Columbia River Project Water Use Plan

KINBASKET RESERVOIR FISH AND WILDLIFE INFORMATION PLAN

Wetland Vegetation

Implementation Year 2

Reference: CLBMON-61

Kinbasket Reservoir: Mica Unit 5 Wetlands Monitoring Program

Study Period: 2013

LGL Limited
environmental research associates
Sidney, BC

August 24, 2014

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KINBASKET AND ARROW LAKES RESERVOIRS

Monitoring Program No. CLBMON-61

Kinbasket Reservoir Wetlands Monitoring Program

Year 2–2013
Draft Report

Prepared for

BC Hydro Generation
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Prepared by

Douglas B. Adama, BSc, RPBio.

Charlene M. Wood, MSc.

Virgil C. Hawkes, MSc, RPBio,

August 24, 2014

LGL LIMITED
environmental research associates

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Cover photos

Kinbasket Reservoir from left to right: Retrieving data loggers from Bush Arm Causeway in November 2013, Wood debris at 754 m ASL Valemount Peatland, inundated wetland at Bush Arm Km 88; Sprague Bay wetland at 754 m ASL © Doug Adama, LGL Limited and Richard Klafki.

August 24, 2014

EXECUTIVE SUMMARY

The construction of Mica Dam and Kinbasket Reservoir resulted in the loss of an estimated 21,946 ha riparian, wetland, and shallow pond habitats. In 2008, BC Hydro undertook an Environmental Assessment (EA) for the construction and operation of two additional turbines in Mica Dam. Generalized Optimization Modeling (GOM) predicted reservoir levels will increase an additional 0.6 meters in every three years out of ten, and that the impacts, if any, will occur in the 753 to 754 meter elevation band. Under the Environmental Assessment Certificate (EAC), BC Hydro is required to assess the impacts resulting from the operation of the additional turbines on wetland habitats in Kinbasket Reservoir. This assessment is being carried out under CLBMON–61: Kinbasket Reservoir Wetlands Monitoring Program. The Terms of Reference (BC Hydro 2012) for CLBMON–61 calls for a Before-After- Control-Impact (BACI) design to address the following management questions:

1. What are the short-term effects of water level changes on wetland vegetation composition or productivity, with emphasis on the 753 to 754 m elevation band?
2. If negative changes in wetland vegetation composition or wetland productivity are detected, which are directly imputable to Mica 5 operations, are there operational changes or mitigative measures that could be implemented to improve wetland integrity (combination of composition and productivity) in Kinbasket Reservoir?

In 2013, a second year of data were collected for CLBMON–61 and are summarized herein. The objectives for Year 2 were to (1) summarize the state of the wetland index sites identified in Year 1, and (2) provide preliminary insight into the expected changes to vegetation composition or wetland productivity. We also assessed the efficacy of the sampling methods.

Aquatic and terrestrial wetlands were sampled at four index sites: Bush River, Km 88, Sprague Bay, and the Valemount Peatland. Index sites included paired impact and reference aquatic wetlands (ponds), as well as terrestrial wetlands stratified across the following elevation bands: 752 to 753 (lower control) m ASL, 753 to 754 (target) m ASL, 754 to 755 m ASL (upper control), and reference wetlands outside the reservoir above 755 m ASL. Terrestrial wetlands were sampled using belt transects and circular plots. Vegetation and ground cover (substrate) data were compared across the four elevation bands. Ponds were stratified to include ponds within the reservoir at approximately 753 m ASL and reference ponds between 756 and 780 m ASL. Macrophyte cover and biomass, water physicochemistry, and pelagic invertebrates were compared across these strata. Continuous water physicochemistry and aquatic metabolism (net ecosystem production (NEP), gross primary production (GPP), and ecosystem respiration (R)) were compared before and after inundation of the 753 m ASL elevation band. Where possible, results were compared against 2012 data.

Similar to 2012, reservoir levels greatly exceeded the 1987 to 2006 historical operating regime, and the 753 m ASL elevation band was inundated 5 times longer (87 versus 16.6 days) than the 1987 to 2006 norm. Fortunately, reservoir levels were slow to fill and the target elevation band did not become inundated until August 15, 2013, permitting data collection at all sites.

Terrestrial wetland communities increased in complexity from sparsely vegetated graminoid communities at lower elevations (752 m ASL) to more diverse shrub-herb communities in the upper elevations (754 m ASL and above). Our assessment of terrestrial wetland data indicates that shrub and substrate (e.g., wood debris) cover

provided the most reliable signal for assessing change over time. Shrub cover increased by 200 per cent with each 1–metre elevation band from 752 to 754 m ASL. Shrub species richness also increased by over 150 per cent across each elevation band from 10 species at 752 ASL to 35 species at 754 m ASL and diversity measures (Shannon and Simpson) differed significantly ($\alpha = 0.10$) across the elevation gradient. These attributes are predicted to decrease with increased flooding of the upper elevation bands.

Wood debris cover was significantly higher in terrestrial wetland transects within the reservoir than in reference transects (13.2 versus 0.4 per cent; $p = 0.01$) and wood debris was associated with a reduction in vegetation cover within the reservoir. In aquatic wetlands, wood debris was detected at a higher frequency within the reservoir than in reference ponds ($p = 0.02$) by a factor of 5. Following the installation of Mica units 5 and 6, the inundation of the 753 m ASL elevation band is predicted to increase in frequency; thus a parallel increase in the accumulation of wood debris in both aquatic and terrestrial wetlands is predicted.

Indicator species analysis (ISA) identified *Carex lenticularis ssp. lipocarpa* (lenticular sedge) as the only indicator species for the 753 m ASL target elevation band (*INDVAL* = 36.4, $p = 0.04$). The identification of lenticular sedge as an indicator is significant because it has been planted extensively throughout Kinbasket Reservoir under CLBWORKS-30 to enhance vegetation in the upper elevations of the reservoir (KES 2012). An increase in flooding frequency and duration may reduce the survival of both planted and naturally occurring lenticular sedge undermining the efforts of the revegetation program to maintain and enhance vegetation in the reservoir.

Diel changes in dissolved oxygen were used to estimate NEP, GPP, and R before and after inundation, uncorrected for atmospheric diffusion. In reference ponds, aquatic metabolism generally remained static across the pre- and post-inundation periods whereas within reservoir ponds, GPP, NEP, and R generally differed pre- and post-inundation. As the values were not corrected for diffusion and responses varied across index sites, we cannot comment on the magnitude of the changes at this time. Nevertheless, our findings indicate changes in NEP, GPP, and R may be useful for assessing the impacts of reservoir operations on primary productivity.

Despite high variability in macrophyte cover across the study area, macrophyte abundance in 2013 was between 1.7 and 9.3 times greater in ponds within the reservoir than in paired reference ponds. Higher abundance of macrophytes in the reservoir may reflect the lack of adjacent forest canopy, which results in greater light availability and macrophyte growth, and/or increased nutrient and mineral input from reservoir inundation. From the data collected, it appears that macrophytes respond sufficiently to reservoir conditions to warrant monitoring; however, our results are based on a small sample size ($n = 8$ ponds). Nevertheless, it is anticipated that increases in water depth and prolonged inundation (resulting from operational changes) will have a negative effect on macrophyte abundance through reduced light penetration and increased wave action. However, if reservoir operations return to the 1987 to 2006 norm, macrophyte abundance in DDZ ponds may increase from the values observed in 2012 and 2013, when the ponds were subjected to prolonged periods of inundation.

Pelagic invertebrate data did not produce any discernible patterns in either 2012 or 2013. This may be related to high natural variability combined with limited sampling frequency (one sample session per year). As such, we do not consider pelagic invertebrates to be useful for assessing the impacts associated with Mica Units 5 and 6, as currently sampled.

Recommendations:

1. We recommend discontinuing the sampling for pelagic invertebrate, as no obvious trends could be determined from the data. In lieu, we recommend obtaining more accurate estimates of primary production and aquatic metabolism (NEP, GPP, and R).
2. Recent advances in sonde technology permit the calculation of reliable metabolic rates (NEP, GPP, and R) from diel fluctuations in dissolved oxygen. Dissolved oxygen sondes were deployed in aquatic wetlands in 2012 and 2013; however, additional instrumentation is required to estimate the diffusion of oxygen into the atmosphere. As oxygen diffusion can exceed hourly NEP, estimates of NEP and GEP without such correction can lead to spurious results. In future years, we recommend installing additional instrumentation to correct for atmospheric diffusion. Because of the complexities involved with equipment installation, data collection, and numerical computations, we recommend that protocols for instrumentation installation and data analysis be prepared.
3. We recommend an additional year of sampling be carried out in 2014 for the following reasons:
    a. The Terms of Reference for CLBMON-61 prescribed a BACI sampling design and specified two years of data to be collected prior to the operation of the new turbines. High reservoir levels limited the sampling in 2012 resulting in an incomplete data set for Year 1. Sampling in 2014 will provide a second year of data as required.
    b. Prolonged inundation (87 days) of the 753 m ASL elevation band in 2013 may affect the composition and productivity of wetlands in this elevation band and impacts may carry over into the post-impact period (2015 and beyond). If this occurs, we will be unable to determine whether these impacts are a result of the 2013 reservoir levels or the installation of the new units.
    c. As reservoir levels are forecast to be considerably lower, sampling in 2014 should provide an opportunity to sample under conditions more similar to the 1987-2006 operating conditions.

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1.0 INTRODUCTION

Wetlands offer many ecosystem services including the control of sediment, flood mitigation, improvement to water quality, wildlife habitat, and the sequestration and long-term storage of carbon dioxide (Mitsch and Gosselink 2007). Notwithstanding their importance, wetlands continue to be degraded or lost due to the development of hydroelectric projects (Junk et al. 2102; Russi et al. 2013). In 1974, the construction of Mica Dam and Kinbasket Reservoir resulted in the loss of 15,527.5 ha of riparian, 5,863 ha of wetland, and 555 ha of shallow pond habitats and less than two per cent of the wetland habitat that existed prior to the construction and inundation remain (Utzig and Schmidt 2011; Adama et al. 2013). Thus, the remaining wetlands in the upper elevations of Kinbasket Reservoir provide rare and unique environments for wetland dependent species in this mountainous landscape.

In 2008, BC Hydro undertook an Environmental Assessment (EA) for the construction and operation of two additional turbines in Mica Dam (Units 5 and 6; BC Hydro 2009). During the EA, it was identified that changes to reservoir operations could negatively impact wetlands, wetland vegetation, and dependent wildlife during the summer re-fill period. A General Optimization Model (GOM) predicted that reservoir levels would be 0.6 m higher in July and August in every three years out of ten (KCB 2009). The model also predicted that the impacts would be restricted to the elevation band spanning 753 to 754 m above sea level (ASL). Under the EA certificate, BC Hydro is required to assess the potential impacts resulting from the construction and operation of these two additional turbines on wetlands in Kinbasket Reservoir.

The CLBMON-61 ToR (BC Hydro 2012) call for a Before-After-Impact-Control (BACI) design to (1) assess potential changes in wetland composition and productivity in Kinbasket Reservoir, and (2) to determine whether any change can be associated with reservoir operations. Year 1 of the study was initiated in 2012; this report provides the annual progress report for Year 2 (2013).

The objectives for Year 2 were to (1) summarize the state of the wetland index sites identified in Year 1, and (2) provide preliminary insight into the expected changes to vegetation composition or wetland productivity. We also assessed the efficacy of the sampling methods.

2.0 MANAGEMENT QUESTIONS AND HYPOTHESES

2.1 Management Questions

To address the uncertainties relating to changes in reservoir operation following the installation of Mica Units 5 and 6, this monitoring program will focus on:

- obtaining measurements of wetland area, composition and productivity that can also be used as parameters for modeling the effects of inundation on plant communities in the 753 to 754 m range (as specified under CLBMON 10); and
- determining key indicators of change in wetland composition and productivity.

The key management questions to be addressed by the monitoring program are:

- What are the short-terms effects of water level changes on wetland vegetation composition or productivity, with emphasis on the 753 to 754 m elevation band?
- If negative changes in wetland vegetation composition or wetland productivity are detected which are directly imputable to Mica 5 operations, are there operational

changes or mitigative measures that could be implemented to improve wetland integrity (combination of composition and productivity) in Kinbasket Reservoir?

2.2 Management Hypotheses

To assess the effects of reservoir operations associated with Mica Unit 5 and 6 on wetland composition and productivity, the following null hypotheses will be tested:

H_{01}: There are no changes in wetland composition in Kinbasket Reservoir over the course of the monitoring period.

H_{1A}: Wetland composition is not affected by reservoir operations.

H_{02}: There are no changes in wetland productivity in Kinbasket Reservoir over the course of the monitoring period.

H_{2A}: Wetland productivity is not affected by reservoir operations.

2.3 Key Water Use Decision

Implementation of the proposed monitoring program will provide information to support decisions around the need to balance storage in Kinbasket Reservoir with impacts on wetland integrity (composition and productivity). Specifically, the program will provide information required to support future decisions around maintaining the current operating regime or modifying operations through adjusting minimum or maximum elevations to sustain reservoir wetlands. The intent is to ensure that wetlands in the upper elevations of the reservoir drawdown area are not adversely affected by incremental changes in reservoir operations attributable to the fifth and sixth turbines in Mica Dam.

3.0 SUMMARY OF PREVIOUS WORK

This study commenced in 2012 with the following objectives:

a) provide a general description of wetlands in the upper elevation of Kinbasket Reservoir;

b) describe and justify the methods used to select index sites for monitoring; and,

c) review the study approach and methods (both field and analytical) to ensure they are appropriate for addressing the management questions and hypotheses.

A site review was undertaken using GIS and existing data to delineate wetland habitats in Kinbasket Reservoir for sampling. Using vegetation mapping from CLBMON-10 (Hawkes et al. 2007, 2010), 102.8 hectares of wetland habitat were identified between the 751 and 755 m ASL elevations with 34.1 hectares occurring in the target elevation band (753–754 m ASL). During the site review, 25 aquatic and 50 terrestrial wetland sites were identified for sampling including 12 aquatic and 13 terrestrial wetland reference sites located outside the reservoir. Aquatic wetlands were defined as permanent shallow waters (i.e., ponds and shallow lakes) and terrestrial wetlands include bog, fen, swamp, or marsh wetland classes as defined by MacKenzie and Moran (2004; Appendix 10.1).

Data collected during field surveys between July 7 and August 22, 2012 included general wetland characteristics, vegetation community composition, water physicochemistry, wood debris, macrophytes biomass, and macroinvertebrates. Terrestrial sampling was stratified across one-meter elevation bands from 752 to 755 m

ASL and at reference sites above the reservoir (> 755 m ASL). Aquatic sampling was stratified between ponds within and above the reservoir. During the sampling period, water levels in Kinbasket Reservoir rose rapidly and flooded many of the sites. Consequently, only 16 aquatic wetlands and 15 terrestrial wetland sites were sampled. Despite this, sufficient data were collected to characterize the wetlands in and adjacent Kinbasket Reservoir and to review the study approach and methodology.

Key findings from Year 1:

- Terrestrial and aquatic wetland communities were quite diverse. Nine terrestrial and twelve aquatic wetland associations were identified using the classifications of Pierce and Jensen (2001), MacKenzie and Moran (2004), and Hawkes et al. (2007).
- In terrestrial wetlands, species richness and diversity increased with elevation from 752 to 755 m ASL. Lower elevation communities tended to be either Willow–Sedge or Swamp Horsetail associations and upper elevation communities were either Willow–Sedge, flood, marsh, and fen associations. Decreasing shrub cover was also observed across the elevation gradient while pteridophyte (e.g., *Equisetum* spp.) and sedge cover increased at the lower elevations (752 –753 m ASL).
- In aquatic wetlands, beaver activity, water depth, water physicochemistry, and organic accumulation (including wood debris) appeared to influence the distribution of aquatic communities. Beaver activity was apparent in 75 per cent of the ponds sampled and appears to be an important wetland forming process in the study area
- pH and conductivity values differed significantly among the three reaches (Bush Arm, Mica Arm, and Canoe Reach) and was likely due to geological differences across the study area. Differences in water physicochemistry were reflected in the distribution of the vegetation communities.
- A higher frequency of wood debris was observed in the benthic sediment of drawdown zone (DDZ) ponds than in reference ponds. This is likely due to the large amounts of wood debris that accumulate annually in the upper elevation of the reservoir.
- Macrophyte biomass did not differ significantly between DDZ ponds and reference ponds.
- Pelagic macroinvertebrate taxa in DDZ ponds did not differ from reference ponds with the exception of Canoe Reach, where the number of taxa documented from DDZ ponds were lower than in upland reference ponds. In almost all cases the relative abundance of the individual taxa detected in 2012 did not differ significantly between DDZ and reference ponds.

Recommendations from Year 1 included:

1) Review the methodology for sampling pelagic and benthic invertebrates, water transparency, and macrophyte biomass samples.
2) Continue the stratified sampling design established in Year 1, stratifying by wetland type (terrestrial and aquatic), by elevation band, and by reach;
3) Focus the monitoring effort to four index sites: Valemount Peatland (Canoe Reach), the Sprague Bay wetlands (Mica Arm), the Km 88 wetlands (Bush Arm), and the wetland complex at the Bush River Causeway (Bush Arm).
4) Investigate the utility of using diel dissolved oxygen measurements to estimate primary productivity.

5) Develop an Index of Wetland Integrity (IWI) for the index sites using metrics to assess taxonomic diversity and richness, structural stage, community structure, primary productivity, secondary productivity, and disturbance.
6) Replace the BACI design prescribed in the Terms of Reference with annual monitoring.

4.0 STUDY AREA

4.1 Kinbasket Reservoir

Located in south eastern B.C., Kinbasket Reservoir is surrounded by the Rocky and Monashee Mountain ranges and is approximately 216 km long Arm (Figure 4-1). The Mica hydroelectric dam, located 135 km north of Revelstoke, B.C., spans the Columbia River and impounds Kinbasket Reservoir. The Mica powerhouse, completed in 1973, has a generating capacity of 1,805 MW, and Kinbasket Reservoir has a licensed storage volume of 12 million-acre feet (MAF; BC Hydro 2007). The normal operating range of the reservoir is between 707.41 m and 754.38 m elevation, but can be operated to 754.68 m ASL with approval from the Comptroller of Water Rights.

The reservoir consists of seven reaches: Beaver Mouth, Kinbasket Reach, Bush Arm, Sullivan Arm, Mica Creek, Wood Arm, and Canoe Reach. The reaches of interest to this study include Bush Arm, Mica Arm, and Canoe Reach (Figure 4-1). The shoreline of the reservoir is generally steep and rocky and wetlands occur on low-lying land on alluvial fans and fluvial or lacustrine terraces.

Kinbasket Reservoir fills in the spring and is typically full by the mid- to late-summer (Figure 4-2). Although there is some year-to-year variation, the general pattern is consistent. In 2012 and 2013 Kinbasket was filled beyond the normal operating maximum (i.e., > 754.38 m ASL) for the first time since 1997.

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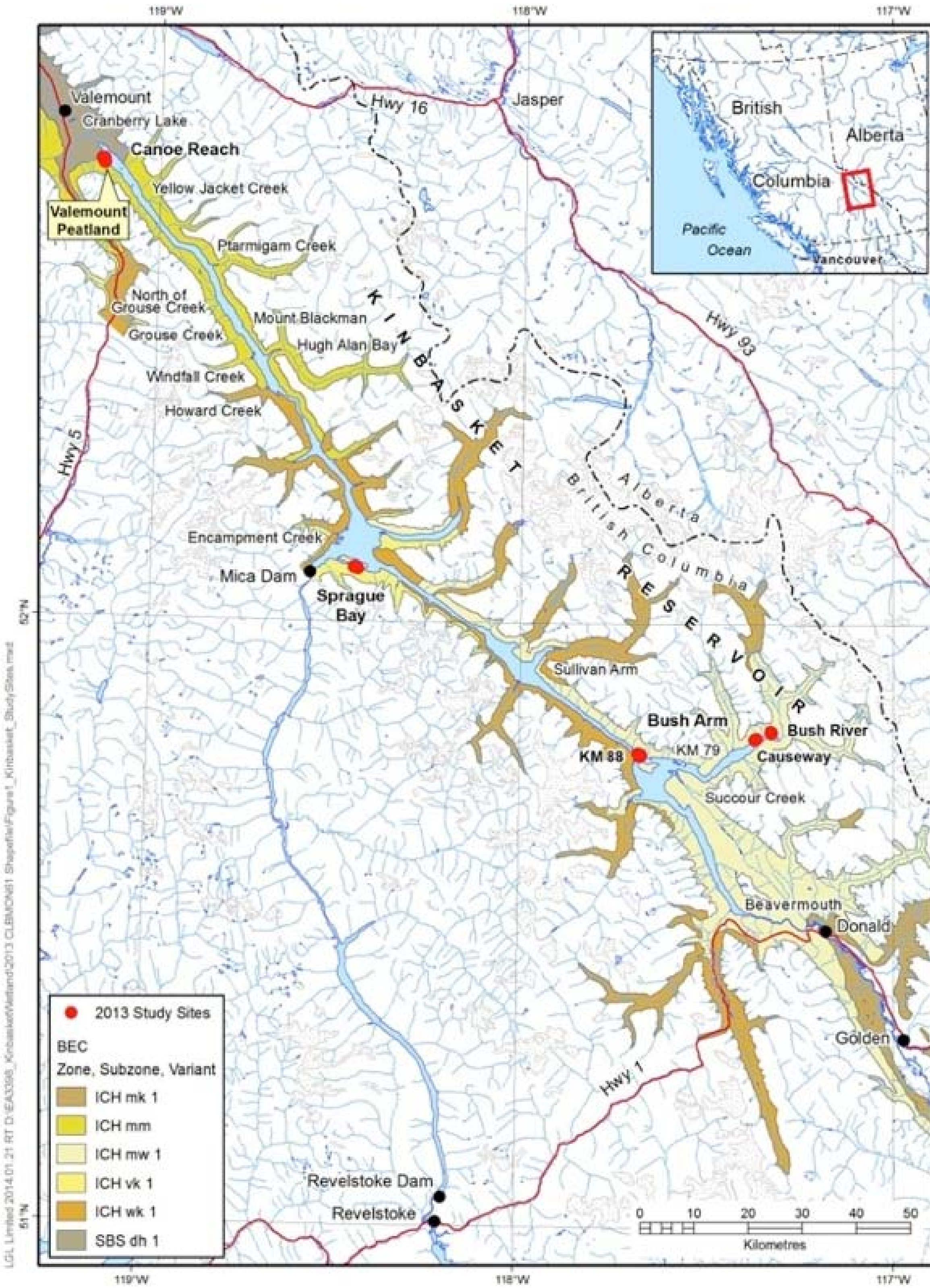

Figure 4-1: Location of the CLBMON-61 index sites in Kinbasket Reservoir, 2013

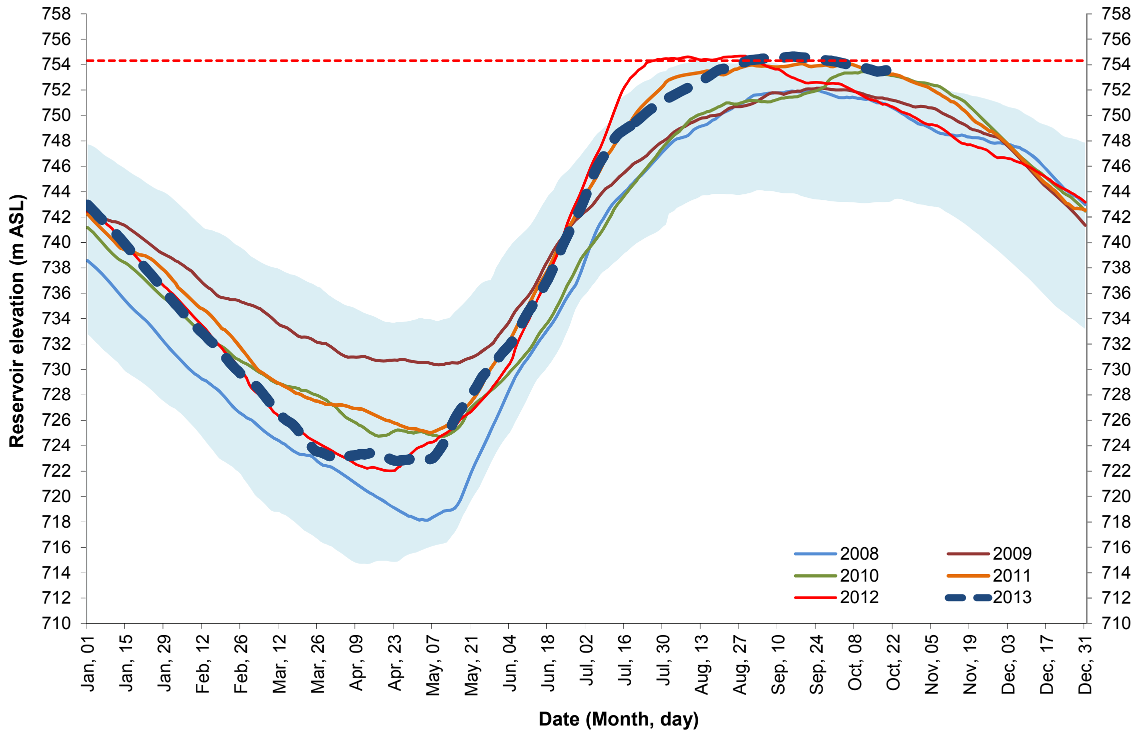

Figure 4-2: Kinbasket Reservoir hydrograph for the period 2008 through 2013. The shaded area represents the 10th and 90th percentile for the period 1976 to 2013; the dashed red line is the normal operating maximum

4.2 Index Sites

Four index sites were identified in Year 1 for long-term monitoring: the Valemount Peatland in Canoe Reach, Sprague Bay in Mica Arm, and Km 88 and the Bush River wetlands in Bush Arm (Figure 4-1).

These sites were selected for the following reasons:

1) they represent the geographic distribution of wetland communities across the study area;
2) they occur across a broad range of environmental conditions (e.g. climate, surficial geology, water chemistry);
3) both the aquatic and terrestrial wetland types occur at each site;
4) suitable aquatic and terrestrial reference wetlands occur nearby;
5) the sites occur across a relatively low elevation gradient, which increases the area between elevation bands for sampling;
6) the sites represent the most intact and highest value wetland habitat in the reservoir.

Descriptions of the index sites along with, maps, and representative images are provided in Appendix 10.2.

5.0 METHODS

5.1 Approach

A detailed description of the monitoring program is provided in Adama et al. (2013) and Adama and Hawkes (2013). In brief, the potential impacts resulting from Mica Units 5 and 6 will be assessed following a Before-After-Control-Impact (BACI) design (as prescribed by BC Hydro) with two years of pre-impact sampling (2012 and 2013) and three years post-impact sampling (2015, 2016, and 2017). The study will employ a repeated measures model to compare community composition, productivity, and physiochemical parameters collected in terrestrial and aquatic wetlands over the study period. Comparisons will be made across “target”, “control”, and “reference” sites (for definitions see Appendix 10.1)

To identify those variables that will most likely respond to changes in reservoir operations, we compared data across the 1-meter elevation gradient (terrestrial wetlands), between DDZ and reference ponds (aquatic wetlands), or in response to inundation (water physicochemistry and pond metabolism). Given the limitations of the monitoring program, we reason that variables that do not differ consistently across these strata are not likely to be useful for assessing changes in wetland integrity associated with Mica 5 and 6.

5.2 Terrestrial Wetland Sampling.

A modified belt-line transect was used to sample terrestrial wetland communities at four elevation strata: lower control (LC) 752–753, target 753–754, and upper control (UC) 754–755 m ASL within the reservoir and in reference sites above 755 m ASL. The location of transects were determined in the four index sites using GIS and were paired at each elevation band. An example of this layout is shown in (Figure 5-1). Transects were repositioned in the field when the GIS coordinates did not correspond to the appropriate sampling location, elevation band or habitat type.

20 m belt transect were sampled along the entire length using ten 2 m X 0.5 m quadrats (Figure 5-2). As standard procedure, we sampled with our backs to the reservoir to ensure the same side of the transect is sampled every year. To assess forest and shrub cover, a circular 100 m^2 plot was established at each transect end using a tape measure and the cover of woody species within the circular plots was visually estimated (Figure 5-2). The location of each transect endpoint (0 m and 20 m) was georeferenced using a Garmin handheld GPS. Rebar was installed into the ground at the transect endpoints and fitted with an orange safety cap.

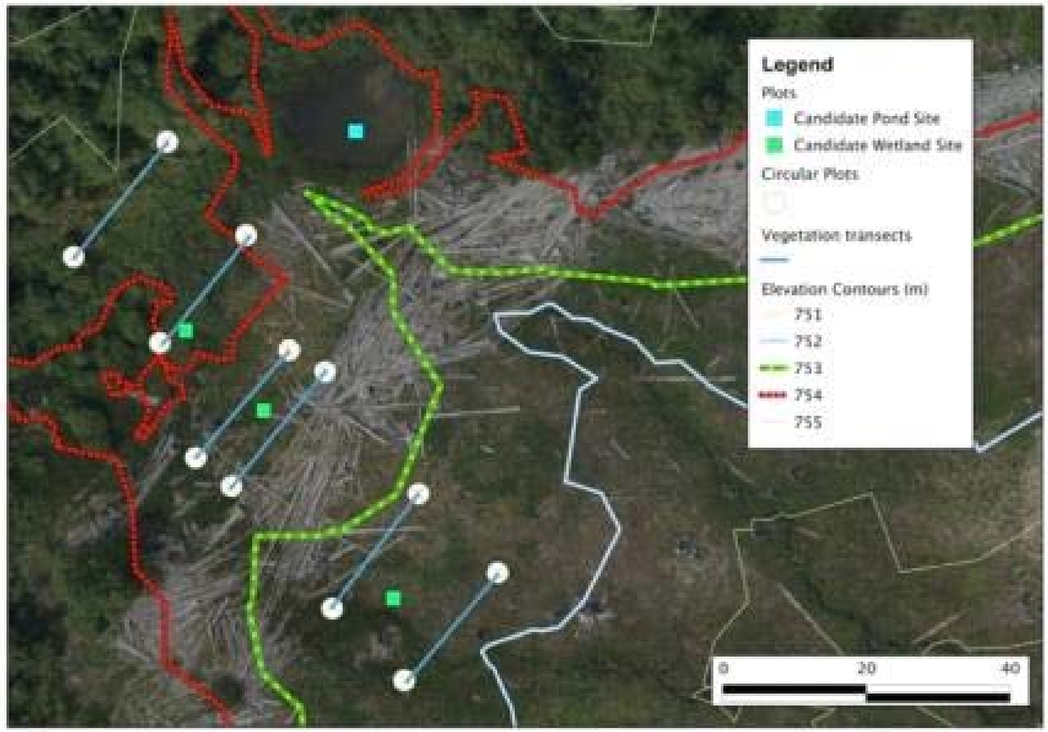

Figure 5-1: Layout of paired 20-meter vegetation transects in Kinbasket Reservoir across three elevation bands (752–753, 753–754, and 754–755 m ASL). A similar pair of transects was established for reference sites outside the reservoir (not shown)

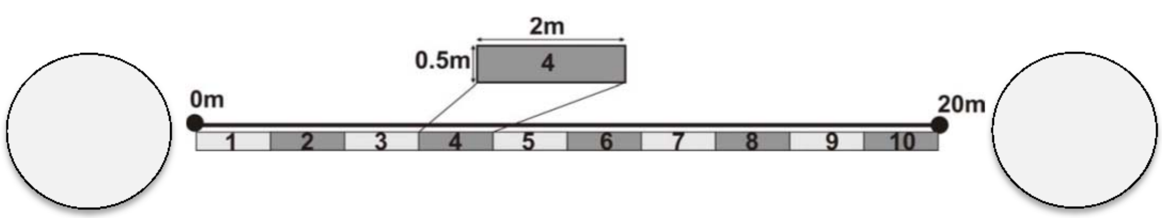

Figure 5-2: Schematic of the belt-line transect quadrat method and 100m^2 circular plots (5.64 m radius) used to sample wetland communities in Kinbasket Reservoir. Note: not drawn to scale

Data sheets were based on the FS882 (3) Vegetation Form (RISC 2010; Appendix 10.3). Vegetation within each quadrat was identified to species, or in some cases, to genus, and the per cent cover to the nearest 1 per cent was visually estimated. Species occurring below 1 per cent were recorded as trace (T) and later converted to 0.1 per cent in the database for numerical analyses. For each species, the total cover was averaged across the ten quadrats to derive a mean cover value along each transect. Cover estimates were stratified by the following vegetation layers:

A: Trees (woody plants greater than 10 m tall)
B1: Tall Shrubs (woody plants 2 m to 10 m tall)
B2: Low Shrubs (woody plants less than 2 m tall)

C: Herbs (forbs and graminoids)
D: Moss, lichen, and seedlings

The ground surface was categorized as either bare soil (mineral, sand, or fines), coarse wood debris, rock, dead organic material, live organic material, or water, and the per cent cover of each surface type recorded for each quadrat. Where possible, a shallow soil sample (less than 50 cm) was collected at one end of each transect using a soil auger. Samples were bagged and labeled for future analysis.

For a visual portrayal of vegetation conditions, photographs were taken at each quadrate along the length of transect and in each of the four cardinal directions at the transect ends. Photos taken of the same vegetation communities over time will provide visual evidence of changes in vegetation and ground covers. The file numbers of all photos were recorded on data forms and images were coded by project number, year, month, date, and photo number (e.g., CLBMON61_20130622_DSC0809.jpg). Additionally, the date and time of the image capture is retained in the EXIF image data.

Terrestrial wetlands were classified following the classifications of Hawkes et al. (2007, and 2010) or MacKenzie and Moran (2004). Communities were classified manually either in situ or after reviewing summaries of the vegetation data collected.

5.3 Aquatic Wetland Sampling

5.3.1.1 Water Physicochemistry

Wetland area, hydrology, and chemistry are essential data for assessing changes in wetland integrity and provide valuable information for interpreting biological data, verifying wetland classification, and diagnosing potential stressors (Finlayson and Davidson 1999; Mitsch and Gosselink 2007; US EPA 2008). Parameters monitored in 2013 included water depth, water transparency, temperature, pH, conductivity, and dissolved oxygen.

Sample stations were established along a transect that bisected the wetland using a small 2.5 m inflatable boat. At a minimum, three sampling stations were established in each pond. At each station, point samples of water temperature, dissolved oxygen, conductivity, and pH were recorded at depths of 10 cm and 30 cm below the surface of the water. Water temperatures, dissolved oxygen, and conductivity measurements were obtained using a YSI model 85 digital multi-parameter meter. pH was obtained using an Oakton 35423-10 EcoTestr pH2. Water transparency was recorded using a 120 cm transparency tube (Dahlgren et al. 2004) and measured to the nearest centimetre. Transparency above 120 cm were recorded as “> 120 cm”.

Water depth was measured using a weighted tape measure and recorded to the nearest cm. Where possible, organic muck depth was estimated by pushing a D-net handle into the sediment as a probe until met with stiff resistance. In deeper water/sediments it was not possible to obtain these measurements if the combined muck and water depth exceeded the length of the net handle (2.75 m).

The presence of wood debris in benthic substrate was determined by probing the surface of the substrate and recording whether the probe struck wood. Four probes were taken at each station at each corner of the boat. Sediment information including texture, colour, and sediment type: organic muck (OM), wood, coarse organic matter (CO), or mineral sediment (MS) was obtained from Ponar grabs.

At each aquatic sampling station, photographs were taken in the four cardinal directions.

Where feasible, photos were also taken from the north, south, east, and west shoreline of the wetland; however, some ponds were too large or too irregular in shape to be photographed this way. A sample data form is provided in Appendix 10.3.

Conductivity (Onset HOBO U24-001) and dissolved oxygen (PME MiniDOT or Onset HOBO U26) sondes were installed in the index ponds to collect continuous data for monitoring changes in water physicochemistry and aquatic metabolism. PME MiniDOT and Onset U26 sondes both employ optode sensors that are less prone to drift and require less frequent calibration than membrane electrodes such as the galvanic and polarographic sensor used in YSI 85 multimeters (Wilcock et al. 2011). The stated accuracy of the PME MiniDOT is +/- 5 per cent dissolved oxygen and +/- 0.1 °C; the accuracy of the Onset HOBO U26 is 0.2 mg/L dissolved oxygen and 0.2°C.

Sondes were installed in 8 ponds (4 in reference ponds and 4 in DDZ ponds) between June 4 and June 13, 2013. The sondes were affixed to ¾" rebar using a pipe clamp between 43 cm and 53 cm below the water surface at depths of 78 to 101 cm. The rebar was fitted with an orange plastic safety (Figure 5-3). Sondes were factory programmed to record data every hour and calibrated as per manufacturer's instructions. They were retrieved from the field on November 20, 2013 and data were downloaded using the manufacture's software (Onset Hoboware and PME miniDOT software). Dissolved oxygen concentration were corrected for altitude (Staehr et al. 2010):

$$\text{Correction factor} = (0.0000005 \times \text{altitude}^2 - 0.0118 \times \text{altitude} + 99.979)/100$$

For quality assurance, we compared the values obtained from sondes to the point sample data acquired simultaneously during the field sampling.

Figure 5-3: Dissolved oxygen and conductivity sondes installed in an aquatic wetland (pond) in Kinbasket Reservoir

5.3.2 Aquatic Macrophytes

Aquatic wetland plants occur in three growth forms (emergent, planmergent, and submergent) making it impossible to sample them using a single technique. Several sampling methods were employed based on growth form and data requirements and are described below. Taxonomies used in the identification of aquatic wetland plant species included:

- George W. Douglas, Del Meidinger and Jim Pojar. 1998-2002. Illustrated Flora of British Columbia.
- Brayshaw, T. C. 2000. Pondweeds, bur-reeds and their relatives of British Columbia: aquatic families of monocotyledons. Royal British Columbia Museum.
- Johnson, D., L. Kershaw, A. MacKinnon, and J. Pojar. 1995. Plants of the western boreal forest and aspen parkland. Lone Pine Publishing, Edmonton, Alta.

Planmergent Communities

Planmergent (floating) communities were sampled in open water (away from the pond edges and beds of emergent communities) using a 1-m^2 quadrat along the port and starboard sides of the boat. The overall per cent cover of vegetation occurring within quadrats were recorded along with the individual cover of each species.

Submergent Communities

Submergent macrophyte communities were sampled in open water but using two methods. First, the per cent cover of each species was estimated visually using a viewing tube (Parsons 2001) in the 1-m^2 quadrats described above. Typically, this method is constrained by water depth and water transparency; however all ponds were less than 2 m in depth and sufficiently clear to sample. This technique was added to our sampling procedures to address concerns raised in 2012 regarding the grapnel method (Adama et al. 2013).

The second method employed a macrophyte grapnel made by binding two garden rakes together to create a double-headed rake (Figure 5-4). With a 5-m rope attached, the grapnel was tossed 1.5 m from the boat and was allowed to settle on the bottom of the pond. Once on the bottom, the grapnel was dragged for ~1 m capturing submergent vegetation within the tines of the rakes. Upon hauling the grapnel into the boat, overall vegetation abundance was estimated in per cent cover based on amount of vegetation that passed across the plane of the rake tines. The relative abundance of each species was determined by sorting through the vegetation and estimating its contribution to the total amount collected. Although grapnels have been widely used for collecting macrophyte sample (Alberta Environment 2006; Hawkes et al. 2011; Gunn et al. 2010, Yin and Kreiling 2011), in 2012 we found it difficult to obtain consistent grapnel samples in depths greater than 1.5 meters and suspected that samples in deeper ponds were not sampled effectively. In 2013, the DDZ pond at Km 88 was sampled by tossing the grapnel from the shore rather than from the boat, resulting in a much higher macrophyte yield from the grapnel.

At each sample station, two grapnel samples were collected (typically one from each side of the boat) as a measure of macrophyte biomass. Samples were bagged and labeled separately for biomass measurement. Biomass samples were stored in an ice cooler until the end of the field day when they were then transferred to a refrigerator. In the lab, the samples were dried at 75 °C for 72 hours. Dry weight (g) of each sample was obtained from a digital balance.

Figure 5-4: Image showing a grapnel sample for aquatic macrophytes.

At each sampling station, macrophyte communities were typed using the planmergent and submergent cover data. As in Adama et al. (2013), we used the classifications of Mackenzie and Moran (2004) for planmergent and emergent communities and Pierce and Jensen (2001) for submergent communities. As many ponds had less than three species, our classifications were often based one or two dominant species.

5.3.3 Pelagic Invertebrates

Pelagic invertebrates were sampled at each station as an index of secondary productivity. Samples were obtained using a fine-meshed aquarium net following the methodology outlined in Fenneman and Hawkes (2012) and Adama and Hawkes (2013). At each sampling point, ten 1 m sweeps were performed at a depth of ~20-30 cm, five on each side of the boat. Samples were pooled into a single Whirl-Pak bag and preserved in 85 per cent ethanol.

Pelagic invertebrate samples were sorted and identified by Thilaka Krishnaraj following a modified Cabin protocol (RISC 2009). Each sample was sorted using a Marchant box (Marchant 1989) and a minimum of 200 invertebrates were extracted from each sample; in samples containing less than 200 specimens, then entire sample were sorted. Diptera, Coleoptera, Hemiptera, Ephemeroptera, Plecoptera, Trichoptera, Megaloptera and Neuroptera were keyed to family while other taxa were keyed to order or phyla level as per the CABIN protocol (McDermott 2012).

5.4 Data Analyses

5.4.1 General Analyses

Terrestrial wetlands were stratified by reach, index site, and elevation band, and comparisons were made using transects or circular plots as replicates. For belt-transects, substrate and vegetation data from nested quadrats were pooled and averaged; circular plots were treated as independent samples. Aquatic wetlands were stratified by reach, index site, and position; subsamples were pooled and averaged for each sampling station.

Data from terrestrial and aquatic wetlands were summarized with box plots using JMP (2013) or R (R Core Team 2013). Box plots are a convenient way of depicting data through quartiles without making any underlying assumptions about statistical distribution (McGill et al. 1978). Boxes represent between 25 per cent and 75 per cent of the data. The horizontal line inside the box is the median. The length of the boxes is their interquartile range (Sokal and Rohlf 1995). A small box indicates that most data are found around the median (small dispersion of the data). The opposite is true for a long box: the data are dispersed and not concentrated around the median. Whiskers extend above and below the box to the largest or smallest observations within the 1.5 interquartile range.

For all statically analyses including the determination of confidence intervals, a critical value of $\alpha = 0.10$ was used. This value was chosen due to the small number of index sites (n =4) and inherent natural variability associated with vegetation per cent cover, water physicochemistry, and invertebrate data.

5.4.2 Cover and Vegetation Data

Differences in cover data across strata were compared using the Kruskal-Wallis rank sum test as a non-parametric alternative to analysis of variance (ANOVA). Post-hoc pairwise tests were corrected for multiple comparisons with the Bonferroni adjustment (α = 0.10 / no. of comparisons). To assess relationships between vegetation cover and environmental variables, we performed non-parametric Spearman rank correlation tests using the 'rcorr' function in the 'Hmisc' package in R (Harrell and Dupont 2013; R Core Team 2013). Spearman correlation tests the null hypothesis that the ranks of one variable do not covary with the ranks of another variable. In contrast to the Pearson correlation, Spearman correlation is not restricted to linear relationships between variables, is less sensitive to strong outliers in the data, and is appropriate for datasets that are unlikely to meet the assumptions of parametric correlation (normality and homoscedasticity).

Species richness and diversity (Pielou's evenness, Shannon diversity index, and Simpson diversity index) were compared across strata. Species richness corresponds to the total number of species recorded in each sample, whereas Pielou's evenness (Pielou 1966) quantifies the similarity in species relative abundance for all species observed and is calculated:

$E = H/H_{max} = (-\Sigma\ (p_i \log p_i))/ \log q$, where q is species richness.

The more E tends towards 1, the more evenly the families are distributed throughout the community. Conversely, a value of E close to zero means that the community is dominated by a relatively small number of species (i.e., the distribution is uneven).

The Shannon and Simpson diversity indices provide information about community composition, combining both the number of species and their relative abundances. The Shannon index includes a "log" to the relative abundance (of species), which reduces the weight of abundant species relative to rare species. Simpson's index squares the relative abundance, which reduces the weight of rare species relative to abundant species.

Similarity in species composition across strata was calculated using the Sørensen similarity coefficient (Sørensen 1948), as follows:

$$\% \text{ Sørensen Similarity} = 2C / (A + B)$$

where A is the number of species present in site one, B is the number of species present in site two, and C is the number of species present in both site one and site two. The higher the % Sørensen Similarity values are between sites the more similar the species compositions are proportionally. A value of 100 indicates that the species compositions are identical whereas a value of 50 % indicates that the sites share 50% of the species. Value of zero indicates that there are no species in common across sites. This coefficient was chosen because it gives higher weight to species presences, which is more informative because species absences do not necessarily reflect environmental differences (Legendre and Legendre 2012).

To test for differences in species composition across strata we used Permutational Multivariate Analysis of Variance tests (PERMANOVA; Anderson 2001; McArdle and Anderson 2001). Where data was available for both 2012 and 2013, two-way PERMANOVA tests were used to test for temporal differences between sites. PERMANOVA analyses test the null hypothesis of no difference in species composition ($\alpha = 0.10$) between groups, based on the Sørensen similarity index. Tests were performed in PC-ORD v. 6.08 (McCune and Mefford 2011) with 4999 permutations. As these tests require a balanced data structure, sample units were randomly selected when missing data resulted in an unbalanced design (e.g., plots with no species).

5.4.3 Water Physicochemistry Data

Differences in water physicochemistry, pond sediment, and wood debris, between aquatic wetlands in and above the reservoir were compared using the Kruskal-Wallis rank sum test as a non-parametric alternative to analysis of variance (ANOVA).

Continuous dissolved oxygen (DO), temperature, and conductivity data were plotted to visually assess these data for obvious trends and anomalies. Since the date and time that the DDZ ponds were inundated were not known, the timing of inundation of the 753 m ASL elevation band was estimated using hydrometric data supplied by BC Hydro, a digital elevation model (2002), and abrupt changes in water physicochemistry near the approximate time of inundation. As we could not be entirely certain of the timing of inundation, we buffered our estimates by 10 days on either side. Differences in water physicochemistry in DDZ and reference ponds before and after the inundation date were explored with box plots and repeated measures ANOVA.

5.4.4 Diel dissolved oxygen data

Daily (diel) changes in DO concentrations can be used to determine primary productivity in aquatic ecosystems (Odum 1956; Staehr et al. 2010; Staehr et al. 2012). With recent advances in sonde technology, reliable metabolic rates including net ecosystem production (NEP), gross primary production (GPP), and ecosystem respiration (ER) can be determined from high frequency sampling of DO. This approach assumes that changes in oxygen concentration reflect the balance between daily photosynthesis and respiration. The production of DO occurs only during daylight hours via photosynthesis whereas respiration is the only metabolic process occurring at night. Thus, NEP, R, GPP can be calculated by measuring temporal changes in DO throughout a 24 hour period from the equations:

$$\Delta O_2/\Delta t = GPP - R + D$$

$$NEP = GPP - R$$

where $\Delta O_2/\Delta t$ is the change in oxygen concentration over time, and D is the exchange of oxygen with the atmosphere via diffusion (Odum 1956; Hoellein et al. 2013). NEP, R, and GPP are expressed in O_2 m^{-3} day^{-1}, which is equivalent to mg O_2 l^{-1} day^{-1}.

NEP and GPP (uncorrected for atmospheric diffusion) were calculated over two 10-day periods pre- and post-inundation buffered 10 days from August 15, 2013 (the estimated date of inundation) to ensure (1) reservoir levels prior to inundation did not affect aquatic metabolism, and (2) to allow sufficient time for the ponds to acclimate to inundation. Thus, the pre inundation period was July 27 to August 5 and the post inundation period was August 26 to September 4, 2013. Equations for calculating NEP and GPP are provided in Appendix 10.4.

Prior to calculating metabolic activity, DO was corrected for altitude (rather than barometric pressure) and data from sondes were reviewed to identify anomalous data. Two types of anomalous data were identified and omitted. First, data from the Sprague Bay DDZ pond was omitted as the pond drained prior to inundation providing unreliable DO data (

Figure 10-27). Second, positive NEP_{hr} values recorded during the hours of darkness were omitted, as photosynthesis cannot occur in the absence of light (Appendix 10.4). Differences between mean GPP, NEP, and R before and after inundation were assessed using the Kruskal-Wallis rank.

5.4.5 Pelagic Invertebrate Data

Taxonomic identification generated 38 distinct taxa (i.e., “lower-level taxa”; family-level and higher taxonomic rank); however, many of the lower-level taxa were infrequently collected among ponds. Analyses were also performed on “higher-level taxa” consisting of 19 distinct taxa by pooling family-level taxonomic determinations within their respective orders. The abundance of invertebrates was standardized to a 100 mL sample volume for analyses. When possible, confidence intervals were provided for means to allow for comparisons of the variance.

To assess patterns in taxon abundance, we calculated and compared mean taxon abundance for 38 lower-level taxa and for the 19 higher-level taxa using Kruskal-Wallis rank sum tests and Bonferroni-adjusted post-hoc pairwise comparisons. To test for contingency in invertebrate occurrence (presence/non-detection) on environmental variables, we created contingency tables and performed Chi-square analyses. Chi-square analyses test the null hypothesis (H_0) of independence between the two descriptors (i.e., the frequency of observations of invertebrates is not influenced by environmental conditions, and thus invertebrates are distributed randomly). For tests with fewer than seven observations per cell, the Fisher’s exact test was used to correct p-values of Chi-square analyses. Both Chi-square and Fisher tests were performed in the base stats package of R (R Core Team 2013).

Dominance plots were created to identify qualitative changes in taxon dominance between DDZ and reference ponds for 2012 and 2013 invertebrate data. Only taxa that occurred in both DDZ and Reference ponds were included in plots, as taxon non-detection may not equate absence. For each higher-level taxon (19 total), we plotted taxon rank abundance by taxon rank frequency of occurrence in reference and DDZ ponds. Taxa that occur near the plot origin are those that are most abundant and occur most frequently among ponds. Indicator Species Analysis was not performed as few relevant associations were found.

We compared lowest-level taxon richness and diversity (Pielou's evenness, Shannon diversity index, Simpson diversity index) for DDZ and Reference ponds for both years (2012-2013), providing confidence intervals for interpretation. We also examined diversity indices for each pond separately to compare pond position in each reach in each year.

Two-way PERMANOVA was performed to assess the effects of pond position, reach, and the interaction (position*reach) on species composition. To achieve a balanced design, we randomly selected 3 sample replicates from each pond (n = 24 samples). Boxplots were created to display the variation in community composition within and between ponds by plotting the Hellinger distance to group centroids for each pond (Hellinger distance varies from 0 to 1, where 1 is the greatest dissimilarity in taxa).

To assess patterns in invertebrate taxa assemblages, we employed a classical, unconstrained ordination approach. We performed a PCA to determine the major compositional variation in pelagic invertebrate taxa, and to examine relationships with environmental variables. Taxa abundances were Hellinger-transformed, whereby each taxon observation was relativized by the total taxon abundance, and square root transformed (Legendre and Gallagher 2001; Legendre and Legendre 2012). Correlations with environmental variables were determined by 999 permutations. Principal components analyses were performed using the vegan package in R (v. 3.0.1; R Core Team 2013).

6.0 RESULTS

6.1 2013 Sampling Effort and Reservoir Levels

Power generation at Mica Dam was halted for much of 2013 to facilitate the construction of the new turbines (Mica 5 and 6). Consequently, Kinbasket Reservoir was forecast to exceed full pool so sampling commenced three weeks earlier than in 2012 to ensure all sites could be accessed. In addition, Kinbasket Reservoir was slower to fill and reservoir levels did not impede our ability to access the monitoring sites (as in 2012). Sampling occurred from June 22 to July 17 (Figure 6-1). For comparison, Table 6-1 shows the disparity in sampling effort across the four index sites between Years 1 and 2. Full pool (754.4 m ASL) was reached on September 02, 2013 (Figure 6-1). Kinbasket Reservoir peaked at 754.63 m ASL on September 16, 2013 and remained above 753 m ASL until mid-November, delaying the retrieval of the water chemistry sondes until November 20, 2013.

In recent years, reservoir elevations have reached and remained at high levels. In 2007 and from 2011 to 2013, the elevations of Kinbasket Reservoir have greatly exceeded the 1987 to 2006 historical average (Figure 6-2). In 2013, the 753 m ASL elevation band was inundated for 5.2 times longer (87 days) than the 1987 to 2006 historical norm (16.6 days). This is problematic as 2013 and 2012 were to provide baseline conditions for assessing change in wetland composition and productivity as required under the BACI study design. Further, residual effects of high reservoir levels may carry over into the post-impact period, confounding the interpretation of results in future years.

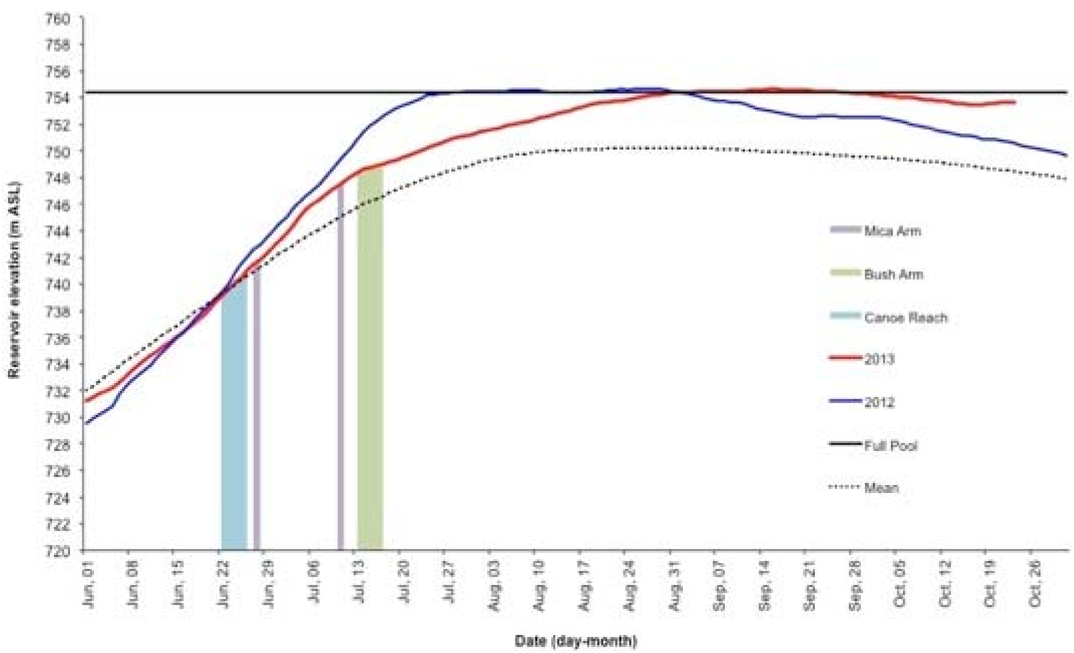

Figure 6-1: Kinbasket Reservoir elevations and the timing of field reconnaissance (April–June) and in situ monitoring (July–August) in 2013. Mean reservoir elevations were calculated from 1977–2013.

Table 6-1. Sampling effort at the four-index sites in 2012 and 2013. DDZ = drawdown zone, REF = reference

Wetland Type	Index Site	Elevation Strata	2012*	2013
Aquatic	Valemount Peatland	DDZ	8	8
		REF	5	5
	Sprague Bay	DDZ	7	5
		REF	6	5
	Km 88	DDZ	3	5
		REF	**0**	5
	Bush River	DDZ	**0**	3
		REF	**0**	5
		Sites Sampled	**5**	**8**
		Stations Sampled	**29**	**41**
Terrestrial	Valemount Peatland	LC	0	4
		Target	0	4
		UC	0	4
		REF	1	4
	Sprague Bay	LC	0	2
		Target	0	2
		UC	0	2
		REF	0	2
	Km 88	LC	0	2
		Target	1	2
		UC	1	2
		REF	0	2
	Bush River	LC	0	2
		Target	0	2
		UC	0	2
		REF	0	2
		Elevation bands Sampled	**3**	**20**
		Transects Sampled	**4**	**40**

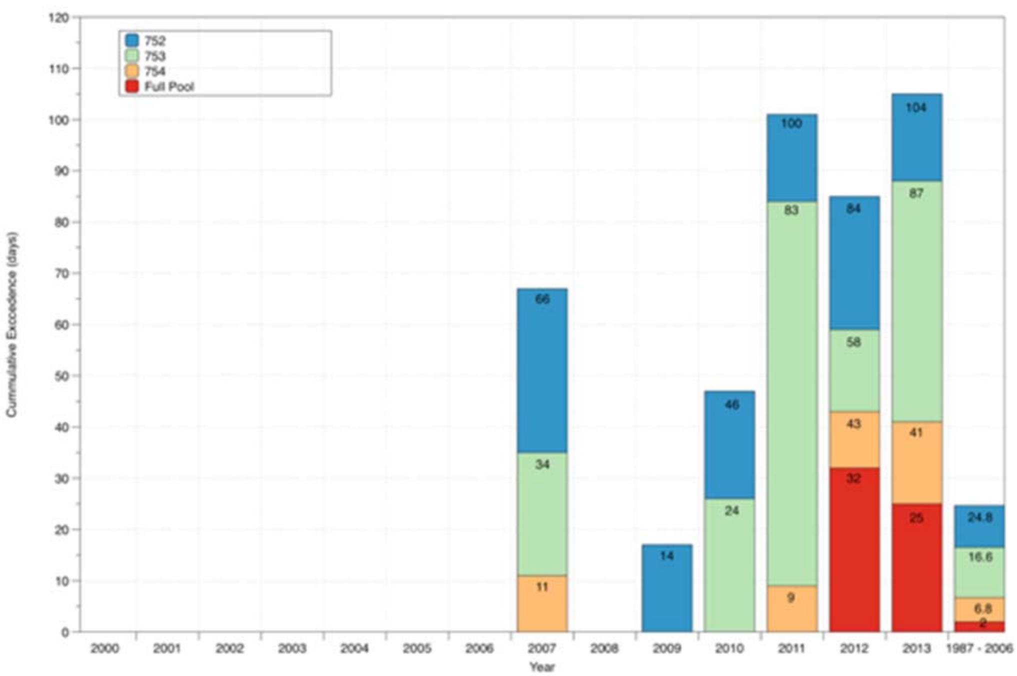

Figure 6-2. Cumulative days Kinbasket Reservoir exceeded elevations of 752, 753, 754, and 754.38 m ASL (full pool). Mean values from 1987 to 2006 are provided to reflect the historical average.

Seasonal temperatures affect water physicochemistry parameters such as water temperature, pH, conductivity, and dissolved oxygen. Mean daily temperatures (MDT) during the sampling period were higher than the 1971-2000 daily mean: 14.8 ºC versus 14.5 ºC in June; 18.0 ºC versus 16.6 ºC in July (Figure 6-3). Extended plots of MDT and precipitation from June 15 to October 31, 2013 are provided in Figure 10-15 (Appendix 10.5). A change in weather with colder temperatures and more precipitation occurred in mid-August, coinciding with the inundation of the 753 m ASL (Figure 10-15).

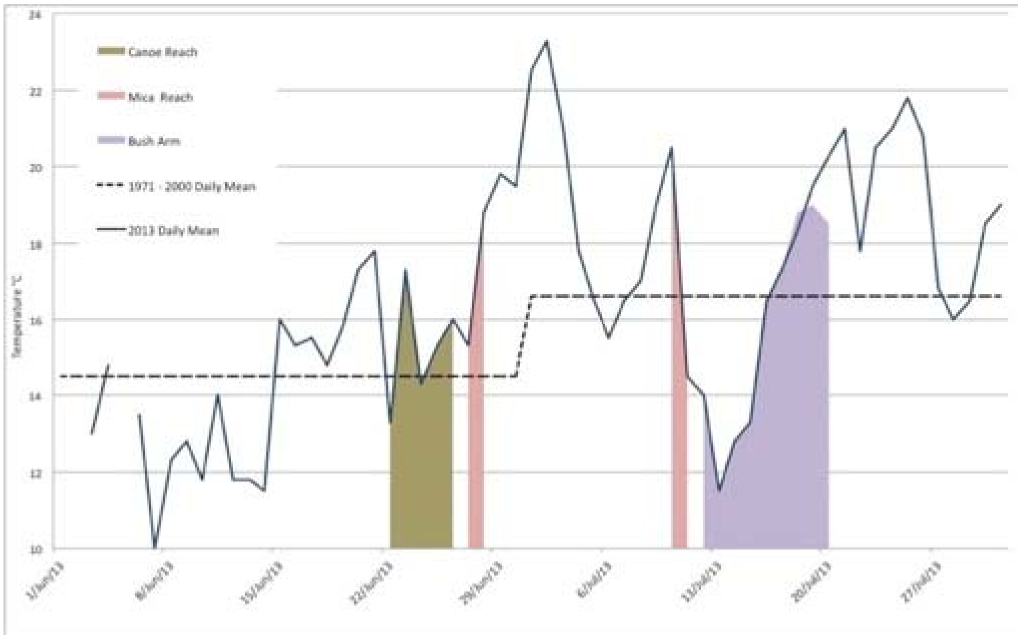

Figure 6-3: Mean daily temperatures at Mica Creek during June and July 2013. Shading indicates the timing of sampling sessions and location. Temperatures for Mica Cr. are shown as it located approximately mid-point in the study area. The dotted line indicates the 1971 – 2000 mean temperature for June and July.

6.2 Terrestrial Wetland Vegetation

6.2.1 Community Classification

Seventeen vegetation associations were identified in 2013 using the classifications of MacKenzie and Moran (2004) and Hawkes et al. (2007) (Table 6-2). Detailed descriptions of the vegetation communities are provided in Appendix 10.2 along with representative images. Several communities fit only loosely into the existing classifications and one community eluded classification altogether (Table 6-2). This is not surprising given the inherent problems with ecological classifications, the complexity and diversity of wetland ecosystems, and the response of vegetation to an unnatural disturbance regime.

Table 6-2. Terrestrial wetland associations corresponding to the sites and elevation strata sampled in Kinbasket Reservoir, 2013

Reach	Site	Stratum	Wetland Association	Source
Canoe Reach	Valemount Peatland	Lower Control	Swamp Horsetail (SH)	Hawkes *et al.* 2007
		Target	Swamp Horsetail (SH)	Hawkes *et al.* 2007
		Upper Control	Swamp Horsetail (SH) and Driftwood (DR)	Hawkes *et al.* 2007
		Reference	Black Spruce–Buckbean–Peat Moss Bog (Wb11) Scrub birch–Buckbean–Shoresedge (Wf07)	MacKenzie and Moran 2004
Mica Arm	Sprague Bay	Lower Control	Pink Spiraea-Carex Sitchensis (Ws50)	MacKenzie and Moran 2004
		Target	Mountain Alder-Pink Spiraea-Sitka Sedge (Ws02)	MacKenzie and Moran 2004
		Upper Control	Unclassified	
		Reference	Scheuchzeria-Peat-moss (Wb12)	MacKenzie and Moran 2004
Bush Arm	Km 88	Lower Control	Buckbean–Slender Sedge (BS) Slender Sedge–Buckbean (Wf06)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
		Target	Willow–Sedge (WS)	Hawkes *et al.* 2007
		Upper Control	Willow–Sedge (WS) Scrub Birch–Water Sedge (Wf02) Lodge Pole Pine–Water Sedge –Peat Moss (Wb07)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
		Reference	Shore Sedge – Buckbean–Hook-moss (Wf08) Hudson Bay Clubrush – Red hook moss (Wf10)	MacKenzie and Moran 2004
	Bush River	Lower Control	Driftwood (DR)	Hawkes *et al.* 2007
		Target	Swamp Horsetail (SH) and Willow-Sedge (WS) Swamp Horsetail–Beaked Sedge (Wm02)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
		Upper Control	Willow–Sedge (WS)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
		Reference	Water Sedge–Beaked Sedge Fen (Wf01)	MacKenzie and Moran 2004

Within the reservoir, the influence of fluctuating water levels was reflected in the composition of wetland communities. Communities in the lower control elevation band tended to be dominated by graminoids, wood debris, or *Spiraea douglasii* – a species that is characteristic of disturbed water receiving sites and fluctuating water tables (Klinka et al. 1989). The upper control elevation band and the reference communities were considerably more complex and diverse than the communities observed at lower elevation communities and were typified by fen, bog and riparian associations (Table 6-2). As expected, communities in the target elevation band were typically transitional between the lower and upper control elevation bands. Exceptions to this were the communities in the Valemount Peatland, where communities within the reservoir were dominated by the Swamp Horsetail (SH) association across all elevations. In addition, accumulations of wood debris in the upper control elevation band likely suppressed the development of more complex communities such as those observed in the other index sites (Appendix 10.2).

6.2.2 Vegetation and Substrate Cover

In general, all of the vegetation layers increased in per cent cover with elevation (Table 10-2; Figure 10-16). Between 752 to 754 m ASL, shrub cover increased by approximately 200 per cent with each 1–metre elevation band from 7.3 to 33.5 per cent, respectively. Shrub cover in the target elevation band was 11.6 per cent. Herb cover varied from 40.9 to 51.4 per cent across the elevation strata. Trees were virtually absent in all elevation bands (< 1 per cent cover). Overall, the strongest differences in vegetation cover across elevation bands were observed for the shrub layer (Table 10-2).

General vegetation cover (e.g., tree, shrub, herb, and moss layers) also varied by index site (Table 10-2; Figure 10-16). At Km 88, average herb per cent cover was nearly twice that of other index sites. Shrub cover was fairly consistent across sites with the exception of the Valemount Peatland, where shrub growth in the upper control elevation band was supressed by high wood debris cover. The per cent cover of the four vegetation layers were all negatively correlated with wood cover (Table 10-3). Herbs and mosses were negatively correlated with dead organic and mineral soil substrate cover. Logically, most layers were positively correlated with live organic matter.

Substrate (ground) cover differed among elevation bands and sites (Table 10-4; Figure 10-17). Per cent cover of live organic matter tended to increase with elevation, while no differences in dead organic matter cover were found (Table 10-4; $p_{LIVE} = 0.009$; $p_{DEAD} > 0.10$). Overall, wood cover was significantly higher in the reservoir than in reference transects (13.2 versus 0.4 per cent; $p = 0.014$).

The upper control had the greatest cover of wood (20.1 per cent; Table 10-4), which was due to large accumulation observed in the Valemount Peatland (57.6 per cent cover at 754 m ASL; Appendix 10.2.1). Large accumulations of wood debris also occurred on the downstream side of the Bush River causeway at 752 m ASL (51.4 per cent cover; Appendix 10.2.4). Wood debris cover in the 753 m ASL transect (3.4 per cent) was lower than either the 752 or the 754 m ASL by factors of 4.6 and 6.6, respectively.

The presence of mineral soils at the Bush River site (Table 10-4) was indicative of the Bush River floodplain, where sediment deposition from the silt laden Bush River occurs. Consequently, the Bush River floodplain tends to supports riverine type wetlands and riparian vegetation (Appendix 10.2.4). The higher presence of standing water at Sprague Bay was a function of the Beaver dam and floating fen complex that comprise this wetland (Appendix 10.2.2).

6.2.3 Species Richness and Diversity

Within the reservoir, the number of shrub species increased with elevation by approximately 200 per cent with each successive elevation band (10, 18, and 35 species, respectively); reference sites supported 23 species. Results of diversity analyses for shrubs were consistent for circular plots and transects (Figure 10-18). Species richness, Shannon diversity, and Simpson diversity were significantly different ($\alpha = 0.10$) across elevation bands. The target elevation band was transitional between the lower control and upper control elevation bands. Species richness, evenness, and diversity of the shrub community were least similar between Bush Arm and Mica Arm and both sites were more similar to Canoe Reach than to each other (Table 10-5).

The number of herb species increased with elevation across the 752, 753 and 754 m ASL elevation bands (46, 54 to 72 species, respectively); reference sites had 73 species. Species richness and diversity of herbs were generally most variable in the upper control elevation band (Figure 10-19). Species richness was highest at Km 88 but evenness and species diversity were similar across sites. Across reaches, Bush Arm had the highest species richness (Table 10-6). This result is consistent with those obtained under CLBMON-10 (Hawkes et al 2013a) and is likely due to the calcareous soils found in Bush Arm. Calcareous soils are known contribute increased species richness though increased phosphorus availability (Brady and Wiel 2008; Ewlad 2003; Peet et al 2003).

6.2.4 Species Composition

Sorensen similarity coefficients for shrubs varied from 23.5 to 60 per cent across elevation bands (Table 6-3). Adjacent elevation bands were always more similar to one another (50 to 60 per cent of species shared) than more distant elevation bands (23 to 44 per cent of species shared). Differences in shrub composition across elevation bands were significant ($p = 0.01$). Shrub communities in the lowest elevation band and reference sites were most distinct ($p < 0.009$; Table 6-4). Shrub composition also differed across index sites (Table 6-3; Table 6-4). The greatest differences in shrub composition occurred between Bush River and Valemount Peatland sites (4.3 per cent species shared). All reaches were significantly different from one another in their shrub composition, with only 25 to 32 per cent similarity (Table 6-3; Table 6-4).

Table 6-3. Shrub species composition in 2013 transects by elevation band, site, and reach. Sørensen similarity coefficients (%) indicate the proportion of species that occur across two sites. Bold indicates values greater than 50 per cent.

Treatment Levels Compared		Sørensen similarity Coefficient (%)
ELEVATION	LC vs. Target	**50.0**
	LC vs. UC	24.2
	LC vs. Reference	23.5
	Target vs. UC	**51.2**
	Target vs. Reference	44.4
	UC vs. Reference	**60.0**
SITE	Bush River vs. Km 88	28.6
	Bush River vs. Sprague	27.3
	Bush River vs. Valemount Peatland	8.0
	Km 88 vs. Sprague	27.3
	Km 88 vs. Valemount Peatland	32.0
	Sprague vs. Valemount Peatland	31.6
REACH:	BUSH vs. CANOE	28.6
	BUSH vs. MICA	25.0
	CANOE vs. MICA	31.6

Table 6-4. Evaluation of differences in shrub composition in 2013 transects. Results of multiple one-way Permutational Multivariate Analysis of Variance (PERMANOVA) tests, based on 4999 permutations using the Sørensen index (presence-absence). Composition was calculated on *n* randomly selected replicates of transects for each variable of interest, where *n* = (1+ total d.f.) / number of levels of each factor. M.C. = multiple comparisons, corrected to *α* = 0.10

Source	d.f.	SS	MS	F	p-value	M.C.
ELEV band	3	2.100	0.700	2.263	**0.009**	
Residual	24	7.422	0.309			Lower Control ≠ Reference
Total	27	9.522				
SITE	3	5.082	1.694	9.788	**< 0.001**	all ≠, except Sprague = Valemount Peatland
Residual	20	3.461	0.173			
Total	23	8.542				
REACH	2	3.271	1.635	7.423	**< 0.001**	
Residual	15	3.305	0.220			all ≠
Total	17	6.576				

Results of the herb composition analyses were generally consistent with the shrubs analysis (Table 6-5; Table 6-6). Adjacent elevation bands were always more similar to one another and communities in the lowest elevation band and reference sites were most distinct. In comparison to shrubs, herb communities tended to be more similar across elevation bands. This suggests that shrubs may provide greater resolution for detecting the impacts of reservoir operation over time. Herb composition also differed among sites (Table 6-3; Table 6-4).

Table 6-5. Evaluation of differences in herb composition in 2013 transects. Results of a multiple one-way PERMANOVA tests based on 4999 permutations using the Sørensen index (presence-absence). Composition was calculated on *n* randomly selected replicates of transects for each variable of interest, where *n* = (1+ total *d.f.*) / number of levels of each factor. M.C. = multiple comparisons, corrected to α = 0.10.

Source	d.f.	SS	MS	F	p-value	M.C.
ELEV band	3	1.288	0.429	1.719	**0.02**	
Residual	36	8.995	0.250			1 LC ≠ 4 Ref
Total	39	10.283				
SITE	3	3.266	1.089	5.399	**< 0.001**	
Residual	28	5.645	0.202			Sprague = Valemount Peatland; otherwise ≠
Total	31	8.911				
REACH	2	1.806	0.903	4.545	**< 0.01**	
Residual	21	4.172	0.199			Bush ≠ Canoe; Bush ≠ Mica
Total	23	5.979				

Table 6-6. Herb species composition (per cent Sørensen Similarity) in 2013 transects by elevation band, site, and reach. VP = Valemount Peatland

Treatment Levels Compared		Sørensen Coefficient (%)
ELEVATION	LC vs. Target	**62**
	LC vs. UC	45.8
	LC vs. Reference	46.7
	Target vs. UC	**55.6**
	Target vs. Reference	**51.6**
	UC vs. Reference	**67.1**
SITE	Bush River vs. Km 88	**50.4**
	Bush River vs. Sprague	32.6
	Bush River vs. Valemount Peatland	28.3
	Km 88 vs. Sprague	40.7
	Km 88 vs. Valemount Peatland	42.0
	Sprague vs. Valemount Peatland	**51.3**
REACH:	BUSH vs. CANOE	39.4
	BUSH vs. MICA	36.6
	CANOE vs. MICA	**51.3**

6.2.5 Indicator Species Analyses

Indicator Species Analyses of shrub (ISA) and herb communities identified only three herbs and no shrub species for the lower and target elevation bands (Table 10-7; Table 10-8). *Scirpus microcarpus* and *Utricularia intermedia* were indicators of the lower control elevation band (*INDVAL* = 40.5, p< 0.10 *INDVAL* = 29.2, p = 0.07). These wetland obligates were present in transects in the Valemount Peatland and Sprague Bay and reflect the influence of presence of standing water and saturated soil condition at these sites (Appendix 10.2). *Scirpus microcarpus* is also an indicator of nitrogen rich soils (Klinka et al. 1989), which likely reflects the high organic content of soils at these sites. *Carex lenticularis ssp. lipocarpa* was the only potential indicator species identified for the target elevation band (*INDVAL* = 36.4, p = 0.04). *C. lenticularis ssp. lipocarpa* occurred at all index sites but was more common in the target elevation bands in the Valemount Peatland and Km 88.

The lack of shrubs as indicators in the lower control and target elevation bands suggests that the species occurring at the lower elevations are generalists and can survive across a wide variety of environmental conditions. More indicator species were identified in the upper control and reference transects (Table 10-7; Table 10-8). The identification of terrestrial herbs (*e.g., Cornus canadensis*, *Fragaria virginiana*, *Pyrola asarifolia*, *Rubus pubescens*, and *Viola glabella*) as indicators in the upper control elevation band indicates a shift to mesic conditions. In contrast, the identification of *Menyanthes trifoliate* and several species of *Carex spp.* in the reference communities signify more hygric conditions.

Among index sites, a greater number of indicator species were identified in Bush Arm (Bush River n= 18; Km 88 n = 23) than in either Sprague Bay (n = 10) or Valemount Peatland (n = 5), likely reflecting higher species richness and higher site fidelity in the Bush Arm transects (Table 10-7; Table 10-8).

6.3 Pond characteristics, water physicochemistry, and metabolism

With few exceptions (DO in the Sprague Bay DDZ pond, and pH, conductivity, and DO in the Valemount Peatland DDZ pond), pond physicochemistry data collected in 2013 were consistent with data from 2012 (Table 6-7). Of the ponds monitored, the Valemount Peatland DDZ pond appeared to differ the most between the two years.

Pond data did not differ between DDZ and Reference ponds in 2013 (Table 6-8) however, these results are not unexpected given small number of ponds sampled (n =8) and the overarching influence of local geology, hydrogeological processes, deil patterns, and weather on water physicochemistry. In 2012, the frequency of wood debris occurring in pond sediment differed between DDZ and reference ponds (p = 0.02, df = 15). The non-significant results for 2013 likely reflect the smaller sample size (p = 0.17, df= 7; Table 6-8).

Table 6-7. General pond characteristics and water physicochemistry data of aquatic wetlands sampled in 2013 and 2012 (in brackets). Asterisks denotes values that differed statistically across years (α = 0.10). REF = reference; DDZ = drawdown zone

Reach Site - Position	Survey Date	n	Elev. (m)	Area (m^2)	Depth (cm)	Mean Sed Depth (cm)	Mean pH	Mean Temp (°C)	Mean DO (mg/l)	Mean Conductivity (µS/cm)	Sediment Type**	Wood Debris (Proportion of wood strikes)
Bush Arm	**2013** **2012**						8.5 (8.2)			254.2 (253.1)		
Bush River Ref	July 16	5	~760	40735	67.4	73.8	8.7	14.6	10.7	128.0	OM/MS	0.0
Bush River DDZ	July 15	3	753	4361	44.7	28.7	8.8	19.4	9.1	228.1	OM/MS	0.0
Km 88 Ref	Aug 23	5	~770	2807	120.2	16.0	8.1	14.3	6.9	308.6	MS	0.0
Km 88 DDZ	July 16	5 (3)	752.7	887	114.6 (119.3)	8.8 (9.5)	8.4 (8.5)	17.4 (18.2)	9.6* (11.4)	331.8 (349.9)	MS	0.13 (0.17)
Mica	**2013** **2012**						6.3 (6.1)			30.9 (33.9)		
Sprague Bay Ref	July 10	5 (6)	757	9251	168.2 (178.0)	107.0 (118.0)	6.1* (5.9)	23.1 (22.9)	2.7 (2.7)	26.6* (22.2)	OM	0.0 (0.2)
Sprague Bay DDZ	July 10	6 (7)	753	378	68.6 (89.4)	74.5 (94.0)	6.4 (6.3)	25.4* (21.9)	5.2* (2.9)	36.0 (42.3)	OM	0.4 (0.14)
Canoe	**2013** **2012**						7.5 (7.2)			84.3 (93.6)		
Peatland Ref	June 24	5 (3)	757	830	85.7 (87.3)	116.1 (104.3)	7.2 (7.1)	9.7* (13.1)	8.2* (9.5)	43.5* (38.8)	OM	0.3 (0.08)
Peatland DDZ	June 26	8 (8)	753.5	8340	80.2 (76.0)	131.7 (101.7)	7.7* (7.2)	19.2* (15.8)	8.7* (6.4)	109.7* (80.5)	OM	0.38 (0.18)

* sediment type: M = Muck; MS = Mineral Sediment; OM = Organic Muck

Table 6-8. Results of Kruskal-Wallis tests for differences aquatic parameters values by position (REF = reference; DDZ = drawdown zone) in 2013. df = 7.

Environmental Attribute	REF (mean ± 90 CI)	DDZ (mean ± 90% CI)	p
Pond Area (ha)	1.37 ± 1.32	3.49 ± 4.7	0.386
Pond depth (cm)	109.3 ± 49.9	82.1 ± 34.0	0.549
Temp (°C)	15.5 ± 6.4	20.4 ± 4.1	0.238
Cond (µS/cm)	127.0 ± 152.0	176.4 ± 154.0	0.504
pH	7.5 ± 1.4	7.8 ± 1.3	0.423
DO (mg/l)	7.1 ± 3.9	8.0 ± 2.6	0.423
Sediment depth (cm)	78.2 ± 53.3	60.9 ± 64.3	0.549
Wood cover (freq)	0.08 ± 0.2	0.23 ± 0.22	0.172

Continuous dissolved oxygen, conductivity, and temperature profiles obtained from sondes generally showed clear diel patterns (Figure 10-23 to 10-30). Following inundation (August 15, 2013), diel fluctuations in temperature, conductivity, and DO were attenuated but the direction of change varied by reach. Sudden drops in water temperature and conductivity corresponded to weather events as observed in mid-July and again in mid-August (Figure 10-15; Figure 10-23 to 10-30). A major change in weather in mid-August coincided with the inundation of DDZ ponds, potentially obscuring the effects of inundation. As in 2012, the Sprague Bay DDZ pond drained prior to inundation as registered by the extreme daily fluctuations in DO, temperature, and conductivity. A change in weather in mid-August coincided with the inundation of DDZ ponds, potentially obscuring the effects of inundation. As in 2012, the Sprague Bay DDZ pond drained prior to inundation as registered by the extreme daily fluctuations in DO, temperature, and conductivity.

Estimates of aquatic metabolism in paired DDZ and Reference ponds over 10-day pre- and post-inundation periods are provided in Table 6-9. As we did not correct for atmospheric diffusion, we are not in a position to comment on the actual numerical values obtained; nonetheless, several trends were apparent.

In general, aquatic metabolism differed in DDZ ponds following inundation, whereas the values tended to remain static in Reference ponds. Metabolic metrics (GPP, NEP, and R) in reference ponds at Km 88 and Bush River were remarkably stable across the sample periods, while GPP and R declined in both ponds following inundation (Table 6-9). NEP also increased in the Bush River DDZ pond.

Patterns in the Valemount Peatland and Sprague Bay were less clear. In the Valemount Peatland, metabolic metrics in both the DDZ and reference ponds were different across the sample periods; however, the metrics shifted in opposite directions. GPP and R both decreased in the reference pond while GPP and R increased and NEP decreased in the DDZ pond (Table 6-9). These trends suggest that aquatic metabolism in these ponds were responding to different conditions or that the ponds responded differently to similar conditions. In the Sprague Bay reference pond, only GPP differed statistically across the two periods. Due to the pond draining, reliable data was lacking for the Sprague Bay DDZ pond.

Differences in metabolic rates also highlighted ecological differences between ponds. For example, metabolic metrics of the Bush River DDZ and reference ponds suggest that these ponds were ecologically dissimilar. Prior to inundation, values of R and GPP for the Bush River DDZ pond were over a magnitude greater than the reference pond (Table 6-9). The low metabolic rates of the Bush River reference pond, as well as lower macrophyte abundance, and high DO are indicative of an oligotrophic wetland. In contrast, high rates of GPP and R observe in the DDZ pond, high macrophyte abundance, and low DO indicate that this pond was more eutrophic. Again, trends were less clear in the Valemount Peatland and Sprague Bay ponds.

Table 6-9. Mean daily pond temperature, DO concentration, Gross Primary Production (GPP), Net Ecosystem Production (NEP), and Respiration (R) before (pre) and after (post) inundation of 753 m ASL. Units for GPP, NEP, R in grams O_2 m^{-3} d^{-1}; DDZ = ponds within Kinbasket Reservoir at 753 m ASL; REF = reference ponds. df =1, α =0.10, 90 per cent confidence intervals. Test statistics between paired DDZ and PER ponds prior to inundation are also provided*.

Site	Position	Period	Temp (°C)	DO (mg/l)	GPP $g\ O_2\ m^{-3}\ d^{-1}$	NEP $g\ O_2\ m^{-3}\ d^{-1}$	R $g\ O_2\ m^{-3}\ d^{-1}$	Trend
Bush River	REF	Pre	17.36	10.24	0.54 ± 0.64	-0.73 ± 0.27	1.27 ± 0.59	GPP =, NEP =, R=
		Post	15.65	10.45	0.63 ± 0.37	-0.73 ± 0.35	1.36 ± 0.49	
					p = 0.47	p = 0.91	p = 0.76	
	DDZ	Pre	20.56	4.64	11.27 ± 4.86	-3.79 ± 2.00	15.06 ± 6.73	GPP ⇓, NEP⇑, R⇓
		Post	9.02	2.40	1.06 ± 0.63	-0.72 ± 0.19	1.78 ± 0.40	
					p < 0.001	**p = 0.004**	**p < 0.001**	
	*DDZ and REF prior to inundation				**p < 0.001**	**p < 0.007**	**p < 0.001**	GPP, NEP, and R ≠ prior to inundation
Km 88	REF	Pre	16.75	6.55	5.00 ± 1.25	-2.59 ± 0.63	7.60 ± 1.70	GPP =, NEP =, R=
		Post	15.46	7.58	4.79 ± 0.99	-2.41 ± 0.80	7.20 ± 1.61	
					p = 0.85	p = 0.82	p = 0.91	
	DDZ	Pre	18.24	8.54	5.67± 1.30	-0.76 ± 0.49	6.43 ± 1.4	GPP ⇓, NEP=, R⇓
		Post	15.56	7.10	1.88 ± 0.51	-0.56 ± 0.26	2.44 ± 0.39	
					p < 0.001	p = 0.71	**p < 0.001**	
	*DDZ and REF prior to inundation				p = 0.29	**p = 0.001**	p = 0.23	NEP ≠ prior to inundation
Valemount Peatland	REF	Pre	13.87	4.99	3.59 ± 0.95	-0.84 ± 0.37	4.43 ± 1.01	GPP ⇓, NEP=, R⇓
		Post	12.70	3.64	1.48 ± 0.82	-1.02 ± 0.29	2.50 ± 0.85	
					p = 0.003	p = 0.55	**p = 0.01**	
	DDZ	Pre	17.96	7.99	1.82 ± 0.65	-0.29 ± 0.25	2.10 ± 0.56	GPP =, NEP⇓, R⇑
		Post	16.45	3.14	3.01 ± 1.45	-2.07 ± 1.61	5.09 ± 1.65	
					p = 0.41	**p = 0.06**	**p = 0.001**	
	*DDZ and REF prior to inundation				**p = 0.01**	**p = 0.06**	**p = 0.002**	GPP, NEP, and R ≠ prior to inundation
Sprague Bay*	REF	Pre	22.32	5.31	5.95 ± 0.80	-1.18 ± 0.68	7.13 ± 1.29	GPP* =, NEP =, R=
		Post	19.04	6.26	4.40 ± 0.99	-2.10 ± 0.89	6.50 ± 1.68	
					p = 0.02	p = 0.16	p = 0.19	

* Sprague Bay DDZ pond data were omitted as the pond had drained prior to inundation.

6.4 Aquatic Macrophytes

6.4.1 Macrophyte communities

Macrophyte communities were considerably less diverse than terrestrial communities; a comparably small number of species were observed in ponds in 2012 and 2013 (20 and 24 species, respectively). In 2013, six aquatic wetland associations were identified following the classifications of Pierce and Jensen (2002) and MacKenzie and Moran (2004; Table 6-10). Community classifications were largely determined by the dominance of three macrophytes: *Nuphar polysepalum*, *Potamogeton pusillus*, and *Chara spp.* A seventh undescribed submergent community dominated by *Potamogeton zosteriformis* was also identified in the Valemount Peatland DDZ pond. In part, these communities correspond to regional differences in water physicochemistry as exemplified by the CHARA communities, reflecting *Chara spp*. requirement for mineral rich water (MacKenzie and Moran 2004). A more thorough description of the aquatic communities is provided in Appendix 10.2.

6.4.2 Macrophyte cover and biomass

Macrophytes were more abundant in the DDZ ponds than in Reference ponds, despite high variability within ponds and across index sites (Table 6-11; Table 6-12; Figure 10-32). Across paired ponds, macrophyte abundance was between 1.7 and 9.3 times greater in DDZ ponds (Table 10-10). As expected, macrophyte biomass and grapnel and submergent cover were highly correlated (Table 10-11); planmergent cover was not correlated with any of the other variables. pH and conductivity were positively correlated with submergent abundance (biomass and cover) while sediment depth was negatively correlated (Table 6-13). However these variables were highly correlated themselves reflecting local influences on pond attributes (e.g., pH and conductivity). Planmergent cover was negatively correlated to the water depth. This was in part due to the abundance of emergent species (E.g., *Equisetum spp.* and *Carex spp.)* in shallow water.

Table 6-10. Aquatic wetland associations in the drawdown zone and reference ponds in Kinbasket Reservoir, 2013.

Reach	Site	Position	Stratum	Wetland Association*	Source
Canoe Reach	Valemount Peatland	DDZ	Submergent	UNKNOWN	Ibid.
			Planmergent	NUPLUT	Pierce and Jensen 2002 MacKenzie and Moran 2004
		REF	Submergent	insufficient data	
			Planmergent	NUPLUT	Pierce and Jensen 2002 MacKenzie and Moran 2004
Mica Arm	Sprague Bay	DDZ	Submergent	POTPUS–SPAANG	Pierce and Jensen 2002
			Planmergent	NUPLUT	Pierce and Jensen 2002 MacKenzie and Moran 2004
		REF	Submergent	insufficient data	
			Planmergent	insufficient data	Pierce and Jensen 2002
Bush Arm	Km 88	DDZ	Submergent	CHARA–RANAQU	Pierce and Jensen 2002 MacKenzie and Moran 2004
			Planmergent	POTRIC	Pierce and Jensen 2002
		REF	Submergent	CHARA–RANAQU	Pierce and Jensen 2002 MacKenzie and Moran 2004
			Planmergent	insufficient data	
	Bush River	DDZ	Submergent	CHARA–POTPUS	Pierce and Jensen 2002 MacKenzie and Moran 2004
			Planmergent	insufficient data	
		REF	Submergent	CHARA	Pierce and Jensen 2002 MacKenzie and Moran 2004
			Planmergent	insufficient data	

* Wetland associations described in Appendix 10.2. CHARA = *Chara spp.* dominated, NUPLUT = *Nuphar polysepala* dominated; POTPUS = *Potamogeton pusillus* dominated; POTRIC = *Potamogeton richardsonii* dominated; RANAAQU = *Ranunculus aquatilis* dominated; SPAANG = *Sparganium angustifolium* dominated. See Pierce and Jensen 2002 and MacKenzie and Moran 2004 for full descriptions.

Table 6-11. Mean macrophyte biomass and submergent and planmergent cover in ponds sampled in 2013; values observed in 2012 provided in brackets.

Reach Site - Position	Survey Date	n	Elev. (m)	Planmergent Cover (%)	Grapnel Cover (%)	Submergent Cover (%)	Macrophyte Biomass (g)
Bush River Ref	July 16	5	~760	39.1	7.5	0	9.6
Bush River DDZ	July 15	3	753	66.7	59.8	17.0	79.5
Km 88 Ref	Aug 23	5	~770	31.5	10.5	1.4	6.3
Km 88 DDZ	July 16	5	752.7	100	92.8	2.7	58.6
		(3)			(0.4)	(0.4)	(25.4)
Sprague Bay Ref	July 10	5	757	0.0	0.5	0	0.1
		(6)			(0)	(0)	(0.8)
Sprague Bay DDZ	July 10	6	753	15.1	16.5	6.3	5.5
		(7)			(16.7)	(3.6)	(31.8)
Peatland Ref	June 24	5	757	0.0	1.6	11.9	0.3
		(3)			(5)	(12.5)	(15.0)
Peatland DDZ	June 26	8	753.5	6.1	11.0	13.6	1.5
		(8)			(22.9)	(17.8)	(37.4)

Table 6-12. Results of Kruskal-Wallis tests for differences in macrophyte biomass and cover by pond position (REF = reference; DDZ = drawdown zone) in 2013. df = 2.; 90 per cent confidence intervals (CI).

Environmental Attribute	REF (mean ± 90% CI)	DDZ (mean ± 90% CI)	df	p
Macrophyte biomass (g)	4.3 ± 7.2	36.3 ± 61.8	1	0.25
Grapnel cover (%)	6.2 ± 6.1	45.6 ± 63.7	1	**0.04**
Submergent macrophyte cover (%)	17.5 ± 32.7	46.8 ± 70.8	1	0.25
Planmergent macrophyte cover (%)	3.3 ± 9.1	8.8 ± 8.4	1	**0.08**

Table 6-13. Correlation coefficients for 2013 macrophyte biomass and grapnel, planmergent, and submergent cover data with environmental variables. Bold indicates significant difference α < 0.1.

	pH	Water Depth	Sediment Depth	DO	Temp	Conductivity	Wood
pH	1.00	**-0.66**	**-0.64**	**0.92**	**-0.55**	**0.78**	-0.41
Depth	**-0.66**	1.00	0.25	**-0.76**	0.29	-0.22	-0.18
Sediment Depth	**-0.64**	0.25	1.00	-0.40	0.16	**-0.89**	0.53
DO	**0.92**	**-0.76**	-0.40	1.00	-0.64	**0.57**	-0.10
Temp	**-0.55**	0.29	0.16	**-0.64**	1.00	-0.35	0.18
Conductivity	**0.78**	-0.22	**-0.89**	**0.57**	-0.35	1.00	-0.47
Wood	-0.41	-0.18	0.53	-0.10	0.18	-0.47	1.00
Planmergent Cover	0.31	**-0.65**	0.11	0.44	-0.24	0.04	0.41
Submergent Cover	**0.72**	-0.19	**-0.83**	**0.55**	-0.06	**0.81**	-0.40
Rake Cover	**0.56**	-0.26	**-0.73**	0.46	0.01	**0.72**	-0.23
Biomass	**0.62**	-0.39	**-0.70**	0.46	0.00	**0.65**	-0.37

6.4.3 Planmergent Vegetation

The abundance and distribution of planmergent macrophytes were sparse and patchy in ponds sampled in 2013. The Bush River DDZ pond and the DDZ and Reference ponds in Valemount Peatland had the highest planmergent cover (Table 10-12); no vegetation in this strata was detected in the Bush River and Sprague Bay Reference pond plots. Of the twelve planmergent species recorded in the index ponds, *N. polysepala* and *P.*

pusillus were the most abundant. *P. pusillus* was detected only in DDZ ponds and *N. polysepala* was equally abundant in both DDZ and Reference ponds. The composition of planmergent communities could not be compared across reservoir position due to insufficient data in Reference ponds. Finally, no species in these strata had higher indicator values in DDZ ponds than in reference ponds ($p > 0.10$).

6.4.4 Submergent Vegetation

The reference and DDZ ponds supported different submergent communities ($p = 0.06$; Table 6-14) and many submergent macrophytes were exclusively to either the DDZ ponds or Reference ponds.

Plots in the Km 88 and Bush River DDZ and Reference ponds had moderate to high submergent cover, which was in part due to the presence of *Chara* spp. The Reference ponds in the Valemount Peatland at Sprague Bay had virtually no submergent vegetation. As expected, submergent communities differed across the four index sites (DDZ ponds only PERMANOVA: $F_{3,8} = 5.49$, $p < 0.001$). *P. pusillus* was the only indicator species of reservoir position and was a potential indicator for DDZ ponds (*INDVAL* = 38.1, $p = 0.002$).

In the ponds sampled, submergent macrophyte species richness (n = 21) was greater than planmergent species (n = 12), although, macrophyte abundance (per cent cover) was highly variable among ponds (Table 10-13).

Table 6-14. Differences in submergent vegetation composition (Sorensen similarity measure). Permutation-based nonparametric MANOVA, with three replicates of reservoir position (lower control, target, and upper control elevation band), nested within each of 2 locations (only testing differences between Bush River and Km 88 sites)

SOURCE	df	SS	MS	F	P-value
Location	1	0.373	0.373	1.274	0.669
Position	2	0.586	0.293	1.804	**0.061**
Residual	8	1.2992	0.162		
Total	11	2.2582			

6.4.5 Inter-annual variation in aquatic vegetation

Macrophyte abundance was compared across ponds sampled both in 2012 and 2013. In a first comparison, grapnel cover and macrophyte biomass differed across years while planmergent cover did not (Table 6-15). In reviewing the data (Table 6-11), we suspect that the grapnel samples from the Km 88 pond were underestimated in 2012 due to problems associated with sampling in water over 1.5 m deep (Adama et al. 2013). A second comparison of grapnel cover and macrophyte biomass with those data removed showed no differences between years.

Ten planmergent species in total occurred in ponds sampled in either 2012 or 2013 (Table 10-14), of which half were only present in a single pond in any given year. The Sprague Bay Reference pond had no recoded species in either year. There was a significant effect of pond on planmergent vegetation composition planmergent macrophyte composition between Sprague Bay DDZ and Valemount Peatland DDZ (Two-way PERMANOVA: $F_{1,16} = 6.29$, $p = 0.001$), but year and the interaction of year*pond were not significant ($F_{1,16} = 1.06$, $p = 0.38$; $F_{1,16} = 0.53$, $p = 0.69$).

Table 6-15. Differences in vegetation composition within ponds sampled in 2012 and 2013. Grapnel and biomass ran with and without data for Km 88 pond. Data excluded for KM 88 due to inconsistent sampling between 2012 and 2013. Bold indicates significant difference α < 0.1.

Variable	Test	Value	Approx. F	NumDF	DenDF	Prob>F
Grapnel cover[1]	Year	0.46	8.69	1	19	**0.008***
	Year *Pond	1.83	8.70	4	19	**<0.001***
Grapnel cover[2]	Year	0.01	0.03	1	17	0.858
	Year *Pond	0.15	0.88	3	17	0.47
Biomass[1]	Year	0.84	16.05	1	19	**<0.001***
	Year *Pond	2.83	13.42	4	19	**<0.001***
Biomass[2]	Year	0.08	0.14	1	17	0.72
	Year *Pond	0.17	13.42	3	17	0.43
Planmergent	Year	0.01	0.27	1	19	0.61
	Year *Pond	0.13	0.59	4	19	0.67

[1] Km 88 ponds included [2] Km 88 ponds excluded

6.5 Pelagic Invertebrates

No trends were apparent in the abundance of pelagic invertebrates among the paired reference and DDZ ponds (Figure 6-4), index sites, or among reaches. Average abundance of pelagic invertebrates varied greatly within and among ponds ranging from a low of 6.2 to a high of 928.2 invertebrates per sample. Variation was high at each pond, resulting in large confidence intervals around each mean.

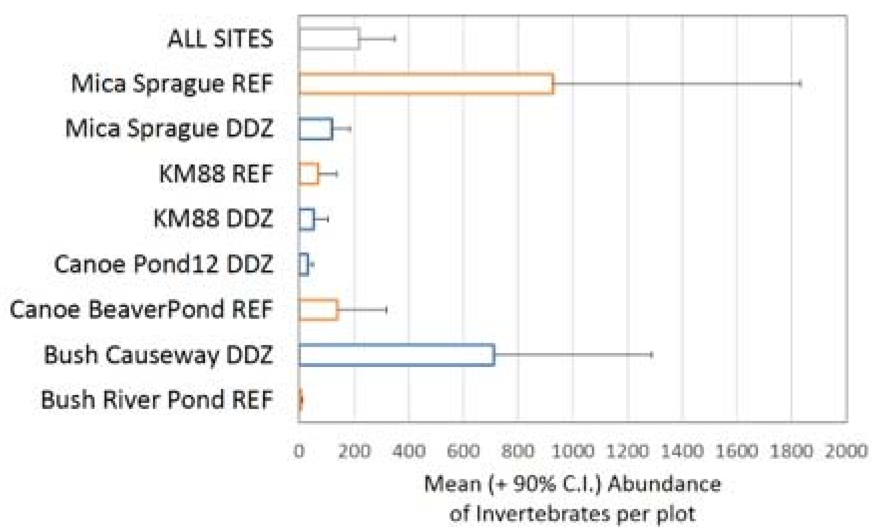

Figure 6-4. Average (Mean + 90% confidence interval) abundance of invertebrate at each pond of Kinbasket Reservoir. Reservoir ponds (DDZ; blue) are listed adjacent to their paired reference ponds (REF; orange)

In both 2012 and 2013, Clitellata (leeches, crayfish worms, and oligochaete worms) and hydrozoan abundances were greater in DDZ ponds and insects and Ostracoda (seed shrimp) were more common in Reference ponds (Figure 6-5; Table 10-16). For other taxa, the observed patterns in abundance across years were inconsistent (Figure 6-5).

For example, Gastropoda (slugs and snails), Arachnida (spiders and mites), and Malacostraca (amphipods and isopods) were more often found in Reference ponds in 2012, whereas in 2013 they were more common in DDZ ponds (Table 10-16). Branchiopoda (water fleas, clam shrimp, and fairy shrimp) were more common in DDZ ponds in 2012, yet the opposite was true in 2013.

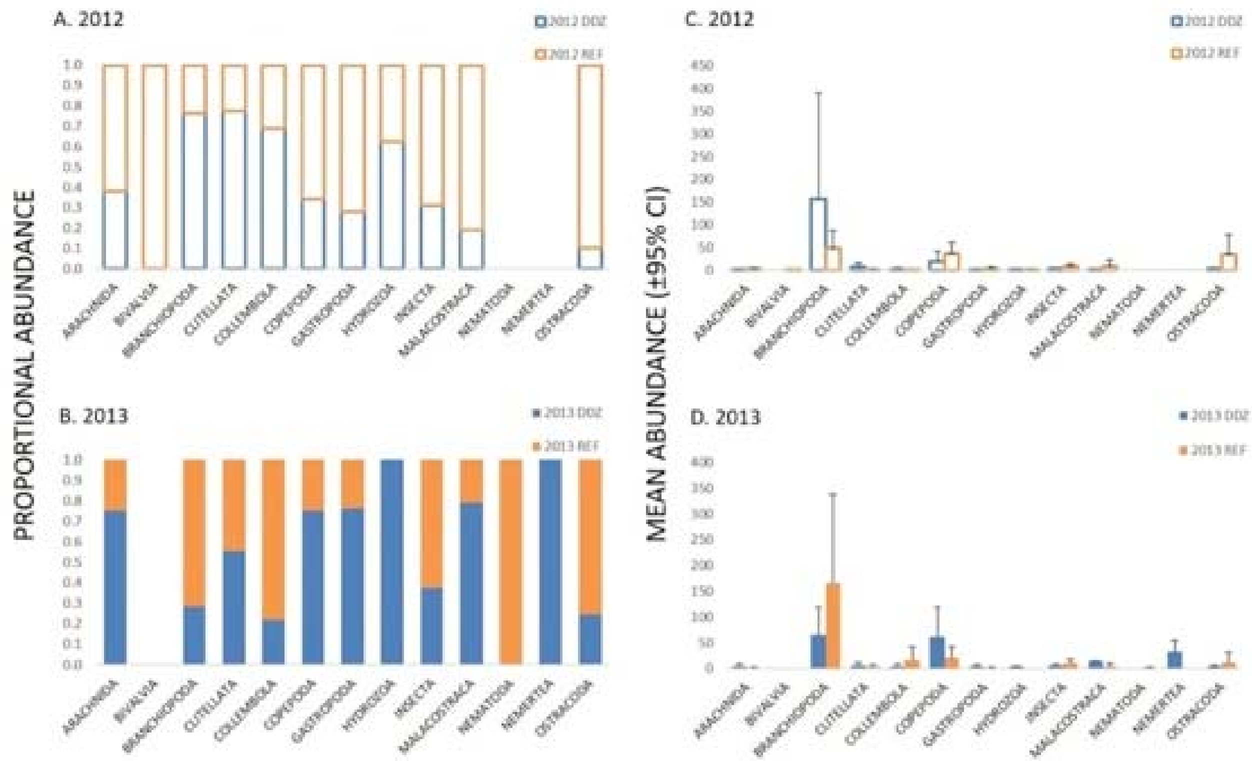

Figure 6-5. **Distribution of taxa in drawdown zone (DDZ) and reference (REF) ponds between monitoring years 2012 (above) and 2013 (below); proportional (relative) abundance (left: A, B) and mean abundance with 90 per cent confidence intervals (right: C, D) are given for each invertebrate taxon**

To assess whether pond position influences invertebrate diversity, we examined the richness, evenness, Shannon diversity, and Simpson diversity of DDZ and Reference ponds in both 2012 and 2013. The number of taxa collected (richness) was similar between 2012 and 2013. A total of 40 taxa were identified from 55 samples collected in 2012 and 38 taxa were identified from 35 samples collected in 2013. No differences in richness, evenness, or diversity were detected between DDZ and Reference ponds within or between years (Figure 6-6), as overall patterns were obscured by the effect of reach.

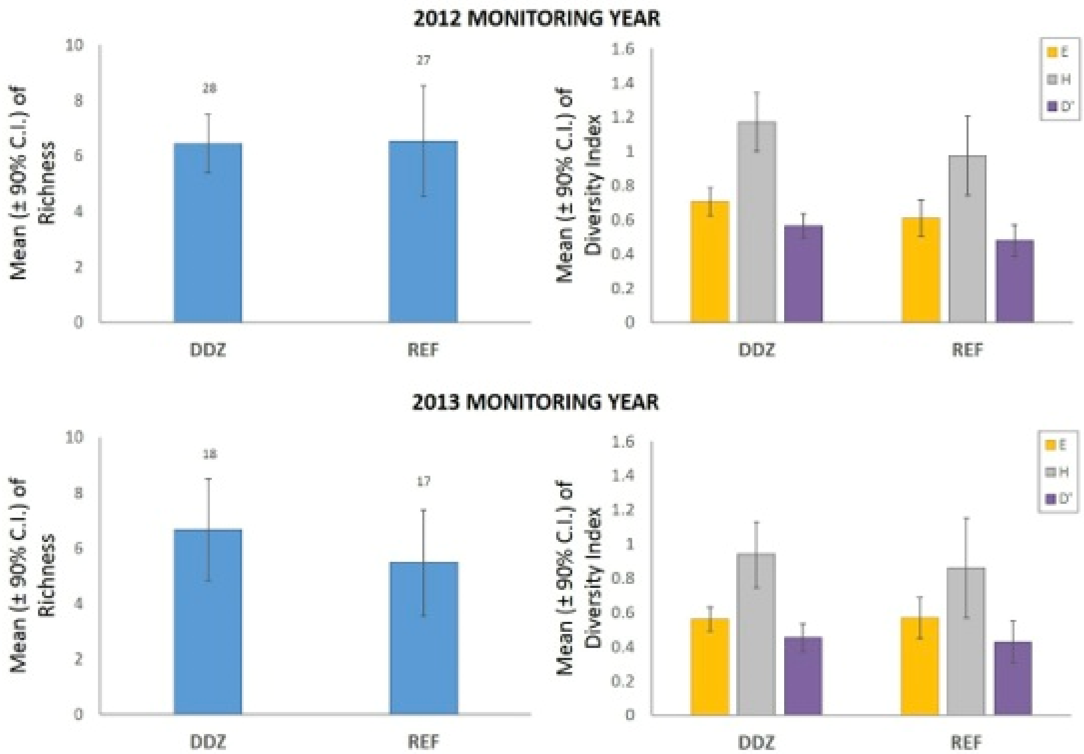

Figure 6-6. Observed number of taxa (lowest-level identified) and diversity for DDZ and Reference ponds in 2012 and 2013. Average taxon richness (S), Pielou's evenness (E), Shannon Diversity (H) and Simpson Diversity (D) are given with 90 % confidence intervals. The number of samples (n) is given above the richness error bars

Community composition varied greatly among sites and reaches. Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson 2001) tests revealed a significant interaction between reach and pond position (Table 6-16; Figure 6-7). For all reaches except Mica Arm, samples from Reference ponds were less similar (higher mean distance) in invertebrate composition than DDZ samples (Figure 10-36). At Mica Arm, the DDZ samples were more dissimilar than their reference counterparts. These results further confirm that communities are highly variable in Kinbasket Reservoir and the influence of pond position on invertebrate composition varies among the index sites.

Table 6-16. Results of a Two-Way Multivariate Analysis of Variance based on 4999 permutations. Composition was calculated in Hellinger distance for 38 species and 24 samples (3 randomly chosen replicates per site)

Source	df	SS	MS	F	p-value
Reach	3	4.043	1.38	1.201	**0.094**
Position	1	1.288	1.288	1.148	0.203
Interaction	3	5.406	1.802	1.606	**0.002**
Residual	16	17.956	1.122		
Total	23	28.693			

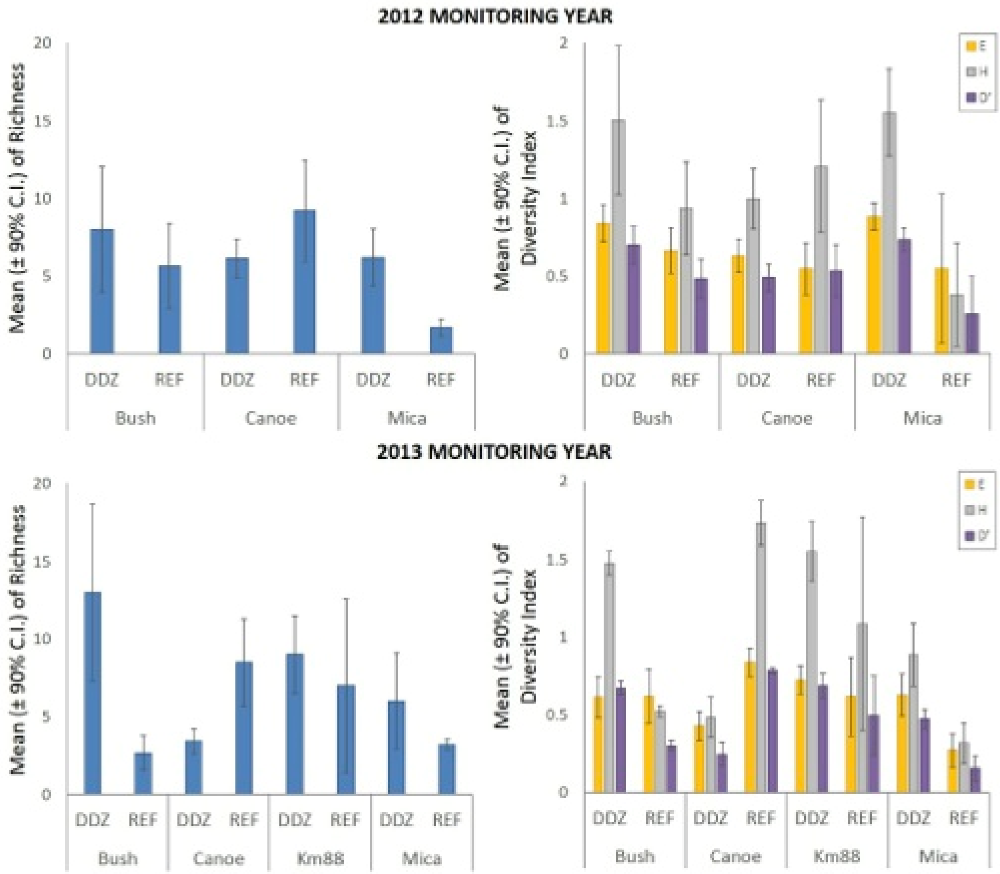

Figure 6-7. Observed number of taxa (lowest-level identified) and diversity by site in each reach for monitoring years 2012 and 2013. Average taxon richness (S), Pielou's evenness (E), Shannon Diversity (H) and Simpson Diversity (D) are given with 90 per cent confidence intervals

To assess the structure of invertebrate communities and relate these to environmental variables, a PCA was performed on these data. The first two axes of the PCA explained 39.8% of the variation in aquatic invertebrates (Figure 6-8). Consistent with the results of the PERMANOVA analysis, the composition of invertebrates showed no clear separation between DDZ ponds and Reference ponds in the PCA ordination overall and invertebrate communities were quite dissimilar between paired DDZ and Reference ponds.

The strongest relationships with environmental variables were found for Cladocera, Isopoda, Diptera, Odonata, Amphipoda, and Coleoptera (Table 10-17). Cladocera distribution was strongly related to water depth and was associated with reference sites at Mica (Sprague) and Km 88 (Figure 6-8). Most taxa were negatively associated with temperature, and were distributed in sites higher in DO and conductivity.

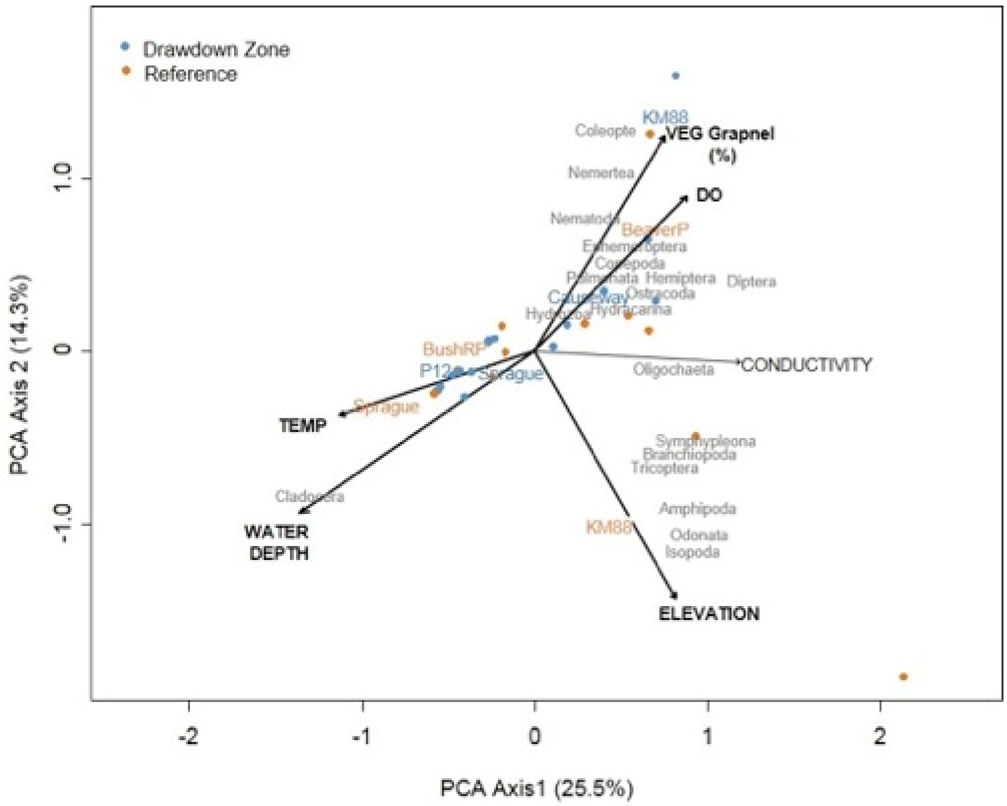

Figure 6-8. **Principal Components Analysis (PCA) of aquatic invertebrate taxa (n = 19).** Points represent each sampling plot at various sites and are colour-coded to indicate pond position (blue = DDZ, orange = REF). Site centroids are plotted by name and colour (Causeway = Bush River DDZ Pond, BushRP= Bush River REF, P12= Valemount Peatland DDZ Pond, BeaverP= Valemount Peatland Reference pond). Direction and strength of environmental relationships are depicted by the overlaid vectors. Taxa are indicated in grey text

7.0 DISCUSSION

The objectives for Year 2 of CLBMON-61 were to summarize the state of the index sites, and provide preliminary insight into the expected changes to vegetation composition or wetland productivity. We also assessed the efficacy of the sampling methods.

7.1 State of the Index Sites

Differences in geology, climate, hydrology and other hydrogeomorphic factors have created a unique complement of wetlands in the Kinbasket region. From our assessment, the state of the index sites changed little between 2012 and 2013. However, assessments for 2012, the pilot study year, were less extensive than in 2013, limiting our ability to compare across years.

7.1.1 Bush River and Km 88

The Km 88 wetland is a small 25 ha complex of fens and beaver dams located north of Bear Island (Figure 10-8). A series of beaver ponds occur above and extend down into the reservoir. Two DDZ ponds are positioned at 752 and 753 m ASL; the reference pond is positioned approximately 500m upslope at 780 m ASL. In 2013, recent beaver activity was not evident. Due to seepage, fens and swamps have developed downstream of the ponds. The largest of these extends from 757 to below 750 m ASL and is the site of the terrestrial wetland transects (Figure 10-8). Wildlife values of the Km 88 wetlands are moderate to high and provide habitat for Black Bear, Beaver, River Otter, Moose, White-Tailed Deer, Western Toad, Columbia Spotted Frog, and a variety of birds (MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012). *Liparis loeselii*, a red listed orchid was documented in the upper control and reference transects. Herbaceous plant cover in the Km 88 transects was nearly twice that of other index sites (Table 10-2) making it one of the most productive terrestrial wetlands in the reservoir.

The Bush River wetlands extend downstream of the confluence of the Bush and Valenciennes Rivers (770 m ASL) to the Bush River causeway (752 m ASL; Figure 10-11, Figure 10-12). The DDZ pond and terrestrial wetlands are located adjacent the causeway (753 m ASL) and are frequently inundated during spring freshet. Wetlands also occur on the west side of the causeway at 752 m ASL and are prone to accumulations of wood debris. The reference wetland is located 3.3 km upstream of the DDZ pond at approximately 760 m ASL. This wetland complex consists of a 4.1 ha shallow lake bounded by fenlands. There is evidence of a natural spring and of old beaver activity.

The wildlife values of Bush River wetlands are high. They are known to provide habitat for Grizzly Bear, Black Bear, Elk, Western Toad, Long-toed Salamander, Columbia Spotted Frog, small mammals, and a variety of birds (MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012). *Liparis loeselii was* documented in the reference transects.

The Bush River and the Km 88 wetlands were influenced by high pH and mineral content (conductivity), which reflects the calcium carbonate rich water in the Central Rocky Mountains. In Bush Arm, submergent cover and macrophyte biomass were over an order of magnitude higher than observed elsewhere, largely due to the presence of *Chara spp*. At both sites, macrophyte abundance was higher in DDZ ponds reflecting either a direct response to disturbance resulting from increased sedimentation and nutrient input or an indirect response such as increased solar radiation resulting from the loss of adjacent riparian vegetation.

Despite their proximity, the terrestrial wetland communities of these two sites were disparate. The terrestrial associations at Km 88 were characteristic of an elevated fenland with high herb cover and deep organic soils, whereas the Bush River associations were characteristic of riverine wetlands dominated by *Salix spp.* and mineral soil. In the lower control transects at the Bush River causeway, high amounts of wood debris have accumulated due to the Bush River causeway, which concentrates wood on the downstream side. During spring freshet, the undersized river channel in the Bush River causeway causes back flooding on the upstream side. Such events were apparent in the sonde data when mean daily water temperatures rapidly decreased and remained 5 to 7 °C colder for several days. It has been suggested that flooding may result in delayed development of amphibian larvae in these otherwise highly productive ponds (Virgil Hawkes, pers. comm.).

7.1.2 Valemount Peatland

The Valemount Peatland is a remnant fenland located at the north end of Kinbasket Reservoir and Canoe Reach, 10km south of Valemount. Approximately 300 hectares in size, the area extends west of the Canoe River channel (745 m ASL) to an elevation 760 m ASL (Figure 10-2); approximately 90 per cent of the wetland complex occurs below full pool (754.4 m ASL). The Valemount Peatland is comprised of vegetation communities that reflect both the historic fenland complex that existed prior to inundation and the elevation gradient within the reservoir (Moody and Carr 2003, Hawkes et al. 2010). As suggested in the site name, peat is the dominant substrate; however, wood debris and wood fragments blanket portions of the remnant fenland excluding vegetation growth (Hawkes et al. 2010). Wildlife use of the Valemount Peatland has been well documented and it is known to provide important habitat for birds, Moose and White-Tailed Deer, Wolves, small mammals, reptiles, amphibians (Western Toad, Columbia Spotted Frog, and Long-Toed Salamander), and rare plants (Ham 2010; MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012).

Although mostly degraded by reservoir operations (Moody and Carr 2003), wetland communities in the Valemount Peatland appeared intermediate between the Bush Arm and Sprague Bay but with the influence of the SBS biogeoclimatic zone (i.e., colder and drier). The DDZ and Reference ponds were both characterized as NUPLUT communities dominated by *Nuphar polysepala.* Water physicochemistry was intermediate between Bush Arm and Sprague Bay and pH was neutral. This reflects the remnant peat accumulation and surficial geology.

Terrestrial vegetation species richness, diversity and community composition in the Valemount Peatland were the lowest of the four index sites indicating that this wetland is less heterogeneous and diverse than previously reported (Moody and Carr 2003; Hawkes et al. 2007). Transects within the reservoir were dominated by a single wetland association (Swamp-Horsetail) and high amounts of wood debris. The higher latitude of the site was reflected by the presence of *Picea mariana* (Black Spruce) in the reference transects.

7.1.3 Sprague Bay

Located 8km east of Mica Dam, the Sprague Bay wetlands are comprised of a narrow fenland/beaver pond complex extending from 760 to 752 m ASL (Figure 10-5). Beaver dams bisect the complex creating a series of ponds, fens, and riparian benches that extend down into the reservoir. The entire complex is approximately 9 hectares of which two-thirds is located above the reservoir and is mostly comprised of a floating fen. Low conductivity and pH in the aquatic wetlands were indicative of deep peat accumulation and low mineral content in the igneous and metamorphic parent materials of the Selkirk Mountains. The aquatic vegetation in the DDZ or reference ponds was dominated by *Potamogeton pusillus* but was sparse overall. The reference wetland consists of a unique floating fen complex supported entirely by a beaver dam that is over two meters high and spans the width of the wetland complex. This complex supported *Scheuchzeria palustris* and the Scheuchzeria-Peat-moss association, which did not occur elsewhere in the study area. Relative to the reference transects, the drawdown zone transects had a higher complement of shrubs including *S. douglasii*. *S. douglasii* is characteristic of disturbed water receiving sites and fluctuating water tables (Klinka et al. 1989). Its abundance in the lower elevation band indicates high levels of disturbance possibly caused by high reservoir level in recent years. Wildlife values have not been thoroughly documented; however, based on our brief observations Black Bear, Beaver, Moose, and a variety of songbirds utilize these wetlands. These wetlands also provide highly productive breeding habitat for the Western Toad (Hawkes and Tuttle 2012).

7.2 Anticipated changes

Potential changes to wetlands resulting from the installation of the new turbines were explored by assessing a variety of wetland characteristics. Our approach was to identify those variables that either differed across the 1-meter elevation gradient (terrestrial wetland characteristics), between DDZ and Reference ponds (aquatic wetland characteristics), or in response to inundation (water physicochemistry and pond metabolism). Given the limitations of the monitoring program, we reason that variables that do not differ consistently across these strata are not likely to be useful for assessing changes in wetland integrity associated with Mica 5 and 6.

7.2.1 Terrestrial Wetlands

Terrestrial wetland communities increased in complexity from sparsely vegetated graminoid associations at lower elevations to more diverse shrub – herb associations in the upper elevations. Communities at lower elevations were dominated by graminoids, wood debris, or *S. douglasii*, whereas the upper control elevation band and reference transects were characterized by species-rich fen, bog and riparian associations. Vegetation cover, species richness, and species diversity increased with elevation and the strongest differences occurred in shrub communities. As expected, the vegetation in the target elevation band was transitional between the lower control and the upper control elevations; however, communities in the target elevation band were more similar to those in the lower elevation band, which likely reflects recent high reservoir elevations and prolonged inundation.

Within the reservoir, differences across a small elevation gradient (3 meters) are not unexpected. Extended periods of inundation create anoxic soil conditions resulting in increased plant stress and reduced survival (Kozlowski 2002; Parent et al. 2008). Plants tolerant of flooding have physiological and anatomical adaptations that allow them to cope with reduced soil oxygen and nutrient availability. At lower reservoir elevations, the distribution of plants likely reflects their ability to tolerate reduced soil oxygen and scouring, while at higher elevations where less flood tolerant species can exist, plant compositions also reflect interspecific competition and adaptations to microsites (e.g., standing water, soil types, etc.). In addition, woody vegetation (shrubs and trees) must also contend with mechanical damage caused by wood debris that abrades the cambium from stems. This has been observed on both native shrubs and planted willow and cottonwood stakes in Kinbasket Reservoir (Hawkes et al. 2013a).

Interestingly, lenticular sedge (*Carex lenticularis ssp. lipocarpa*) was the only potential indicator species (shrub or herb) identified for the target elevation band. Lenticular sedge occurred at all the index sites but was more common in the lower elevation bands in the Valemount Peatland and Km 88 reflecting its ability to withstand moderate periods of inundation at these sites. The identification of this species as a potential indicator is significant because it has been intensively planted throughout Kinbasket Reservoir under CLBWORKS-30 to enhance vegetation in the upper elevations of the reservoir (KES 2012); to date approximately 425,000 lenticular sedge have been planted. Although none of the sites that were sampled under CLBMON-61 were planted previously, an increase in flooding frequency and duration of the target elevation band could reduce the survival of both planted and naturally occurring lenticular sedge undermining the efforts of the revegetation program to maintain vegetation in the upper elevations of Kinbasket Reservoir.

Shrub richness and cover in the 752 and 753 m ASL elevation bands were almost 200 per cent lower than those observed in the upper control elevation band or in reference sites. Composition analyses for both herbs and shrubs found stronger similarities between the lower two elevation bands, likely reflecting recent periods of prolonged inundation. Between 2011 and 2013, the 752 and 753 m ASL elevation bands were inundated for an average of 96 and 76 days respectively, whereas the 754 m ASL elevation band was inundated for 31 day on average (Figure 6-2). While it is premature to establish a firm cause and effect relationship between prolonged inundation and vegetation patterns from CLBMON-61 data, these observations are supported by Hawkes et al. (2013b) that show reductions in plant diversity and abundance in Kinbasket Reservoir following high reservoir elevations in 2007 and 2011.

Although differences in both shrub and herb communities were observed across the 1-meter elevation bands, shrub abundance, species richness, and composition will likely provide the most reliable results for assessing change over time. Reiterating our observations (Adama et al. 2013), we predict that if the frequency of annual flooding increases in the target elevation band, the wetland communities at this elevation will come to further resemble communities presently found in the lower control elevation band. We also predict a drop in species diversity and richness as hydrophytic species out-compete some of the less adapted species and that shrubs will likely be further reduced and may even be eliminated over time.

7.2.2 Aquatic Macrophytes

Despite high variability in macrophyte abundances across the study area, macrophyte abundance was between 1.7 and 9.3 times greater in DDZ ponds than in paired reference ponds. In addition, submergent macrophytes were almost an order of magnitude more abundant in the Bush Arm ponds than in either the Sprague Bay or Valemount Peatland whereas planmergent vegetation was highest in the Valemount Peatland but lowest in Sprague Bay.

Factors that influence the ecology and growth of aquatic macrophytes include light availability, water chemistry, nutrient availability, temperature, sediment, hydrology, and disturbance (e.g., wave action and inundation) (Cronk and Fennessey 2002; Bornette and Puijalon 2011; Kisson et al. 2013). Due to small sample sizes and high correlation between factors, it is difficult to attribute observed patterns to any particular factor; however, the ecology of some macrophytes (e.g., *Chara spp.*) underscores the importance of water chemistry.

Two factors that may account for increased macrophyte abundance in DDZ ponds include light interception and sedimentation. Ponds in Kinbasket Reservoir are largely depauperate of adjacent upland vegetation resulting in maximum exposure to solar radiation. Canfield and Hoyer (1988) and Fletcher et al. (2000) report increases in macrophyte abundance following the removal of 50 to 60 per cent of the adjacent forest canopy, attributing this to increased light penetration. Julian et al. (2011) report a 60 per cent increase in macrophyte biomass and 77 percent increase in macrophyte cover in open stream reaches in comparison to forested reaches.

The deposition of fine sediments following inundation (by the reservoir) may also contribute to increased macrophyte abundance in DDZ ponds. Nutrient availability in natural systems can be limiting for aquatic macrophytes and sedimentation provides an important means of nutrient renewal to the littoral zone. Following short periods of inundation, the growth of submersed macrophytes may be stimulated by the import of fine-textured inorganic materials such as clays and silts (Barko et al. 1991). However, due to reduced light availability from suspended sediments and other factors, prolonged periods of inundation and increased water depths can have a detrimental effect on macrophytes, particularly submergents (Bornette et al. 2011). Increased water depth can also negatively affect planmergent species through wave action (Lacoul and Freedman 2006).

From the data collected in 2012 and 2013, it appears that indicate macrophytes respond sufficiently to reservoir conditions to warrant monitoring; however, our results are based on a small sample size. Nevertheless, it is anticipated that increases in water depth and prolonged inundation (resulting from operational changes) will have a negative effect on macrophyte abundance. However, if reservoir operations return to the 1987 to 2006 norm, macrophyte abundance in DDZ ponds will likely increase from the values observed in 2012 and 2013, when the ponds were subjected to prolonged periods of inundation.

To date, we have not measured photosynthetically active radiation (PAR) or estimated canopy cover adjacent the DDZ and Reference ponds. We recommend doing so in future year as these variables may help corroborate our observations regarding light interception.

7.2.3 Wood Debris

In Kinbasket Reservoir, wood debris causes mechanical damage to shrubs and trees and site disturbance through surface scouring, displaces existing terrestrial and aquatic vegetation, and reduces habitat values for wildlife (Hawkes et al. 2013a). Our data show that higher amounts of wood debris occur in aquatic and terrestrial wetlands within Kinbasket Reservoir than in wetlands adjacent the reservoir. Following the installation of Mica units 5 and 6, the frequency of inundation of the 753 m ASL elevation band is predicted to increase, which may result in a parallel increase in the accumulation of wood debris at this elevation.

Since 2008, high-resolution aerial imagery has been collected biennially for several areas in Kinbasket Reservoir. Mapping the extent of wood debris in the reservoir through GIS in conjunction with field data would provide the necessary information to estimate annual accumulation rates of wood debris at various elevations in the reservoir (Woodall and Williams 2008; Hawkes et al. 2013a). Acquiring this information would provide an additional means to assess the potential impacts of Mica 5 and 6 on wetlands and will benefit other WLR studies (CLBMON-09, CLBMON-10, CLBMON-11A, and CLBWORKS-01).

7.2.4 Aquatic Metabolism

One of the primary objectives of CLBMON-61 is to assess the impacts from Mica 5 and 6 on productivity resulting from changes in reservoir operations. Diel changes in DO can be used to estimate net ecosystem production (NEP), ecosystem respiration (R), and gross primary production (GPP), which are fundamental metrics of ecosystem metabolism (Cole et al. 2000; Staehr et al. 2010; Hoellein et al. 2013). Metabolic metrics are considered to be good indicators of ecosystem integrity (Allen and Castillo 2007; Mitsch and Gosselink 2007; van der Valk 2012). Wetlands with high rates of GPP are characterized by high periphyton or macrophyte biomass, which occur under high light and nutrient regimes. Conversely, heavily shaded and oligotrophic wetlands generally have very low GPP. Sites with high rates of R are normally characterized by high macrophyte and algae biomass, and/or large inputs of organic matter, and are prone to low minimum DO concentrations. NEP is a measurement of the net gain or loss of energy over a period of time. Positive NEP rates are indicative of autotrophic ecosystems, which accumulate energy (i.e., macrophyte biomass), whereas negative NEP rates are indicative of heterotrophic ecosystems, which lose energy through heterotrophic respiration.

In 2013, uncorrected estimates of aquatic metabolism were calculated in DDZ and Reference ponds in Kinbasket Reservoir, before and after inundation. These calculations were performed to assess whether differences could be observed in metabolic metrics in response to inundation, and to explore the technical requirements for determining these metrics. The results also provided insight into the ecological characteristics of the ponds and how metabolic metrics respond to inundation.

Trends observed in 2013 indicate that metabolic metrics in DDZ ponds were affected by inundation, whereas Reference ponds tended to be more stable across the sampling period. Following inundation, GPP and R decreased in the Km 88 and Bush River DDZ ponds, whereas GPP and R remained unchanged in the Reference ponds. This pattern is consistent with the findings of Cooper et al (2013) who report that R and GPP were lower in wetlands with increased hydrologic exposure (wave action, water depth, and current). They suggest that higher metabolic rates of more sheltered wetlands support higher rates of nutrient cycling, organic matter production, and decomposition than exposed wetlands. A consequence of inundation is increased hydrologic exposure, thus GPP and R are predicted to decrease in the Bush Arm wetlands following inundation.

Patterns in the Valemount Peatland and Sprague Bay were less clear. These ponds are dystrophic with deep organic sediments. Lauster et al (2006) found that shallow dystrophic lakes functioned differently and had higher rates of R than non-dystrophic lakes. They also found DO data from sondes to be sensitive to the presence and distribution of macrophytes and organic matter, particularly along lake margins. Both of these factors may have contributed to the patterns observed in 2013; however, the mechanisms that affect aquatic metabolism in these ponds may require further investigation and are unlikely to be explained with a single year of data.

Metabolic metrics were not corrected for atmospheric diffusion and estimates were likely confounded by additional assumptions made during computation (Section 5.4.5).

Correcting for diffusion requires simultaneous measurements of wind speed, PAR (photosynthetic active radiation), barometric pressure, temperature, and water depth (Cole et al. 2000; Staehr et al. 2010) and instrumentation to collect this data should be installed in future years. Because of the complexities involved with equipment installation, data collection, and numerical computations (none of which are trivial), we recommend that protocols for instrumentation installation and data analysis be prepared.

7.2.5 Pelagic Invertebrates

Pelagic invertebrates appear to be of low utility for assessing the effects of reservoir operations, as sampled under CLBMON-61. Few reliable patterns between ponds in the drawdown zone and reference sites were found. Further, taxonomic similarity within sites suggests that invertebrate communities more readily reflect local conditions than reservoir position. This is not unexpected as aquatic invertebrate communities are extremely sensitive to water physicochemistry (e.g., DO and conductivity: Spieles and Mitsch 2000; temperature and pH: Suren et al. 2010; Simaika and Samway 2011), and these parameters were strongly associated with reach and site. Thus, differences in water physicochemistry across sites obscured the variation observed between DDZ and Reference ponds rendering such comparisons to be unreliable for assessing changes in reservoir operations.

Although it is generally agreed that macroinvertebrates can be good indicators of wetland integrity (U.S. EPA 2002), pelagic invertebrate communities are greatly influenced by daily and seasonal weather patterns (Hann 1996; Cardinale and Burton 1997). This likely explains the variability observed in pelagic invertebrate communities between 2012 and 2013. As such, we do not consider pelagic invertebrates to be useful for assessing the impacts associated with Mica Units 5 and 6 following the current methodology (single sample per year). Multiple samples obtained prior to and following inundation would greatly improve reliability; however, this would not be economically feasible given the remoteness of the study area.

8.0 CONCLUSION AND RECOMMENDATIONS

8.1 Summary

With the collection of a full suite of data in Year 2, we were able to confirm and elaborate on the general patterns that were observed in Year 1. We also applied a common sense approach to assess both the efficacy of the various methods and to predict what the potential impacts from Mica 5 and 6 may be. For terrestrial wetlands, this approach entailed comparisons across 1-meter elevation bands in the upper elevation of the reservoir; for aquatic wetlands this entailed comparisons between DDZ and Reference ponds, or before and after inundation.

In summary, we found support for monitoring terrestrial wetland vegetation (particularly the shrub communities), pond metabolism, aquatic macrophytes, and wood debris; however, we found pelagic invertebrates to be too variable following the current methodology. Factors identified as confounding the overall study design included: 1) high reservoir levels in 2012 and 2013 that prevented the collection of data under baseline (1987 – 2006) conditions; 2) residual effects of recent high reservoir levels on wetland vegetation, which may confound the interpretation of results in future years; 3) insufficient data in 2012 to comply with the BACI study design; 4) small sample sizes; 5) and high variability in part due to differences in geology and water physicochemistry across the study area. The recommendations that follow address some of these concerns as well as some of the methodological issues raised in the report.

8.2 Recommendations

1. We recommend discontinuing the collection of pelagic invertebrate data as no obvious trends could be determined from the data collected to date. In lieu, we recommend obtaining more accurate estimates of primary productivity and aquatic metabolism (NEP, GPP, and R).
2. Recent advances in sonde technology permit the calculation of reliable metabolic rates (NEP, GPP, and R) from diel fluctuations in dissolved oxygen. Dissolved oxygen sondes were deployed in aquatic wetlands in 2012 and 2013; however, additional instrumentation is required to estimate the diffusion of oxygen into the atmosphere. As oxygen diffusion can exceed hourly NEP, estimates of NEP and GEP without such correction can lead to spurious results. In future years, we recommend installing additional instrumentation to correct for atmospheric diffusion. Because of the complexities involved with equipment installation, data collection, and numerical computations, we recommend that an operating procedure be prepared.
3. We recommend an additional year of sampling be carried out in 2014 for the following reasons:
    a. The Terms of Reference for CLBMON-61 prescribed a BACI sampling design and specified two years of data to be collected prior to the operation of the new turbines. High reservoir levels limited the sampling in 2012 resulting in an incomplete data set for Year 1. Sampling in 2014 will provide a second year of data as required.
    b. Prolonged inundation (87 days) of the 753 m ASL elevation band in 2013 may affect the composition and productivity of wetlands in this elevation band and impacts may carry over into the post-impact period (2015 and beyond). If this occurs, we will be unable to determine whether these impacts are a result of the 2013 reservoir levels or the installation of the new units.
    c. As reservoir levels are forecast to be considerably lower, sampling in 2014 should provide an opportunity to sample under conditions more similar to the 1987-2006 operating conditions.

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10.0 APPENDIX

10.1 Definitions

Definitions are provided to ensure that the terminology used in this report is understood. The definitions are presented in logical, not alphabetical, order.

Wetland – "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment" (National Wetlands Working Group 1988).

For this study, we distinguish between two types of wetlands that do not occur under the BC or Canadian wetland classification systems (Table 4-1):

1) Terrestrial wetland – includes the bog, fen, swamp, or marsh wetland classes as defined under the Canadian Wetland Classification Scheme (National Wetlands Working Group 1988) and MacKenzie and Moran (2004).

2) Aquatic wetland – Aquatic wetlands are permanently flooded "shallow-water" wetlands that are dominated by rooted, submerged and planmergent (floating) aquatic plants (Moran and MacKenzie 2004). These communities typically occur in standing water less than 2 m deep and are associated with permanent still or slow-moving water bodies such as ponds, shallows lake or lake margins. The term pond is used interchangeably with aquatic wetland.

Pond – used interchangeably with aquatic wetland and includes shallow lakes (< 2m deep).

Reach – Seven reaches within Kinbasket Reservoir are recognized: Canoe Reach, Mica Arm, Wood Arm, Sullivan Arm, Kinbasket Reach, Beaver Mouth, and Bush Arm. Canoe Reach, Mica Arm, and Bush Arm are the focus of this study.

Position – refers to whether a wetland, site, or transect is located within the footprint of Kinbasket Reservoir (elevation ≤ 754.4 m ASL; **DDZ**) or outside/above (> 754.4 m ASL; **REF**).

Target Site/Target Wetland – wetlands or sites within the 753 to 754 m ASL elevation band (Figure 10-1).

Control Site – (not to be confused with a BACI "Control") wetlands within the reservoir but not within the 753 to 754 m ASL elevation band. In terrestrial wetlands control sites are located in either the 752 to 753 or 754 to 755 m ASL elevation bands.

Upper Control – wetlands within the 754 to 755 m ASL elevation band.

Lower Control – wetlands within the 752 to 753 m ASL elevation band.

Reference Site/Reference Wetland (REF) – wetlands above 755 m ASL.

Index Site (Site) – wetlands to be monitored under CLBMON-61 including wetlands within the drawdown zone that will be impacted by reservoir activity as well as reference wetlands. For aquatic wetlands, an index site will be a discrete pond. For terrestrial wetlands, an index site will include the control, target and reference elevation bands.

Table 10-1: The relationship between the CLBMON-61 wetland type and the Canadian and BC wetland classification systems (National Wetlands Working Group 1988; MacKenzie and Moran 2004).

CLBMON 61 Wetland Type	NWWG Site Class	BC Wetland Associations*	Environmental Characteristics	Vegetation Types
Terrestrial Wetland	Bog	Wb associations (e.g., Wb01)	Ombrotrophic □pH < 5.5□ > 40 cm fibric/mesic peat	Sphagnum mosses, ericaceous shrubs, and conifers
	Fen	Wf associations (e.g., Wf01)	Groundwater-fed□pH > 5.0 > 40 cm fibric/mesic peat	Deciduous shrubs, sedges, and□brown mosses
	Swamp	Ws associations (e.g., Wb01)	Mineral soils or well-humified peat Temporary shallow flooding (0.1–1.0 m)□Significant water flow	Conifers, willows, alders, forbs, grasses, leafy mosses
	Marsh	Wm associations (e.g., Wb51)	Mineral soils or well-humified peat Protracted shallow flooding (0.1–2.0 m)	Large emergent sedge, grass, forb, or horsetail species
Aquatic Wetlands (ponds)	Shallow Waters	Various descriptions	Permanent deep flooding (0.5–2 m)	Planmergent and submerged macrophytes; emergent vegetation < 10% cover

***MacKenzie and Moran (2004)**

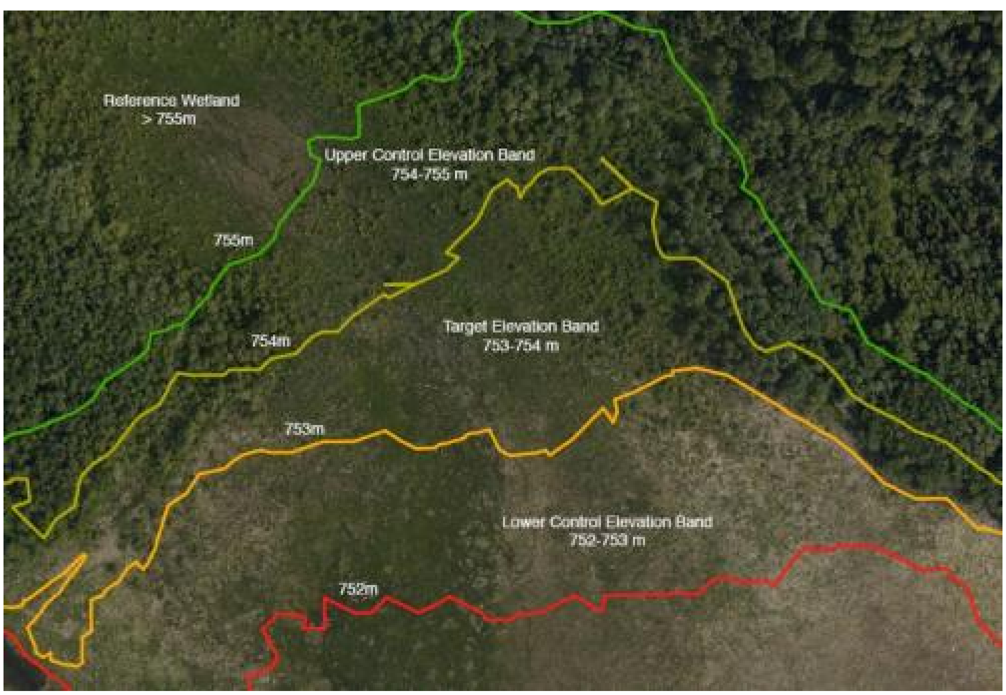

Figure 10-1: Sampling strata of an index monitoring site showing target, control, and reference elevation bands

Vegetation Community/association – plant assemblages characterized by similar species composition and per cent cover. Vegetation communities are delineated into vegetation polygons. Includes definition of dominant species.

Transect – sampling unit for sampling terrestrial wetlands.

Sample stations – Sampling location within aquatic wetlands/ponds.

Wetland integrity – To have integrity, a wetland should be relatively unimpaired across a range of characteristics and spatial and temporal scales. Ecological integrity can also be defined as the "structure, composition, and function of an ecosystem as compared to reference ecosystems operating within the bounds of natural or historic disturbance regimes" (Faber-Langendoen et al. 2008).

Wetland composition – The relative abundance of different flora and fauna species that characterize the structure of the biological community of a wetland. Composition can be expressed as per cent cover, per cent biomass, or the relative abundance (per cent) of species.

Wetland productivity – Primary productivity is the capture and storage of solar energy by autotrophic plants via photosynthesis. Secondary productivity involves the transfer and storage of primary production to higher trophic levels (e.g., heterotrophs). For the purposes of CLBMON-61, we use vegetative biomass as a measure of primary productivity and the diversity and abundance of aquatic macroinvertebrates as a measure of secondary productivity. Adapted from Sala and Austin (2000).

Wetland complex – a grouping of two or more adjacent or connected wetland community's sharing a common water source.

BACI (Before-After-Control-Impact) – A repeated measures study design with spatial replication of treatment and control sites and temporal replication with measurements before and after a treatment application or impact. Under CLBMON-61, "target" sites can be thought of as "treatment sites" for the purposed of the BACI study design.

Control (BACI) – A "control" under a BACI study is a spatial replicate of a treatment (target) site.

Pre-Installation – the time period prior to the completion and operation of Units 5 and 6.

Pre-Impact – the time period prior to when reservoir operations have been influenced by the construction and operation of Units 5 and 6.

Post-Impact – the time period after reservoir operations have been influenced by the construction and operation of Units 5 and 6.

Gross Primary Production (GPP) – *Gross primary production* is the amount of chemical energy as biomass that primary producers create in a given length of time. (GPP is sometimes confused with Gross Primary *productivity*, which is the rate at which photosynthesis or chemosynthesis occurs)

Net Ecosystem Production (NEP) – the total metabolic balance of an ecosystem; the difference between gross primary production and respiration

Respiration (R) or Ecosystem Respiration (ER) – is the sum of all cellular respiration occurring by the living organisms in a specific ecosystem.

Pond Metabolism– pond metabolism represents how energy is created (primary production) and used (ecosystem respiration) within an aquatic wetland.

10.2 Index Sites

10.2.1 Valemount Peatland, Canoe Reach

The Valemount Peatland is a remnant fenland located at the north end of Kinbasket Reservoir and Canoe Reach, 10km south of Valemount. Approximately 300 hectares in size, the area extends west of the Canoe River channel (745 m ASL) to an elevation 760 m ASL (Figure 10-2); approximately 90 per cent of the wetland complex occurs below full pool (754.4 m ASL). The Valemount Peatland is comprised of vegetation communities that reflect both the historic fenland complex that existed prior to inundation and the elevation gradient within the reservoir (Moody and Carr 2003, Hawkes et al. 2010). As suggested in the site name, peat is the dominant substrate; however, wood debris and wood fragments blanket portions of the remnant fenland excluding vegetation growth (Hawkes et al. 2010). Wildlife use of the Valemount Peatland has been well documented and it is known to provide important habitat for birds, Moose and White-Tailed Deer, Wolves, small mammals, reptiles, amphibians (Western Toad, Columbia Spotted Frog, and Long-Toed Salamander), and rare plants (Ham 2010; MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012). The terrestrial wetland communities observed in the four elevation bands are depicted in Figure 10-3 and are described below. A description of the aquatic wetlands and macrophyte communities follows.

Lower control (752 – 753 m ASL):

- Vegetation cover was low (13.9 % herb and 1.1 % shrub); substrate was dominated by moss (59.6 %), accompanied by fine organic dead matter (22.5%), water (8.5%), and wood (6.3%).
- *Equisetum fluviatile* was the dominant herb (5.8 %), accompanied by *Carex aquatilis*, *Comarum palustre, Scirpus microcarpus*, and *Menyanthes trifoliate* between 1 and 3 %. With a cover of 1.0 %, *Salix pedicularis* was the dominant shrub.
- Classification of the community corresponded most closely to the Swamp Horsetail (SH) association of Hawkes et al. (2007) but due to the influence of reservoir did not fit well into the classification Mackenzie and Moran (2004).

Target (753 – 754 m ASL):

- Substrate, vegetation composition and cover was similar to the low control elevation band but with a greater cover of water. Vegetation cover was low (13.8 % herb and 0.8 % shrub); substrate was dominated by moss (48.9%) accompanied by fine organic dead matter (17.7%), water (17.7%), and wood (8.1%).
- *C. aquatilis* and *C. palustre*, and *E. fluviatile* were the dominant herbs (4.4%, 3.9%, and 3.8%, respectively), accompanied by *M. trifoliate*, *Utricularia intermedia*, and *S. microcarpus. S. pedicularis* was the dominant shrub with a cover of 0.4%.
- Classification of the community corresponded most closely to the SH association of Hawkes et al. (2007) but due to the influence of reservoir did not fit well into the classification Mackenzie and Moran (2004).

Upper control (754 – 755 m ASL)

- Vegetation composition and cover of the upper control elevation band was similar to the low control and target elevation bands despite differences in substrate cover. Vegetation cover was low (16.5 % herb and 0.6 % shrub); substrate was dominated by wood (57.6%) accompanied by fine organic dead matter (17.5%), live organic (16.5%) and water (7.1%).

- *Comarum palustre*, *M. trifoliate,* and *C. aquatilis* were the dominant herbs (6.6%, 4.8%, and 3.3%, respectively) accompanied by *E. fluviatile* and *Calamagrostis Canadensis*. *Salix lucida* was the dominant shrub with a cover of 0.4%.
- Classification of the community corresponded most closely to the SH and Driftwood associations of Hawkes et al. (2007) but due to the influence of reservoir did not fit well into the classification Mackenzie and Moran (2004).

Reference (755 – 757 m ASL):

- Vegetation composition and cover of the reference community was distinctly different from the communities within the reservoir. Herb and shrub cover were much higher in reference community (43.7 % and 24.5, respectively); substrate was dominated by moss (63.2%) accompanied by fine organic dead matter (19.5%), live organic excluding moss (9.1 %) and water (6.6%). Wood cover was < 0.01 %.
- *Menyanthes trifoliate* was the dominant herb (35.8%) accompanied by *Carex lasiocarpa* (3.1 %) and *C. aquatilis* (2.7%). Dominant shrubs included *Betula pumila* (11.6 %) and *Myrica gale* (6.9%) accompanied by *Rhododendron groenlandicum* (3.6%), *Picea mariana* (2.5%), and *Oxycoccus oxycoccos* (1.3%).
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely with the Black Spruce–Buckbean–Peatmoss Bog (Wb11) and Scrub birch–Buckbean–Shoresedge (Wf07) associations of Mackenzie and Moran (2004).

Aquatic Wetlands

Numerous small ponds occur throughout the complex, although many are merely shallow depressions (< 30 cm) in the eroding peat (Hawke and Tuttle 2012). A deep pond occurs at the south end of the complex in the target elevation band and was identified as an index pond for aquatic sampling in 2012 (Adama et al. 2012, Figure 10-4). This pond was 0.83 ha in size and has a mean depth of 80 cm; pH was neutral (pH 7.2). The dominant planmergent vegetation in this pond was *Nuphar polysepala* accompanied by *Potamogeton natans*. The submergent vegetation was sparse and dominated by *Potamogeton zosteriformis*. The community was classified as the NUPLUT association following the classifications of both Mackenzie and Moran (2004) and Pierce and Jensen (2001). The submergent community did not fit within either classification and was tentatively described as the POTZOS association. A thin margin of emergent vegetation (*Carex aquatilis)* borders the interior edge of the pond. The pond was surrounded by a low layer (< 2 m tall) of vegetation characteristic of the 753 m ASL terrestrial wetland community. The origin of this pond is uncertain; however, the presence of a concrete slab at the west end indicates anthropogenic modification that predates the reservoir.

A small beaver pond at 757 m ASL provides a paired reference to the monitoring pond within the reservoir (Figure 10-4). This pond was 0.08 ha in size and has a mean depth of 86 cm; pH was slightly alkaline (pH 7.7). The dominant planmergent vegetation in the reference pond was *Nuphar polysepala* accompanied by *Potamogeton natans*. The submergent vegetation was virtually absent except along the margins of the ponds where an extensive stand of emergent vegetation occurs, dominated by *Carex aquatilis*. The community was classified as the NUPLUT community following the classifications of both Mackenzie and Moran (2004) and Pierce and Jensen (2001). Tall dense riparian shrub and mixed forest surrounded the pond.

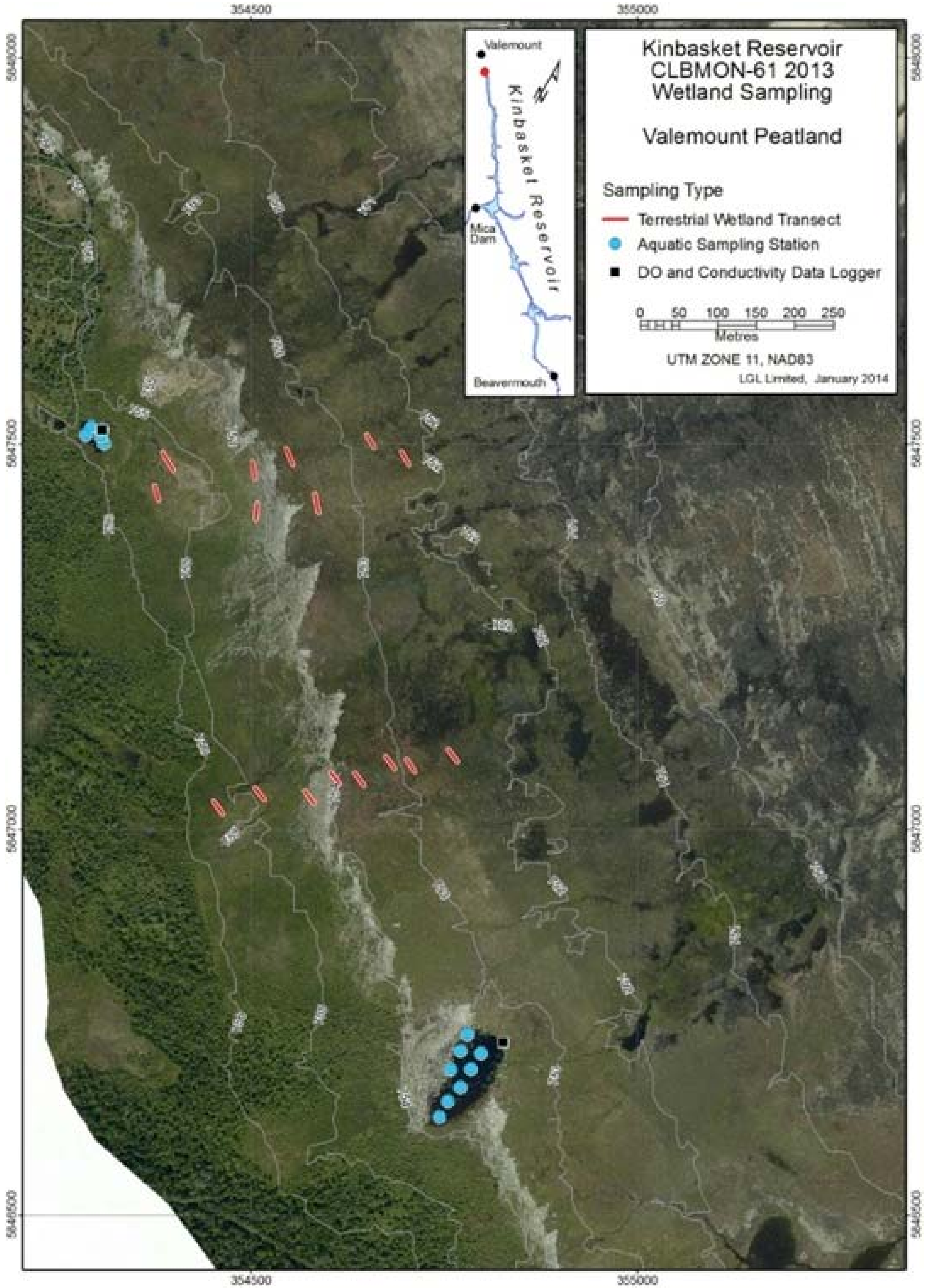

Figure 10-2: Location of aquatic and terrestrial wetland sampling sites in the Valemount Peatland (Canoe Reach), 2013.

Lower Control 752 m ASL

Target Elevation Band 753 m ASL

Upper Control 754 m ASL

Reference Community 755 m ASL

Figure 10-3: Representative images of terrestrial wetland transects in the Valemount Peatland.

Figure 10-4: Representative images of paired index ponds in the Valemount Peatland: pond within the reservoir @ 753 m ASL (above), reference pond @ 756 m ASL (below).

10.2.2 Sprague Bay Wetlands, Mica Arm

Located 8km east of Mica Dam, the Sprague Bay wetlands are comprised of a narrow fenland/beaver pond complex extending from 760 to 752 m ASL (Figure 10-5). Beaver dams bisect the complex creating a series of ponds, fens, and riparian benches that step down into the reservoir. The entire complex is approximately 9 hectares of which two-thirds is located above the reservoir and is mostly comprised of a floating fen. Wildlife values have not been thoroughly documented; however, based on our brief observations Black Bear, Beaver, Moose, and a variety of songbirds utilize these wetlands. These wetlands also provides highly productive breeding habitat for the Western Toad (Hawkes and Tuttle 2012). The terrestrial wetland communities observed in the four elevation bands are depicted in Figure 10-6 and are described below. A description of the aquatic wetlands and macrophyte communities follows.

Lower control (752 – 753 m ASL):

- Transects in the lower control bisected the flood zone of a beaver pond. Vegetation cover was low with a prominent shrub layer (5.6 % herb and 22.2 % shrub); substrate was dominated by fine organic dead matter (58.9%), accompanied by live organic (26.3 %) and wood (11.7%).
- *Carex aquatilis and Scirpus microcarpus* were the dominant herbs (2.7 % and 2.2%); *Spiraea douglasii* was the single dominant shrub species (22.2%).
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely with the Pink Spiraea-Carex sitchensis (Ws50) association of Mackenzie and Moran (2004).

Target (753 – 754 m ASL):

- Vegetation cover was moderate (42.8 % herb and 32.1 % shrub); substrate was dominated by live organic (67.9%), dead organic (17.9 %) and moss (10.3%).
- *Carex aquatilis, Lysichiton americanus, S. microcarpus* and *C. palustre* were the dominant herbs (15.3%, 10.6%, 8.8%, and 4.2%); *Spiraea douglasii* was the dominant shrub (30.0%) accompanied by *Alnus incana*, *Cornus stolonifera,* and *Lonicera involucrate*
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely to the Ws50 association of Mackenzie and Moran (2004).

Upper control (754 – 755 m ASL)

- Vegetation cover was moderately low (22.5 % herb and 16.4 % shrub); substrate was dominated by moss (60.3%) accompanied by live organic (30.5%), and organic dead matter (8.5%).
- *Comarum* palustre (7.2%) was the dominant herb accompanied by *Platanthera dilatata*, *C. lasiocarpa, Lysichiton americanus*, *Trientalis europaea*, *Lycopus americanus*, and *Eriophorum angustifolium* (all between 1% – 4%). *Spiraea douglasii* was the dominant shrub (12.7%) accompanied by *Alnus incana* and *M. gale* (1 % – 3%).
- The vegetation community did not fit into the classification of Hawkes et al. (2007) nor into the classification of Mackenzie and Moran (2004).

Reference (755 – 757 m ASL):

- Vegetation cover was moderately low with no shrub cover (20.3 % herb and 0.0 %

shrub); substrate was dominated by moss (57.5%) accompanied by water (17.0%), organic dead matter (12.9%), and live organic (12.6%).

- *Menyanthes trifoliate* was the dominant herb (13.0%) accompanied by *Carex limosa, C. utriculata, Comarum palustre*, and *Scheuchzeria palustris*.
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely with the Scheuchzeria-Peat-moss (Wb12) association of Mackenzie and Moran (2004).

Aquatic Wetlands

The reference complex is comprised of a floating fen adjacent an acidic beaver pond (pH 6.1; Figure 10-7). The pond was 0.9 ha and had a mean depth of 168 cm. A 2-meter high beaver dam supports the entire complex, which has likely been maintained by Beaver for many decades. Aquatic vegetation was sparse with only *Potamogeton pusillus* detected infrequently and in low abundance (< 1%). *Nuphar polysepala* occurs in some areas of the pond but was not detected in our plots. A mature cedar-hemlock forest bounded the southwest shoreline and the Sprague Bay FSR ran along the northeast shoreline accompanied by a thin margin of forest and shrub.

Downstream of the reference pond, a complex of beaver ponds extends into the reservoir from 755 to 752 m ASL. The total area of the open water was 0.04 ha; mean depth was 68 cm. The planmergent vegetation in the DDZ pond complex included *Potamogeton pusillus, Nuphar polysepala* accompanied by *Equisetum fluviatile* and *Sparganium angustifolium.* These species were also present in the submergent layer along with *Utricularia macrorhiza*. The aquatic vegetation community was tentatively classified as NUPLUT –SPARANG (Pierce and Jensen 2001). A dense alder shrub community surrounded the margin of the pond complex.

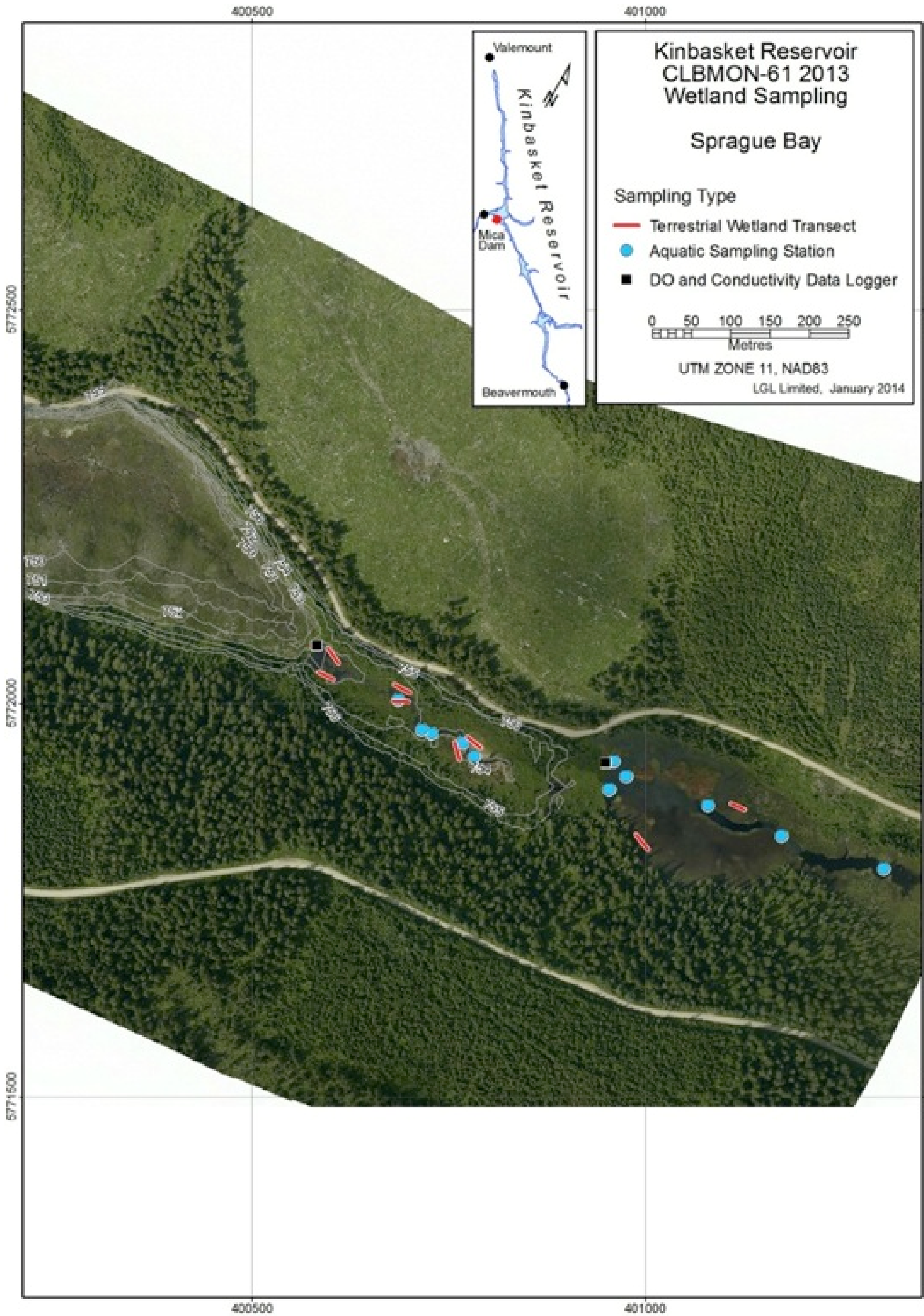

Figure 10-5: Location of aquatic and terrestrial wetland sampling sites at Sprague Bay (Mica Arm), 2013

Lower Control 752 m ASL

Target Elevation Band 753 m ASL

Upper Control 754 m ASL

Reference Community 755 m ASL

Figure 10-6: Representative images of the terrestrial wetland transects at Sprague Bay.

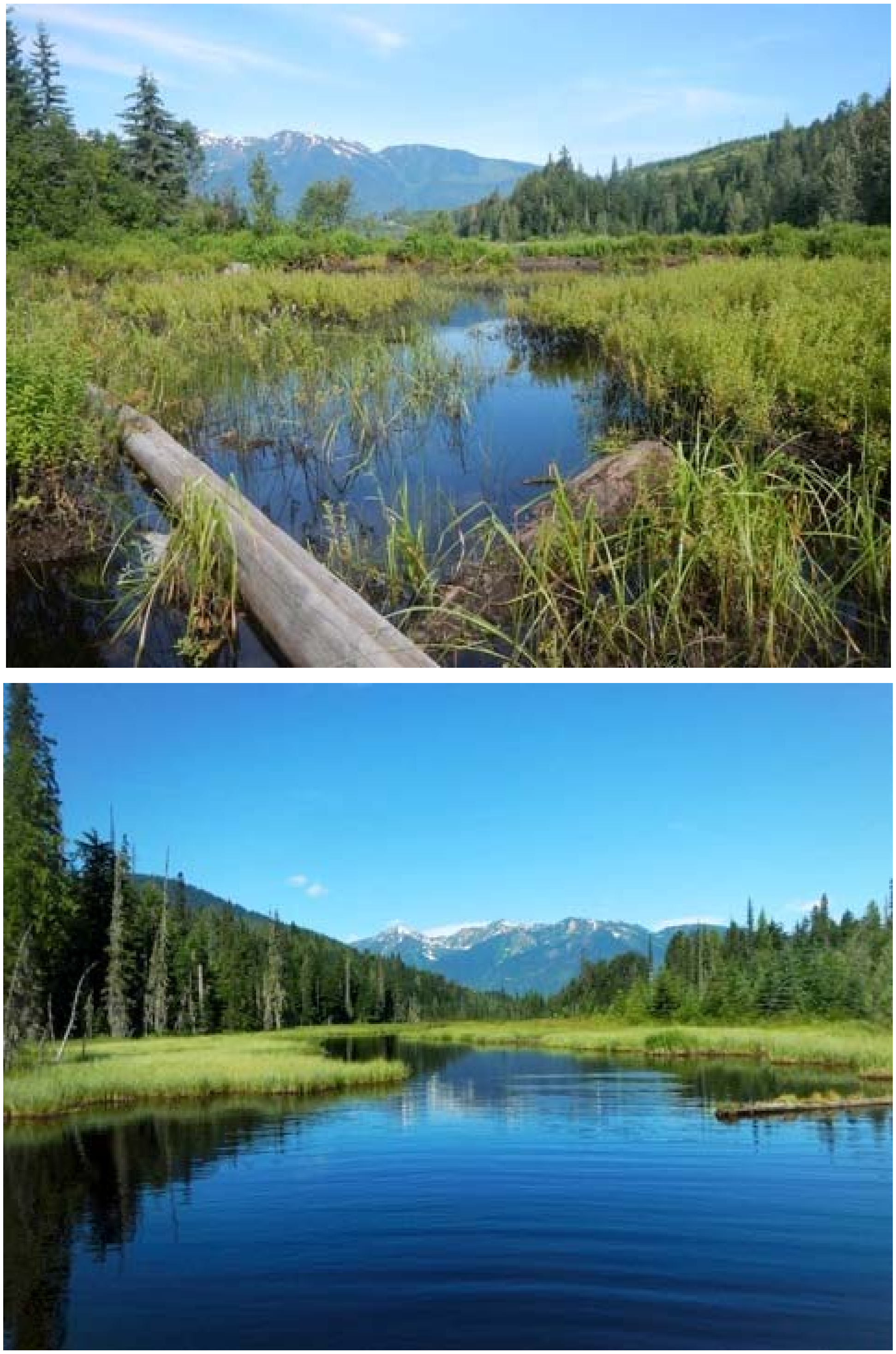

Figure 10-7: Representative images of paired index ponds at Sprague Bay: pond within the reservoir @ 753 m ASL (above), reference pond @ 756 m ASL (below).

10.2.3 Km 88 Wetlands, Bush Arm

The Km 88 wetland is a small 25 ha complex of fens and beaver dams located north of Bear Island in Kinbasket Reservoir (Figure 10-8). A series of beaver ponds occur above and step down into the reservoir. The DDZ ponds are positioned at 752 and 753 m ASL; the reference pond is positioned approximately 500m upslope at 780 m ASL. In 2013, recent beaver activity was not evident. Due to seepage, fens and swamps have developed downstream of the ponds. The largest of these extends from 757 to below 750 m ASL and is the site of the terrestrial wetland transects (Figure 10-8). Wildlife values of the Km 88 wetlands are moderate to high and provide habitat for Black Bear, Beaver, River Otter, Moose, White-Tailed Deer, Western Toad, Columbia Spotted Frogs, and a variety of birds (MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012). *Liparis loeselii*, a red listed orchid was documented in the terrestrial wetland transects (Ibid). The terrestrial wetland communities observed in the four elevation bands are depicted in Figure 10-11 and are described below. A description of the aquatic wetlands and macrophyte communities follows.

Lower control (752 – 753 m ASL):

- Vegetation in the lower elevation band was high in herb cover (68.0 %) and very low in shrub cover (0.03 %); substrate was dominated by moss (53.6%), accompanied by live organic (30.0 %), and water (15.7%).
- The vegetation cover in this elevation band reflected the hydric site conditions. Menyanthes trifoliate was the dominant herb (58.8%) accompanied by *Utricularia intermedia*, *Eleocharis elliptica*, *Equisetum arvense*, *Utricularia macrorhiza*, and *Phalaris arundinacea*.
- The vegetation community corresponded to the Buckbean–Slender Sedge association of Hawkes et al. (2007) and the Slender Sedge–Buckbean association of Mackenzie and Moran (2004).

Target (753 – 754 m ASL):

- Vegetation cover in the target elevation band was similar to the lower control band but with more shrub (65 % herb and 1.6 % shrub). Substrate cover was also similar to the lower control band and was dominated by moss (57.1%), accompanied by live organic (12.5 %), and water (12.9%). The cover of dead organic matter and wood increased in the target elevation band but were still low (8.5% and 3.1%).
- The vegetation cover in this elevation band also reflected hydric site conditions although the composition of subdominant species differed from the lower elevation band. Menyanthes trifoliate was the dominant herb (53.9%) accompanied by *Typha latifolia* (10.4%), *Equisetum palustre*, *Utricularia intermedia*, *Eleocharis elliptica*, *Equisetum arvense*, *Utricularia macrorhiza*, and *Phalaris arundinacea*. Prominent shrubs included *Betula pumila* and *Salix pedicellaris* (1.2 % and 0.6%).
- The vegetation community corresponded to the WS association of Hawkes et al. (2007) but did not fit into the classifications of Mackenzie and Moran (2004).

Upper control (754 – 755 m ASL)

- Vegetation cover in the upper control elevation band was moderately high. Herb cover was similar to the lower control and target elevation bands (63.5%) but shrub cover was much higher (44.7%). Substrate cover was dominated by moss and live organic (48.9% and 40.5 %), accompanied by minor amounts of dead organic and wood (3.6% and 1.6%).
- *Menyanthes trifoliate* (23.9%), *Petasites frigidus var. sagittatus* (16.9%), and

Trichophorum alpinum (16.8%) were the dominant herbs accompanied by *E. fluviatile* (6.4%), *Carex interior* (4.1%), *Maianthemum stellatum* (3.2%), and *Lysichiton americanus* (3%). A diverse shrub layer was dominated by *Betula pumila* (15.2%), *Rhododendron groenlandicum* (12.1%), accompanied by *Thuja plicata* (7.3%), *Picea engelmannii x glauca* (4.5%), and *Alnus incana ssp. tenuifolia* (2.6%). The red listed orchid *Liparis loeselii* was recorded in this elevation band.

- The vegetation community corresponded to the WS association of Hawkes et al. (2007) and loosely corresponded to the Scrub Birch–Water Sedge (Wf02) and Lodge Pole Pine–Water Sedge–Peatmoss (Wb07) associations of Mackenzie and Moran (2004).

Reference (755 – 757 m ASL):

- Vegetation cover in the reference community (a fen meadow) was high and was dominated by herbs (75.6%); shrub cover was only 3.2 %. Substrate cover was dominated by moss and live organic matter (56.5% and 23.5 %), accompanied by minor amounts of dead organic and water (3.3% and 1.8%).
- *Menyanthes trifoliate* was the dominant herb (72.9%) accompanied by *Trichophorum alpinum* (6.9%), *E. palustre* (2.9%), *Cicuta douglasii* (2.0%), and *Eleocharis elliptica* (2.0 %). *Betula pumila* (2.9%) was the dominant shrub. The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely to the Shore Sedge – Buckbean – Hook-moss (Wf08) and Hudson Bay Clubrush–Red hook moss (Wf10) associations of Mackenzie and Moran (2004).

Aquatic Wetlands

The Km 88 DDZ ponds comprise two small remnant beaver ponds that occur between 752 and 753 m ASL. The DDZ ponds were 0.1 ha in size and had a mean depth of 139 cm; pH was alkaline (pH 8.4). Planmergent vegetation was sparse with *Potamogeton richardsonii* and *E. fluviatile* (3.5% and trace); Submergent cover was 100 % and diverse with *Potamogeton pusillus* (28.5%), *Chara* (24.0%), *Rana aquatilis* (24.0%), and *Myriophyllum sp.* (22.0%), *Potamogeton richardsonii* (1.5%), and *Utricularia macrorhiza* (trace). The macrophyte community was classified as a CHARA–RANAQU community (Pierce and Jensen 2001; Mackenzie and Moran 2004). An herb community dominated by *Phalaris arundinacea* surrounded the DDZ ponds. Mixed forests occurred upslope of the east and west shoreline.

The Km 88 reference pond is a beaver pond located approximately 500 meters upslope (~780m ASL) of the DDZ pond along the same watercourse. The reference pond was 0.4 ha and had a mean depth of 120 cm; pH was alkaline (pH 8.1). Planmergent vegetation was sparse with *Carex utriculata* (1.3 %) and *Eleocharis palustre* (trace); *Chara* (19.1%) was the dominant submergent and was accompanied by *Rana aquatilis* (9.1%), *Carex utriculata* (1.7 %) and *Eleocharis palustre* (trace). The macrophyte community was classified as a CHARA–RANAQU community (Pierce and Jensen 2001; Mackenzie and Moran 2004). Dense stands of riparian shrub and conifers surrounded the margins of the pond.

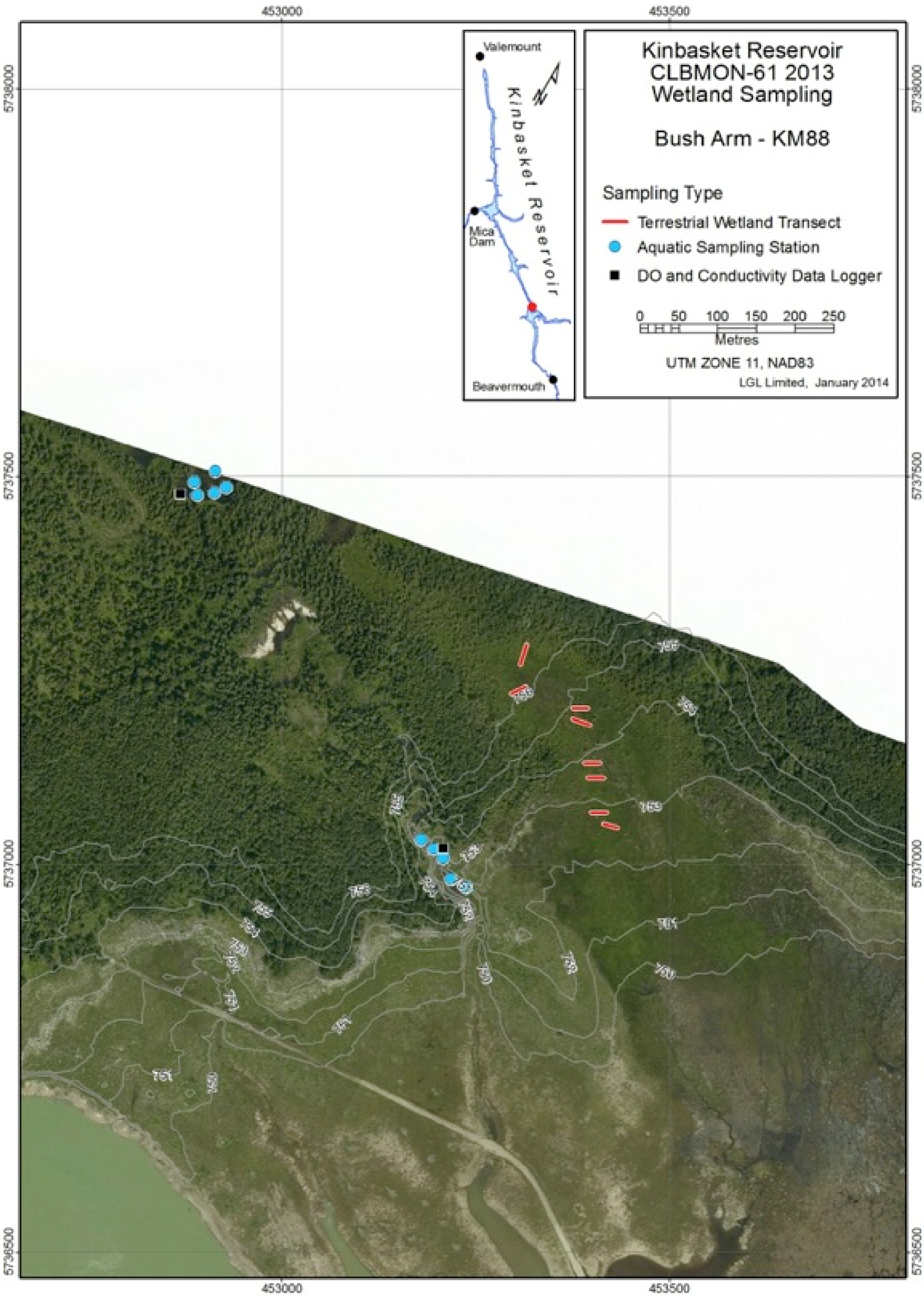

Figure 10-8: Location of aquatic and terrestrial wetland sampling sites at Km 88 (Bush Arm), 2013.

Lower Control 752 m ASL

Target Elevation Band 753 m ASL

Upper Control 754 m ASL

Reference Community 755 m ASL

Figure 10-9: Representative images of terrestrial wetland transects at Km 88, 2013.

Figure 10-10: Representative images of paired index ponds at Km 88, 2013: pond within the reservoir @ 752 m ASL (above), reference pond @ 780 m ASL (below).

10.2.4 Bush River Wetlands, Bush Arm

The Bush River wetlands occur downstream of the confluence of the Bush and Valenciennes Rivers (770 m ASL) to the Bush River causeway (752 m ASL; Figure 10-11, Figure 10-12). The DDZ pond and terrestrial wetlands are located adjacent the causeway (753 m ASL) and are frequently inundated during spring freshet. Wetlands also occur on the west side of the causeway at 752 m ASL and lower and are prone to accumulations of wood debris. The reference wetland occurs 3.3 km upstream of the DDZ pond. This wetland complex consists of a 4.1 ha shallow lake bounded by fenlands that extend to the Bush River There is evidence of a natural spring and of old beaver activity. The wildlife values of Bush River wetlands are high. They are known to provide habitat for Grizzly Bear, Black Bear, Elk, Western Toad, Long-toed Salamander, Columbia Spotted Frog, small mammals, and a variety of birds (MacInnis et al. 2011; van Oort et al. 2012; Hawkes et al. 2012). *Liparis loeselii was* documented in the reference wetland.

The terrestrial wetland communities observed in the four elevation bands are depicted in Figure 10-13 and are described below. A description of the aquatic wetlands and macrophyte communities follows.

Lower control (752 – 753 m ASL):

- The two transects in the lower control elevation band occurred on sites with quite different ground cover. The ground cover on transect 61.35 was primarily mineral soil (85.1%); whereas wood was the primary ground cover on transect 61.36 (96.5%). Vegetation cover was low on both transects (6.6% and 1.7%, respectively). Shrubs cover was low (0.16 % transect 61.35) to none (transect 61.36).
- *Carex* and *Equisetum spp.* were dominant in both transects: *C. saxatilis* and *C. viridula* occurred in 61.35 at 2.6%; *C. aquatilis*, *E. palustre*, and *E. fluviatile* all occurred (0.5%) in 61.36.
- The vegetation communities in the lower control band corresponded to Driftwood (DR) and Willow Shrub (WS) associations of Hawkes et al. (2007) but did not fit into the classification of Mackenzie and Moran (2004).

Target (753 – 754 m ASL):

- As in the lower control elevation band, the two transects in the target elevation band occurred on sites with quite different ground cover. The ground cover on transect 61.33 was primarily moss (88.5%); whereas the ground cover on transect 61.34 was predominately organic dead matter (51%) and mineral soil (33.5%). Vegetation cover overall was similar among the two transects (19.1% and 10.6 %); mean herb and shrub covers were 14.9 % and 3.3 %.
- *Carex utriculata* was the dominant herb in transect 61.33, accompanied by *E. fluviatile, C. aquatilis*, and *E. palustre* between 1% and 4 %; Salix farriae was the dominant shrub (2.9%). *C. viridula,* was the dominant herb in transect 61.34, accompanied by *C. saxatilis* and *Eleocharis mamillata* between 1% and 3 %; an unidentified Salix was the dominant shrub (2.6 %).
- The vegetation community in the target elevation band corresponded to Swamp Horsetail (SH) and Willow Shrub (WS) associations of Hawkes et al. (2007). Transect 61.33 corresponded to the Swamp Horsetail–Beaked Sedge (Wm02) association of Mackenzie and Moran (2004).

Upper control (754 – 755 m ASL)

- Vegetation cover in this elevation band was moderate (19.6 % herb and 25.9 % shrub); substrate was dominated by moss (49.0%) accompanied by organic dead matter, (27.1%), live organic (22.5%) matter, and wood (1.5%).
- *Equisetum palustre* (12.1%) was the dominant herb accompanied by *Rubus pubescens* (4.8%), *C. aquatilis* (2.0%), and *Fragaria virginiana* (1.9%). A diverse shrub community included *Salix commutate* (8.2%) and *S. farriae* (8.0%), accompanied by *Populus trichocarpa*, *Rosa acicularis, Lonicera involucrate*, *Cornus stoloniferous,* and *S. maccalliana* between 1 and 4%.
- The vegetation community corresponded to the Willow-Shrub (WS) associations of Hawkes et al. (2007) and Mackenzie and Moran (2004).

Reference (755 – 757 m ASL)

- Vegetation cover in this elevation band was moderate (33.1 % herb and 8.6 % shrub); substrate was dominated by dead and live organic matter (55.0% and 42.3%) accompanied by moss (8.8%) and wood (2.3%).
- *Carex utriculata* was the dominant herb (20.0%) accompanied by *C. lenticularis ssp. lipocarpa*, *C. lasiocarpa, C. intermedia, Packera plattensis*, and *Eleocharis mamillata* between 1 and 4%. *Salix farriae* (4.6 %) and an unidentified *Salix sp*. (4.0%) were the most prominent shrubs. The red listed orchid *Liparis loeselii* occurred in the reference transects.
- The terrestrial vegetation community did not corresponded to the classifications of Hawkes et al. (2007) and Mackenzie and Moran (2004). The plant association and edatopic characterization indicates this community may be transitional between the Wf01 and Willow-Sedge community.

Aquatic Wetlands

The DDZ pond was located immediately adjacent the Bush River causeway to the east. The pond was 0.4 ha in area and had an average depth of 44.7 cm; pH was alkaline (pH 8.7) and had the characteristics of a marl wetland. The planmergent macrophytes in the DDZ pond included *Potamogeton pusillus* and *Equisetum fluviatile* (10.0% and 3.9%); submergent species included *Chara* (54.6%) and *P. pusillus* (11.7%), and *Equisetum fluviatile* (0.4%). The macrophyte community was classified as a *Chara* community (Pierce and Jensen 2001; Mackenzie and Moran 2004). Terrestrial vegetation varied along the perimeter of the pond from exposed mudflat, riparian shrub, and coniferous forest. The Bush Rush FSR bounded the western shoreline.

The reference pond is located 3.2 km upstream of the DDZ pond. The pond was 4.1 ha and had an average depth of 67 cm; pH was alkaline (pH 8.7) and had the characteristics of a marl wetland. Planmergent or emergent macrophytes did not occur in the sample plots and *Chara* was the dominant submergent (34.4%) accompanied by *Stuckenia pectinata* and an unidentified species of *Eleocharis*, both < 1%. The macrophyte community was classified as a *Chara* community (Pierce and Jensen 2001; Mackenzie and Moran 2004). Adjacent vegetation varied from closed canopy coniferous forest to fen wetlands.

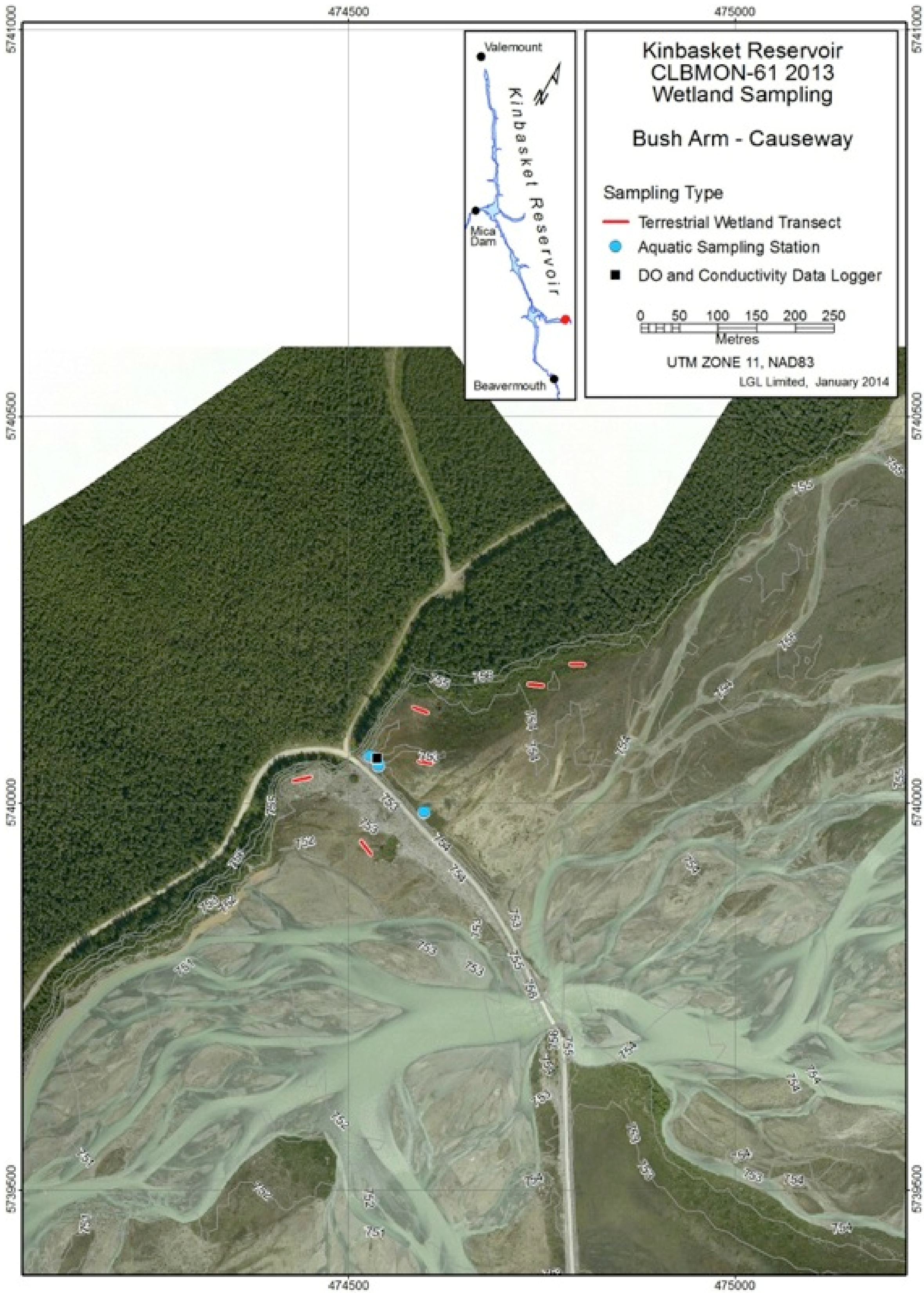

Figure 10-11: Location of aquatic and terrestrial wetland sampling sites at the Bush River causeway, 2013.

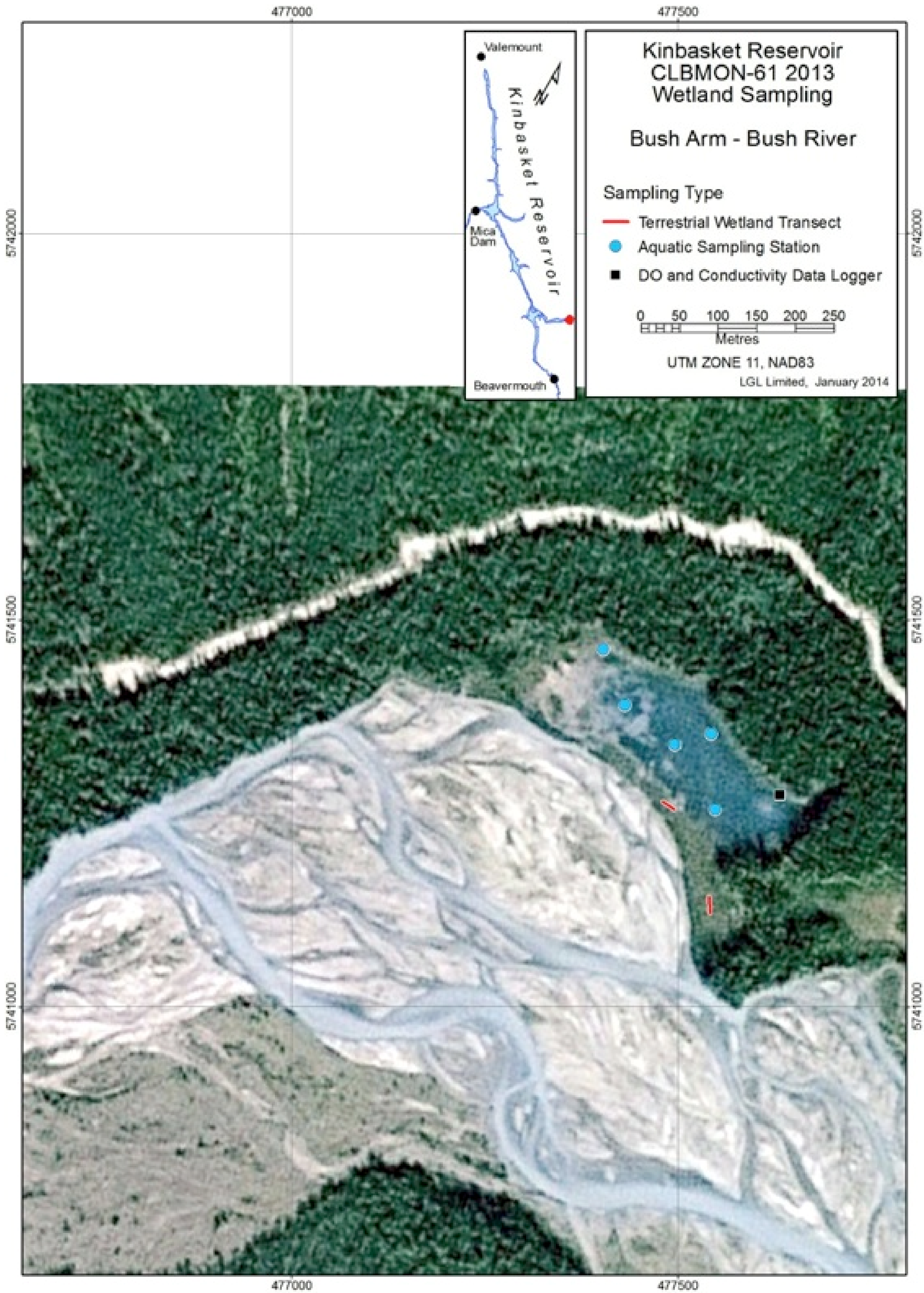

Figure 10-12: Location of aquatic and terrestrial wetland sampling sites at the Bush River reference wetland, 2013. Image from GoogleEarth.

Lower Control 752 m ASL

Target Elevation Band 753 m ASL

Upper Control 754 m ASL

Reference Community 755 m ASL

Figure 10-13: Representative images of the terrestrial wetland transects along the Bush River, 2013.

Figure 10-14: Representative images of paired index ponds along the Bush River, 2013: pond within the reservoir @ 753 m ASL (above), reference pond @ 760 m ASL (below).

10.3 Data Forms

Project ID	CLBMON-10: Kinbasket Reservoir Inventory of Vegetation Resources					
Date		Transect #		Quadrat #		Tran Brg
Surveyors		VCC (2007 / 08)		Photo Nos.		

Vegetation Cover	%	TREE LAYER (A)									
Tree Layer (A)		Spp	A1	A2	A3	Tot	Spp	A1	A2	A3	Tot
Shrub Layer (B)											
Herb Layer (C)											
Moss / Seedling (D)											

SHRUB LAYER (B)											
Spp Code	B1	B2	Tot	Spp Code	B1	B2	Tot	Spp Code	B1	B2	Tot

HERB LAYER (C)						Moss Layer		NOTES
Spp Code	%	Spp Code	%	Spp Code	%	Spp Code	%	

SUBSTRATE (Must Equal 100%)	Type (General) Rock ☐ Cobble ☐ Gravel ☐ Sand ☐ Silt ☐ Fines ☐ Wood ☐						
Organic Matter – Live		Organic Matter - Dead		Decay Wood		Bedrock	
Rock		Mineral Soil		Water		Other	

LGL PROJECT: EA3398 SURVEY: CLBMON-61 Kinbasket Wetlands Assessment DATE: ________-________-________ TIME: ______:______

SURVEY CODE: ______________ STUDY AREA: ______________________ POND/WETLAND #: ________________ PLOT #: ________________

LOCATION: WAYPOINT #: ______________ UTM: 11 ____________________ / ____________________ SURVEYORS: ____________________

WATER PHYSIOCOCHEMISTRY:
SURFACE: DO: ____________ WATER TEMP: ______________ CONDUCTIVITY: _____________ PH: ________________
DEPTH: (CM) ___________ DO: ____________ WATER TEMP: ______________ CONDUCTIVITY: _____________ PH: ________________

WATER DEPTH: ____________________ SUBSTRATE DEPTH: ____________________ TOTAL DEPTH: ______________ SECCHI DEPTH: ___________
SUBSTRATE TYPE: ________________ PROBE: NORTH: _____________ EAST: ______________ SOUTH: ________________ WEST: __________________

INVERT COLLECTION LABELS: BENTHIC: ________________ PELAGIC: ______________________
BIOMASS COLLECTION? Y ☐ N ☐ BIOMASS COLLECTION LABEL: ______________________

VEGETATION PRESENT? Y ☐ N ☐ FISH PRESENT? Y ☐ N ☐ AMPHIBIANS PRESENT? Y ☐ N ☐ SPECIES: __________________________

Type	#	*VEG ABUND	% Cover	SPECIES	**REL ABUND	% COVER	Strata Code

Substrate Codes

F = Fines (Clay/Silt)
S = Sand
SG = Small Gravel
LG = Large Gravel
C = Cobble
B = Boulder
BR = Bedrock
M = Muck
CO = Coarse Organic Detritus
W = Wood

Abundance Codes

*VEG Abundance

1 Trace
2 Small
3 Large

**Relative Abundance in Sample

T Trace
1 <10%
2 10-20%
3 20-50%
4 50-75%
5 75-100%

Strata Codes

B = Benthic
S = Submergent
F = Floating
E = Emergent

10.4 Calculation of NEP and GPP from diel dissolved oxygen data.

Primary productivity was calculated and expressed as g O_2 m^{-2} day^{-1}. Where:

$\Delta O_2/\Delta t$ = GPP – R– D
O_2 = rate of change in dissolved oxygen (note: mg/l = g/m^2)
t = time (hour)
GPP = rate of primary production
R = rate of respiration
D = rate of oxygen exchange from other factors

Primary productivity was not corrected for D (diffusion, advection, and other factors) and the equation was simplified to:

$$\Delta O_2/\Delta t = GPP - R$$

Where:

GPP(mg/l(day)) = $NEP_{daytime}$ + $R_{daytime}$
$NEP_{daytime}$ = NEP_{hr} (mg O_2 /l(hr)) during daylight hours x dayfraction x 24
NEP_{hr} (mg O_2 /l(hr)) = ΔO_2 (mg O_2 /l(hr))

$R_{daytime}$ (mg O_2 /l(daylight period)) = R_{day} x 24 x dayfraction
R_{day} (mg O_2 /l(day)) = R_{hr} x 24
R_{hr} (mg O_2)/l(hr) = mean NEP_{hr} during darkness
NEP = GPP – (R_{day} + R_{night})
GPP = $NEP_{daytime}$ + $R_{daytime}$
NEP(mg/l(day)) = GPP – R_{day}
dayfraction = proportion of a 24-h period when it is light (light hours /24)

Day light hours were estimated using the Center for Biosystems Modelling (CBM) modified Schoolfields equation (Forsythe et al. 1995) for latitude (L) and date (J):

$$\theta = 0.2163108 + 2\tan^{-1}[0.9671396\tan[0.00860 \times (J - 186)]], \quad (1)$$

$$\phi = \sin^{-1}[0.39795\cos\theta], \quad (2)$$

$$D = 24 - \frac{24}{\pi}\cos^{-1}\left[\frac{\sin\frac{p\pi}{180} + \sin\frac{L\pi}{180}\sin\phi}{\cos\frac{L\pi}{180}\cos\phi}\right], \quad (3)$$

R_{day} was calculated from mean NEP_{hr} during the hours of darkness and extrapolated over a 24-hr period. Positive NEP_{hr} ($\Delta O_2/\Delta t$) values during the hours of darkness were omitted, as photosynthesis cannot occur in the absence of light. Positive NEP_{hr} can result from advective transport of DO and can cause net increases in hourly NEP during darkness leading to negative R-values, which is theoretically erroneous (Staehr et al. 2010).

10.5 Weather Data

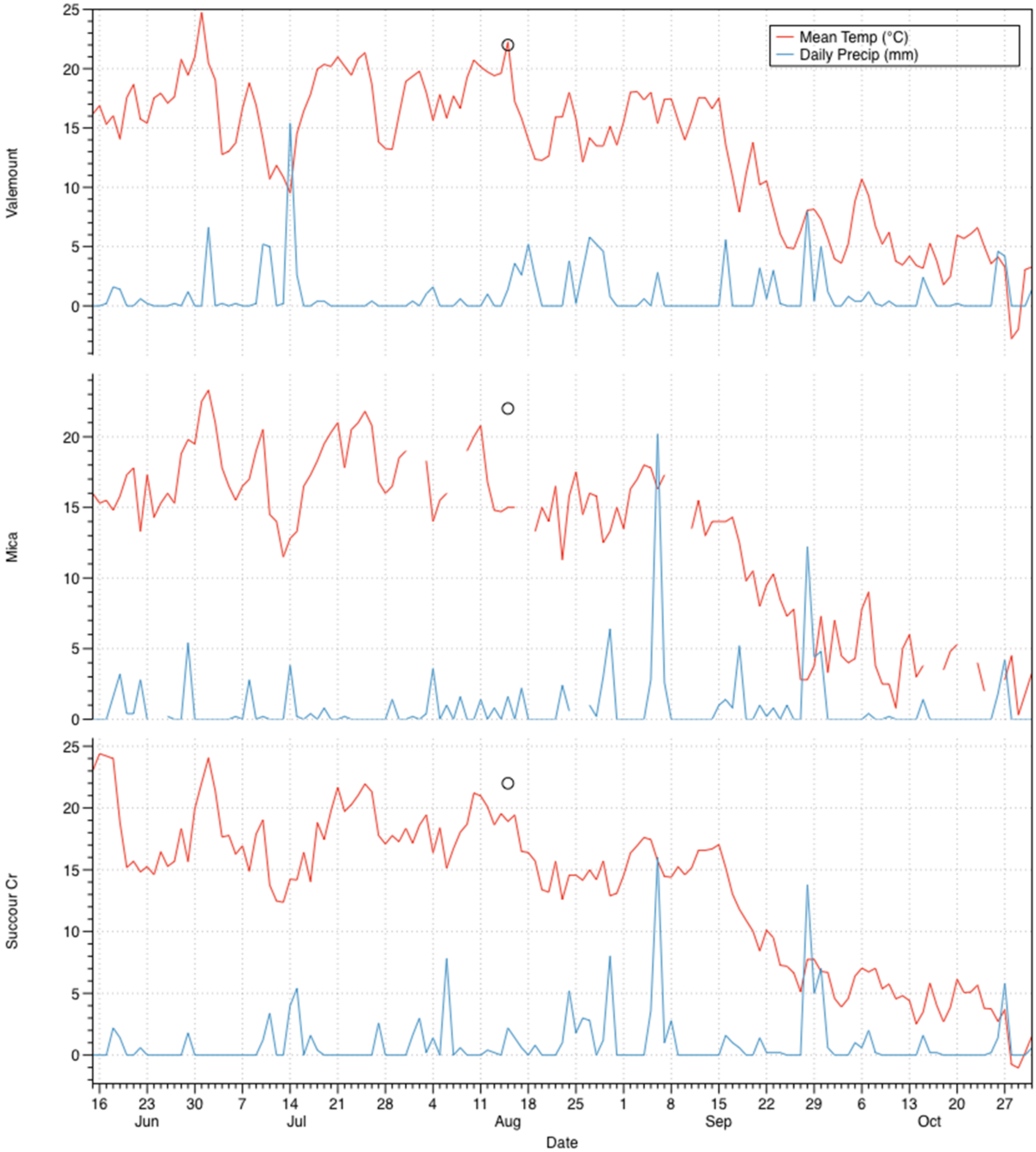

Figure 10-15. Mean daily temperature (°C) and precipitation (mm) at weather stations located at Succour Cr. (Bush Arm), Mica Dam, and the town of Valemount (Canoe Reach). Estimated date of inundation of 753 m ASL is denoted with a circle (August 15th, 2013)

10.6 Supplementary results for the analyses of terrestrial wetland vegetation data

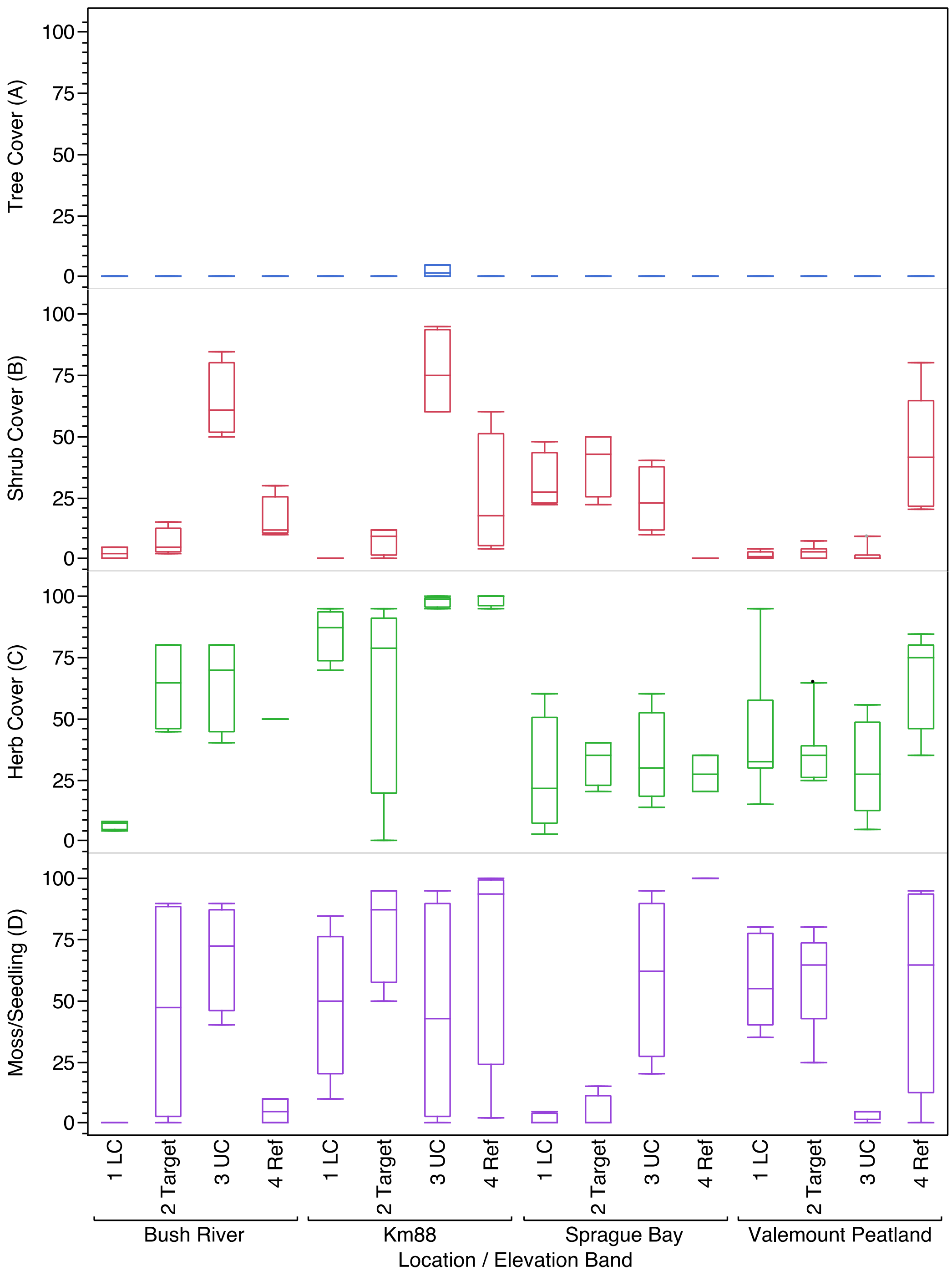

Figure 10-16. Box plots of tree, shrub, herb, and moss per cent cover by elevation and site from circular plot data. n = 4 plots per elevation band per location except in the Valemount Peatland where n = 8 plots.

Table 10-2. Results of non-parametric Kruskal-Wallis and multiple comparisons for differences in vegetation cover (%) by elevation band and site (α = 0.10) with 90 % confidence intervals. M.C. = significant differences in mean ranks, ordered from highest (a) to lowest (c).

	Tree % Cover		Shrub % Cover		Herb % Cover		Moss % Cover	
TREATMENT	Mean	M.C.	Mean	M.C.	Mean	M.C.	Mean	M.C.
ELEVATION BAND df = 3	p = 0.28		**p = 0.03**		**p = 0.09**		**p = 0.08**	
Lower control	0.0	-	7.2 ± 5.1	b	40.9 ± 12.5	b	32.8 ± 12.3	b
Target	0.0	-	11.6 ± 6.1	b	46.4 ± 9.7	b	49.5 ± 13.6	ab
Upper Control	0.4 ± 0.5	-	33.5 ± 13.1	a	51.4 ± 12.3	ab	36.2 ± 14.7	b
Reference	0.0	-	25.5 ± 9.5	a	62.0 ± 10.4	a	58.0 ± 17.1	a
SITE df = 3	p = 0.14		**p = 0.07**		**p < 0.001**		**p = 0.07**	
Bush River	0.0	b	22.2 ± 11.8	a	46.4 ± 11.9	b	30.0 ± 16.6	b
Km 88	0.5 ± 0.6	a	27.2 ± 14.8	a	86.3 ± 10.8	a	61.6 ± 16.4	a
Sprague Bay	0.0	b	23.6± ± 8.0	a	30.0 ± 6.7	c	41.3 ± 19.8	ab
Valemount Peatland	0.0	b	12.2 ± 6.4	b	44.1 ± 7.0	b	43.9 ± 9.9	ab

Table 10-3. Spearman Rank Correlations (r_s) between vegetation layer cover (%) and substrate cover (%). Significant correlations (α = 0.10) are indicated in bold.

	Live OM*		Dead OM*		Wood		Mineral Soil		Water	
	r_s	p	r_s	p	r_s	p	r_s	p	r_s	p
Trees	-0.050	0.323	0.040	0.484	0.070	0.176	-0.010	0.794	0.080	0.119
Shrubs	0.350	**<0.001**	0.100	**0.052**	-0.290	**<0.001**	-0.010	0.834	-0.380	**<0.001**
Herbs	0.570	**<0.001**	-0.230	**<0.001**	-0.400	**<0.001**	-0.200	**<0.001**	-0.080	0.121
Moss	0.740	**<0.001**	-0.320	**<0.001**	-0.390	**<0.001**	-0.170	**<0.01**	-0.100	**0.051**

* OM = Organic matter

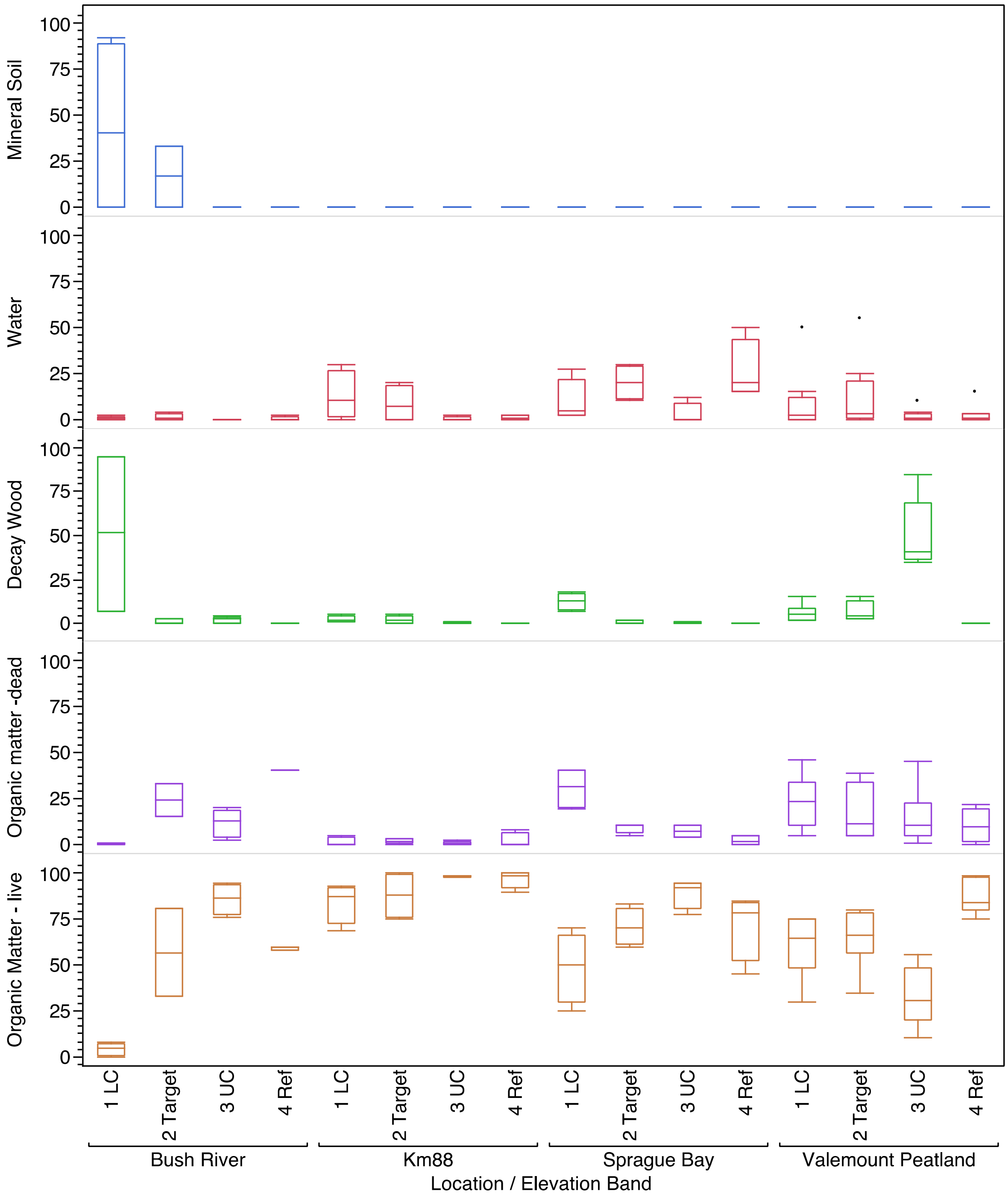

Figure 10-17. Box plots of ground cover (per cent) by elevation and site from circular plots. n = 4 plots per elevation band per location except in the Valemount Peatland where n = 8 plots.

Table 10-4. Results of non-parametric Kruskal-Wallis and multiple comparisons for differences in substrate per cent cover by elevation band and site (α = 0.10). M.C. = significant differences in mean ranks, ordered from highest (a) to lowest (c); OM = organic matter.

	Live OM % Cover		Dead OM % Cover		Wood % Cover		Mineral Soil % Cover		Water % Cover	
TREATMENT	Mean	M.C.	Mean	M.C.	Mean	M.C.	Mean	M.C.	Mean	M.C.
ELEVATION BAND df = 3	p = 0.01		p = 0.70		p < 0.001		p = 0.24		p = 0.01	
Lower control	51.9 ± 11.7	c	16.0 ± 6.2	-	15.6 ± 10.7	ab	8.6 ± 10.3	-	8.1 ± 5.1	a
Target	68.9 ± 7.3	bc	13.9 ± 4.9	-	3.4 ± 1.7	bc	3.3 ± 3.9	-	11.0 ± 5.5	a
Upper Control	67.9 ± 12.3	ab	9.9 ± 4.2	-	20.1 ± 10.6	ab	0.0	-	1.6 ± 1.3	b
Reference	80.5 ± 6.4	a	12.9 ± 6.1	-	0.4 ± 0.02	c	0.0	-	6.7 ± 4.8	ab
SITE df = 3	p < 0.001		p < 0.001		p = 0.01		p = 0.01		p = 0.001	
Bush River	51.6 ± 14.6	c	19.0 ± 7.1	b	13.7 ± 14.0	ab	14.9 ± 13.2	a	0.6 ± 0.5	c
Km 88	91.8 ± 4.4	a	1.4 ± 1.0	a	1.5 ± 0.7	c	0.0	b	5.7 ± 4.0	bc
Sprague Bay	70.2 ± 8.7	b	12.1 ± 5.5	b	3.3 ± 2.6	bc	0.1	b	14.7 ± 6.1	a
Valemount Peatland	61.4 ± 7.2	bc	16.6 ± 4.1	b	15.6 ± 6.8	a	0.1	b	6.5 ± 4.0	b

Table 10-5. Results of non-parametric Kruskal-Wallis Rank Sum Tests for differences in mean ranks of shrub species richness (S), evenness (E), Shannon Diversity Index (H), and Simpson Diversity Index (D') for 2013. Significant differences from post hoc multiple comparison tests are indicated by lower-case letters (α = 0.10): differing letters indicate significant differences and same letters indicate non-significance. Subscript c = circular plot data. Bold indicates p-values below 0.10.

ELEVATION (df = 3)	**H**	**P-value**	**Lower Control 752**	**Target 753**	**Upper Control 754**	**REF 755 +**
Number of Species	-	-	10	18	35	23
S_c	7.53	0.**056**	b	ab	a	a
E_c	5.19	0.158	-	-	-	-
H_c	9.73	**0.021**	b	ab	ab	a
D'_c	7.75	**0.051**	-	-	-	-
REACH (df = 2)	**H**	**P-value**	**Bush**	**Mica**	**Canoe**	
Number of Species	-	-	40	9	15	
S_c	**9.76**	**0.008**	a	b	b	
E_c	2.54	0.280	-	-	-	
H_c	**8.32**	**0.016**	a	b	ab	
D'_c	**7.67**	**0.022**	a	b	ab	
SITE (df = 3)	**H**	**P-value**	**Bush River**	**Km 88**	**Sprague Bay**	**Valemount Peatland**
Number of Species	-	-	23	25	9	11
S_c	**9.92**	**0.042**	a*	a*	a	a
E_c	3.85	0.426	-	-	-	-
H_c	**9.27**	**0.055**	**-**	**-**	**-**	**-**
D'_c	**8.56**	**0.073**	**-**	**-**	**-**	**-**

*even though the test was significant, no differences between groups were found

Table 10-6. Results of non-parametric Kruskal-Wallis Rank Sum Tests for differences in mean ranks of herb species richness (S), evenness (E), Shannon Diversity Index (H), and Simpson Diversity Index (D') for 2013 transect data. Significant differences from post hoc multiple comparison tests are indicated by lower-case letters (α = 0.10): differing letters indicate significant differences and same letters indicate non-significance. Bold indicates significant difference α < 0.1.

ELEVATION (df = 3)	H	P-value	Lower Control	Target	Upper Control	Reference
Number of Species	-	-	46	54	72	73
S	1.21	0.750	-	-	-	-
E	3.42	0.332	-	-	-	-
H	1.78	0.619	-	-	-	-
D'	4.49	0.213	-	-	-	-
REACH (df = 2)	**H**	**P-value**	**Bush**	**Mica**	**Canoe**	
Number of Species	-	-	95	42	36	
S	**8.82**	**0.012**	**a**	**ab**	**b**	
E	1.22	0.543	-	-	-	
H	2.34	0.311	-	-	-	
D'	0.78	0.676	-	-	-	
SITE (df = 3)	**H**	**P-value**	**Bush River**	**Km 88**	**Sprague Bay**	**Valemount Peatland**
Number of Species	-	-	50	77	36	42
S	15.47	**0.004**	**b**	**a**	**b**	**b**
E	5.20	0.267	-	-	-	-
H	2.50	0.644	-	-	-	-
D'	1.48	0.831	-	-	-	-

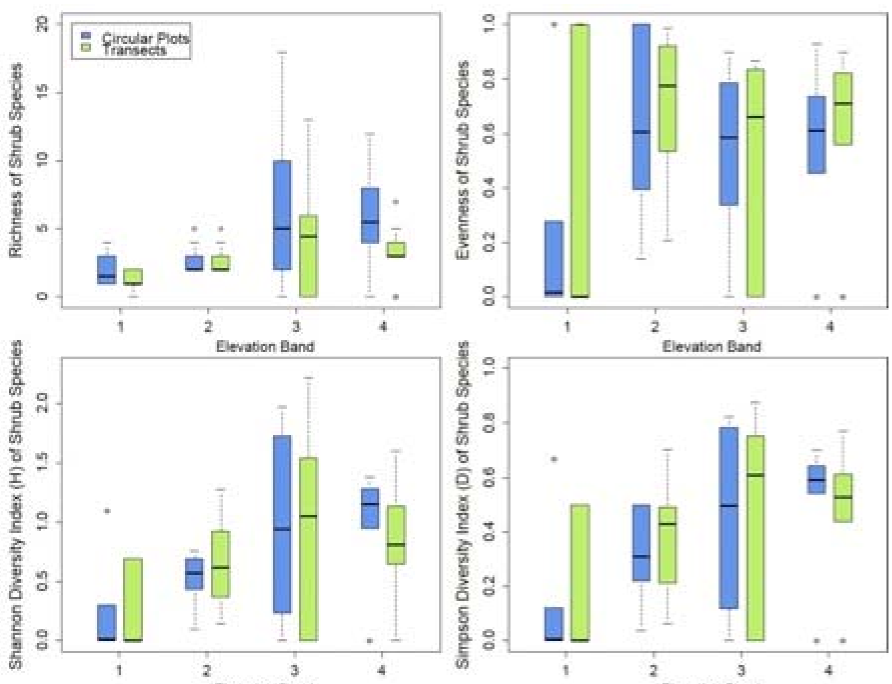

Figure 10-18. Variation in shrub species richness, evenness, and Shannon and Simpson diversity by elevation band in 2013. Data from circular plots and transects presented separately.

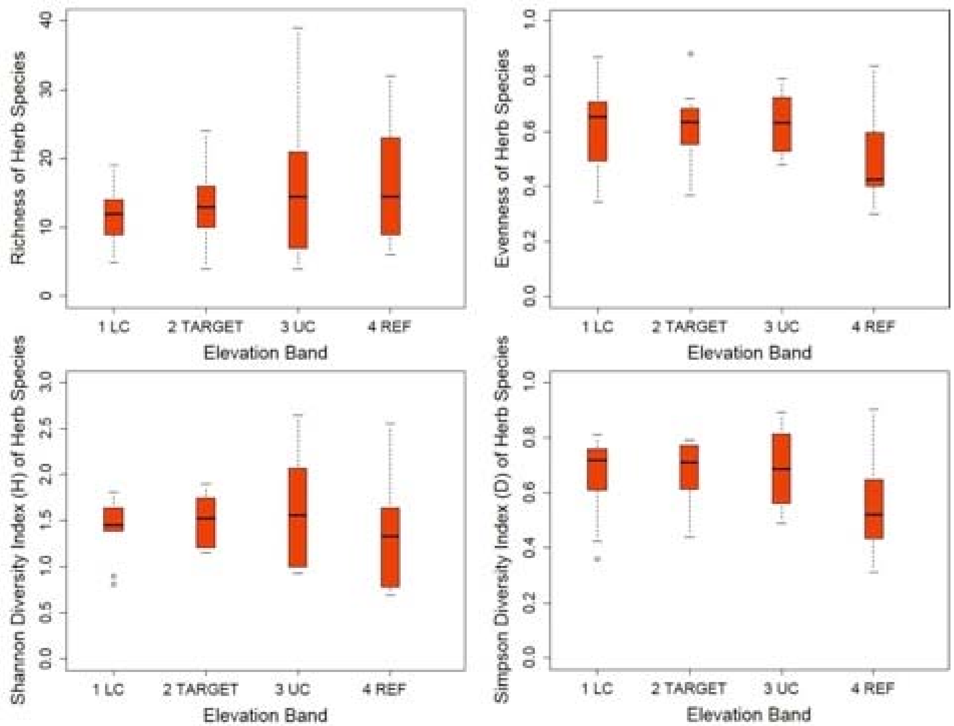

Figure 10-19. Variation in herb species richness, evenness, and diversity by elevation band in 2013.

Table 10-7. Shrub indicator species of each elevation band and site. Values correspond to indicator values (*INDVAL*) that were significant (α = 0.10) after 4999 Monte-Carlo randomizations of Indicator Species Analyses.

	Lower Control	Target	Upper Control	Reference	Bush River	Km 88	Sprague Bay	Valemount Peatland
CIRCULAR PLOTS:								
ABIELAS						37.5		
ALNUINC2			54.1				53.8	
BETUPUM				37.6		46.8		
CORNSTO			58.8					
LONIINV			42.0					
MYRIGAL				53.7				35.5
OXYCOXY				33.3				
PICEENE				31.1		40.7		
RHODGRU								
ROSAACI			47.0		35.8			
SALIX sp.			25					
SALIBRA					50			
SALICOM					75			
SALIFAR					87.5			
SALIPED						61.4		
SPIRDOU							85.7	
THUJPLI						49.8		
TRANSECTS:								
ALNUINC2			51.6					
BETUPUM				42		55.6		
CORNSTO				35.4				
JUNICOM				28.6				
MYRIGAL								
POPUTRI				28.6				
ROSAACI			42.9					
SALICOM			28.4		57.1			
SALIFAR					85.7			
SALILUC								30.8
SALIXPED								54.6
SALIX SP.					55.8			
SPIRDOU							100	
THUJPLI			38.1			33.1		

Table 10-8. Herb indicator species of each elevation band and site. Values correspond to indicator values (*INDVAL*) that were significant (α = 0.10) after 4999 Monte-Carlo randomizations of Indicator Species Analyses.

	ELEVATION BAND				SITE			
SPECIES CODE	Lower Control	Target	Upper Control	Reference	Bush River	Km 88	Sprague Bay	Valemount Peatland
AGROGIG						49.1		
CALACAN			55.3					
CALLPAL					50			
CAREAQU							56.1	
CARECHO								25
CAREFLA						34.4		
CAREGYN				35.6				
CAREINT						49.7		
CARELAS								
CARELEN		38.0						
CARELIM				30.3				
CARETEN								37.5
CAREUTR					68			
CICUDOU						68.3		
COMAPAL							43.7	
CORNCAN			30.0					
DROSROT							30.3	
ELEOELL						87.5		
ELEOMAM								
EPILPAL						37.5		
EQUIFLU								50.8
EQUIHYE					45.3			
EQUIPAL					46.8			
FRAGVIR			32.0		33.9			
GALITRD							37.1	
JUNCNOD						87.5		
LIPALOE						28.1		
LYCOAME							37.5	
LYSIAME							25.1	
MAIASTE						62		
MENYTRI				49.1		76.3		
MIMUGUT						50		
PACKPLA					34.2			
PARNFIM						50		
PRUNVUL					50			
PYROASA			30.0					
RHINMIN					37.5			
RUBUARC						34.5		
RUBUPUB			36.5		29.1			
SCHEPAL							50	
SCIRMIC	40.0							
SIUMSUA								25
SYMPBOR						46.0		
TRIAGLU						28.1		
TRICALP						37.5		
TRIEEUR							36.7	
TRIGMAR							47.0	
TYPHLAT						50		
UTRIINT	35.3					42.9		
VIOLGLA			30.2					
ZIGAELE						50		

Table 10-9. Wetland species observed in terrestrial wetland transects, 2013.

Tree Layer (A)		
	PICEENE	Picea engelmannii x glauca
	THUJPLI	Thuja plicata
Shrub Layer (B)		
	ABIELAS	Abies lasiocarpa
	ALNUINC2	Alnus incana
	AMELALN	Amelanchier alnifolia
	BETUNAN	Betula nana
	BETUOCC	Betula occidentalis
	BETUPUM	Betula pumila
	CORNSTO	Cornus stolonifera
	DASIFRU	Dasiphora fruticosa
	GAULHIS	Gaultheria hispidula
	JUNICOM	Juniperus communis
	LONICIL	Lonicera ciliosa
	LONIINV	Lonicera involucrata
	MENZFER	Menziesia ferruginea
	MYRIGAL	Myrica gale
	OXYCOXY	Oxycoccus oxycoccos
	PICEENE	Picea engelmannii x glauca
	PICEMAR	Picea mariana
	POPUTRE	Populus tremuloides
	POPUTRI	Populus balsamifera ssp. trichocarpa
	RHODGRO	Rhododendron groenlandicum
	RIBETRI	Ribes triste
	ROSAACI	Rosa acicularis
	SALI SP	Salix sp.
	SALI SP3	Salix sp.
	SALIBRA	Salix brachycarpa
	SALICOM	Salix commutata
	SALIDRU	Salix drummondiana
	SALIEXI	Salix exigua
	SALIFAR	Salix farriae
	SALIGLA	Salix glauca
	SALILUC	Salix lucida ssp.lasiandra
	SALIMAC	Salix maccalliana
	SALIMEL	Salix melanopsis
	SALIPED	Salix pedicellaris
	SALISCO	Salix scouleriana
	SALISIT	Salix sitchensis
	SALISOL	Salix sp.
	SALIX SP	Salix sp.
	SHEPCAN	Shepherdia canadensis
	SPIRBET	Spiraea betulifolia
	SPIRDOU	Spiraea douglasii
	THUJPLI	Thuja plicata
	TRIEEUR	Trientalis europaea
	VACCMYR	Vaccinium myrtilloides or Vaccinium myrtillus
	VIBUEDU	Viburnum edule

Herb Layer (C)

AGROCAP	Agrostis capillaris	JUNCNOD	Juncus nodosus
AGROGIG	Agrostis gigantea	LINNBOR	Linnaea borealis
ANAPMAR	Anaphalis margaritacea	LIPALOE	Liparis loeselii
ANGEGEN	Angelica genuflexa	LYCOAME	Lycopus americanus
ANTEPUL	Antennaria pulcherrima	LYSIAME	Lysichiton americanus
CALACAN	Calamagrostis canadensis	LYSITHY	Lysimachia thyrsiflora
CALLHER	Callitriche hermaphroditica	MAIASTE	Maianthemum stellatum
CALLPAL	Calla palustris	MAIATRI	Maianthemum trifolium
CAREAQU	Carex aquatilis	MENTARV	Mentha arvensis
CAREATH	Carex atherodes	MENYTRI	Menyanthes trifoliata
CAREAUR	Carex aurea	MIMUGUT	Mimulus guttatus
CARECAN	Carex canescens	MUHLGLO	Muhlenbergia glomerata
CARECAP	Carex capillaris	MYRIGAL	Myrica gale
CARECHO	Carex chordorrhiza	PACKPLA	Packera paupercula
CARECUS	Carex cusickii	PARNFIM	Parnassia fimbriata
CAREDIS	Carex disperma	PARNKOT	Parnassia kotzebuei
CAREFLA	Carex flava	PETASAG	Petasites frigidus var.sagittatus
CAREGYN	Carex gynocrates	PHALARU	Phalaris arundinacea
CAREINT	Carex interior	PLATAQU	Platanthera aquilonis
CARELAS	Carex lasiocarpa	PLATDIL	Platanthera dilatata
CARELEN	Carex lenticularis ssp.lipocarpa	POA PAL	Poa palustris
CARELEP	Carex leptalea	PRUNVUL	Prunella vulgaris
CARELIM	Carex limosa	PYROASA	Pyrola asarifolia
CARELIV	Carex livida	RHINMIN	Rhinanthus minor
CAREMAG	Carex magellanica	RUBUARC	Rubus arcticus
CAREPAU	Carex pauciflora	RUBUPAR	Rubus parviflorus
CARESAR	Carex sartwellii	RUBUPUB	Rubus pubescens
CARESAX	Carex saxatilis	SANIMAR	Sanicula marilandica
CARESIM	Carex simulata	SCHEPAL	Scheuchzeria palustris ssp. americana
CARETEN	Carex tenuiflora	SCIRMIC	Scirpus microcarpus
CAREUTR	Carex utriculata	SCUTGAL	Scutellaria galericulata
CAREVIR	Carex viridula	SIUMSUA	Sium suave
CAREX SP	Carex sp.	SPARNAT	Sparganium natans
CASTMIN	Castilleja miniata	SPIRROM	Spiranthes romanzoffiana
CICUDOU	Cicuta douglasii	STELLON	Stellaria longifolia
CICUMAU	Cicuta maculata	SYMPBOR	Symphyotrichum boreale
COMAPAL	Comarum palustre	SYMPFOL	Symphyotrichum foliaceum
CORNCAN	Cornus canadensis	SYMPLAE	Symphyotrichum laeve
CYPRPAR	Cypripedium parviflorum	TARAOFF	Taraxacum officinale
DESCCES	Deschampsia cespitosa	TOFIPUS	Tofieldia pusilla
DROSANG	Drosera anglica	TRIAGLU	Triantha glutinosa
DROSROT	Drosera rotundifolia	TRICALP	Trichophorum alpinum
ELEOARC	Eleocharis sp.	TRIEEUR	Trientalis europaea
ELEOELL	Eleocharis elliptica	TRIGMAR	Triglochin maritima
ELEOMAM	Eleocharis mamillata	TRIGPAL	Triglochin palustris
EPILLEP	Epilobium leptophyllum	TYPHLAT	Typha latifolia
EPILPAL	Epilobium palustre	Unk Dicot	Unknown Sp
EQUIARV	Equisetum arvense	Unk grass1	Unknown Sp
EQUIFLU	Equisetum fluviatile	UTRIINT	Utricularia intermedia
EQUIHYE	Equisetum hyemale	UTRIMAC	Utricularia macrorhiza
EQUIPAL	Equisetum palustre	UTRIMIN	Utricularia minor
EQUIVAR	Equisetum variegatum	VIOLADU	Viola adunca
ERIOANG	Eriophorum angustifolium	VIOLGLA	Viola glabella
ERIOVIR	Eriophorum viridicarinatum	VIOLMAC	Viola macloskeyi
ERUCGAL	Erucastrum gallicum	VIOLPAL	Viola palustris
FRAGVIR	Fragaria virginiana	ZIGAELE	Zigadenus elegans
GALITRD	Galium trifidum		
GALITRI	Galium triflorum		
GAULHIS	Gaultheria hispidula		
GEOCLIV	Geocaulon lividum		
GEUMMAC	Geum macrophyllum		
GLYCSTR	Glyceria striata		
JUNCALP	Juncus alpinoarticulatus		
JUNCENS	Juncus ensifolius		

10.7 Supplementary results for the analyses of pond physicochemistry

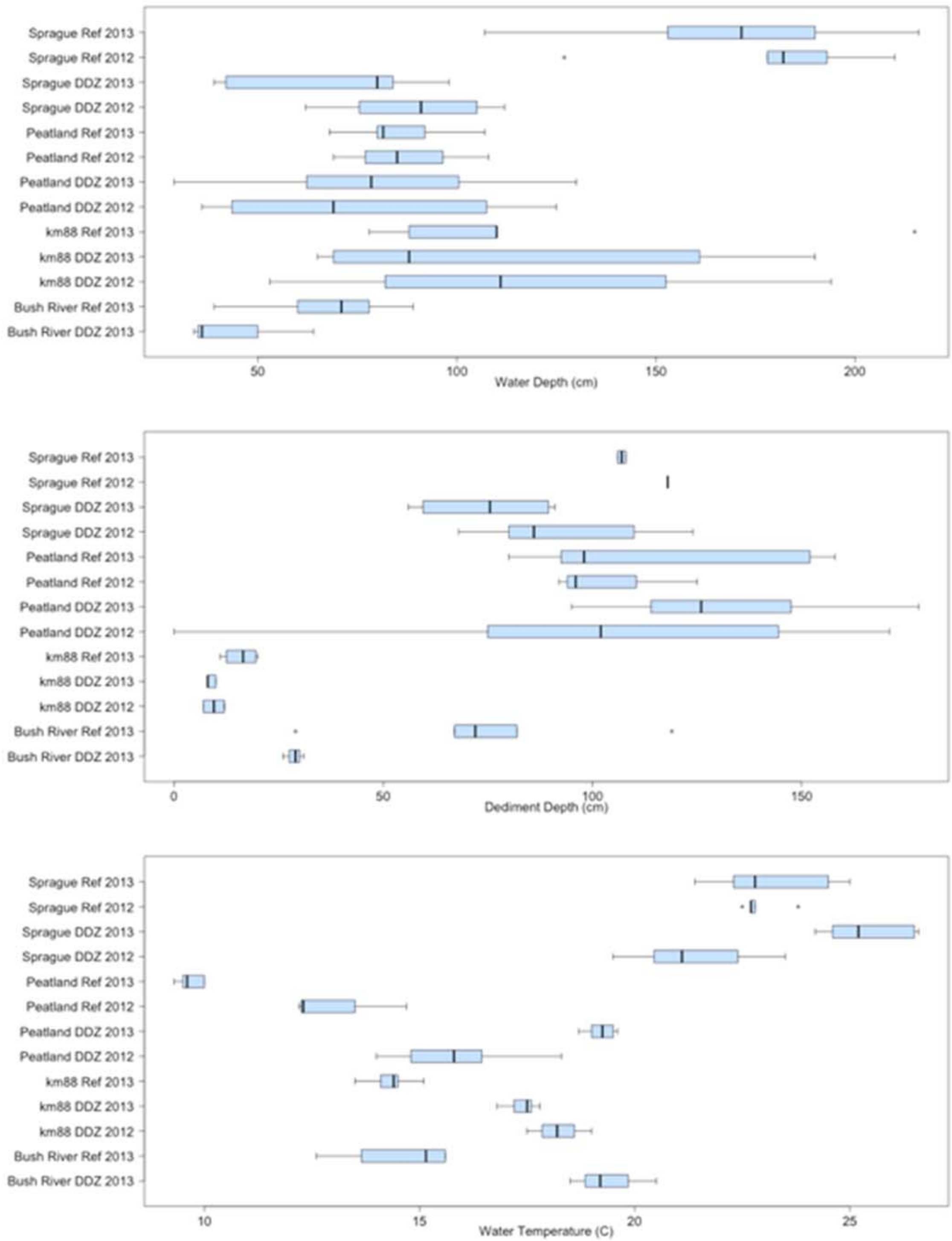

Figure 10-20. Box plots of water depth, sediment depth, and water temperature in ponds sampled in 2013. Data for ponds sampled in 2012 also included.

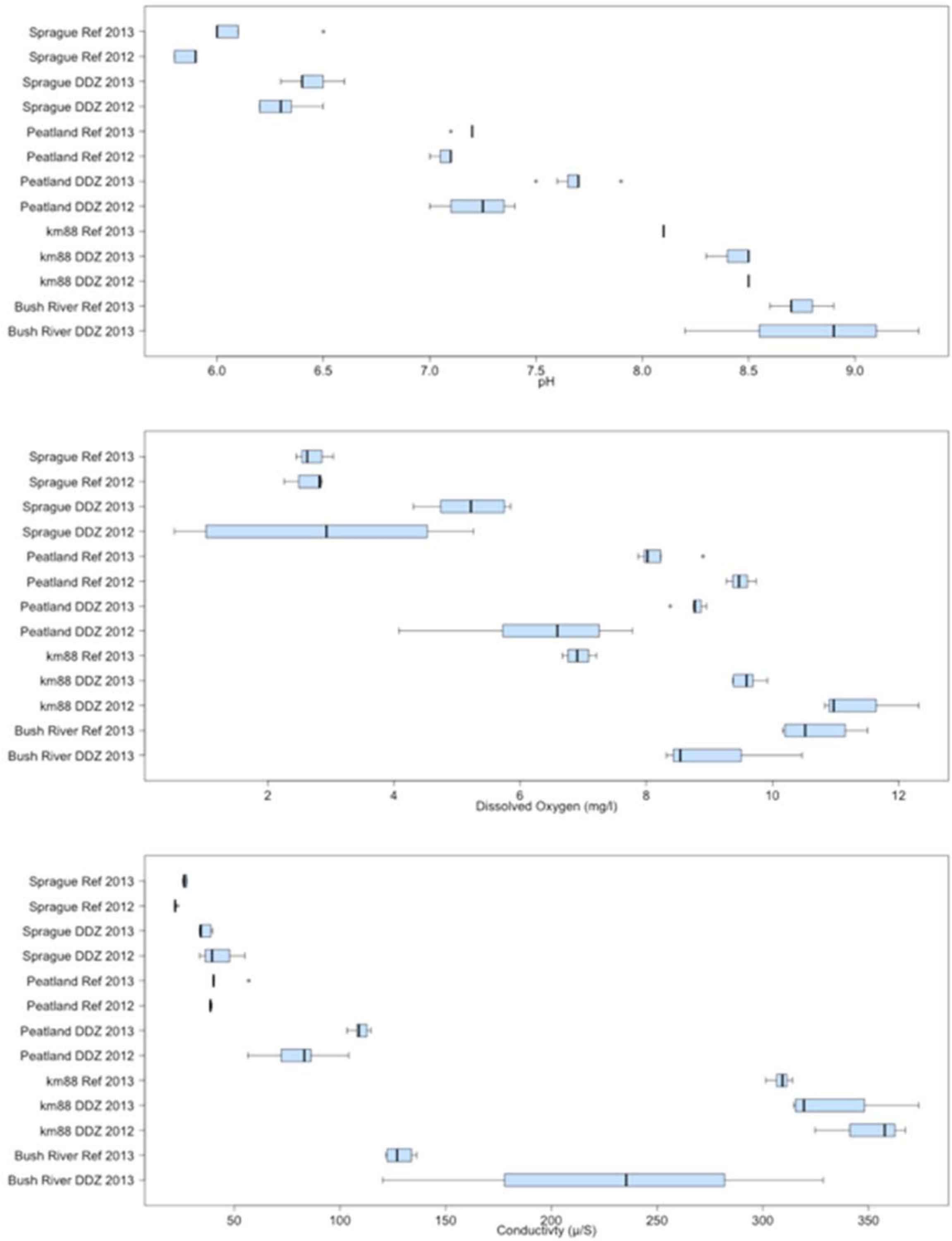

Figure 10-21. Box plots of pH, dissolved oxygen, and conductivity in ponds sampled in 2013 Data for ponds sampled in 2012 also included.

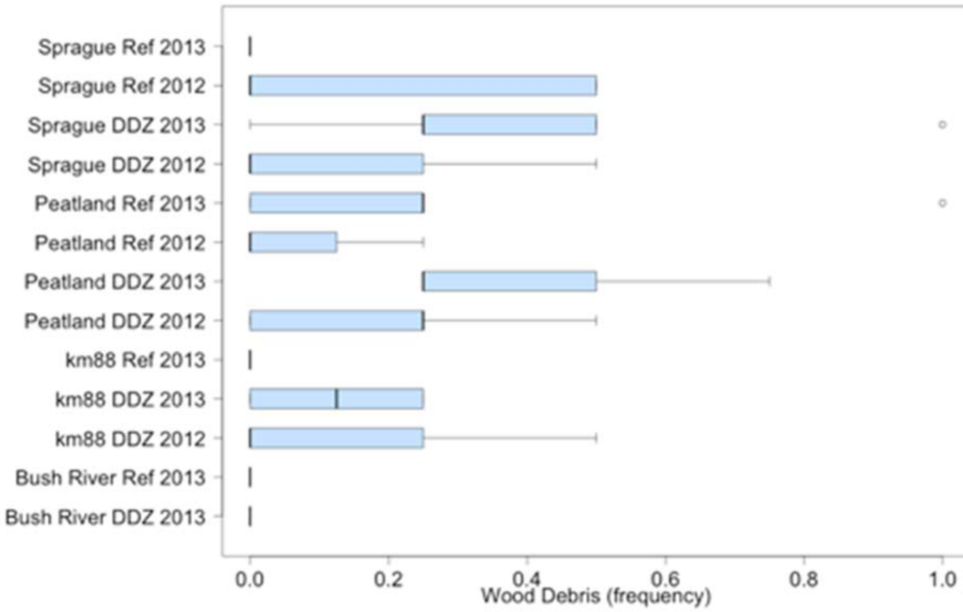

Figure 10-22. Box plots of the frequency of wood debris detected in pond sediment in ponds sampled in 2013. Data for ponds sampled in 2012 also included

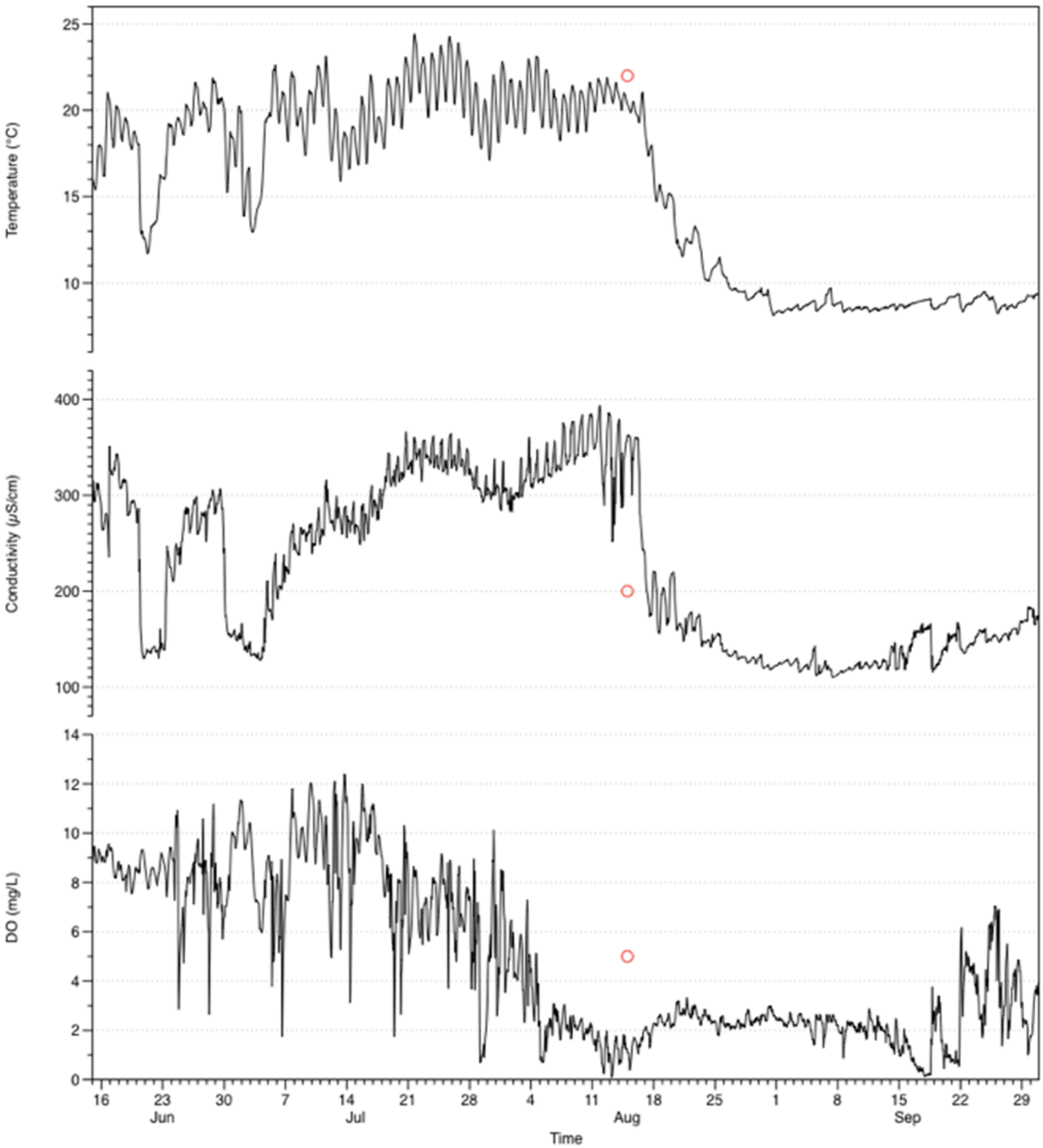

Figure 10-23. Temperature, conductivity, and dissolved oxygen for Bush River reference pond, 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL.

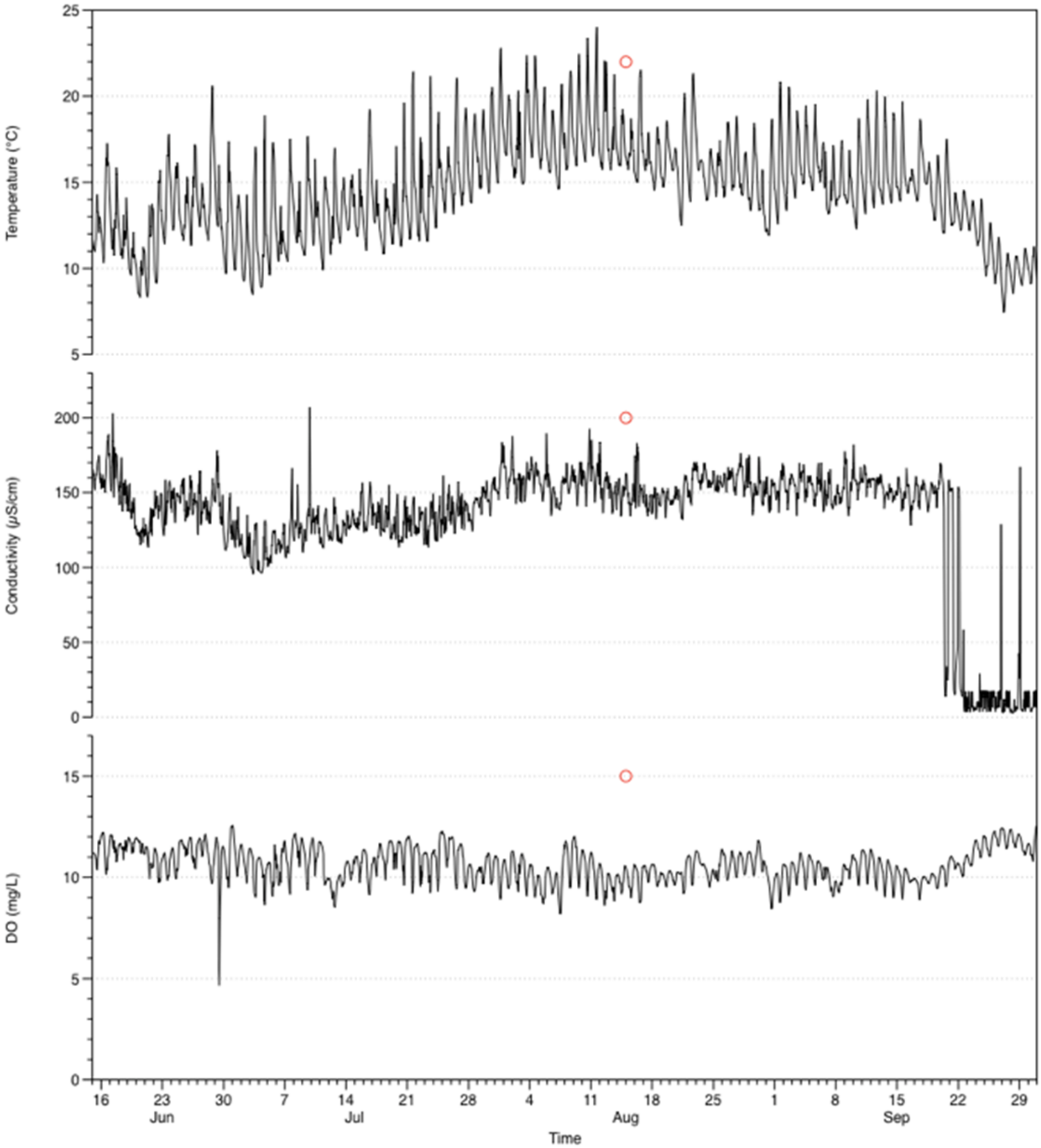

Figure 10-24. Temperature, conductivity, and dissolved oxygen for Bush River reference pond, 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL.

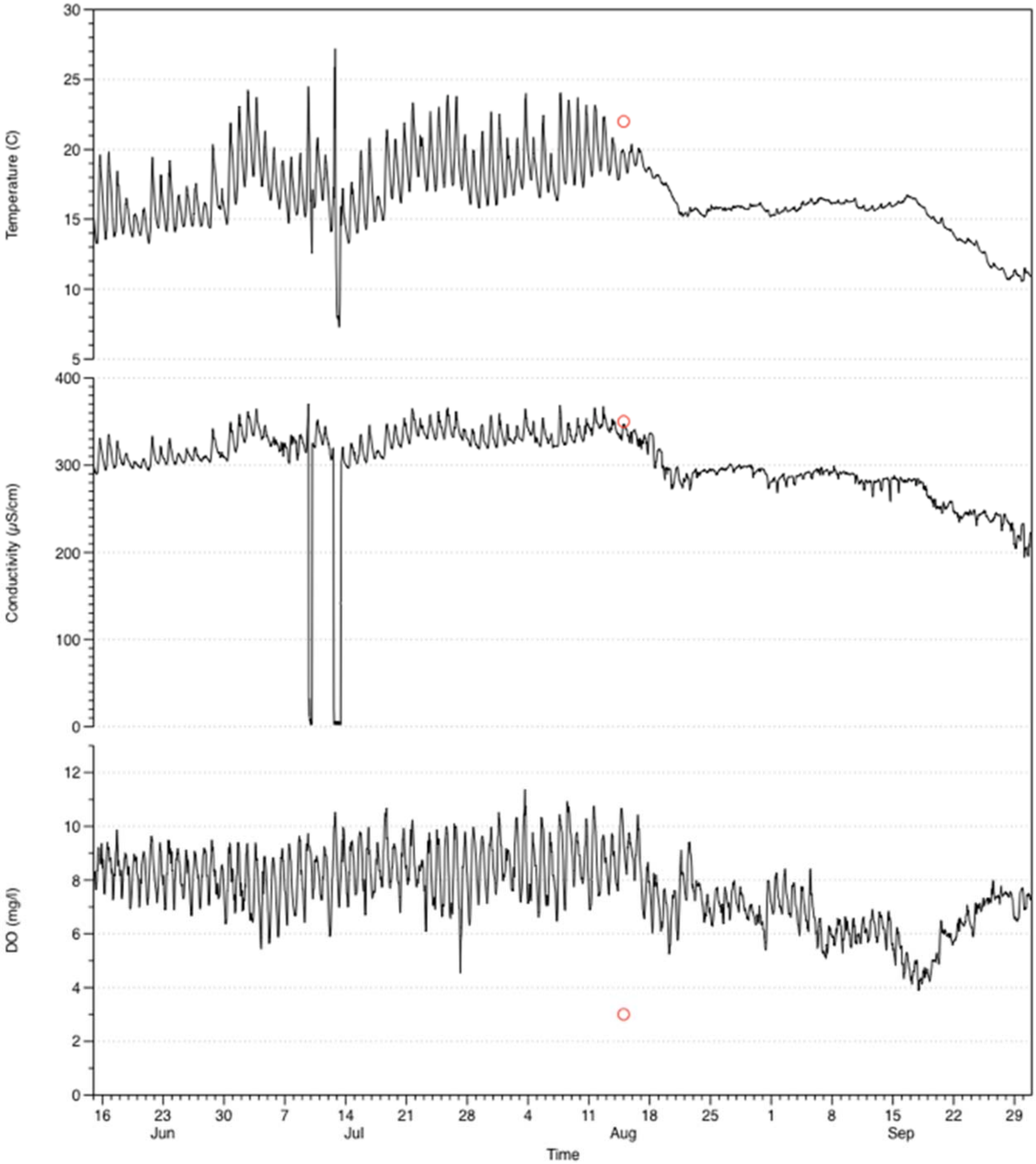

Figure 10-25. Temperature, conductivity, and dissolved oxygen for the Km 88 DDZ pond (Bush Arm), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL

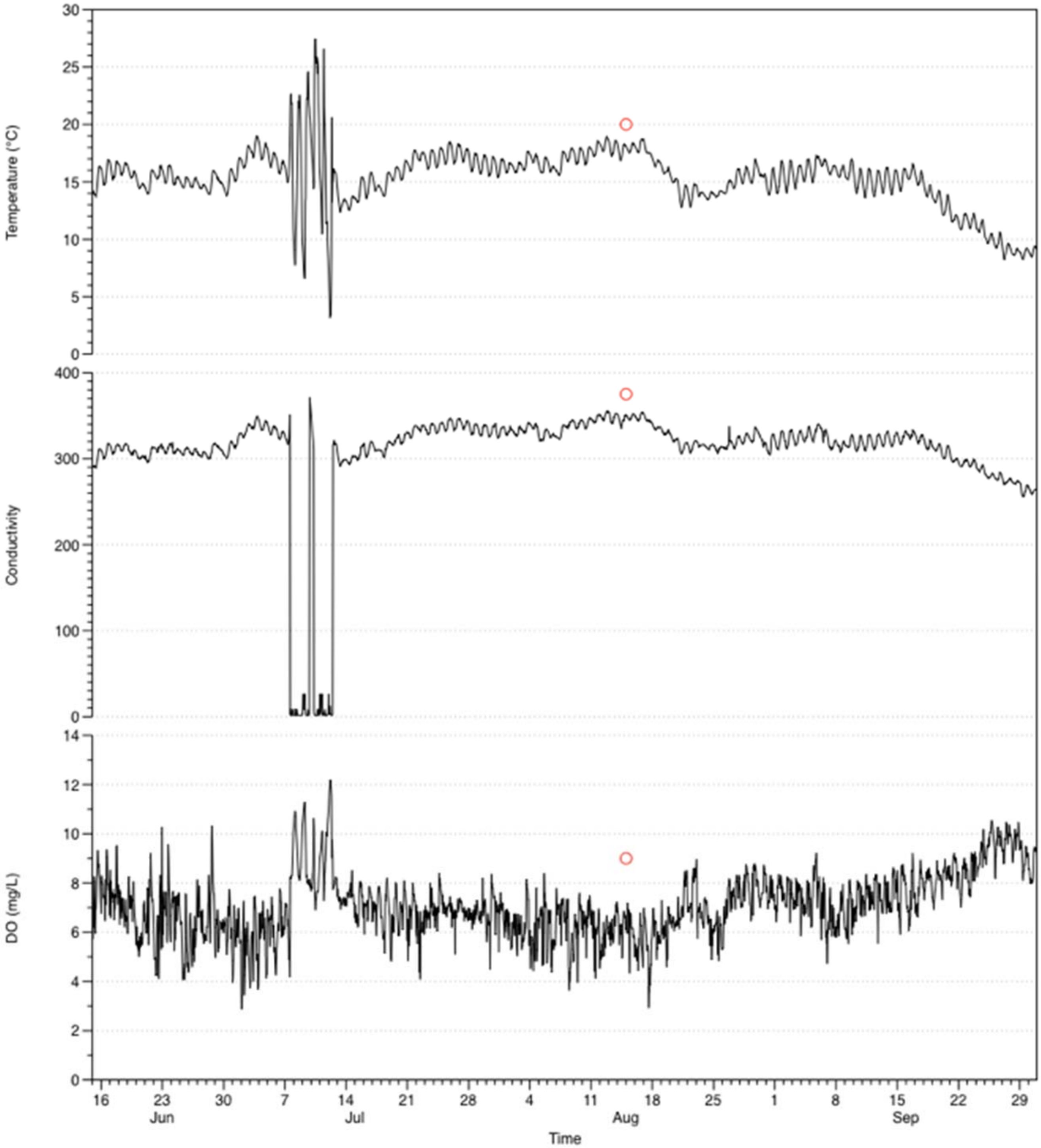

Figure 10-26. Temperature, conductivity, and dissolved oxygen for the reference pond at Km 88 (Bush Arm), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL

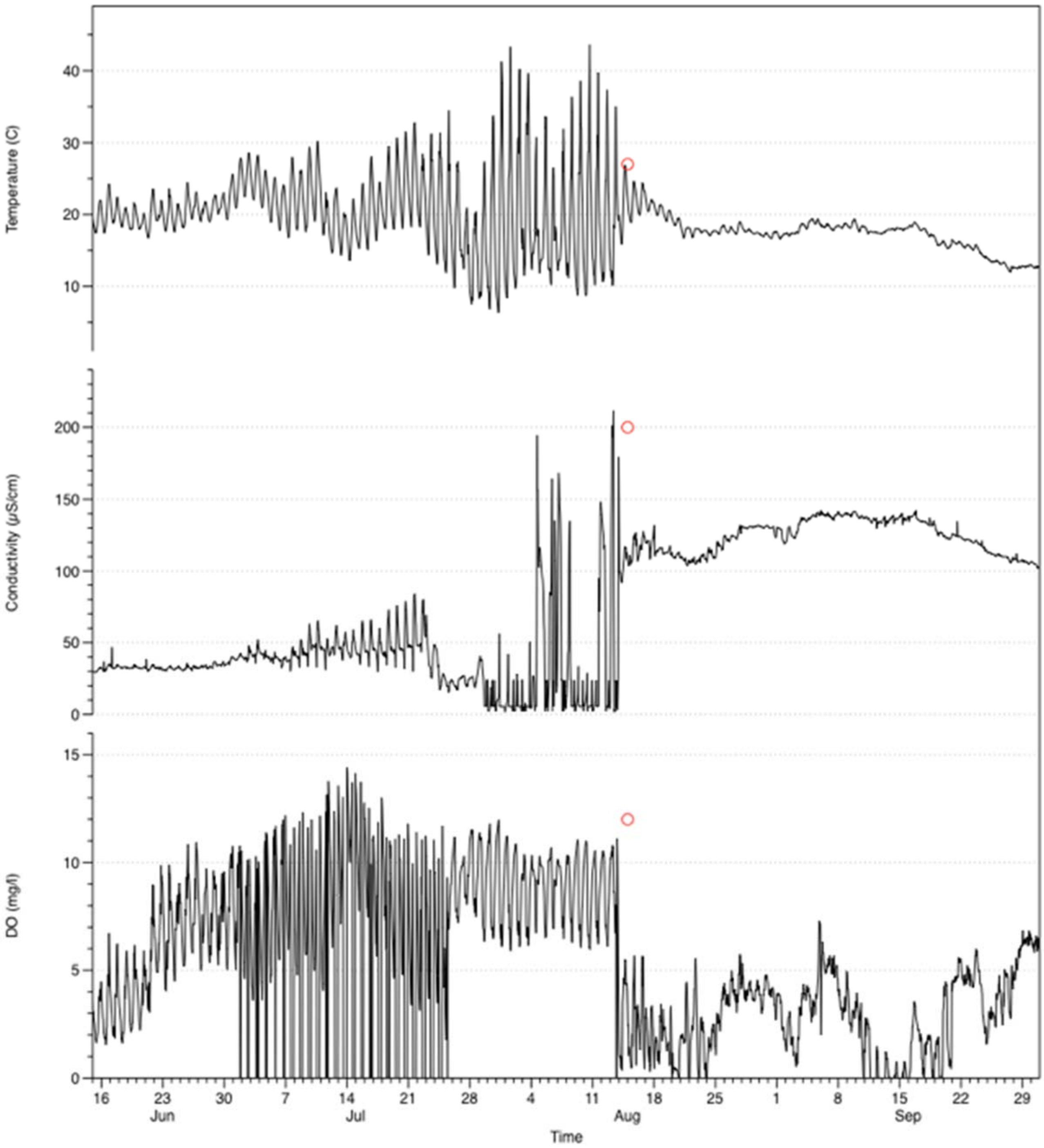

Figure 10-27. Temperature, conductivity, and dissolved oxygen for the DDZ pond at Sprague Bay (Mica Arm), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL. The large swings in conductivity between July 28 and Aug 15, 2014 indicate the sensor was exposed to air

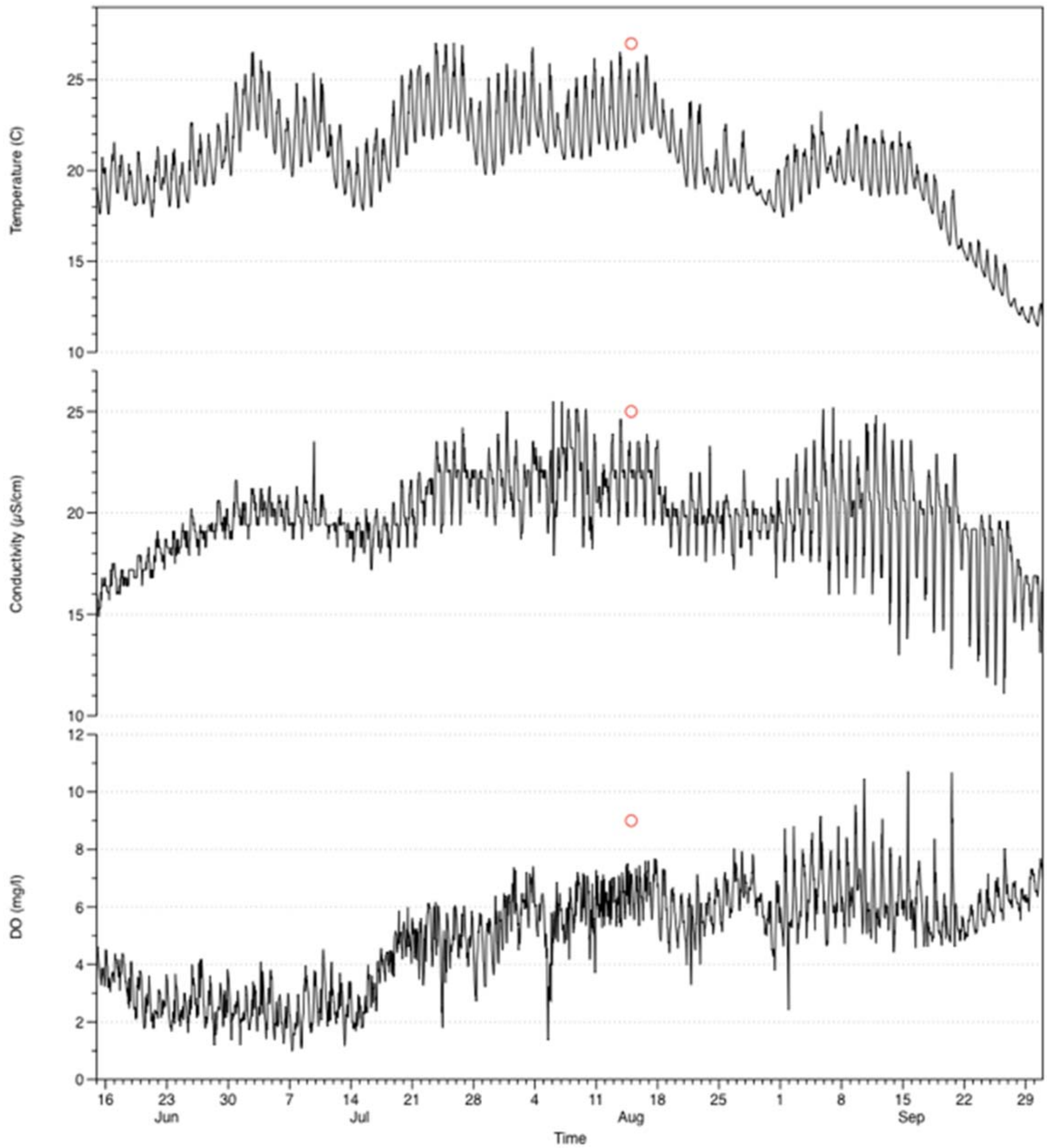

Figure 10-28. Temperature, conductivity, and dissolved oxygen plots for the reference pond at Sprague Bay (Mica Arm), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL

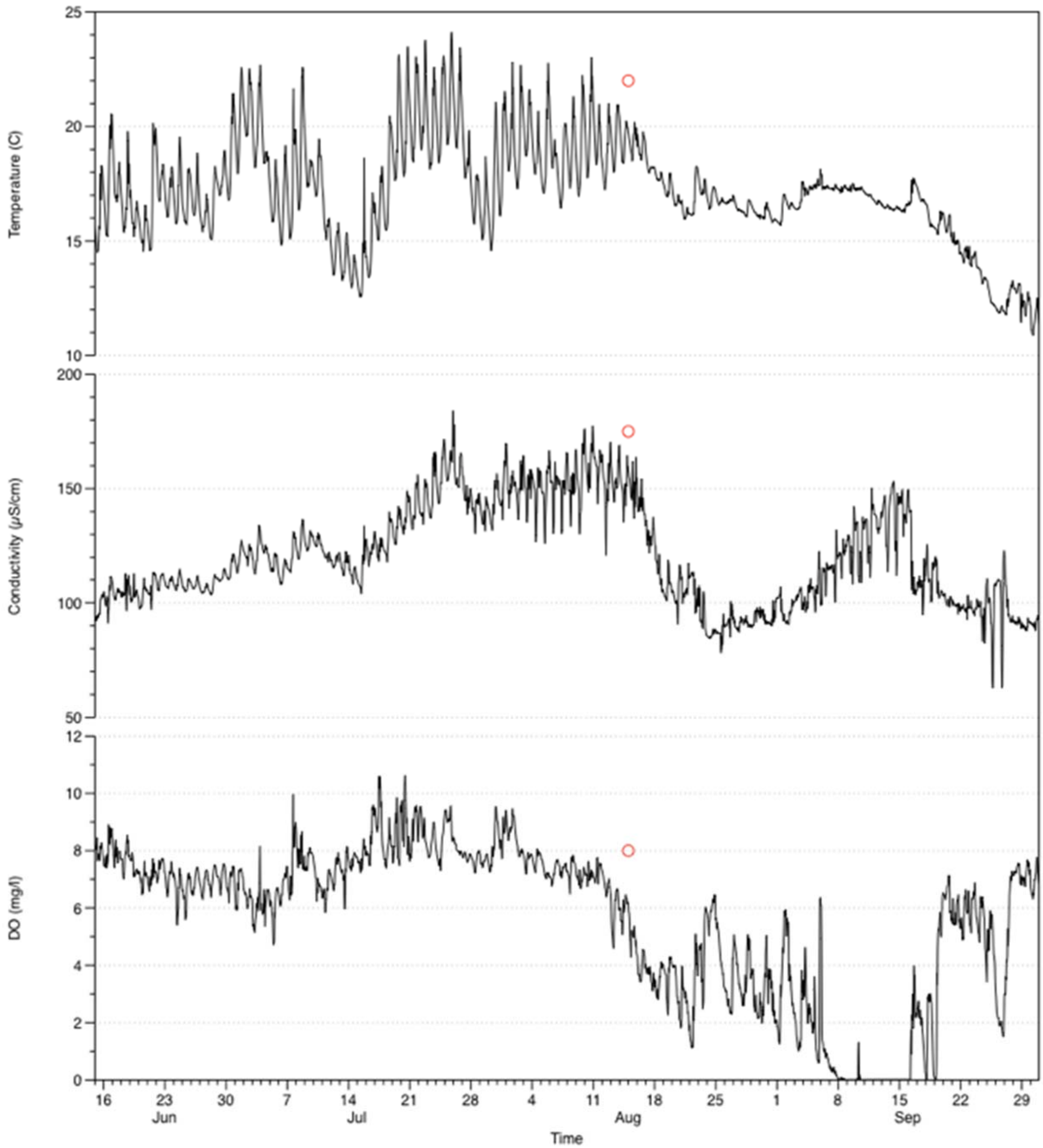

Figure 10-29: Temperature, conductivity, and dissolved oxygen for the DDZ Pond at the Valemount Peatland (Canoe Reach), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL

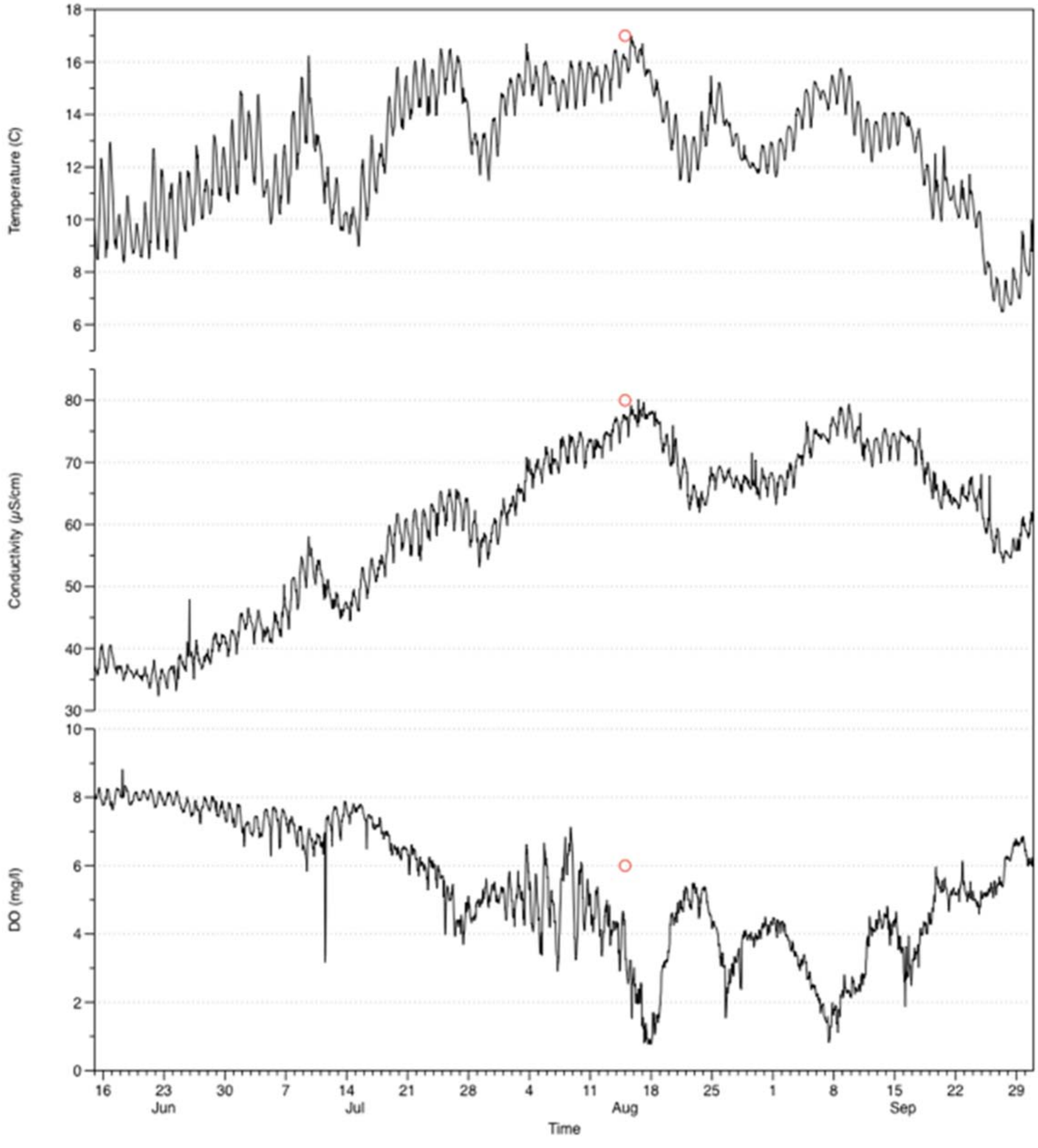

Figure 10-30. Temperature, conductivity, and dissolved oxygen for the reference pond at the Valemount Peatland (Canoe Reach), 2013. Circles denote the approximate time at which the reservoir reached 753 m ASL

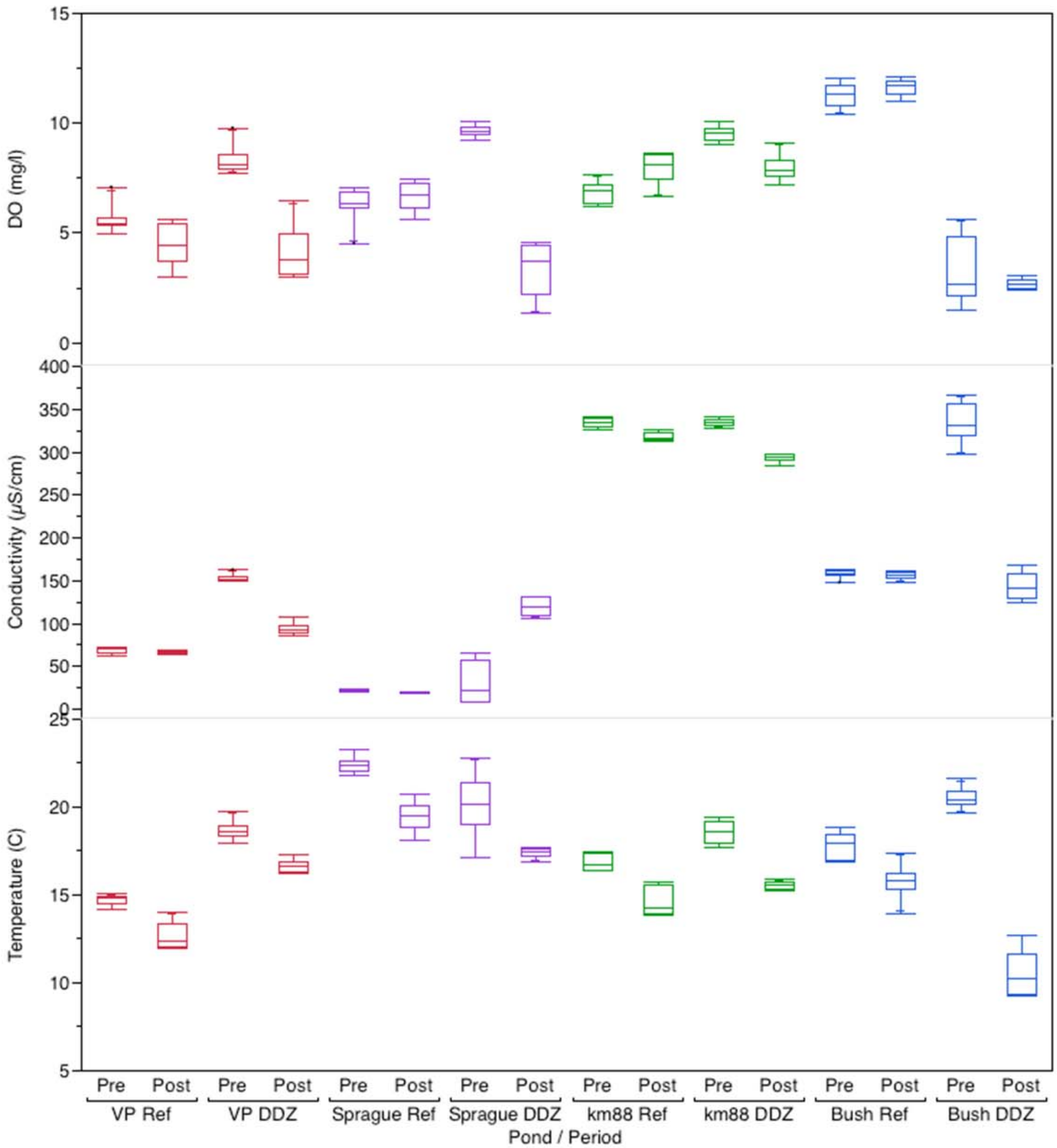

Figure 10-31. Box plots of temperature, conductivity, and dissolved oxygen before (pre) and after (post) inundation of the 753 m ASL elevation band.

10.8 Supplementary results for the analyses of aquatic vegetation data

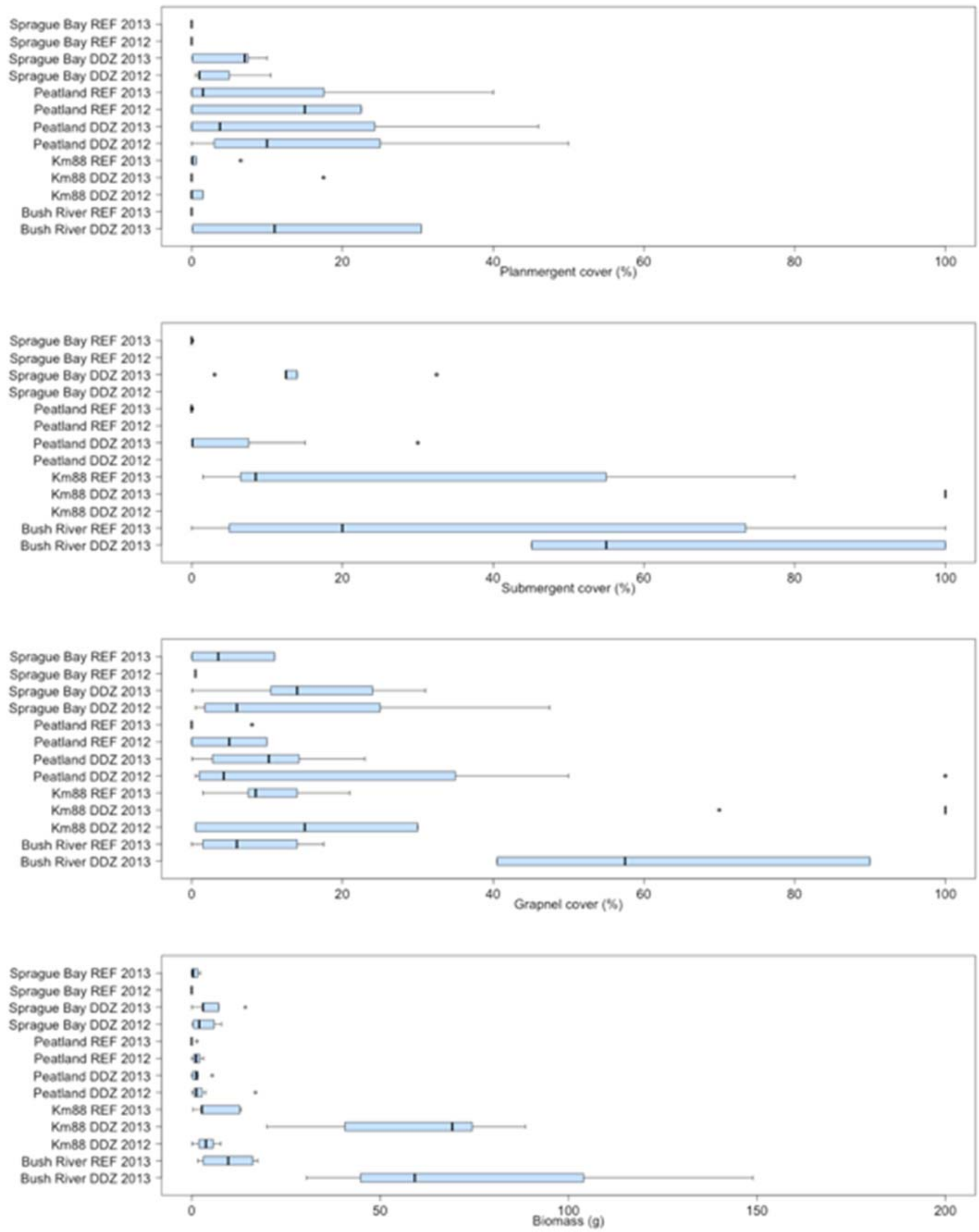

Figure 10-32: Box plots of the frequency of macrophyte cover and biomass in ponds sampled in 2013. Data for ponds sampled in 2012 also included

Table 10-10. Ratio of macrophyte biomass, grapnel submergent and planmergent cover between paired DDZ and Reference ponds (DDZ/REF).

Index Site	Planmergent Cover	Submergent Cover	Grapnel Cover	Macrophyte Biomass
Bush River	-	1.7	8.03	8.25
Km 88	2.44	3.3	8.95	9.31
Peatland	1.08	-	6.02	5.09
Sprague Bay	-	-	3.28	6.86
Mean	1.76	2.49	7.67	7.55
Std. Dev	0.96	1.15	2.5	1.82

Table 10-11. Correlation coefficient and R^2 values (in brackets) for 2013 macrophyte biomass and grapnel, planmergent, and submergent cover data.

	Rake	Submergent cover	Planmergent cover
Biomass	**0.81** **(0.66)**	**0.71** **(0.51)**	-0.02 (0.04)
Grapnel cover	-	**0.84** **(.68)**	0.04 (0.00)
Submergent cover	-	-	-0.07 (0.00)

Table 10-12. Mean per cent cover of planmergent vegetation species in 2013 aquatic wetland plots.

	Bush R. DDZ	Bush R. REF	Km 88 DDZ	Km 88 REF	VP DDZ	VP REF	Sprague DDZ	Sprague REF
CAREAQU	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
CAREUTR	0.0	0.0	0.0	1.3	0.0	0.0	0.0	0.0
COMAPAL	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ELEOPAL	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
EQUIFLU	3.9	0.0	0.0	0.0	0.0	0.0	1.5	0.0
MENYTRI	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
NUPHPOL	0.0	0.0	0.0	0.0	12.5	11.5	0.6	0.0
POTANAT	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0
POTAPUS	10.0	0.0	0.0	0.0	0.0	0.0	2.8	0.0
POTARIC	0.0	0.0	3.5	0.0	0.0	0.0	0.0	0.0
SCHOTAB	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SPARANG	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Average % Cover / plot	**13.9**	**0**	**3.5**	**1.4**	**12.8**	**11.8**	**4.9**	**0**

Table 10-13. Mean percent cover of submerged vegetation species in each plot during 2013 underwater visual surveys

	Bush R. DDZ	Bush R. REF	Km 88 DDZ	Km 88 REF	VP DDZ	VP REF	Sprague DDZ	Sprague REF
CAREAQU	0.0	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0
CAREUTR	0.0	0.0	0.0	1.7	0.0	0.0	0.0	0.0
CHARA	54.6	35.9	24.0	19.1	0.0	0.0	0.0	0.0
COMAPAL	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0
ELEOCH sp.	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0
ELEOPAL	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
EQUIFLU	0.4	0.0	0.0	< 0.1	0.0	0.0	< 0.1	0.0
MENYTRI	0.0	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0
MYRIOsp	0.0	0.0	8.5	0.0	< 0.1	0.0	0.0	0.0
MYRIVER	0.0	0.0	13.5	0.0	0.0	< 0.1	0.0	0.0
NUPHPOL	0.0	0.0	0.0	0.0	0.6	0.0	0.4	0.0
POTAFRI	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0
POTANAT	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0
POTAPUS	11.7	0.0	28.5	0.0	0.0	0.0	13.1	< 0.1
POTARIC	0.0	0.2	1.5	0.0	< 0.1	0.0	0.0	0.0
POTAZOS	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0
RANUAQU	0.0	0.0	24.0	9.2	0.0	0.0	0.0	0.0
SCHOTAB	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0	0.0
SPARANG	0.0	0.0	0.0	< 0.1	0.0	0.0	1.4	0.0
STUCFIL	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0
STUCPEC	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
Unknown	0.0	0.0	0.0	0.0	< 0.1	0.0	0.0	0.0
UTRIMAC	0.0	0.0	< 0.1	0.0	< 0.1	0.0	< 0.1	0.0
Average % Cover / plot	**66.7**	**39.7**	**100**	**30.3**	**5.7**	**< 0.1**	**14.9**	**< 0.1**

Table 10-14. Abundance (per cent cover) of planmergent species in 2012 and 2013 visual surveys by Pond (n = 5). VP = Valemount Peatland; DDZ = drawdown zone; REF = reference

Species Code	Km 88 DDZ Pond		VP DDZ Pond 12		VP REF Beaver Pond		Sprague Bay DDZ Pond		Sprague Bay REF Pond	
	2012	2013	2012	2013	2012	2013	2012	2013	2012	2013
n plots:	**3**	**5**	**8**	**8**	**3**	**5**	**5**	**5**	**5**	**6**
CAREAQU	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
CARELEN	0.0	<0.1	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
EQUIFLU	0.0	0.0	0.0	0.0	0.0	0.0	0.3	1.5	0.0	0.0
MENYTRI	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
NUPHPOL	0.0	0.0	9.5	12.5	12.5	11.5	2.1	0.6	0.0	0.0
PERSAMP	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
POTANAT	0.0	0.0	1.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0
POTAPUS	0.0	0.0	1.0	0.0	0.0	0.0	0.6	2.8	0.0	0.0
POTARIC	0.0	3.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SPARANG	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total % Cover / plot	**0.4167**	**3.51**	**13.919**	**12.763**	**12.5**	**11.81**	**2.938**	**4.94**	**0**	**0**

Table 10-15. Wetland vegetation observed in aquatic wetland plots, 2013.

Code	*Scientific Name*	Origin	Habitat	Form	Common Name
CAREAQU	*Carex atherodes*	Native	Wetland	Emergent	sedge
CAREUTR	*Carex lasiocarpa*	Native	Wetland	Emergent	sedge
CARELEN	*Carex lenticularis ssp.lipocarpa*	Exotic	Wetland	Emergent	sedge
Chara sp.	*Chara spp.*	Native	Aquatic	Submerged	muckgrass
CICUMAC	*Cicuta maculata*	Native	Terrestrial	Forb	spotted water hemlock
COMAPAL	*Comarum palustre*	Native	Wetland	Emergent	marsh cinquefoil
ELIOCH	*Eliocharis S.*		Wetland		
EQUIFLU	*Equisetum fluviatile*	Native	Wetland	Emergent	swamp horsetail
Green Algae	*Algae sp.*		Aquatic	Submerged	
HIPPVUL	*Hippuris vulgaris*	Native	Wetland	Emergent	mares' tail
MENYTRI	*Menyanthes trifoliata*	Native	Wetland	Emergent	bogbean
Moss	*Moss sp.*		Terrestrial	Moss	
MYRI_SP	*Myriophyllum sp.*		Aquatic	Submerged	milfoil
MYRISPI	*Myriophyllum spicatum*	Exotic	Aquatic	Submerged	eurasian water-milfoil
MYRIVER	*Myriophyllum verticillatum*	Native	Aquatic	Submerged	bracted water-milfoil
NUPHPOL	*Nuphar polysepala*	Native	Aquatic	Floating - Rooted	Rocky Mountain Pond-lily
PERSAMP	*Persicaria amphibia*	Native	Aquatic	Floating - Rooted	water smartweed
POTA_SP	*Potamogeton sp*	Native	Aquatic	Submerged	pondweed
POTAGRA	*Potamogeton gramineus*	Native	Aquatic	Submerged	grass-leaved pondweed
POTANAT	*Potamogeton natans*	Native	Aquatic	Submerged	floating-leaved pondweed
POTAPRA	*Potamogeton praelongus*	Native	Aquatic	Submerged	long-stalked pondweed
POTAPUS	*Potamogeton pusillus*	Native	Aquatic	Submerged	small pondweed
POTARIC	*Potamogeton richardsonii*	Native	Aquatic	Submerged	richardson's pondweed
POTAZOS	*Potamogeton zosteriformis*	Native	Aquatic	Submerged	eel-grass pondweed
RANUAQU	*Ranunculus aquatilis*	Native	Aquatic	Submerged	water crowfoot
Rumex	*Rumex occidentalis*	Native	Wetland	Emergent	dock
Salix Spp	*Salix sp.*		Terrestrial	Shrub	
SCHOTAB	*Schoenoplectus tabernaemontani*	Native	Wetland	Emergent	soft-stemmed bulrush
SPARG_SP	*Sparganium sp*		Aquatic	Submerged	
SPARGANG	*Sparganium angustifolium*	Native	Aquatic	Submerged	narrow-leaved bur-reed
Stuckenia	*Stuckenia sp.*	Native	Aquatic	Submerged	pondweed
UTRIINT	*Utricularia intermedia*	Native	Aquatic	Submerged	flat-leaved bladderwort
UTRIMAC	*Utricularia macrorhiza*	Native	Aquatic	Submerged	greater bladderwort
UTRIMIN	*Utricularia minor*	Native	Aquatic	Submerged	lesser bladderwort

10.9 Supplementary results for the analyses of pelagic invertebrate data

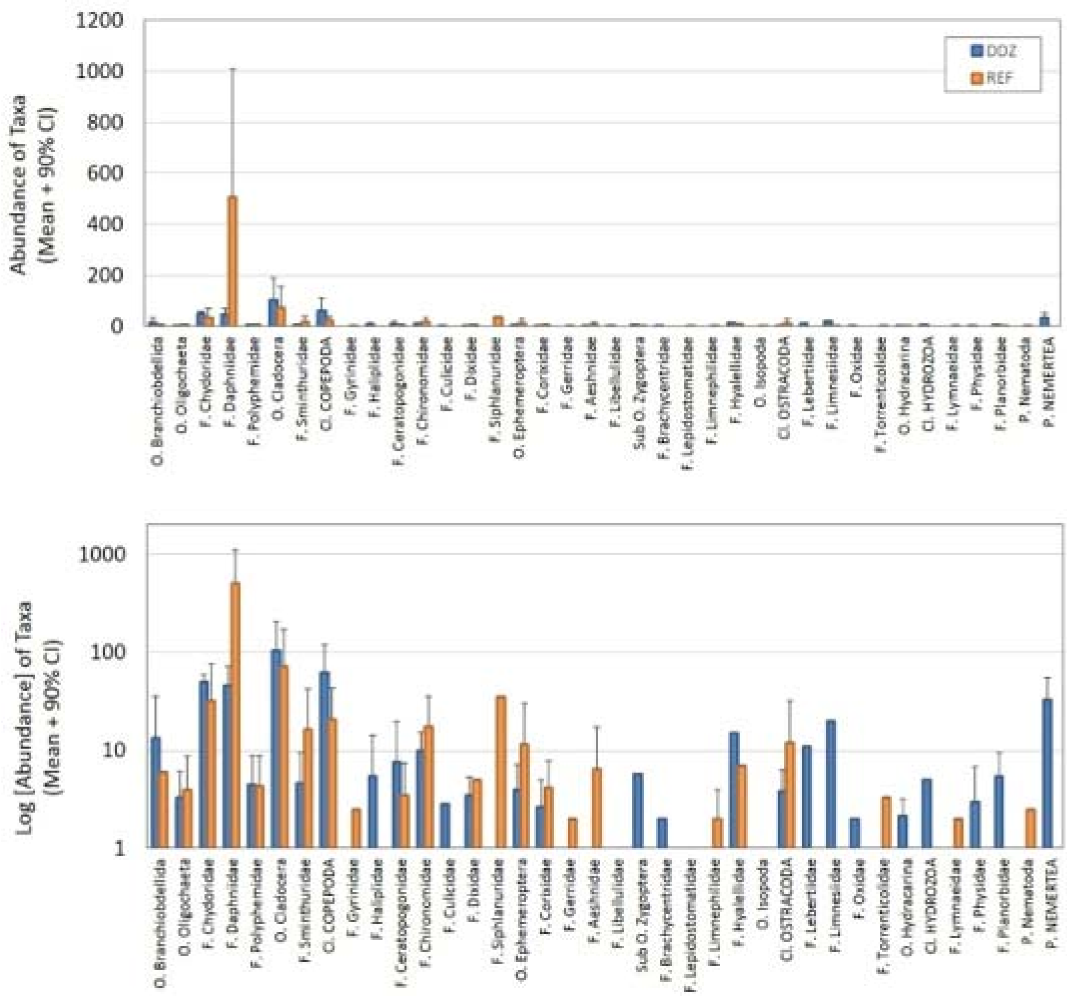

Figure 10-33. Average abundance (A) and log-transformed abundance (B) of lower-level taxon identified (38 taxa) per sample with 90% confidence intervals. P = Phylum, Cl = Class, O = Order, F = Family

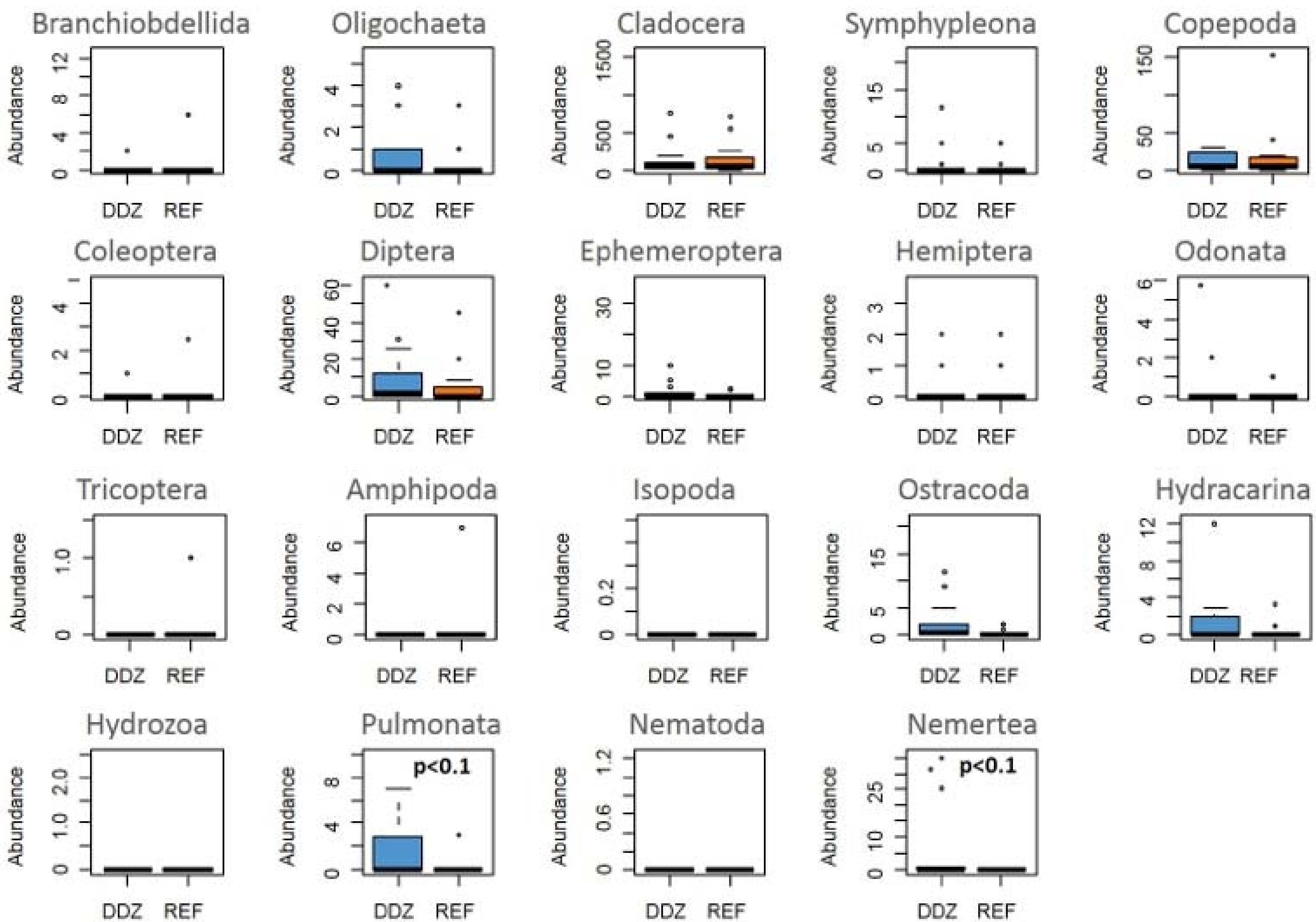

Figure 10-34. Boxplots for 19 higher-level invertebrate taxa by pond position (DDZ = reservoir pond; REF = reference pond). P-values are given for taxa with significant differences in mean rank abundance (according to Kruskal-Wallis non-parametric tests; α = 0.10)

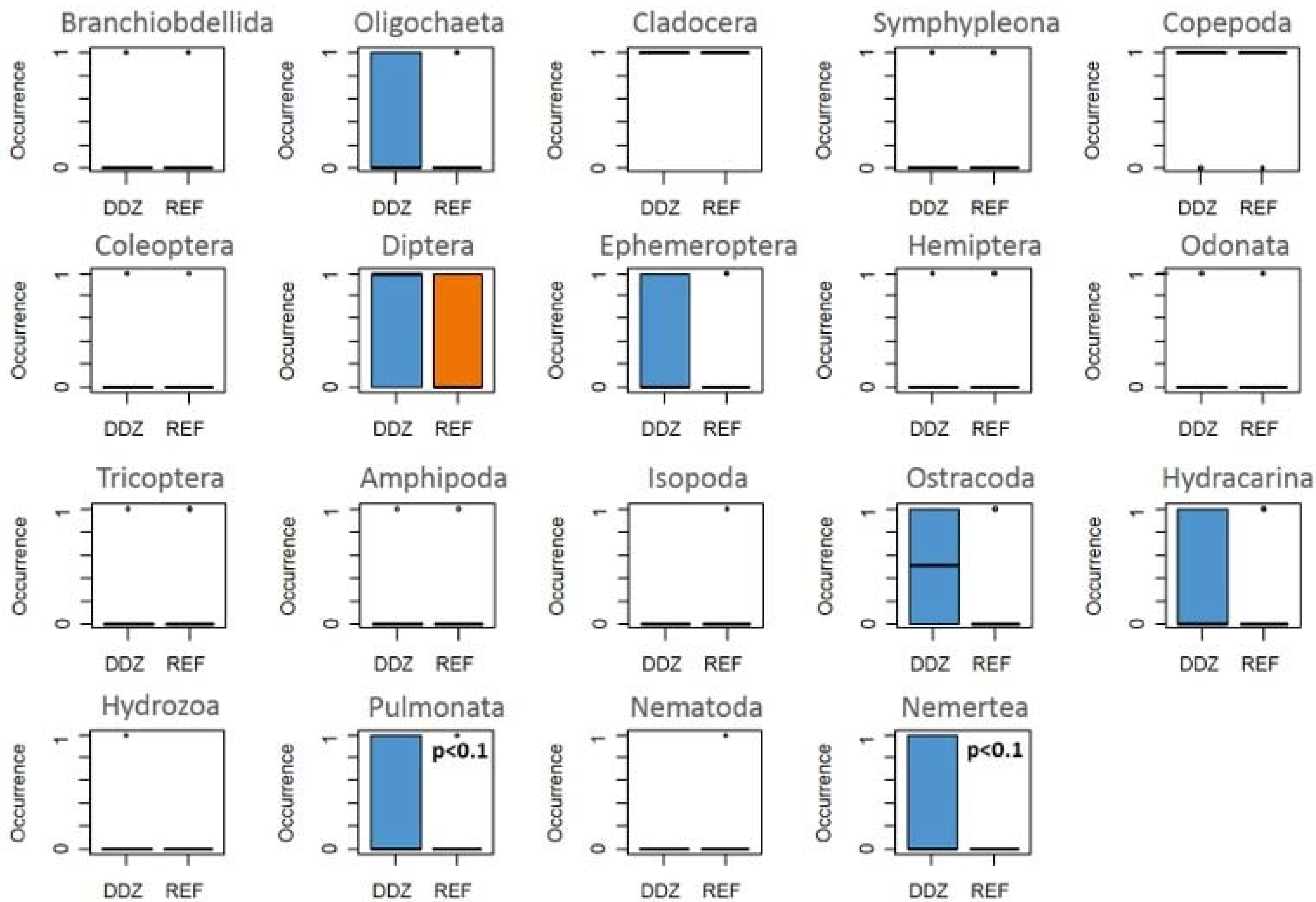

Figure 10-35. Boxplots of occurrence (presence/non-detection) for 19 higher-level invertebrate taxa by pond position (DDZ = reservoir pond; REF = reference). P-values are given for taxa with significant differences in mean rank abundance (according to Fisher's Exact Tests; α = 0.10)

Table 10-16. Average abundance (with 90 per cent confidence intervals) of taxa in Drawdown zone (DDZ) and Reference (REF) ponds across both monitoring years (2012-2013). Taxa were grouped at higher-taxonomic levels (Class and higher) in order to compare between years

	DDZ 2012		REF 2012		DDZ 2013		REF 2013	
TAXON	Mean	*90% CI*	Mean	*90% CI*	Mean	*90% CI*	Mean	*90% CI*
Arachnida	2.5	1.2	4.0	2.3	4.8	3.2	1.6	1.0
Bivalvia			1.0	0.0				
Branchiopoda	157.4	195.1	48.5	32.6	65.9	43.8	166.1	144.7
Clitellata	8.6	6.2	2.5	1.6	5.9	4.8	4.7	2.6
Collembola	2.9	2.4	1.3	0.3	4.6	4.0	16.3	22.0
Copepoda	19.6	19.5	37.5	20.6	61.8	47.8	20.6	18.7
Gastropoda	1.9	0.8	4.8	2.2	4.9	2.7	1.5	0.8
Hydrozoa	3.3	1.5	2.0	0.0	5.0	0.0		
Insecta	4.2	1.2	9.2	4.5	6.3	2.1	10.5	6.0
Malacostraca	2.3	2.2	9.8	12.0	15.0	0.0	4.0	4.9
Nematoda							2.5	0.0
Nemertea					32.9	18.8		
Ostracoda	4.2	1.9	36.0	35.8	3.8	2.1	11.9	16.8

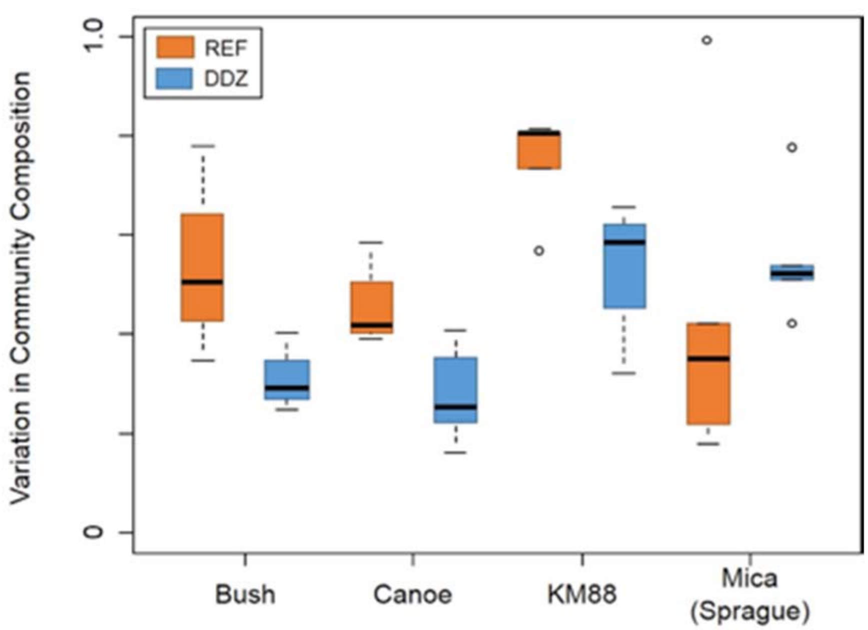

Figure 10-36. Variance in invertebrate community composition (Hellinger distance) for each site by reach (2013 monitoring data). Higher values indicate less similarity in taxa composition among replicate samples collected at each site

Table 10-17. Taxon relationships with each principal-components axis of the ordination

TAXON	PCA 1	PCA 2	R^2	P-VALUE
Cladocera	**-0.85534**	**-0.51807**	**0.7844**	**0.001**
Isopoda	**0.69985**	**-0.71429**	**0.7438**	**0.001**
Diptera	**0.96649**	**0.25669**	**0.7048**	**0.001**
Odonata	**0.74806**	**-0.66363**	**0.7021**	**0.001**
Amphipoda	**0.7925**	**-0.60988**	**0.6226**	**0.005**
Coleoptera	**0.50001**	**0.86602**	**0.519**	**0.004**
Symphypleona	**0.90802**	**-0.41892**	**0.4792**	**0.012**
Branchiopoda	**0.89344**	**-0.44919**	**0.4437**	**0.016**
NEMERTEA	**0.49387**	**0.86953**	**0.4065**	**0.018**
Tricoptera	**0.84467**	**-0.53529**	**0.3655**	**0.019**
Oligochaeta	**0.99461**	**-0.10368**	**0.2924**	**0.004**
Ephemeroptera	**0.77846**	**0.62769**	**0.2681**	**0.025**
COPEPODA	**0.83317**	**0.55302**	**0.2498**	**0.004**
OSTRACODA	**0.89092**	**0.45416**	**0.1802**	**0.04**
Hemiptera	**0.8345**	**0.55101**	**0.2306**	**0.047**
Pulmonata	**0.80837**	**0.58867**	**0.1943**	**0.056**
Nematoda	**0.41448**	**0.91006**	**0.2059**	**0.072**
Hydracarina	0.92255	0.38587	0.1162	0.137
HYDROZOA	0.89913	0.43767	0.0477	0.146

Table 10-18. Environmental relationships with each principal-components axis

VARIABLE	PCA 1	PCA 2	R2	P-VALUE
Depth	**-0.84404**	**-0.53628**	**0.2644**	**0.016**
VEG grapnel cover	**0.5453**	**0.83824**	**0.2721**	**0.019**
Elevation	**0.39992**	**-0.91655**	**0.2328**	**0.02**
VEG biomass	**0.56229**	**0.82694**	**0.209**	**0.033**
DO	**0.70418**	**0.71002**	**0.2012**	**0.026**
Temp	**-0.94783**	**-0.31876**	**0.1892**	**0.025**
Cond	**0.99914**	**-0.04138**	**0.1553**	**0.056**
VEG submergent cover	**0.43868**	**0.89865**	**0.1615**	**0.059**
pH	0.91146	0.41139	0.1249	0.109
Pond Area	-0.92878	-0.37064	0.1005	0.166
VEG Planmergent cover	0.33171	0.94338	0.1071	0.171
Reservoir Position	0.11035	-0.99389	0.0603	0.394
Pond Depth	-0.7121	-0.70208	0.0207	0.741
Wood presence	-0.35118	0.93631	0.0241	0.747