TOTAL MAXIMUM DAILY LOAD DEVELOPMENT IN TAMPA BAY TRIBUTARIES: CONSIDERATION OF THE FACTORS INFLUENCING THE RELATIONSHIP BETWEEN NUTRIENTS AND DISSOLVED OXYGEN AND CHLOROPHYLL a

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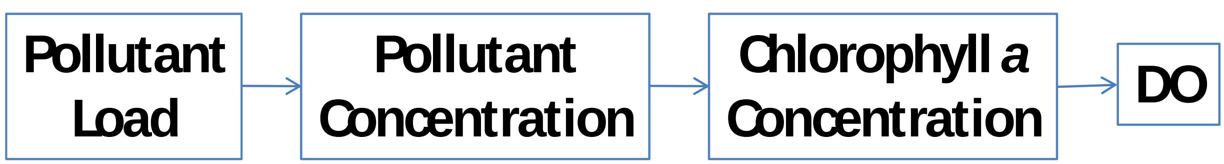
A Total Maximum Daily Load (TMDL) has been defined as a scientific determination of the maximum amount of a given pollutant that a surface water can absorb and still meet the water quality standards that protect human health and aquatic life. This definition is essentially an attempt to define assimilative capacity using endpoints based on the Florida water quality standards (e.g., those standards for dissolved oxygen and nutrient impairments). The conceptual model that has been conveyed in the proposed TMDLs is that anthropogenic nutrient sources increase the concentrations of phytoplankton in the water column and the decomposition of this organic material reduces the dissolved oxygen levels in the surface water. This is a completely plausible and reasonable though somewhat simplistic conceptual model. Establishing a TMDL requires the determination of a causative agent and this conceptual model is often used to justify the selection of the agent. However, a host of other processes outside this model influence chlorophyll and dissolved oxygen levels in these surface waters. In particular, the current development of TMDLs for Tampa Bay area tributaries has neglected to consider the impact of hydrology, alterations to stream morphology or to fully characterize the nature of the waterbody under study within the context of physical alterations within the watershed. Understanding the biogeochemical effects of stream alterations is critical in estimating the waterbody's ability to assimilate pollutants.

Typically, the causative agent addressed in a TMDL is assumed to be either a nutrient (i.e., nitrogen or phosphorus) or, in the case of a DO TMDL, biological oxygen demand (BOD). Many of the TMDLs that have been proposed for a series of tributaries to Tampa Bay have been based on the following conceptual model:



It is well known that at each of the connections between the boxes shown above there are any number of confounding factors that can significantly influence the quantitative relationship between pollutant loading and DO. This is especially true in flowing waters (Hynes, 1970; Allan, 1995; Hauer and Lambert, 1996; Wetzel, 2001).

Figure 1 presents a comprehensive conceptual model of the relationship between pollutant loading and DO in flowing waters. There is a critical link between the hydrologic features of a flowing waterbody and its watershed and other factors such as nutrient loading, DO, and water temperature. These include the effects due to groundwater and tributary inputs and upstream impoundments.

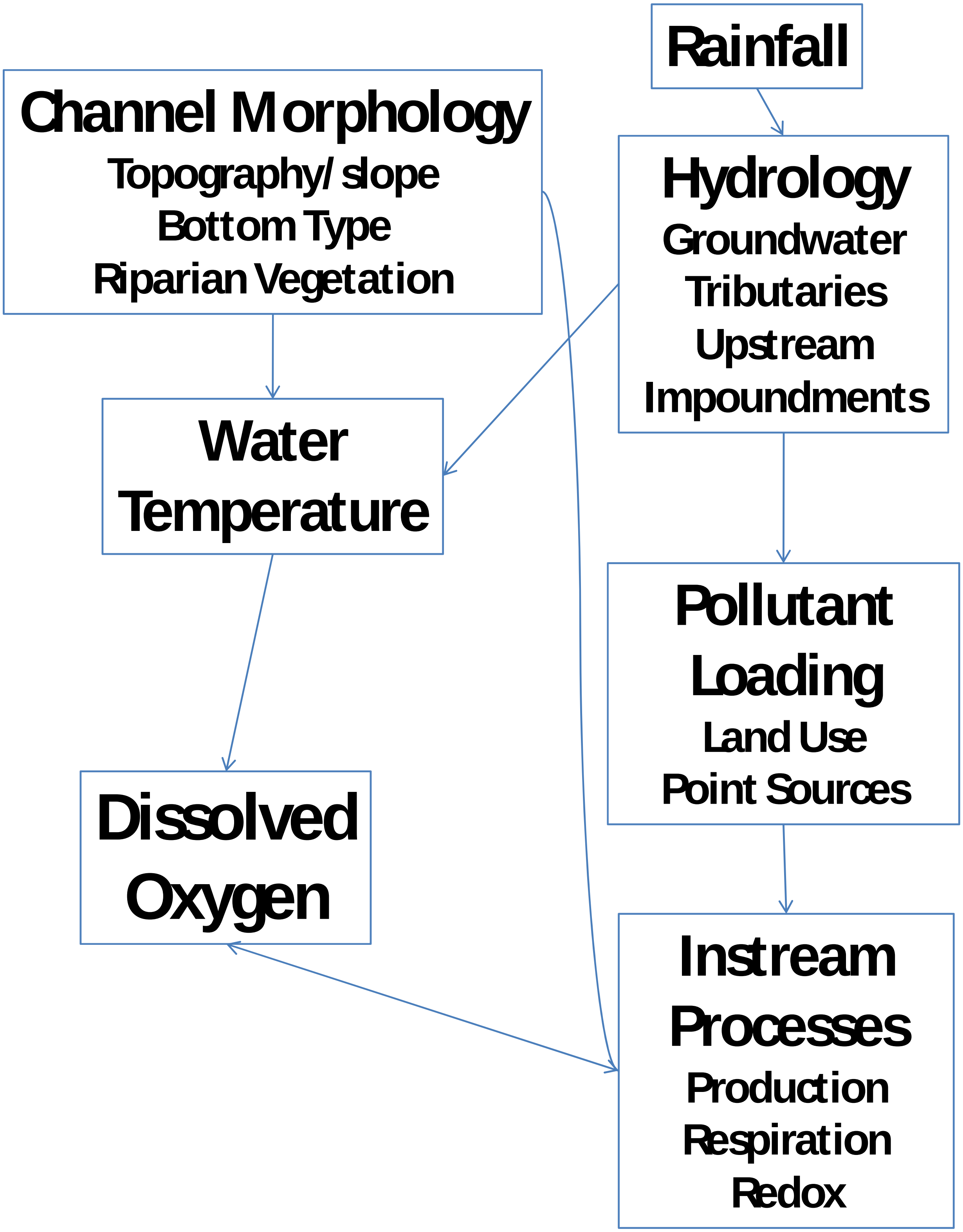


Figure 1. Conceptual model of the relationship between pollutant loading and DO and the factors that can significantly affect this relationship in flowing waters.

Stream flow during periods of little or no rainfall can be sustained by contributions from groundwater, or base flow. Base flow occurs when the potentiometric surface of the Floridan aquifer, which represents the pressure of groundwater below a confining layer, or the free surface elevation of the surficial aquifer, is higher than the bottom of a river or stream channel, allowing water to enter the channel and become stream discharge through upward vertical leakage or horizontal flow, as shown in Figure 2.

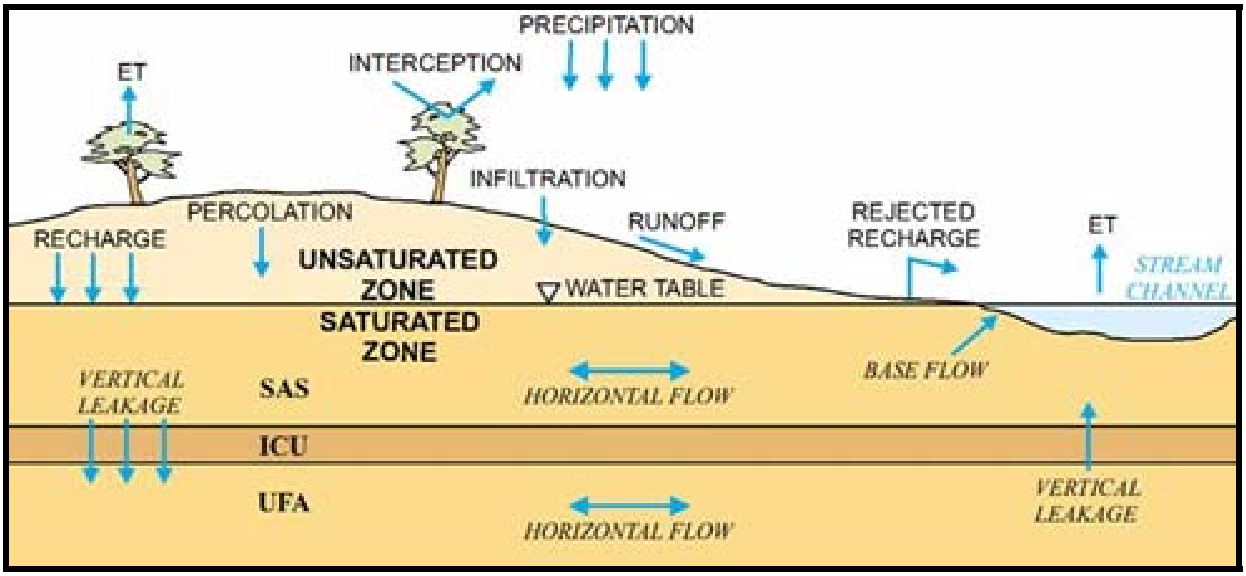


Figure 2. Typical water budget with base flow (from: Yobbi, 2000) (SAS – surficial aquifer system; ICU – intermediate confining unit; UFA – Upper Floridan aquifer).

The Tampa Bay watershed contains a range of hydrogeologic and land use conditions throughout areas overlying the Floridan aquifer system. Hydrologic confining conditions and karst features are especially prevalent in the Floridan aquifer system in the northern Tampa Bay watershed (Miller, 1986).

Many studies have investigated base flow contributions to Tampa Bay area rivers and streams. Computer modeling and empirical analyses have indicated that the 1987 average annual base-flow in a short reach of the Hillsborough River between USGS gaging stations 02301990 (Hillsborough River above Crystal Springs, near Zephyrhills, Florida) and 02303000 (Hillsborough River near Zephyrhills, Florida) was in the range of 45 - 50 cfs (Crandall, 2007). Average daily discharge in this reach for 1987 was 220 cfs, so base flow contributed over 20 percent of the total river flow at that point.

Based on the volume of water that base flow contributes, it is logical that base flow can also alter stream discharge water chemistry. Many studies have examined the contributions of base flows on stream water quality. Trommer et al. (1999) found that base flow made a significant increase in ion concentrations in Lake Ward and the Bradenton River in Manatee County. Lewelling (1997) found high base flow contributions to Horse Creek in the Peace River basin, with nitrogen (organic plus ammonia) concentrations frequently exceeding 2 mg/L. Jones et al. (2000) report nitrate levels up to 3.1 mg/L at Lithia Springs on the Alafia River in Hillsborough County. He states that "*Such quantities of a limiting nutrient discharged into rivers and estuaries may have a significant negative effect on the health of an aquatic system.*"

Base flow also has generally lower dissolved oxygen (DO) concentrations than surface water. Marchman (2000) conducted a sampling study of the Apalachicola Bay watershed and found base flow with periodic depressed dissolved oxygen levels and moderate increases in nutrients such as ammonia nitrogen, total Kjeldahl nitrogen, and nitrate/nitrite, possibly indicative of sewage contamination, perhaps from leaking and poorly maintained septic tanks or aging treatment systems, cross connections, or illicit connections.

The potential effects of base flow on surface waters have also been recognized by environmental protection agencies. FDEP (2008) states on its Groundwater Protection website (emphasis added):

> *"The water quality of base flow is now also considered an equally important ground water use to ensure the support of aquatic life in surface waterbodies. Identifying and quantifying ground water contributions where substances with extensive natural or anthropogenic abundances in geological deposits coexist with high percentages of base flow are also important* ***in evaluating impaired surface waters****."*

The effect of land use and resultant nonpoint source pollutant loads along with point source loads on primary production and respiration has been well-documented. These instream processes directly affect DO concentrations. Other instream biogeochemical processes influence the concentration of nutrients during transport through stream networks, resulting in a net decrease downstream (Peterson et al., 2001). It was estimated that 60% of the N entering a watershed's stream network may be retained (permanently via denitrification or temporarily through biotic sequestration) within streams of the eastern United States (Seitzinger et al., 2002). Alteration of nutrient concentrations by riverine processes during transport also changes the timing, quantity, and quality (labile vs. refractory) of nutrients exported to downstream ecosystems (Mulholland, 2004). Understanding the processes controlling the temporal variation in nutrient delivery and availability is critical in coastal environments where nutrient enrichment may have many negative impacts including eutrophication, habitat degradation and decreased biodiversity (Boesch et al., 2001; Nixon, 1995).

Many studies have concluded that the main stressor to stream and creek ecosystems is human population density in the watershed and the associated impacts to ecosystems (Holland et al., 2004; Peterson et al., 2001). While the importance of headwater streams for the downstream success of coastal rivers and estuaries has been long studied (Nixon, 1995; Boesch et al., 2001; Rothenberger et al., 2009), unfortunately there are still very few criteria to determine whether an impacted stream can be restored to replace the functions of the original ecosystem.

When considering stream function and its relationship to DO levels, one must consider anthropogenic changes associated with urbanization which link increases in human population density with increases in the amount of impervious cover and overall degradation of stream environmental quality and biological integrity (Holland et al., 2004). Land use changes in coastal areas are linked to elevated instream nutrient concentrations and degradation of headwater streams (Basnyat et al., 1999; Allan, 2004; Poor, 2007; Rothenberger et al., 2009). Principal mechanisms by which land use influences stream ecosystems include sedimentation, nutrient enrichment, contaminant pollution, hydrologic alteration, riparian clearing and loss of large woody debris. Increased development accompanied by changes in impervious area (due to compaction of native soils and construction of impermeable surfaces) and modification of drainage systems result in drastic changes in watershed hydrology. Peak storm flows are greater and base flow is reduced due to decreased infiltration. Greater flow velocities during storm events lead to increased bed scouring and movement of channel sediments. This can also impact the biogeochemical functioning of the stream. In a study by Jones et al. (2001), landscape metrics, particularly the integrity of riparian corridors, explained 65-84% of variation in yields of N, P and suspended sediments for 78 catchments across the five-state Mid-Atlantic Highland region. Research suggested that severe biological degradation occurs when more than 30% of the watershed is converted to impervious cover (Schueler, 1994; Arnold and Gibbons, 1996). A study by Sutherland and others in 2002 on the effects of land cover on streams showed that in areas with greater than 20% non-forested land cover, suspended sediment and bedload transport increased sharply, which correlated with marked decrease in important fish species. A study by Holland et al. (2004) specifically investigated the impact of degradation of natural systems to understand indicators to creek systems. In the study, twenty-three creeks draining watersheds representative of forested, suburban, urban and industrial development were sampled from 1994 to 2002. Data from the study showed that when the level of imperviousness was less than 10-20%, physical, chemical, and biological indicators of tidal creeks were similar to those reported for pristine conditions and ecosystems were able to sustain high levels of environmental quality (Holland et al., 2004). When the level exceeds 20%, physical, chemical, and biological processes were altered or impacted (Holland et al., 2004).

Clearly, several physical factors can also be important determinants of DO in flowing waters. Particularly, water temperature, flow, and the physical nature of the stream channel can directly affect DO concentrations and the instream processes that also significantly affect DO. First, the solubility of oxygen in water is a function of the temperature of that water - higher solubility occurring at lower water temperatures. Temperature follows typical seasonal variation but can also be affected by flow, runoff from surfaces that can retain heat, and the extent of shading offered by the occurrence of riparian vegetative canopy. Other channel characteristics that can potentially affect DO are bottom type and slope.

In addition to external factors, internal factors such as changes to the physical structure of headwater streams can impact stream function and biological integrity. Streams are being altered such that natural channels with complex heterogeneity and meandering reaches are being replaced by straight, homogenous drainage ditches. Channel morphology is an important determinant of transient solute storage and nutrient retention in headwater streams. A study by Gucker and

Boechat (2004) investigated stream morphology controls on ammonium retention in tropical headwaters. Twelve first-order stream reaches with similar water chemistry and sediment structure were classified as swamp, step-pool reaches, meandering or run. Altered streams were more characteristic of the run classification with straight homogenous drainage ditches. The four factors studied for each morphology were transient solute storage zone size, residence time, uptake potential of nutrients and pollutants. Of the four types, the run channel morphology had the lowest function in all four categories, with retention of ammonium 50-80% less than swamp, step-pool or meandering ecosystems (Gucker and Boechat, 2004).

Data suggest that flowing water ecosystems are strongly affected by anthropogenic influence across many temporal and spatial scales. Management of these ecosystems must take into account rehabilitation of the heterogeneity in stream morphology and riparian vegetation as well as the use of best management practices which may be specific to an area to restore the functioning of ecosystems and provide humans with resources and aesthetic and recreational benefits (Aldridge et al., 2007).

In the following pages, we examine relationships between dissolved oxygen and chlorophyll and potential causative agents used in Florida Department of Environmental Protection's (FDEP) assessment of surface waters in accordance with EPA's Clean Water Act. Using these parameters and data collected throughout Tampa Bay, we will identify stressor - response relationships and identify potential endpoints that may serve as alternatives to the endpoints identified by FDEP. Based on these analyses, we recommend alternative methods to the development of a TMDL.

ANALYTICAL APPROACH

The FDEP Impaired Water Rule (IWR) dataset (Run 37) was obtained from the FDEP and all Group 1 and Group 2 waterbodies in the Tampa Bay Basin were subset from the IWR database, to include all tributaries, both estuarine and freshwater. The Cycle 2 Verified Impaired list for the Tampa Bay Basin was obtained and all WBIDs verified as impaired for dissolved oxygen and nutrients were identified. These WBIDs were then identified within the IWR subset database (heretofore referred to as IWR_TB) to identify "Impaired" and "Unimpaired" WBIDs for dissolved oxygen and nutrients. Southwest Florida Water Management District land use data were also obtained and the percentage of each of 12 major land use categories listed in Table 1 was calculated for each WBID. Logistic regression was then used to estimate the probability of a WBID to be classified as impaired based on its land use characteristics. Descriptive plots and statistical analysis were conducted on these data to compare relationships between stressors and responses in Impaired and Unimpaired waters to elucidate potential differences in the assimilative capacity of the waters in the Tampa Bay area.

Table 1. Major land use classifications used to characterize WBID land use in Tampa Bay WBIDs.

Urban non-residential	Utilities, Commercial & Transportation
Agricultural	Low Density Residential
Pasture & Rangeland	Medium Density Residential
Upland Vegetation	High Density Residential
Water	Barren Lands
Wetlands	Mining

DO levels were different in waters classified as impaired in both Class 3F and Class 3M streams as evidenced in the cumulative distribution plots of Figure 3. In neither case did the grouped data meet the 10 percent rule for either 5 mg/L (3F) or 4 mg/L(3M). This artifact is assumed to be due to the inclusion of a margin of error in the statistical test used in the determination of impairment (binomial test statistic) or an artifact of the sampling which may not have qualified the samples for inclusion in the assessment. Regardless, it is clear that unimpaired WBIDs typically have higher DO values compared to impaired WBIDs.

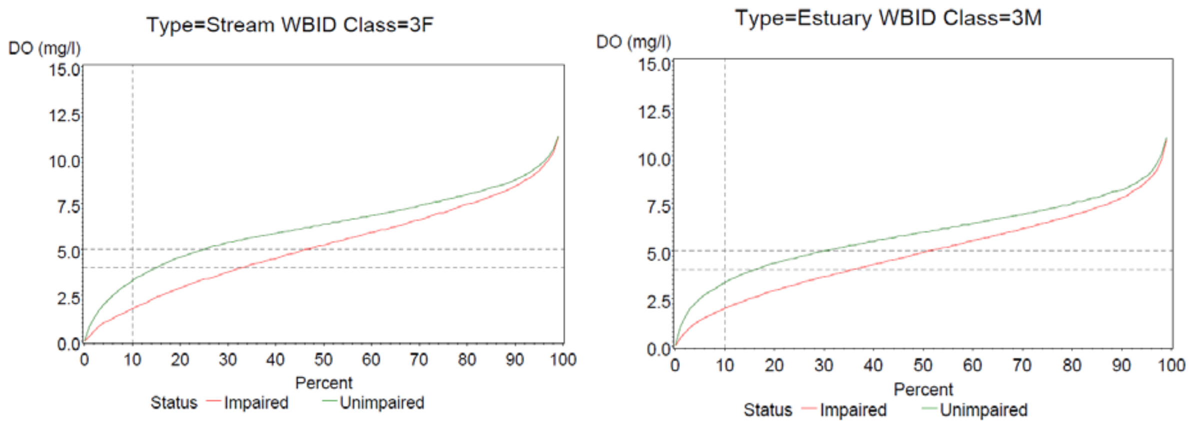


Figure 3. Cumulative distribution of dissolved oxygen concentrations in impaired and unimpaired waters for freshwater (3F) streams (left) and estuarine (3M) waters (right).

Analysis of land use demonstrated that increased urban land use increased the probability that a Class 3F stream WBID would be characterized as impaired for dissolved oxygen (Figure 4). This finding indicates that urbanized streams are significantly more likely to be classified as Impaired based on the FDEP water quality standards for class 3 freshwater streams. The same analysis performed on Class 3 estuarine waters which include tidal tributaries to Tampa Bay suggested that increased percentage of wetlands in the WBID was protective of being classified as impaired (Figure 4).

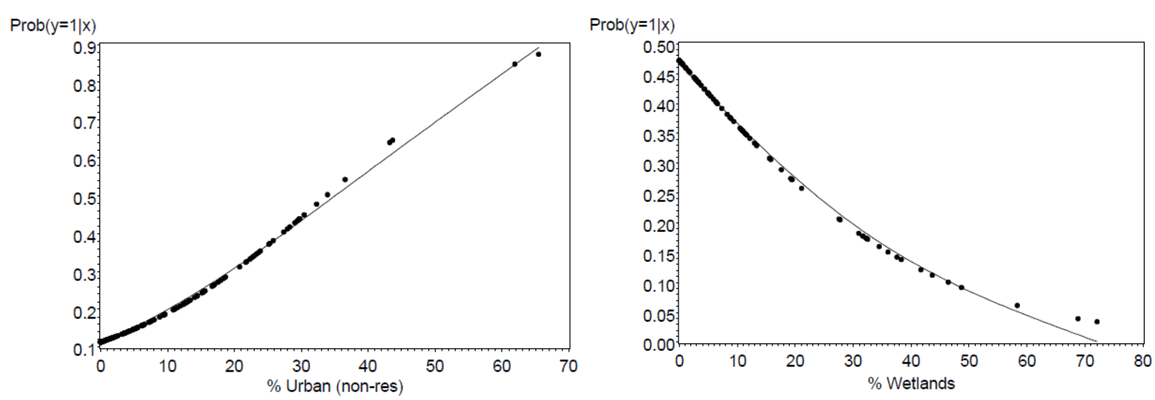


Figure 4. Probability of a WBID impairment for dissolved oxygen as a function of the percent of the WBID characterized as Urban in 3F streams (left) and wetlands in 3M streams (right).

These finding are reasonable and expected outcomes. While based on simplistic univariate relationships, the results suggest that estuarine WBIDs with a larger proportion of wetlands have higher ecological function. This is supported by the finding that unimpaired WBIDS have higher hydrologic loadings (Figure 5) and total nitrogen loading (Figure 6). The Wilcoxon rank sum test was used to compare these statistically and these difference were found to be highly statistically significant ($P<0.001$). Clearly, the volume, timing and variability in hydrologic load combined with the residence time of those waters influence the in-stream processes that combine with other factors in governing the dissolved oxygen level in these water bodies. Similar boxplots comparing differences with many other potential causative agents are provided in Appendix 1. Results of the Wilcoxon rank sum test results (Table 2) suggest that many other potential causative agents may influence the classification of a waterbody as impaired including the colored dissolved organic matter, pH, chlorophyll, and the delivery ratio (a measure of the TN concentration of the incoming hydrologic load). Also interesting was that Impaired WBIDs tended to have higher water temperatures than Unimpaired WBIDs.

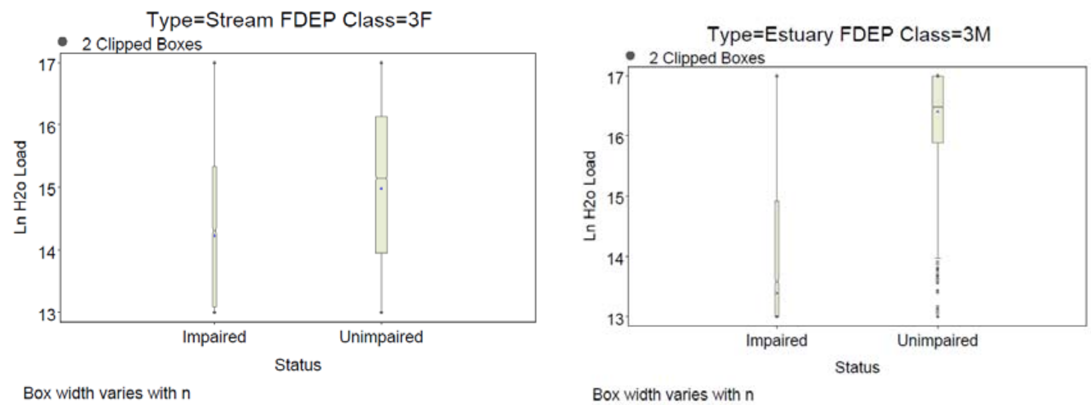


Figure 5. Distribution of modeled hydrologic loads (natural log transformed freshwater load in cubic meters per month) in impaired and unimpaired water class 3 WBIDs.

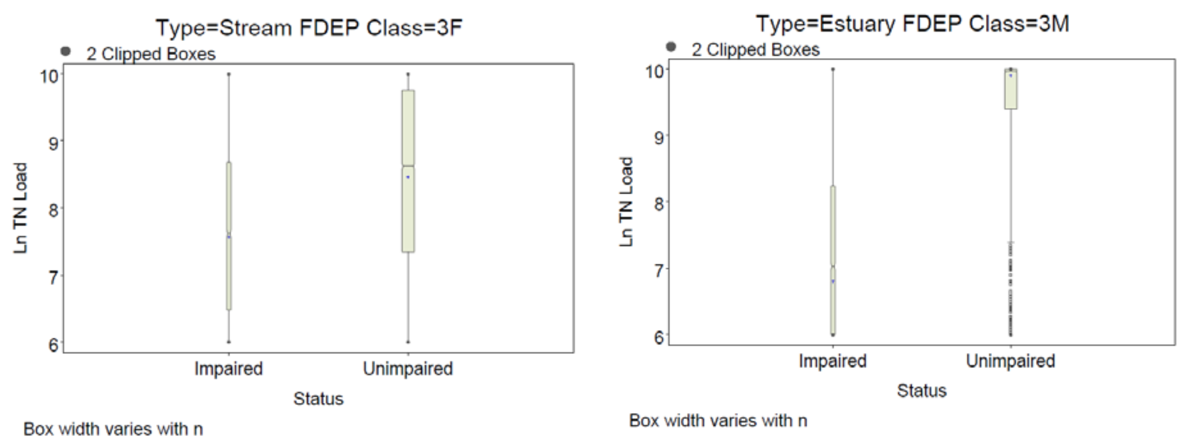


Figure 6. Distribution of modeled TN loads (natural log transformed TN in kilograms per month) in impaired and unimpaired water class 3 WBIDs.

Table 2. Results of Wilcoxon Rank Sum test between impaired and unimpaired WBIDs in Class 3M and 3F streams.		
Parameter	WBID Class	
	3M	3F
ln_Hydrologic Load (m^3/month)	<0.001	<0.001
ln_TN Load (kg/month)	<0.001	<0.001
DO (mg/L)	<0.001	<0.001
pH (SU)	<0.001	NS
Color (PCU)	<0.001	<0.001
Chlorophyll a (μg/L)	<0.001	<0.001
Corrected Chlorophyll a (μg/L)	<0.001	<0.001
Alkalinity (mg/L)	<0.001	<0.001
TN (mg/L)	<0.001	NS
TP (mg/L)	<0.001	<0.001
Organic N (mg/L)	<0.001	<0.001
Temperature (ºCelsius)	<0.001	<0.001
DO Deficit (% saturation – DO(mg/L))	<0.001	<0.001

Regression relationships were developed to estimate the change in dissolved oxygen concentration as a function of these potential causative agents (Table 3). The WBID status was included as a classification variable to describe differences in the mean dissolved oxygen concentrations between impaired WBIDS for the same unit of the response variable. Analysis of covariance was used to test the relationship between dissolved oxygen values and potential causative agents while including the class variable of status to test for statistical significance in difference in the classification of the waterbody as Impaired or Unimpaired. Results suggest that these differences in dissolved oxygen were observed irrespective of which potential causative agent was selected. For example, the relationship between DO and TN and TP concentrations and loads in 3F and 3M streams reflect this observation (Figure 7 and 8). These plots show that DO concentrations are higher in unimpaired streams given similar TN or TP concentrations. The most apparent difference is that at lower concentrations impaired streams show both higher highs and lower lows than in the unimpaired streams. Also, despite higher TN and TP loads in unimpaired streams, DO concentrations are generally similar in both groups of streams.

Table 3. Results of analysis of covariance on dissolved oxygen levels for potential causative agents.						
Class	Parameter	Intercept	Slope	Prob (t)	Status	Prob (t)
3F	DO % Saturation	-0.0789	0.0881	<0.001	-0.0508	<0.001
3F	Alkalinity	6.0580	0.0009	NS	-0.6297	<0.001
3F	Chlorophyll *a*	6.4916	-0.0045	<0.001	-1.0039	<0.001
3F	Corrected Chlorophyll *a*	6.0048	-0.0064	<0.001	-0.9860	<0.001
3F	Color	7.2887	-0.0110	<0.001	-1.3269	<0.001
3F	ln Hydrologic Load	8.1038	-0.1076	<0.001	-1.4169	<0.001
3F	ln TN Load	7.0146	-0.0620	<0.001	-1.3866	<0.001

Table 3. Results of analysis of covariance on dissolved oxygen levels for potential causative agents.						
3F	Organic N	7.1394	-0.8966	<0.001	-1.0737	<0.001
3F	pH	-3.1342	1.2992	<0.001	-1.0929	<0.001
3F	Temperature	10.5468	-0.1893	<0.001	-0.9231	<0.001
3F	TN	6.5033	-0.1300	<0.001	-1.0800	<0.001
3F	TP	6.2121	0.0828	<0.001	-1.0326	<0.001
3M	DO % Saturation	0.1336	0.0719	<0.001	0.0707	<0.001
3M	Alkalinity	5.6079	0.0000	NS	-0.8535	<0.001
3M	Chlorophyll *a*	5.8060	0.0008	NS	-0.9978	<0.001
3M	Corrected Chlorophyll *a*	5.8309	0.0015	NS	-1.1245	<0.001
3M	Color	6.1728	-0.0072	<0.001	-1.1458	<0.001
3M	ln Hydrologic Load	7.0452	-0.0879	<0.001	-1.0881	<0.001
3M	ln TN Load	6.4632	-0.0868	<0.001	-1.0913	<0.001
3M	Organic N	5.4177	-0.4237	<0.001	-0.4669	<0.001
3M	pH	-10.7364	2.1393	<0.001	-0.6568	<0.001
3M	Temperature	10.5246	-0.1861	<0.001	-0.9168	<0.001
3M	TN	6.0193	-0.3728	<0.001	-0.9907	<0.001
3M	TP	5.8435	-0.1609	<0.001	-1.1647	<0.001

Other plots were also constructed by month to eliminate the potential confounding effects of temperature on dissolved oxygen values. The influence of temperature identified in the analysis of impaired and unimpaired WBIDs is seen primarily during the spring and summer months (Figure 9). This eliminates the potential bias due to the temporal differences in data collection. Comparing DO percent saturation resolved much of the difference observed between impaired and unimpaired WBIDs with an order of magnitude smaller difference in DO between impaired and unimpaired WBIDs. This indicates that temperature differences between impaired and unimpaired waters may result in an approximately 1 mg/L difference in dissolved oxygen levels.

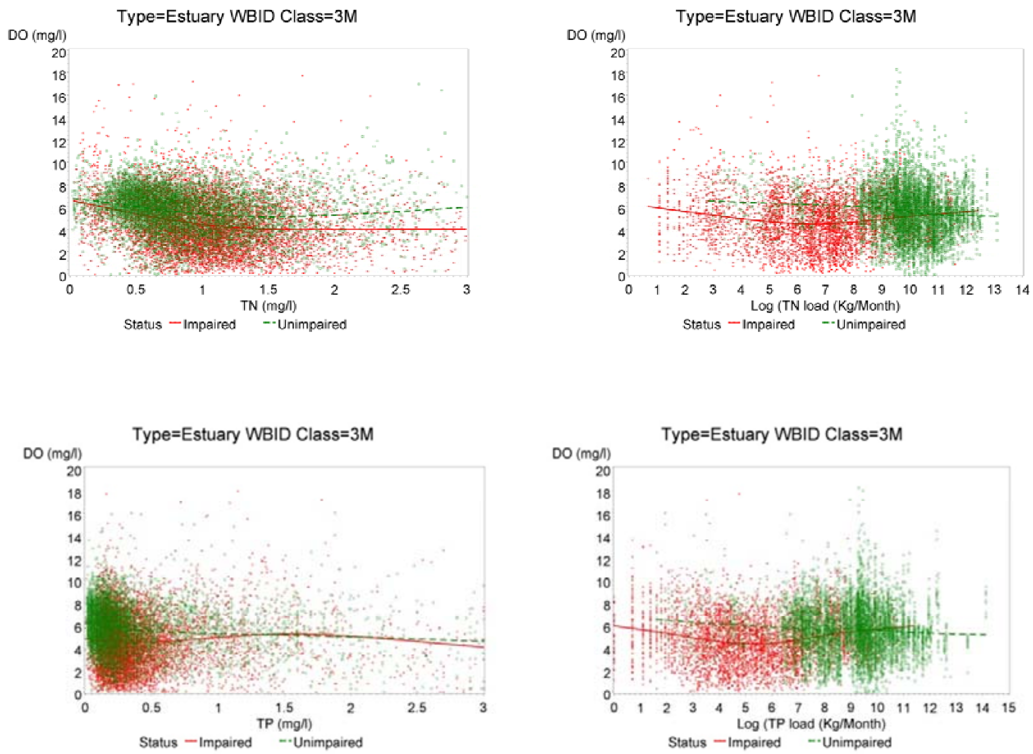


Figure 7. Relationship between DO and TN concentrations and loads in 3M and 3F WBIDs.

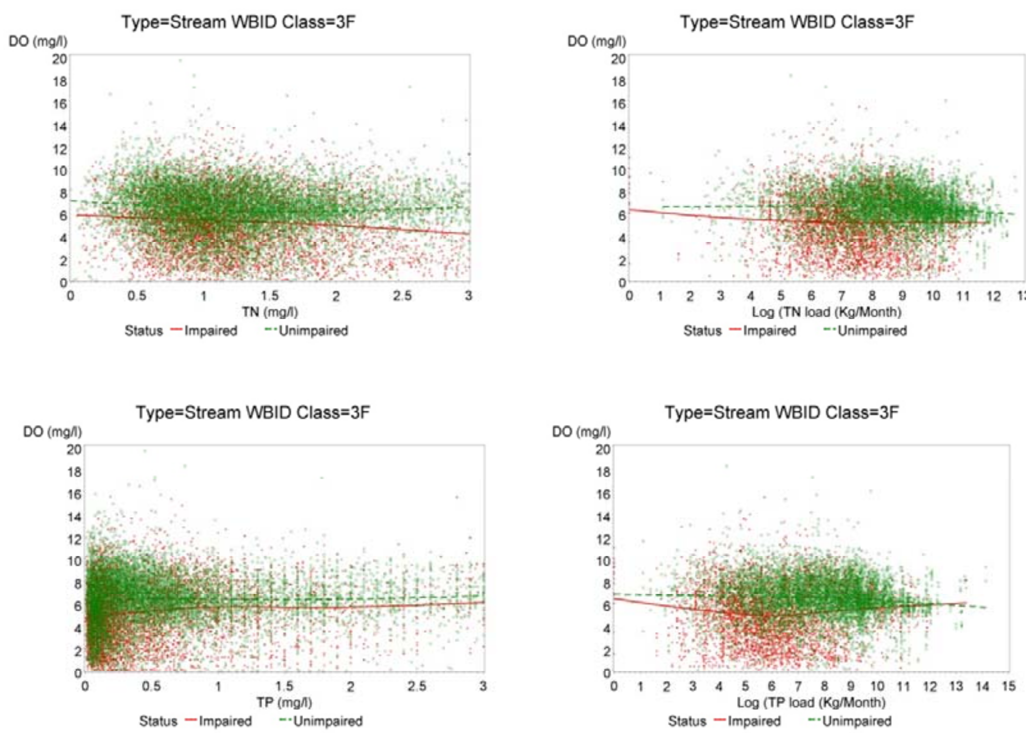


Figure 8. Relationship between DO and TP concentrations and loads in 3M and 3F WBIDs.

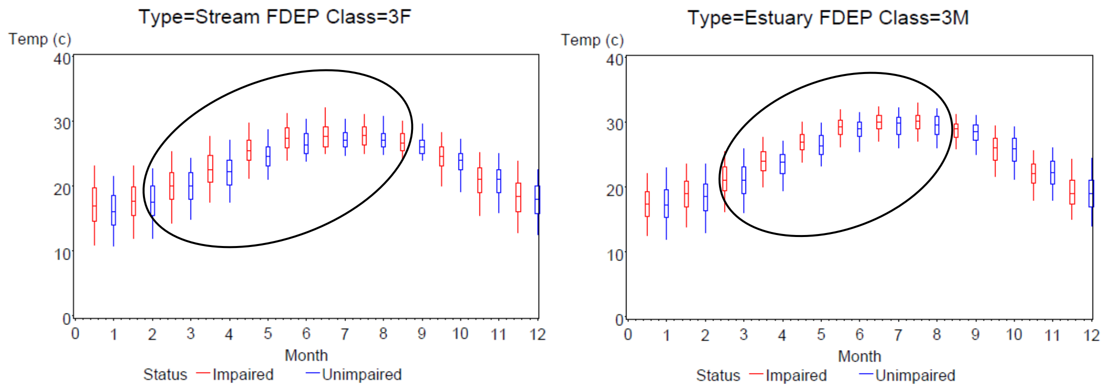


Figure 9. Comparison of water temperature in impaired and unimpaired 3M and 3F waters.

SUMMARY AND RECOMMENDATIONS

The analysis described above was performed as an attempt to identify unifying principles that drive relationships between anthropogenic sources and dissolved oxygen concentrations in waters classified as impaired by FDEP compared to unimpaired WBIDs. The results of the analysis found that the degree of urbanization within the WBID of freshwater streams had a substantial negative effect on probability for the WBID to be classified as impaired (i.e., an increased probability of impairment). In estuarine waters (excluding major Tampa Bay segments), the percentage of wetlands in the WBID had a protective effect (i.e., a reduced likelihood of being classified as impaired). Unimpaired WBIDs had higher hydrologic loads, higher TN loads and higher delivery ratios (expressed as TN load/hydrologic load). This finding indicates that something other than simply pollutant loading is responsible for a WBID being classified as impaired for dissolved oxygen. Water temperatures were higher in Impaired WBIDs. The ability for water to hold oxygen is known to be a physical function of water temperature and can also be the result of increased urbanization of the watershed. Correcting for temperature by using dissolved oxygen percent saturation ameliorated the differences in dissolved oxygen levels between impaired and unimpaired WBIDs and should be considered a reasonable alternative criterion that corrects for the effects of water temperature at the time of sampling.

The current development of TMDLs for Tampa Bay area tributaries has neglected to consider the impact of physical alterations to the waterbodies under consideration. The results presented above suggest that these physical alterations both within the waterbody and within the watershed play an important role in the ability of the waterbody to function as a natural system. Alterations to stream morphology and hydrology impact the ability of a stream to properly function like a natural stream and must be considered when establishing a TMDL. Many of the streams identified as impaired have been significantly altered to convey water away from urbanized areas. These alterations have negatively affected the function of these streams and it is unclear that pollutant load reductions alone will result in improved conditions. It becomes clear that there are many factors that must be accounted for to derive an explanatory relationship between dissolved oxygen concentrations and anthropogenic effects on Class 3 waters in the Tampa Bay watershed.

Alterations to stream morphology associated with urbanization or agriculture are common issues in the Tampa Bay region. These alterations have dramatically influenced the volume, timing and residence times of hydrology. Alterations to streambeds have compromised an important ecologic function of the instream processes that control the assimilation and processing of nutrients in the stream itself. Unless these processes are restored it is unclear that pollutant reductions alone will result in increased water quality as measured by dissolved oxygen and chlorophyll concentrations. The BMAP process can and should continue in impaired waterbodies with altered streams but focus should be on restoring stream function where possible to restore the assimilative capacity of the streams as pollutant reductions alone will likely not result in the waters meeting the FDEP criteria established based on a reference system approach. Where stream restoration is not possible it is recommended that FDEP consider changing the designated use of highly altered systems to reflect their status as compromised in their function as a natural stream. This does not mean that the stream has no ecological value but rather that a reference system approach is not valid as an assessment for evaluating the potential ecological function of the system.

Given the analyses and discussion presented above, we offer the following recommendations. If either the observed impairment, particularly those due to DO exceedances, can be attributed to natural phenomena or cannot be attributed to a causative agent or pollutant, then consideration should be given to re-categorizing the existing impairment into either a 4c or 4d subcategory. Any management actions to reduce nutrient or BOD loadings to impaired WBIDs that fall into these categories would require a significant investment to address nonpoint source loads. If other factors, such as water temperature or hydrologic loads, significantly influence whether nonpoint source load reductions can be reasonably expected to achieve the designated uses in these WBIDs, then further analyses are obviously warranted before proceeding with implementation of the TMDL and the commitment of local government resources.

Additionally, we agree with the FDEP proposition in a recent series of TMDLs for WBIDs within the Tampa Bay and Hillsborough River basins that protecting Tampa Bay by setting TMDLs within contributing basins is appropriate. However, it should be recognized that protection of Tampa Bay does not require that every WBID draining to the bay meet the same water quality targets as those established for the bay. Through the Reasonable Assurance process, it was determined that maintaining the hdyrologically-normalized 2003-2007 segment TN loads is protective of water quality in each segment (Tampa Bay Estuary Program and Janicki Environmental, 2009). Using this logic, the TMDL for an impaired WBID within the Tampa Bay watershed could be defined as the 2003-2007 hydrologically-normalized TN loads from the WBID and its watershed to Tampa Bay. This would result in a cohesive management plan for Tampa Bay and its tributaries.

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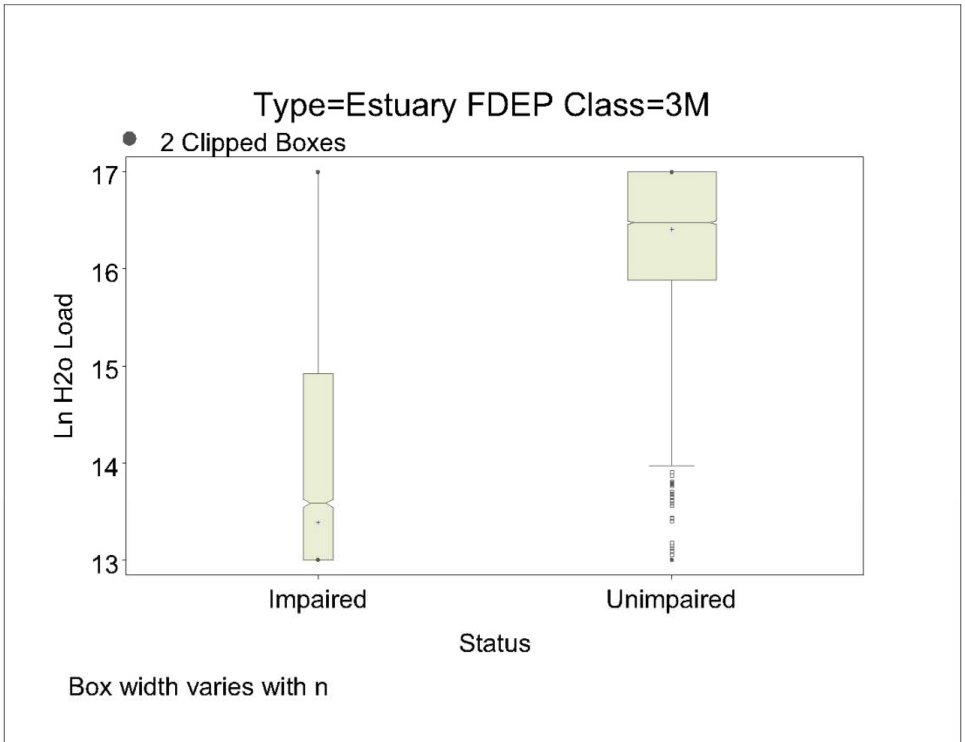
APPENDIX 1.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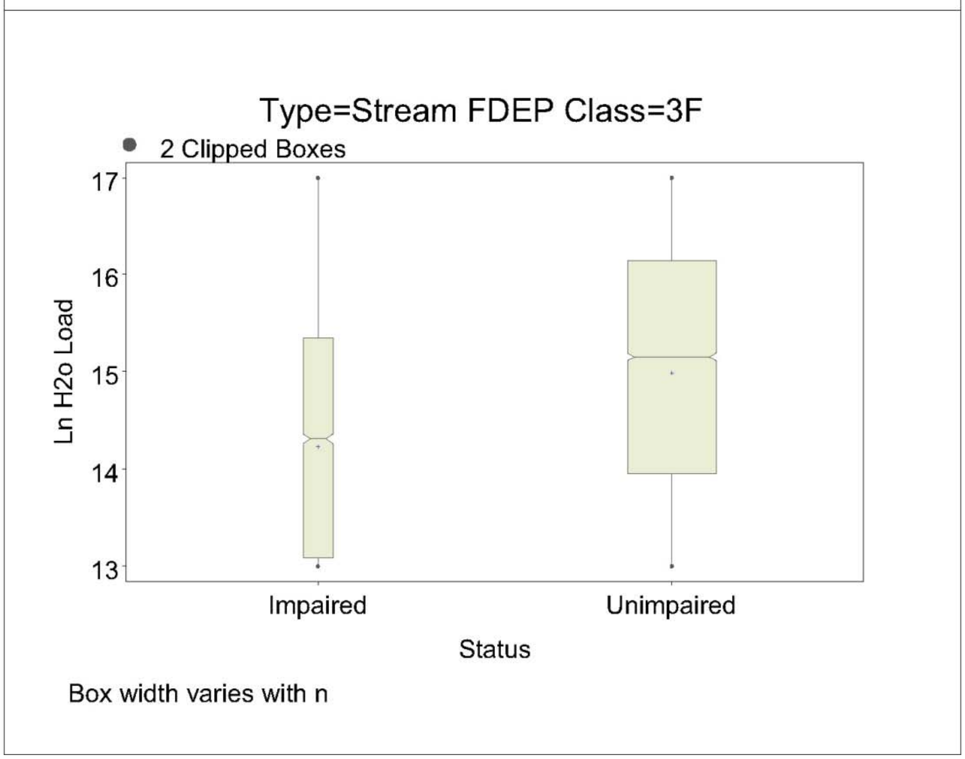
2 Clipped Boxes
17
16
15
14
13
Ln H2o Load
Impaired
Unimpaired
Status
Box width varies with n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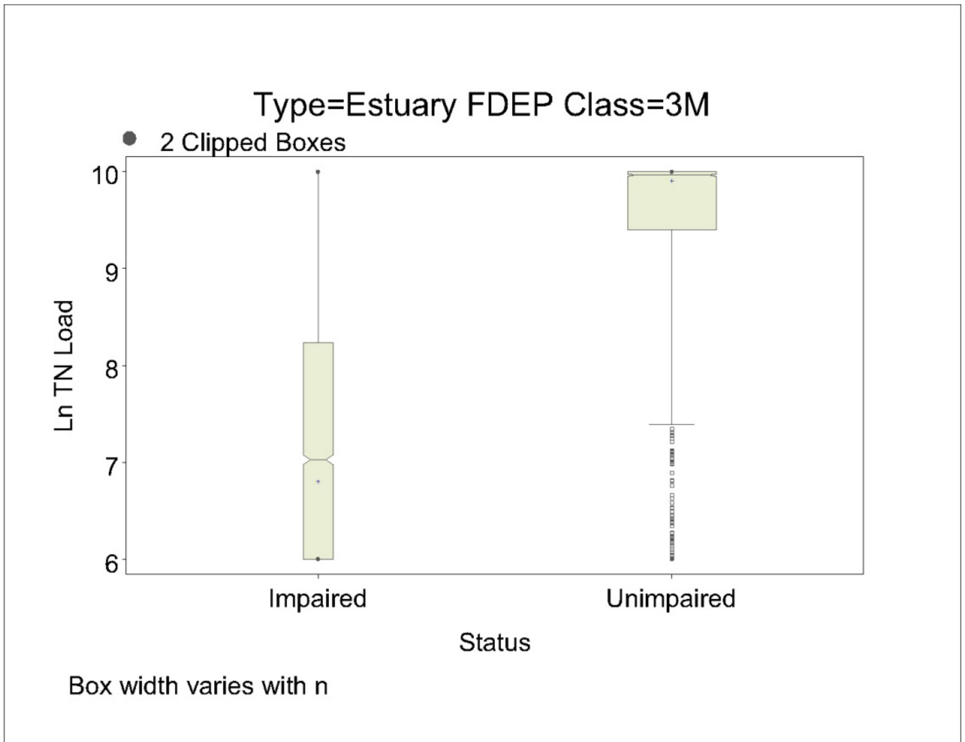
16
15
14
13
Ln H2o Load
Impaired
Unimpaired
Status
Box width varies with n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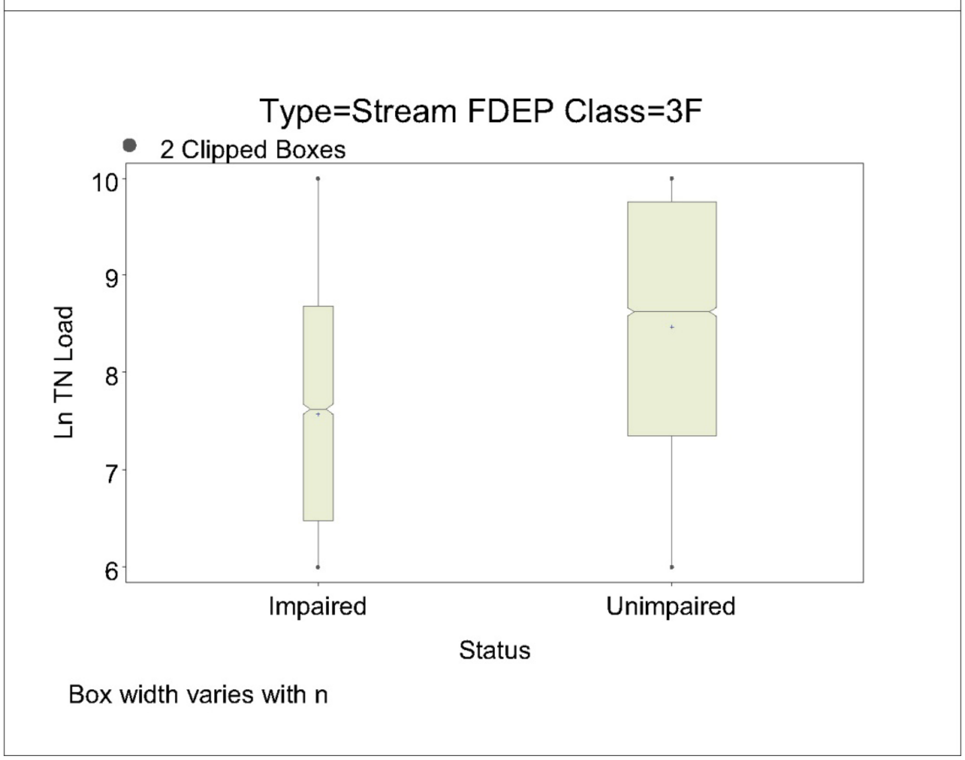
2 Clipped Boxes
10
9
8
7
6
Ln TN Load
Impaired
Unimpaired
Status
Box width varies with n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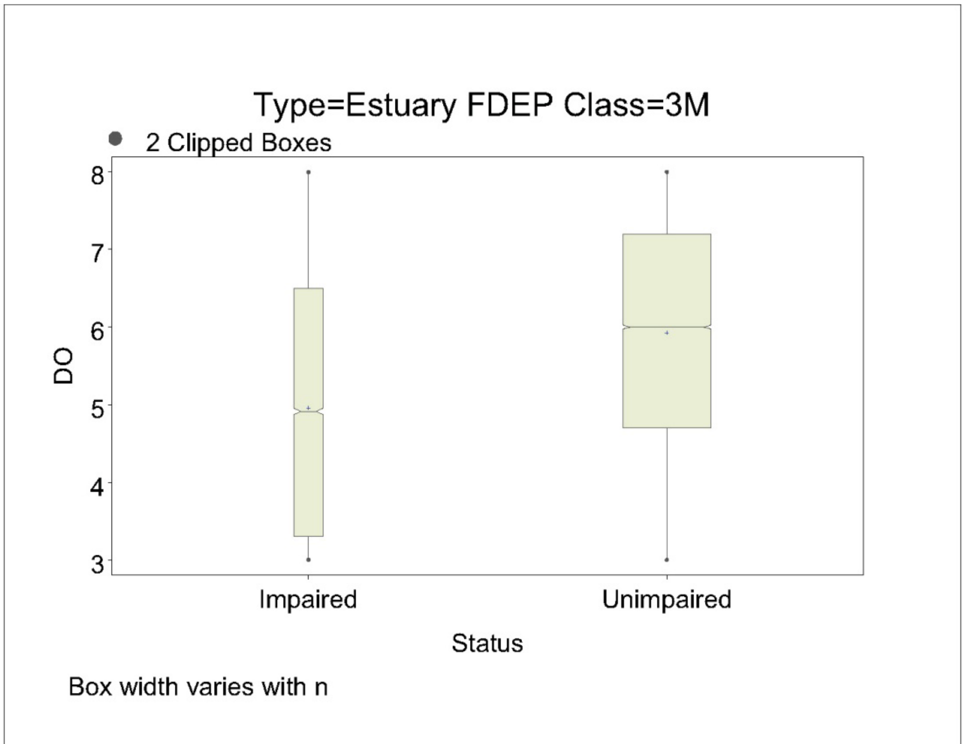
9
8
7
6
Ln TN Load
Impaired
Unimpaired
Status
Box width varies with n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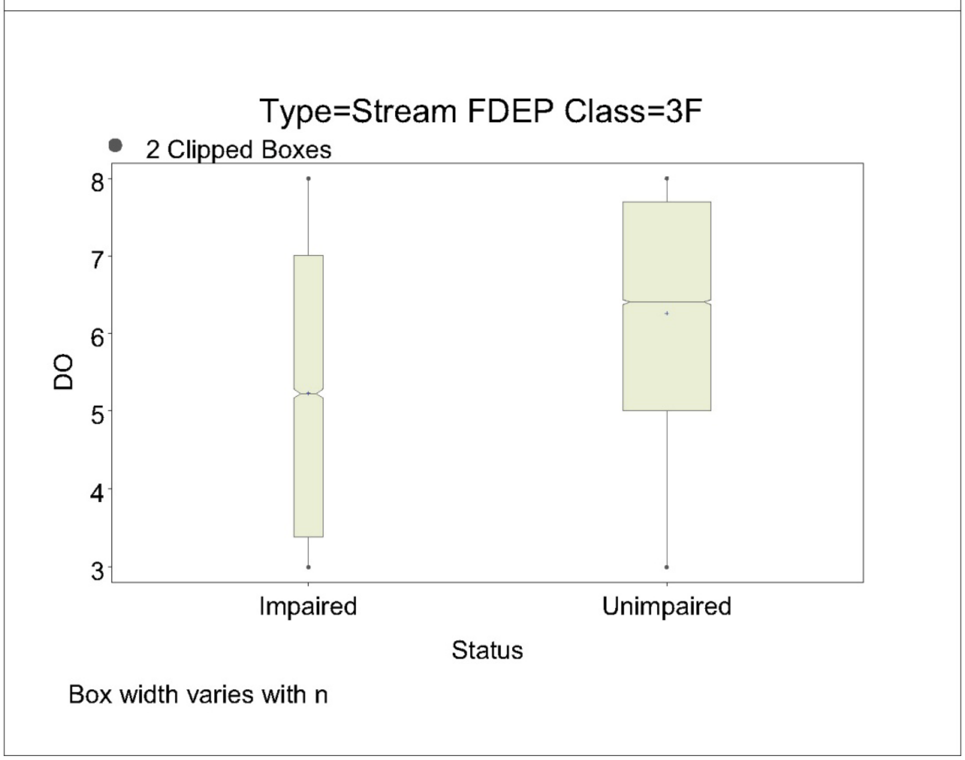
2 Clipped Boxes
8
7
6
5
4
3
DO
Impaired
Unimpaired
Status
Box width varies with n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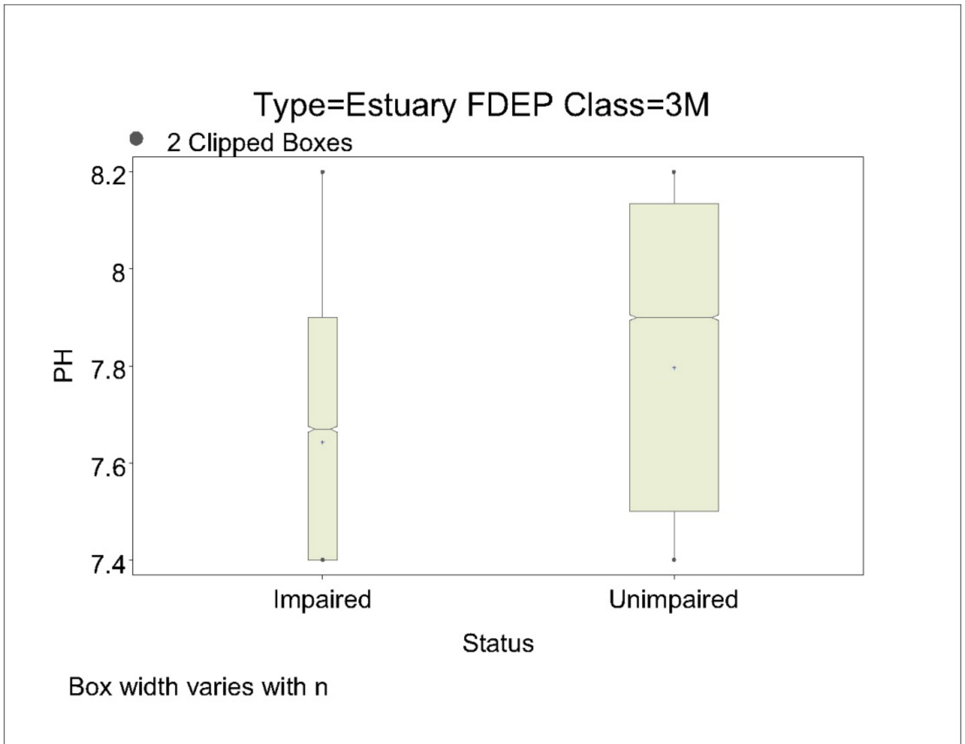
2 Clipped Boxes
8
7
6
5
4
3
DO
Impaired
Unimpaired
Status
Box width varies with n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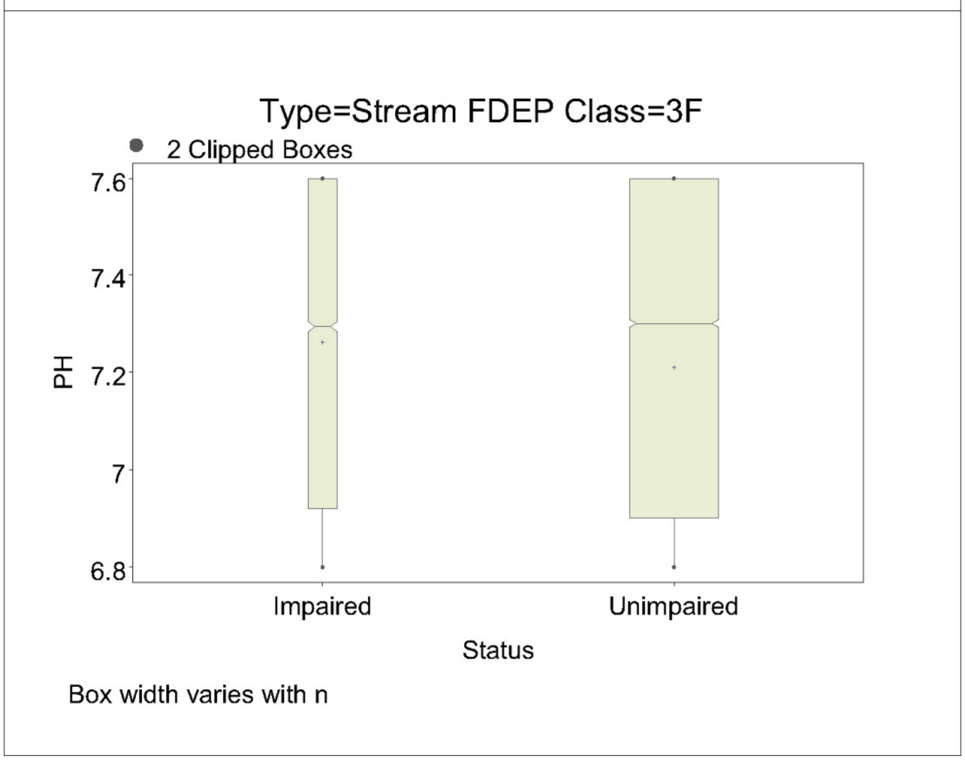
2 Clipped Boxes
8.2
8
7.8
7.6
7.4
PH
Impaired
Unimpaired
Status
Box width varies with n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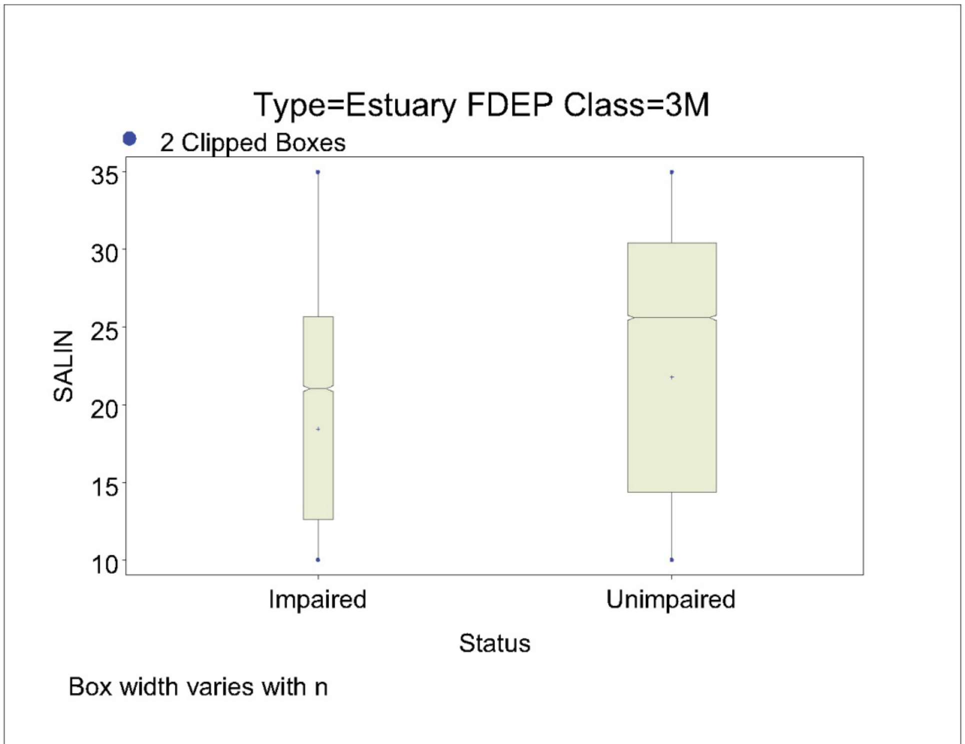
7.6
7.4
7.2
7
6.8
PH
Impaired
Unimpaired
Status
Box width varies with n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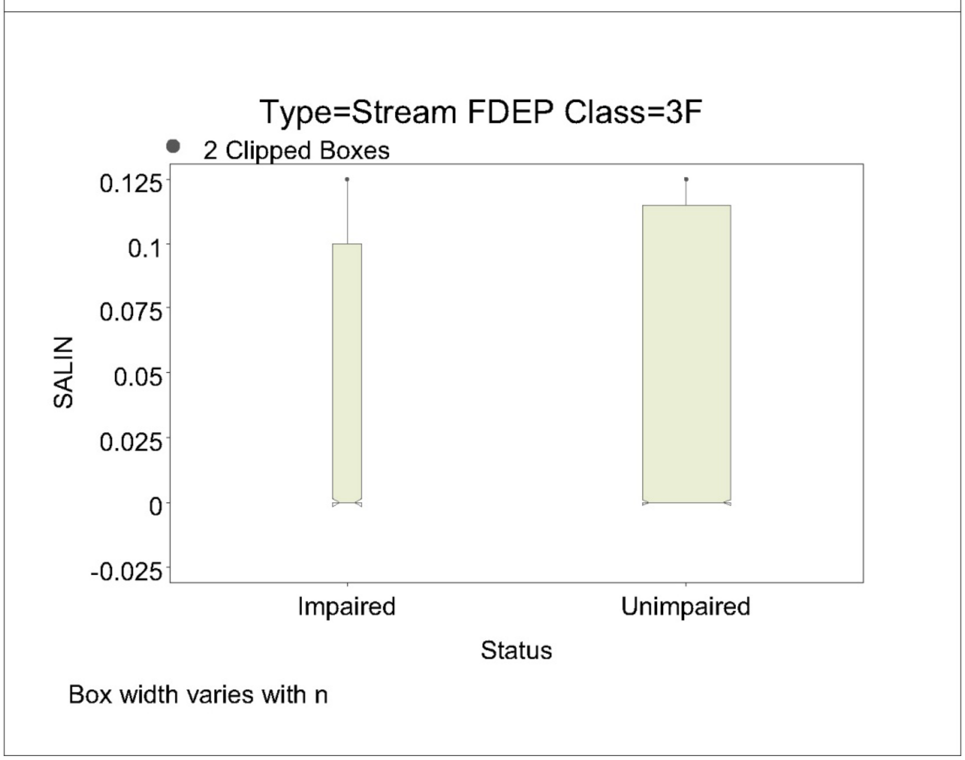
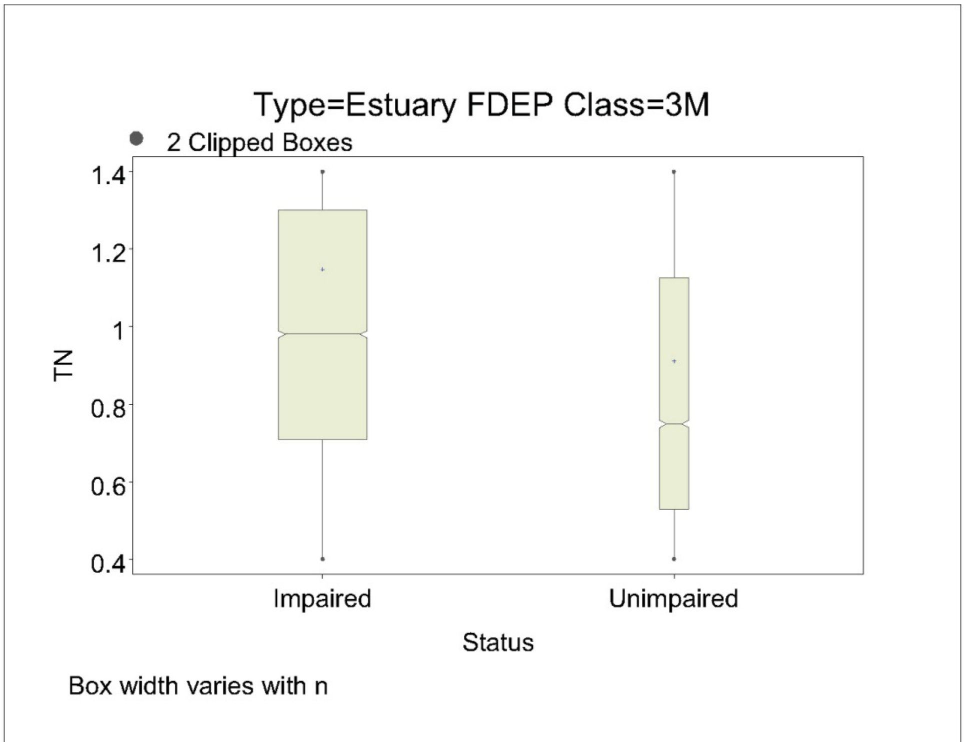
2 Clipped Boxes
35
30
25
20
15
10
SALIN
Impaired
Unimpaired
Status
Box width varies with n


Type=Stream FDEP Class=3F
2 Clipped Boxes
0.125
0.1
0.075
0.05
0.025
0
-0.025
SALIN
Impaired
Unimpaired
Status
Box width varies with n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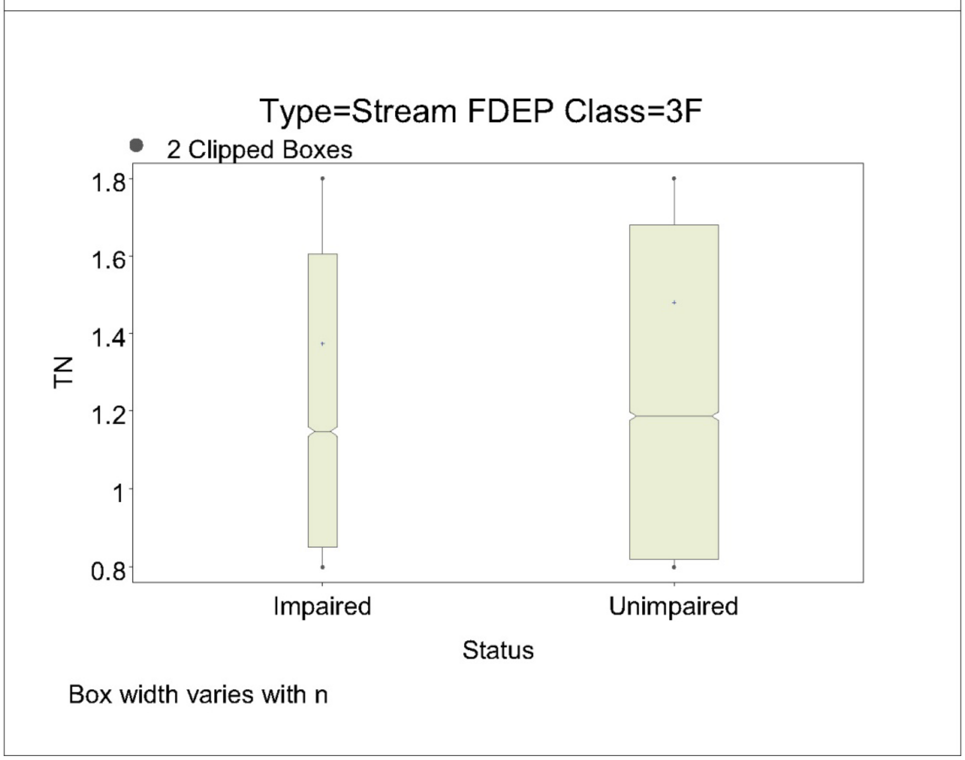
2 Clipped Boxes
1.4
1.2
1
0.8
0.6
0.4
TN
Impaired
Unimpaired
Status
Box width varies with n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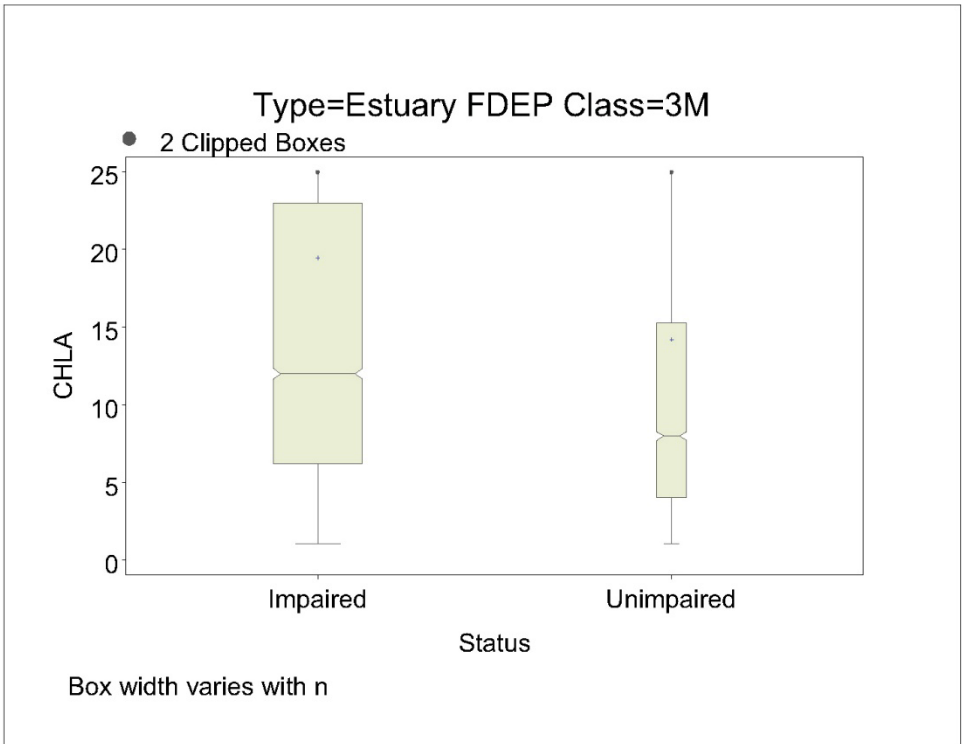
1.8
1.6
1.4
1.2
1
0.8
TN
Impaired
Unimpaired
Status
Box width varies with n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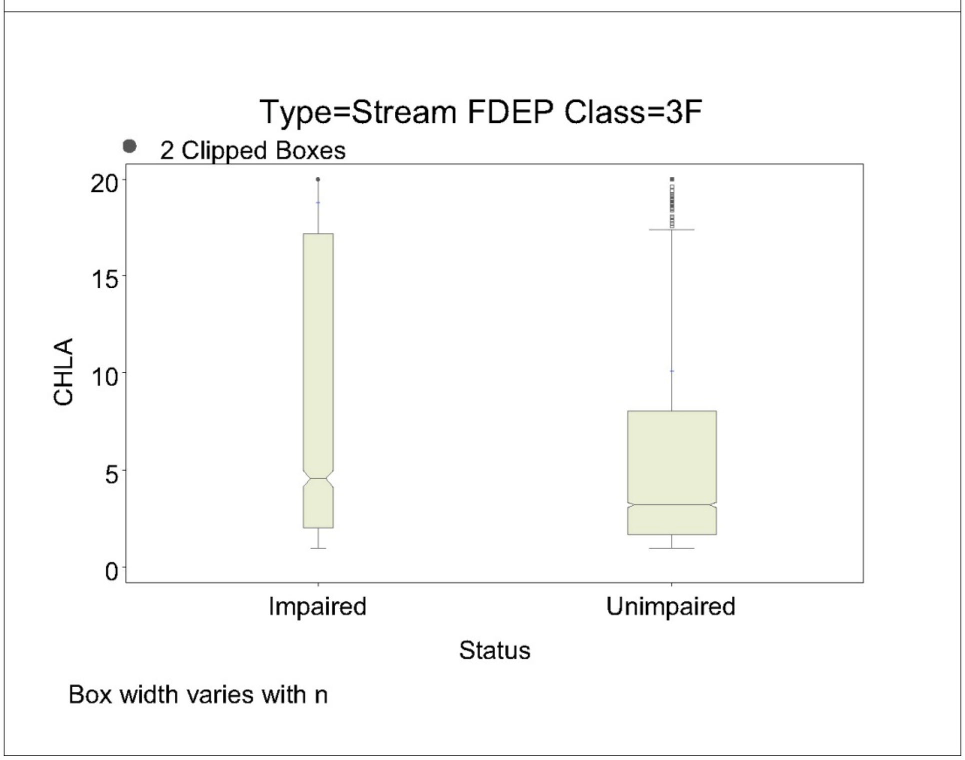
2 Clipped Boxes
25
20
15
10
5
0
CHLA
Impaired
Unimpaired
Status
Box width varies with n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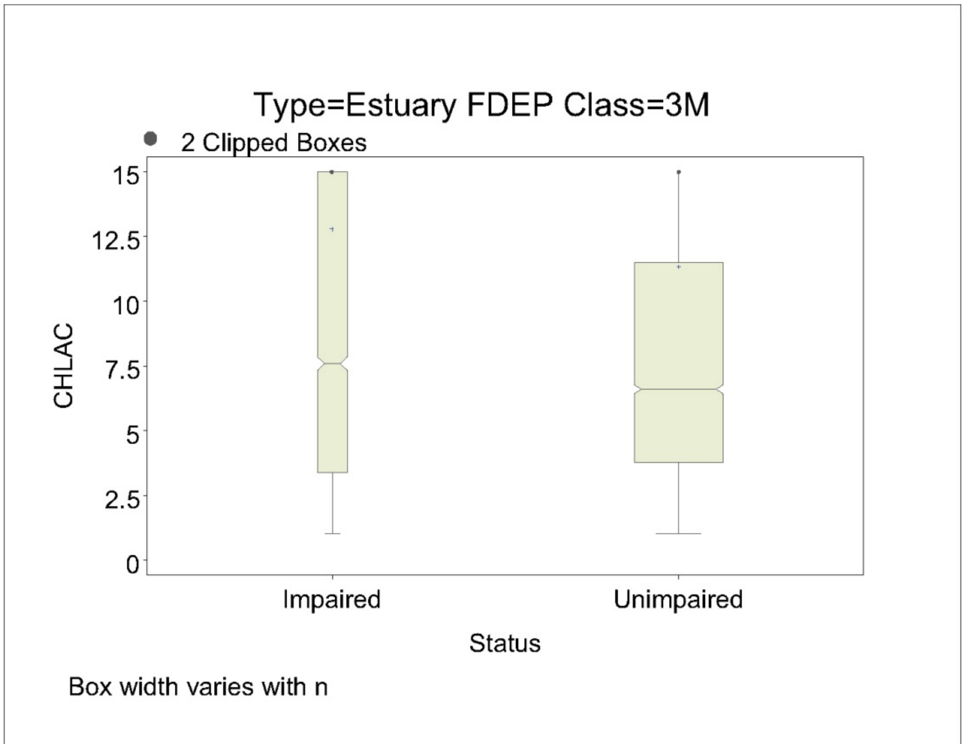
2 Clipped Boxes
20
15
10
5
0
CHLA
Impaired
Unimpaired
Status
Box width varies with n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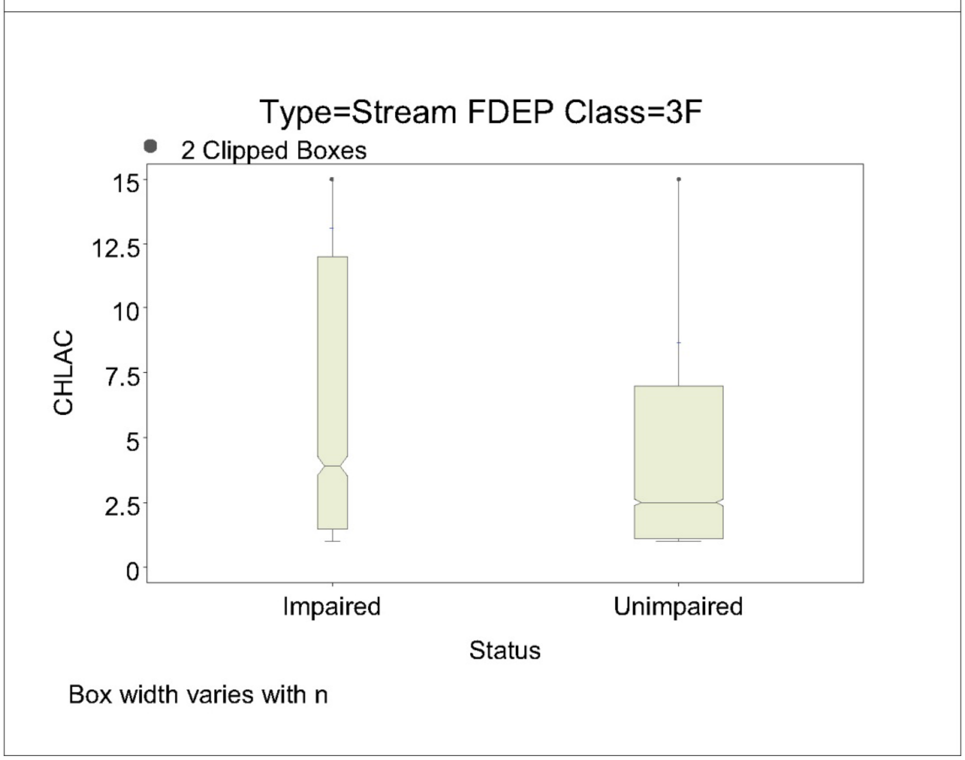
2 Clipped Boxes
15
12.5
10
7.5
5
2.5
0
CHLAC
Impaired
Unimpaired
Status
Box width varies with n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2 Clipped Boxes
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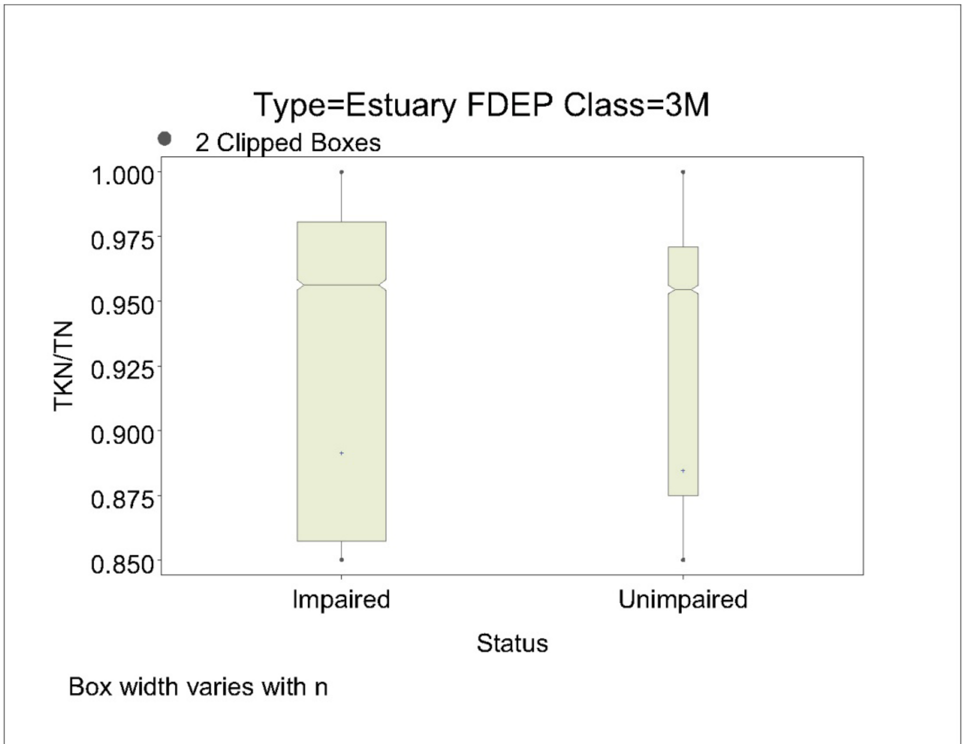
12.5
10
7.5
5
2.5
0
CHLAC
Impaired
Unimpaired
Status
Box width varies with n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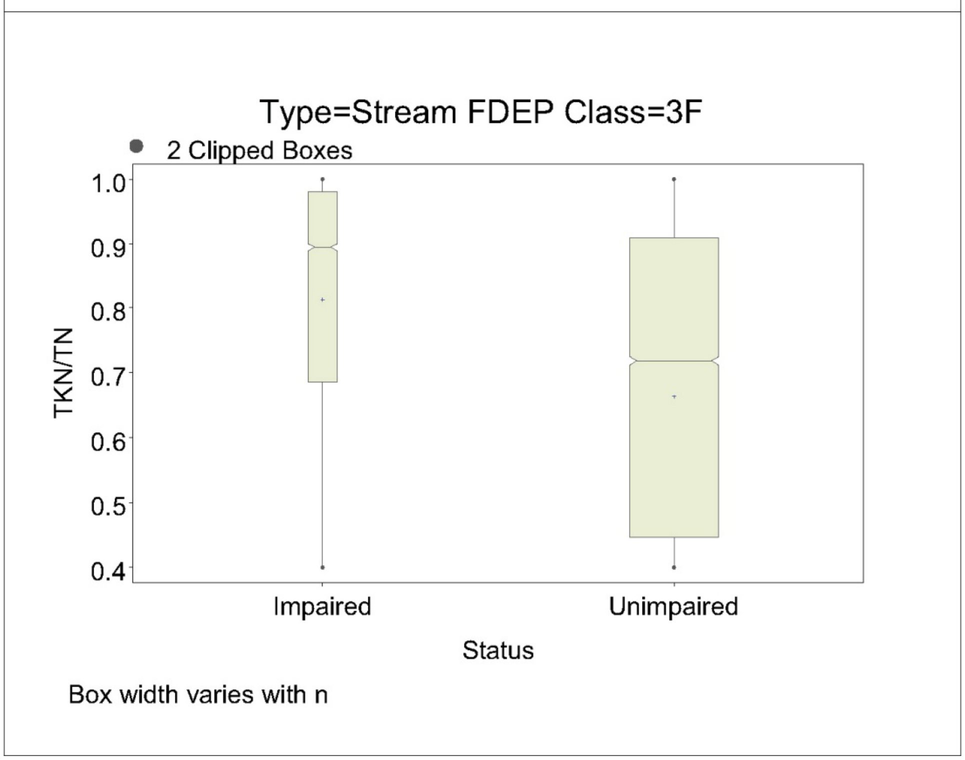
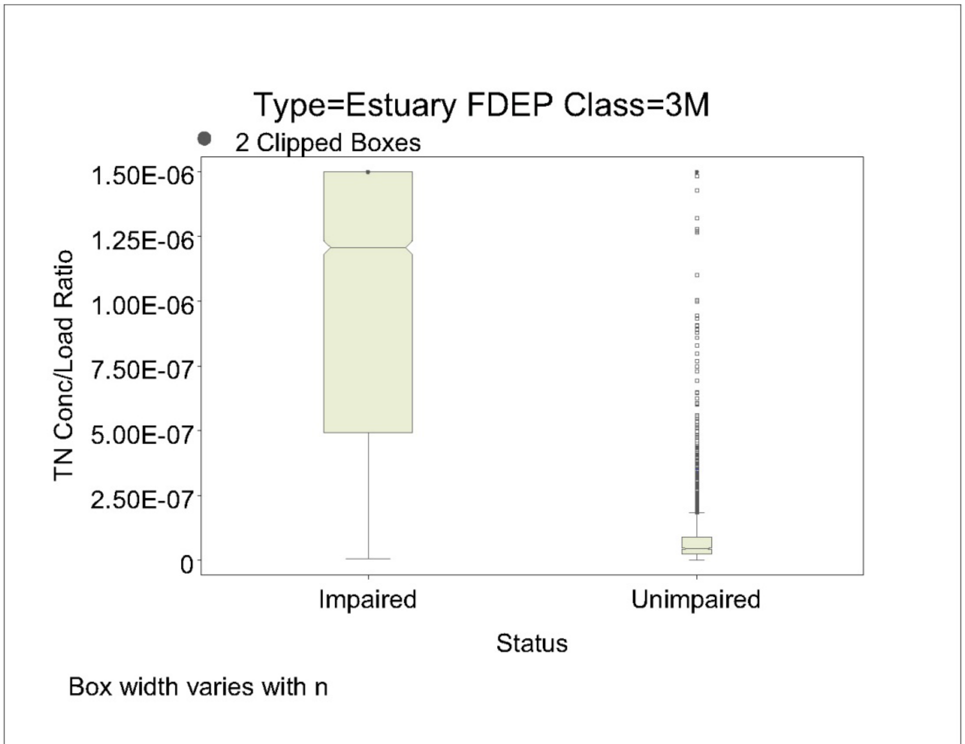
2 Clipped Boxes
1.000
0.975
0.950
0.925
0.900
0.875
0.850
TKN/TN
Impaired
Unimpaired
Status
Box width varies with n


Type=Stream FDEP Class=3F
2 Clipped Boxes
1.0
0.9
0.8
0.7
0.6
0.5
0.4
TKN/TN
Impaired
Unimpaired
Status
Box width varies with n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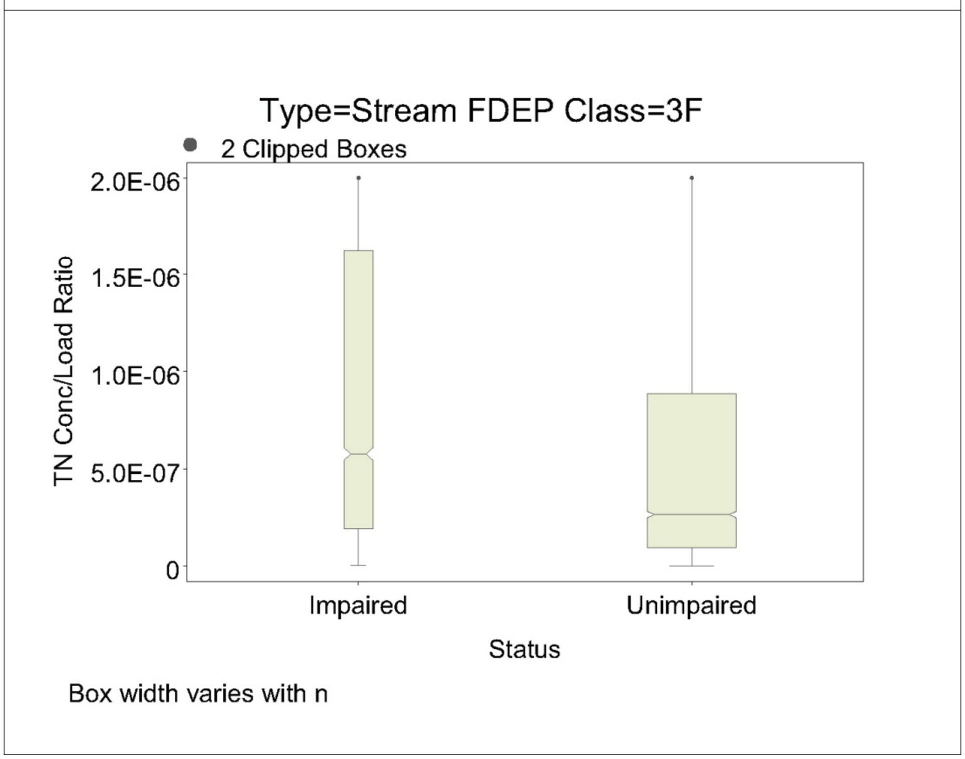
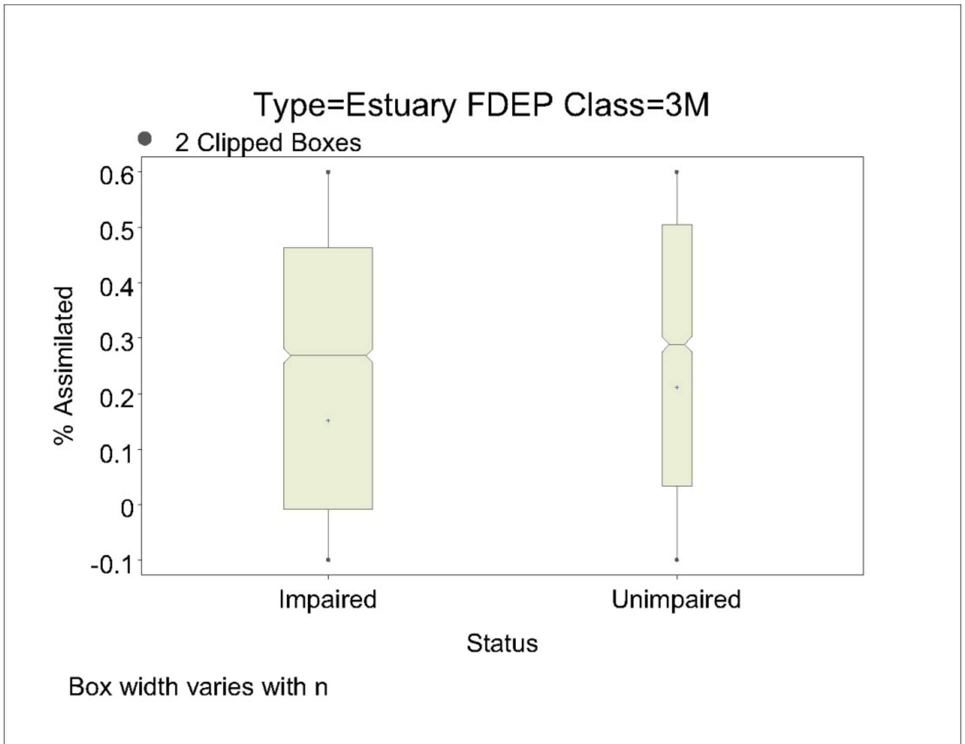
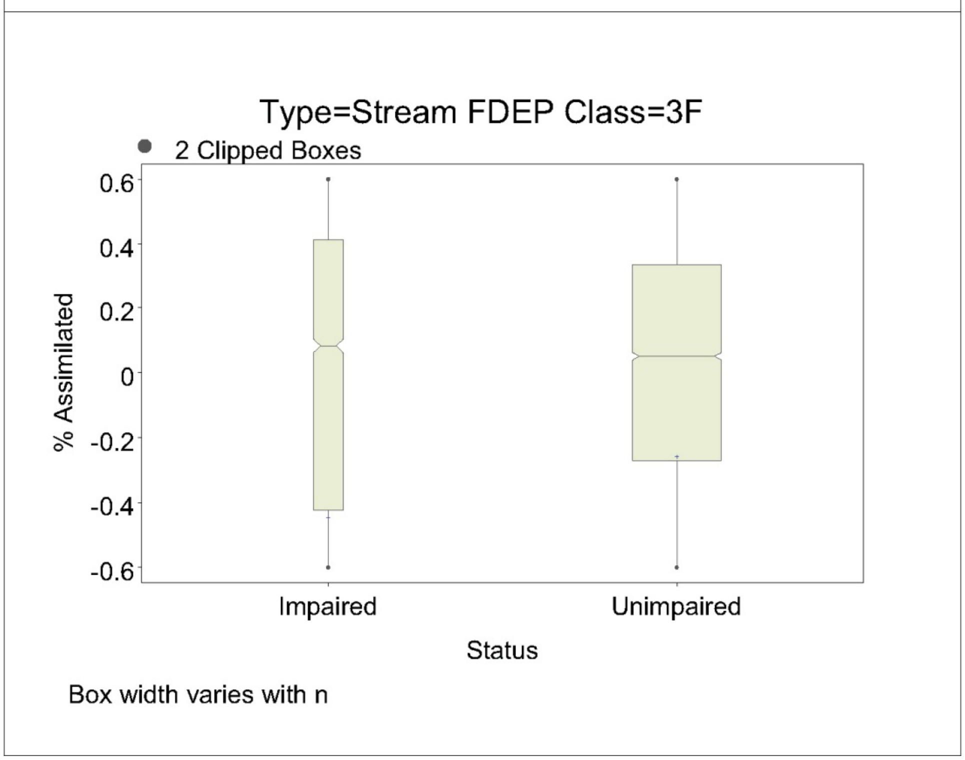
2 Clipped Boxes
1.50E-06
1.25E-06
1.00E-06
7.50E-07
5.00E-07
2.50E-07
0
TN Conc/Load Ratio
Impaired
Unimpaired
Status
Box width varies with n

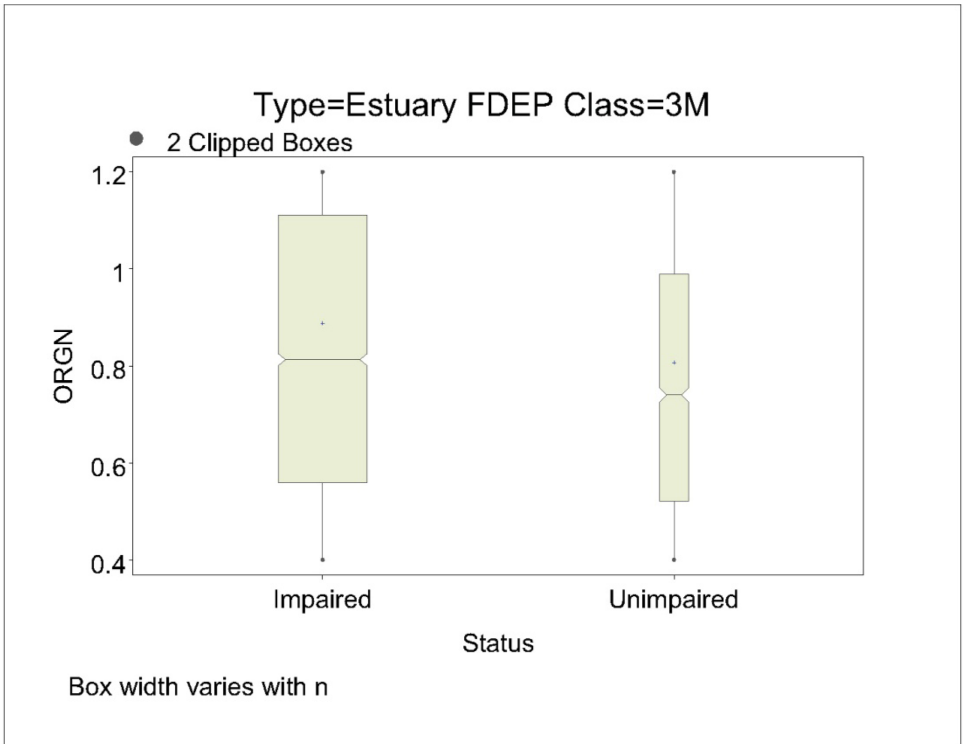

Type=Stream FDEP Class=3F
2 Clipped Boxes
2.0E-06
1.5E-06
1.0E-06
5.0E-07
0
TN Conc/Load Ratio
Impaired
Unimpaired
Status
Box width varies with n

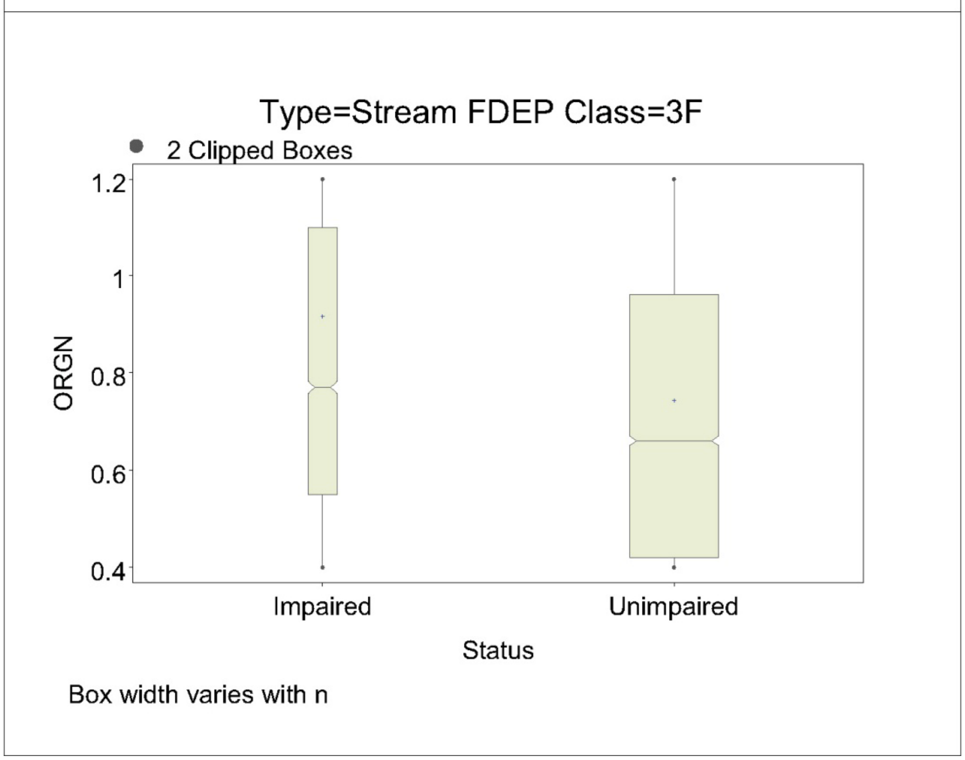


Type=Estuary FDEP Class=3M
2 Clipped Boxes
0.6
0.5
0.4
0.3
0.2
0.1
0
-0.1
% Assimilated
Impaired
Unimpaired
Status
Box width varies with n



Type=Stream FDEP Class=3F
2 Clipped Boxes
0.6
0.4
0.2
0
-0.2
-0.4
-0.6
% Assimilated
Impaired
Unimpaired
Status
Box width varies with n


Type=Estuary FDEP Class=3M
2 Clipped Boxes
1.2
1
0.8
0.6
0.4
ORGN
Impaired
Unimpaired
Status
Box width varies with n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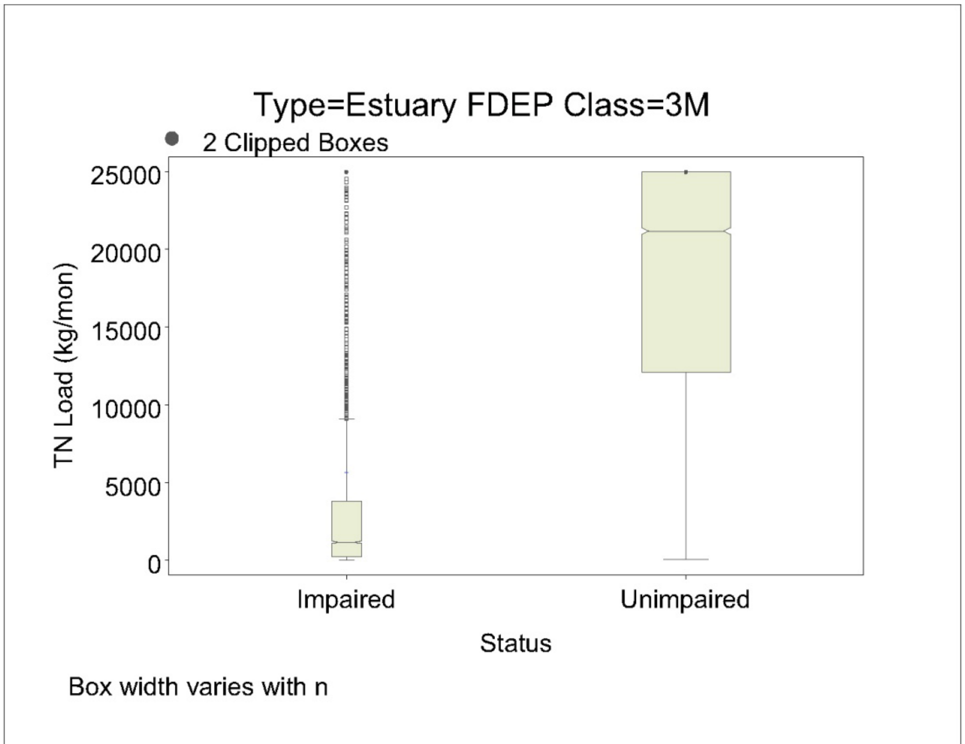
2 Clipped Boxes
1.2
1
0.8
0.6
0.4
ORGN
Impaired
Unimpaired
Status
Box width varies with n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2 Clipped Boxes
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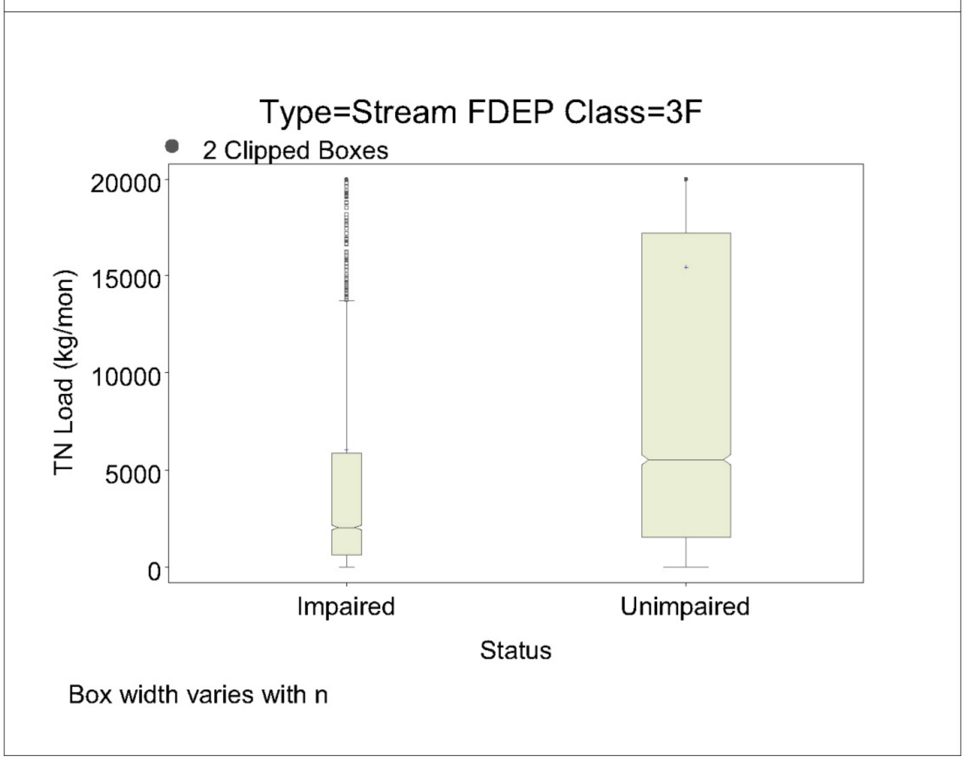
20000
15000
10000
5000
0
TN Load (kg/mon)
Impaired
Unimpaired
Status
Box width varies with n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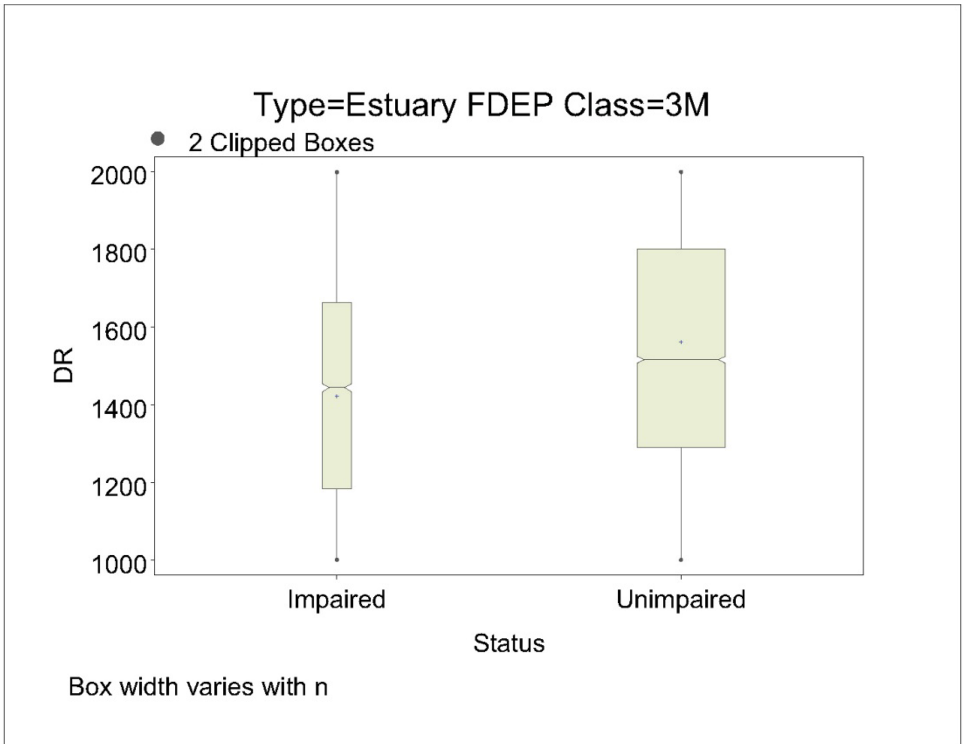
2 Clipped Boxes
20000
15000
10000
5000
0
TN Load (kg/mon)
Impaired
Unimpaired
Status
Box width varies with n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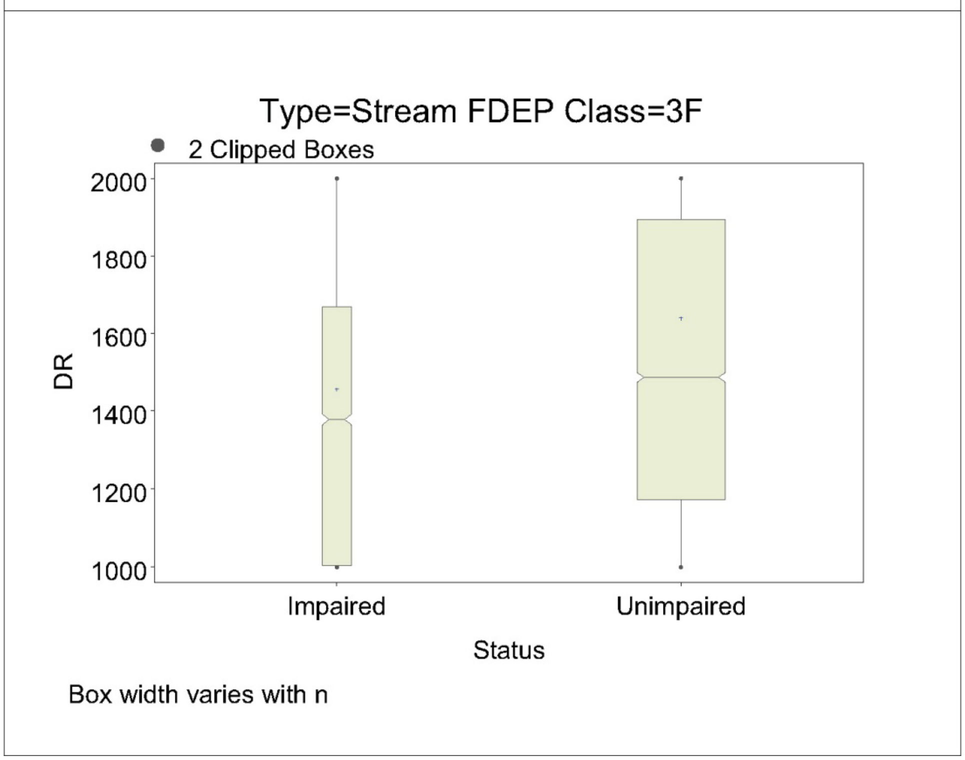
2 Clipped Boxes
2000
1800
1600
1400
1200
1000
DR
Impaired
Unimpaired
Status
Box width varies with n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2 Clipped Boxes
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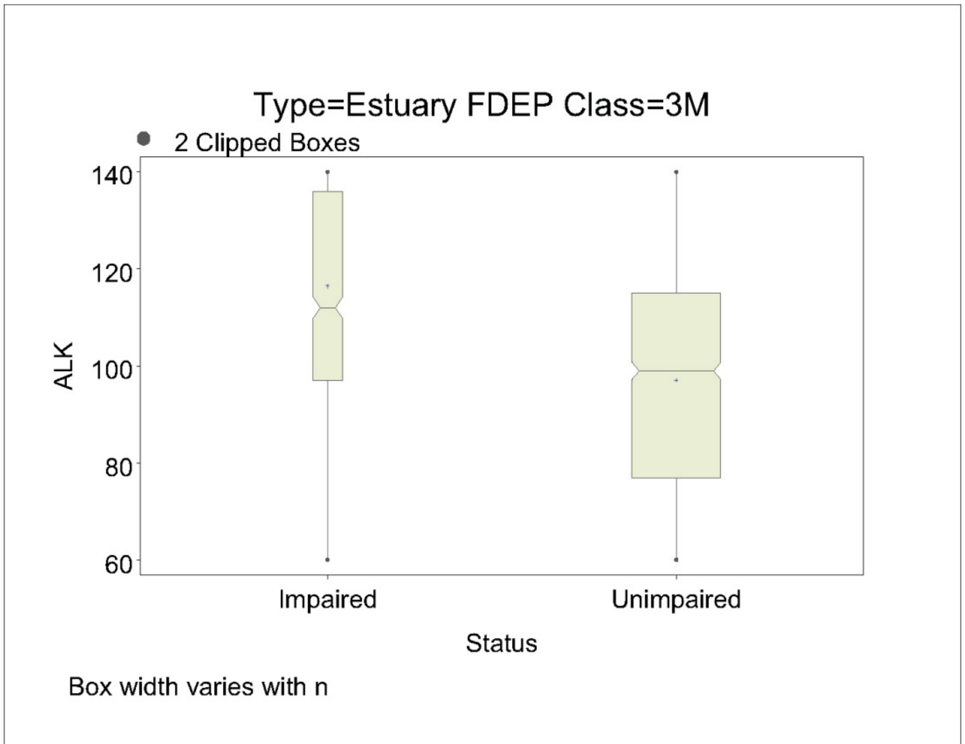
1800
1600
1400
1200
1000
DR
Impaired
Unimpaired
Status
Box width varies with n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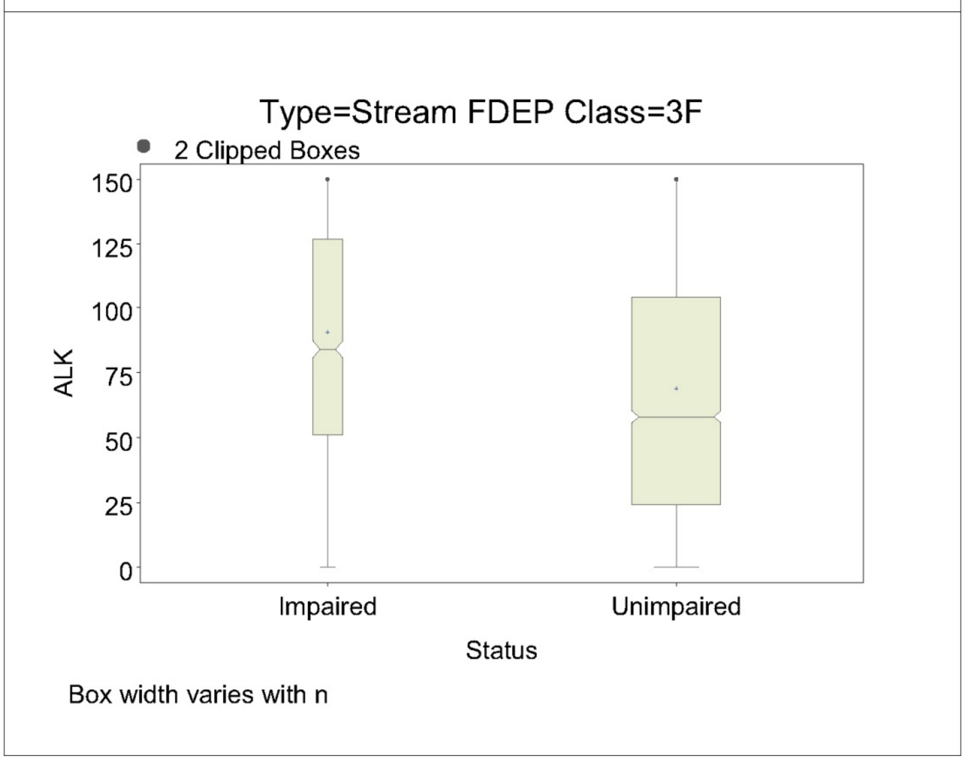
2 Clipped Boxes
140
120
100
80
60
ALK
Impaired
Unimpaired
Status
Box width varies with n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2 Clipped Boxes
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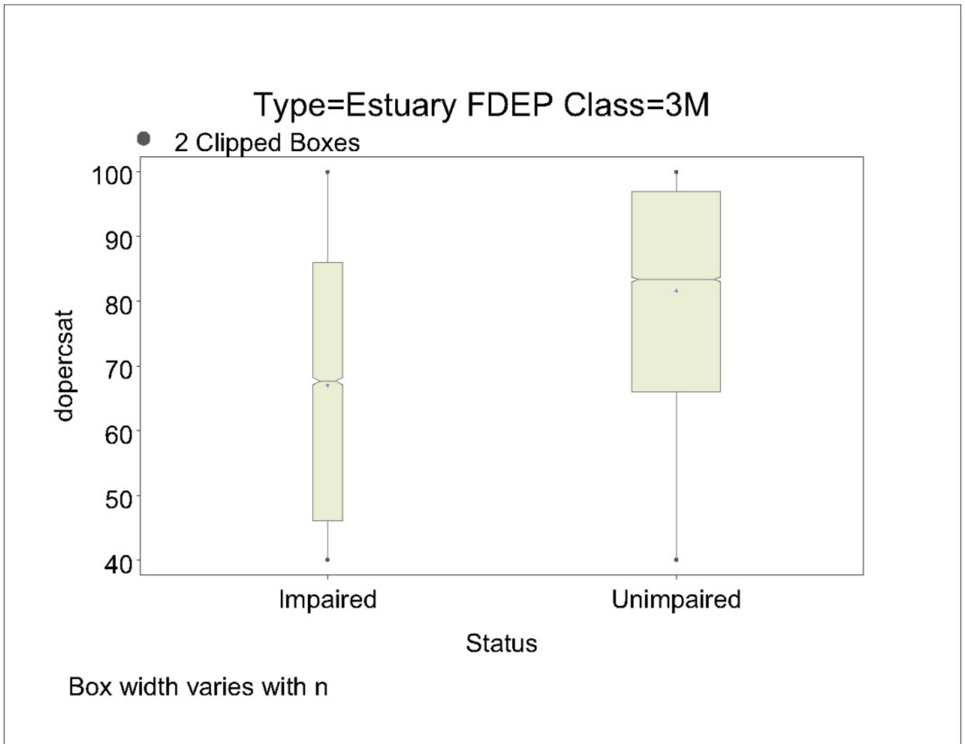
125
100
75
50
25
0
ALK
Impaired
Unimpaired
Status
Box width varies with n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2 Clipped Boxes
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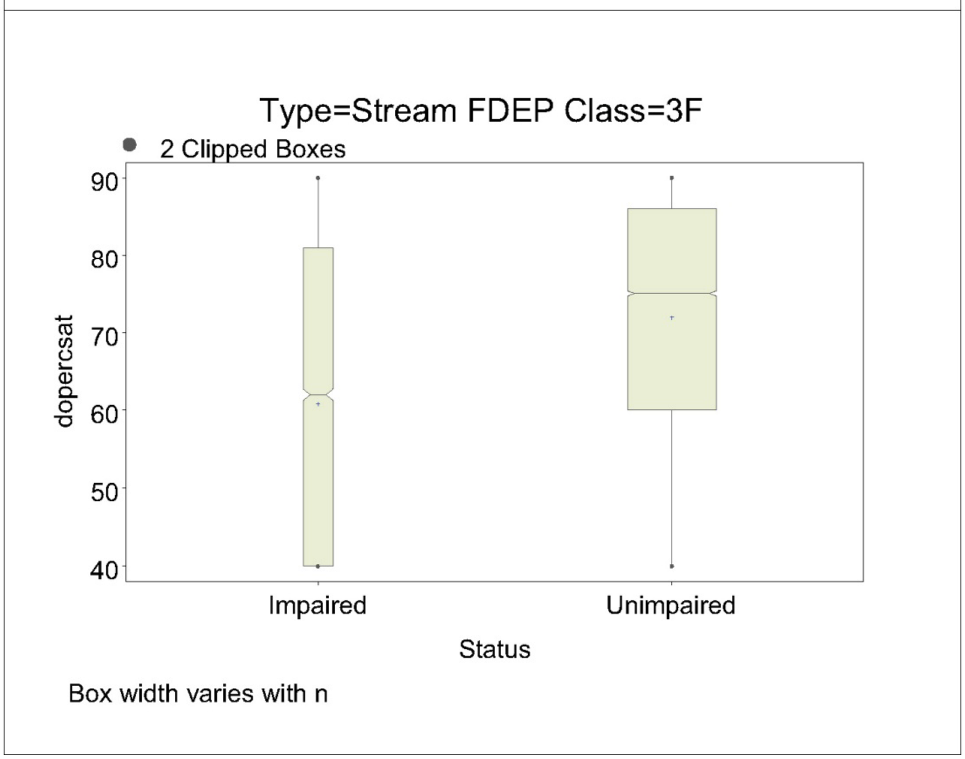
90
80
70
60
50
40
dopercsat
Impaired
Unimpaired
Status
Box width varies with n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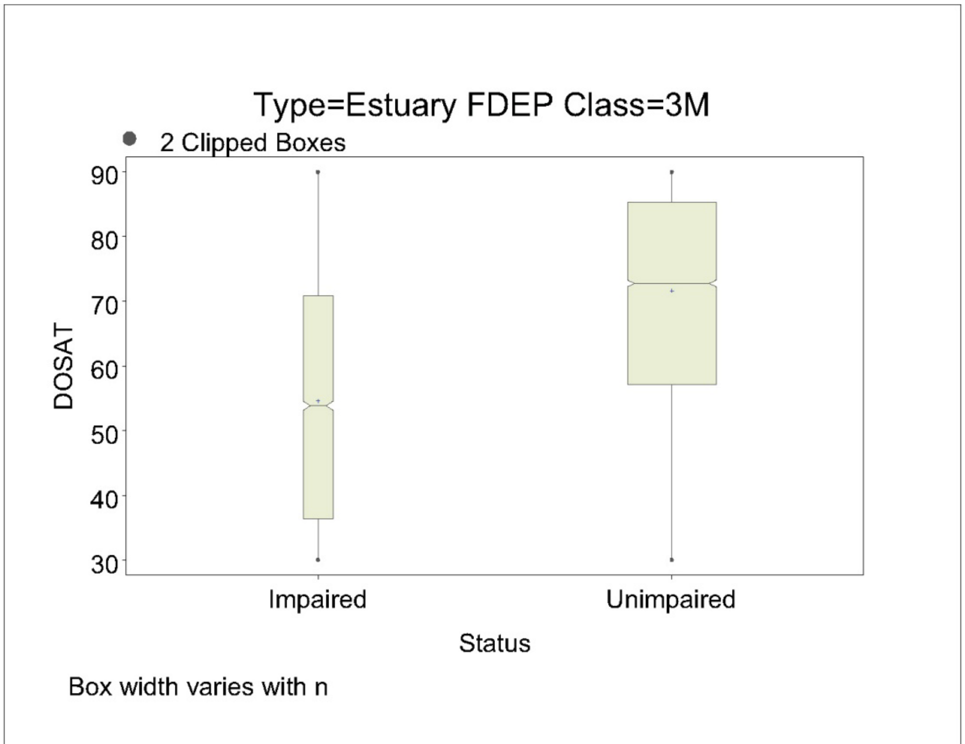
2 Clipped Boxes
90
80
70
60
50
40
dopercsat
Impaired
Unimpaired
Status
Box width varies with n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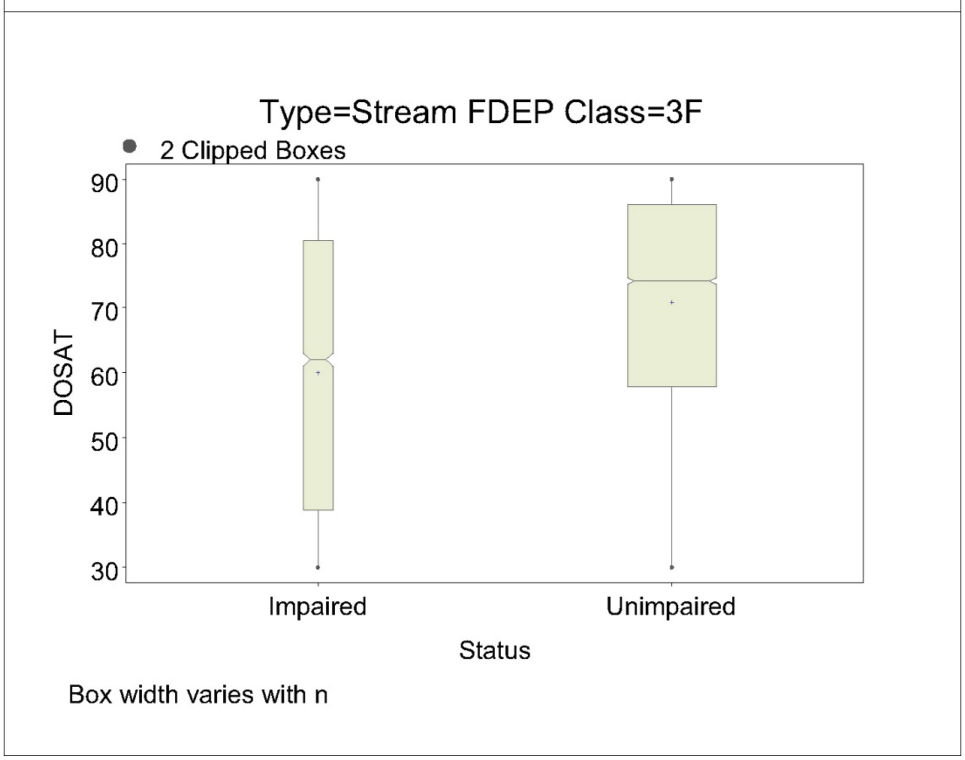
2 Clipped Boxes
90
80
70
60
50
40
30
DOSAT
Impaired
Unimpaired
Status
Box width varies with n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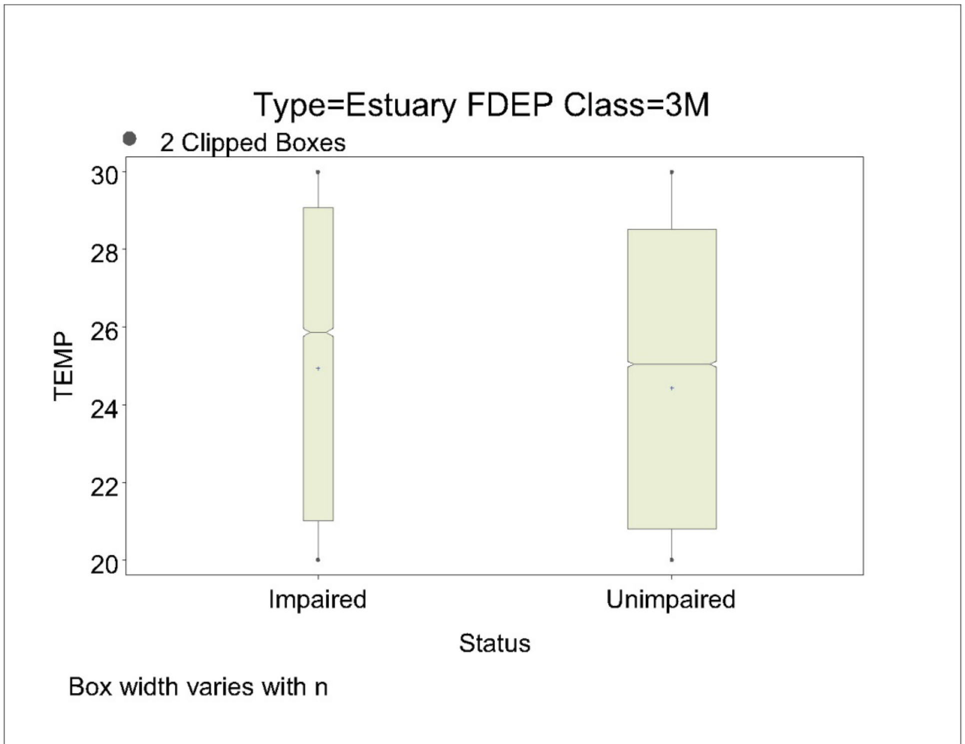
2 Clipped Boxes
90
80
70
60
50
40
30
DOSAT
Impaired
Unimpaired
Status
Box width varies with n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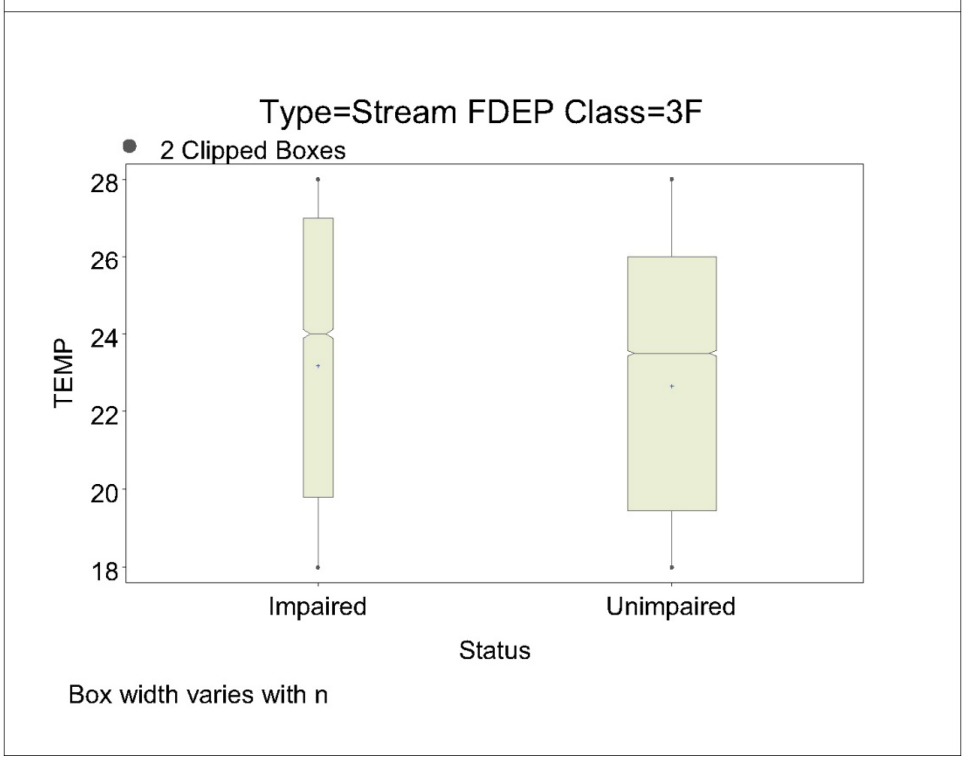
2 Clipped Boxes
30
28
26
24
22
20
TEMP
Impaired
Unimpaired
Status
Box width varies with n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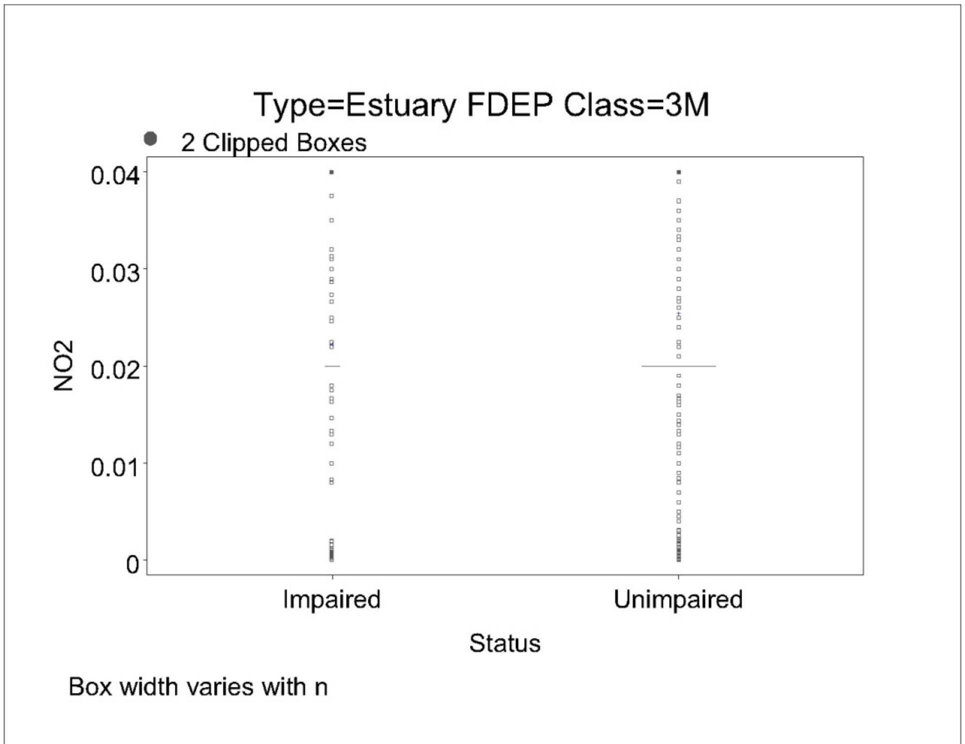
2 Clipped Boxes
28
26
24
22
20
18
TEMP
Impaired
Unimpaired
Status
Box width varies with n



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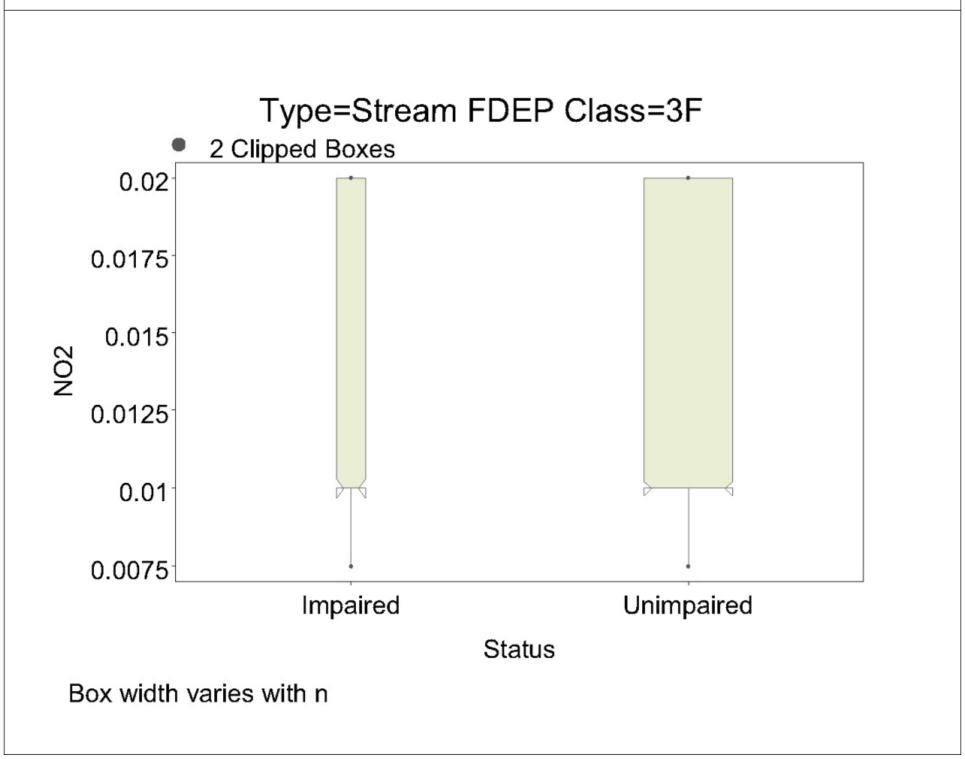
2 Clipped Boxes
0.04
0.03
0.02
0.01
0
NO2
Impaired
Unimpaired
Status
Box width varies with n



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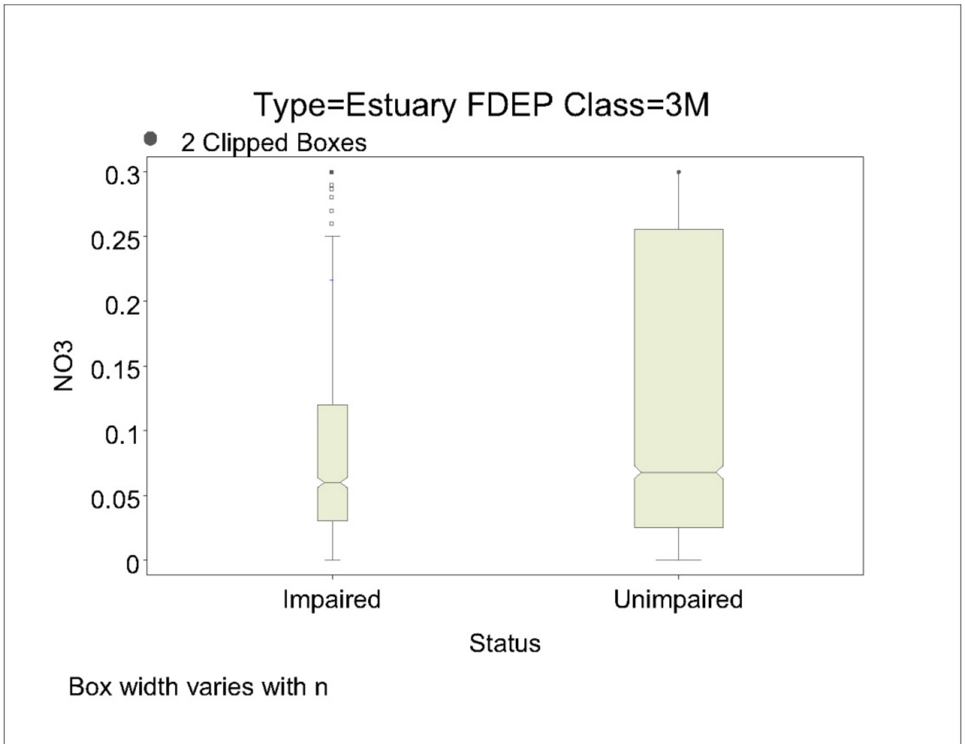
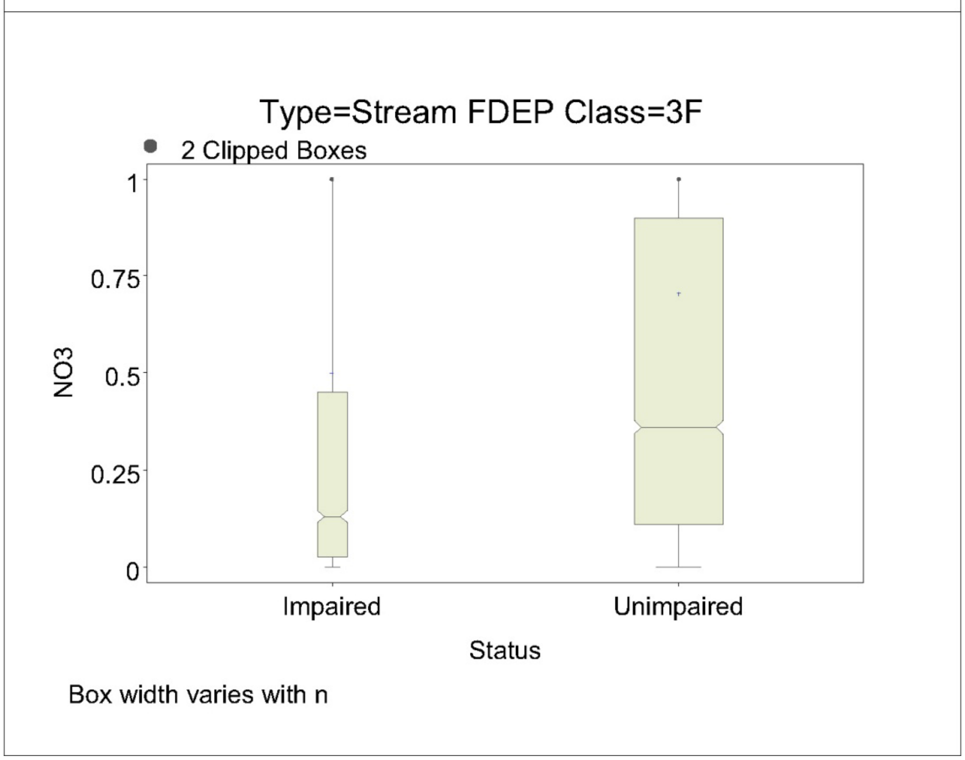
2 Clipped Boxes
0.02
0.0175
0.015
0.0125
0.01
0.0075
NO2
Impaired
Unimpaired
Status
Box width varies with n


Type=Estuary FDEP Class=3M
2 Clipped Boxes
0.3
0.25
0.2
0.15
0.1
0.05
0
NO3
Impaired
Unimpaired
Status
Box width varies with n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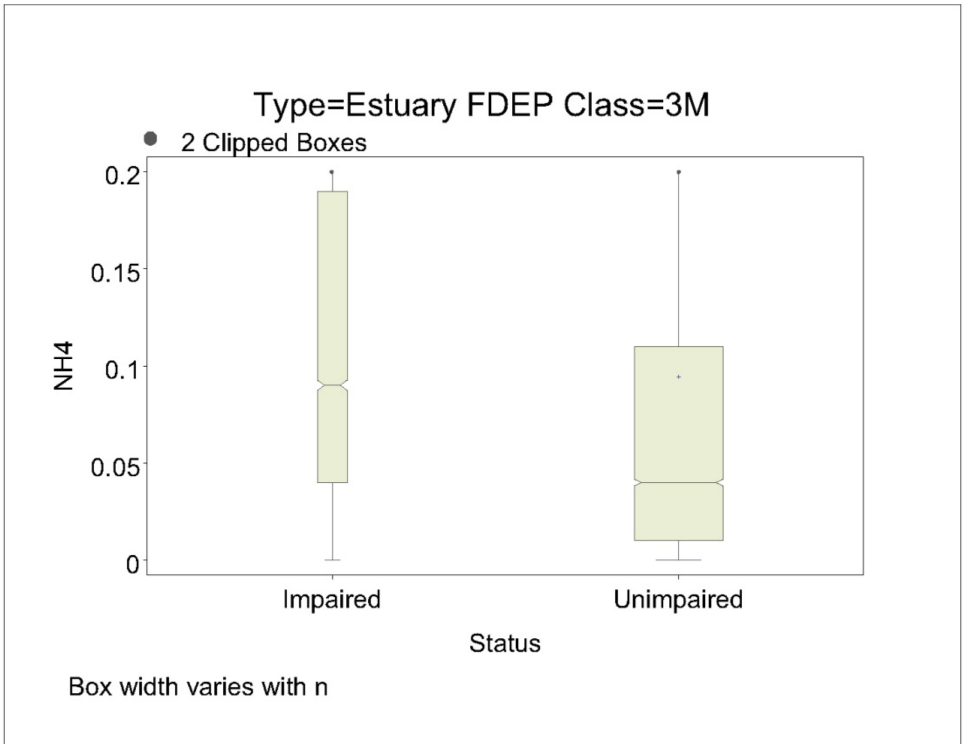
2 Clipped Boxes
1
0.75
0.5
0.25
0
NO3
Impaired
Unimpaired
Status
Box width varies with n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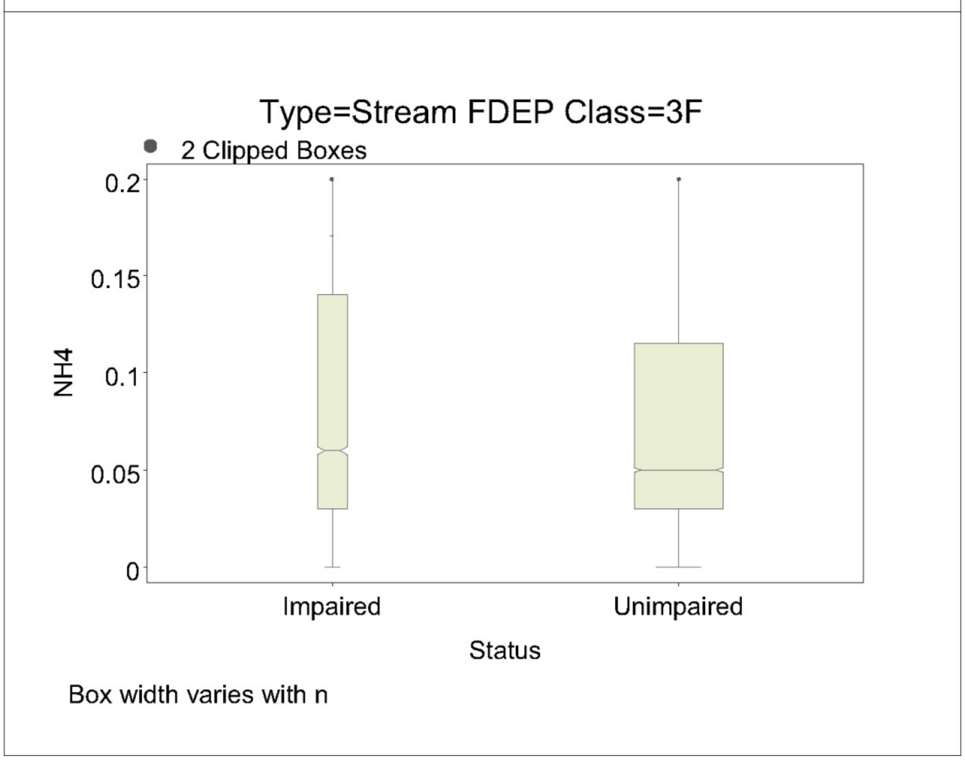
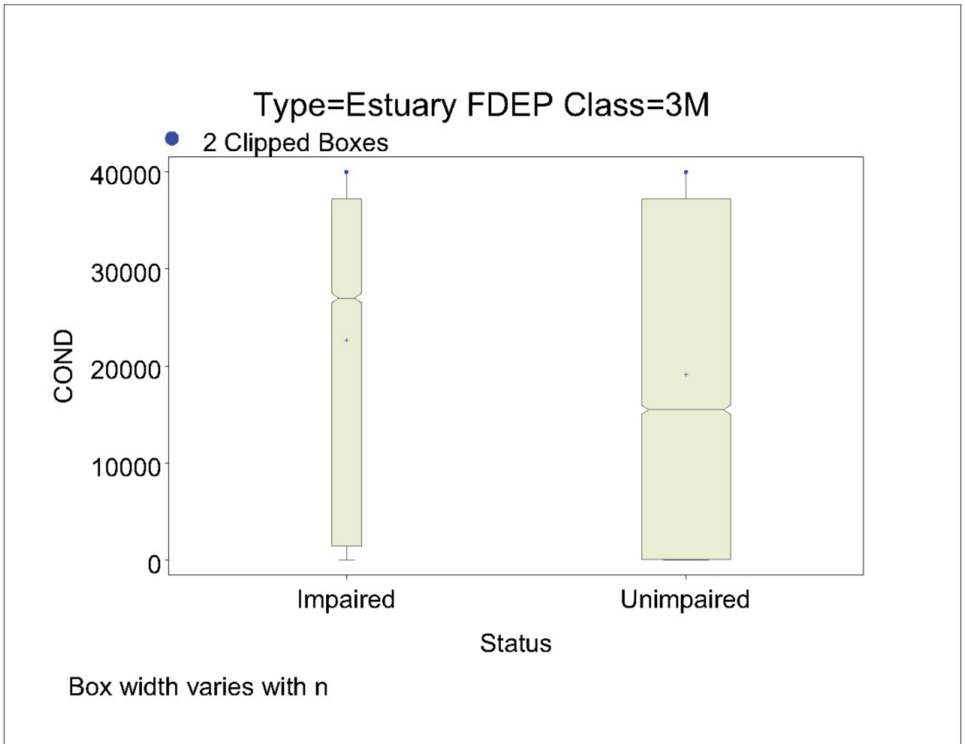
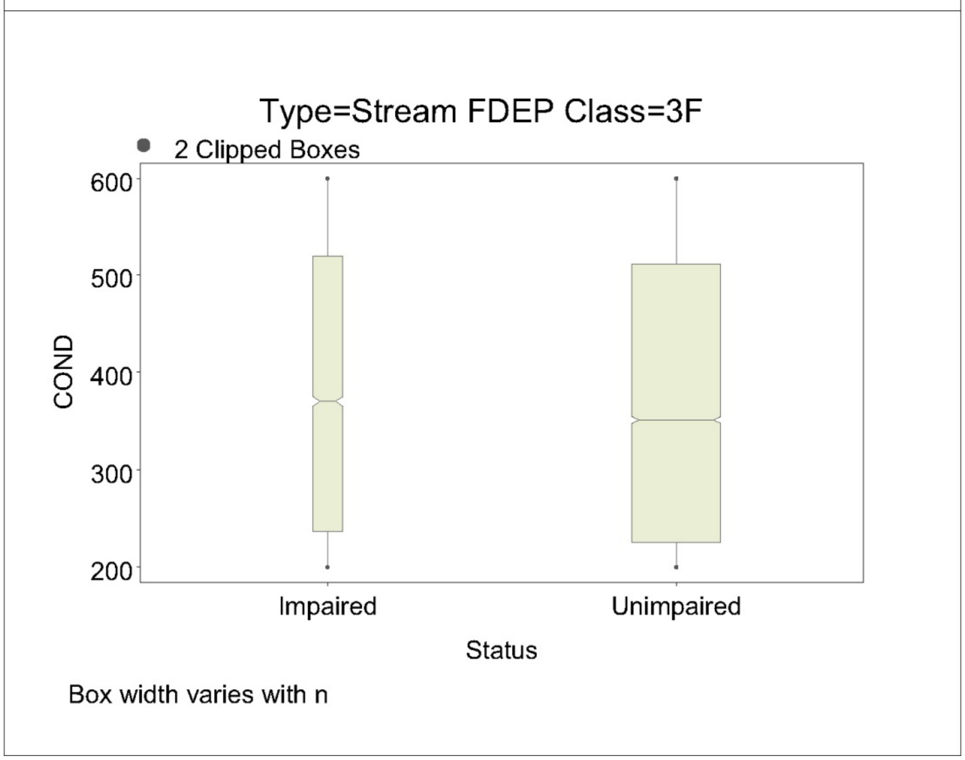
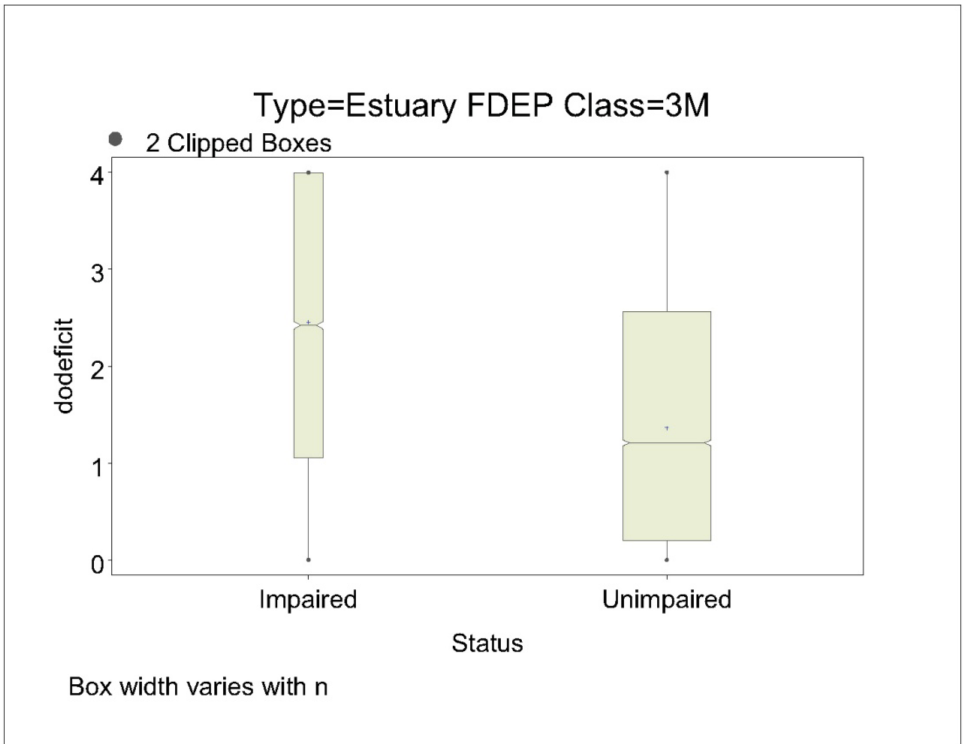
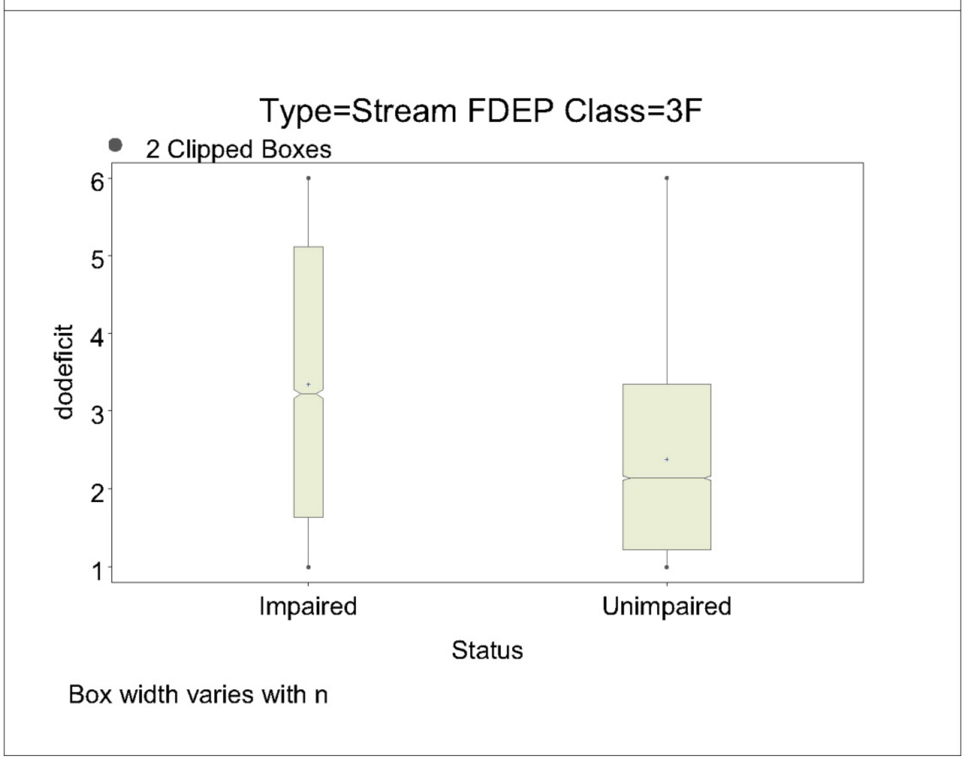
2 Clipped Boxes
0.2
0.15
0.1
0.05
0
NH4
Impaired
Unimpaired
Status
Box width varies with n


Type=Stream FDEP Class=3F
2 Clipped Boxes
0.2
0.15
0.1
0.05
0
NH4
Impaired
Unimpaired
Status
Box width varies with n


Type=Estuary FDEP Class=3M
2 Clipped Boxes
40000
30000
20000
10000
0
COND
Impaired
Unimpaired
Status
Box width varies with n


Type=Stream FDEP Class=3F
2 Clipped Boxes
600
500
400
300
200
COND
Impaired
Unimpaired
Status
Box width varies with n


Type=Estuary FDEP Class=3M
2 Clipped Boxes
dodeficit
4
3
2
1
0
Impaired
Unimpaired
Status
Box width varies with n


Type=Stream FDEP Class=3F
2 Clipped Boxes
dodeficit
6
5
4
3
2
1
Impaired
Unimpaired
Status
Box width varies with n