

Evaluation of the Bay of Quinte BUI 13 degradation of phytoplankton and zooplankton populations using a functional food web approach (prepared September 2017)

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EVALUATION OF THE BAY OF QUINTE BUI 13 DEGRADATION OF PHYTOPLANKTON AND
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(PREPARED SEPTEMBER 2017)

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Abstract

Currie, W.J.S., Bowen, K.L., Niblock, H.A., Fitzpatrick, M.A.J., Rozon, R., Munawar, M., Koops, M.A.. Proposed Targets for Evaluation of the Bay of Quinte BUI 13 Degradation of phytoplankton and zooplankton populations using a functional food web approach (prepared September 2017). Can. Manuscr. Rep. Fish. Aquat. Sci. 3263: x + 138 p.

The evaluation of Beneficial Use Impairment (BUI) 13 “*Degradation of phytoplankton and zooplankton populations*” has been problematic in that there has previously been no consensus on the approach to assessment. The DFO Great Lakes Laboratory for Fisheries and Aquatic Sciences applies a food-web function based approach using the 4 decades of plankton and fishes monitoring data in the Bay of Quinte using the set of 26 proposed metrics for phytoplankton, zooplankton populations, and trophic biomass ratios, which allows the assessment of this BUI for the Bay of Quinte Remedial Action Plan. While there are decreases to phosphorus concentrations over time we continue to observe very strong bottom-up effects controlling biomass of phytoplankton and fishes, but only weak top-down factors in the upper Bay of Quinte indicated by trophic ratios and path analysis. Impairments are indicated across phytoplankton, zooplankton and trophic ratio metrics strongly indicating BUI 13 remains impaired within the Bay of Quinte. Recommendations for a monitoring plan is provided.

Résumé

Currie, W.J.S., Bowen, K.L., Niblock, H.A., Fitzpatrick, M.A.J., Rozon, R., Munawar, M., Koops, M.A.. Proposed Targets for Evaluation of the Bay of Quinte BUI 13 Degradation of phytoplankton and zooplankton populations using a functional food web approach (prepared September 2017). Can. Manuscr. Rep. Fish. Aquat. Sci. 3263: x + 138 p.

L'évaluation de l'atteinte à l'utilisation bénéfique (AUB) 13 « Dégradation des populations de phytoplancton et de zooplancton » a été problématique en ce qu'il n'y avait pas eu auparavant de consensus sur l'approche d'évaluation. Le Laboratoire des Grands Lacs pour les pêches et les sciences aquatiques du MPO applique une approche basée sur la fonction du réseau trophique en utilisant les 4 décennies de données de surveillance du plancton et des poissons dans la baie de Quinte en utilisant l'ensemble de 26 mesures proposées pour le phytoplancton, les populations de zooplancton et les ratios de biomasse trophique. , ce qui permet l'évaluation de ce BUI pour le plan d'action correctif de la baie de Quinte. Bien qu'il y ait des diminutions des concentrations de phosphore au fil du temps, nous continuons d'observer de très forts effets ascendants contrôlant la biomasse du phytoplancton et des poissons, mais seulement de faibles facteurs descendants dans la partie supérieure de la baie de Quinte indiqués par les rapports trophiques et l'analyse des trajectoires. Des dégradations sont indiquées dans les mesures du phytoplancton, du zooplancton et du rapport trophique, ce qui indique fortement que la AUB 13 reste dégradée dans la baie de Quinte. Des recommandations pour un plan de surveillance sont fournies.

A food web assessment of BUI 13 degradation of phytoplankton and zooplankton populations for the Bay of Quinte.

Section I: Trophic Ratios and a Food Web Approach

The following section will include two approaches for the analysis of impairment for *Phytoplankton and Zooplankton Populations* in the Bay of Quinte as outlined in Currie et al. (2023). *Proposed targets for evaluation of the Bay of Quinte BUI 13 Degradation of phytoplankton and zooplankton populations using a functional food web approach.* The first method in Part I is an analysis of changes to trophic food web biomass ratios (Jeppesen et al. 1997a) over time and along the nutrient gradient from the upper eutrophic Bay of Quinte to the more oligotrophic lower bay (Fig. 1.1). Aquatic food webs tend to be more efficient when systems are increasingly nutrient limited (oligotrophic) so that ratios between successive levels in a food chain will increase (McCauley and Kalff 1981) hence they can be useful in tracking recovery of a eutrophic system. Part II will present the results of path analysis (Grace et al. 2010) that will show the trophic linkages between members of a simplified Bay of Quinte food web.

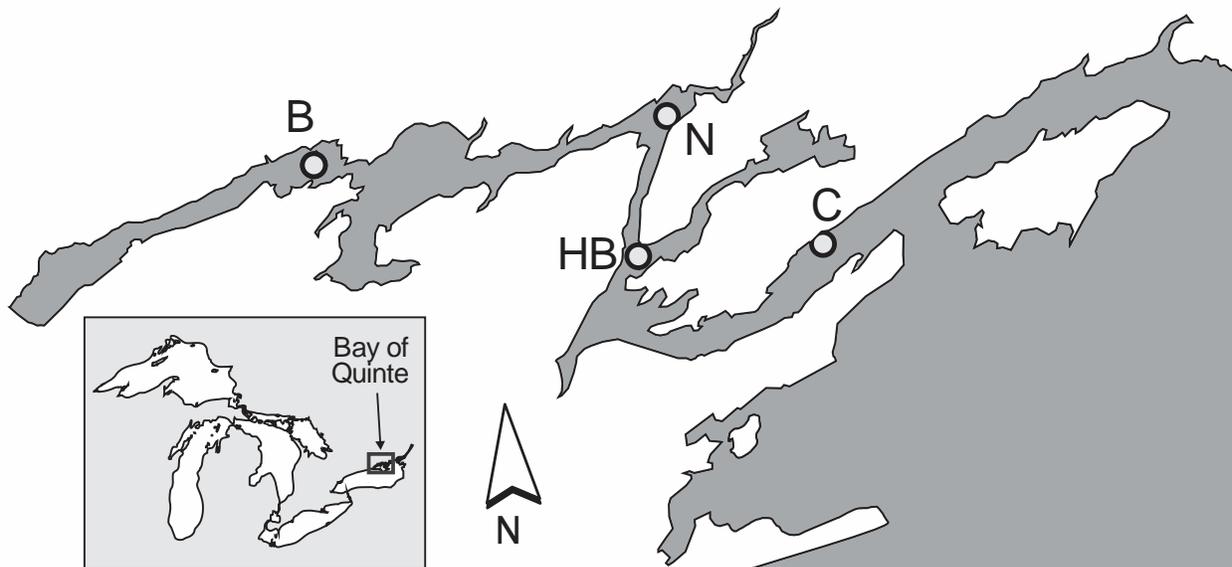


Figure 1.1: Map of sampling locations on the Bay of Quinte.

1.1 Trophic Ratios

Targets:

Demonstrate that the efficiency of energy and nutrient transfer through the food web from phytoplankton to zooplankton to fishes is consistent with a functional shallow freshwater ecosystem of similar trophic status, as demonstrated through the following suite of trophic ratios describing bottom-up and top-down food web processes:

1.	phytoplankton biomass to TKN	P:TKN
2.	zooplankton biomass to TKN	Z:TKN
3.	planktivorous fish biomass to TKN	PlankF:TKN
4.	zooplankton biomass to phytoplankton biomass	Z:P
5.	Z:P with TP and TKN	Z:P with TKN
6.	<i>Daphnia</i> biomass to phytoplankton biomass	Daph:P
7.	Z:P with planktivorous fish biomass	Z:P with PlankF
8.	planktivorous fish biomass to zooplankton biomass	PlanktF:Z
9.	planktivorous fish biomass to <i>Daphnia</i> biomass	PlanktF:Daph
10.	planktivorous fish biomass to phytoplankton biomass	PlanktF:P
11.	piscivorous fish biomass to zooplankton biomass	Pisc:Z
12.	piscivorous fish biomass to phytoplankton biomass	Pisc:P
13.	yellow perch to white perch biomass	YPerch:WPerch

Targets 1 – 6 are fundamentally bottom-up relationships while targets 7 – 13 are top-down.

1.2 Why is TP not used?

Although total phosphorous (TP) is generally viewed as the limiting nutrient in freshwater systems, when TP is present in high concentrations through sediment release (P enriched), nitrogen can become limiting in eutrophic systems (Elser et al. 2007, Conley et al. 2009). This is the case in the Bay of Quinte where Total Kjeldahl Nitrogen (TKN) is a significantly better predictor of total phytoplankton biomass (Table 1.1) (also found in Nicholls and Hurley 1989), and all the other phytoplankton groups (except Chrysophyceae). Therefore TKN will be used as the driver of production in most of the food web analysis, however TP relationships will still be referred to when comparing with previous research. Effectively this makes no difference since the intent of using TP in previous studies was that phosphorus was the limiting nutrient and strongest correlate to phytoplankton growth. See Section 3 for analysis of phytoplankton communities for discussion of nitrogen limitation during summer.

Table 1.1: ANOVA results of Total Phytoplankton Biomass by nutrient driver.

<i>Analysis of Variance</i>				
Source	DF	Sum of Squares	Mean Square	F Ratio
Model	2	1542011271	771005635	178.7296
Error	149	642757735	4313810.3	Prob > F
C. Total	151	2184769006		<.0001

Source	DF	Sum of Squares	F Ratio	Prob > F
TKN	1	122517830	28.4013	<.0001
TP	1	35910045	8.3244	0.0045

1.3 Time Stanzas

Shallow water systems are particularly prone to sudden switching between two stable states: a turbid phase dominated by phytoplankton, and a clear-water phase with increased macrophyte growth and reduced phytoplankton biomass (Scheffer et al. 1993, Scheffer and Carpenter 2003). The change between these phases is known as regime shifts or alternative stable states and can occur rapidly on time scales less than a year, driven by changes in nutrients, chemistry, climate or water levels. Transition from a turbid to clear phase occurred in the Bay of Quinte in 1994 – 1995 (Nicholls et al. 2011, Currie and Frank 2015). These phases were each split into two based on a discriminant analysis of the Bay of Quinte biogeochemistry data (primarily nutrients and water clarity measures; Currie and Frank 2015) to give 4 time-stanzas. These groupings will be used throughout this report.

Turbid Phase (1972 – 1994)

- 1) “**High P**” 1972 – 1982 was dominated by high nutrient levels and planktivorous fishes along with the transition to P-controls (high phytoplankton biomass)
- 2) “**Peak Walleye**” 1983 – 1994 was a stable period of climate, with intermediate nutrient levels which stabilized the system (high plankton, piscivorous fishes) and the start of submerged aquatic vegetation (SAV) regrowth

Clear Phase (1995 – 2014)

- 3) “**Clear Transition**” 1995 – 2000 began with a rapid transition to a clear-phase system, with higher clarity (deeper Secchi) and rapid expansion of macrophytes, most likely driven by the record cold winter of 1994
- 4) “**Climate Variability**” 2001 – 2015 shows much increased variability in precipitation and temperature and the expansion of zebra mussels and invasive predatory cladocerans in the upper bay

Outlier years will be identified where abiotic climatic conditions or biotic conditions were radically different. For instance, the year 2000 was an outlier year, with very high precipitation and a cool summer that led to recruitment failure in some of the zooplankton and fish species.

1.3 Bottom-Up and Top-Down Drivers: Regressions and Ratios

There are some strong underlying relationships for both bottom-up (TKN) and top-down (fishes) relationships in the Bay of Quinte. When the upper and middle bay annual data are pooled, regressions indicated that biomass was positively related to TKN concentration for phytoplankton, zooplankton and planktivorous fish biomass (Table 1.2, Fig. 1.2). Piscivorous fish biomass in contrast was unimodal with a peak biomass at ~0.6 – 0.7 mg/L TKN and reduced biomass at larger and smaller values. This corresponds to TP values of ~0.035 – 0.045 mg/L. Regressions for the lower bay are also significant, but the slopes are different than found in the upper and middle bay, and inclusion in the overall relationship reduces the fit.

Table 1.2: Top-down and bottom-up relationships for biomass and biomass ratios using least-square regression for the entire time period 1972 – 2014.

Relationship	Slope	Equation	r ² and p-value
P to TKN	+	-6255.19 + 21138.182*TKN	0.53, p<0.0001
Z to TKN	+	-7.74 + 338.64*TKN	0.18, p<0.0001
PlanktF to TKN	+	-49.65 + 139.69*TKN	0.56, p<0.0001
Z:P to TKN	-	0.064 - 0.0516*TKN	0.13, p=0.001*
Z:P to Dgalea	+	0.027 + 0.00022*Dgalea	0.48, p<0.0001
Log(PlankF:Z) to Dgalea	-	-0.54 - 0.0037*Dgalea	0.24, p<0.0001
Log(Pisc:PlankF) to Dgalea	+	-0.84 + 0.0041*Dgalea	0.13, p<0.001
Log(YPerch:WPerch) to TKN	-	2.21 - 3.78*TKN	0.39, p<0.0001
Log[Pisc] to TKN	na	0.50 + 0.74*TKN - 6.62*(TKN-0.6) ² **	0.26, p<0.0001

* slope for HB site more negative than B and N **quadratic fit.

The slope of the phytoplankton:TKN relationship (photosynthetic efficiency) is not significantly different among sites in the Bay of Quinte. This suggests that additional limitations beyond nutrients (e.g. light) are similar for all sites and top-down control on phytoplankton biomass is limited. The zooplankton to TKN relationship has much more variability (Z:TKN, r²=0.18) than found for phytoplankton (P:TKN, r²=0.53) and while the Z:TKN slopes are similar between the upper and middle bay sites, the lower bay has a much lower slope indicative of higher efficiency in this more oligotrophic site. Zooplankton biomass is driven by factors other than nutrients compared to phytoplankton or planktivorous fish biomass.

Step-wise regression was run for total phytoplankton (PhytoBM) biomass, zooplankton biomass (ZoopBM) and planktivorous fish biomass (PlankFishBM) to explore potential drivers (Table 1.3). TKN explained 57% of the PhytoBM (TP added 2% but was not a significant variable), with “other planktivore biomass” adding only 4% more, illustrating the strong bottom-up effect of nutrients on phytoplankton biomass. The explanation of ZoopBM was more complex, with

abiotic effects of Trent River discharge (May – Sept) and minimum air temperature (May – Aug) accounting for 60% of explained variance, with Alewife (not significant) adding less than 1%. The stepwise regression for log(PlankFishBM) indicated that food and habitat factors were important with 40% of the model explained by *Bosmina* sp. biomass, 13% by Secchi depth and 6% by TKN.

Most previous published relationships for trophic regressions used log-transformed data to improve fits and encompass a range of trophic conditions (McCauley and Kalff 1981, McQueen et al. 1986, Pace 1986, Peters 1986). To compare Quinte to these findings the relationship of biomass of the major biotic groupings to both TKN and TP was analyzed (Table 1.4). The slope of the Z:nutrient biomass slope is just over half of that for P:nutrient for both TKN and TP. The slope of the planktivorous fishes cannot be compared directly in this way because the fish biomass estimates, while consistent across years, cannot be converted into a volumetric value (e.g. mg m^{-3}) as occurs for phytoplankton and zooplankton.

Table 1.3: Stepwise regressions for total phytoplankton, zooplankton and planktivorous fish biomass of selected abiotic and biotic factors for all Bay of Quinte sites from 1972 – 2014.

Total Phytoplankton Biomass

Variable	Estimate	F-ratio	p>F
Intercept	-8079.6726	0.000	1
TKN	24695.8404	27.782	4.8e-6
Other_PlanktivoreBM	-2612.84	7.930	0.06279

Source	DF	Sum of Squares	Mean Square	F Ratio
Model	2	1402681678	701340839	159.6441
Error	142	623827462	4393151.1	Prob > F
C. Total	144	2026509140		<.0001*

RMSE = 2399.50, $r^2=0.606$, $r^2(\text{adjusted}) = 0.585$, BIC = 766.3

Total Zooplankton Biomass

Variable	Estimate	F-ratio	p>F
Intercept	806.732913	0.000	1
TrentDischargeMay-Sept	-0.1355602	23.255	0.00042
May-Aug_MeanMin_Temp	-41.192323	7.025	0.02116

Source	DF	Sum of Squares	Mean Square	F Ratio
Model	2	53375.53	26687.8	3.4805
Error	113	866463.81	7667.8	Prob > F
C. Total	115	919839.34		0.0341*

RMSE = 40.410, $r^2=0.661$, $r^2(\text{adjusted}) = 0.605$, BIC = 161.0

Planktivorous Fish Biomass

Variable	Estimate	F-ratio	p>F
Intercept	1.96582763	0.000	1
Bosmina BM	0.00287637	9.177	0.01146
Secchi	-0.242611	7.621	0.01854
TKN	-0.4083359	2.888	0.11732

Source	DF	Sum of Squares	Mean Square	F Ratio
Model	3	5.524363	1.84145	32.4066
Error	104	5.909633	0.05682	Prob > F
C. Total	107	11.433997		0.0341*

RMSE = 0.0476, $r^2=0.682$, $r^2(\text{adjusted}) = 0.595$, BIC = -39.9

Table 1.4: Log-log trophic biomass regressions for phytoplankton, zooplankton and planktivorous fish with TKN and TP ($\mu\text{g/L}^{-1}$) in the pooled upper and middle Bay of Quinte sites (Belleville, Napanee and HB) for comparison with previous published relationships.

Least fit equation	fit and prob
$\text{Log}[P] = -1.69 + 1.97*\text{Log}[\text{TKN } \mu\text{g/L}]$	$r^2=0.55$, $p<0.0001$
$\text{Log}[Z] = -0.86 + 1.12*\text{Log}[\text{TKN } \mu\text{g/L}]$	$r^2=0.18$, $p<0.0001$
$\text{Log}[\text{PlankFish}] = -4.89 + 2.29*\text{Log}[\text{TKN } \mu\text{g/L}]$	$r^2=0.57$, $p<0.0001$
$\text{Log}[P] = 2.10 + 1.06*\text{Log}[\text{TP } \mu\text{g/L}]$	$r^2=0.51$, $p<0.0001$
$\text{Log}[Z] = 1.37 + 0.56*\text{Log}[\text{TP } \mu\text{g/L}]$	$r^2=0.11$, $p=0.0021$
$\text{Log}[\text{PlankFish}] = -0.50 + 1.25*\text{Log}[\text{TP } \mu\text{g/L}]$	$r^2=0.47$, $p<0.0001$

Top-down relationships for the Bay are slightly weaker, but still very significant. In particular, relationships with the large herbivorous cladoceran *Daphnia galeata mendotae* are stronger than almost any other member of the food web, with almost half of the variance explained for the Z:P ratio (Fig 1.3). The Z:P ratio is often used as an indicator of planktivory in a system, but in Quinte, this ratio still has a large bottom-up component given the significant relationship to TKN.

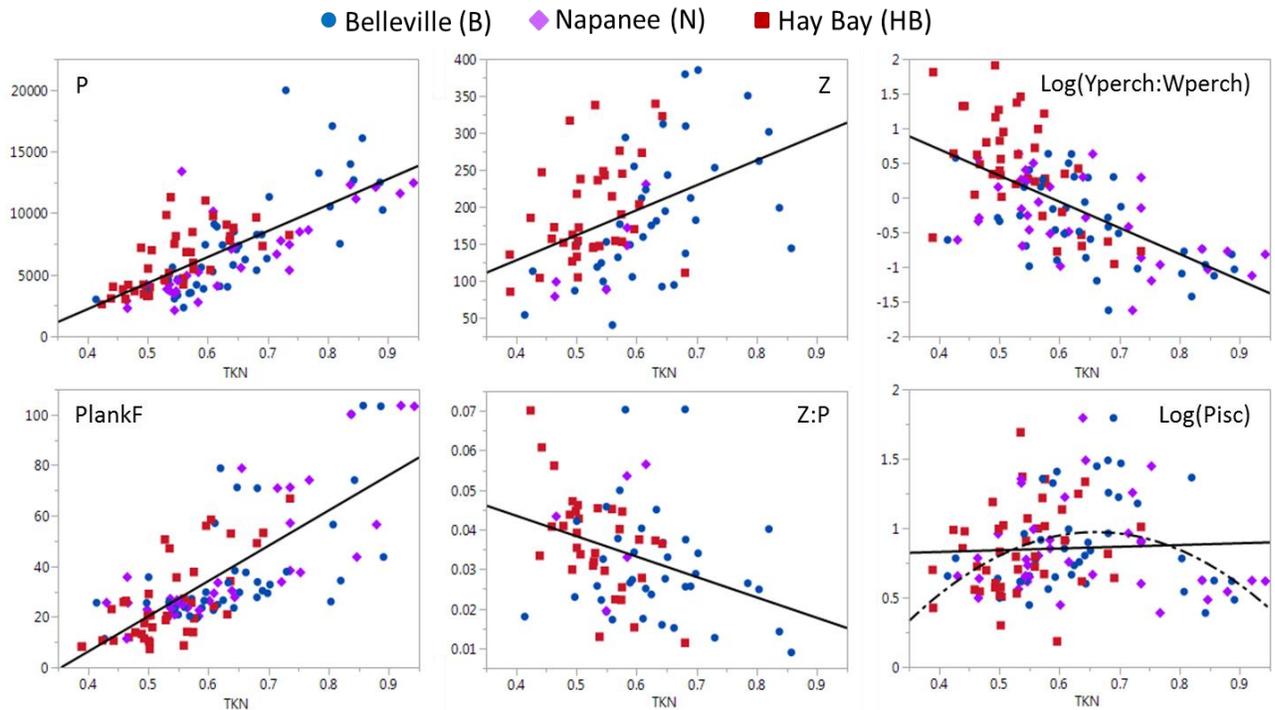


Figure 1.2: Scatter plots for the pooled upper and middle Bay of Quinte data used for regressions with TKN ($\text{mg}\cdot\text{L}^{-1}$) in Table 1.2. Solid black line is least squared fit, the dash-dot line is a quadratic fit for Log(Pisc).

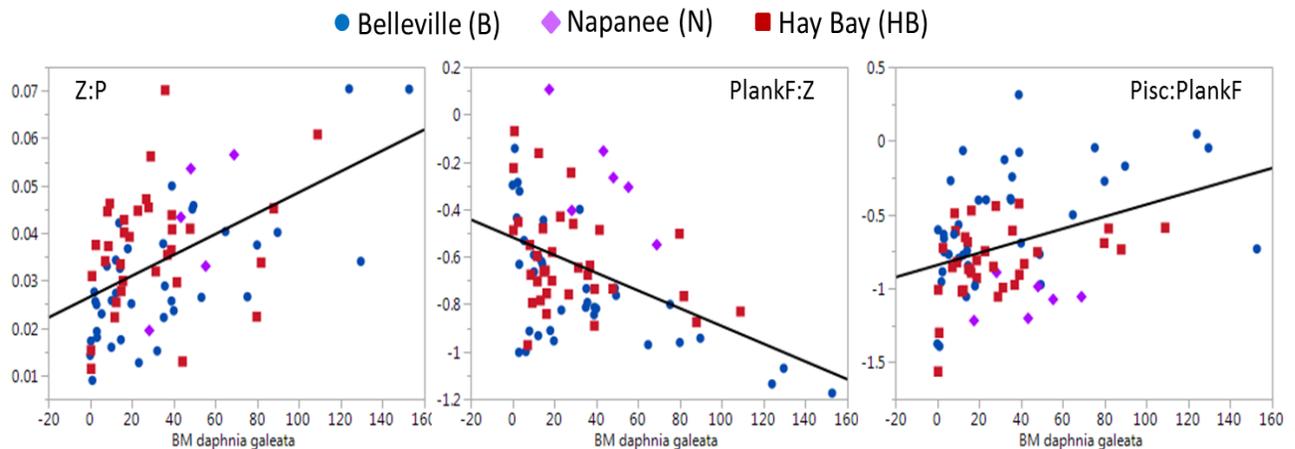


Figure 1.3: Scatter plots for the pooled upper and middle Bay of Quinte data used for regressions with *Daphnia galeata mendotae* in Table 1.2. Solid black line is least squared fit.

Trophic ratios for the Bay of Quinte over the phases and time stanzas are shown in Table 1.5. The Phytoplankton:TKN (P:TKN photosynthetic yield) relationship is apparent for both the upper and middle bay, and the ratio has decreased during the clear phase (1995 – 2014). The lower bay has exhibited a steadier change from stanza 1 to 4. This suggests there is less phytoplankton biomass per unit TKN over time. This reduced ratio is likely due to a decline in nitrogen availability, particularly in the upper bay, during late summer when peak phytoplankton biomass occurs. The ratio of N:P initially increased until the 1992, but then declined and has been steady since the onset of the clear phase (Currie and Frank 2015). The phytoplankton biomass peak is now reduced from the turbid phase but TKN has not reduced proportionally. There has been little change in this ratio since the beginning of the clear phase, but may decrease if summer phosphorus release from the organic sediment is reduced over time.

The zooplankton biomass relationship with TKN has declined in the upper bay over time, but only significantly after phosphorous controls, unlike phytoplankton which had a quick transition with the onset of the clear phase. There was no significant change at Hay Bay and only a small but significant gradual change at Conway. This concurs with the finding that zooplankton populations are resistant to change with nutrient status except at the extremes (Jeppesen et al. 1997a). This reinforces the use of the Z:P biomass ratio as a bottom up indicator in Quinte because phytoplankton biomass is very much tied to nutrients, but zooplankton biomass is not.

Total planktivorous fish biomass is more strongly associated with TKN and bottom-up drivers than zooplankton biomass. This ratio decreased at all sites indicating that as nutrients decrease, planktivorous fish biomass is increasingly reduced. This is particularly true for undesirable planktivores such as Alewife, White Perch and Gizzard Shad. Yellow perch tend to increase with nutrient concentration, so are in this respect, more like a predatory fish. This is not surprising in the Bay of Quinte where Yellow Perch are particularly large and likely to be more piscivorous. Piscivorous fish biomass in the upper and middle bays does not follow a linear relationship with nutrients, instead having a unimodal relationship peaking at ~ 0.65 mg/L TKN. Since Yellow Perch are favoured at lower TKN values, they tend to be a larger proportion of the piscivore biomass as nutrients decline.

Table 1.5: Trophic biomass ratios during the turbid and clear phases in the Bay of Quinte and across the four time stanzas. Significance using Student's t-test given by: * <0.05 , ** <0.01 , *** <0.001 . Stanzas not connected by the same letter are significantly different. When ratios are compared against another biomass factor, the mean values of the turbid and clear phases are given with the slope, intercept and r^2 of the least squares fit.

Ratio	Site	Turbid 72-94	Clear 95-14	Sig.	Stanza			
					1	2	3	4
Log(Phytoplankton:TKN)	B	4.10	3.88	***	A	A	B	B
	HB	4.10	3.91	***	A	A	B	B
	C	3.72	3.40	***	A	AB	BC	C
Log(Zooplankton:TKN)	B	2.51	2.35	**	A	AB	AB	B
	HB	2.58	2.52	-	A	A	B	B
	C	2.33	2.19	*	A	A	A	A
Log(PlanktivorousFish:TKN)	B	1.81	1.64	**	A	B	B	B
	HB	1.74	1.48	***	A	B	B	C
	C	2.08	1.35	***	A	B	C	C
Zooplankton:Phytoplankton	B	0.0285	0.0320	-	A	A	A	A
	HB	0.0306	0.0438	**	A	AB	BC	C
	C	0.0465	0.0669	*	A	AB	AB	B
<i>Daphnia</i> :Phytoplankton	B	0.0101	0.0110	-	A	A	A	A
	HB	0.0075	0.0151	***	A	AB	B	C
	C	0.0060	0.0156	***	A	A	B	B
Z:P with TKN [mean]/slope/intercept/(r^2)	B	[0.029]	[0.032]	-				
		-0.051	0.041	-				
		0.064 (0.08)	0.001 (0.03)	-				
HB	[0.031]	[0.044]	*					
	-0.071	-0.161	-					
	0.072 (0.10)	0.122 (0.26)*	-					
C	[0.047]	[0.067]	*					
	0.113	0.153	-					
	0.01 (0.01)	0.026 (0.01)	-					
Z:P with PlanktivorousFish [mean]/slope/intercept/(r^2)	B	[0.029]	[0.032]	-				
		-.0003	.0004	-				
		0.043 (0.29)*	0.022 (0.01)	-				
HB	[0.031]	[0.044]	*					
	-.0002	-.0003	-					
	0.037 (0.06)	0.049 (0.03)	-					
C	[0.047]	[0.067]	*					
	-.0002	-.0016	-					
	0.054 (0.02)	0.078 (0.03)	-					
Log(PlanktivorousFish:Z)	B	-0.72	-0.73	-	A	B	BC	C
	HB	-0.51	-0.69	**	A	B	B	B
	C	0.009	-0.21	**	A	AB	B	B
Log(PlanktivorousFish: <i>Daphnia</i>)	B	-0.15	-0.23	-	A	AB	BC	C
	HB	-0.18	-0.53	**	A	B	B	B
	C	0.69	-0.15	***	A	B	C	C

Table 1.5 continued

Log(PlanktivorousFish:Phyto)	B	-2.24	-2.29	-	A	A	AB	B
	HB	-2.35	-2.40	-	A	AB	B	B
	C	-1.65	-2.04	***	A	B	B	B
Log(PiscivorousFish:Z)	B	-1.24	-1.32	-	A	AB	B	B
	HB	-1.30	-1.50	*	A	AB	AB	B
	C	-0.47	-0.62	-	A	AB	AB	B
Log(PiscivorousFish:Phyto)	B	-2.85	-2.97	-	A	AB	B	C
	HB	-2.87	-2.90	-	A	A	AB	B
	C	-1.82	-2.07	-	A	A	A	B
Log(YPerch:WPerch) with TKN [mean]/slope/intercept/(r ²)	B	[-0.03] -2.83 1.41 (0.22)*	[-0.43] 0.64 -0.44 (0.01)	- -				
	HB	[0.095] -7.64 4.63 (0.46)*	[0.540] 1.01 0.32 (0.01)	- *				
	C	[2.15] -13.0 4.77 (0.34)*	[0.82] 1.18 1.79 (0.00)	* -				
Log(YPerch:WPerch) with Dgalea [mean]/slope/intercept/(r ²)	B	[-.678] -.0001 -0.675 (0.00)	[-.083] 0.0021 -0.152 (0.03)	*** -				
	HB	[0.305] .0140 0.053 (0.24)*	[0.821] 0.0029 0.721 (0.02)	- -				
	C	[0.785] 0.199 0.644 (0.07)	[2.059] 0.0297 1.930 (0.03)	- -				
Log(YPerch:WPerch)	B	-0.658	-0.083	**	A	B	BC	C
	HB	0.095	0.804	***	A	A	A	B
	C	0.508	2.056	***	A	A	B	C

The Z:P ratio did increase significantly with the transition from the turbid to clear phase in the middle and lower bay, but not in the upper bay (B). This is unexpected since clear water phases have higher Z:P ratios (Jeppesen et al. 1999). The relationship is similar and stronger for the Z:*Daphnia* ratio. If however, the Z:P is attempted to be explained by TKN or PlanktivorousFishes, there is no significant difference over time. The slope of the Z:P line in the upper bay changes from negative to positive during the clear phase, but no change is seen in the HB or C data and all relationships are not significant. This is in part due to the high degree of annual variability in Z:P within each site with CVs ranging from 30 – 50%. Even if individual pair-wise comparisons are run on Z:P with TKN or PlanktivorousFish, there is no change with stanza. The interaction effects for the ANOVA of Z:P with TKN or PlanktivorousFish is not large

indicating that other factors must be responsible for the variability of the Z:P ratio at all sites in Quinte. A step-wise regression of Z:P indicated that Trent River discharge, annual mean temperatures and May-Aug max temperatures and annual primary production accounted for more than 50% of the variation so abiotic factors clearly play a major role.

If the least square fit equations from Table 1.4 are used to predict the overall Z:P ratio for all sites within the Bay, the prediction line can be compared to the measured values found at each site (Fig. 1.4). The overall model fits the data reasonably well, but compared to the logarithmic fit of the measured site values [$Z:P = -0.023\ln(TP)+0.16$, $r^2=0.26$], overestimates the Z:P ratio. This is due mostly to the values at Belleville which have more than 73% of the points below the line, whereas HB has 50% below and C has 42%.

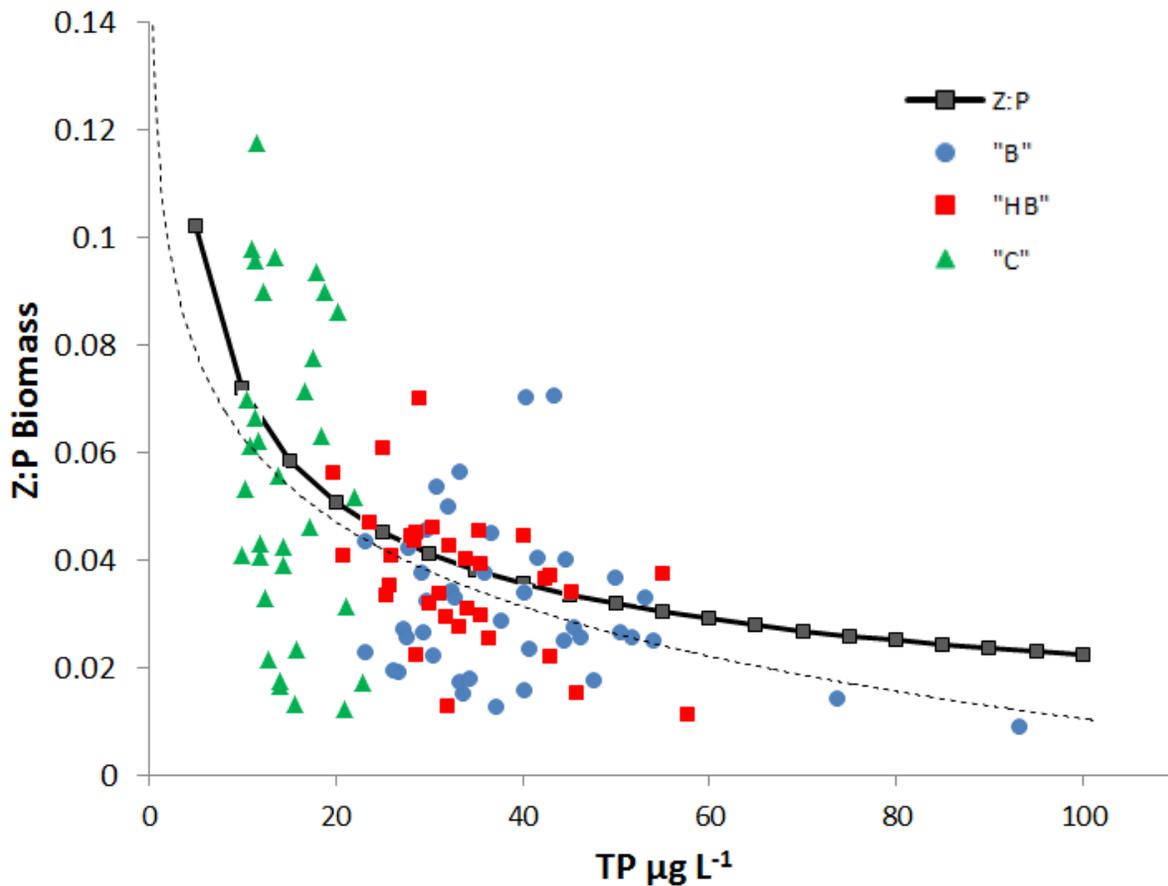


Figure 1.4: The Z:P ratio predicted by overall regressions to concentration of total phosphorus ($TP \mu\text{g L}^{-1}$) found in Table 1.4, which indicates increased trophic efficiency with reduced nutrient concentration. Measured Z:P values from B, HB and C are plotted and the logarithmic line of best fit is given by the dotted line.

Piscivorous fish ratios to zooplankton and phytoplankton are insignificant or weak with turbid vs clear phase, but show an increase after phosphorus controls. Little change is evident since 1983 even though both zooplankton and phytoplankton biomass have decreased steadily during this period. This is not entirely unexpected since piscivores often have a weak top-down effect and the Bay of Quinte has migratory piscivorous fish biomass from Lake Ontario. Biomass of predators in large lake systems can be low enough to have negligible top-down effects, for example, in Lake Washington, WA, the total limnetic fish population was estimated to consume only 1.7% of annual zooplankton production (Eggers et al. 1978).

Unlike many studies using trophic ratios where individual lakes are compared to each other, the Bay of Quinte is a system with an open boundary to Lake Ontario. This results in both interactions with the abiotic factors in Lake Ontario (e.g. nitrate, calcium etc.), but also with the abiotic deepwater transport of Lake Ontario species into the lower and middle bay, and migration of fishes that are non-resident to the Bay of Quinte (Bowlby and Hoyle 2011). This is particularly true for piscivore biomass which is dominated by Walleye in the upper bay, but also historically for White Perch and Alewife (Ridgway et al. 1990).

There is clearly a top-down effect on the Quinte food-web, though the linkage to predators is weak. To investigate this further, the YPerch:WPerch ratio was used to examine differential trends in Yellow Perch (increasing over time) to White Perch (decline after phosphorus controls then steady). Yellow Perch larval stages and juveniles are known to select larger zooplankton (MacDougall et al. 2001). White Perch diet is similar, but consumes more zooplankton by percentage and more per individual while Yellow Perch consume more benthos and fishes (Parrish and Margraf 1990, Parrish and Margraf 1994, MacDougall et al. 2001). As Yellow Perch increase in size they become piscivorous and Bay of Quinte Yellow Perch are known to be large (Hoyle et al. 2012).

Yellow Perch and White Perch biomass are negatively correlated which is not unexpected since their diets often overlap (Parrish and Margraf 1994). The YPerch:WPerch ratio showed relationships with both TKN and *Daphnia galeata mendotae*, a large crustacean grazer known to be highly selected by planktivores (Jeppesen et al. 2003, Mehner 2008, Friederichs et al. 2011). This ratio then has the convenience of providing used both a top-down and bottom-up indicator, but since planktivorous fishes increase in eutrophic conditions, the metric is unidirectional. There is still a great deal of annual variation in the metric, but all areas of the Bay

of Quinte show an increase in the ratio with the lower bay showing the highest magnitude (and steady increase over time), though Hay Bay lagged in this metric until the most recent stanza. This could be due to the mixture of fish community of both shallow water upper bay fishes and deeper water Lake Ontario fishes. White Perch biomass in the upper bay was reduced during the start of the clear phase but has increased in the most recent stanza to levels near the turbid phase. This pattern is not seen in the middle and lower bay where White Perch biomass has been consistently less than Yellow Perch biomass since the end of phosphorus controls in 1978.

2.1 Path Analysis

Path analysis, also known as Structural Equation Modeling (SEM) (Arhonditsis et al. 2006, Grace et al. 2010) is a method to identify potential causal linkages between factors in a constructed model. In this case, we produced a very simplified Bay of Quinte pelagic food web model to investigate positive or negative associations between groups in the food web. It is easiest to think of path analysis as an exploratory tool that uses correlation or regression analysis to partition the variance between boxes within the model. These paths can be unidirectional or bidirectional. In a food web model, the path can be assumed to be unidirectional if there is an expectation of only uptake from one trophic level to the next, but food web interactions are often bidirectional, so to be consistent, correlations between the boxes was used. This model will be used to identify changes that occurred in the trophic ratios or species composition when the Bay of Quinte transitioned from a turbid (1972 – 1994) to clear (1995 – 2014) phase.

Since the lower bay (Conway) has a very different species composition from the upper and middle bays (B, N, HB), especially in the fish community, it has been excluded from the path analysis. The boxes in the path analysis can be comprised of constructed variables which aggregate measures together into functional groups or drivers. Rather than impose this on the model, we chose instead to use only measured biomass values in the model and determined the linkages between them. No abiotic effects were added to the model other than TKN.

To simplify the model, some highly generalized groupings were applied (Fig. 1.5, 1.6). Since there was no clear effect of phytoplankton composition during the analysis of trophic ratios, only one class of total phytoplankton was used (Tphyto). Macrophyte densities (SAV/km²) as coverage (km²) were only available for the upper bay but were expected to be a factor in the

transition to the clear phase, so this metric was applied to the upper and middle bays. Since a size effect on zooplankton is expected (Jeppesen et al. 1997b, Bertolo et al. 2005, Mehner et al. 2008), particularly from compositional changes from *Daphnia* to smaller taxa such as *Bosmina*, zooplankton were grouped into large (ZoopLG) and small (ZoopSM). The large grouping consisted of all *Daphnia* species, predatory cladocerans (e.g. *Leptodora*, *Cercopagis* etc.), and large predatory copepods (e.g. *Mesocyclops*). All other zooplankton were grouped in the small category. Planktivorous fishes were limited to the four main species which account for most of the biomass: Alewife, Gizzard Shad (GizzSh), White Perch (Wperch), and Yellow Perch (Yperch). White Perch and Yellow Perch were added because there is some expectation of resource competition between them. Alewife and Gizzard Shad were added because they were both a major component of the total fish biomass during the 1970s in Quinte and because high biomasses of these fishes are not a desirable fish community endpoint. Because information on benthic biomass was not available consistently for the whole dataset, benthivorous fishes were also excluded even though it has been shown that the YOY of these fishes can be a major component of planktivory in shallow water systems. All potential piscivores were grouped together. No data were available on YOY biomass in the Bay of Quinte so this is an additional source of uncertainty in the model.

All mean biomass values decreased from the turbid to the clear phase (Table 1.6), except SAV (macrophytes) (+84%) and Yellow Perch (+47%). TKN changed the least (-23%) while Gizzard Shad (-89%) and Alewife (-91%) changed the most. The covariance values almost all increased in value. The interactions with strongly negative covariances during the turbid phase were the ones expected to have strong grazing interactions: ZoopLG <-> Tphyto, GizzSh <-> ZoopSM, Alewife <-> ZoopSM, Wperch <-> ZoopSM, Alewife <-> ZoopLG, GizzSh <-> ZoopLG, Wperch <-> ZoopLG. These all have much reduced negative interactions (near zero) during the clear phase. Reduced top-down effects from piscivores on planktivores were less than from planktivores on zooplankton.

During the clear phase, the paths of primary production significantly shifted to macrophytes, though the path through phytoplankton is still dominant. Large zooplankton decreased in biomass but relative to small zooplankton the paths are stronger. White Perch are now the dominant planktivore, and a more important link to piscivores with Alewife and Gizzard Shad biomass at extreme lows. Yellow Perch are increasingly similar to piscivores in their linkages and increase in biomass during the clear phase.

1972-1994

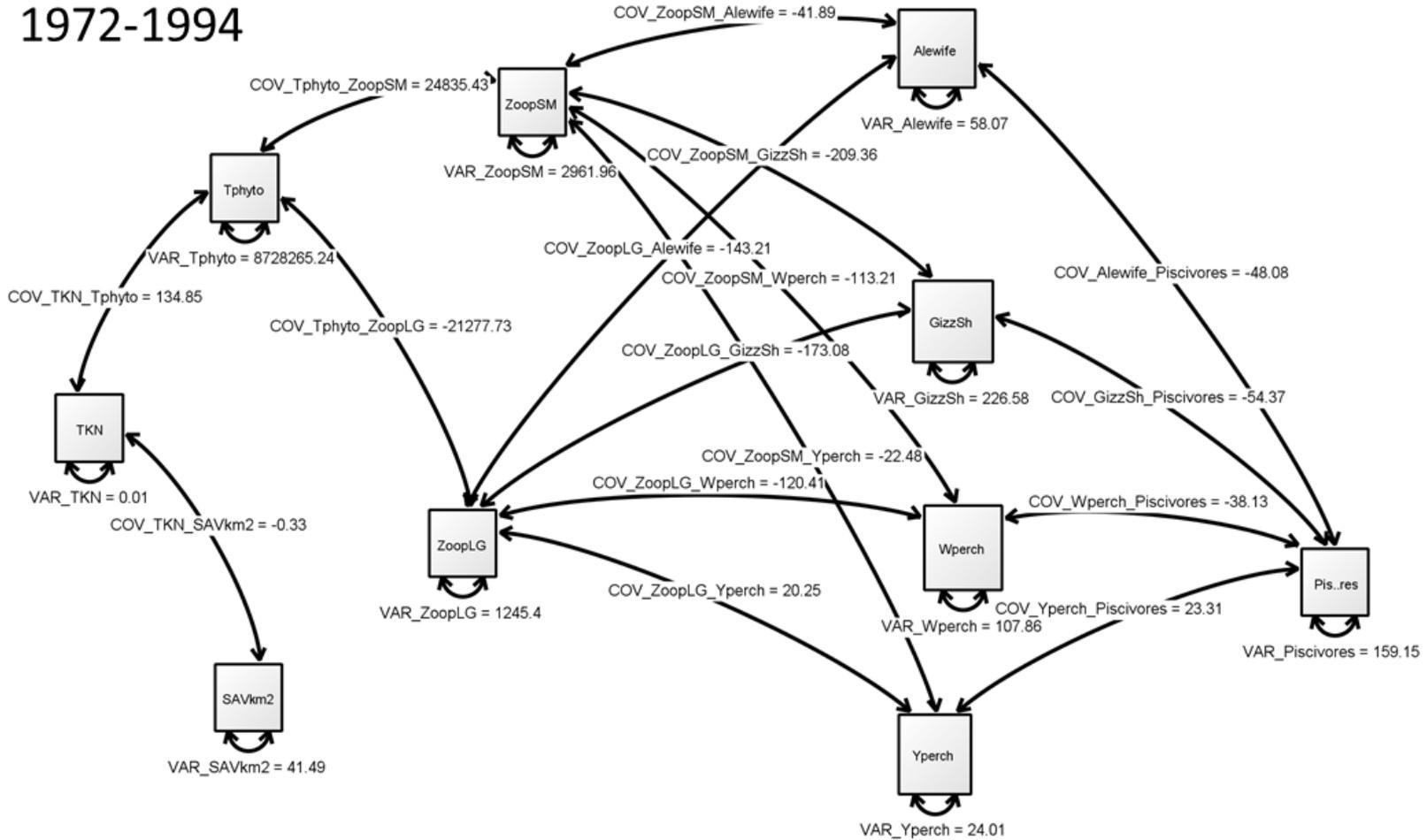


Figure 1.5: Path analysis for the annual upper and middle Bay of Quinte pelagic food web biomasses from the turbid phase (1972 – 1994). Negative linkages suggest ecological interactions such as competition or predation (higher predators predict lower prey biomass), while positive linkages suggest facilitation with intensified values indicating a stronger connection. Model $\chi^2=130.19$, $df=29$, $p<0.001$, $RMSEA=0.237$, $CFI=0.39$). Detailed covariance values can be found in Table 1.6.

1995-2014

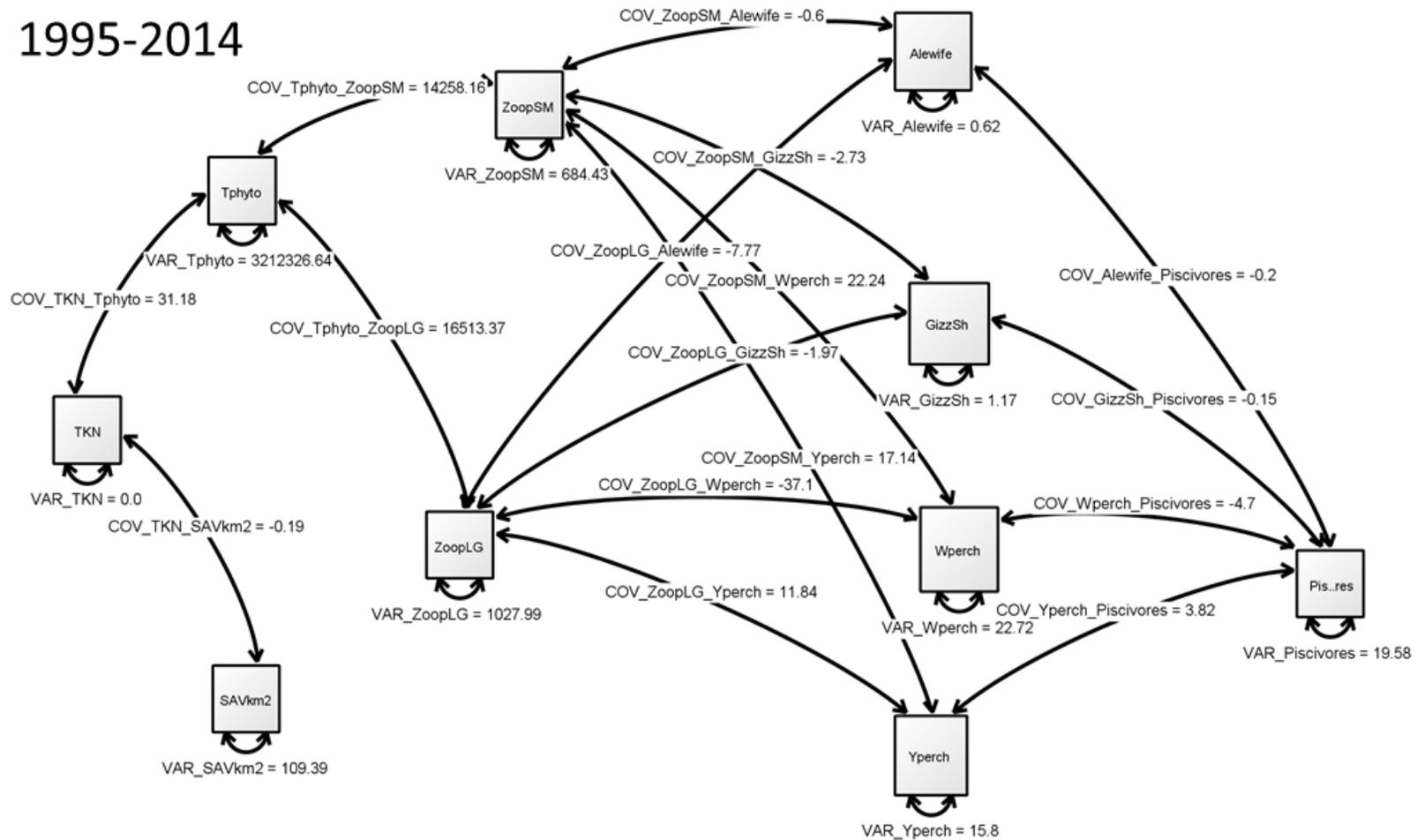


Figure 1.6: Path analysis for the upper and middle Bay of Quinte pelagic food web from the turbid phase (1972 – 1994). Negative linkages suggest ecological interactions such as competition or predation (higher predators predict lower prey biomass), while positive linkages suggest facilitation with intensified values indicating a stronger connection. Model $X^2=201.32$, $df=29$, $p<0.001$, $RMSEA=0.237$, $CFI=0.00$). Detailed values can be found in Table 1.6.

Table 1.6: Variance and covariance estimates, standard errors, mean biomass and percent difference in mean biomass between the turbid and clear phases from the path analysis of the simplified Bay of Quinte food web illustrated in Figure 1.4 and 1.5.

PATH	Turbid Phase			Clear Phase			%Diff
	Covariance	SE	MeanBM	Covariance	SE	MeanBM	
TKN <-> TKN	0.00602	1.44	0.687	0.00199	1.44	0.522	-24.0
Tphyto <-> Tphyto	8728265.2	1.44	8869	3212326.6	1.55	4295	-51.6
SAVkm2 <-> SAVkm2	41.49	1.82	34.66	109.39	1.75	63.77	84.0
ZoopSM <-> ZoopSM	2962.0	1.90	165.56	684.43	1.65	94.88	-42.7
ZoopLG <-> ZoopLG	1245.4	1.90	70.50	1027.99	1.65	52.91	-24.9
Alewife <-> Alewife	58.07	1.47	8.22	0.6249	1.44	0.91	-88.8
GizzSh <-> GizzSh	226.58	1.47	14.95	1.1715	1.44	1.28	-91.4
Wperch <-> Wperch	107.86	1.47	18.15	22.720	1.44	8.87	-51.1
Yperch <-> Yperch	24.01	1.47	6.66	15.805	1.44	9.76	46.6
Piscivore <-> Piscivore	159.15	1.44	13.58	19.577	1.44	6.59	-51.5
Tphyto <-> TKN	134.85	1.02		31.18	1.09		
SAVkm2 <-> TKN	-0.333	1.29		-0.1889	1.23		
ZoopLG <-> Tphyto	-21277.7	1.34		16513.4	1.23		
ZoopSM <-> Tphyto	24835.4	1.34		14258.2	1.23		
Alewife <-> ZoopSM	-41.89	1.40		-0.596	1.16		
GizzSh <-> ZoopSM	-209.36	1.40		-2.726	1.16		
Wperch <-> ZoopSM	-113.21	1.40		22.24	1.16		
Yperch <-> ZoopSM	-22.48	1.40		17.14	1.16		
Piscivore <-> Alewife	-48.08	1.04		-0.2004	1.02		
Piscivore <-> GizzSh	-54.37	1.04		-0.1533	1.02		
Piscivore <-> Wperch	-38.13	1.04		-4.7013	1.02		
Piscivore <-> Yperch	23.31	1.04		3.8240	1.02		
Alewife <-> ZoopLG	-143.21	1.40		-7.7689	1.16		
GizzSh <-> ZoopLG	-173.08	1.40		-1.9723	1.16		
Wperch <-> ZoopLG	-120.41	1.40		-37.101	1.16		
Yperch <-> ZoopLG	20.25	1.40		11.842	1.16		

Table 1.7: Summary results of trophic metrics for Bay of Quinte Sites, upper bay (B), middle bay (HB) and lower bay (C). Number of arrows ↓ indicates strength ($p < 0.05$, 0.01, 0.001, or not significant) and direction. Checkmark indicates metric meets criteria for upper bay.

Trophic Metric	Site			Expected Trend		Comment
	B	HB	C			
P:TKN	↓↓↓	↓↓↓	↓↓↓	↓	✓	All sites
Z:TKN	↓↓	ns	↓	↑ or ns	✗	B and C only
PlankF:TKN	↓↓	↓↓↓	↓↓↓	↓	✓	All sites, B weakest
Z:P	ns	↑↑	↑	↑	✗	All sites except B
Daph:P	ns	↑↑↑	↑↑↑	↑	✗	All sites except B
PlankF:Z	ns	↓↓	↓↓	↓	✗	All sites except B
PlankF:Daph	ns	↓↓	↓↓↓	↓	✗	All sites except B
PlankF:P	ns	ns	↓↓	↓	✗	Not upper middle bay
Pisc:Z	ns	ns	↓	↓	✗	Not upper middle bay
Pisc:P	ns	ns	ns	↓	-	Weak top-down
YPerch:WPerch	↑↑	↑↑↑	↑↑↑	↑	✓	All sites, B weakest
Z:P with TKN	+	-	+	-	ns	Trend incorrect
Z:P with PlankF				-	ns	All slopes ~ 0
YPerch:WPerch with TKN	+	+	+	+	ns	Trend correct
YPerch:WPerch with D.gal		-	-	-	ns	Trend correct HB, C

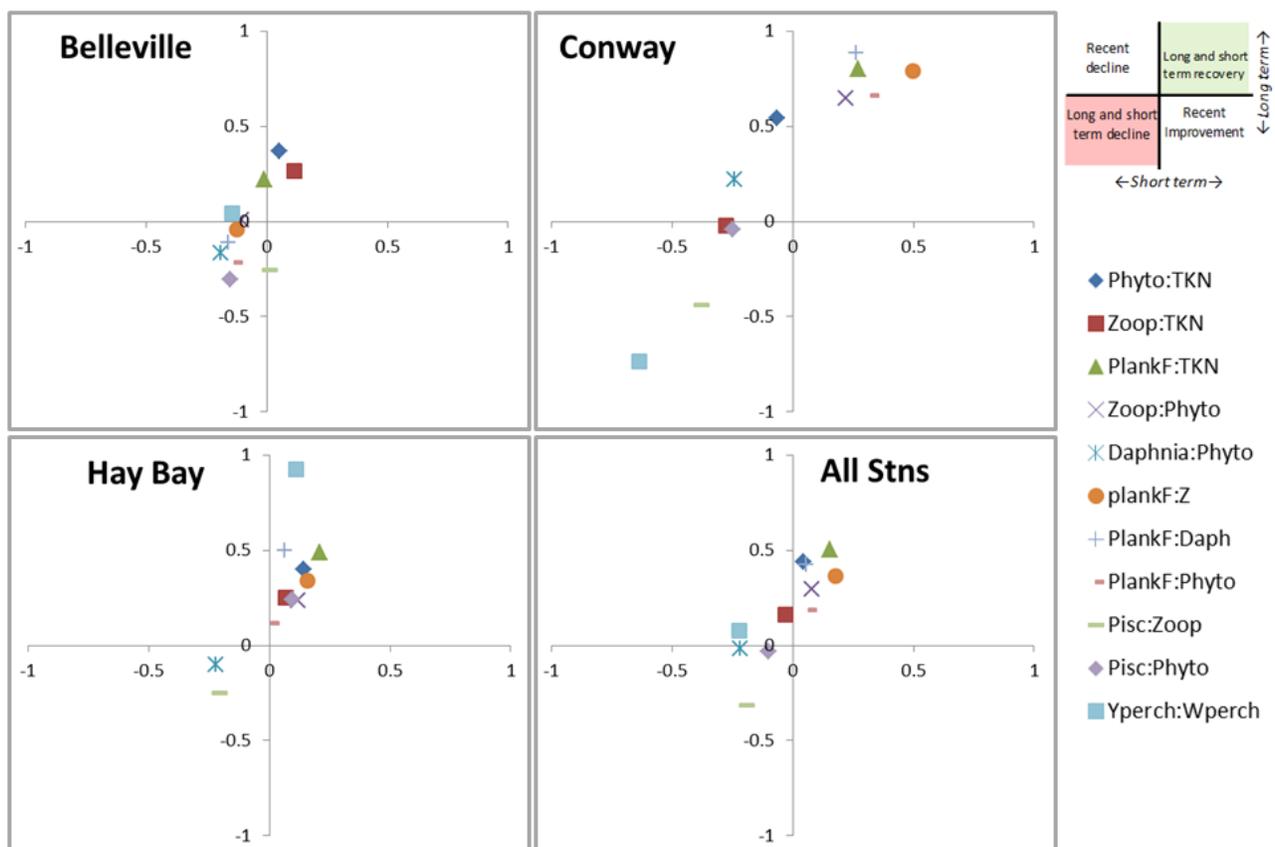


Figure 1.7: Ordination of food web metrics showing changes in the short term (x-axis, last 5 years relative to stanza 4 mean) and long term (y-axis, 5 year mean relative to overall mean).

3.1 Discussion

Multivariate techniques are particularly applicable to the study of aquatic food webs and their drivers (Gannon and Stemberger 1978, Peters and Downing 1984, Nicholls and Hurley 1989, Bertolo et al. 2005, Gebrehiwot et al. 2017) and have been key to the fundamental papers that form the basis of the use of trophic linkages in aquatic ecology (McCauley and Kalff 1981, McQueen et al. 1986, Pace 1986, Peters 1986, Jeppesen et al. 2003). Systems with divergent trophic status (a nutrient gradient) have been shown to have very different relationships between standing stocks of biomass (McCauley and Kalff 1981, Jeppesen et al. 2005). Oligotrophic systems are expected to be more efficient in their use of nutrients, and the ratios between successive members of the food-web tend to be higher (Jeppesen et al. 1997a, Havens 2014). Oligotrophic systems also are known for more efficient nutrient recycling through the microbial food web and tend to have increased numbers of linkages within a food web compared to eutrophic systems (Munawar et al. 2011, Havens 2014). The more direct the trophic relationships are (more “food chain” rather than “food web”), the better the ratio will respond to changes in the system (Drenner and Hambright 2002) and the number of levels (odd vs. even) is expected to change the expectations of ultimate community structure (Mazumder 1994). Regardless, a functional food web is expected to transfer energy from primary producers up to higher trophic levels in ratios that are highly influenced by abiotic factors (precipitation, temperature, mixing etc.) and the physical structure of the ecosystem, but should be comparable to similar systems.

A particular characteristic of shallow water systems is the potential for sudden changes from turbid to clear alternative stable states (Scheffer et al. 1993, Jeppesen et al. 1999, Muylaert et al. 2003). As the name suggests, the turbid phase is dominated by high phytoplankton biomass and reduced submerged macrophytes, while the clear phase has much reduced phytoplankton, and more macrophytes as the light penetrates to deeper depths. A change to clear phase tends to be associated with reduced benthivorous and planktivorous fishes and often increased zooplankton biomass. This change is not necessarily controlled by nutrients and can occur over a wide range of nutrient conditions. Thus, it is possible to have a eutrophic clear phase system or an oligotrophic turbid phase system (see Muylaert et al. 2003), so shallow water systems can cross boundaries of relationships that are not necessarily classified by their trophic status.

It is often difficult to detect differences in plankton community structure, especially at one location over time. Plankton in particular can be quite resilient in that one species may dominate on any given year, and the dominant species may change across years (Steele and Frost 1977), while the total biomass is unaffected (Jeppesen et al 1997a, Bertolo et al. 2005). Sampling a spatially and temporally patchy community structure is expected to produce high variability and reduced predictive power (Steele 1978, Levin 1992, Folt and Burns 1999). Functional or size classification systems (Sprules and Holtby 1979, Gebrehiwot et al. 2017) are often more efficient than taxonomic ones in cases such as these (e.g. herbivores, predators) since species guilds can be comprised of species which have highly overlapped niches and plankton are never truly at equilibrium. This forms the basis of one of the most famous papers in limnology “The Paradox of the Plankton” (Hutchinson 1961) which uses this rationale to explain why so many species can coexist against competition.

The path analysis is a useful tool for exploring relationships between factors, in this case the biomass groupings in the simplified Bay of Quinte pelagic model. The analysis is clear that the top-down piscivory and planktivory effects are certainly reduced during the clear phase and that bottom-up drivers dominate which has been shown in other modeling efforts in eutrophic systems (Kinter and Ludsin 2012). There is a very strong change for the upper and middle bay sites from a turbid phase dominated by Alewife and Gizzard Shad and high biomass of small sized zooplankton, to a clear phase with Yellow Perch and White Perch and a more equal grouping of small and larger zooplankton (particularly *Daphnia* sp.). This matches some of the expectations that total biomass of zooplankton would be less affected, but the size composition would be more altered by reductions in planktivorous fish biomass (Jeppesen et al. 1997b, Hessen et al. 2006, Mehner et al. 2008). Measures of planktivory have changed over time, given by the increase in covariance from very strong negative values (negative association, so an increase in one led to reduction in the other) in the turbid phase to near zero in the clear phase. Interestingly, the large zooplankton group showed changes in covariance from highly negative to much lower values with Alewife, Gizzard Shad and White Perch, but their overall biomass was reduced much less than observed for small-sized zooplankton. The large zooplankton to TKN covariance changed from strongly negative in the turbid phase to positive in the clear phase while small zooplankton covariance became less positive. This indicates that *Daphnia* were likely more efficient at converting the reduced phytoplankton biomass than smaller zooplankton (Shapiro and Wright 1984, Vanni 1986). The phytoplankton to TKN covariance was significantly lower in the clear phase and the macrophyte to TKN covariance

was less negative illustrating the change to increased macrophytes during the clear phase. One unexpected change was the increase in Yellow Perch biomass (the only positive change other than macrophytes), with covariance with small zooplankton changing from weakly negative to weakly positive during the clear phase. The covariance with large zooplankton had little change with weak positive values. This suggests that Yellow Perch are not very strong planktivores, and might be more piscivorous or benthivorous, which may be due to their large size in the Bay of Quinte.

In spite of the natural variability in plankton systems, we have determined some of the underlying relationships in the Bay of Quinte. The bottom-up drivers (nutrients, light) continue to be dominant at all sites in the Bay of Quinte. This is not surprising since eutrophication has been identified as a major issue in this Area of Concern. Unlike Nicholls and Hurley (1989) who found no relationship between total phytoplankton biomass and TP unless all stations were pooled, in adding more years to the analysis, there has been sufficient range of nutrients to give a significant regression for all sites. We also found that TKN was a far better predictor of phytoplankton biomass than TP. The underlying regressions were pooled for B, N and HB since their slopes were not significantly different from each other. This confirms the observations in Currie and Frank (2015) that HB was not different from B for nutrients, seston or chlorophyll-a values in the clear phase. Photosynthetic yield, given by the P:TKN ratio related strongly only with turbid vs. clear phase in the upper and middle bays, and did increase with depth. This differs from studies on oligotrophic lakes where the P:nutrient ratio did not change with nutrient status (Bertolo et al. 2005, Hessen et al. 2006, Friederichs et al. 2011). Nicholls and Dillon (1978) discuss the factors which can lead to the wide range of Chl:TP relationships, but our values for phytoplankton biomass to TP fall within their overall model. The P:TKN ratio is higher in the lower bay as expected given its oligo-mesotrophic status (Jeppesen et al. 2003), but is not different between the upper and middle bay sites.

The regression of zooplankton to TKN, while significant, has a much reduced r^2 compared to phytoplankton. This is not surprising given that the step-wise regression found that the abiotic factors of May-Aug Trent River discharge and May-Aug minimum temperatures explained most of the variability of zooplankton biomass. That being said, as in other oligotrophic-eutrophic lakes (Nicholls and Dillon 1978, McCauley and Kalff 1981, Pace 1986), the slope of the zooplankton biomass to nutrient regression relationship is only about half of the value found for phytoplankton.

The relationship between zooplankton to phytoplankton biomass has been extensively studied in a wide range of systems from ultra-oligotrophic large lakes to shallow water hyper-eutrophic wetlands. The ratio of Z:P decreases with increases in total phosphorus (Cryer et al. 1986, Sager and Richman 1991, Jeppesen et al. 1997a, Jeppesen et al. 2003), planktivorous fishes (Jeppesen et al. 2003, Havens and Beaver 2013), depth (Jeppesen et al. 1997a, Jeppesen et al. 2003) and clearwater phase (Jeppesen et al. 1999, Muylaert et al. 2003). The Z:P ratios, when phytoplankton biomass is converted from wet-weight to dry-weight using a 0.15 factor (Winberg 1971), are closest to those found for eutrophic systems (0.1 – 0.2) (Jeppesen et al. 2003), except for Conway which transitioned into the oligotrophic range (0.35) after the onset of the clear phase. Z:P values for coastal Lake Ontario are about 0.4, roughly twice that found in the upper bay. The Z:P ratio increased at HB and C between turbid and clearwater phases as expected, but did not in the upper bay. Depth does play a part because zooplankton are expected to be more exposed to predation in shallow depths since they are concentrated in smaller volumes for planktivores to search. However, the effect of depth on Z:P is expected to be greatest from 0 – 3 m and much reduced for depths similar to the upper bay (4 – 5 m). Therefore there is much less zooplankton per unit phytoplankton in the upper Bay of Quinte than expected.

The consistent slopes of the sites confirm the reliable difference in the phytoplankton and zooplankton biomass (Z:P) with lake trophy (Jeppesen et al. 1997a). The ratio of zooplankton to phytoplankton biomass has been found to decline sharply in shallow lakes above a total phosphorus (TP) threshold of 0.1 mg L⁻¹ (Jeppesen et al. 1997a, Hessen et al. 2006), but this is higher than occurs in the Bay of Quinte, and we found that the decline occurred at a much lower value of TP ~ 20 ugL⁻¹ (Figure 1.4). This suggests an inefficient conversion of phytoplankton to zooplankton biomass which is inhibited at a more intermediate nutrient level than found in other studies, though planktivory likely plays a major role in the overall reduction of zooplankton biomass.

The impact of eutrophication on plankton community structure is less well known. For instance, in lakes with TP >0.1 mg L⁻¹, cyanobacteria have been found to dominate with 70% of the total phytoplankton biomass (Watson et al. 1997), though Quinte phytoplankton biomass is dominated by filamentous diatoms (see Section 2). The biomass size spectrum in the Bay of Quinte suggests nutrients control the overall biomass but trophic interactions and perturbations

affect the structure of the relationship (Minns et al. 1987). We find that both biotic and abiotic factors are important in forming the structure of the food web (Folt and Burns 1999), and the efficiencies at which energy is transferred upwards. Abiotic factors were very important in determining zooplankton biomass. The Z:P ratio was one of the weakest relationships in the analysis, with only 13% of the variance explained using TKN as the bottom-up driver. This was less than measured for phytoplankton (53%), which is not surprising, but also good for the planktivorous fishes (56%). Fish biomass with lake trophy has been well studied including relationships with TP (Hanson and Leggett 1982).

The availability of nutrients is a major factor in the species composition of aquatic systems. In the Bay of Quinte, high nutrient conditions lead to high phytoplankton biomass, low zooplankton biomass, high undesirable planktivorous fish biomass (Alewife, White Perch and Gizzard Shad) and low piscivorous fish biomass (Stanza 1). Intermediate nutrients tend to have higher piscivorous fish (Peak Walleye – Stanza 2), reduced planktivorous fish, higher zooplankton and lower phytoplankton biomass. Lower nutrient conditions result in low piscivorous fish biomass, intermediate planktivorous fish biomass, reduced zooplankton and reduced phytoplankton biomass (Stanza 3-4). This relationship is in part driven by highly organic sediment which results in re-mineralized phosphorus during the heat of summer (Jeppesen et al. 1998, Søndergaard et al. 2003) which produces peak phytoplankton biomass, but in filamentous and colonial forms which are not generally ideal for zooplankton grazing (Burns 1987, Haney 1987). In spite of the classification of 1995 – 2014 as a “clear phase”, this increased summer release of phosphorus in the upper bay is usually a characteristic of turbid phase systems (Jeppesen et al. 1999).

Though zooplankton were thought to have the greatest impact on phytoplankton in low nutrient systems (McQueen et al. 1986), trophic transfer efficiency through grazing by herbivorous zooplankton, especially *Daphnia*, on phytoplankton has been shown to be maximized during mesotrophic conditions (Elser and Goldman 1991, Gaedke and Straile 1994). This has been supported with observations of increased *Daphnia* percentages at TP concentrations of ~ 90 mg m⁻³ (Jeppesen et al. 2003, Friederichs et al. 2011). This suggests that as systems move from a eutrophic to mesotrophic state, the percentage of *Daphnia* should increase, and then decrease as it moves to an oligotrophic state. Strus and Hurley (1992) found no relationship between *Daphnia galeata mendotae* and chlorophyll in the Bay of Quinte, but in our analysis *D. galeata* has increased in percentage during the clear phase and low chlorophyll values (see

Section 3). *Daphnia* are highly efficient grazers of phytoplankton, ingesting bacteria, pico and nanoplankton sized cells (Muylaert et al. 2003, Onandia et al. 2015), but are less efficient when feeding on large filamentous or colonial phytoplankton (DeMott et al. 2001). *Daphnia* are accepted as a more efficient forager (Shapiro and Wright 1984, Vanni 1986), reducing *Bosmina*, rotifers and copepod biomass through food competition when planktivory pressure is low (DeMott and Kerfoot 1982). In contrast, mesocosm studies have shown that copepods can be effective at suppressing large-sized phytoplankton (Sommer et al. 2001) and cyclopoid copepods have been more competitive in eutrophic systems dominated by filamentous cyanobacteria (Haney 1987). Clearance rates of *Daphnia* are generally lower in eutrophic systems, likely due to food saturation and interference with their feeding apparatus (Peters and Downing 1984, Onandia et al. 2015). No zooplankton feeding studies have been carried out for the Bay of Quinte communities, so it is unknown how much impact there is to the handling rate due to the dominance of filamentous forms of phytoplankton.

Studies that have directly measured grazing rates of zooplankton have shown that as the nutrient gradient becomes more mesotrophic, grazing rates can approach the growth rates of phytoplankton (Sager and Richman 1991). Cladocera are less efficient grazers in eutrophic systems which are often dominated by filamentous or colonial cyanobacteria forms (Burns 1987). *Daphnia* growth rates are expected to be maximum at mesotrophic TP values (Persson et al. 2007) and food requirements increase with temperature (Peters and Downing 1984, Dziuba et al. 2017), requiring them to expend more energy in foraging (Lehman 1988). Water column temperature closely tracks air temperature in shallow water systems such as the Bay of Quinte upper bay (Currie and Frank 2015) so it is expected that, all other considerations aside, cladocerans would be large and require moderate food concentrations to flourish (Peters and Downing 1984).

Increases in planktivorous fishes decrease the Z:P ratio, zooplankton size, % calanoid copepods, and % *Daphnia*. High planktivory can increase the Chl:TP ratio by reducing zooplankton grazing, and the effects are greater in shallow lake systems (Northcote 1988, Jeppesen et al. 1997b, Hessen et al. 2006, Mehner et al. 2008, Havens and Beaver 2013). Previous studies have shown the Z:P ratio decreases along a low to high nutrient gradient when both separate lakes are considered (McQueen et al. 1986, Jeppesen et al. 1997a) or in embayments (Sager and Richman 1991). Shallow lakes tend to have lower Z:P ratios, especially when macrophytes are abundant and zooplankton individual weights are lower (Cryer

et al. 1986, Jeppesen et al. 2004). Interestingly, in paired wetlands in Belgium, the Z:P ratio (even higher for *Daphnia*:P) was significantly increased in clearwater, macrophyte dominated lakes with reduced fish biomass, regardless of the nutrient concentration (Muyllaert et al. 2003). While shallow water systems are known for rapid switching between turbid and clear phases (Scheffer et al. 1993), these systems may also gradually lose plant biomass and become phytoplankton dominated during eutrophication processes with resulting changes to zooplankton composition (Sayer et al. 2010), so a lack of drastic alteration may not indicate that system change is not occurring. Increased macrophytes can change phytoplankton functional groups, zooplankton composition and behaviour (Cryer et al. 1986, Gebrehiwot et al. 2017) and fish behaviours (Persson and Crowder 1997, Jeppesen 1997b). Changes in planktivorous fishes can also result in increased biomass and changes in composition of zooplankton and behaviour in littoral regions (Jacobsen et al. 1997, Balayla et al. 2017, Gebrehiwot et al. 2017) and can contribute to the switch from turbid to clear phase in lakes (Drenner and Hambright 2002). Studies which sample the zooplankton and phytoplankton communities in the macrophyte dominated littoral regions in the Bay of Quinte would assist in determining if these sorts of changes have occurred.

There is established evidence that planktivorous fishes can exert control over zooplankton composition and sometimes size (See Section 3), but often do not have any effect on total zooplankton biomass (Cryer et al. 1986, Bertolo et al. 2005, Mehner et al. 2008), and have only rarely been shown to have any effect on phytoplankton biomass (Drenner and Hambright 2002, Bertolo et al. 2005). However, the percentage of *Daphnia* approaches zero when planktivorous fish biomass exceeds a threshold (McQueen et al. 1986, Jeppesen et al. 2003). Predation on crustacean zooplankton also tends to promote rotifer communities (Hunt and Matveev 2005), though rotifer biomass in the upper bay Bay of Quinte is a small percentage of the total zooplankton biomass suggestive of a limited top-down effect (Section 3).

In studies of large numbers of Canadian Shield lakes, cladocerans were found to be larger when piscivores were present (Currie et al. 1999), but other studies in oligotrophic systems found no relationship of zooplankton biomass to TP though *Daphnia* and *Holopedium* increased in lakes with piscivores (Bertolo et al. 2005). Both of these suggest weak top-down effects in oligotrophic systems, and Pinel-Alloul et al. (1995) found only 15% of the 50% total variation in zooplankton was explained by fishes. However, the Bay of Quinte is not oligotrophic and top-down effects, particularly by planktivorous fishes are expected to be stronger than found in the

lower bay or Lake Ontario due to increased trophic status and the fact that shallow systems decrease the search volume and depth refuge of prey (Jeppesen et al. 1997a).

Data on planktivorous fish diets in the Bay of Quinte are unfortunately rare. Strus and Hurley (1992) found that Alewife from 1979 – 1986 were more likely to select for cyclopoid copepodites, *Chydorus sphaericus*, *Eubosmina coregoni*, and *Bosmina longirostris*, similar to diets found in Green Bay, WI (Gannon 1976). Even though *Chydorus* is small, it is highly pigmented and thought to be more visible to predation. *Daphnia* was a minor prey item (<1%) but *Daphnia* were rare during these periods and zooplankton populations in Quinte were dominated by smaller species. More planktivory is expected than is captured by the planktivorous fish biomass. Planktivory by invertebrate predators can match that found in vertebrates (Northcote 1988). Benthivores increase phytoplankton by bioturbation (Persson 1997) and their YOY densities can be extremely high, increasing predation pressure on zooplankton (Kahn 2003, Friederichs et al. 2011).

Benthivorous fish biomass is dominated by Common Carp, Freshwater Drum, Brown Bullhead, and White Sucker in the upper and middle Bay of Quinte. Stanza 1 was characterized by Common Carp and Drum. The other stanzas had only occasional years with very high Common Carp, with stanza 2 – 3 having more variety of fishes. White Sucker and Freshwater Drum have been fairly consistent in biomass through the entire project. Predation by Age-0+ fishes can be as important as in adults (Cryer et al. 1986) and soft-finned fishes can dominate YOY in shallow water systems (Drenner and Hambright 2002, Friederichs et al. 2011). Additionally, piscivores such as Walleye depend on zooplankton prey during their larval stages. Unfortunately, consistent data on biomass and diet of YOY and larval fishes in the Bay of Quinte is lacking.

The control of planktivorous fishes by piscivorous fishes is thought to occur only during mesotrophic conditions (Persson et al. 1992, Jeppesen et al. 2000). Walleye dominate the biomass of piscivores in the Bay of Quinte, and have been shown to consume YOY Alewife (Ridgway et al. 1990). In Quinte, peak biomass of piscivores did show a slight effect on cladoceran size (see Section 3), which coincides with the decrease in planktivorous fish biomass during the 1980s. Unlike surveys of many lakes in Europe and New Zealand (Jeppesen et al. 1997a, Jeppesen et al. 1999, Jeppesen et al. 2003), we did not find increased piscivore biomass in the clear phase in the Bay of Quinte. Further, we did not find increased piscivore biomass with reduced nutrient loadings which is fundamental to the AOC fish IBI

(Minns et al. 1994). We did however note that the Yellow:White Perch ratio was higher in the clear phase, which may be another indication that Yellow Perch are more piscivorous in Quinte.

Trophic cascades (Carpenter and Kitchell 1988) would suggest that zooplankton biomass increases with increased piscivory of planktivores, which is supported in part by the highest values of zooplankton being found in stanza 2 (Peak Walleye) in the Bay of Quinte. However, the increase in zooplankton more closely tracks the decrease in planktivore biomass which is lagged by several years before the increase in piscivores, and the years of peak piscivores do not have significantly lower planktivore biomass. We also did not see an increase in the piscivore to phytoplankton biomass ratio in any of the sites as predicted by McQueen et al. (1986) or any changes in the Z:P ratio during the period of high piscivory. Other indicators of top-down control of zooplankton or phytoplankton were also not significant in the upper or middle bay sites suggesting weak effects. This confirms findings in other Canadian and European lakes (Bertolo et al. 2005, Mehner 2010, Friederichs et al. 2011). Top-down effects were evident in the lower bay, but the communities of fishes, dominated by populations sourced from Lake Ontario, are very different than found in the upper bay. This matches previous findings suggesting that control of zooplankton or phytoplankton by piscivores is likely to succeed only in mesotrophic systems (Drenner and Hambright 2002). The expectation is that as the upper bay of Quinte transitions into a more mesotrophic system, there should be a measureable effect of top-down metrics which is currently lacking.

Summary

Overall we see very strong bottom-up effects controlling biomass of phytoplankton and fishes, but little effect of top-down factors in the upper Bay of Quinte (Table 1.7). There has been a shift from high biomass values for Alewife and Gizzard Shad during the earlier turbid phase to White Perch and Yellow Perch during the clear phase. In spite of this, zooplankton biomass ratios show little grazing effects by planktivorous fish or with changes in piscivorous fish biomass. Some compositional changes (e.g. in *Daphnia galeata mendotae*) indicate that grazing effects on the zooplankton community are occurring, but the changes are not significant in the upper bay, though they are significant and strong in the middle and lower bay. There has been little change in the ratio metrics even over the long-term (the entire Project Quinte time-series) at the upper bay sites, and no recent improvement since 2001 (Fig. 1.7, Table 1.7). This lack of change indicates that the trophic efficiency of the upper Bay of Quinte has not yet improved over time as it has in the middle bay site, even though many of the biomass and nutrient

measures are not significantly different between Belleville and Hay Bay. There was an expectation of strong changes, particularly to the Z:P ratio in the upper bay with the change to the clear phase after 1994, however this has not occurred. This is likely due to the underlying effect of summer sediment phosphorus release into the mixed, shallow water system, whereas Hay Bay is deeper and is generally stratified through the summer. This sediment-mediated release of nutrients is driving nitrogen limitation (given by the strong TKN relationship as opposed to TP in the upper and middle bays) and promoting the growth of filamentous and colonial phytoplankton which are not being efficiently converted into zooplankton biomass. As sediment phosphorus values decrease over time, this effect is likely to be moderated and the Z:P ratios should increase to those found in typical eutrophic-mesotrophic environments.

Section II: BUI 13 Assessment: Phytoplankton

In our recommended targets document (Currie et al. 2023), we proposed two targets for assessing phytoplankton composition with a total of five associated measures (Table 2.1). The following assessment of phytoplankton targets includes analyses from 4 sites. Emphasis is given to the upper bay sites at Belleville (station B) and Napanee (station N). However trends from sites at Hay Bay (station HB or middle bay) and Conway (Station C, or lower bay near the interface with Lake Ontario) are also presented for comparison and to provide a holistic picture of the bay (Fig. 1.1). The aim here is to show whether or not there are long term improvements in the bay based on these revised criteria to inform our assessment of the status of the phytoplankton and zooplankton populations BUI.

Table 2.1: Proposed targets and measures to assess phytoplankton composition under BUI 13.

Target	Measure
Demonstrate a positive change in phytoplankton species composition with a decrease in nuisance and eutrophic (and noxious) indicator species that may impair food web function.	<ol style="list-style-type: none">1. A decrease in nuisance phytoplankton to less than 50% of the biomass (i.e. $< 2.5 \text{ g m}^{-3}$ in the upper bay offshore based on the existing target for BUI 8 of phytoplankton biomass of $4 - 5 \text{ g m}^{-3}$)2. A decreasing trend in filamentous and colonial / mucilaginous Cyanobacteria (both in terms of biomass and relative composition)3. A decreasing trend in filamentous diatoms (biomass and % composition).
Show an increasing trend in phytoplankton taxa that provide an ample food resource for zooplankton and other grazers.	<ol style="list-style-type: none">1. An Increase in the more edible phytoplankton taxa to approximately 30% of the biomass2. An increase in the biomass of Chrysophyceae with an ideal target of $>15\%$ of total phytoplankton biomass

2.1 Methods

The phytoplankton data used in this assessment were collected under the long term monitoring program Project Quinte by MOECC (1973 – 1999) and DFO (2000 – 2015) from the upper, middle and lower bay stations listed above. Integrated whole water samples were collected from either the euphotic zone in unstratified conditions or the epilimnion during stratification. Details of this sampling program are given in each of the Project Quinte Annual Reports (e.g., Munawar et al, 2017). Phytoplankton samples were fixed with acidified Lugol's iodine upon collection. From 1973 – 2011, phytoplankton identification and enumeration was conducted following a modified version of the Utermöhl (1958) inverted microscope technique (Nicholls et al. 2002).

From 2012 – 2015, the HPMA (2-hydroxypropyl methacrylate) technique described by Crumpton (1987) was used. These techniques are broadly compatible.

While the May – October sampling season has been maintained throughout Project Quinte, there have been changes in both frequency and site locations from year to year. From 1972 – 1982, sampling was approximately weekly (21 – 24 per station per year). From 1983 onwards, the frequency was reduced to bi-weekly (11 – 14 per station per year). A total of 2399 phytoplankton samples were collected for this analysis. Beginning in 1978, phytoplankton samples at some sites were physically pooled creating a single composite sample that was then analysed with the intention that data were representative of the station mean over the growing season (see Nicholls and Carney, 2011 for complete details).

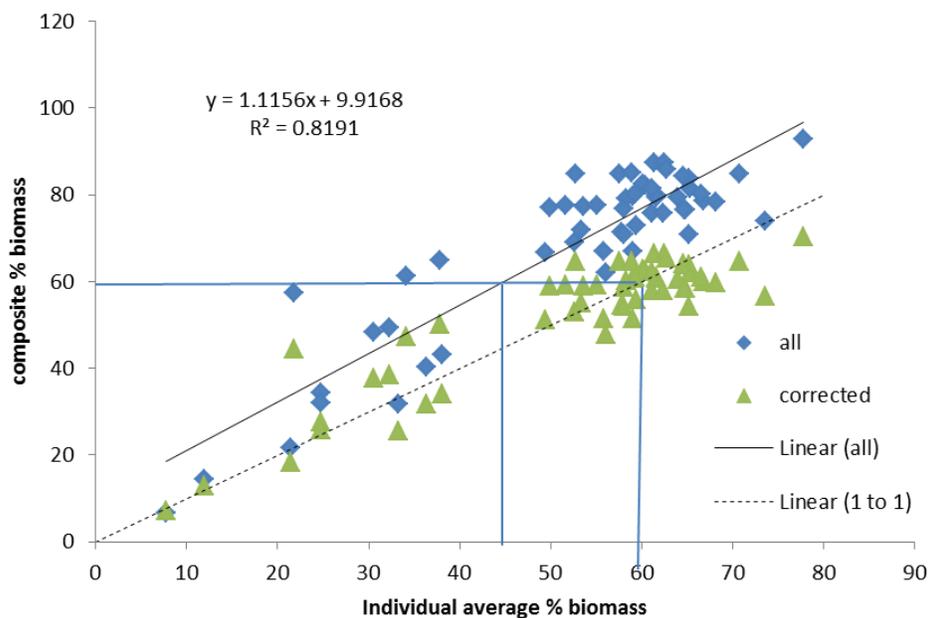


Figure 2.1: Comparison of nuisance algae (% biomass) measured by composite sample (blue diamond) or regression-corrected composite (green triangle) with the arithmetic means of individually counted samples. Blue lines indicate the composite samples equivalent as individually counted average pre correction (45%) and post correction (60%). Similar analyses (not shown) were undertaken for all target measures.

For the purposes of data analyses, we used May – October arithmetic means where available representing 35 years at Belleville, 18 years at Napanee, 12 years at Hay Bay and 17 years at Conway. In cases where only a single composite phytoplankton sample was analysed and/or an arithmetic mean could not be computed because too few individual dates were enumerated (i.e. at a frequency of less than 1 per month), we used composite phytoplankton samples to

complete the analysis (B=8, N=14, HB=27, C=20). The individual and composite samples analysed are summarized in the Appendix to this section.

There were a total of 59 sample years where both arithmetic means of phytoplankton samples and composite phytoplankton samples were available. These data were used to construct a series of linear regressions comparing the results of the arithmetic mean (x-axis) to the composite sample (y-axis) which showed departure from the 1 to 1 line (e.g. Fig. 2.1). These regressions were computed for all measures on the basis of both biomass and % biomass. When the sample types were found to be strongly correlated ($r^2 > 0.7$), we used the regression equation to develop correction factors for the composite data which were then applied to the subsequent analyses; otherwise, no correction factor was applied. Table 2.2 summarizes the regressions.

Table 2.2: Relationships between phytoplankton targets measured as averaged individual date (x axis) and physically composite sample (y axis) slope and r^2 values.

Target Measure	units	slope	r^2
Nuisance algae	Biomass	$y = 0.9008x + 251.69$	0.82
	% biomass	$y = 1.1156x + 9.9168$	0.82
Filamentous Cyanobacteria	Biomass	$y = 0.8683x + 57.69$	0.73
	% biomass	$y = 1.1973x + 0.2676$	0.82
Colonial Cyanobacteria	Biomass	$y = 0.7919x + 32.627$	0.90
	% biomass	$y = 1.2323x - 0.6579$	0.77
Filamentous diatoms	Biomass	$y = 0.8657x + 275.04$	0.82
	% biomass	$y = 1.3475x - 3.8518$	0.82
Edible phytoplankton	Biomass	$y = 0.7837x + 66.584$	0.71
	% biomass	$y = 0.9423x - 7.0027$	0.71
Chrysophyceae	Biomass	$y = 0.5738x + 20.203$	0.33
	% biomass	$y = 0.6778x - 0.9745$	0.62

Once the composite data were corrected, each target and measure (nuisance algae, edible algae, etc.) were analysed for trends in biomass and % biomass using a one way ANOVA by station across four time stanzas (1. 1972 – 1982, 2. 1983 – 1994, 3. 1995 – 2000, 4. 2001 – 2015) as defined in Section I. A Tukey-Kramer comparison between stanzas was applied to determine differences between stanzas.

For each phytoplankton measure at each station, we ran a CuSUM (cumulative sum) analysis to observe general changes in the annual data over time (see also Nicholls et al. 2002). The

CuSUM requires an assigned weight “ ω ” to be subtracted from each individual measure. For ω , we used either the numerical value associated with the target measure (Table 2.1) or where no value is specified, the average of the measure over the full time period. Each plotted CuSUM shows deviation from the target, either positive or negative. As deviation away from the target increases, the slope of the plotted line increases (positive or negative). Similarly, as deviation away from the target decreases (i.e. the indicator begins to stabilize), the slope of the line approaches zero. When a change in deviation from the mean occurs, there is a delay in the change in CuSUM direction. If an indicator is slipping towards a previous state (i.e., poor community health), the CuSUM will cease to plateau and will once again increase or decrease.

Trend analyses were undertaken for all measures at all sites to show whether or not a given measure is moving towards the target (positive) or away from the target (negative). The idea is to take a more nuanced approach to the assessment than a simple Yes (target met) or No (target not met) conclusion. Short term (x-axis) and long term (y-axis) trends were computed as:

$$X = (\text{AVG}_{\text{Last 5 yrs}} - \text{AVG}_{\text{Stanza 4}}) / (\text{AVG}_{\text{Stanza 4}}); Y = (\text{AVG}_{\text{Stanza 4}} - \text{AVG}_{\text{All Years}}) / (\text{AVG}_{\text{All years}})$$

The coordinates are mapped on a Cartesian plane with each quartile representing (clockwise):

1. Long and short term recovery (+,+), 2. Recent improvement (+,-), 3. Long and short term decline (-,-), 4. Recent decline (-,+).

2.2 Assessment of Phytoplankton Targets

2.2.1 Assessment of Target 1: Decrease in species that may impair food web function.

2.2.1a A decrease in nuisance phytoplankton to less than 50% of the biomass ($< 2.5 \text{ g m}^{-3}$ in the upper bay offshore based on the existing target for BUI 8 of phytoplankton biomass of $4 - 5 \text{ g m}^{-3}$)

ANOVA:

At B, the biomass of nuisance phytoplankton decreased from an average ($\pm 1 \text{ SE}$) of $7.5 \pm 0.9 \text{ g m}^{-3}$ in the first time stanza (1972 – 82) to $2.9 \pm 0.2 \text{ g m}^{-3}$ in the most recent stanza (2001-2015) corresponding with declines in phosphorus loadings and concentrations (Fig. 2.2). Although this is a dramatic and statistically significant decline ($p < 0.0001$), 10 of 15 years in stanza 4 remain above the target of 2.5 g m^{-3} . The proportion of nuisance algae did not show a significant change from stanza 1 ($63.5 \pm 1.8 \%$) to stanza 4 ($59.4 \pm 1.9 \%$) and still remains above the target of 50% (Fig. 2.3).

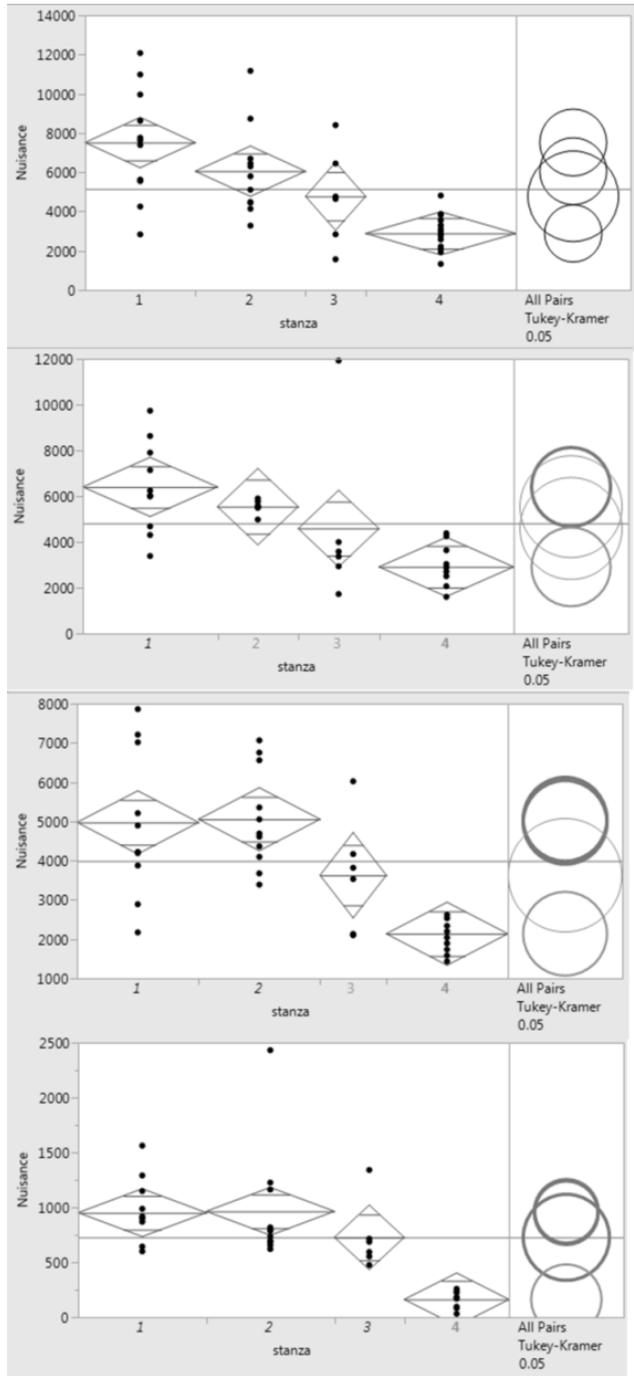
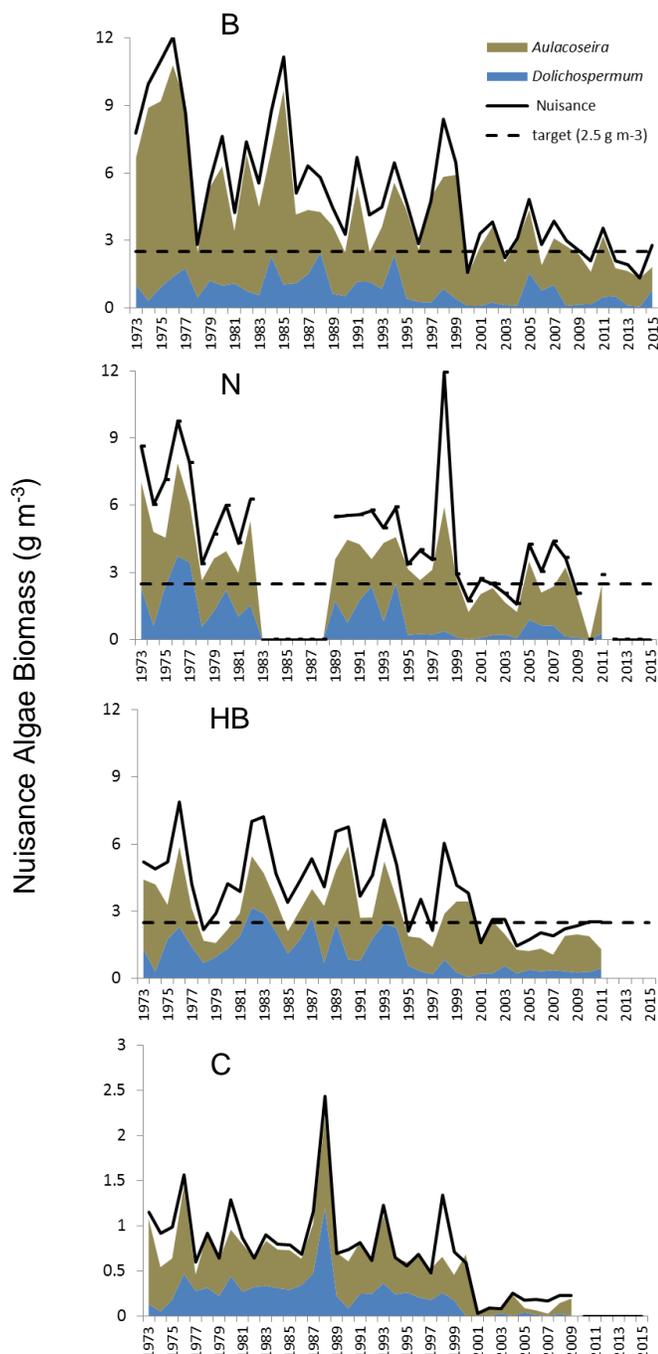


Figure 2.2: Left: May – October mean biomass (g m^{-3}) of nuisance algae with proportion of biomass contributed by *Aulacoseira* and *Dolichospermum* indicated. Right: ANOVA analysis of left by time stanza (biomass in mg m^{-3}). Circles indicate range of and amount of overlap between time stanzas.

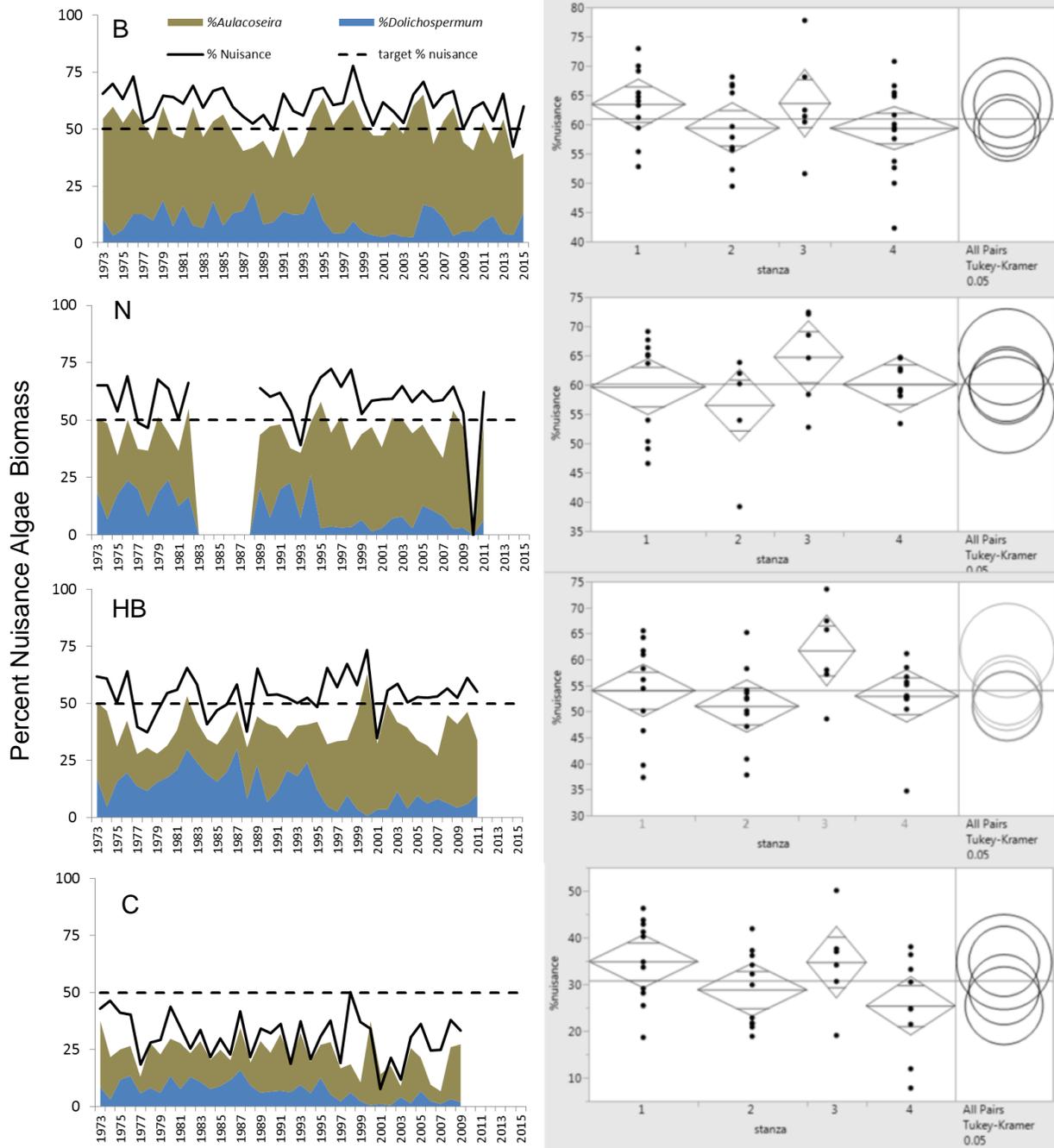


Figure 2.3: Left: May – October mean % contribution to total biomass by nuisance algae with proportion of biomass contributed by *Aulacoseira* and *Dolichospermum* indicated. Right: ANOVA analysis of left by time stanza. Circles indicate range of and amount of overlap between time stanzas.

Nuisance phytoplankton biomass has also fallen significantly at N, from $6.4 \pm 0.6 \text{ g m}^{-3}$ in stanza 1 to $2.9 \pm 0.3 \text{ g m}^{-3}$ in Stanza 4 ($p=0.0045$), but still remains slightly above the target of 2.5 g m^{-3} (Fig. 2.2) and 7 of 10 years in stanza 4 are over the target. However, the proportion of nuisance

algae has not changed significantly over time accounting for approximately 60% of the biomass across time stanzas and above the target of 50% (Fig. 2.3).

HB showed a significant ($p < 0.0001$) downward trend in nuisance algal biomass from $5.0 \pm 0.5 \text{ g m}^{-3}$ in stanza 1 to $2.1 \pm 0.1 \text{ g m}^{-3}$ in stanza 4 (Fig. 2.2) with only 4 of 11 years exceeding the target. The trend in % composition was not significant, with nuisance phytoplankton accounting for slightly more than 50% of the total biomass in all stanzas (Fig. 2.3). At C, which is heavily influenced by the offshore waters of Lake Ontario, total nuisance biomass declined significantly ($p < 0.0001$) from $0.9 \pm 0.09 \text{ g m}^{-3}$ (stanza 1) to $0.2 \pm 0.03 \text{ g m}^{-3}$ (stanza 4) (Fig. 2.2) but did not show a significant trend in terms of the proportion of nuisance algae (Fig. 2.3).

CuSUM:

The CuSUM analyses of nuisance phytoplankton reveal similar trends to the ANOVA (Fig. 2.4). In the upper bay (B, N), the sharply increasing slope from 1973 to 1978 tracks the elevated nuisance biomass prior to phosphorus abatement. By 1998, the slopes appear to level off suggesting that nuisance biomass is stabilizing near the target measure of 2.5 g m^{-3} . The middle bay site, HB, closely resembles the trends at N. At C, the negative slope of the CuSUM functions simply reflects the fact that this site has never had an excess of nuisance algae in the years measured. Anecdotally, the amount of filamentous algae in the zooplankton nets collected at C has increased in recent years.

As a percent of the total phytoplankton community biomass, the nuisance algae in the upper bay have consistently been over the 50% target, seen as a continual increase in the CuSUM plot (Fig. 2.4). In the middle bay the slope of the CuSUM is very shallow and there have been periods in the late 1970s and mid 1980s when the proportion of nuisance phytoplankton was below 50%. C, in the lower bay is consistently below the 50% measure for nuisance algae.

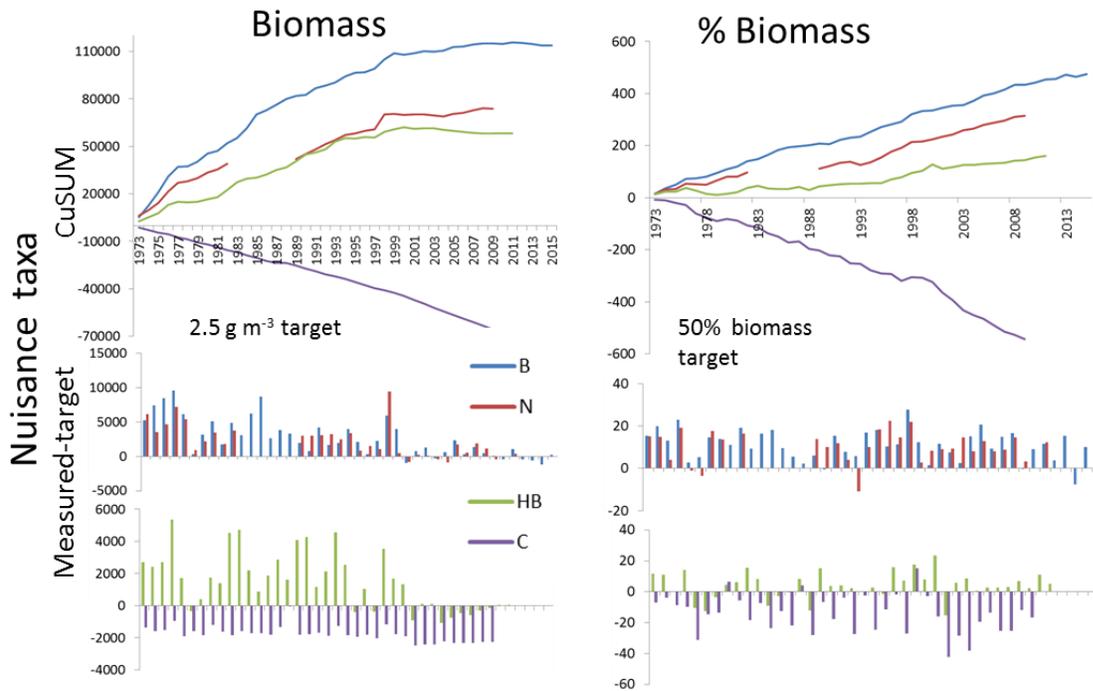


Figure 2.4: Left: Nuisance algae biomass; Right: Nuisance algae % biomass; Each set of plots shows in top panel: Cumulative sum of measure – target by year; lower panels: measured value – target value by year (0 line target = measured, hope for ≤ 0 values for these measures). Target used in calculations indicated as text on graphs.

2.2.1b A decreasing trend in filamentous and colonial Cyanobacteria (both in terms of biomass and relative composition)

ANOVA:

In the upper bay site at B, the biomass of filamentous Cyanobacteria have declined significantly ($p < 0.0001$) from an average (± 1 SE) of $1.7 \pm 0.2 \text{ g m}^{-3}$ in stanza 1 (1972 – 1982) to $0.5 \pm 0.1 \text{ g m}^{-3}$ in stanza 4 (2001 – 2015), shown in Figure 2.5 and Table 2.3. The relative contribution of filamentous Cyanobacteria also declined significantly from $18.2 \pm 1.8 \%$ of the total phytoplankton biomass in stanza 1 to $9.4 \pm 1.9 \%$ in stanza 4 (Fig. 2.6). Colonial Cyanobacteria did not show a significant change in terms of biomass ($\approx 0.2 - 0.3 \text{ g m}^{-3}$) but did show a significant increase ($p = 0.0044$) in their relative contribution from $2.4 \pm 0.4 \%$ of total biomass in stanza 1 to $7.2 \pm 1.2 \%$ in stanza 4.

Table 2.3. Stanza means (\pm SE), one way ANOVA significance value and comparison of means (Tukey-Kramer HSD) for phytoplankton metrics expected to decrease.

Target	Measure	Site	Stza 1 72-83	Stza 2 84-94	Stza 3 95-00	Stza 4 01-15	Sig.	Stanza			
								1	2	3	4
Nuisance Phytoplankton	Biomass (<2.5 g m ⁻³)	B	7.5 \pm 0.9	6.1 \pm 0.7	4.8 \pm 1.0	2.9 \pm 0.2	<0.0001	B	AB	A	A
		N	6.4 \pm 0.6	5.5 \pm 0.1	4.6 \pm 1.5	2.9 \pm 0.3	0.0045	B	AB	AB	A
		HB	5.0 \pm 0.5	5.1 \pm 0.4	3.6 \pm 0.6	2.1 \pm 0.1	<0.0001	B	AB	A	A
		C	1.0 \pm 0.1	1.0 \pm 0.2	0.7 \pm 0.1	0.2 \pm 0.0	<0.0001	B	A	A	A
	% Biomass (<50%)	B	63.5 \pm 1.8	59.4 \pm 1.9	63.6 \pm 3.6	59.4 \pm 1.9	ns				
		N	59.7 \pm 2.7	55.8 \pm 3.7	64.8 \pm 3.2	60.1 \pm 1.1	ns				
		HB	54.1 \pm 2.9	51.1 \pm 2.3	61.7 \pm 3.6	53.0 \pm 2.1	ns				
		C	34.9 \pm 2.7	28.9 \pm 2.4	34.8 \pm 4.1	25.4 \pm 3.5	ns				
Filamentous Cyanobacteria	Biomass	B	1.7 \pm 0.2	2.3 \pm 0.3	0.4 \pm 0.1	0.5 \pm 0.1	<0.0001	B	B	A	A
		N	3.1 \pm 0.5	2.6 \pm 0.4	0.2 \pm 0.1	0.4 \pm 0.1	<0.0001	B	B	A	A
		HB	2.9 \pm 0.4	2.7 \pm 0.3	0.4 \pm 0.1	0.4 \pm 0.0	<0.0001	B	B	A	A
		C	0.4 \pm 0.1	0.4 \pm 0.1	0.2 \pm 0.0	0.0 \pm 0.0	<0.0001	B	AB	A	A
	% Biomass	B	18.2 \pm 1.8	23.9 \pm 2.2	7.2 \pm 1.2	9.4 \pm 1.9	<0.0001	B	B	A	A
		N	29.0 \pm 2.5	26.7 \pm 5.1	4.0 \pm 0.7	8.6 \pm 1.7	<0.0001	B	B	A	A
		HB	33.0 \pm 2.3	29.2 \pm 2.8	6.8 \pm 2.2	8.5 \pm 1.2	<0.0001	B	B	A	A
		C	18.3 \pm 1.8	12.6 \pm 1.2	5.8 \pm 2.5	3.3 \pm 0.7	<0.0001	C	C	B	A
Colonial Cyanobacteria	Biomass	B	0.2 \pm 0.0	0.2 \pm 0.0	0.6 \pm 0.4	0.3 \pm 0.1	ns				
		N	0.2 \pm 0.1	0.1 \pm 0.0	1.7 \pm 1.1	0.5 \pm 0.1	ns				
		HB	0.2 \pm 0.1	0.1 \pm 0.0	1.3 \pm 0.5	0.5 \pm 0.1	0.0006	B	B	B	A
		C	0.0 \pm 0.0	0.0 \pm 0.0	0.3 \pm 0.1	0.1 \pm 0.0	0.0131	B	B	AB	A
	% Biomass	B	2.4 \pm 0.4	2.7 \pm 0.4	6.0 \pm 2.3	7.2 \pm 1.2	0.0044	B	B	AB	A
		N	3.0 \pm 0.4	2.8 \pm 0.4	14.3 \pm 4.4	11.3 \pm 1.6	0.0004	B	B	A	A
		HB	2.9 \pm 0.3	2.6 \pm 0.2	17.8 \pm 4.7	12.4 \pm 1.9	<0.0001	B	B	A	A
		C	2.0 \pm 0.2	2.8 \pm 0.3	13.6 \pm 5.3	8.0 \pm 1.9	0.0013	B	B	AB	A
Filamentous Diatoms	Biomass	B	6.3 \pm 0.8	3.9 \pm 0.6	3.8 \pm 0.6	2.2 \pm 0.2	<0.0001	B	AB	B	A
		N	3.7 \pm 0.4	3.7 \pm 0.8	3.1 \pm 0.6	2.1 \pm 0.2	0.0454	B	AB	AB	A
		HB	2.4 \pm 0.3	3.0 \pm 0.5	2.1 \pm 0.4	1.5 \pm 0.2	0.0149	B	AB	AB	A
		C	0.6 \pm 0.1	0.6 \pm 0.1	0.4 \pm 0.1	0.1 \pm 0.0	<0.0001	B	A	A	A
	% Biomass	B	53.3 \pm 2.7	40.4 \pm 2.5	54.5 \pm 1.5	45.5 \pm 3.0	0.0063	B	AB	A	A
		N	37.6 \pm 2.3	38.8 \pm 5.4	50.3 \pm 3.4	43.7 \pm 2.7	0.0454	A	A	A	A
		HB	28.6 \pm 2.4	32.6 \pm 2.7	39.9 \pm 5.6	37.3 \pm 2.6	ns				
		C	26.7 \pm 3.0	30.0 \pm 2.4	22.2 \pm 3.3	16.3 \pm 2.7	0.0093	B	AB	AB	A

Similar trends were apparent at N. The biomass of filamentous Cyanobacteria declined significantly ($p < 0.0001$) from $3.1 \pm 0.5 \text{ g m}^{-3}$ in Stanza 1 to $0.4 \pm 0.1 \text{ g m}^{-3}$ in stanza 4 (Fig. 2.5). The decline in relative contribution was also significant from $29.0 \pm 2.5 \%$ to $8.6 \pm 1.7 \%$. With respect to colonial forms of Cyanobacteria, biomass did not change significantly across the time stanzas however the proportion of colonial Cyanobacteria did increase significantly ($p = 0.0004$) from $3.0 \pm 0.4 \%$ in stanza 1 to $11.3 \pm 1.6 \%$ in stanza 4 (Fig. 2.6).

At the remaining sites outside of the AoC (HB, C), similar declines in the biomass and relative composition of filamentous Cyanobacteria were apparent between Stanzas 1 and 4 ($p < 0.0001$; Fig. 2.5, 2.6). However, both of these sites also showed a significant increase in the biomass of colonial Cyanobacteria in stanza 3 (1995 – 2000; $P < 0.0001$) as well as significant increases in the relative contribution of colonial Cyanobacteria beginning in Stanza 3 (Fig. 2.5, 2.6).

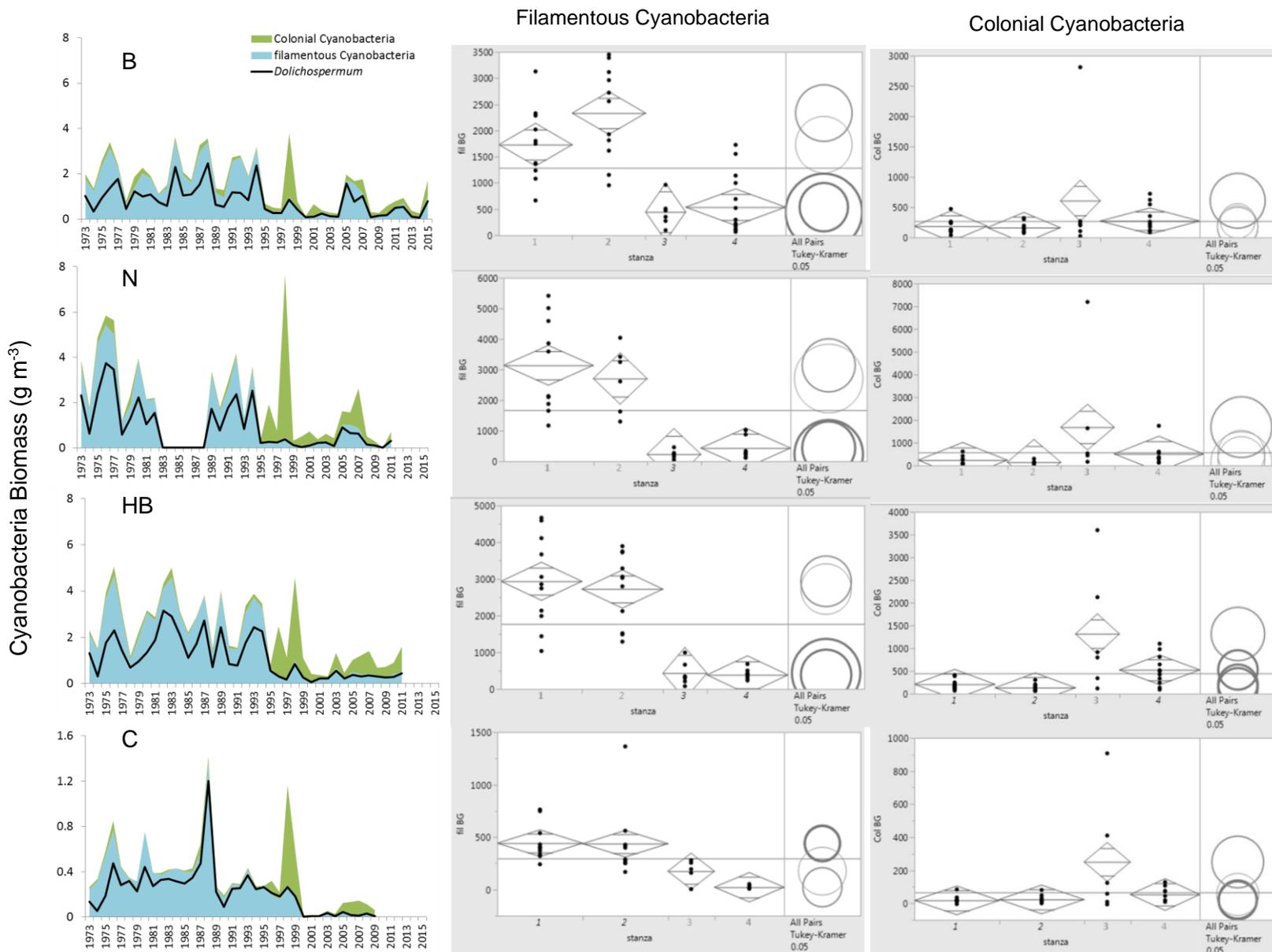


Figure 2.5: Left panel: Cyanobacteria biomass (g m^{-3}) divided by filamentous (blue) and colonial (green) forms. Black line indicates amount contributed by *Dolichospermum*. Centre: ANOVA analyses of filamentous and Right: colonial Cyanobacteria biomass (May–Oct average, mg m^{-3}) by time stanza. Circles indicate range of and amount of overlap between time stanzas.

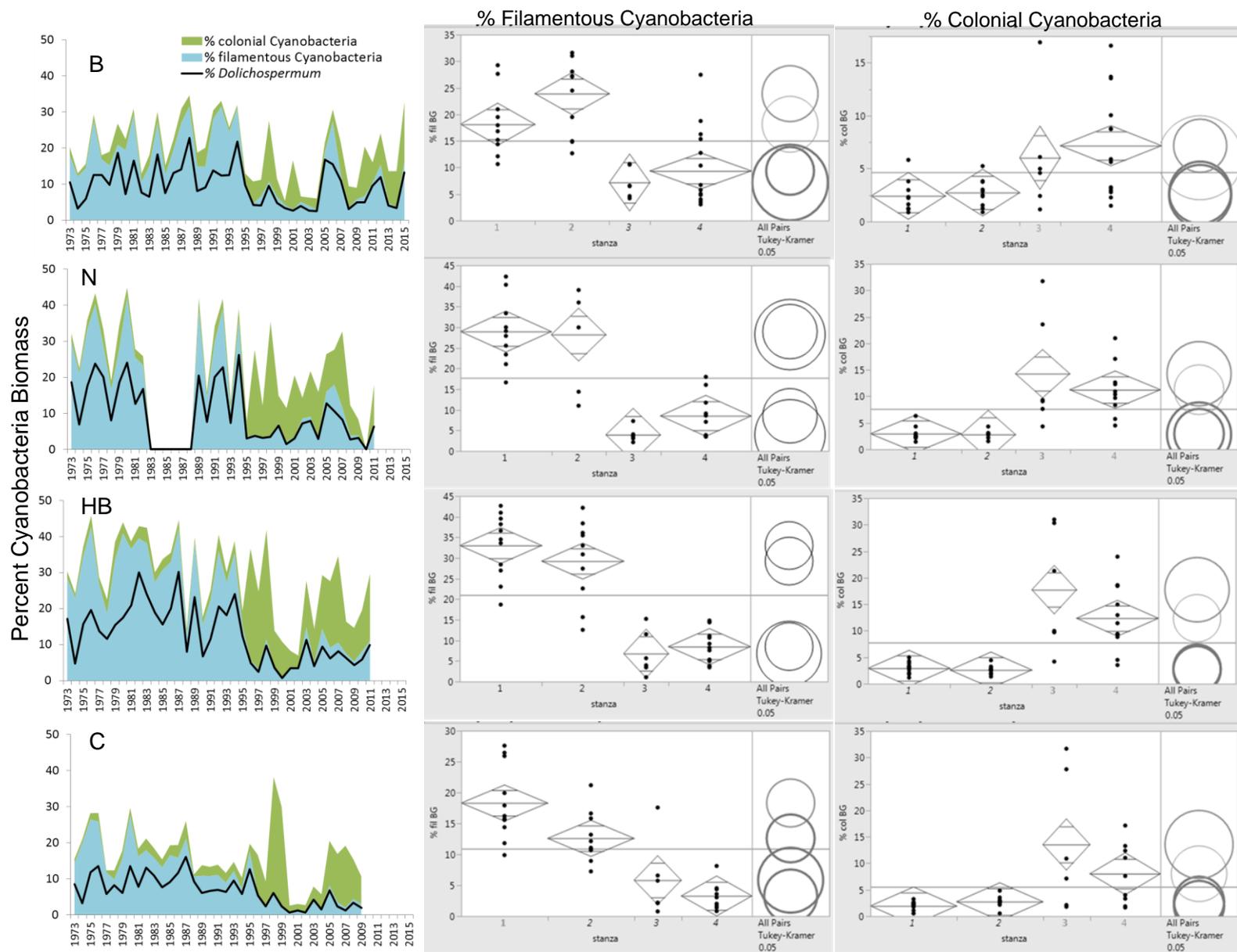


Figure 2.6: May – Oct mean percent Cyanobacteria of total phytoplankton biomass divided by filamentous (blue) and colonial (green) forms (left). Black line indicates amount contributed by *Dolichospermum*. ANOVA analyses of filamentous (centre) and colonial (right) Cyanobacteria biomass May – Oct average (% biomass) by time stanza. Circles indicate range of and amount of overlap between time stanzas.

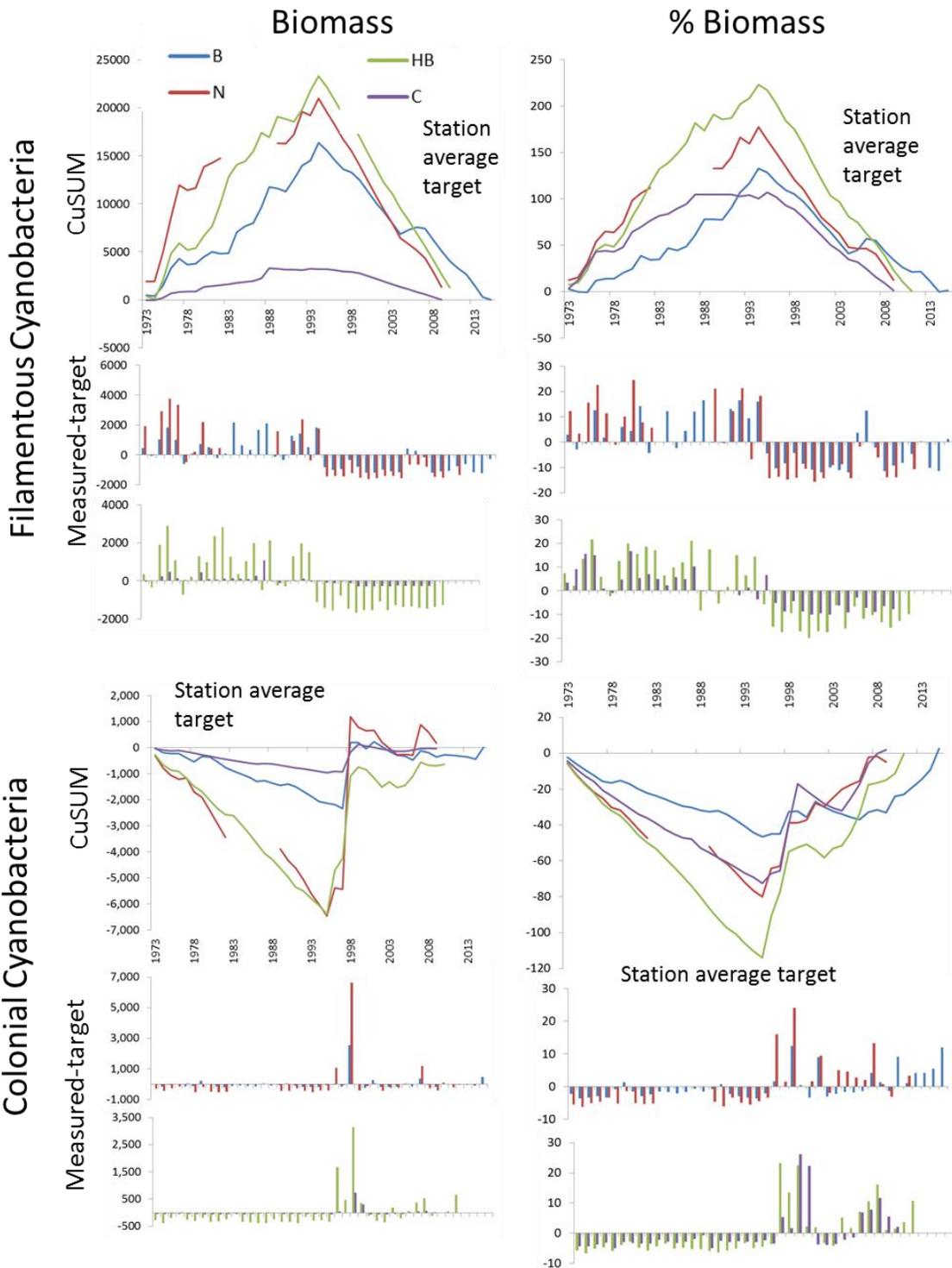


Figure 2.7: Top left: Filamentous Cyanobacteria biomass; top right: Filamentous Cyanobacteria % biomass; bottom left: Colonial Cyanobacteria biomass; bottom right: Colonial Cyanobacteria % biomass. Each set of plots shows in top panel: Cumulative sum of measure – target by year; lower panels: measured value – target value by year. Target used in CuSUM calculations indicated as text on graphs.

CuSUM:

With respect to the CuSUM analysis, filamentous and colonial Cyanobacteria biomass show opposite trends in the residual plot which is apparent at all stations, although the magnitude of change at C is considerably less than the other sites. A sharp change in direction can be seen beginning in 1995, the start of the clear phase (Fig. 2.7). Filamentous Cyanobacteria biomass and % biomass were above average but variable in the early years (1973 – 1994), then shifted to consistently lower levels. In contrast, colonial Cyanobacteria biomass and % biomass were comparatively low from the 1970s to the 1990s but some dramatic spikes in biomass were observed in the mid to late 1990s. Despite this, biomass of colonial forms has generally trended closer to the long term average while the proportion has fluctuated widely since 1995.

*2.2.1c A decreasing trend in filamentous diatoms (biomass and % composition).***ANOVA:**

Throughout Project Quinte the largest proportion of phytoplankton biomass and nuisance algae in the Bay of Quinte has belonged to species of *Aulacoseira*, a filamentous diatom (Fig. 2.3). In the upper bay, filamentous diatom biomass at B has declined significantly ($p < 0.001$) from an average (± 1 SE) of $6.3 \pm 0.8 \text{ g m}^{-3}$ in stanza 1 (1972 – 1982) to 3.9 ± 0.6 in Stanza 2 (1984 – 1994) and nominally to $2.2 \pm 0.2 \text{ g m}^{-3}$ in stanza 4 (2001 – 2015) (Fig. 2.8, Tab. 2.3). Likewise, significant changes in the relative contribution of filamentous diatoms occurred between Stanzas 1 ($53.3 \pm 2.7\%$) and 2 ($40.4 \pm 2.5\%$) ($p < 0.01$); stanza 4 fell in between at $45.5 \pm 3.0\%$ (Fig. 2.9). At N, there is no significant trend in the biomass of filamentous diatoms among the time stanzas although stanza 4 at $2.1 \pm 0.3 \text{ g m}^{-3}$ is considerably lower than stanza 1 at $3.7 \pm 0.4 \text{ g m}^{-3}$ (Fig. 2.8). There does appear to be an increasing trend in the relative contribution of filamentous diatoms beginning in stanza 3 ($p < 0.05$) compared to stanzas 1 and 2. Having said that, the mean value for the current stanza (4) of $43.7 \pm 2.7\%$ is only marginally higher than the $37.6 \pm 2.3\%$ observed in stanza 1 (Fig. 2.9). The sites at HB and C also showed significant declining trends in the biomass of filamentous diatoms (Fig. 2.8). There was no significant change in relative composition at HB, however C did show a significant downward trend in % biomass ($p < 0.05$; Fig. 2.9).

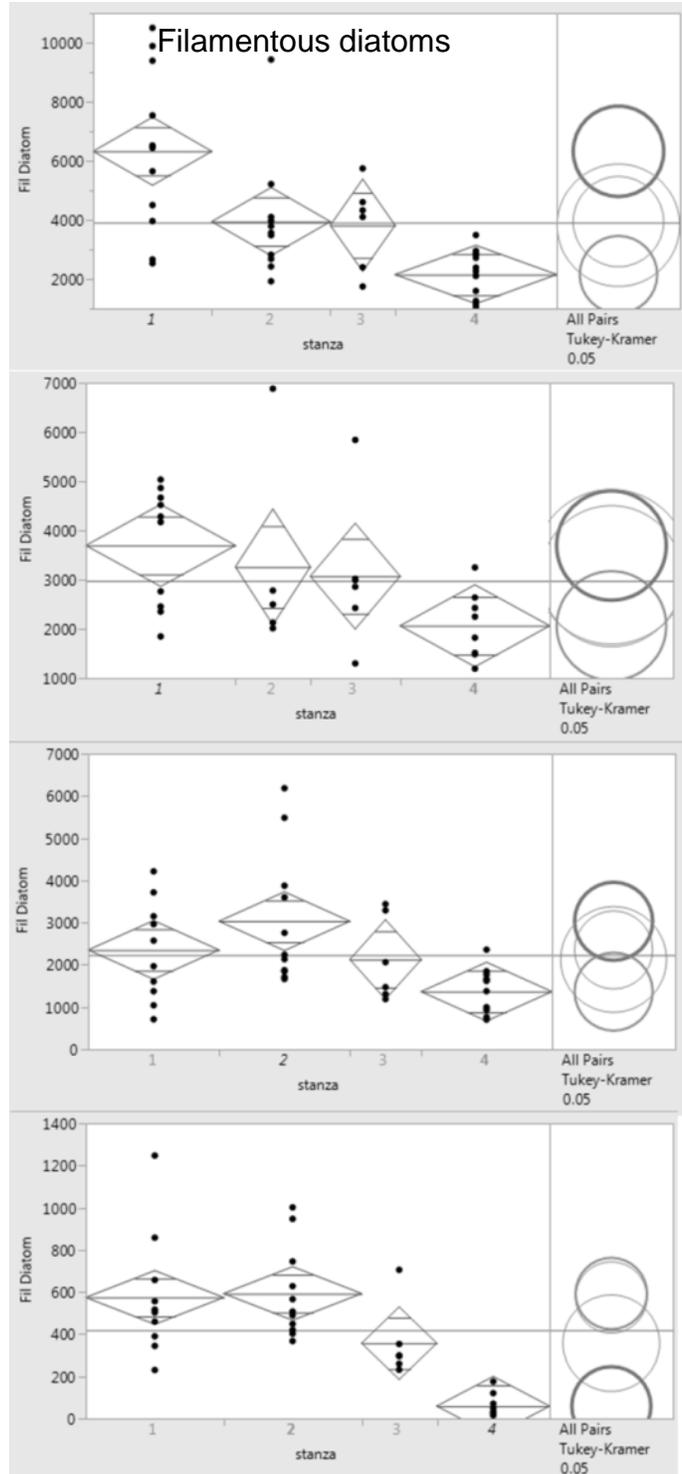
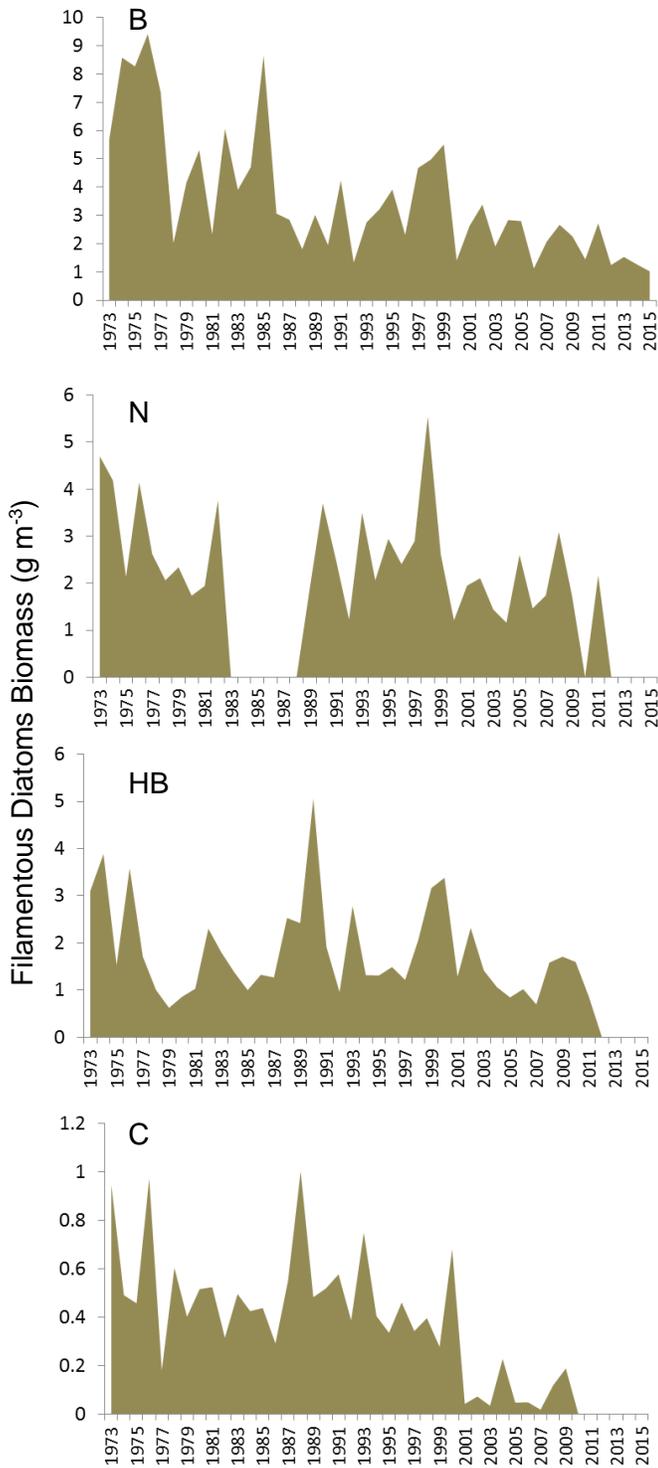


Figure 2.8: Left: May – Oct mean filamentous diatoms biomass ($g\ m^{-3}$). Right: ANOVA analysis of filamentous diatoms biomass by stanza (biomass in $mg\ m^{-3}$). Circles indicate range of and amount of overlap between time stanzas.

Percent Filamentous Diatoms Biomass

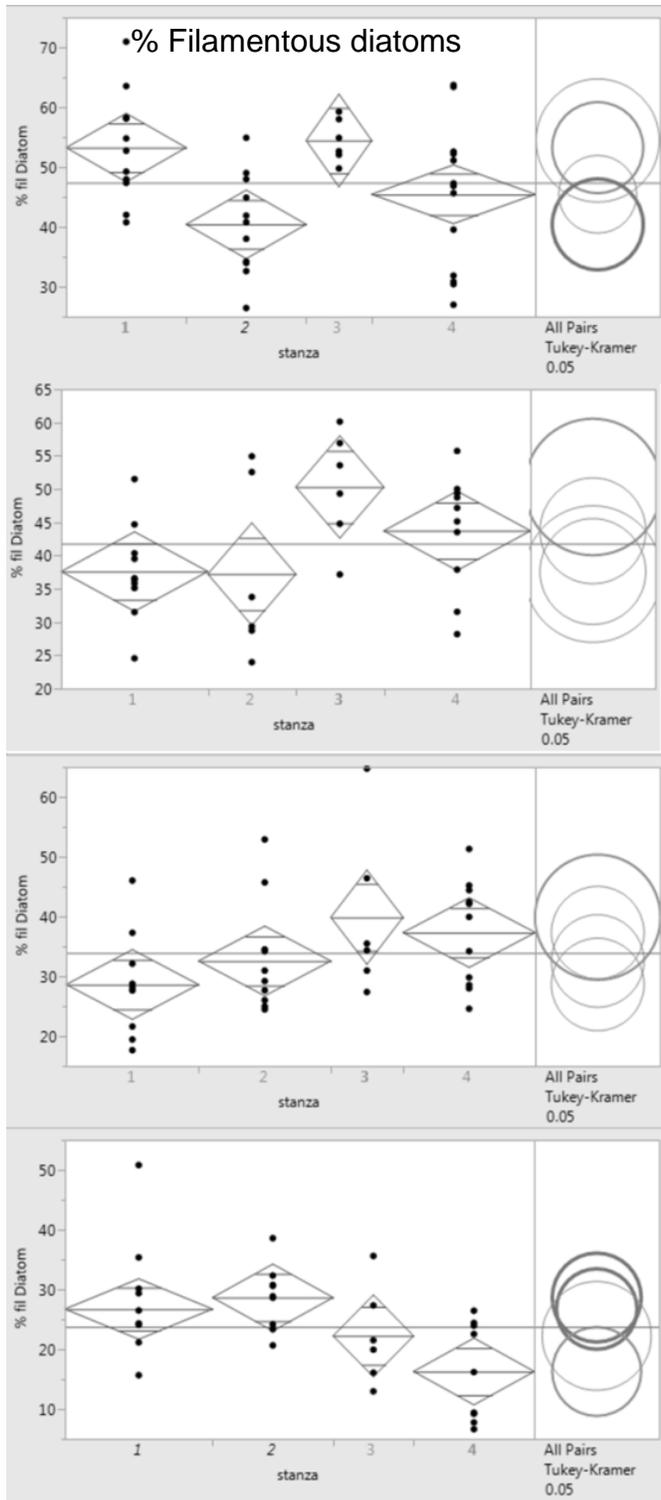
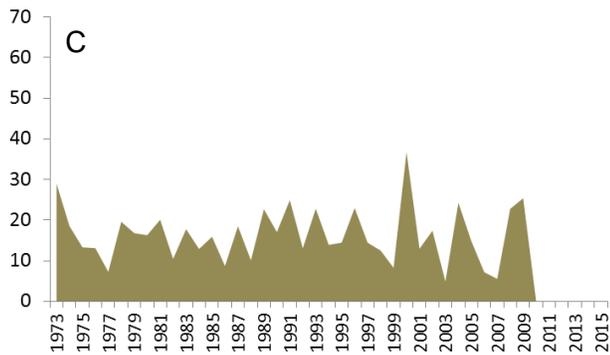
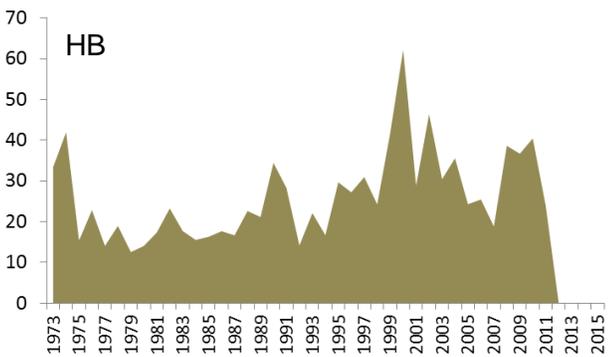
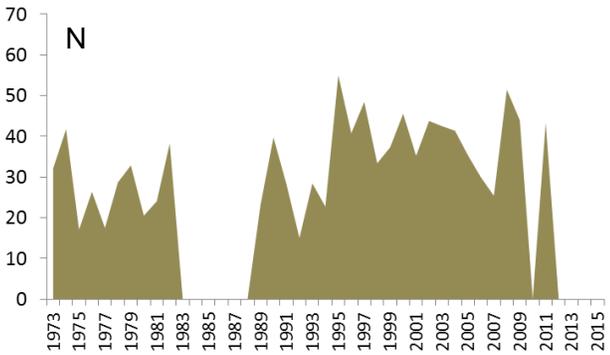
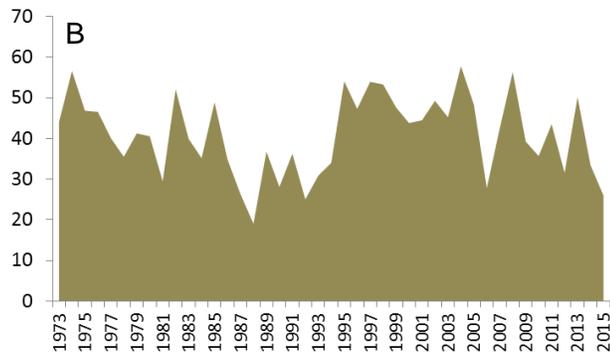


Figure 2.9: Left: May – Oct mean filamentous diatoms percent biomass. Right: ANOVA analysis of filamentous diatoms biomass by stanza. Circles indicate range of and amount of overlap between time stanzas.

CuSUM:

For this analysis, the CuSUM was weighted against the average biomass of filamentous diatoms over the study period. The dome shape of the function, most pronounced at B in the upper bay, indicates a long term decline (Fig. 2.10). The pattern is less pronounced at the other sites, but the trend is apparent, the biomass of filamentous diatoms is decreasing. The proportion of filamentous diatoms is highly variable, fluctuating back and forth around the long term average.

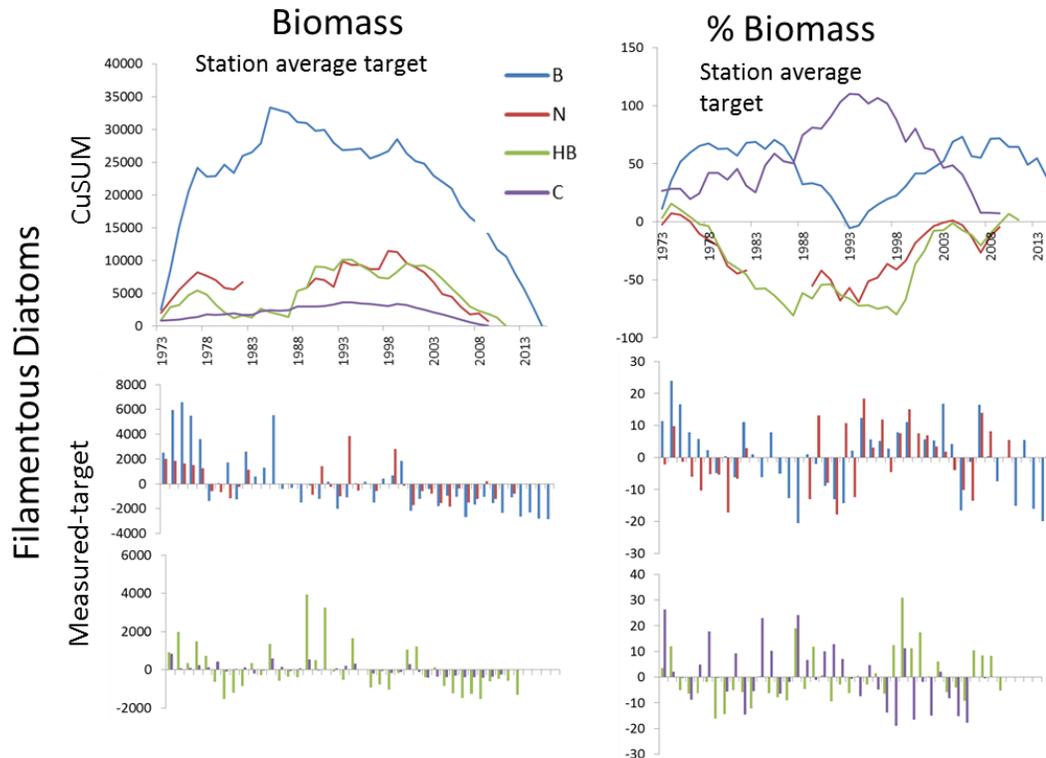


Figure 2.10: Left: Filamentous diatoms biomass; Right: Filamentous diatoms % biomass; Each set of plots shows in top panel: Cumulative sum of measure – target by year; lower panels: measured value – target value by year (0 line target = measured, hope for ≤ 0 values for these measures). Target used in calculations indicated as text on graphs.

2.2.2 Assessment of Target 2: Increase in food resource for grazers.

2.2.2a An Increase in the more edible phytoplankton taxa to over 30% of the biomass

ANOVA:

The upper bay site B shows a decreasing trend in the biomass of edible phytoplankton over time, from an average (± 1 SE) of 0.9 ± 0.2 g m⁻³ in stanza 1 (1972 – 1982) to 0.6 ± 0.04 g m⁻³ in Stanza 4 (2001 – 2015) which is significant at $p=0.002$ (Fig. 2.11). However, there has also been a concurrent and significant ($p=0.0047$) increase in the proportion of edible algae at B from $16.4 \pm 2.0\%$ in stanza 1 to

23.5 ± 1.8% in stanza 4 (Fig. 2.12, Table 2.4). At N, we also observe a similar decline in edible phytoplankton biomass from 1.1 ± 0.2 g m⁻³ in stanza 1 to 0.4 ± 0.04 in Stanza 4 and significant at p=0.0008 (Fig. 2.11). There was no significant difference in the relative composition at N; the proportion of edible algae hovered around 20% (Fig. 2.12).

With respect to the other long term monitoring sites, both HB and C showed significant declines in edible phytoplankton biomass since the 1970s (p<0.01; Fig. 2.11); however no significant changes were observed in relative composition (Fig. 2.12, Table 2.4).

Table 2.4 Stanza means (±SE), one way ANOVA significance value and comparison of means (Tukey-Kramer HSD) for phytoplankton metrics expected to increase.

Target	Measure	Site	Stza 1	Stza 2	Stza 3	Stza 4	p	Stanza			
			72-83	84-94	95-00	01-15		1	2	3	4
Edible Phytoplankton	% Biomass (>30%)	B	16.4 ± 2.0	15.2 ± 1.1	20.0 ± 2.3	23.5 ± 1.8	0.0047	B	B	AB	A
		N	18.9 ± 2.1	15.2 ± 1.1	20.5 ± 2.9	22.2 ± 1.0	ns				
		HB	22.9 ± 2.9	21.8 ± 2.5	18.8 ± 1.0	25.4 ± 2.0	ns				
		C	31.3 ± 2.3	29.8 ± 3.9	29.1 ± 1.6	33.8 ± 3.4	ns				
Chrysophyceae	% Biomass (>15%)	B	1.4 ± 0.3	1.4 ± 0.3	4.9 ± 0.4	5.8 ± 0.6	<0.0001	B	B	A	A
		N	0.9 ± 0.1	1.0 ± 0.2	2.0 ± 0.7	4.5 ± 0.3	<0.0001	B	B	B	A
		HB	1.2 ± 0.2	2.2 ± 0.7	1.5 ± 0.5	2.6 ± 0.3	ns				
		C	2.6 ± 0.4	4.2 ± 0.7	5.2 ± 0.6	10.5 ± 0.9	<0.0001	B	B	B	A

CuSUM:

With regards to edible phytoplankton taxa, the biomass CuSUM has the characteristic dome shape indicating consistent declines over time at all sites, both within and outside of the AoC (Fig. 2.13). This is expected given the reductions in total phosphorus concentrations since the 1970s (P abatement). In the upper bay at B, there has been a slight uptick in the biomass of edible phytoplankton post 2010 which is encouraging. The proportion of edible phytoplankton has levelled off at the upper bay site B but still remains below the target of 30% most years. At N, the proportion of edible phytoplankton has increased starting in 1999 and the trend has been a slow movement toward the target (Fig. 2.13). In the lower bay the relative contribution has been over 30% at various points in the 70s, 80s and it continues to fluctuate widely around the 30% target. The middle bay continues to move away from the target.

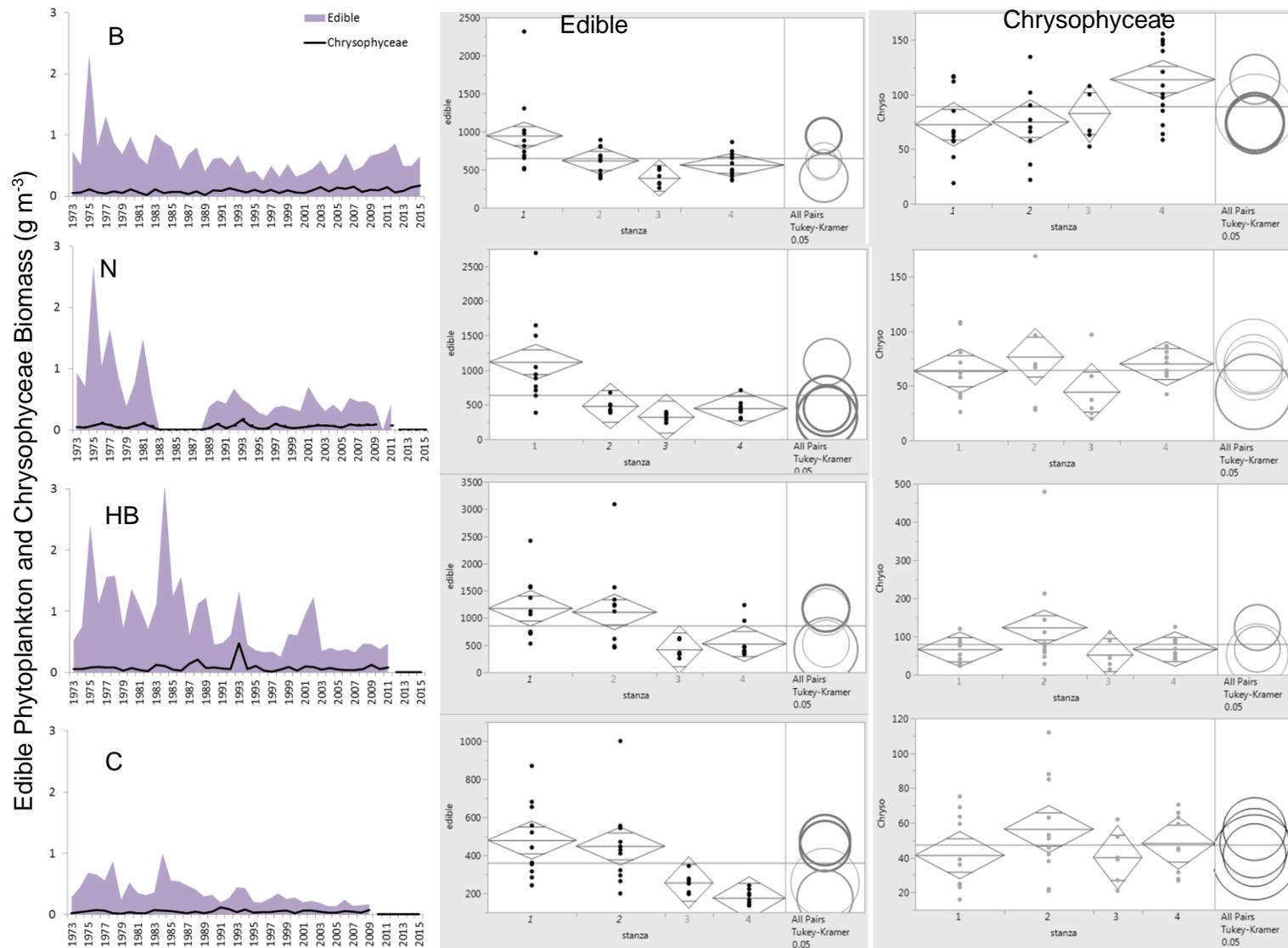


Figure 2.11: Left: Biomass (g m^{-3}) of highly edible phytoplankton (purple) and Chrysophyceae biomass (black line) by station. Centre: ANOVA of highly edible biomass by stanza (biomass in mg m^{-3}). Right: ANOVA of Chrysophyceae biomass by stanza (biomass in mg m^{-3}). Circles indicate range of and amount of overlap between time stanzas.

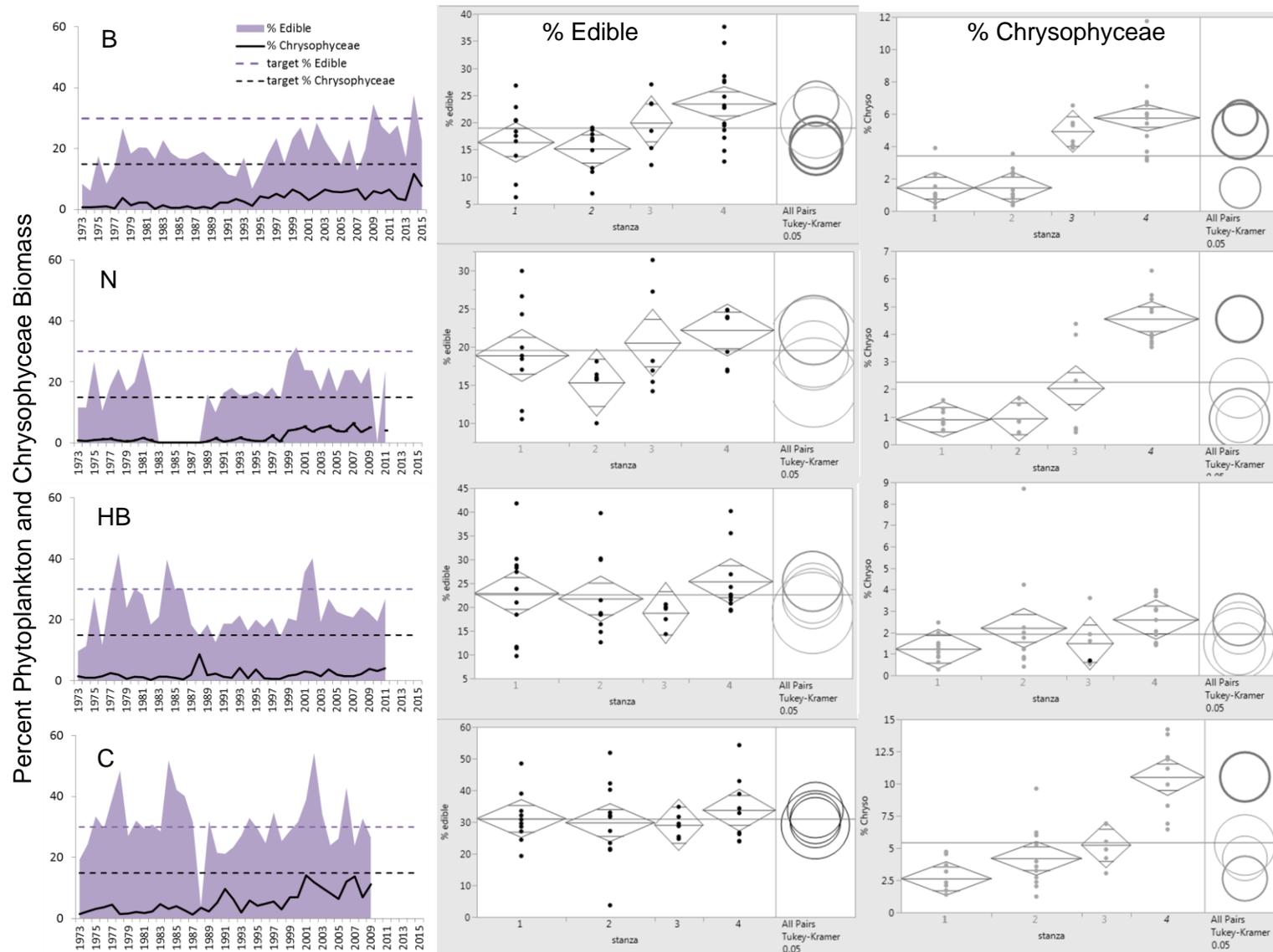


Figure 2.12: Left: Percent of total phytoplankton biomass of highly edible phytoplankton (purple) and Chrysophyceae (black line) by station. Centre: ANOVA of highly edible % biomass by stanza. Right: ANOVA of Chrysophyceae % biomass by stanza. Circles indicate range of and amount of overlap between time stanzas.

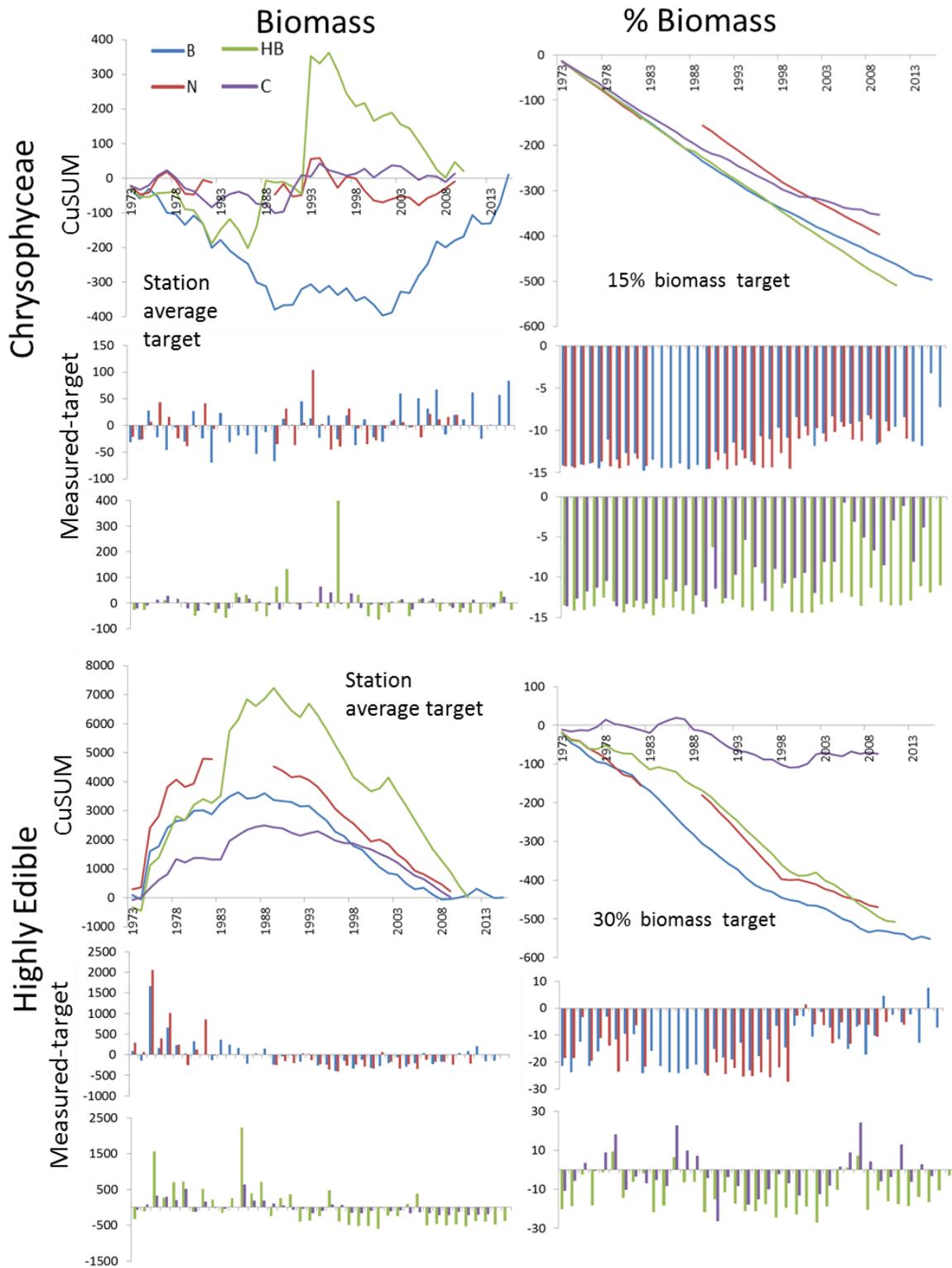


Figure 2.13: Top left: Chrysophyceae biomass; Top right: Chrysophyceae % biomass; Bottom left: Highly edible algae biomass; Bottom right: Edible algae % biomass. Each set of plots shows in top panel: Cumulative sum of measure – target by year; Lower panels: measured value – target value by year. Target used in CuSUM calculations indicated as text on graphs.

2.2.2b An increase in the biomass and relative composition of Chrysophyceae with an ideal target of > 15% of total biomass

Considering the upper bay sites, B showed a significant increase in Chrysophyceae biomass ($p < 0.01$) in stanza 4 (2001 – 2015) compared to stanza 1 and 2 with the current value of $0.1 \pm 0.01 \text{ g m}^{-3}$ (Fig. 2.11). Likewise, a significant increase in the relative composition of Chrysophyceae was observed for stanza 3 and 4 ($p < 0.0001$). For the most recent stanza (4), Chrysophyceae account for $5.8 \pm 0.6 \%$ compared to $1.4 \pm 0.3 \%$ in the first stanza (1972 – 1982) (Fig. 2.12, Table 2.4). N has not shown any significant change in Chrysophyceae biomass ($< 0.1 \text{ g m}^{-3}$) since the 1970s, but the relative composition has increased significantly to $4.5 \pm 0.3 \%$ in stanza 4 ($p < 0.0001$) compared to $0.9 \pm 0.1 \%$ in stanza 1 and similar values in the other stanzas (Fig. 2.11, 2.12).

No significant trends in either Chrysophyceae biomass or relative composition were detected at HB (Fig. 2.11, 2.12). However at C, a significant increase in the relative contribution of Chrysophyceae was detected in stanza 4 ($p < 0.0001$) although no change in biomass was detected (Fig. 2.11, 2.12).

CuSUM:

The CuSUM for B shows an increasing trend in Chrysophyceae biomass (inverted dome shape) and the residual plot shows that it is particularly evident since 2001 (Fig. 2.13). At the other sites, there is little long term change in the biomass of Chrysophyceae. The sole exception is the spike observed at HB in 1993, which was primarily species of *Mallomonas*. Zooplankton populations crashed in 1993 so it is likely it was not being eaten (Fig. 3.1). With respect to the % contribution of Chrysophyceae, the CuSUM plot shows that none of the stations are meeting the target and only C, influenced by Lake Ontario comes close (Fig. 2.13). Having said that, both upper bay sites are moving towards the target, following a trend that began in the late 1990s which is positive for the AoC. There is no obvious change at HB, which falls well below the target.

2.2.3 Assessment of Target Measures: Trend Analysis

The results of the short term vs long term trend analysis are presented in Figure 2.14. At the upper bay site B, 7 of 10 indicators fall in the upper right quartile showing long and short term improvements. The proportion of Chrysophyceae and the biomass of filamentous Cyanobacteria

showed the most improvement. Filamentous Cyanobacteria (%) and colonial Cyanobacteria biomass showed declines in the short term (upper left quartile). Only 1 measure, % colonial Cyanobacteria, falls completely in the bottom left quartile, indicating both long and short term decline.

Also in the upper bay, station N shows fewer positive trends in the short term along with several neutral measures (i.e. no change) over the last 5 years measured. Long and short term improvements are seen in the biomass and relative proportion of filamentous Cyanobacteria as well as the proportion of Chrysophyceae. The proportion of colonial Cyanobacteria appears to be getting worse in the long term; nuisance phytoplankton biomass, % filamentous diatoms and biomass of colonial Cyanobacteria appear to be slipping in the short term.

In the middle bay (HB), there is only one measure that has improved in the long and short term (% Chrysophyceae). Two measures (colonial Cyanobacteria biomass and % biomass) have declined in the long and short term. Although most measures have improved over the long term, 4 of them are in decline over the short term.

The lower bay is influenced by the upper bay but can also be affected by incursions of water from Lake Ontario. Any nutrient that is re-suspended or algal bloom that forms in the upper bay can be transported to the lower bay. While all measures (except colonial Cyanobacteria relative biomass) have improved over the long term at C, most are declining in the short term. Unlike the other stations, phytoplankton hasn't been assessed at C since 2009. Therefore the short term at this station doesn't match the other stations exactly. Given the anecdotal observations of blooms in the last few years, declines at C are suspected to have continued.

Phytoplankton indicators

- ◇ % Edible
- Nuisance(g)
- Filamentous Cyanobacteria (g)
- Colonial Cyanobacteria (g)
- ▼ Filamentous diatoms (g)
- △ % Chrysophyceae
- % Nuisance
- % Filamentous Cyanobacteria
- % Colonial Cyanobacteria
- ▼ % Filamentous diatoms

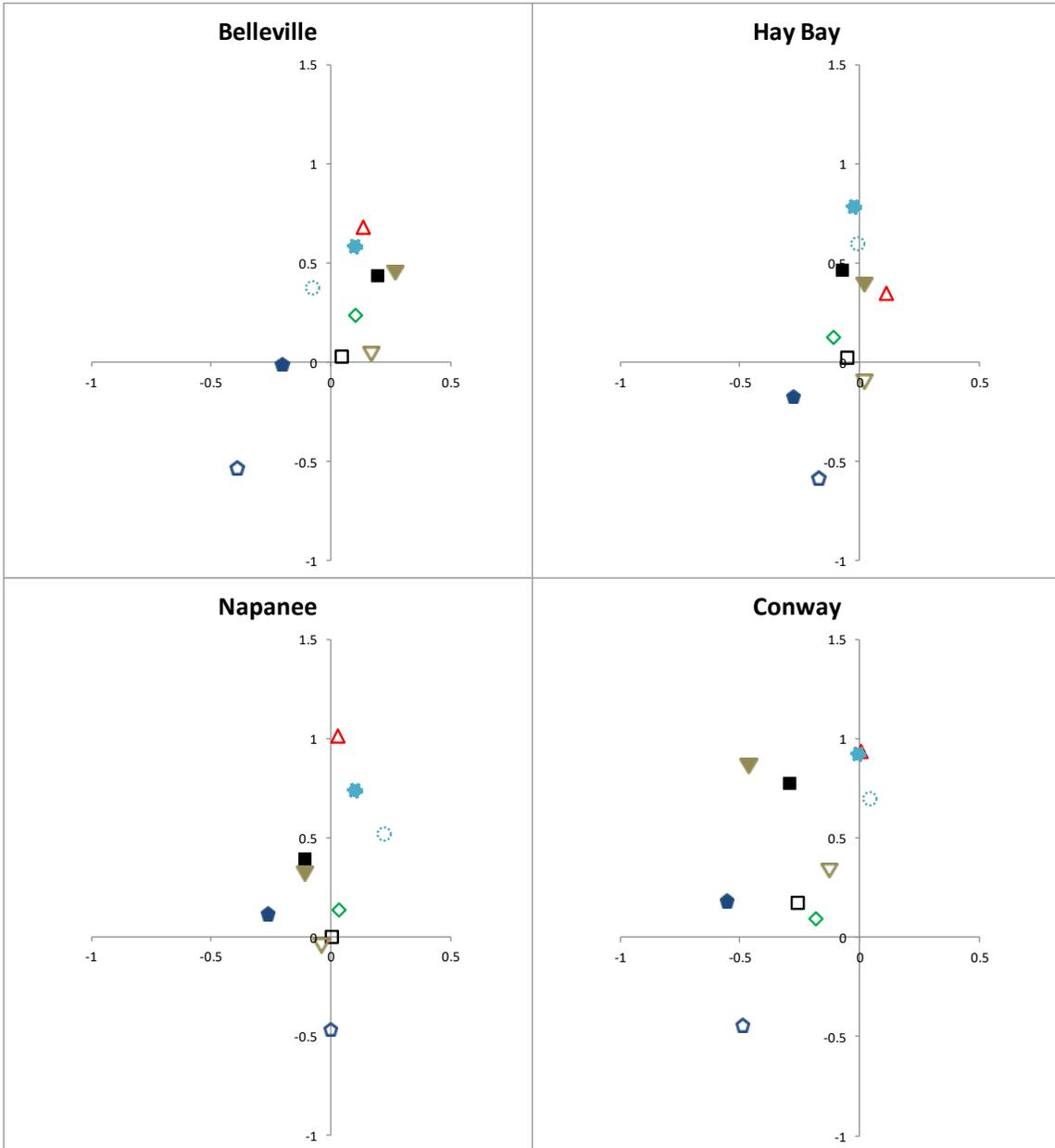
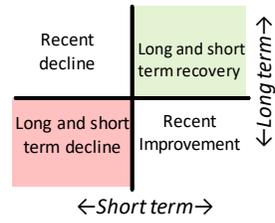


Figure 2.14: Plot phytoplankton metrics showing trends in the short term (x-axis, last 5 years relative to stanza 4 mean) and long term (y-axis, 5 year mean relative to overall mean). Note that short-term change at Conway is calculated differently since it was last sampled in 2009.

2.3 Conclusions: Phytoplankton Measures in the upper Bay of Quinte

The final assessment will focus on the long term monitoring sites of Belleville (B) and Napanee (N). Belleville was sampled consistently from 1973 – 2015 (n=43). Napanee was sampled from 1973-1982, 1989-2009, and 2011 (n=32). Long term trends refer to the entire data set for each station; short term trends refer to the last 5 years sampled from each site (B: 2011-2015; N: 2006-2009, 2011). Note that the start and end points (1973 and 2015) reflect the available data.

2.3.1 Conclusions regarding Target 1: Decrease in species that may impair food web function.

2.3.1a. A decrease in nuisance phytoplankton to less than 50% of the biomass (< 2.5 g m⁻³ in the upper bay offshore based on the existing target for BUI 8 of phytoplankton biomass of 4-5 g m⁻³)

The term nuisance algae can be applied to a variety of taxa, but in the Bay of Quinte we are referring specifically to those known to produce algal blooms. That includes filamentous diatoms (e.g. *Aulacoseira* spp.), filamentous Cyanobacteria (e.g. *Dolichospermum* spp.) and colonial Cyanobacteria (*Aphanocapsa* sp., *Cyanodictyon* sp., *Microcystis* sp.).

The average May – October mean biomass of the last time stanza (2001 – 2015) has not met the target of 2.5 g m⁻³ for nuisance algae at either B or N although both were near to it at 2.9 g m⁻³. This does represent a statistically significant decline in nuisance algal biomass at both sites from levels of >6.0 g m⁻³ observed in the 1970s. The CuSUM analysis for the upper bay sites also shows that nuisance algae biomass has been stabilizing around the target of 2.5 g m⁻³ since the late 1990s. The short term trend in nuisance phytoplankton is mixed. At B, this measure was met in 3 of the last 5 years (2012, 2013, and 2014) and the overall short term trend is positive. At N, the short term trend was negative and the target was met only once (2009) in the last 5 years measured.

Nuisance phytoplankton, however still consistently account for approximately 60% of the total biomass at both upper bay sites, well above the 50% target. At B, the short term trend has shown improvement but is driven by one unusually low year (2014) when nuisance phytoplankton accounted for only 42.3% of the total biomass; there is no long term trend. At N, there is no trend in relative composition in either the short or long term. It is worth noting that during the 2001 – 2015 time stanza there were 2 years where the target was met at B (≤ 50%) and another 2 that were close (≈53%); likewise, there was one year at N that was close to the

target ($\approx 53\%$). This indicates the target for nuisance phytoplankton is achievable even if it is not being consistently met.

Overall, the results for this measure are mixed. There is an improving long term trend at both upper bay sites towards reduced nuisance phytoplankton biomass and the target measure is certainly within reach. This speaks to the success of the phosphorus abatement program. However, the short term trend is improving at B, but declining at N. For % biomass of nuisance phytoplankton, there are no long term trends at either B or N. The short term trend at B is improving but skewed by a single observation, and there is no discernable short term trend at N. With continued phosphorus abatement, this measure is likely achievable.

2.3.1b A decreasing trend in filamentous and colonial Cyanobacteria (both in terms of biomass and relative composition)

Filamentous and colonial Cyanobacteria are both important components of the broader category of nuisance phytoplankton. The intent of this measure is to look at the overall trends in biomass and relative composition rather than trying to achieve hard targets. Many of the Cyanobacteria species observed in the upper and middle bay are nitrogen fixers that begin to emerge in large quantities during the latter part of July. This phenomenon is characteristic of phosphorus enriched, culturally eutrophic environments. Both nitrogen and phosphorus are essential for algal growth and in oligotrophic conditions phosphorus is usually the limiting element. But in eutrophic phosphorus-enriched environments, nitrogen becomes limiting and creates an ecological niche for N-fixing Cyanobacteria (e.g. Vollenweider, 1968).

For biomass of filamentous Cyanobacteria (primarily species of *Dolichospermum*) the short and long term trends are positive at both sites in the upper bay. At both upper bay sites biomass decreased sharply in 1995 (as seen in the CuSUM) and remained low; the ANOVA shows that Stanzas 3 and 4 (spanning 1995 – 2015) are significantly lower than stanzas 1 and 2 (1973 – 1994). The long term trend in relative biomass of filamentous Cyanobacteria is positive at both sites; however, B is showing a negative trend in the short term while N remains positive. Overall, the biomass and relative composition of filamentous Cyanobacteria appear to be declining over the long term.

Colonial Cyanobacteria (including species of *Microcystis*, *Gloeotrichia*, etc.) at both upper bay sites have not shown any long term trend in biomass since the 1970s; year after year the biomass has been more or less the same with the exception of a huge spike in biomass in 1998.

In the short term, the trend is negative at both sites, with 2007 being a particularly bad year for colonial and filamentous Cyanobacteria at B and N. For relative biomass, the long term trend is negative at both upper bay sites, while the short term trend is also negative at B. At N, no short term trend is evident. This means that colonial Cyanobacteria as a group have not significantly changed in biomass while almost all other types of phytoplankton have decreased and as a consequence, their proportion has increased over time.

Overall, the results of this measure are mixed. The trends in filamentous Cyanobacteria are encouraging with long term declines in biomass being the most important. For colonial cyanobacteria, there has been no change in biomass discernable by stanza and its proportion is either increasing or holding steady, but definitely not declining.

2.3.1c A decreasing trend in filamentous diatoms (biomass and % composition)

Filamentous diatoms, mainly *Aulacoseira* spp., have consistently formed the largest component of the nuisance algae and indeed the entire phytoplankton biomass since monitoring began. There are few exceptions. Like the measure for filamentous and colonial Cyanobacteria, the intent here is to consider the trend rather than focus on hard targets. With respect to biomass, the ANOVA shows a significant decline in biomass since the 1970s at both upper bay sites. Likewise, the CuSUM shows that filamentous diatom biomass has been below the long term average since 1999. Overall these are positive long term trends. The short term trend at B is positive (2011 – 2015); while N trends slightly negative (2006 – 2011). It is worth noting that filamentous diatom biomass is similar at B and N, so the most recent trends from N might also be positive (i.e. 2011 – 2015), but the samples have not been counted. The relative biomass of filamentous diatoms showed no long term trend at B and a slightly negative trend at N. The short term trend was positive at B and no trend was observed at N.

Again the results are mixed. The long term trend in filamentous diatom biomass is positive in the upper bay, since it has declined significantly over the past forty years which is attributable to phosphorus abatement. However, there has been little movement in % biomass over time as other types of phytoplankton also respond to synergistic ecosystem changes.

2.3.2 Conclusions regarding Target 2: Increase in food resource for grazers.

2.3.2a An increase in the more edible phytoplankton taxa to over 30% of the biomass

Edible algae are defined on the basis of size and shape with smaller unicellular forms being the most likely to be grazed by zooplankton (e.g. Cyr and Pace, 1992). A healthy food web must have a significant biomass of edible phytoplankton; we would anticipate that the proportion of edible algae will increase as a system recovers from eutrophy even if the actual biomass remains stable or shows a slight decline.

While the average proportion of edible phytoplankton in stanza 4 (2001 – 2015) at $\approx 23\%$ does not meet the target of 30% it is certainly in range and the target has been met once in the last 5 measures at B. At B, the ANOVA shows a significant increase in stanza 4 compared to stanza 1 and 2 with a similar but not significant pattern at N. The CuSUM analysis shows that both sites are moving towards the 30% measure. The short term trend remains positive at B, while N does not show any trend. These measures show that the upper bay is improving with an increasing proportion of edible algae apparent over the long term although the change is much more dramatic at B than at N.

2.3.2b An increase in the biomass and relative composition of Chrysophyceae with an ideal target of over 15% of total biomass

Chrysophyceae are a class of phytoplankton that are particularly important in the transfer of autochthonous production in oligotrophic systems and often used as an indicator of ecosystem health (Munawar et al. 1978; Nicholls et al. 1986). For a system like the Bay of Quinte, with a long history of eutrophication, we would expect that the proportion of Chrysophyceae would increase over time if the system is in a recovery phase. Chrysophyceae have been shown to account for approx. 15 – 35 % of the total phytoplankton biomass in some of the more efficiently operating food webs in the Great Lakes.

The proportion of Chrysophyceae has increased significantly at both upper bay sites since the 1970s. However, at current levels (stanza 4) of 5.8% at B and 4.5% at N, it falls well below the 15% target. Similarly, the CuSUM plot is linear and not approaching the target, which suggests that reaching the target may be a far-off goal. Having said that, both B and N have improved considerably since the 1970s on this measure and the short and long term trends are encouraging.

2.3.4 Summary

It is clear from the results that although the targets are not completely being met in the 2001 – 2015 time period, the long term trends are mainly positive at both stations (results summarized in Tables 2.5 – 2.8). Additionally, the short term trend is better at B than at N, but this could be an artifact of the mismatch in dates analysed. Overall the targets for phytoplankton composition in the Bay of Quinte Area of Concern suggest that the phytoplankton community continues to be impaired but the overall trends show an improvement in most of the measures. As indicated in section 2.2, conditions have generally deteriorated in the middle and lower bay, especially in the short term.

Table 2.5: Assessment of station B with regards to target 1: decrease in nuisance and eutrophic (and noxious) indicator species that may impair food web function. Asterisk (*) denotes statistical significance. Numbers refer to stanzas.

Measure		Meeting Target? Stanza 4 Average	Meeting Target? # out of Last 5 Samples	Long Term Trend	Short Term Trend
Nuisance Phytoplankton	Biomass (<2.5 g m ⁻³)	No (2.9 ± 0.2)	3 of 5	Positive* (4 <1,2)	Positive
	% Biomass (<50%)	No (59.4 ±1.9)	1 of 5	No Trend (4 = 1)	Positive
Filamentous Cyanobacteria	Biomass	N/A	N/A	Positive * (4,3<1,2)	Positive
	% Biomass	N/A	N/A	Positive * (4,3<1,2)	Negative
Colonial Cyanobacteria	Biomass	N/A	N/A	No Trend	Negative
	% Biomass	N/A	N/A	Negative * (4>1,2)	Negative
Filamentous Diatoms	Biomass	N/A	N/A	Positive * (4<1)	Positive
	% Biomass	N/A	N/A	No Trend	Positive

Table 2.6: Assessment of station B with regards to target 2: increase in phytoplankton taxa that are likely to provide a food resource for zooplankton and other grazers. Asterisk (*) denotes statistical significance.

Measure		Meeting Target? Stanza 4 Average	Meeting Target? # out of Last 5 Samples	Long Term Trend	Short Term Trend
Edible Phytoplankton	% Biomass (30%)	No (23.5 ± 1.8)	1 of 5	Positive * (4>1,2)	Positive
Chrysophyceae	% Biomass (10%)	No (5.8 ± 0.6)	0 of 5	Positive * (4,3 >1,2)	Positive

Table 2.7: Assessment of station N with regard to target 1; decrease in nuisance and eutrophic (and noxious) indicator species that may impair food web function. Asterisk(*) denotes statistical significance.

Measure		Meeting Target? Stanza 4 Average	Meeting Target? # Out Of Last 5 Samples	Long Term Trend	Short Term Trend
Nuisance Phytoplankton	Biomass (<2.5 g m ⁻³)	No (2.9 ± 0.3)	1 of 5	Positive*(4 <1)	Negative
	% Biomass (<50%)	No (60.1 ±1.1)	0 of 5	No trend (4 = 1)	Neutral
Filamentous Cyanobacteria	Biomass	N/A	N/A	Positive * (4,3<1,2)	Positive
	% biomass	N/A	N/A	Positive * (4,3<1,2)	Positive
Colonial Cyanobacteria	Biomass	N/A	N/A	No Trend	Negative
	% biomass	N/A	N/A	Negative * (4,3>1,2)	Neutral
Filamentous Diatoms	Biomass	N/A	N/A	Positive (4<1)	Slight Negative
	% biomass	N/A	N/A	Slight Negative	Neutral

Table 2.8: Assessment of station N with regard to target 2; increase in phytoplankton taxa that are likely to provide a food resource for zooplankton and other grazers. Asterisk(*) denotes statistical significance.

Measure		Meeting Target? Stanza 4 Average	Meeting Target? # Out Of Last 5 Samples	Long Term Trend	Short Term Trend
Edible Phytoplankton	% biomass (30%)	No (22.2 ± 1.0)	0 of 5	Slight Positive	Neutral
Chrysophyceae	% biomass (10%)	No (4.5 ± 0.3)	0 of 5	Positive * (4 >1,2,3)	Slight Positive

2.4 Target Measures, Phosphorus and Nitrogen

Phosphorus and nitrogen are important elements that regulate phytoplankton growth. In culturally eutrophic environments like the Bay of Quinte, elevated phosphorus concentrations promote phytoplankton growth to the point where nitrogen (in the form of nitrate) becomes depleted and in turn promotes the growth of nitrogen fixing Cyanobacteria. As a consequence, efforts to control eutrophication have focussed on reducing phosphorus to limit phytoplankton growth and circumvent the possibility of Nitrogen limitation (e.g. Vollenweider, 1968; Schindler, 1978). The Bay of Quinte has been subject to a phosphorus management strategy since 1978 and it remains one of the key management activities of the Remedial Action Plan to control eutrophication. While nitrogen is not specifically part of the strategy, it is worth considering how both N and P relate to the phytoplankton community target measures.

Total phosphorus (TP) and total Kjeldahl nitrogen (TKN) are conservative indicators in that they are present and detectable even if bioavailable forms have been depleted. Part 1 of this document (Section I) showed that total phytoplankton biomass had a strong positive relationship with both TP and TKN. The revised target measures, which focus on different aspects of the phytoplankton community (Target 1: nuisance algae, colonial/filamentous Cyanobacteria, filamentous diatoms, Target 2: edible algae, Chrysophyceae) should also respond in a similar manner. The point here is to show which, if any, of these measures behave differently from total phytoplankton biomass in relation to TP or TKN. Most measures responded in a positive linear fashion to both TP and TKN (Tables 2.9 and 2.10). The exceptions were % edible algae (Fig. 2.15) and % Chrysophyceae (Fig. 2.16) which responded negatively to both increasing TP and increasing TKN; this is expected as both were predicted to improve with decreasing nutrient concentrations. Colonial Cyanobacteria showed no response at all to either nutrient (Fig. 2.17).

Table 2.9. Relationships between phytoplankton target measures (y axis) and Total Kjeldahl Nitrogen (x axis) slope and r^2 values. All lines are significant ($p < 0.0001$).

Target	units	Regression equation	r^2
Nuisance phytoplankton	Biomass	Nuisance = $-3394.092 + 13325.757 \cdot \text{TKN}$	0.69
	% biomass	%nuisance = $19.0173 + 61.2898 \cdot \text{TKN}$	0.49
Filamentous Cyanobacteria	Biomass	filamentousBG = $-1531.707 + 5214.57 \cdot \text{TKN}$	0.44
	% biomass	% filamentousBG = $-1.685 + 33.296 \cdot \text{TKN}$	0.24
Filamentous diatoms	Biomass	Fil Diatoms = $-2519.43 + 9332.1857 \cdot \text{TKN}$	0.58
	% biomass	% fil Diatoms = $16.1758 + 39.5455 \cdot \text{TKN}$	0.25
Edible phytoplankton	Biomass	edible = $-104.252 + 1388.913 \cdot \text{TKN}$	0.25
	% biomass	% edible = $39.3458 - 30.6267 \cdot \text{TKN}$	0.34
Chrysophyceae	% biomass	Log(% Chrysophytes) = $2.475 - 3.128 \cdot \text{TKN}$	0.32

Table 2.10: Relationships between phytoplankton targets (y axis) and Total Phosphorus (x axis) slope and r^2 values.

Target	units	Regression equation	r^2
Nuisance phytoplankton	Biomass	Nuisance = $-623.9322 + 129576.35 \cdot \text{TP}$	0.65
	% biomass	%nuisance = $120.06454 + 19.3529 \cdot \text{Log}(\text{TP})$	0.52
Filamentous Cyanobacteria	Biomass	filamentous BG = $-329.9065 + 47161.552 \cdot \text{TP}$	0.36
	% biomass	% filamentous BG = $6.04822 + 299.352 \cdot \text{TP}$	0.19
Filamentous diatoms	Biomass	Fil Diatoms = $-769.7992 + 96469.911 \cdot \text{TP}$	0.62
	% biomass	% fil Diatoms = $24.0989 + 393.481 \cdot \text{TP}$	0.24
Edible phytoplankton	Biomass	edible = $101.642 + 15997.281 \cdot \text{TP}$	0.33
	% biomass	% edible = $31.9521 - 266.910 \cdot \text{TP}$	0.25
Chrysophyceae	% biomass	Log(% Chrysophytes) = $1.8233 - 30.385 \cdot \text{TP}$	0.30

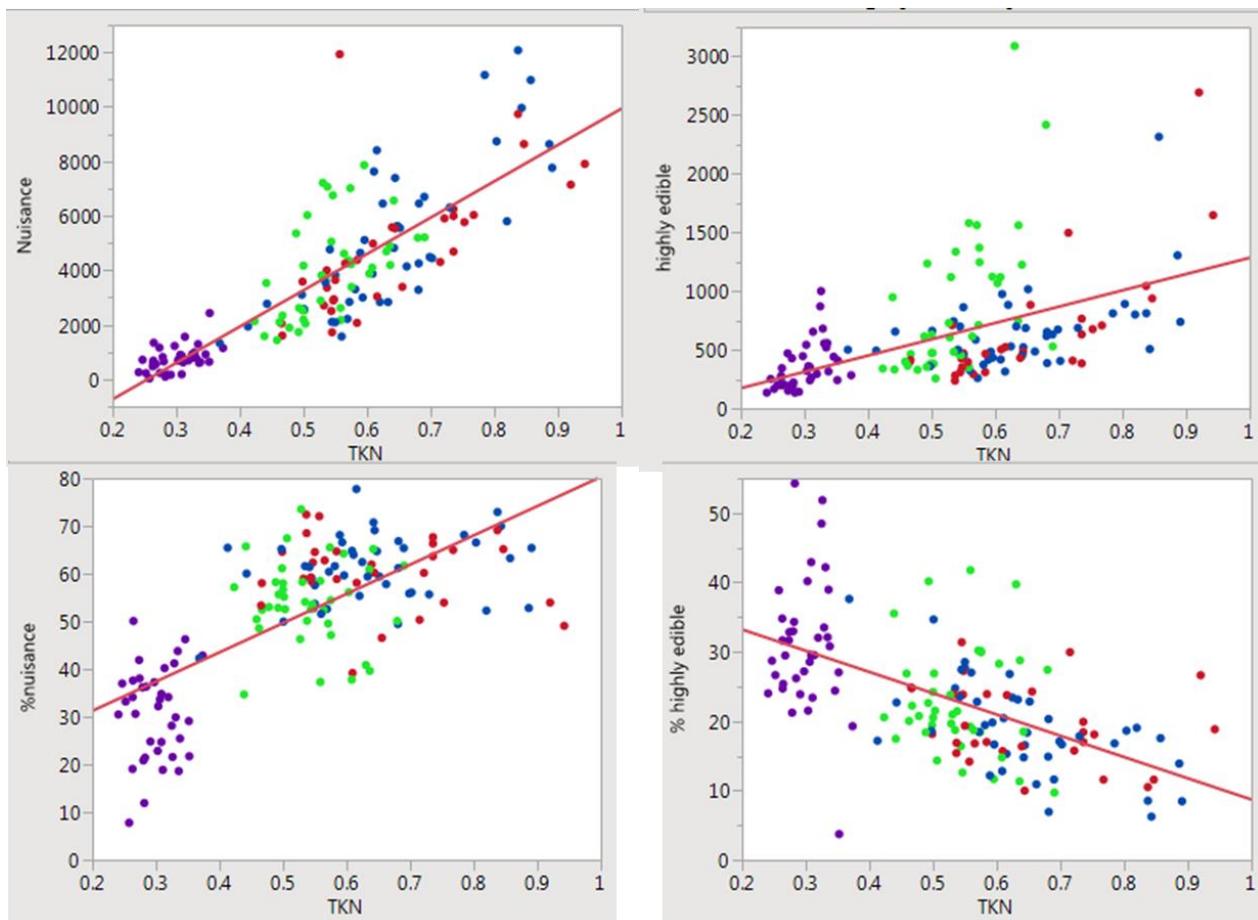


Figure 2.15: Nuisance (left) and edible (right) biomass (mg m^{-3} , upper) and % biomass (lower) by total Kjeldahl nitrogen concentration (mg m^{-3}). Stations indicated by colour: blue=B, red=N, green=HB, purple=C.

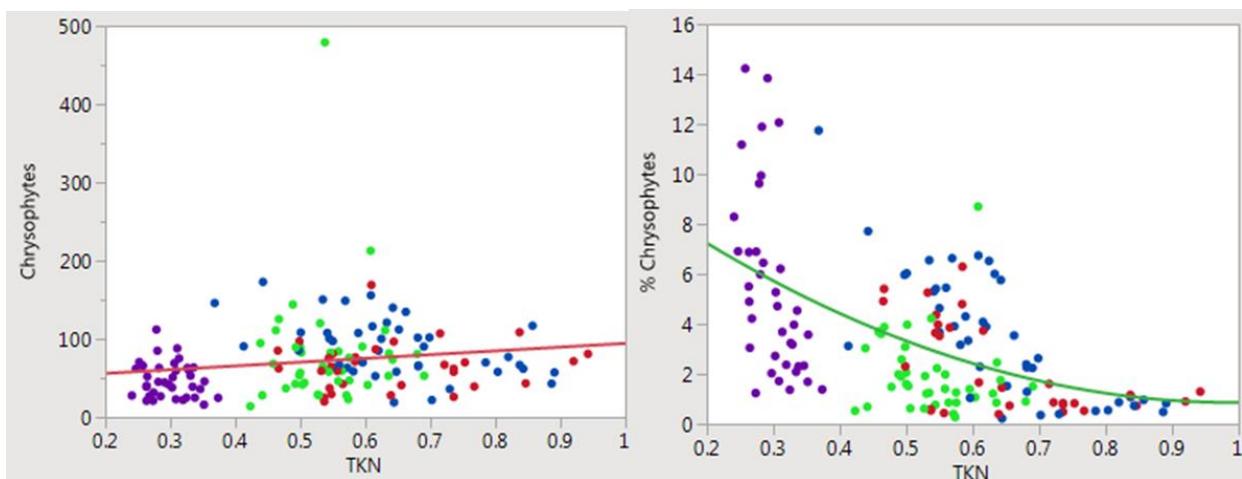


Figure 2.16: Chrysophyceae biomass (mg m^{-3} , left) and % biomass (right) by total Kjeldahl nitrogen concentration (mg m^{-3}). Stations indicated by colour: blue=B, red=N, green=HB, purple=C.

With respect to TKN, the strongest correlations (Table 2.9) were observed between: nuisance phytoplankton biomass ($r^2=0.69$), filamentous diatom biomass ($r^2=0.58$), % nuisance algae ($r^2=0.49$) and filamentous Cyanobacteria biomass ($r^2=0.44$). Similarly, the strongest correlations with TP (Table 2.10) were: nuisance phytoplankton biomass ($r^2=0.65$), filamentous diatom biomass ($r^2=0.62$) and % nuisance algae ($r^2=0.52$). These groupings represent species that tend to thrive in eutrophic conditions. The remaining correlations (r^2) ranged from 0.2 – 0.36 for measures associated with both TP and TKN.

Not all measures responded in a linear fashion to nutrients. For % Chrysophyceae, we observed an inverse curvilinear relationship between both TP and TKN (Fig. 2.16, TP data not shown). Historic work in the Bay of Quinte has reported similar results regarding % Chrysophyceae and TP (Nicholls et al. 1986). Similar relationships with % Chrysophyceae and TP have also been observed in European lakes (Sandgren 1988; Sandgren and Walton 1995). These and other studies have not found significant relationships between Chrysophyceae biomass and TP (e.g. Watson et al. 1997, Sandgren 1988). One reason for this apparent contradiction is that Chrysophyceae tend to be grazed more as nutrient loadings increase and any response to nutrients tends to be muted. Measures that do not respond linearly to nutrient levels remind us that the phytoplankton community is affected by multiple factors; simple measures of nutrients cannot tell the whole story.

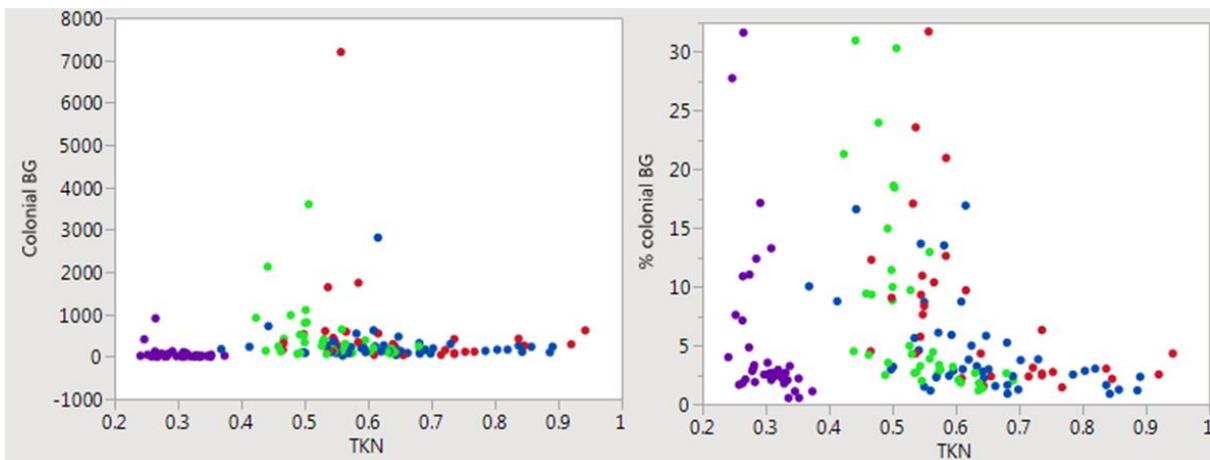


Figure 2.17: Colonial Cyanobacteria biomass (mg m^{-3} , left) and % biomass (right) by total Kjeldahl nitrogen concentration (mg m^{-3}). Stations indicated by colour: blue=B, red=N, green=HB, purple=C.

2.5 Future Monitoring

Taxonomic analysis of phytoplankton samples can be time consuming and costly. One of the attempts to reduce costs in Project Quinte was to produce a single composite sample by pooling 13 samples collected biweekly from May – October. We showed in our analyses that this approach tends to overestimate the importance of larger commonly occurring taxa and underestimate smaller and less frequently occurring taxa. Given the error associated with the composite sample approach, we do not recommend this approach be taken.

An alternative approach would be to reduce the sampling frequency to have fewer samples analyzed. We considered 4 possible scenarios:

1. One sample per year (mid-August);
2. Seasonal sampling (3X per year, early June, late August, early October);
3. Monthly (6 samples, May – October, mid-month);
4. Biweekly (13 samples May –October).

For this exercise, we repeated the 1 way ANOVA measures from our assessment of phytoplankton composition targets at B, only we altered the sampling frequency for each set of ANOVAs. All measures were assessed for long term changes based on sampling intervals of once per year, once per season, monthly and biweekly (control). The question we are asking is: would we have achieved the same results with a different sampling frequency?

Table 2.11 summarizes these results. It is apparent that monthly sampling will produce the same result as biweekly sampling but the trend is lost when sampling frequency is reduced to seasonal or annual scales. Given these results, we recommend that phytoplankton samples be collected at least monthly and counted individually to obtain seasonal average values that can be used to assess each Target measure.

Table 2.11: Analysis of Phytoplankton target measures (ANOVA test of differences) using different sampling frequencies. Yes * indicates significant differences. NS indicates not significant.

Measure	unit	Sampling frequency			
		Once	Seasonally	Monthly	Biweekly
Nuisance algae	biomass	NS	NS	yes *	yes *
Nuisance algae	% biomass	NS	NS	NS	NS
Fil Cyanobacteria	biomass	yes *	yes *	yes *	yes *
Fil Cyanobacteria	% biomass	yes *	yes *	yes *	yes *
Colonial Cyanobacteria	biomass	NS	NS	NS	NS
Colonial Cyanobacteria	% biomass	-	-	-	-
Filamentous diatoms	biomass	NS	yes*	yes*	yes*
Filamentous diatoms	% biomass	NS	NS	yes*	yes*
Edible algae	biomass	NS	NS	yes *	yes *
Edible algae	% biomass	NS	NS	yes *	yes *
Chrysophyceae	biomass	NS	NS	NS	NS
Chrysophyceae	% biomass	NS	NS	yes *	yes *

Appendix: *Phytoplankton data available by station and year. I= May to October individual date samples counted, C= composite sample counted*

	B	N	HB	C
1973	I	I	I	I
1974	I	I	I	I
1975	I	I	I	I
1976	I	I	I	I
1977	I	I	I	I
1978	I	C	C	C
1979	I	C	C	C
1980	I + C	C	C	C
1981	I + C	C	I + C	C
1982	C	C	C	C
1983	C		C	C
1984	C		C	C
1985	C		C	C
1986	C		C	C
1987	C		C	C
1988	C		I + C	I + C
1989	C	C	I + C	C
1990	I + C	I + C	I + C	I + C
1991	I + C	C	C	C
1992	I + C	C	C	C
1993	I + C	C	C	C
1994	I + C	C	C	C
1995	I + C	C	C	C
1996	I + C	C	C	C
1997	I + C	C	C	C
1998	I + C	C	C	C
1999	I + C	I + C	C	C
2000	I + C	I + C	I + C	I + C
2001	I + C	I + C	C	I + C
2002	I + C	I + C	C	I + C
2003	I + C	I + C	C	I + C
2004	I + C	I + C	C	I + C
2005	I + C	I + C	C	I + C
2006	I + C	I + C	C	I + C
2007	I + C	I + C	C	I + C
2008	I + C	I + C	C	I + C
2009	I + C	I + C	C	I + C
2010	I + C		I + C	
2011	I + C	I + C	I + C	
2012	I			
2013	I			
2014	I			
2015	I			
	B	N	HB	C
I total	35	18	12	17
C total	32	27	34	32
overlap	24	13	7	12

Section III: Quinte BUI 13 – Zooplankton Assessment

Current BUI Status: Impaired

BUI#13 Delisting Target:

“Demonstrate that the upper Bay of Quinte phytoplankton and zooplankton community structure do not differ significantly from an unimpaired Great Lakes / control site of comparable physical and chemical characteristics.”

3.1 Zooplankton Background Information:

Zooplankton fill a key role in the aquatic food web as they form an important link between primary producers and larger heterotrophs such as predatory invertebrates and planktivorous fishes. As a result, zooplankton are subject to both top-down and bottom-up drivers. In the Bay of Quinte, zooplankton have been sampled biweekly from early May to October at three monitoring stations [upper bay – Belleville (B), middle bay – Hay Bay (HB) and lower bay – Conway (C)] since 1975 (Fig. 1.1). Details of this sampling program are given in Cooley et al. (1986), Bowen and Johannsson (2011), Johannsson and Bowen (2012) and each of the Project Quinte Annual Reports (e.g., Bowen and Rozon, 2017). Monitoring of zooplankton tracks species composition, size structure, abundance, biomass and productivity in an effort to understand the response of the community to changes in the controlling factors and assess its ‘health’ as a component of the ecosystem.

The Bay of Quinte is a productive environment for zooplankton, and their densities and biomass values in are high compared to other Great Lakes systems. In Stanza 4, total May to October biomass averaged $137 \pm 18 \text{ mg m}^{-3}$ in the upper bay, and $45 \pm 12 \text{ mg m}^{-3}$ in the lower bay. The upper bay biomass is about three times higher than Saginaw Bay in Lake Huron (adapted from Pothoven et al. 2013), two times higher than western Lake Erie, and 2.5 to 5 times higher than coastal sites in Lake Ontario (K. Bowen, DFO, unpublished data). Cladocerans dominate the zooplankton community in the Bay of Quinte (Fig. 3.1), with the highest percentages in the upper bay. When averaged over the season for the 1995 to 2014 period, cladocerans comprised 79.5% of total zooplankton biomass at B, 74.8% at HB and 54.1% at C. The most dominant cladocerans are the bosminids *Bosmina* and *Eubosmina*, *Daphnia retrocurva*, *D. galeata mendotae* and *Chydorus sphaericus*. Cyclopoid copepods have generally declined in the Bay of Quinte since the 1980s (Fig. 3.1), and over the 1995 to 2014 period comprised on average 12.1% of biomass at B, 17.7% at HB and 22.0% at C. The most dominant cyclopoids

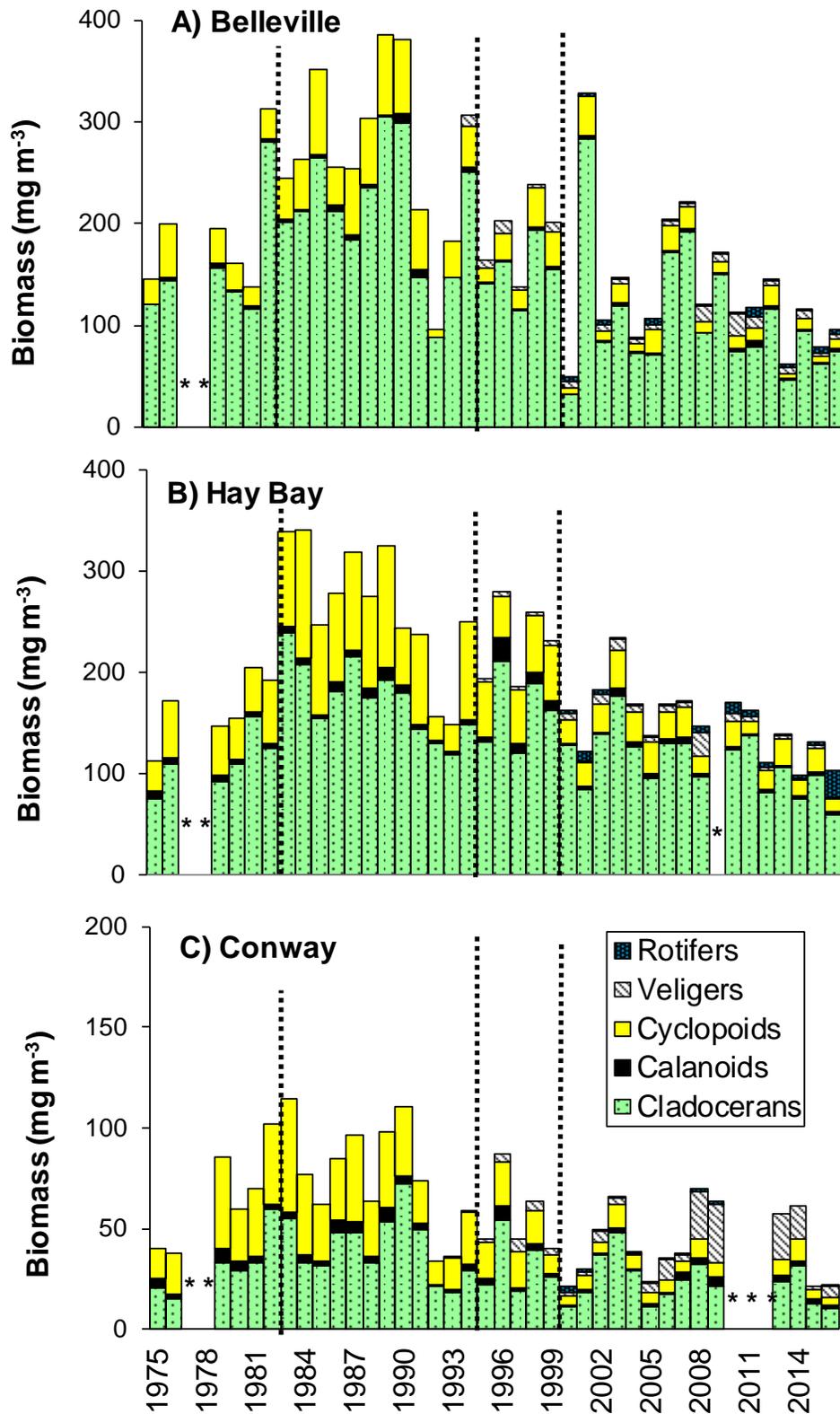


Figure 3.1: May to October mean biomass of dominant zooplankton groups in the Bay of Quinte. Dotted lines separate the time stanzas, and * represent years that were not sampled.

are *Mesocyclops edax*, *Tropocyclops extensus* and *Acanthocyclops vernalis* in the upper and middle reaches. *A. vernalis* declines down the length of the bay and is replaced by *Diacyclops thomasi* in the deeper waters. Calanoid copepods are generally uncommon in the upper bay (1.6%) and middle bay (3.4%) as they tend to prefer cooler, deeper systems. At C they comprise 6.6% of total biomass. Dreissenid mussels invaded the system in the mid-1990s, and now dreissenid veliger larvae are common in the zooplankton, especially in the lower bay. Other zooplankton that have invaded the system include the predatory fishhook water flea *Cercopagis* (first detected in 1998) and the spiny water flea *Bythotrephes* (first detected in 2006). Both of these species are most common in the lower bay.

3.2 Determining the Suitability of Zooplankton Indicators

Stanza Description

To examine changes in the Bay of Quinte zooplankton over time, zooplankton data were split into the following time stanzas:

- 1) *Turbid Phase – High Phosphorus* (1975 – 1982; 1977 and 1978 were not sampled)
- 2) *Turbid Phase – Peak Walleye pre-dreissenid, post-nutrient control* (1983 – 1994),
- 3) *Clear Phase – Clear Transition* (1995 – 1999)¹ – expanded macrophyte coverage,
- 4) *Clear Phase – Climate Variability* (2001 – 2015) – recent times characterized by increased variability in precipitation and temperature and the widespread establishment of zebra mussels in the upper bay. This time stanza includes impacts from both invasive *Dreissena* and the predatory cladocerans *Cercopagis* and *Bythotrephes*.

For all tests we used annual May to October mean values. Initially, as a data exploration exercise at each of the three stations, regression analyses were carried out for each of the potential zooplankton measures for the entire 1975 to 2014 time series.

Methods for Regression Analyses of Zooplankton Indicators

A number of studies have assessed the usefulness of zooplankton as indicators of environmental conditions in a variety of temperate aquatic systems, particularly with respect to trophic status (e.g., Attayde and Bozelli, 1998; Čeirāns, 2007; Gannon and Stemberger, 1978; Gulati, 1983). To identify the effectiveness of the zooplankton indicators chosen for the Bay of Quinte BUI13 outlined in Currie et al. (2023), we performed a suite of least squares regression analyses of our long-term data at the three main stations, B, HB and C using Systat V11 (Table 3.1). In addition to the chosen indicators, we included many other zooplankton measures to look

¹ 2000 was excluded from analyses due to extraordinary cool and rainy conditions driving down zooplankton populations in the upper bay. For example there was nearly a complete recruitment failure in *Daphnia* that year.

for relationships with eutrophication parameters (Table 3.2). The zooplankton indicators or metrics selected for the BUI assessment are shown in bold. These results will help assess the status of the Quinte zooplankton community, and potentially identify other useful crustacean zooplankton and rotifer indicators that could be used in the future. It will also identify insensitive zooplankton parameters that do not clearly respond to environmental changes.

Note that percentages of individual cladoceran taxa, total cyclopoids and total calanoids represent the biomass of that group relative to total crustacean biomass, whereas the percentages for individual calanoid and cyclopoid taxa are relative to adult cyclopoid and adult calanoid biomass, respectively. These dependent zooplankton variables were regressed against the following independent variables: fish planktivore biomass (as defined in Currie et al. 2023), predatory cladoceran biomass (sum of *Bythotrephes*, *Cercopagis*, *Leptodora* and *Polyphemus*), several eutrophication parameters (secchi depth, TP, TKN, chlorophyll, total phytoplankton biomass, edible phytoplankton biomass), total primary production (post-1987), edible (2-20 μm) primary production (post-1999), bacterial biomass (post-1999), May to August mean air temperature, and May to August total precipitation.

The regression analyses were repeated using only stanza 1 and stanza 3 data (see stanza description below) as there were fewer confounding factors such as higher densities of invasive dreissenid mussels and predatory cladocerans that occurred in stanza 4. These analyses were done for each station separately, all stations combined, and B and HB combined (Table 3.2). Note that when all three stations are combined, we cover a larger trophic gradient of conditions, but there are also habitat differences between the warm, shallow upper bay, and the deeper, cooler, thermally stratified lower bay. Some taxa may also be responding to these changes in habitat. Due to these confounding habitat influences when the lower bay was included, we chose to focus mainly on relationships at B and HB.

It must be noted however that correlation does not necessarily mean causation. Interpretation is also challenged by the fact that both eutrophication (bottom-up influences) and planktivory (top-down influences) were high in the early period and declined together over time. High planktivory is often associated with eutrophic systems, and domination by the typical assemblage of planktivores (e.g. Alewife, White Perch, Gizzard Shad) is generally not considered to be a desired fish community. A preferred fish community is one where piscivores (usually considered “desirable” species such as Walleye, bass species and Northern Pike) are high enough to keep

the planktivore numbers in check, resulting in a lessened impact on the zooplankton community. More abundant large herbivorous zooplankton should then be able to graze down phytoplankton to a greater degree. This is the top-down trophic cascade response described by Carpenter et al. (1985) and Mazumder (1994) and is described in more detail in Section 1.

Table 3.1: Least squares regression results (slopes and r^2 values) for each zooplankton indicator correlated with independent variables for planktivory and eutrophication. Only data from stanza 1 (1975 to 1982) and stanza 3 (1995 to 1999) were used. Positive relationships are green and negative ones are pink. Note that total and edible phytoplankton biomass data were not consistently available at HB and C (shaded in grey). NS indicates the regression was not significant and "all" represents the three stations combined. * indicates an outlier was removed.

	stn	Planktivore Biomass		Secchi Depth		Total Phosphorus		TKN		Chlorophyll		Edible Phyto Biomass		Total Phyto Biomass	
		slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2
% <i>D. galeata mendotae</i>	B	-0.38	0.51		NS	-603	0.50	-100	0.38	-1.49	0.44		NS		NS
	HB	-0.62	0.59	18.40	0.59	-855	0.47	-136	0.31	-1.30	0.50				
	C		NS		NS		NS		NS		NS				
	B HB	-0.38	0.43	16.62	0.33	-569	0.39	-73	0.25	-1.32	0.44		NS		NS
	all	-0.26	0.25		NS		NS		NS		NS		NS		
Cladoceran mean size	B	-0.005	0.63	0.280	0.40	-7.2	0.85	-1.3	0.73	-0.017	0.66	-1E-05	0.67	-1E-04	0.67
	HB		NS	0.08	0.47	-5.2	0.53	-0.1	0.66	-0.007	0.70				
	C	-0.002	0.68	0.033	0.53	-10.1	0.76	-1.5	0.72	-0.020	0.63				
	B HB	-0.003	0.52	0.125	0.58	-6.0	0.59	-0.8	0.55	-0.011	0.58	-1E-04	0.68	-1E-04	0.62
	all	-0.003	0.50		NS	-3.2	0.35			-0.006	0.28	-1E-04	0.64	-1E-04	0.45
% Large <i>Daphnia</i>	B	-0.005	0.88		NS	-10.4	0.88	-1.7	0.81	-0.022	0.65				
	HB		NS		NS	-13.6	0.85	-1.8	0.70		NS				
	C	0.006	0.83	-0.063	0.56	-26.8	0.68		NS	0.048	0.77				
	B HB	-0.005	0.60	0.160	0.35	-9.6	0.80	-1.1	0.61	-0.017	0.55		NS	-1E-04	0.71
	all		NS		NS	-3.7	0.20		NS		NS		NS		NS
<i>Chydorus</i> Biomass	B	0.11	0.49	-9.92	0.50	183	0.54	37	0.60	0.43	0.44		NS	0.001	0.57
	HB		NS		NS		NS		NS		NS				
	C		NS		NS		NS		NS		NS				
	B HB	0.07	0.18	-3.63	0.21	121	0.22	17	0.18	0.29	0.29		NS	0.001	0.48
	all	0.07	0.15	-2.04	0.35	172	0.49	20	0.52	0.39	0.53		NS	0.001	0.61
% <i>A. vernalis</i>	B	0.50	0.56		NS	822	0.59	150	0.55	1.73	0.41	0.03	0.65		NS
	HB		NS		NS	690	0.43	103	0.39		NS				
	C		NS		NS		NS		NS		NS				
	B HB	0.51	0.48	-18.23	0.24	896	0.60	142	0.61	1.32	0.28	0.019	0.33		NS
	all	0.44	0.37	-7.42	0.29	820	0.69	90	0.63	1.47	0.46	0.02	0.34		NS
% <i>Mesocyclops</i>	B	-0.55	0.43		NS	-716	0.29	-153	0.46		NS		NS		NS
	HB	-1.00	0.50	34.10	0.66	-1565	0.51	-252	0.53	-2.72	0.73				
	C		NS	2.12	0.42		NS	-111	0.72		NS				
	B HB	-0.53	0.33	28.07	0.37	-737	0.26	-105	0.21	-2.12	0.46	-0.027	0.50		NS
	all		NS		NS		NS	59.7	0.14		NS	-0.024	0.33		NS
Shannon Diversity Index	B	-0.005	0.58	0.390	0.47	-7.28	0.52	-1.2	0.37	-0.022	0.69	-2E-04	0.55		NS
	HB	-0.014	0.67	0.410	0.63	-22.5	0.74	-3.3	0.65	-0.033	0.69				
	C		NS		NS		NS		NS		NS				
	B HB	-0.007	0.52	0.410	0.61	-11.6	0.54	-1.7	0.46	-0.029	0.67		NS		NS
	all	-0.005	0.21		NS		NS		NS		NS		NS		NS

Table 3.2: Least squares linear regression results for various zooplankton parameters correlated with fish planktivore biomass and eutrophication parameters (Secchi depth, TP, TKN and chlorophyll a). The percentages of individual cladoceran taxa, total calanoids and total cyclopoids were relative to total crustacean zooplankton biomass, whereas the percentages of individual cyclopoid and calanoid taxa were relative to total adult cyclopoid and calanoid biomass, respectively. BM represents biomass and blank cells indicate there was no significant regression. The Daphnia ratio is the ratio of *D. galeata mendotae* to *D. retrocurva* biomass, and the copepod ratio is the ratio of calanoid to cyclopoid biomass. Log TR is log transformed trophic ratio (calanoid biomass to cyclopoid plus cladoceran biomass).

Zooplankton Parameter	Belleville		Hay Bay		Conway		all stations		B and HB	
	Fish Plank ^a	Eutro. ^b								
Cladocerans										
Daphnia BM	-		-	---	-			+	-	--
% Daphnia	-	---	-	---	-	---	-		-	--
<i>D. galeata</i> BM	-	--	-	---			-		-	---
% <i>D. galeata</i>	-	--	-	---			-		-	---
<i>D. retrocurva</i> BM				-	-	-		+++		
% <i>D. retrocurva</i>					-	---				
Daphnia ratio*	-	---	-	---			-		-	---
Cladoceran length	-	---		---	-	---	-	--	-	---
Daphnia length	-	--		-					-	--
% Daphnia >0.75 mm	-	--		--	+	+		-	-	---
% Daphnia >1.0 mm	-	--		--			-		-	--
<i>Bosmina</i> BM				+						
% <i>Bosmina</i>			+	++						
<i>Eubosmina</i> BM							+	+++		+
% <i>Eubosmina</i>		+		++			+	+++	+	+++
<i>Eubos</i> : <i>Bos</i> *							+	++	+	++
<i>Bosminid</i> BM			+	+			+	+++		+
<i>Chydorus</i> BM	+	+++					+	+++	+	+++
% <i>Chydorus</i>	+	+++		++			+	+++	+	+++
<i>Diaphanosoma</i> BM							+	+++	+	++
% <i>Diaphanosoma</i>						-		+		
<i>Ceriodaphnia</i> BM				-						--
% <i>Ceriodaphnia</i>						--				
Cyclopoids										
Cyclopoid BM						+		++		
% Cyclopoid	+	+						---		
<i>Diacyclops</i> BM				-		+		---		--
% <i>Diacyclops</i>								---		
<i>A. vernalis</i> BM		++			+	++	+	+++	+	++
% <i>A. vernalis</i>	+	++		++			+	+++	+	+++
<i>Mesocyclops</i> BM			-	---					-	---
% <i>Mesocyclops</i>	-	--	-	---		--		+	-	---
<i>Tropocyclops</i> BM					+	+++				
% <i>Tropocyclops</i>			+	++				+		
cyclopoid nauplii BM								+		

Zooplankton Parameter	Belleville		Hay Bay		Conway		all stations		B and HB	
	Fish Plank ^a	Eutro. ^b								
Calanoids										
Calanoid BM										--
% Calanoid								---		--
Eurytemora BM										
% Eurytemora										
<i>S. oregonensis</i> BM	-	-		--			-		-	---
% <i>S. oregonensis</i>		-	-	---	-		-	---	-	---
<i>L. siciloides</i> BM										
% <i>L. siciloides</i>			+	+++				+++		
% minutus			+				+			
calanoid nauplii BM	-	--		---	-	--	-	-	-	---
copepod ratio							-	---		--
log Trophic Ratio								---		--
Diversity Index	-	---	-	---			-		-	---

* one outlier removed

^a Fish Planktivore Biomass = trawl plus gill net biomass from OMNR

^b **Eutrophic Indicators**

+++	negative for secchi and positive for TP, TKN, chlorophyll
++	indicates two or three of the above correlations were significant
+	indicates one of the above correlations were significant
---	positive for secchi and negative for TP, TKN, chlorophyll
--	indicates two or three of the above correlations were significant
-	indicates one of the above correlations were significant

Summary of Zooplankton Correlation Analyses

The following summarize the results of the zooplankton correlation analyses for B and HB combined, using Stanzas 1 and 3 data only. Taxa in bold and marked with * are the chosen indicators:

Increase with higher fish planktivory and eutrophy:

- **Chydorus biomass*** and % *Chydorus*
- % *Eubosmina*
- *Eubosmina* to *Bosmina* ratio
- *Diaphanosoma* biomass
- *Acanthocyclops vernalis* biomass and % ***Acanthocyclops vernalis****

Decrease with higher fish planktivory and eutrophy:

- *Daphnia* biomass and % *Daphnia*
- *Daphnia galeata mendotae* biomass² and % ***Daphnia galeata mendotae*^{2*}**
- *D. galeata* to *D. retrocurva* ratio²
- **Mean cladoceran length***
- Mean *Daphnia* length
- % very large *Daphnia* (>1.0 mm in length)
- % **large *Daphnia**** (>0.75 mm in length)
- *Mesocyclops* biomass and % ***Mesocyclops*^{3*}**
- *Skistodiaptomus oregonensis* biomass and % *Skistodiaptomus oregonensis*
- Calanoid nauplii biomass
- **Diversity Index***

Weakly increase with higher eutrophy but insensitive to planktivory:

- *Eubosmina* biomass

Decrease with increased eutrophy but insensitive to planktivory:

- *Ceriodaphnia* biomass
- *Diacyclops* biomass
- Total Calanoid biomass and % Total Calanoid
- Calanoid to Cyclopoid biomass ratio
- Log Trophic Ratio

² When all stations were combined, *D. galeata* responded negatively to planktivory, but not trophic status

³ When all stations were combined, % *Mesocyclops* showed a weak positive response to eutrophy (TKN only). This species prefers the warm shallower conditions in the upper bay. Although it showed a negative response to higher eutrophy within each station over time, it is likely mostly responding to planktivory.

Relatively insensitive to trophic state and planktivory:

- *Daphnia retrocurva* biomass⁴ and *Daphnia retrocurva* %⁴
- *Bosmina* biomass and % *Bosmina*⁵
- % *Dlaphanosoma*
- % *Ceriodaphnia*
- Total Cyclopoid biomass⁶ and % Total Cyclopoid⁵
- % *Diacyclops*⁵
- *Tropocyclops* biomass and % *Tropocyclops*
- Cyclopoid nauplii biomass
- *Eurytemora* biomass and % *Eurytemora*
- *Leptodiptomus siciloides* biomass and % *Leptodiptomus siciloides*⁷
- % *Leptodiptomus minutus*

It should be noted that several parameters that were relatively insensitive to changes in trophic status in the Bay of Quinte have been used as indicators in other systems. For example, the small cladoceran *Bosmina* has been used as an indicator for many years, but it is also plagued by taxonomic uncertainties (Gannon and Stemberger, 1987). While *Bosmina* is one of the most dominant zooplankton taxa in Quinte and its densities are high relative to more open water areas, it did not correlate with nutrients or chlorophyll. Another indicator commonly utilized is the trophic ratio, the ratio of calanoids to cyclopoids plus cladocerans (Gannon and Stemberger, 1987), but in Quinte it shows no trophic relationships within stations. The gradient observed in trophic ratio down the bay more likely relates to habitat differences.

⁴ When all stations were combined, *D. retrocurva* biomass showed a strong positive response to eutrophy, as it tends to be more abundant in the upper and middle bays. However, at B over time, it showed no response, and at HB, it showed a small negative response (secchi only). At C, *D. retrocurva* biomass and % *D. retrocurva* showed a stronger negative response to both eutrophy and planktivory.

⁵ % *Bosmina* showed a positive response to planktivory and eutrophy only at HB

⁶ When all stations were combined, cyclopoid biomass showed a positive response to eutrophy, whereas % cyclopoids, *Diacyclops* BM and % *Diacyclops* showed a strong negative response. This is because the proportion of cyclopoids, particularly *Diacyclops* is highest in the less eutrophic lower bay.

⁷ When all stations were combined, % *L. siciloides* showed a strong positive response to eutrophication. This was also seen at HB, but not the other stations. This species tends to prefer warm eutrophic embayments and it is sometimes used as an indicator of eutrophy.

ANOVA Results – Station and Stanza Differences

For each of the zooplankton parameters tested above, significant differences among stations for each of the time stanzas (Table 3.3), and stanza differences for each of the stations (Table 3.4) were determined using ANOVA (Systat 11). A p-value ≤ 0.05 was considered to be significant. Results for the chosen indicators (bold) are described in more detail in the assessments that follow in Section 3.3.

Cumulative Sum Analyses

As part of the assessment of each of the chosen zooplankton indicators, we ran a CuSum (cumulative sum) analysis at each station to observe changes in the annual data over time. To calculate the CuSum, the assigned weight “ ω ” (in this case, the average for each indicator for each station from 1975 – 2015) was subtracted from each annual mean and then the cumulative sum of the ‘annual mean- ω ’ was plotted. This analysis tends to smooth the data and make longer term trends more apparent.

Each plotted CuSum shows deviation from the mean, either positive or negative. As deviation away from the mean increases, the slope of the plotted lines increases. Similarly, as deviation away from the mean decreases, the slope of the line approaches zero. When values are below the mean, the CuSum slope becomes negative and when values are above the mean the CuSum slope becomes positive.

When a change in deviation from the mean occurs, there is a delay in the change in CuSum direction since the new values must account for the cumulative sum of opposing values from previous years. Once an indicator begins to stabilise, this is reflected in the CuSum slope approaching zero. However, if an indicator shows slipping towards a previous state (i.e., poor community health), the CuSum will cease to plateau and will once again increase or decrease depending on how the indicator is deviating away from the mean. CuSum results for the chosen indicators are described in more detail in the assessments that follow in Section 3.3.

Rotifer Regression Analyses

Rotifers were collected at B, HB and C in the Bay of Quinte starting in 2000 (stanza 4 only). Although rotifers were collected on each sampling date, only annual seasonal composites were enumerated due to budgetary constraints and taxonomist availability. We repeated the crustacean zooplankton correlation analyses described above to examine relationships between

rotifer taxa and eutrophic parameters in the Bay of Quinte (Table 3.5). These analyses included all but the rare rotifer taxa for each of the three stations, plus all three stations combined. The group “eutrophic species” is the density sum of *Anuraeopsis fissa*, *Brachionus angularis*, *Filinia* sp., *Keratella cochlearis tecta*, *Pompholyx sulcata* and *Trichocerca* sp. (Gannon and Stemberger, 1978).

There were fewer significant correlations for individual stations, as the trophic gradients were not as great within stations as they were across stations (Tables 3.6 and 3.7). When all stations were combined and the trophic gradient was greater, the following taxa were identified as potential indicators. Those in bold with an * were chosen as the best rotifer indicators.

Rotifers which increase with higher eutrophy:

- Density *Keratella cochlearis*
- Density *Pompholyx sulcata* and % ***Pompholyx sulcata****
- Density *Trichocerca cylindrica* and % *Trichocerca cylindrica*
- Density *Trichocerca multicroinis* and % *Trichocerca multicroinis*
- Density *Trichocerca rousseleti* and % *Trichocerca rousseleti*
- Density *Trichocerca pusilla* and % *Trichocerca pusilla*
- Density *Trichocerca* sp. and % ***Trichocerca* sp.***
- Density eutrophic species and % **eutrophic species***

Rotifers which decrease with higher eutrophy:

- % ***Kellicottia longispina****
- % ***Synchaeta kitina****

In summary, the best indicators for increasing eutrophic conditions in the Bay of Quinte appear to be higher percentages of “eutrophic species”, *Trichocerca* sp. and *Pompholyx sulcata*, and for decreasing trophy we would expect to see increasing percentages of *Kellicottia longispina* and *Synchaeta kitina*. As all the *Trichocerca* species in the Bay of Quinte respond positively to eutrophy, using the genus level (*Trichocerca* sp.) rather than species examines this group as a whole.

Table 3.3: Significant differences (shaded cells) among Project Quinte stations for each zooplankton parameter and time stanza examined using ANOVA. NS indicates no significant difference ($p>0.05$). * Bold zooplankton parameters were chosen as metrics for the BUI assessment. BM represents biomass and “–” indicates that parameter was not tested.

Taxa	Stanza 1		Stanza 2		Stanza 3		Stanza 4	
	BM	% BM						
Cladocera	C< B, HB	C< HB<B	C< B, HB	C< HB<B	NS	C< B, HB	C< B, HB	C<HB<B
<i>Daphnia</i> sp.	C, HB<B	C, HB<B	C< B, HB	C< B, HB	C< B, HB	C< B	C< B, HB	NS
<i>Daphnia galeata</i>*	NS	NS	C< B, HB	C< B, HB	C< HB	NS	NS	NS
<i>Daphnia retrocurva</i>	C, HB<B	C, HB<B	C< B, HB	NS	NS	NS	C< B, HB	NS
<i>Bosmina</i>	NS	B<C	NS	B, HB<C	B, C<HB	NS	C< B, HB	NS
<i>Eubosmina</i>	C< HB<B	C< HB<B	C< HB<B	C, HB<B	C< B, HB	C<B	C<B	HB<B
<i>Chydorus</i>*	C< B, HB	C< B, HB	C< B, HB	C< B, HB	NS	NS	C<HB	NS
predatory cladocerans	C<HB	--	NS	--	NS	--	C< B, HB	--
ratio <i>Eubos</i> : <i>Bosmina</i>	NS	--	C, HB<B	--	C, HB<B	--	NS	--
Bosminid biomass	C< B, HB	--						
ratio DGM:retro	NS	--	C<B	--	NS	--	NS	--
Cladocera size*	NS	--	C< B, HB	--	NS	--	NS	--
<i>Daphnia</i> mean length	NS	--	C< B, HB	--	C< B, HB	--	NS	--
% Large <i>Daphnia</i>*	NS	--	NS	--	C< B, HB	--	NS	--
% <i>Daphnia</i> >1.0 mm	NS	--	C<B	--	C< B	--	NS	--
Total Cyclopoid	NS	B<HB<C	B, C<HB	B<HB<C	B, C<HB	B, HB<C	C<B<HB	B<HB<C
Cyclopoid nauplii	B, C<HB	--	B, C<HB	--	C<HB	--	C< B, HB	--
<i>Diacyclops</i>	B, HB<C	B<HB<C	B, HB<C	C< B, HB	B, HB<C	B, HB<C	B<HB<C	B, HB<C
<i>Tropocyclops</i>	NS	B, C<HB	NS	NS	NS	NS	C< B, HB	C< B, HB
<i>Mesocyclops</i>*	C<B	C<B	C< B, HB	C< B, HB	C<HB	C< B, HB	C<HB	C< B, HB
<i>A. vernalis</i>*	HB, C<B	C<B	C<B	C< B, HB	C< B	C< B	C<B	C<B
Total Calanoid	B<HB	B<C	B, C<HB	B, HB<C	B, C<HB	B<HB, C	B<HB	B, HB<C
Calanoida nauplii	NS	--	NS	--	NS	--	NS	--
<i>Eurytemora affinis</i>	NS	HB, C<B	NS	NS	NS	NS	NS	NS
<i>Epischura lacustris</i>	NS	NS	NS	NS	NS	NS	B, HB<C	B, HB<C
<i>L. minutus</i>	NS							
<i>L. sicilis</i>	NS	B, HB<C	NS	B<C	NS	NS	NS	NS
<i>L. siciloides</i>	NS	B, C<HB	NS	NS	NS	NS	NS	NS
<i>L. macrurus</i>	NS	B, HB<C	NS	NS	NS	NS	NS	NS
<i>S. oregonensis</i>	NS	NS	NS	B<C	NS	NS	NS	NS
copepod ratio	NS	--	NS	--	NS	--	B, HB<C	--
log Trophic Ratio	B<HB,C	--	B<HB<C	--	B<HB,C	--	B<HB<C	--
Diversity Index*	NS	--	NS	--	NS	--	C<HB	--
Rotifers	--	--	--	--	--	--	C<HB	--
Rotifer to Zoopl. Ratio	--	--	--	--	--	--	NS	--
Veligers	--	--	--	--	NS	HB<C	NS	C< B, HB

Table 3.4: Significant differences (shaded cells) among Project Quinte stanzas for each zooplankton parameter and station examined using ANOVA. NS indicates no significant difference ($p>0.05$). * Bold zooplankton parameters were chosen as indicators for the BUI assessment. BM represents biomass and “-” indicates that parameter was not tested.

Taxa	Biomass			% by Biomass		
	B	HB	C	B	HB	C
Cladocera	3,4<2	1,4<2	4<2	NS	1,2<4	1,2<4
<i>Daphnia</i> sp.	4<2	1,4<2,3	NS	NS	1<3,4	1,2<3,4
<i>Daphnia galeata</i>*	NS	1<3	NS	NS	1<3,4	1,2<4
<i>Daphnia retrocurva</i>	4<1, 2	1, 4 <2	4 < 2	4<1	NS	1,2<3
<i>Bosmina</i>	NS	NS	4<1, 2	1,2<4	2<4	NS
<i>Eubosmina</i>	4<1, 2; 3<2	3,4<2	NS	4<1,2	4<1,2	NS
<i>Chydorus</i>*	3,4<1,2	NS	NS	NS	NS	2<4
predatory cladocerans	NS	2,3,4<1	1,2,3<4	-	-	-
ratio <i>Eubos</i> : <i>Bosmina</i>	4<1	4<2	NS	-	-	-
Bosminid biomass	3,4<2	NS	4<1; 3,4<2	-	-	-
ratio DGM:retro	NS	NS	1,2<4	-	-	-
Cladocera size*	NS	1<2,3,4; 4<3	1<3,4; 2<4	-	-	-
<i>Daphnia</i> mean length	NS	NS	2<4	-	-	-
% Large <i>Daphnia</i>*	NS	NS	2<1	-	-	-
% <i>Daphnia</i> >1.0 mm	NS	NS	NS	-	-	-
Total Cyclopoid	1,3,4<2	1,3,4<2	3,4<1,2	4<2	4<1,2; 3<2	4<1,2,3; 3<1
Cyclopoid nauplii	4<2	4<2	4<1,2,3	-	-	-
<i>Diacyclops</i>	1,3,4<2	4<2	4<2, 2,3,4 <1	NS	NS	4<1,2,3
<i>Tropocyclops</i>	NS	NS	4<2	4<1,2	4<1,2	NS
<i>Mesocyclops</i>*	4<2	1<2,3	NS	NS	1<2,3,4	1,2,3<4
<i>A. vernalis</i>*	4<2	4<2	4<1,2,3	4<1,2	4<1,2	NS
Total Calanoid	NS	4<3	NS	NS	2<3	2<4
Calanoida nauplii	1,2<3,4	4<3; 1,2<3,4	1<3,4; 2<4	-	-	-
<i>Eurytemora affinis</i>	NS	NS	NS	NS	NS	NS
<i>Epischura lacustris</i>	NS	NS	2<4	NS	NS	1,2<4
<i>L. minutus</i>	NS	NS	NS	NS	NS	NS
<i>L. sicilis</i>	NS	NS	NS	NS	NS	NS
<i>L. siciloides</i>	NS	NS	NS	NS	4<1,2	NS
<i>L. macrurus</i>	NS	NS	2,3,4<1	NS	NS	2,3,4<1
<i>S. oregonensis</i>	NS	1,2,4<3	NS	1,2<3,4	1<3	1<2,3
copepod ratio	2<4	2<3,4	1,2<4	-	-	-
log Trophic Ratio	NS	2<3	NS	-	-	-
Diversity Index*	NS	1<3	NS	-	-	-

Table 3.5: Least squares linear regression results for Quinte rotifer taxa (density and percent composition in terms of density) against a suite of independent eutrophic variables, including inverse Secchi depth, TP, TKN chlorophyll and 2-20 μm primary production. Taxa in bold show the most promise as indicators for eutrophy.

Rotifer Taxon	Rotifer Density				% Composition			
	B	HB	C	all	B	HB	C	all
<i>Ascomorpha ovalis</i>	+		-	+		--		
<i>Asplanchna</i> sp.								
<i>Conochilus unicornis</i>				+				
<i>Filinia longiseta</i>		+		+		+		+
<i>Filinia</i> sp.				+				+
<i>Gastropus stylifer</i>					-	+		
<i>Kellicottia longispina</i>								--
<i>Keratella cochlearis</i>	+	+		+++		++		
<i>Keratella cochlearis tecta</i>				+	+			+
<i>Keratella quadrata</i>								
<i>Polyarthra dolichoptera</i>								
<i>Polyarthra major</i>		-				-		
<i>Polyarthra remata</i>		+						
<i>Polyarthra</i> sp.		+						
<i>Polyarthra vulgaris</i>		+			-			
<i>Pompholyx sulcata</i>	++	+		++	++	+		++
<i>Synchaeta kitina</i>		+						---
<i>Synchaeta pectinata</i>					-	--		
<i>Synchaeta</i> sp.		+						
<i>Trichocerca cylindrica</i>				++		++		+++
<i>Trichocerca multicroinis</i>				+++	++	+		++
<i>Trichocerca pusilla</i>	++		+	+++	++	+		+++
<i>Trichocerca rousseleti</i>	+	+		++	++	+		++
<i>Trichocerca</i> sp.	+	+	+	+++	++			+++
Eutrophic Species	+	+	+	++	++	++		+++

Eutrophic parameters tested were inverse secchi, TP, TKN, chlorophyll and 2-20 μm production

+++	positive for 4-5 of the eutrophic parameters
++	positive for 2-3 of the eutrophic parameters
+	positive for 1 of the eutrophic parameters
---	negative for 4-5 of the eutrophic parameters
--	negative for 2-3 of the eutrophic parameters
-	negative for 1 of the eutrophic parameters

Table 3.6: Least squares regression results (slopes and r^2 values) for potential rotifer indicator taxa (density) and independent variables for planktivory and eutrophication for stanza 4 (2000 to 2015). Blank cells were not significant. "All" indicates all three stations were combined. Positive relationships are green and negative ones are pink.

Density (No.L ⁻¹)	stn	Planktivore Biomass		Secchi		Total Phosphorus		TKN		Chlorophyll		DOC		Bacteria Biomass		2-20 production	
		slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2
<i>Keratella cochlearis</i>	B									5.91	0.26						
	HB															4.98	0.77
	C																
	all	1.44	0.1	-9.01	0.16	1302	0.13	145.1	0.23	3.73	0.27	9.62	0.11			2.67	0.34
<i>Pompholyx sulcata</i>	B					391	0.35			1.24	0.71						
	HB															1.28	0.95
	C																
	all					211	0.25	16.9	0.18	0.55	0.36						
<i>Trichocerca cylindrica</i>	B																
	HB																
	C																
	all							11.1	0.12	0.42	0.31						
<i>Trichocerca multirinis</i>	B																
	HB																
	C																
	all			-0.49	0.12	99	0.21	9.0	0.2	0.29	0.42						
<i>Trichocerca pusilla</i>	B					238	0.33	30.0	0.36	0.72	0.62						
	HB																
	C			-0.25	0.49												
	all	0.12	0.2	-0.56	0.13	129	0.31	11.8	0.31	0.31	0.39	0.78	0.17				
<i>Trichocerca rousseleti</i>	B									2.15	0.56						
	HB															0.38	0.81
	C																
	all					249	0.15	24.8	0.16	0.72	0.24						
<i>Trichocerca sp.</i>	B									5.89	0.11						
	HB							81.8	0.30								
	C			-2.32	0.42												
	all			-4.16	0.28	901	0.41	80.6	0.4	2.35	0.28	4.84	0.27				
Eutrophic Species	B									8.23	0.68						
	HB															2.83	0.84
	C			-2.44	0.41												
	all					1314	0.24	107.0	0.24	3.3	0.39						

Table 3.7: Least squares regression results (slopes and r^2 values) for potential rotifer indicator taxa (percent composition by density) and independent variables for planktivory and eutrophication for stanza 4 (2000 to 2015). Blank cells were not significant. "All" indicates all three stations were combined.

percent composition by density	stn	Planktivore Biomass		Secchi		Total Phosphorus		TKN		Chlorophyll		DOC		Bacteria Biomass		2-20 production		
		slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	slope	r2	
<i>Kellicottia longispina</i>	B														1E-05	0.34		
	HB																	
	C																	
	all	-0.09	0.22	0.52	0.26	-68.8	0.19			-0.12	0.15	-0.7	0.28					
<i>Pompholyx sulcata</i>	B					208	0.43			0.4	0.32							
	HB														0.28	0.26	0.81	
	C																	
	all					105	0.29	7.72	0.2	0.22	0.3							
<i>Synchaeta kitina</i>	B																	
	HB																	
	C																	
	all			1.16	0.15	-167	0.15	-14.7	0.13	-0.35	0.14							
<i>Trichocerca cylindrica</i>	B																	
	HB			-1.77	0.31					0.13	0.29							
	C																	
	all			-0.22	0.15	0.016	0.18	3.27	0.17	0.12	0.45							
<i>Trichocerca multirinis</i>	B					57.5	0.35	7.12	0.35	0.16	0.51							
	HB									0.24	0.54							
	C																	
	all					36.2	0.19	3.15	0.17	0.1	0.31							
<i>Trichocerca pusilla</i>	B					119	0.43	13.9	0.4	0.22	0.3	1.41	0.28	-0.001	0.4			
	HB																	
	C									12.6	0.37							
	all	0.01	0.16	-0.34	0.14	63.6	0.3	6.68	0.29	0.15	0.26	0.52	0.21	-0.001	0.14			
<i>Trichocerca rousseleti</i>	B							23	0.35	0.49	0.47							
	HB							22.3	0.34			2.06	0.36					
	C																	
	all					85.65	0.23	8.34	0.21	0.21	0.24							
<i>Trichocerca sp.</i>	B					619	0.54	63	0.33	1.44	0.41							
	HB																	
	C																	
	all			-1.72	0.2	363	0.41	31	0.35	0.85	0.48	2.11	0.2					
Eutrophic species	B					889	0.55	83.6	0.4	2.06	0.69							
	HB							62.2	0.3	1.3	0.41			-0.005	0.88			
	C																	
	all			-2.25	0.22	548	0.49	40.8	0.4	1.15	0.57	2.6	0.2					

Table 3.8. Significant differences (shaded cells) among Project Quinte stations for each rotifer parameter (Stanza 4 only) using ANOVA. NS indicates no significant difference ($p>0.05$). * Bold rotifer parameters were chosen as potential metrics for the BUI assessment. % Comp represents percent composition in terms of density and “–” indicates that parameter was not tested.

Rotifer Taxon	density	% Comp.
<i>Ascomorpha ovalis</i>	NS	NS
<i>Asplanchna</i> sp.	NS	NS
<i>Conochilus unicornis</i>	NS	NS
<i>Filinia longiseta</i>	NS	NS
<i>Filinia</i> sp.	NS	NS
<i>Gastropus stylifer</i>	NS	NS
<i>Kellicottia longispina</i>	NS	B<C
<i>Keratella cochlearis</i>	C, HB<B	HB<B
<i>Keratella cochlearis tecta</i>	NS	NS
<i>Keratella quadrata</i>	NS	NS
<i>Polyarthra dolichoptera</i>	NS	NS
<i>Polyarthra major</i>	NS	NS
<i>Polyarthra remata</i>	B<HB	NS
<i>Polyarthra</i> sp.	NS	NS
<i>Polyarthra vulgaris</i>	NS	NS
<i>Pompholyx sulcata</i>	NS	NS
<i>Synchaeta kitina</i>	NS	HB<C
<i>Synchaeta pectinata</i>	NS	NS
<i>Synchaeta</i> sp.	NS	HB<C
<i>Trichocerca cylindrica</i>	NS	NS
<i>Trichocerca multigrinis</i>	NS	NS
<i>Trichocerca pusilla</i>	C<B	NS
<i>Trichocerca rousseleti</i>	NS	NS
<i>Trichocerca</i> sp.	NS	NS
Eutrophic Species	C<B	C<HB
All Species	C<B	--

For each rotifer parameter tested above, significant differences among stations in Stanza 4 were determined using ANOVA (Systat 11). A p-value ≤ 0.05 was considered to be significant (Table 3.8). Results for the chosen indicators (bold) are described in more detail in the assessments that follow.

3.3 Assessment against Zooplankton Indicators

Four of the zooplankton indicators combine both eutrophication (bottom-up) and planktivory (top-down) influences. Background information and the rationale for their inclusion are described in more detail in Currie et al. (2023). These indicators are:

- 1) Cladoceran mean size
- 2) *Chydorus* biomass
- 3) Percent *Acanthocyclops vernalis*
- 4) Shannon Diversity Index

Three indicators are primarily driven by top down influences (planktivory). These are:

- 5) Percent Large *Daphnia* (>0.75 mm)
- 6) Percent *Daphnia galeata mendotae*
- 7) Percent *Mesocyclops*

The remaining indicator is driven primarily by eutrophication:

- 8) Rotifer Community Structure

In the following sections, each indicator is assessed to provide evidence for the determination of zooplankton impairment.

3.3.1 Cladoceran mean size

Metric: An increase in cladoceran mean size.

Rationale: The cladoceran mean size metric addresses both top-down and bottom up impacts. Mean cladoceran length is sensitive to taxonomic composition. Reduction in size is an indicator of both eutrophic conditions and high planktivory, as more eutrophic systems are dominated by small cladocerans such as *Bosmina*, *Eubosmina* and *Chydorus*. An increase in mean size suggests that planktivore abundance and trophic status have decreased.

Notes: Prior to 1995 in the Bay of Quinte, zooplankton size was determined retrospectively using seasonal composite samples. Unfortunately from Stanza 1, archived samples were only

available for measurement from 1979 and 1982 at HB, and 1976, 1979 and 1982 at B and C. Therefore we have very small sample sizes from Stanza 1 for this parameter.

Stanza Differences: Overall the cladoceran mean length relationship is largely driven by a few low points early in the time series when planktivory was very high (Fig. 3.2). It has also been quite variable at all three stations in Stanza 4. At B and C, mean cladoceran size was significantly lower in stanza 1 (Table 3.4). At HB, it was lower in stanza 1 than in 2 and 3, but size in stanza 4 was not different from any of the other stanzas (Fig. 3.2).

Station Differences: Overall the cladoceran mean length relationship is largely driven by a few low points early in the time series when planktivory was very high (Fig. 3.3). Despite spatial differences in trophic status, cladoceran mean size was not significantly different among the three stations for most of Project Quinte (stanzas 1, 3 and 4). In stanza 2, C was < B and HB (Table 3.3; Fig. 3.3).

Regression Relationships: Cladoceran mean length significantly decreases with increasing eutrophy, as shown by negative correlations for chlorophyll, TP, TKN and planktivore biomass using stanza 1 and 3 data at B and HB (Fig. 3.4). When all three stations were examined individually, this indicator correlates negatively with planktivory at B and C, but the relationship at HB isn't significant (Table 3.1). Many of the recent points at B and HB fall below those from stanza 3, showing a general reduction in mean size in many of the recent years. This slippage is shown by the negative slope in the CuSum plots in Stanza 4 (Fig. 3.2). In contrast, mean size has increased slightly at C in the last few years, as shown by the positive CuSum slope since the mid-1990s. This likely corresponds to a recent drop in planktivory at this station due to lower Alewife numbers.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD was 0.42 ± 0.05 mm for B and HB combined. Values <0.37 mm were considered unusually poor (greater than 1 SD below the mean), and occurred in the recent stanza in 2004, 2011 and 2013 at B, and 2006 and 2011 at HB (Table 3.9). The Hamilton Harbour cladoceran mean size between 2002 and 2012 was very similar to Quinte (0.42 ± 0.02 mm), whereas in the Kingston Basin of Lake Ontario the average was higher ($0.54 \pm .03$) from 2007 to 2012 (K. Bowen, DFO, unpublished data).

Summary: This indicator has shown improvement from Stanza 1 but has declined in recent years.

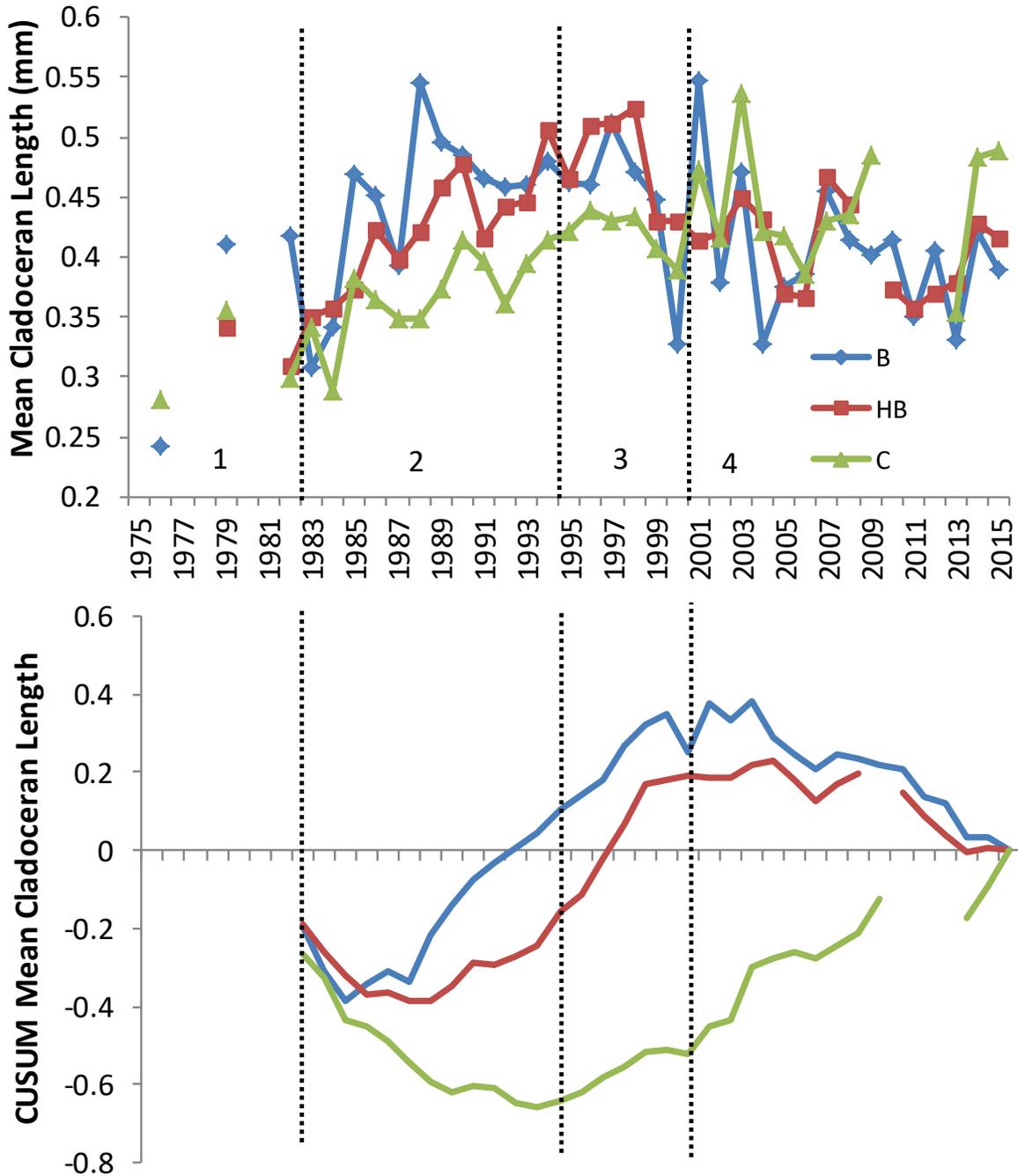


Figure 3.2: Changes in cladoceran mean size (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines. Note the missing years in Stanza 1.

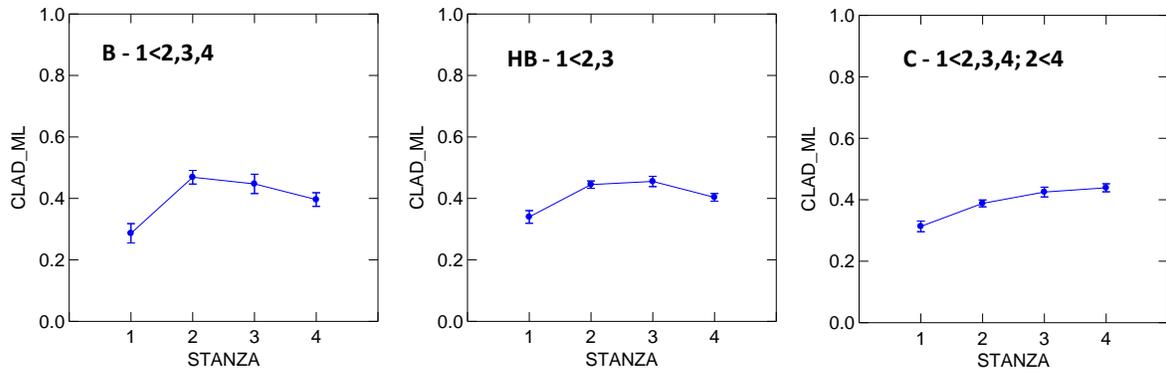


Figure 3.3: Cladoceran mean size \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph.

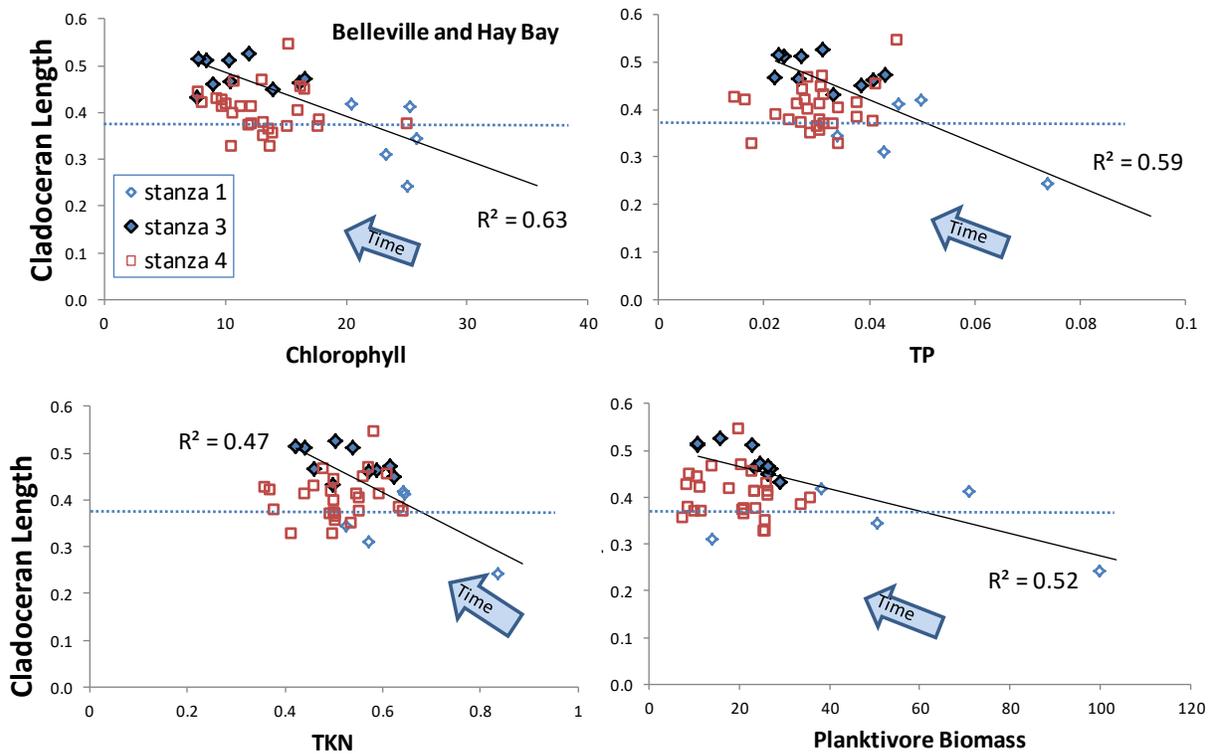


Figure 3.4: Regression relationships for cladoceran mean length plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes, including r^2 values, show the relationships for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. Also shown are the stanza 4 points to illustrate recent trends. The dotted line shows the 1995 to 2015 mean value -1 standard deviation for B and HB combined. Points falling below this line are unusually poor.

3.3.2 *Chydorus* Biomass

Metric: Decreased *Chydorus* biomass relative to the 1970s and early 1980s.

Rationale: *Chydorus sphaericus* is a small herbivorous littoral cladoceran that is abundant in the Bay of Quinte, particularly in the Upper Bay. *C. sphaericus* has been proposed as an indicator of eutrophic conditions (Gannon and Stemberger, 1978, Gulati, 1983; Pejler, 1983). It is not a preferred prey species for planktivorous fishes due to its small size.

Stanza Differences: *Chydorus* biomass has significantly dropped at B since stanza 1 and 2 (Fig. 3.5 and 3.6). Although not significantly different, this same pattern was seen at HB. There were no significant differences among stanzas at C. The highest *Chydorus* biomass (30.9 mg m^{-3}) was observed at HB in 1983 and some of the lowest *Chydorus* biomass values were observed at B in the early 2000s. The negative slope of the CuSum plots for HB and B shows declining values starting in about 1990 and 1995, respectively, until about the early to mid-2000s. Since about 2005, year-to-year fluctuations in *Chydorus* biomass have been relatively high, with some years above 10 mg m^{-3} . At B, values have again dropped below 5 mg m^{-3} since 2012. *Chydorus* biomass has been relatively stable at C over the entire time series, as shown by the relatively flat CuSum line at this station.

Station Differences: *Chydorus* biomass is generally about an order of magnitude lower at C than at HB and B (Fig. 3.5), a difference that was statistically significant in stanzas 1 and 2, but not in stanza 3. In stanza 4, C was only lower than HB. It may not be appropriate to use this indicator at C because the open waters of the lower bay are more oligotrophic and lake-influenced which is not the preferred habitat of this littoral species.

Regression Relationships: The regressions between *Chydorus* biomass and the eutrophication parameters at HB were not significant. Therefore we only included B in the regressions in Figure 3.7, which show *Chydorus* biomass is positively related to chlorophyll, TP, TKN and planktivory. Positive correlations with all these factors were also found when data from all three Quinte stations were combined (Table 3.1), illustrating this species' ability to tolerate more eutrophic conditions and high planktivory.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD (B and HB combined) was $5.6 \pm 3.9 \text{ mg m}^{-3}$. *Chydorus* biomass $>9.5 \text{ mg m}^{-3}$ was considered unusually

poor (mean + 1 SD) and occurred frequently in stanzas 1 and 2 (Table 3.9). During the recent stanza, poor values occurred in 2005, 2007 and 2011 at B, and 2003, 2005 and 2006 at HB. During this period, *Chydorus* comprised on average 3.8% of crustacean biomass at B and HB.

Although *Chydorus* tend to prefer eutrophic nearshore areas and embayments, biomass values in other systems in Ontario tend to be considerably lower (K. Bowen, DFO, unpublished data). For example, biomass of this species averaged 4.3 mg.m⁻³ in Hamilton Harbour between 2002 and 2014 (1.5% of the total biomass) in this highly eutrophic system. In other productive systems, including the west Basin of Lake Erie in the 1990s, Cook's Bay in Lake Simcoe, Penetang Harbour in Georgian Bay and Weller's Bay in Prince Edward County (all sampled in 2012), *Chydorus* biomass averaged <0.3 mg m⁻³. The only system sampled by DFO that was similar to or higher than the upper Bay of Quinte was Prince Edward County's West Lake in 2012 (11.2 mg m⁻³ or 16.0% of crustacean biomass). *Chydorus* biomass in highly eutrophic Lake Võrtsjärv in Estonia was also higher than the Bay of Quinte, ranging from 14.6 to 33.8 mg m⁻³ (adapted from Haberman and Haldna, 2014, using the B mean weight). This species was often the dominant cladoceran in Lake Võrtsjärv.

Summary: *Chydorus* biomass has generally shown long-term improvement (decline) since the 1970s. Although there has been year-to-year variability, there has been little continued improvement in this indicator since the mid-1990s.

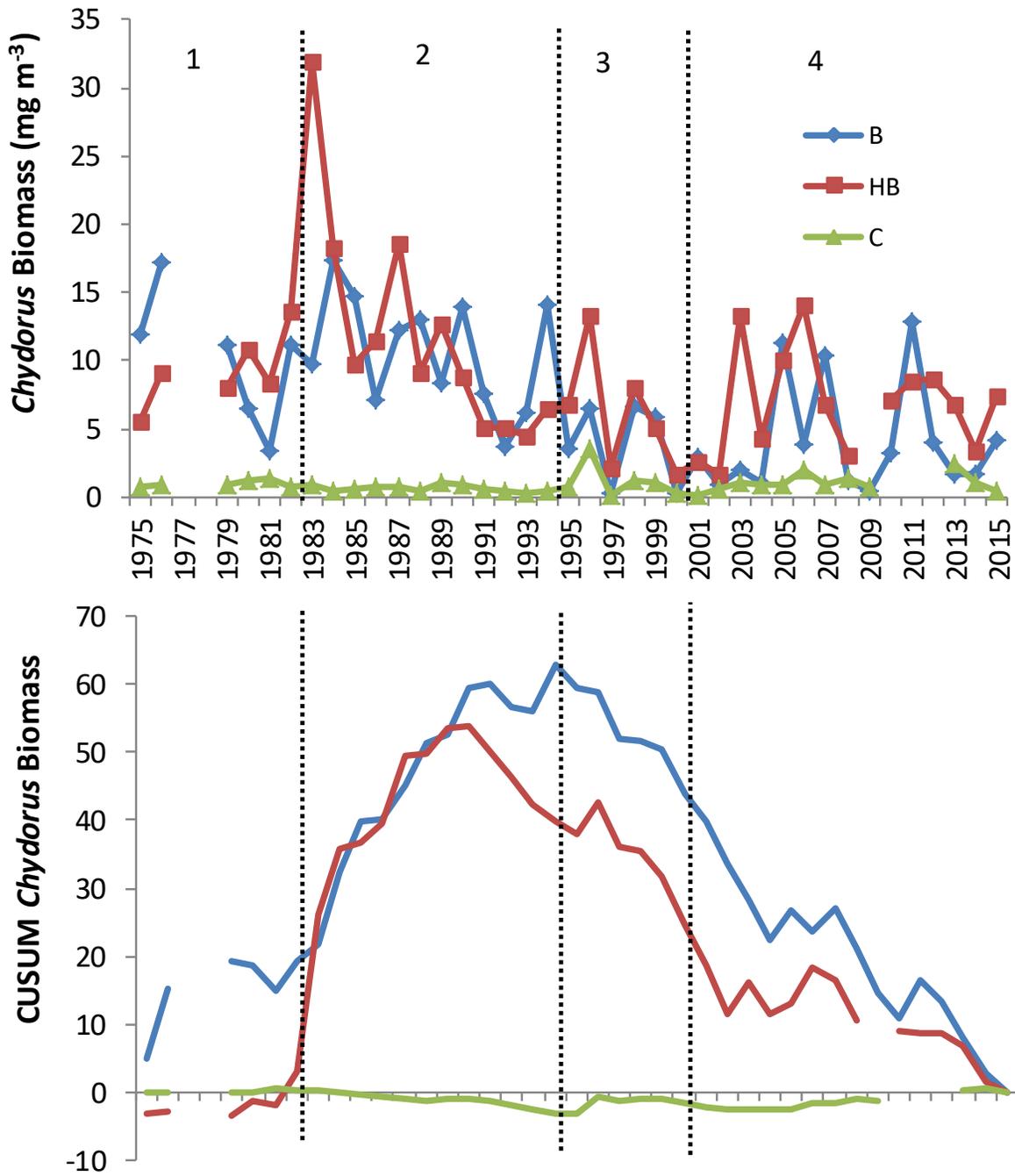


Figure 3.5: Changes in Chydorus biomass (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

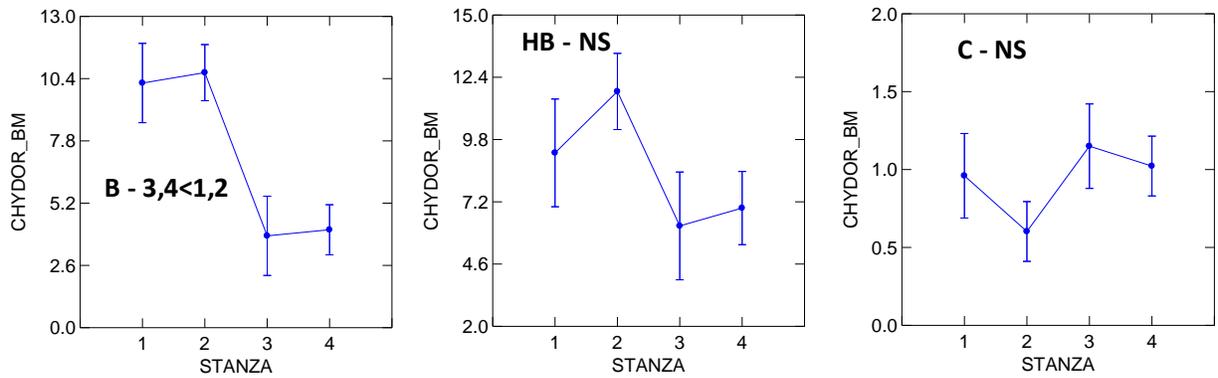


Figure 3.6: Chydorus biomass (mg m^{-3}) \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

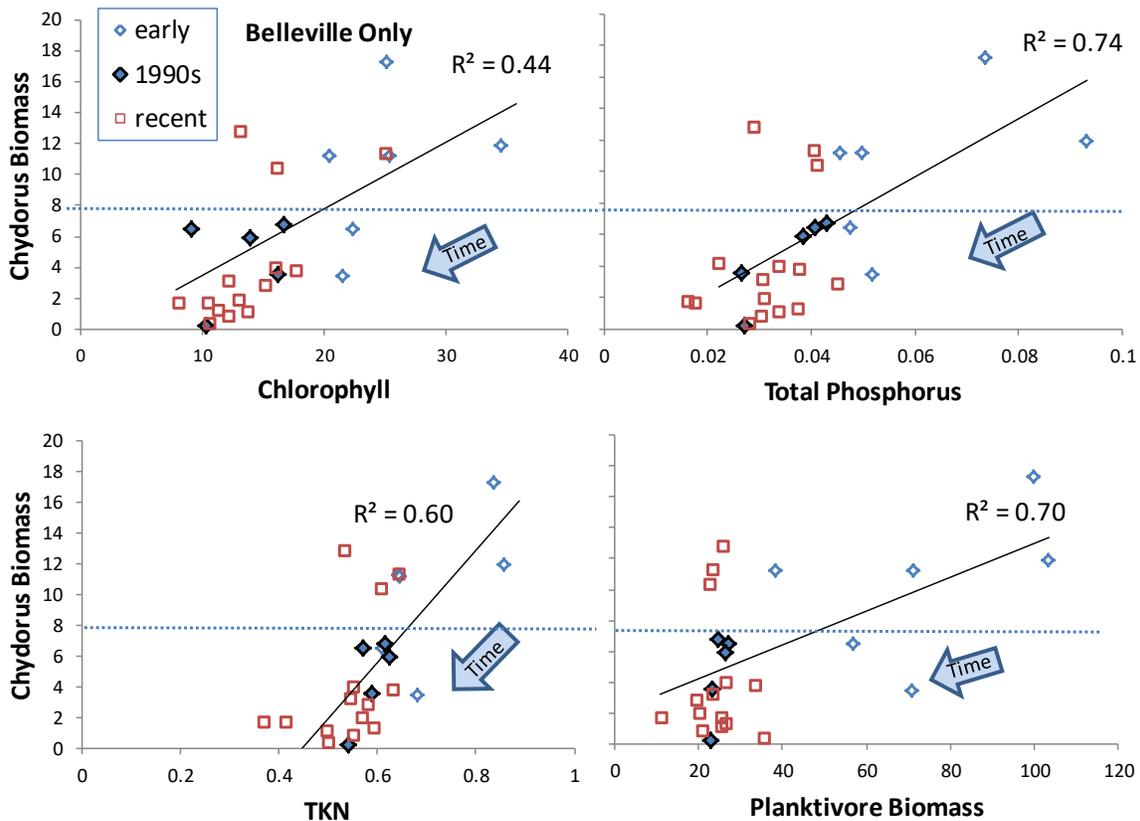


Figure 3.7: Regression relationships for Chydorus biomass plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B only. Relationships were not significant at HB. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value - 1 standard deviation.

3.3.3 Percent *Acanthocyclops vernalis*

Metric: Reduced proportion of *Acanthocyclops vernalis* relative to total adult cyclopoid biomass.

Rationale: *Acanthocyclops vernalis* (sometimes called *Cyclops vernalis*) is a medium-sized cyclopoid copepod that is generally only found in eutrophic embayments such as Saginaw Bay, Green Bay, as well as the western basin of Lake Erie, and as such it is considered to be a good indicator of eutrophy in the Great Lakes (Gannon and Stemberger, 1978). This species is likely less vulnerable to fish predation than the larger *Mesocyclops* due to size selective planktivory.

Notes: Overall, cyclopoid biomass has declined in the Bay of Quinte since the 1990s (Fig. 3.8). This decline has also occurred in Lake Ontario (Rudstam et al, 2015). We calculated the percent of *A. vernalis* biomass relative to adult cyclopoid biomass, as this appeared to work better as an indicator than biomass. This allows us to look at changes in the relative composition of the cyclopoid community independent of changes in overall biomass.

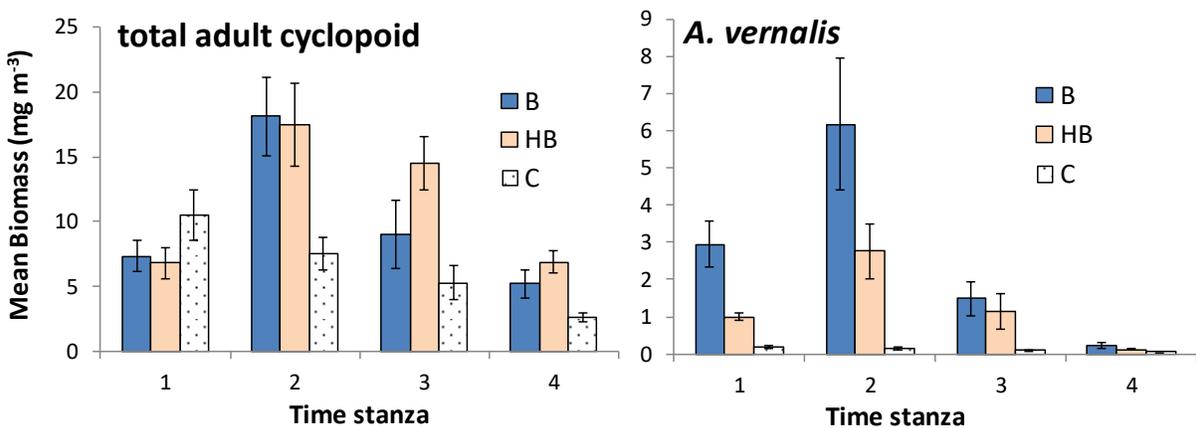


Figure 3.8: Mean biomass of adult cyclopoids and *Acanthocyclops vernalis* in the Bay of Quinte in the four time stanzas. Standard errors are given. Note the different scales.

Stanza Differences: At B and HB, percent *A. vernalis* was significantly less in stanza 4 than in stanzas 1 and 2 (Fig. 3.8 and 3.9). At C, there were no differences (Fig. 3.10). Overall % *A. vernalis* has declined over time.

Station differences: Compared to the upper bay, biomass of this species is about an order of magnitude lower at C due to its more oligotrophic nature and proximity to the open waters of Lake Ontario (Fig. 3.8). In stanzas 1, 3 and 4, percent *A. vernalis* at C was significantly less than B. In stanza 2, the percentage at C was less than both B and HB. Although not statistically

different, percent *A. vernalis* was on average about two times higher in the upper bay than the middle bay.

Regression Relationships: When B and HB are combined, percent *A. vernalis* positively correlates with eutrophication parameters (chlorophyll, TP, TKN, planktivory), and negatively correlates with Secchi (Fig. 3.11, Table 3.1). At B alone, all but Secchi were significant, and at HB, only TP and TKN were significant. There were no significant relationships with eutrophy at C, where this taxon is much less common. This indicator of eutrophic conditions has shown the greatest response at B, a moderate response at HB, and no response at C.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD at B and HB combined was 6.4 ± 9.0 %. There have been no unusually poor values (>15.4 %, which is the mean + 1 SD) during the last time stanza at either station (Fig. 3.10, Table 3.9). Poor values were common in stanzas 1 and 2, especially at B. This is one of the only indicators where stanza 4 has shown a continued downward trend relative to the 1990s, as shown by the negative CuSum slopes for HB and B in Figure 3.9. It is possible that other factors, such as increases of the predatory invader *Cercopagis* at HB, have contributed to this decline. *Cercopagis* has been shown to negatively impact juvenile copepods (Benoit et al., 2002), although why it would disproportionately affect *A. vernalis* is unclear. Furthermore, *Cercopagis* populations are generally quite low at B, so it is unlikely that this invasive species is responsible for the downward trend in the upper bay.

This cyclopoid is a common species in shallow, productive systems across temperate North America. Bay of Quinte *A. vernalis* appear to be at intermediate levels compared to other studies. For example in western Lake Erie, *A. vernalis* comprised 21.4% of adult cyclopoid biomass (Makarewicz, 1993), whereas in Hamilton Harbour, this value was only 2.9% (Bowen and Currie, 2017). It was a dominant species in hypereutrophic Onondaga Lake in New York (Meyer and Effler, 1980). Although it was absent from Weller's Bay and West Lake in Prince Edward County in 2012, *A. vernalis* comprised 15.4% of adult cyclopoid biomass in Cook's Bay of Lake Simcoe, but only 1.1% in Penetang Harbour (K. Bowen, DFO, unpublished data).

Summary:

This indicator has shown steady improvement in the upper and middle bays over time, and values have continued to decline in the recent stanza.

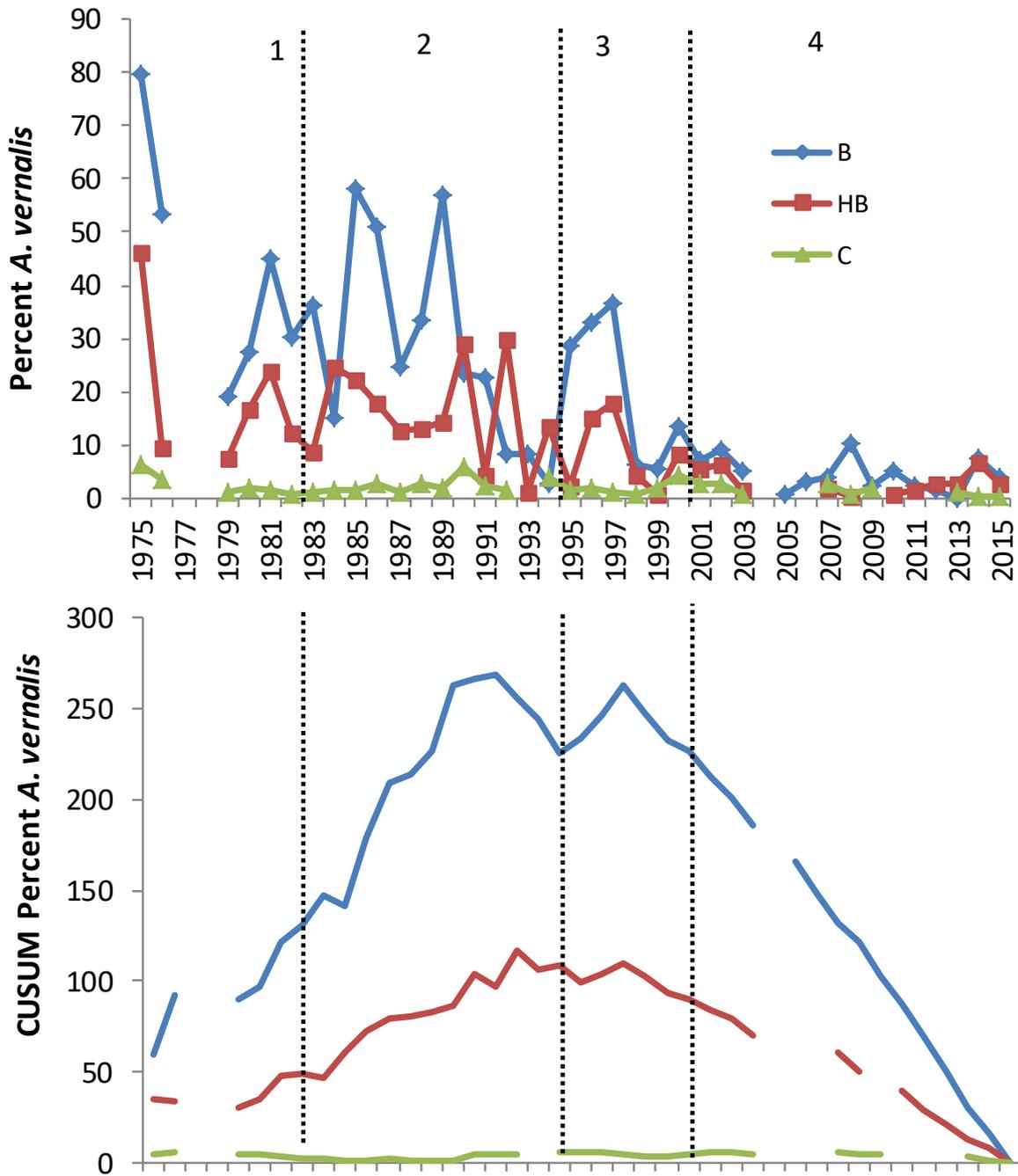


Figure 3.9: Changes in percent *A. vernalis* (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

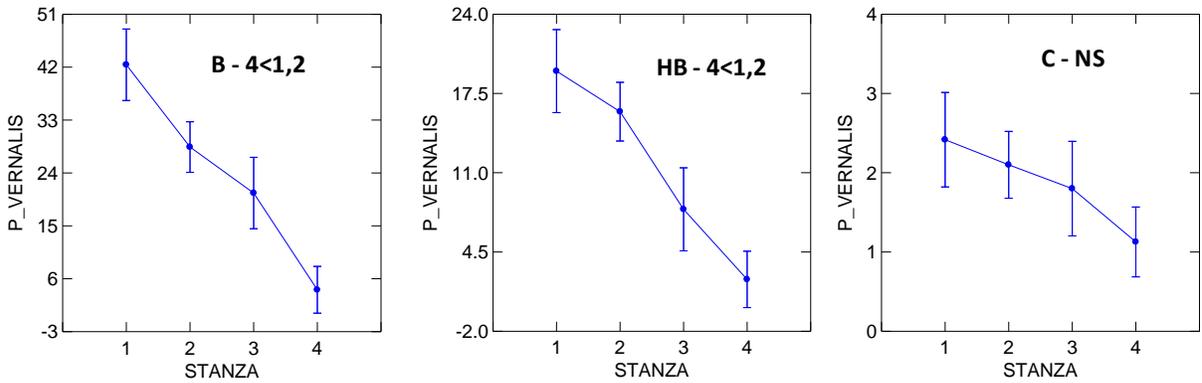


Figure 3.10: Percent *A. vernalis* biomass \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

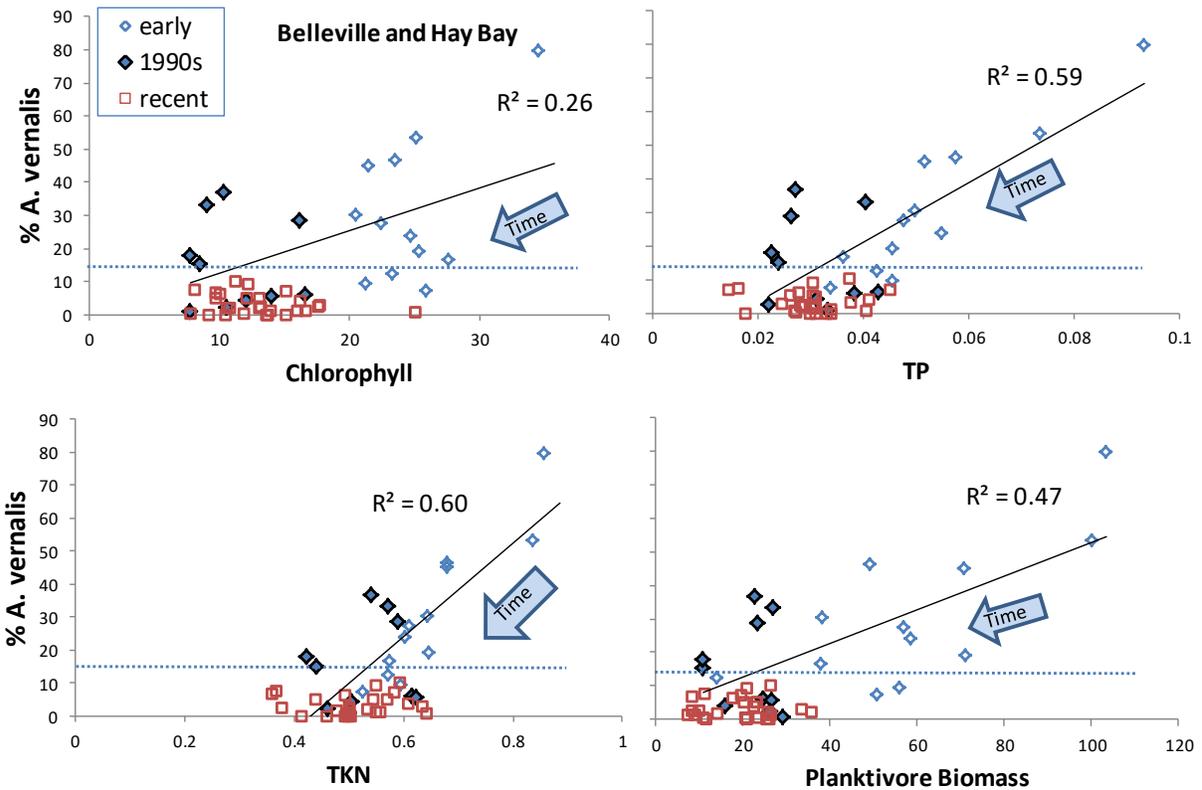


Figure 3.11: Regression relationships for percent *A. vernalis* plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value + 1 standard deviation for B and HB combined.

3.3.4 Shannon Diversity Index

Metric: A trend toward increasing diversity index scores in the zooplankton community over time.

Rationale: Biodiversity represents the variety and heterogeneity of organisms found within an ecosystem or community. A diversity index is a mathematical measure of species diversity that combines species richness and abundance. We chose the widely used Shannon's diversity Index (H'), with higher values representing a more diverse community. However, diversity indices only describe community structure, and not how well the ecosystem functions (Heip et al., 1998). Degraded systems dominated by one or a few tolerant species have lower diversity scores. Diversity scores are expected to increase in the Bay of Quinte over time as conditions improve.

Notes: Our calculation methods, including allocation of juvenile copepods, are described in Bowen and Currie (2017). Unlike most of the indicators, the diversity index is based on zooplankton density, not biomass.

Stanza Differences: Shannon diversity index values have not radically changed over time in the Bay of Quinte (Fig. 3.12), and diversity scores were not different among stanzas at B and C (Fig. 3.13). At HB, stanza 1 scores were significantly lower than stanza 3. Some of the very low values at C in stanza 4 are related to high densities of dreissenid veligers.

Station differences: In the first three time stanzas, there were no differences in diversity scores among stations (Table 3.3). In Stanza 4, C had significantly lower scores than HB, again due to domination by dreissenid veligers in the lower bay since 2008.

Regression Relationships: The Shannon Diversity Index decreases with increasing eutrophy and planktivory at B and HB, and for the two stations combined (Table 3.1). There are significant negative correlations with TP, TKN, chlorophyll and planktivory (Fig. 3.14), and positive correlations with Secchi. However, there were no significant correlations at C.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD was 2.0 ± 0.2 at B and HB combined. Years with unusually poor diversity index scores (<1.8) were 1975, 1979, 1980, 1982, 2002, 2004, 2008, 2009, 2010 and 2013 at B and 1976, 1981, 2008 and

2011 at HB. In the last stanza, most of the years with low values had zooplankton communities numerically dominated by a few small taxa such as *Bosmina*, veligers and *Ceriodaphnia*.

H' values typically fall between 1.5 and 3.5 in most ecological systems, and rarely exceed 4. The Bay of Quinte mean score of 2.0 was higher than those observed for zooplankton communities in other shallow productive systems in Ontario (K. Bowen, DFO, unpublished data). In Hamilton Harbour, Shannon Diversity averaged 1.68 between 2002 and 2014. In 2012, H' was 1.73 in West Lake, and 1.70 in Penetang Harbour. Values in western Lake Erie in the 1990s and Cook's Bay of Lake Simcoe in 2012 were similar to Quinte, with scores of 1.90 and 1.96, respectively.

Zooplankton diversity in the upper Bay of Quinte was often higher than values reported from other eutrophic systems in the literature. For example, in strongly eutrophic Lake Võrtsjärv in Estonia, the Shannon Diversity index for zooplankton plus rotifers showed a decreasing trend over time as eutrophy increased, ranging from about 1.9 in the 1970s to ~1.5 in 2010s (Haberman and Haldna 2014). In Lake Bracciano, Italy, H' scores for zooplankton ranged from 0.47 to 2.92 (Ferrara et al. 2002). Wang et al. (2012) reported scores of 0.3 to 1.1 across Lake Dianchi, a large shallow eutrophic lake in China.

Summary: This indicator has been relatively stable over the time series. Overall, it does not appear to be overly sensitive as an indicator. Diversity scores in the Bay of Quinte are equivalent to, or often higher than those obtained from other eutrophic systems.

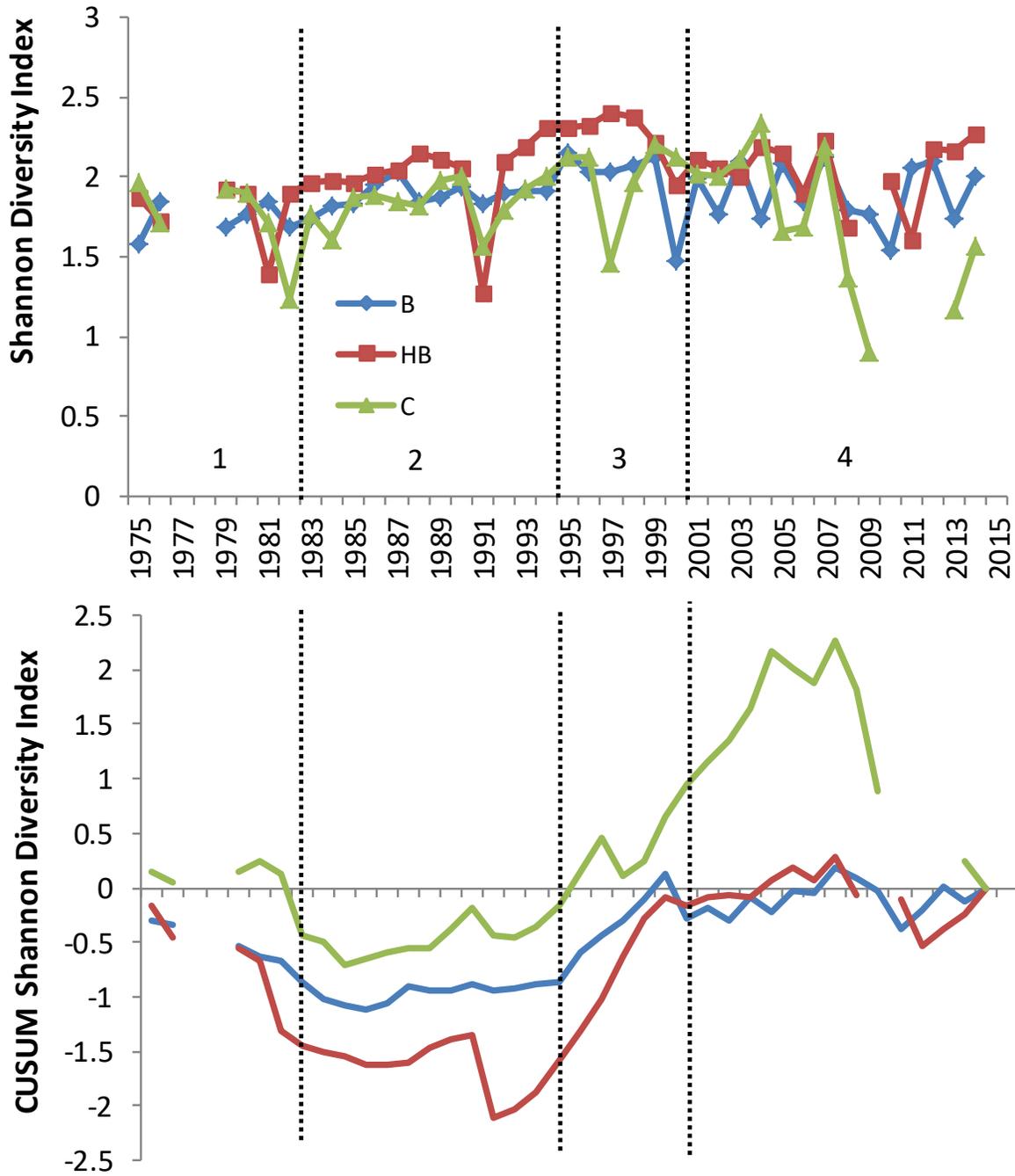


Figure 3.12: Changes in Shannon Diversity Index (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

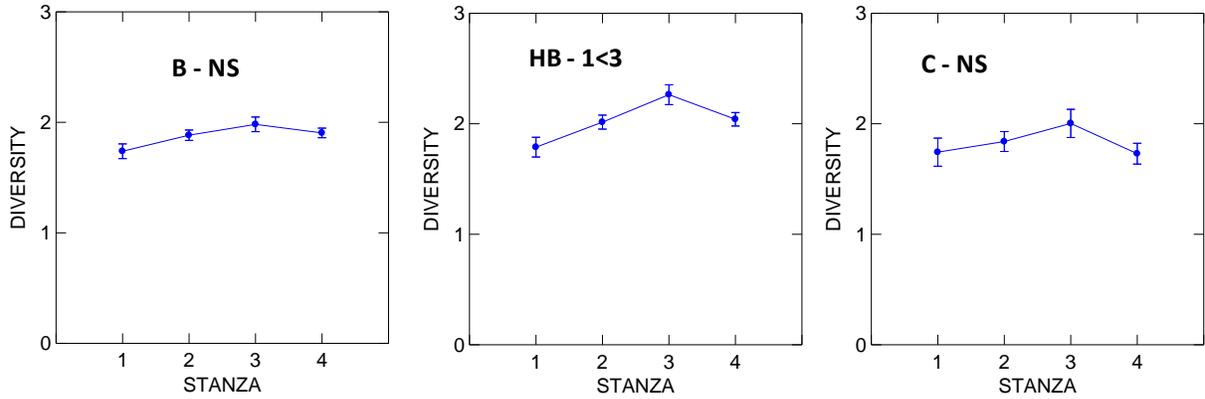


Figure 3.13: Shannon Diversity Index \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

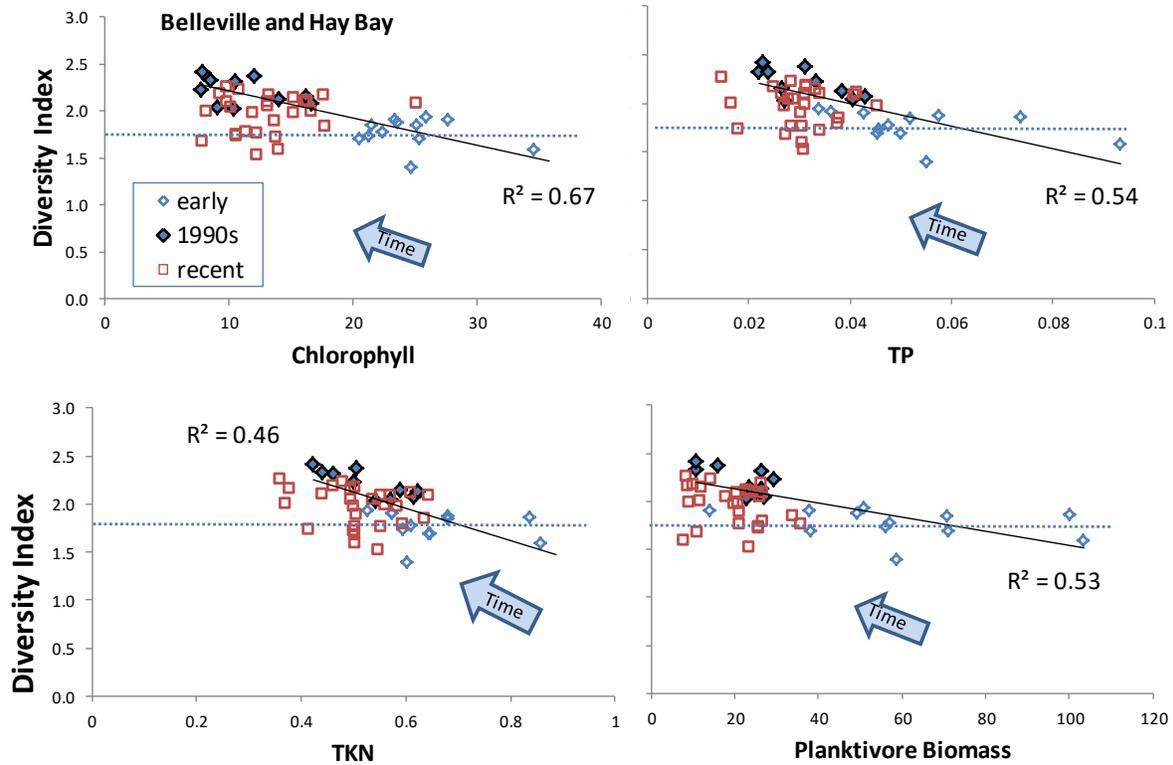


Figure 3.14: Regression relationships for Shannon Diversity Index (dependent variable) plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value -1 standard deviation for B and HB combined.

3.3.5 Percent Large *Daphnia*

Metric: A trend toward increasing large *Daphnia* over time.

Rationale: The percentage of large *Daphnia* (>0.75 mm) is expected to increase as the Bay of Quinte becomes less eutrophic and the abundance of planktivorous fishes decline. *Daphnia* are among the largest herbivorous zooplankton found in the Bay of Quinte, and their size and abundance can be reduced due to size-selective planktivory by fishes.

Notes: Unfortunately from Stanza 1, *Daphnia* measurements were only available from 1979 and 1982 at HB, and 1976, 1979 and 1982 at B and C. Therefore we have very small sample sizes from Stanza 1 for this parameter.

Stanza Differences: At B and HB, there were no significant differences among time stanzas, although the percentage of large *Daphnia* tended to be lowest in stanza 1 (Fig. 3.15 and 3.16). The low sample sizes in Stanza 1 may have led to the lack of significant differences. At C the pattern was reversed, with the percentage greatest in stanza 1.

Station Differences: In stanzas 1, 2 and 4, the three stations were not different. In stanza 3, there were significantly fewer large *Daphnia* at C.

Regression Relationships: When B and HB stations are combined, there are significant negative relationships with chlorophyll, TP, TKN and planktivory (Fig. 3.17). Therefore large *Daphnia* negatively correlate with increasing eutrophy, although increasing planktivory probably plays a strong role as well. As with the other size indicators, some of the values in the recent stanza have been lower than those during stanza 3. The results at Conway are somewhat contradictory.

Mean Values: For B and HB combined, the 1995 to 2015 mean \pm SD was $56 \pm 8\%$ large *Daphnia* relative to all *Daphnia*. Unusually poor values (<48%) occurred in 1976, 1979, 1982, 2002, 2004 and 2013 at B; and 1982, 2006, 2011 and 2012 at HB. The CuSum plots show that this parameter has been relatively stable at all three stations in stanzas 2 and 4, except for a recent increase at C (Fig. 3.15). As this value is rarely reported in other published zooplankton studies, no comparisons to literature values are available.

Summary: Although there was some early improvement in the proportion of large *Daphnia* in the upper and middle reaches of Quinte, this parameter has plateaued since the mid -1980s. It has recently shown an upward trend at C.

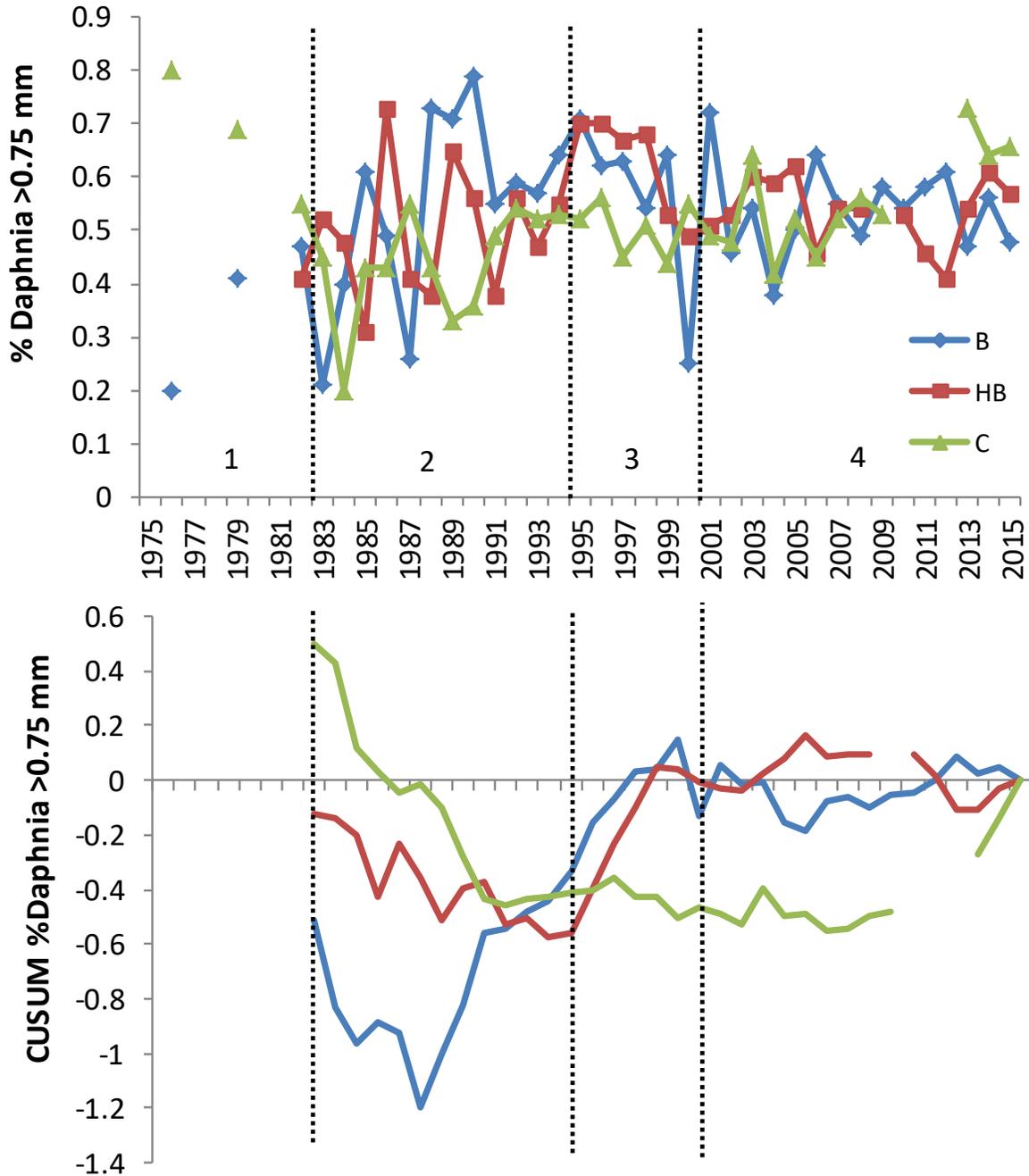


Figure 3.15: Changes in percent *Daphnia* >0.75 mm (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

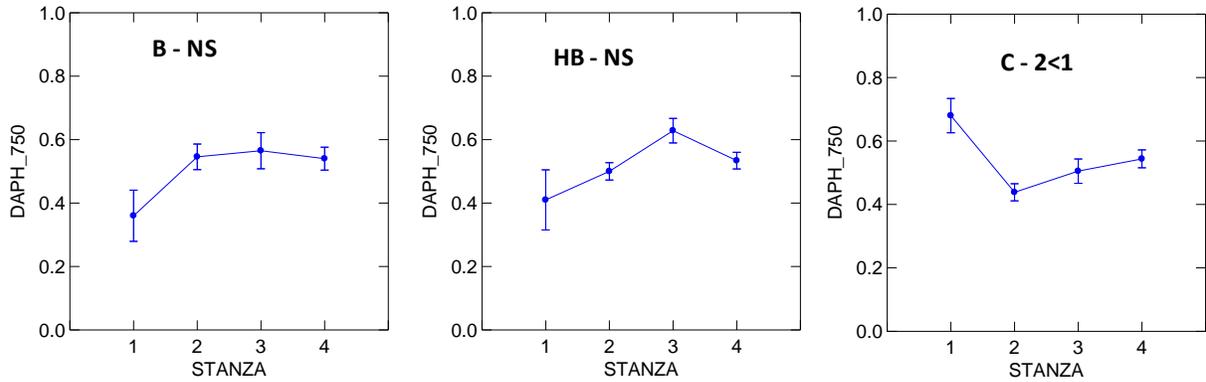


Figure 3.16: Percent *Daphnia* >0.75 mm \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

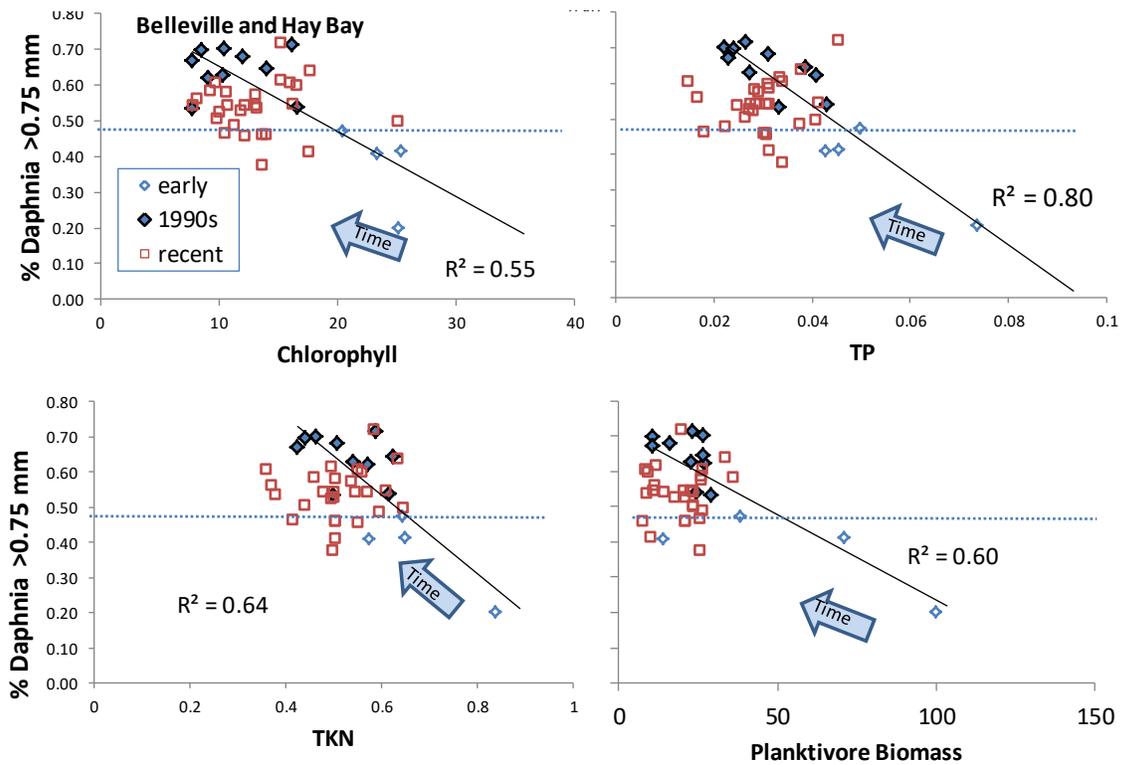


Figure 3.17: Regression relationships for percent *Daphnia* >0.75 mm (dependent variable) plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value - 1 standard deviation for B and HB combined.

3.3.6 Percent *Daphnia galeata mendotae*

Metric: A trend toward increasing % *D. galeata mendotae*.

Rationale: The percent of *D. galeata mendotae* biomass relative to total crustacean biomass also relates to cladoceran mean size and percent large *Daphnia*. This species is considered by some as an indicator of oligotrophy (Pejler, 1983), although their numbers are often controlled by planktivory. An increase in the proportion of this species indicates that planktivory is declining, and to a lesser degree, eutrophy is decreasing. Furthermore, *D. galeata mendotae* may be more resistant to invertebrate predators such as *Cercopagis* due to its larger size.

Stanza Differences: Percent *D. galeata mendotae* was generally low in stanza 1 and has increased since, although this difference was not significant at B. At B and HB, it has remained relatively unchanged since stanza 2 (Fig. 3.18 and 3.19). At C, it has increased over time.

Station Differences: In stanzas 1, 3 and 4, stations were not different and in stanza 2, the proportion of *D. galeata mendotae* was lowest at C.

Regression Relationships: This indicator was significantly related to eutrophication parameters and planktivory at B and HB, but these relationships were not significant at C. Percent *D. galeata mendotae* decreases with increasing chlorophyll, TP, TKN and planktivory, and increases with increasing Secchi (Fig. 3.20). This metric is negatively associated with increasing eutrophy, but may in fact be responding most strongly to planktivory, as individuals of this species tend to be large and are desirable prey for fishes. It has increased since stanza 1 when both eutrophy and planktivory were highest. When all stations were combined in the Bay of Quinte, this species negatively correlated with high planktivory, but not with eutrophy (Table 3.1). This species also appears to be sensitive to high flow rates and low water temperatures, as shown by their reproductive failure and low densities in the upper bay in 2000, an unusually cool and wet summer.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD at B and HB combined was 20.2 ± 12.2 % of crustacean biomass. Unusually poor values (<7.9 %) occurred during most years prior to 1986 at both B and HB. More recently, years with few *D. galeata mendotae* were 1998, 2002, 2004, 2013, 2014 at B and 1999 and 2011 at HB.

Generally this percentage is higher or similar to other productive Great Lakes systems. In Hamilton Harbour in the 2000s, *D. galeata mendotae* averaged 12.9% of biomass across the season (Bowen and Currie, 2017), and in Sodus Bay in 2004 it averaged 31.4% (adapted from Makarewicz and Lewis, 2005). Both are eutrophic Lake Ontario embayments. In the mid-1980s it averaged 5.9% in productive western Lake Erie, 13.2% in the central basin, and 11.6% in the more oligotrophic eastern basin (Makarewicz, 1993). In Saginaw Bay, a productive embayment in Lake Huron, this species reached 24.6% of biomass when planktivore abundance was low (2009 – 2010), but only 5.4% when Alewife were abundant (adapted from Pothoven et al., 2013).

Summary: Although there has been some improvement relative to stanza 1, this metric has been highly variable year to year. There have been no dramatic improvements since the mid-1980s.

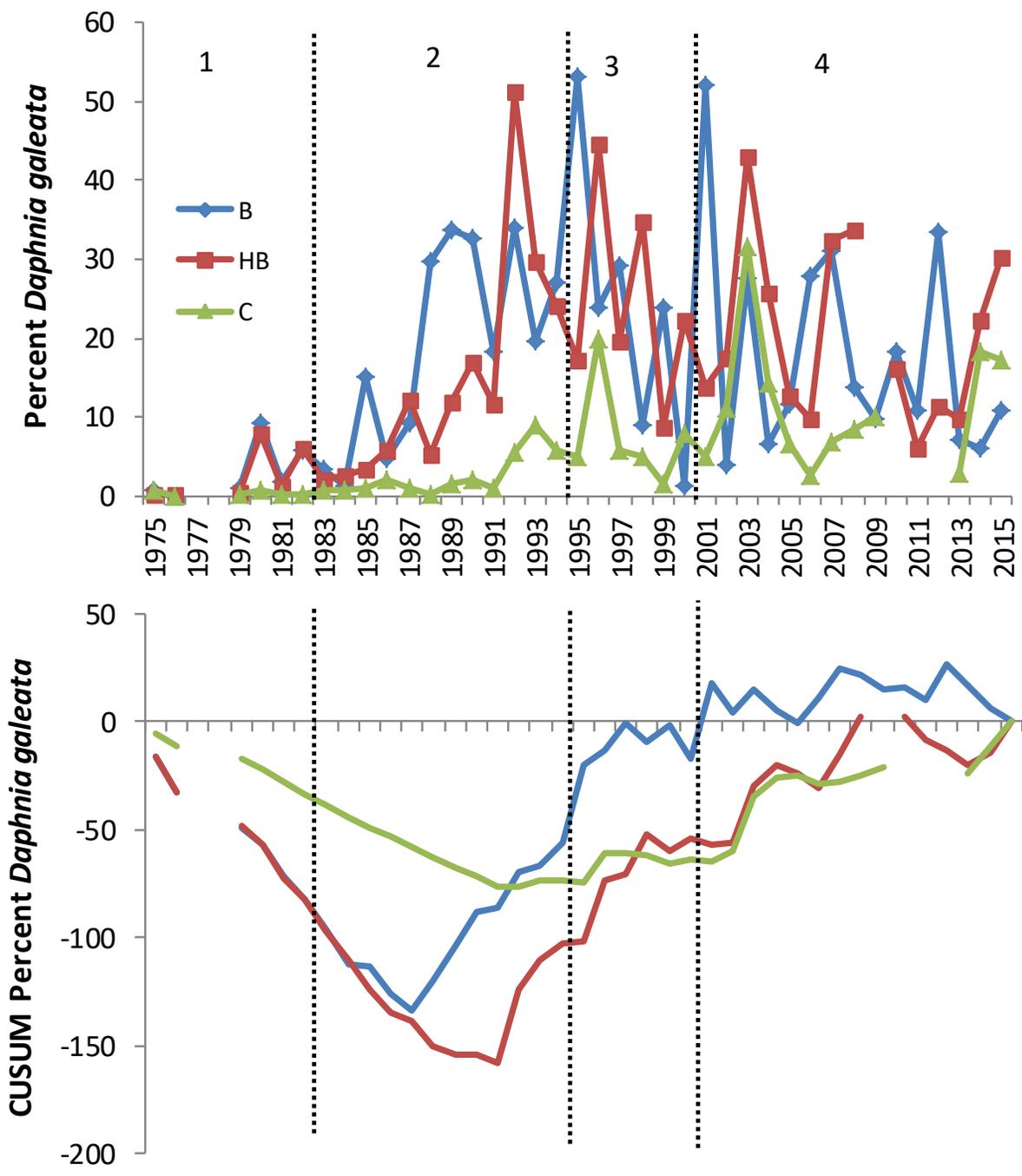


Figure 3.18: Changes in percent *Daphnia galeata mendotae* (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

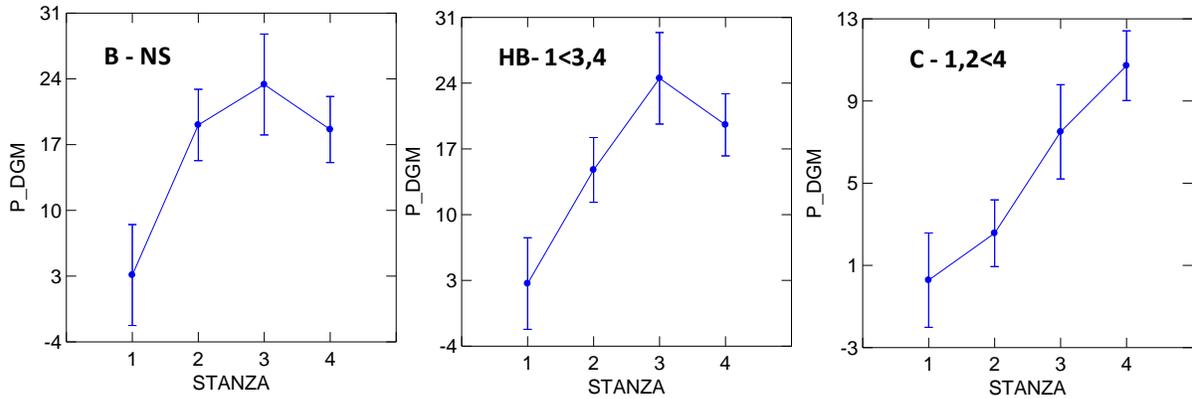


Figure 3.19: Percent *Daphnia galeata mendotae* \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

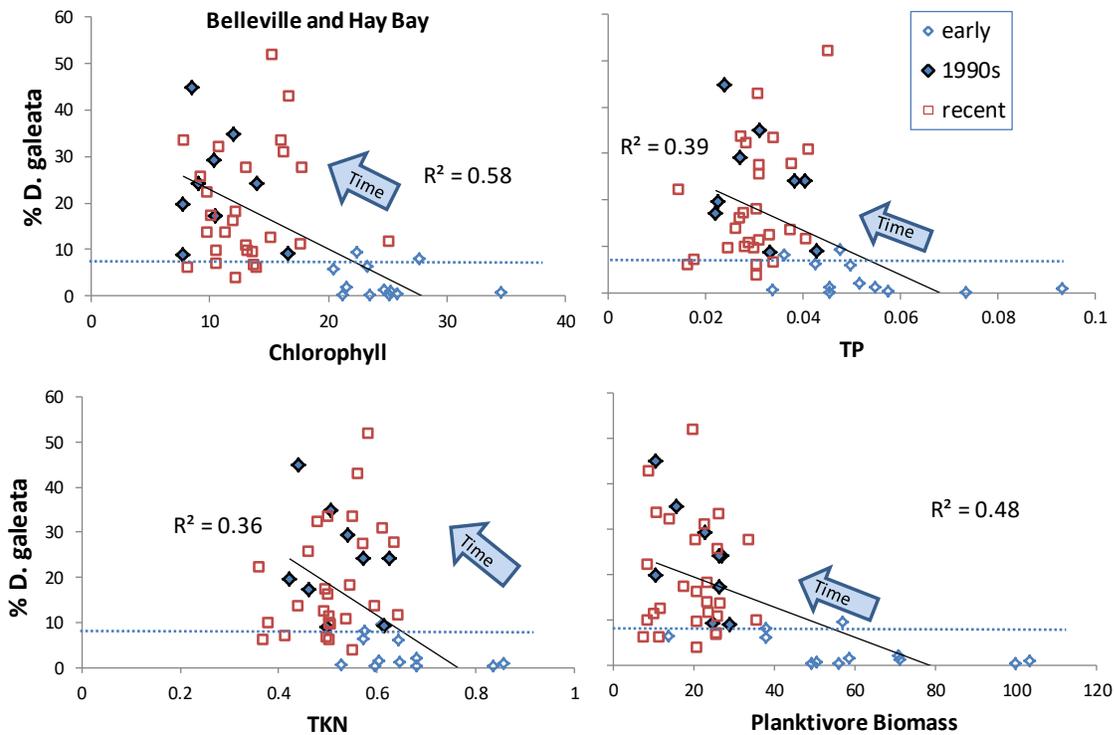


Figure 3.20: Regression relationships for % *Daphnia galeata mendotae* (dependent variable) plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value -1 standard deviation for B and HB combined.

3.3.7 Percent *Mesocyclops*

Metric: An increase in the proportion of *Mesocyclops* relative to other cyclopoids.

Rationale: *Mesocyclops* is a large cyclopoid copepod that is common in the Bay of Quinte. Although *Mesocyclops* is typically considered a eutrophic species, because it is so strongly controlled by high planktivory, we are considering an increase in the proportion of *Mesocyclops* an indicator of reduced planktivory in the Bay of Quinte.

Notes: This metric is calculated as the percent of *Mesocyclops edax* biomass relative to adult cyclopoid biomass.

Stanza Differences: The overall trend is for percent *Mesocyclops* to increase over time (Fig. 3.21 and 3.22). At B, there were no significant differences among stanzas, although values generally increased between stanza 1 and 3. At HB, stanza 1 was significantly lower, and at C, stanzas 1, 2 and 3 were significantly less than 4.

Station differences: In stanza 1, percent *Mesocyclops* at C was significantly lower than B, and in the stanzas since, values at C were lower than both B and HB.

Regression Relationships: Percent *Mesocyclops* correlates positively with Secchi depth and negatively with eutrophication parameters (TP, TKN, chlorophyll) and planktivory when B and HB are combined (Fig. 3.23; Table 3.1) and at HB alone. At B, only planktivory, TP and TKN are significant. At C, only Secchi and TKN are significant. Although *Mesocyclops* showed a negative response to eutrophy when the stations were examined separately (a decline over time), this is probably more of a response to planktivory, as high planktivore biomass co-occurred with more eutrophic conditions.

Mean Values and Comparison to Other Studies: The 1995 to 2015 mean \pm SD value was 61.9 ± 16.4 % at B and HB combined. Unusually low values (<45.5 %) occurred in 1975, 1976, 1981, 1995, 2004, 2011 and 2013 at B; and 1975 to 1982 and 2011 at HB. Despite year-to-year fluctuations, the CuSum plot suggests percent *Mesocyclops* began deviating above the overall mean in the upper and middle bays starting in the early 1990s, and plateaued starting in about 2010 (Fig. 3.21).

This species is a common component of temperate eutrophic systems. For example, *Mesocyclops edax* was one of the most dominant crustacean zooplankton in small eutrophic lakes in Ontario (Brandl and Fernando 1979), Florida (Wyngaard et al., 1982) and Estonia (Haberman and Haldna, 2014). It was also the most abundant cyclopoid in Hamilton Harbour (67% of adult biomass) (Bowen and Currie, 2017), Sodus Bay (88% of adult biomass) (Makarewicz and Lewis, 2005) and the western basin of Lake Erie (54% of adult biomass) (Makarewicz, 1993). It is rarely encountered in the less productive open water of Lake Ontario.

Summary: Generally this indicator of fish planktivory has shown improvement since the early time stanza in the Bay of Quinte, but has been relatively stable in the upper and middle reaches since the early 1990s. At C, improvements were more gradual in the first three stanzas, and the largest improvement was evident in Stanza 4. This may relate to generally lower Alewife numbers in the open waters of Lake Ontario in the last decade.

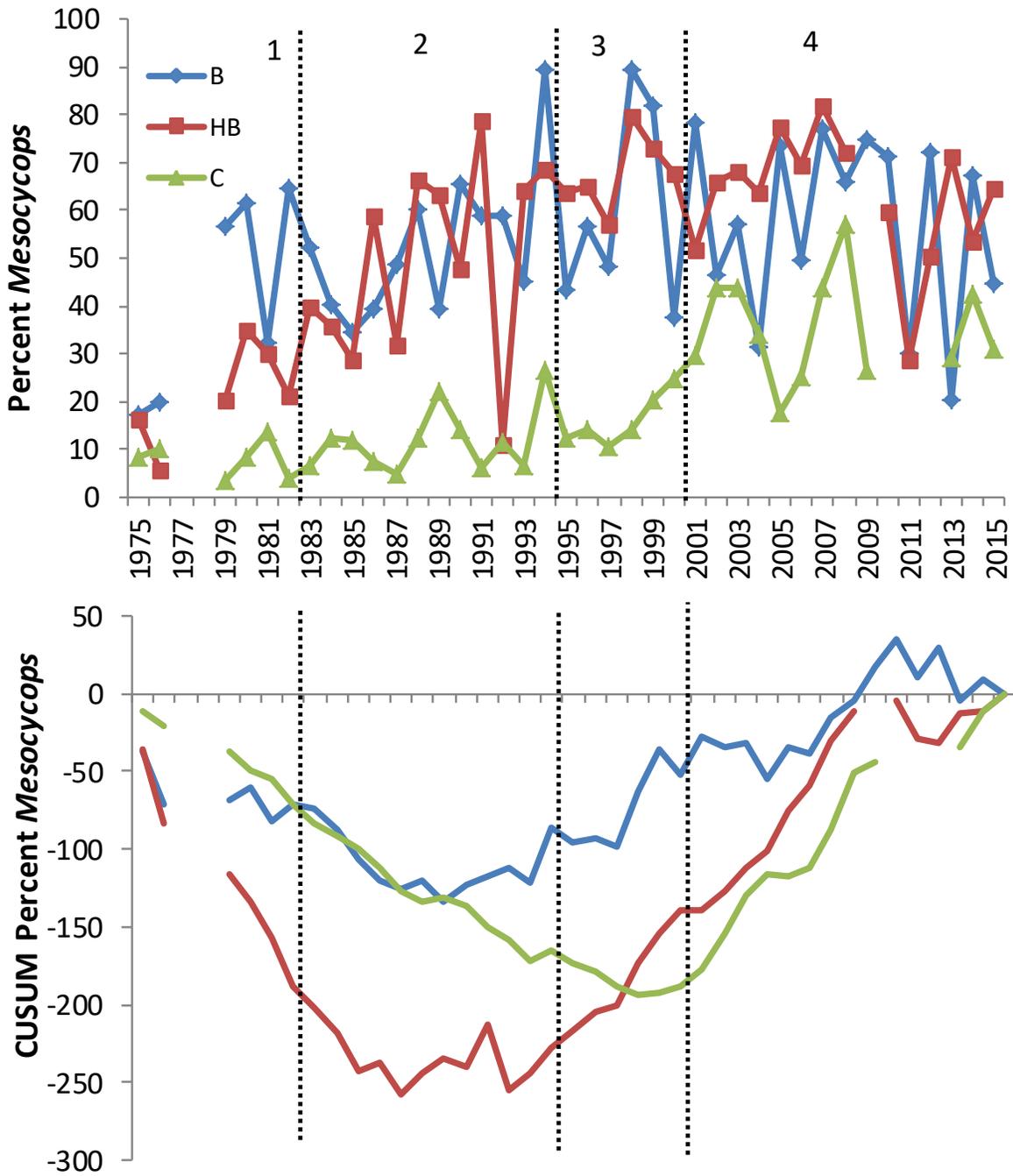


Figure 3.21: Changes in percent Mesocyclops (top) and the CUSUM trends (bottom) over time at the three stations, with time stanzas 1 to 4 shown by the dotted lines.

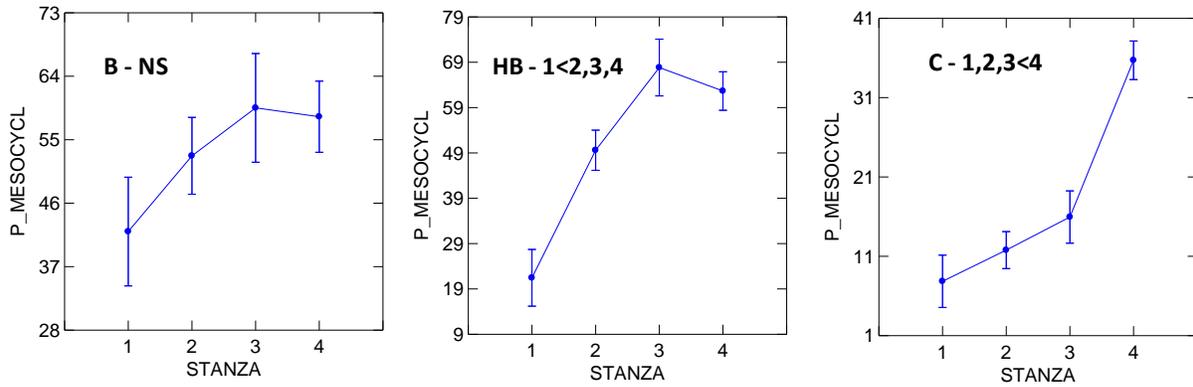


Figure 3.22: Percent Mesocyclops \pm standard error during each of the four time stanzas at the three stations, B, HB and C. Significant differences among stanzas are indicated on each graph. NS indicates there were no differences among stanzas.

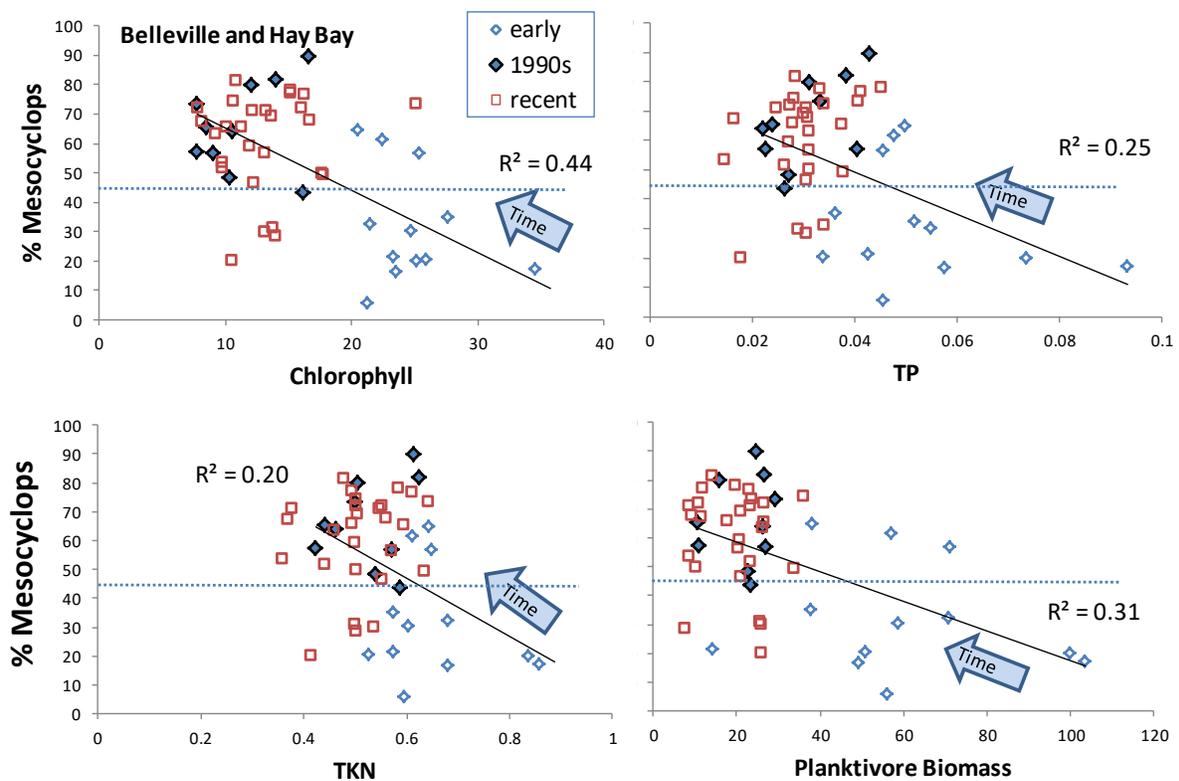


Figure 3.23: Regression relationships for percent Mesocyclops (dependent variable) plotted against eutrophication parameters (chlorophyll, TP, TKN and planktivore biomass) for B and HB combined. The slopes are for stanza 1 and stanza 3 points only, and the arrow shows the direction of movement over time. The stanza 4 points illustrate recent trends. The dotted line shows the 1995 to 2015 mean value - 1 standard deviation for B and HB combined.

3.3.8 Rotifers

Metric: A declining proportion of eutrophic rotifer taxa.

Rationale: Rotifers are small non-crustacean zooplankton that can be very abundant in freshwater systems worldwide. Rotifers are among the most widely accepted zooplankton indicator taxa (Čeirāns, 2007; Gannon and Stemberger, 1978; Sládeček, 1983). As the trophic state shifts to less eutrophic conditions in the Bay of Quinte, we expect to see a declining proportion of the eutrophic taxa.

Notes: In the Bay of Quinte, rotifers have only been collected since 2000 (Stanza 4) and therefore available rotifer data are coincident with a narrow range in trophic conditions. The group “eutrophic species” is the density sum of *Anuraeopsis fissa*, *Brachionus angularis*, *Filinia* sp., *Keratella cochlearis tecta*, *Pompholyx sulcata* and *Trichocerca* sp.

Station differences: There were only a few rotifer taxa that were significantly different among stations (Table 3.8). The densities of all taxa combined, the group “eutrophic species”, *Trichocerca pusilla* and *Keratella cochlearis* were significantly lower at C than B, and *Polyarthra remata* was lower at B than at HB. In terms of percent composition by density, % “eutrophic species” was lower at C than HB, and % *K. cochlearis* was lower at HB than B. For the species that prefer less eutrophic conditions, *Kellicottia longispina* was lower at B than C, , and *Synchaeta kitina* was lower at HB than C.

Regression Relationships Regression relationships for Quinte rotifers are described in Tables 3.5 to 3.7. Most of the common rotifers found in the Bay are ubiquitous taxa that are also dominant at other Great Lakes locations (e.g., Makarewicz and Lewis, 2015, Barbiero and Warren, 2011). During the 2000 to 2015 time period, when all stations are combined, the trophic gradient is sufficient to expect a rotifer response. In Quinte, the proportions of the group “eutrophic species”, *Trichocerca* sp. and *Pompholyx sulcate*, are related to eutrophic conditions. Conversely, *Kellicottia longispina* and *Synchaeta kitina* are indicators of decreasing eutrophy. These percentages are calculated relative to total rotifer density.

Mean Values and Comparison to Other Studies: Numerically, the most dominant rotifers in the Bay of Quinte are *Keratella cochlearis*, *Polyarthra dolichoptera* and *Polyarthra vulgaris* at all three stations, *Conochilus unicornis* at B and HB, and *Synchaeta kitina* at C (Fig. 3.24). Total

rotifer density (\pm SE) has averaged $184 \pm 26 \text{ L}^{-1}$ at B, $177 \pm 36 \text{ L}^{-1}$ at HB and $79 \pm 13 \text{ L}^{-1}$ at C. Unfortunately longer term trends in Quinte rotifer densities are largely unknown, as the only rotifer survey prior to 2000 that we know of was carried out in 1985 (Nogrady 1988). In 1985, seasonal rotifer density averaged 1690 L^{-1} in the Upper Bay (average of Trenton and Northport stations), 755 L^{-1} in the Middle Bay at Picton and 551 L^{-1} in the Lower Bay at Conway (Fig. 3.25). Therefore, relative to stanza 4, 1985 densities were about 9 times higher at B, 4 times higher at HB and 7 times higher at C. The dominant taxa in the mid-1980s were similar to the 2000s (*K. cochlearis*, *P. dolichoptera*, *P. remata*, *P. vulgaris* and *Synchaeta* sp.). In addition to declining trophic status, the recent drop in rotifer density may be caused by competition and predation by *Dreissena* (Pace et al., 1998), *Cercopagis* predation (Makarewicz and Lewis, 2015) and competition with large *Daphnia*. *Daphnia*, who are more efficient grazers of phytoplankton, may also physically damage rotifers while feeding (Gilbert, 1988).

Although rotifers are more numerically abundant than macrozooplankton, they comprise on average only around 3% of the total biomass due to their much smaller size. The proportion of rotifer biomass is not significantly different among stations. The current rotifer proportion in Quinte is similar to that observed in Hamilton Harbour (Bowen and Currie 2017) and in Western Lake Erie (Barbiero and Warren 2011).

Except for a few unusually high years, the proportions of the indicator taxa identified above have been relatively constant at the Quinte stations since 2000 (Fig. 3.26). The proportion of eutrophic species (\pm SE) has averaged $12.2 \pm 2.4\%$ at B, $11.8 \pm 1.6\%$ at HB and $4.4 \pm 0.7\%$ at C. Years when this proportion was unusually high ($>20\%$) were 2001, 2005 and 2007 at B; and 2003 and 2006 at HB. The CuSUM plot (Fig. 3.27) shows that values have been decreasing at B since about 2007, and at HB since about 2013. *Trichocerca* sp. comprised the majority of these eutrophic taxa, and this genus comprised $9.6 \pm 1.8\%$ at B, $9.6 \pm 1.6\%$ at HB and $3.9 \pm 0.6\%$ at C. Unusually poor years ($>16\%$) were the same as for eutrophic species. Corresponding values for *P. sulcata* were $1.8 \pm 0.7\%$, $1.8 \pm 0.4\%$ and $0.4 \pm 0.2\%$. Poor years ($>4\%$) were 2001, 2005, 2007 and 2012 at B; and 2007 and 2011 at HB.

Kellicottia longispina and *Synchaeta kitina* are indicators of decreasing eutrophy in the Bay of Quinte and were more abundant in the lower Bay (Fig. 3.25). The former averaged $0.3 \pm 0.1\%$ at B, $0.7 \pm 0.3\%$ at HB and $2.2 \pm 0.9\%$ at C, and the latter averaged $2.8 \pm 0.7\%$ at B, $1.9 \pm 0.4\%$ at HB and $7.1 \pm 2.5\%$ at C.

Many of the rotifer taxa identified as indicators of eutrophic conditions in the Great Lakes by Gannon and Stemberger (1978) have been absent or very rare in the Bay of Quinte since 2000. These include species the genus *Brachionus*, *Polyarthra euryptera* and *Anuraeopsis fissa*. Another eutrophic species, *Filinia longiseta* is commonly identified in Quinte, but at very low densities. Although most of these were found in the Bay of Quinte in 1985, they were not dominant species even at that time (Nogrady, 1988). Other eutrophic taxa, *Trichocerca cylindrica*, *T. multicornis* and *Pompholyx sulcata* are common Quinte rotifers in the 2000s. *Keratella cochlearis f. tecta* is commonly found, but its densities are much lower than the dominant form *Keratella cochlearis*.

Summary: Rotifer density and the proportion of eutrophic rotifer taxa are higher at B and HB relative to C. Although rotifer data prior to 2000 is very limited, densities appear to have declined in Stanza 4 relative to the mid-1980s. Overall the rotifer community is not indicative of a highly eutrophic assemblage.

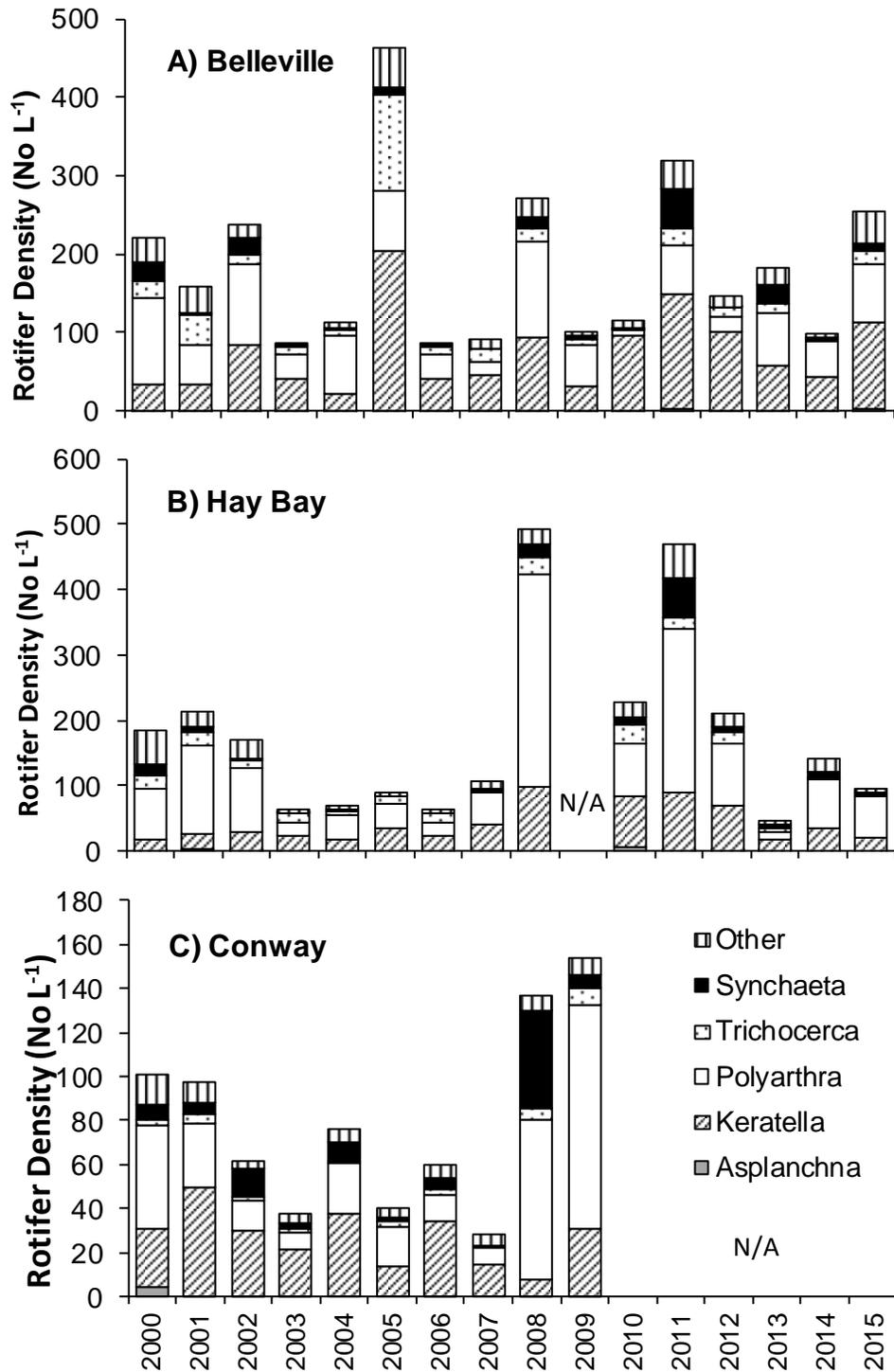


Figure 3.24: Densities of dominant rotifer taxa at the three stations in the Bay of Quinte. Note the different scales at C, and that no samples were collected at C after 2009, and at HB in 2009.

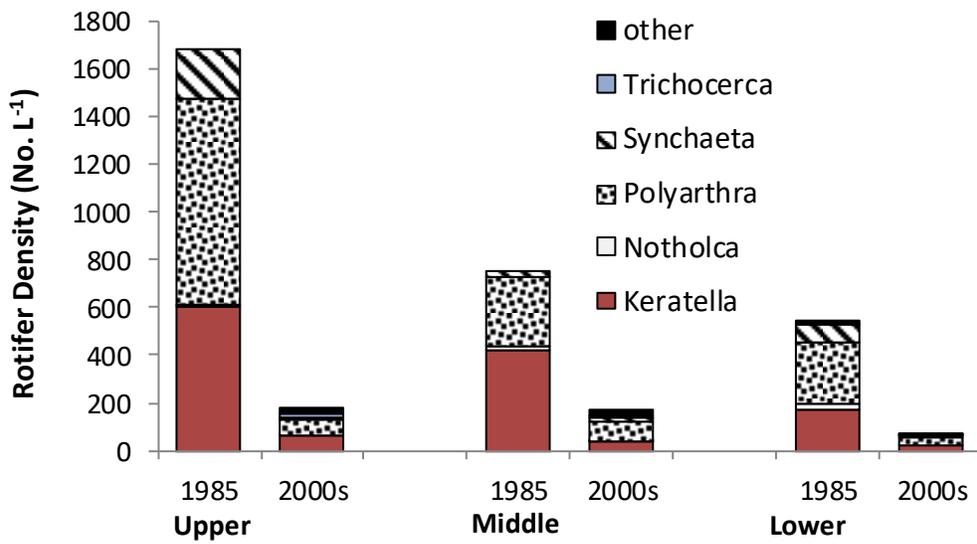


Figure 3.25: Densities of dominant rotifer groups in the 2000s compared to 1985 at the upper, middle and lower reaches of the Bay of Quinte (adapted from Nogrady, 1988).

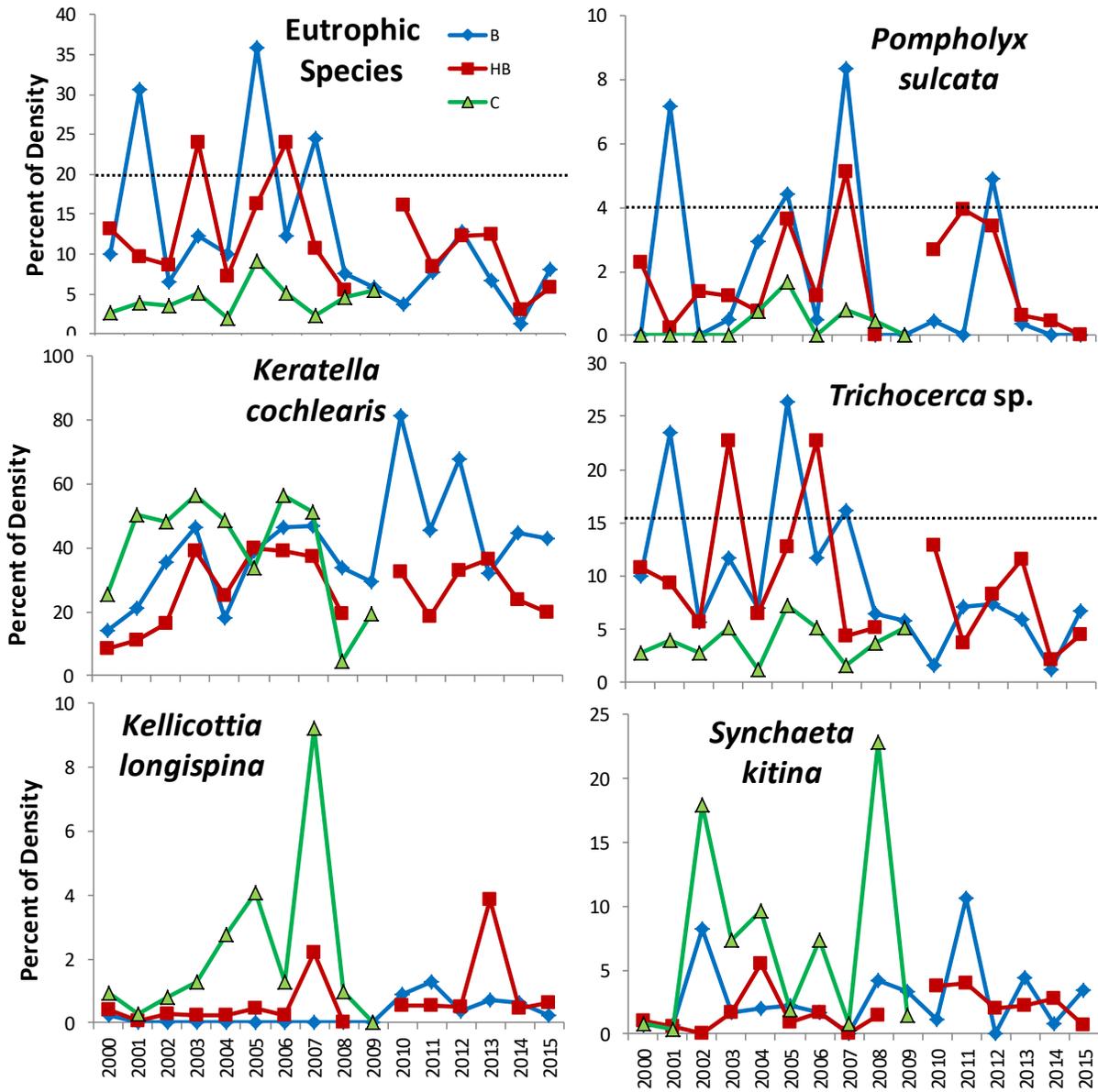


Figure 3.26: Percent by density of selected rotifer taxa at the three stations in the Bay of Quinte. Note that no samples were collected at C after 2009, and at HB in 2009. The dotted line is the mean plus 1 SD.

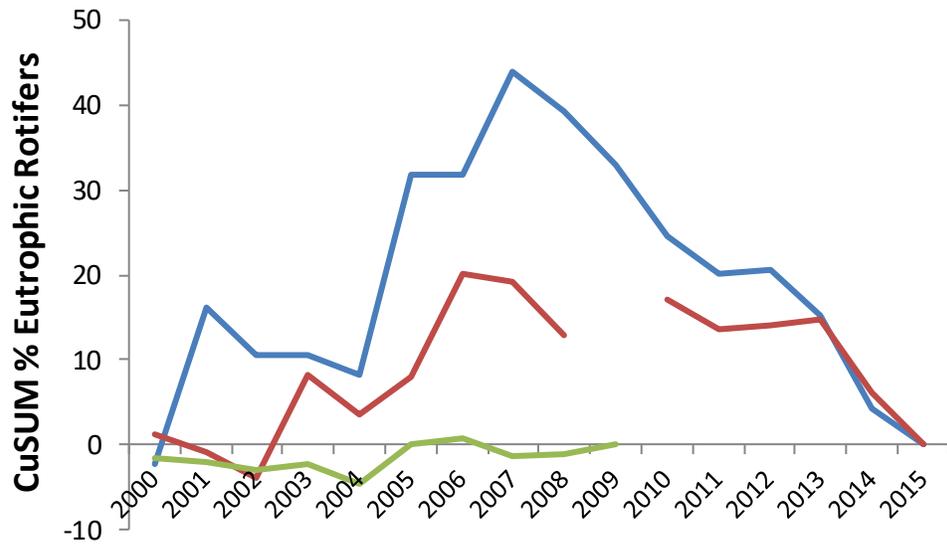


Figure 3.27: CUSUM trends over time for percent eutrophic rotifer taxa at the three stations in the Bay of Quinte.

3.4 Summary of Zooplankton Indicators:

Good and poor years for each of the zooplankton indicators in the upper and middle bays are shown in Table 3.9. Threshold values for each indicator are the mean of the 1995 to 2015 period for B and HB combined, plus or minus one standard deviation, depending on the expected direction of the indicator. For example, values below 0.37 mm were considered poor for cladoceran mean length. The number of “poor” scores obtained each year (out of a possible 8 in the recent stanza) were tallied to illustrate which years the zooplankton community was especially poor. “Bad” years where 3 or more indicators scored in the poor range were 1975, 1976, 1979, 1981-1987, 2000, 2002, 2004, 2011, and 2013 at B, and 1975 – 1982, 1984 – 1987, 2006, and 2011 at HB. Generally there were more “poor” years in stanza 1 and the first part of stanza 2, and in the recent stanza, there have been more “poor” years at B than HB. Conversely, green shading indicates that the metric has been better than average.

A trend analysis ordination was also created for all associated measures which compares trends in the short term (last 5 years measured vs. Stanza 4) compared to the long term (stanza 4 vs all years). The trend analyses is meant to highlight how all the indicators are performing in both the short and long term at all stations (Fig. 3.28). These values were computed as:
$$X = (AVG_{\text{Last 5 yrs}} - AVG_{\text{Stanza 4}}) / (AVG_{\text{Stanza 4}}); Y = (AVG_{\text{Stanza 4}} - AVG_{\text{All Years}}) / (AVG_{\text{All years}})$$
A point falling in the “Slipping” quadrant indicates there has been improvement in the long-term, but not in the short-term, “lagging” indicates backwards direction in both the short and long-term, “improving” indicates short-term but not long-term improvement, and “recovering” indicates improvement in both the short and long-term.

Each of the indicators are summarized in the following paragraphs:

Cladoceran mean size: this indicator showed improvement from the 1970s and early 1980s at B and HB. It was highest at these stations during the mid-1980s to about 1998. Overall, its central position in Fig. 3.28 suggests that it has been relatively constant in the short term (5 years) and long-term (stanza 4) at all stations.

Chydorus Biomass: At B and HB, *Chydorus* biomass was highest in stanzas 1 and 2 and has generally shown long-term improvement. *Chydorus* biomass is about an order of magnitude lower at C, where it has not really improved over time. Except for three high years in the last decade, *Chydorus* biomass at B has generally been below the 1995 to 2015 mean. Although there has been year-to-year variability, the short-term indicates slippage at B (Fig. 3.28). *Chydorus* biomass in the upper and middle bays of Quinte is generally higher than in other shallow productive systems in Ontario.

Percent *Acanthocyclops vernalis*: This indicator has shown steady long-term improvement in the upper and middle bays, and values have continued to improve in the short-term at B (Fig. 3.28). It likely is not a very effective indicator at C, where it comprises a much lower proportion of cyclopoid biomass.

Shannon Diversity Index: This metric has been relatively stable over the time series in the Bay of Quinte, as shown by its central position in the ordinations in Fig. 3.28. Recent slippage at C is associated with high densities of dreissenid veligers. Overall, it does not appear to be overly sensitive as an indicator. Scores in the Bay of Quinte are equivalent to, or often higher than those obtained from other eutrophic systems.

Percent Large *Daphnia*: Although there was some early improvement in the proportion of large *Daphnia* in the upper and middle reaches of Quinte, this parameter has stagnated since the mid-1980s (Fig. 3.28). It has recently shown improvement at C.

Percent *Daphnia galeata mendotae*: In upper and middle Bay of Quinte, the proportion of *Daphnia galeata mendotae* biomass has increased since the mid-1970s and early 1980s when planktivory was very high. Long-term improvement has been minimal at B and HB, but greater at C (Fig. 3.28). Although there has been considerable year to year variation, this metric has shown recent slippage at B and HB.

Percent *Mesocyclops*: Generally this indicator of fish planktivory has shown improvement since the early time stanza in the Bay of Quinte, but has been relatively stable in the upper and

middle reaches since the early 1990s. However, there has been some slippage at B and HB in the last 5 years. It has shown long term improvement at C, but little change in the short-term.

Rotifers: Rotifer density and the proportion of eutrophic rotifer taxa are higher at B and HB relative to C. Although rotifer data prior to 2000 is very limited, densities appear to have declined in Stanza 4 relative to the mid-1980s. However, taxonomic composition is similar in the two time periods. The proportion of eutrophic species appears to have declined somewhat since about 2007 in the upper bay. Some of the most widely recognized eutrophic indicator taxa (e.g. *Brachionus*, *Filinia* and *Anuraeopsis fissa*) are found infrequently in the Bay of Quinte, and overall the rotifer community is not indicative of a highly eutrophic assemblage.

Overall Summary: For most of the indicators, there has been improvement relative to stanza 1 when nutrients and planktivory were very high. However, there has generally been little ongoing improvement in the last decade or so, and in some cases evidence of slippage. Until dramatic changes in trophic status are achieved, the zooplankton community is likely to remain relatively unchanged. Based on this suite of indicators, the zooplankton community has improved over the long-term, however, there are signs that these improvements have declined in recent years.

Table 3.9: Zooplankton indicator summary for B and HB. Mean values ± 1 SD are given for the 1995 to 2015 period for B and HB combined. The desired direction for each indicator is given. Cells shaded in pink were the “poor” years ($> \text{mean} + 1$ SD for *A. vernalis* and *Chydorus*; $< \text{mean} - 1$ SD for the rest); whereas the green cells were the “good” years ($< \text{mean} - 1$ SD for *A. vernalis* and *Chydorus*; $> \text{mean} + 1$ SD for the rest). For the white cells, the annual mean fell within the mean ± 1 SD range. No data were available for grey cells. The number of indicators which fell in the “poor” range were tallied for each year.

year	Belleville									Hay Bay								
	cladoceran Mean Size	<i>Chydorus</i> BM	% <i>A. vernalis</i>	Diversity index	<i>Daphnia</i> >0.75 mm	% <i>Daphnia</i> galeata	% <i>Mesocyclops</i>	% Eutrophic Rotifers	No. "poor"	cladoceran Mean Size	<i>Chydorus</i> BM	% <i>A. vernalis</i>	Diversity index	<i>Daphnia</i> >0.75 mm	% <i>Daphnia</i> galeata	% <i>Mesocyclops</i>	% Eutrophic Rotifers	No. "poor"
desired	+	--	--	+	+	+	+	--		+	--	--	+	+	+	+	--	
mean-SD	0.37	1.6	-2.6	1.82	0.48	7.9	45.5	4.1		0.37	1.6	-2.6	1.82	0.48	7.9	45.5	4.1	
mean+SD	0.48	9.5	15.4	2.25	0.65	32.4	78.3	20.0		0.48	9.5	15.4	2.25	0.65	32.4	78.3	20.0	
1975									5									3
1976									6									3
1979									5									3
1980									2									3
1981									3									4
1982									5									5
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2010									1									0
2011									3									5
2012									0									1
2013									5									0
2014									1									0
2015									1									0

Zooplankton indicators

- ◆ chyd_BM
- Daph_>750
- ▲ P_Meso
- ★ Diversity
- %_DGM
- clad_mean_length
- ▼ P_vernalis

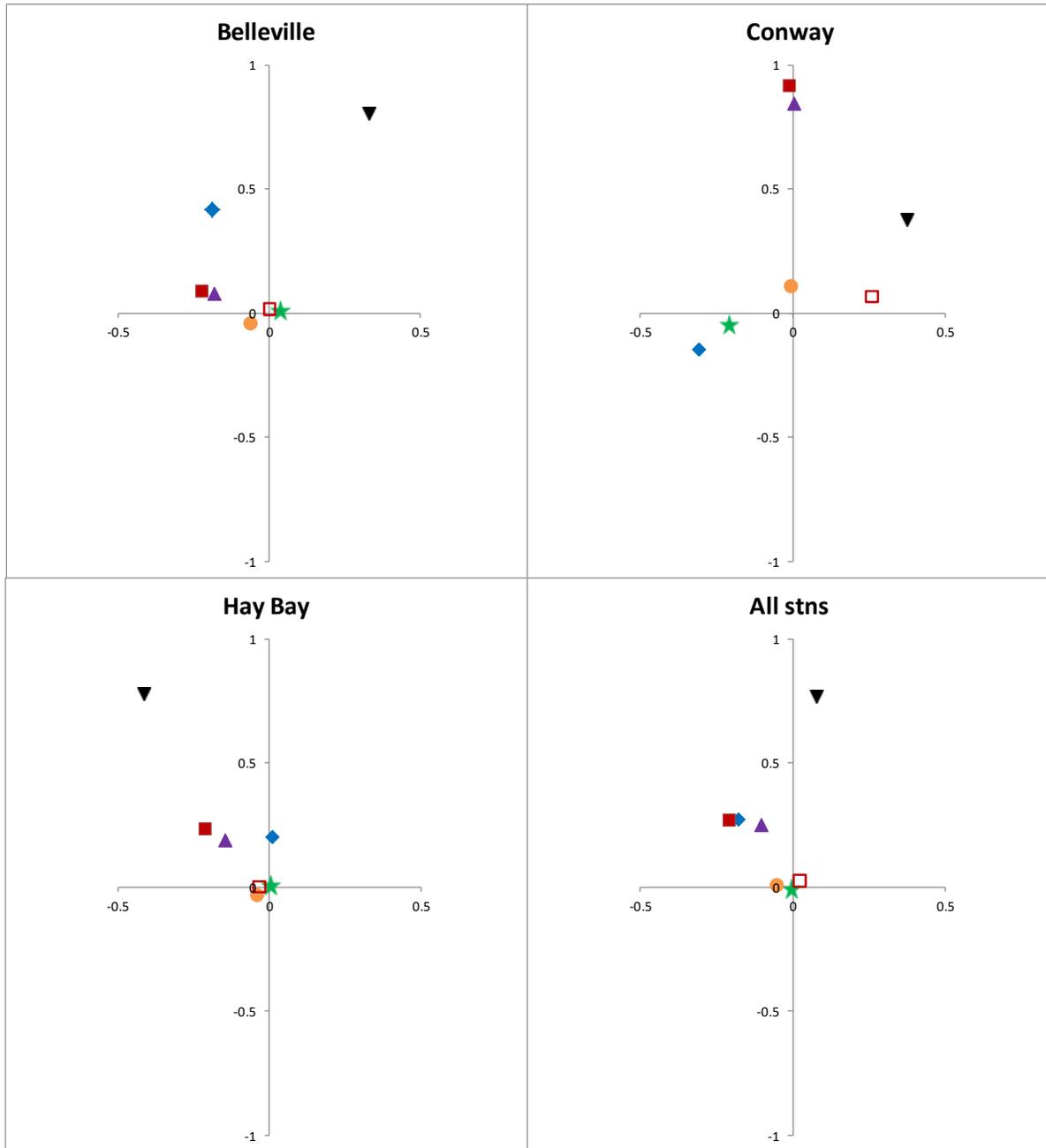
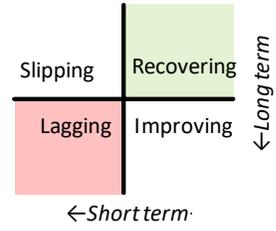


Figure 3.28: Ordination by station of zooplankton metrics showing changes in the short term on the x-axis (last 5 years relative to stanza 4 mean) and long term on the y-axis (stanza 4 mean relative to overall mean).

3.5 Future Monitoring of Zooplankton in the Bay of Quinte:

3.5.1 Overview

While zooplankton monitoring during Project Quinte from 1975 to 2015 was based on biweekly sampling from early May to the end of October (usually 13 cruises), this may not be feasible moving forward. The minimum required for achieving accurate zooplankton data is monthly sampling (Currie et al. 2015).

Moving forward, we would expect to see a mean annual *Chydorus* biomass of less than 10 mg m⁻³ for most years. We would also suggest removing the Shannon Diversity Index as an indicator due to its apparent insensitivity and taxonomic complexities in its calculation.

It is recognized that future monitoring programs in the Bay of Quinte may be faced with tighter budgetary constraints, and that it may not be possible to contract every sample to enumeration by a trained zooplankton taxonomist. In the future, one possibility may be to have individual samples (discrete dates) counted by a trained staff member of the organization carrying out the monitoring program. Identification of taxa would be to the lowest possible level based on the ability of the person conducting the counts, but would likely not be to species level for the more difficult taxa. A seasonal composite sample comprised of half of each of the individual date samples could be prepared and sent to a taxonomist for complete species identification and measurements. Since 2000, only rotifer seasonal composites have been counted, and it is recommended to continue with this approach. An analysis of previous seasonal composite zooplankton samples compared to individual date samples follows.

3.5.2 Enumeration of Seasonal Composites

For taxonomic enumeration, zooplankton annual or seasonal composite samples are sometimes created by combining samples collected on a number of sampling dates. This may be done due to budgetary constraints or lack of availability of taxonomic expertise. While they may provide a reasonable estimate of annual biomass, density and taxonomic composition, seasonal variability is lost and the resulting data may not be as accurate. The total species richness counts are also usually less on composite samples simply because the number of animals identified and counted overall is much lower.

Prior to 1995 in the Bay of Quinte, zooplankton seasonal composite samples were analyzed with the primary goal of obtaining size data. In addition to measuring animals, densities and biomass of individual taxa in the seasonal composites were also calculated. However, the identification of several of our indicator taxa (*D. galeata mendotae* and *A. vernalis* in particular) appeared to be inconsistent for some samples. When these were omitted and percentages were calculated for the remaining samples, the seasonal composite values and the means of the individual date samples were compared for each station using a paired t-test (Fig. 3.29). There were no significant differences for % *D. galeata mendotae*, % *Mesocyclops* and % *A. vernalis*. For *Chydorus* biomass, the only significant difference was at HB.

For these same indicators, least squares regression analyses for the individual dates compared to the seasonal composites were completed (Fig. 3.30). The best agreement between the two counting methods was obtained for *Chydorus* biomass and % *Mesocyclops*, with r^2 values of 0.82 and 0.76, respectively. Agreement for % *D. galeata mendotae* and % *A. vernalis* was much poorer. These were the two taxa with the most uncertainty regarding proper identification in the seasonal composites.

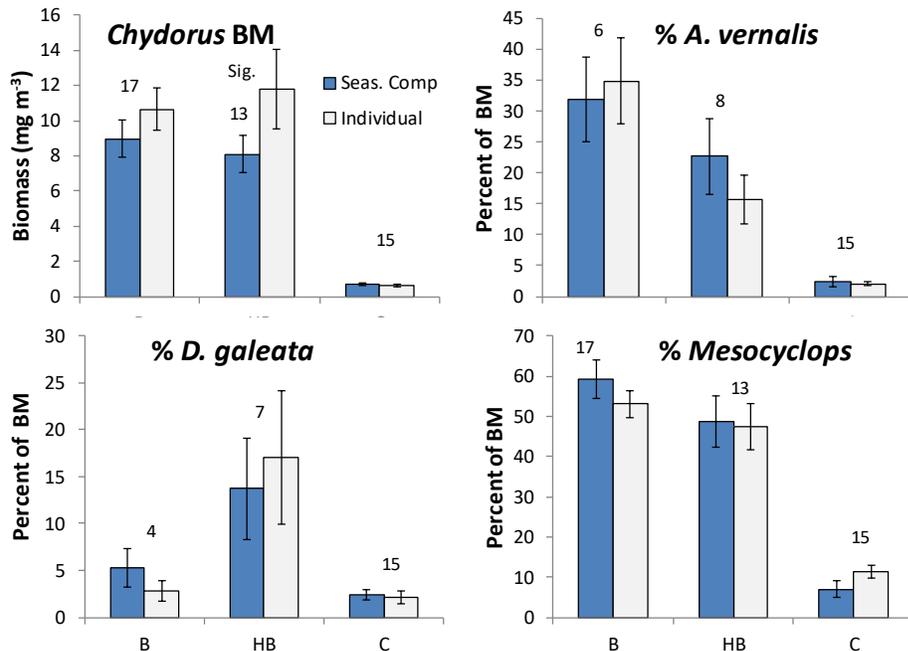


Figure 3.29: Comparison of seasonal composite samples and the means of individual dates for chosen zooplankton metrics. Sample sizes for the paired t-tests are given above each bar pair.

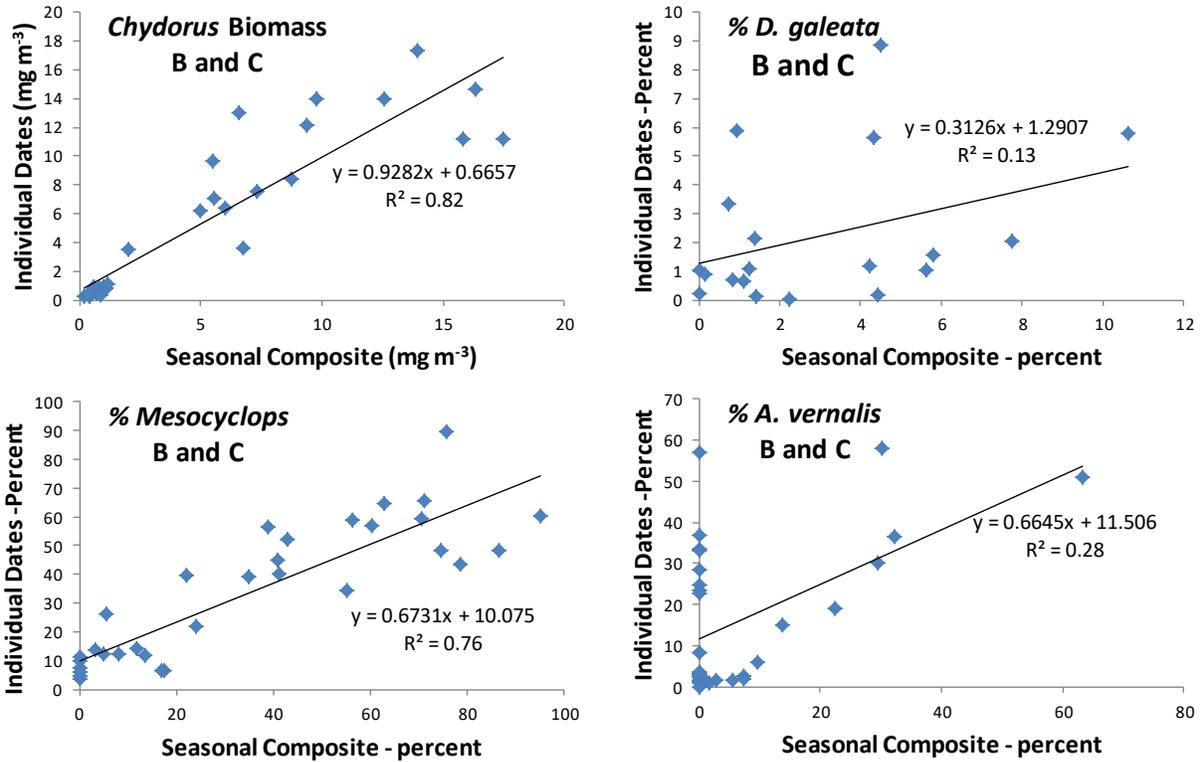


Figure 3.30: Regression analyses (individual dates vs seasonal composites) for chosen zooplankton metrics (1976 to 1995), B and C only.

Overall, these analyses suggest that enumerating seasonal composite samples can provide reasonable estimates for the indicators tested, provided adequate care is taken to ensure a sufficient number of animals are enumerated in the composite samples to provide sound estimates.

3.5.3 Zooplankton Indicators - Ease of Use

Cladoceran Mean Size - is easy to determine, as it only requires zooplankton identification to the cladoceran level and length measurements of each individual.

***Chydorus* Biomass** - It is easy to identify *Chydorus* in zooplankton samples provided the dissecting scope used is of sufficient power (at least 40X magnification).

% *Acanthocyclops vernalis* – this is a more challenging indicator as it requires greater taxonomic expertise to identify this species.

Shannon Diversity Index – This is a more challenging metric to calculate in a manner consistent with past work in the Bay of Quinte, as it requires taxonomic identification to the lowest possible level. Given its difficulty and lack of sensitivity, we recommend dropping this indicator.

% Large *Daphnia* - It is easy to determine the percentage of large *Daphnia* (those >0.75 µm) as all that is required is to identify zooplankton to the *Daphnia* level and measure the length of each individual.

% *Daphnia galeata mendotae* – This metric is more challenging as identifying preserved *Daphnia* to species can be difficult. It partly depends on the shape of the head, which is variable. The ocellus of *D. galeata mendotae*, an identifying feature, can fade in preserved samples. It is easiest to spot in unpreserved or freshly preserved specimens.

% *Mesocyclops*. *Mesocyclops edax* is relatively easy to identify.

Rotifers - Rotifers are generally more difficult to identify than crustacean zooplankton (especially to the species level) and require a good compound microscope and a trained taxonomist. As all the *Trichocerca* species in the Bay of Quinte respond positively to eutrophy, using the genus level (*Trichocerca* sp.) rather than species simplifies identification. *K. longispina* is an easy rotifer to identify.

Status of BUI13 and Recommendations for Future Monitoring

There has been improvement in the status of the phytoplankton and zooplankton populations in the Bay of Quinte over the long-term (compared to pre-phosphorus controls), but there is evidence of continued deterioration in the status in the upper and middle bay since 2001.

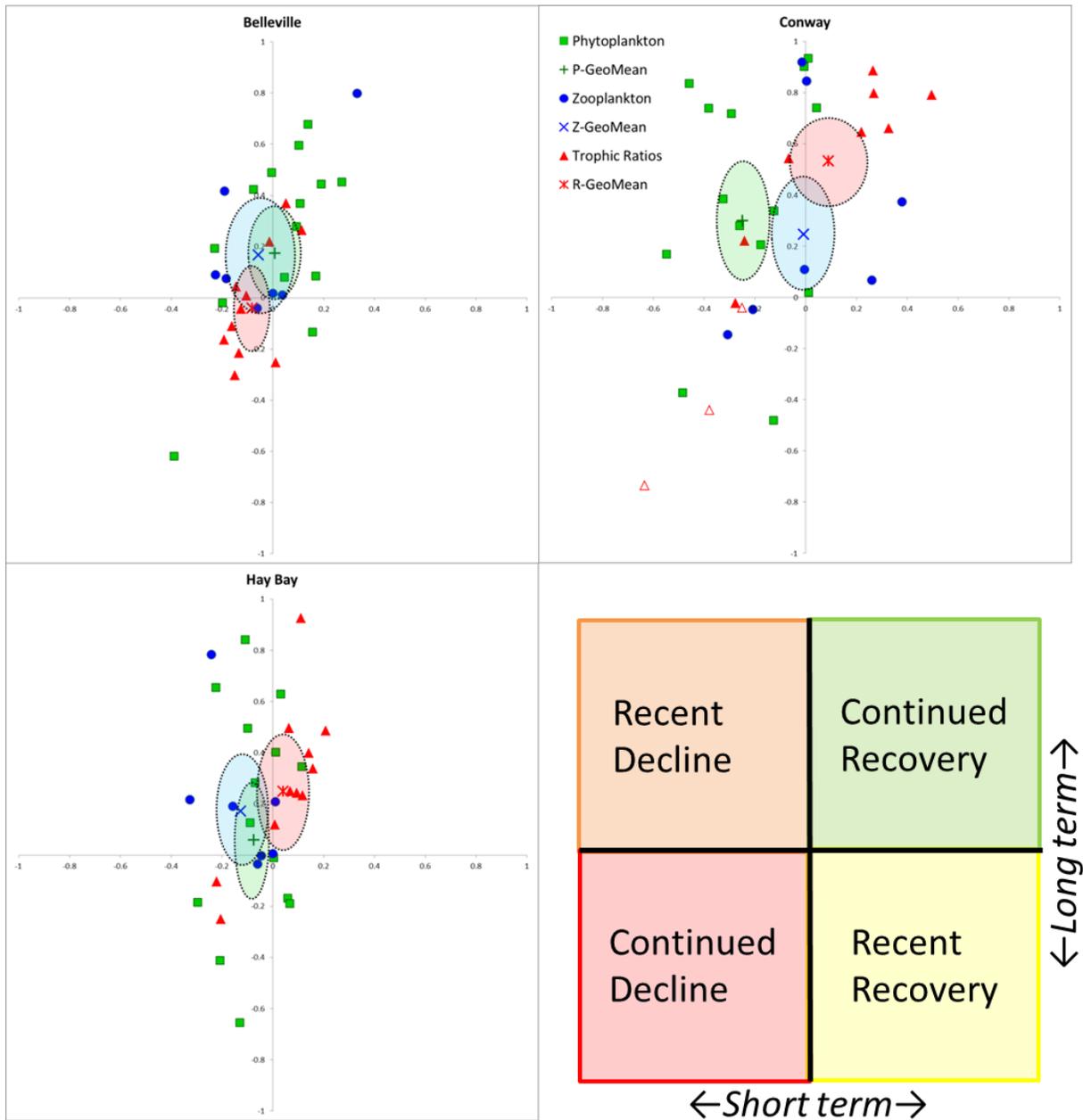


Figure 4.1: Ordination plots of short-term (since 2001) vs. long-term (since 1972) improvement of the metrics normalized to mean (where improvement of 1 is an improvement equal to the mean) for phytoplankton, zooplankton and trophic ratios for upper (Belleville), middle (Hay Bay) and lower (Conway) Bay of Quinte. Shaded ellipses indicate 95% confidence limits. Unfilled red triangles for Conway indicate metrics using piscivores which were excluded from the trophic ratio ellipse due to issues of non-resident fishes biasing the biomass estimates.

We identified 13 trophic metrics, 5 phytoplankton specific metrics and 8 zooplankton metrics and applied a weight-of-evidence approach to determine change over time (Fig. 4.1). We used an ordination technique to determine the relative improvement of these metrics over long time periods (the entire Project Quinte time-series) and recently since 2001. The upper bay exhibited no long-term improvement in trophic (food-web ratio) metrics and a short-term decline. Phytoplankton and zooplankton metrics indicated long-term improvement but no change or slight decline more recently. Middle bay exhibited improvement in trophic ratios over the long-term and slight continued improvement in the short-term. Zooplankton indicated improvement over long-term but short-term declines. Little change occurred in the phytoplankton metrics and improvements were much less in the long-term compared to upper bay indicating that the middle bay has declined to the point where there is little difference from the upper bay post-phosphorus controls. The lower bay indicated significant improvement over the long-term, but only slight or no improvement in zooplankton and trophic ratios and a significant decline in phytoplankton metrics in recent years.

Overall we see very strong bottom-up effects controlling biomass of phytoplankton and fishes, but only weak top-down factors in the upper Bay of Quinte. There has been a shift from high planktivore biomass values for Alewife and Gizzard Shad during the earlier turbid phase to White Perch and Yellow Perch during the clear phase (after 1994). In spite of this, zooplankton biomass ratios show little grazing effects by planktivorous fishes or with changes in piscivorous fish biomass. Some compositional changes (e.g. in *Daphnia galeata*) indicate that grazing effects on the zooplankton community are occurring, but the changes are not significant in the upper bay, though they are significant and strong in the middle and lower bay. This lack of change indicates that the trophic efficiency of the upper Bay of Quinte has not yet improved over time as it has in the middle bay site, even though many of the biomass and nutrient measures are not significantly different now between Belleville and Hay Bay. There was an expectation of strong changes, particularly to the Z:P ratio in the upper bay with the change to the clear phase after 1994, however this did not occur. This is likely due to the underlying effect of summer sediment phosphorus release into the mixed, shallow water system, whereas Hay Bay is deeper and is generally stratified through the summer. This sediment-mediated release of nutrients is driving nitrogen limitation (given by the strong TKN relationship as opposed to TP in the upper and middle bays) and promoting the growth of filamentous and colonial phytoplankton which are not efficiently converted into zooplankton biomass. As sediment

phosphorus values decrease over time, this effect is likely to be moderated and the Z:P ratios should increase to those found in typical eutrophic-mesotrophic environments.

The zooplankton and phytoplankton metrics show improvement from the period of pre-phosphorus controls but have declined since the 1980s. The Bay of Quinte went through a state change between 1994 – 1995 from a turbid to a clear phase system and as expected with this, total algal biomass decreased sharply in 1995. Zooplankton biomass did not increase during the clear phase, but relative composition of *Daphnia galeata*, a large zooplankton grazer, has increased post-phosphorus controls at all sites except Belleville. The relative biomass of smaller less efficient grazers that thrive in eutrophic environments (*Chydorus sphaericus* and *Acanthocyclops vernalis*) have declined. Most other metrics indicated little change over time in the upper bay, but some long-term improvements in the middle and lower bay, though little change since the onset of the clear phase.

After declines due to phosphorus controls, nuisance algal biomass, in particular, filamentous cyanobacteria and diatoms, have been stable since the 1990s. In contrast colonial cyanobacteria biomass has not changed since the 1970s in spite of overall biomass declines. These forms of algae are difficult for zooplankton to handle and interfere with the transfer of energy to the upper trophic levels. Furthermore, there is an indication of production being exported from the upper bay into the lower bay in recent years given the decrease in the metrics. This is likely in part due to the summer nitrogen limitation in the upper bay which results in phosphorus being exported into the middle and lower bay where nitrogen is no longer limiting. This is supported by the recent phytoplankton improvements in Belleville being greater than those found at Napanee and deterioration indicated in the middle and lower bay, with scums and algal blooms being present in the lower bay.

Overall, improvements to the upper Bay of Quinte plankton food-web composition and function have improved only over the period of pre to post-phosphorus controls, and have since stalled with little indication of improvement since 1985 when the AOC was designated. Increased trophic efficiency (transfer to higher trophic levels) has been seen in the trophic ratios in the middle and lower bay though zooplankton and phytoplankton metrics have declined or shown no improvement, likely due to the export of phosphorus and potential production away from the upper bay where nitrogen limitation occurs. The expectation is that the upper bay will exhibit trophic improvements and likewise lead to recovery in the middle and lower bay zooplankton

and phytoplankton community measures when the excess organic sediment phosphorus is reduced. Given the lack of improvement in the metrics since the 1990s, and evidence of recent decline, the status of BUI 13 for the Bay of Quinte should remain unchanged as impaired. If an Area in Recovery designation is desired, a monitoring plan should be instituted to determine if the recent declines in status show improvement.

Future Monitoring

The ideal situation in establishing a monitoring plan would be to sample phytoplankton and zooplankton biweekly, but this is expensive and labour intensive, and it is understood that budgets will likely constrain this. Zooplankton and phytoplankton in particular, are notoriously temporally variable and reductions in sampling may give trends or compositional changes which are not genuine. It has been determined that monthly sampling May – October is an effective trade-off of sampling resolution and costs. There is a potential for limited taxonomy to be done in-house for zooplankton with composite samples sent out for detailed taxonomic analysis and biomass estimates. However, composite samples are not recommended for phytoplankton samples which showed bias and overestimated counts and biomass of larger algae. Samples can be collected and archived if preserved appropriately, so if there are indications that condition is of concern, the samples can always be counted by a taxonomist at a later date. Thus the preferred sampling intensity would be: 1) biweekly, 2) monthly, 3) seasonal, 4) single sample in August (when diversity is highest).

We also recommend that sampling continue at the major Bay of Quinte stations B, N, HB and C if possible. There has been indication of recent lapsing of status (particularly in phytoplankton) in the middle and lower bay (HB and C) and it will be important to track this. One possibility may be to pair samples in the littoral region with the long-term stations collected at mid-channel. This is being evaluated by GLLFAS to determine if improvements in the upper bay may be found in the littoral zone's increase in macrophytes as habitat for zooplankton. It should also be noted that there are indications that the upper bay of Quinte may switch back to a stable turbid phase. If this occurs it is recommended that biweekly sampling be prioritized because major ecosystem changes are expected when this occurs.

Several data gaps have been noted. Foremost are estimates of planktivory, particularly by larval and YOY fishes in the upper bay. The lack of response of zooplankton and trophic ratio metrics may in part be due to the uptake of biomass by these fishes. There is a weak top-down

response on the community from planktivorous fishes, and an accurate measure of their abundance and diet is likely to improve the results. This is additionally important because the biomass of planktivorous fishes is just as well related to nutrients as phytoplankton, but because the biomass increases much less with a given value of nutrient, total fish biomass is likely to continue to decrease under nutrient controls. This drop in biomass is already being seen in the zooplankton data which has had all-time low biomass values in recent years, which may be problematic for larval fish recruitment. Secondly, the role of macrophytes on the distribution and composition of zooplankton (and larval fishes) may help to explain the invariance of trophic ratios in the upper bay. Finally, a study of the nutrient limitation on phytoplankton in the upper and middle bay is needed. Minimum N:P ratios have not changed since 1990 and it is not known if nitrogen limitation is driving the biomass and composition of phytoplankton to filamentous forms during the peak bloom period in July – August. Nitrogen augmentation experiments can determine this.

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