Macroinvertebrate Communities in Great Salt Lake Impounded Wetlands, Their Relationship to Water and Sediment Chemistry and to Plant Communities and Proposed Modifications to the MMI

Report to

Jordan River/Farmington Bay Water Quality Council

By

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Introduction

Following comments by various stakeholders, and the Division of Water Quality the Jordan River/Farmington Bay WQC initiated a research program focused on the health of Great Salt Lake wetlands to determine if these important habitats support waterfowl and shorebirds that depend on these resources. Intense studies began in 2004. The goal was to identify and test biological indicators of the plant, macroinvertebrate and avian communities that would be useful for assessing the health and functionality of these wetlands. We have simultaneously measured physical characteristics and chemical constituents with quantitative measures of the plant and macroinvertebrate communities. We continue to search for several potential stressors that might be influencing the wetlands' biological condition and functionality.

During these past years and particularly in this study, we have enlisted experts in relevant environmental disciplines to collect and evaluate appropriate, co-located, sediment and water physical and chemical data with plant, macroinvertebrate and avian community measures. Data collection and analysis from the 2004 through 2006 field seasons were summarized in the Phase 1 progress report (Miller and Hoven 2007). Detailed plant community analysis and shorebird nesting habitat utilization, nesting success and food habit studies indicated that the fringe sheetflow (shorebird) habitat supported a densely populated community of nesting American avocets, black necked stilts and other shorebirds. Reproductive/nesting success and chick survival was among the highest rates measured in the world, particularly in areas where depredation programs were employed. In addition, preferred food items, including midge larvae and corixid (water boatman) aquatic bugs were abundant except in areas that had experienced intense pesticide spraying for mosquito control. In those areas, diets switched to plant parts, particularly seeds (Cavitt 2007). Conversely, Miller and Hoven (2007) identified potential issues of concern for the impounded wetlands including: the degree of development, density, and health of the submerged aquatic vegetation (SAV); timing of late summer or fall SAV senescence (in relation to the arrival of staging and migrating waterfowl); production/availability of dietary components of waterfowl (invertebrate and plant items), both during spring nesting and fall staging; the presence and potential adverse affects of surface mats of algae or duckweed; and whether nutrients or other chemical constituents (sulfides, ammonia or toxic metals) may be related to different community characteristics within and among different impoundments.

We report here on the continued research, which includes field studies performed during the 2010 field season. We intensively focused on chemical and metal analysis in the water column, sediment pore water and whole sediment digestions in order to support three primary objectives: 1) describe the macroinvertebrate community composition among ponds that were suspected of being stressed by various environmental factors and compare these impoundments against those that were considered reference or least-impacted; 2) Expand the data base beyond water column nutrients to other

chemical constituents and metals in surface water, pore water and whole sediment to investigate the potential for other factors that may influence the biologic community; and 3) During the same season investigate the relationship and value of macroinvertebrate species to the waterfowl that use these impoundments to feed, rear their young and provide essential food resources during fall staging. This last objective is intended to identify potential metrics which provide direct linkage to the designated beneficial uses (identified as: support for waterfowl and shorebirds, other water oriented wildlife and the aquatic life in their food chain). Evaluation for part of the third objective included harvesting adult "dabbling" ducks (the primary users of these impoundments), and identifying, weighing and counting waterfowl stomach and esophageal contents and performing other measures and observations related to the general health of birds that were sacrificed. These analyses were reported earlier (Wilson et al. 2011). The present report addresses macroinvertebrate relative abundance, density and biomass in the ponds where the waterfowl were harvested and provides an evaluation of stomach contents vs the presence, abundance and biomass of specific food items during the spring nesting and fall staging seasons. Statistical analyses were performed to evaluate the relationships between environmental chemistry and the plant and macroinvertebrate community. We also include a brief discussion of the health and condition of the macroinvertebrate community in terms of complexity and whether appropriate wetland niches and functional feeding groups are represented in these ponds. Finally we suggest additional/alternative metrics to be used in the assessment of beneficial use support. These metrics are chosen on the basis that they are essential components of a healthy wetland ecosystem or demonstrate direct strong linkage to the waterfowl in terms of dietary value or essential habitat or water quality characteristics that are essential to support the macroinvertebrate community. These potential metrics are discussed in the context of recent reports that also identify biological indicators and assess these metrics against environmental stressors of concern.

Methods

Seven impoundment wetlands were studied during the summer and fall of 2010 Samples. These ponds included Farmington Bay WMA Unit 1 (FB1) and Unit 2 (FB2), Ambassador W1 (AM), Newstate Pond 47 (N47), Bear River Migratory Bird Refuge 4C (BR), Public Shooting Grounds Pintail (PN) and Widgeon (WD) ponds. Macroinvertebrate samples were collected monthly from June through September, 2010 and were carefully co-located with the SAV measurements (Hoven et al. 2011) and water column, pore water and sediment chemistry and metal analyses (Johnson et al. 2011). One-square-meter samples were collected in triplicate using a sweep net. These samples were composited and preserved in formalin or ethanol. Most samples were collected off-shore in order to sample the SAV community. Triplicate samples helped to account for local heterogeneity. Emergent vegetation near the shoreline was sampled to compare species lists and density between emergent and submerged vegetation habitats. Sampling in AM was performed in three habitats: 1) at sediment depths of 5 cm, to sample

the true benthic organisms, 2) within the SAV community by sweeping the bottom of the net across the sediment/water interface and through the standing SAV and 3) within the emergent vegetation near the shoreline by bumping the bottom of the sweep net along the sediment surface and root systems and against the plant stems. A comparison of the abundance and biomass of macroinvertebrates among these habitats will be addressed in greater detail in a future report. Samples were sorted, identified, counted and weighed in the laboratory of Ecoanalysts, Inc. of Moscow, ID. Appropriate multivariate and bivariate statistics were performed to compare species distribution among ponds and to understand the relationships with the chemicals, metals and nutrients in the sediment, pore water and water column that we measured. The relationship between macroinvertebrate abundances and habitat (SAV) structure was also evaluated.

Results

A complete list of invertebrate taxa collected from all seven ponds over each of 4 months is presented in Table 1. This list is the most detailed taxonomic survey of any study yet performed on Great Salt Lake wetlands. Combined with the chemical and aquatic plant measurements, it allowed us to perform a detailed ecological, toxicological and statistical evaluation of the macroinvertebrate assemblages. Data is summarized by date, sample location and abundance (number per m^2) in Table 2. Taxa groupings also allow for easy identification and reporting of biomass and abundances of the different functional feeding groups. Tables of dry weights are listed in Appendix D.

Monthly sampling across all ponds provided for a detailed comparison of taxa richness, and abundance and their response to environmental conditions. Figure 1 illustrates a comparison of taxa richness for each pond across the sampling season. Taxa richness was quite similar across ponds and the sampling season. The only exceptions were the low number in AM during June, FB1 during July and all of the samples collected from N47. Spraying for mosquito control using a broad-spectrum adulticide (usually Trumpet EC®) was the likely cause of the low taxa numbers for FB1 and AMB and because of the rapid return to more average values by the next month. N47 has consistently shown low taxa richness and abundance among all sampling events since sampling began in 2005 (Miller and Hoven 2007, Gray, 2009 and 2010). PN had the highest single monthly value for taxa richness which occurred in July. This increase in taxa was largely due to an increase in the number of coleopteran (beetle) and odonate (damselfly) taxa (Table 3). However, many of these taxa were absent from subsequent samples.

Table 1. List of taxa found among the seven study ponds.

Ephemeroptera	Baetidae
	Caenis sp.
	Callibaetis sp.
	Leptophlebiidae
Odonata	Aeshnidae
	Coenagrionidae
	Libellulidae
	Sympetrum sp.
Hemiptera	*Corisella* sp.
	Corixidae
	Notonecta sp.
	Notonectidae
	Sigara sp.
Coleoptera	Dytiscidae
	Enochrus sp.
	Haliplus sp.
	Hydrophilidae
	Hydroporinae
	Hygrotus sp.
	Laccophilus sp.
	Scirtidae
	Stictotarsus sp.
	Tropisternus sp.
Diptera-Chironomidae	*Ablabesmyia* sp.
	Apedilum sp.
	Chironomus sp.
	Cladotanytarsus sp.
	Corynoneura sp.
	Cricotopus sp.
	Cryptochironomus sp.
	Glyptotendipes sp.
	Microchironomus sp.
	Paratanytarsus sp.
	Procladius sp.
	Psectrocladius sp.
	Tanypodinae
	Tanypus sp.
	Tanytarsini
	Tanytarsus sp.
Diptera	*Bezzia/Palpomyia* sp.
	Ceratopogoninae
	Dasyhelea sp.
	Dolichopodidae
	Ephydridae
	Sphaeromias sp.
	Stratiomyidae
Trichoptera	Leptoceridae
	Oecetis sp.
	Triaenodes sp.
Lepidoptera	Lepidoptera
Gastropoda	Gastropoda
	Gyraulus sp.
	Lymnaeidae
	Physa sp.
	Planorbidae
	Radix auricularia[1]
	Stagnicola sp.
Annelida	*Erpobdella* sp.
	Helobdella sp.
	Helobdella stagnalis
	Oligochaeta
	Theromyzon sp.
Acari	*Arrenurus* sp.
	Limnesia sp.
	Pionidae
Crustacea	*Hyalella* sp.
	Ostracoda
Other Organisms	Nematoda
	Turbellaria

[1]An invasive snail species. Only one individual was collected.

Table 2. Major taxon groups and dates sampled of macroinvertebrates found in the seven study ponds surrounding Great Salt Lake.

	Site Name	Farmington Bay Unit 1	Farmington Bay Unit 1	Farmington Bay Unit 1	Farmington Bay Unit 1	Farmington Bay Unit 1	Farmington Bay Unit 2	Farmington Bay Unit 2	Farmington Bay Unit 2	Farmington Bay Unit 2	Farmington Bay Unit 2
	Jar#/Replicate				#1-3 BENTHOS!	(#4 SHORE!)			(#1-3 Benthos)	#4 Shoreline	
	Collection Date	06-17-2010	07-16-2010	09-02-2010	10-01-2010	10-01-2010	06-22-2010	07-26-2010	09-02-2010	09-02-2010	09-28-2010
	Sampled Area (m²)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	Percent Subsampled	1.19	1.79	5.21	78.74	100.00	1.39	3.57	9.52	35.46	7.14
Ephemeroptera	(MAYFLIES)	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Odonata	(DRAGON AND DAMSEL FLIES	0.00	391.06	422.26	10.16	25.00	215.83	840.34	42.02	84.60	196.08
Hemiptera	(WATERBOATMAN)	84.03	8,994.41	1,094.05	0.00	150.00	1,582.73	196.08	21.01	2.82	224.09
Coleoptera	(BEETLES)	0.00	167.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Diptera-Chironomidae	(MIDGES)	27,647.06	2,122.91	2,706.33	397.51	1.00	30,791.37	1,848.74	2,563.03	59.22	938.38
Diptera	(NON-MIDGE FLIES)	0.00	837.99	0.00	0.00	0.00	0.00	532.21	0.00	28.20	0.00
Trichoptera	(CADDISFLIES)	0.00	0.00	19.19	2.54	0.00	0.00	0.00	10.50	0.00	0.00
Lepidoptera	MOTHS AND BUTTERFLIES)	0.00	0.00	0.00	66.04	8.00	0.00	0.00	0.00	0.00	0.00
Gastropoda	(SNAILS)	1,764.71	1,899.44	959.69	13.97	4.00	935.25	364.15	84.03	160.74	882.35
Annelida	(SEGMENTED WORMS)	84.03	2,234.64	3,474.09	1.27	0.00	575.54	1,680.67	2,363.45	225.61	0.00
Acari	(WATER MITES)	336.13	55.87	499.04	118.11	14.00	503.60	336.13	955.88	0.00	126.05
Crustacea	(MOSTLY HYALELLA OR SCUDS)	4,873.95	12,513.97	1,285.99	1.27	0.00	2,302.16	5,378.15	63.03	930.63	42.02
Other Organisms	(Nematoda, Turbellaria)	0.00	0.00	19.19	0.00	1.00	0.00	0.00	0.00	0.00	0.00
	TOTAL	43,781.51	30,111.73	11,036.47	655.32	204.00	38,489.21	15,210.08	6,775.21	1,548.22	7,759.10

Table 2. Continued.

	Site Name	Pintail	Pintail	Pintail	Pintail	Pintail	Pintail	Widgeon
	Jar#/Replicate	(#1-2 Benthos)	#3 Shoreline					Pond was dry in July
	Collection Date	06-22-2010	06-22-2010	07-29-2010	08-09-2020	08-31-2010	09-29-2010	06-22-2010
	Sampled Area (m²)	1.00	1.00	1.00	1.00	1.00	1.00	1.00
	Percent Subsampled	5.95	16.67	7.94	8.33	8.93	79.37	26.18
Ephemeroptera	(MAYFLIES)	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Odonata	(DRAGON AND DAMSEL FLIES	100.84	12.00	1,020.15	480.19	2,889.14	94.49	38.20
Hemiptera	(WATERBOATMAN)	857.14	821.84	793.45	324.13	358.34	78.12	99.31
Coleoptera	(BEETLES)	0.00	24.00	138.54	180.07	11.20	2.52	0.00
Diptera-Chironomidae	(MIDGES)	6,773.11	779.84	1,536.52	4,141.66	1,254.20	23.94	1,550.80
Diptera	(NON-MIDGE FLIES)	0.00	0.00	163.73	12.00	11.20	1.26	3.82
Trichoptera	(CADDISFLIES)	0.00	0.00	176.32	144.06	0.00	0.00	30.56
Lepidoptera	MOTHS AND BUTTERFLIES)	0.00	24.00	226.70	0.00	55.99	2.52	0.00
Gastropoda	(SNAILS)	991.60	1,421.72	1,586.90	816.33	851.06	456.09	171.89
Annelida	(SEGMENTED WORMS)	184.87	89.98	176.32	84.03	347.14	0.00	7.64
Acari	(WATER MITES)	16.81	6.00	12.59	24.01	11.20	45.36	0.00
Crustacea	(MOSTLY HYALELLA OR SCUDS)	0.00	0.00	50.38	12.00	0.00	1.26	7.64
Other Organisms	(Nematoda, Turbellaria)	50.42	6.00	0.00	0.00	0.00	0.00	0.00
	TOTAL	9,142.86	3,185.36	6,977.33	6,518.61	6,282.19	708.08	2,055.00

Table 2. Continued.

	Site Name	Ambassador W1	Ambassador W1	Ambassador W1	Ambassador W1	Ambassador W1	Ambassador w1
	Jar#/Replicate			**A-C Benthic**	**D-F Canopy**	**G Shoreline**	
	Collection Date	**06-14-2010**	**07-29-2010**	**09-03-2010**	**09-03-2010**	**09-03-2010**	**09-27-2010**
	Sampled Area (m²)	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**
	Percent Subsampled	**75.76**	**0.74**	**2.38**	**2.38**	**18.76**	**6.55**
Ephemeroptera	**(MAYFLIES)**	1.00	1.00	1.00	1.00	1.00	1.00
Odonata	**(DRAGON AND DAMSEL FLIES**	0.00	405.41	756.30	168.07	351.81	442.75
Hemiptera	**(WATERBOATMAN)**	0.00	675.68	0.00	0.00	63.97	152.67
Coleoptera	**(BEETLES)**	1.32	135.14	0.00	42.02	5.33	0.00
Diptera-Chironomidae	**(MIDGES)**	632.26	14,189.19	1,974.79	5,168.07	47.97	137.40
Diptera	**(NON-MIDGE FLIES)**	3.96	1,756.76	84.03	42.02	21.32	15.27
Trichoptera	**(CADDISFLIES)**	0.00	0.00	0.00	0.00	0.00	0.00
Lepidoptera	(M**OTHS AND BUTTERFLIES)**	0.00	0.00	0.00	0.00	0.00	15.27
Gastropoda	**(SNAILS)**	0.00	10,540.54	13,865.55	10,294.12	762.26	6,900.76
Annelida	**(SEGMENTED WORMS)**	36.96	59,864.86	10,462.18	9,075.63	1,167.38	91.60
Acari	**(WATER MITES)**	0.00	135.14	210.08	630.25	5.33	45.80
Crustacea	**(MOSTLY HYALELLA OR SCUDS)**	2.64	1,891.89	336.13	1,344.54	767.59	198.47
Other Organisms	(Nematoda, Turbellaria)	2.64	0.00	126.05	0.00	26.65	0.00
	TOTAL	**679.78**	**90,540.54**	**27,815.13**	**26,764.71**	**3,256.93**	**8,015.27**

Table 2. Continued

	Site Name	**NS #47**	**NS #47**	**NS #47**	**NS #47 (#1-2 Benthos)**	**NS #47 #3 Shoreline**	**BR4C**	**BR4C**
	Jar#/Replicate							
	Collection Date	**06-16-2010**	**07-22-2010**	**09-01-2010**	**10-07-2010**	**10-07-2010**	**06-24-2010**	**07-29-2010**
	Sampled Area (m²)	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**
	Percent Subsampled	**0.78**	**1.49**	**38.17**	**100.00**	**100.00**	**3.87**	**10.72**
Ephemeroptera	**(MAYFLIES)**	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Odonata	**(DRAGON AND DAMSEL FLIES**	0.00	671.14	7.86	1.00	49.00	387.60	1,184.70
Hemiptera	**(WATERBOATMAN)**	0.00	872.48	47.16	36.00	421.00	258.40	270.52
Coleoptera	**(BEETLES)**	0.00	0.00	0.00	0.00	0.00	51.68	9.33
Diptera-Chironomidae	**(MIDGES)**	43,974.36	11,610.74	157.19	14.00	7.00	11,524.55	1,613.81
Diptera	**(NON-MIDGE FLIES)**	0.00	0.00	0.00	0.00	0.00	0.00	9.33
Trichoptera	**(CADDISFLIES)**	0.00	0.00	0.00	0.00	0.00	25.84	37.31
Lepidoptera	M**OTHS AND BUTTERFLIES)**	0.00	0.00	0.00	3.00	0.00	0.00	55.97
Gastropoda	**(SNAILS)**	256.41	805.37	7.86	11.00	17.00	25.84	83.96
Annelida	**(SEGMENTED WORMS)**	32,948.72	25,436.24	1,288.97	4.00	0.00	956.07	998.13
Acari	**(WATER MITES)**	0.00	0.00	0.00	0.00	1.00	0.00	0.00
Crustacea	**(MOSTLY HYALELLA OR SCUDS)**	128.21	2,684.56	0.00	2.00	18.00	51.68	214.55
Other Organisms	(Nematoda, Turbellaria)	0.00	0.00	15.72	0.00	0.00	25.84	0.00
	TOTAL	**77,307.69**	**42,080.54**	**1,524.76**	**71.00**	**513.00**	**13,436.69**	**4,981.34**

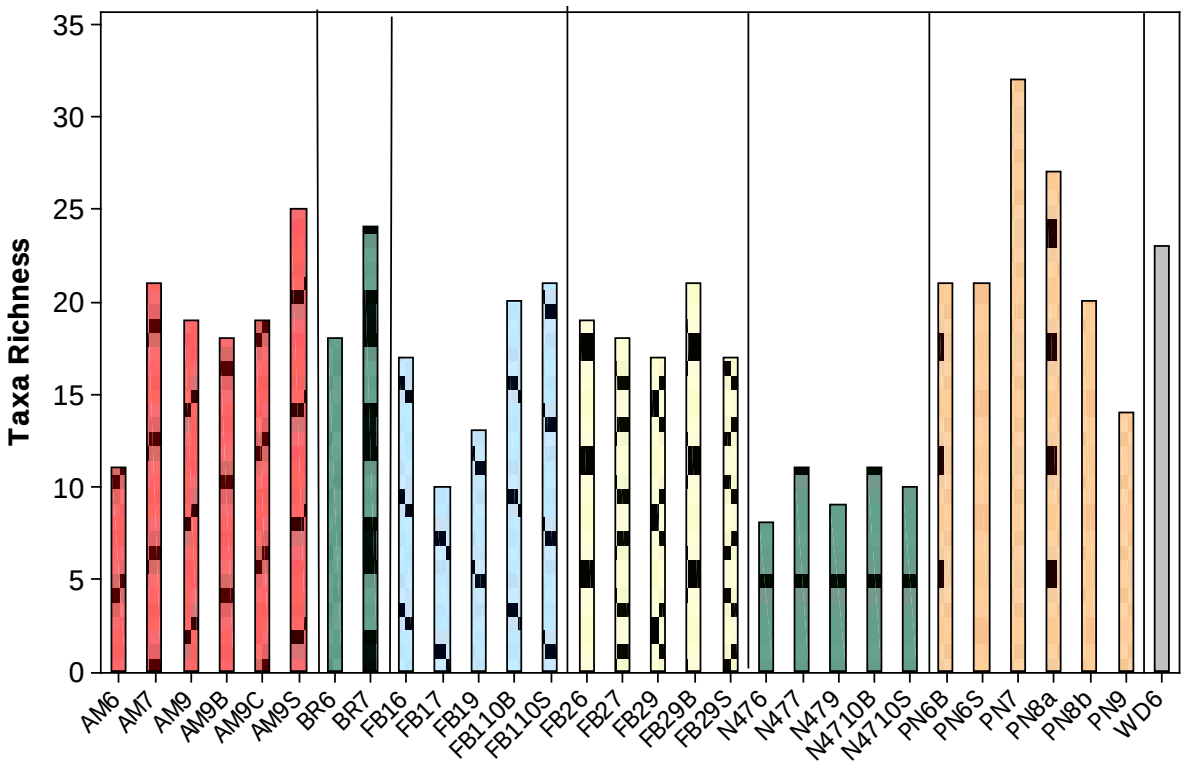


Figure 1. A summary of taxa richness (total number of taxa per m^2) for each study pond across the four months of the growing season. The ponds include Ambassador W1 (AM) Bear River Migratory Bird Refuge pond 4C (BR), Farmington Bay Wildlife Management Area Unit 1 (FB1) and Unit 2 (FB2), Newstate Pond 47 (N47), Public Shooting Grounds Wildlife Management Area Pintail (PN) and Widgeon Pond (WD). The number after the pond name represents the month of sample collection. The additional letters after the month represent a benthic sample (B) which included the approximate top 5 cm of the sediment, a sample which included sweeping the net just above the sediment surface to included only the SAV stand (C), and a sample collected along the shoreline, performed by rapidly bumping the sweep net against the roots and stems of the emergent vegetation. Where "post script" letters were not included, samples were collected as described for a benthic sample. See text above for further sampling details.

We also graphed total abundance of all macroinvertebrates collected during each sampling (Figure 2). Note that all samples represented a square meter of pond sediment surface so these values represent a direct comparison of density among ponds.

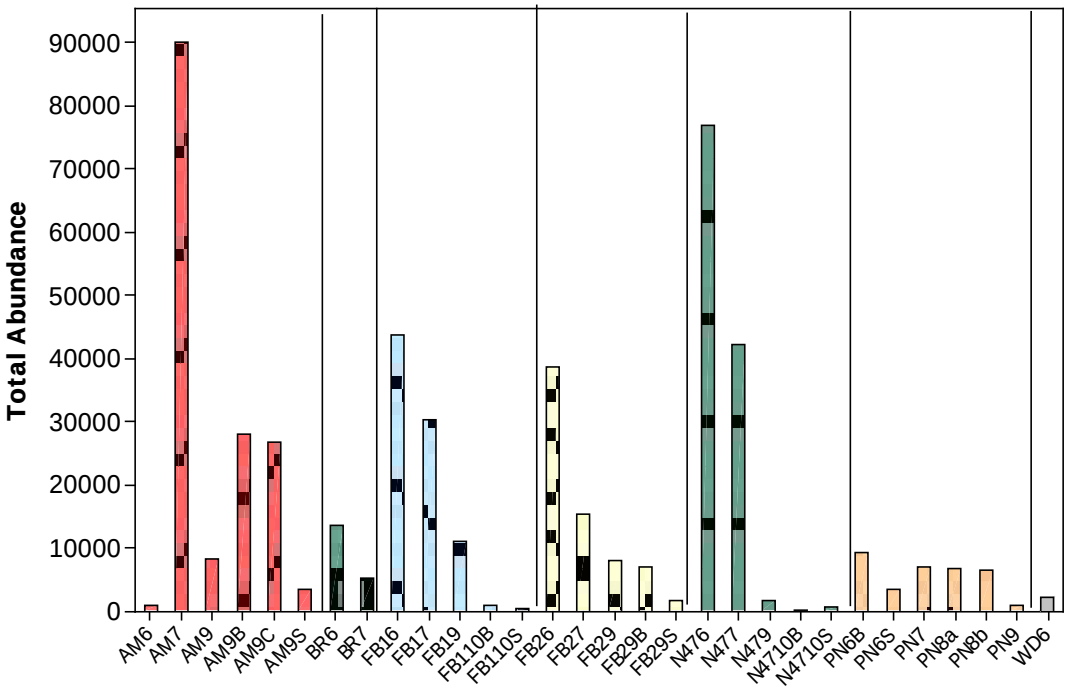


Figure 2. Total abundance (m^{-2}) of taxa collected across the study ponds and across the sampling season. The ponds include Ambassador W1 (AM) Bear River Migratory Bird Refuge pond 4C (BR), Farmington Bay Wildlife Management Area Unit 1 (FB1) and Unit 2 (FB2), Newstate Pond 47 (N47), Public Shooting Grounds Wildlife Management Area Pintail (PN) and Widgeon Pond (WD). The number after the pond name represents the month of sample collection. The additional letters after the month represent a benthic sample (B) which included the approximate top 5 cm of the sediment, a sample which included sweeping the net just above the sediment surface to included only the SAV stand (C), and a sample collected along the shoreline, performed by rapidly bumping the sweep net against the roots and stems of the emergent vegetation. Where "post script" letters were not included, samples were collected as described for a benthic sample. See text above for further sampling details.

Total abundance varies dramatically between ponds and even between seasons. AMB-6 was low in both number of taxa and abundance in June but these values recovered within 30 days with abundance increasing by several thousand fold. The abundance of all taxa in AM increases between June (AMB-6) and July (AMB-7) – reaching the highest density of individuals among all ponds and sample dates. Although numbers of all taxa increased between AMB-6 and AMB-7, the greatest increase occurred for midges and oligochaete worms. Although N47 had the lowest overall number of taxa (Figure 1), it had the second highest abundance of individuals among all of the ponds and sample dates. The relative abundance in N47 and AM in the July samples were quite similar (Table 2, Figure 6). Also notable, while taxa richness does not

decline substantially through the season, there is a large decline in abundance of organisms for all ponds as summer and fall progressed (Figures 1 and 2).

Figure 3 shows the total weight of the samples collected in each pond for each sampling event. Total sample weights were generally comparable to the abundance data in Figure 2. The major differences were the result of a growing snail population that added proportionately more weight (per individual) than their contribution to the total abundance of individuals. Between spring and fall, abundance declined dramatically in all ponds. This may have important implications for the feeding habits of waterfowl between spring nesting and fall staging seasons (Wilson et al. 2011; see below) and it was therefore informative to further break down the data to individual taxa abundance and biomass (Figures 4 and 5). As noted above, higher biomass was observed in the spring samples and diminished as summer progressed. The only exceptions were increases in biomass of gastropods (snails), (i.e. FB2, PN, and AMB), or a population of large leaches that developed in July at BR. Snails are comparatively long lived and typically increase in abundance and biomass throughout the summer.

Notably, except for AMB, large numbers and biomass of midges occurred during spring in all ponds. FB1 and FB2 were quite similar in June and July but then FB1 showed remarkable loss in biomass in September and October. FB2 started with much lower biomass in June, but biomass remained consistent throughout the summer. The only noticeable change was a reduction in midges and a large increase in snails. WD and PN, have served as reference ponds over the past seven years (Miller and Hoven 2007, Hoven and Miller 2009), primarily because they have consistently contained low concentrations of phosphorus, dense macrophyte growth of spiral witchgrass (*Ruppia cirrhosa*) and sago pond weed (*Stuckenia pectinatus*) and some invertebrate taxa that are considered sensitive - at least to organic pollution (e.g. Hillsenhoff 1989). In the present study, macroinvertebrate biomass in PN was intermediate between FB 1 and FB2 and was largely dominated by snails. WD showed similar characteristics to PN (both are located in Public Shooting Grounds WMA) during the June sample. Unfortunately, with the scarcity of water due to irrigation diversions, WD was allowed to go dry before the July sample was collected.

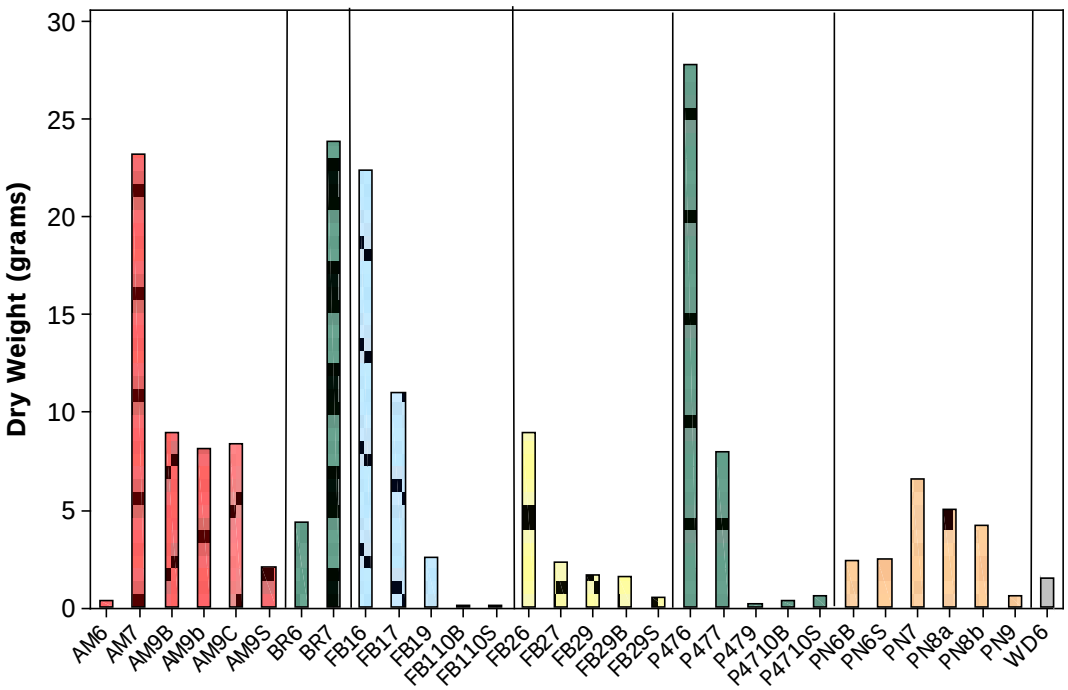


Figure 3. The distribution of total biomass across the sample ponds and sampling events throughout the summer. The ponds include Ambassador W1 (AMB) Bear River Migratory Bird Refuge pond 4C (BR), Farmington Bay Wildlife Management Area Unit 1 (FB1) and Unit 2 (FB2), Newstate Pond 47 (N47), Public Shooting Grounds Wildlife Management Area Pintail (PN) and Widgeon Ponds (WD). The number after the pond name represents the month of sample collection. The additional letters after the month represent a benthic sample (B) which included the approximate top 5 cm of the sediment, a sample which included sweeping the net just above the sediment surface to included only the SAV stand (C), and a sample collected along the shoreline (S), performed by rapidly bumping the sweep net against the roots and stems of the emergent vegetation. Where "post script" letters were not included, samples were collected as described for a benthic sample. See text above for further sampling details.

The pond AMB had the lowest biomass of all ponds during the June sampling, but developed a large and more diverse community and biomass by July (Figure 6). The increase in biomass was due to an increase in the numbers of snails. N47, another duck club impoundment is located very near the terminus of the Jordan River and, along with AMB, had been considered to be one of the most "stressed" ponds among the Farmington Bay impoundments.

Figure 4. Summary of monthly biomass data for major taxonomic groups. Widgeon became dry by mid-July and Bear River Pond 4C became dry by mid August.

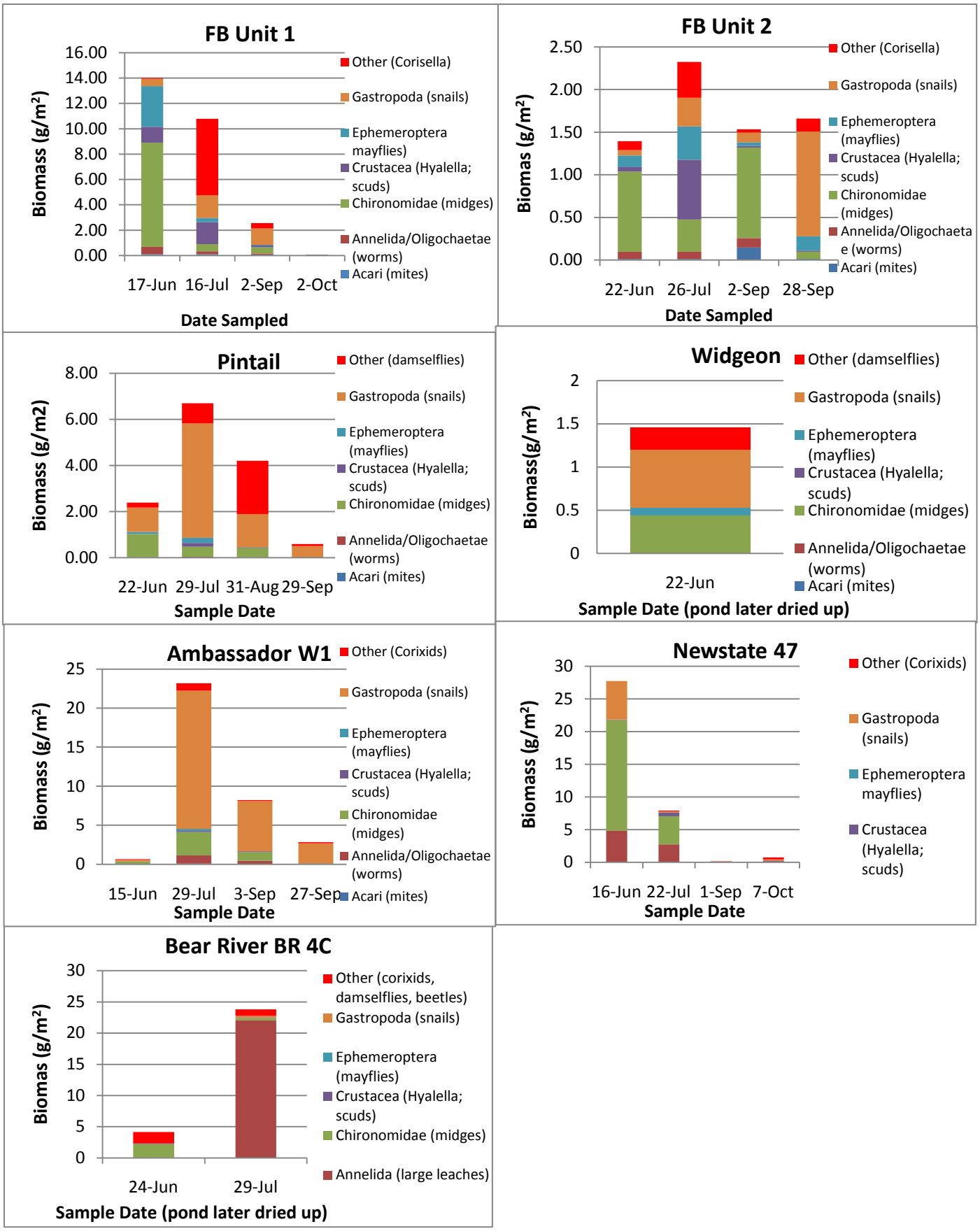
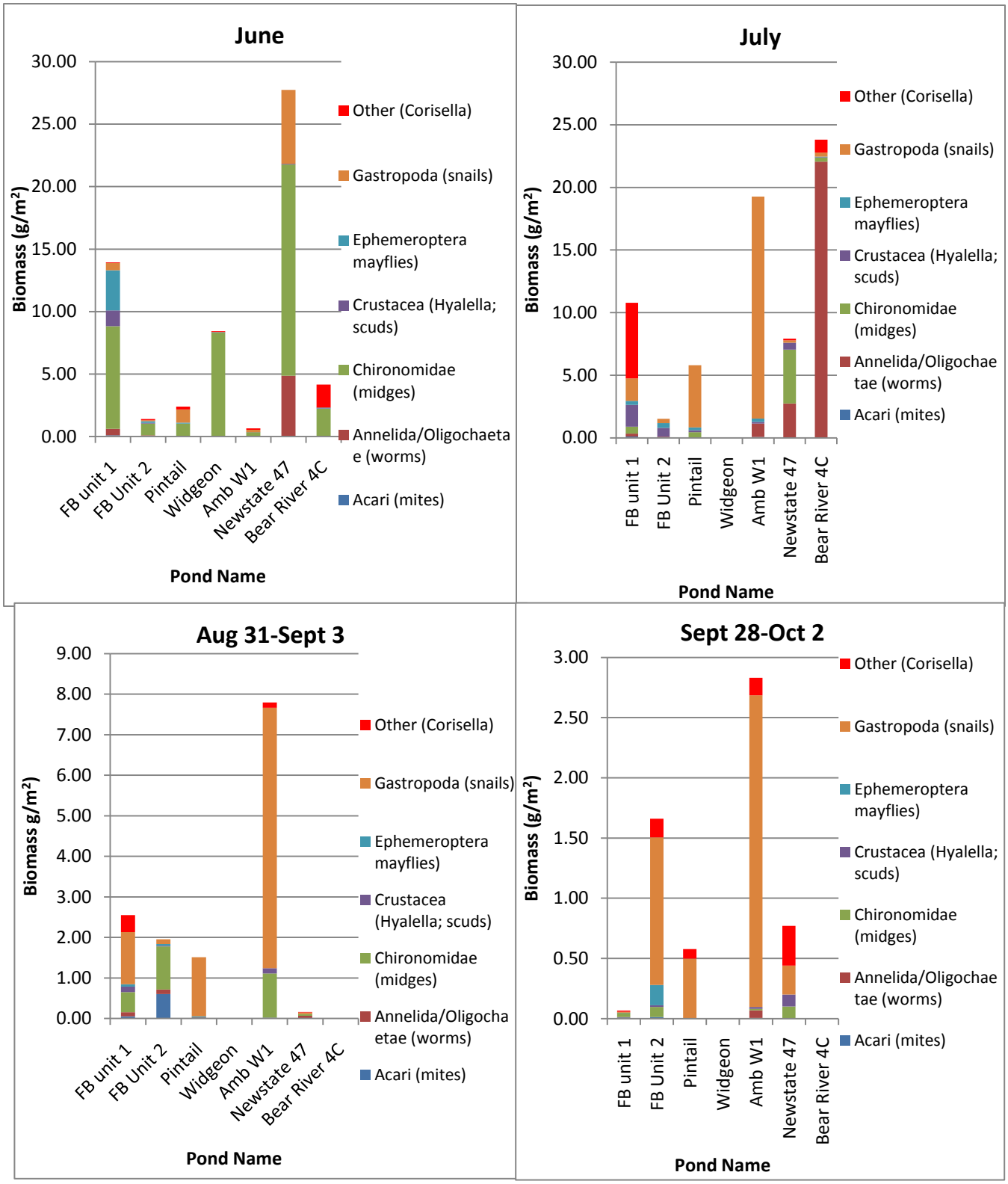


Figure 5. A comparison of the biomass of taxa across all study ponds for each sampling month.



However, as opposed to AMB, N47 had the highest June biomass of all the study ponds. Yet, it rapidly declined in biomass as summer progressed. These two different reactions might be difficult to explain with regard to natural processes, except that the SAV tends to die off as the season progresses and the associated habitat structure is lost. Particularly of note is that the density of SAV branches diminishes earlier in N47 than that at AM (see Hoven et al. 2011). Also, these ponds are frequently treated for mosquito control. The aggressive mosquito abatement program includes both a highly species-specific bacterial enzyme, noted as BTI, from *Bacillus thuringiensis israelenis* that acts as an intestinal poison to mosquitoes, as well as several broad spectrum pesticides (such as Trumpet EC®). Basically, where numerous mosquito larvae are found, BTI is used because it is very specific to mosquito larvae and secondarily to midge larvae. Where numerous adults are observed, a more broad-spectrum adulticide (usually Trumpet EC®), that is toxic to most other wetland macroinvertebrate species. For example, it is possible that AMB was treated with one or both types of pesticides, while it is also possible that N47 was treated several times later in the summer (several incidences of spraying were observed by club managers). The local mosquito abatement districts will be contacted to acquire spray records. Alternatively, however, an intense investigation of water column, pore water and sediment chemistry and metals was performed as part of this study in order to determine the potential for internal sediment processes, or legacy issues of sediment metal contamination to also be responsible for differences observed among ponds or throughout the growing season (Johnson et al . 2011; see below). BR, located in the Bear River Migratory Bird Refuge, had been considered to be intermediate in trophic status in that it had lower nutrient concentrations than Farmington Bay ponds, but considerably higher concentrations than PN or WD. Because of these characteristics, we had hypothesized that BR would provide for abundant plant biomass and abundant and diverse macroinvertebrate biota. The BR June sample was relatively very low in biomass and although the July sample increased substantially, it was dominated by a large species of leach. As with WD, BR dried up shortly after the July sample. The remaining nominal reference site, PN, experienced the typical decline in both macroinvertebrate biomass and abundance by the September sampling period.

We also compared taxa biomass among sample ponds according to each sample month (Figure 5). This more clearly displayed the temporal changes between the different ponds. Most apparent is the dominance by midges during spring. Although species lists shift during the remainder of summer, snails or leaches dominate the biomass during the remainder of the growing season. Also notable, from month to month, there is a consistent decline in macroinvertebrate biomass within all of the ponds. In each of the figures the term “other” was used to denote the weight of usually more rare taxa. In each case, this group largely consisted of the hemipterans, Corisella and other Corixidae, generally common predators in wetland environments.

We repeated this effort by plotting the relative abundance and density of dominant taxa both by individual ponds and according to each sampling period (Figures 6 and 7).

Figure 6. Abundance of individuals in major taxa compared across study ponds for each sampling period. Ambassador W1 supported most individuals although species dominance and diversity varied between months.

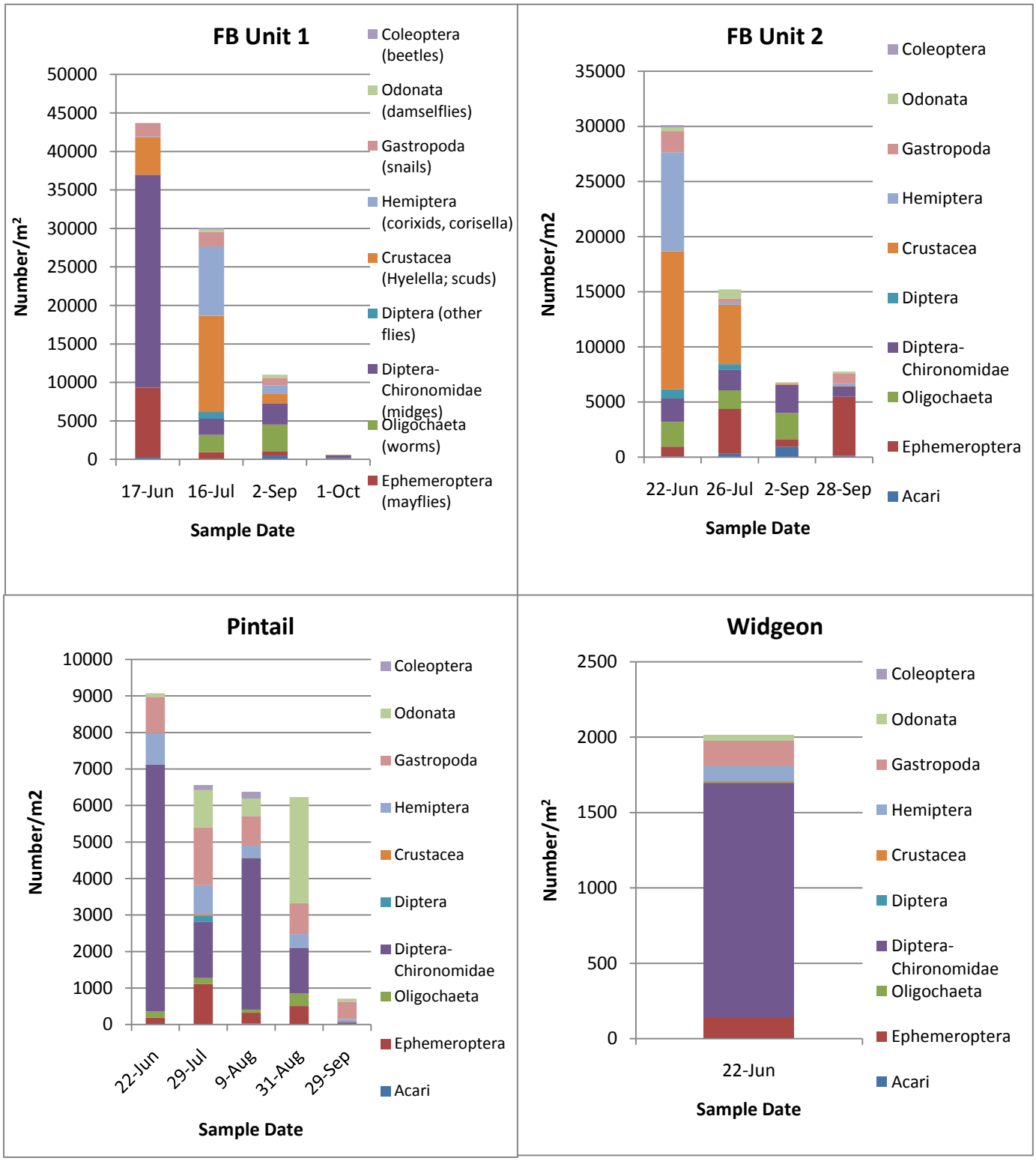


Figure 6. Continued.

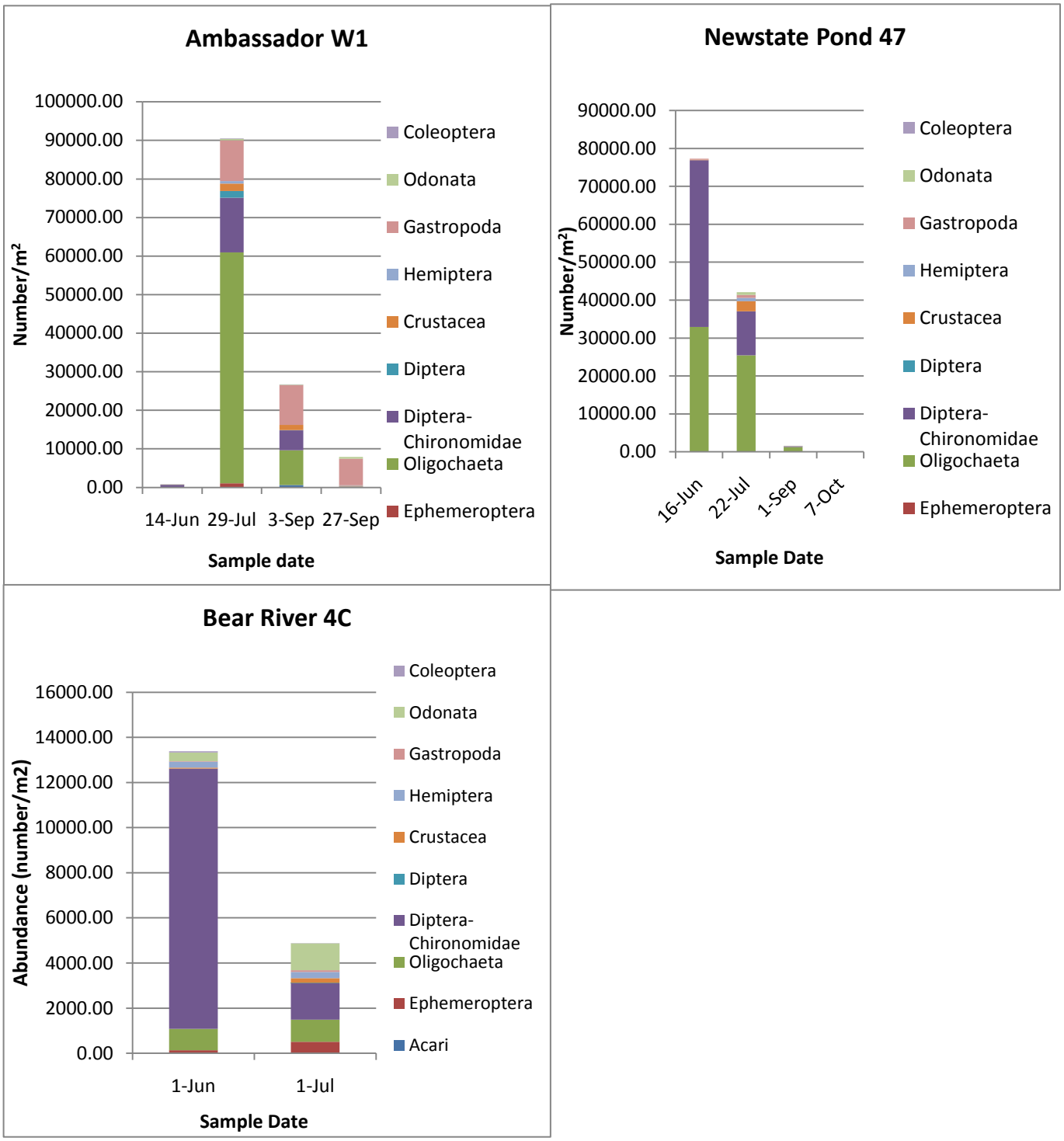
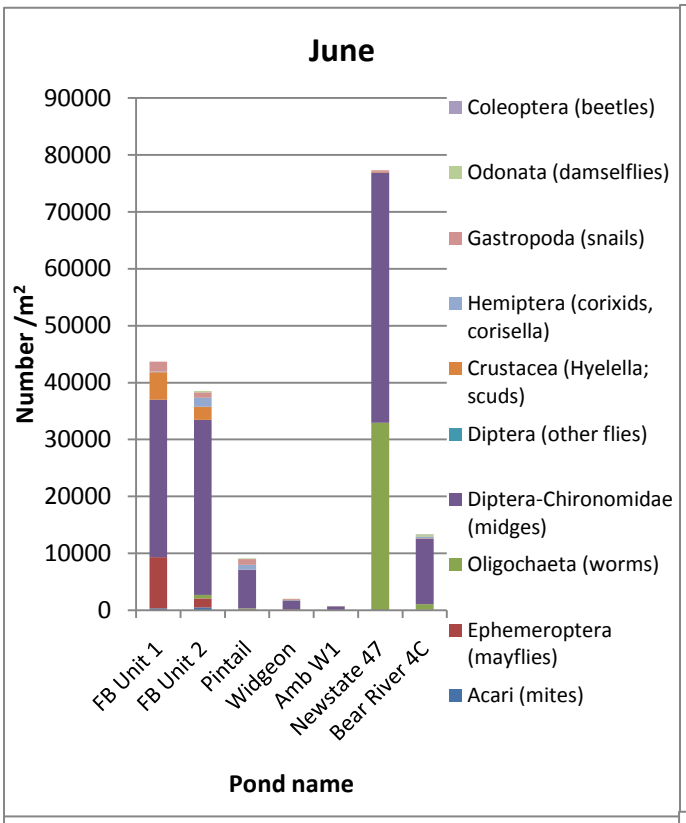
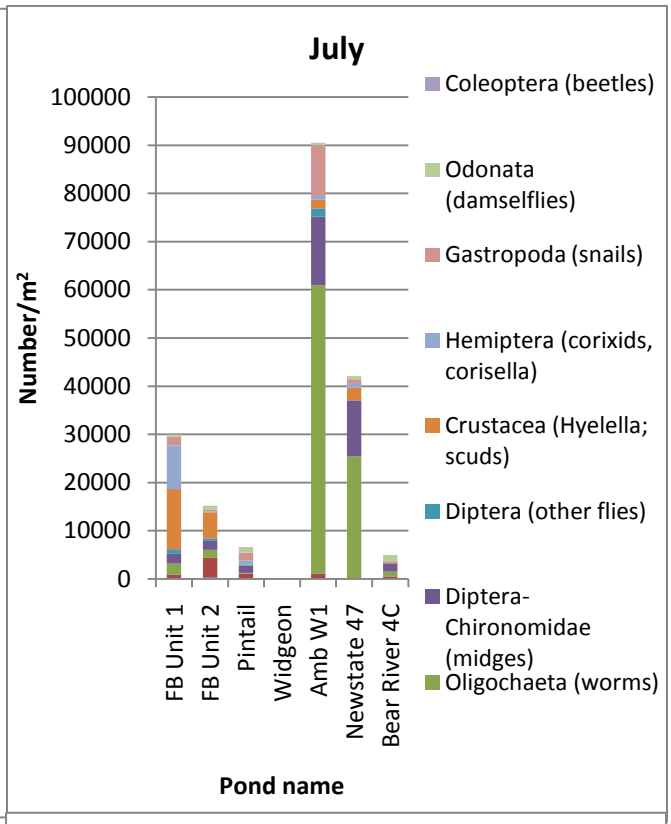
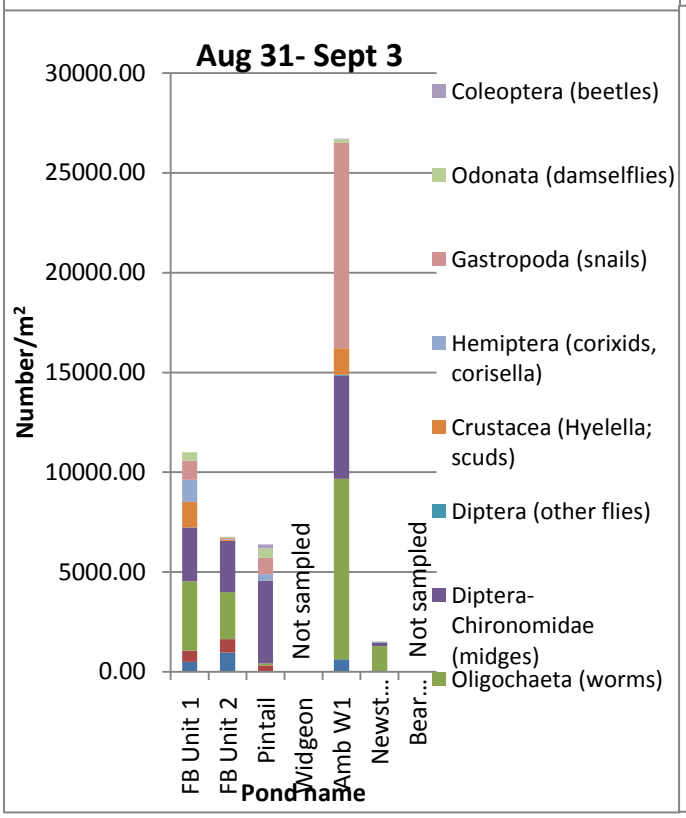
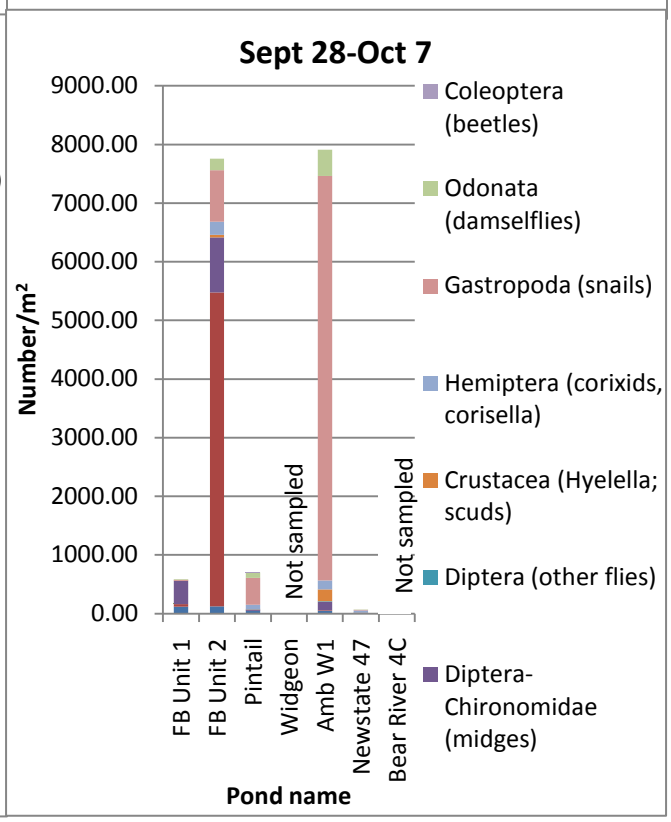


Figure 7. Abundance of individuals in major taxa compared among impoundments for each sampling period. Ambassador W1 supported most individuals although species dominance and diversity varied between months.


June
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
Number /m²
FB Unit 1
FB Unit 2
Pintail
Widgeon
Amb W1
Newstate 47
Bear River 4C
Pond name
Coleoptera (beetles)
Odonata (damselflies)
Gastropoda (snails)
Hemiptera (corixids, corisella)
Crustacea (Hyelella; scuds)
Diptera (other flies)
Diptera-Chironomidae (midges)
Oligochaeta (worms)
Ephemeroptera (mayflies)
Acari (mites)


July
100000
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
Number/m²
FB Unit 1
FB Unit 2
Pintail
Widgeon
Amb W1
Newstate 47
Bear River 4C
Pond name
Coleoptera (beetles)
Odonata (damselflies)
Gastropoda (snails)
Hemiptera (corixids, corisella)
Crustacea (Hyelella; scuds)
Diptera (other flies)
Diptera-Chironomidae (midges)
Oligochaeta (worms)


Aug 31- Sept 3
30000.00
25000.00
20000.00
15000.00
10000.00
5000.00
0.00
Number/m²
Not sampled
Not sampled
FB Unit 1
FB Unit 2
Pintail
Widgeon
Amb W1
Newst...
Bear...
Pond name
Coleoptera (beetles)
Odonata (damselflies)
Gastropoda (snails)
Hemiptera (corixids, corisella)
Crustacea (Hyelella; scuds)
Diptera (other flies)
Diptera-Chironomidae (midges)
Oligochaeta (worms)


Sept 28-Oct 7
9000.00
8000.00
7000.00
6000.00
5000.00
4000.00
3000.00
2000.00
1000.00
0.00
Number/m²
Not sampled
Not sampled
FB Unit 1
FB Unit 2
Pintail
Widgeon
Amb W1
Newstate 47
Bear River 4C
Pond name
Coleoptera (beetles)
Odonata (damselflies)
Gastropoda (snails)
Hemiptera (corixids, corisella)
Crustacea (Hyelella; scuds)
Diptera (other flies)
Diptera-Chironomidae (midges)

This facilitates the observation and understanding of both the temporal and spatial differences within and between ponds of the actual numbers of individuals. The seasonal changes in numbers are also very striking. Whereas biomass can be altered (in this case, sustained) because of the increasing numbers and growth of snails, the total counts of individuals within each taxon clearly declines as the summer and fall progress.

The significance of these seasonal changes can also be discussed in relationship to the seasonal changes in waterfowl diet (i.e. see Wilson et al. 2011). During the nesting season, when macroinvertebrates are abundant and particularly midges, waterfowl prefer an animal diet, and particularly midges. The known omnivory of waterfowl supports the paradigm - that they are not particularly selective in their diet. However, in the fall, waterfowl have switched to a mostly plant diet (drupelets of SAV or bulrush seeds). It has long been thought that this switch to plant items in the fall is the desire for the high carbohydrate/energy component of these seeds (Chamberlain 1959; Anderson and Low 1976; Winslow 2003). However, it is also known that these birds must complete their annual molt and rebuild muscle/protein in preparation for the fall migration. No other known studies have concentrated on the comparison of the waterfowl diet vs food availability. In the present study, the sheer decline in numbers and biomass macroinvertebrates would certainly reduce the availability of these organisms as prey and consequently force a herbivorous diet. Fortunately, the late summer and fall crop of drupelets and other seeds are abundantly available. However, in one pond, FB1, midges comprised a large portion (21%; second only to Asmatales seeds) of the waterfowl diet (Wilson et al. 2011). This raises the question and challenges the paradigm that waterfowl prefer a diet of plant seeds. Although the midge population had declined throughout the summer, FB1 had an unusually high number and biomass of midges, as compared to the other ponds. It may simply be that this abundance was high enough to make it energetically feasible for birds to seek this higher protein diet, as well as the traditionally expected seed diet. Plans are being developed to continue this important research to compare the availability of different food items as sources of energy (i.e. plant seeds vs invertebrates) with the different compartments or types of nutrition that these items provide and finally with the availability of these food items in their environment.

For the most part, the relative abundance correlated with the relative biomass of taxa. One notable exception was in FB2 where there were greater numbers of organisms during June vs July. However, there was a greater biomass during July than June. The species lists and relative abundances were mostly similar and actual number of individuals decreased. Therefore, the greater biomass during July was likely due to growth of the individuals and particularly for snails. Increase in biomass in other ponds was also likely due to increases in snail growth as well as overall growth by other organisms (i.e. the growth of leaches in BR4C).

Non-metric Multidimensional Scaling Ordination

The next goal of our evaluation was to determine if such differences among ponds or seasonal changes were the result of chemical differences among ponds or chemical changes as the summer season progressed. We briefly mentioned above that retrieving the spray records from the mosquito abatement districts is essential in understanding potential ecological changes/degradation among these ponds. In addition, however, we suspected basic chemical characteristics from sediment biogeochemical processes associated with natural organic processes of primary production and decomposition as well as metal deposition and sediment recycling from the known legacy mining and refining activities in the Jordan River watershed.

As described above abundances of several dozen macroinvertebrate taxa were determined for each pond. This allowed us to explore their multivariate relationships. For initial exploratory, visual analyses, ordination techniques are often superior for explaining relationships of assemblages and communities than hypothesis testing approaches (McCune and Grace 2002). In general, ordination is the ordering of objects along axes according to their similarities. The main objective of ordination is data reduction and expressing many-dimensional relationships into a small number of easily interpretable dimensions (axes). This amounts to extracting the strongest correlation structure in the data (using correlation in the broad sense). The correlation structure is then used to position objects in ordination space. Objects close in the ordination space are generally more similar than objects distant in the ordination space (McCune and Mefford 2011).

Several types of ordination exist; non-metric multidimensional scaling (NMS) was used for this data. NMS has been shown to be robust for ordination of species composition (e.g., Kenkel and Orloci 1986, Ludwig and Reynolds 1988) and is often more useful than other ordination techniques because, among other things, it avoids the assumption of linear relationships among variables.

NMS Methods

We explored several dozen NMS scenarios using different distance measures and numbers of axes and then compared these with randomized data Monte Carlo simulations. We then conducted a post hoc analysis of coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space. This provided estimates

of the amount of variability in the data explained by each of the ordination axes. We used the computer program PC-Ord (McCune and Mefford 2011) for NMDS ordination.

Macroinvertebrate NMS

We averaged macroinvertebrate abundances from the samples in each site collected at different times and habitats. This resulted in 7 locations: AMB, FB1, FB2, PIN, BR, WDG, and P47 with an average abundance for each taxon. Averaging samples at each pond allowed us to include taxa that occurred seasonally and gave us an idea of all of the taxa that occurred at a site (given our sampling methods). Taxa aren't always collected at a site due to various biological, ecological, and sampling reasons. Averaging also reduced 'noise' resulting from many samples with large variability. Our original, un-averaged NMS model had 3 dimensions and a higher stress level and was therefore not as useful.

We square root transformed abundances to reduce the wide range of values (a standard procedure for count data) and removed taxa that only occurred at one site. Ordination methods are highly influenced by this phenomenon (rare and uncommon taxa). For example, *Radix auricularia,* an invasive snail, only occurred at one pond (N47) and there was only one individual. When it was left in the ordination it had a strong influence on the results.

Our best NMS model had a 2-dimensional solution using a Sorenson's distance measure. This model resulted in a final stress of < 0.1 and final instability of 0.00 at 43 iterations. McCune and Grace (2002) suggested that most ecological assemblage data sets will have NMS solutions with stress between 10 and 20 and that values in the lower half of this range are quite satisfactory. Final stress $\leq$ 5 is considered "an excellent representation with no prospect of misinterpretation" [Clarke's (1993) "rules of thumb" for NMS in McCune and Grace (2002)]. Our post hoc analysis of coefficients of determination gave an R^2 of 0.47 for Axis 1 and 0.37 for Axis 2. Both axes cumulatively explained 0.84 of the variability in our macroinvertebrate assemblage data. Figure 8 shows the relationship of the macroinvertebrate assemblages in the seven ponds in 2010.

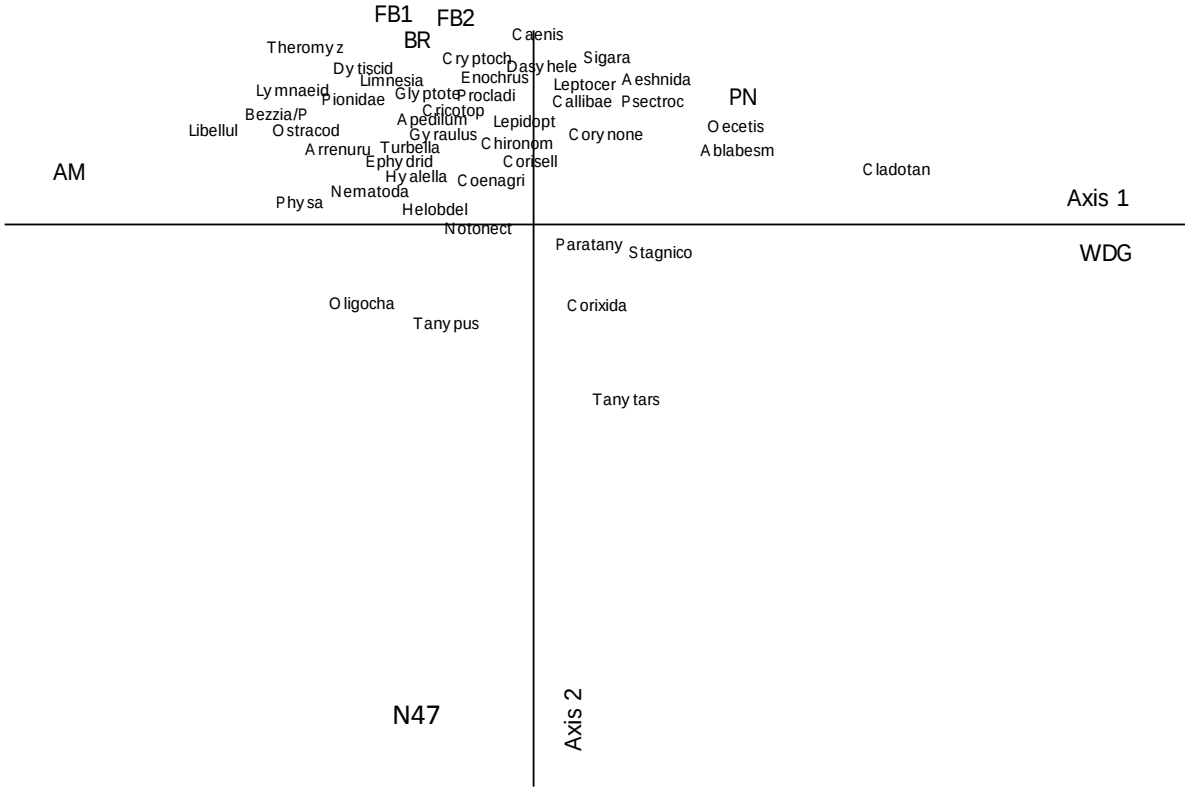


Figure 8. NMS ordination Axes 1 and 2 of macroinvertebrate assemblages by site. Explanation of sample IDs: AM=Ambassador W1, N47=Newstate 47, FB1 & FB2=Farmington Bay Unit 1 & 2, BR=Bear River Unit 4C, WD=Widgeon, PN=Pintail. 1=June, 2=July, 3=August, 4=September. A=AM (morning) sample, P=PM (afternoon) sample.

Most taxa were shared among most of the ponds. For example, high numbers and biomass of midges and most other dipterans were a common characteristic (Figures 2-7). Several other important taxa, characterized as members of various functional feeding groups, were also common among several ponds. Examples include: collector/gatherers (mayflies, midges, most other dipterans and Hyalella), shredders (Triaenodes [Trichoptera], the dipteran *Cricotopus* sp.[Chironomidae], scrapers (snails) and predators (dragonflies and damselflies, some dipterans, most hemipterans and most coleopterans). Collector/gatherers are represented by

several taxa and are considered a rather broad niche. Collector/gatherers are a well-represented niche among all Great Salt Lake impoundments. This characteristic adds resiliency to ecosystem function against various environmental stressors.

The macroinvertebrate ordination results are very similar to the ordination results for the surface water chemistry by Johnson et al. (2011). For illustration, we have included this graph below (Figure 9). Logically, ponds in similar geographic locations, or with similar source water, group together. The largest grouping (highest variance among ponds) occurred among the ponds in Bear River Bay. Particularly, the BRBR receives the great majority of its water from the Bear River, which includes some irrigation return flow. Conversely, PSG ponds (Pintail and Widgeon) receive most of their water from saline springs with the remainder being primarily from irrigation return flow. Chemical and metal concentrations among ponds were the result of these source waters (Johnson et al. 2011). Equally notable, N47 and AMB showed similarity between their taxa lists. These two duck club ponds also share similar source water. However, as mentioned previously, N47 is located near the terminus of the Jordan River while Ambassador Duck Club receives its water from the Surplus Canal, a major diversion of the Jordan River located approximately 20 miles (28 km) upstream from the New state Duck Club diversion. This separation may have been at a great-enough distance upstream to allow differences in watershed characteristics p, but more likely is the result of legacy remnants of the industrial/smelting/refining activities and discharges that occurred differently downstream from the diversions. Notably, Jaccards Similarity Index showed that these sites were dissimilar (Table 3) and the MDS ordination has N47 far away from the other sites, so it is chemically dissimilar.

The source water for the Farmington Bay complex, including FB1 and FB2 is water that is diverted to the State Canal approximately 5 miles (8 km) upstream from the New State Duck Club diversion. Even these short distances between diversion points has resulted in large differences in background water and sediment chemistry as well as biological measures (Figure 9).

Similarity in ordinations of invertebrate assemblages and water chemistry suggests that macroinvertebrate assemblages are strongly influenced by surface water chemistry. For example, invertebrate assemblages in FB1 and FB2 overlapped substantially and are similar to that for BR. PN and WD oriented similarly along axis 2 (Figure 8). Invertebrate assemblages in AMB and N47 contained different species composition (see Figures 4-7) but because the chemical composition of these two ponds showed high similarity (Figure 8), the difference in invertebrate assemblages was likely due to plant habitat characteristics (see below). As mentioned above, the very low numbers for biomass and abundance in AMB during June could have been the result of pesticide spraying. Yet, the rapid recovery of species composition and

Table 3. Most similar and least similar sites based on Jaccards Similarity Index (values can range from 0.00 least similar to 1.00 completely similar)(using Chao estimators and Estimate S computer program)(abundances square root transformed).

Most Similar			Least Similar		
Site 1	Site 2	Jaccard	Site 1	Site 2	Jaccard
AM9	AM9B	0.86	N4710B	BR6	0.25
FB26	FB27	0.85	N476	N4710B	0.25
FB110B	FB29	0.84	AM9	N4710B	0.25
FB26	FB29	0.84	FB1	N477	0.26
FB19	FB26	0.83	FB5	FB10	0.26
FB27	AM7	0.83	FB19	N4710B	0.27
FB29	AM9B	0.82	WD6	N4710B	0.27
AM7	AM9	0.82	PN7	N479	0.27
FB26	AM7	0.81	AM6	N4710B	0.27
FB19	FB29	0.80	AM9B	N4710S	0.27
FB19	FB27	0.80	PN7	N4710B	0.28
AM7	AM9C	0.80	PN8b	AM6	0.28
PN9	AM9S	0.79	AM9S	N477	0.28
FB19	AM7	0.79	FB110B	PN6S	0.28
AM6	N476	0.78	FB5	PN8a	0.28
FB17	FB19	0.76	PN8b	N4710S	0.28
AM9	AM9C	0.76	FB10	N4710B	0.29
N476	N477	0.76	AM9	N4710S	0.29
AM7	AM9B	0.75	FB1	N476	0.29
FB19	BR7	0.75	FB27	N4710B	0.29
FB9	AM7	0.75	FB17	N4710B	0.29

Table 3. Continued					
Most Similar			Least Similar		
Site 1	Site 2	Jaccard	Site 1	Site 2	Jaccard
FB27	FB29	0.75	FB26	N4710B	0.3
FB110B	FB10	0.75	FB5	AM9S	0.3
FB17	AM7	0.74	FB5	BR7	0.3
AM7	BR7	0.74	FB26	N4710S	0.3
FB29	AM7	0.73	N4710B	BR7	0.3
FB1	AM9S	0.72	PN8b	N476	0.3
FB1	FB26	0.72	AM7	N4710S	0.31
FB19	AM9	0.72	FB27	N479	0.31
FB9	AM9C	0.72	FB9	N4710S	0.31
PN8a	BR7	0.72	FB5	PN8b	0.31
BR6	BR7	0.72	FB9	PN6S	0.31
FB26	PN8a	0.72	PN8a	N4710B	0.31
FB26	AM9B	0.71	FB17	FB5	0.31
FB17	FB27	0.71	FB19	FB5	0.31
AM6	N479	0.71	FB1	PN6S	0.31
FB19	FB110B	0.71	FB5	FB27	0.31
AM9B	AM9C	0.71	FB29	N4710B	0.32
FB26	AM9	0.70	FB110B	WD6	0.32
FB19	AM9B	0.70	FB9	WD6	0.32
FB27	FB9	0.70	FB29	PN6S	0.33
PN8b	PN9	0.70	PN9	AM7	0.33
FB29	AM9	0.70	PN7	N477	0.33
FB26	AM9C	0.69	AM9B	N479	0.34

Table 3. Continued					
Most Similar			Least Similar		
Site 1	Site 2	Jaccard	Site 1	Site 2	Jaccard
FB26	BR7	0.69	PN6S	PN9	0.34
FB9	AM9	0.69	N4710S	BR7	0.34
PN8a	AM7	0.69	PN8a	N4710S	0.35
PN8a	PN8b	0.69	WD6	AM7	0.35
FB26	FB10	0.69	FB5	PN6S	0.35
FB27	AM9C	0.68	FB27	PN6S	0.35
FB29	FB10	0.68	WD6	AM9	0.35

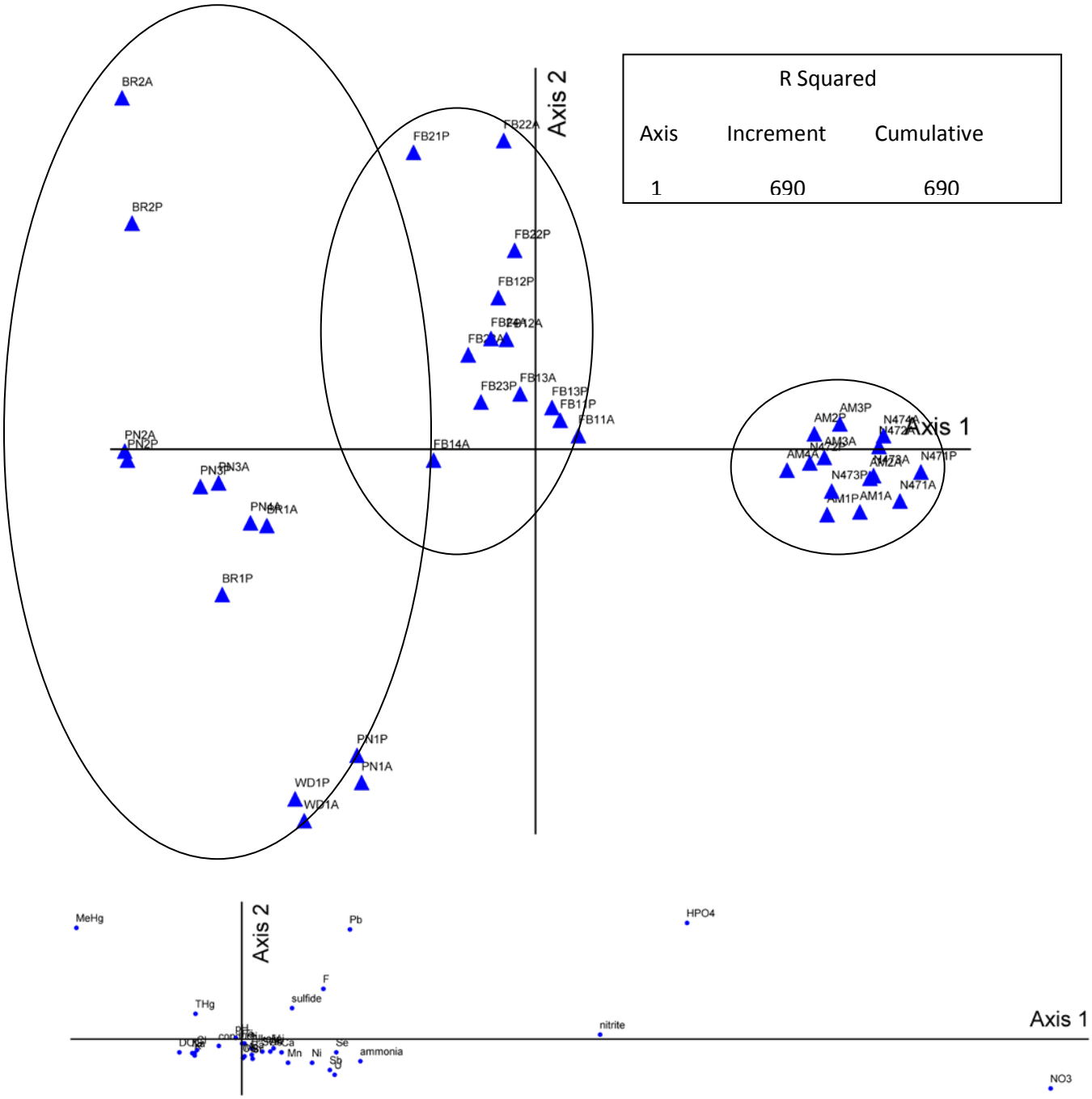


Figure 9. Ordination results for surface water chemistry. Axis 1 explained 69% of variance and Axis 2 explained 18.2%. Axis 3 is shown in Appendix 1. Explanation of sample IDs: AM=Ambassador W1, N47=Newstate 47, FB1 & FB2=Farmington Bay Unit 1 & 2, BR=Bear River Unit 4C, WD=Widgeon, PN=Pintail. 1=June, 2=July, 3=August, 4=September. A=AM (morning) sample, P=PM (afternoon) sample.

biomass in this pond by the July sampling suggests that the stressor was removed. Similarly, the decline in numbers and biomass in FB1 starting in the September sample and reduction to very low numbers and biomass by the Oct 2 sample, could be primarily the result of expected annual emergence to adulthood. However, an unidentified stressor, perhaps late-season mosquito spraying may also be influencing FB1. Additionally, however, Hoven (2011) found that SAV in FB1 remained relatively low throughout the growing season and even declined further between the August and September samples. Contrarily, in the adjacent pond, FB2, biomass and number of macroinvertebrate individuals actually increased during the late season. Notably, the taxa group with the largest increase in numbers was the mayflies, while the taxa responsible for the increase in biomass were snails, both of which are phytophilous taxa (see Figures 4-7). Accordingly, the low density of SAV throughout the summer – followed by a further reduction in September, to being nearly absent, undoubtedly represented a serious absence of macroinvertebrate habitat. This is noteworthy when comparing the species composition and biomass between FB1 and FB2.

The invertebrate assemblage in the Public Shooting Grounds ponds PN and WD oriented fairly closely together on the same axis (Figure 8) and the surface water chemistry was very similar between these two ponds, particularly in June (Figure 9; Figures 4-7).

Correlation Analysis

In pursuing these trends in ordination, we performed correlation analysis between specific chemicals, nutrients and metals found in the surface water, pore water and whole sediments and individual invertebrate taxa. Where correlations were significantly negative ($p<0.1$), more detailed analyses were performed, including literature review and comparisons with state water quality criteria to evaluate the potential for toxicity.

There are two strong sets of correlations for the nutrients and other inorganic constituents. First, there is a strong negative correlation to phosphate (PO_4) P across several (15) taxa (Table 3). Second, concurrent with the strong negative relationship, there is an exact opposite strong positive correlation to Na, K, and Cl, and the metal Li, to nearly the exact same taxa of invertebrates. Our initial interpretation is that the raw PO_4 data, both laboratory [ion chromatography (detection limit – 0.02 mg/L)] and the field Chemetrics analysis (detection limit = 0.1 mg/L) were nearly always below detection limits at PN. But equally consistent, PN had high salt concentrations as compared to the rest of the study ponds. It might seem logical that these numerous taxa might have this “aversion” to phosphate. However, there is no logic why these same taxa would have affinity to salt.

Further, PN was the pond that contained the most taxa that negatively correlated with PO_4, including odonates (dragonflies and damselflies) and coleopterans (beetles). The greatest number of the beetle taxa (5) occurred during the July sampling in PN and dropped off again to

one beetle taxon by August. The other taxa that correlated with low PO_4 and high salts, the mayflies (*Callibaetis* sp. and *Caenis* sp.), and coleopterans occurred in similar or greater number in FB1 (mean PO_4 value of 0.33 mg/L; diel maximum > 3 mg/L) than in PN (Table 3, Appendix A, Appendix E). In order to sort this out further, we performed similar correlation analyses just on the field (Chemetrics) data which included only the nutrient fractions and H_2S (Table 4). In this analysis no taxa correlated negatively to PO_4. Rather, there were several significant negative correlations to H_2S.

We reviewed the diel DO data (appendix E) and found that while DO curves varied between PN and most Farmington Bay Ponds, and BR during the June sampling (i.e. PN DO minimum remained higher than most other ponds (Figure E-7), PN experienced similar DO minima as the Farmington Bay ponds and BR during the July sampling (Figure E-8, E-7, E-6 and E-5; minimum PN DO reached 1.5 on August 3, 2010). Yet, most of the diverse beetle taxa appeared in PN during July – when DO minima was actually a little lower than FB1 and FB2 (Figure E-6 and E-5) and similar to N47 on July 23 (Figure E4). Pond N47 has long been considered the most stressed of all study ponds (i.e. lowest SAV coverage and lowest macroinvertebrate diversity). This evaluation suggests that DO is not associated with the sensitivity of these macroinvertebrate taxa. Similarly, pH is generally higher in PSG ponds than FB ponds – due to high alkalinity and very high photosynthetic rates (DO has consistently approached or exceeded 20 mg/L in early August). Further, during all of these temporal changes in taxa richness, PO_4 always remained below detection limits in Pintail while it continued to spike to near 3 mg/L in most FB ponds (Appendix E).

With the apparent weak relationship between various measures of macroinvertebrates and nutrients or dissolved oxygen, it is not likely that these parameters are the drivers for macroinvertebrate community characteristics. Instead, we looked for other parameters, such as metals, sulfides or DO or habitat differences that co-varied with the nutrients or the salts as an explanation.

Table 3. Water column nutrients and inorganic constituent correlations to macroinvertebrate taxa and p-values. Samples were averaged for sites. The original taxa list from Table 1 was used in the analysis. Taxa that are not listed did not show a significant correlation to the chemicals analyzed. Correlations and p-values were only listed if p<0.1.

	Callibaetis	Leptophl	Aeshnida	Coenagr	Sympetru	Sigara	Dytiscid	Enochrus	Haliplus	Hydrophi	Hydropor	Hygrotus	Scirtida	Stictota	Tropiste	Chironom
Sulfide							-0.89									
							0.018									
Nitrite	-0.75					-0.82										
	0.086					0.046										
phosphat	-0.945	-0.915	-0.782		-0.915				-0.915	-0.915	-0.915	-0.915	-0.915	-0.915	-0.915	
	0.004	0.011	0.066		0.011				0.011	0.011	0.011	0.011	0.011	0.011	0.011	
ammonia							0.883									
							0.02									
Alkalinity							-0.964									
							0.002									
F							-0.848	-0.815								
							0.033	0.048								
Cl	0.992	0.874	0.829		0.874	0.764			0.874	0.874	0.874	0.874	0.874	0.874	0.874	
	0	0.023	0.041		0.023	0.077			0.023	0.023	0.023	0.023	0.023	0.023	0.023	
NO3						-0.947										-0.782
						0.004										0.066
HPO4	-0.796					-0.925		-0.908								
	0.058					0.008		0.012								
SO4							-0.849									
							0.032									
Na	0.989	0.905	0.845	0.731	0.905				0.905	0.905	0.905	0.905	0.905	0.905	0.905	
	0	0.013	0.034	0.099	0.013				0.013	0.013	0.013	0.013	0.013	0.013	0.013	
Mg			0.778				-0.76									
			0.069				0.079									
K	0.985	0.879	0.772	0.802	0.879				0.879	0.879	0.879	0.879	0.879	0.879	0.879	
	0	0.021	0.072	0.055	0.021				0.021	0.021	0.021	0.021	0.021	0.021	0.021	
Ca						-0.862	-0.744	-0.752								

Table 3. Continued.

	Corynone	Cryptoch	Paratany	Procladi	Tanypodi	Dasyhele	Sphaerom	Stratiom	Leptocer	Oecetis	Lepidopt	Planorbi	Stagnico	Oligocha	Limnesia	Pionidae
sulfide												-0.887			0.768	0.731
												0.018			0.074	0.099
nitrite						-0.75								0.895		
						0.086								0.016		
phosphat			-0.859		-0.915			-0.915		-0.905	-0.838		-0.782	0.747		
			0.028		0.011			0.011		0.013	0.037		0.066	0.088		
ammonia			-0.354			-0.738						0.865				
			0.491			0.094						0.026				
Alkalini	-0.749	-0.888										-0.98				
	0.087	0.018										0.001				
F		-0.806		-0.895								-0.867				
		0.053		0.016								0.025				
Cl					0.874			0.874		0.972	0.832			-0.891		
					0.023			0.023		0.001	0.04			0.017		
NO3				-0.735										0.94		
				0.096										0.005		
HPO4										-0.823	-0.857			0.792		
										0.044	0.029			0.06		
SO4		-0.962		-0.773								-0.928				
		0.002		0.071								0.008				
Na					0.905			0.905	0.739	0.978	0.84		0.748	-0.851		
					0.013			0.013	0.093	0.001	0.036		0.087	0.032		
Mg																
K					0.879			0.879	0.82	0.991	0.899			-0.823		
					0.021			0.021	0.046	0	0.015			0.044		
Ca	-0.806	-0.934		-0.897					-0.775			-0.835				
	0.053	0.006		0.015					0.07			0.039				
	0.053	0.006		0.015					0.07			0.039				

Table 4. June macroinvertebrate and field water column (Chemetrics) correlations. Cells colored in yellow represent significance at (P< 0.1). No taxa were negatively correlated to PO_4. However, eight taxa correlated negatively with H_2S. The only significant positive correlations were with NO_3.

Taxon	NH4-N	H2S-S	NO3-N	PO4-P		Taxon	NH4-N	H2S-S	NO3-N	PO4-P
Caenis	0.554	0.328	-0.399	0.127		Tanytarsini	-0.435	0.184	0.927	-0.265
	0.197	0.473	0.375	0.811			0.329	0.692	0.003	0.611
Callibaetis	0.019	0.254	-0.503	-0.459		Tanytarsus	-0.479	0.312	-0.366	-0.577
	0.967	0.582	0.25	0.36			0.277	0.495	0.419	0.231
Coenagrionid	-0.34	-0.117	-0.488	-0.684		Bezzia/Palpo	-0.045	-0.161	0.215	0.624
	0.456	0.803	0.267	0.134			0.924	0.73	0.643	0.185
Corisella	0.484	0.329	-0.324	0.218		Dasyhelea	-0.435	0.803	-0.223	-0.503
	0.271	0.471	0.479	0.679			0.329	0.03	0.632	0.31
Corixidae	-0.287	-0.28	-0.31	-0.65		Sphaeromias	-0.045	-0.161	0.215	0.624
	0.533	0.543	0.499	0.162			0.924	0.73	0.643	0.185
Sigara	-0.321	-0.066	-0.307	-0.503		Leptoceridae	-0.116	-0.47	-0.208	NA
	0.483	0.888	0.503	0.31			0.804	0.287	0.655	
Dytiscidae	-0.045	-0.161	0.215	0.624		Oecetis	-0.435	0.803	-0.223	-0.503
	0.924	0.73	0.643	0.185			0.329	0.03	0.632	0.31
Enochrus	-0.116	-0.47	-0.208	NA		Triaenodes	-0.435	0.803	-0.223	-0.503
	0.804	0.287	0.655				0.329	0.03	0.632	0.31
Ablabesmyia	-0.302	0.833	-0.287	-0.44		Gastropoda	-0.435	0.184	0.927	-0.265
	0.51	0.02	0.533	0.382			0.329	0.692	0.003	0.611
Apedilum	0.142	0.097	-0.467	0.02		Gyraulus	-0.164	0.106	-0.497	-0.708
	0.762	0.837	0.29	0.971			0.726	0.821	0.256	0.115
Chironomus	0.18	0.003	0.308	0.816		Physa	-0.044	0.723	-0.355	-0.445
	0.699	0.995	0.502	0.048			0.925	0.066	0.435	0.377
Cladotanytar	-0.331	-0.208	-0.32	-0.691		Planorbidae	-0.116	-0.47	-0.208	NA
	0.468	0.655	0.484	0.129			0.804	0.287	0.655	
Corynoneura	-0.456	-0.323	-0.465	-0.781		Stagnicola	-0.385	-0.078	-0.342	-0.745
	0.304	0.479	0.293	0.067			0.394	0.868	0.453	0.089
Cricotopus	0.65	-0.098	-0.373	0.387		Helobdella	-0.506	0.539	-0.331	-0.74
	0.114	0.834	0.41	0.448			0.247	0.211	0.468	0.092
Cryptochiron	-0.116	-0.47	-0.208	NA		Oligochaeta	-0.463	0.085	0.944	-0.211
	0.804	0.287	0.655				0.295	0.856	0.001	0.688
Glyptotendip	0.048	-0.775	-0.468	-0.29		Theromyzon	0.569	0.087	-0.223	0.257
	0.918	0.041	0.289	0.578			0.183	0.852	0.632	0.624
Paratanytars	-0.183	-0.482	-0.244	-0.512		Hyalella	0.879	0.086	-0.329	0.482
	0.695	0.273	0.597	0.299			0.009	0.854	0.472	0.333
Procladius	0.004	-0.595	-0.531	-0.357		Ostracoda	0.363	0.723	-0.451	-0.039
	0.993	0.159	0.22	0.487			0.424	0.067	0.31	0.942
Psectrocladi	0.517	0.281	-0.313	0.252		Nematoda	-0.235	-0.696	-0.159	-0.088
	0.235	0.541	0.495	0.63			0.613	0.082	0.734	0.868
Tanypus	-0.503	-0.043	0.918	-0.212						
	0.25	0.927	0.003	0.686						

Correlation analysis between macroinvertebrates and metals and sulfides

Table 5 displays the correlation between each of the metals and sulfides with each invertebrate taxon found among the study ponds. In summary, we found two taxa, the mayfly, Caenis and the corixids (*Corisella* and other corixids) correlated negatively with Cr, Ni, Cu, Ag and Cd. However, the other 14 taxa that had correlated negatively with PO_4 and positively with Na, K and Cl, did not relate significantly to these metals (Table 5). Further however, Cr, Ni, Cu, Ag, Cd, Be, Pb and Fe all correlated negatively with 6-10 total taxa groups, including several predacious taxa that are typically identified as being phytophilous (living in association with the SAV). Beryllium strongly negatively correlated to 7 taxa, many of which taxa are commonly associated with sediments (Table 3). Similar relationships with Be also occurred with the sediment and pore water relationships to the macroinvertebrates (see below). Notably, Lead (Pb) negatively correlated to the common damselfly, Coenagrionidae, a predator, two predacious midge taxa, three trichopteran shredder taxa, and one snail (scraper) taxon (Table 3). Overall, the many different metals found to be negatively correlated to several different taxa, offers substantial evidence that metals in the water column may generate large differences in the relative abundance of these wetland ponds (Table 5). This suggests that the absence of certain beetle and damselfly taxa in Farmington bay ponds relative to PN could be linked to elevated concentrations of metals and sulfides as compared to PN.

Table 5. Correlations between water column metals and macroinvertebrate taxa and p-values. Samples were averaged for sites. Correlations and p-values were only listed if p<0.1.
The original taxa list from Table 1 was used in the analysis. Taxa that are not listed did not show a significant correlation to the chemicals analyzed.

	Baetidae	Caenis	Callibaetis	Leptophl	Aeshnida	Coenagri	Sympetru	Corisella	Corixidae	Sigara	Dytiscidae	Haliplus	Hydrophi	Hydropor	Hygrotus	Scirtida
MeHg	0.762															
	0.078															
Li			0.972	0.926	0.856	0.79	0.926					0.926	0.926	0.926	0.926	0.926
			0.001	0.008	0.03	0.062	0.008					0.008	0.008	0.008	0.008	0.008
Be		-0.803						-0.887								
		0.054						0.018								
Al						0.804					0.78					
						0.054					0.067					
Ti								-0.785								
								0.064								
Cr		-0.826						-0.733								
		0.043						0.097								
Fe											0.89					
											0.017					
Co								-0.917								
								0.01								
Ni		-0.806						-0.758								
		0.053						0.08								
Cu		-0.748						-0.734								
		0.087						0.097								
Zn									0.956							
									0.003							
As					-0.853											
					0.031											
Se		-0.914														
		0.011														
Sr									0.845							
									0.034							
Mo									0.761							
									0.079							
Ag		-0.88						-0.8								
		0.021						0.056								
Cd		-0.788						-0.872								
		0.062						0.023								
Sb					-0.859											
					0.029											
Ba					0.534				0.89		-0.744					
					0.275				0.018		0.09					
Tl									-0.779		0.852					
									0.068		0.031					
Pb						-0.906										
						0.013				0.077						
U						-0.781										

Table 5. Continued.

	Stictota	Tropiste	Ablabesm	Chironom	Cladotan	Corynone	Cricotop	Cryptoch	Glyptote	Microchi	Paratany	Procladi	Psectroc	Tanypodi	Tanytars	Bezzia/P
THg						0.815				-0.793						
						0.048				0.06						
MeHg						0.765	0.931	0.746	0.791							
						0.076	0.007	0.089	0.061							
Li	0.926	0.926			0.926									0.926		
	0.008	0.008			0.008									0.008		
Be			-0.835							-0.744						
			0.039							0.09						
Ti										-0.854						
										0.031						
V										0.857						
										0.029						
Cr			-0.902													
			0.014													
Fe			-0.777													
			0.069													
Co										-0.948						
										0.004						
Ni				-0.929						-0.744						
				0.007						0.09						
Cu				-0.883												
				0.02												
Zn							-0.749				0.835				0.812	
							0.086				0.039				0.05	
As																
Se			-0.824	-0.885												
			0.044	0.019												
Sr				-0.795				-0.844				-0.77				
				0.059				0.035				0.073				
Mo				-0.821			-0.816	-0.863								
				0.045			0.048	0.027								
Ag			-0.931													
			0.007													
Cd			-0.835							-0.742						
			0.039							0.091						
Sb													-0.783			
													0.065			
Ba								-0.858			0.812					
								0.029			0.05					
Tl															-0.768	0.76
															0.074	0.08
Pb						-0.886		-0.806								
						0.019		0.053								
U						-0.961		-0.772								

Table 5. Continued

	Dasyhele	Ephydrid	Stratiom	Leptocer	Oecetis	Lepidopt	Gastropo	Lymnaeid	Planorbi	Stagnico	Oligocha	Theromyz	Limnesia	Pionidae	Hyalella	Turbella
THg							-0.793	-0.852								
							0.06	0.031								
MeHg																
Li			0.926	0.773	0.964	0.828				0.788	-0.794					
			0.008	0.072	0.002	0.042				0.063	0.059					
Be	-0.927					0.355	-0.744			0.251			-0.75	-0.736	-0.933	
	0.008					0.49	0.09						0.086	0.095	0.007	
Al				0.885					0.819				-0.806	-0.858		
				0.019					0.046				0.053	0.029		
Ti							-0.854	-0.747								
							0.031	0.088								
V							0.857	0.837				0.802			0.754	
							0.029	0.038				0.055			0.084	
Cr	-0.917												-0.915	-0.91	-0.798	
	0.01												0.01	0.012	0.057	
Fe	-0.747			0.752					0.863				-0.784	-0.758		
	0.088			0.085					0.027				0.065	0.081		
Co	-0.755						-0.948	-0.833						-0.468	-0.769	
	0.082						0.004	0.04							0.074	
Ni							-0.744								-0.788	
							0.09								0.063	
Cu		0.806								0.735					-0.828	
		0.053								0.096					0.042	0.052
Zn										0.859						
										0.028						
As											0.75					
											0.086					
Se	-0.858										0.818					
	0.029										0.047					
Ag	-0.945												-0.878	-0.835	-0.886	
	0.004												0.021	0.038	0.019	
Cd	-0.932						-0.742						-0.773	-0.778	-0.893	
	0.007						0.091						0.072	0.069	0.016	
Sb					-0.735						0.776					
					0.096						0.069					
Ba									-0.775	0.769						
									0.07	0.074						
Pb				-0.957	-0.8	-0.753			-0.794							
				0.003	0.056	0.084			0.059							
U				-0.779				0.758								
Positive	2	2	1	4												
Negative	6	6	7	6												

Sediment and pore water comparisons to macroinvertebrates demonstrated somewhat different relationships. Correlations between pore water metal concentrations and macroinvertebrate taxa were not as strong. Li and Na continued to show a strong positive correlation (to 20 taxa). Only Be, Sb and Cd showed a strong negative relationship to multiple species (Appendix A). However, several metals did demonstrate a negative relationship to several taxa.

Sulfide (H_2S) is common in anaerobic environments (Kadlec and Wallace 2010) and can accumulate as a result of organic (protein) decomposition or from sulfate-containing minerals in exposed geologic formations or well or spring water. Sulfate reduction to H_2S is most often biologically mediated (Kadlec and Wallace 2010). The high sulfide concentrations in PN are likely due to reduced sulfur released from local anaerobic spring water (Gwynn 2006) and to a lesser extent to organic decomposition. We examined the potential effect of sulfide in surface water and pore water among these ponds. Unfortunately, field techniques for measuring H_2S only include a detection limit of 0.2 mg/L. However, if there are few interferences (i.e. low turbidity), the Chemetrics instrument will report values much lower. Therefore, reported values below 0.2 mg/L should be considered with that caveat. Nevertheless, with the aquatic life standard for H_2S at 2 ug/L (as listed in the Utah state water quality standards), these values should be considered as an indication of a potentially toxic condition. For example, values in all ponds ranged to 100x or more than the standard. FB1 and FB2 ranged up to 0.33 and 0.5 mg/L respectively, AMB ranged to 0.19 mg/L and N47 ranged to 0.28 mg/L. Pintail ranged to somewhat less with a peak value of 0.16 mg/L during June, 0.12 in early August and 0.15 mg/L in late August. Based on these measurements, although H_2S concentrations in PN were slightly below recognized instrument detection limits, they are an indication that the low aquatic life standard for sulfide is frequently, if not regularly exceeded even in the reference pond throughout the summer. Therefore, it is possible that the overall dominance of productive wetlands by species that are tolerant to organic enrichment (i.e. Hilsenhoff et al. 1989; see additional discussion below), could be the result of consistent elevated concentrations of H_2S, as well as other related factors such as low DO and elevated pH, that may nearly continuously exceed aquatic life criteria. As such, it is also possible that the slightly lower concentrations of H_2S in PN relative to FB ponds may be responsible for the somewhat greater species richness and diversity observed during the July sampling period. Confirmation of these high H_2S concentrations using laboratory analysis with lower detection limits should be a high priority. Secondly, there is sufficient evidence to warrant inclusion of H_2S among the potential stressors in future MMI development.

Unfortunately, with the focus on the presence and potential toxicity of H_2S and metals, and the ability to only extract finite quantities of pore water from the upper strata of the sediment, we did not have enough sample volume for pore water nutrient analysis. Correlations between

pore water chemistry and macroinvertebrates demonstrated a similar relationship as the surface water between the macroinvertebrates and Na and Li, but not with K. Chloride was not measured in pore water. Again, there is no known reason why these relationships might occur on a nutrient basis, suggesting that they are likely a co-variant with some other constituent. Of the toxic metals, only Be and Sb demonstrated strong negative correlations with multiple taxa (17 with Be and 14 with Sb). However numerous other metals demonstrated a negative correlation to various taxa.

The whole-sediment samples also had fewer correlations between metals and macroinvertebrate taxa. Zn, Cu and Fe negatively correlated to 5-6 taxa. However, most of these taxa reside in the SAV rather than on the sediment surface. Also notable, sediment concentrations of these metals show perhaps a stronger inverse relationship to these phytophilous taxa than water column concentrations of these metals. This suggests a secondary relationship between sediment metal concentrations and macroinvertebrates that reside in the SAV structure. For example, Fe and Cu water column concentrations were below the chronic aquatic life criteria, so this negative relationship is unexpected. However, Fe and Cu, both known to be toxic to plants, were found to have a negative relationship with total SAV coverage and shoot density (Hoven et al. 2011). This suggests that the negative effect on macroinvertebrates may be due to the negative effect on plant productivity and hence a loss of habitat for phytophilous invertebrates.

In addition to the potential impact on SAV productivity, there are at least four additional metals that occur in greater concentrations in Farmington Bay ponds than in Pintail. These include Pb, Se, As and total Hg and MeHg. Total Hg is similar between all of the study ponds (ranging between 0.3 and 2 ng/L; Appendix C). However, methyl Hg is substantially higher throughout most of the Farmington Bay ponds (Appendix C). Utah's water quality standard does not have separate standards for total vs methyl mercury. However, it is well known that methyl mercury is considered to be at least 100 times more toxic that elemental mercury because it is readily absorbed into biological systems. Lead is of equal or greater concern because it is also lipophilic, also bioaccumulates and actually exceeded the 4-day average standard of 2.5 ug/L in most samples in most FB ponds (Appendix C). Se and As concentrations were not in excess of their respective criteria. However, the ability of inorganic forms of many of these metals to be additive or even synergistic in toxicity is well known (Miller and Mackay 1980, Lloyd 1961). Therefore, with Pb occurring in excess of its criterion and three other metals that are known to be additive in toxicity also occurring in much higher concentrations than at PSG ponds, it is clearly possible that these metals could be responsible for differences in the macroinvertebrate composition. This is particularly true for mercury, where the methyl mercury is known to biomagnify by about 10 fold for each successive link in the foodchain (e.g. Bowles et al. 2001). As such, a review of the taxa differences between PN and the Farmington Bay ponds indicated a

greater number of predatory beetle taxa and damselflies in Pintail (i.e. higher level in the food chain), a pond with much less methyl mercury than in Farmington Bay ponds. The elevated methyl mercury and/or the elevated concentrations of other toxic metals, particularly Pb, in Farmington Bay ponds could be responsible for this discrepancy.

We also organized the relationship between metal concentrations and H_2S in the three environmental media (water column, pore water and sediment) and various biological metrics, including tuber biomass, total bottom coverage of SAV, branch density per m^2, drupelet biomass and two general groups of macroinvertebrates, phytophilous and benthic (associated with the sediment surface of subsurface) (Table 6). It is perhaps most important to note the metals that were negatively correlated to multiple biological metrics. For example, while various metals in surface water, pore water and whole sediment samples were negatively correlated to SAV coverage and shoot branch density, tuber and drupelet productivity, they and other metals were also negatively correlated to phytophilous macroinvertebrates. In short, lower numbers of macroinvertebrate individuals and total taxa may be related to less habitat availability as well as direct toxicity from metals and this condition may certainly be a major reason why macroinvertebrate species composition at N47 is so vastly different from the rest of the ponds. In pursuing this line of evidence we also assembled mean sediment pore water and whole sediment metals from all ponds (Appendix B). These samples were collected at sunrise – to obtain a hypothetical diel minimum and again at about 1600 hours – to obtain a hypothetical diel maximum. Samples were also collected in the upper strata (top 10 cm) and lower strata (10-25 cm in depth; Johnson et al. 2011). These samples and analyses were performed for each monthly visit. Between the pore water and whole-sediment fractions, the only major difference that might explain the poor plant metrics is the very high pore water (dissolved) Fe concentrations relative to all of the other ponds (see appendix B). As these values have been shown to negatively correlate to both plant and macroinvertebrate metrics, Fe could be the primary reason for these “stressed” indicators in N47. Notwithstanding, MeHg, Pb and possibly Cr may also be linked to the decline in various SAV metrics. One or more of these metals may be enough to elicit the “tipping point” and further research is obviously necessary to elucidate these important signs of SAV stress. Future studies need to focus on additional sample sites or otherwise establish gradients of these stressors to identify critical thresholds. With this more intensive experimental study design additional analysis, such as “change point” analysis may help in understanding such thresholds.

Table 6. Relationship of significantly (P<0.1) negatively correlated metals and H_2S measured within the water column, pore water and sediment compartments to measurements tuber (g/m2), SAV (total % cover), SAV branch density, drupelet (g/m2) and macroinvertebrate density. Plant data were obtained from Hoven et al. (2011). Macroinvertebrate taxa were subdivided into those that were mostly associated with SAV (phytophilous) and those mostly associated with the sediment surface or subsurface (benthic).

	Biological Compartment					
Physical media sampled	Tubers	SAV % cover	SAV branch density	Drupelets	Macroinvertebrate taxa	
					Phytophilous	Benthic
Water Column		V, U	V, U	Pb, Ni, As, Se, Mo, Sb	Cu, Pb, Fe, Be, Cr, Ag, Ni, Ti, Ba, Sb, H_2S	Cu, Pb, Hg, Se, Be, Cr, Sr, Ag, Ba, Ti, Mo, H_2S
Pore Water	Ti, Fe, H_2S	Ti, Co, H_2S		Mn, Pb, Fe, Se, Ni, As, Ti,	Fe, Be, Cd, Ag, Se, Cr, Sb, H_2S	Be, Se, Pb, Cr, Sb, H_2S
Sediment	MeHg, HgT, T, Pb,	MeHg, HgT, Cr, Fe, Mo	Fe, Cr, Ba, Mo, V	MeHg, HgT, Fe, Cu, Zn, Pb, Ti, Ag, Ba, Ag,	Fe, Be, Cd, Ag, Cu, Zn, Cr, As, MeHg, Sb, Ti, Co	Be, Se, Cd, Ag, MeHg, Sb, Pb

Relationships between macroinvertebrates and habitat

Macroinvertebrate assemblages are primarily governed by the quality and suitability of the habitat. Habitat suitability is dependent on the relationships and interactions between water quality and habitat structure. The most diverse macroinvertebrate assemblages occur in locations with good water quality and a mixture of habitats. If either is absent, then macroinvertebrate assemblage diversity will be lowered.

Heterogeneous habitats

A central tenet in ecology is that, all else being equal, heterogeneous habitats will have more taxa diversity than homogenous habitats. This is obvious in open water wetlands where relatively homogenous silt and mud (2-dimensional) substrates typically have less macroinvertebrate diversity than do multi-storied (3-dimensional) macrophyte canopies (e.g. Tarkowska-Kukuryk 2010 and others). Macrophyte habitats can increase macroinvertebrate diversity in several ways: 1) considerably increased total surface area compared to relatively flat silt and mud substrates; 2) substantially increased amounts of periphyton food resources growing on the surface of the macrophytes, 3) increased security for prey taxa and cover for predators, 4) increased surface area for egg laying and development, and for a few macroinvertebrate taxa, 5) increased food resources due to direct consumption of the macrophyte tubers, leaves, drupelets, flowers, or seeds and associated epiphytes.

Water quality, macrophyte assemblages, and macroinvertebrate assemblages

In the collaborative study, Hoven et al. (2011), we demonstrated that plant assemblages were directly correlated to water quality.

Briefly, these metrics included SAV bottom coverage (as percent cover), density of SAV branches, dry weight of drupelets produced per square meter and tubers produced per square meter. Nutrients and metal concentrations in the water column that were negatively correlated ($P<0.1$) to one or more of these plant metrics include: Cl, NO_3, PO_4, H_2S, SO_4, Ca, Ni, Se, As, Mo, Sb, U and Pb. Metal concentrations in the pore water that were negatively correlated to these plant metrics include: Ti, Mn, Fe, Ni, As, Se, and Pb. Pore water H_2S was also negatively correlated to drupelet production. Most of these correlations occurred between the metals and drupelets or shoots. Metals concentrations in whole sediment that negatively correlated to these plant metrics include: THg, MeHg, Pb, Cr, Fe, Cu, V, Ag, Sb, Ti and Zn.

Macroinvertebrate assemblages however, can be affected both directly and indirectly by water quality. Direct affects are the lethal and sublethal affects on the individual taxa, which may be occurring in our study impoundments (discussed above). Indirectly, we found various water column and sediment pore water constituents to have a negative impact on macrophytes (Hoven et al. 2011) which then may be directly influencing macroinvertebrate assemblages, as previously discussed. Consequently, macroinvertebrate assemblage diversity is dependent on both water quality and habitat heterogeneity (i.e. macrophytes); both of which are interrelated and not mutually exclusive.

We presented NMS ordination of macroinvertebrate communities among ponds in Figure 8 above. Here we also present the plant community ordination among the study ponds (Figure 10). The relationships of the macroinvertebrates and plant assemblages are very similar, with only a few differences. The most obvious is that the macroinvertebrate and the plant assemblages in N47 are much different than in the other ponds. Also, both the macroinvertebrate and plant assemblages in the FB1 and FB2 sites are more similar to each other than they are to AM. However, macroinvertebrate assemblages in AM are less similar to PN, WD, and BR than are the plant assemblages. The differences in macroinvertebrate assemblages in WD can be attributed to only one sample event in June 2010.

This comparison illustrates that, for the most part, the macroinvertebrate and the plant assemblages are similarly aligned with few differences. Comparisons of NMS ordination for water chemistry and the macroinvertebrate and the plant assemblages also show similar relationships, suggesting that all three are intimately linked. However, the ordinations do not allow us to infer that macroinvertebrate assemblages are more determined by water and sediment chemistry or by plant assemblages. Further analysis will likely reveal that it is the interaction of both.

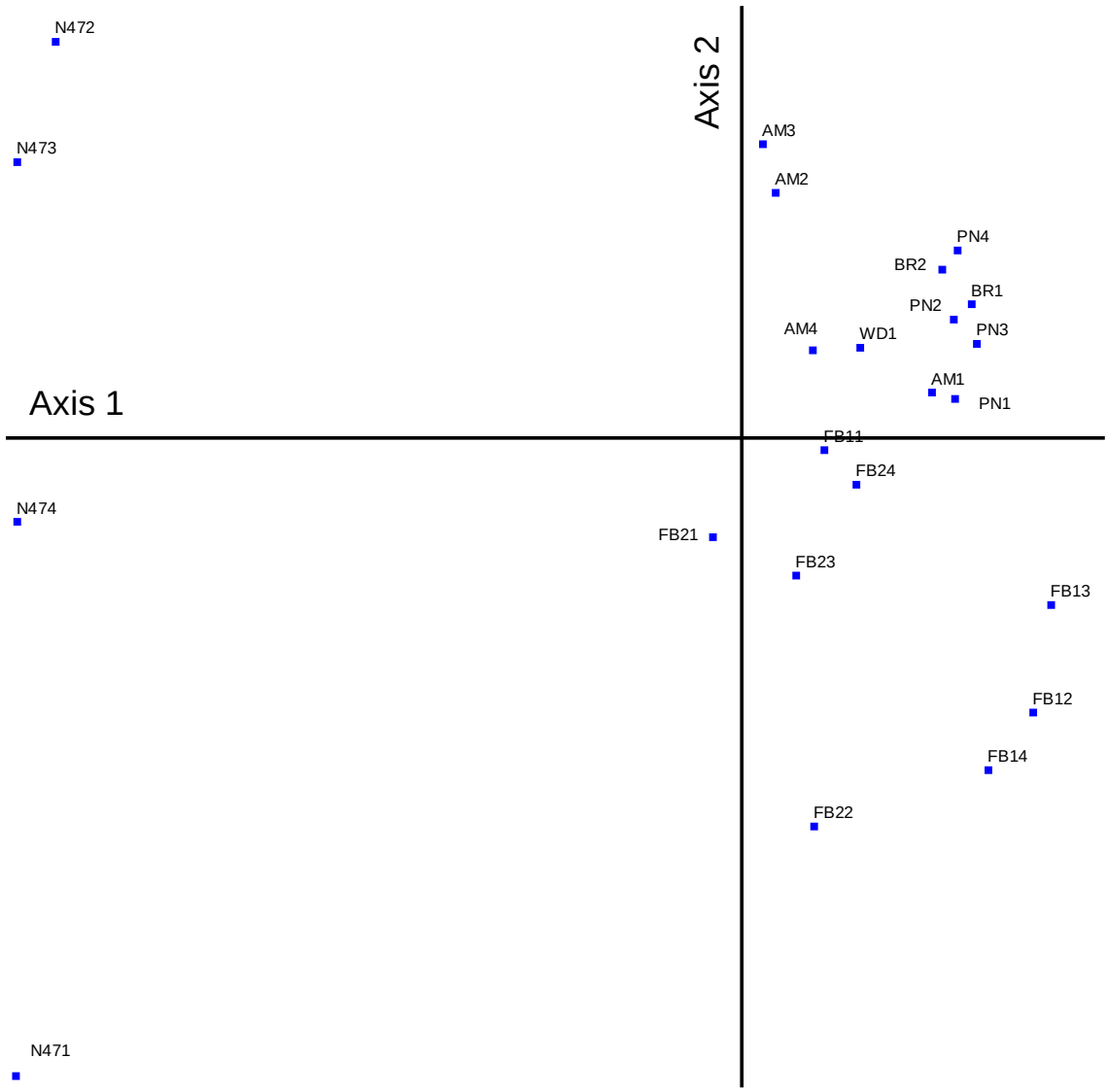


Figure 10. NMS ordination Axes 1 and 2 of plant communities by site during 2010. AM = Ambassador W1, BR = Bear River Migratory Bird Refuge Unit 4C, FB = FB WMA Units 1 and 2, N47 = New State Pond 47, PN = Public Shooting Grounds Pintail Pond outlet, WD = Public Shooting Grounds Widgeon Pond inlet.

Macroinvertebrate assemblages in wetland ponds and lakes in the GSL area have been described as "base" assemblages associated with the shallow water profundal (benthic) habitats (Barnes and Toole 1981, Gray 2009, Gray 2011, Shiozawa and Barnes 1977) and "phytophilous" assemblages more closely associated with the macrophyte community (Cyr and Downing 2006, Feldman 2001, Gray 2009, Gray 2011).

We expanded on this concept by comparing individual macroinvertebrate taxa abundances (square root transformed) (N = 22 comparable samples) with seven plant metrics: Tubers (g/ m^2) (log generalized), LEMI, %Surface Mat, % SAV, SAV shoots (#/m^2) (log generalized), % Surface Light- subcanopy, and Drupelets (g m^2) (log generalized) using Pearson's correlation coefficients and associated p-values to determine if there were relationships between macroinvertebrate taxa and the macrophyte community (Table 7). Pearson's correlation coefficients are based on linear relationships so they may not have fully captured the relationship between macroinvertebrate taxa abundance and plant metrics if the relationship was other than linear. To compensate, we accepted p-values of < 0.10 as significant relationships worth examining further.

Correlations between macroinvertebrate taxa and plant metrics are in Table 7. Significant relationships ($p < 0.10$) are highlighted in yellow. Our results are similar to Gray (2011) who combined individual macroinvertebrate taxa into a PMI [phytophilous (associated with macrophytes) macroinvertebrates] category and then compared PMI with a single categorical metric, SAV cover, however we elected to compare individual taxa with several plant metrics for higher resolution. This higher resolution allows for better insights into the relationships between macroinvertebrates and plant communities in the ponds.

As was anticipated, many of the phytophilous taxa were positively correlated with macrophyte metrics including; *Callibaetis* sp. (Ephemeroptera), Aeshnidae (Odonata), Coenagrionidae (Odonata), *Notonecta* sp. (Hemiptera), Ephydridae (Diptera), Stratiomyidae (Diptera), Leptoceridae (Trichoptera) *Oecetis* sp. (Trichoptera), an unidentified Lepidopteran taxon, *Stagnicola* sp. (Gastropoda), Ostracoda, and *Sigara* sp. (Hemiptera) (Table 7). Habitat preferences for several of these taxa are not well known which illustrates the need to further research in this area. Three additional taxa were also positively correlated with macrophyte metrics; *Chironomus* sp. (Chironomidae), *Cladotanytarsus* sp. (Chironimidae) and *Arrenurus* sp. (Acari) (Table 7). Habitat preferences for these three taxa are poorly understood. Further research and analysis is needed to determine which of the above taxa would be the most useful indicators of high quality macrophyte habitat.

Oligochaetes were negatively correlated with Drupelets (g m^2) (Table 7). *Limnesia* sp. (Acari), Pionidae (Acari), and *Hyalella* sp. (Amphipoda) were negatively correlated with % SAVT (Table 7). Little is known about habitat characteristics of Acari (mites), although *Hyalella* sp. was

shown to occur in greater abundances in ponds that contained filamentous algae and duckweed (Gray 2011). *Physa* sp. (Gastropoda) and *Stagnicola* sp. (Gastropoda) were negatively correlated with %Surface Light-subcanopy (Table 7).

Other metrics that have been commonly used as measures of community structure or health in streams is a species diversity calculation, such as the Shannon-Weaver and Hillsenhoff's (1989) pollution tolerance index of biological integrity or IBI. Figure 11 displays the Shannon-Weaver Index for every sampling event performed in each wetland complex. The lowest value, AM-6 reflects the low numbers and taxa richness discussed above (Figures 1-7). The other pond that generally scored low was N47. This also reflects the low taxa richness measured in this pond (Figure 1). Although somewhat variable seasonally, Shannon-Weavers Diversity did not demonstrate any additional differences or trends. Thus, except for large differences, which are identified with other and perhaps more sensitive metrics, the Shannon Weaver index does not seem sufficiently sensitive as a predictor of stress in these ponds. The Hillsenhoff Biotic Index (HBI) is a tool developed specifically to identify taxa that are tolerant of "organic enrichment" or specifically, tolerance to low dissolved oxygen. Figure 12 shows the results of assembling the HBI for our study ponds. As noted in previous reports (e.g. Miller and Hoven 2007) and in appendix D in this report, all of these study ponds experience tremendous DO swings (ranging from < 1 to > 15 mg/L). Note that higher HBI scores indicate greater dominance of tolerant taxa – due to more organic enrichment/lower DO conditions. Notably, AM-6 also showed the greatest stress in this metric. This occurred even though DO at that time was not particularly lower that the other ponds (Appendix D). This temporary impact was also identified in multiple measurements and metrics (Figures 1-7, 9 and 10).

Table 7. Pearson correlations and associated p-values between macroinvertebrate taxa and seven plant metrics: Tubers (g m^2), LEMI, %Surface Mat, %SAV, SAV shoots (#/m^2), % Surface Light- Subcanopy, and Drupelets (g m^2) (N = 22 comparable samples) (significant relationships at $p < 0.10$ are highlighted in yellow).

	Tubers (g m^2)	LEMI	%Surface Mat	%SAV	SAV shoots (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Caenis sp.	-0.04	-0.25	-0.02	-0.12	-0.09	0.02	0.15
	0.85	0.24	0.95	0.59	0.66	0.91	0.48
Callibaetis sp.	0.15	-0.19	-0.21	0.44	0.48	-0.22	0.62
	0.49	0.37	0.33	0.03	0.02	0.30	0.01
Aeshnidae	0.21	-0.12	-0.17	0.33	0.33	-0.28	0.70
	0.33	0.57	0.44	0.12	0.12	0.20	0.01
Coenagridae	0.21	-0.01	-0.09	0.40	0.35	-0.33	0.23
	0.34	0.97	0.66	0.05	0.10	0.12	0.29
Corisella sp.	0.05	-0.09	-0.06	-0.23	-0.07	0.09	-0.08
	0.79	0.66	0.76	0.30	0.75	0.68	0.69
Corixidae	0.05	0.32	0.32	0.11	0.01	-0.22	0.24
	0.80	0.14	0.13	0.60	0.96	0.31	0.28
Notonecta sp.	0.48	-0.12	-0.16	0.49	0.44	-0.20	0.02
	0.02	0.58	0.45	0.02	0.03	0.35	0.89

Table 7. Continued

	Tubers (g m^2)	LEMI	%Surface Mat	%SAV coverage	SAV shoots (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Notonectidae	0.01	0.32	0.25	0.04	0.16	-0.36	0.08
	0.94	0.14	0.24	0.84	0.46	0.09	0.70
Sigara sp.	-0.04	-0.16	-0.22	0.19	0.04	0.07	0.41
	0.82	0.45	0.31	0.38	0.85	0.75	0.05
Enochrus sp.	0.01	-0.12	-0.17	0.09	-0.03	-0.03	0.23
	0.98	0.57	0.43	0.69	0.88	0.88	0.29
Ablabesmyia sp.	-0.14	-0.18	0.12	-0.02	-0.03	0.01	0.26
	0.53	0.41	0.57	0.92	0.87	0.94	0.23
Apedilum sp	-0.15	-0.24	-0.11	0.07	-0.02	0.12	-0.14
	0.49	0.28	0.60	0.74	0.91	0.58	0.52
Chironomus sp.	-0.16	-0.09	0.12	-0.12	-0.13	0.37	-0.07
	0.45	0.68	0.59	0.58	0.54	0.08	0.75
Cladotanytarsus sp.	-0.07	-0.09	-0.11	0.19	-0.03	-0.01	0.38
	0.72	0.66	0.60	0.38	0.87	0.93	0.07
Corynoneura sp.	-0.22	-0.15	-0.20	0.22	-0.16	0.13	0.08
	0.31	0.48	0.36	0.32	0.47	0.56	0.71
Cricotopus sp.	-0.11	-0.10	0.11	0.04	0.02	-0.02	-0.08

Table 7. Continued

	Tubers (g m^2)	LEMI	%Surface Mat coverage	%SAV coverage	SAV shoots (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Cryptochironomus sp	-0.06	-0.16	-0.21	0.32	-0.01	0.07	-0.01
	0.78	0.45	0.32	0.13	0.99	0.72	0.93
Glyptotendipes sp.	-0.05	-0.24	-0.12	0.19	0.13	-0.02	-0.04
	0.80	0.26	0.57	0.37	0.55	0.92	0.83
Paratanytarsus sp.	-0.05	0.13	0.08	0.01	-0.15	-0.10	0.26
	0.80	0.54	0.69	0.93	0.49	0.65	0.22
Procladius sp.	0.01	-0.33	-0.18	0.15	-0.05	0.02	0.13
	0.96	0.12	0.40	0.50	0.82	0.92	0.55
Psectrocladius sp.	-0.12	-0.09	0.12	-0.10	-0.07	0.08	0.05
	0.57	0.69	0.58	0.64	0.73	0.71	0.80
Tanypus sp.	-0.20	0.37	0.31	-0.25	-0.18	0.32	-0.34
	0.35	0.08	0.15	0.24	0.41	0.13	0.11
Tanytarsus sp.	-0.06	-0.16	-0.19	0.51	0.06	0.10	-0.04
	0.77	0.46	0.39	0.01	0.77	0.63	0.83
Bezzia/Palpomyia sp.	0.11	-0.04	-0.08	0.29	0.46	-0.30	-0.23
	0.59	0.85	0.71	0.19	0.02	0.17	0.29

Table 7. Continued

	Tubers (g m^2)	LEMI	%Surface Mat coverage	%SAV coverage	SAV shoots (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Dasyhelea sp.	0.04	-0.12	-0.17	-0.02	0.11	-0.18	0.05
	0.85	0.59	0.43	0.89	0.61	0.41	0.81
Ephydridae	-0.06	-0.07	-0.10	0.29	0.22	-0.23	0.73
	0.78	0.73	0.64	0.18	0.31	0.28	0.00
Stratiomyidae	0.31	-0.12	-0.16	0.47	0.51	-0.29	0.80
	0.14	0.59	0.45	0.02	0.01	0.18	0.01
Leptoceridae	0.01	-0.14	-0.19	0.55	0.26	-0.14	0.50
	0.98	0.53	0.38	0.01	0.23	0.50	0.02
Oecetis sp.	0.05	-0.20	-0.23	0.45	0.39	-0.07	0.64
	0.80	0.37	0.28	0.03	0.06	0.75	0.01
Lepidoptera	0.08	-0.15	-0.21	0.34	0.29	-0.17	0.57
	0.70	0.49	0.3	0.11	0.18	0.42	0.01
Gyraulus sp.	0.29	-0.31	-0.21	0.07	0.23	-0.30	0.31
	0.17	0.15	0.32	0.72	0.28	0.17	0.14
Lymnaeidae	0.07	-0.11	-0.16	-0.34	-0.23	0.04	-0.11
	0.73	0.60	0.47	0.11	0.30	0.84	0.60

Table 7. Continued

	Tubers (g m^2)	%LEMI	%Surface Mat coverage	%SAV coverage	SAV Branch Density (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Physa sp.	0.05	0.01	-0.02	0.03	0.21	-0.35	-0.17
	0.80	0.98	0.92	0.89	0.33	0.10	0.42
Stagnicola sp.	-0.08	0.29	0.23	0.32	0.23	-0.36	0.69
	0.76	0.18	0.30	0.13	0.28	0.09	0.01
Helobdella sp,	-0.05	-0.13	-0.16	0.08	-0.02	-0.31	0.28
	0.81	0.55	0.46	0.72	0.90	0.16	0.20
Oligochaeta	-0.08	0.29	0.21	-0.04	0.14	-0.04	-0.40
	0.69	0.18	0.33	0.82	0.52	0.85	0.05
Theromyzon sp.	0.09	-0.09	0.13	-0.03	-0.08	-0.05	-0.01
	0.68	0.66	0.54	0.88	0.71	0.80	0.99
Arrenurus sp.	0.66	-0.17	-0.22	0.13	0.39	0.14	0.26
	0.01	0.44	0.32	0.56	0.06	0.51	0.23
Limnesia sp.	-0.16	-0.16	-0.10	-0.37	-0.26	0.28	0.12
	0.47	0.46	0.62	0.08	0.23	0.19	0.56
Pionidae	-0.16	-0.22	-0.03	-0.4	-0.26	0.17	-0.01
	0.45	0.30	0.88	0.06	0.23	0.43	0.97

Table 7. Continued							
	Tubers (g m^2)	%LEMI	%Surface Mat coverage	%SAV coverage	SAV Branch Density (#/m^2)	% Surface Light-Subcanopy	Drupelets (g m^2)
Hyalella sp	-0.08	0.03	0.15	-0.39	-0.28	0.01	-0.14
	0.71	0.86	0.50	0.06	0.20	0.95	0.51
Ostracoda	0.10	-0.09	0.01	0.15	0.35	-0.29	-0.17
	0.65	0.66	0.95	0.50	0.10	0.18	0.44
Nematoda	-0.11	0.03	-0.02	-0.08	-0.28	-0.02	-0.06
	0.60	0.86	0.90	0.69	0.19	0.89	0.79

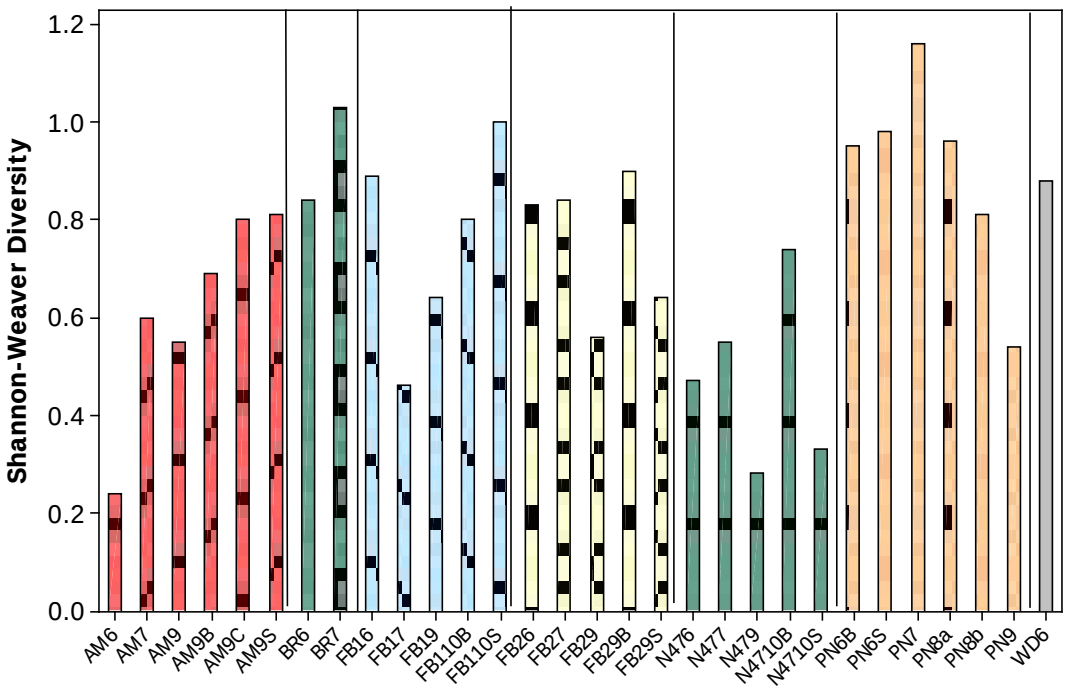


Figure11. Shannon's Diversity Index calculated for each sampling event. X-axis labels are defined as for Figure 1.

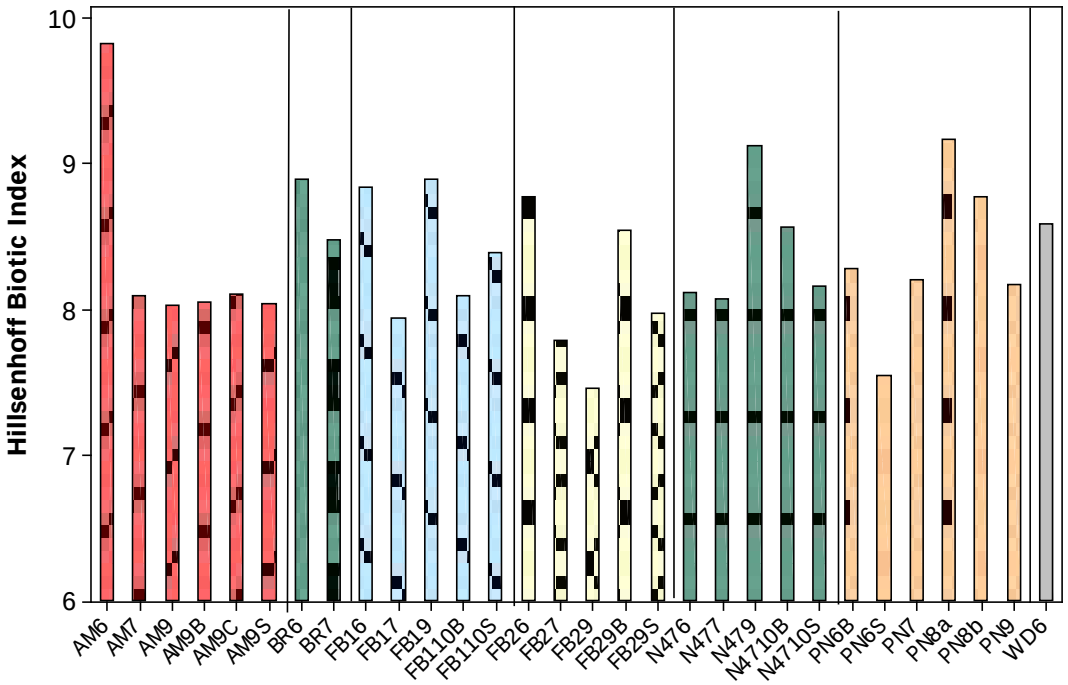


Figure 12. Hillsenhoff Biotic Index of Biotic Integrity calculated for each sampling event. This index is based on known tolerance of individual taxa to low DO conditions – as caused by organic enrichment. Note higher values indicate greater numbers of tolerant taxa. X-axis labels are defined as for Figure 1.

Implications for developing an impounded wetland MMI for Great Salt Lake

The development of various indices of biological integrity (IBI) has been in progress for about a century with the great majority being developed for stream ecosystems. Several modifications of this theme have since been developed. One of the most cited references in this field, Karr and Chu (1999) recommends selecting metrics that respond systematically to stressors along a gradient of increasing intensity. Following this recommendation, Barbour et al. (1999) summarized two basic approaches to performing biological assessments: 1) develop a multimetric index using an *a priori* approach – in which previous knowledge or a well formed hypothesis guides the identification and sampling of reference sites and test sites. Data analyses include univariate and multivariate analysis (e.g. clustering and similarity analysis, MANOVA) on various arrangements of species composition. Some of the more common metrics for streams include taxa richness, percent dominant taxa, percent EPT taxa, % coleopterans, % of different functional groups such as shredders, collectors or predators. Studies relating macroinvertebrate feeding ecology with environmental condition suggested functional-feeding metrics can detect stressors at multiple scales (Cummins 1974, Kerans and Karr 1994, Wallace and Webster 1996). Percent predators may be useful because they are usually longer living (univoltine) and may be more sensitive to contaminants that biomagnify (e.g. methyl mercury and lead). Percent gatherer genera are typically the most abundant functional-feeding group found in streams (Wallace and Webster 1996). They are also the most abundant in our impounded wetlands studies. The ultimate goals of developing these various metrics are based on taxa or groups of taxa that are known to be sensitive to degraded water quality or to habitat quality. For this to be successful, however, it has recently been acknowledged that intense habitat analysis needs to be performed in order to avoid the confusion of covariance between physical stressors (such as sedimentation, embededness, riparian degradation, etc.) that are also known to cause biological degradation (Barbour et al. 1999, Kaufman et al. 1999) and for wetland environments, plant community structure presents a similar or parallel stressor gradient (Uzarski et al 2004, Gray, 2011).

The second approach to performing biological assessments uses an *a posteriori* approach. In this approach classes or bins are arranged according data distribution. The sites are then classified according to “clusters” of similar species groups. These sites also need to be organized or “clustered” into groups possessing similar geographical and physical habitat attributes as well. Use of discriminant models to characterize the similarity of these abiotic factors is critical in order to establish reference condition. Aggregate species composition lists can then be appropriately assigned to the reference condition and are noted as the “expected” (E) list. Species lists obtained (observed) in the (test) sites in the sampling groups can then be compared to the reference (E) values (expressed as O/E) to determine the percentage of similarity or deviation from the reference condition.

Care must be taken not to misinterpret this value by arbitrarily assigning a cause that has not been carefully evaluated and shown to be related to an observed response (Karr and Chu 1999, Hawkins et al. 2000). For example, a Pearson’s Correlation should be performed to test for

autocorrelation between specific metrics (as above) and potential environmental stressors or between O/E scores and various geographical, habitat or water quality stressors. NMS or other multivariate techniques such as Principle Components Analyses, Detrended Correspondence Analysis or Linear Regression tests should also be performed to establish the stressor response relationship and to determine their degree of dependency. After a strong relationship is determined, a particular metric becomes a candidate for membership in the multimetric index.

In one example, an intensive study of the lacustrine wetlands of the Great Lakes, Uzarski, et al. (2004) used this type of approach. Chemical/physical and land use and cover parameters and plant community composition (stressors) were measured each time that macroinvertebrate samples were collected. They arranged wetlands according to four general vegetation zones (related to depth or permanence of surface water), ranging from *Juncus* and wet meadow zones to *Typha*, lily and *Sparganium* zones with the *Scirpus* zones falling in the middle. In general, principal components analysis of chemical/physical and land use and cover data (at the basin and lake scales) indicated disturbance gradients that explained the type of vegetation zone. In turn, vegetation type, and to a lesser extent water quality parameters (pH, DO and soluble reactive P) indicated disturbance gradients that explained the macroinvertebrate community. Ultimately, vegetation type appeared to explain the relationship best with communities shifting from those found in *Scirpus* zones (with a seemingly higher production:respiration ratio, less organic sediment accumulation and increased fetch and pelagic mixing) to those found in lily and *Typha* communities (with seemingly lower production:respiration ratios, more organic sediment accumulation and decreased fetch and pelagic mixing). In other words, the systematic process of evaluating potential biological attributes/metrics by their response to potential landscape/physical/chemical characteristics (stressors) using correlation analysis, DCA and PCA is critical in analyzing biological data.

Other examples have followed this general path (Koontz 2010, Lougheed et al. 2007). Lougheed et al. (2007) evaluated wetland plant, epiphytic diatoms and zooplankton metrics against land use and water quality characteristics in Midwest depressional wetlands.

One must keep in mind that unlike multi-metrics and RIVPACs-type models that have been in the development stage for decades, wetlands assessments using these techniques have just begun. These early attempts and developing IBIs have been applied to lacustrine Great Lakes (Uzarski et al. 2004, Burton et al. 1999, Burton et al. 2002, Burton et al. 2004); Missouri River riparian wetlands (Koontz 2010); Midwest depressional wetlands (Lougheed et al. 2007); and the Florida Everglades (King and Richardson 2002) for which there is also concern that those wetlands are descending into a degraded condition. However, those other wetlands remain in a largely hydrologically and often botanically natural condition and the applicable MMIs can be constructed to measure and understand relatively subtle differences or changes in taxa lists or functional groups, and the environmental stressors (adjacent land use, aquatic habitat structure and chemical characteristics) that drive the composition and functionality of the respective wetland. Hence, the major stressors are often land coverage, agricultural or urban development (buffer zone infringement) that can influence natural vegetation zones and local water chemistry. In turn, even different aquatic vegetation zones can drastically alter

macroinvertebrate communities (Uzarski et al. 2004, Burton et al. 1999, Burton et al. 2002 and Burton et al. 2004). We suggest that these are relatively subtle disturbance gradients compared to those imposed by diking/impounding, followed by intensive hydrologic and vegetative management of hundreds of thousands of acres of lacustrine habitats surrounding freshwater and brackish water tributaries to Great Salt Lake. Unlike the more natural wetlands, these impoundments are intensively managed for water depth and emergent vegetation control in order to maintain a monoculture of SAV species of *Stuckenia* or *Ruppia* for waterfowl support. Further, these ponds are often heavily treated with broad-spectrum adulticide and larvicide pesticides for mosquito control. This is the unnatural reality for Great Salt Lake impounded wetlands. The source water, dominated by agricultural and urban runoff and POTW discharges, and in the case of Public Shooting Grounds, dominated by saline spring water, has been dammed, diverted, and carefully controlled for SAV culture. These ponds represent perhaps the most extreme modification from natural conditions and management intensity anywhere in North America and perhaps worldwide. As a result, the subtle regional and land surface cover stressors will likely never be detected. Further, these ponds were constructed and have been managed for this purpose for 70 to 80 years. With the management objective of growing lush stands of SAV to feed nesting, staging and migrating waterfowl, the sediments contain up to 40 cm of a long legacy of organic and nutrient-enriched deposits from senesced vegetation as well as toxic metal remnants of mining, smeltering and refining practices focused in the Jordan River Watershed.

Nevertheless, because of these management goals, including the production of thick beds of SAV; and in spite of the decades of resource extraction, urbanization and accumulation of organically enriched sediments that are contaminated with metals, both naturally (lake deposits) and anthropogenically, these impoundments are extremely successful at supporting the nesting, rearing and staging of millions of waterfowl each year (Wilson et al. 2011). It was with the basic understanding of these characteristics that we designed our study to evaluate sediment, pore water and water column chemical constituents and physical habitat as stressors, against multiple plant and macroinvertebrate metrics that are ecologically meaningful and applicable to beneficial use support.

Therefore, it is prudent to seek a balance between metrics that describe functional health of the ecosystem as well as those that reflect a direct measure of beneficial use support (i.e. abundance of direct food items), as well as habitat and water quality needs for those organisms (SAV and macroinvertebrates) that are a component of the waterfowl diet.

As a background to the metrics development process, the Utah Division of Water Quality (2010) listed the following important requirements for MMI development:

The basic steps in creating local or regional wetlands MMI (Karr and Chu 1999) are as follows:

- Select the appropriate biotic assemblage for quantification. Appropriate examples would be wetland plants or macroinvertebrates.

- For a particular biotic assemblage (e.g., plants or macroinvertebrates), select candidate metrics that are grounded in ecological theory. Each metric should describe aspects of composition (i.e., abundance, richness), condition (fecundity, composition based on pollution tolerance), or functional organization (i.e., functional feeding groups, microhabitat preference) that are known to reflect key properties of the ecosystem.
- Test and evaluate the chosen metrics (measurements of the assemblages that show predictable and measurable responses to stressor gradients). In wetlands, typical stressors would be salinity, DO, nutrients, pollutants (e.g., urban runoff metals, agricultural pesticides) or the presence of carp or other invasive species. Metrics selected for the final MMI should respond predictably to variations along these stressor gradients.
- Ensure that the metrics are not strongly correlated. Evaluate the metric against potential covariates that may alter our interpretation of the data. This will lower the risk potentially overweighting a single line of evidence.
- Combine the metrics into an MMI. Typically, measured attributes of the assemblages are ranked (e.g., 1 through 5, 1 through 10) or continuously rescaled to remove the effects of different units. Once metrics have been transformed to remove the effects of different units or measures that move in different directions (i.e., scores get higher with stress versus scores that decrease with stress) the scores of each metric can be summed to create an overall MMI. For each assemblage that is examined, multiple metrics are typically used to create an overall MMI score.
- Test the MMI. Databases can be split to create an MMI from a subset of wetlands and to test the application on another, local set. Also, IBI scores can be compared to various stressor gradients to see if they behave similarly.

These are the basic tenets for MMI development that are accepted in the scientific community engaged in impact assessment and water quality criteria development. Under this premise, we offer several suggestions for modifying DWQ's current draft assessment MMI:

1. DWQ's draft MMI mixes individual biological metrics (response variables) with potential stressor variables including, nutrients, pH, DO, turbidity (as Chl a), SAV bottom coverage). These stressor variables should be removed from the list of candidate biological/response variables and included among the range of stressor variables (Karr and Chu 1999). These stressor gradients should be identified and developed separately for comparison with the response variables using NMS, Pearson Correlations, PCA, DCA, linear regressions, or changepoint regressions or other appropriate statistical means of identifying and testing relationships.
2. The MMI contains no metrics for waterfowl use. Because waterfowl nesting, rearing and staging are the most critically important management goal for these ponds and as an over-arching goal for GSL wetlands as a component of Western Hemispheric Shorebird Reserve System, such metrics are seriously missing from DWQ's MMI.

3. Dissolved oxygen and pH are known to vary drastically in a diel pattern. Grab samples for these parameters are just not representative. DO minima may be useful as well as pH maxima, but DWQ has not identified these as potential metrics. Similarly, P and N have been shown to have very little influence on most SAV and macroinvertebrates metrics (see above). Rather, water column, pore water and whole sediment metal and sulfide concentrations are much stronger drivers for various SAV metrics (Hoven et al. 2011, the present study).
4. Although DWQ's wetland assessment framework document infers the negative relationship between surface mates and SAV health, the surface mat metrics are not correlated to any of the SAV metrics (Hoven et al. 2011). Evaluation of light extinction measurements against SAV canopy and subcanopy density indicates SAV are well adapted to low light conditions. In addition, surface mat development and disappearance is extremely ephemeral (mats may bleach, senesce and sink or be blown away from sampling areas by frequent high winds). Therefore the presence, duration and affect on plant and macroinvertebrate populations appear to be minimal and are nearly impossible to measure and does not provide a useful metric. In addition, regardless of the extent of mat coverage during summer, they have nearly entirely disappeared by the end of August-early September. In addition, although the ancillary influence of surface mats on water temperature, DO pH, etc. may be measurable, the proposed hypothesis by USGS (that of inhibiting methyl mercury photodegradation has not been supported (Johnson and Carling, in preparation).

5. The cited SAV metrics are too variable and too simplified to fully appreciate the ecological importance particularly for waterfowl support. Different ponds develop SAV beds at different rates, depending on water depth, temperature, and recent hydrologic changes (drying, flooding). Consequently, they may also senesce in the fall at different times, regardless of nutrient concentrations or mat accumulation. Instead, we suggest the additional metrics measured by Hoven et al. (2011) many of which have direct implication and value as waterfowl dietary components. Further, many of these proposed metrics are aligned with water column, pore water and sediment H_2S and several toxic metals which may play a role in stress and early senescence. This needs to be investigated further.

6. Although the statements made in DWQ's MMI document – that macroinvertebrates are a critical component of the waterfowl diet are valid, none of the proposed metrics include specific taxa that are important to the waterfowl diet, particularly during nesting season (i.e. not even amphipods were found in the waterfowl diet-even though they were collected in the ponds; Wilson et al. 2011). Rather, the diet was largely comprised

of midge and other dipteran pupae during spring and plant seeds, SAV drupelets and tubers in the fall. Also notable, midge pupae were heavily used by waterfowl in FB Unit 1 in the fall. While metrics such as total taxa, Simpson's Diversity, coleopteran taxa and percent Ephemeroptera are well known, metrics for stream systems, and are perhaps useful for oligotrophic or pristine wetlands, these metrics have little relevance to management goals for these ponds (i.e. support for waterfowl). DWQ's metrics should be adjusted to include a value of bird counts and the presence and biomass of known dietary components. DWQ needs to be aware that waterfowl management goals may not align well with the water quality management goals and the paradigm of the social value of pristine or oligotrophic waters. By the very definition, such oligotrophic waters will support less plant and macroinvertebrate biomass and hence will logically support fewer waterfowl than a more productive mesotrophic wetland. In turn, this may actually have negative impacts to the beneficial use of support for waterfowl and shorebirds and the aquatic life in their food chain. One final note, we need to recognize that the many decades focused on culturing thick beds of SAV has left a legacy of 20-50 cm of organic and nutrient-rich sediments deposited from the annual senescence of SAV. The one nuisance created by these management goals is the regular (on a diel basis) hypoxia (from high sediment oxygen demand) and resultant dissolution and release of nutrients, metals, ammonia and sulfides back into the water column (see Appendix E). The significance of the sediment release is substantial. For example, ortho-P concentrations are consistently 1.5 to 3 times higher than in the Jordan River source water. This has become the set point or reference point for impounded wetlands surrounding Great Salt Lake. These anoxic conditions promote the production of ammonia and H_2S and chemical reduction/ionization of various metalloids which consistently have demonstrated adverse impacts to various plant and macroinvertebrate metrics and measures. Preferably, the scientific and management communities need to be aware of this and develop measures, metrics and management strategies that identify, measure and mitigate the adverse impacts of these current sediment conditions. One promising strategy may be to occasionally drain and dry the sediment surface. This has the potential to facilitate the oxidation of organic matter, oxidize, precipitate and reduce the toxicity of most metals, oxidize iron and promote co-precipitation with phosphate, reduce ammonia production and eliminate sulfides.

Summary and proposed metrics

1. There were obvious seasonal and spatial differences among the study ponds. In general, total taxa richness did not vary throughout the summer, although the relative abundance and total number and biomass decreased significantly as summer and fall progressed.
2. A series of Non-metric Multi-dimensional Scaling ordinations indicated that water column, pore water and whole sediment chemistry influenced the initial and seasonal differences within and among ponds. Although our analyses suggest that various chemical factors are strongly correlated with biological response, we still need to determine which factor(s) are the most critical in driving wetland condition and whether macroinvertebrate and SAV tissues are accumulating and or concentrating toxic levels of metals from the aquatic environment. Briefly, with the exception of PO_4, water column nutrients did not influence various SAV metrics (SAV coverage, tuber and drupelet biomass, branch density, and % surface light reaching the subcanopy). This seems logical in that virtually all rooted macrophytes, including SAV, obtain the great majority of their nutrients from sediments (Adams and Prentki 1982 and others). Water column PO_4 was negatively correlated with SAV coverage, but not to the other metrics. Notably, there was no correlation between nutrients and algal surface mats. However, N and P were correlated to the presence of duckweed. What promotes the dominance of algae vs duckweed when surface mats do form is unknown. Miller and Hoven (2007) speculated that the type of surface mat may be driven by nutrient ratios, particularly in that N limitation may favor duckweed because they are known to participate in a symbiotic relationship with N-fixing Cyanobacteria, such as *Lyngbia* sp. Most notable, a growing data base continues to demonstrate that the great majority of these nutrients originate from the sediments of these ponds rather than tributary sources such as the Jordan River and we suspect this is also the case for most of the metals that were measured. Sediment-derived nutrients and organic matter are also released in dissolved form by growing macrophytes (McRoy et al. 1972; Twilley et al. 1977) and during the senescence and decay processes that occurs throughout the growing season (Adams and Prentki 1982). In particular, dissolved nutrients released by decaying macrophytes are relatively reactive or in metabolizable forms and make an important contribution toward the nutrient and organic matter that drive pelagic production (Carpenter 1980). These nutrients are in the most biologically available form and thus are readily available for both bacterial and algal growth. In turn, the two forms of surface mat, the filamentous alga *Cladophora* sp. that develops as periphyton or as an epiphyte, and later floats to the surface, and duckweed are immediately nourished by this diel release of nutrients.

Again however, these mats did not negatively affect the SAV coverage or tuber or drupelet development as it was noted that *Ruppia* and *Stuckenia* sp. are quite shade-tolerant (Hoven et al. 2011). Secondly, although PO_4 appeared to be negatively correlated to multiple macroinvertebrate taxa, we believe this to be an artifact of the statistical analysis and consistent "non-detect" results obtained in PN and WD (i.e. these same analyses showed positive correlations to Na, K and Cl; a totally nonsensical relationship for these invertebrate taxa). Finally, when we performed correlation analysis on just the nutrient and sulfide data, there were only 2 chironomid taxa that showed a negative relationship to PO_4 while 8 taxa were negatively correlated to H_2S.

3. Correlations to metal and sulfide concentrations in the water column, sediment pore water and whole sediment samples indicate that these factors are much more likely candidates driving macroinvertebrate and plant community structure than surface mat coverage or water column nutrients. For example, several metals showed significant negative correlations to individual invertebrate taxa and to several plant metrics. In turn concentrations of several of these toxic metals were elevated in Farmington Bay ponds compared to PN and WD. Lead in particular, most often exceeded its aquatic life criterion value. In addition, Total Hg, MeHg, Cr, Fe, Cu, Zn, Mo, Se, Mn, Ni, As, U, Ti, V, Ag, Sb, Tl and H_2S were found to negatively correlate to various SAV metrics, including, SAV coverage, and branch density and drupelet biomass as well as several phytophilous invertebrate taxa. Finally, a review of the sediment pore water indicated that N47 had vastly higher pore water Fe than any other pond, which may be directly toxic or indicate an additive affect to the other toxic metals in that impoundment's sediments. The reported toxicity of Fe to *Stuckenia pectinatus* (Van Wijck et al. 1992) and the presence of Fe in the pore water of the root zone provide convincing evidence that Fe plays an important role in the early senescence in N47. Further, Fe and several other metals have both a direct affect on the presence and abundance of macroinvertebrates as well as a secondary affect by impacting habitat structure.

Proposed Metrics

These results provide additional insights into defining appropriate metrics that more accurately reflect essential ecological linkages and the physical and toxicological factors or stressors that are responsible for the changes in plant and macroinvertebrate community composition. These metrics also have significant relevance to waterfowl support.

1. SAV branch density or % of bottom covered by SAV.

Both are a direct measure of habitat availability of phytophilous invertebrates. A habitat structure metric will reflect the availability and diversity of habitat that would explain abundance and diversity of macroinvertebrates and zooplankton. If possible, this metric should be measured at least 2-3 times per summer in order to understand interrelations between habitat availability, macroinvertebrate community structure and beneficial use support. More detail of developing a habitat structure metric is provided in Hoven et al. (2011).

2. Tuber biomass in mid to late summer

   This is known waterfowl dietary item during fall staging.

3. Drupelet biomass from mid to late summer

   This is also a preferred dietary item during fall staging.

4. Numbers or biomass of midges during May and June and again in August/September

   Midges and other perhaps less common invertebrates are a primary dietary item during spring nesting and were found to be important in fall waterfowl diets - apparently if there is sufficient abundance.

5. A measure of the relative abundance of macroinvertebrate functional feeding groups

   This will identify whether major niches are being filled and can be a sensitive measure of the influence of various environmental stressors.

6. Observations of bird presence and abundance prior to the hunting season

   This provides documentation of supporting the beneficial use of these ponds.

Suggested Additional Stressor Gradient Measures

1. Water column and sediment metals (the ICP suite) and sulfides.

   Many of these metals, particularly Fe, Pb and methyl Hg and sulfides are negatively correlated to the plant metrics and several invertebrate taxa. These taxa occupy a range of ecosystem functions, including members of all major feeding groups. These could be divided into a water column chemistry and sediment chemistry metric.

2. Water column and sediment nutrients.

   Although nutrients are generally poorly correlated to the various plant metrics (i.e. only to SAV coverage), their ability to diffuse back into the water column from sediments and to influence surface mat (duckweed) growth, make this an important metric to monitor as the presence and abundance of surface mats may be subject to change in the future.

3. Total SAV coverage, branch density and drupelet biomass or density.

   Although these are useful metrics related to plant health and the waterfowl diet, they have been shown to significantly correlate to the presence and abundance of several phytophilous macroinvertebrates.

Monitoring for these additional metrics and potential stressors is vital in recognizing important ecological linkages as well as measures that are critically important in assuring that the beneficial uses of these ponds are being met.

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Appendix A

Correlations between macroinvertebrate taxa and pore water chemical values. All correlations are significant at $p < 0.10$.

	sulfide	sulfate	THg	MeHg	Li	Be	Na	Mg	Al	K	Ca	Ti	V	Cr	Se	Sr	Mo	Ag	Cd	Sb	Ba	Pb
Baetidae																		-0.87				
Caenis								-0.89	-0.78		-0.85						-0.83					
Callibae					0.91	-0.81	0.86					0.80									0.88	
Leptophl			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Aeshnida					0.94	-0.92	0.92					0.99				0.76			-0.91	-0.98	0.93	
Coenagri																						
Libellul																						
Sympetru			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Corisell																						
Corixida													-0.79			0.85	0.75					
Notonect																						
Sigara											-0.78				-0.81							s
Dytiscid		-0.88								-0.90			0.94									
Enochrus																						
Haliplus			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Hydrophi			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Hydropor			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Hygrotus			0.80		0.96	-0.99	0.88				-0.19					0.90				-0.79	0.98	
Laccophi																						0.91
Scirtida			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Stictota			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Tropiste			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Ablabesm								-0.91			-0.80											
Apedilum																						
Chironom								-0.76									-0.82					-0.80
Cladotan			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Corynone		-0.76		0.76						-0.94			0.75									
Cricotop																						
Cryptoch		-0.95		0.78						-0.87			0.90		-0.78							
Glyptote										-0.84			0.77		-0.89							
Microchi																						
Paratany					0.83	-0.91	0.78					0.79	-0.56			0.90					0.82	
Procladi	0.25	-0.89		0.52		0.18		-0.38	-0.71	-0.58	-0.59	-0.15	0.83	-0.44	-0.90	-0.39	-0.78	0.16	0.17	0.25	-0.01	-0.51
Psectroc													-0.25		-0.28		-0.30					
Tanypodi			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Tanypus	-0.87								0.77					0.91								
Tanytars														0.74	0.79		0.83					
Bezzia/P																						
Ceratopo																						0.91
Dasyhele								-0.89			-0.74											
Dolichop																						0.91
Ephydrid			0.93													0.77						
Sphaerom																						0.91
Stratiom			0.80		0.96	-0.99	0.88					0.93				0.90				-0.79	0.98	
Leptocer				0.87																		
Oecetis					0.89	-0.74	0.85														0.80	
Lepidopt																						
Gastropo																						

Appendix A. Continued.

Gyraulus	0.84																					
Lymnaeid																						
Physa																						sp
Planorbi		-0.93		0.85						-0.90			0.93									
Radix	-0.80								0.91					0.90	0.87		0.84					
Stagnico					0.89	-0.97	0.82					0.90				0.94				-0.80	0.89	
Erpobdel																						0.91
Helobdel																						0.90
Oligocha							-0.75	0.79	0.83		0.87				0.80							
Theromyz																						
Arrenuru																						
Limnesia														-0.76								
Pionidae	0.76													-0.77								
Hyalella			-0.77																			
Ostracod				-0.75																		0.76
Nematoda																						
Turbella			0.93													0.78						
	sulfide	sulfate	THg	MeHg	Li	Be	Na	Mg	Al	K	Ca	Ti	V	Cr	Se	Sr	Mo	Ag	Cd	Sb	Ba	Pb
Positve	3	5	14	5	17	1	17	1	3	0	1	15	6	3	4	18	3	1		1	17	7
Negative	2	0	1	1	0	17	1	5	2	6	6	1	3	3	4	1	4	1		14	1	1

Appendix B.

Macroinvertebrate and sediment elements correlations R values and p-values

significant highlighted in yellow for Mercury

	Caenis	Callibae	Aeshnida	Coenagri	Libellul	Corisell	Corixida	Notonect	Notonect_1	Sigara	Enochrus	Hydropor	Ablabesm	Apedilum	Chironom	Corynone	Cricotop	Cryptoch	Glyptote	Paratany	Procladi	Psectroc	Tanypus
%LOI	0.42	-0.172	-0.563	-0.27	0.649	0.396	-0.792	-0.466	0.052	-0.118	0.176	0.524	0.137	0.606	-0.233	-0.268	0.487	0.074	0.646	-0.642	0.6	-0.255	-0.478
	0.481	0.782	0.323	0.66	0.236	0.509	0.11	0.429	0.934	0.851	0.777	0.364	0.826	0.279	0.706	0.662	0.406	0.906	0.239	0.243	0.285	0.679	0.415
THg	0.068	-0.963	-0.911	-0.905	0.342	0.07	-0.576	-0.908	0.097	-0.677	-0.463	-0.03	-0.051	0.182	0.604	-0.575	0.263	-0.17	0.043	-0.864	-0.306	-0.308	0.507
	0.914	0.009	0.031	0.035	0.573	0.911	0.309	0.033	0.877	0.209	0.432	0.961	0.935	0.769	0.281	0.31	0.669	0.784	0.945	0.059	0.616	0.614	0.383
MeHg	-0.021	-0.954	-0.614	-0.842	0.028	-0.095	-0.27	-0.777	-0.054	-0.591	-0.623	-0.424	0.033	-0.062	0.821	-0.357	0.142	-0.061	-0.284	-0.631	-0.632	-0.034	0.761
	0.973	0.012	0.27	0.074	0.964	0.879	0.66	0.122	0.931	0.294	0.262	0.477	0.958	0.921	0.089	0.556	0.819	0.922	0.643	0.253	0.252	0.957	0.135
Li	0.795	0.159	-0.062	0.021	0.557	0.525	-0.822	-0.287	-0.34	0.361	0.289	0.236	0.629	0.74	-0.334	0.269	0.744	0.606	0.691	-0.469	0.808	0.303	-0.811
	0.108	0.799	0.921	0.973	0.33	0.364	0.088	0.64	0.575	0.551	0.638	0.703	0.255	0.152	0.583	0.662	0.149	0.278	0.196	0.426	0.098	0.621	0.096
Be	0.076	-0.638	-0.868	-0.6	0.652	0.084	-0.686	-0.694	0.118	-0.619	-0.257	0.471	-0.165	0.459	0.095	-0.528	0.355	-0.268	0.461	-0.831	0.061	-0.431	0.051
	0.904	0.246	0.056	0.284	0.233	0.893	0.201	0.194	0.849	0.265	0.677	0.423	0.791	0.436	0.879	0.361	0.558	0.663	0.434	0.081	0.922	0.468	0.936
Na	0.362	0.926	0.806	0.779	-0.158	0.258	0.152	0.655	-0.218	0.852	0.599	0.005	0.408	0.101	-0.6	0.634	0.077	0.506	0.176	0.589	0.658	0.442	-0.786
	0.549	0.024	0.1	0.121	0.8	0.675	0.807	0.23	0.725	0.067	0.286	0.993	0.495	0.872	0.285	0.251	0.901	0.384	0.777	0.296	0.228	0.456	0.115
Mg	0.871	0.159	0.124	0.067	0.523	0.444	-0.795	-0.271	-0.578	0.423	0.145	0.046	0.8	0.78	-0.266	0.507	0.862	0.784	0.658	-0.452	0.687	0.587	-0.791
	0.055	0.799	0.843	0.914	0.366	0.454	0.108	0.659	0.307	0.478	0.816	0.941	0.104	0.12	0.665	0.383	0.061	0.116	0.227	0.445	0.2	0.298	0.111
Al	0.182	-0.524	-0.798	-0.514	0.7	0.159	-0.751	-0.65	0.072	-0.504	-0.163	0.515	-0.075	0.546	-0.012	-0.445	0.431	-0.171	0.557	-0.817	0.213	-0.371	-0.116
	0.769	0.365	0.105	0.376	0.188	0.798	0.144	0.235	0.909	0.387	0.794	0.375	0.904	0.341	0.984	0.453	0.468	0.784	0.33	0.091	0.73	0.539	0.853
K	-0.223	0.149	-0.408	0.089	0.375	0.072	-0.142	0.084	0.469	-0.222	0.318	0.81	-0.504	0.163	-0.502	-0.472	-0.15	-0.517	0.414	-0.082	0.408	-0.689	-0.311
	0.719	0.811	0.496	0.887	0.534	0.909	0.82	0.893	0.425	0.72	0.602	0.097	0.387	0.793	0.389	0.422	0.81	0.373	0.488	0.895	0.495	0.198	0.61
Ca	-0.061	0.574	0.39	0.722	0.69	-0.376	-0.095	0.495	-0.447	-0.014	-0.071	0.816	-0.114	0.678	-0.962	0.525	0.386	-0.076	0.858	0.057	0.304	0.231	-0.786
	0.922	0.312	0.516	0.168	0.198	0.533	0.879	0.396	0.45	0.982	0.909	0.092	0.855	0.208	0.009	0.363	0.521	0.903	0.063	0.928	0.618	0.708	0.115
Ti	0.028	-0.342	-0.644	-0.266	0.813	-0.044	-0.628	-0.421	-0.008	-0.549	-0.214	0.736	-0.218	0.628	-0.277	-0.315	0.417	-0.294	0.709	-0.696	0.188	-0.347	-0.242
	0.964	0.573	0.241	0.665	0.094	0.944	0.257	0.48	0.99	0.338	0.729	0.156	0.725	0.256	0.652	0.606	0.484	0.631	0.18	0.191	0.763	0.567	0.694
V	0.67	-0.502	-0.625	-0.586	0.588	0.48	-0.984	-0.812	-0.179	-0.124	-0.052	0.098	0.475	0.638	0.18	-0.172	0.714	0.382	0.499	-0.893	0.42	0.04	-0.266
	0.216	0.389	0.26	0.3	0.297	0.413	0.003	0.095	0.774	0.842	0.934	0.876	0.419	0.247	0.772	0.782	0.175	0.525	0.392	0.041	0.481	0.949	0.665
Cr	0.471	-0.779	-0.82	-0.818	0.479	0.368	-0.879	-0.951	-0.047	-0.368	-0.229	-0.011	0.298	0.445	0.445	-0.391	0.551	0.185	0.281	-0.954	0.119	-0.117	0.1
	0.424	0.121	0.089	0.09	0.415	0.543	0.05	0.013	0.94	0.542	0.711	0.986	0.626	0.453	0.452	0.515	0.335	0.766	0.647	0.012	0.849	0.851	0.873
Mn	0.406	0.372	0.213	0.431	0.792	-0.017	-0.544	0.101	-0.574	0.125	-0.013	0.608	0.31	0.891	-0.745	0.538	0.731	0.314	0.945	-0.293	0.544	0.4	-0.9
	0.497	0.537	0.731	0.469	0.11	0.978	0.343	0.872	0.311	0.841	0.984	0.277	0.612	0.043	0.148	0.35	0.161	0.607	0.015	0.633	0.343	0.505	0.038
Fe	-0.101	-0.886	-0.979	-0.856	0.283	0.068	-0.45	-0.798	0.334	-0.713	-0.328	0.122	-0.274	0.051	0.516	-0.759	0.05	-0.385	-0.004	-0.742	-0.264	-0.565	0.53
	0.872	0.046	0.004	0.064	0.645	0.914	0.447	0.106	0.583	0.176	0.59	0.845	0.655	0.936	0.373	0.137	0.936	0.522	0.995	0.151	0.668	0.321	0.358
Co	0.057	-0.798	-0.94	-0.76	0.526	0.095	-0.65	-0.806	0.16	-0.669	-0.327	0.286	-0.151	0.33	0.317	-0.608	0.293	-0.265	0.282	-0.862	-0.084	-0.44	0.259
	0.928	0.105	0.018	0.136	0.362	0.88	0.235	0.099	0.797	0.216	0.592	0.64	0.808	0.588	0.603	0.277	0.633	0.667	0.646	0.06	0.894	0.458	0.674
Ni	-0.144	-0.536	-0.758	-0.41	0.746	-0.173	-0.529	-0.498	0.056	-0.745	-0.38	0.667	-0.361	0.498	-0.102	-0.443	0.304	-0.45	0.562	-0.73	-0.067	-0.437	0.028
	0.818	0.352	0.137	0.493	0.148	0.78	0.36	0.393	0.928	0.149	0.528	0.219	0.551	0.393	0.871	0.455	0.619	0.447	0.324	0.162	0.915	0.462	0.964
Cu	-0.109	-0.908	-0.875	-0.769	0.495	-0.15	-0.504	-0.783	0.039	-0.83	-0.588	0.196	-0.229	0.274	0.415	-0.528	0.263	-0.344	0.193	-0.843	-0.397	-0.338	0.465
	0.861	0.033	0.052	0.128	0.396	0.81	0.386	0.117	0.951	0.082	0.297	0.752	0.711	0.655	0.487	0.36	0.669	0.571	0.756	0.073	0.508	0.578	0.43
Zn	-0.095	-0.921	-0.888	-0.796	0.463	-0.119	-0.505	-0.802	0.062	-0.814	-0.567	0.162	-0.216	0.246	0.451	-0.55	0.247	-0.332	0.158	-0.841	-0.391	-0.349	0.485
	0.879	0.026	0.044	0.108	0.433	0.849	0.386	0.103	0.921	0.094	0.319	0.795	0.727	0.69	0.446	0.337	0.689	0.586	0.799	0.074	0.515	0.565	0.407
As	0.034	-0.736	-0.731	-0.563	0.779	-0.192	-0.671	-0.703	-0.221	-0.764	-0.614	0.405	-0.1	0.607	0.12	-0.267	0.54	-0.208	0.542	-0.916	-0.218	-0.134	0.121
	0.957	0.157	0.161	0.323	0.121	0.757	0.215	0.186	0.721	0.133	0.271	0.499	0.872	0.278	0.848	0.664	0.347	0.738	0.346	0.029	0.724	0.829	0.846
Se	-0.571	-0.088	-0.494	0.004	0.417	-0.308	0.04	0.066	0.429	-0.608	-0.067	0.849	-0.787	0.089	-0.392	-0.54	-0.252	-0.815	0.338	-0.129	-0.051	-0.749	0.029
	0.314	0.888	0.398	0.995	0.485	0.614	0.949	0.916	0.471	0.276	0.915	0.069	0.114	0.887	0.514	0.348	0.682	0.093	0.578	0.836	0.936	0.145	0.963
Sr	-0.247	0.857	0.678	0.953	0.264	-0.386	0.344	0.849	-0.246	0.225	0.157	0.642	-0.229	0.275	-0.968	0.545	-0.007	-0.142	0.507	0.532	0.263	0.177	-0.688
	0.689	0.063	0.208	0.012	0.668	0.52	0.571	0.069	0.69	0.716	0.801	0.243	0.711	0.654	0.007	0.343	0.991	0.819	0.383	0.356	0.669	0.776	0.199
Mo	0.363	0.605	0.569	0.68	0.595	-0.116	-0.287	0.371	-0.693	0.321	0.009	0.448	0.374	0.767	-0.804	0.795	0.642	0.416	0.814	0.001	0.473	0.618	-0.924
	0.549	0.28	0.316	0.206	0.29	0.853	0.639	0.539	0.194	0.598	0.988	0.449	0.536	0.13	0.101	0.108	0.242	0.486	0.094	0.999	0.421	0.266	0.025
Ag	-0.09	-0.889	-0.824	-0.722	0.568	-0.203	-0.531	-0.767	-0.065	-0.84	-0.65	0.226	-0.196	0.36	0.362	-0.436	0.344	-0.309	0.271	-0.868	-0.408	-0.252	0.411
	0.885	0.044	0.086	0.168	0.317	0.743	0.357	0.13	0.918	0.075	0.235	0.715	0.752	0.552	0.549	0.463	0.571	0.613	0.659	0.056	0.495	0.683	0.491
Cd	-0.226	-0.763	-0.843	-0.604	0.61	-0.24	-0.453	-0.628	0.078	-0.871	-0.544	0.452	-0.387	0.341	0.176	-0.529	0.224	-0.491	0.343	-0.773	-0.327	-0.438	0.33
	0.714	0.134	0.073	0.281	0.275	0.697	0.444	0.256	0.901	0.054	0.343	0.445	0.52	0.574	0.777	0.359	0.718	0.401	0.572	0.125	0.591	0.461	0.587
Sb	-0.566	-0.759	-0.729	-0.54	0.343	-0.48	-0.054	-0.435	0.191	-0.97	-0.649	0.334	-0.636	0.014	0.28	-0.576	-0.103	-0.72	0.04	-0.506	-0.664	-0.515	0.642
	0.32	0.137	0.162	0.348	0.572	0.413	0.931	0.464	0.759	0.006	0.236	0.582	0.249	0.982	0.648	0.31	0.87	0.17	0.949	0.385	0.221	0.375	0.243
Ba	-0.565	0.353	0.109	0.567	0.574	-0.626	0.201	0.51	-0.076	-0.419	-0.208	0.959	-0.647	0.357	-0.833	0.102	-0.03	-0.623	0.625	0.124	-0.059	-0.235	-0.347
	0.321	0.56	0.862	0.319	0.311	0.258	0.745	0.38	0.904	0.483	0.738	0.01	0.238	0.555	0.079	0.871	0.962	0.261	0.26	0.843	0.924	0.704	0.568
Tl	-0.394	-0.804	-0.835	-0.628	0.443	-0.327	-0.26	-0.574	0.181	-0.93	-0.583	0.361	-0.516	0.139	0.285	-0.609	0.033	-0.615	0.146	-0.655	-0.503	-0.517	0.533
	0.512	0.101	0.078	0.256	0.455	0.592	0.673	0.312	0.77	0.022	0.302	0.551	0.373	0.824	0.642	0.275	0.958	0.27	0.815	0.23	0.388	0.373	0.355
Pb	-0.254	-0.945	-0.865	-0.811	0.274	-0.189	-0.302	-0.735	0.174	-0.845	-0.583	0.055	-0.335	0.028	0.561	-0.626	0.046	-0.442	-0.054	-0.703	-0.552	-0.424	0.682
	0.68	0.015	0.059	0.096	0.656	0.761	0.621	0.157	0.779	0.071	0.302	0.93	0.582	0.964	0.325	0.259	0.942	0.456	0.931	0.186	0.335	0.477	0.205
U	0.642	-0.009	-0.218	-0.053	0.765	0.324	-0.871	-0.371	-0.386	0.067	0.054	0.421	0.458	0.86	-0.363	0.198	0.802	0.413	0.828	-0.636	0.635	0.223	-0.724
	0.243	0.988	0.724	0.932	0.132	0.595	0.054	0.538	0.521	0.914	0.931	0.48	0.438	0.062	0.548	0.749	0.103	0.489	0.084	0.249	0.249	0.718	0.167

Appendix B Continued

	Tanytars	Tanytars_1	Bezzia/P	Dasyhele	Ephydrid	Stratiom	Oeceti s	Lepido pt	Gastrop o	Gyraul us	Lymna eid	Physa	Stagnic o	Helobd el	Helobd el	Oligoch a	Therom yz	Arrenu ru	Limnes ia	Pionid ae	Hyalella	Ostracod	*Nemat oda*	Turbella
%LOI	-0.726	0.318	0.7	0.681	0.072	-0.562	-0.462	-0.271	-0.492	0.727	0.768	0.699	-0.747	-0.398	0.502	0.25	0.783	0.614	0.498	0.761	0.471	0.849	*0.405*	0.066
	0.165	0.602	0.188	0.205	0.908	0.324	0.434	0.659	0.4	0.164	0.129	0.189	0.147	0.506	0.389	0.685	0.117	0.27	0.394	0.135	0.423	0.069	*0.498*	0.916
THg	0.177	-0.342	0.297	0.067	-0.599	-0.972	-0.982	-0.889	0.48	-0.238	0.252	0.303	-0.876	0.457	-0.067	0.708	0.259	-0.281	0.217	0.303	0.511	0.293	*-0.111*	-0.605
	0.776	0.573	0.627	0.915	0.286	0.006	0.003	0.044	0.414	0.699	0.682	0.62	0.052	0.439	0.915	0.18	0.675	0.647	0.726	0.62	0.379	0.633	*0.859*	0.28
MeHg	0.529	-0.601	-0.134	-0.213	-0.777	-0.759	-0.786	-0.86	0.733	-0.706	-0.197	-0.153	-0.558	0.818	-0.451	0.518	-0.201	-0.727	0.105	0.01	0.371	-0.218	*-0.494*	-0.779
	0.36	0.284	0.83	0.731	0.122	0.137	0.115	0.061	0.159	0.183	0.75	0.806	0.329	0.091	0.446	0.371	0.745	0.164	0.867	0.988	0.539	0.725	*0.397*	0.12
Li	-0.972	0.273	0.383	0.923	0.028	-0.26	-0.038	-0.078	-0.828	0.678	0.557	0.311	-0.495	-0.274	0.225	-0.329	0.562	0.476	0.832	0.941	0.519	0.547	*0.013*	0.025
	0.006	0.657	0.524	0.025	0.964	0.673	0.952	0.901	0.083	0.209	0.329	0.611	0.397	0.655	0.716	0.588	0.324	0.418	0.081	0.017	0.37	0.34	*0.983*	0.968
Be	-0.244	0.147	0.728	0.269	-0.135	-0.843	-0.869	-0.66	0.031	0.356	0.532	0.743	-0.875	-0.032	0.438	0.716	0.55	0.313	0.254	0.497	0.379	0.775	*0.36*	-0.143
	0.692	0.813	0.163	0.662	0.829	0.073	0.056	0.226	0.961	0.557	0.356	0.15	0.052	0.959	0.46	0.174	0.336	0.608	0.68	0.395	0.529	0.123	*0.552*	0.819
Na	-0.572	0.312	-0.218	0.39	0.453	0.758	0.9	0.78	-0.769	0.432	0.032	-0.255	0.568	-0.492	0.033	-0.864	0.024	0.368	0.206	0.148	-0.148	-0.127	*-0.001*	0.457
	0.314	0.609	0.724	0.517	0.444	0.138	0.038	0.12	0.129	0.467	0.96	0.679	0.318	0.4	0.957	0.059	0.969	0.543	0.74	0.812	0.812	0.838	*0.999*	0.439
Mg	-0.914	0.188	0.214	0.883	-0.079	-0.195	0.042	-0.141	-0.814	0.489	0.302	0.1	-0.391	-0.025	0.039	-0.477	0.303	0.262	0.938	0.94	0.493	0.336	*-0.248*	-0.081
	0.03	0.761	0.73	0.047	0.899	0.754	0.946	0.821	0.094	0.404	0.621	0.873	0.515	0.968	0.95	0.416	0.62	0.67	0.019	0.018	0.399	0.58	*0.687*	0.897
Al	-0.404	0.221	0.76	0.399	-0.071	-0.791	-0.784	-0.583	-0.135	0.484	0.601	0.766	-0.868	-0.127	0.484	0.606	0.619	0.417	0.35	0.604	0.408	0.834	*0.383*	-0.078
	0.5	0.721	0.136	0.506	0.91	0.111	0.117	0.302	0.829	0.409	0.284	0.131	0.057	0.839	0.409	0.279	0.265	0.485	0.564	0.281	0.495	0.079	*0.525*	0.901
K	-0.368	0.567	0.739	0.152	0.564	-0.105	-0.183	0.193	-0.297	0.802	0.668	0.827	-0.251	-0.801	0.804	0.429	0.689	0.864	-0.214	0.125	-0.09	0.835	*0.908*	0.559
	0.542	0.319	0.153	0.808	0.322	0.866	0.769	0.756	0.628	0.103	0.218	0.084	0.683	0.103	0.101	0.471	0.199	0.059	0.729	0.841	0.886	0.079	*0.033*	0.327
Ca	-0.614	0.966	0.712	0.126	0.842	0.356	0.259	0.237	-0.78	0.842	-0.055	0.652	0.221	-0.459	0.827	-0.042	-0.032	0.84	0.145	0.277	-0.506	0.677	*0.587*	0.84
	0.27	0.008	0.177	0.84	0.074	0.556	0.674	0.701	0.119	0.074	0.931	0.233	0.721	0.436	0.084	0.946	0.96	0.075	0.816	0.652	0.385	0.209	*0.298*	0.075
Ti	-0.442	0.489	0.914	0.285	0.208	-0.609	-0.669	-0.467	-0.256	0.646	0.491	0.914	-0.693	-0.245	0.711	0.629	0.516	0.614	0.248	0.526	0.151	0.952	*0.573*	0.2
	0.456	0.403	0.03	0.642	0.738	0.275	0.217	0.427	0.677	0.239	0.401	0.03	0.194	0.691	0.178	0.255	0.373	0.271	0.687	0.363	0.809	0.013	*0.312*	0.747
V	-0.605	-0.071	0.42	0.752	-0.408	-0.817	-0.641	-0.618	-0.297	0.316	0.586	0.37	-0.931	0.103	0.067	0.17	0.592	0.132	0.774	0.898	0.754	0.547	*-0.109*	-0.414
	0.28	0.91	0.482	0.143	0.495	0.091	0.244	0.266	0.627	0.604	0.299	0.54	0.021	0.869	0.915	0.785	0.293	0.833	0.125	0.038	0.141	0.34	*0.861*	0.489
Cr	-0.273	-0.263	0.346	0.508	-0.584	-0.971	-0.854	-0.8	0.068	0.04	0.489	0.32	-0.994	0.3	-0.048	0.414	0.493	-0.099	0.587	0.694	0.745	0.428	*-0.165*	-0.589
	0.657	0.669	0.569	0.382	0.302	0.006	0.066	0.104	0.914	0.949	0.404	0.6	0.001	0.624	0.939	0.489	0.398	0.875	0.298	0.194	0.149	0.472	*0.791*	0.296
Mn	-0.885	0.739	0.657	0.555	0.488	0.031	0.083	-0.001	-0.91	0.83	0.172	0.561	-0.157	-0.307	0.608	-0.18	0.19	0.71	0.598	0.712	-0.037	0.699	*0.295*	0.486
	0.046	0.153	0.228	0.332	0.404	0.96	0.894	0.998	0.032	0.082	0.781	0.325	0.8	0.616	0.277	0.772	0.76	0.179	0.287	0.177	0.952	0.189	*0.63*	0.407
Fe	0.218	-0.256	0.383	-0.013	-0.441	-0.905	-0.954	-0.739	0.512	-0.119	0.388	0.433	-0.826	0.221	0.088	0.813	0.397	-0.112	0.005	0.168	0.419	0.396	*0.136*	-0.447
	0.725	0.677	0.525	0.984	0.457	0.035	0.012	0.154	0.378	0.849	0.519	0.467	0.085	0.721	0.888	0.094	0.508	0.857	0.993	0.788	0.482	0.51	*0.827*	0.45
Co	-0.065	-0.059	0.575	0.187	-0.329	-0.929	-0.951	-0.763	0.236	0.126	0.463	0.595	-0.91	0.142	0.251	0.755	0.477	0.088	0.218	0.414	0.451	0.608	*0.203*	-0.336
	0.918	0.925	0.31	0.763	0.589	0.023	0.013	0.134	0.702	0.84	0.432	0.29	0.032	0.82	0.684	0.14	0.416	0.889	0.724	0.488	0.446	0.276	*0.744*	0.581
Ni	-0.183	0.372	0.867	0.073	0.1	-0.701	-0.816	-0.604	0.013	0.436	0.372	0.88	-0.71	-0.081	0.639	0.813	0.396	0.442	0.1	0.357	0.102	0.859	*0.532*	0.093
	0.769	0.537	0.057	0.907	0.872	0.187	0.092	0.28	0.983	0.463	0.538	0.049	0.179	0.897	0.246	0.094	0.509	0.456	0.872	0.555	0.87	0.062	*0.356*	0.882
Cu	0.187	-0.111	0.494	-0.069	-0.386	-0.908	-0.997	-0.886	0.44	-0.115	0.159	0.497	-0.8	0.398	0.162	0.843	0.173	-0.121	0.115	0.237	0.295	0.446	*0.079*	-0.392
	0.764	0.858	0.398	0.912	0.521	0.033	0	0.045	0.459	0.854	0.798	0.395	0.104	0.506	0.795	0.073	0.781	0.846	0.854	0.702	0.63	0.451	*0.899*	0.514
Zn	0.199	-0.151	0.462	-0.06	-0.418	-0.918	-0.998	-0.885	0.46	-0.139	0.176	0.468	-0.81	0.404	0.127	0.833	0.188	-0.148	0.115	0.233	0.324	0.42	*0.057*	-0.425
	0.749	0.808	0.434	0.924	0.483	0.028	0	0.046	0.436	0.823	0.778	0.427	0.096	0.5	0.838	0.08	0.762	0.813	0.854	0.706	0.595	0.481	*0.927*	0.476
As	-0.129	0.174	0.71	0.111	-0.189	-0.847	-0.921	-0.868	0.092	0.172	0.16	0.67	-0.803	0.311	0.372	0.739	0.18	0.122	0.335	0.479	0.24	0.666	*0.16*	-0.196
	0.836	0.78	0.18	0.859	0.761	0.07	0.026	0.056	0.883	0.782	0.797	0.216	0.102	0.61	0.538	0.153	0.772	0.845	0.582	0.414	0.697	0.219	*0.797*	0.752
Se	0.005	0.571	0.798	-0.252	0.55	-0.179	-0.391	-0.032	0.042	0.562	0.357	0.886	-0.202	-0.54	0.84	0.756	0.383	0.696	-0.453	-0.144	-0.33	0.783	*0.927*	0.545
	0.994	0.315	0.106	0.682	0.337	0.773	0.515	0.959	0.946	0.324	0.555	0.045	0.745	0.348	0.075	0.139	0.525	0.192	0.443	0.818	0.588	0.117	*0.023*	0.342
Sr	-0.384	0.863	0.376	-0.094	0.943	0.755	0.642	0.632	-0.666	0.679	-0.184	0.354	0.639	-0.588	0.669	-0.262	-0.169	0.742	-0.178	-0.107	-0.708	0.335	*0.571*	0.945
	0.523	0.06	0.533	0.881	0.016	0.14	0.243	0.252	0.22	0.208	0.767	0.559	0.246	0.298	0.217	0.671	0.785	0.151	0.774	0.864	0.18	0.582	*0.315*	0.015
Mo	-0.786	0.72	0.408	0.424	0.555	0.352	0.4	0.209	-0.929	0.688	-0.098	0.295	0.187	-0.243	0.46	-0.436	-0.086	0.583	0.518	0.528	-0.223	0.413	*0.144*	0.555
	0.115	0.171	0.496	0.477	0.332	0.561	0.505	0.736	0.023	0.199	0.875	0.63	0.764	0.694	0.436	0.463	0.89	0.302	0.371	0.36	0.719	0.489	*0.817*	0.332
Ag	0.146	-0.052	0.531	-0.062	-0.355	-0.895	-0.989	-0.915	0.384	-0.09	0.099	0.518	-0.787	0.436	0.192	0.827	0.114	-0.105	0.168	0.278	0.26	0.47	*0.064*	-0.362
	0.815	0.934	0.357	0.921	0.557	0.04	0.001	0.03	0.524	0.886	0.875	0.372	0.114	0.463	0.758	0.084	0.855	0.867	0.787	0.651	0.673	0.425	*0.919*	0.55

Appendix B Continued

Cd	0.095	0.134	0.694	-0.1	-0.129	-0.808	-0.944	-0.769	0.311	0.117	0.216	0.709	-0.734	0.184	0.421	0.91	0.237	0.14	0.022	0.213	0.136	0.646	*0.339*	-0.136
	0.879	0.83	0.193	0.873	0.836	0.098	0.016	0.128	0.61	0.852	0.727	0.18	0.158	0.767	0.481	0.032	0.702	0.822	0.972	0.731	0.828	0.239	*0.577*	0.827
Sb	0.505	0.035	0.49	-0.505	-0.107	-0.617	-0.846	-0.676	0.632	-0.176	-0.069	0.532	-0.444	0.294	0.31	0.983	-0.052	-0.065	-0.341	-0.214	-0.129	0.367	*0.315*	-0.112
	0.385	0.956	0.402	0.385	0.864	0.268	0.071	0.21	0.253	0.777	0.912	0.356	0.453	0.631	0.611	0.003	0.934	0.917	0.575	0.73	0.836	0.543	*0.606*	0.857
Ba	-0.169	0.943	0.811	-0.319	0.906	0.278	0.03	0.179	-0.332	0.697	-0.082	0.823	0.241	-0.498	0.966	0.423	-0.054	0.821	-0.349	-0.144	-0.726	0.718	*0.872*	0.903
	0.786	0.016	0.096	0.601	0.034	0.65	0.962	0.774	0.585	0.191	0.895	0.087	0.697	0.393	0.007	0.478	0.931	0.088	0.565	0.818	0.165	0.172	*0.054*	0.036
Tl	0.33	0.031	0.568	-0.301	-0.166	-0.751	-0.927	-0.741	0.518	-0.063	0.104	0.604	-0.623	0.249	0.331	0.967	0.122	0.004	-0.171	-0.009	0.044	0.489	*0.313*	-0.172
	0.588	0.961	0.318	0.622	0.79	0.144	0.023	0.152	0.371	0.919	0.868	0.281	0.262	0.687	0.586	0.007	0.845	0.995	0.783	0.988	0.944	0.403	*0.609*	0.782
Pb	0.43	-0.265	0.314	-0.254	-0.457	-0.843	-0.959	-0.834	0.662	-0.323	0.068	0.341	-0.685	0.459	0.023	0.865	0.078	-0.288	-0.079	0.006	0.237	0.246	*0.022*	-0.462
	0.47	0.666	0.607	0.68	0.439	0.073	0.01	0.079	0.224	0.596	0.913	0.574	0.202	0.437	0.97	0.058	0.901	0.639	0.899	0.992	0.701	0.69	*0.972*	0.433
U	-0.905	0.399	0.614	0.796	0.082	-0.411	-0.265	-0.28	-0.745	0.715	0.501	0.538	-0.608	-0.203	0.404	-0.053	0.514	0.538	0.784	0.935	0.425	0.731	*0.144*	0.077

Appendix C. Laboratory analysis of water column constituents in the study impoundments.

Sample ID	sample code	Date	sulfide	nitrite	ammonia	conductivity	pH	DOC	Alkalinity	F	Cl	NO3	HPO4	SO4	Na	Mg	K	Ca
Below detection: reported as 1/2 detection limit			mg/L	mg/L	mg/L	uS/cm		mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
No data: value is average for site																		
Probe data from DEQ (only one measurement on sample date)								Filt	Unf	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt
AMDC-W1 June AM	AM1A	6/1/2010	0.05	0.06	0.20	829	8.10	14.5	180	0.51	141.5	1.02	0.67	145.0	135.50	43.79	9.82	90.12
AMDC-W1 June PM	AM1P	6/1/2010	0.05	0.05	0.20	1042	10.06	10.8	176	0.48	210.2	1.02	0.44	159.3	164.57	50.22	12.60	104.93
AMDC-W1 July AM	AM2A	7/29/2010	0.05	0.29	0.19	1736	8.35	7.1	229	0.71	253.1	1.70	0.93	224.9	183.62	50.60	15.70	76.62
AMDC-W1 July PM	AM2P	7/29/2010	0.05	0.13	0.18	1879	8.94	6.8	218	0.67	264.3	1.33	0.64	221.7	189.19	49.64	15.51	75.87
AMDC-W1 Aug AM	AM3A	8/17/2010	0.06	0.21	0.11	1537	8.45	7.2	228	0.78	280.3	2.43	0.71	228.5	192.44	53.30	16.82	75.74
AMDC-W1 Aug PM	AM3P	8/17/2010	0.06	0.09	0.35	1684	9.14	2.4	215	0.69	281.4	2.35	0.51	229.2	190.35	52.02	16.37	74.14
AMDC-W1 Sep	AM4A	9/21/2010	0.08	0.17	0.15	1620	8.62	10.5	223	0.76	302.9	1.37	0.74	238.1	202.43	55.65	17.56	74.61
N-47 June AM	N471A	6/1/2010	0.04	0.13	0.28	815	8.05	8.4	190	0.36	159.7	1.55	0.28	105.6	113.65	39.00	7.56	107.32
N-47June PM	N471P	6/1/2010	0.04	0.13	0.28	815	8.05	8.0	182	0.41	148.1	1.25	0.37	101.7	110.01	38.59	7.58	105.80
N-47 July AM	N472A	7/28/2010	0.06	0.19	0.15	1843	7.57	5.3	229	1.21	250.1	4.06	0.71	206.9	175.30	49.50	15.70	79.38
N-47 July PM	N472P	7/28/2010	0.05	0.08	0.10	1843	8.41	10.4	217	0.99	198.9	2.74	0.49	173.6	182.86	50.77	15.71	81.50
N-47 Aug AM	N473A	8/17/2010	0.05	0.19	0.22	2831	7.77	4.8	232	0.99	254.1	3.42	0.71	209.3	188.59	53.48	16.65	79.45
N-47 Aug PM	N473P	8/17/2010	0.06	0.07	0.40	3264	8.79	8.8	216	0.82	263.7	3.50	0.53	214.6	190.13	53.92	16.11	79.45
N-47 Sep	N474A	9/21/2010	0.04	0.14	0.55	1583	7.98	4.3	220	1.21	271.4	3.38	0.60	220.6	196.72	54.42	17.53	82.53
FB-1 June AM	FB11A	6/2/2010	0.06	0.02	0.25	1255	8.96	23.8	169	0.63	234.4	0.01	0.30	63.8	172.41	41.39	11.69	61.21
FB-1 June PM	FB11P	6/2/2010	0.06	0.01	0.25	1255	8.96	12.1	134	0.52	193.6	0.01	0.08	52.0	144.16	33.50	8.40	61.72
FB-1 July AM	FB12A	7/29/2010	0.05	0.03	0.09	2167	8.41	12.4	197	0.62	429.1	0.01	0.46	121.8	296.00	34.32	18.90	63.62
FB-1 July PM	FB12P	7/29/2010	0.06	0.04	0.05	2128	8.65	15.2	177	0.46	428.5	0.01	0.52	118.8	280.81	30.95	18.04	62.36
FB-1 Aug AM	FB13A	8/18/2010	0.14	0.03	0.35	2337	8.79	12.4	240	0.66	426.3	0.01	0.31	161.6	304.21	56.64	20.16	62.27
FB-1 Aug PM	FB13P	8/18/2010	0.06	0.03	0.67	2319	8.99	8.6	225	0.57	361.5	0.01	0.29	141.1	298.88	55.07	19.75	62.67
FB-1 Sep	FB14A	9/22/2010	0.07	0.03	0.10	1810	9.20	12.5	212	0.43	416.2	0.01	0.01	125.7	271.93	56.53	18.90	48.29
FB-2 June PM	FB21P	6/15/2010	0.07	0.03	0.03	1081	9.97	22.8	65	0.61	200.5	0.01	0.18	42.9	180.41	34.45	11.21	48.82
FB-2 July AM	FB22A	7/29/2010	0.67	0.04	0.06	1657	8.64	11.9	242	0.83	326.9	0.01	0.35	127.4	228.17	49.15	15.57	64.28
FB-2 July PM	FB22P	7/29/2010	0.10	0.05	0.18	1744	8.91	12.3	248	0.94	329.0	0.01	0.52	126.7	228.53	47.98	15.14	60.87
FB-2 Aug AM	FB23A	8/18/2010	0.09	0.03	0.13	2141	9.33	11.8	206	0.80	429.0	0.01	0.22	162.5	294.69	59.12	18.20	52.97
FB-2 Aug PM	FB23P	8/18/2010	0.06	0.03	0.10	2244	9.42	12.4	234	0.79	422.4	0.01	0.14	163.4	277.05	58.27	17.41	60.17
FB-2 Sep	FB24A	9/22/2010	0.06	0.04	0.09	2420	9.41	4.3	230	1.19	523.3	0.01	0.34	219.6	367.11	80.99	25.60	54.95
BR-4C June AM	BR1A	6/1/2010	0.01	0.01	0.12	1948	9.78	28.9	52	0.26	403.0	0.01	0.04	59.9	318.12	63.90	26.04	27.09
BR-4C June PM	BR1P	6/1/2010	0.01	0.01	0.12	1948	9.78	24.3	51	0.31	587.9	0.01	0.01	80.0	476.64	75.26	34.23	28.82
BR-4C July AM	BR2A	7/28/2010	0.01	0.01	0.12	1401	9.50	15.7	125	0.16	383.0	0.01	0.05	34.1	262.94	14.10	18.37	20.83
BR-4C July PM	BR2P	7/28/2010	0.01	0.01	0.12	1663	9.48	20.3	117	0.17	390.3	0.01	0.01	35.7	304.35	16.38	20.88	21.96
PSG widgeon June AM	WD1A	6/1/2010	0.06	0.02	0.12	2656	8.8	23.6	328	0.01	793.7	0.01	0.01	146.5	702.10	75.41	38.60	105.83
PSG widgeon June PM	WD1P	6/1/2010	0.07	0.03	0.12	2656	8.8	27.9	336	0.01	717.4	0.01	0.01	131.8	698.06	75.15	37.50	93.97
PSG-pintail June AM	PN1A	6/1/2010	0.03	0.01	0.12	3169	8.04	26.3	340	0.72	811.2	0.01	0.01	156.8	704.13	76.46	40.43	99.93
PSG-pintail June PM	PN1P	6/1/2010	0.06	0.02	0.12	3169	8.04	32.9	314	0.52	800.2	0.01	0.01	156.0	672.48	75.20	38.13	98.56
PSG-Pintail July AM	PN2A	7/28/2010	0.07	0.02	0.01	5287	9.62	26.1	193	0.91	1464.9	0.01	0.01	205.5	1124.34	40.65	48.98	27.68
PSG-Pintail July PM	PN2P	7/28/2010	0.07	0.03	0.09	6970	9.97	33.0	126	0.63	1379.3	0.01	0.01	203.5	1156.19	41.60	49.53	27.41
PSG-Pintail Aug AM	PN3A	8/17/2010	0.06	0.03	0.16	4831	9.52	20.9	238	0.79	1462.2	0.01	0.01	203.0	1072.93	59.12	45.06	26.66
PSG-Pintail Aug PM	PN3P	8/17/2010	0.06	0.03	0.09	5698	9.46	22.1	235	0.54	911.1	0.01	0.01	135.6	1151.82	57.93	47.37	27.53
PSG-Pintail Sep	PN4A	9/21/2010	0.05	0.03	0.24	4812	9.21	9.5	322	0.53	1082.0	0.01	0.01	282.6	855.82	106.62	41.88	39.54

Appendix C. Continued

Sample ID	sample code	Date	THg	MeHg	Li	Ti	V	Mn	Fe	Ni	Cu	As	Se	Sr	Sb	Ba	Pb	U
Below detection: reported as 1/2 detection limit			ng/L	ng/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
No data: value is average for site																		
Probe data from DEQ (only one measurement on sample date)			Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt	Filt
AMDC-W1 June AM	AM1A	6/1/2010	0.49	0.03	83.12	5.73	4.49	33.51	65.90	2.22	3.21	9.93	1.07	869.64	1.37	69.72	0.90	8.26
AMDC-W1 June PM	AM1P	6/1/2010	0.64	0.03	100.51	6.59	5.48	32.81	65.90	3.21	3.21	12.74	1.07	1012.48	1.54	79.12	1.39	9.16
AMDC-W1 July AM	AM2A	7/29/2010	0.43	0.03	110.12	3.91	3.65	33.27	64.29	2.06	2.95	14.60	1.10	931.00	1.09	54.57	4.22	6.11
AMDC-W1 July PM	AM2P	7/29/2010	0.30	0.03	110.91	3.75	4.77	10.03	64.40	2.01	3.12	12.64	1.05	917.52	0.97	51.23	3.95	5.46
AMDC-W1 Aug AM	AM3A	8/17/2010	0.95	0.02	118.20	3.58	4.40	18.67	65.13	2.11	3.52	12.81	1.06	960.44	0.96	50.50	3.89	5.93
AMDC-W1 Aug PM	AM3P	8/17/2010	0.49	0.02	117.02	3.45	5.97	9.28	64.41	2.04	3.63	12.18	1.14	933.39	0.95	48.21	3.78	5.77
AMDC-W1 Sep	AM4A	9/21/2010	0.57	0.14	126.51	3.97	4.35	48.03	71.25	2.21	2.81	12.02	0.98	969.34	0.86	48.20	3.38	5.74
N-47 June AM	N471A	6/1/2010	0.67	0.07	58.71	6.02	4.79	38.19	64.70	3.36	2.89	7.60	1.20	822.21	1.25	76.87	0.94	6.03
N-47June PM	N471P	6/1/2010	0.31	0.04	58.03	5.64	4.22	38.48	64.70	2.58	2.89	7.46	1.20	810.09	1.37	73.47	1.88	5.65
N-47 July AM	N472A	7/28/2010	0.36	0.04	103.18	2.93	4.73	32.42	60.43	1.92	2.17	10.81	1.13	919.63	0.92	51.81	3.85	5.32
N-47 July PM	N472P	7/28/2010	0.56	0.05	106.35	3.58	5.50	22.02	59.10	2.02	4.10	11.69	1.17	928.52	1.09	52.48	4.65	5.46
N-47 Aug AM	N473A	8/17/2010	0.70	0.01	115.48	3.69	4.75	39.56	63.04	1.98	2.28	11.56	1.15	960.51	0.85	52.17	3.93	5.39
N-47 Aug PM	N473P	8/17/2010	0.74	0.02	117.62	3.77	6.16	24.64	76.24	1.95	3.94	12.25	1.38	963.30	1.00	51.89	5.17	5.62
N-47 Sep	N474A	9/21/2010	0.41	0.02	120.86	4.11	4.89	27.20	64.60	1.92	1.96	10.23	1.16	977.48	0.74	49.78	3.15	5.65
FB-1 June AM	FB11A	6/2/2010	0.64	0.12	75.29	4.81	7.20	14.80	52.40	4.11	1.73	11.25	0.39	536.55	1.23	59.68	0.30	9.06
FB-1 June PM	FB11P	6/2/2010	0.64	0.04	59.14	4.70	6.45	13.59	52.40	1.75	1.73	9.21	0.39	456.83	1.11	51.89	0.31	8.61
FB-1 July AM	FB12A	7/29/2010	0.69	0.19	124.99	4.72	4.79	13.93	46.91	1.26	1.12	7.33	0.36	542.09	0.59	53.34	2.53	3.72
FB-1 July PM	FB12P	7/29/2010	0.44	0.25	116.72	5.25	4.74	9.24	46.91	1.33	1.39	7.26	0.41	504.73	0.51	52.85	2.63	3.42
FB-1 Aug AM	FB13A	8/18/2010	0.98	0.10	140.24	4.21	7.06	13.98	53.36	1.46	2.26	7.88	0.34	637.22	0.66	40.67	2.54	5.32
FB-1 Aug PM	FB13P	8/18/2010	0.52	0.05	137.15	4.31	7.67	10.24	54.94	1.89	1.76	7.93	0.44	621.52	0.68	45.17	2.72	5.28
FB-1 Sep	FB14A	9/22/2010	0.27	0.06	125.85	2.62	8.96	10.49	60.00	1.13	2.15	9.28	0.41	484.77	1.03	50.41	2.72	8.18
FB-2 June PM	FB21P	6/15/2010	0.84	0.18	85.49	2.50	3.04	8.57	58.56	0.99	2.61	6.98	0.43	383.87	0.29	19.13	2.40	2.10
FB-2 July AM	FB22A	7/29/2010	1.91	1.07	112.57	4.82	3.11	16.60	51.38	1.24	1.25	6.47	0.25	647.85	0.30	68.22	2.46	2.17
FB-2 July PM	FB22P	7/29/2010	1.18	0.80	114.48	5.18	3.52	10.72	53.13	1.25	1.61	6.69	0.40	613.16	0.39	62.40	2.55	1.85
FB-2 Aug AM	FB23A	8/18/2010	1.07	0.39	154.75	4.63	4.59	9.75	54.15	1.55	2.82	7.27	0.47	648.47	0.58	48.81	2.41	2.66
FB-2 Aug PM	FB23P	8/18/2010	0.93	0.19	149.93	8.23	4.63	9.77	56.14	2.28	3.42	7.53	0.57	677.31	0.65	59.45	2.76	2.75
FB-2 Sep	FB24A	9/22/2010	1.49	0.44	219.73	5.99	2.85	16.70	62.42	2.13	3.44	8.74	0.47	710.04	0.63	12.73	2.59	4.87
BR-4C June AM	BR1A	6/1/2010	0.85	0.13	172.44	4.09	5.80	13.81	82.50	1.81	2.21	13.59	0.39	480.74	0.87	40.09	0.05	2.77
BR-4C June PM	BR1P	6/1/2010	0.78	0.03	233.07	3.55	5.21	12.63	82.50	1.60	2.21	14.84	0.39	544.55	0.88	44.86	0.05	3.00
BR-4C July AM	BR2A	7/28/2010	1.58	0.49	150.89	3.93	2.96	8.23	58.93	0.49	1.87	5.43	0.35	211.79	0.21	24.92	2.23	0.23
BR-4C July PM	BR2P	7/28/2010	1.05	0.42	192.62	10.06	3.14	10.34	58.93	1.17	2.56	5.79	0.42	244.57	0.69	27.18	2.31	0.31
PSG widgeon June AM	WD1A	6/1/2010	0.84	0.11	322.12	7.29	6.82	44.39	65.20	2.67	3.57	10.02	0.45	1864.28	0.80	69.18	0.05	3.59
PSG widgeon June PM	WD1P	6/1/2010	0.31	0.13	311.64	5.64	3.90	23.75	65.20	2.36	3.57	9.88	0.45	1741.76	0.81	51.03	0.05	3.58
PSG-pintail June AM	PN1A	6/1/2010	0.43	0.11	316.34	4.66	3.94	42.11	65.20	2.41	3.57	11.67	0.45	1782.74	0.86	126.17	0.05	6.33
PSG-pintail June PM	PN1P	6/1/2010	0.72	0.15	307.61	6.90	7.69	29.35	65.20	2.63	3.57	10.84	0.45	1711.36	0.87	122.63	0.05	5.65
PSG-Pintail July AM	PN2A	7/28/2010	0.42	0.05	546.61	2.50	1.54	8.62	60.61	0.49	3.00	4.22	0.46	742.39	0.17	42.55	2.27	1.08
PSG-Pintail July PM	PN2P	7/28/2010	1.03	0.04	636.49	2.75	1.64	7.27	64.04	0.49	3.85	3.74	0.47	841.10	0.22	47.23	2.20	0.98
PSG-Pintail Aug AM	PN3A	8/17/2010	0.93	0.03	559.12	3.07	1.70	8.34	64.81	0.49	3.70	5.03	0.43	897.39	0.27	60.99	2.39	1.74
PSG-Pintail Aug PM	PN3P	8/17/2010	1.39	0.03	596.32	3.05	2.44	7.36	67.32	0.49	4.34	4.85	0.50	927.78	0.34	63.29	2.25	2.02
PSG-Pintail Sep	PN4A	9/21/2010	1.15	0.08	494.95	3.57	1.45	8.15	68.95	0.49	2.99	7.78	0.40	1138.98	0.21	53.34	2.21	2.62

Appendix D. average sediment pore water and whole sediment concentrations of metals and H_2S among all study ponds.

Average pore water concentrations of H2S and metals among 6 study ponds across all study months. Pore water in Widgeon Pond was not measured.

	H2S	THg	MeHg	Li	Be	Na	Mg	Al	K	Ca	Ti	V	Cr	Mn	Fe
		ng/L	ng/L	ug/L	ug/L	mg/L	mg/L	ug/L	mg/L	mg/L	ug/L	ug/L	ug/L	ug/L	ug/L
N47	0.3	1.57	0.09	164.24	0.59	286.79	53.64	221.47	21.54	94.77	35.84	5.31	1.20	234.64	1634.10
AM	9.1417	2.16	0.05	136.87	0.62	177.08	60.75	375.16	20.95	90.69	26.49	6.86	1.92	211.64	166.50
FB1	8.1271	1.49	0.07	181.44	0.72	350.91	83.81	348.29	28.84	69.51	21.57	7.31	1.74	77.68	36.02
FB2	6.4	1.62	0.07	212.45	0.57	437.85	81.31	277.90	33.97	61.76	17.64	6.24	0.96	64.35	38.43
BR	NM	1.76	0.07	176.92	0.64	321.95	75.29	333.78	27.92	73.99	21.90	6.80	1.54	117.89	80.32
PN	7.4188	2.60	0.10	588.67	ND	1151.64	158.45	256.72	68.14	92.36	14.64	5.24	0.96	93.00	74.64

	Co	Ni	Cu	Zn	As	Se	Sr	Mo	Ag	Cd	Sb	Ba	Tl	Pb	U
	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	mg/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L	ug/L
N47	1.54	3.20	2.75	3160.06	10.88	0.33	1005.08	10.14	0.33	0.46	0.50	85.06	ND	0.96	0.83
AM	1.20	2.79	2.85	4888.49	2.89	0.19	1074.25	9.46	0.33	0.36	0.50	123.78	ND	2.59	1.00
FB1	1.64	1.95	2.69	4915.39	3.24	0.14	686.82	8.95	0.33	0.47	0.69	102.40	ND	0.43	0.99
FB2	1.03	1.67	3.98	4502.73	2.13	0.13	686.68	9.06	ND	ND	0.13	130.93	ND	0.45	0.38
BR	1.29	2.13	3.18	4768.87	2.76	0.15	815.92	9.16	ND	ND	0.44	119.03	ND	1.16	
PN	1.29	2.03	8.04	3579.12	3.55	0.16	2887.59	9.52	ND	ND	ND	608.48	ND	0.52	0.90

Average whole sediment metal concentrations among 5 study ponds across all study months. Pore water in Widgeon Pond and Bear River 4C were not measured.

	THg	MeHg	Li	Be	Na	Mg	Al	K	Ca	Ti	V	Cr	Mn	Fe
	ng/g	pg/g	mg/kg	ug/kg	g/kg	g/kg	g/kg	g/kg	g/kg	mg/kg	mg/kg	mg/kg	mg/kg	g/kg
N-47	197.2	99.7	16.3	298.8	0.5	7.7	2.7	0.9	54.2	37.2	8.2	4.9	165.7	4.3
AM	135.2	15.2	38.6	474.3	0.8	12.6	4.2	1.2	112.8	60.0	12.5	6.1	352.8	4.0
FB-1	98.5	17.7	41.3	331.7	1.0	13.7	3.2	1.0	63.2	41.8	13.0	6.3	243.9	3.4
FB-2	103.8	42.3	41.9	290.2	1.0	17.1	2.9	0.8	81.3	40.3	13.0	6.3	318.0	2.6
PN	10.1	4.5	25.4	172.1	1.5	10.8	2.0	0.9	94.9	32.1	6.2	2.2	264.9	1.3

	Co	Ni	Cu	Zn	As	Se	Sr	Mo	Ag	Cd	Sb	Ba	Tl	Pb	U
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	ug/kg	g/kg	ug/kg	ug/kg	mg/kg	ug/kg	mg/kg	ug/kg	mg/kg	mg/kg
N-47	2.0	4.9	60.2	254.3	5.9	67.7	0.2	26.9	55.3	2.1	18.1	77.0	289.8	577.1	0.6
AM	2.5	6.9	63.7	244.4	9.0	110.6	0.6	81.1	62.7	3.7	15.9	117.2	311.0	217.0	2.3
FB-1	2.0	4.8	20.2	82.0	4.7	61.8	0.3	46.7	26.6	0.9	7.7	69.2	91.2	69.7	1.7
FB-2	1.8	4.7	28.4	108.6	6.6	44.1	0.3	91.1	40.4	1.0	8.4	73.6	95.2	82.3	2.0
PN	1.1	3.7	3.4	11.6	2.7	58.8	0.8	83.1	9.9	0.2	6.0	94.2	37.7	8.2	0.9

Appendix E.

Diel patterns of Sulfide, Ammonia and Nutrients

Introduction

The impounded wetlands of Farmington Bay WMA and private duck club ponds have been identified as being nutrient-rich since sampling began in 2004 (Miller and Hoven 2007). Phosphorus concentrations in grab samples from various Farmington Bay impoundments have ranged from about 0.5 mg/L to approximately 4 mg/L. It has long been suspected that the source of these nutrients has been the Jordan River. However, samples collected from the Jordan River near its terminus have been consistently less than impoundment concentrations. For example, ortho-P concentrations during 2010 at Burnham Dam [approximately 1.2 km upstream from Newstate pond 47 (NS47) and approximately 7 km upstream from Farmington Bay WMA Unit 1 (FB1)], have averaged only 0.38 mg/L (range = 0.23 to 0.79 mg/L; JR/FBWQC data). Similarly, the average ortho-P measured in the Legacy Nature Preserve, about 5 km upstream from Burnham Dam (approximate 0.4 km downstream from the South Davis South POTW) during 2008-2009 was 0.51 mg/L (range = 0.13 – 1.05 mg/L). Further, the limited diel DO measurements that had been performed previous to 2010 indicated that DO in nearly all of the study impoundments in Farmington Bay, as well as designated reference impoundments in Public Shooting Grounds and the Bear River Migratory Bird Refuge (BRBR) routinely fell to or nearly to 0.0 mg/L each night. The elevated pond concentrations relative to the inflow concentrations and the daily hypoxia experienced in the ponds suggests the pond sediments themselves are contributing substantial amounts of nutrients to the overlying water column. Specific mechanisms that would facilitate this transfer include the daily swings in the oxidation/reduction (red-ox) potential may play an important role in nutrient, H_2S and ammonia dynamics. For example, the ionic bonding and precipitation of ferric phosphate (Fe^{+++} PO_4^{---}) under oxidized conditions, followed by their dissociation and resolubilization (as a result of reducing Fe^{+++} to Fe^{++};) under the evening and early morning hypoxia may be occurring at the sediment/water interface (Mortimer 1941). Ortho-phosphate can also be involved in the formation of the mineral hydroxyapatite ($Ca_5(PO_4)_3(OH)$), although this mineral does not likely participate in the diel exchange of PO_4 because the pH needs to fall below about 6 in order to get the mineral to dissolve (e.g. Haobo and Darvell 2010). Similarly, nitrogen is also easily recycled in the sediments, as a result of decomposition and deamination (dissociation of the NH_3 group from larger molecules) of organic matter (Kemp et al. 1990). Subsequent diffusion from the sediments readily occurs to the overlying water and particularly if the overlying water is hypoxic (Mortimer, 1941, Hutchinson 1957). If the overlying water column is oxic ammonia is readily nitrified to nitrate (NO_3^-). Both of these nitrogen compounds are readily assimilated by bacteria and plants (phytoplankton in the water column), although nitrate has to be reduced to NH_4 before it can be reincorporated into biological compounds (mostly proteins and nucleic acids), a process called amination). If nitrates are not assimilated by plants or microbes and are exposed to anaerobic conditions, denitrification by another set of anaerobic bacteria occurs. This multi-step process is generally written as $NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2$, with each step of reduction being performed by a different group of bacteria. H_2S undergoes similar transformations between anaerobic and aerobic conditions. A common group of bacteria,

known as sulfate reducing bacteria, or SRB, utilize S and a terminal electron acceptor by reducing SO_4^{--} to H_2S or HS^- .

In order to increase our understanding of the importance of sediment as a source of nutrients, and the patterns of release (potentially following diel aerobic and anaeroabic cycles), we collected samples from specially constructed chambers (designed to reduce the turbulence and sediment disturbance during sampling) and from the ambient water column at 3 to 4-hour intervals.

Methods and Materials

Table 1 provides wetland sampling sites and general descriptions. Ambassador W1 (AMB), New State Pond 47 (N 47) , and Farmington Bay's Units 1 and 2 (FB 1 and FB 2 respectively) receive inflows from the Urban Wasatch Front via Utah Lake, the Jordan River and its tributaries, to the State Canal and Surplus Canal. Bear River Pond 4C (BR), Pintail (PN) and Widgeon (WD) ponds are located in Bear river Bay. BR receives most of its water from the Bear River. PN, and WD are located in Public Shooting Grounds WMA and receive the majority of their water from nearby saline springs with supplemental supply from irrigation return flow.

Table 1. Wetland sampling sites and general description.

Site/pond location	Flow regime	
Ambassador W1	Inflow	Upper Jordan River via Surplus Canal & upstream wetlands/ponds via Surplus Canal
outlet	Outflow	drains through additional duck club ponds, and to Farmington Bay
New State #47	Inflow	Lower Jordan River approximately 1.5 km below Burnham Dam
inlet	Outflow	Through additional duck club ponds, to Farmington Bay WMA and to Farmington Bay
Unit 1, FBWMA	Inflow	Jordan River via the State Canal
outlet	Outflow	Farmington Bay
Unit 2, FBWMA	Inflow	Jordan River via the State Canal
outlet	Outflow	Farmington Bay
Bear River 4C	Inflow	Bear River
outlet	Outflow	Bear River Bay
Pintail, PSG WMA	Inflow	Salt Creek via upstream wetlands in PSG WMA
outlet	Outflow	Widgeon pond
Widgeon, PSG WMA	Inflow	Pintail pond
outlet	Outflow	drains through to Bear River Bay

The study ponds were visited three times during the summer and fall of 2010. Sampling was timed to coincide with sediment sampling (Johnson et al. 2011), plant sampling (Hoven et al. 2011) and macroinvertebrate sampling described in this report.

Special chambers were constructed of clear Plexiglas. These chambers were approximately 30 cm tall, open at the bottom and fitted with a lid that contained sampling ports for inserting pipettes. However, after the initial two tests (at Ambassador and Newstate, June 14/15 and June 17/18 respectively), the lids were elevated approximately 2 cm above the lip of the chamber because we suspected that the head space was becoming dominated by sediment-derived gasses (including H_2S) from ebullition and hence causing an unnatural increase in water column concentrations because of increased partial pressure in the head space (as a function of Henry's law). Also, where pond depth was much greater than about 30 cm a taller chamber (approximately 60 cm tall) was used so that the sediment and overlying water column could be completely isolated from ambient conditions.

Chambers were set in triplicate in order to account for sediment heterogeneity and variability. One of the three chambers was fitted with an *In situ*® multi-parameter data recording sonde that was fitted with an optical DO probe, pH, conductivity and temperature probes. The only exception was that the sondes used for Pintail and Widgeon monitoring were fitted only with temperature and DO probes. Data were recorded for a minimum of 24 hours. Water samples were extracted every 3 – 4 hours and analyzed for sulfide, ammonia, nitrite, nitrate and ortho-phosphate using Chemetrics® field testing colorimetric kits. The analyses in these kits are based on EPA standard methods. Ambient probe data and Chemetrics samples were also collected.

Results

Table E-2 provides dissolved oxygen, pH, and temperature data measured during the 24-hour sampling events for each pond. The June 14-15 measurements at AMB experienced the largest diel swing in DO, pH and temperature of any ponds and during any sampling event (Figure E-1). AMB (approximately 1500 ha in size) receives the benefit of settling/removing the great majority of the Jordan River suspended solids in an upstream pond (approximately 2800 ha). This typically results in greater clarity of the water – enhancing the photosynthetic potential of the SAV and phytoplankton in AMB as compared to the other ponds. The other Farmington Bay impoundments receive water from the State Canal (FB 1 and FB 2) or directly from the lower Jordan River (N47). However, it should also be noted that the June sampling event occurred during the spawning period for the common carp. Hence, each pond experienced episodes of increased turbidity from carp foraging and spawning activity.

Table E-2. Average, minimum, and maximum DO, pH, and temperature.

			DO (mg/L)				pH				Temp. (deg. C)			
Pond	**Date**	**Probe**	**avg**	**min**	**max**	**spread**	**avg**	**min**	**max**	**spread**	**avg**	**min**	**max**	**spread**
Ambassator W1	June 14-15, 2010	Ambient	11.75	0.92	23.52	22.60	9.2	8.1	10.2	2.1	21.1	13.7	29.7	16.0
		Chamber	8.58	0.54	17.21	16.67					22.1	14.9	30.8	15.9
Ambassator W1	July 23-24, 2010	Ambient	6.03	0.03	14.93	14.90	8.5	8.2	9.1	0.9	26.4	22.1	30.3	8.2
		Chamber	4.33	-0.08	12.46	12.54	8.5	8.2	9.1	0.9	25.7	22.2	29.6	7.4
Ambassator W1	Sept. 24-25, 2010	Chamber	8.13	3.71	14.89	11.18	8.7	8.5	9.0	0.5	17.8	14.4	23.1	8.7
New State #47	June 17-18, 2010	Ambient	7.31	5.26	11.06	5.80	8.6	8.4	8.9	0.5	16.2	14.1	22.6	8.5
		Chamber	4.63	0.14	8.44	8.30					18.4	14.5	23.7	9.2
New State #47	July 22-23, 2010	Ambient	6.01	1.05	12.12	11.07	8.1	7.8	8.6	0.8	24.3	20.8	28.7	7.9
		Chamber	6.00	0.77	11.71	10.94	8.1	7.7	8.5	0.8	25.0	21.3	29.8	8.6
New State #47	Sept 28-29, 2010	Ambient	6.89	4.51	12.32	7.81	8.3	8.1	8.7	0.5	18.7	17.0	20.9	3.9
		Chamber	7.17	4.04	14.18	10.14	8.1	7.9	8.5	0.6	18.8	17.0	21.3	4.3
Unit 1, FB	July 26, 2010	Ambient	4.32	2.18	5.96	3.78	8.6	8.5	8.7	0.2	27.6	25.6	29.8	4.1
		Chamber	3.43	1.93	5.60	3.67					27.6	25.7	29.5	3.8
Unit 1, FB	Sept. 27-28, 2010	Ambient	10.00	8.03	12.12	4.09	9.6	9.5	9.6	0.1	20.5	18.8	22.5	3.6
		Chamber	7.24	1.99	12.23	10.24	9.3	9.1	9.5	0.4	20.1	19.0	21.9	2.9
Unit 2, FB	June 18-19, 201	Chamber	8.92	5.63	13.89	8.26					20.4	19.5	22.5	3.0
Unit 2, FB	July 26, 2010	Ambient	8.91	6.59	11.54	4.95	9.1	9.1	9.2	0.2	28.3	26.4	30.0	3.6
Unit 2, FB	Sept. 27-28, 2010	Ambient	21.71	19.19	26.00	6.81					21.6	20.1	23.4	3.4
Bear River 4C	June 23-24, 2010	Ambient	9.82	7.54	12.47	4.93	10.4	10.3	10.6	0.3	19.5	18.5	20.3	1.8
		Chamber	12.29	9.83	14.30	4.47					20.0	18.8	22.8	3.9
		Large	11.15	8.88	13.54	4.66					19.5	18.4	21.3	2.9
Bear River 4C	July 24-25, 2010	Ambient	5.47	1.97	9.80	7.83	9.9	9.7	10.2	0.5	23.8	19.2	29.4	10.2
		Chamber	3.58	-0.09	5.79	5.88	9.8	9.7	10.0	0.3	22.5	19.5	25.2	5.7
		Large	4.77	3.73	6.66	2.93					22.3	19.4	24.9	5.5
Pintail	June 22-23, 2010	Ambient	12.31	7.34	17.38	10.04					20.2	16.2	24.6	8.4
		Chamber	11.74	7.17	15.08	7.91					20.6	16.7	24.8	8.1
Widgen	June 22-23, 2010	Ambient	12.80	7.34	18.53	11.19					20.2	17.0	23.8	6.8
		Chamber	11.67	9.04	13.90	4.86					20.8	17.8	24.4	6.6

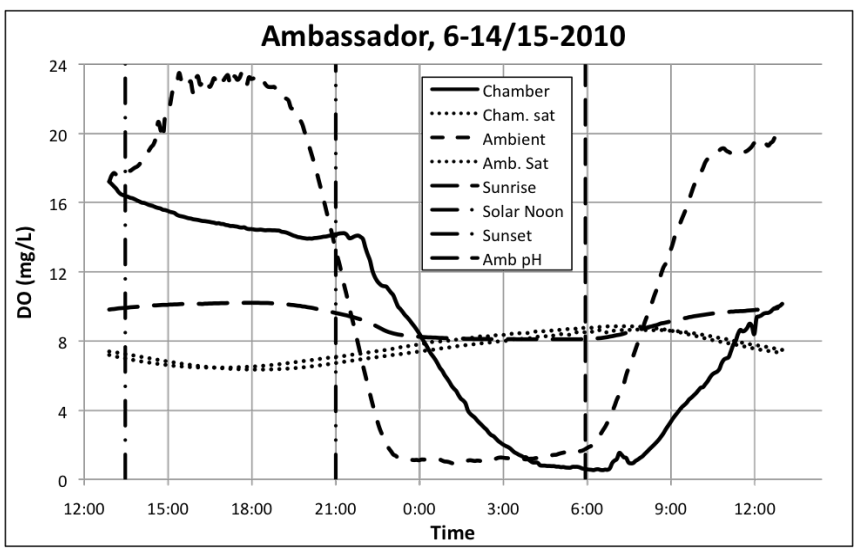


Figure E-1. Diurnal DO and pH fluctuations in AMB during the June sampling event.

The fluctuation in pH is the result of photosynthesis and the assimilation of carbon dioxide (CO_2), carbonic acid, (H_2CO_3) and bicarbonate, (HCO_3^-) by plants. The depletion of CO_2, H_2CO_3 and HCO_3^- reduces the buffering capacity and results in an increase in ambient pH. During nighttime hours, heterotrophic and autotrophic respiration reverses this pattern and results in a net increase in dissolved CO_2, H_2CO_3, and HCO_3^- , which causes a decrease in pH. Diurnal swings in pH were observed in all ponds during all sampling events.

In comparison, Farmington Bay WMA FB1 and FB2 experienced the smallest diurnal DO, temperature and pH fluctuations. A typical example of this condition is displayed in Figure E-2. This was despite the fact that the entire source water is also the Jordan River, (delivered through the State Canal, a diversion near the terminus of the Jordan River). Notably, even with this narrower range, the DO did not fall as low as that in AMB. There are likely two linked reasons for this discrepancy; 1) The greater turbidity inhibited primary productivity relative to AMB; and (2) The lower plant biomass responded with lower respiration rates. A third and important reason is that the Farmington Bay ponds are generally deeper (25-40 cm). Hence, the larger mass of water is not as responsive to daytime solar heating and similarly, reduces the effectiveness of convective or evaporative heat loss during evening hours. Diel data plots of the remaining study ponds are displayed below.

Generally, DO probes that were placed outside the chambers recorded similar diel fluctuations in DO, pH and temperature to those inside the chambers (e.g. Figure E-3 (B)). However, there were some cases where both DO and pH diverged between the ambient and chamber measurements. This could be the result of decreased light in the chambers or lower rates of photosynthesis (because plants were generally excluded from the chambers), or both. FB2 had peculiar DO signatures during the June and September sampling events where no obvious diurnal pattern was observed (Figure E-6). During both of these sampling events, the DO levels steadily decreased in a near-linear fashion. Unfortunately, a probe was not set outside the chamber so we could not determine if this pattern was an artifact of the chamber or reflected the actual pond conditions.

The entire set of chamber and ambient pond data are displayed below. The following discussion is based on notable highlights of these data sets.

The three spikes following a DO deficit in the FB 2 June sampling event correspond to nutrient sampling events, and are most likely the result of ambient water with higher DO replacing the water removed for sampling (Figure E-11). The DO probe monitoring ambient conditions during the September sampling event in FB 2 shows a highly supersaturated DO concentration of 21.7 mg/L, but very odd trends have been observed in this pond (Figure E-6). During the June sampling event we observed large amounts of gas bubbles being released from the sediment or from the algal surface mat as we maneuvered the float tube to the sampling site. During daytime, it is possible that these gas bubbles contained oxygen that was retained by the surface mat – as DO had reached supersaturated conditions. This might also be the cause of the apparent oxygen supersaturation that persisted during evening hours (Figure E-6, ambient), although we have typically observed ponds that experience supersaturated daytime DO values also experience hypoxia in late evening and early morning hours because of the high respiration rates associated with such high primary production rates.

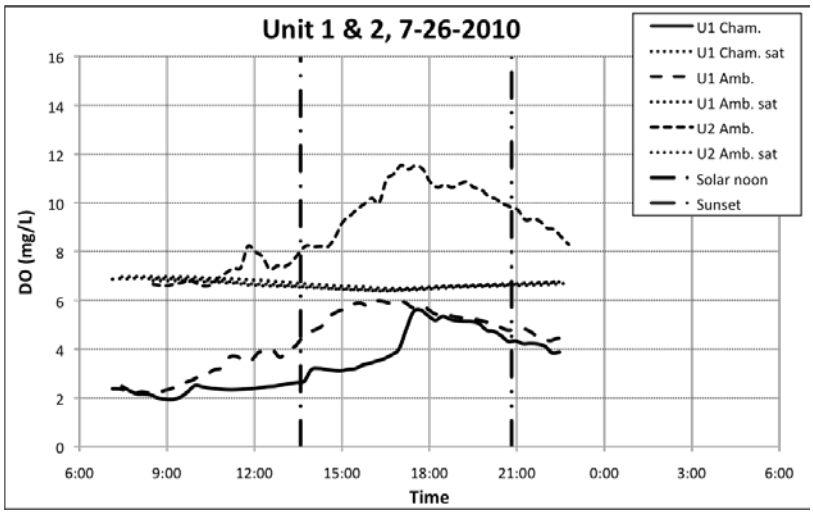
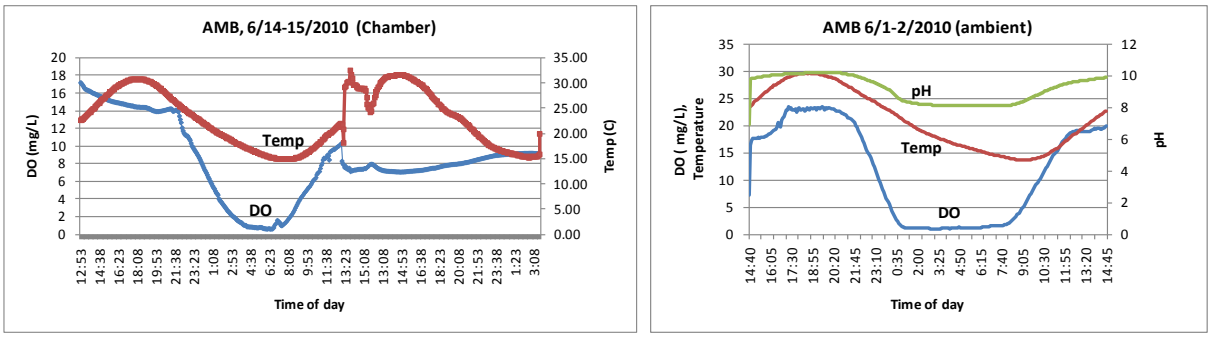


Figure E-2. Dissolved oxygen tracing for FB1 and FB2 impounds. Note the diel range is much narrower than in AMB (Figure 1).

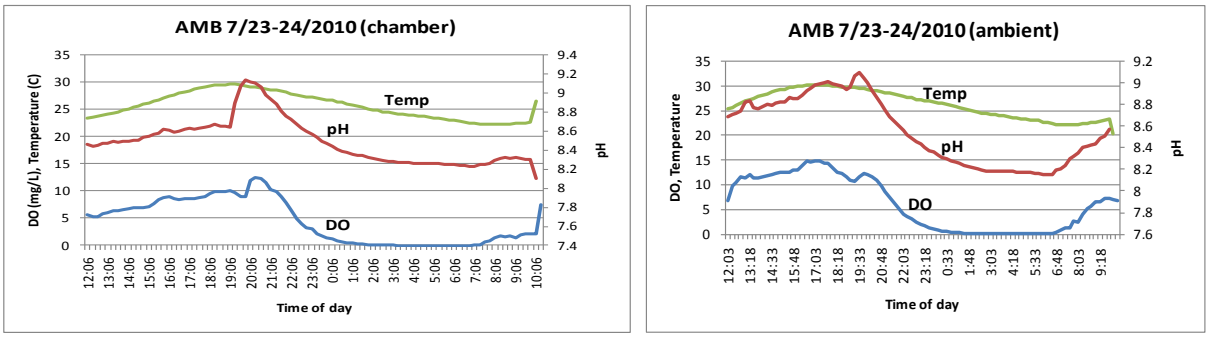
The fact that we were not able to sample during windy conditions in FB1 and FB2 created a bias in terms of ambient reaeration rates. Windy conditions increase atmospheric oxygen transfer across the air-water interface and we were only able to effectively sample in more calm/stagnant conditions. In addition, the relatively consistent and diurnally stable water temperatures of these large ponds may result in less convective dissolved oxygen and ion transport (Schmid et al., 2005). For example, it is possible that floating algal mats could contribute to daytime supersaturation by inhibiting transfer of the "excess" dissolved oxygen to the atmosphere. Similarly, when oxygen deficits developed during evening and early morning hours such mats could inhibit atmospheric oxygen transfer back to the water column.

There was also a lot of variability between ponds. For example, neither the ambient probe nor the chamber probe reached DO saturation during daylight in FB1 during the July sampling event (Figure E-5), but on the same day FB2 was supersaturated (Figure E-6). In this example, and through years of observations of similar variations, we suspect that such differences are the result of different timing of surface algal blooms among the ponds and their subsequent collapse. For example, the algal mats, consisting primarily of Chladorphora, persist for perhaps 5-10 day. At that time the filaments had become severely bleached and the cells are clearly dying – probably as a result of the sensitivity of Chladophora to the high levels of UV radiation at the surface of the ponds. This decomposing algae sinks to the bottom and undoubtedly contributes to oxygen depletion. However, the release of nutrients from this decomposition process has also been observed (see ammonia and phosphorus figures below). Finally, the elevated nutrient concentrations and deeper light penetration may be conducive to a new algal bloom, although a second algal surface mat as extensive as the first has not been observed.

a.



b.



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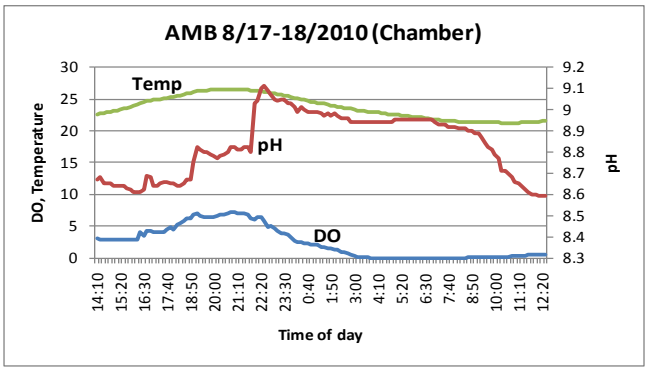

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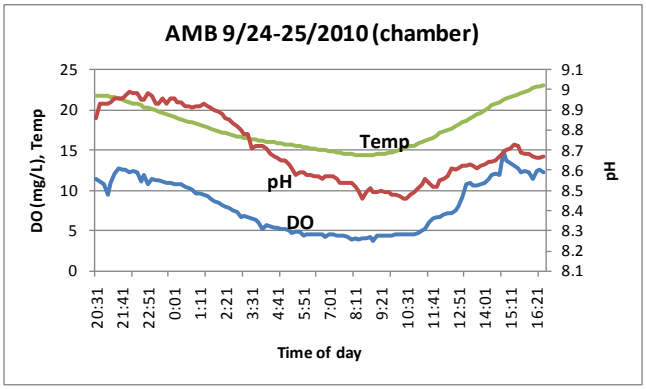


Figure E-3. Diel data sonde recordings in Ambassador W1 during 2010. (a). During the June sample, the sonde in the chamber had only temperature and DO probes. The ambient chamber also included pH. Also note data was collected from the chamber for approximately 1.5 days, during which the chamber was disturbed during water sample collection at about 1200 hours on the second day. (b). During July, all parameters were measured in the test chamber and in the ambient water column. Note similarity in values and ranges for each parameter. (c). During the August sample the sonde that was placed in the ambient water column malfunctioned. (d). During the September sample, the temperature had cooled, the DO fluctuation was not as extreme and the DO did not dip nearly as low as the previous months.

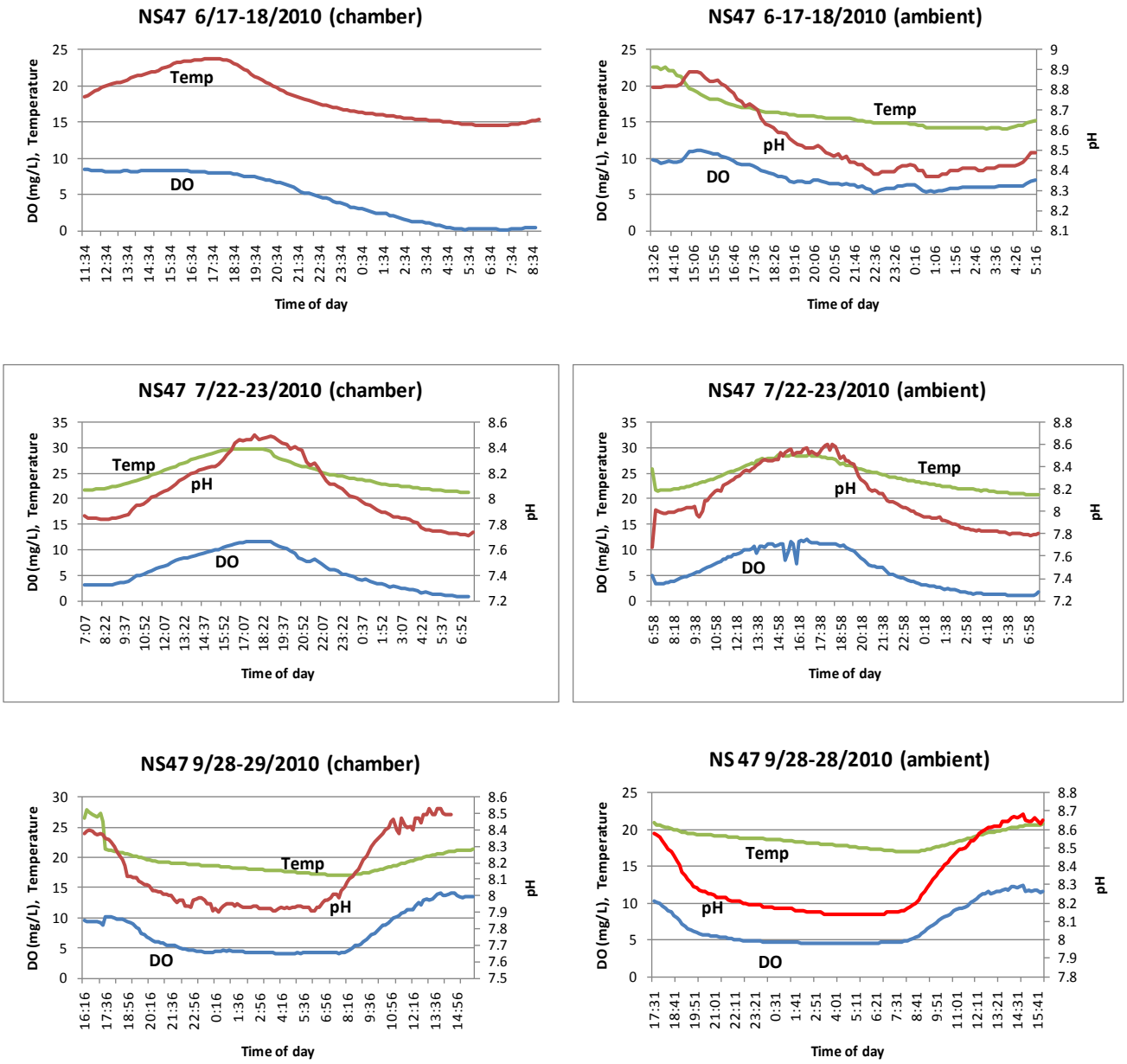


Figure E-4. Diel data sonde recordings in New State Duck Club, pond number 47 during 2010. Readings were highly consistent between the chamber and ambient environments during all sampling events. The slight increase in DO, temperature and pH during the early-morning hours in the June ambient readings suggests that the internal clock in the data sonde may have been accidently set a few hours fast.

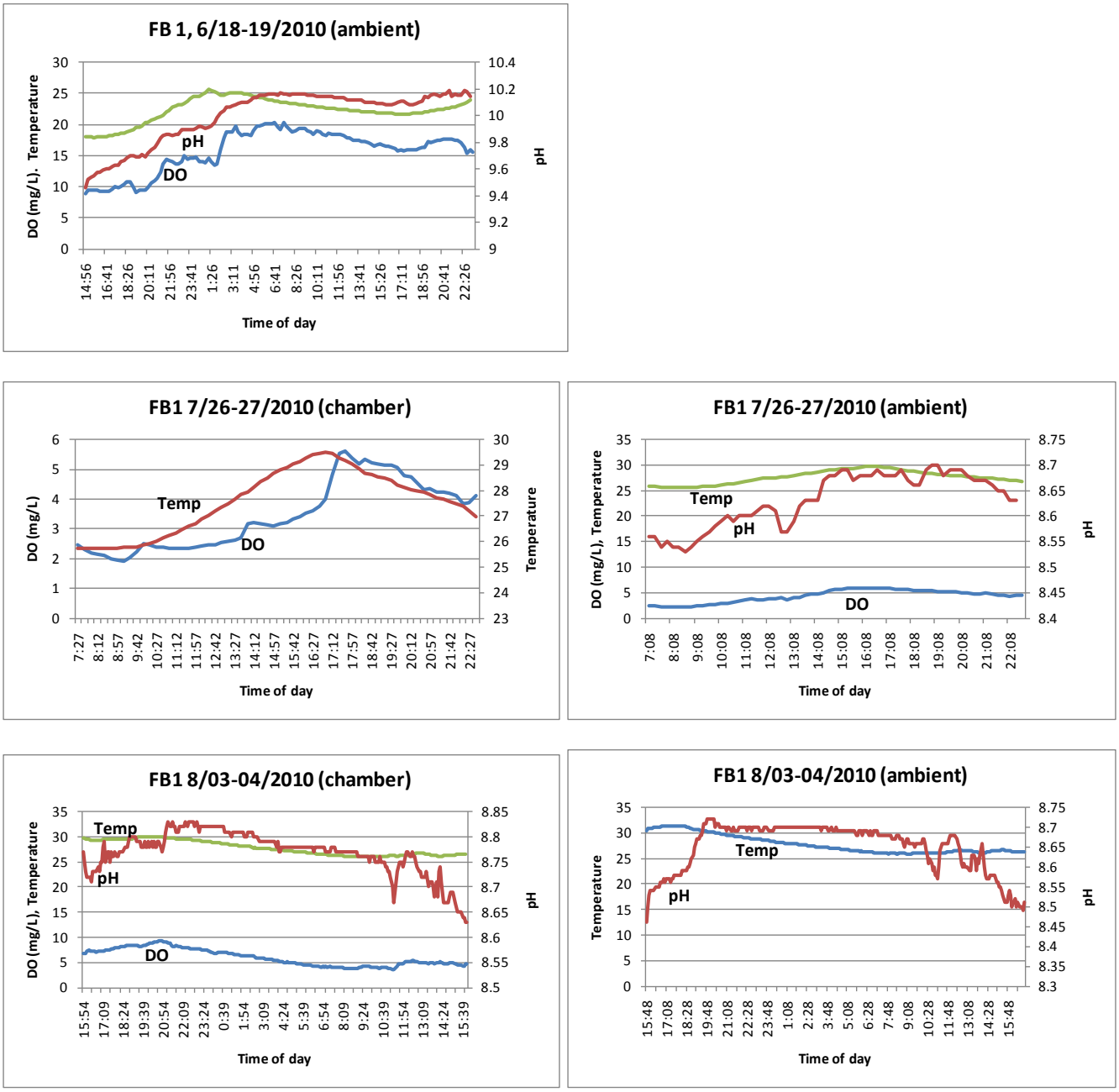
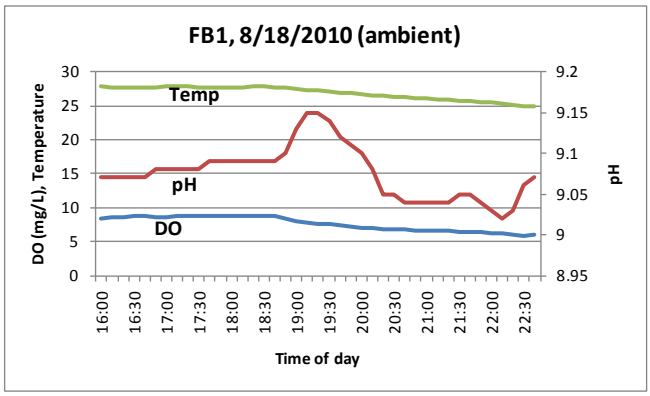
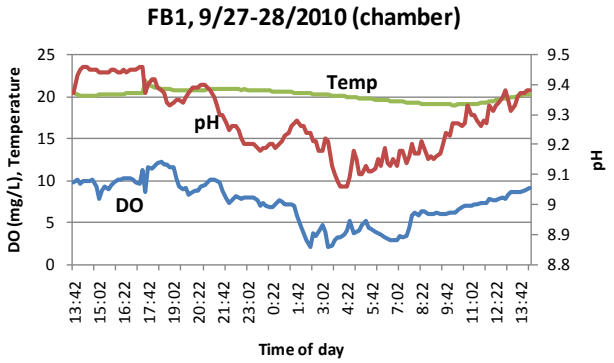


Figure E-5. Monthly diel data sonde recordings in Farmington Bay WMA Unit 1 impoundment. In most cases temperature, pH and DO followed similar patterns, including similar maximum and minimum values. In one unusual case, June 18-19, DO remained supersaturated for the entire monitoring period. See text for further details.





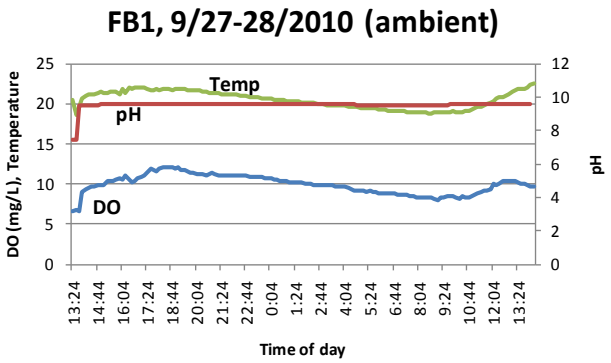
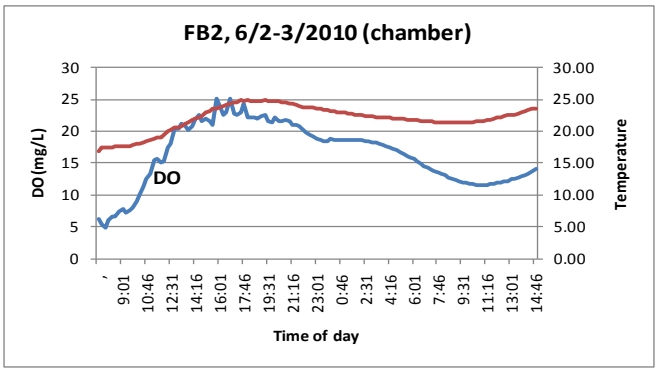


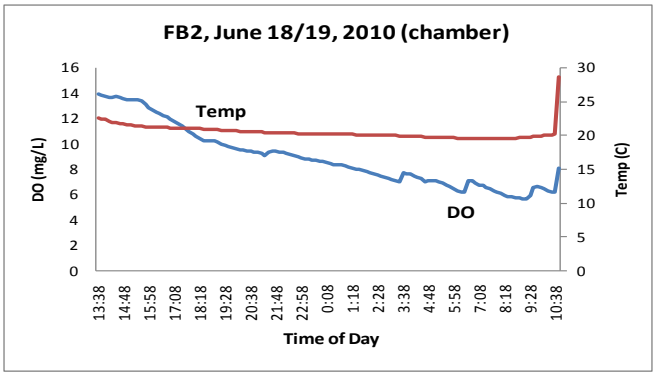
Figure E-5. Con't. Diel DO, pH and temperature plots in Farmington Bay Unit 1 during August and September. While temperatures were similar, the DO in the chamber dropped to a lower level during the early morning. Also, it appears that the pH probe was not functioning properly in the ambient test.

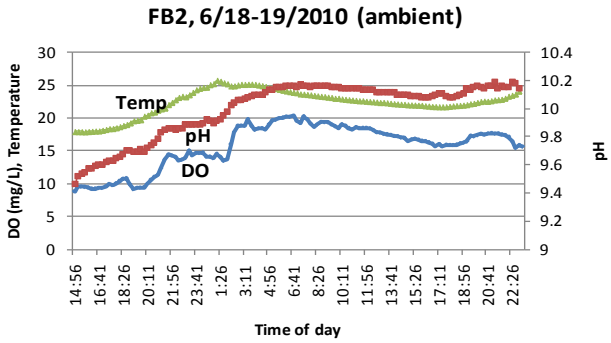
Overall, FB2 sampling results were quit similar to FB1 results. During the June sampling, DO reached nearly 200% of saturation and pH values ranged to 10. Notably, neither pH nor DO reached these high levels during the remainder of the summer.

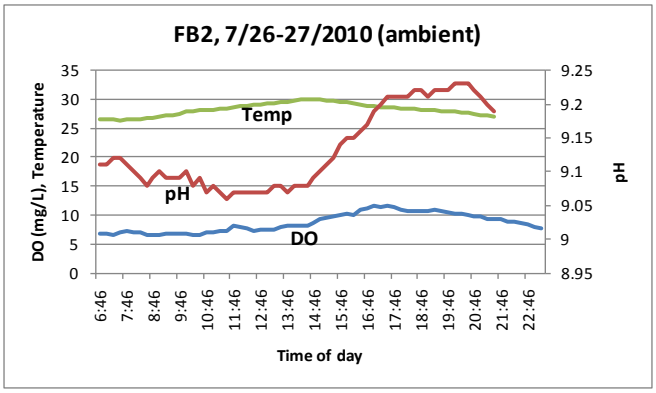
Notwithstanding these general trends, the DO and pH diel patterns in FB1 and FB2 remained somewhat unpredictable. One reason for this unpredictability is the fact that these ponds are so large. Even light winds can create sufficient turbulence to stir the entire water column, resuspend sediments, shift surface algal mats and even tip over the sampling chambers. On multiple occasions sampling needed to be abandoned due to waves breaking onto the float tube and stirring the chamber water.

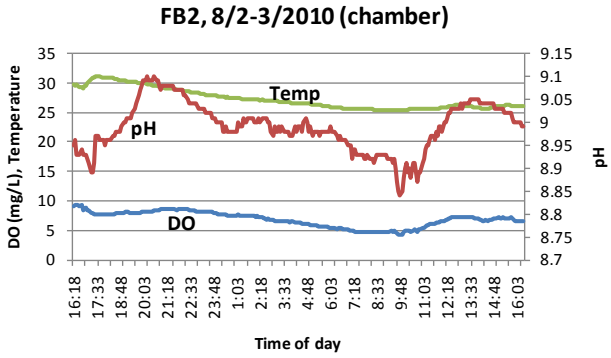
The Public Shooting Grounds impoundments, Pintail (PN) and Widgeon (WD) produced more typical sinusoidal diel DO and pH signatures (Figures E-7 and E-8). In PN the ambient data for June showed much wider fluctuations for temperature and DO than in the chamber. The chamber tended to dampen and smooth the temperature and DO. The DO only fell to about 7.1 while the ambient DO fell to near zero during the early morning hours of both sampling days. Temperature showed similar wide fluctuations. Because the chamber didn't get as warm as the ambient temperature, we suspect that the chamber was at least partially shaded by the thick growth of a mix of widgeon grass (*Ruppia maritima*) and sego pondweed (*Stuckenia pectinatus*). This would also explain why the DO did not experience as great a fluctuation either. The probe placed in the chamber was not fitted with a pH probe.

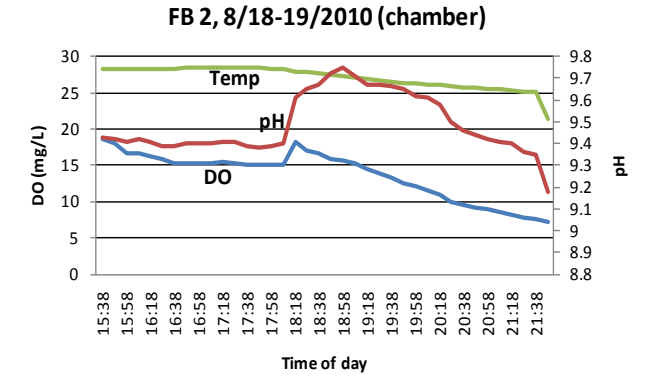












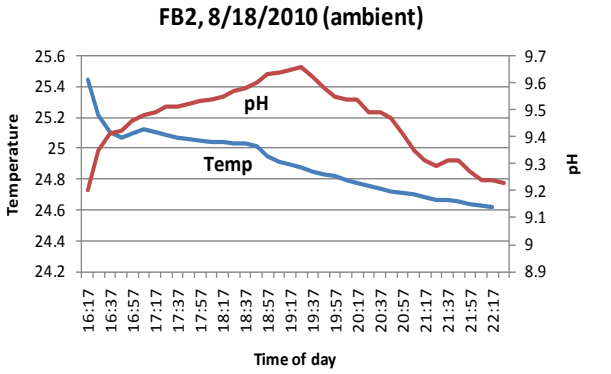


Figure E-6. Monthly data sonde recordings for Farmington Bay WMA Unit 2. Most sampling events included typical diel patterns such as afternoon peeks in DO, pH and temperature, while in other recordings such patterns were not so typical. For example, DO values in the August chamber samples started to decline prior to the pH values. The sondes malfunctioned during the September sampling.

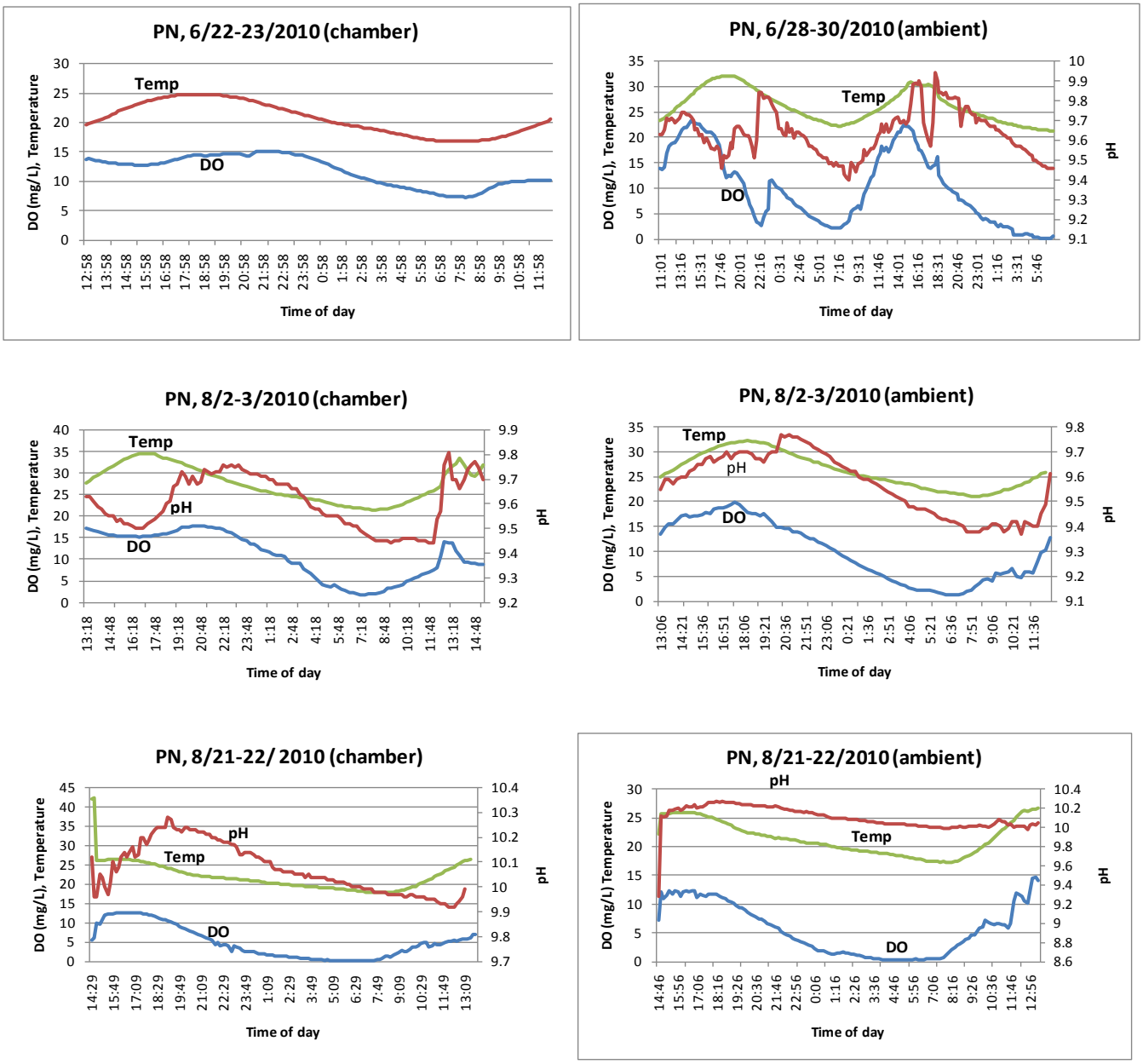


Figure E-7. Data sonde recordings for Public Shooting Grounds WMA impoundment, Pintail. Temperature, pH and DO exhibited more typical and yet much wider diel fluctuations than in our other study ponds. Unlike the larger and deeper Farmington Bay WMA ponds, these large fluctuations continued into August. The pond had dried up by early September.

The chamber in WD also appeared to smooth and dampen temperature and DO as compared to the ambient probe. Notably the ambient DO remained well about 5 mg/L during the June 22-23 sampling, yet, one week later the DO dipped to near zero during the early morning hours. WD had nearly completely dried by the end of July, precluding any additional sampling.

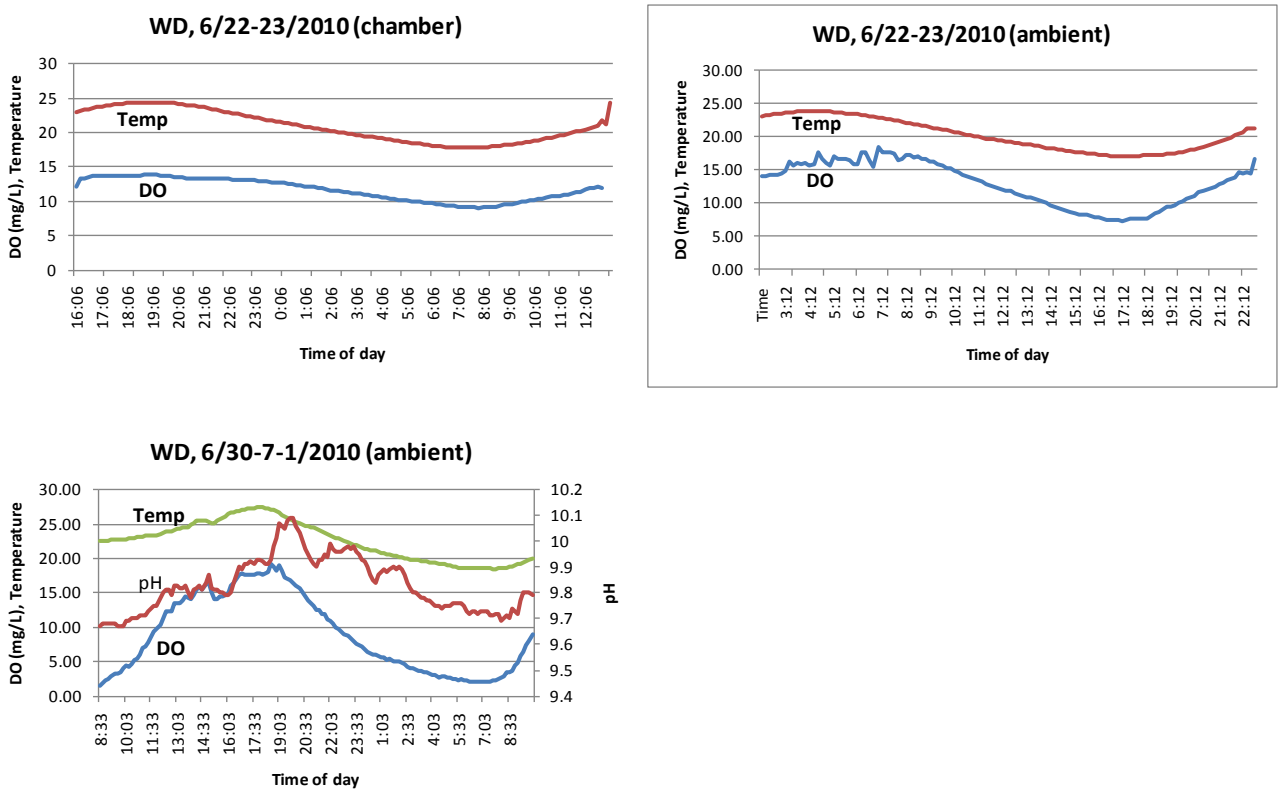


Figure E-7. Data sonde recordings for Public Shooting Grounds WMA impoundment, Widgeon. Note extremely high pH (exceeding 10.0) and very low DO (approaching 0.0 mg/L), occurred in a similar fashion as many of the Farmington Bay impoundments.

The Bear River impoundment 4C (BR) was visited during June and July. The pond had dried up by early August. Both the chamber and the ambient plots did not exhibit typical sinusoidal patterns (Figure E-8). For example, the pH data continued to decline during the midday and the DO did not show a typical response to photosynthesis. Notably, BR is larger than either FB1 or FB2 and was equally deep during the June and July sampling events. Therefore, similar divergence from typical or expected diel patterns, such as the continued decline of pH and DO, even until near 1200 hrs, appears to be common in BR as well as FB1 and FB2.

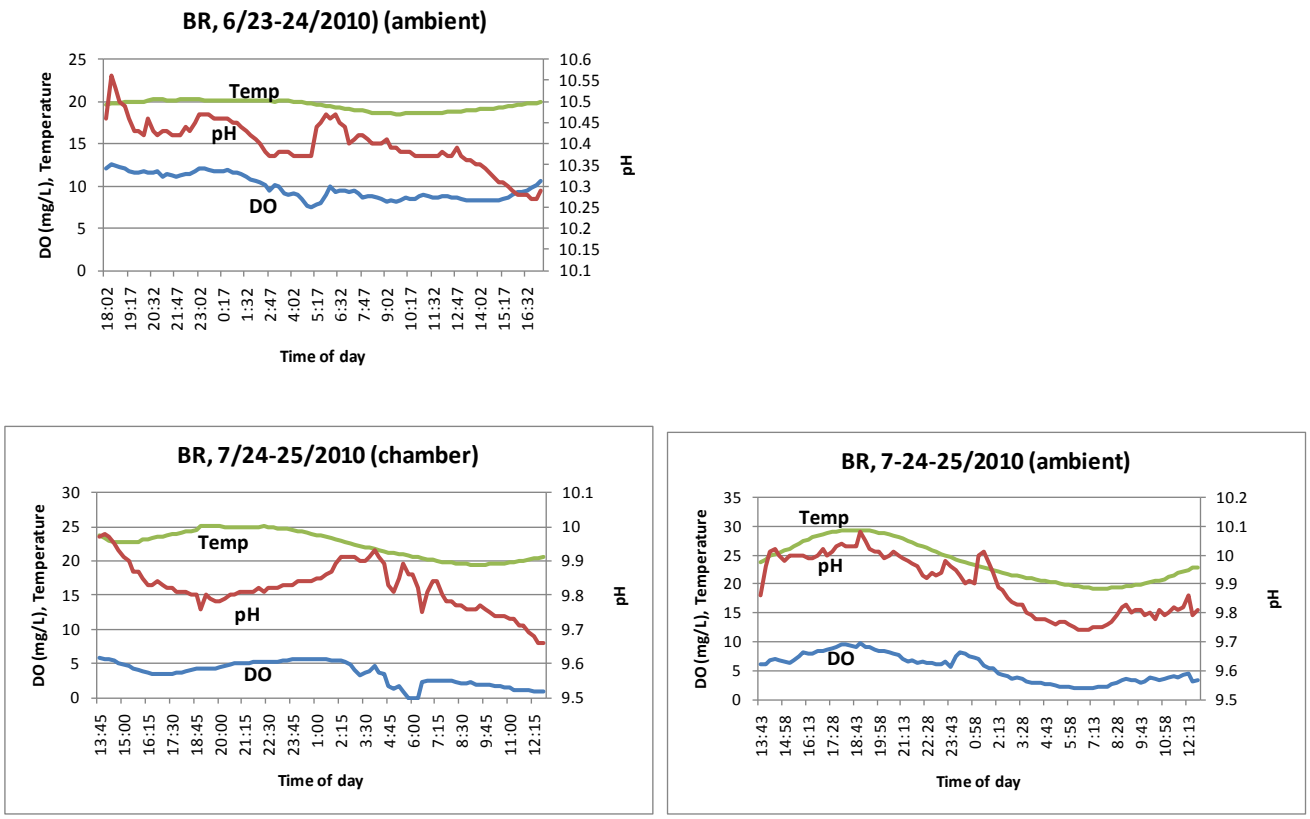


Figure E-8. Dissolved oxygen, pH and temperature plots in Bear River Migratory Bird Refuge impoundment 4-C (BR). This pond also was too dry to sample by mid August.

PN and WD have been used as "control" impoundments since the GSL impoundment studies began in 2004. This decision was made primarily on the basis of the very low water column nutrient and particularly phosphorus concentrations (see report above, Miller and Hoven 2007). Despite largely non-detectable nutrient concentrations and particularly phosphorus, these ponds continually had very high densities of SAV. This reveals one of the key characteristics of shallow, SAV-dominated ponds and impoundments; that, as with emergent marsh species, SAV obtains the great majority of nutrients from the sediments and can still reach very high plant densities. A second paradigm shift from these studies is that we had originally assumed that the somewhat greater diversity of macroinvertebrate taxa was due to the general lack of nutrients. However, as presented above and in Gray (2011), more diverse populations of macroinvertebrate populations are mostly associated with the diversity of habitat structure afforded by dense stands of SAV. The Public Shooting Grounds ponds have consistently contained such dense plant communities that recently became co-dominated by widgeon grass and Stuckenia as well as considerable patches of the macroalga, Chara. Also, the widgeon grass does not appear to senesce as early in the fall as sego pondweed. This can explain why the invertebrate community was somewhat more diverse during the late fall sampling.

The average seasonal diurnal dissolved inorganic nitrogen, sulfur, and orthophosphate concentrations are provided in Table E-3. These measurements were taken for several reasons: 1) to understand diel

Table E-3. Average diel dissolved inorganic nutrient concentrations for all study ponds for all samples collected during 2010.

		(mg/L)						
Pond		**NH_4-N**	**NO_2-N**	**NO_3-N**	**TIN**	**H_2S-S**	**SO_4-S**	**PO4-P**
Ambassador W1	avg, cham	1.02	0.11	0.63	1.76	0.07	49.8	0.91
June, July, August, September	avg, amb	0.77	0.13	0.67	1.50	0.09	45.3	0.92
New State #47	avg, cham	0.78	0.15	1.17	2.09	0.11	44.0	0.58
June, July, September	avg, amb	0.81	0.15	1.61	2.55	0.07	63.3	0.69
Unit 1, FB	avg, cham	0.82	0.07	0.13	0.91	0.12	41.3	0.54
June, August, September	avg, amb	0.79	0.07	0.12	0.76	0.12	41.0	0.56
Unit 2, FB	avg, cham	0.82	0.07	0.10	0.84	0.13	40.3	0.42
June, August, September	avg, amb	0.69	0.06	0.10	0.74	0.06	39.8	0.28
Bear River 4C	avg, cham	0.56		0.10		0.09	12.2	
June, July	avg, amb	0.55		0.09		0.07	12.8	
Pintail	avg, cham	0.83	0.06	0.08	0.99	0.07	59.7	0.05
June, July, August, September	avg, amb	0.89	0.06	0.07	0.96	0.08	58.0	0.06
Widgen	avg, cham	0.64		0.08	0.67	0.13	40.3	0.05
June	avg, amb	0.57		0.08		0.05	36.5	0.03

variability and importance of missed information when data are collected by grab samples alone, for example during mid-day; 2) to determine if the observed patterns in concentration are related to diel

patterns of DO and pH and whether hypoxic conditions the most conducive to sediment release of P, ammonia, sulfide and various toxic metals (e.g. see Mortimer, 1941, 1942, Hutchinson 1957); and 3) to determine whether water column concentrations of metals, sulfide or ammonia are related to sediment or pore water concentrations. The diel variability of ammonia, P, and sulfide are discussed below. The relationship between sediment and pore water concentrations of metals and sulfide to water column concentrations are reported and discussed by Johnson et al. (2011). It should be noted that all species are reported as mg/L nitrogen as N, Sulfur as S and phosphorus as P. Table E-3 provides a general characterization of these dynamic and complex impoundments and allows comparison between the ambient pond water and chamber water environments.

As noted above, the ambient pH in AMB, FB1, FB2, PN, WD and BR during the June reached very near or exceeded 10.0 shortly before sunset. As an example, a pH of 10 and a temperature of 30° C, most of the measured chamber and ambient total ammonia concentrations of 1.2 and 0.5 respectively, would be in the toxic unionized form. Accordingly, using EPA's equation to calculate Utah's warm water (3B, 3C and 3D) acute standard for ammonia:

$$mg/l\ as\ N\ (Acute) = (0.411/(1+10^{7.204-pH})) + (58.4/(1+10^{pH-7.204}))$$

ammonia commonly exceeded the warmwater standard. For example, at a pH of 9, the warmwater ammonia standard = 1.3 mg/L, while at a pH of 10 the warmwater standard = 0.50 mg/L. Further, if the ammonia standard were calculated according to EPA's criteria document for somewhat warmer water (>10C) a factor for temperature is included. The acute criteria equation (where no mussels are present) is:

$$CMC = 0.826 * \left(\frac{0.0489}{1 + 10^{7.204-pH}} + \frac{6.95}{1 + 10^{pH-7.204}}\right) * MIN\left(12.09, 6.018 * 10^{0.036*(25-T)}\right)$$

Where *CMC* is the "Criterion Maximum Concentration",

MIN (12.09, 6.018 * $10^{\wedge 0.036*(25-T)}$) = the lower value derived by using either 12.09 or 6.018 as the multiplier and T = °C.

Examples of the acute ammonia criteria values with different pH values and three temperature regimes (either 10, 20 or 30 C), is presented in Table E-4. Similarly, Figure E-9 illustrates how sensitive both the acute and chronic criteria are to warm temperature and elevated pH. Using either equation, and particularly for the EPA (2009) equation, it is likely that these concentrations of ammonia at these measured pH values violated the acute ammonia criterion for some period of time every day in all of our study ponds, including the reference ponds, PN and WD. The only exception may be perhaps NS47. Typically, where and when the impoundments exceed about 0.5 mg/L ammonia N and pH values exceed about 9.5 and the temperature exceeds about 25C, the acute ammonia criterion would be violated. This is undoubtedly one of the driving stressors that dictate the macroinvertebrate species composition in these and many other wetlands. Ultimately, it is also possible that additional stressors, including high pH, temperature, H_2S, ammonia and several metals combine simultaneously to exacerbate stressful conditions.

Table E-4. Acute ammonia criteria or Criterion Maximum Concentration (CMC) in relation to pH and seasonal temperature values commonly present in Great Salt Lake impounded wetlands.

pH	temp (°C)	CMC (mg-N/L)	temp (°C)	CMC (mg-N/L)	temp (°C)	CMC (mg-N/L)
10	30	0.20	20	0.45	10	1.03
9.5	30	0.27	20	0.63	10	1.44
9	30	0.52	20	1.19	10	2.72
8.5	30	1.25	20	2.87	10	6.57
8	30	3.29	20	7.53	10	17.25
7.5	30	7.77	20	17.81	10	40.80
7	30	14.11	20	32.32	10	74.03

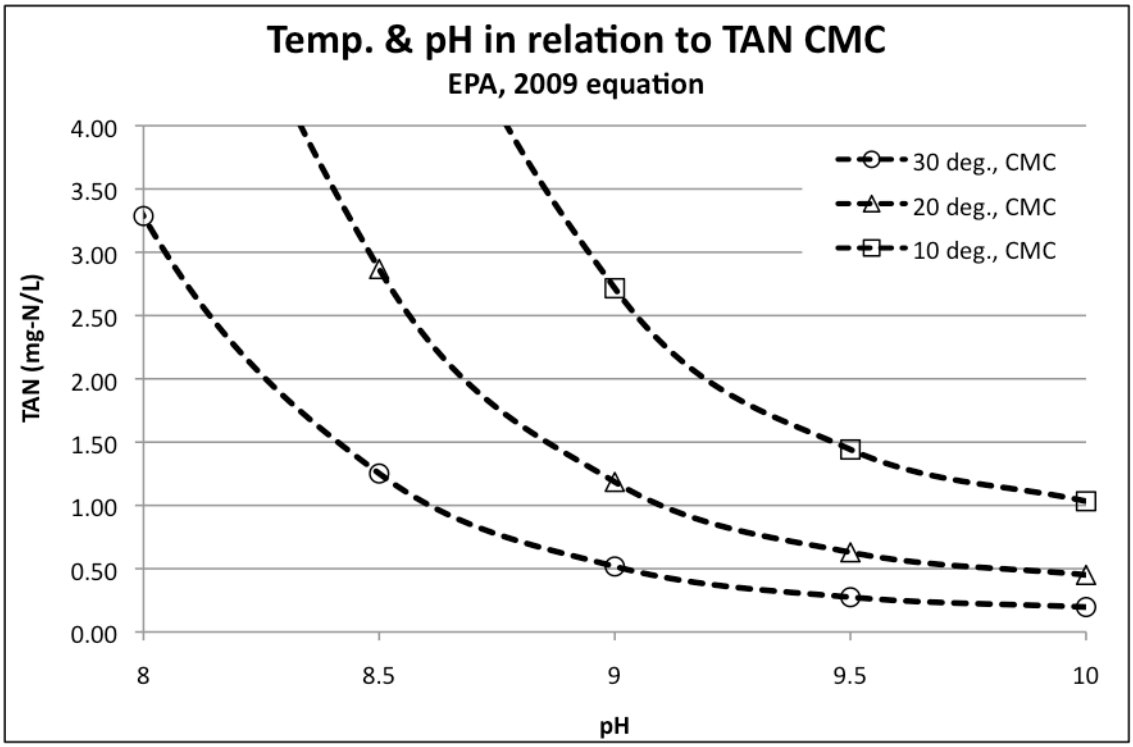


Figure E-9. Total ammonia nitrogen (TAN) criterion maximum concentration (CMC) in relation to pH and temperature.

These stressors have resulted in macroinvertebrate community structure that is comprised of relatively few taxa that are typically characterized as tolerant (i.e. see Hilsenhoff 1987 or Bode et al. 1991), of these conditions. Yet, these conditions are typical of Great Salt Lake impounded wetlands. Notably, such combined water quality and biological data for wetlands are scarce in the literature. However, similar taxa lists, accompanied by low DO (periodically reaching 0 mg/L), and high pH (periodically > 9) have been found in wetlands in other states that have even been designated as least-impacted reference wetlands (e.g. Heimann and Femmer 1998).

In NS47 ammonia behaved similar to that in AMB (Figures E-12 and E-13). Values commonly exceeded 1 mg/L. Notably however, pH values in NS47 did not reach the high values that occurred in AMB (Figures E-3 and E-4). Consequently, the ammonia acute criteria value was not exceeded – perhaps providing a somewhat less stressful concentration of ammonia. Within Farmington Bay WMA, FB1 had similar ammonia concentrations as in AMB and NS47. Ammonia concentrations in the reference impoundments PN and WD and in BR were also very similar to the Farmington Bay impoundments.

The Ambassador and Newstate impoundments tended to have the highest concentrations of total inorganic nitrogen (TIN), which is the sum of ammonia-N, nitrite-N, and nitrate-N (Table E-3). Elevated nitrate concentrations were the primary contributor to TIN in these ponds. Because ammonia concentrations were

very dynamic it is possible that the high nitrate concentrations are the result of an active nitrifying bacterial population capable of oxidizing ammonia released from the organic-rich sediments. In addition however, Jordan River nitrate concentrations near its terminus or in the Surplus Canal (a major Jordan River diversion at 2100 S) may have added to the nitrate loading. For example, nitrate in the Jordan River and the surplus canal typically range from about 1 to about 7 mg/L (mean for Burnham Dam = 4.0 and for the Surplus Canal = 4.1 mg/L; DWQ STORET data, 2005-2009) – concentrations much higher than in the Farmington Bay impoundments that receive this water (summer monthly average concentrations ranging from 0.1 – 1.17 mg/L). With the exception of NS47, the impoundment concentrations of nitrate are typically 1 to 2 orders of magnitude less than the source water. Comparatively, ammonia concentrations in the lower Jordan River range from about 0.1 to about 0.7 mg/L (e.g. mean ammonia at Center St = 0.31, below South Davis Sewer District plant (in Legacy Nature Preserve) = 0.17 and at Burnham Dam the average ammonia concentration = 0.38; STORET data, 2005-2009). These source-water values are much lower (1/2 to 1/3) that of the impoundment ammonia concentrations. These data suggest that there are two separate, but linked processes occurring: 1) the rapid reduction in nitrate concentrations in the ponds, compared to the source water, suggests that denitrification is readily occurring. Assimilation by plants and bacteria may also be occurring, but because ammonia (the most biologically preferred form of nitrogen), is in much higher concentrations than nitrate, actual biological uptake of nitrate is likely of minimal importance; 2) Similarly, because ammonia is so readily assimilated by algae and bacteria (Hutchinson 1957), its presence in high concentrations (typically 0.5 to 1.5 mg/L among our study impoundments; Figures E-10, E-13, E-16, E-19, E-22 and E-26) – and particularly in comparison to nitrate - suggests that ammonia is continually released from the sediments at rates greater than nitrification. Because of this dynamic nature and importance of ammonia and nitrate in aquatic ecosystems, both as a nutrient and a toxicant, additional research should be performed to elucidate this relationship.

Interestingly, FB1 and FB2 tended to have TIN concentrations lower than the nearby Ambassador and Newstate ponds. The lower TIN in these impoundments is the result of lower concentrations of nitrate relative to the other N species. Following the same logic as expressed above, it is possible that these ponds were experiencing even higher rates of denitrification, the reduction of nitrate to N_2 gas or the fact that FB1 and FB2 are much larger in size also provides greater residence time for biological processes to occur. Ammonia concentrations in FB1 and FB2 were comparable to NS47 and AMB, but nitrate-N concentrations were much lower and resembled those found in the reference wetlands, PN and WD. These impoundments, along with BR tended to have the majority of the dissolved nitrogen as ammonia-N. The high ammonia concentrations in these ponds, not being linked to source waters with high ammonia, offers further support for the conclusion that ammonia is being released from the degradation of organic material within the sediments.

Diel average sulfate-S concentrations where typically within the range of 40 mg/L, except for PN, where higher concentrations were routinely observed and at the Bear River site that had characteristically low, (12 mg/L sulfate-S). As noted above Public Shooting Grounds impoundments are primarily supplied by saline springs and secondarily by irrigation return flows while BR source water is exclusively the Bear River, whose headwaters are in the Uinta Mountains. Sulfate concentrations increased in all ponds, as did TDS in general, as diminishing inflows and evaporation became a dominant hydrologic feature of these ponds.

As with ammonia, sulfide concentrations in these impoundments were likely also consistently exceeding its freshwater criterion. Although dissolved sulfide concentrations were below detection limits of the Chemetrics field instruments (0.2 mg/L), logical measurement results were provided by the Chemetrics photometer. Further, most humans are capable of detecting the odor of H_2S, gas (the “rotten egg” smell) at concentrations near 0.5 ug/L (parts per billion) or lower. The odor of hydrogen sulfide gas was readily detected when sediments

were disturbed and periodically throughout the evening and early morning hours during our sampling events, providing evidence for the presence of sulfide. Therefore, all results are reported in the tables and graphs. Notwithstanding these frequent limitations, spikes in H_2S concentrations were observed in most of the ponds – suggesting that the spikes are real and values that were recorded below the described detection limit are useful. Notably, the aquatic life standard for sulfide is 4 ug/L. Therefore, any concentration reported near the detection limit reflects a criterion exceedence by about an order of magnitude.

Understanding whether ammonia concentrations or fluxes or those of sulfides or phosphorus occurred in a predictable diel pattern were also important to document. Figures E10 through E26 show the results of field measurements performed at approximate 3-hour intervals for a 24-hour sampling event. Notably, although there were considerable fluctuations in all three constituents, such variations did not follow a predictable diel pattern as with DO, pH or temperature (Figures E-10 through E-26).

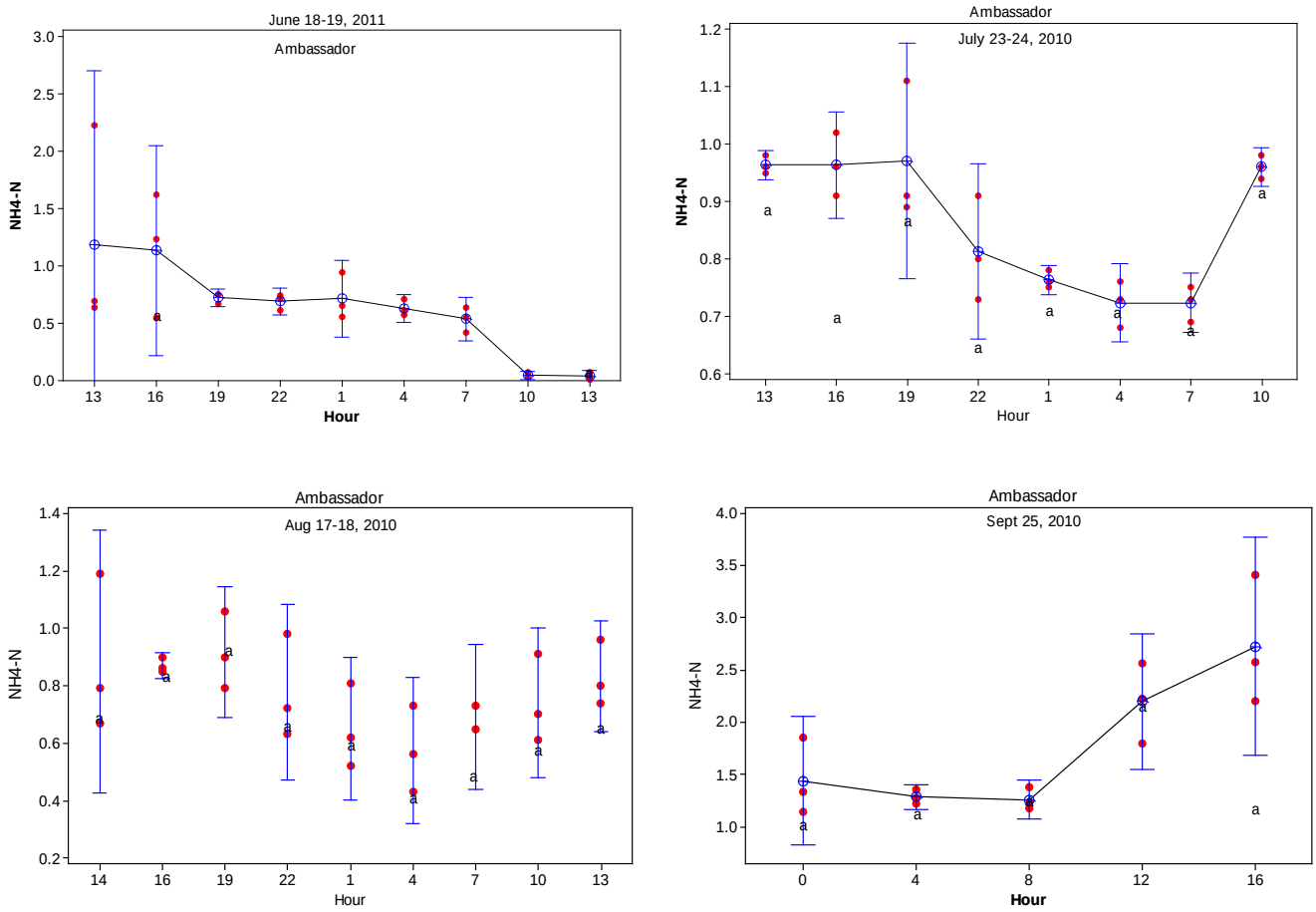


Figure E-10. Monthly measurements of ammonia in AMB. Red dots are measured values within each of three replicate chambers. Error bars indicate the 90% CI. The letter (a) indicates the ambient water column concentration measured at the same time.

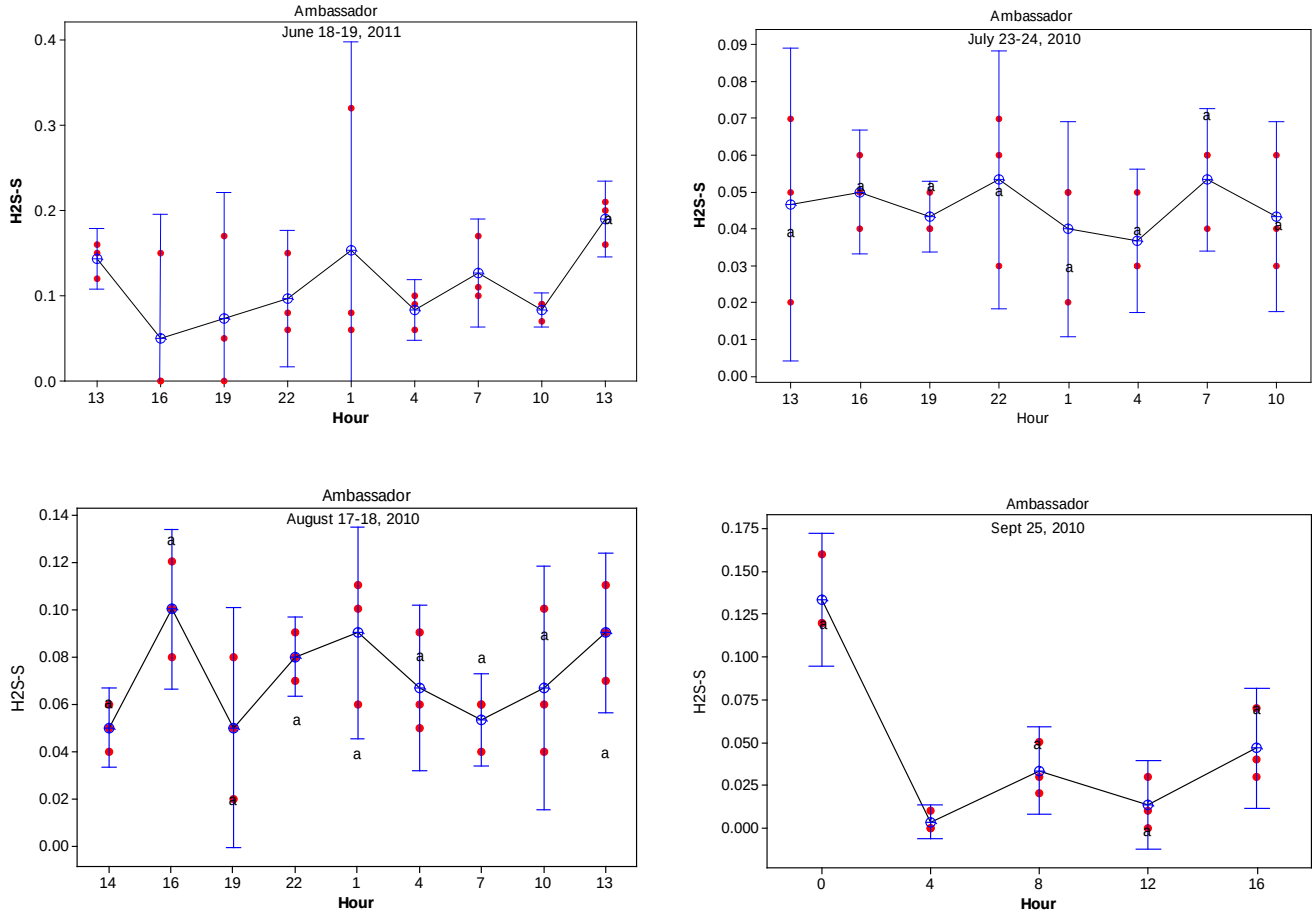


Figure E-11. Monthly measurements of H_2S in AMB. The instrument detection limit was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

We also tested the hypothesis that the phosphate ion would exhibit a diel pattern of equal proportion to, but opposite that of the diel DO pattern. This is because the diel pattern of DO reached or nearly reached 0.0 mg/L, and the well-known association and dissociation of phosphate^{---} ion to Fe^{+++} and Fe^{++}, respectively, that occurs as DO approaches 0.0 mg/L. However, this was not the case. Although phosphate exhibited a large amount of variability, these patterns were random with respect to the time of day (Figure E-12). Many of the highest

concentrations actually occurred during daylight hours, when maximum photosynthesis/primary production and elevated DO are occurring. The presence of such high concentrations of ortho-phosphate is uncommon and generally very transient in nature because, as with ammonia, this is the most preferred form of phosphorus and therefore is readily assimilated by phytoplankton and bacteria. This is notable on three counts, 1) the abundance of water column ortho-P suggests relatively little nutrient uptake/removal occurring by the phytoplankton, bacteria or the SAV; 2) this supports the findings of (Carignan and Kalff 1980 and others) where it was determined that both emergent and submergent macrophytes obtain the great majority of their nutrients from the sediments. In other words, lush growth of SAV occurs regardless of the very low/non-detectable phosphate concentrations in Public Shooting Grounds impoundments or the elevated phosphate concentrations of most Farmington Bay impoundments; In addition, several chlorophyll a samples (the best surrogate for phytoplankton density) throughout these wetlands have indicated that relatively low concentrations exist (circa 5-10 ug/L; : Utah DWQ data). This would typically be unexpected with regard to the high ammonia and ortho-P concentrations- that could easily support many times that amount of Chlorophyll a; and 3) Therefore, the elevated concentrations of these actively assimilated forms of N and P suggest that the sources of both ammonia and phosphate are continuous and are likely the result of ongoing decomposition of the organic material such as from the previous year's production. This conclusion is supported by the fact that the average (n=11) Jordan River phosphate-P concentration at in the Surplus Canal (Upstream from the Ambassador diversion) was 0.45 mg/L during 2010 (Jordan River Water Quality Council data; analyzed at Central Valley Water Reclamation Facility). Phosphate-P concentrations were generally much higher than this with average values throughout the summer at 1.5-3 times higher than the river concentrations. This provides additional evidence of the importance of sediment as the source for phosphorus as well as ammonia in the nutrient budget for these ponds.

Phosphate concentrations were similar among all of the ponds except the two reference ponds, PN and WD. Phosphate in these two impoundments was commonly below detection limits and never exceeded about 0.2 mg/L (Figures E-23 and E-25). It was the initial measurements of low P concentrations in PN and WD and prompted us to choose these as reference ponds when this overall study was initiated in 2004.

Concentrations of ammonia and H_2S in BR were similar to the Farmington Bay impoundments (Figures E-26 and E-11, E-14 and E-17). The one sampling event for phosphate in BR indicated that phosphate was slightly less than those in Farmington Bay impoundments by higher than the Public Shooting Grounds ponds, PN and WD. The exceptionally high ambient ammonia value (2.39 mg/L) as well as the relatively high chamber values in BR in August (measured at 0400 hours), demonstrates the dynamic characteristics of ammonia release from the sediments. Notably, these elevated concentrations were short lived as later measurements showed a substantial decline of ammonia in both the chambers and the ambient water column. Concentrations were quite consistent throughout the summer season. The lowest concentrations generally occurred during the evening hours which does not support the notionthat the greatest ammonia production occurs when DO is minimal. Because of the limitation of water availability, the Utah Division of Wildlife Resources could not maintain water in WD after June.

Concerns regarding the "build-up" of gases at the top of the chamber prompted us to provide openings at the top of each chamber to provide some additional mixing with ambient water for all subsequent sampling events. Again, the purpose of using the chambers in this study was to prevent disturbance of the sediments while samples were being collected and otherwise emulate ambient conditions and sediment/water interactions. Measurements of rates of sediment flux of these compounds will be integrated in future studies.

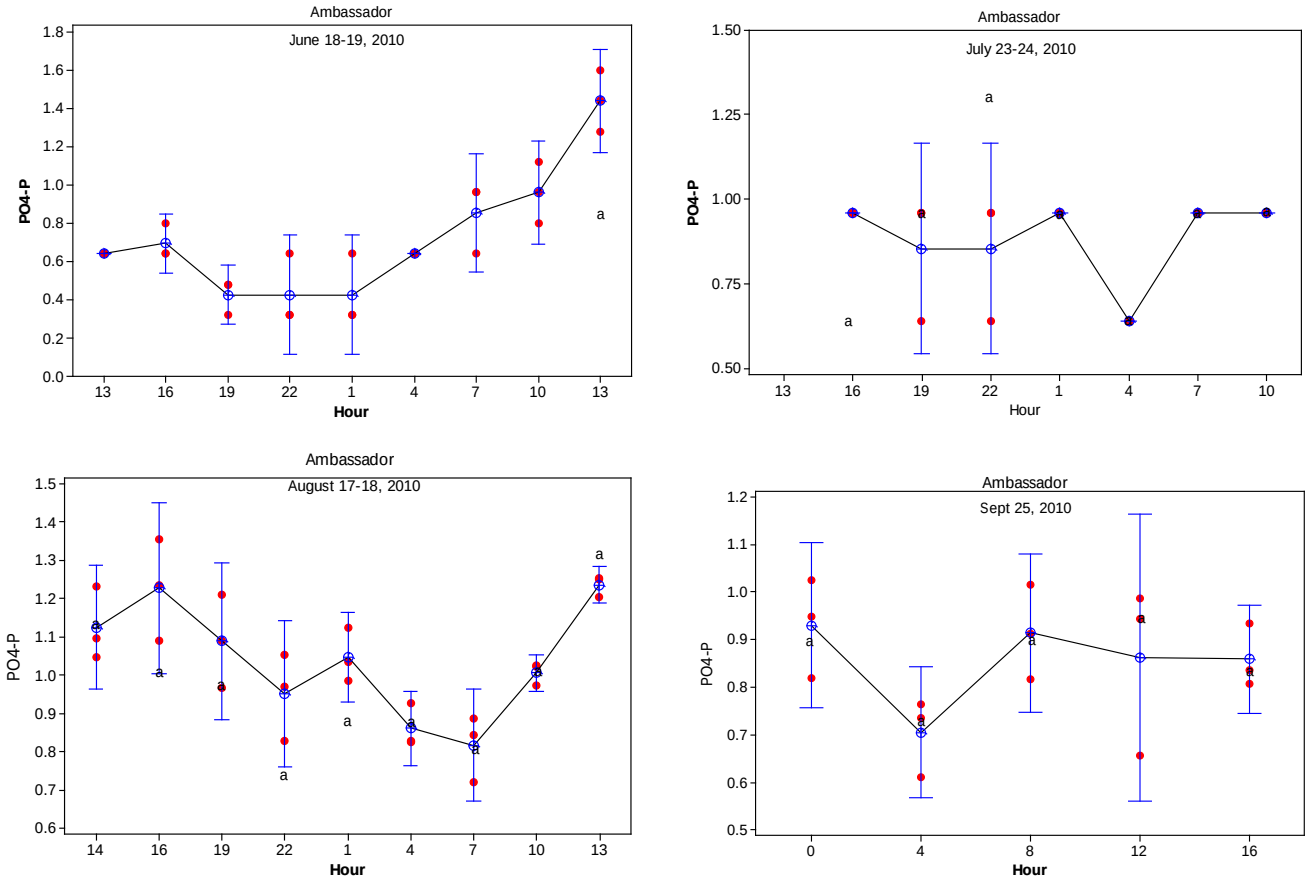


Figure E-12. Monthly measurements of phosphate in AMB. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

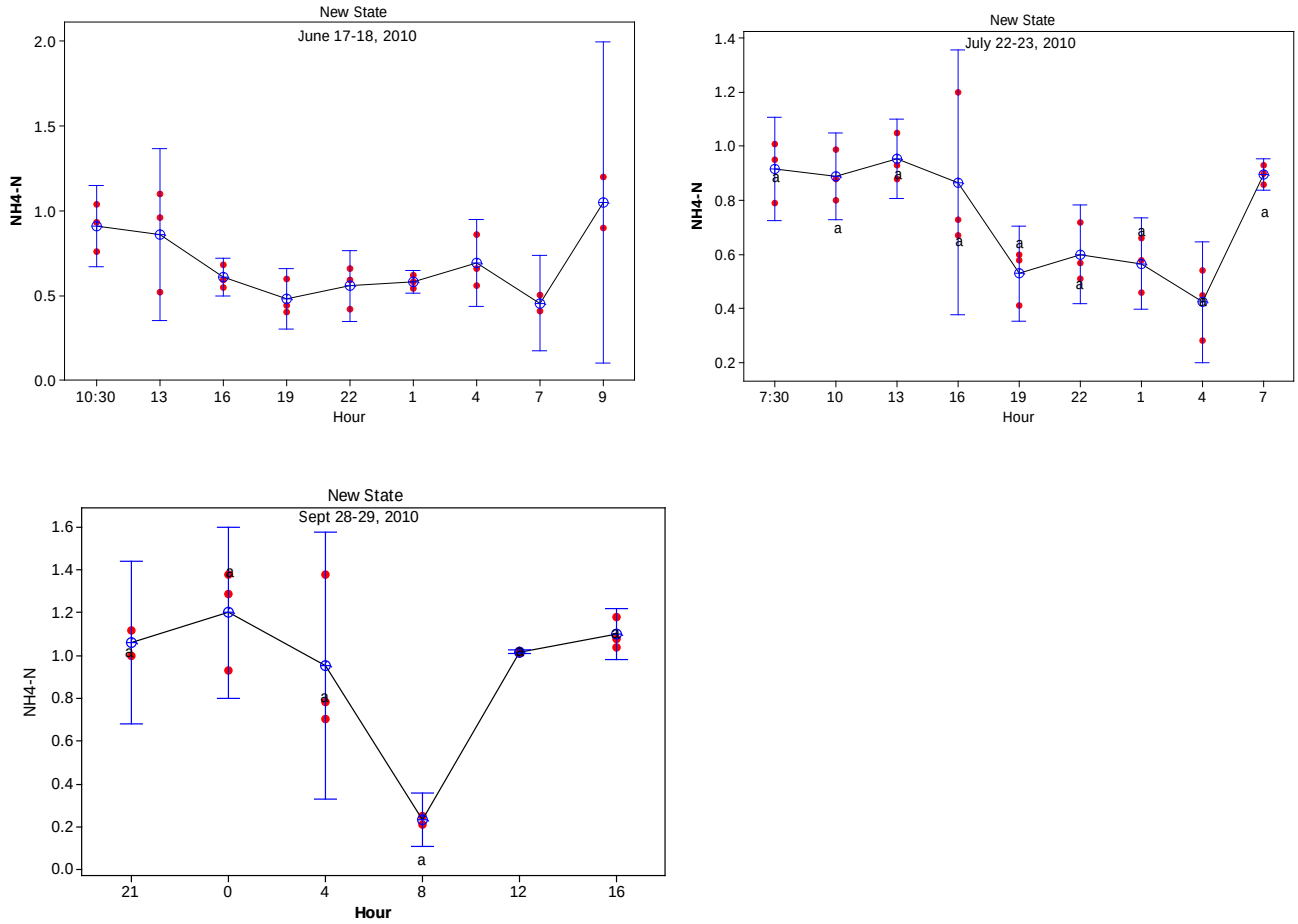


Figure E-13. Monthly measurements of ammonia in NS47. Red dots are measured values within each of three replicate chambers. Error bars indicate the 90% CI. The letter (a) indicates the ambient water column concentration measured at the same time.

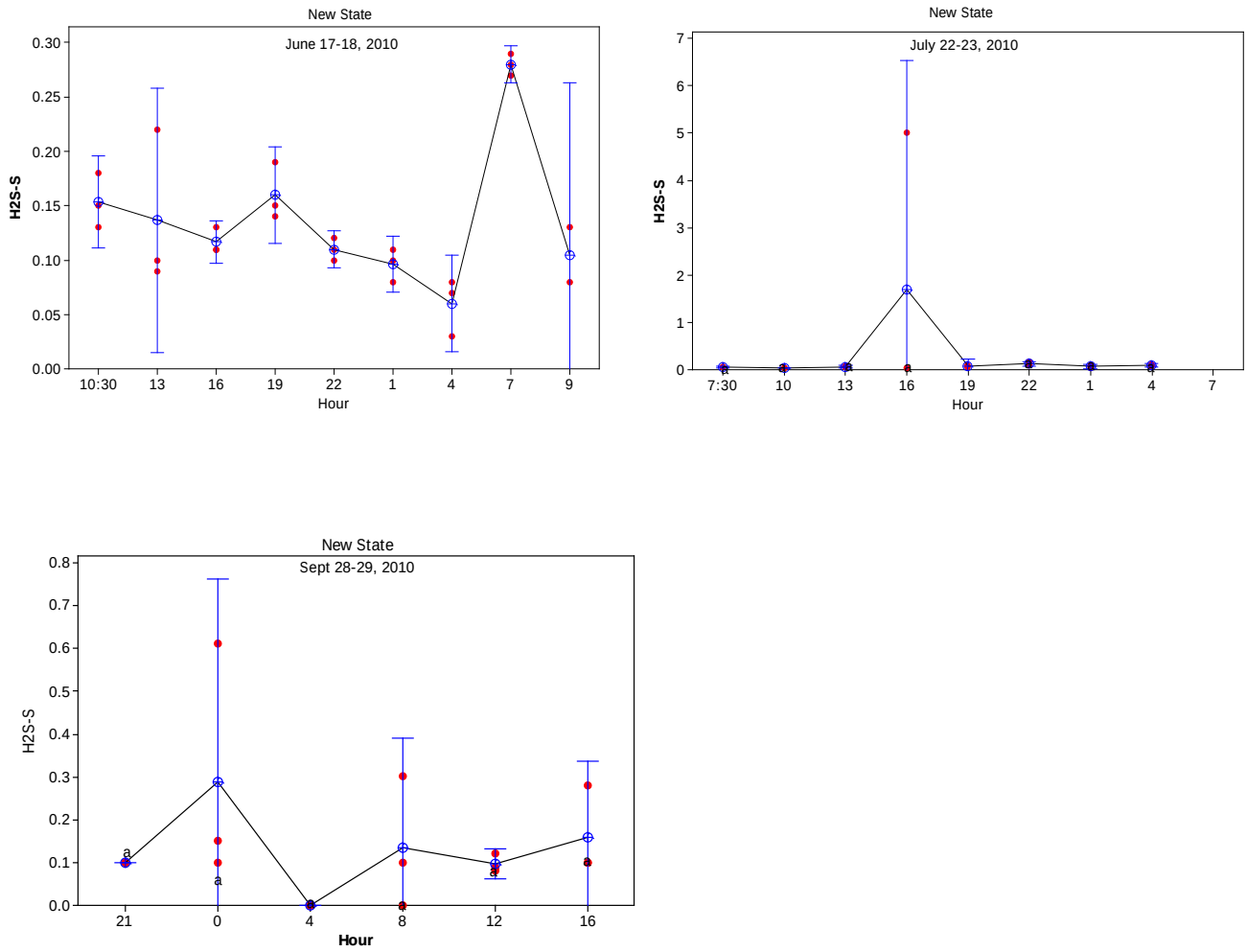


Figure E-14. Monthly measurements of H_2S in NS47. Instrument detection limit was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

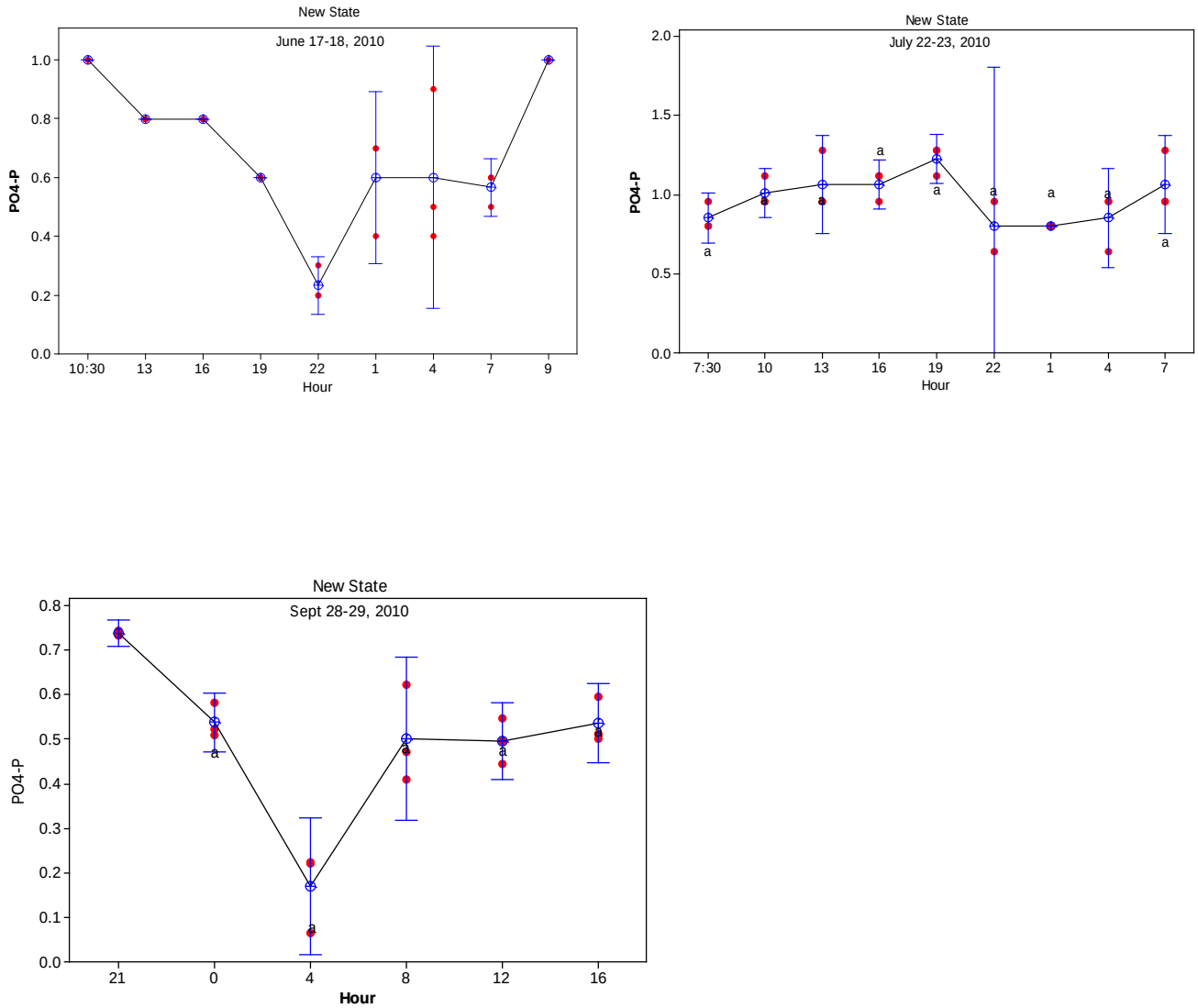


Figure E-15. Monthly measurements of phosphate in NS47. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

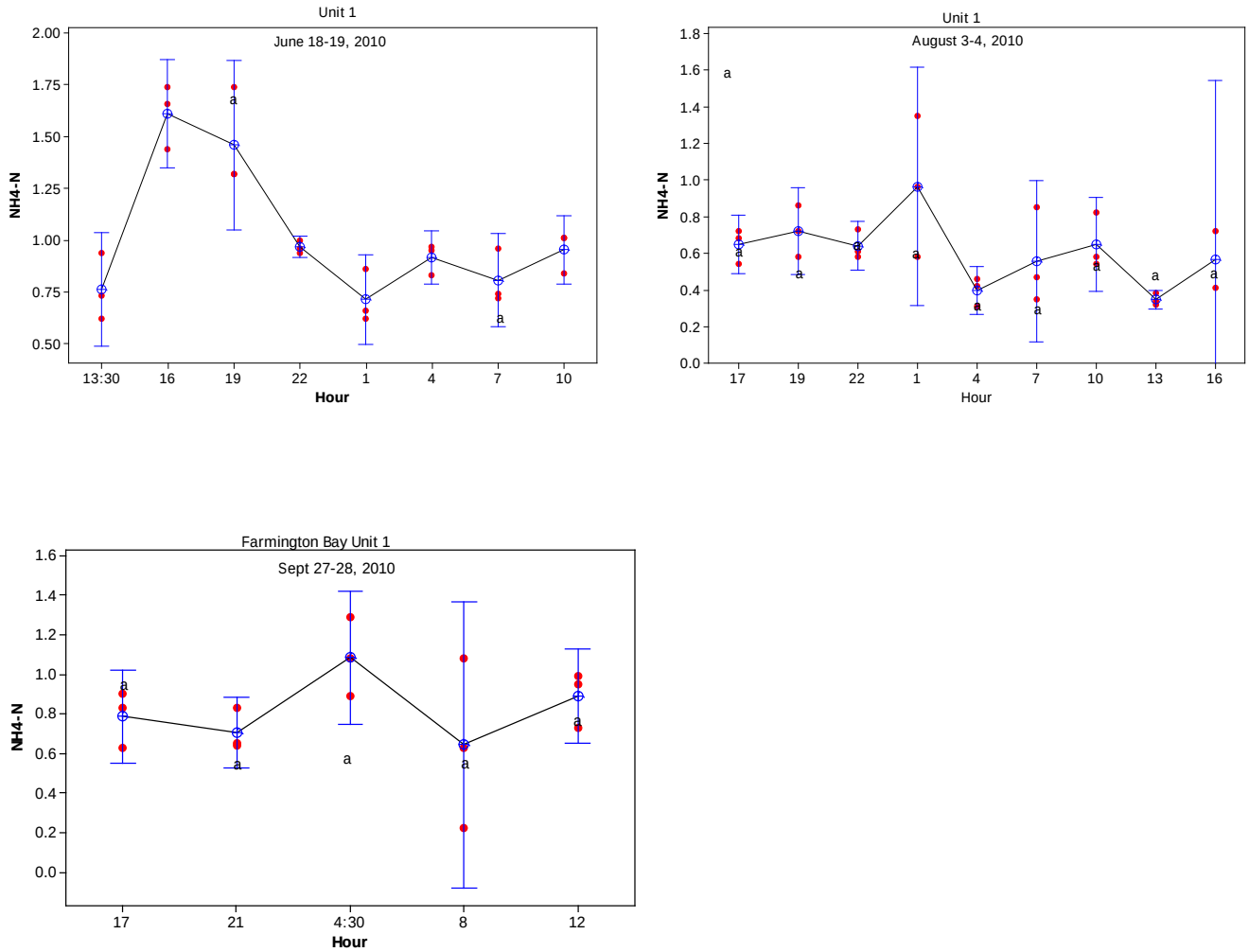


Figure E-16. Monthly measurements of ammonia in FB1. Red dots are measured values within each of three replicate chambers. Error bars indicate the 90% CI. The letter (a) indicates the ambient water column concentration measured at the same time.

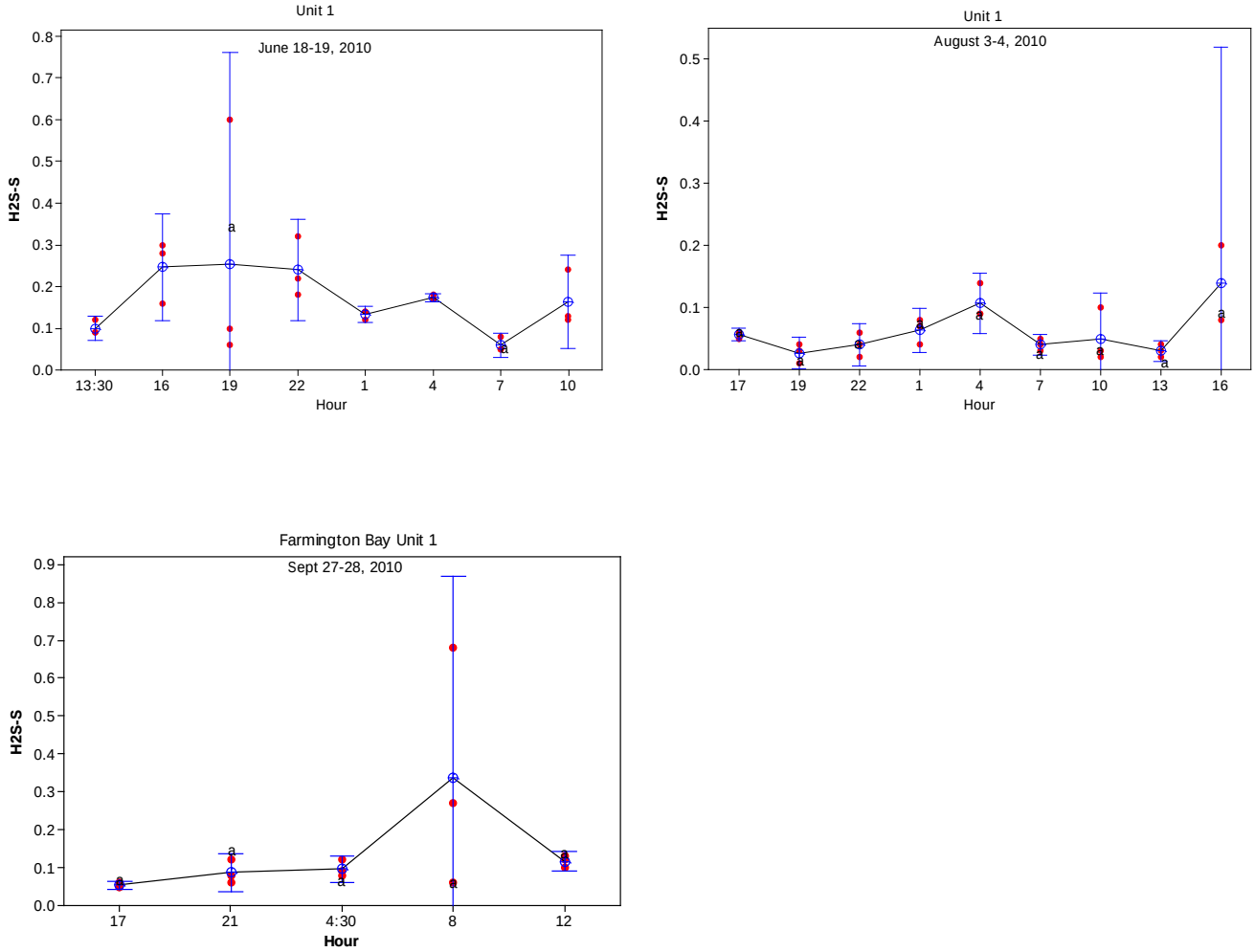


Figure E-17. Monthly measurements of H_2S in FB1. Instrument detection limit was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

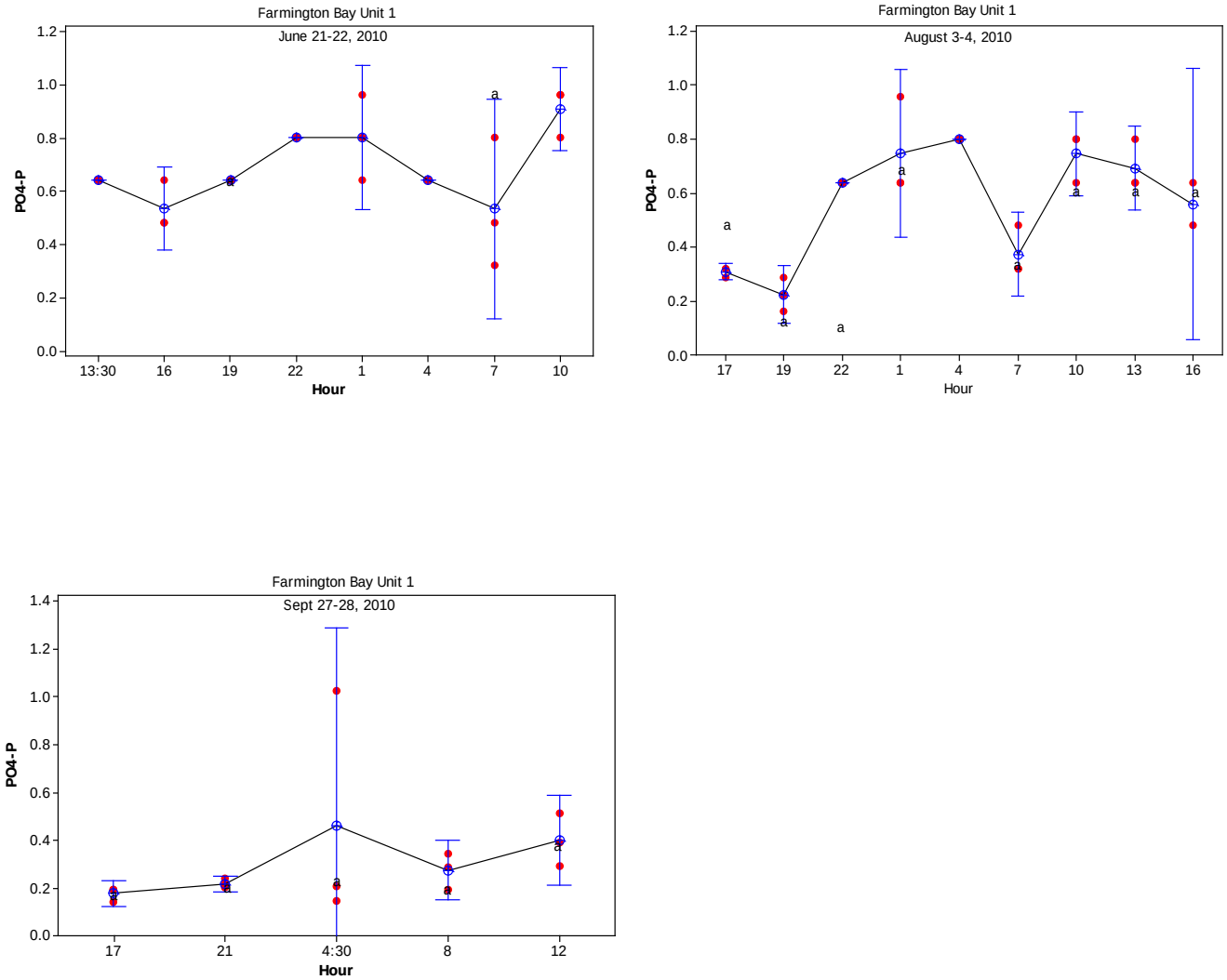


Figure E-18. Monthly measurements of phosphate-P in FB1. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

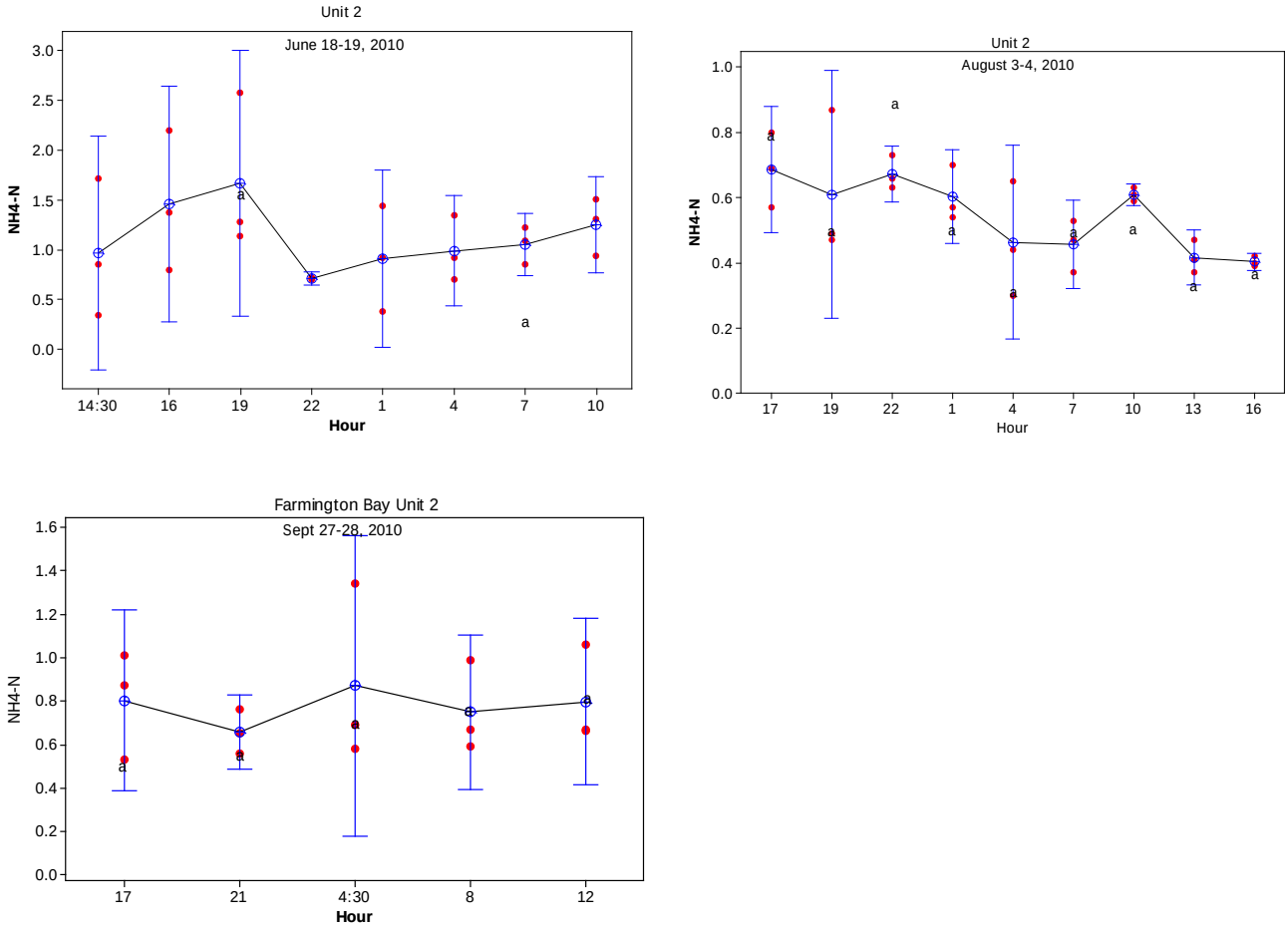


Figure E-19. Monthly measurements of ammonia in FB2. Red dots are measured values within each of three replicate chambers. Error bars indicate the 90% CI. The letter (a) indicates the ambient water column concentration measured at the same time.

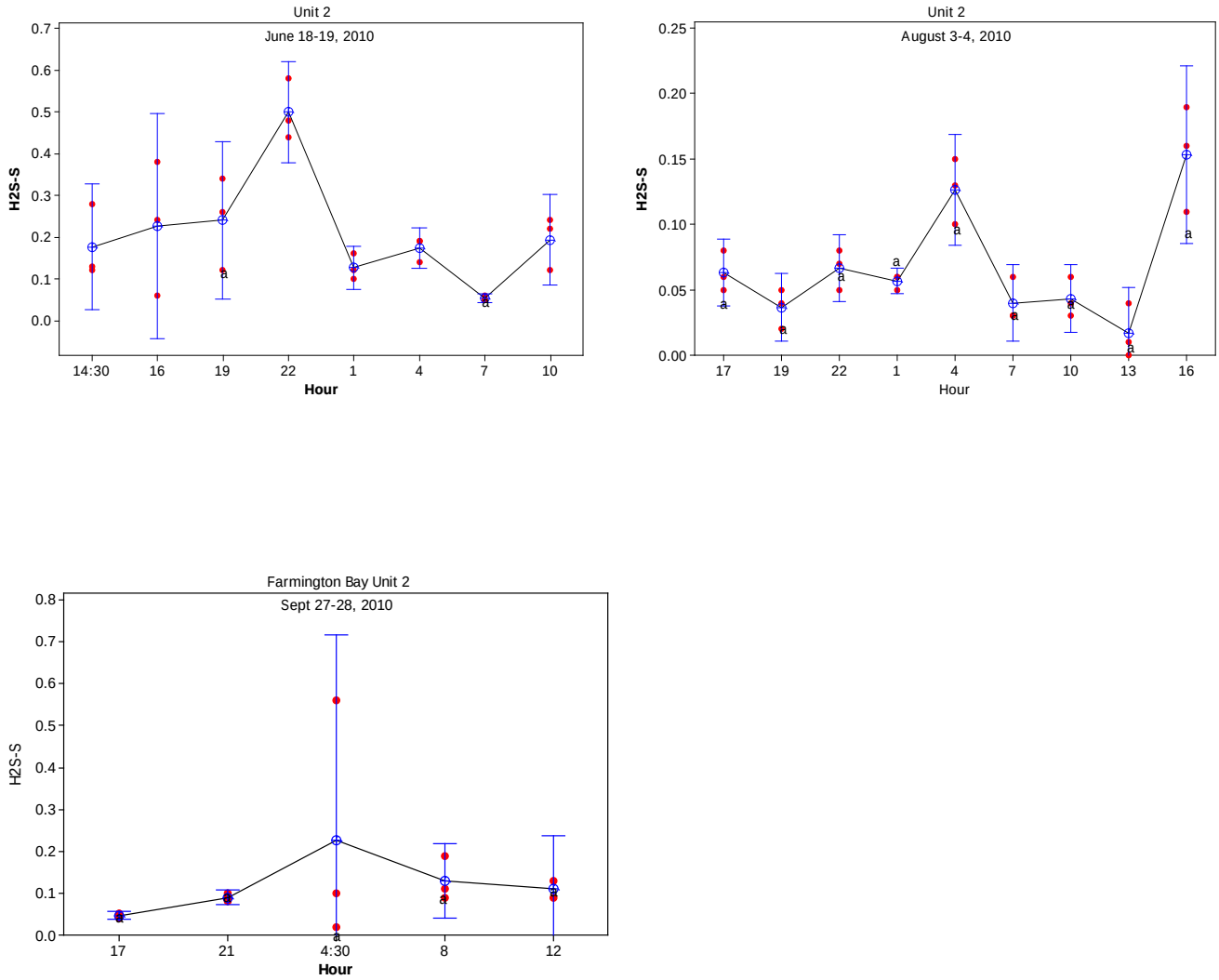


Figure E-20. Monthly measurements of H_2S in FB2. Instrument detection limit was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

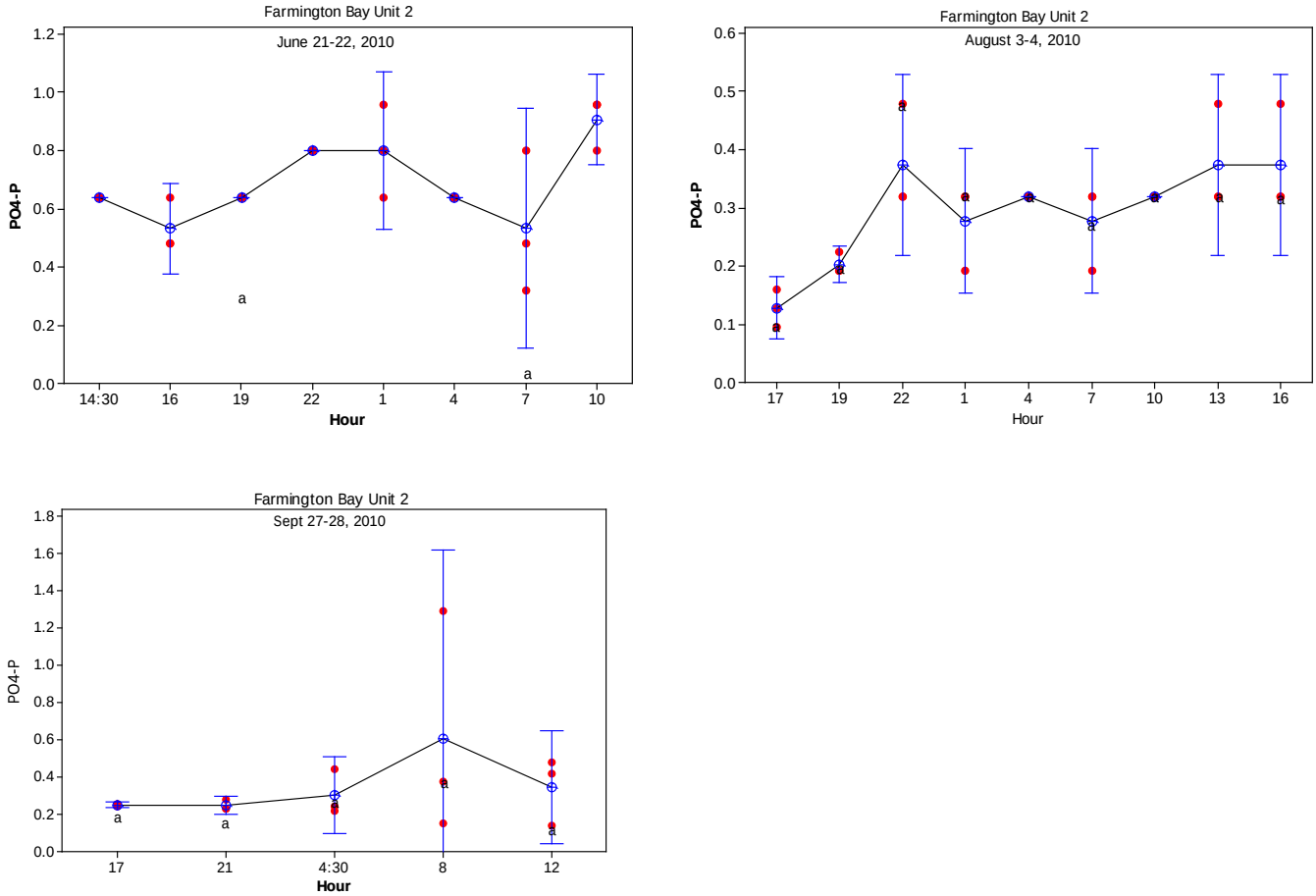


Figure E-21. Monthly measurements of phosphate in FB2. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

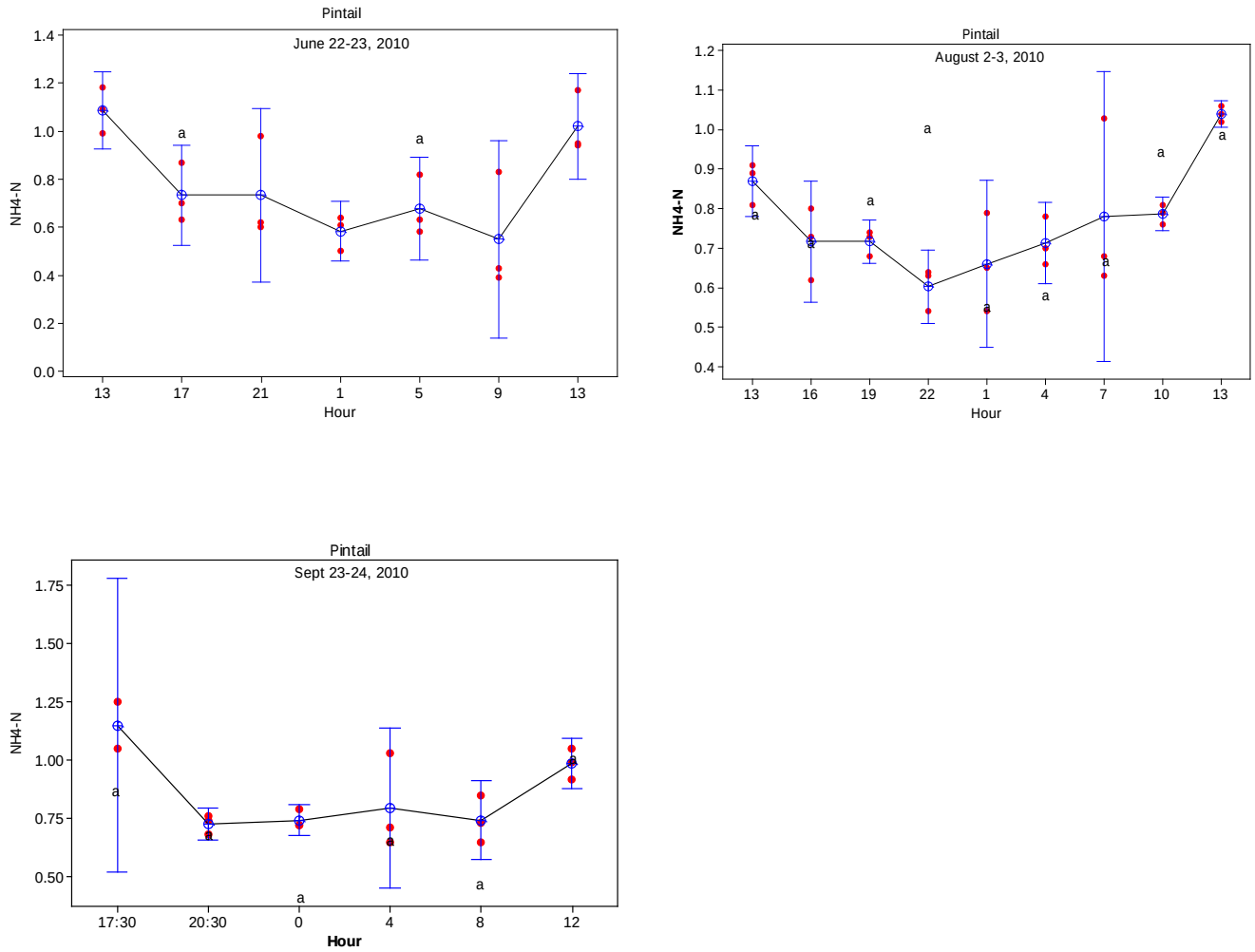


Figure E-22. Monthly measurements of ammonia in PN. Red dots are measured values within each of three replicate chambers. Error bars indicate the 90% CI. The letter (a) indicates the ambient water column concentration measured at the same time.

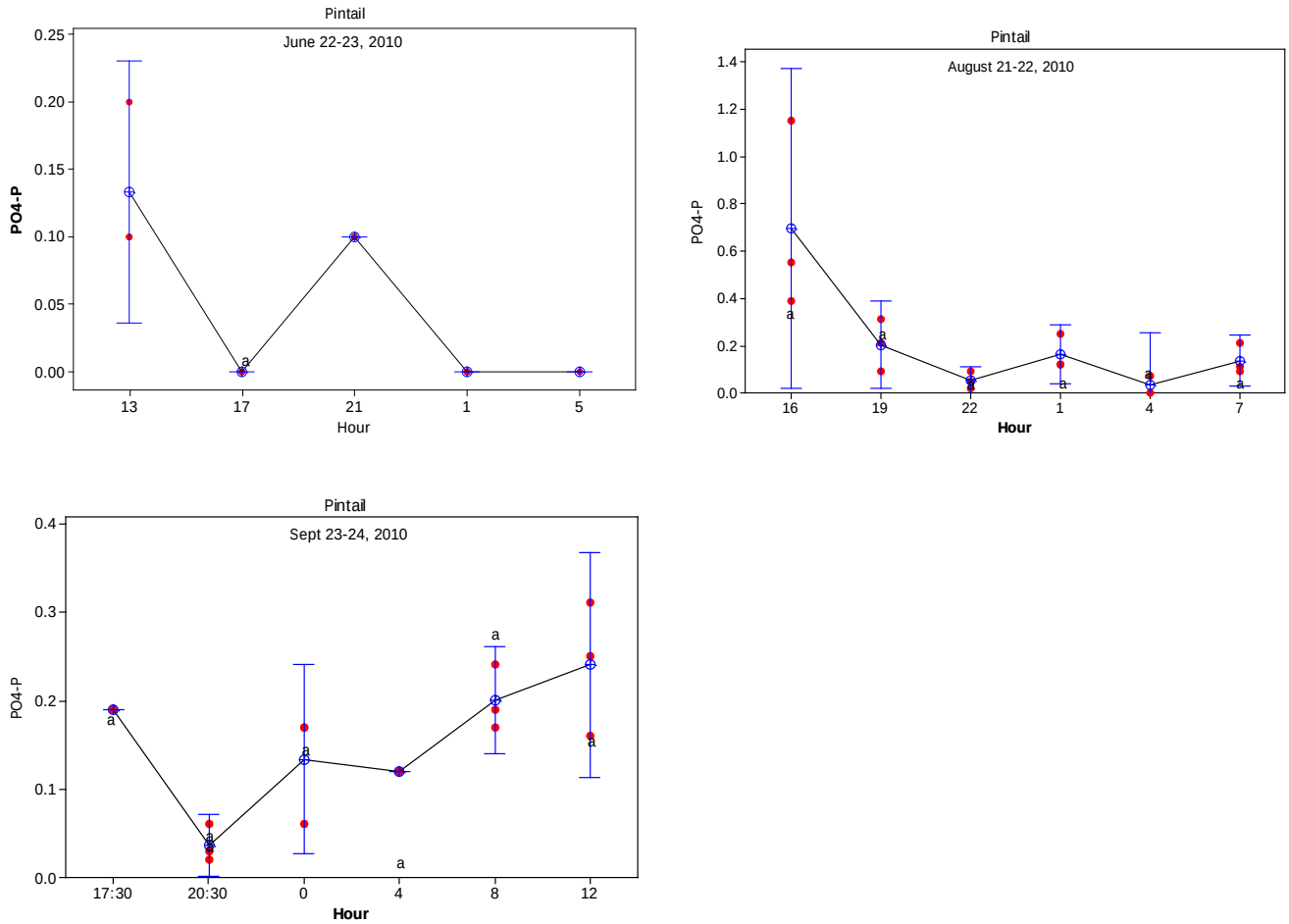


Figure E-23. Monthly measurements of phosphate in PN. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

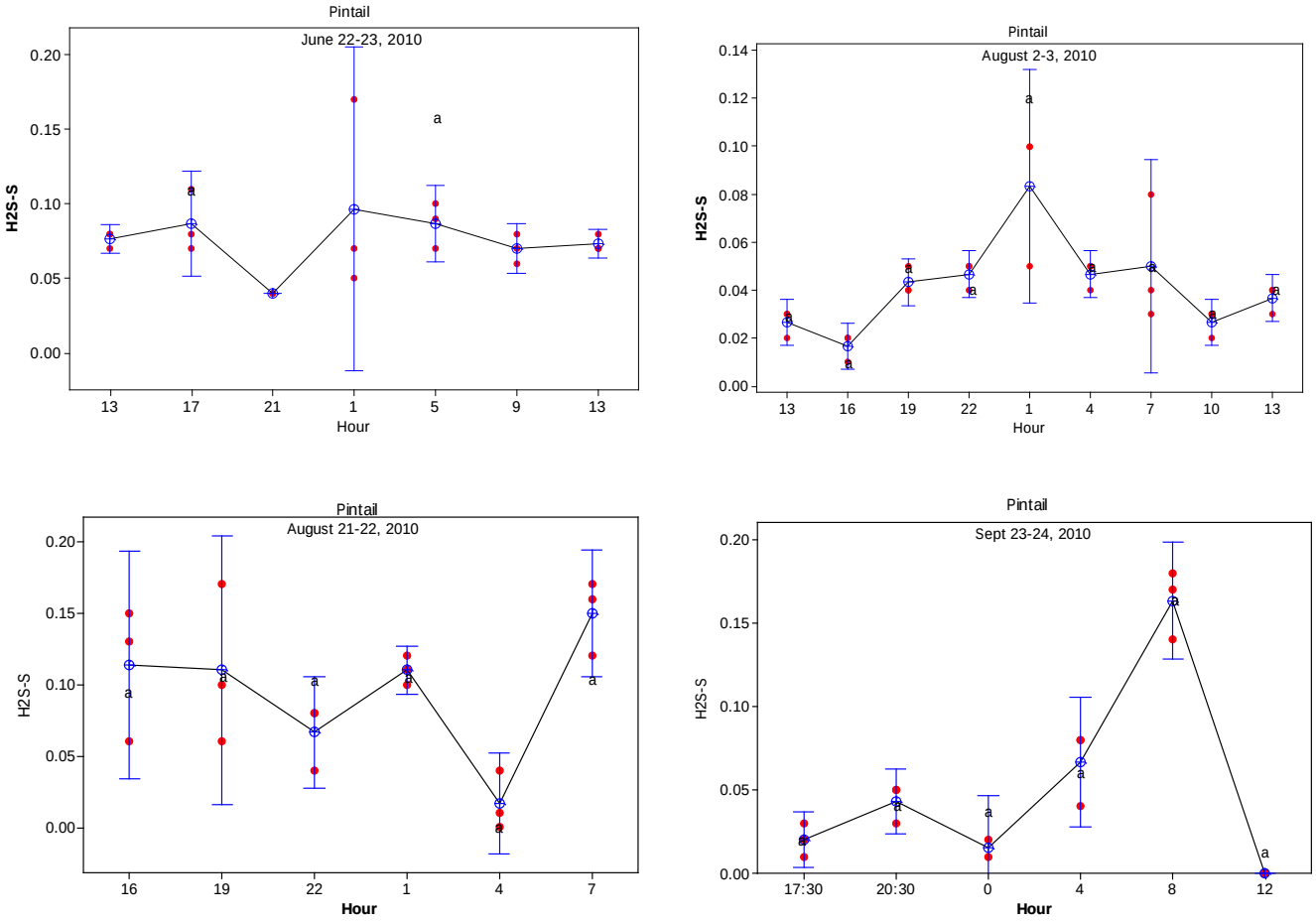


Figure E-24. Monthly measurements of H_2S in PN. Instrument detection limit was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

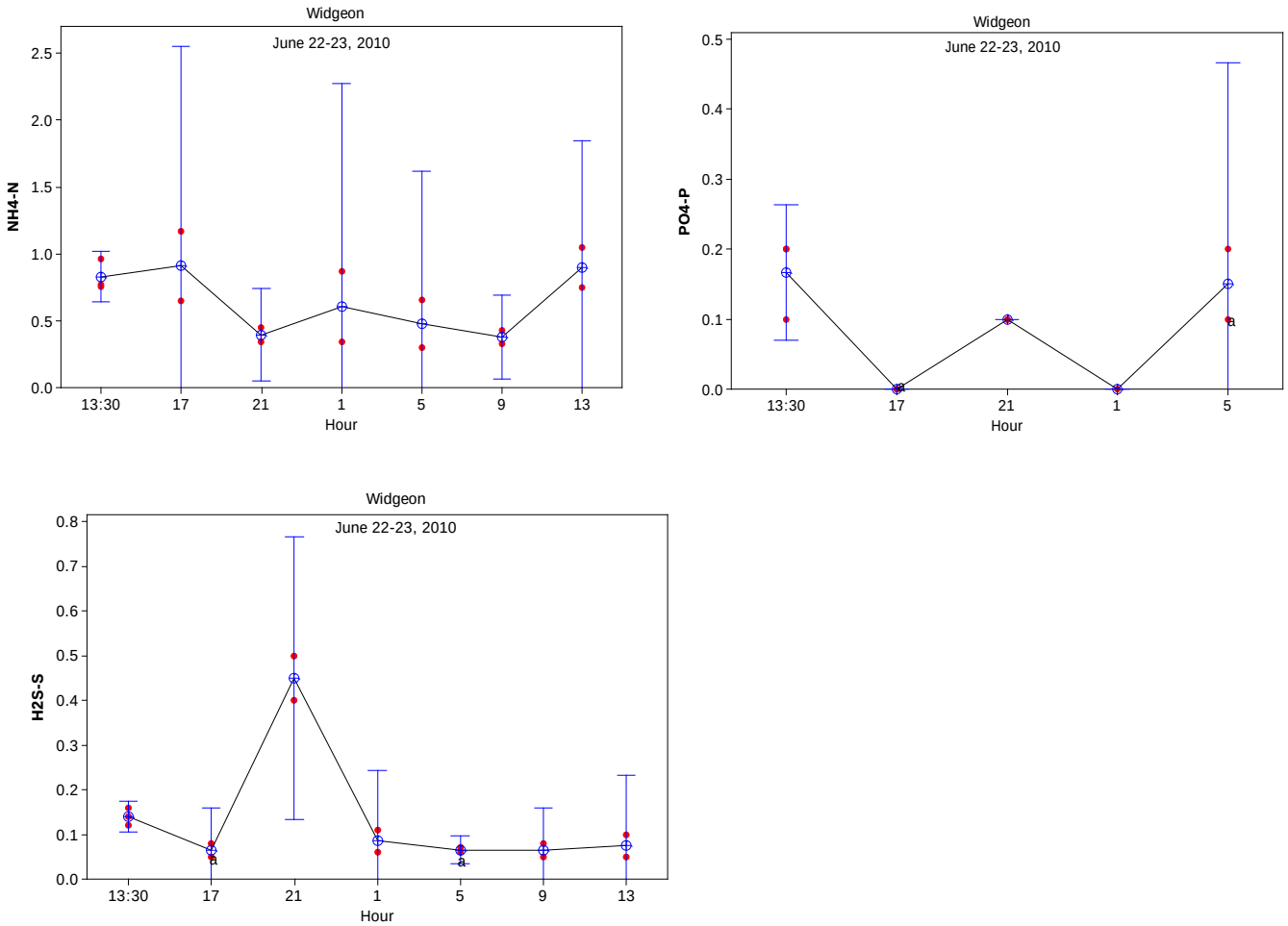


Figure E-25. June measurements of ammonia, phosphate and H_2S in WD. WD was effectively dry by July. Instrument detection limit for H_2S was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers. The letter (a) indicates values measured in the ambient water column.

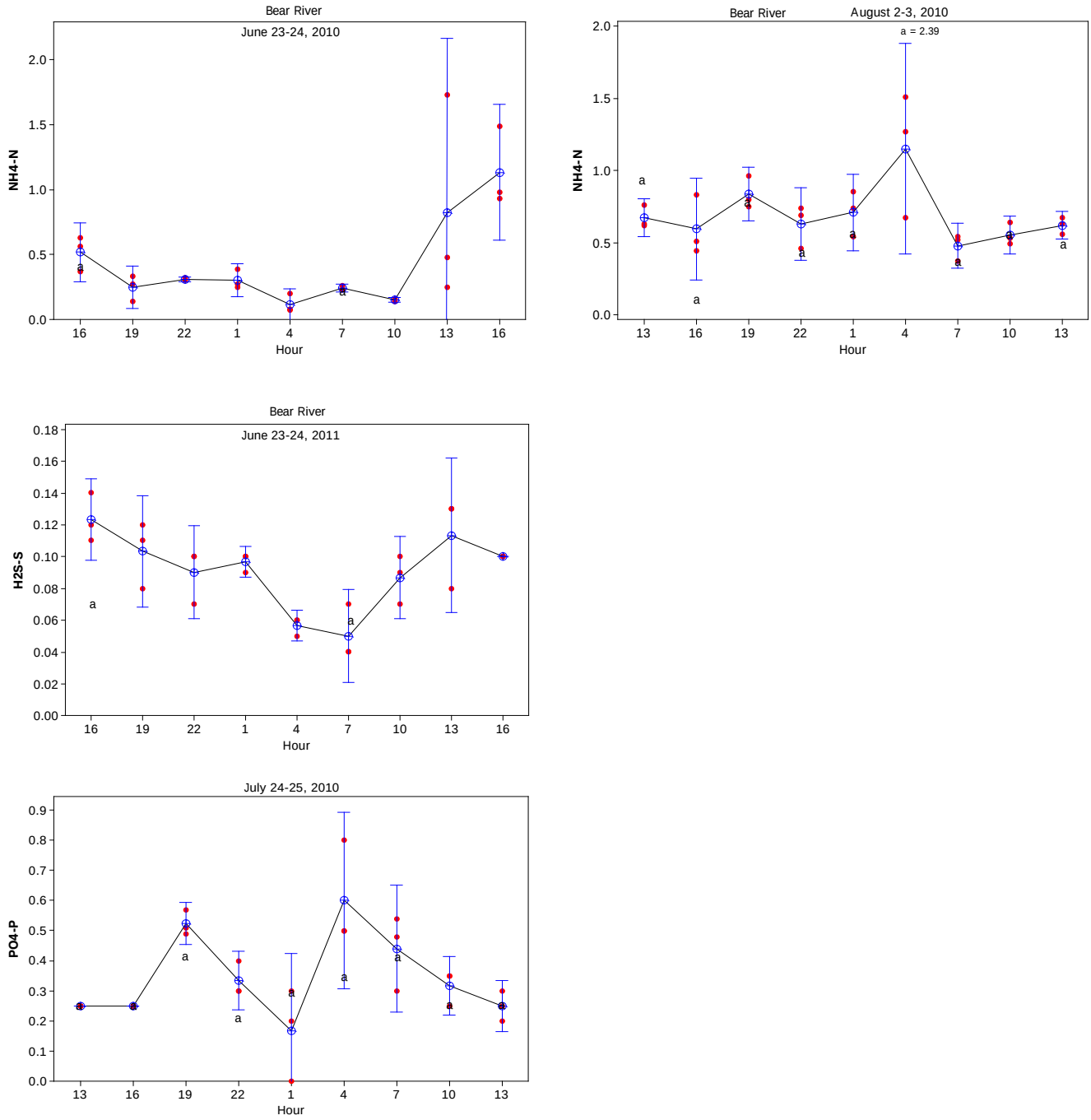


Figure E-26. Measurements of ammonia, phosphate and H_2S in BR. This impoundment was effectively dry by early August. Instrument detection limit for H_2S was 0.2 mg/L. However, as described in the text, values were reported below this concentration. Red dots are measured values and error bars (90% CI) are indicated for the sampling chambers.

Tables E-5 to E-7 provide the average, minimum, and maximum nitrogen, sulfur, and phosphate concentrations measured during each diurnal sampling event. Table E- 5 provides nutrient information for the Ambassador and Newstate ponds. Blank spaces indicate that the parameter was not measured or was measured infrequently.

Table E-5. Average, minimum and maximum N, P, and S chamber concentrations measured at Ambassador and Newstate impoundments.

Pond		NH_4-N	NO_2-N	NO_3-N	TIN	H_2S-S	SO_4-S	PO_4-P
Ambassador W1	avg, cham	0.65	0.07	0.30	1.02	0.11	24.5	0.74
June 14-15, 2010	min, cham	0.05	0.04	0.23	0.46	0.05	9.7	0.44
	max, cham	1.18	0.11	0.35	1.58	0.19	36.3	1.38
Ambassador W1	avg, cham	0.86	0.11	0.90	1.87	0.05	35.0	0.92
July 23-24, 2010	min, cham	0.72	0.08	0.64	1.62	0.04	9.7	0.67
	max, cham	0.97	0.14	1.41	2.49	0.05	63.4	1.00
	avg, amb	0.77	0.11	0.79	1.67	0.05	32.9	0.95
	min, amb	0.65	0.09	0.10	1.00	0.03	9.3	0.67
	max, amb	0.93	0.13	1.18	2.22	0.07	76.4	1.33
Ambassador W1	avg, cham	0.77	0.15	0.62	1.54	0.07	77.5	1.08
Aug. 17-18, 2010	min, cham	0.57	0.11	0.22	1.18	0.05	69.3	0.85
	max, cham	0.92	0.24	0.87	1.85	0.10	85.4	1.29
	avg, amb	0.64	0.14	0.77	1.55	0.07	78.8	1.02
	min, amb	0.38	0.09	0.28	1.05	0.02	63.6	0.75
	max, amb	0.91	0.19	1.36	2.20	0.13	92.5	1.40
Ambassador W1	avg, cham	1.78	0.11	0.72	2.62	0.05	62.0	0.89
Sept. 24-25, 2010	min, cham	1.26	0.09	0.36	1.99	0.00	51.1	0.73
	max, cham	2.73	0.14	1.19	3.37	0.13	72.2	0.97
	avg, amb	1.36	0.12	0.81	2.29	0.05	69.6	0.89
	min, amb	1.02	0.09	0.41	1.98	0.00	51.1	0.77
	max, amb	2.11	0.17	1.47	2.64	0.12	77.6	1.00

Table E-5 Continued								
New State #47	avg, cham	0.67	0.09	0.60	1.36	0.14		0.22
June 17-18, 2010	min, cham	0.46	0.03	0.23	0.99	0.06		0.08
	max, cham	0.91	0.13	1.15	2.19	0.28		0.33
New State #47	avg, cham	0.74	0.11	1.90	2.73	0.07	56.3	1.01
July 22-23, 2010	min, cham	0.42	0.09	1.17	2.25	0.04	31.8	0.83
	max, cham	0.95	0.19	2.77	3.78	0.13	67.2	1.28
	avg, amb	0.70	0.11	1.99	2.76	0.07	53.4	0.96
	min, amb	0.42	0.08	0.90	1.81	0.03	16.1	0.67
	max, amb	0.89	0.15	2.82	3.63	0.11	68.3	1.33
New State #47	avg, cham	0.93	0.25	1.01	2.19	0.13	75.8	0.52
Sept 28-29, 2010	min, cham	0.23	0.12	0.46	1.11	0.00	66.3	0.18
	max, cham	1.20	0.63	2.06	3.64	0.29	89.8	0.77
	avg, amb	0.92	0.20	1.24	2.35	0.07	73.2	0.41
	min, amb	0.05	0.12	0.57	1.14	0.00	60.6	0.07
	max, amb	1.45	0.50	2.13	3.74	0.13	84.9	0.54

Table 6 provides average, minimum and maximum measured nutrient values, H_2S and sulfate in Farmington Bay WF's Units 1 and 2.

Table 6. Average, minimum and maximum N, P, and S chamber concentrations measured at Farmington Bay ponds Unit 1 and Unit 2

Pond		NH_4-N	NO_2-N	NO_3-N	TIN	H_2S-S	SO_4-S	PO_4-P
Unit 1, FBWMA	avg, cham	1.02				0.17	18.0	0.70
June 18-19, 2010	min, cham	0.71				0.06	11.3	0.56
	max, cham	1.61				0.25	21.7	0.83
	avg, amb	1.19				0.20	19.8	0.83
	min, amb	0.66				0.06	18.7	0.67
	max, amb	1.72				0.34	21.0	1.00
Unit 1, FBWMA	avg, cham	0.61	0.06	0.10	0.77	0.06	28.8	0.59
Aug. 3-4, 2010	min, cham	0.35	0.03	0.00	0.53	0.03	11.2	0.23
	max, cham	0.96	0.12	0.21	1.16	0.14	40.3	0.83
	avg, amb	0.49	0.05	0.09	0.62	0.05	30.1	0.58
	min, amb	0.29	0.02	0.01	0.45	0.01	11.2	0.13
	max, amb	0.64	0.07	0.14	0.77	0.10	46.5	1.00
Unit 1, FBWMA	avg, cham	0.82	0.07	0.15	1.05	0.14	77.2	0.32
Sept. 27-28, 2010	min, cham	0.64	0.04	0.10	0.85	0.05	68.0	0.18
	max, cham	1.09	0.12	0.33	1.45	0.34	81.4	0.48
	avg, amb	0.67	0.08	0.14	0.90	0.09	73.1	0.25
	min, amb	0.51	0.04	0.08	0.74	0.05	59.8	0.19
	max, amb	0.94	0.15	0.23	1.13	0.15	84.0	0.38
Unit 2, FBWMA	avg, cham	1.13				0.21	23.0	0.61
June 18-19, 201	min, cham	0.71				0.05	14.0	0.24
	max, cham	1.67				0.50	29.0	1.11
	avg, amb	0.87						
	min, amb	0.24						
	max, amb	1.50						

Table 6. Continued.								
Unit 2, FBWMA	avg, cham	0.55	0.06	0.08	0.69	0.06	27.0	0.29
Aug. 3-4, 2010	min, cham	0.40	0.02	0.00	0.55	0.02	17.6	0.13
	max, cham	0.69	0.09	0.15	0.86	0.13	41.1	0.39
	avg, amb	0.53	0.05	0.09	0.63	0.05	33.9	0.30
	min, amb	0.33	0.03	0.04	0.44	0.01	21.8	0.10
	max, amb	0.88	0.07	0.16	0.97	0.10	45.1	0.50
Unit 2, FBWMA	avg, cham	0.78	0.08	0.13	0.99	0.12	71.1	0.37
Sept. 27-28, 2010	min, cham	0.66	0.04	0.09	0.79	0.05	47.6	0.26
	max, cham	0.87	0.16	0.16	1.08	0.23	78.3	0.63
	avg, amb	0.65	0.08	0.12	0.85	0.06	65.9	0.22
	min, amb	0.48	0.03	0.08	0.60	0.00	24.0	0.13
	max, amb	0.81	0.16	0.17	1.11	0.10	88.9	0.34

Dissolved ammonia continually dominated TIN in FB1 and FB2, and nitrate concentrations were significantly less than those observed in the Ambassador and Newstate ponds. Multiple dissolved sulfide spikes were observed where concentrations exceeded detection limits during the June sampling event in both FB1 and FB2, and during the September sampling event in FB1 (Figures E-17 and E-20). The only other confirmed sulfide spike (exceeding the detection limits), occurred within WD (Figure E-25). As noted above, such concentrations exceeded the sulfide standard by at least an order of magnitude.

Dissolved sulfate concentrations increased throughout the summer months, likely the result of concentration through evaporation. Interestingly, phosphate concentrations declined throughout the summer in both of the Farmington Bay ponds. This could be the result of P assimilation by surface mats of duckweed or the filamentous alga, Chladophora. Depending upon the frequency and intensity of wind storms, these mats are frequently removed from the surface and accumulate among the shoreline emergent vegetation or as windrows at the ponds' edge. These actions have been observed throughout the many years of our studies. Again, the majority of the sampling challenges encountered during the 2010 diel wetland sampling efforts were encountered in FB1 and FB2 where frequent high winds shifted surface mats, knocked over chambers and stirred up unstable sediments. Several tests needed to be restarted or otherwise abandoned.

Table E-7 provides average, minimum, and maximum nutrient concentrations measured in the northern ponds located within the Bear River Wildlife Refuge and Public Shooting Grounds. These wetland ponds are unique since they receive influent from the Bear River and various saline springs.

Table E-7. Average, minimum and maximum N, P, and S chamber concentrations measured at Bear River and Public Shooting Grounds

Pond		NH_4-N	NO_2-N	NO_3-N	TIN	H_2S-S	SO_4-S	PO_4-P
Bear River 4C	avg, cham	0.43		0.10		0.09	12.7	
June 23-24, 2010	min, cham	0.12		0.08		0.05	9.7	
	max, cham	1.13		0.11		0.12	15.7	
	avg, amb	0.36		0.11		0.07	12.8	0.03
	min, amb	0.26		0.09		0.06	10.0	0.02
	max, amb			0.13		0.07	15.7	0.03
Bear River 4C	avg, cham	0.69	0.09	0.10	0.88	0.10	11.6	0.11
July 24-25, 2010	min, cham	0.48	0.05	0.06	0.67	0.06	10.1	0.06
	max, cham	1.15	0.18	0.17	1.26	0.19	15.9	0.20
	avg, amb	0.74	0.09	0.08	0.91	0.08	12.7	0.09
	min, amb	0.15	0.04	0.04	0.29	0.04	9.9	0.01
	max, amb	2.39	0.20	0.12	2.47	0.14	15.3	0.13
Pintail, PSG WMA	avg, cham	0.77		0.07		0.08	42.0	0.04
June 22-23, 2010	min, cham	0.55		0.05		0.04	27.7	0.03
	max, cham	1.09		0.08		0.10	56.3	0.04
	avg, amb			0.07		0.14		
	min, amb			0.06		0.11		
	max, amb			0.08		0.16		
Pintail, PSG WMA	avg, cham	0.77	0.07	0.09	0.93	0.04	46.3	0.05
Aug. 2-3, 2010	min, cham	0.60	0.01	0.05	0.71	0.02	36.4	0.02
	max, cham	1.04	0.15	0.21	1.23	0.08	49.8	0.08
	avg, amb	0.79	0.06	0.09	0.94	0.05	45.0	0.03
	min, amb	0.56	0.00	0.04	0.63	0.01	31.8	0.02
	max, amb	1.04	0.11	0.21	1.14	0.12	57.7	0.06

Table E-7. Continued

Pintail, PSG WMA	avg, cham	0.97	0.05	0.09	1.10	0.09	55.4	0.07
Aug. 21-22, 2010	min, cham	0.63	0.03	0.04	0.74	0.02	52.4	0.01
	max, cham	1.45	0.09	0.14	1.65	0.15	60.0	0.23
	avg, amb	1.05	0.04	0.07	1.14	0.08	56.4	0.05
	min, amb	0.52	0.03	0.04	0.60	0.00	50.2	0.02
	max, amb	2.04	0.06	0.10	2.15	0.11	70.9	0.12
Pintail, PSG WMA	avg, cham	0.80	0.07	0.06	0.93	0.08	95.1	0.05
Sept. 23-24, 2010	min, cham	0.73	0.04	0.01	0.82	0.00	79.9	0.01
	max, cham	0.99	0.11	0.09	1.19	0.16	122.0	0.08
	avg, amb	0.67	0.08	0.06	0.82	0.06	100.5	0.05
	min, amb	0.34	0.03	0.01	0.45	0.02	69.7	0.01
	max, amb	1.03	0.14	0.08	1.22	0.16	117.0	0.09
Widgeon, PSG WMA	avg, cham	0.64		0.08	0.67	0.13	40.3	0.05
June 22-23, 2010	min, cham	0.38		0.06	0.38	0.07	32.7	0.03
	max, cham	0.91		0.10	1.01	0.45	48.0	0.06
	avg, amb	0.57		0.08			36.5	0.03
	min, amb	0.38		0.05			20.1	0.02
	max, amb	0.76		0.10			53.0	0.03

Electrical conductivity (EC), a common surrogate for total dissolved solids, was measured several times in all of the impoundments (Table E-8). EC increased in all of the ponds after spring runoff and residence times and evaporation rates increased. Water availability in Bear River Bird Refuge and the Public Shooting Grounds is particularly limited during the agricultural irrigation season which frequently causes these ponds to completely dry by mid to late summer. During this drying process carp are also concentrated nearer to the outlet culverts. This concentrated carp activity greatly increases turbidity and reduces the healthy appearance of the submerged vegetation.

Notably, the SAV responds very favorably after these ponds have been refilled. For example, after FB1 was dried intentionally during the summer of 2008, the Stuckenia regrew with greater bottom coverage, later

senescence and greater production of seeds (Hoven 2010a, 2010b). Similarly, the macroinvertebrate community responded with greater diversity and greater numbers of known sensitive taxa (Gray 2010). In addition, phosphorus and ammonia values remained below concentrations measured in previous years (the oxidized sediments likely prevented the dissolution of nutrients), and surface mats of duckweed or Chladophora were nearly absent (Hoven 2010a and 2010b). Further, this positive response lasted for the following two years with the DO consistently remaining above zero. Notably these dry conditions occur in Public Shooting Grounds and most Bear River Bird Refuge ponds nearly every year for one to two months. This annual oxidation of the sediments may be the reason which DO rarely reach zero, the SAV is consistently thick and healthy and the macroinvertebrate community generally consists of sensitive taxa.

Finally, the benefits of regular (every 3-4 years) drying of the impoundments, on a rotational basis, should continue to be evaluated as a management tool to retain nutrients, sulfides and metals in sediments and promote healthy growth of SAV.

Table E-8. Average, minimum & maximum ORP & conductivity concentrations

			ORP (mV)			Conductivity		
Pond	**Date**	**Probe**	**avg**	**min**	**max**	**avg**	**min**	**max**
Ambassador W1	June 14-15, 2010	Ambient	261	130	400	935	789	1082
		Chamber						
Ambassador W1	July 23-24, 2010	Ambient	130	110	210	1820	1598	1940
		Chamber	128	30	180	1627	1071	1755
Ambassador W1	Sept. 24-25, 2010	Chamber	-22	-80	10	1429	1303	1592
New State #47	June 17-18, 2010	Ambient	267	210	300	775	732	920
		Chamber						
New State #47	July 22-23, 2010	Ambient	235	60	280	1480	1371	1631
		Chamber	262	180	320	1663	1437	1847
New State #47	Sept 28-29, 2010	Ambient	42	30	50	1483	1439	1529
		Chamber	16	0	40			
Unit 1, FB	July 26, 2010	Ambient	253	70	290	2058	1920	2159
		Chamber						
Unit 1, FB	Sept. 27-28, 2010	Ambient	-6	-70	0	2308	2245	2397
		Chamber	-35	-140	20	2174	2093	2214
Unit 2, FB	June 18-19, 201	Chamber						
Unit 2, FB	July 26, 2010	Ambient	151	40	170	1655	1589	1717
Unit 2, FB	Sept. 27-28, 2010	Ambient						
Bear River 4C	June 23-24, 2010	Ambient	185	10	420	546	410	579
		Chamber						
		Large						
Bear River 4C	July 24-25, 2010	Ambient	100	60	200	1201	1075	1358
		Chamber	68	-80	90	1040	981	1097
		Large						
Pintail	June 22-23, 2010	Ambient						
		Chamber						
Widgen	June 22-23, 2010	Ambient						
		Chamber						

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