth of ~ 77 m (~ 3 m above the bottom) and conveyed to a heat-exchange facility through a 3.2 km intake pipe, and returned through a 154 m outfall pipe (with a multi-port diffuser) to Cayuga Lake's southern shelf (Figure 1). The project is intended to reduce the consumption of fossil fuel, eliminate the related emissions of chlorofluorocarbons, and over the long-term reduce cooling costs. The volume of lake water circulated through the LSC system is variable, depending on the campus demand for cooling; the permitted flow rate is 2 $m^3 \cdot s^{-1}$. The Environmental Impact Statement (EIS) for the LSC facility estimated a 3 to 7% increase in the existing TP load to the shelf associated with its operation (Stearns and Wheler 1997). The impact of this added phosphorus load on algal growth was estimated to be low and no discernable impact on clarity was projected (Stearns and Wheler 1997). During its first six years of operation (2000 – 2005), the LSC facility has contributed an estimated 3 to 7% of the total external load of TP to the southern portion of Cayuga Lake during the May – October interval (UFI 2006).

The discharge permit for the LSC facility requires ambient lake monitoring, with a focus on the potential for impact on trophic state indicators. The surrogate measures of trophic state specified are those that are widely applied, TP, Secchi disc transparency (SD), and Chl. The LSC discharge permit offers the following guidance with respect to in-lake monitoring and impact detection (SPDES No. NY 024 4741, Part I, Section II):

A. Resource Monitoring

> In-lake monitoring will be required to show that the levels of total phosphorus and chlorophyll-a in the lake segment, as described, have not increased. Additionally Secchi Disc transparencies shall also be monitored. If trending shows a statistically significant increasing concentration for total phosphorus over time, the outfall will have to be reevaluated. Reevaluation is discussed in Section C. For chlorophyll-a, results must be presented in a summary report showing comparisons to pre-discharge years. Statistically significant changes will trigger outfall reevaluation. Secchi disc data must also be reported. Should clarity show a statistically significant trending decline, the outfall will have to be reevaluated.
>
> Cornell will submit to this Department, within six months of EDP, an approvable monitoring plan for these parameters listed. Monitoring shall cover the entire growing season, from April 1 to October 31. Additionally, temperature shall also be trended with the data collected. Temperature monitoring shall be year round. All reports of data collected shall be submitted in a coherent and understandable manner, with cumulative trending of all data points over the permit life. Monitoring shall be done at least twice a month, in-lake, in the discharge segment of the lake, as defined above. Additionally, data must be collected from at least two separate locations in the lake, in portion described above. It is encouraged that more are collected and analyzed. An annual report of in-lake monitoring must be submitted to the Department for review and approval by April 1 of the following year.

C. Outfall Reevaluation

> Should the water quality of the discharge area of Cayuga Lake be proven to have deteriorated because of the addition of the Cornell LSC outfall, the outfall location and discharge parameters must be reevaluated. Any statistically significant trend of increasing parameters will require reevaluation. If reevaluation of the outfall is required, Cornell has six months to determine causes and present the methods for ceasing further lake detriment and for restoring problems created by the LSC outfall. Possible alternatives would include, but not be limited to, moving the outfall to a location 'over the shelf' of the southern end of the lake, or treatment for phosphorus.

Application of TP and SD as trophic state indicators implicitly assumes that particulate forms of phosphorus exist predominantly as phytoplankton, and that concentrations of phytoplankton regulate SD, respectively (Carlson 1977). Effler et al. (2002) demonstrated that clay minerals and quartz, received from the watershed, and $CaCO_3$, that is produced internally, are the primary regulators of turbidity ($T_n$) and SD in the southern portion of Cayuga Lake. These inorganic particles also represent most of the particulate phosphorus and are primarily responsible for the higher $T_n$, lower SD, and higher TP on the southern shelf compared to the deep water region (Effler et al. 2002). Matthews et al. (2002) found Chl to be the preferred indicator of trophic state for this system. Secchi disc transparency is a systematically flawed measure of clarity on the southern shelf of Cayuga Lake because of the shallowness of this area. The disc is often visible lying on the lake bottom. Matthews et al. (2002) recommended $T_n$ instead of SD as a measure of clarity on the shelf.

## 3. Methods

### 3.1. Monitoring Program Design and Measurements

Data from five lake sites are used in this analysis. Four sites are located in the southern end (sites 1, 4, 5 and 7) and one (site 8) located further north (Figure 1). Site 8 is included as a reference location representative of main lake conditions. Sites 1, 3, 4, 5 and 7 are considered to be located on the shelf (i.e., depths < 6 m). Sites 1 and 7 are located in the vicinity of the LSC discharge along the east shore. Site 1 and 7 are located northwest and southwest of the LSC discharge, respectively. Sites 3, 4 and 5 represent conditions in the central, western, and northern portions of the shelf, respectively.

Three water quality parameters (TP, Chl, and $T_n$) are considered in detail here. Total phosphorus was measured according to standard methods (APHA 1996). Chlorophyll **a** was measured according to Parsons et al. (1984). Turbidity was measured with a calibrated *HACH 2100AN* turbidimeter (APHA 1992). Lake sampling was conducted bi-weekly, over the July-October interval of 1998, and for the April-October period of 1999, 2000, 2001, 2002, 2003, 2004 and 2005. Additional weekly sampling was conducted from May to August of 2000, bracketing start-up of the LSC facility in early July. A total of 131 sampling surveys were conducted over these eight years, 38 in the pre LSC start-up period and 93 in the post start-up period. Composite samples, formed from equal volumes of sub-samples collected at depths of 0, 2, and 4 m, were collected at sites 5 and 8. Composite samples for sites 1, 3, 4 and 7 were formed from equal volumes of sub-samples collected at depths of 0 and 2 m.

Precision of sampling, sample handling and laboratory analyses was assessed by a program of field replicates. Samples for laboratory analyses were collected in triplicate at site 1 on each sampling day. Triplicate samples were collected at one other station each monitoring trip. This station was rotated each sampling trip through the field season. Precision was high for the triplicate sampling/measurement program, as represented by the average values of the coefficient of variation for the six study years (Table 1). Variability was similar for the three parameters considered here (Table 1). The magnitude of uncertainty associated with these measurements should be recognized when interpreting the analyses that follow.

Table 1. Precision for triplicate sampling/measurement program for key parameters for 1998 - 2005, represented by the average coefficient of variation for the six study years.

| Parameter | Site 1 | Rotating Site |
|---|---|---|
| total phosphorus | 0.10 | 0.09 |
| chlorophyll **a** | 0.13 | 0.12 |
| turbidity | 0.13 | 0.11 |

### 3.2. Statistical Design and Analysis

#### *3.2.1. The Before-After-Control-Impact Design*

The BACI design is used here to evaluate changes in trophic indicators following LSC start-up. In this design, paired samples are collected at control and impact locations on multiple dates before and after LSC start-up. The analysis is based on differences between the control site and the impact site paired by sampling date, and a two-sample t-test is conducted comparing the before and after differences (Stewart-Oaten et al. 1986). This is equivalent to a test for a time by site interaction using a two-factor analysis of variance (Smith et al. 1993). The objective is to determine if the mean difference between impact and control locations has changed coincident with the intervention. Simple statistical comparisons (e.g., t-test) of before versus after start-up data for individual impact sites are inappropriate for this evaluation. This is because changes in water quality parameters are likely to occur over time with or without operation of the LSC facility. Thus, data from control sites are used to account for natural temporal variations not associated with LSC operation. The approach we have adopted is to apply the BACI analysis to multiple control-impact site pairs. More complex and potentially more powerful statistical methods could be used to evaluate potential impacts (e.g., Underwood 1992, 1994). The BACI approach adopted here can be viewed as a 'least common denominator' analysis – it can be simplified no further. This offers the advantage of transparency, allowing regulators and the public to follow what was done. The efficacy of the BACI design for assessing impacts to the southern portion of Cayuga Lake associated with operation of the LSC facility was discussed by Matthews et al. (2002).

A statistically significant change in the difference between control and impact sites is evidence of a change coincident with LSC start-up, but not necessarily evidence that LSC operation is the cause of the change. Even if the control and impact sites are similar prior to the impact, there is no guarantee that this similarity would persist over time absent of LSC operation. The difficulty of assigning a cause to an observed change is common to observational studies.

Discovering the cause of any observed changes in trophic indicators in the southern end of Cayuga Lake is particularly complicated because of the potential for simultaneous changes in multiple drivers not associated with LSC start-up and operation. Matthews et al. (2002) discussed a number of these potentially confounding factors including: (1) natural variation in meteorological conditions, (2) changes in treatment at wastewater treatment plants, and (3) the uncertain effects of zebra mussel populations.

It is also important to note that statistical significance is not equivalent to biological significance (Scheiner and Gurevitch 2001). Large studies conducted on populations that vary little may detect very small, biologically unimportant effects as significant. Conversely, biologically meaningful effects can go undetected if sample sizes are small or natural spatial or temporal variability is high. Sample sizes for this study are large, 38 and 93 for the pre and post start-up intervals, respectively. The three variables of interest (Chl, TP and $T_n$) have exhibited substantial variability on the southern shelf of Cayuga Lake, both among sites and over time at individual sites (Matthews et al. 2002, UFI 1999-2007). Power analyses conducted by Matthews et al. (2002) on pre start-up data established that the BACI analysis would detect a 30% change in Chl with a probability of 0.7 at $\alpha = 0.05$. The probability of detecting a 30% change in Chl increases to about 0.8 if evaluated at $\alpha = 0.10$. Similar results were obtained for TP and $T_n$ (UFI, unpublished results). Thus, the statistical design adopted here is appropriate for the detection of changes on the order of about 30% or greater, though substantially smaller effects will likely be judged statistically insignificant. These power analyses were conducted on uncorrected *p*-values and are not representative of the lower statistical power that results from the Bonferroni or Benjamini and Hochberg (1995) adjustments for multiple comparisons.

### *3.2.2. Outlier Analysis*

An outlier analysis (Section 6) was performed on untransformed TP, Chl, and $T_n$ data using Grubbs's test statistic for outliers (Sokal and Rohlf 1995). Grubbs's test statistic is $(Y_1 - Y)/s$, where $Y_1$ is the suspected outlier, $Y$ is the sample mean, and $s$ is the sample standard deviation. Critical values for Grubbs's test statistic are presented in Sokal and Rohlf (1995). This test was performed on a site-by-site basis for both the before LSC and after LSC periods. Deviations from the mean found to be significant at a two-tailed probability of 0.05 were identified and the veracity of these measurements was investigated using other supporting information. Statistical outliers that could not be explained with available data were identified, and the BACI analysis was conducted both with and without these data points.

### *3.2.3. Selection of Impact and Control Sites and Significance Levels*

Application of the BACI design begins with *a priori* selection of suitable impact and control sites. Impact sites should be located within the area potentially affected by LSC operation. Not surprisingly, potential impacts are most likely in the immediate vicinity (e.g., diffuser "mixing zone") of the LSC discharge (Stearns and Wheler 1997). Further, there is some limited information that suggests a predominant counterclockwise flow pattern in the southern end of Cayuga Lake, particularly following major runoff events (Oglesby 1978). This flow pattern is expected to cause the LSC discharge to move to the north, in the direction of site 1 (Figure 1). Further, model projections included in the **Draft Environmental Impact Statement** (Stearns and Wheler 1997) predicted that the LSC discharge would usually move in a northerly direction, making site 1 the most likely location for detection of impacts associated with LSC. Although site 7 is located southwest (i.e., upstream) of the LSC discharge, it is also treated as an impact site in this analysis because of its proximity to the LSC outfall. In addition, the average of sites 1, 3, 4 and 5 (designated as DMR) is treated as an impact area. This is the spatial averaging used in Discharge Monitoring Reports (DMR) submitted to NYSDEC and intended to represent conditions on the shelf.

Control sites should be located in areas subject to the same temporal and spatial variability found at the impact sites, but outside the area of potential impact (Stewart-Oaten et al. 1986, Underwood 1994). If impacts are limited to a small area near the LSC discharge, another site located on the shelf could serve as a suitable control (e.g., site 4; Figure 1). If, however, the estimated scale of impact is different than indicated by Stearns and Wheler (1997), and changes occur over the entire shelf, a significant change could go undetected because the control site would also be affected (Underwood 1994). Ideally, another shallow area adjacent to wastewater treatment plants and major tributaries (i.e., similar to the south shelf of Cayuga Lake) would be designated as a control. Because such an area is not available, a deep-water site in Cayuga Lake (e.g., site 8; Figure 1) is also considered here as a control location.

Running the BACI analysis on multiple impact-control pairs constitutes a series of pairwise comparisons. Without adjustment, the familywise error rate (FWER), or cumulative Type-I error rate (the probability of a false positive) increases as the number of comparisons increases (Kuehl 1994). Although it is generally accepted that some control on the Type I error rate should be exercised when multiple comparisons are conducted, this position has not been adopted universally by researchers (Rothman 1990, Stewart-Oaten 1995, O'Keefe 2003). The experimentwise Type-I error rate is typically controlled at some desired level (e.g., $\alpha = 0.05$) using the Bonferroni adjustment or some other accepted method. In the Bonferroni adjustment, the desired experimentwise error rate $\alpha$ is divided by the number of individual tests. Thus, if six comparisons are made and an overall $\alpha = 0.05$ is desired, the $\alpha$-level for individual comparisons becomes 0.0083 (0.05/6 = 0.0083). The Bonferroni adjustment is widely considered to be conservative because it exerts very strong control of the Type-I error rate, and other more powerful methods exist for controlling the false discovery rate (e.g., Rice 1989, Benjamini and Hochberg 1995). However, any method that controls for Type I errors necessarily increases the probability of Type II errors, or failures in the detection of real effects. An adjustment procedure for multiple comparisons developed by Benjamini and Hochberg (1995) controls the false discovery rate (FDR), which is the probability of rejecting the null hypothesis (no effect) in any one specific comparison where the null is actually true. This method is the most powerful of all those available to correct for multiple comparisons and has been promoted as a compromise between outright refusal to control for multiplicity, which maximizes Type I error, and strict adherence to FWER control, which minimizes power (Waite and Campbell 2006, Matsunaga 2007). Because we don't want to compromise the ability of this analysis to identify real effects, the number of comparisons has been limited to six. A significance level of 0.05 ($\alpha = 0.05$) has been chosen for the purpose of evaluating permit compliance. Thus, individual t-tests with Benjamini-Hochberg adjusted $p$-values < 0.05 are considered "statistically significant" and constitute strong evidence of a change in water quality coincident in time with start-up of the LSC facility. Bonferroni and Benjamini-Hochberg adjusted $p$-values were computed using version 2.7.2 of R: A Language and Environment for Statistical Computing (R Development Core Team 2008).

Impact-control site pairings for the BACI analysis were not determined prior to collection and preliminary analysis of the data. Therefore, we cannot eliminate the possibility that selection of the pairwise comparisons was biased by knowledge of the results. This issue bears directly on the appropriate interpretation of the *p*-values presented. In the interest of transparency, both adjusted and unadjusted *p*-values are presented in this report. However, the unadjusted *p*-values are confounded by the fact that the pairwise comparisons presented here are a small subset of those actually conducted. Approximately 20 individual pairwise comparisons were conducted at $\alpha = 0.05$, resulting in a probability of 0.64 that at least one statistically significant (unadjusted $p<0.05$) difference would be detected by chance alone. This rate of false detection would be considered unacceptably high in a scientific study. Thus, Benjamini-Hochberg adjusted *p*-values are presented as the most appropriate indicator of statistically significant changes.

Based on careful consideration of the issues outlined above, the New York State Department of Environmental Conservation and Cornell University have selected seven impact-control pairs for the BACI analysis: (1) impact—site 1, control—site 4, (2) impact—site 1, control—site 8, (3) impact—site 7, control—site 4, (4) impact—site 7, control—site 8, (5) impact—site 4, control—site 8, (6) impact—site 5, control—site 4, and (7) DMR—site 8. These seven pairings allow for the testing of a number of hypotheses related to potential changes in trophic indicators in southern Cayuga Lake following LSC start-up. Pairings No. 1 and No. 3 provide tests of whether or not levels of TP, Chl and $T_n$ increased disproportionately at sites 1 and 7 compared to site 4 following LSC start-up. If, however, site 4 was also impacted, this test could fail to detect a substantial change in water quality. Pairings No. 2 and 4 overcome this problem by using site 8, which is located well beyond the zone of influence of the LSC discharge, as the control. If any of the first four pairings produce a statistically significant result, pairings No. 5 and 6 become important as checks for consistency and for defining the spatial extent of the impact. Pairing No. 7 compares water quality changes on the shelf relative to those at the deep water reference site 8. Adjustments for multiple comparisons were not applied to the DMR-site 8 pairing because this test was considered to be chosen *a priori*. Although water quality data continued to be collected through 2008, only data collected through 2005 is included in this analysis because of potentially confounding effects related to a 50% reduction in TP loading from the Ithaca Area Wastewater Treatment Plant beginning in 2006.

#### ***3.2.4.*** *Assumptions of Statistical Tests*

Values for TP ($\mu g \cdot L^{-1}$), Chl ($\mu g \cdot L^{-1}$) and $T_n$ (NTU) were determined from samples collected on the same day for replicate surveys conducted before and after start-up of the LSC facility. The data were log-transformed to achieve additivity and reduce autocorrelation (Stewart-Oaten et al. 1986) and differences ($\Delta$) were calculated between the values at impact and control locations for each sampling date. Statistical significance is determined from two-tailed Welch t-tests comparing the before and after differences for the various control-impact pairs. STATISTICA version 6 (data analysis software system; StatSoft, Inc. 2003) was used for statistical analyses.

The two-sample t-test conducted on the before and after differences is subject to the usual assumptions for such a test: the observations (differences) are assumed independent, and the sample sizes are assumed large enough so that the distribution of the mean differences (before and after periods) is approximately normally distributed. Variances of the differences in the two time periods need not be assumed equal if the Welch t-test (Snedecor and Cochran 1980) is used. The normality assumption is likely satisfied by the data typically encountered in a BACI study unless numerous zeros are present (as may occur for abundance data). Applying a logarithmic transformation, which is a common practice with environmental data (Eberhardt and Thomas 1991, Stewart-Oaten et al. 1992, Osenberg et al. 1994), reduces skewness of the data prior to calculating differences, and taking differences still further diminishes skewness (Stewart-Oaten et al. 1992). Sample sizes for both the before and after periods exceed 30 in this study; consequently, the Central Limit Theorem further contributes to approximate normality of the distribution of the mean differences. If the normality assumption is problematic, the Mann-Whitney (also called Wilcoxon) non-parametric two-sample test or a randomization test (Carpenter et al. 1989) can be used in place of the Welch t-test. Although these alternatives do not invoke a normality assumption, they do require the other assumptions of the Welch t-test (Stewart-Oaten et al. 1992). Because the normality assumption was not satisfied for all of the impact-control distributions (Appendix 1), Mann-Whitney tests were conducted for all impact-control pairs. The Mann-Whitney tests yielded results consistent with the Welch t-tests, and no significant (Benjamini & Hochberg adjusted $p < 0.05$) pre-post differences were observed (see Appendix 4).

The independence assumption is likely more of a concern than the normality assumption because of possible serial correlation of the differences over the sampling times. The effect of positive serial correlation is to inflate the Type I error rate of the test. In other words, positive serial correlation will make it more likely to detect a LSC effect that does not exist. Stewart-Oaten et al. (1986) emphasize that it is the differences that must be uncorrelated, not the observations over time at each individual sampling station. Serial correlation of the differences is not expected to be nearly as strong as serial correlation in the observations obtained at each individual site. Even if serial correlation is found to be statistically significant, conclusions of the BACI test remain valid if the serial correlation is small; e.g., lag-1 $r < 0.3$ (Stewart-Oaten et al. 1986).

Another assumption of the BACI analysis is 'additivity' of time and location effects. Violation of this assumption may cause the BACI test to lose power because the differences are highly variable or inflate the actual Type I error of the test (Stewart-Oaten et al. 1986, Smith et al. 1993). Stewart-Oaten et al. (1986) suggest employing Tukey's (1949) test for non-additivity, although Smith et al. (1993) note that the test for additivity is sensitive to serial correlation. If non-additivity exists, a log-transformation often diminishes or eliminates the problem. Stewart-Oaten et al. (1992) discuss more details of the additivity assumption. After log transformation and differencing (impact-control), the data were checked against the assumptions for a Welch t-test (normality, temporal independence, additivity). Evaluations of the normality, independence, and additivity assumptions are presented in Appendixes 1, 2, and 3, respectively.

#### ***3.2.5.*** *Calculation and Interpretation of Effect Size*

For the BACI design, the test for an impact associated with LSC start-up is an interaction test: does the mean difference between the control and impact site before LSC start-up differ from the mean difference after start-up? Because the key test is one of interaction, defining a readily interpretable effect size is more difficult than when the test is a comparison of means (rather than mean differences). The rationale for choosing and interpreting effect size is as follows. In the Before period, suppose the mean of the control site is $x$, and that the mean of the impact site is $a\%$ higher, $(1+a)x$. Both the control and impact site means increase $b\%$ in the After period, so the control site has mean $(1+b)x$, and the impact site has mean $(1+a)(1+b)x$. This $b\%$ increase for both control and impact sites is consistent with additivity on a log-transformed scale. If an impact is present, suppose it further increases the mean of the impact site in the After period by an additional $c\%$, so the impact site mean is $(1+a)(1+b)(1+c)x$. On the log-transformed scale, the difference in means in the Before period is then $\log(x)\log[(1+a)x] = \log[x/(1+a)x] = \log[1/(1+a)]$, and the difference in means in the After period is $\log[(1+b)x] - \log[(1+a)(1+b)(1+c)x] = -\log[1/(1+a)(1+c)]$. Finally, the interaction test would evaluate the difference of the differences in the Before and After period, resulting in $-\log[1/(1+a)(1+c)] - \log[(1/(1+a)] = -\log[1/(1+c)]$. Thus, the effect size $c$ is the percent increase in a variable associated with the impact. This formulation of the effect size is consistent with a multiplicative effects model that motivates analysis on the logarithmic scale.

The calculation and interpretation of effect size is illustrated here using a hypothetical example. Suppose that in the Before period the mean chlorophyll **a** concentration of the control site ($x$) is 5 $\mu g \cdot L^{-1}$ and the mean chlorophyll **a** concentration of the impact site ($y$) is 20% higher, $y = (1+ 0.20)x = 6\ \mu g \cdot L^{-1}$. Both the control and impact site means increase 10% in the After period, so the control site has mean $(1+0.10)5\ \mu g \cdot L^{-1} = 5.5\ \mu g \cdot L^{-1}$, and the impact site has mean $(1+0.20)(1+0.10)5\ \mu g \cdot L^{-1} = 6.6\ \mu g \cdot L^{-1}$. Suppose that a perturbation in the After period causes the mean chlorophyll **a** concentration of the impact site to increase by an additional 20%, so the impact site mean is $(1+0.20)(1+0.10)(1+0.20)5\ \mu g \cdot L^{-1} = 7.92\ \mu g \cdot L^{-1}$. In the BACI analysis, we evaluate the increase at the impact site over and above any increases that also occurred at the control site. On the log-transformed scale this increase is $\log(7.92\ \mu g \cdot L^{-1}) - \log(6.6\ \mu g \cdot L^{-1}) = 0.0792$. We can see that this is equivalent to the formulation of effect size described above, $-\log[1/(1+0.20)] = 0.0792$. Thus, the effect size $c$ can be determined by solving $0.0792 = -\log[1/(1+c)]$ for $c$, which yields $c = 0.20$.

## 4. Simple Before-After Comparison of Mean Values

A simple comparison of mean values for the pre and post LSC start-up intervals is presented here as a way to characterize water quality changes in the southern portion of Cayuga Lake. This analysis is presented outside of the BACI results (Section 7) because it is not an appropriate method for assessing potential impacts associated with the LSC facility. Changes in water quality would be expected to occur with or without LSC operation. In fact, chlorophyll **a** (Chl) concentrations decreased 35% from 1994 – 1996 to 1998 – 1999 in the absence of LSC or any other documented perturbation (UFI 2007). Furthermore, total phosphorus (TP) and Chl concentrations increased on the southern shelf in 2006 despite a 50% decrease in TP loading from IAWWTP. Thus, the simple comparison of mean values presented here should be interpreted as a snapshot in time of a complex ecosystem and not as a cause and effect analysis.

Average chlorophyll **a** concentrations increased in the post start-up period at eight of the nine sites monitored in this study (Figure 2). At site 2 Chl decreased by 4% and the increase at site 1 was just 1%. Increases at the remaining sites (3, 4, 5, 6, 7, 8, LSCI) ranged from 16 to 49%. The largest increases occurred at site 4 (31%) and site 7 (49%). Temporal variability, as represented by the standard deviation, also increased at sites 4 and 7 in the post start-up period (Figure 2). This is not surprising when the unbalanced sample sizes (38 in the pre start-up period and 93 in the post start-up period) and skewed distributions are considered. In other words, there was a greater probability of observing extreme conditions in the post start-up interval. When median values were considered rather than means, increases in Chl at sites 4 and 7 were 7% and 39%, respectively. Chl concentrations at southern shelf sites (1, 2, 3, 4, 5, 7) were not markedly higher than at deep water sites (6, LSCI, 8).

Average TP concentrations increased in the post start-up period at eight of the nine sites monitored in this study (Figure 3). At site 2 TP decreased by 22%, and modest 5-7% increases were observed at sites 1, 3, 5 and LSCI. Increases at the remaining sites (4, 6, 7, 8) ranged from 13 to 15%. Total phosphorus concentrations were higher at southern shelf sites than at deep water sites. The highest TP concentrations were observed at shallow sites located in the vicinity of wastewater discharges (1, 2, 3, 7). Higher TP in the shallow area compared to the deep water region has been attributed to contributions of inorganic tripton delivered to Cayuga Lake from Fall Creek and Cayuga Inlet (Effler et al. 2002).

Average turbidity ($T_n$) levels increased modestly (< 1 NTU) in the post start-up period at sites 1, 3 and 7 (Figure 4). Equally modest decreases in $T_n$ occurred at sites 2, 4 and 5. Turbidity levels were essentially unchanged at sites 6, 8, and LSCI. Turbidity levels were higher at southern shelf sites than at deep water sites. The highest $T_n$ levels were observed at shallow south shelf sites (1, 2, 3, 7). Higher $T_n$ in the shallow area compared to the deep water region has been attributed to contributions of inorganic tripton delivered to Cayuga Lake from Fall Creek and Cayuga Inlet (Effler et al. 2002).

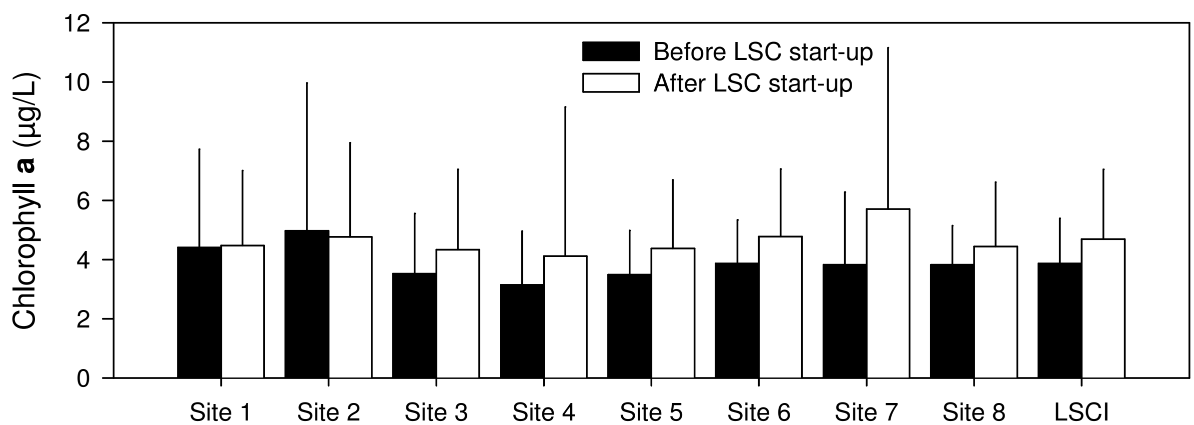


Figure 2. Average chlorophyll **a** concentrations for the periods before (7/9/98 – 6/29/00) and after (7/6/00 – 10/24/05) start-up of the LSC facility. Error bars represent plus one standard deviation of the mean. Symmetrical lower error bars are implied.

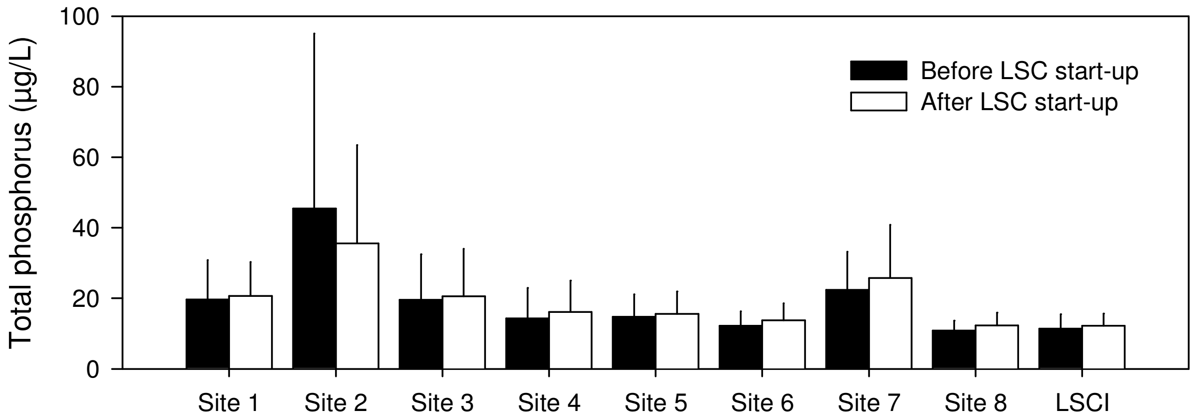


Figure 3. Average total phosphorus concentrations for the periods before (7/9/98 – 6/29/00) and after (7/6/00 – 10/24/05) start-up of the LSC facility. Error bars represent plus one standard deviation of the mean. Symmetrical lower error bars are implied.

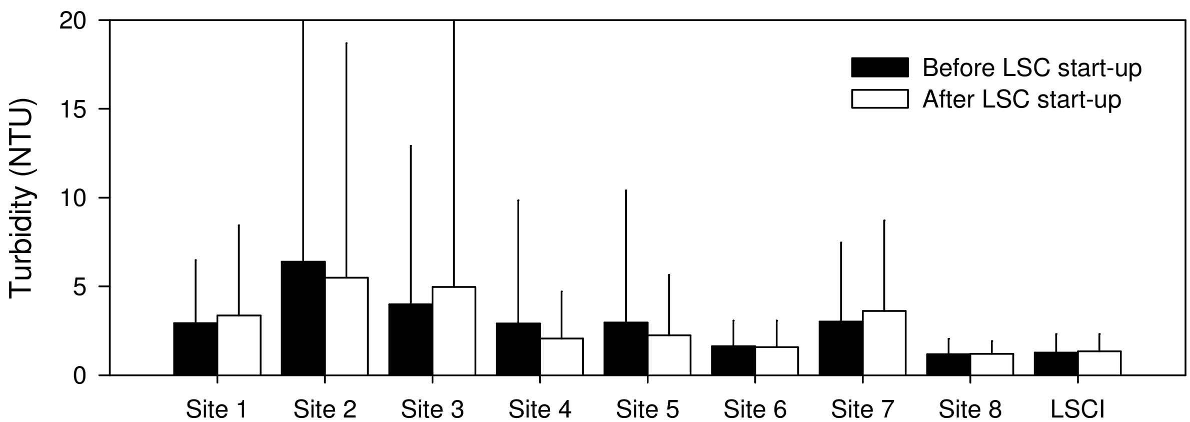


Figure 4. Average turbidity levels for the periods before (7/9/98 – 6/29/00) and after (7/6/00 – 10/24/05) start-up of the LSC facility. Error bars represent plus one standard deviation of the mean. Symmetrical lower error bars are implied.

## 5. Daily Average Flow Rates for LSC, 2000 - 2005

The time series of daily average flow rates for the LSC facility depicts a recurring seasonal pattern driven primarily by ambient environmental temperatures and the associated demand for cooling (Figure 5). Flow rates were lowest in the winter months, increased during spring, peaked during mid-summer, and decreased during fall. The highest flow rates were typically observed during July and August. Average flow rates for July-August were 22 MGD in 2000, 25 MGD in 2001, 33 MGD in 2002, 32 MGD in 2003, 28 MGD in 2004, and 33 MGD in 2005.

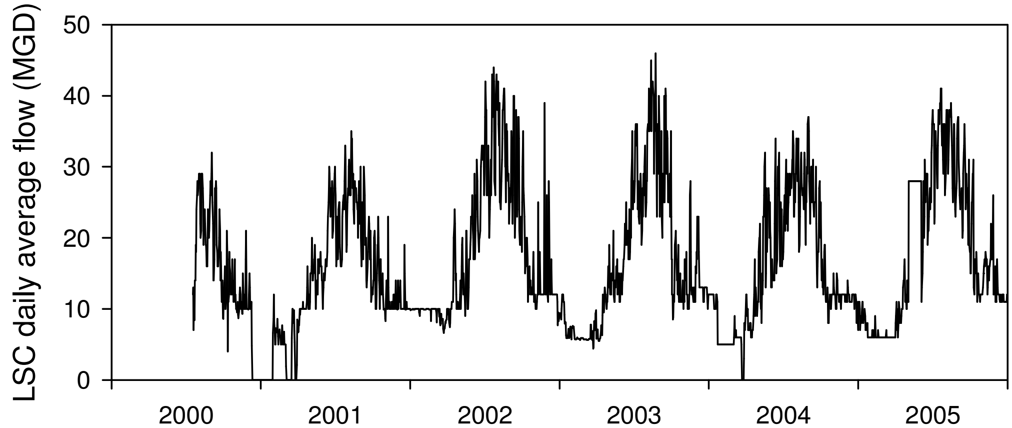


Figure 5. Daily average flow rates for the LSC facility, 2000 – 2005. Daily flow rates are averages of hourly measurements and represented in units of millions of gallons per day (MGD).

## 6. Results of Outlier Analysis

An outlier analysis was performed to identify spurious data that could have deleterious effects on the BACI analysis. Only statistical outliers that could not be explained by other supporting data were considered to be outliers in the BACI analysis. Data points identified as statistical outliers by the Grubbs' test statistic are listed in Tables 2, 3, and 4. Thirteen of the 648 Chl measurements (2%) from sites 1, 4, 5, 7 and 8 were identified as statistical outliers (Table 2). It's interesting that 12 of the 13 extreme values were from samples collected in late July, September, and October. In addition, ten outliers were from the shallow (i.e., < 5 m deep) sites 1, 4, and 7. We hypothesize that dense macrophyte beds that develop annually during summer in the shallow areas of the southern shelf interfered with collection of representative samples. Sampling was conducted from a motorboat, which may have shredded macrophytes and disturbed attached algae. This could result in false high Chl concentrations by inclusion of material other than phytoplankton in the samples. *In situ* fluorometric chlorophyll **a** measurements that were taken at the time of water sampling provide an objective basis to evaluate these extreme Chl concentrations. Six of the statistical outliers were within the 99% prediction interval defined by linear regression of laboratory Chl on in situ Chl, suggesting that these measurements were reasonably representative of ambient conditions (Figure 6). Five of the statistical outliers were outside of the 99% confidence interval, which indicates that they did not accurately reflect ambient levels of Chl (Figure 5). Paired *in situ* measurements were not available for two of the statistical outliers (site 4 on 8/26/04, site 7 on 9/5/02). Because these two statistical outliers could not be compared to *in situ* data, they were treated as outliers in the analyses that follow. The BACI analysis was conducted both with and without the outliers identified by shading in Table 2.

Table 2. Statistical outliers for chlorophyll **a** measurements from sites 1, 4, 5, 7 and 8 identified by the Grubbs' statistic. Shaded data points could not be verified by *in situ* fluorometric data.

| Site | Date | Chl (µg/L) | Supporting Information |
|---|---|---|---|
| 1 | 7/30/98 | 15.6 | Exceeds 99% prediction interval from in situ fluorometry |
| 1 | 6/15/00 | 17.1 | Exceeds 99% prediction interval from in situ fluorometry |
| 1 | 8/17/00 | 14.6 | Within 99% prediction interval from in situ fluorometry |
| 4 | 7/30/98 | 8.1 | Within 99% prediction interval from in situ fluorometry |
| 4 | 8/26/04 | 28.5 | In situ fluorometry data unavailable |
| 4 | 9/9/04 | 37.3 | Exceeds 99% prediction interval from in situ fluorometry |
| 5 | 7/27/00 | 11.7 | Within 99% prediction interval from in situ fluorometry |
| 5 | 7/31/03 | 14.0 | Within 99% prediction interval from in situ fluorometry |
| 7 | 6/29/00 | 12.0 | Within 99% prediction interval from in situ fluorometry |
| 7 | 9/21/00 | 24.1 | Exceeds 99% prediction interval from in situ fluorometry |
| 7 | 9/5/02 | 32.9 | In situ fluorometry data unavailable |
| 7 | 8/10/04 | 30.6 | Exceeds 99% prediction interval from in situ fluorometry |
| 8 | 7/31/03 | 11.0 | Within 99% prediction interval from in situ fluorometry |

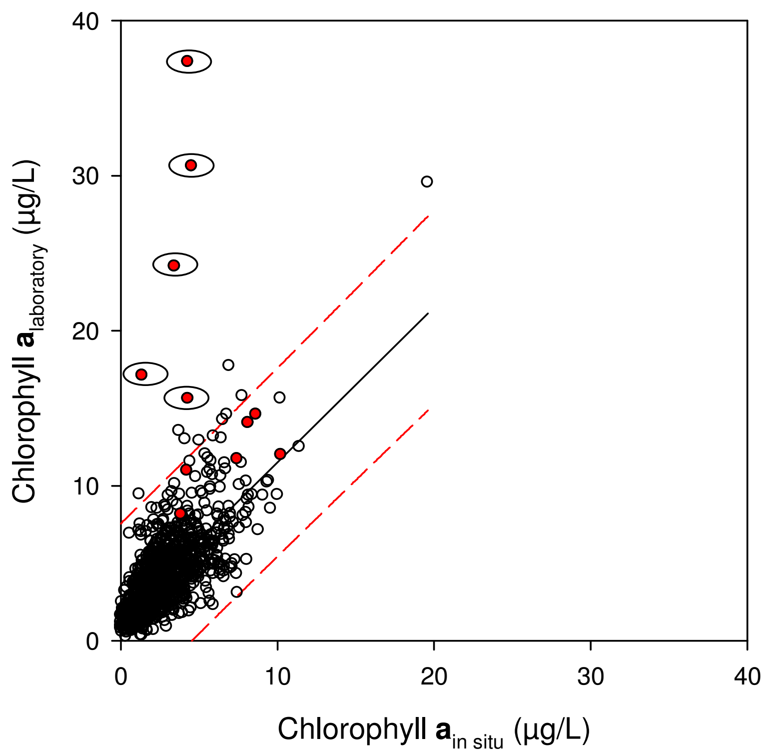


Figure 6. Evaluation of the relationship between laboratory measurements of spectrophotometric chlorophyll **a** and *in situ* measurements of fluorometric chlorophyll **a**. The 99% prediction interval is identified by the red dashed lines. Red data points were identified as statistical outliers by the Grubbs' statistic. Circled data points are statistical outliers that fall outside of the 99% prediction interval.

Fourteen of the 648 TP measurements (2%) and 20 of the 653 $T_n$ measurements (3%) from sites 1, 4, 5, 7 and 8 were identified as statistical outliers (Tables 3 and 4). All of the statistical outliers identified for TP and $T_n$ were adequately explained by supporting limnological information. In most cases high levels of TP and $T_n$ were associated with runoff events, which have been documented to cause major increases in inorganic tripton (i.e., clays) concentrations in the southern end of Cayuga Lake (Effler et al. 2002).

Table 3. Statistical outliers for total phosphorus measurements from sites 1, 4, 5, 7 and 8 identified by the Grubbs' statistic.

| **Site** | **Date** | **TP (µg/L)** | **Supporting Information** |
|---|---|---|---|
| 1 | 7/9/98 | 70.8 | Runoff event |
| 1 | 8/17/00 | 55.6 | Chl also high, possible whiting event |
| 4 | 4/6/00 | 42.3 | Runoff event |
| 4 | 7/6/00 | 44.5 | Slightly elevated turbidity |
| 4 | 4/24/03 | 47.6 | Elevated turbidity, following runoff event |
| 4 | 8/26/04 | 45.6 | Elevated Chl, low Secchi disc, following seiche |
| 5 | 6/22/00 | 42.4 | Runoff event |
| 5 | 4/5/01 | 45.4 | Runoff event |
| 5 | 5/16/02 | 37.4 | Runoff event |
| 7 | 5/11/00 | 56.9 | Elevated turbidity, following runoff event |
| 7 | 10/23/03 | 112.3 | Elevated Chl, turbidity, low Secchi disc |
| 7 | 10/29/03 | 70.6 | Elevated SRP, turbidity, low Secchi disc |
| 7 | 8/10/04 | 70.6 | Elevated Chl |
| 8 | 8/10/04 | 31.1 | High value at LSC intake site (21.1 ug/L) |

Table 4. Statistical outliers for turbidity measurements from sites 1, 4, 5, 7 and 8 identified by the Grubbs' statistic.

| Site | Date | $T_n$ (NTU) | Supporting Information |
|---|---|---|---|
| 1 | 4/6/00 | 16.1 | Runoff event |
| 1 | 5/16/02 | 21.5 | Runoff event |
| 1 | 7/29/04 | 40.6 | Runoff event |
| 4 | 4/6/00 | 40.5 | Runoff event |
| 4 | 4/5/01 | 11.2 | Runoff event |
| 4 | 5/16/02 | 17.9 | Runoff event |
| 4 | 4/24/03 | 10.9 | Runoff event |
| 5 | 6/22/00 | 46.8 | Runoff event |
| 5 | 4/5/01 | 26.7 | Runoff event |
| 5 | 5/16/02 | 15.4 | Runoff event |
| 7 | 4/6/00 | 14.8 | Runoff event |
| 7 | 5/11/00 | 23.9 | Runoff event |
| 7 | 4/1/02 | 18.2 | Elevated flows, low Secchi disc (0.6 m) |
| 7 | 5/16/02 | 30.2 | Runoff event |
| 7 | 10/23/03 | 31.1 | Elevated flows, low Secchi disc (0.5 m) |
| 8 | 5/18/00 | 3.6 | Runoff event |
| 8 | 6/22/00 | 3.7 | Runoff event |
| 8 | 6/28/01 | 3.7 | Runoff event |
| 8 | 5/16/02 | 3.4 | Runoff event |
| 8 | 4/14/05 | 3.6 | Elevated flows, low Secchi disc (2.0 m) |

# 7. BACI Results

## 7.1. Chlorophyll

Time series of log-transformed chlorophyll **a** concentrations (Chl) and differences calculated for the impact-control pairs are presented in Figure 6. Chlorophyll **a** concentrations in southern Cayuga Lake exhibited a recurring seasonal pattern, with low values in spring followed by increases through the summer months and decreases during the fall (Figure 7a-e). Peak Chl concentrations usually occurred during July, August and September. This general pattern has been observed for all monitored sites over the eight study years. Temporal variations were correlated ($0.47 < r < 0.84$) among the five sites. No long-term trends of increasing or decreasing Chl are readily apparent from time series of log-transformed data (Figure 7a-d).

The distinct seasonality apparent for the individual sites was reduced for the impact-control differences (Figure 7f-k). No conspicuous long-term trends were apparent from the time series of impact-control differences. The Shapiro-Wilk test was used to evaluate significant ($\alpha$ = 0.05) deviations from normality (Appendix 1). The normality assumption was met for ten of the 14 impact-control distributions (see Appendix 1). The following distributions were non-normal in the post start-up period: site 1 – site 4, site 1 – site 8, site 7 – site 4, site 7 – site 8. This is not expected to be an important issue because the t-test is generally robust to deviations from normality and sample sizes are large enough to invoke the Central Limit Theorem. Differences calculated from the site 7 – site 4 pairing showed significant negative serial correlation in the pre-LSC period (lag 1 $r$ = -0.32) and positive correlation during the post-LSC period (lag 1 $r$ = 0.25) (Appendix 2). Positive serial correlation was observed for the following impact-control pairings during the post start-up period: site1 – site 4 (lag 1 $r$ = 0.30), site 4 – site 8 (lag 1 $r$ = 0.25), site 5 – site 4 (lag 1 $r$ = 0.20). Serial correlation was low ($r < 0.2$) for the other impact-control pairs. No serious violations of the additivity assumption were detected based on the weak ($r^2 \leq 0.2$) relationships observed between the differences and averages for log Chl (Appendix 3).

Summary statistics and $p$-values are presented in Table 5 for the six selected impact-control pairs. Differences between the pre and post start-up intervals were not statistically significant (Benjamini & Hochberg $p > 0.25$) for any of the impact-control pairs at the nominal significance level of 0.05. Results computed from non-parametric tests (Mann-Whitney U-test) also failed to indicate statistically significant changes in Chl for any of the control-impact pairings (p > 0.6; Appendix 4). The strongest evidence for a post-start-up increase in Chl comes from the site 7 – site 4 ($p$ = 0.255) and site 7 – site 8 ($p$ = 0.255) pairings. These pairings indicate a 23-24% increase at site 7 relative to sites 4 and 8. The power analysis conducted on pre start-up data (Matthews et al. 2002) indicated that the BACI analysis would likely find increases in Chl of this magnitude (23 and 24%) to be statistically insignificant. In contrast, the site 1 – site 4 and site 1 – site 8 pairings indicated a modest (4-5%) decrease in Chl at site 1 relative to sites 4 and 8. This finding indicates that the apparent increase in Chl at site 7 did not extend north (i.e., downstream) of the LSC discharge. The DMR – site 8 pairing indicated a small (4%) statistically irrelevant ($p$ = 0.613) increase on the shelf relative to the main lake.

Table 5. Results from Welch t-tests comparing log-transformed chlorophyll **a** for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs (outliers not removed). Standard deviation is abbreviated $S_d$. Raw $p$-values, Bonferroni-adjusted $p$-values, and Benjamini & Hochberg (1995) adjusted $p$-values (B&H) are presented. Effect size represents the percent change in chlorophyll **a** at the impact site relative to the control site.

| Impact-Control Pairing | Pre start-up log(μg·L$^{-1}$) | | Post start-up log(μg·L$^{-1}$) | | $p$-value | | | Effect size (%) |
|---|---|---|---|---|---|---|---|---|
| | mean | $S_d$ | mean | $S_d$ | raw | Bonferroni | B&H | |
| site 1-site 4 | 0.139 | 0.247 | 0.119 | 0.316 | 0.708 | 1.000 | 0.850 | -4 |
| site 1-site 8 | 0.020 | 0.258 | -0.002 | 0.194 | 0.646 | 1.000 | 0.850 | -5 |
| site 7-site 4 | 0.080 | 0.192 | 0.170 | 0.361 | 0.069 | 0.414 | 0.255 | +23 |
| site 7-site 8 | -0.047 | 0.260 | 0.047 | 0.308 | 0.085 | 0.510 | 0.255 | +24 |
| site 4-site 8 | -0.123 | 0.221 | -0.123 | 0.294 | 0.995 | 1.000 | 0.995 | 0 |
| site 5-site 4 | 0.078 | 0.184 | 0.112 | 0.251 | 0.402 | 1.000 | 0.804 | +8 |
| DMR-site 8 | -0.057 | 0.160 | -0.040 | 0.164 | 0.613 | - | - | +4 |

The BACI analysis was also conducted following omission of the seven Chl outliers identified in **Section 5**. Summary statistics and *p*-values are presented in Table 6 for the six selected impact-control pairs. With outliers removed, differences between the pre and post start-up intervals were not statistically significant (Benjamini & Hochberg $p > 0.17$) for any of the impact-control pairs at the nominal significance level of 0.05. Results computed from non-parametric tests (Mann-Whitney U-test) also failed to indicate statistically significant changes in Chl for any of the control-impact pairings ($p > 0.7$; Appendix 4). Again, the strongest evidence for a post-start-up increase in Chl comes from the site 7 – site 4 ($p = 0.174$) and site 7 – site 8 ($p = 0.398$) pairings. Chl concentrations increased 27% at site 7 relative to sites 4. Relatively modest (5-17%) increases were indicated by the site 1 – site 4, site 1 – site 8, site 7 – site 8, and site 5 – site 4 pairings. The DMR – site 8 pairing indicated a small (7%) statistically irrelevant ($p = 0.343$) increase on the shelf relative to the main lake.

Table 6. Results from Welch t-tests comparing log-transformed chlorophyll **a** for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. The 8 outliers identified in Table 2 have been omitted from this analysis. Standard deviation is abbreviated $S_d$. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented. Effect size represents the percent change in chlorophyll **a** at the impact site relative to the control site.

| Impact-Control Pairing | Pre start-up $\log(\mu g \cdot L^{-1})$ | | Post start-up $\log(\mu g \cdot L^{-1})$ | | *p*-value | | | Effect size (%) |
|---|---|---|---|---|---|---|---|---|
| | mean | $S_d$ | mean | $S_d$ | raw | Bonferroni | B&H | |
| site 1-site 4 | 0.107 | 0.186 | 0.143 | 0.274 | 0.395 | 1.000 | 0.593 | +9 |
| site 1-site 8 | -0.025 | 0.178 | -0.002 | 0.194 | 0.544 | 1.000 | 0.653 | +5 |
| site 7-site 4 | 0.079 | 0.192 | 0.183 | 0.317 | 0.029 | 0.174 | 0.174 | +27 |
| site 7-site 8 | -0.047 | 0.260 | 0.021 | 0.276 | 0.199 | 1.000 | 0.398 | +17 |
| site 4-site 8 | -0.123 | 0.221 | -0.143 | 0.262 | 0.668 | 1.000 | 0.668 | -5 |
| site 5-site 4 | 0.078 | 0.184 | 0.132 | 0.211 | 0.155 | 0.930 | 0.398 | +13 |
| DMR-site 8 | -0.072 | 0.151 | -0.042 | 0.164 | 0.343 | - | - | +7 |

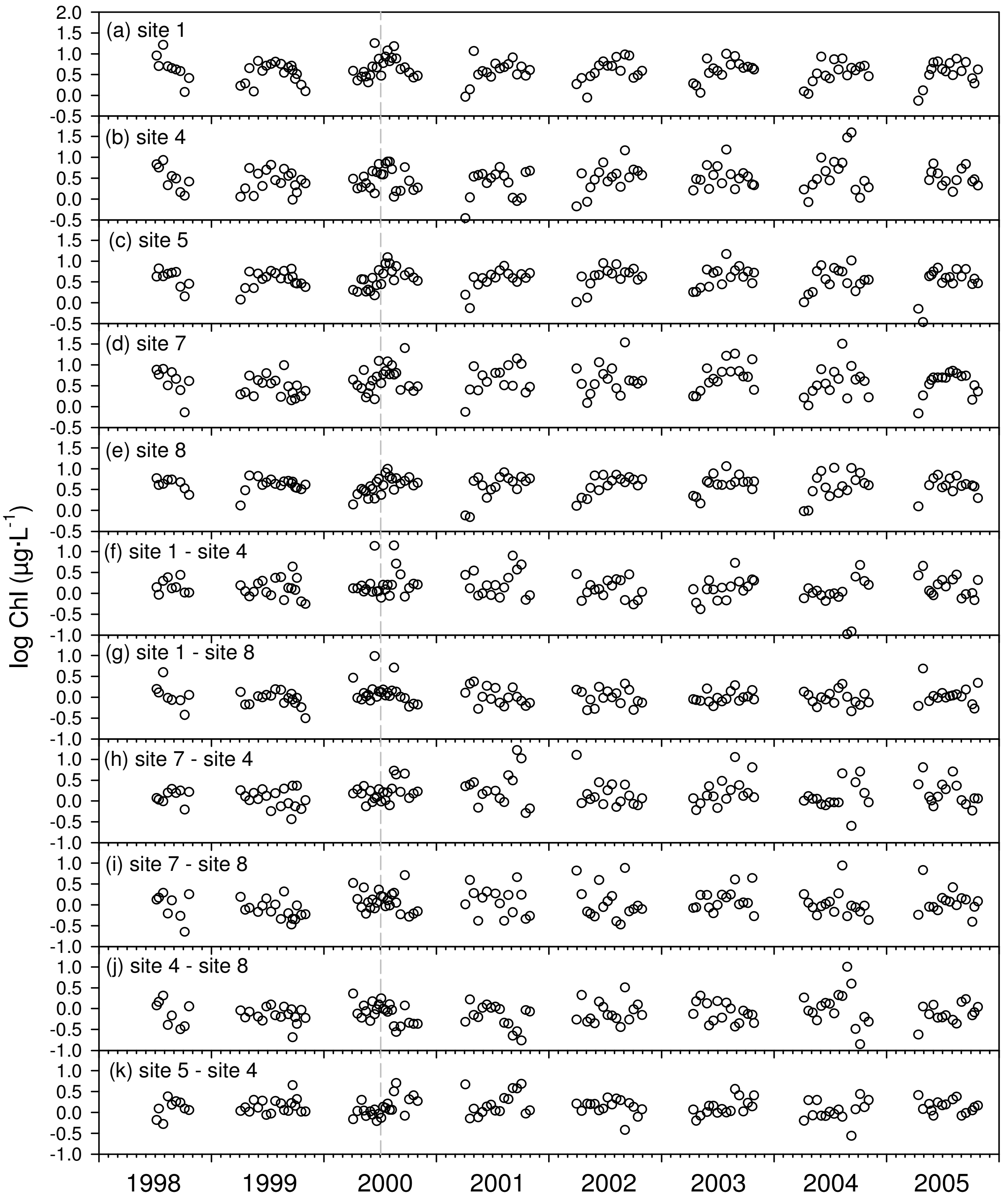


Figure 7. Time series of log Chl concentrations and impact-control differences for the 1998 – 2005 interval: (a) site 1, (b) site 4, (c) site 5, (d) site 7, (e) site 8, (f) site 1 - site 4, (g) site 1 - site 8, (h) site 7 - site 4, (i) site-site 8, (j) site 4 - site 8, and (k) site 5 - site 4. Start-up of the LSC facility is identified by the dashed gray line.

### 7.2. Total Phosphorus

Time series of log-transformed total phosphorus concentrations (TP) and differences calculated for the impact-control pairs are presented in Figure 8. In contrast to Chl, TP did not exhibit a recurring seasonal pattern in southern Cayuga Lake (Figure 8a-e). The highest TP concentrations on the shelf have generally been observed during periods of high runoff (UFI 1999, 2000, 2001, 2002, 2003, 2004, 2005, 2006) when terrigenous inputs of inorganic tripton are greatest. Effler et al. (2002) found that ~ 50% of the TP on the shelf over the June – October interval of 1999 was tripton. The time period included in this study was particularly dry and it is likely that contributions from tripton to the TP pool are even higher during high flow intervals (Effler et. al. 2002). Total phosphorus concentrations have been relatively high during spring (Figure 8a-e), indicative of contributions from terrigenous inputs. Summertime peaks in TP (Figure 8a-e) that coincided with peaks in Chl (Figure 7a-e) were observed at various sites over the eight-year study period. However, correlations between paired measurements of log TP and log Chl for the five sites were weak ($r < 0.4$). Temporal variations in log TP were correlated ($0.32 < r < 0.68$) among the four sites. The strongest correlation ($r = 0.68$) was between sites 1 and 7 and the weakest (0.32) between sites 7 and 4. No long-term trends of increasing or decreasing TP are readily apparent from time series of log-transformed data (Figure 8a-e).

No distinct seasonality or conspicuous long-term trends were apparent for the impact-control differences (Figure 8f-k). The Shapiro-Wilk test was used to evaluate significant ($\alpha = 0.05$) deviations from normality (Appendix 1). The normality assumption was met for six of the 14 impact-control distributions (see Appendix 1). Three distributions were non-normal in the pre start-up period (site 1 – site 8, site 4 – site 8, DMR – site 8) and five distributions were non-normal in the post start-up period (site 1 – site 4, site 1 – site 8, site 7 – site 8, site 4 – site 8, site 5 – site 4). Although eight of the 14 impact-control distributions violated the normality assumption (Appendix 1), this is not expected to invalidate the results of t-tests. The t-test is generally robust to deviations from normality and sample sizes are large enough to invoke the Central Limit Theorem. Significant positive serial correlation was evident for the DMR – site 8 pairing during the pre-LSC interval (lag 1 $r = 0.30$). Differences calculated from the site 7 – site 4 (lag 1 $r = 0.21$) and site 5 – site 4 (lag 1 $r = 0.23$) pairings showed significant positive serial correlation in the post-LSC period (Appendix 2). Serial correlation was low ($r < 0.2$) for the other impact-control pairs. The additivity assumption was violated for pairings with site 8 as the control site (Appendix 3). This is a result of the disproportionate impact of runoff events on TP concentrations on the shelf compared to site 8.

Summary statistics and *p*-values are presented in Table 7 for the six selected impact-control pairs. Differences between the pre and post start-up intervals were not statistically significant (Benjamini & Hochberg $p > 0.97$) for any of the impact-control pairs at the nominal significance level of 0.05. Results computed from non-parametric tests (Mann-Whitney U-test) also failed to indicate statistically significant changes in TP for any of the control-impact pairings ($p > 0.9$; Appendix 4). Effect sizes were small (<10%) and there was no evidence for substantial changes in TP following start-up of the LSC facility. The DMR – site 8 pairing indicated a small (4%) statistically irrelevant ($p = 0.546$) decrease on the shelf relative to the main lake.

Table 7. Results from Welch t-tests comparing log-transformed total phosphorus for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. Standard deviation is abbreviated $S_d$. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented. Effect size represents the percent change in total phosphorus at the impact site relative to the control site.

| Impact-Control Pairing | Pre start-up $\log(\mu g \cdot L^{-1})$ | | Post start-up $\log(\mu g \cdot L^{-1})$ | | *p*-value | | | Effect size (%) |
|---|---|---|---|---|---|---|---|---|
| | mean | $S_d$ | mean | $S_d$ | raw | Bonferroni | B&H | |
| site 1-site 4 | 0.151 | 0.219 | 0.120 | 0.231 | 0.475 | 1.000 | 0.972 | -7 |
| site 1-site 8 | 0.216 | 0.161 | 0.201 | 0.170 | 0.648 | 1.000 | 0.972 | -3 |
| site 7-site 4 | 0.207 | 0.218 | 0.205 | 0.251 | 0.975 | 1.000 | 0.975 | 0 |
| site 7-site 8 | 0.270 | 0.170 | 0.287 | 0.181 | 0.633 | 1.000 | 0.972 | +4 |
| site 4-site 8 | 0.076 | 0.191 | 0.081 | 0.200 | 0.895 | 1.000 | 0.975 | +1 |
| site 5-site 4 | 0.043 | 0.162 | 0.008 | 0.171 | 0.286 | 1.000 | 0.972 | -8 |
| DMR-site 8 | 0.156 | 0.132 | 0.140 | 0.133 | 0.546 | - | - | -4 |

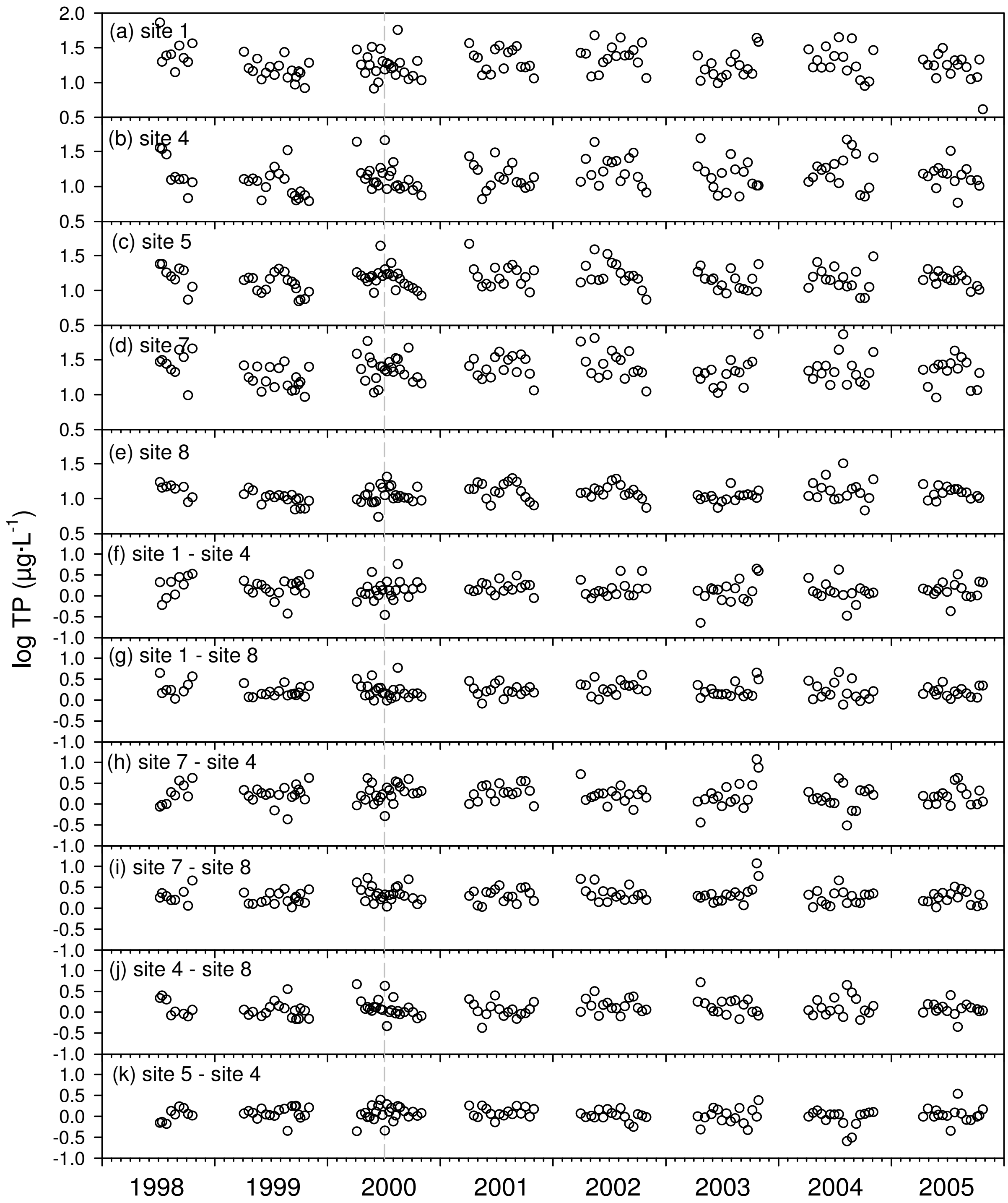


Figure 8. Time series of log TP concentrations and impact-control differences for the 1998 – 2005 interval: (a) site 1, (b) site 4, (c) site 5, (d) site 7, (e) site 8, (f) site 1 - site 4, (g) site 1 - site 8, (h) site 7 - site 4, (i) site-site 8, (j) site 4 - site 8, and (k) site 5 - site 4. Start-up of the LSC facility is identified by the dashed gray line.

### 7.3. Turbidity

Time series of log-transformed turbidity values ($T_n$) and differences calculated for the impact-control pairs are presented in Figure 9. Turbidity values have varied widely, both among sites and over time at individual sites (Figure 9a-e). Effler et al. (2002) found that inorganic tripton, rather than phytoplankton biomass, is the primary regulator of $T_n$ (clarity) in southern Cayuga Lake, and that the higher levels of these constituents (particularly clay minerals) on the shelf are responsible for the generally higher $T_n$ (lower clarity) values observed in this portion of the lake. Elevated turbidity on the shelf is not surprising considering its location with respect to major tributaries and the susceptibility of this shallow area to wind driven resuspension (Figure 1). The extremely high $T_n$ values that accompany major runoff events (Figure 9a-e; see also UFI 1999, 2000, 2001, 2002, 2003, 2005, 2005, 2006) serve to inflate temporal variability and cause mean values to be highly uncertain. Compared to the shelf sites, the deep water sampling location (sites 8) exhibits much lower variability and, in general, substantially lower $T_n$ values (Figure 9e). Paired measurements of log $T_n$ and log TP were positively correlated for all five sites ($r > 0.43$). The coupling between these two variables was particularly strong ($r > 0.6$) for the shelf sites. Temporal variations in log $T_n$ were positively correlated ($0.44 < r < 0.79$) among the four shelf sites. No long-term trends of increasing or decreasing $T_n$ levels are readily apparent from these time series (Figure 9a-e).

No obvious long-term trends were apparent from the time series of impact-control differences (Figure 9f-k). The normality assumption was not met for twelve of the 14 impact-control pairs (Appendix 1). For the reasons stated above, this is not considered a serious issue for the t-tests conducted as part of the BACI analysis. Impact-control differences for log $T_n$ were significantly serially correlated for the following impact-control pairs in the post start-up period: site 7 – site 4, site 7 – site 8, site 4 – site 8 (Appendix 2). The additivity assumption was violated for pairings with site 8 as the control site (Appendix 3). As with TP, this is a result of the disproportionate impact of runoff events on the shelf compared to site 8.

Summary statistics and *p*-values are presented in Table 8 for the six selected impact-control pairs. Differences between the pre and post start-up intervals were not statistically significant (Benjamini & Hochberg $p > 0.67$) for any of the impact-control pairs at the nominal significance level of 0.05. Results computed from non-parametric tests (Mann-Whitney U-test) also failed to indicate statistically significant changes in $T_n$ for any of the control-impact pairings ($p > 0.2$; Appendix 4). The strongest evidence for a post-start-up increase in $T_n$ comes from the site 7 – site 4 ($p = 0.678$) and site 7 – site 8 ($p = 0.678$) pairings. These pairings indicate a 24-25% increase at site 7 relative to sites 4 and 8. Modest increases in $T_n$ were indicated for site 1 relative to control sites, which suggests that the apparent increase in $T_n$ at site 7 did not extend north (i.e., downstream) of the LSC discharge. The DMR – site 8 pairing indicated a small (4%) statistically irrelevant ($p = 0.711$) increase on the shelf relative to the main lake.

Table 8. Results from Welch t-tests comparing log-transformed turbidity for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. Standard deviation is abbreviated $S_d$. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented. Effect size represents the percent change in turbidity at the impact site relative to the control site.

| Impact-Control Pairing | Pre start-up log(NTU) | | Post start-up log(NTU) | | *p*-value | | | Effect size (%) |
|---|---|---|---|---|---|---|---|---|
| | mean | $S_d$ | mean | $S_d$ | raw | Bonferroni | B&H | |
| site 1-site 4 | 0.187 | 0.317 | 0.205 | 0.319 | 0.769 | 1.000 | 0.923 | +4 |
| site 1-site 8 | 0.288 | 0.312 | 0.313 | 0.291 | 0.683 | 1.000 | 0.923 | +6 |
| site 7-site 4 | 0.152 | 0.419 | 0.245 | 0.331 | 0.226 | 1.000 | 0.678 | +24 |
| site 7-site 8 | 0.253 | 0.430 | 0.351 | 0.313 | 0.216 | 1.000 | 0.678 | +25 |
| site 4-site 8 | 0.107 | 0.323 | 0.108 | 0.301 | 0.992 | 1.000 | 0.992 | 0 |
| site 5-site 4 | 0.086 | 0.307 | 0.056 | 0.221 | 0.585 | 1.000 | 0.993 | -7 |
| DMR-site 8 | 0.203 | 0.269 | 0.222 | 0.240 | 0.711 | - | - | +4 |

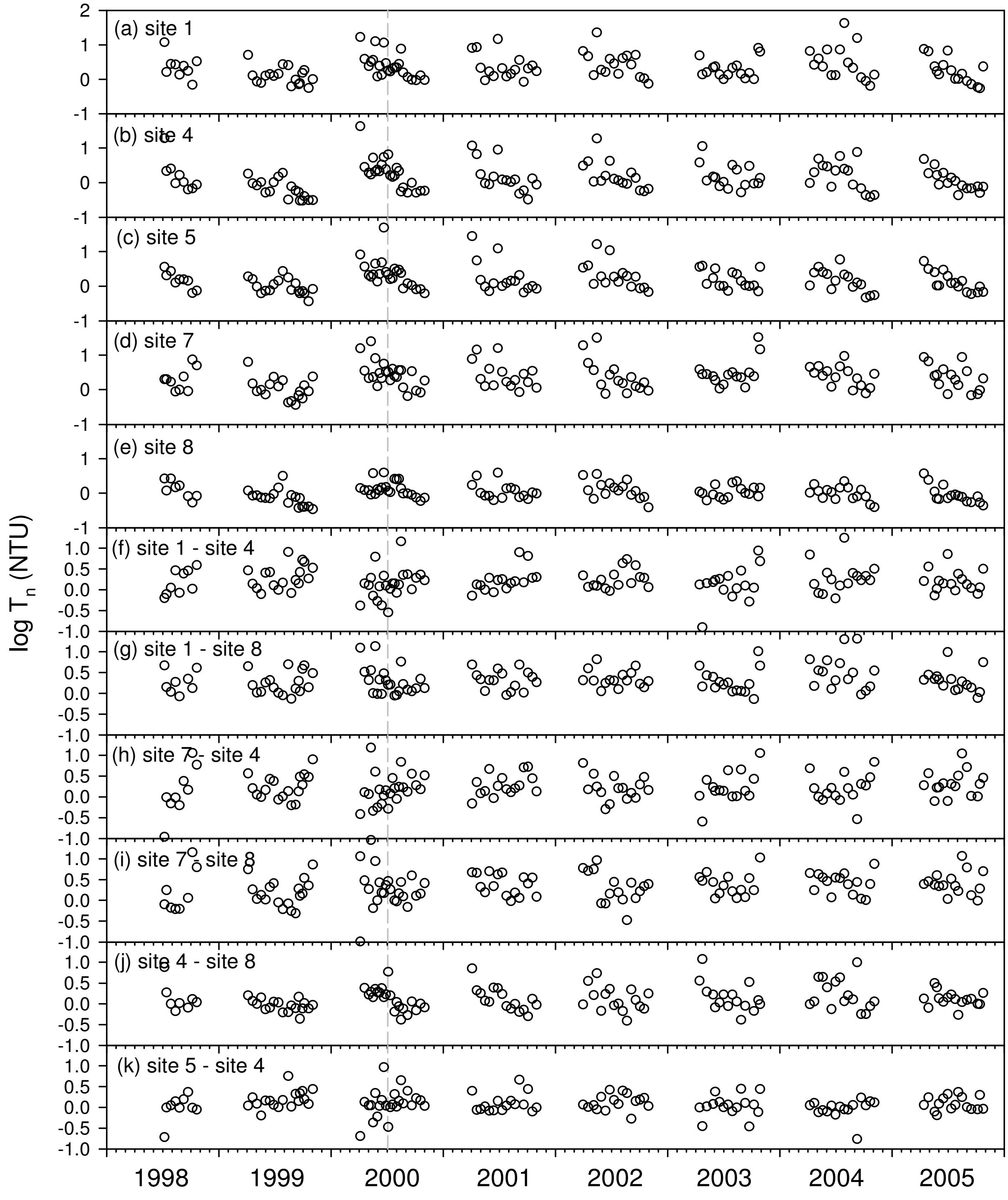


Figure 9. Time series of log $T_n$ concentrations and impact-control differences for the 1998 – 2005 interval: (a) site 1, (b) site 4, (c) site 5, (d) site 7, (e) site 8, (f) site 1 - site 4, (g) site 1 - site 8, (h) site 7 - site 4, (i) site-site 8, (j) site 4 - site 8, and (k) site 5 - site 4. Start-up of the LSC facility is identified by the dashed gray line.

# 8. Limnological Interpretation

## 8.1. Introduction

The primary objective of this report was to determine if statistically significant changes occurred in three water quality parameters (chlorophyll **a**, total phosphorus, turbidity) in the southern portion of Cayuga Lake coincident in time with start-up of Cornell's Lake Source Cooling (LSC) facility. Quantitative decision criteria, determined by appropriate statistical models and techniques are widely desired. However, the effective design, analysis, and interpretation of environmental monitoring studies are formidable tasks fraught with difficult statistical and limnological judgments. Examples of difficult judgments encountered in the BACI analysis presented here include the selection of appropriate control-impact comparisons and the choice of a suitable balance between the costs of Type I and Type II errors. Furthermore, statistical impact analysis cannot, by itself, establish definitively the cause of a documented change in water quality. Discovering the cause of any observed water quality changes in the southern end of Cayuga Lake is particularly complicated because of the potential for simultaneous changes in multiple drivers not associated with LSC operation. Natural variation in meteorological conditions, improvements in treatment at domestic wastewater treatment plants, and the uncertain effects of zebra mussel populations are potential confounding variables.

The issues encountered here are characteristic of observational studies and are not unique to the BACI analysis for the LSC facility. Researchers have advocated a variety of alternate methods for interpreting data from such studies, including the use of graphical presentation, the informal use of statistical tests, expert judgment, and common sense (Stewart-Oaten 1995, Murtaugh 2002). Ultimately, responsible assessment of potential water quality impacts associated with LSC operation requires the integration of limnological and statistical analyses. The purpose of this section is to focus on limnological interpretations rather than *p*-values derived from hypothesis tests of questionable validity. The following issues are considered: the relative contributions of LSC, wastewater treatment plants, and tributaries to material loading to southern Cayuga Lake; the systematic increase in soluble reactive phosphorus concentrations (SRP) in the hypolimnion of Cayuga Lake; hydraulic loading and residence time in the southern shelf of Cayuga Lake; biological significance of observed water quality changes; and potential effects of zebra mussel populations.

## 8.2. Material Loading Contributions to the Southern Shelf of Cayuga Lake

The potential for operation of the LSC facility to effect water quality conditions in the southern shelf of Cayuga Lake is regulated to a large extent by material loading contributions. The primary constituent of concern is TP, as Chl and $T_n$ levels were routinely much lower in the hypolimnion of Cayuga Lake (i.e., LSC effluent) than in the receiving waters of the southern shelf (UFI 1999-2006). Although TP concentrations in the LSC effluent have been routinely lower than those of the southern shelf, soluble reactive phosphorus concentrations (SRP) have been a factor of 2 to 5 higher in the LSC discharge. SRP is a component of total dissolved phosphorus (TDP) that is usually assumed to be immediately available to support phytoplankton growth. Thus, the LSC discharge has the potential to increase phytoplankton biomass (i.e., Chl) on the southern shelf. This potential impact was acknowledged in the **Environmental Impact Statement** (Stearns and Wheler 1997) prepared for the LSC facility, which stated "The estimated potential cumulative increase in concentration of chlorophyll **a** is approximately 2.5 µg/L (range 1.25 to 5 µg/L) over the June to October period. Even if the potential increase in phytoplankton biomass is restricted to the region of the outfall, the increase in chlorophyll **a** would be very small."

The potential for phosphorus loading from LSC to affect phytoplankton biomass is small relative to phosphorus inputs from wastewater treatment plants and tributaries. Following upgrades in treatment at the Ithaca Area and Cayuga Heights WWTPs and under the low flow conditions of 2007, LSC represented about 7.5% of the phosphorus loading to the southern shelf. The maximum contribution of LSC to phosphorus loading on a monthly basis was 13% during August of 2006 and 2007 (Cornell University 2008). The tributaries have been the dominant source of phosphorus and turbidity to shelf, particularly during high runoff years (Effler et al. 2002, 2008). Despite the phosphorus loading received from local sources, summer average Chl concentrations have not been substantially higher on the shelf compared to bounding pelagic waters because of the high flushing rate of the shelf promoted by mixing with pelagic waters (UFI 2008).

### 8.3. Increases in Soluble Reactive Phosphorus in the Hypolimnion of Cayuga Lake

The relative contribution of LSC to phosphorus loading to the shelf has increased since 2005 as a result of improved treatment at WWTPs and increased loading from LSC. Increased TP loading to the shelf from the LSC effluent during 2004-2007 was attributable to an increase in effluent TP concentrations relative to 2000-2003 (Cornell University 2008). The average TP concentration in the LSC effluent was 29% higher during 2004-2007 than during 2000-2003. This increase was caused by a 78% increase in SRP concentrations in the hypolimnetic source waters for LSC (Figure 10). On an annual average basis, SRP concentrations increased from 4.2 µg/L in 2002 to 9.6 µg/L in 2007. The annual average SRP concentration decreased somewhat in 2008 to 8.6 µg/L. The SRP concentrations observed to date continue to be lower than the conservative value of 20 µg/L adopted in the **Environmental Impact Statement** for projection of potential impacts related to the LSC facility (Stearns and Wheler 1997). Because we lack a causal mechanism for these increases in SRP, it is inappropriate to extrapolate historic trends into the future. Hypolimnetic SRP measurements from 1994-1996 (Stearns and Wheler 1997) suggest levels even higher than those observed during 2004-2007 (Figure 10). This supports the position that the 2004-2007 SRP increase is part of a longer-term temporal pattern of increases and decreases that are unrelated to LSC operation. The timing of this increase (3-4 years after LSC startup) is also inconsistent with an LSC-related impact.

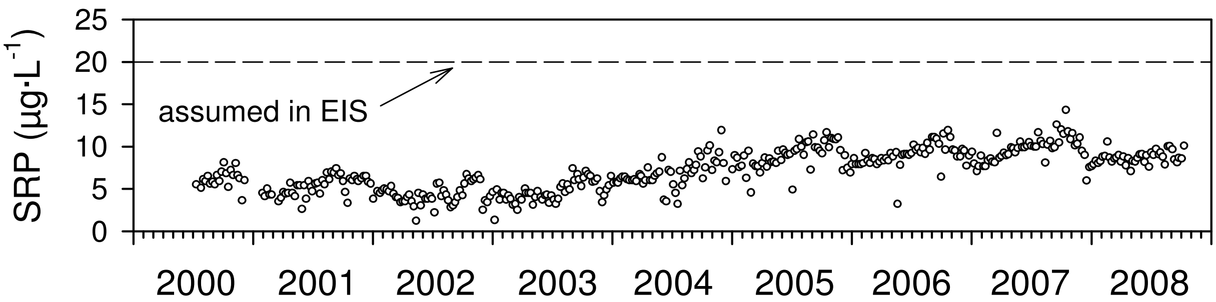


Figure 10. Time series of SRP concentrations measured weekly in the LSC effluent for the 2000 – 2008 interval. LSC effluent concentrations are representative of the hypolimnetic source water (UFI 2007). The dashed line represents the SRP concentration of 20 µg/L used in the **Environmental Impact Statement** to assess potential impacts of the LSC facility (Stearns and Wheler 1997).

An unambiguous explanation for the apparent increases in TP, SRP, and $T_n$ in the lake's hypolimnion has not been identified. In large deep lakes such as Cayuga, changes in hypolimnetic water quality are expected to occur over long time scales, on the order of decades rather than years. Temporary increases in $T_n$ and the particulate fraction of TP in bottom waters can be caused by plunging turbid inflows and internal waves or seiches. However, hypolimnetic SRP levels are generally considered to reflect lake-wide metabolism rather than local effects. Soluble reactive phosphorus is produced during microbial decomposition of organic matter and often accumulates in the hypolimnia of stratifying lakes during summer. Increases in primary production (phytoplankton growth) and subsequent decomposition could cause increases in SRP levels, but noteworthy increases in chlorophyll concentrations (phytoplankton biomass) have not been observed. Longer intervals of thermal stratification, increased hypolimnetic temperatures or depletion of dissolved oxygen could also cause higher concentrations of SRP in the bottom waters. Such changes have not been observed. The apparent increase in hypolimnetic SRP concentrations may represent a short-term anomaly rather than a long-term trend. Regardless, this phenomenon has the potential to affect phosphorus loading to the shelf from the LSC effluent and should be diligently monitored in the future in order to discern the permanence and significance of these changes.

### 8.4. Hydraulic Loading and Residence Time in the Southern Shelf of Cayuga Lake

Substantial interannual variations in hydrologic loading to the southern shelf occurred over the 1998-2007 interval, driven primarily by natural variations in runoff from tributaries (UFI 2008). These variations in runoff have caused conspicuous water quality signatures in the shelf, as manifested in higher levels of TP, $T_n$, and Chl in high runoff years (Cornell University 2008). It is noteworthy that 1999, the only complete year of pre start-up data, was the lowest runoff year of the 1998 – 2005 interval. Runoff was also relatively low in 2001 and 2005, and relatively high in 2000, 2002, 2003, and 2004. The average inflow rate to the shelf for the April-October interval increased approximately 11% following start-up of the LSC facility in 2000 (UFI 2008). However, the LSC effluent contributes more than 50% of the inflow to the shelf during low runoff summertime periods (UFI 2003). UFI (2008) reported that the shelf is flushed rapidly, approximately once per day during the May – October interval. The high flushing rate of the shelf is caused primarily by exchange with water from the main lake. Wind-induced upwelling events occur commonly on the shelf and promote exchange with metalimnetic and hypolimnetic waters of the pelagic zone. The rapid exchange of shelf waters with those of the main lake discourages the development of gradients in Chl from local phosphorus loading to the shelf.

Preliminary numerical simulations conducted with an unvalidated three dimensional (3-D) free surface hydrodynamic model suggest that the exchange of water in and out of the shelf is affected by both inflow events and the internal wave oscillations (personal communication with E. Cowen, Cornell University). Preliminary model results also indicate that transport in-out of the shelf is lower in magnitude during non-inflow events at site 7 than at sites 1 and 4. The relative isolation of site 7 from the pelagic waters may contribute to the higher levels of Chl, TP, and $T_n$ at this location. However, water quality conditions at site 7 are also likely to be influenced by the proximity of this site to WWTP discharges. The effects of the LSC effluent on flow patterns in the shelf have not been documented, although it has undoubtedly increased flushing rates on the shelf during low runoff periods.

### 8.5. Biological Significance of Observed Water Quality Changes

The strongest evidence for water quality changes following LSC start-up was a 27% increase in Chl at site 7 relative to site 4 (Table 6) and a 24% increase in $T_n$ at site 7 relative to site 8 (Table 8). Although the magnitude of these effects was found to be statistically insignificant, it is important to evaluate the potential biological significance of these changes. Admittedly, the assessment of biological significance is no less complex or subjective than the assessment of statistical significance. The 27% increase in Chl at site 7 relative to site 4 is equivalent to an additional increase of approximately 1 μg/L at site 7. An increase of this magnitude would likely be deemed noteworthy in the pelagic waters of an oligotrophic lake, but probably not in the littoral zone of a heavily impacted mesotrophic lake. The 27% increase in $T_n$ at site 7 relative to site 8 is equivalent to an additional increase of approximately 1 NTU at site 7. Again, an increase of this magnitude might be considered noteworthy in the pelagic waters of an oligotrophic lake, but it would probably not be a major cause for concern in the anthropogenically perturbed littoral zone of a mesotrophic lake. In other words, variations of a similar magnitude are not uncommon in lakes.

The apparent large scale increase in Chl depicted in Figure 2 is perhaps more noteworthy, and an obvious cause for this change is not apparent. However, it is important to note that higher Chl concentrations were measured on the shelf during 1994 – 1996. Chl concentrations on the shelf decreased 35% from 1994 – 1996 to 1998 – 1999 in the absence of any documented perturbation. The recent increase in Chl on the shelf has resulted in a rebound to levels observed during 1994-1996. The large spatial extent of this change is not consistent with the type of localized impact that might be expected as a result of LSC operation. It is likely that the association between LSC start-up and this increase in Chl is coincidental rather than causal.

### 8.6. Potential Effects of Zebra Mussel Populations

Dense populations of zebra mussels have been demonstrated to cause major changes in common measures of water quality associated with various aspects of their metabolism. In particular, conspicuous reductions in Chl and increases in clarity have been documented for a number of aquatic ecosystems in North America associated with the filter feeding of this exotic invader (e.g., Effler et al. 1996, Caraco et al. 1997). Effects on phosphorus cycling are less straightforward; e.g., decreases in the particulate fraction may result from filter feeding while excretion causes increases in the dissolved fraction (Arnott and Vanni 1996, Effler et al. 1997). Hecky et al. (2003) developed a conceptual model called the nearshore shunt that describes the redirection of nutrient and energy flow caused by dense zebra mussel populations. The nearshore shunt has been linked to degraded water quality in nearshore zones in many parts of the Laurentian Great Lakes, including the proliferation of the benthic filamentous alga *Cladophora*. Improved water clarity associated with the filter feeding of zebra mussels resulted in increased macrophyte habitat, species richness, frequency of occurrence, and community composition in nearby Oneida Lake (Zhu et al. 2006). Dense macrophyte beds can be a nuisance to boaters and are often considered a symptom of degraded water quality. Shifts in the density of Cayuga Lake's zebra mussel population could cause larger changes in trophic state indicators than any associated with management actions to reduce inputs of phosphorus.

## 9. Summary

A Before-After-Control-Impact (BACI) design was applied to in-lake monitoring data collected over the 1998 – 2005 interval for three water quality parameters (chlorophyll **a**, total phosphorus, turbidity) to determine whether statistically significant changes occurred in the southern portion of Cayuga Lake coincident in time with start-up of Cornell's Lake Source Cooling (LSC) facility. The BACI analysis included data from six in-lake sites, and seven impact-control pairings were considered. Statistical significance was assessed using *p*-values adjusted for multiple comparisons according to the method of Benjamini and Hochberg (1995), which is the most powerful of the widely used methods to control Type I errors or the occurrence of false positives. No statistically significant ($\alpha = 0.05$) impacts were found for the three water quality parameters on any of the seven impact-control pairs. Nonparametric test results (Mann-Whitney U-test; Appendix 4) also failed to reveal statistically significant ($\alpha = 0.05$) changes in water quality.

The strongest evidence for a water quality impact coincident in time with LSC start-up was indicated by increases of approximately 25% in chlorophyll **a** concentrations (Chl) and turbidity levels ($T_n$) at site 7 relative to sites 4 and 8. The magnitude of these increases (~ 1 µg/L, ~ 1 NTU) was small and changes of this size are common in moderately productive lakes. The apparent increase in Chl at site 7 was less than the potential cumulative increase of 1.25 to 5 µg/L estimated in the **Environmental Impact Statement** (Stearns and Wheler 1997). The absence of similar increases in total phosphorus concentrations (TP) complicates interpretation of these results because both phytoplankton and the particles that cause turbidity contain phosphorus. Interpretations are further confounded by the absence of substantial increases in Chl or $T_n$ at site 1, perhaps the site most likely to be impacted by the LSC discharge. Preliminary hydrodynamic modeling suggests transport in-out of the shelf is lower in magnitude during non-inflow events at site 7 than at sites 1 and 4. The relative isolation of site 7 from the pelagic waters may contribute to the higher levels of Chl, TP, and $T_n$ at this location. The effects of the LSC effluent on flow patterns in the shelf have not been documented, although it has undoubtedly increased flushing rates on the shelf during low runoff periods. Average Chl concentrations increased in the post LSC start-up period at eight of the nine sites monitored in this study. Because of the relatively small contributions of the LSC discharge to TP loading on the southern shelf (< 10%), it is unlikely that LSC was the direct cause of these increases.

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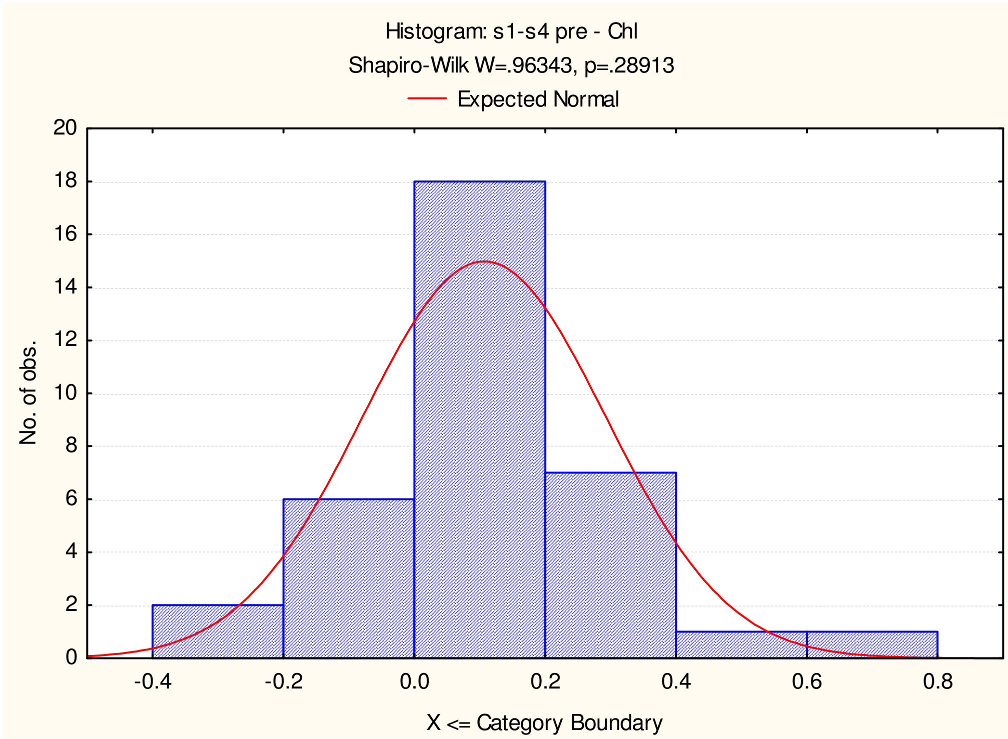
# Appendix 1 - Normality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0.6
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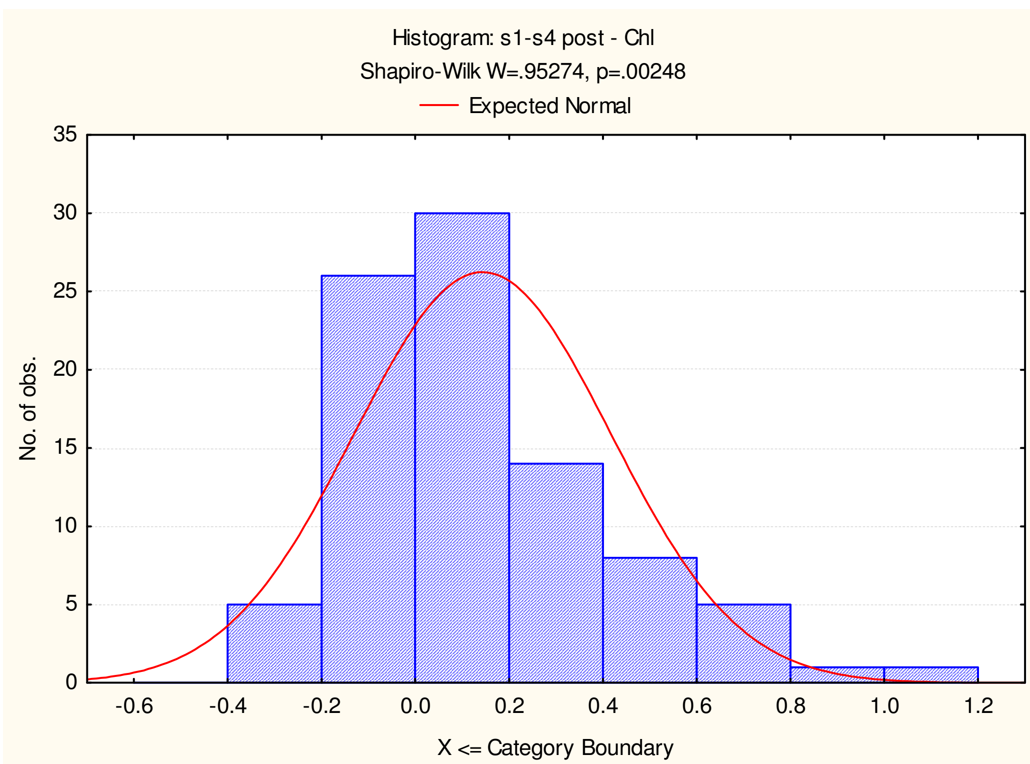
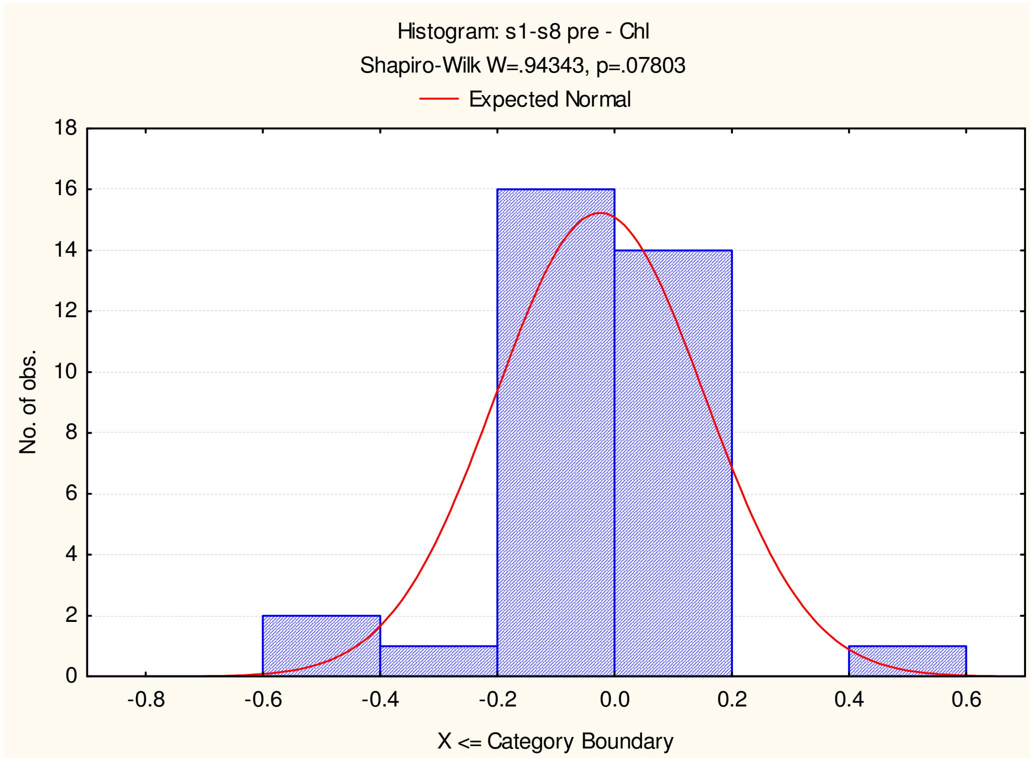
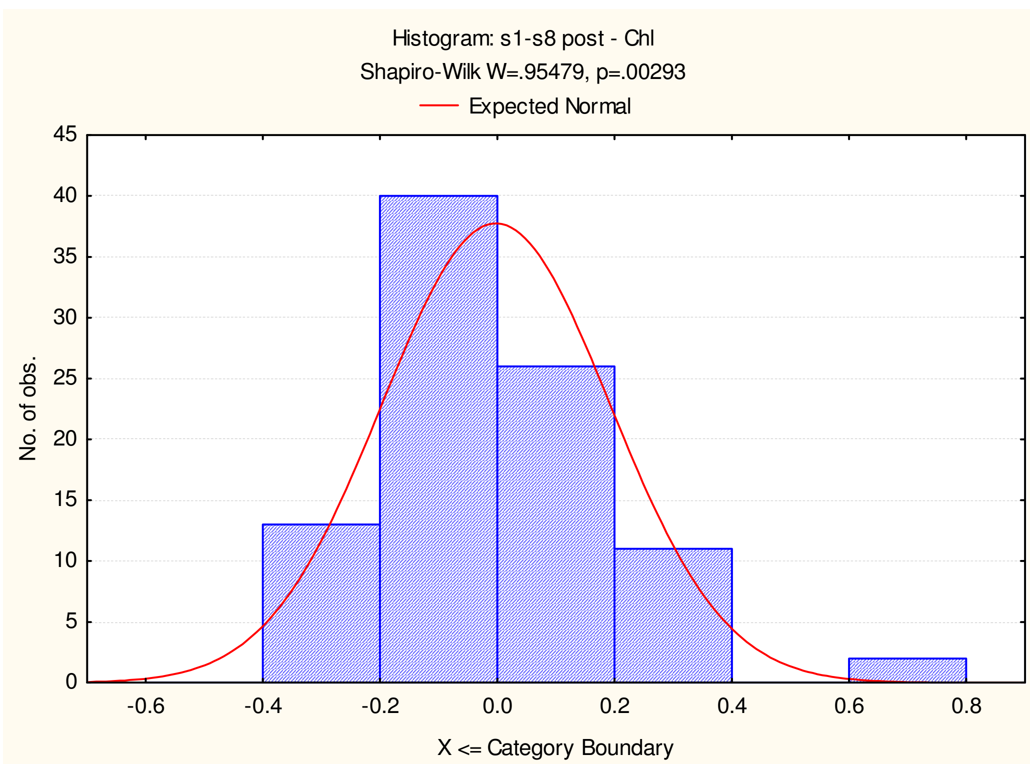
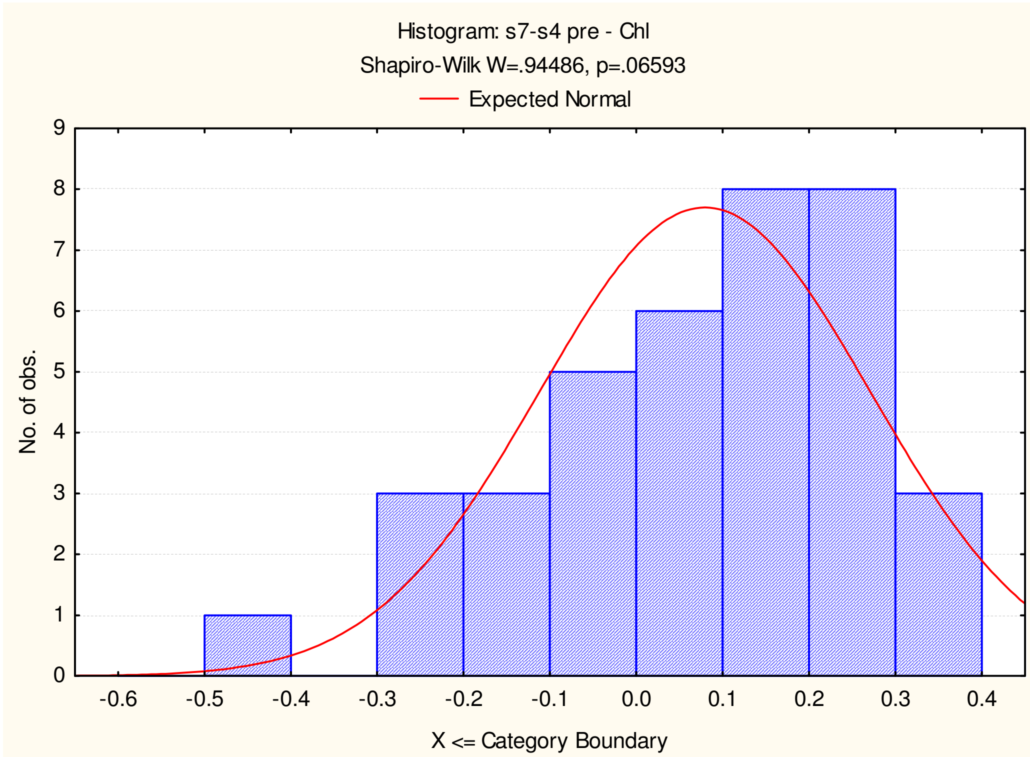
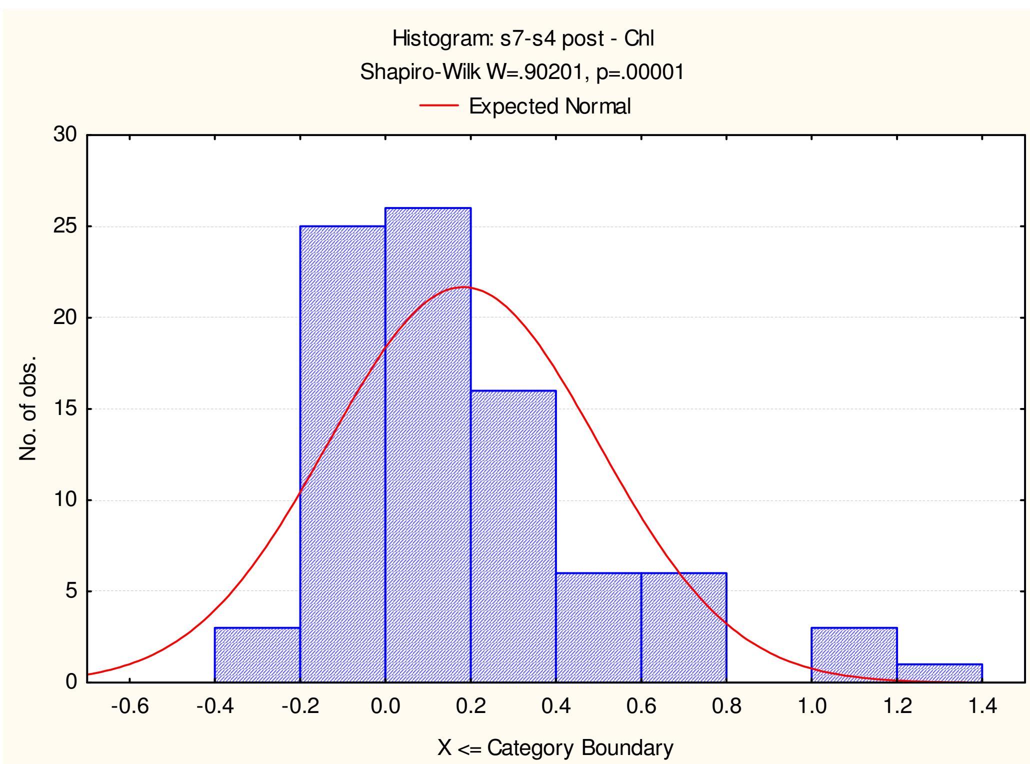
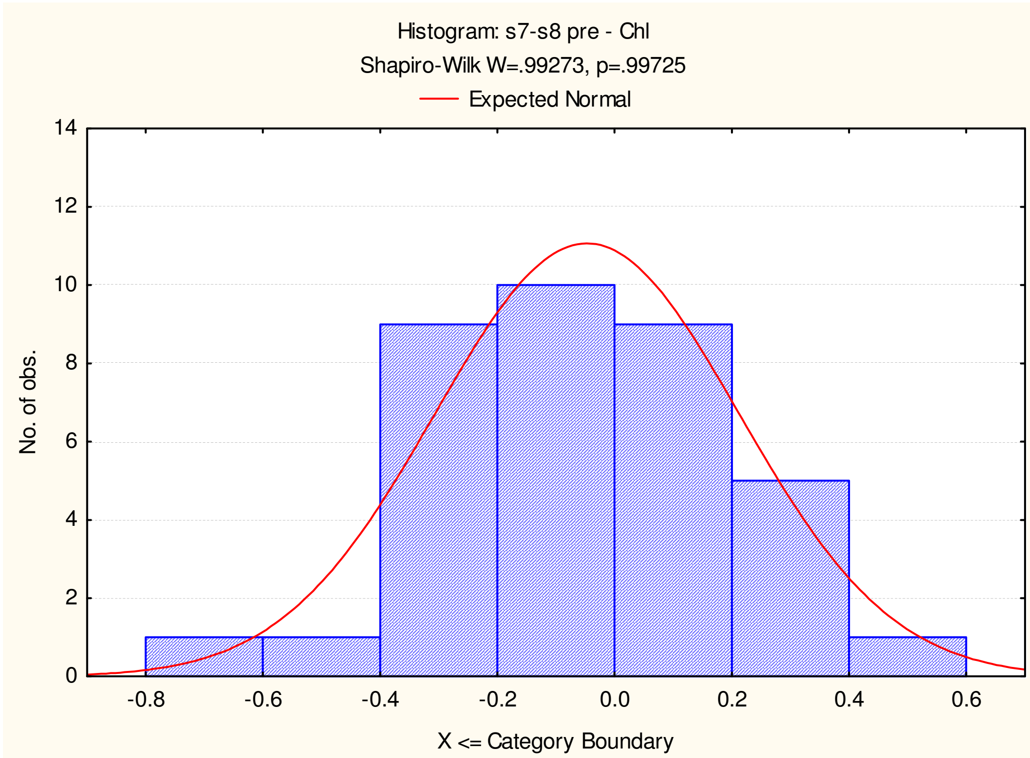

Histogram: s1-s4 post - Chl
Shapiro-Wilk W=.95274, p=.00248
Expected Normal
No. of obs.
X <= Category Boundary
35
30
25
20
15
10
5
0
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2

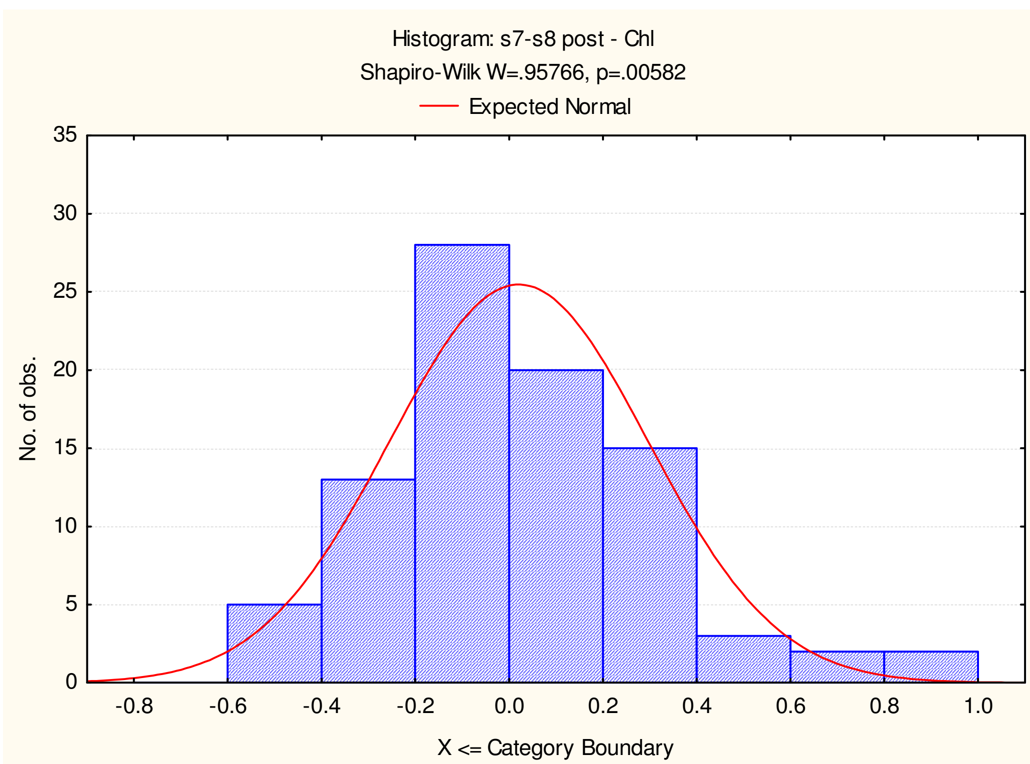

Histogram: s1-s8 pre - Chl
Shapiro-Wilk W=.94343, p=.07803
Expected Normal
No. of obs.
18
16
14
12
10
8
6
4
2
0
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
X <= Category Boundary

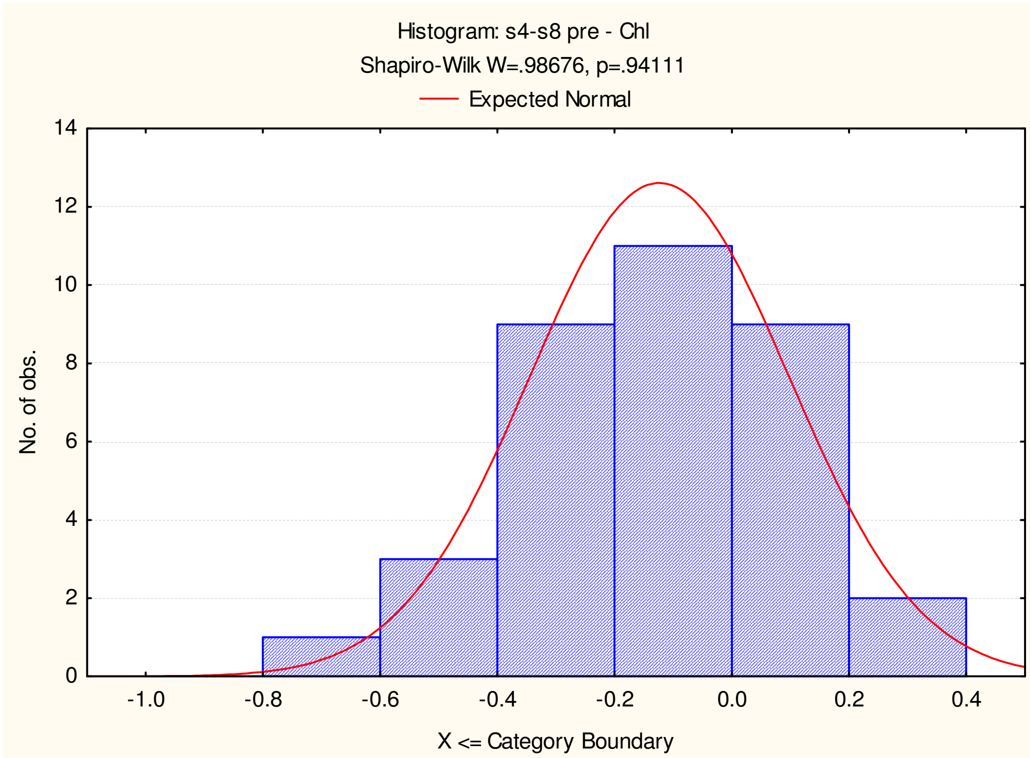

Histogram: s1-s8 post - Chl
Shapiro-Wilk W=.95479, p=.00293
Expected Normal
No. of obs.
45
40
35
30
25
20
15
10
5
0
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
X <= Category Boundary

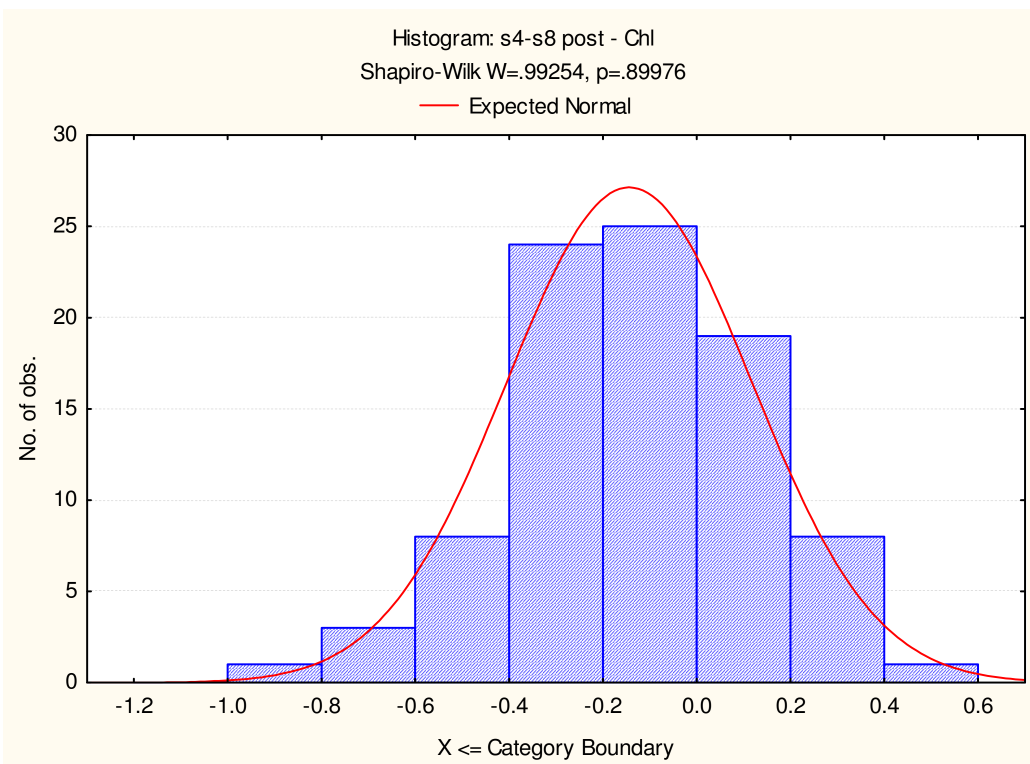

Histogram: s7-s4 pre - Chl
Shapiro-Wilk W=.94486, p=.06593
Expected Normal
No. of obs.
0
1
2
3
4
5
6
7
8
9
-0.6
-0.5
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
X <= Category Boundary

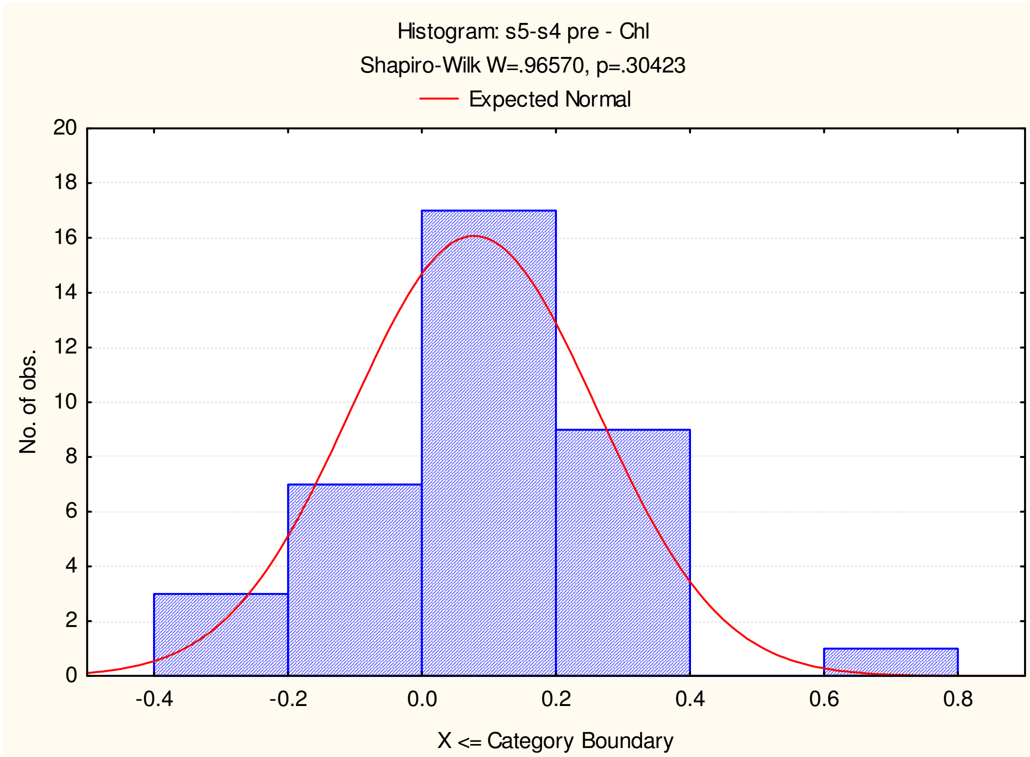

Histogram: s7-s4 post - Chl
Shapiro-Wilk W=.90201, p=.00001
Expected Normal
No. of obs.
0
5
10
15
20
25
30
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
X <= Category Boundary

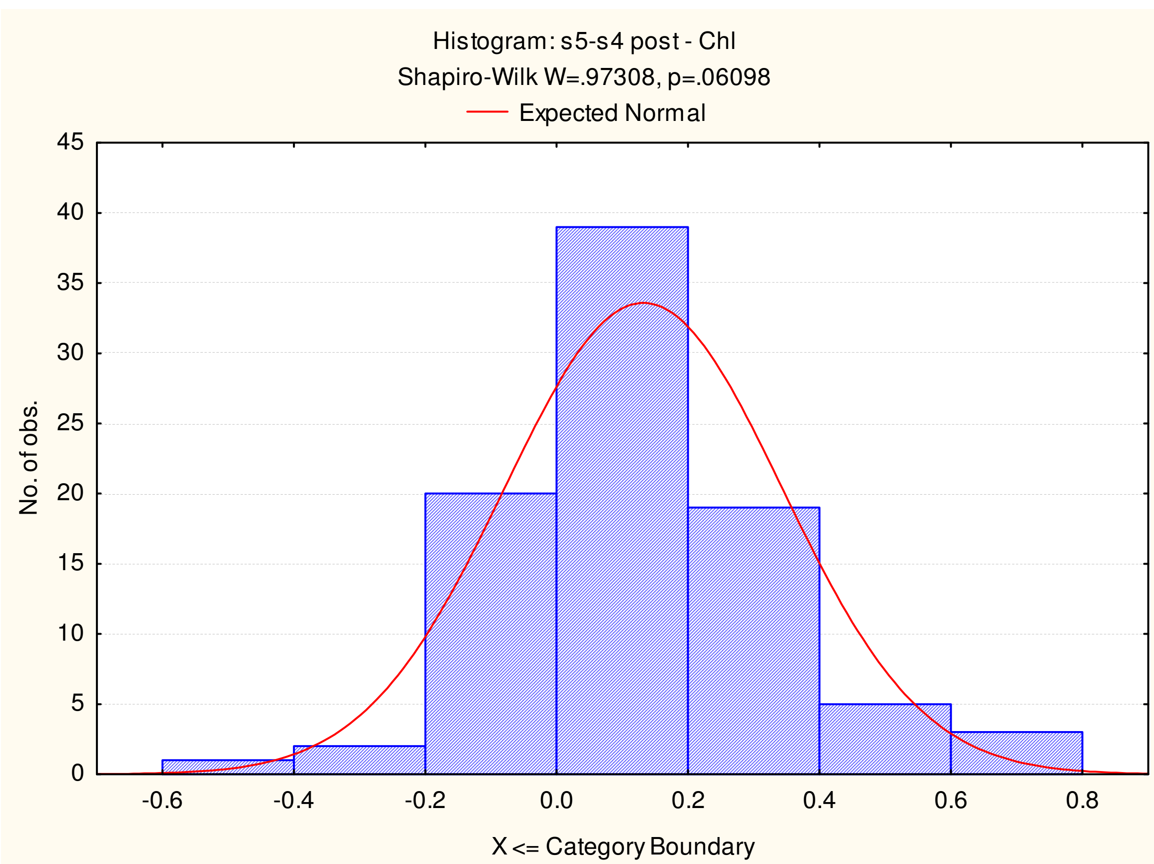

Histogram: s7-s8 pre - Chl
Shapiro-Wilk W=.99273, p=.99725
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
X <= Category Boundary

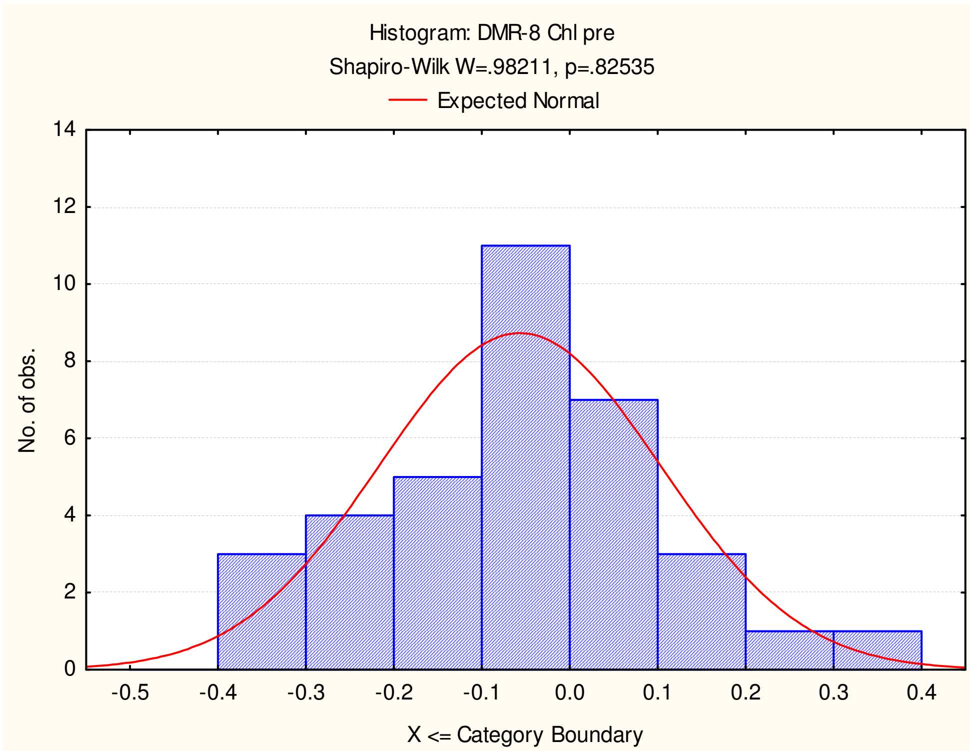

Histogram: s7-s8 post - Chl
Shapiro-Wilk W=.95766, p=.00582
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
X <= Category Boundary

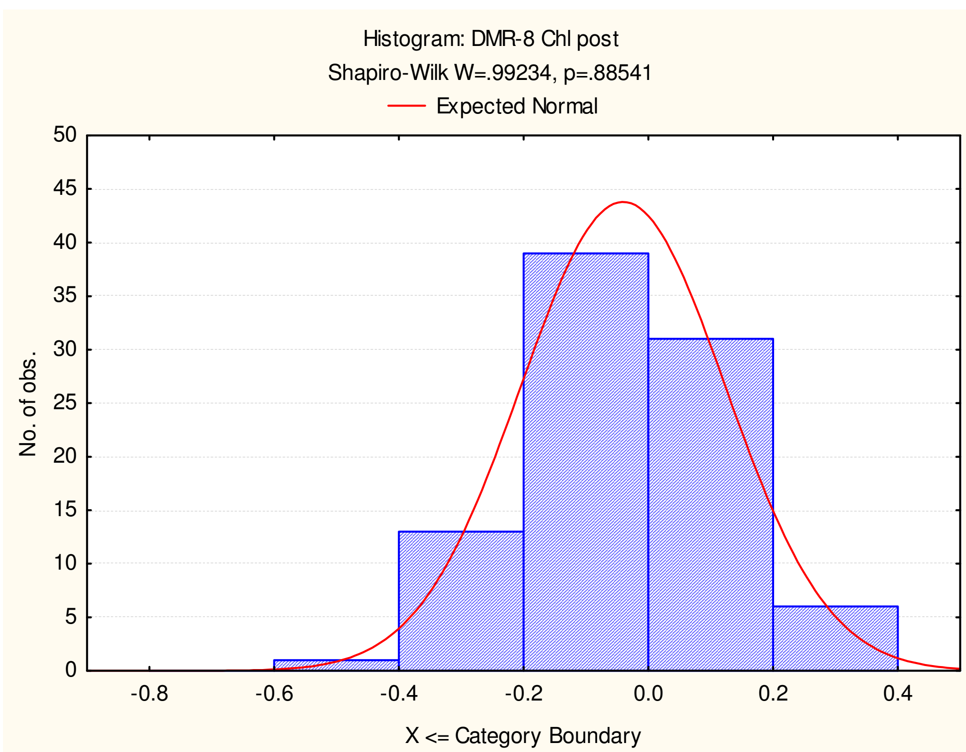

Histogram: s4-s8 pre - Chl
Shapiro-Wilk W=.98676, p=.94111
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
-1.0
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
X <= Category Boundary

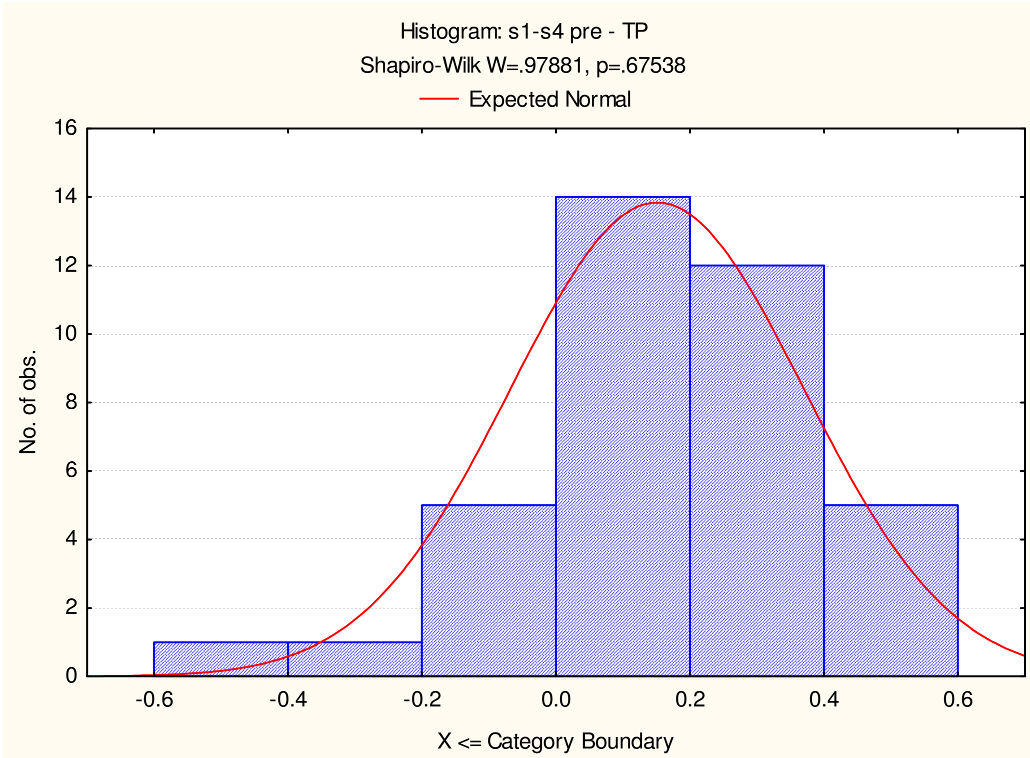

Histogram: s4-s8 post - Chl
Shapiro-Wilk W=.99254, p=.89976
Expected Normal
No. of obs.
0
5
10
15
20
25
30
-1.2
-1.0
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
X <= Category Boundary

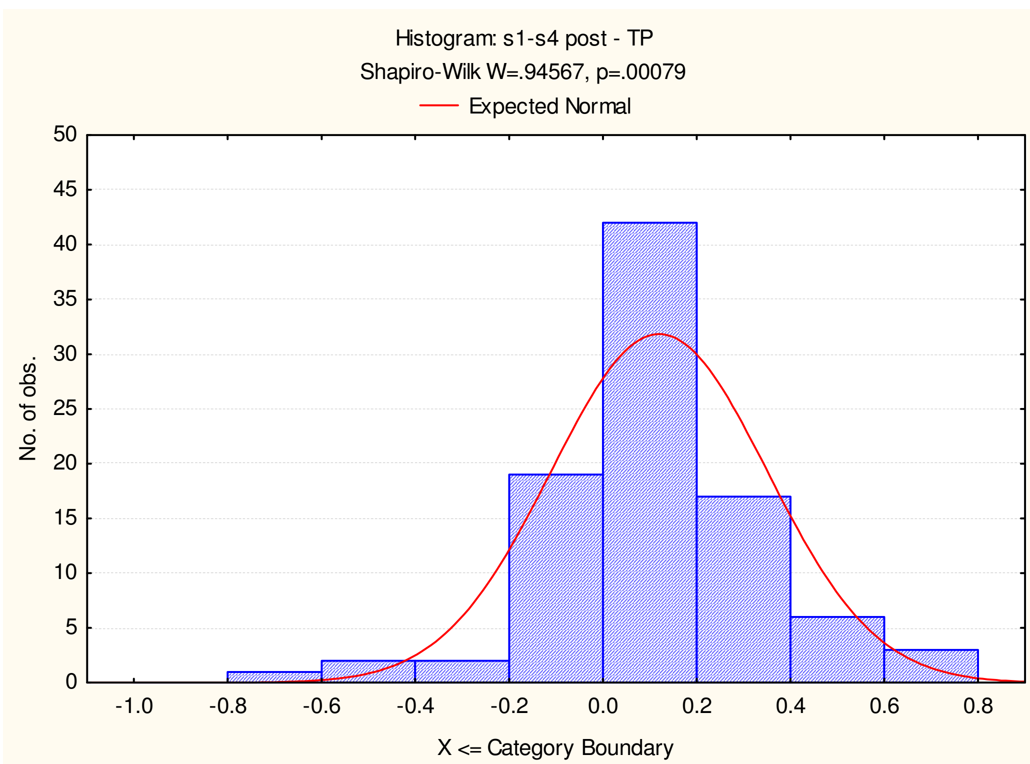

Histogram: s5-s4 pre - Chl
Shapiro-Wilk W=.96570, p=.30423
Expected Normal
No. of obs.
X <= Category Boundary
0
2
4
6
8
10
12
14
16
18
20
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8

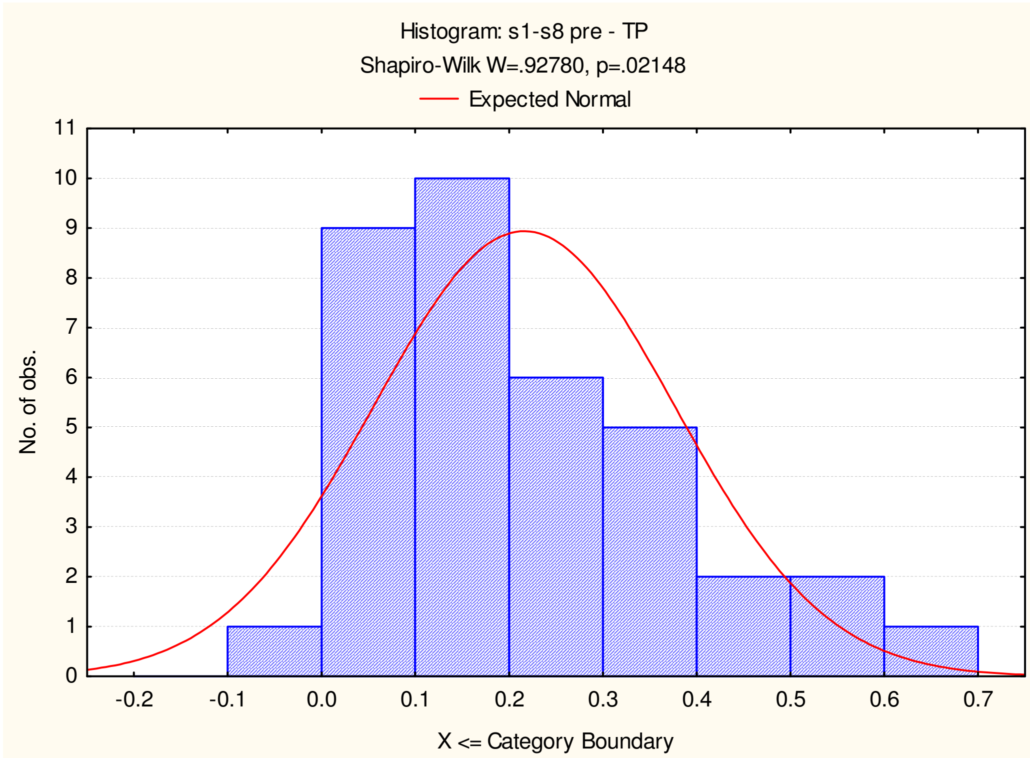

Histogram: s5-s4 post - Chl
Shapiro-Wilk W=.97308, p=.06098
Expected Normal
No. of obs.
X <= Category Boundary
0
5
10
15
20
25
30
35
40
45
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8

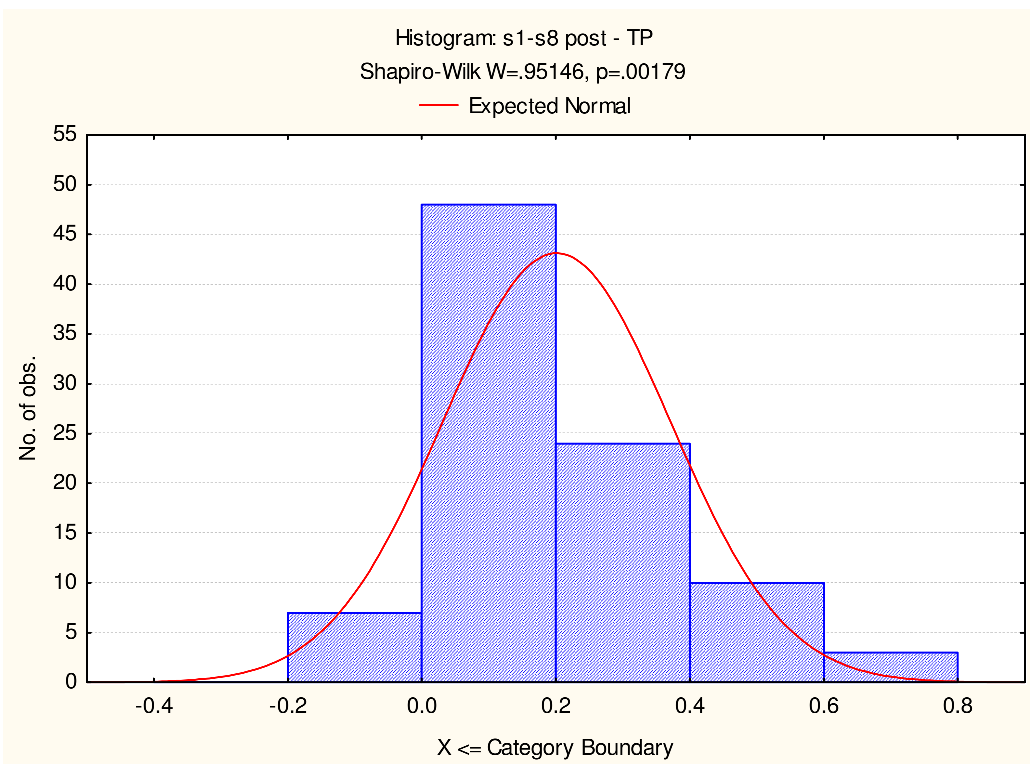

Histogram: DMR-8 Chl pre
Shapiro-Wilk W=.98211, p=.82535
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
-0.5
-0.4
-0.3
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
X <= Category Boundary

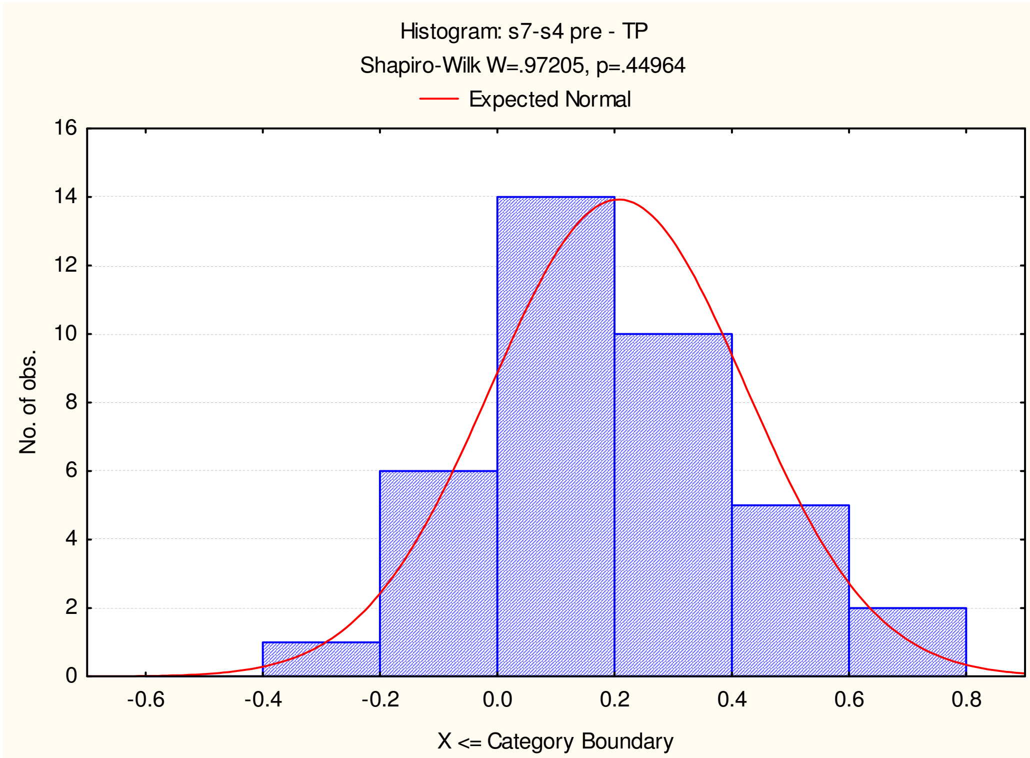

Histogram: DMR-8 Chl post
Shapiro-Wilk W=.99234, p=.88541
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
40
45
50
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
X <= Category Boundary

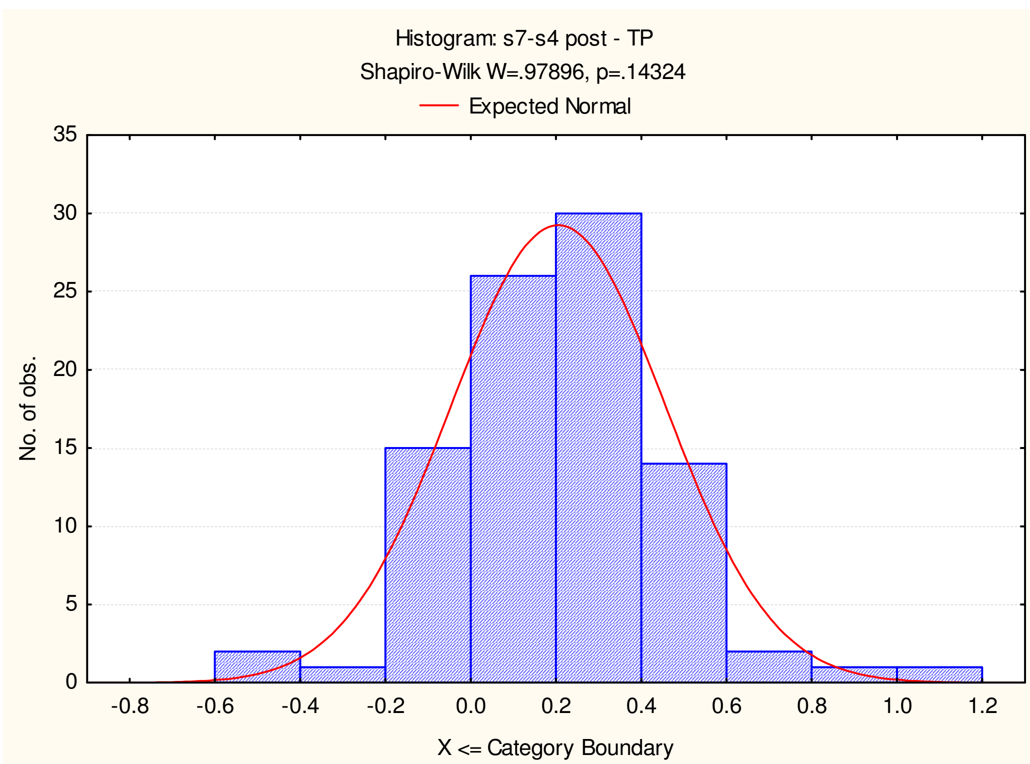

Histogram: s1-s4 pre - TP
Shapiro-Wilk W=.97881, p=.67538
Expected Normal
No. of obs.
16
14
12
10
8
6
4
2
0
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
X <= Category Boundary

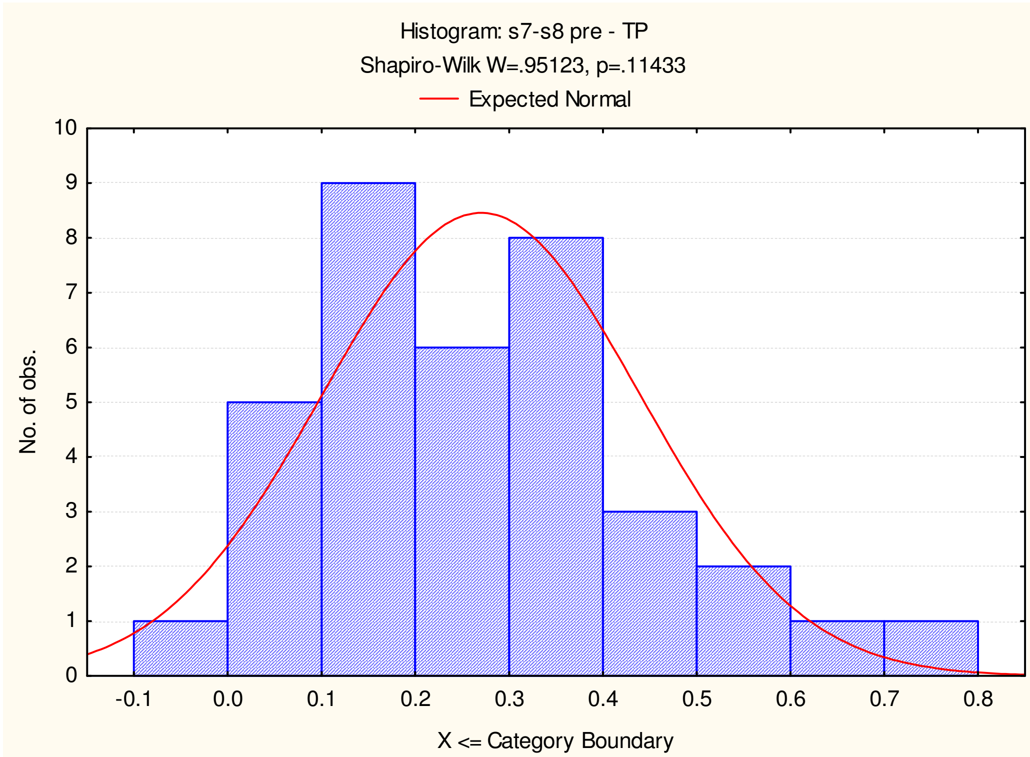

Histogram: s1-s4 post - TP
Shapiro-Wilk W=.94567, p=.00079
Expected Normal
No. of obs.
50
45
40
35
30
25
20
15
10
5
0
-1.0
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
X <= Category Boundary

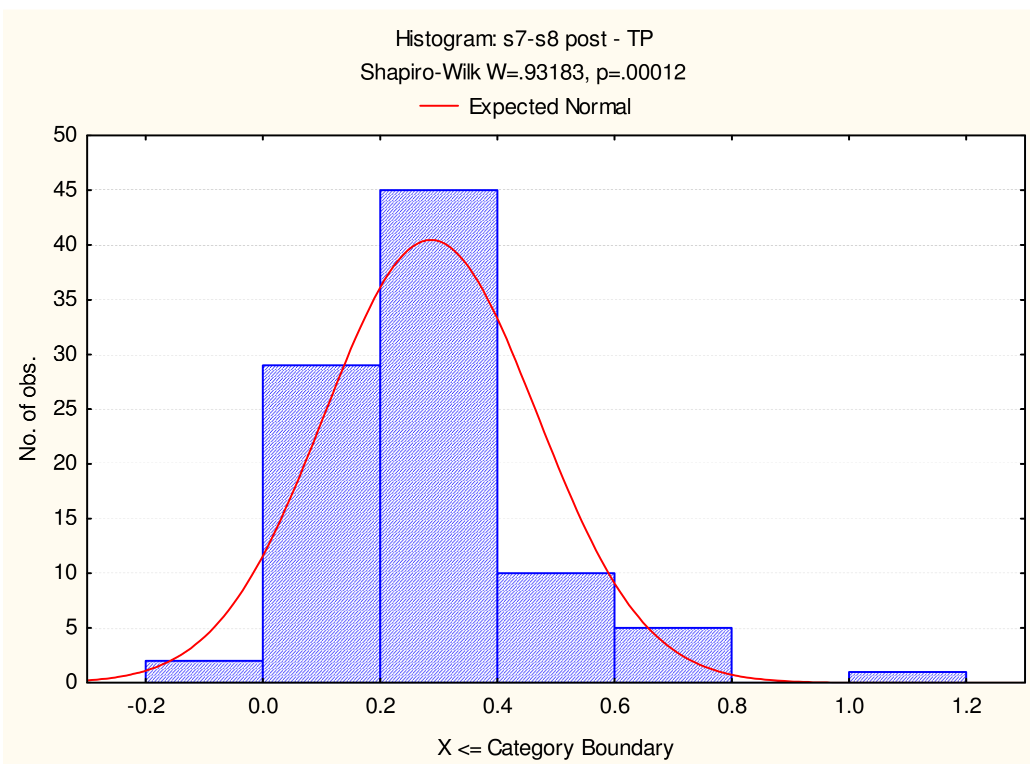

Histogram: s1-s8 pre - TP
Shapiro-Wilk W=.92780, p=.02148
Expected Normal
No. of obs.
0
1
2
3
4
5
6
7
8
9
10
11
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
X <= Category Boundary

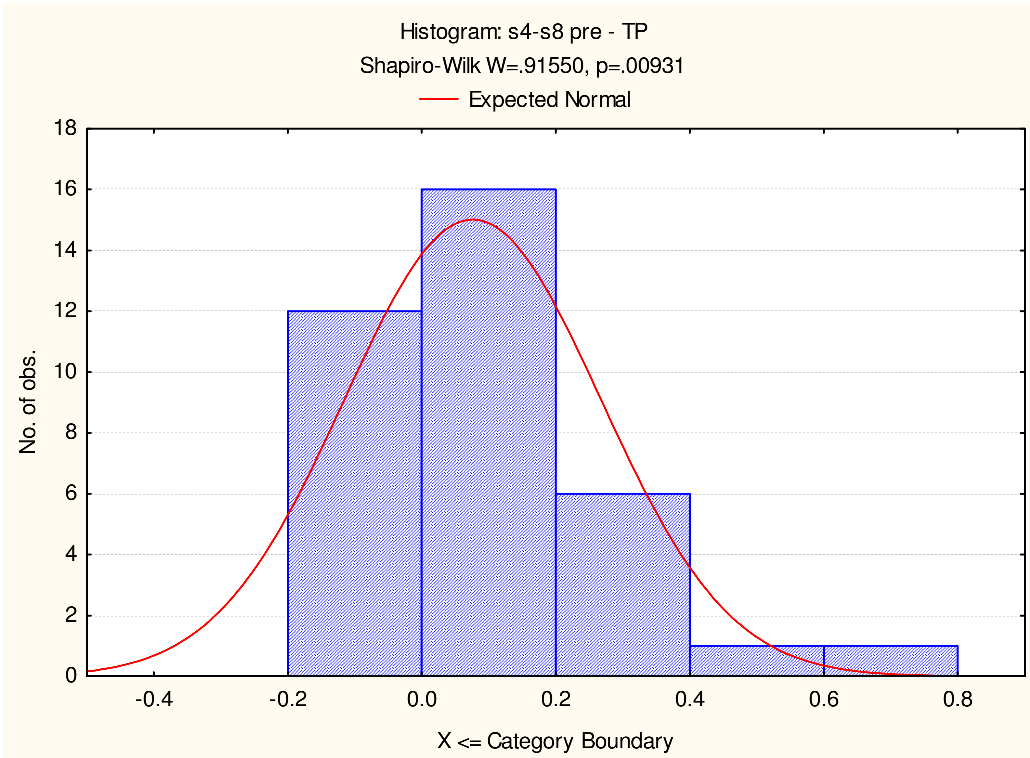

Histogram: s1-s8 post - TP
Shapiro-Wilk W=.95146, p=.00179
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
40
45
50
55
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
X <= Category Boundary

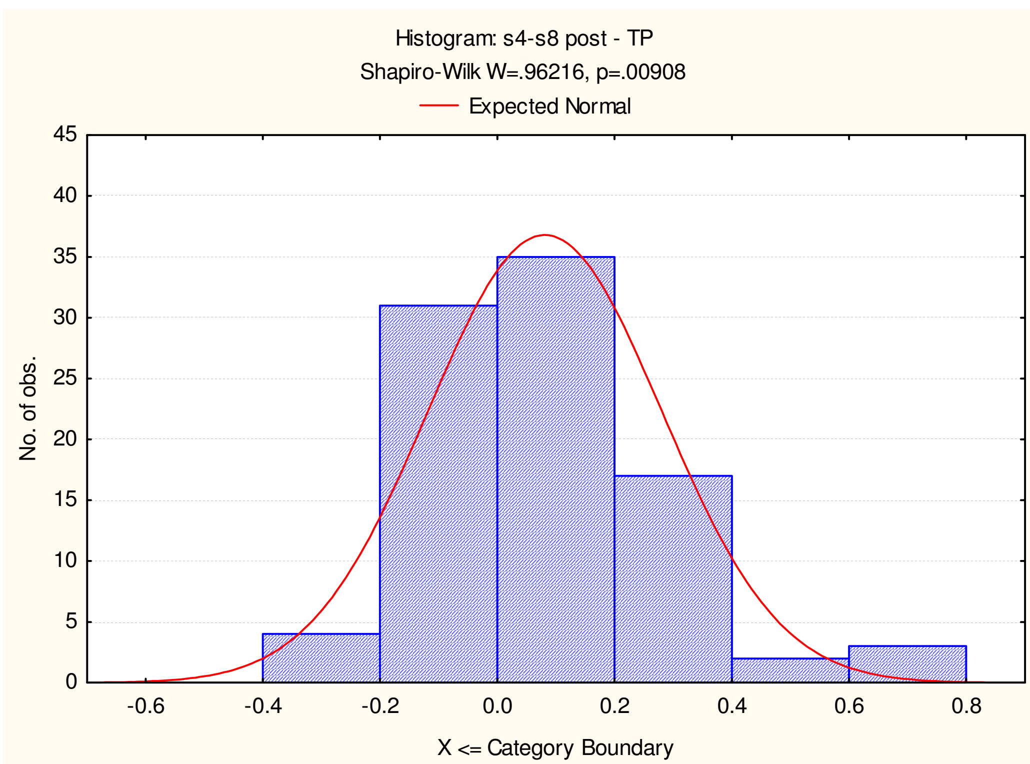

Histogram: s7-s4 pre - TP
Shapiro-Wilk W=.97205, p=.44964
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
16
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
X <= Category Boundary

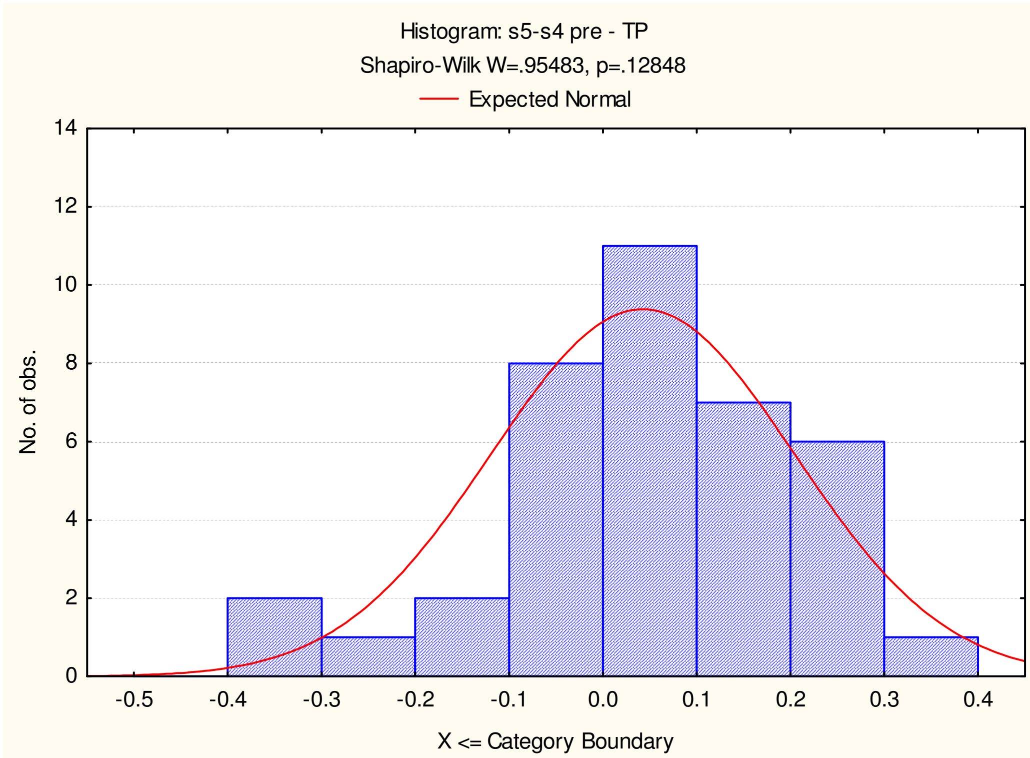

Histogram: s7-s4 post - TP
Shapiro-Wilk W=.97896, p=.14324
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
X <= Category Boundary

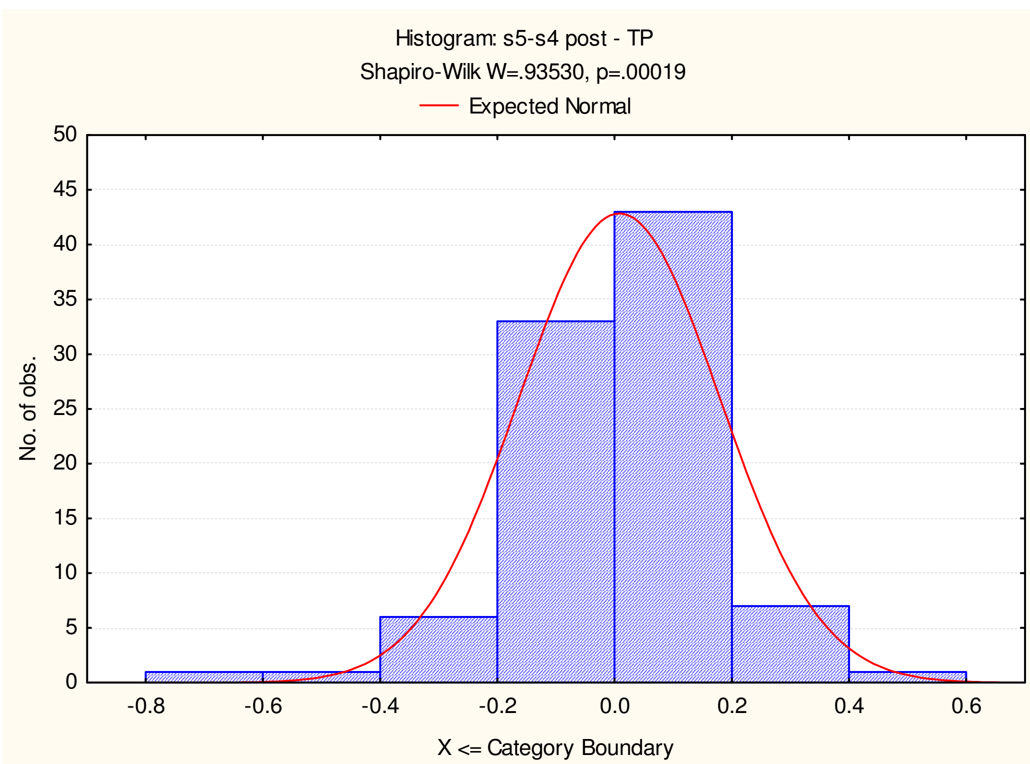

Histogram: s7-s8 pre - TP
Shapiro-Wilk W=.95123, p=.11433
Expected Normal
No. of obs.
X <= Category Boundary
0
1
2
3
4
5
6
7
8
9
10
-0.1
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8

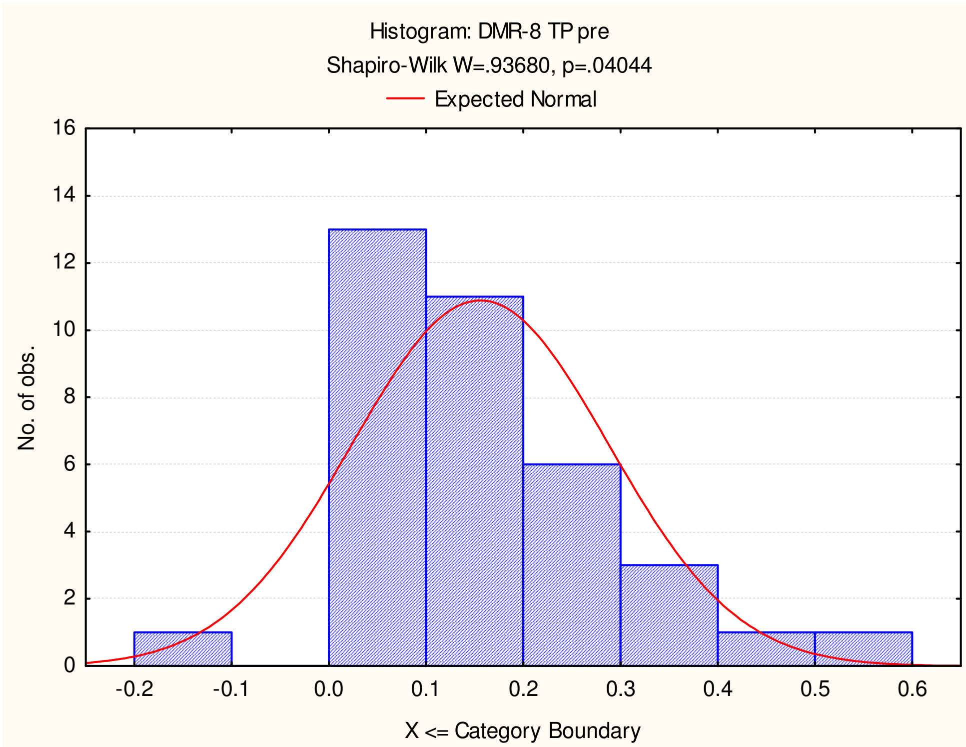

Histogram: s7-s8 post - TP
Shapiro-Wilk W=.93183, p=.00012
Expected Normal
No. of obs.
X <= Category Boundary
0
5
10
15
20
25
30
35
40
45
50
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2

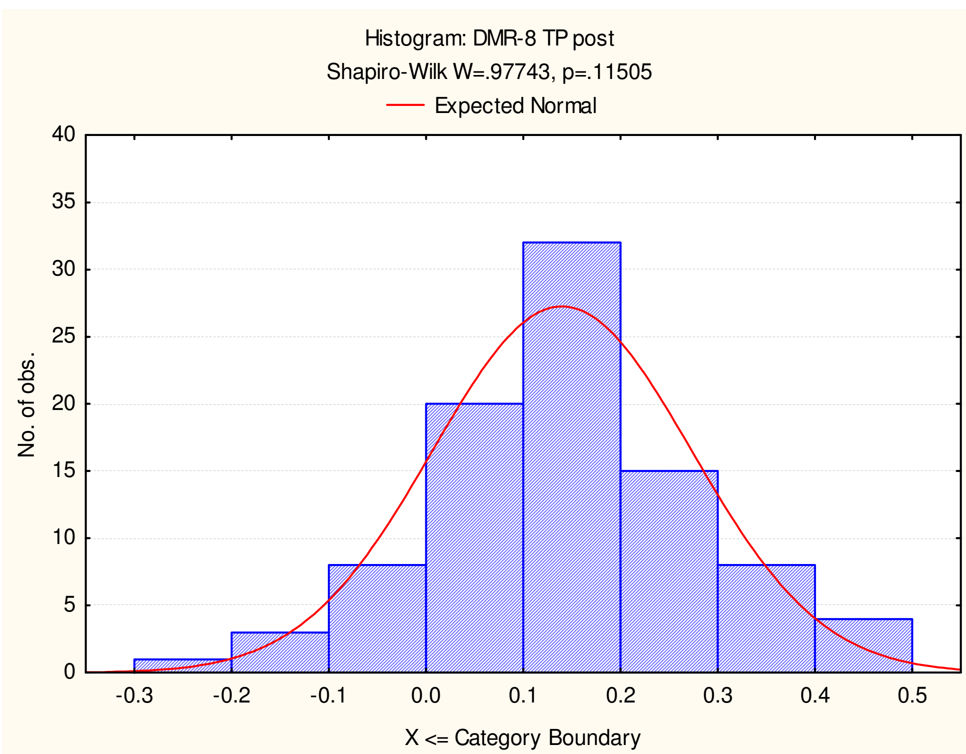

Histogram: s4-s8 pre - TP
Shapiro-Wilk W=.91550, p=.00931
Expected Normal
No. of obs.
X <= Category Boundary

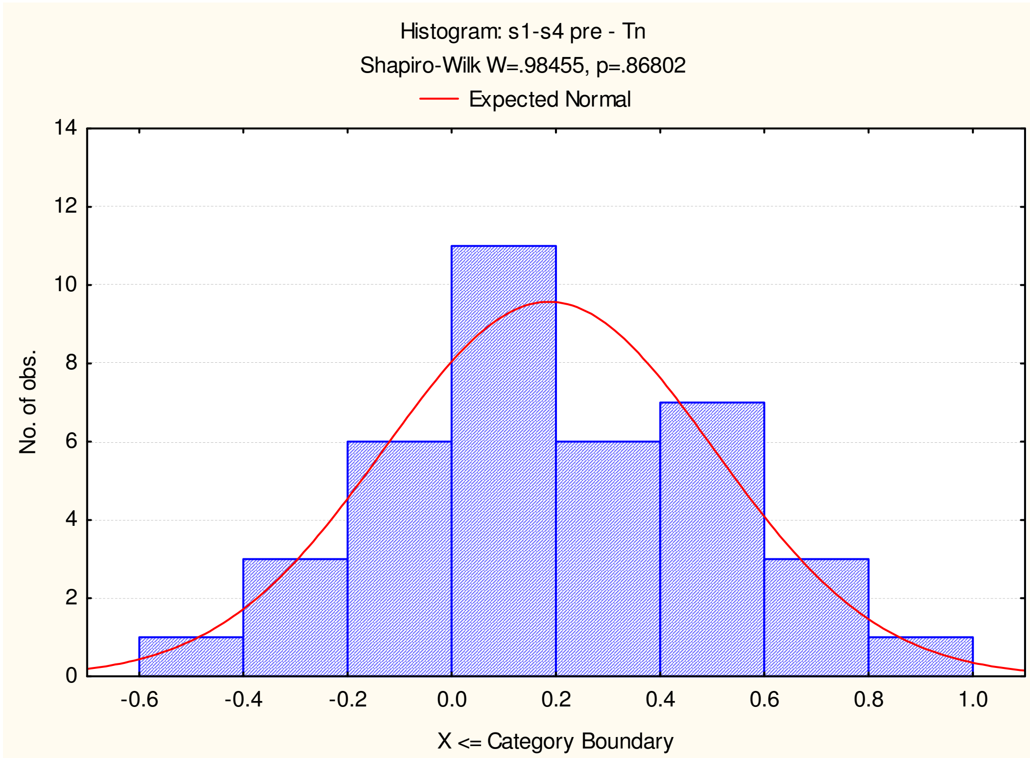

Histogram: s4-s8 post - TP
Shapiro-Wilk W=.96216, p=.00908
Expected Normal
No. of obs.
X <= Category Boundary

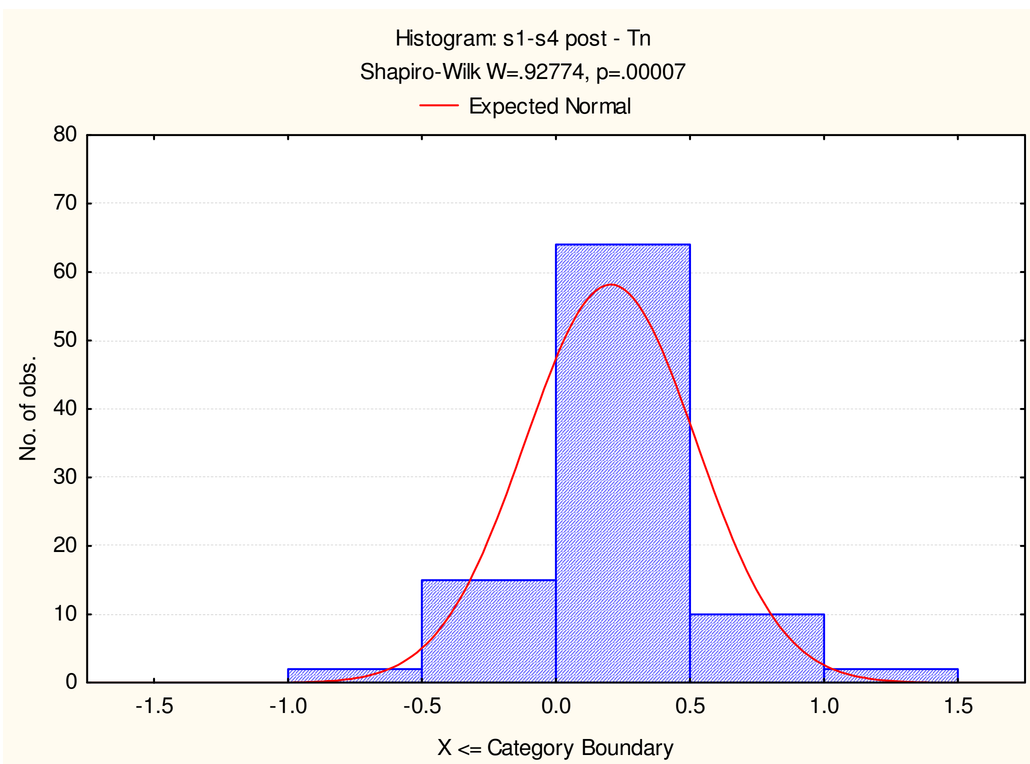

Histogram: s5-s4 pre - TP
Shapiro-Wilk W=.95483, p=.12848
Expected Normal
No. of obs.
X <= Category Boundary

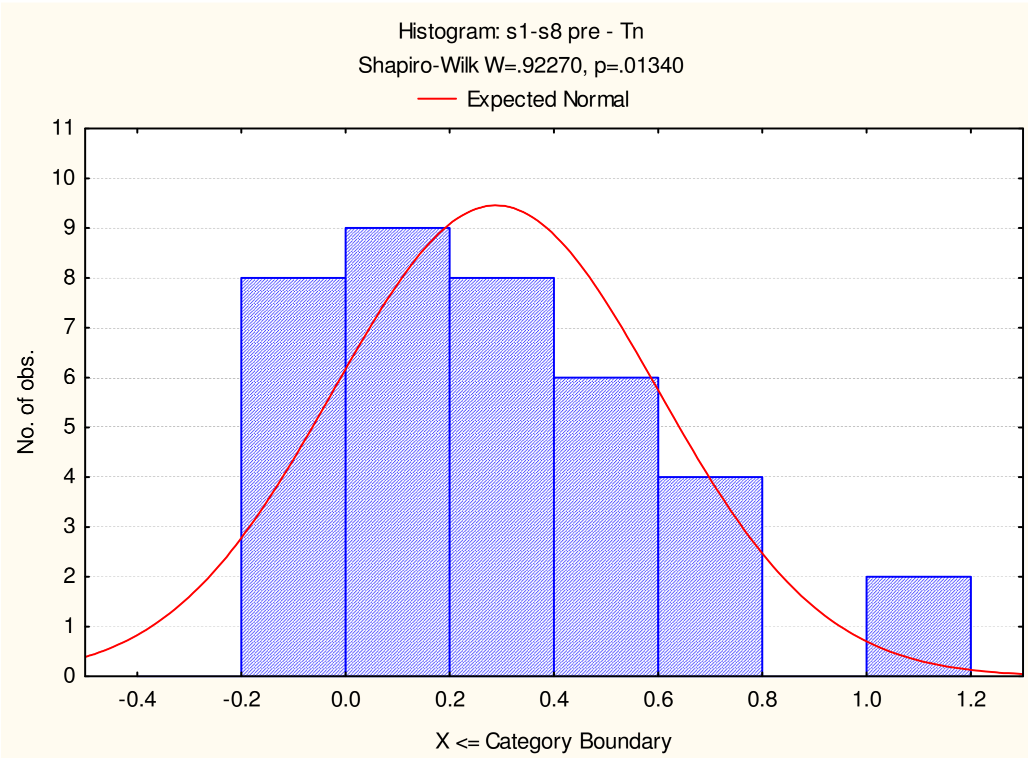

Histogram: s5-s4 post - TP
Shapiro-Wilk W=.93530, p=.00019
Expected Normal
No. of obs.
X <= Category Boundary

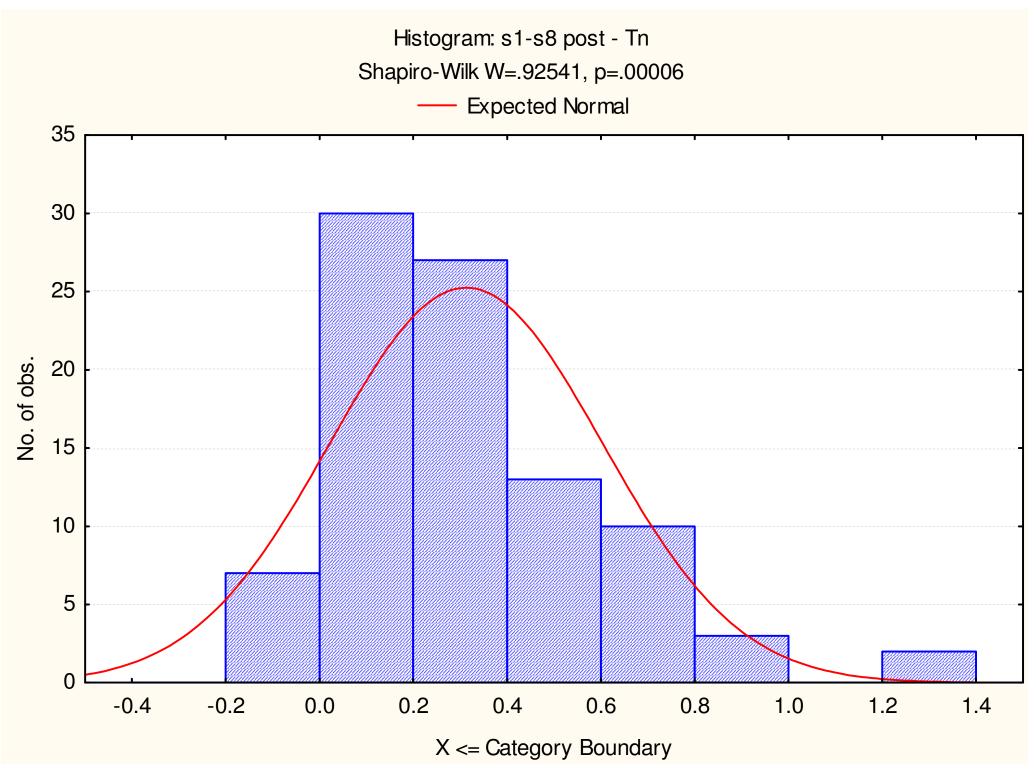

Histogram: DMR-8 TP pre
Shapiro-Wilk W=.93680, p=.04044
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
16
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
0.5
0.6
X <= Category Boundary

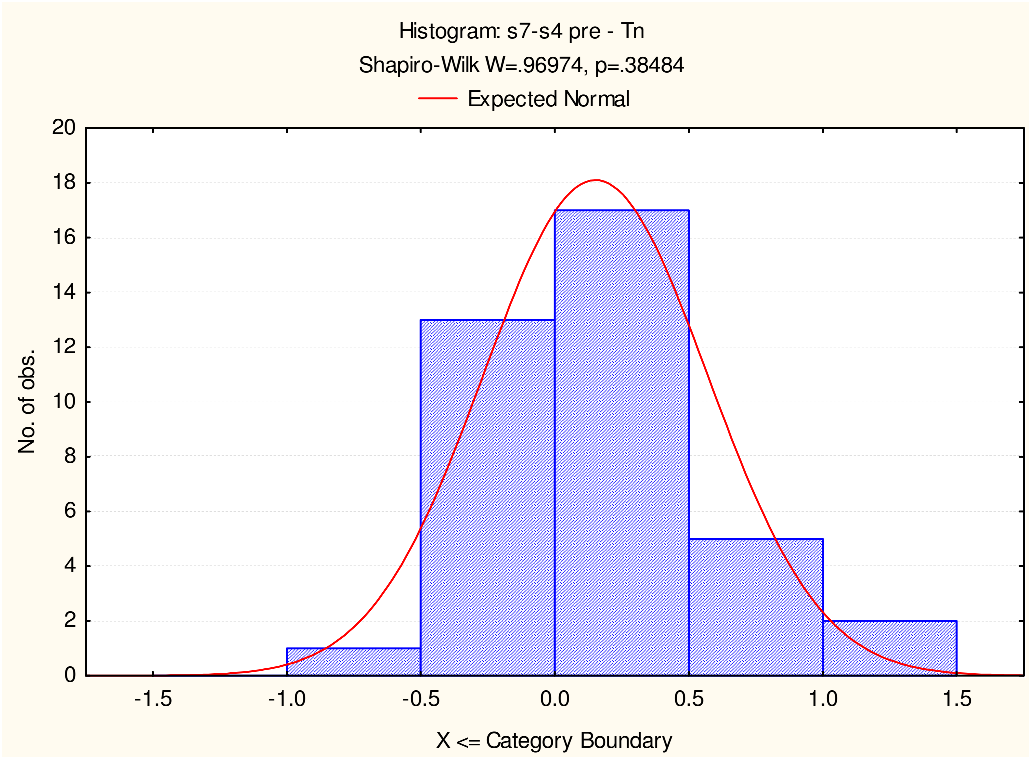

Histogram: DMR-8 TP post
Shapiro-Wilk W=.97743, p=.11505
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
40
-0.3
-0.2
-0.1
0.0
0.1
0.2
0.3
0.4
0.5
X <= Category Boundary

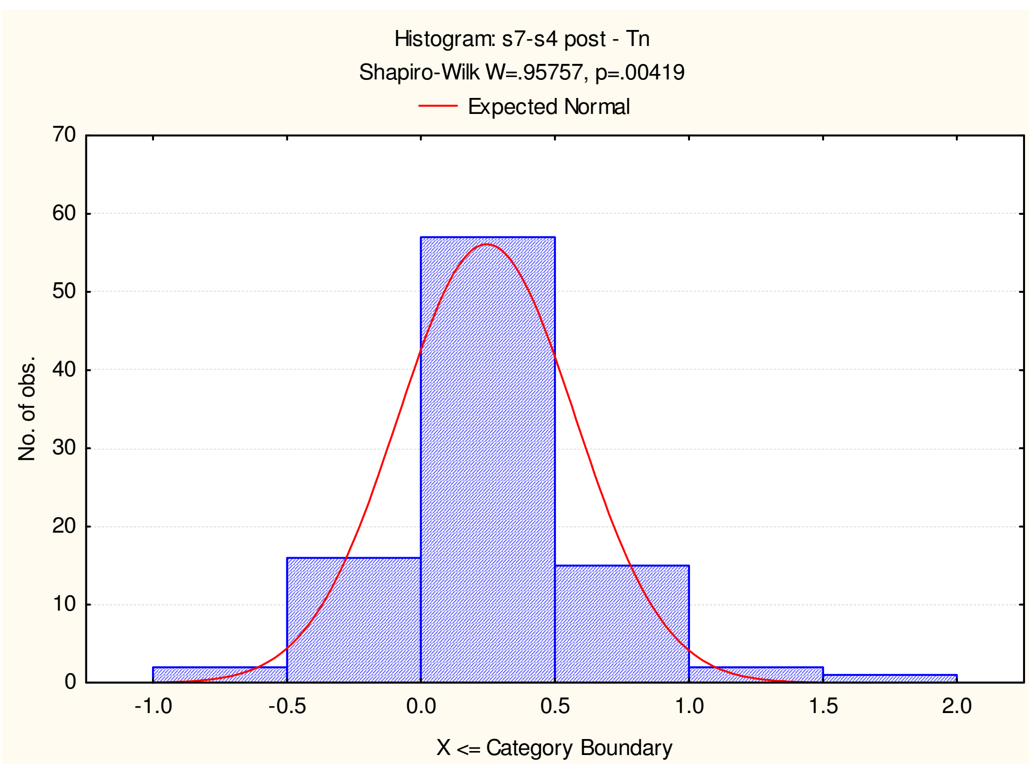

Histogram: s1-s4 pre - Tn
Shapiro-Wilk W=.98455, p=.86802
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
X <= Category Boundary

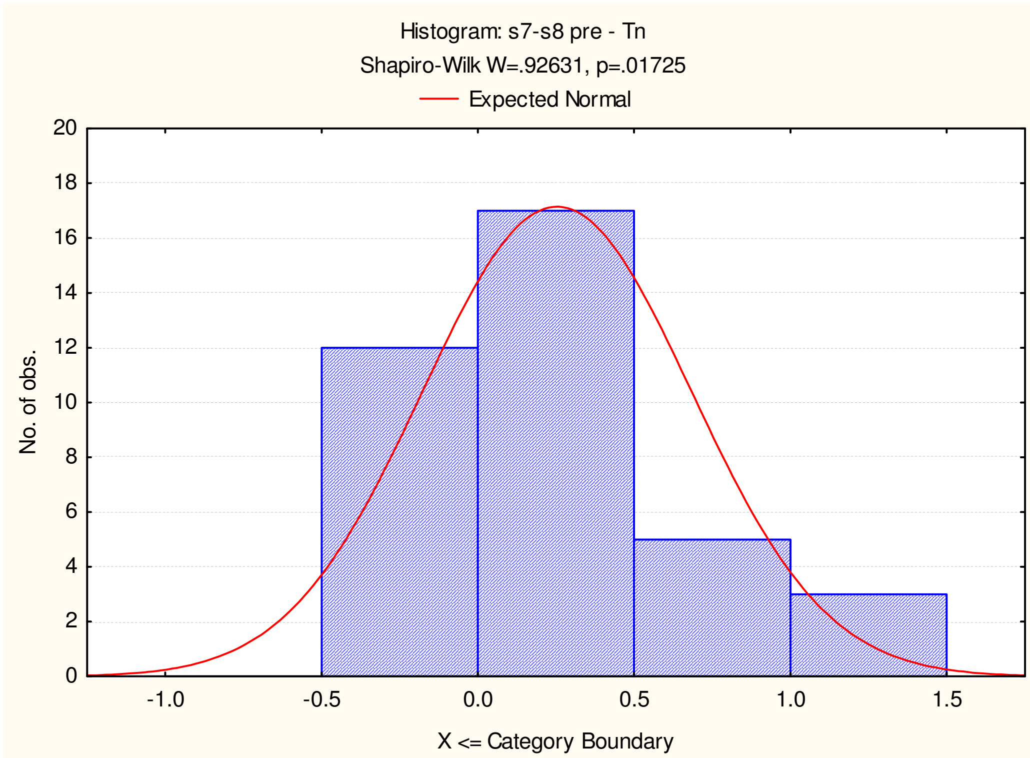

Histogram: s1-s4 post - Tn
Shapiro-Wilk W=.92774, p=.00007
Expected Normal
No. of obs.
0
10
20
30
40
50
60
70
80
-1.5
-1.0
-0.5
0.0
0.5
1.0
1.5
X <= Category Boundary

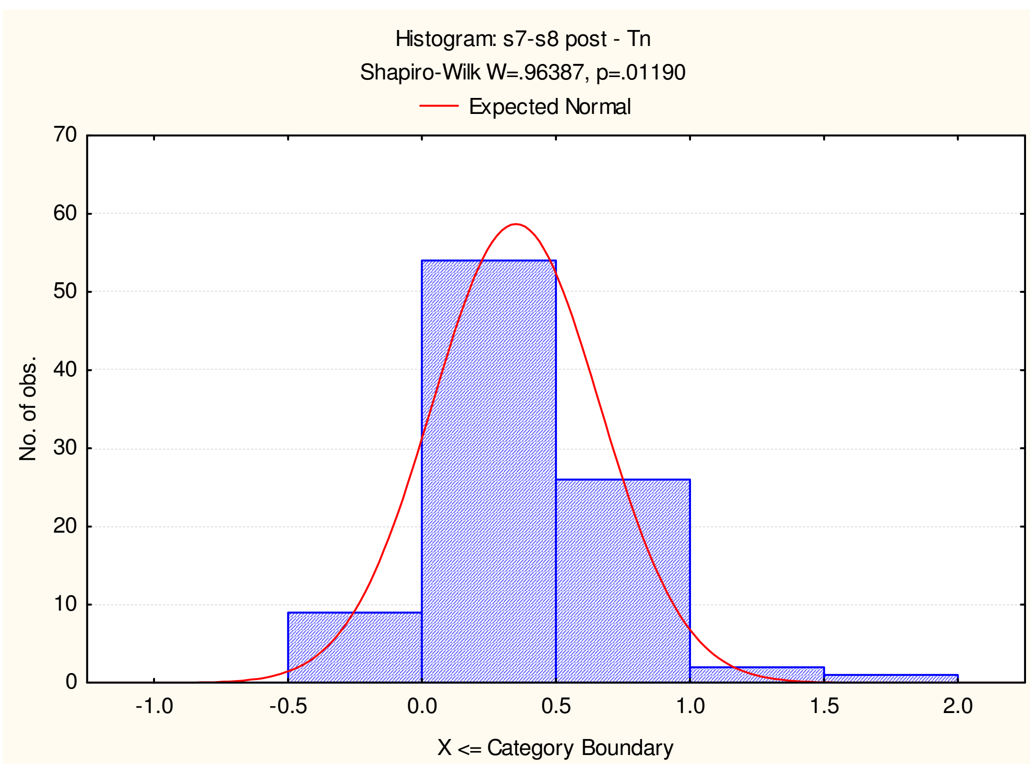

Histogram: s1-s8 pre - Tn
Shapiro-Wilk W=.92270, p=.01340
Expected Normal
No. of obs.
X <= Category Boundary
0
1
2
3
4
5
6
7
8
9
10
11
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2

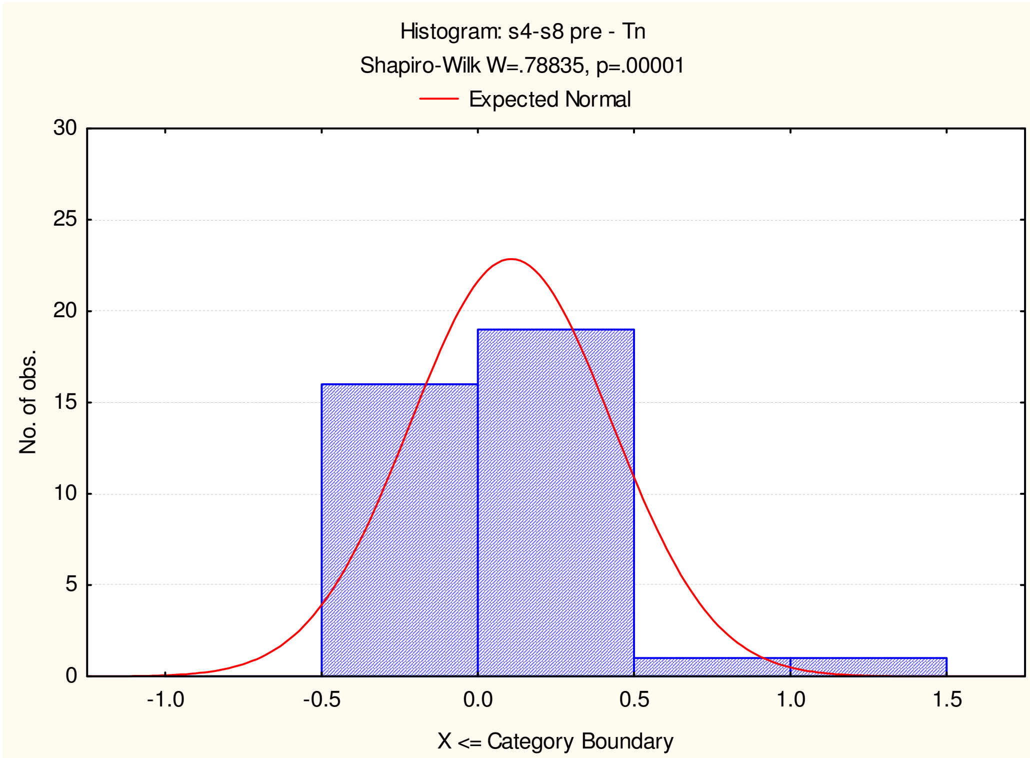

Histogram: s1-s8 post - Tn
Shapiro-Wilk W=.92541, p=.00006
Expected Normal
No. of obs.
X <= Category Boundary
0
5
10
15
20
25
30
35
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4

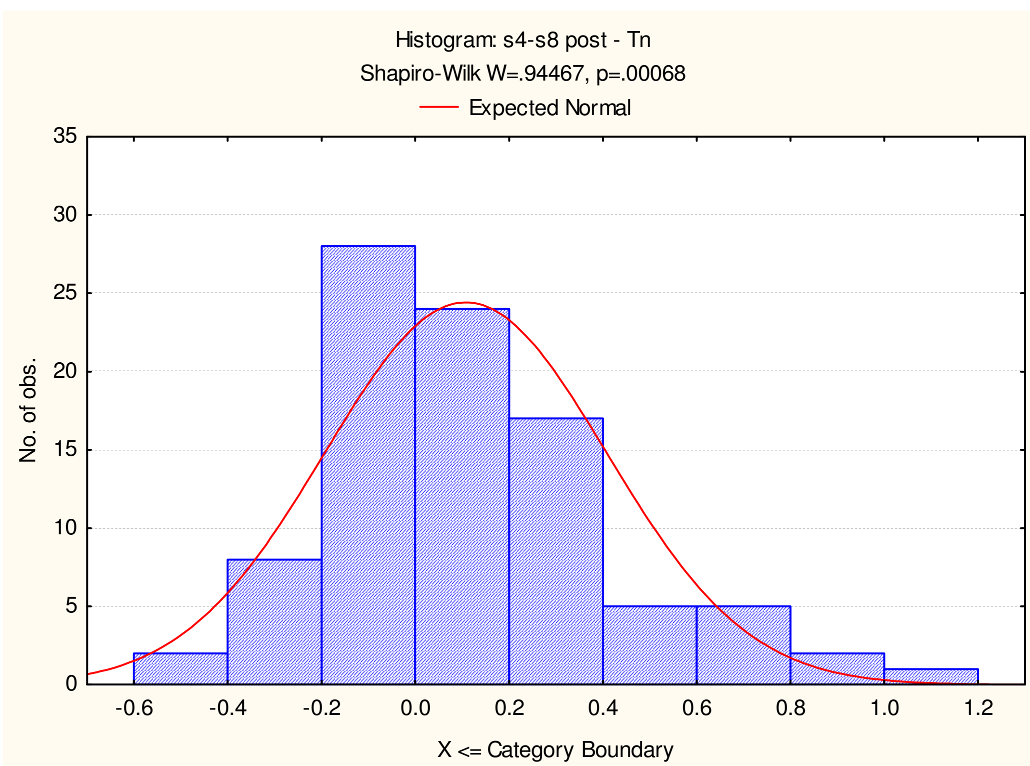

Histogram: s7-s4 pre - Tn
Shapiro-Wilk W=.96974, p=.38484
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
16
18
20
-1.5
-1.0
-0.5
0.0
0.5
1.0
1.5
X <= Category Boundary

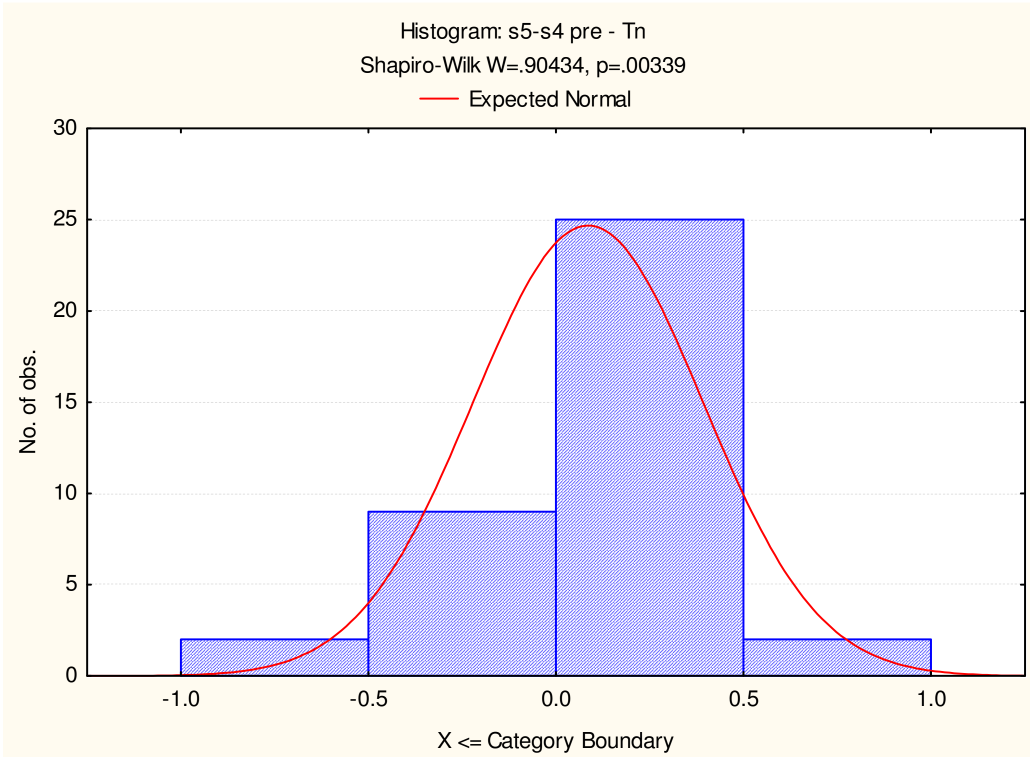

Histogram: s7-s4 post - Tn
Shapiro-Wilk W=.95757, p=.00419
Expected Normal
No. of obs.
0
10
20
30
40
50
60
70
-1.0
-0.5
0.0
0.5
1.0
1.5
2.0
X <= Category Boundary

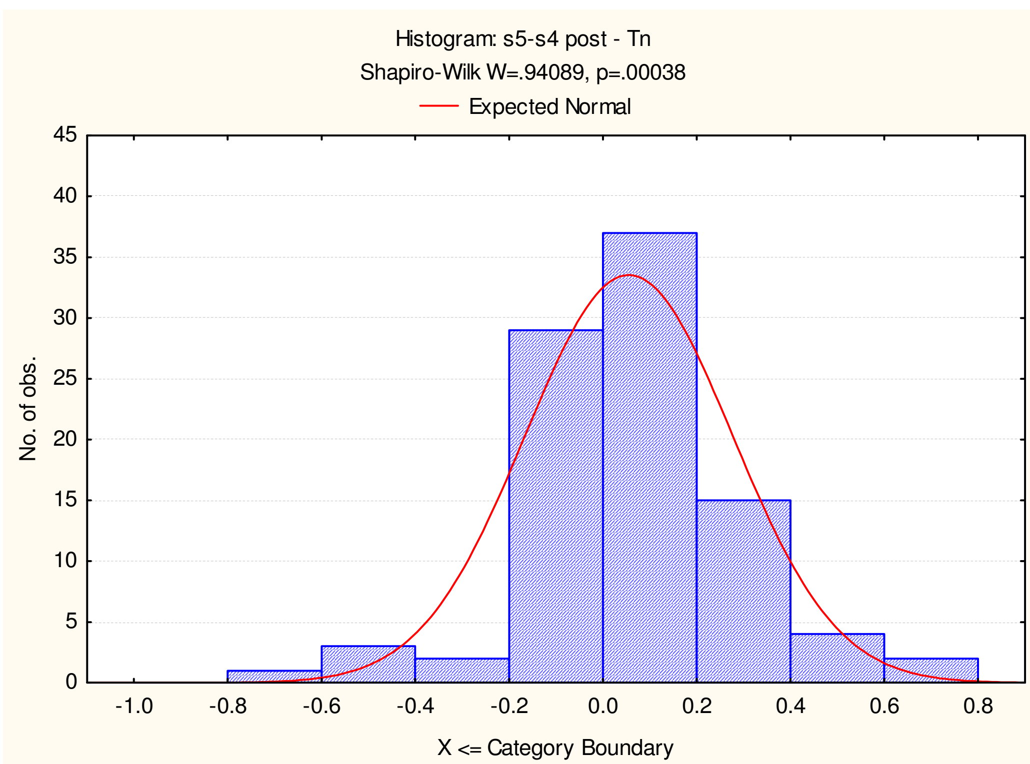

Histogram: s7-s8 pre - Tn
Shapiro-Wilk W=.92631, p=.01725
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
16
18
20
-1.0
-0.5
0.0
0.5
1.0
1.5
X <= Category Boundary

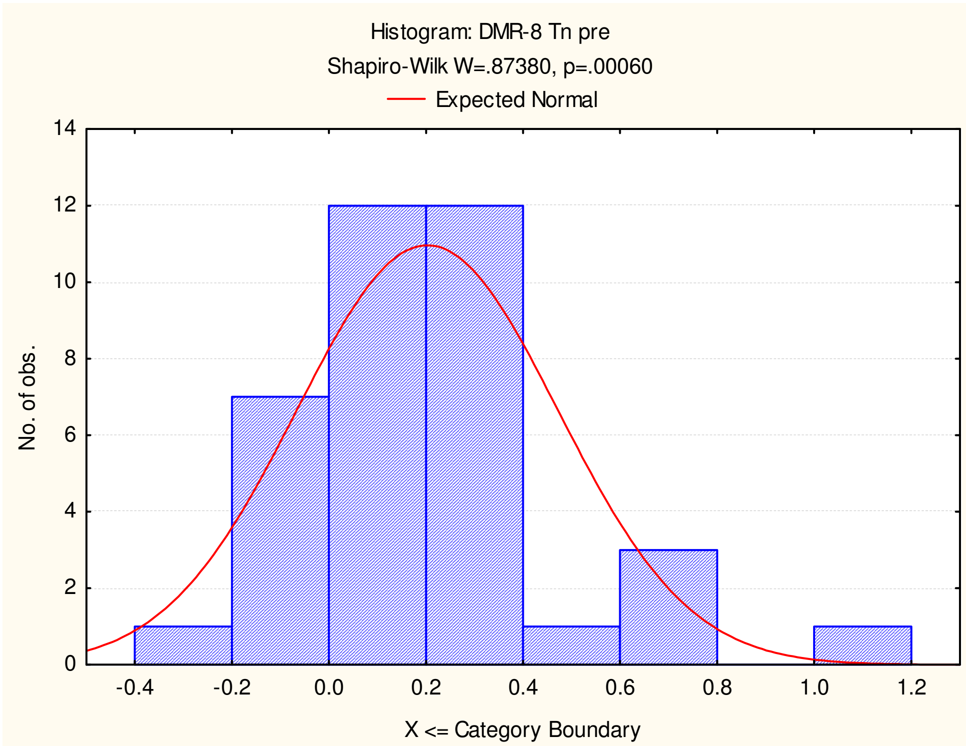

Histogram: s7-s8 post - Tn
Shapiro-Wilk W=.96387, p=.01190
Expected Normal
No. of obs.
0
10
20
30
40
50
60
70
-1.0
-0.5
0.0
0.5
1.0
1.5
2.0
X <= Category Boundary

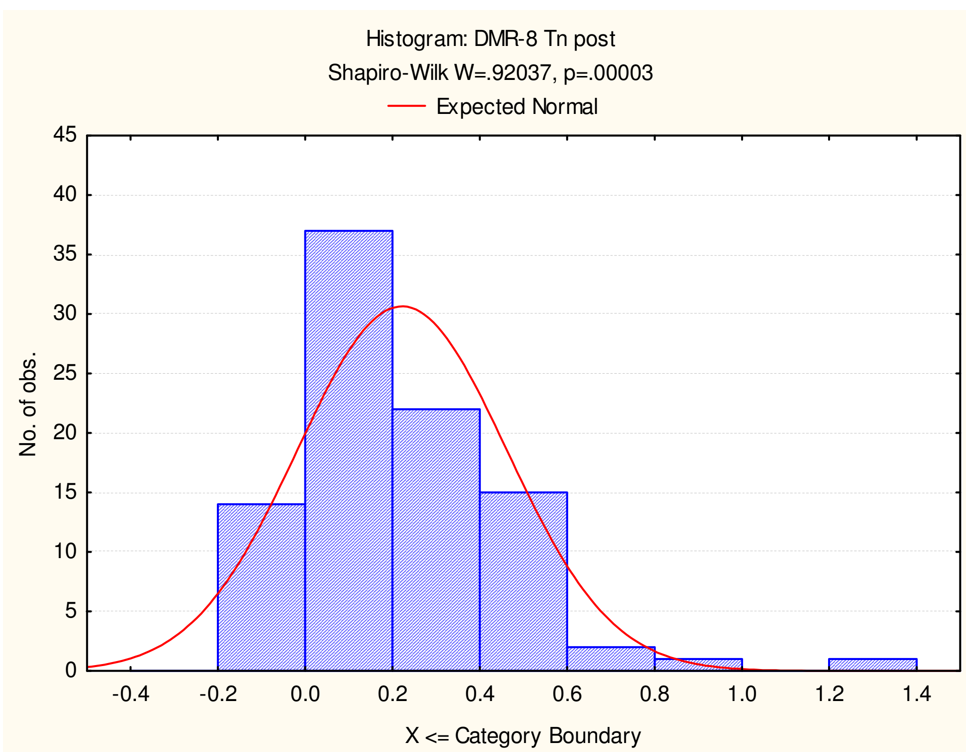

Histogram: s4-s8 pre - Tn
Shapiro-Wilk W=.78835, p=.00001
Expected Normal
No. of obs.
0
5
10
15
20
25
30
-1.0
-0.5
0.0
0.5
1.0
1.5
X <= Category Boundary


Histogram: s4-s8 post - Tn
Shapiro-Wilk W=.94467, p=.00068
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
X <= Category Boundary

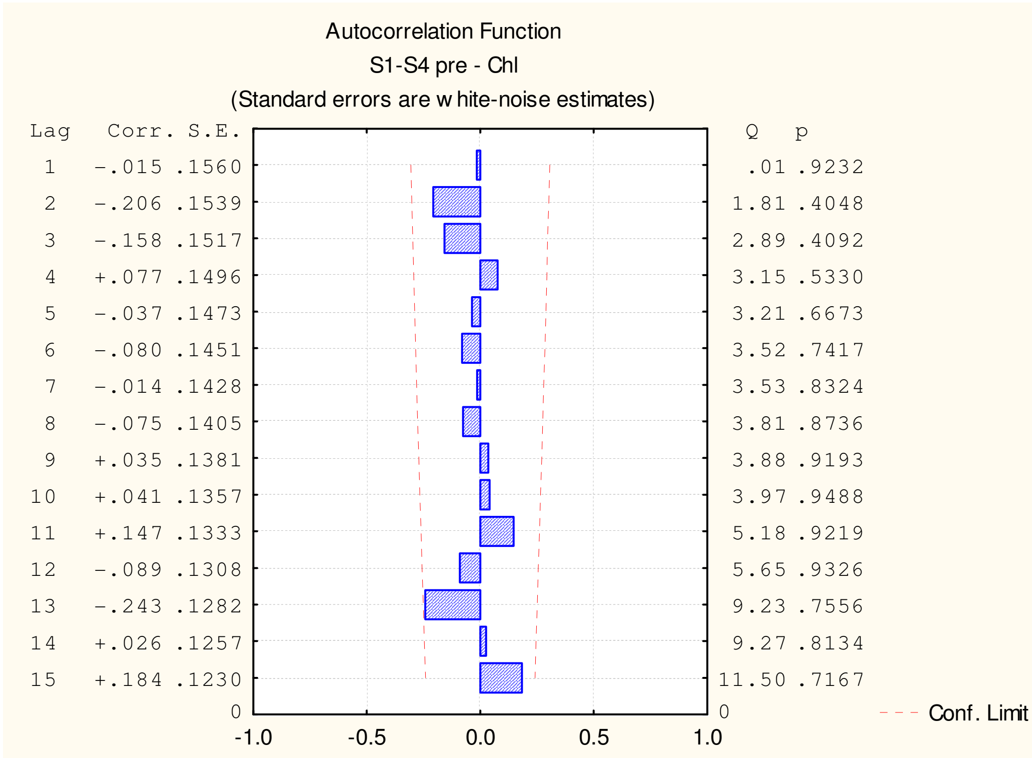

Histogram: s5-s4 pre - Tn
Shapiro-Wilk W=.90434, p=.00339
Expected Normal
No. of obs.
30
25
20
15
10
5
0
-1.0
-0.5
0.0
0.5
1.0
X <= Category Boundary

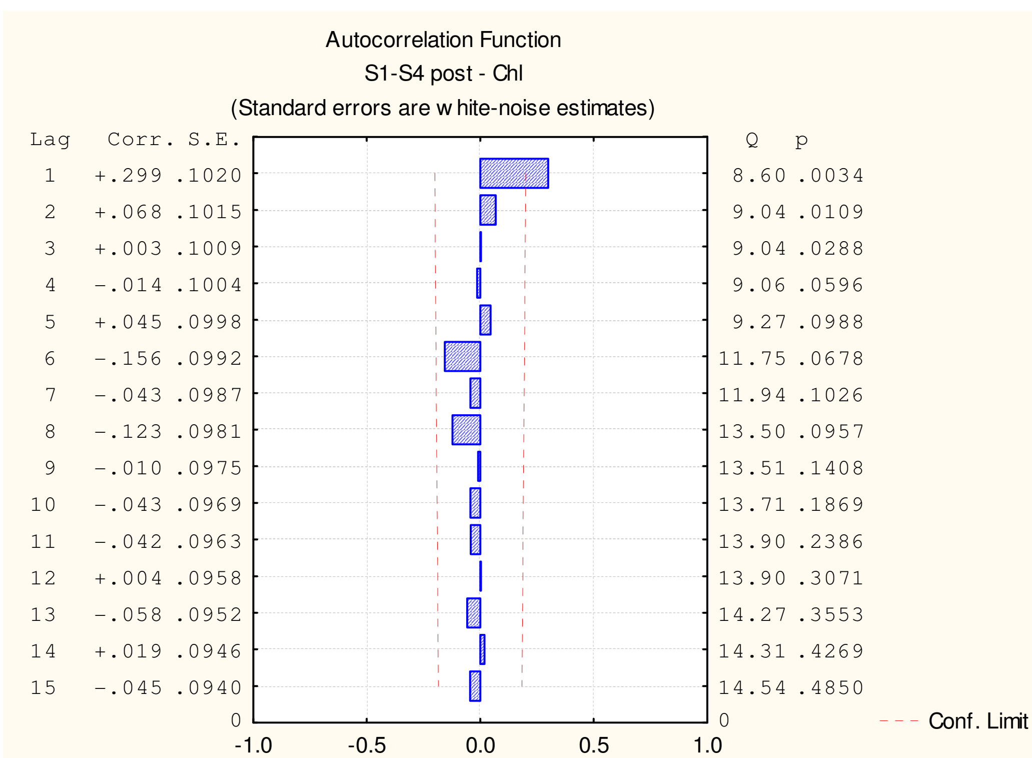

Histogram: s5-s4 post - Tn
Shapiro-Wilk W=.94089, p=.00038
Expected Normal
No. of obs.
45
40
35
30
25
20
15
10
5
0
-1.0
-0.8
-0.6
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
X <= Category Boundary

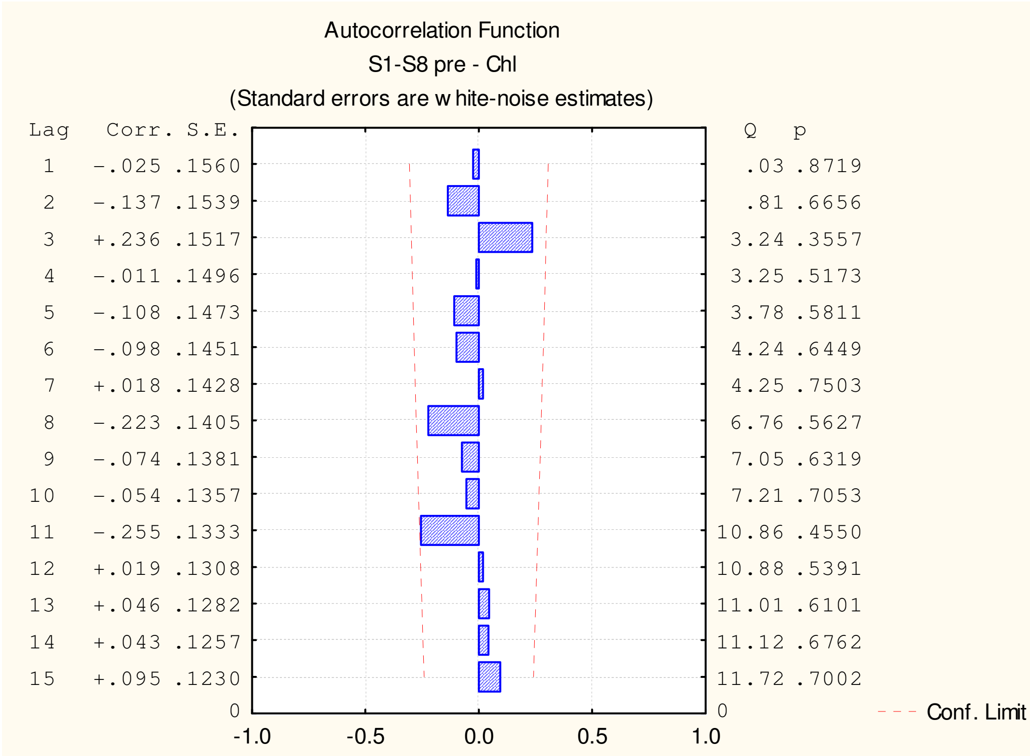

Histogram: DMR-8 Tn pre
Shapiro-Wilk W=.87380, p=.00060
Expected Normal
No. of obs.
0
2
4
6
8
10
12
14
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
X <= Category Boundary

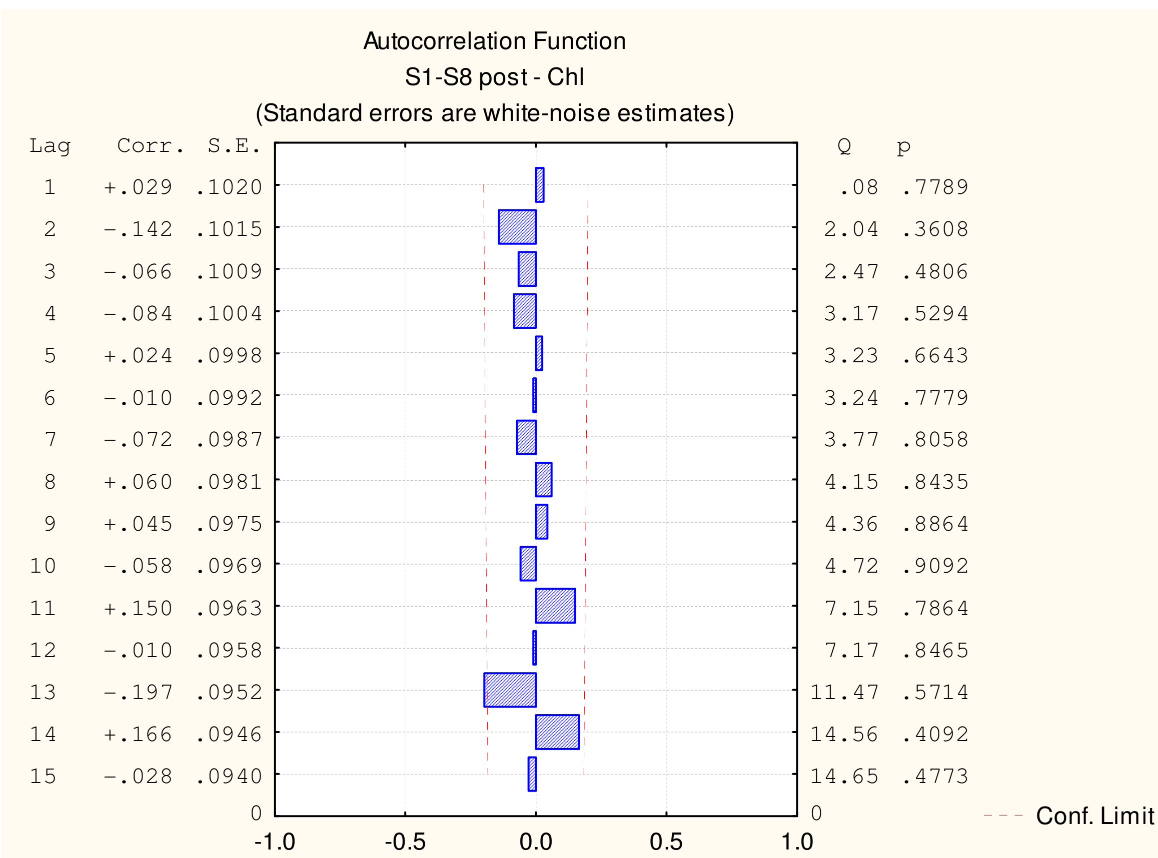

Histogram: DMR-8 Tn post
Shapiro-Wilk W=.92037, p=.00003
Expected Normal
No. of obs.
0
5
10
15
20
25
30
35
40
45
-0.4
-0.2
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
X <= Category Boundary

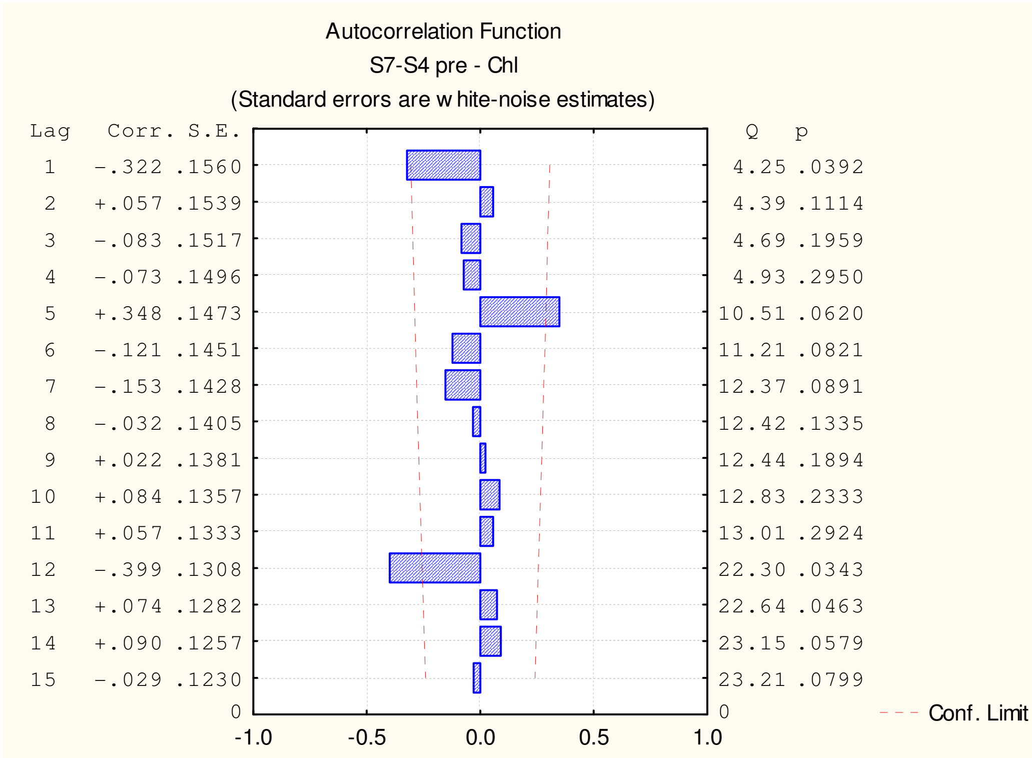
# Appendix 2 - Independence

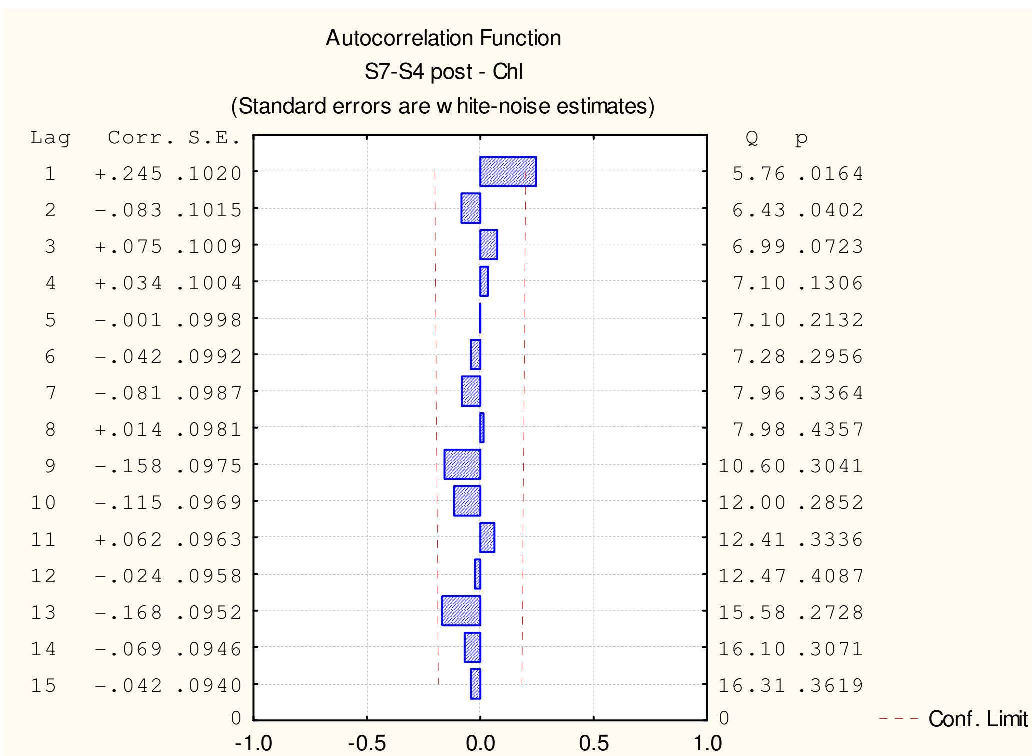

Autocorrelation Function
S1-S4 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 -.015 .1560
2 -.206 .1539
3 -.158 .1517
4 +.077 .1496
5 -.037 .1473
6 -.080 .1451
7 -.014 .1428
8 -.075 .1405
9 +.035 .1381
10 +.041 .1357
11 +.147 .1333
12 -.089 .1308
13 -.243 .1282
14 +.026 .1257
15 +.184 .1230
Q p
.01 .9232
1.81 .4048
2.89 .4092
3.15 .5330
3.21 .6673
3.52 .7417
3.53 .8324
3.81 .8736
3.88 .9193
3.97 .9488
5.18 .9219
5.65 .9326
9.23 .7556
9.27 .8134
11.50 .7167
0
-1.0
-0.5
0.0
0.5
1.0
0
Conf. Limit

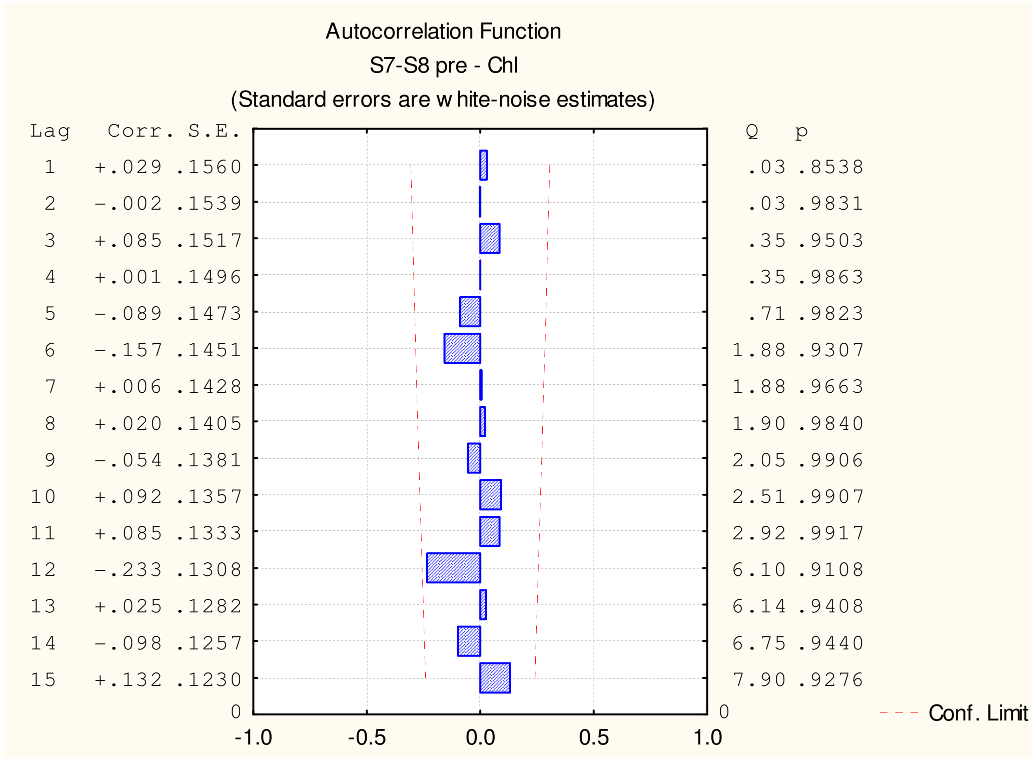

Autocorrelation Function
S1-S4 post - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 +.299 .1020
2 +.068 .1015
3 +.003 .1009
4 -.014 .1004
5 +.045 .0998
6 -.156 .0992
7 -.043 .0987
8 -.123 .0981
9 -.010 .0975
10 -.043 .0969
11 -.042 .0963
12 +.004 .0958
13 -.058 .0952
14 +.019 .0946
15 -.045 .0940
Q p
8.60 .0034
9.04 .0109
9.04 .0288
9.06 .0596
9.27 .0988
11.75 .0678
11.94 .1026
13.50 .0957
13.51 .1408
13.71 .1869
13.90 .2386
13.90 .3071
14.27 .3553
14.31 .4269
14.54 .4850
0
-1.0
-0.5
0.0
0.5
1.0
0
Conf. Limit


Autocorrelation Function
S1-S8 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.025 .1560 .03 .8719
2 -.137 .1539 .81 .6656
3 +.236 .1517 3.24 .3557
4 -.011 .1496 3.25 .5173
5 -.108 .1473 3.78 .5811
6 -.098 .1451 4.24 .6449
7 +.018 .1428 4.25 .7503
8 -.223 .1405 6.76 .5627
9 -.074 .1381 7.05 .6319
10 -.054 .1357 7.21 .7053
11 -.255 .1333 10.86 .4550
12 +.019 .1308 10.88 .5391
13 +.046 .1282 11.01 .6101
14 +.043 .1257 11.12 .6762
15 +.095 .1230 11.72 .7002
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S8 post - Chl
(Standard errors are white-noise estimates)
Lag Corr. S.E. Q p
1 +.029 .1020 .08 .7789
2 -.142 .1015 2.04 .3608
3 -.066 .1009 2.47 .4806
4 -.084 .1004 3.17 .5294
5 +.024 .0998 3.23 .6643
6 -.010 .0992 3.24 .7779
7 -.072 .0987 3.77 .8058
8 +.060 .0981 4.15 .8435
9 +.045 .0975 4.36 .8864
10 -.058 .0969 4.72 .9092
11 +.150 .0963 7.15 .7864
12 -.010 .0958 7.17 .8465
13 -.197 .0952 11.47 .5714
14 +.166 .0946 14.56 .4092
15 -.028 .0940 14.65 .4773
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S7-S4 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.322 .1560 4.25 .0392
2 +.057 .1539 4.39 .1114
3 -.083 .1517 4.69 .1959
4 -.073 .1496 4.93 .2950
5 +.348 .1473 10.51 .0620
6 -.121 .1451 11.21 .0821
7 -.153 .1428 12.37 .0891
8 -.032 .1405 12.42 .1335
9 +.022 .1381 12.44 .1894
10 +.084 .1357 12.83 .2333
11 +.057 .1333 13.01 .2924
12 -.399 .1308 22.30 .0343
13 +.074 .1282 22.64 .0463
14 +.090 .1257 23.15 .0579
15 -.029 .1230 23.21 .0799
0
0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S7-S4 post - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.245 .1020 5.76 .0164
2 -.083 .1015 6.43 .0402
3 +.075 .1009 6.99 .0723
4 +.034 .1004 7.10 .1306
5 -.001 .0998 7.10 .2132
6 -.042 .0992 7.28 .2956
7 -.081 .0987 7.96 .3364
8 +.014 .0981 7.98 .4357
9 -.158 .0975 10.60 .3041
10 -.115 .0969 12.00 .2852
11 +.062 .0963 12.41 .3336
12 -.024 .0958 12.47 .4087
13 -.168 .0952 15.58 .2728
14 -.069 .0946 16.10 .3071
15 -.042 .0940 16.31 .3619
0
0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S7-S8 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 +.029 .1560 .03 .8538
2 -.002 .1539 .03 .9831
3 +.085 .1517 .35 .9503
4 +.001 .1496 .35 .9863
5 -.089 .1473 .71 .9823
6 -.157 .1451 1.88 .9307
7 +.006 .1428 1.88 .9663
8 +.020 .1405 1.90 .9840
9 -.054 .1381 2.05 .9906
10 +.092 .1357 2.51 .9907
11 +.085 .1333 2.92 .9917
12 -.233 .1308 6.10 .9108
13 +.025 .1282 6.14 .9408
14 -.098 .1257 6.75 .9440
15 +.132 .1230 7.90 .9276
0
0
Conf. Limit
-1.0
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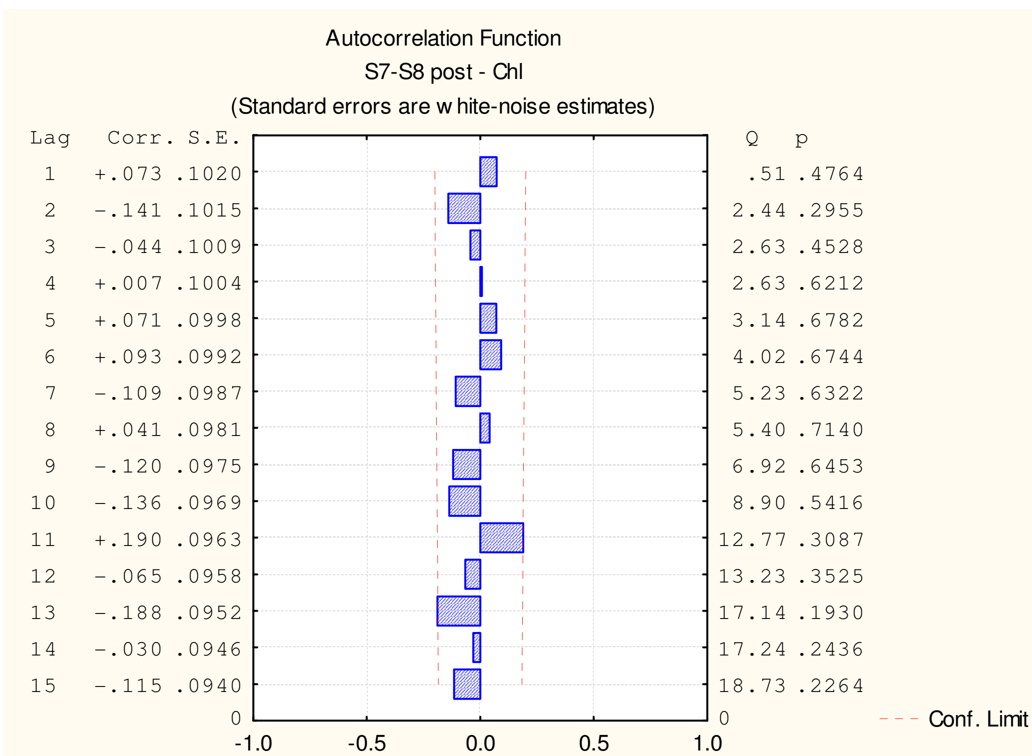
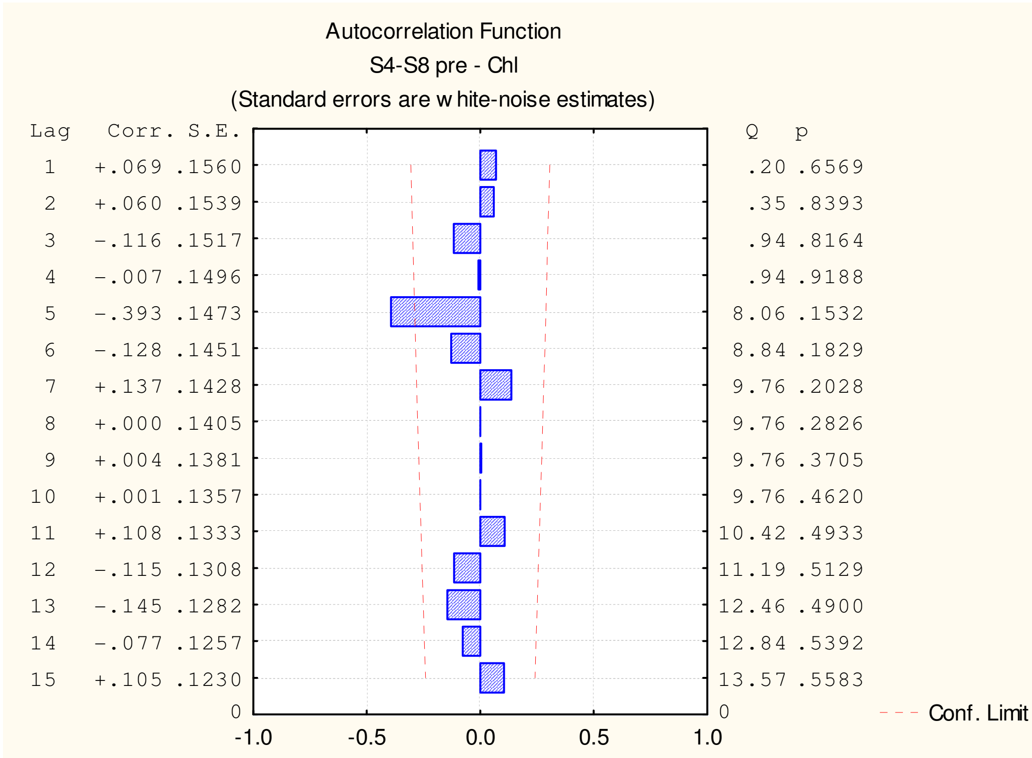
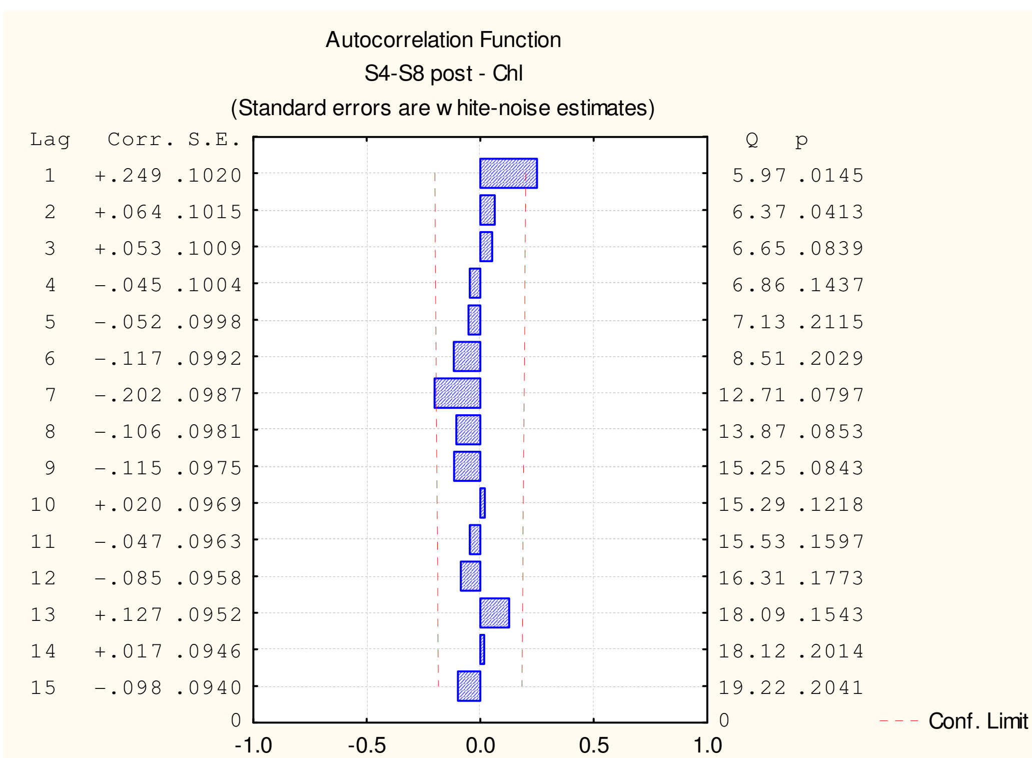
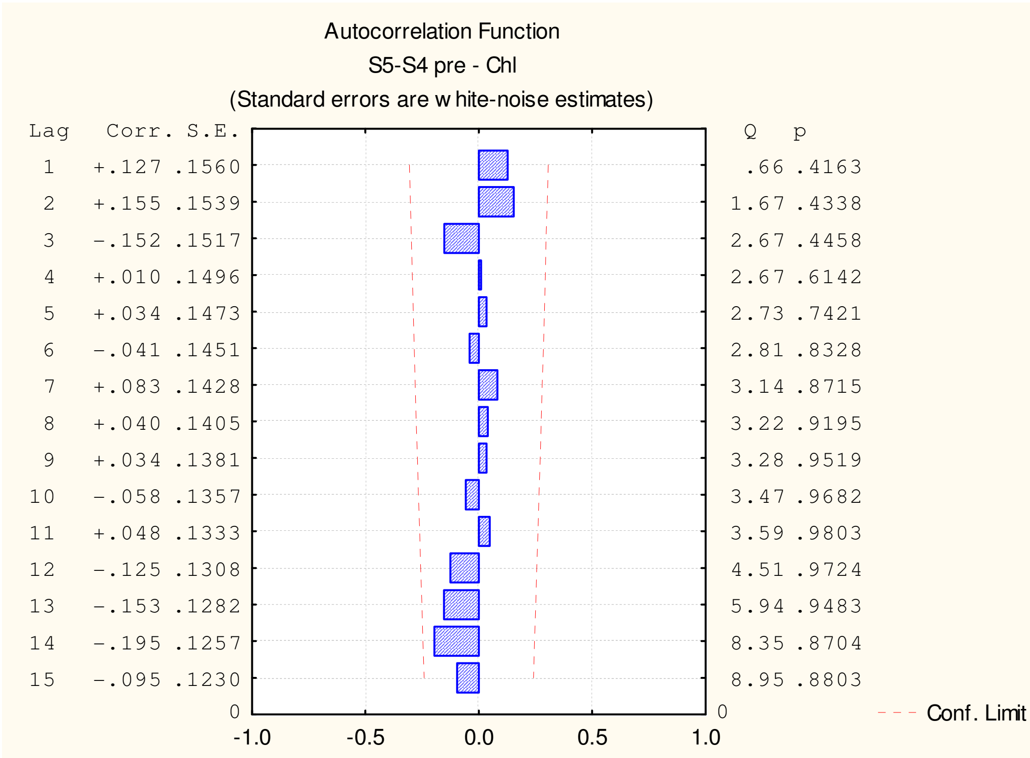
0.0
0.5
1.0


Autocorrelation Function
S7-S8 post - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 +.073 .1020 .51 .4764
2 -.141 .1015 2.44 .2955
3 -.044 .1009 2.63 .4528
4 +.007 .1004 2.63 .6212
5 +.071 .0998 3.14 .6782
6 +.093 .0992 4.02 .6744
7 -.109 .0987 5.23 .6322
8 +.041 .0981 5.40 .7140
9 -.120 .0975 6.92 .6453
10 -.136 .0969 8.90 .5416
11 +.190 .0963 12.77 .3087
12 -.065 .0958 13.23 .3525
13 -.188 .0952 17.14 .1930
14 -.030 .0946 17.24 .2436
15 -.115 .0940 18.73 .2264
0
0
Conf. Limit
-1.0
-0.5
0.0
0.5
1.0


Autocorrelation Function
S4-S8 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.069 .1560 .20 .6569
2 +.060 .1539 .35 .8393
3 -.116 .1517 .94 .8164
4 -.007 .1496 .94 .9188
5 -.393 .1473 8.06 .1532
6 -.128 .1451 8.84 .1829
7 +.137 .1428 9.76 .2028
8 +.000 .1405 9.76 .2826
9 +.004 .1381 9.76 .3705
10 +.001 .1357 9.76 .4620
11 +.108 .1333 10.42 .4933
12 -.115 .1308 11.19 .5129
13 -.145 .1282 12.46 .4900
14 -.077 .1257 12.84 .5392
15 +.105 .1230 13.57 .5583
0
0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S4-S8 post - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.249 .1020 5.97 .0145
2 +.064 .1015 6.37 .0413
3 +.053 .1009 6.65 .0839
4 -.045 .1004 6.86 .1437
5 -.052 .0998 7.13 .2115
6 -.117 .0992 8.51 .2029
7 -.202 .0987 12.71 .0797
8 -.106 .0981 13.87 .0853
9 -.115 .0975 15.25 .0843
10 +.020 .0969 15.29 .1218
11 -.047 .0963 15.53 .1597
12 -.085 .0958 16.31 .1773
13 +.127 .0952 18.09 .1543
14 +.017 .0946 18.12 .2014
15 -.098 .0940 19.22 .2041
0
0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S5-S4 pre - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.127 .1560 .66 .4163
2 +.155 .1539 1.67 .4338
3 -.152 .1517 2.67 .4458
4 +.010 .1496 2.67 .6142
5 +.034 .1473 2.73 .7421
6 -.041 .1451 2.81 .8328
7 +.083 .1428 3.14 .8715
8 +.040 .1405 3.22 .9195
9 +.034 .1381 3.28 .9519
10 -.058 .1357 3.47 .9682
11 +.048 .1333 3.59 .9803
12 -.125 .1308 4.51 .9724
13 -.153 .1282 5.94 .9483
14 -.195 .1257 8.35 .8704
15 -.095 .1230 8.95 .8803
0 0
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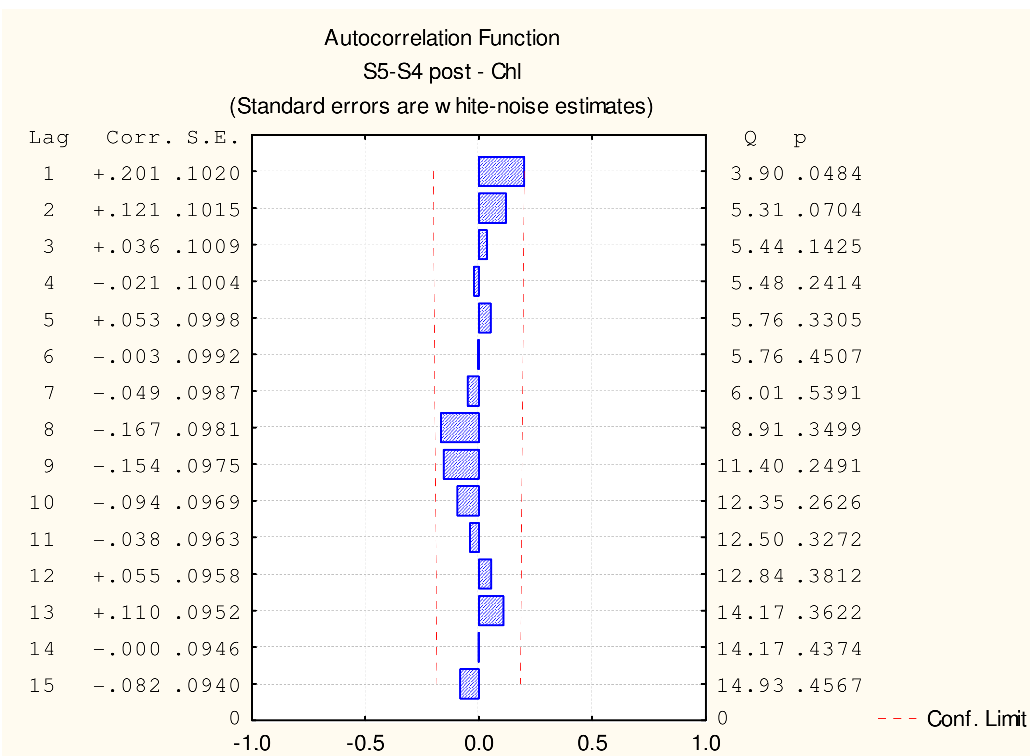
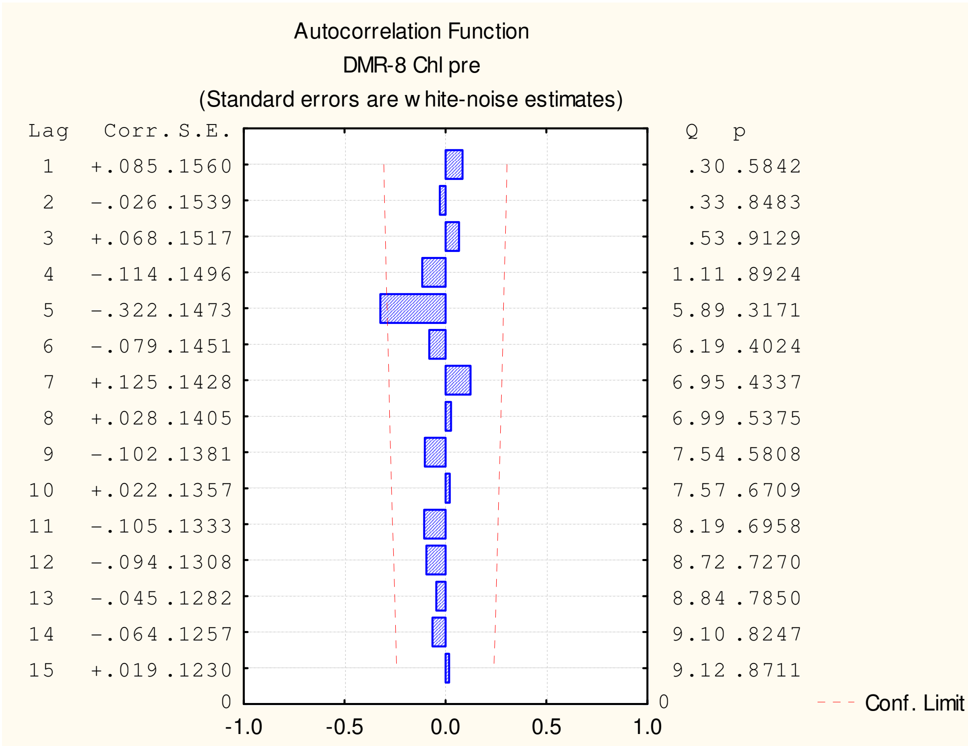
Conf. Limit


Autocorrelation Function
S5-S4 post - Chl
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.201 .1020 3.90 .0484
2 +.121 .1015 5.31 .0704
3 +.036 .1009 5.44 .1425
4 -.021 .1004 5.48 .2414
5 +.053 .0998 5.76 .3305
6 -.003 .0992 5.76 .4507
7 -.049 .0987 6.01 .5391
8 -.167 .0981 8.91 .3499
9 -.154 .0975 11.40 .2491
10 -.094 .0969 12.35 .2626
11 -.038 .0963 12.50 .3272
12 +.055 .0958 12.84 .3812
13 +.110 .0952 14.17 .3622
14 -.000 .0946 14.17 .4374
15 -.082 .0940 14.93 .4567
0 0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
DMR-8 Chl pre
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 +.085 .1560 .30 .5842
2 -.026 .1539 .33 .8483
3 +.068 .1517 .53 .9129
4 -.114 .1496 1.11 .8924
5 -.322 .1473 5.89 .3171
6 -.079 .1451 6.19 .4024
7 +.125 .1428 6.95 .4337
8 +.028 .1405 6.99 .5375
9 -.102 .1381 7.54 .5808
10 +.022 .1357 7.57 .6709
11 -.105 .1333 8.19 .6958
12 -.094 .1308 8.72 .7270
13 -.045 .1282 8.84 .7850
14 -.064 .1257 9.10 .8247
15 +.019 .1230 9.12 .8711
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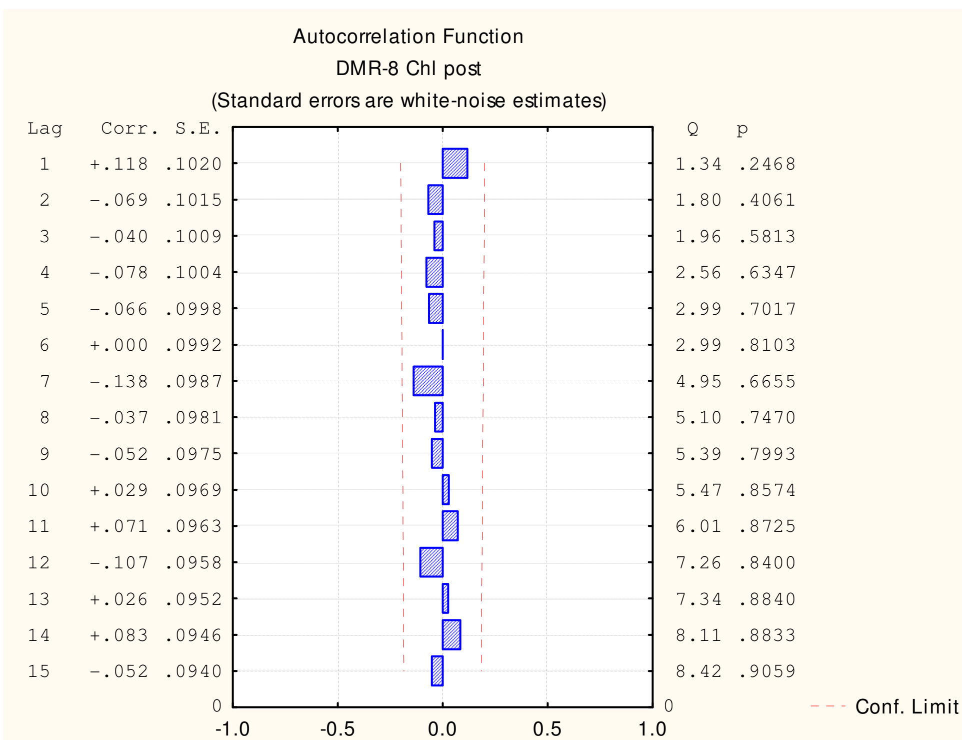
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
DMR-8 Chl post
(Standard errors are white-noise estimates)
Lag Corr. S.E.
Q p
1 +.118 .1020 1.34 .2468
2 -.069 .1015 1.80 .4061
3 -.040 .1009 1.96 .5813
4 -.078 .1004 2.56 .6347
5 -.066 .0998 2.99 .7017
6 +.000 .0992 2.99 .8103
7 -.138 .0987 4.95 .6655
8 -.037 .0981 5.10 .7470
9 -.052 .0975 5.39 .7993
10 +.029 .0969 5.47 .8574
11 +.071 .0963 6.01 .8725
12 -.107 .0958 7.26 .8400
13 +.026 .0952 7.34 .8840
14 +.083 .0946 8.11 .8833
15 -.052 .0940 8.42 .9059
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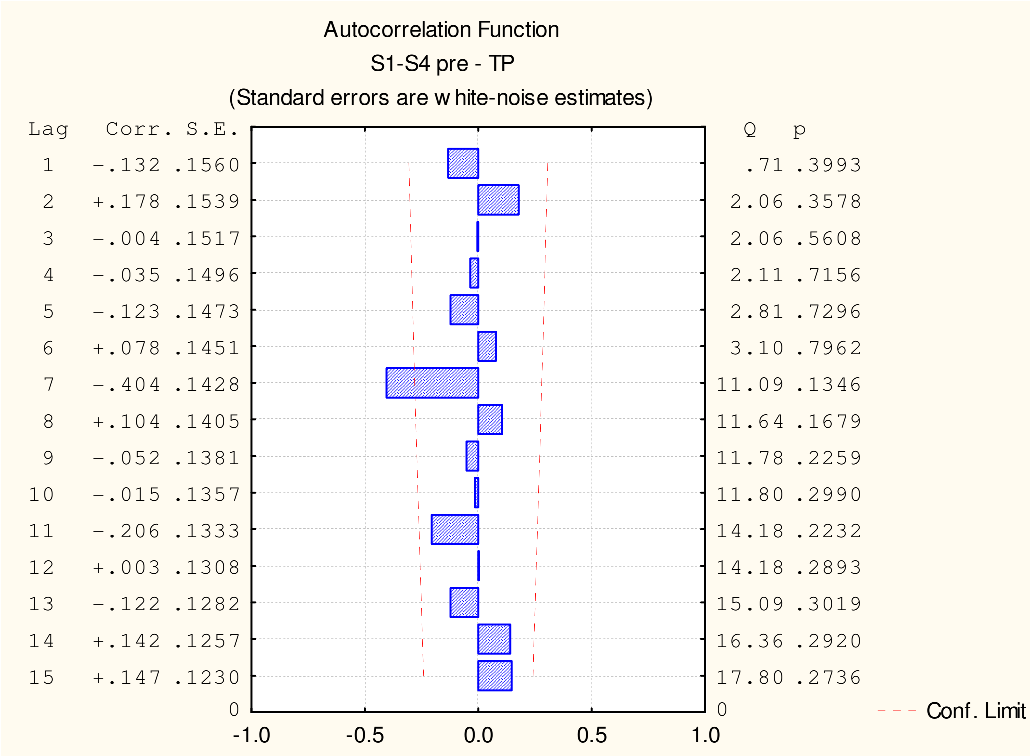
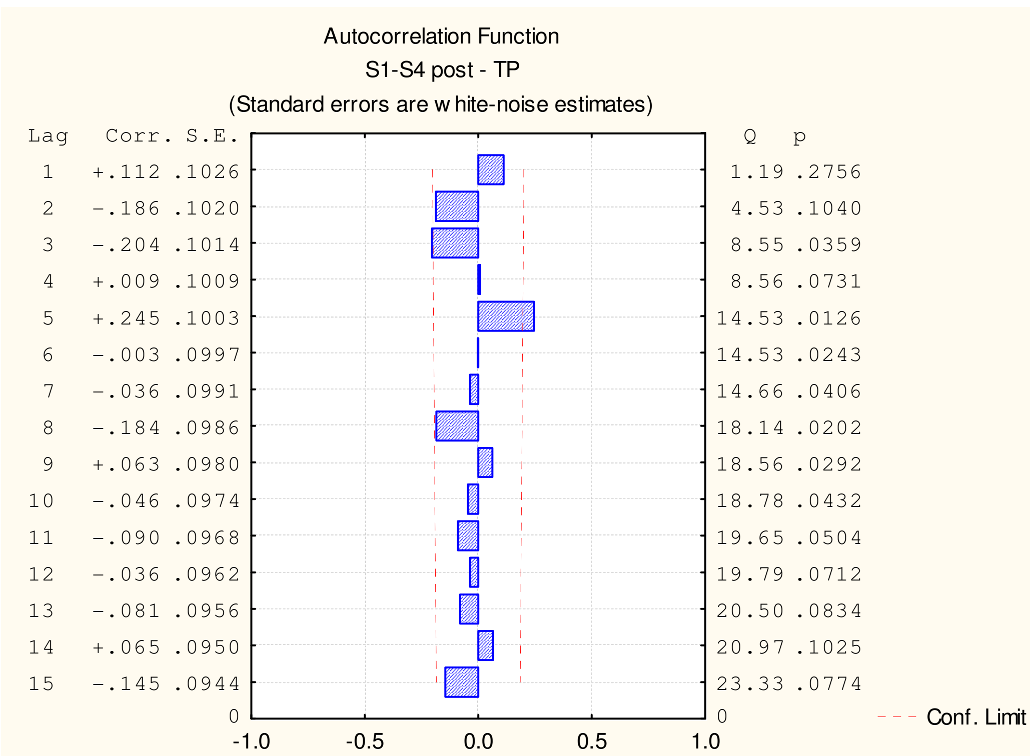
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S4 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 -.132 .1560
2 +.178 .1539
3 -.004 .1517
4 -.035 .1496
5 -.123 .1473
6 +.078 .1451
7 -.404 .1428
8 +.104 .1405
9 -.052 .1381
10 -.015 .1357
11 -.206 .1333
12 +.003 .1308
13 -.122 .1282
14 +.142 .1257
15 +.147 .1230
Q p
.71 .3993
2.06 .3578
2.06 .5608
2.11 .7156
2.81 .7296
3.10 .7962
11.09 .1346
11.64 .1679
11.78 .2259
11.80 .2990
14.18 .2232
14.18 .2893
15.09 .3019
16.36 .2920
17.80 .2736
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S4 post - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 +.112 .1026
2 -.186 .1020
3 -.204 .1014
4 +.009 .1009
5 +.245 .1003
6 -.003 .0997
7 -.036 .0991
8 -.184 .0986
9 +.063 .0980
10 -.046 .0974
11 -.090 .0968
12 -.036 .0962
13 -.081 .0956
14 +.065 .0950
15 -.145 .0944
Q p
1.19 .2756
4.53 .1040
8.55 .0359
8.56 .0731
14.53 .0126
14.53 .0243
14.66 .0406
18.14 .0202
18.56 .0292
18.78 .0432
19.65 .0504
19.79 .0712
20.50 .0834
20.97 .1025
23.33 .0774
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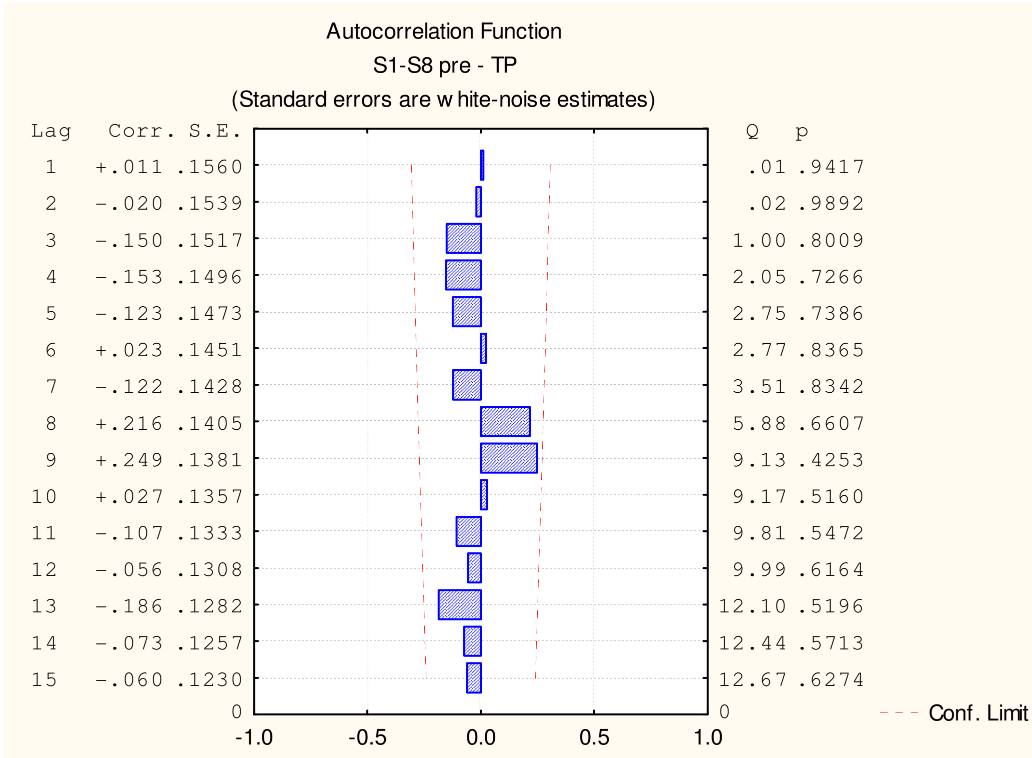
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S8 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.011 .1560 .01 .9417
2 -.020 .1539 .02 .9892
3 -.150 .1517 1.00 .8009
4 -.153 .1496 2.05 .7266
5 -.123 .1473 2.75 .7386
6 +.023 .1451 2.77 .8365
7 -.122 .1428 3.51 .8342
8 +.216 .1405 5.88 .6607
9 +.249 .1381 9.13 .4253
10 +.027 .1357 9.17 .5160
11 -.107 .1333 9.81 .5472
12 -.056 .1308 9.99 .6164
13 -.186 .1282 12.10 .5196
14 -.073 .1257 12.44 .5713
15 -.060 .1230 12.67 .6274
0
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-0.5
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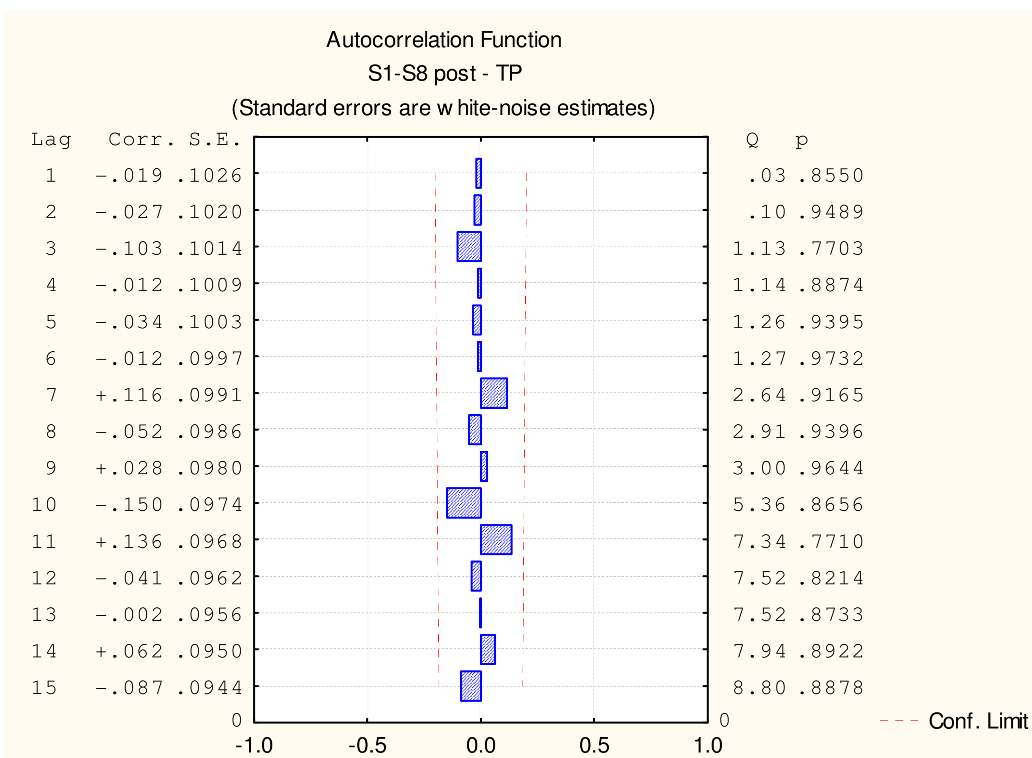
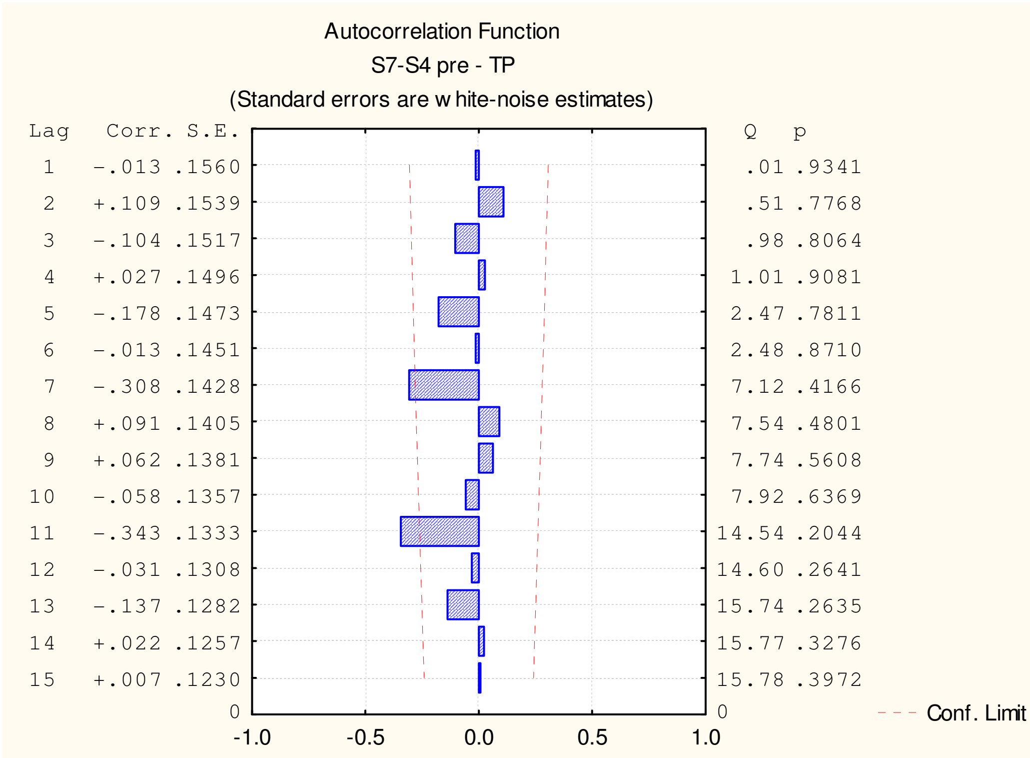
Conf. Limit


Autocorrelation Function
S1-S8 post - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.019 .1026 .03 .8550
2 -.027 .1020 .10 .9489
3 -.103 .1014 1.13 .7703
4 -.012 .1009 1.14 .8874
5 -.034 .1003 1.26 .9395
6 -.012 .0997 1.27 .9732
7 +.116 .0991 2.64 .9165
8 -.052 .0986 2.91 .9396
9 +.028 .0980 3.00 .9644
10 -.150 .0974 5.36 .8656
11 +.136 .0968 7.34 .7710
12 -.041 .0962 7.52 .8214
13 -.002 .0956 7.52 .8733
14 +.062 .0950 7.94 .8922
15 -.087 .0944 8.80 .8878
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S7-S4 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.013 .1560 .01 .9341
2 +.109 .1539 .51 .7768
3 -.104 .1517 .98 .8064
4 +.027 .1496 1.01 .9081
5 -.178 .1473 2.47 .7811
6 -.013 .1451 2.48 .8710
7 -.308 .1428 7.12 .4166
8 +.091 .1405 7.54 .4801
9 +.062 .1381 7.74 .5608
10 -.058 .1357 7.92 .6369
11 -.343 .1333 14.54 .2044
12 -.031 .1308 14.60 .2641
13 -.137 .1282 15.74 .2635
14 +.022 .1257 15.77 .3276
15 +.007 .1230 15.78 .3972
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0.5
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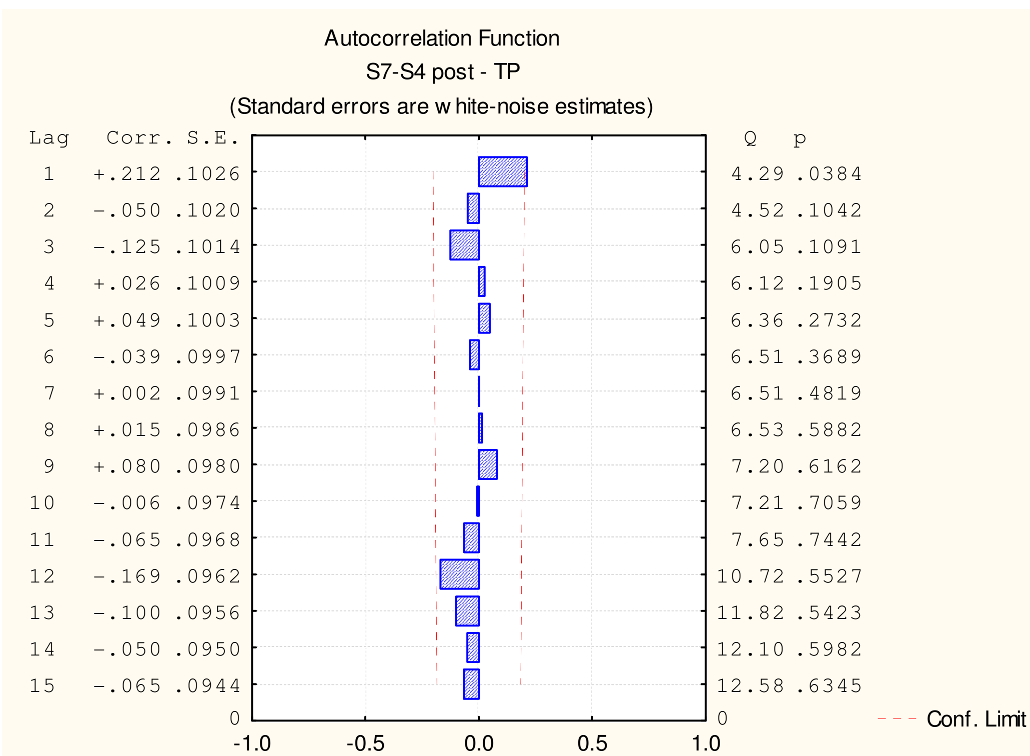
Conf. Limit


Autocorrelation Function
S7-S4 post - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.212 .1026 4.29 .0384
2 -.050 .1020 4.52 .1042
3 -.125 .1014 6.05 .1091
4 +.026 .1009 6.12 .1905
5 +.049 .1003 6.36 .2732
6 -.039 .0997 6.51 .3689
7 +.002 .0991 6.51 .4819
8 +.015 .0986 6.53 .5882
9 +.080 .0980 7.20 .6162
10 -.006 .0974 7.21 .7059
11 -.065 .0968 7.65 .7442
12 -.169 .0962 10.72 .5527
13 -.100 .0956 11.82 .5423
14 -.050 .0950 12.10 .5982
15 -.065 .0944 12.58 .6345
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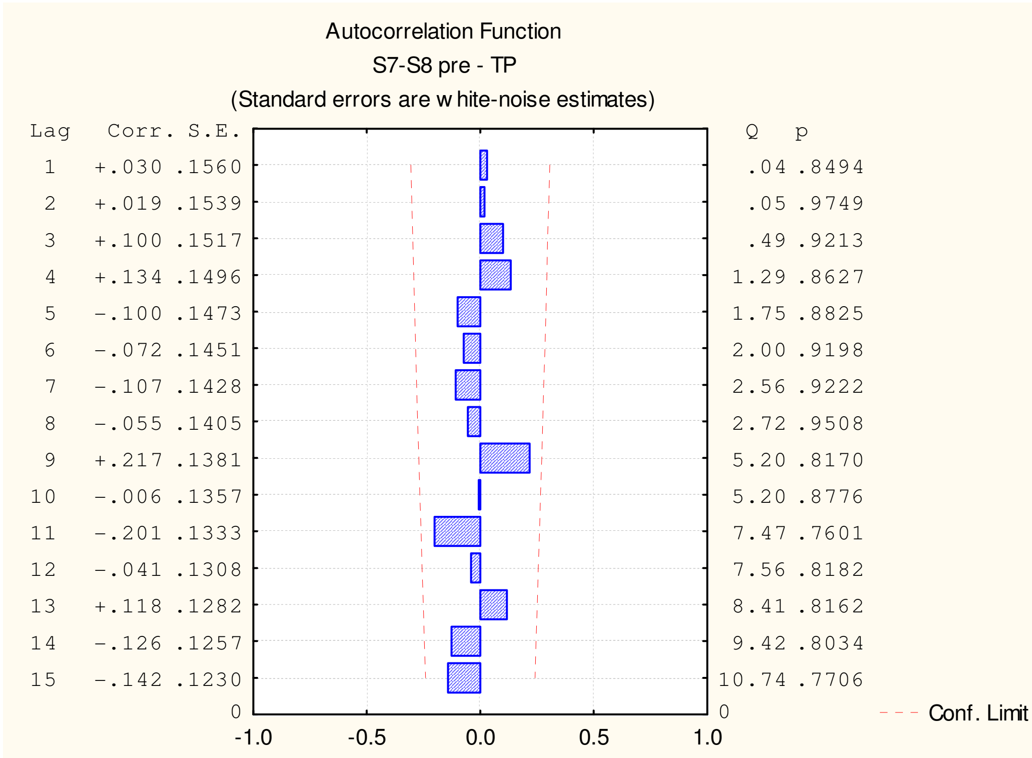
Conf. Limit


Autocorrelation Function
S7-S8 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.030 .1560 .04 .8494
2 +.019 .1539 .05 .9749
3 +.100 .1517 .49 .9213
4 +.134 .1496 1.29 .8627
5 -.100 .1473 1.75 .8825
6 -.072 .1451 2.00 .9198
7 -.107 .1428 2.56 .9222
8 -.055 .1405 2.72 .9508
9 +.217 .1381 5.20 .8170
10 -.006 .1357 5.20 .8776
11 -.201 .1333 7.47 .7601
12 -.041 .1308 7.56 .8182
13 +.118 .1282 8.41 .8162
14 -.126 .1257 9.42 .8034
15 -.142 .1230 10.74 .7706
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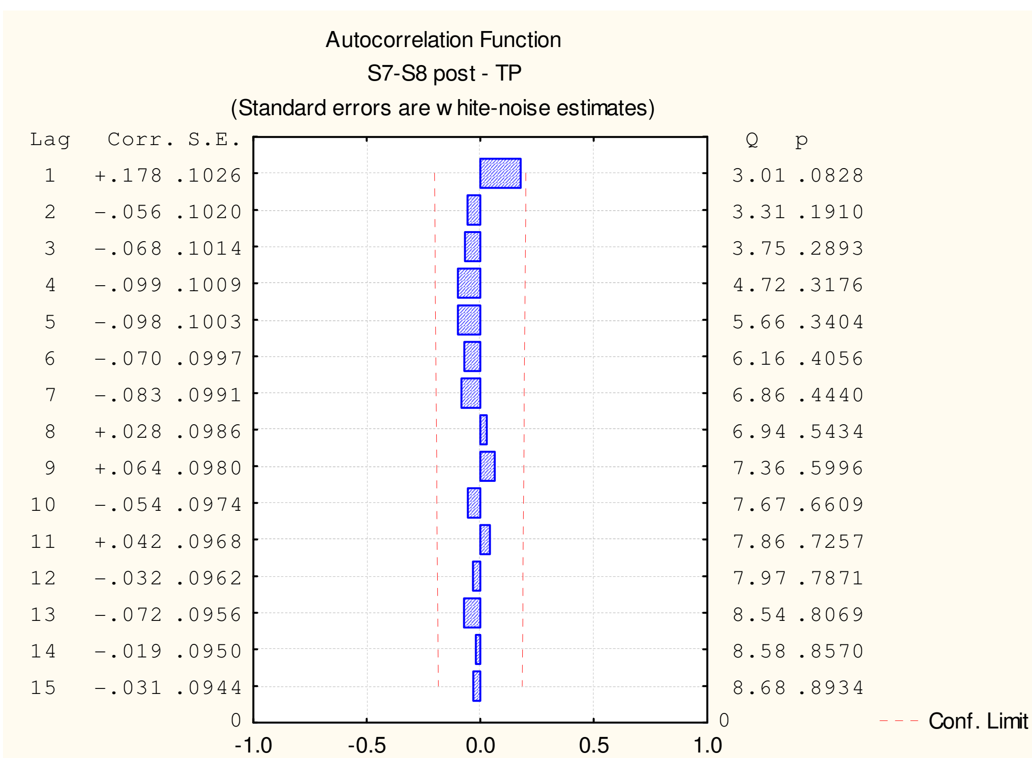
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S7-S8 post - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.178 .1026 3.01 .0828
2 -.056 .1020 3.31 .1910
3 -.068 .1014 3.75 .2893
4 -.099 .1009 4.72 .3176
5 -.098 .1003 5.66 .3404
6 -.070 .0997 6.16 .4056
7 -.083 .0991 6.86 .4440
8 +.028 .0986 6.94 .5434
9 +.064 .0980 7.36 .5996
10 -.054 .0974 7.67 .6609
11 +.042 .0968 7.86 .7257
12 -.032 .0962 7.97 .7871
13 -.072 .0956 8.54 .8069
14 -.019 .0950 8.58 .8570
15 -.031 .0944 8.68 .8934
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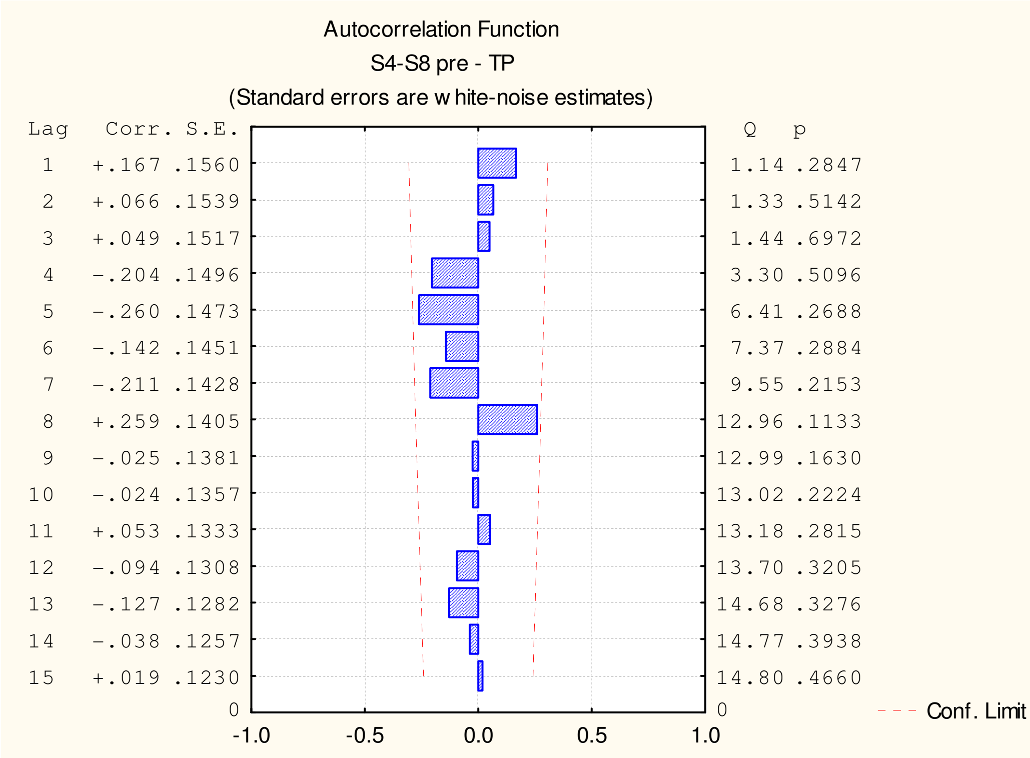
Conf. Limit


Autocorrelation Function
S4-S8 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 +.167 .1560
2 +.066 .1539
3 +.049 .1517
4 -.204 .1496
5 -.260 .1473
6 -.142 .1451
7 -.211 .1428
8 +.259 .1405
9 -.025 .1381
10 -.024 .1357
11 +.053 .1333
12 -.094 .1308
13 -.127 .1282
14 -.038 .1257
15 +.019 .1230
Q p
1.14 .2847
1.33 .5142
1.44 .6972
3.30 .5096
6.41 .2688
7.37 .2884
9.55 .2153
12.96 .1133
12.99 .1630
13.02 .2224
13.18 .2815
13.70 .3205
14.68 .3276
14.77 .3938
14.80 .4660
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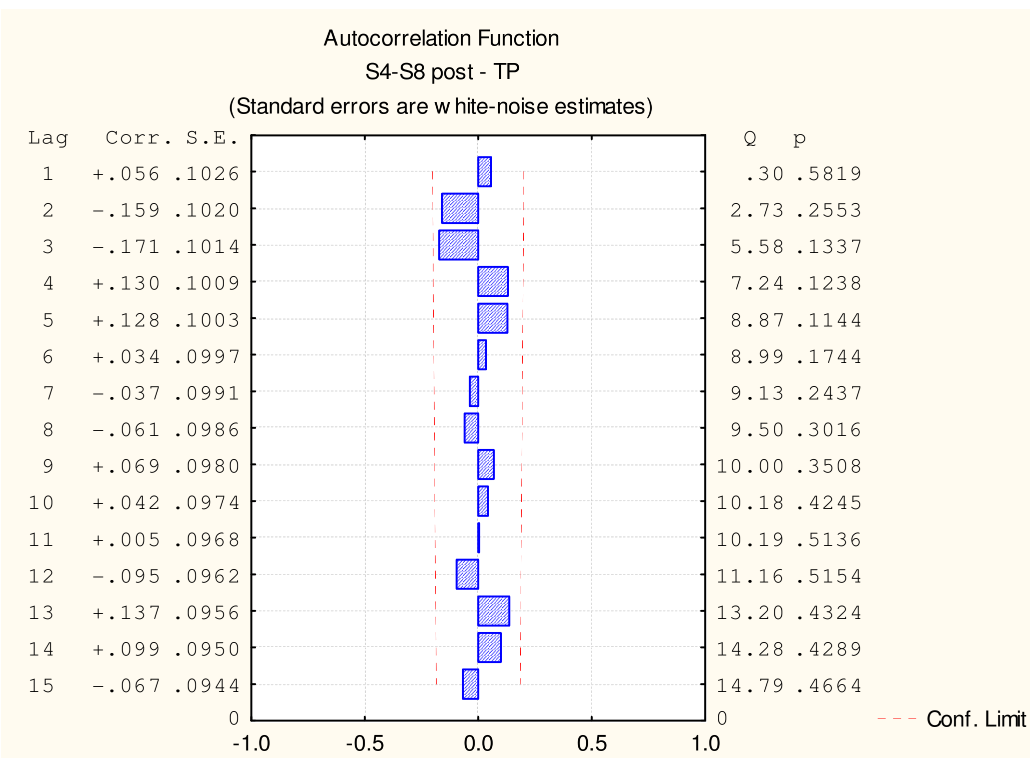
Conf. Limit


Autocorrelation Function
S4-S8 post - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
1 +.056 .1026
2 -.159 .1020
3 -.171 .1014
4 +.130 .1009
5 +.128 .1003
6 +.034 .0997
7 -.037 .0991
8 -.061 .0986
9 +.069 .0980
10 +.042 .0974
11 +.005 .0968
12 -.095 .0962
13 +.137 .0956
14 +.099 .0950
15 -.067 .0944
Q p
.30 .5819
2.73 .2553
5.58 .1337
7.24 .1238
8.87 .1144
8.99 .1744
9.13 .2437
9.50 .3016
10.00 .3508
10.18 .4245
10.19 .5136
11.16 .5154
13.20 .4324
14.28 .4289
14.79 .4664
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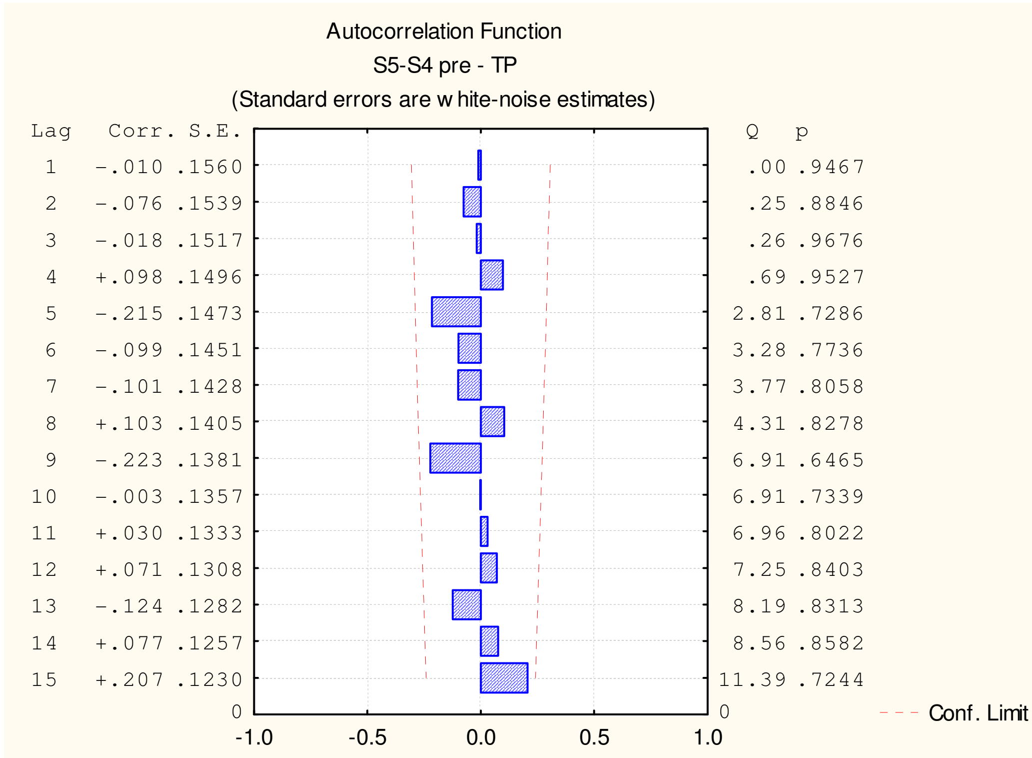
1.0
Conf. Limit


Autocorrelation Function
S5-S4 pre - TP
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 -.010 .1560 .00 .9467
2 -.076 .1539 .25 .8846
3 -.018 .1517 .26 .9676
4 +.098 .1496 .69 .9527
5 -.215 .1473 2.81 .7286
6 -.099 .1451 3.28 .7736
7 -.101 .1428 3.77 .8058
8 +.103 .1405 4.31 .8278
9 -.223 .1381 6.91 .6465
10 -.003 .1357 6.91 .7339
11 +.030 .1333 6.96 .8022
12 +.071 .1308 7.25 .8403
13 -.124 .1282 8.19 .8313
14 +.077 .1257 8.56 .8582
15 +.207 .1230 11.39 .7244
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Conf. Limit
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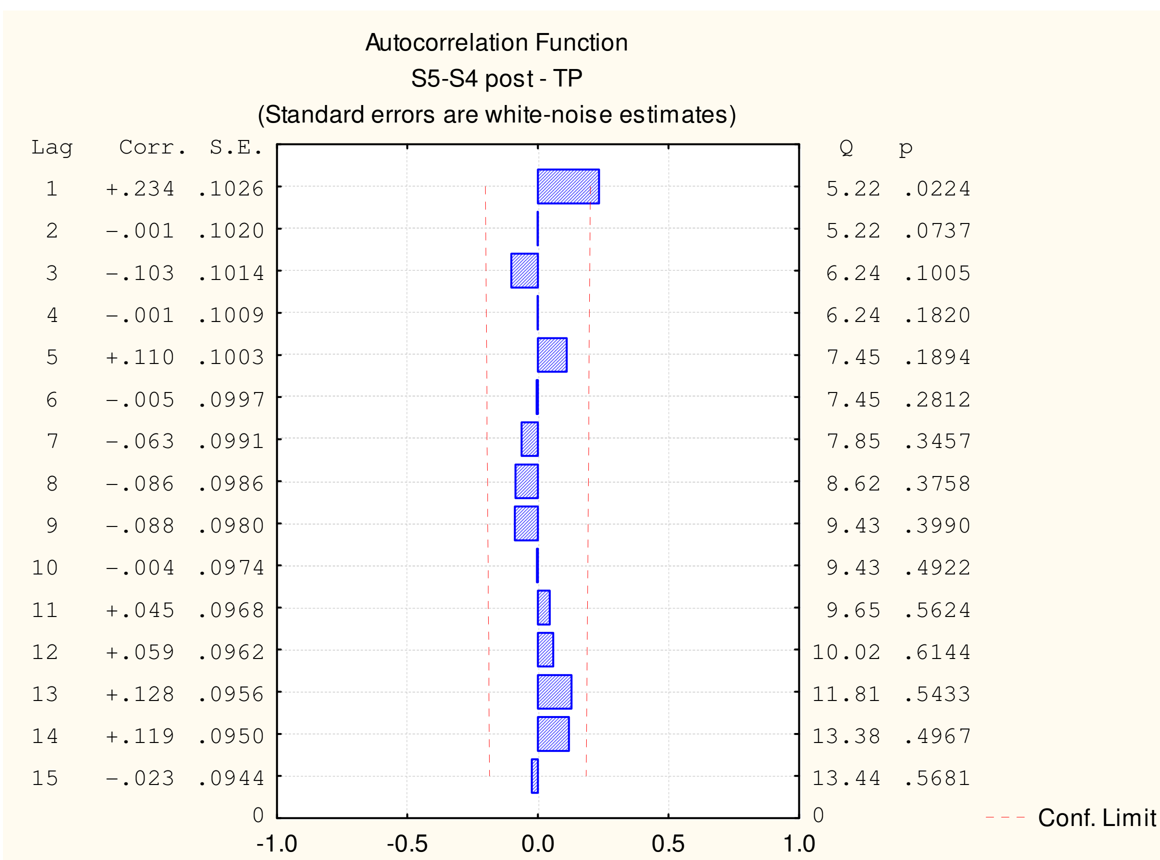
Autocorrelation Function
S5-S4 post - TP
(Standard errors are white-noise estimates)
Lag Corr. S.E.
Q p
1 +.234 .1026 5.22 .0224
2 -.001 .1020 5.22 .0737
3 -.103 .1014 6.24 .1005
4 -.001 .1009 6.24 .1820
5 +.110 .1003 7.45 .1894
6 -.005 .0997 7.45 .2812
7 -.063 .0991 7.85 .3457
8 -.086 .0986 8.62 .3758
9 -.088 .0980 9.43 .3990
10 -.004 .0974 9.43 .4922
11 +.045 .0968 9.65 .5624
12 +.059 .0962 10.02 .6144
13 +.128 .0956 11.81 .5433
14 +.119 .0950 13.38 .4967
15 -.023 .0944 13.44 .5681
0
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Conf. Limit
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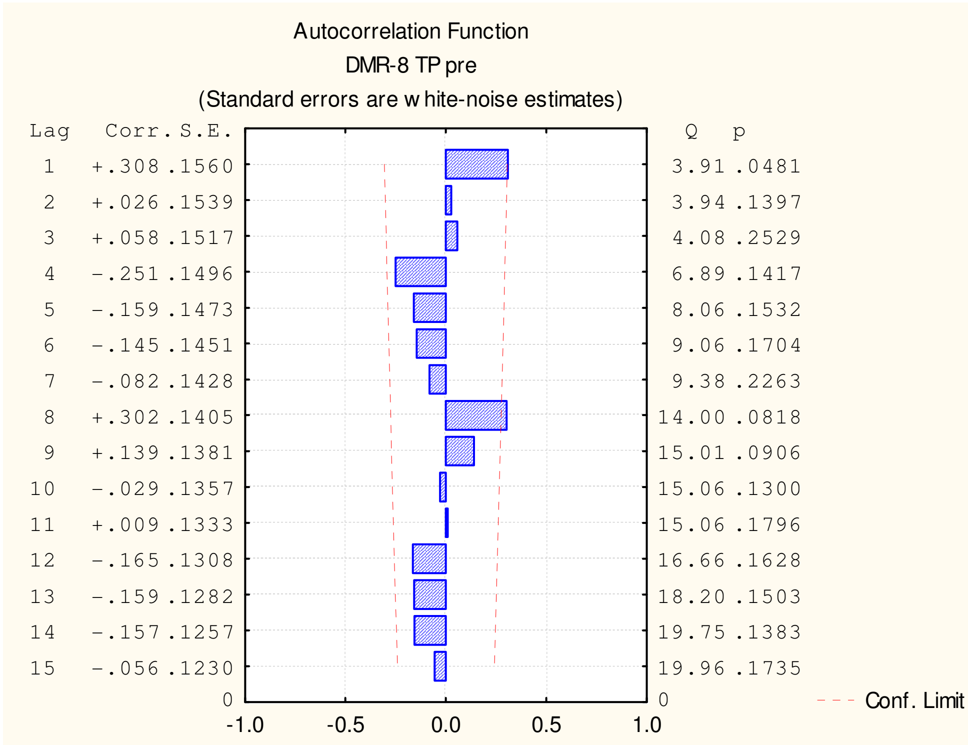
1.0


Autocorrelation Function
DMR-8 TP pre
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.308 .1560 3.91 .0481
2 +.026 .1539 3.94 .1397
3 +.058 .1517 4.08 .2529
4 -.251 .1496 6.89 .1417
5 -.159 .1473 8.06 .1532
6 -.145 .1451 9.06 .1704
7 -.082 .1428 9.38 .2263
8 +.302 .1405 14.00 .0818
9 +.139 .1381 15.01 .0906
10 -.029 .1357 15.06 .1300
11 +.009 .1333 15.06 .1796
12 -.165 .1308 16.66 .1628
13 -.159 .1282 18.20 .1503
14 -.157 .1257 19.75 .1383
15 -.056 .1230 19.96 .1735
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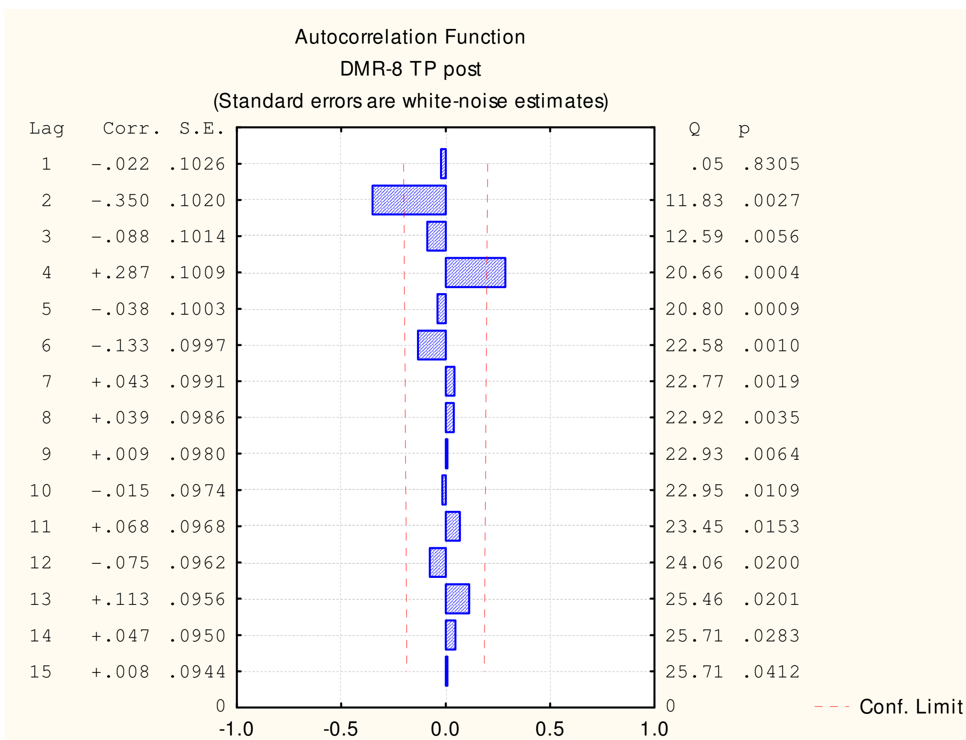
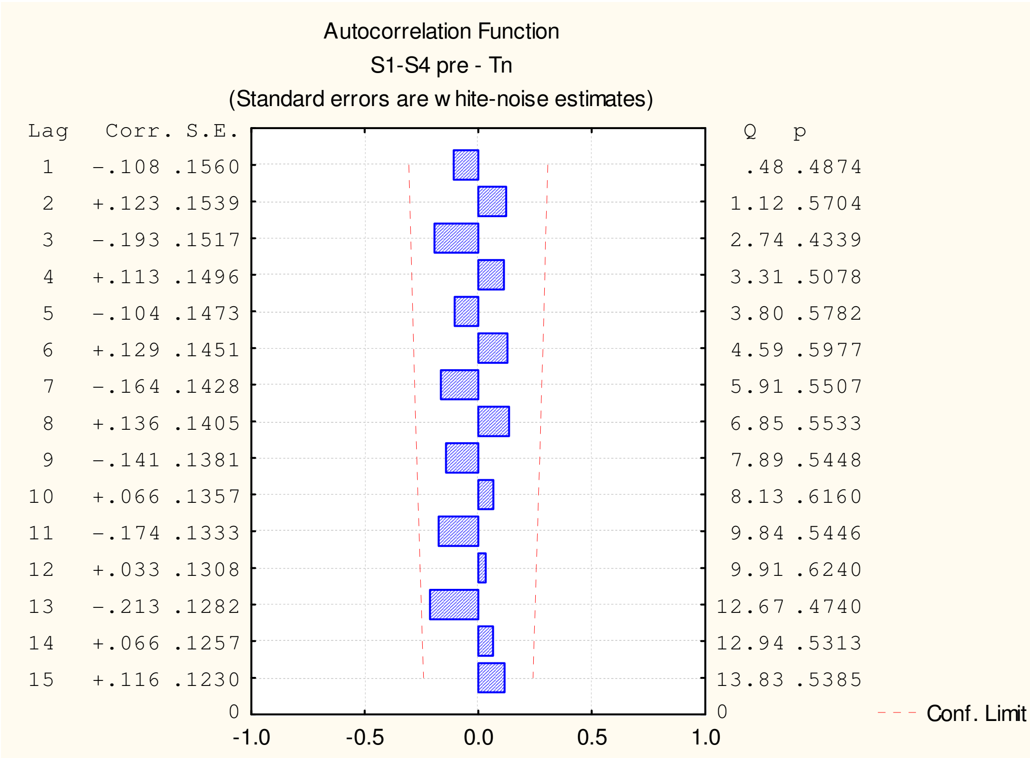
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
DMR-8 TP post
(Standard errors are white-noise estimates)
Lag Corr. S.E. Q p
1 -.022 .1026 .05 .8305
2 -.350 .1020 11.83 .0027
3 -.088 .1014 12.59 .0056
4 +.287 .1009 20.66 .0004
5 -.038 .1003 20.80 .0009
6 -.133 .0997 22.58 .0010
7 +.043 .0991 22.77 .0019
8 +.039 .0986 22.92 .0035
9 +.009 .0980 22.93 .0064
10 -.015 .0974 22.95 .0109
11 +.068 .0968 23.45 .0153
12 -.075 .0962 24.06 .0200
13 +.113 .0956 25.46 .0201
14 +.047 .0950 25.71 .0283
15 +.008 .0944 25.71 .0412
0 0
-1.0 -0.5 0.0 0.5 1.0
Conf. Limit


Autocorrelation Function
S1-S4 pre - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.108 .1560 .48 .4874
2 +.123 .1539 1.12 .5704
3 -.193 .1517 2.74 .4339
4 +.113 .1496 3.31 .5078
5 -.104 .1473 3.80 .5782
6 +.129 .1451 4.59 .5977
7 -.164 .1428 5.91 .5507
8 +.136 .1405 6.85 .5533
9 -.141 .1381 7.89 .5448
10 +.066 .1357 8.13 .6160
11 -.174 .1333 9.84 .5446
12 +.033 .1308 9.91 .6240
13 -.213 .1282 12.67 .4740
14 +.066 .1257 12.94 .5313
15 +.116 .1230 13.83 .5385
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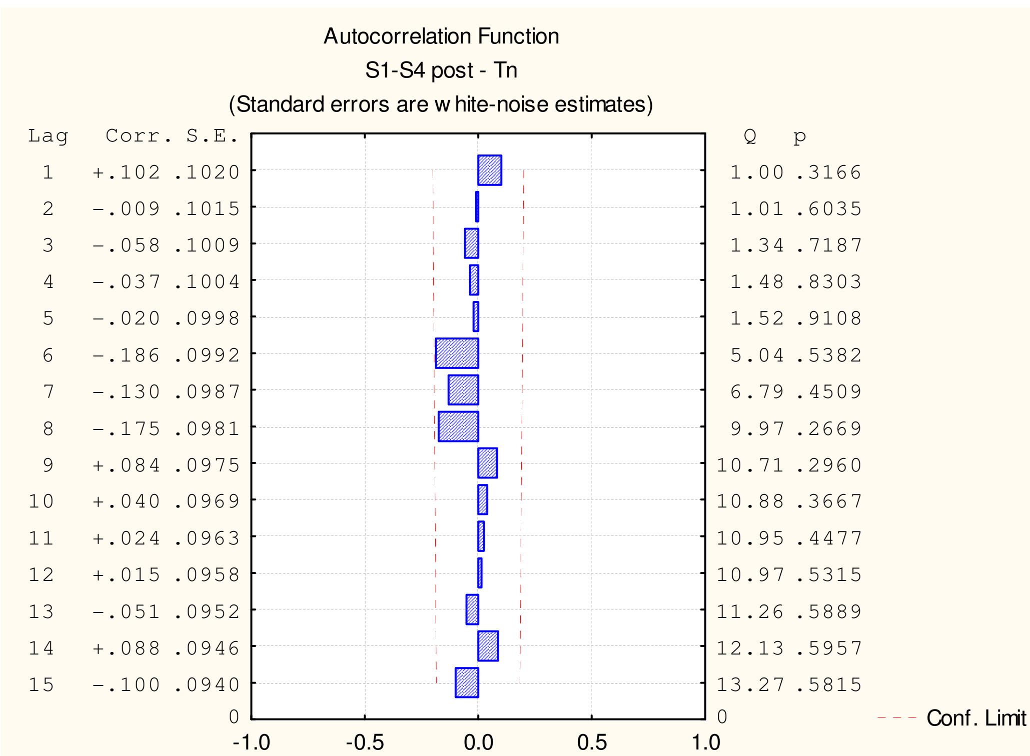
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S4 post - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.102 .1020 1.00 .3166
2 -.009 .1015 1.01 .6035
3 -.058 .1009 1.34 .7187
4 -.037 .1004 1.48 .8303
5 -.020 .0998 1.52 .9108
6 -.186 .0992 5.04 .5382
7 -.130 .0987 6.79 .4509
8 -.175 .0981 9.97 .2669
9 +.084 .0975 10.71 .2960
10 +.040 .0969 10.88 .3667
11 +.024 .0963 10.95 .4477
12 +.015 .0958 10.97 .5315
13 -.051 .0952 11.26 .5889
14 +.088 .0946 12.13 .5957
15 -.100 .0940 13.27 .5815
0
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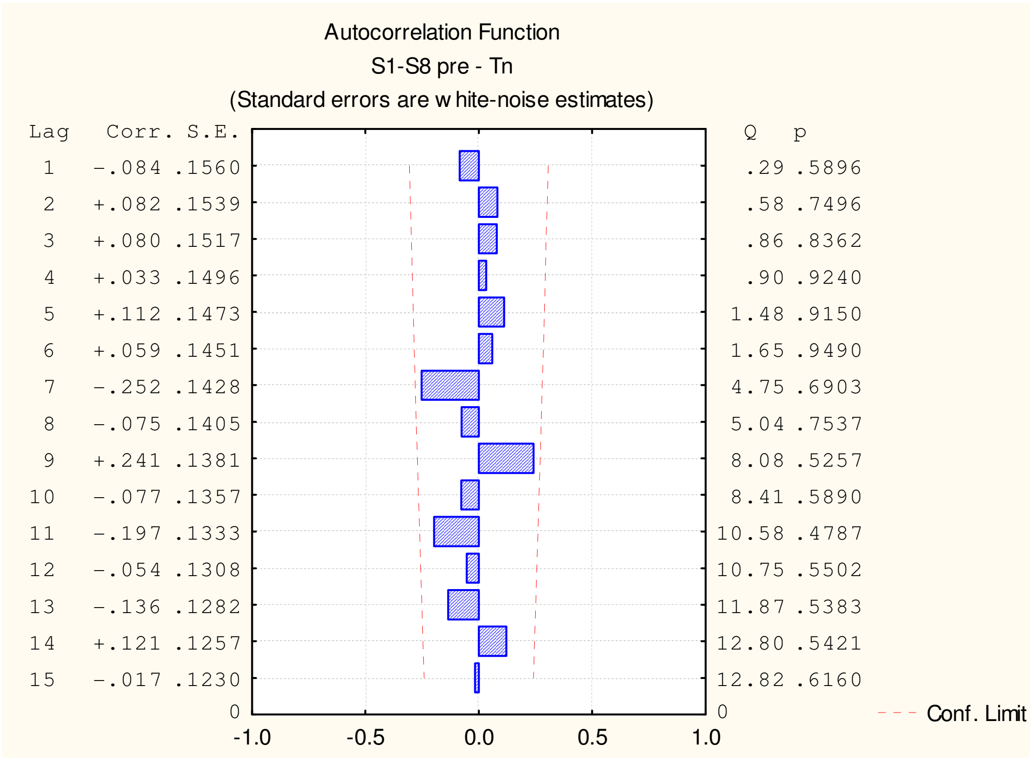
-0.5
0.0
0.5
1.0
Conf. Limit


Autocorrelation Function
S1-S8 pre - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 -.084 .1560 .29 .5896
2 +.082 .1539 .58 .7496
3 +.080 .1517 .86 .8362
4 +.033 .1496 .90 .9240
5 +.112 .1473 1.48 .9150
6 +.059 .1451 1.65 .9490
7 -.252 .1428 4.75 .6903
8 -.075 .1405 5.04 .7537
9 +.241 .1381 8.08 .5257
10 -.077 .1357 8.41 .5890
11 -.197 .1333 10.58 .4787
12 -.054 .1308 10.75 .5502
13 -.136 .1282 11.87 .5383
14 +.121 .1257 12.80 .5421
15 -.017 .1230 12.82 .6160
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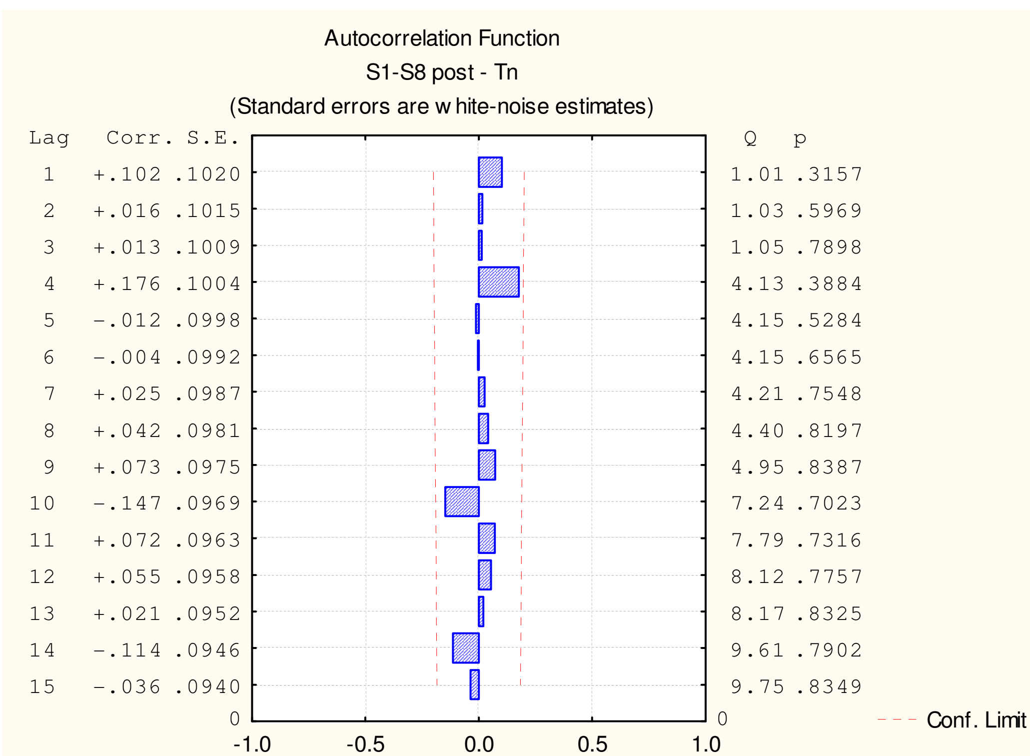
Conf. Limit


Autocorrelation Function
S1-S8 post - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.102 .1020 1.01 .3157
2 +.016 .1015 1.03 .5969
3 +.013 .1009 1.05 .7898
4 +.176 .1004 4.13 .3884
5 -.012 .0998 4.15 .5284
6 -.004 .0992 4.15 .6565
7 +.025 .0987 4.21 .7548
8 +.042 .0981 4.40 .8197
9 +.073 .0975 4.95 .8387
10 -.147 .0969 7.24 .7023
11 +.072 .0963 7.79 .7316
12 +.055 .0958 8.12 .7757
13 +.021 .0952 8.17 .8325
14 -.114 .0946 9.61 .7902
15 -.036 .0940 9.75 .8349
0 0
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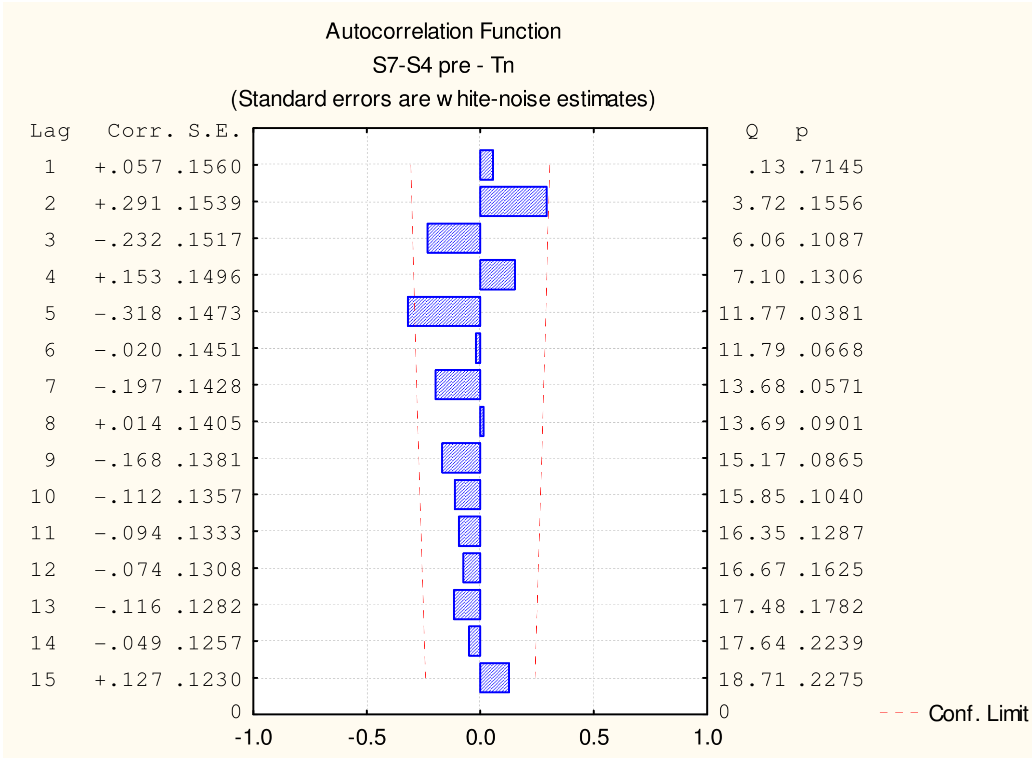
Conf. Limit


Autocorrelation Function
S7-S4 pre - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.057 .1560 .13 .7145
2 +.291 .1539 3.72 .1556
3 -.232 .1517 6.06 .1087
4 +.153 .1496 7.10 .1306
5 -.318 .1473 11.77 .0381
6 -.020 .1451 11.79 .0668
7 -.197 .1428 13.68 .0571
8 +.014 .1405 13.69 .0901
9 -.168 .1381 15.17 .0865
10 -.112 .1357 15.85 .1040
11 -.094 .1333 16.35 .1287
12 -.074 .1308 16.67 .1625
13 -.116 .1282 17.48 .1782
14 -.049 .1257 17.64 .2239
15 +.127 .1230 18.71 .2275
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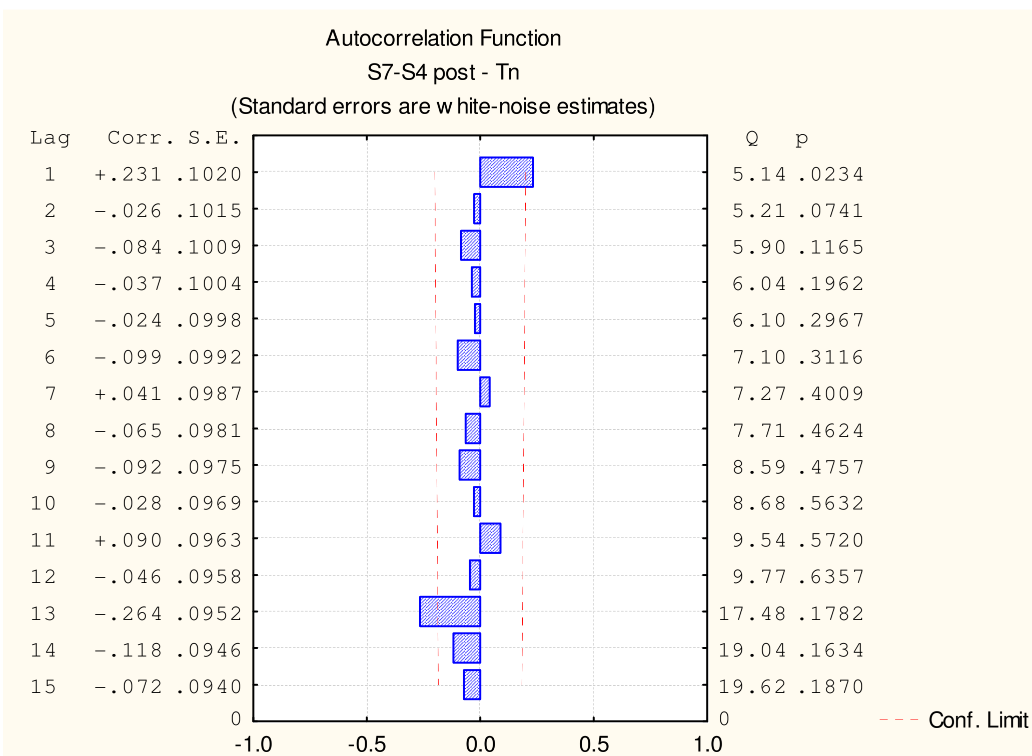
Conf. Limit


Autocorrelation Function
S7-S4 post - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.231 .1020 5.14 .0234
2 -.026 .1015 5.21 .0741
3 -.084 .1009 5.90 .1165
4 -.037 .1004 6.04 .1962
5 -.024 .0998 6.10 .2967
6 -.099 .0992 7.10 .3116
7 +.041 .0987 7.27 .4009
8 -.065 .0981 7.71 .4624
9 -.092 .0975 8.59 .4757
10 -.028 .0969 8.68 .5632
11 +.090 .0963 9.54 .5720
12 -.046 .0958 9.77 .6357
13 -.264 .0952 17.48 .1782
14 -.118 .0946 19.04 .1634
15 -.072 .0940 19.62 .1870
0
0
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-0.5
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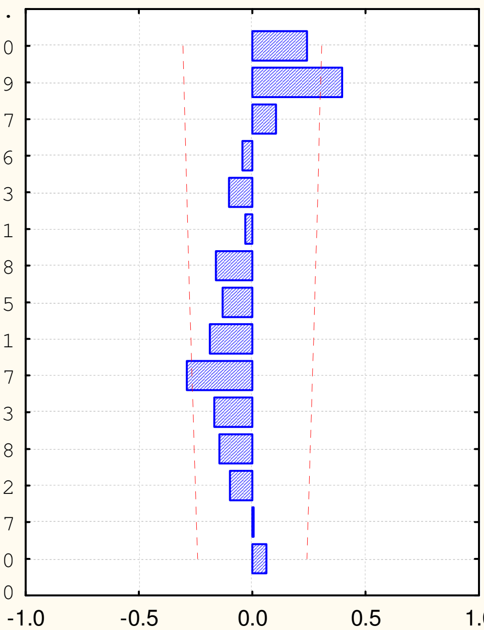
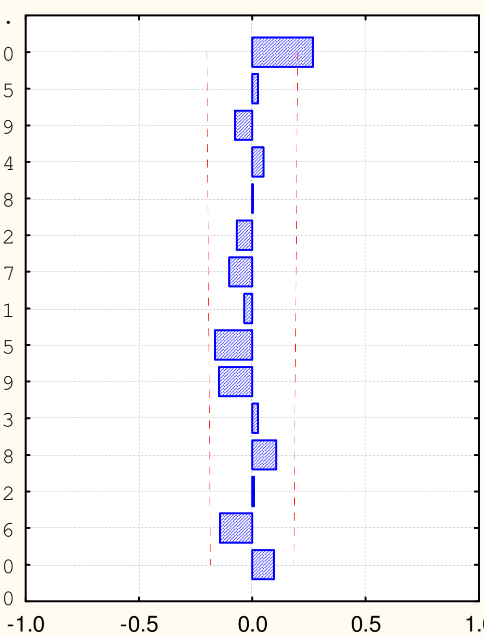
0.5
1.0
Conf. Limit

Autocorrelation Function
S7-S8 pre - Tn
(Standard errors are w hite-noise estimates)

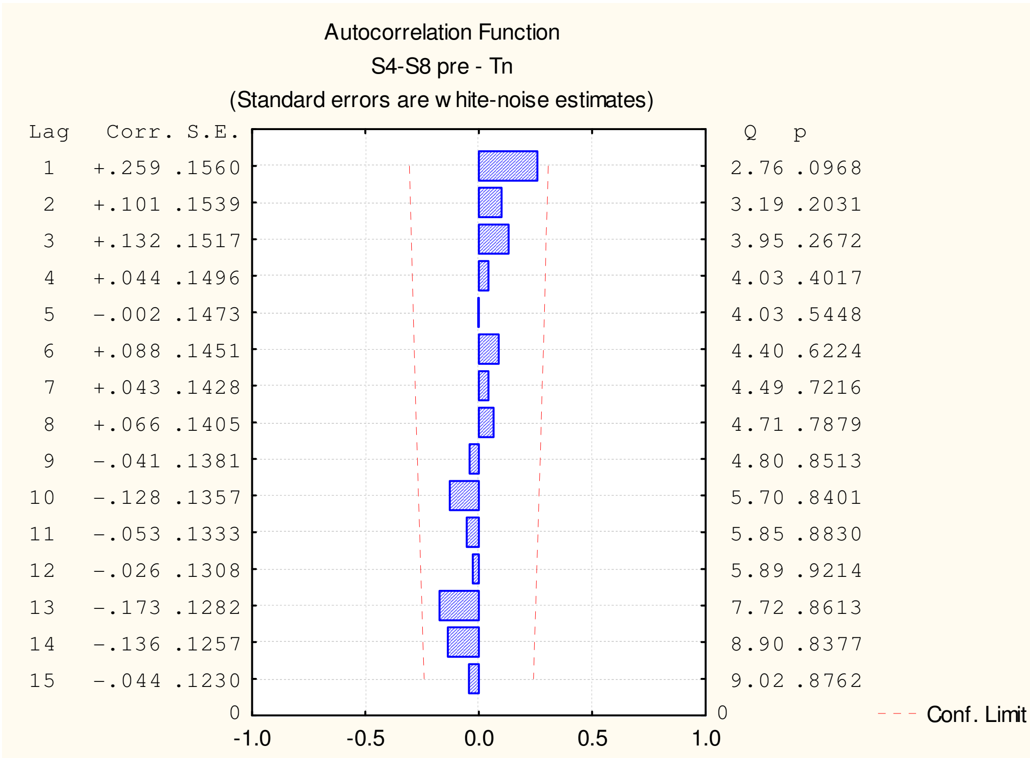



| Lag | Corr. | S.E. | Q | p |
|---|---|---|---|---|
| 1 | +.242 | .1560 | 2.40 | .1214 |
| 2 | +.397 | .1539 | 9.05 | .0108 |
| 3 | +.104 | .1517 | 9.52 | .0231 |
| 4 | -.043 | .1496 | 9.61 | .0476 |
| 5 | -.102 | .1473 | 10.09 | .0727 |
| 6 | -.031 | .1451 | 10.14 | .1191 |
| 7 | -.161 | .1428 | 11.40 | .1220 |
| 8 | -.130 | .1405 | 12.26 | .1398 |
| 9 | -.187 | .1381 | 14.10 | .1190 |
| 10 | -.289 | .1357 | 18.63 | .0453 |
| 11 | -.167 | .1333 | 20.20 | .0427 |
| 12 | -.144 | .1308 | 21.42 | .0446 |
| 13 | -.099 | .1282 | 22.01 | .0552 |
| 14 | +.007 | .1257 | 22.02 | .0784 |
| 15 | +.063 | .1230 | 22.28 | .1009 |

Autocorrelation Function
S7-S8 post - Tn
(Standard errors are w hite-noise estimates)




| Lag | Corr. | S.E. | Q | p |
|---|---|---|---|---|
| 1 | +.269 | .1020 | 6.94 | .0084 |
| 2 | +.026 | .1015 | 7.01 | .0301 |
| 3 | -.077 | .1009 | 7.58 | .0555 |
| 4 | +.050 | .1004 | 7.83 | .0982 |
| 5 | +.002 | .0998 | 7.83 | .1661 |
| 6 | -.069 | .0992 | 8.31 | .2162 |
| 7 | -.101 | .0987 | 9.36 | .2279 |
| 8 | -.034 | .0981 | 9.48 | .3033 |
| 9 | -.164 | .0975 | 12.31 | .1966 |
| 10 | -.148 | .0969 | 14.64 | .1460 |
| 11 | +.025 | .0963 | 14.70 | .1965 |
| 12 | +.106 | .0958 | 15.93 | .1945 |
| 13 | +.008 | .0952 | 15.94 | .2526 |
| 14 | -.141 | .0946 | 18.17 | .1992 |
| 15 | +.097 | .0940 | 19.23 | .2036 |


Autocorrelation Function
S4-S8 pre - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 +.259 .1560 2.76 .0968
2 +.101 .1539 3.19 .2031
3 +.132 .1517 3.95 .2672
4 +.044 .1496 4.03 .4017
5 -.002 .1473 4.03 .5448
6 +.088 .1451 4.40 .6224
7 +.043 .1428 4.49 .7216
8 +.066 .1405 4.71 .7879
9 -.041 .1381 4.80 .8513
10 -.128 .1357 5.70 .8401
11 -.053 .1333 5.85 .8830
12 -.026 .1308 5.89 .9214
13 -.173 .1282 7.72 .8613
14 -.136 .1257 8.90 .8377
15 -.044 .1230 9.02 .8762
0
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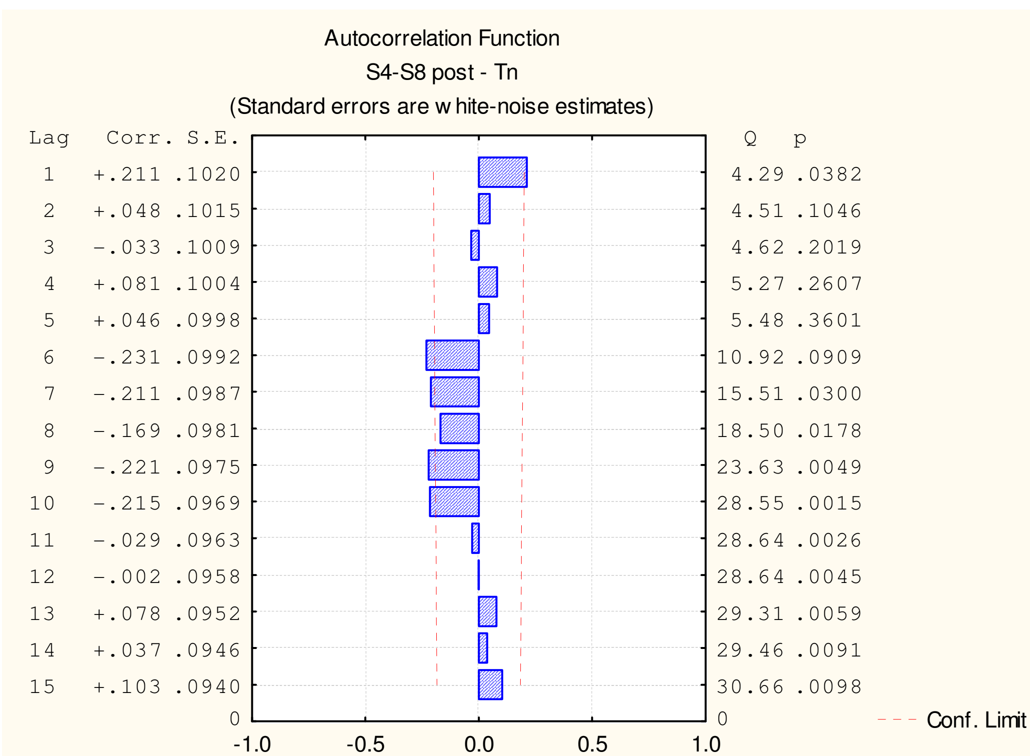
Conf. Limit


Autocorrelation Function
S4-S8 post - Tn
(Standard errors are w hite-noise estimates)
Lag Corr. S.E.
Q p
1 +.211 .1020 4.29 .0382
2 +.048 .1015 4.51 .1046
3 -.033 .1009 4.62 .2019
4 +.081 .1004 5.27 .2607
5 +.046 .0998 5.48 .3601
6 -.231 .0992 10.92 .0909
7 -.211 .0987 15.51 .0300
8 -.169 .0981 18.50 .0178
9 -.221 .0975 23.63 .0049
10 -.215 .0969 28.55 .0015
11 -.029 .0963 28.64 .0026
12 -.002 .0958 28.64 .0045
13 +.078 .0952 29.31 .0059
14 +.037 .0946 29.46 .0091
15 +.103 .0940 30.66 .0098
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-0.5
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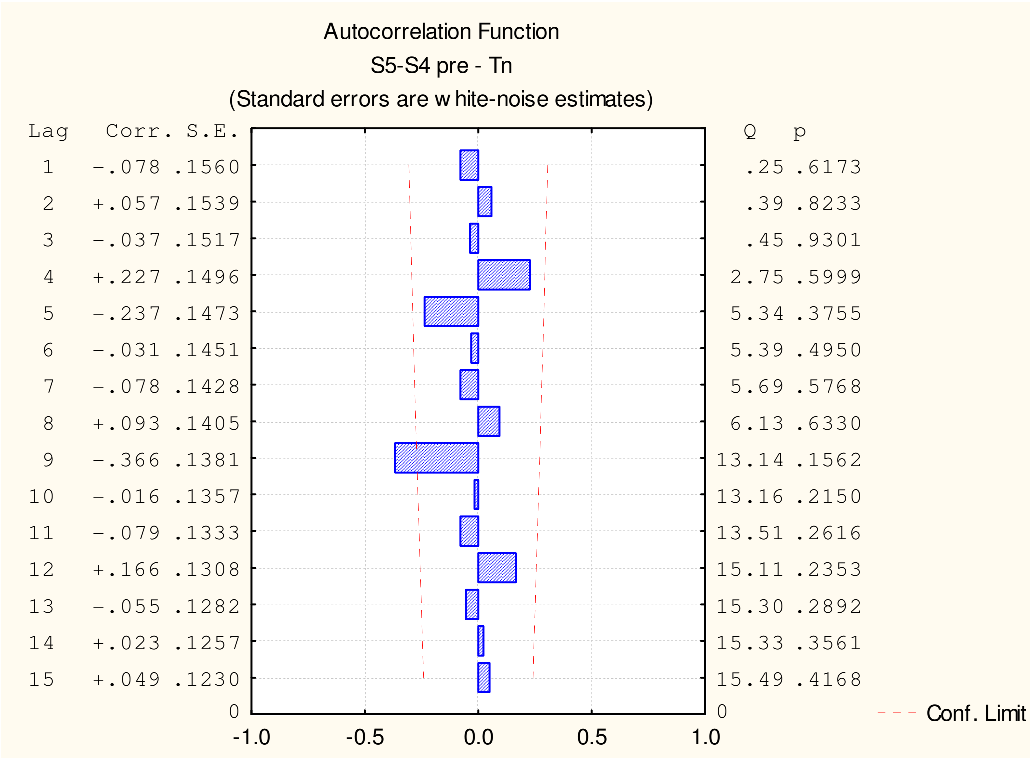
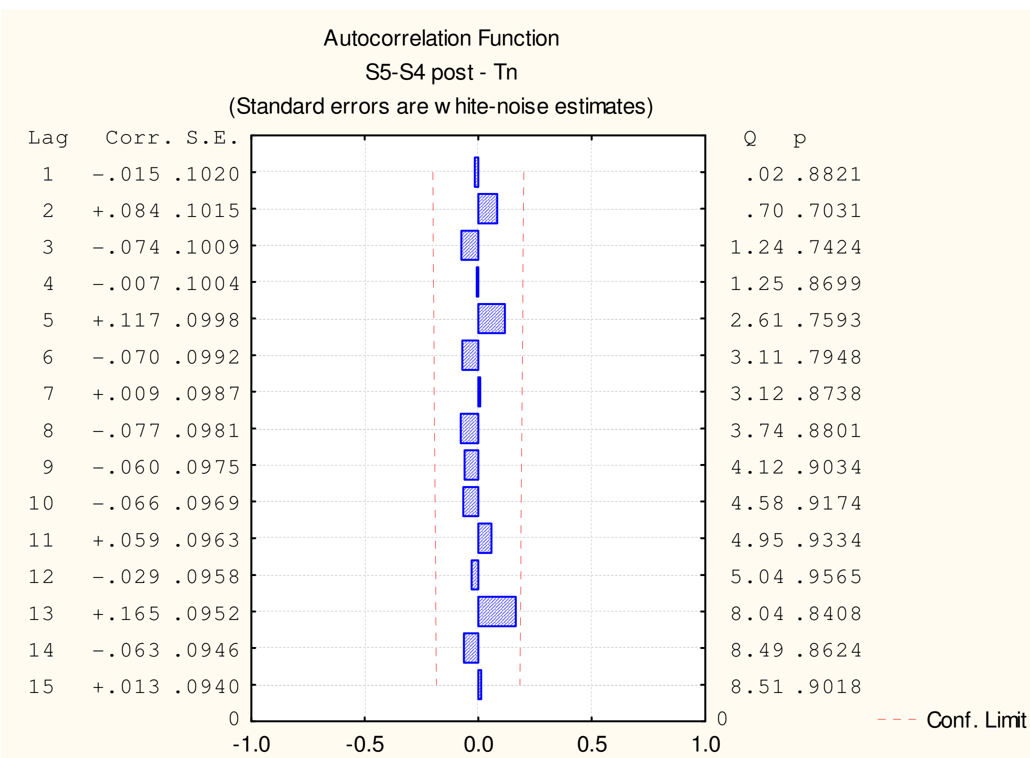
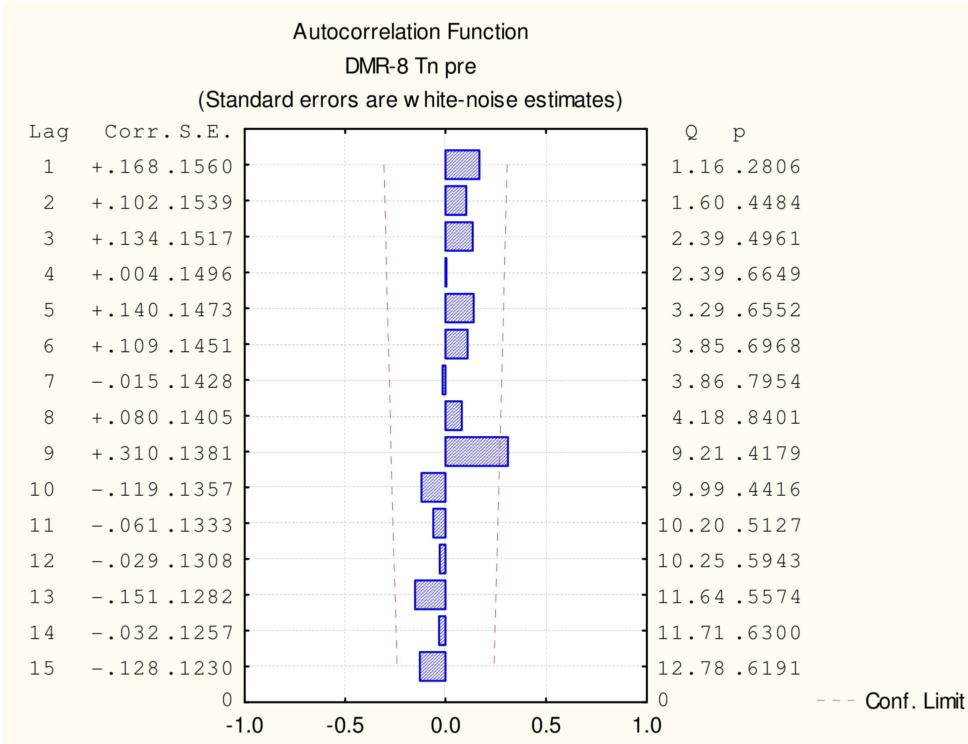
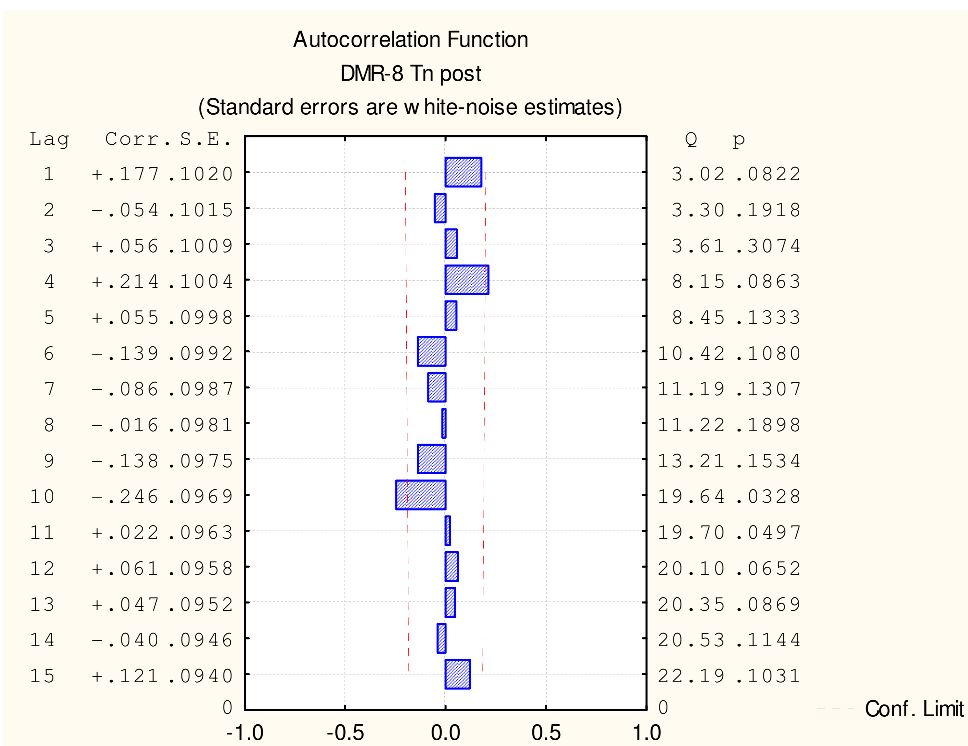
Conf. Limit

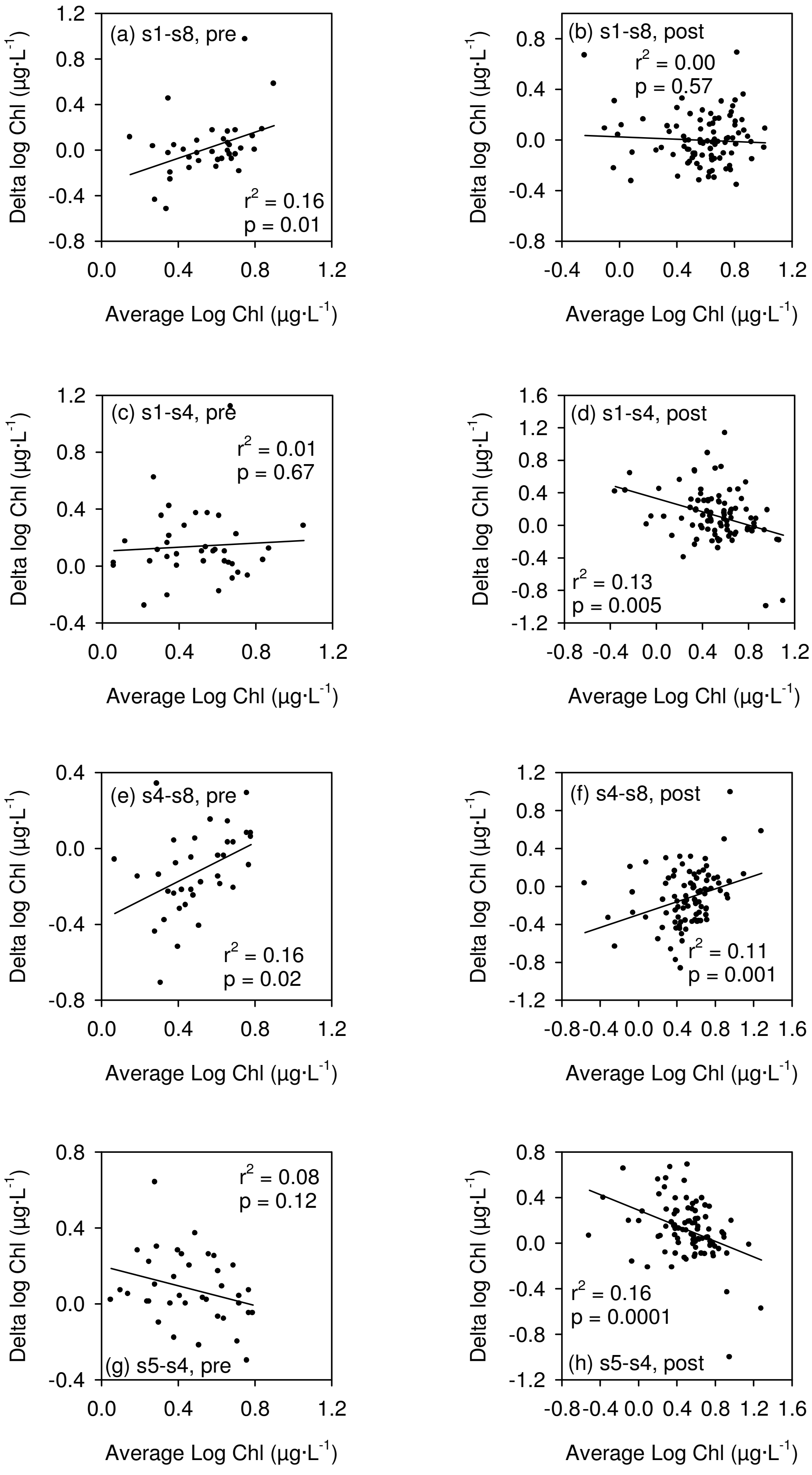

Autocorrelation Function
S5-S4 pre - Tn
(Standard errors are white-noise estimates)
Lag Corr. S.E. Q p
1 -.078 .1560 .25 .6173
2 +.057 .1539 .39 .8233
3 -.037 .1517 .45 .9301
4 +.227 .1496 2.75 .5999
5 -.237 .1473 5.34 .3755
6 -.031 .1451 5.39 .4950
7 -.078 .1428 5.69 .5768
8 +.093 .1405 6.13 .6330
9 -.366 .1381 13.14 .1562
10 -.016 .1357 13.16 .2150
11 -.079 .1333 13.51 .2616
12 +.166 .1308 15.11 .2353
13 -.055 .1282 15.30 .2892
14 +.023 .1257 15.33 .3561
15 +.049 .1230 15.49 .4168
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit

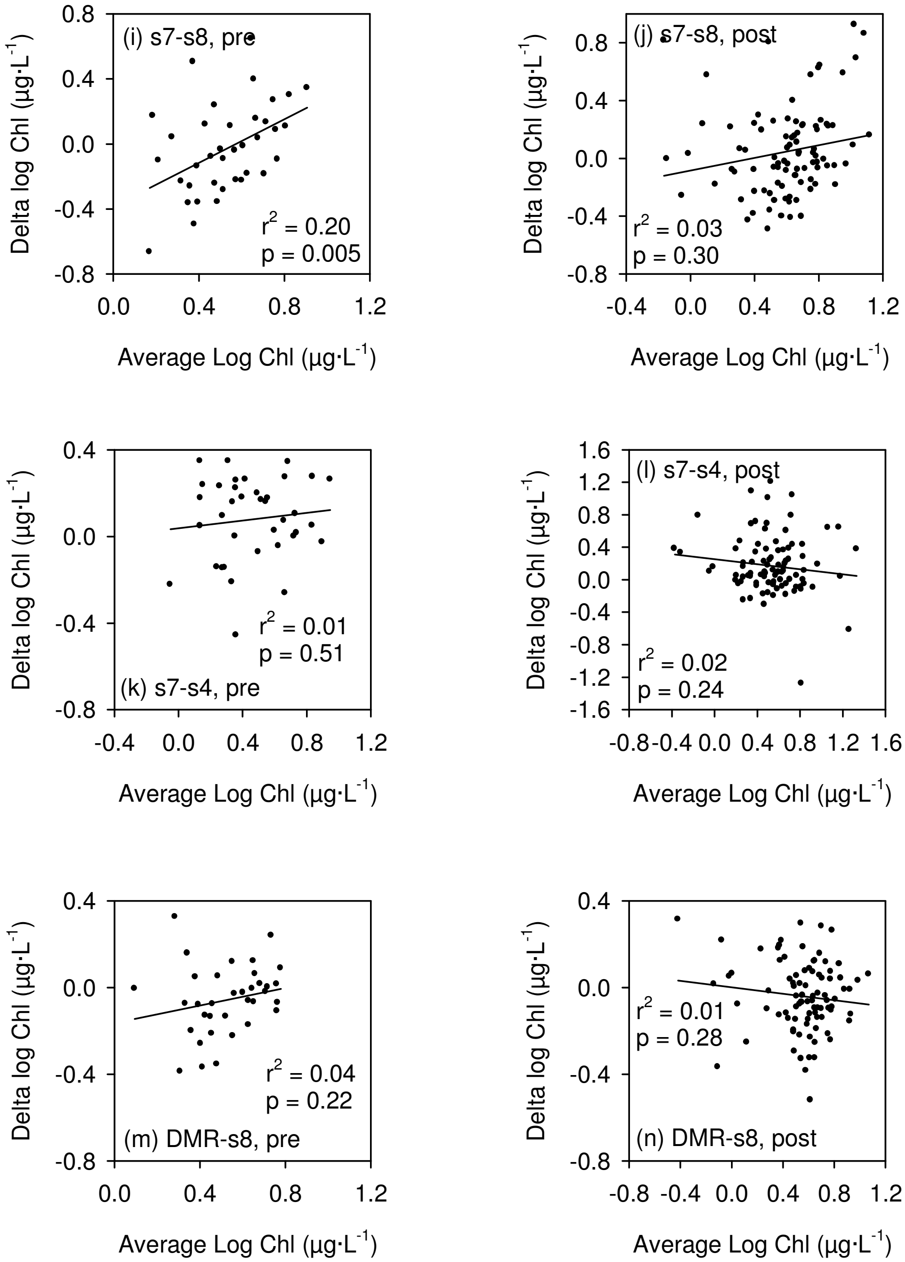

Autocorrelation Function
S5-S4 post - Tn
(Standard errors are white-noise estimates)
Lag Corr. S.E. Q p
1 -.015 .1020 .02 .8821
2 +.084 .1015 .70 .7031
3 -.074 .1009 1.24 .7424
4 -.007 .1004 1.25 .8699
5 +.117 .0998 2.61 .7593
6 -.070 .0992 3.11 .7948
7 +.009 .0987 3.12 .8738
8 -.077 .0981 3.74 .8801
9 -.060 .0975 4.12 .9034
10 -.066 .0969 4.58 .9174
11 +.059 .0963 4.95 .9334
12 -.029 .0958 5.04 .9565
13 +.165 .0952 8.04 .8408
14 -.063 .0946 8.49 .8624
15 +.013 .0940 8.51 .9018
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit

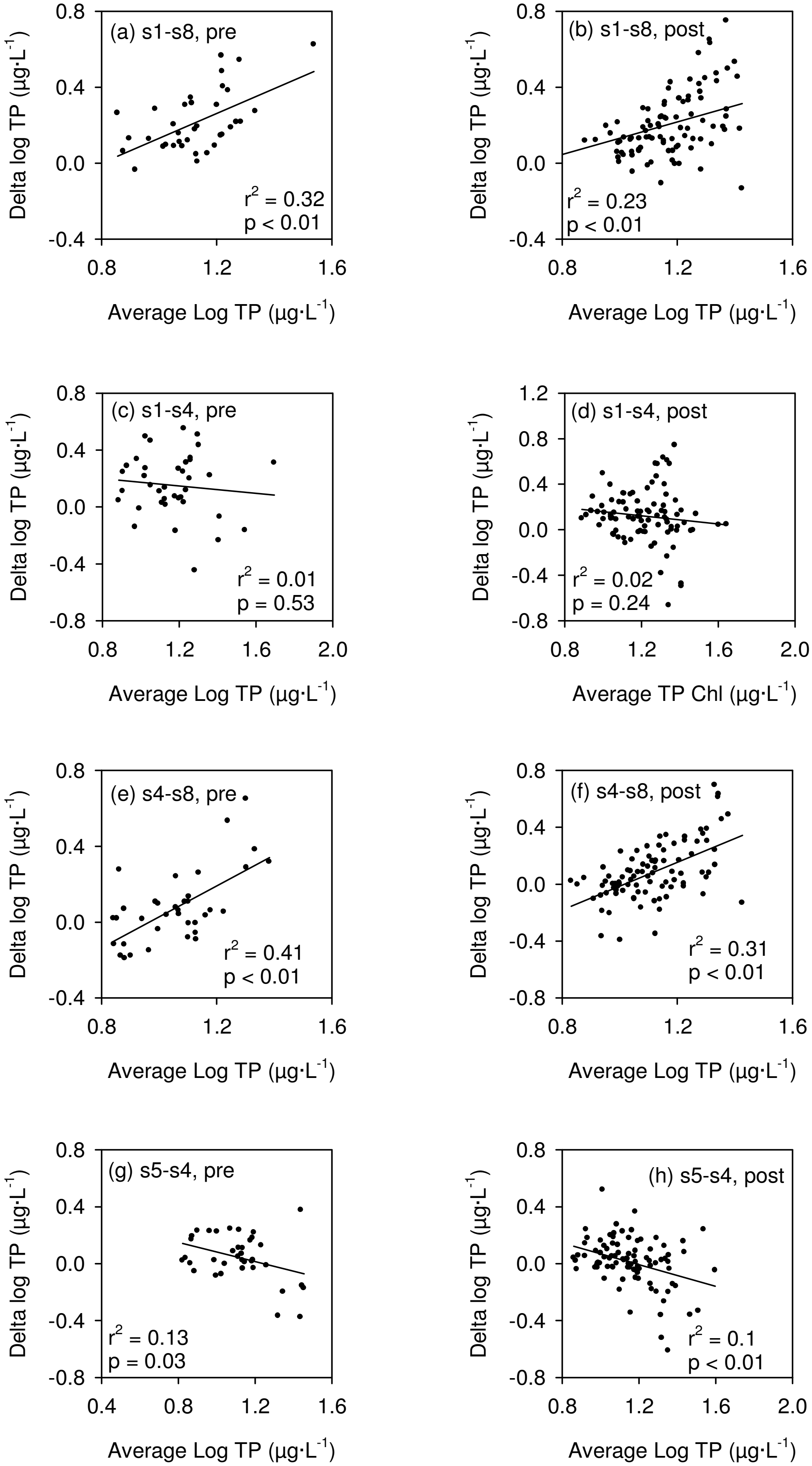

Autocorrelation Function
DMR-8 Tn pre
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.168 .1560 1.16 .2806
2 +.102 .1539 1.60 .4484
3 +.134 .1517 2.39 .4961
4 +.004 .1496 2.39 .6649
5 +.140 .1473 3.29 .6552
6 +.109 .1451 3.85 .6968
7 -.015 .1428 3.86 .7954
8 +.080 .1405 4.18 .8401
9 +.310 .1381 9.21 .4179
10 -.119 .1357 9.99 .4416
11 -.061 .1333 10.20 .5127
12 -.029 .1308 10.25 .5943
13 -.151 .1282 11.64 .5574
14 -.032 .1257 11.71 .6300
15 -.128 .1230 12.78 .6191
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit

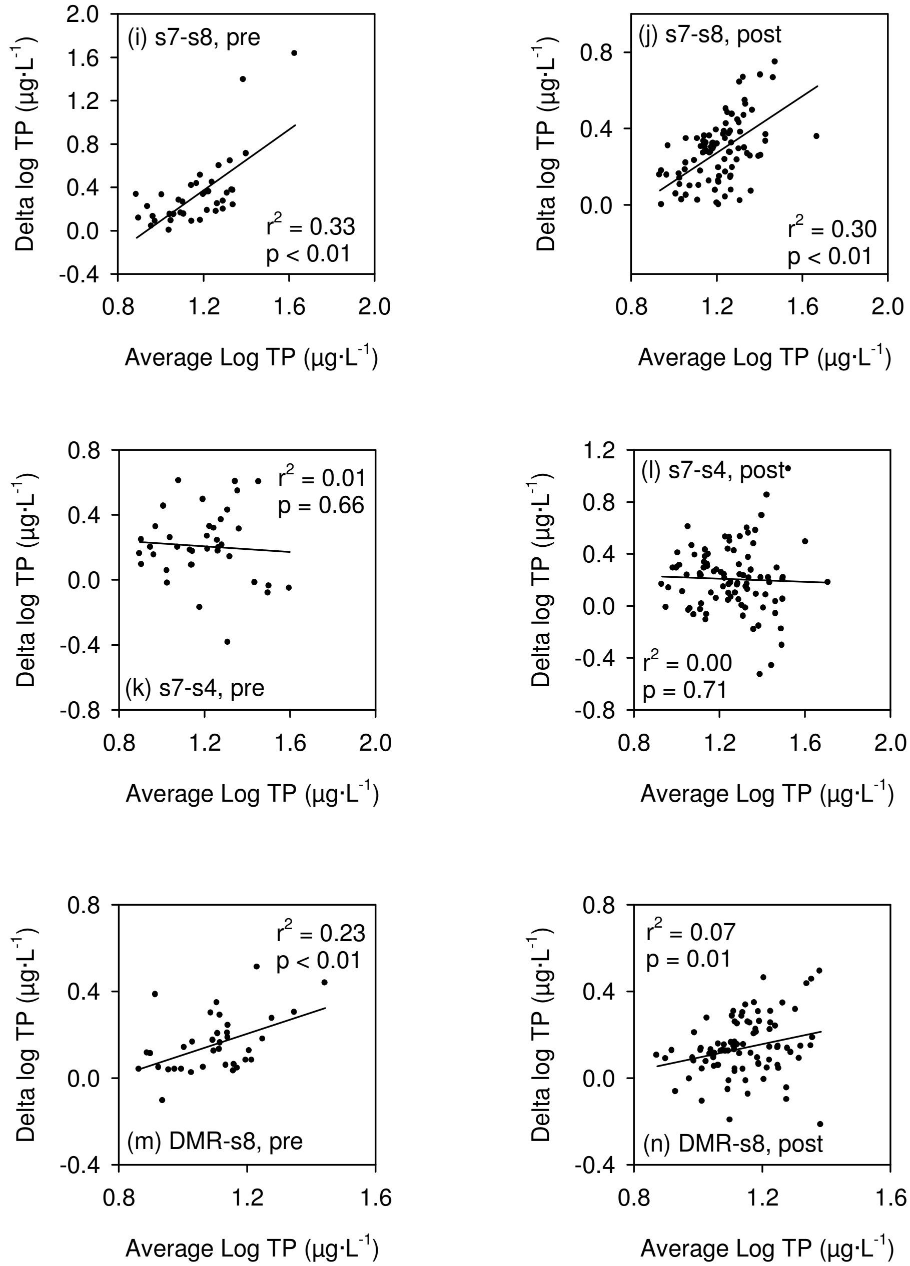

Autocorrelation Function
DMR-8 Tn post
(Standard errors are w hite-noise estimates)
Lag Corr. S.E. Q p
1 +.177 .1020 3.02 .0822
2 -.054 .1015 3.30 .1918
3 +.056 .1009 3.61 .3074
4 +.214 .1004 8.15 .0863
5 +.055 .0998 8.45 .1333
6 -.139 .0992 10.42 .1080
7 -.086 .0987 11.19 .1307
8 -.016 .0981 11.22 .1898
9 -.138 .0975 13.21 .1534
10 -.246 .0969 19.64 .0328
11 +.022 .0963 19.70 .0497
12 +.061 .0958 20.10 .0652
13 +.047 .0952 20.35 .0869
14 -.040 .0946 20.53 .1144
15 +.121 .0940 22.19 .1031
0
0
-1.0
-0.5
0.0
0.5
1.0
Conf. Limit

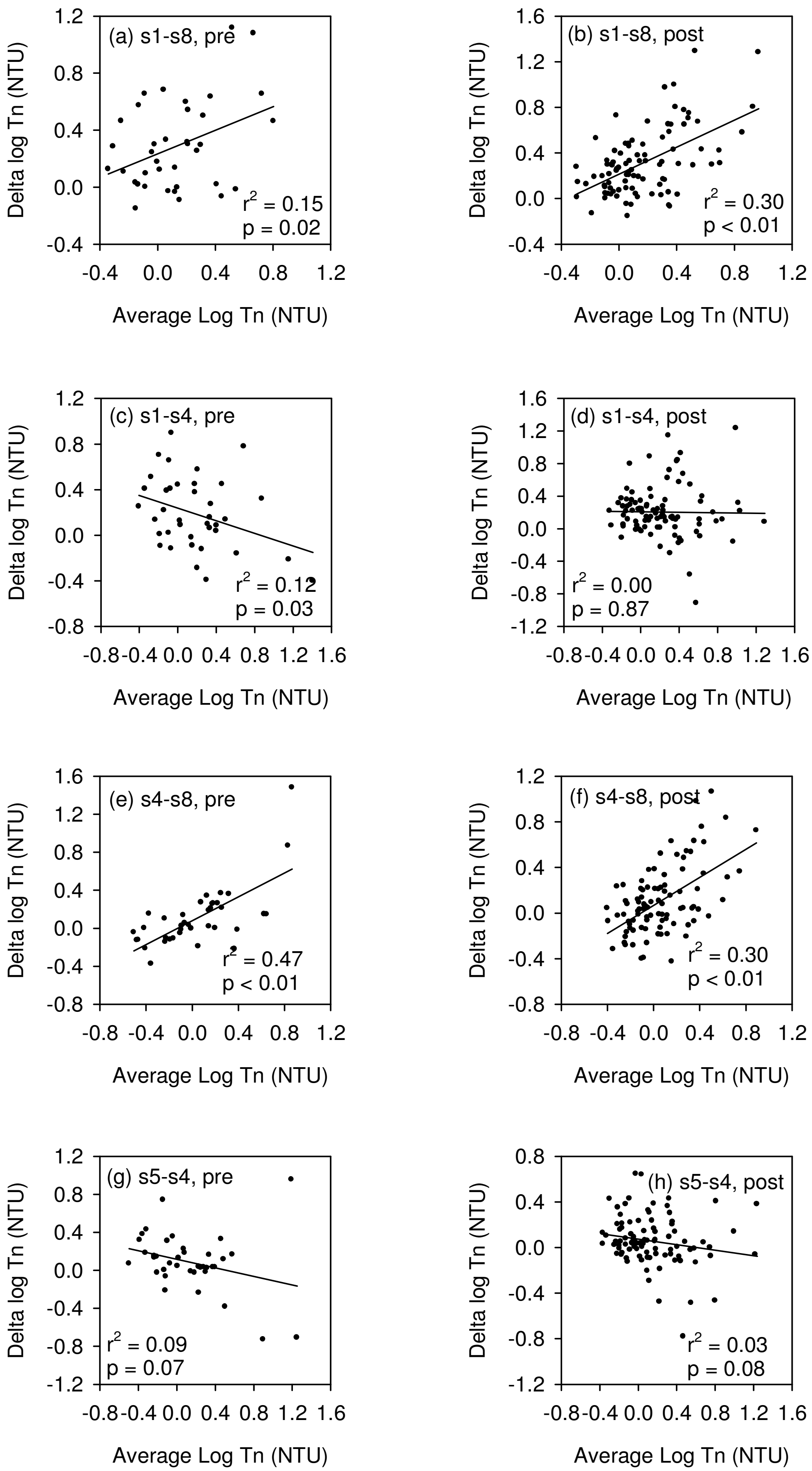
# Appendix 3 - Additivity

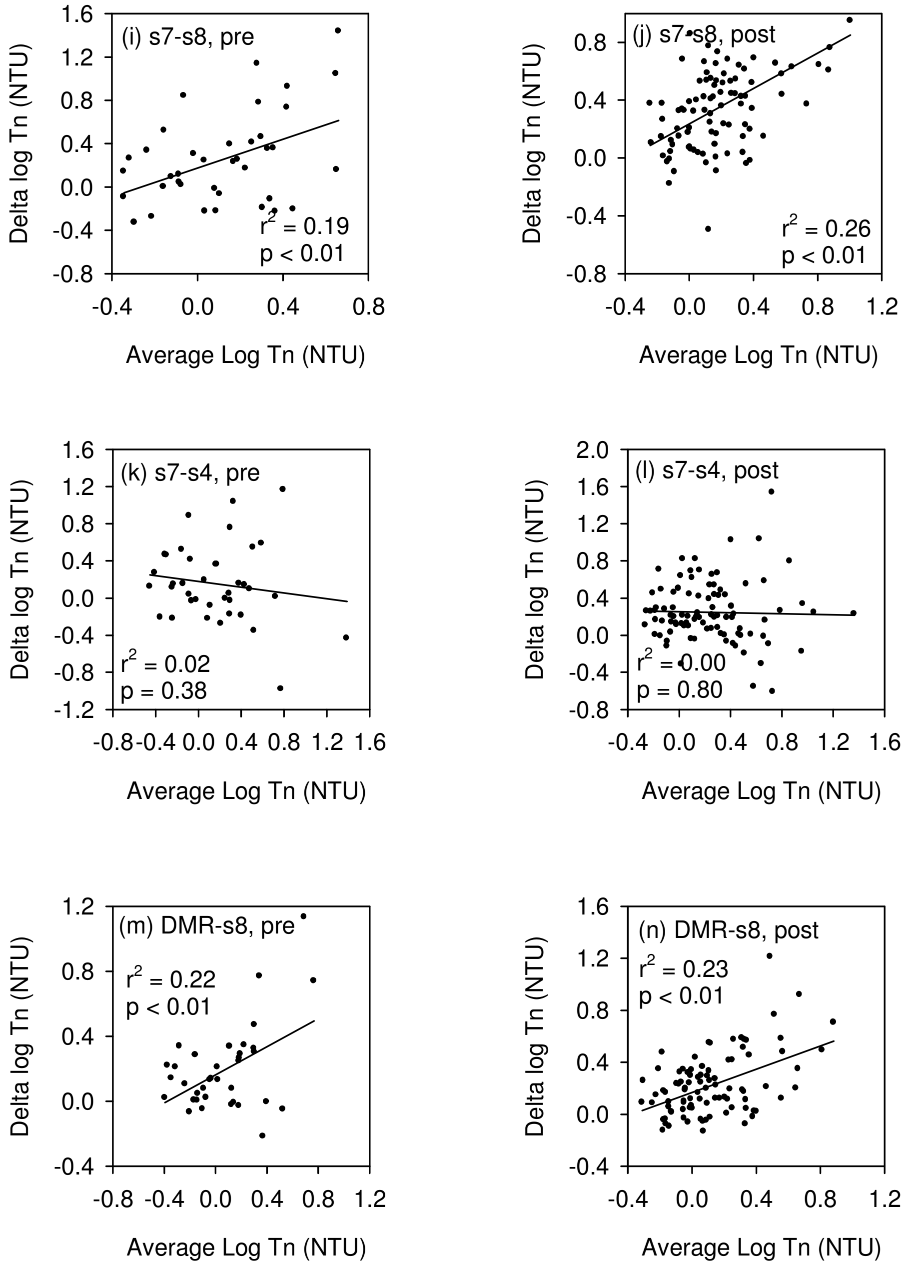

(a) s1-s8, pre
r2 = 0.16
p = 0.01
(b) s1-s8, post
r2 = 0.00
p = 0.57
(c) s1-s4, pre
r2 = 0.01
p = 0.67
(d) s1-s4, post
r2 = 0.13
p = 0.005
(e) s4-s8, pre
r2 = 0.16
p = 0.02
(f) s4-s8, post
r2 = 0.11
p = 0.001
(g) s5-s4, pre
r2 = 0.08
p = 0.12
(h) s5-s4, post
r2 = 0.16
p = 0.0001
Delta log Chl (μg·L-1)
Average Log Chl (μg·L-1)


(i) s7-s8, pre
$r^2 = 0.20$
$p = 0.005$
(j) s7-s8, post
$r^2 = 0.03$
$p = 0.30$
(k) s7-s4, pre
$r^2 = 0.01$
$p = 0.51$
(l) s7-s4, post
$r^2 = 0.02$
$p = 0.24$
(m) DMR-s8, pre
$r^2 = 0.04$
$p = 0.22$
(n) DMR-s8, post
$r^2 = 0.01$
$p = 0.28$
Delta log Chl ($\mu g \cdot L^{-1}$)
Average Log Chl ($\mu g \cdot L^{-1}$)


(a) s1-s8, pre
r2 = 0.32
p < 0.01
(b) s1-s8, post
r2 = 0.23
p < 0.01
(c) s1-s4, pre
r2 = 0.01
p = 0.53
(d) s1-s4, post
r2 = 0.02
p = 0.24
(e) s4-s8, pre
r2 = 0.41
p < 0.01
(f) s4-s8, post
r2 = 0.31
p < 0.01
(g) s5-s4, pre
r2 = 0.13
p = 0.03
(h) s5-s4, post
r2 = 0.1
p < 0.01
Delta log TP (μg·L-1)
Average Log TP (μg·L-1)
Average TP Chl (μg·L-1)


(i) s7-s8, pre
$r^2 = 0.33$
$p < 0.01$
(j) s7-s8, post
$r^2 = 0.30$
$p < 0.01$
(k) s7-s4, pre
$r^2 = 0.01$
$p = 0.66$
(l) s7-s4, post
$r^2 = 0.00$
$p = 0.71$
(m) DMR-s8, pre
$r^2 = 0.23$
$p < 0.01$
(n) DMR-s8, post
$r^2 = 0.07$
$p = 0.01$
Delta log TP (µg·L$^{-1}$)
Average Log TP (µg·L$^{-1}$)


(a) s1-s8, pre
$r^2 = 0.15$
$p = 0.02$
(b) s1-s8, post
$r^2 = 0.30$
$p < 0.01$
(c) s1-s4, pre
$r^2 = 0.12$
$p = 0.03$
(d) s1-s4, post
$r^2 = 0.00$
$p = 0.87$
(e) s4-s8, pre
$r^2 = 0.47$
$p < 0.01$
(f) s4-s8, post
$r^2 = 0.30$
$p < 0.01$
(g) s5-s4, pre
$r^2 = 0.09$
$p = 0.07$
(h) s5-s4, post
$r^2 = 0.03$
$p = 0.08$
Delta log Tn (NTU)
Average Log Tn (NTU)


(i) s7-s8, pre
r2 = 0.19
p < 0.01
(j) s7-s8, post
r2 = 0.26
p < 0.01
(k) s7-s4, pre
r2 = 0.02
p = 0.38
(l) s7-s4, post
r2 = 0.00
p = 0.80
(m) DMR-s8, pre
r2 = 0.22
p < 0.01
(n) DMR-s8, post
r2 = 0.23
p < 0.01
Delta log Tn (NTU)
Average Log Tn (NTU)

# Appendix 4 – Nonparametric Analysis

Table 1. Results from Mann-Whitney U-tests comparing log-transformed chlorophyll **a** for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented.

| Impact-Control Pairing | *p*-value | | |
|---|---|---|---|
| | raw | Bonferroni | B&H |
| site 1-site 4 | 0.838 | 1.000 | 0.854 |
| site 1-site 8 | 0.683 | 1.000 | 0.854 |
| site 7-site 4 | 0.338 | 1.000 | 0.676 |
| site 7-site 8 | 0.215 | 1.000 | 0.676 |
| site 4-site 8 | 0.854 | 1.000 | 0.854 |
| site 5-site 4 | 0.290 | 1.000 | 0.676 |
| DMR-site 8 | 0.632 | - | - |

Table 2. Results from Mann-Whitney U-tests comparing log-transformed chlorophyll **a** for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. The 8 outliers identified in Table 2 have been omitted from this analysis. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented.

| Impact-Control Pairing | *p*-value | | |
|---|---|---|---|
| | raw | Bonferroni | B&H |
| site 1-site 4 | 0.745 | 1.000 | 0.894 |
| site 1-site 8 | 0.950 | 1.000 | 0.950 |
| site 7-site 4 | 0.366 | 1.000 | 0.732 |
| site 7-site 8 | 0.327 | 1.000 | 0.732 |
| site 4-site 8 | 0.703 | 1.000 | 0.894 |
| site 5-site 4 | 0.201 | 1.000 | 0.732 |
| DMR-site 8 | 0.376 | - | - |

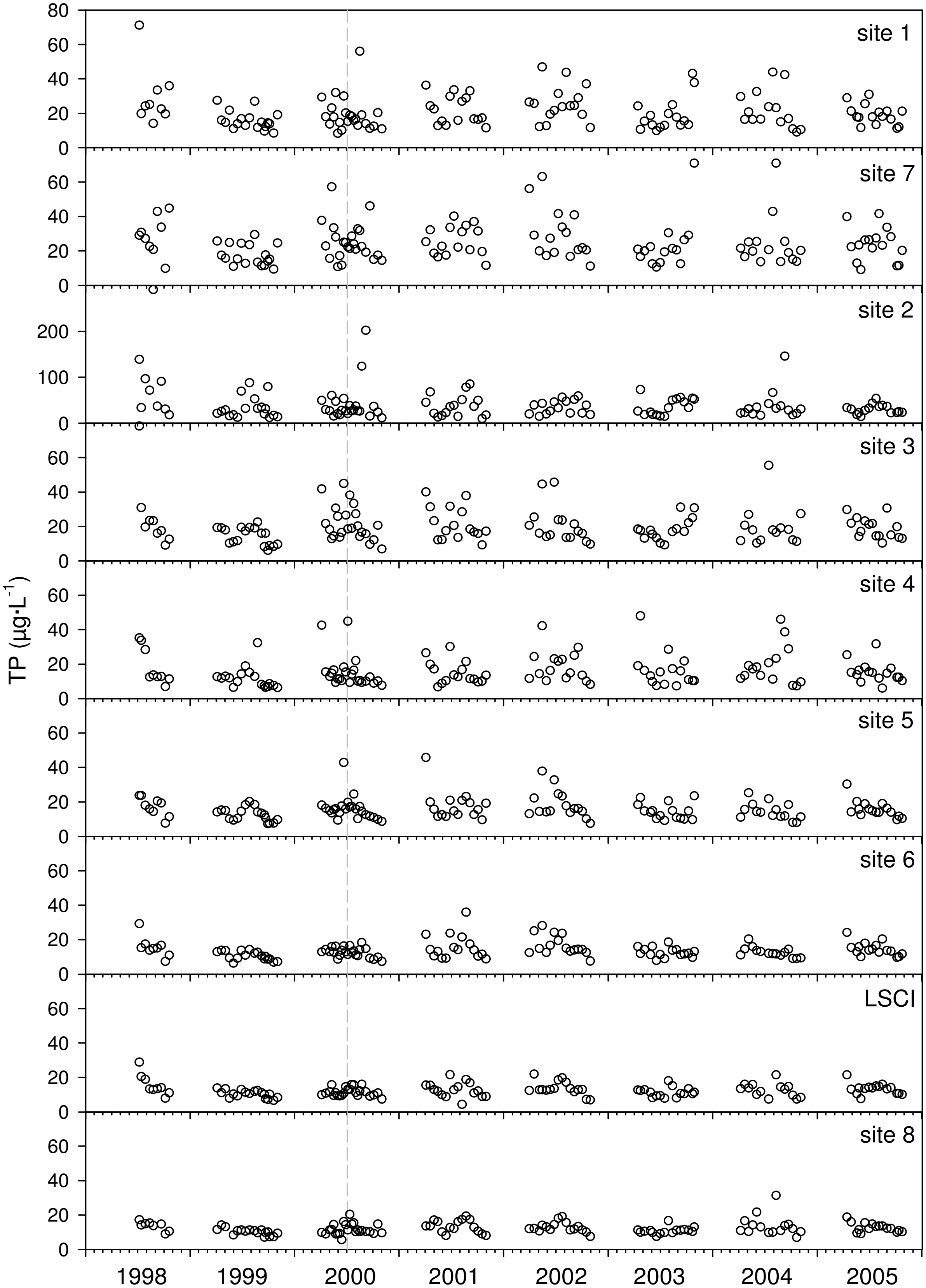
Table 3. Results from Mann-Whitney U-tests comparing log-transformed total phosphorus for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented.

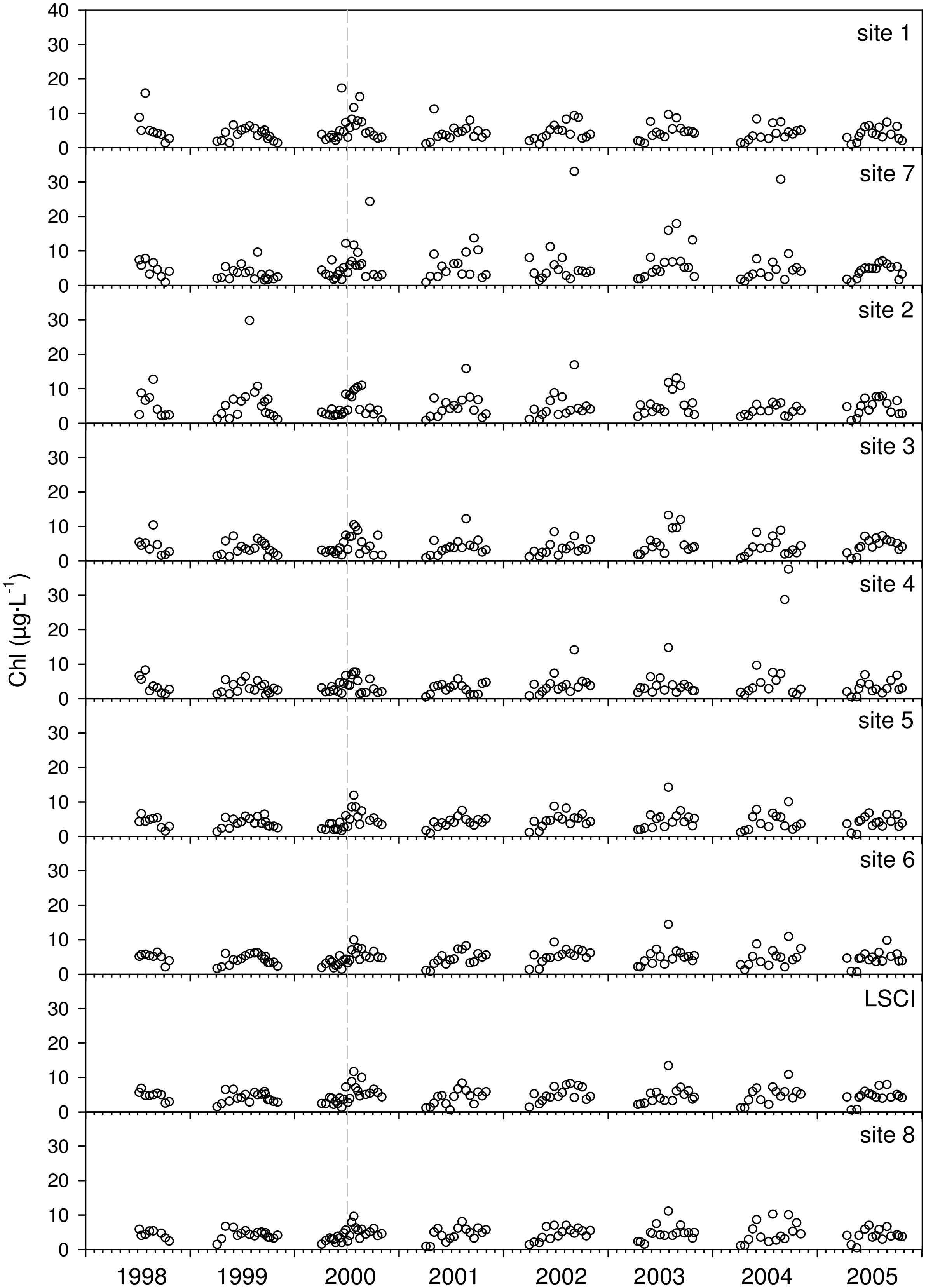
| Impact-Control Pairing | *p*-value | | |
|---|---|---|---|
| | raw | Bonferroni | B&H |
| site 1-site 4 | 0.362 | 1.000 | 0.917 |
| site 1-site 8 | 0.656 | 1.000 | 0.917 |
| site 7-site 4 | 0.917 | 1.000 | 0.917 |
| site 7-site 8 | 0.648 | 1.000 | 0.917 |
| site 4-site 8 | 0.823 | 1.000 | 0.917 |
| site 5-site 4 | 0.262 | 1.000 | 0.917 |
| DMR-site 8 | 0.872 | - | - |

Table 4. Results from Mann-Whitney U-tests comparing log-transformed turbidity for the pre start-up (7/9/98 – 6/29/00) and post start-up (7/6/00 – 10/24/05) intervals for the selected impact-control pairs. Standard deviation is abbreviated $S_d$. Raw *p*-values, Bonferroni-adjusted *p*-values, and Benjamini & Hochberg (1995) adjusted *p*-values (B&H) are presented.

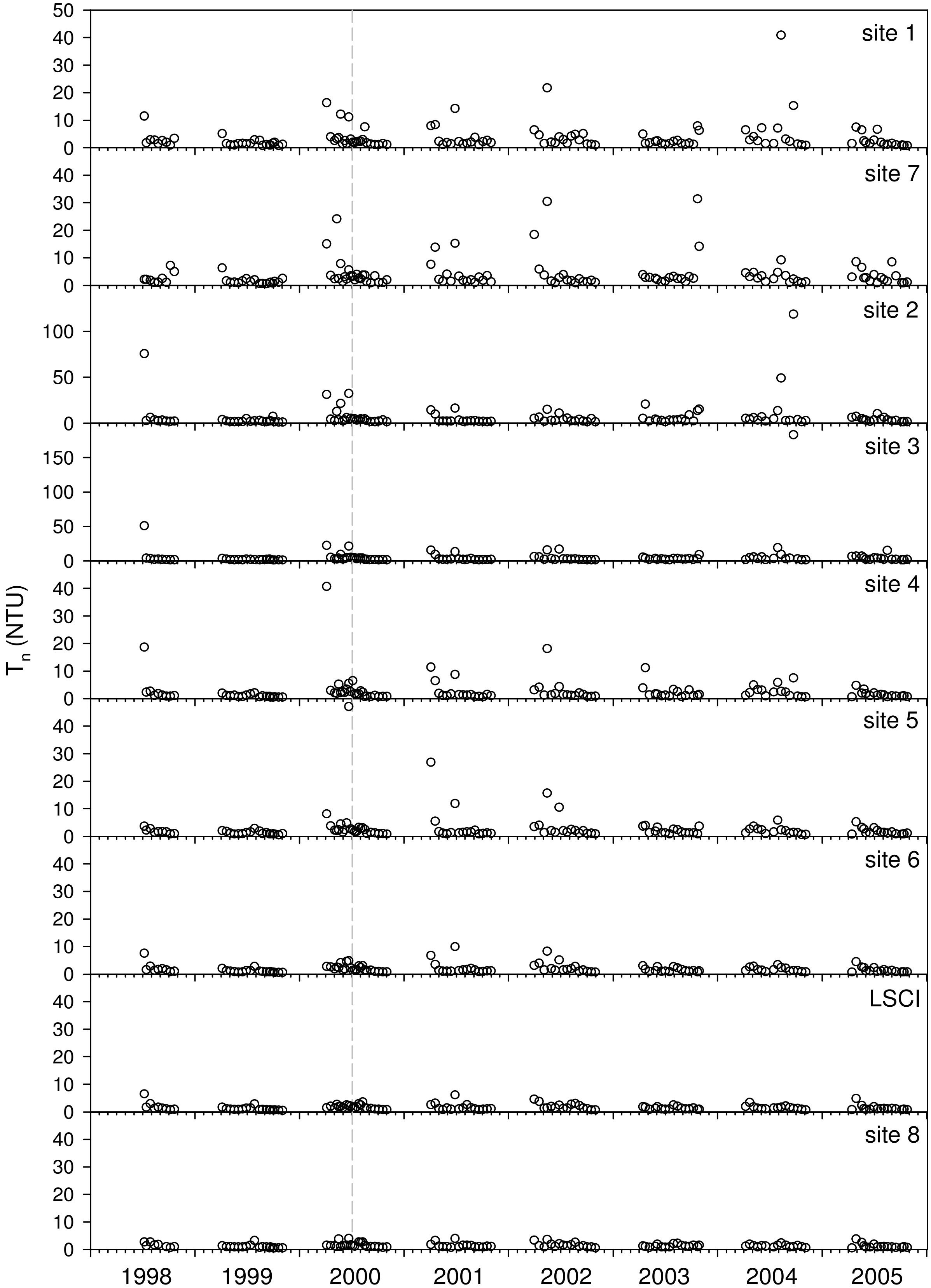
| Impact-Control Pairing | *p*-value | | |
|---|---|---|---|
| | raw | Bonferroni | B&H |
| site 1-site 4 | 0.803 | 1.000 | 0.932 |
| site 1-site 8 | 0.485 | 1.000 | 0.728 |
| site 7-site 4 | 0.104 | 0.624 | 0.312 |
| site 7-site 8 | 0.043 | 0.258 | 0.258 |
| site 4-site 8 | 0.932 | 1.000 | 0.932 |
| site 5-site 4 | 0.299 | 1.000 | 0.598 |
| DMR-site 8 | 0.515 | - | - |

# Appendix 5 – Raw Time Series Plots


site 1
site 7
site 2
site 3
site 4
site 5
site 6
LSCI
site 8
TP (µg·L-1)
1998
1999
2000
2001
2002
2003
2004
2005



Chl (µg·L⁻¹)
site 1
site 7
site 2
site 3
site 4
site 5
site 6
LSCI
site 8
0
10
20
30
40
1998
1999
2000
2001
2002
2003
2004
2005



site 1
site 7
site 2
site 3
site 4
site 5
site 6
LSCI
site 8
$T_n$ (NTU)
1998
1999
2000
2001
2002
2003
2004
2005