Columbia River Project Water Use Plan

Kinbasket Reservoir Fish and Wildlife Information Plan

Mica Unit 5 Wetlands Monitoring Program

Implementation Year 4

Reference: CLBMON-61

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KINBASKET AND ARROW LAKES RESERVOIRS
Monitoring Program No. CLBMON-61
Kinbasket Reservoir Wetlands Monitoring Program




Year 4–2015

Final Report

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EXECUTIVE SUMMARY

This report summarizes the third and fourth year (2014 and 2015) of CLBMON61, which is a seven-year study to assess the operational impacts of Mica Units 5 and 6 on wetlands in Kinbasket Reservoir. The study follows a Before-After-Impact-Control (BACI) design to address the following management questions:

1. What are the short-term effects of water level changes (associated with Mica Unit 5 and 6) on wetland vegetation composition or productivity, with emphasis on the 753 to 754m 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?

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 (control) aquatic wetlands, as well as terrestrial wetlands stratified across the following elevation bands: 752 to 753 m, 753 to 754 m, 754 to 755 m, and reference wetlands outside the reservoir above 755 m. Terrestrial wetlands were sampled using belt transects and circular plots. Vegetation and ground cover (substrate) were compared across the four elevation bands. Aquatic wetlands were stratified to include ponds within the reservoir at approximately 753 m ASL and reference ponds located above the reservoir (between 756 and 780 m ASL). Macrophyte biomass, water physicochemistry, and aquatic metabolism were compared across these strata. The before-impact period for the BACI design commenced in 2012 and extended to 2014. The after-impact period began in 2015 and will include sampling in 2016 and 2017. In 2015, the reservoir levels peaked at 750.8 m and, consequently, the predicted impact (inundation) to the 753-754 m elevation band did not occur.

Using vegetation data from 2014 and 2015, eighteen wetlands communities were redescribed using the wetland classification of MacKenzie and Moran (2004) and community classification of Hawkes et al (2007). LIDAR data obtained in 2014 indicated significant errors in the original DEM resulting in the mischaracterization of communities in 2013. In revisiting the classification of terrestrial wetland communities, five successional pathways were identified for terrestrial wetlands at the four index sites. Wetlands generally progressed from marshes to fens and then to either bogs, swamps, or riparian flood communities; however, the specific communities types and the nuances of each pathway were influenced by site characteristics such as hydrology, geology, topography, soil type, organic accumulations, nutrient availability, climate, and disturbance regime. These communities will be reassessed in the final study year.

The analyses of transect and circular plot data indicated that total herb cover and total shrub cover did not differ from 2013 to 2015. We suggest that after successive years of high reservoir levels terrestrial wetlands communities below 754 m may be at a state of equilibrium dominated primarily by hydrophilic flood tolerant species. Consequently, we do not expect to detect significant changes in vegetation abundance except after years of lower reservoir levels when flood intolerant species re-establish.

Biomass measurements from macrophytes standing crop were obtained as an index of primary productivity in aquatic wetlands (ponds). Linear Mixed Effects modelling indicated

that macrophyte biomass was influenced by location (index site), reservoir position (above or within the reservoir), and year. Macrophyte biomass was over 800 per cent greater in ponds from Bush Arm (Km88 and Bush River) than in Sprague Bay or the Valemount Peatland and was 300 per cent greater in ponds within the reservoir (DDZ) than in reference ponds (REF). Surficial geology and water chemistry (i.e., dissolved minerals and alkalinity) account for differences in macrophyte biomass across index sites and water temperature account for differences in macrophyte biomass between DDZ and REF pond, although sedimentation and light availability may also play a role. Macrophyte biomass increased at similar rates in both DDZ and REF ponds from 2012 to 2015. During this period, daily mean temperatures early in the growing season (May and June) increased from 0.67 to 1.05 ℃ annually from 2012 to 2015, which accounts for these increases.

Diel changes in dissolved oxygen were used to estimate net ecosystem production (NEP), ecosystem respiration (R), and gross primary production (GPP) in aquatic wetlands (ponds) . Linear Mixed Effects modelling indicated that metabolic activity was influenced by index site, pond position, and year. Variability in metabolic activity across index sites was likely function of wetland type and local water chemistry. The short-term effects of inundation on aquatic metabolism were examined by comparing pre- and post-inundation periods in 2014. Our results indicate that increases in flooding associated with Mica 5 and 6 may result in a short-term decrease in wetland primary productivity (GPP and NEP); however, the response may depend on the water physicochemistry and trophic characteristics of the ponds.

For redundancy, we recommend deploying two DO loggers in each pond. Currently, only a single DO logger is deployed in each pond and the loss of data from a single DO unit is significant. Installing a second DO logger in each pond will also reduce noise in the data (negative GPP and positive R values).

Management Question	**Hypotheses**	**Status**
1) What are the short-terms effects of water level changes (due to Mica Units 5 and 6)* on wetland vegetation composition or productivity, with emphasis on the 753 to 754m elevation band? - Our results indicate that increases in flooding associated with Mica 5 and 6 may result in a short-term decrease in wetland primary productivity (GPP and NEP); however, the response may depend on the water physicochemistry and trophic characteristics of the ponds 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? - Too early to assess.	H01: There are no changes in wetland composition in Kinbasket Reservoir over the course of the monitoring period. H1A: Wetland composition is not affected by reservoir operations. - Too early to assess. H02: There are no changes in wetland productivity in Kinbasket Reservoir over the course of the monitoring period. H2A: Wetland productivity is not affected by reservoir operations. - Our results indicate that increases in flooding associated with Mica 5 and 6 may result in a short-term decrease in wetland primary productivity (GPP and NEP); however, long-term effects have not been assessed.	It is too early to fully assess the impacts of Mica 5 (and 6) on wetland composition and productivity with certainty. Units 5 and 6 became operational on January 28, 2015 and December 22, 2015, respectively and in 2015, reservoir levels did not exceed 751 m and did not influence the elevation 753 m elevation band. Additionally, high unrepresentative reservoir levels during the before impact period may confound our ability to detect changes resulting from operational changes associated with Mica 5 and 6.

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1.0 INTRODUCTION

Wetlands offer 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. 2012; 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. Less than two per cent of the wetland habitat that existed prior to the flooding of the reservoir remain (Utzig and Schmidt 2011; Adama et al. 2013). The remaining wetlands in Kinbasket Reservoir provide rare and unique low elevation habitat for wetland dependent species in a 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 could be 0.6m higher in July and August in every three years out of ten (KCB 2009) with both units operational. The model also predicted that the impacts would be restricted to the elevation band spanning 753 to 754m 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) prescribed a Before-After-Control-Impact (BACI) study to (1) assess potential changes in wetland composition and productivity in Kinbasket Reservoir in the elevation band of 753-754m, and (2) to determine whether any change can be associated with reservoir operations. The BACI design called for the collection of two years of before-impact data and three-years of after-impact data. Years 1 and 2 were carried out in 2012 and 2013; however, high reservoir levels in 2012 precluded the collection of data and an additional year of before impact sampling was carried in 2014 (although no reporting was required). The first year of after-impact data collection commenced in 2015 and will be followed by 2 more years of sampling in 2016 and 2017. This report summarizes the data collected in 2014 and 2015 (Years 3 and 4). Definitions for various terms used in this report are provided in Section 9.1: Appendix A. A summary of work completed in previous years is provided in Section 9.2 Appendix B.

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 754m range (as specified under CLBMON 10 addendum): 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 754m 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 STUDY AREA

3.1 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 3-1). The Mica hydroelectric dam, located 135 km north of Revelstoke, B.C., spans the Columbia River and impounds Kinbasket Reservoir. 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 3-1). 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.

3.2 Reservoir operation

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 2009). Construction on Unit 5 began in 2012 and was in-service in early 2015. Construction on Unit 6 began in 2014 and the target in-service date was late 2015.

The normal operating range of the reservoir is between 707.41m and 754.38m elevation, but can be operated to 754.68m with approval from the Comptroller of Water Rights. Normally Kinbasket Reservoir fills in the spring and is full by the mid- to late-summer (Figure 3-2). Water levels drop during the winter due to demand for electricity and the reservoir typically falls to the lowest levels in the spring (April) prior to the onset of the freshet.

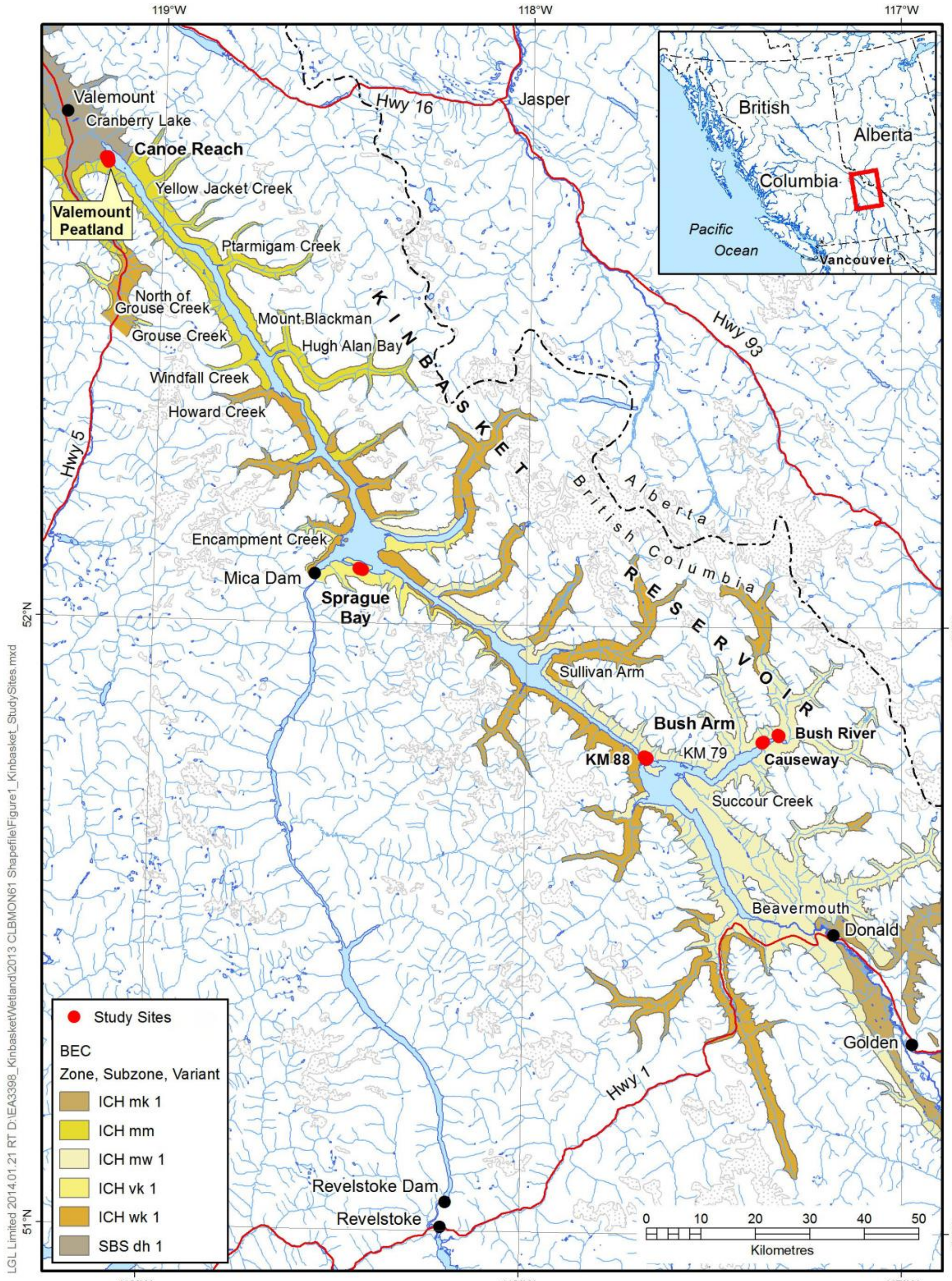

Figure 3-1: Location of the CLBMON-61 index sites in Kinbasket Reservoir

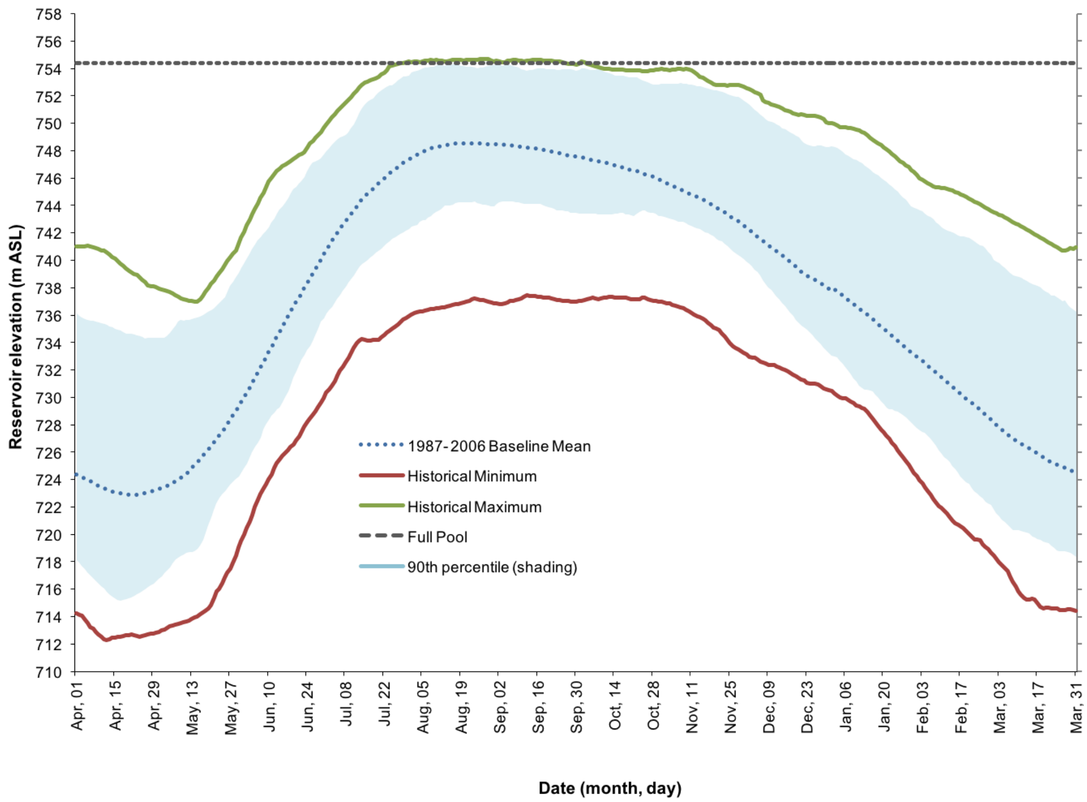

Figure 3-2: Baseline mean elevations for Kinbasket Reservoir (1987–2006) with minimum, maximum, 10 to 90th percentile operating ranges, and licensed maximum elevation (full pool).

3.3 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 3-1). 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.

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) to an elevation 760m (Figure 9-2): approximately 90 per cent of the wetland complex occurs below full pool (754.4 m). The Valemount Peatland is comprised of vegetation communities that reflect both the historical 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).

Located 8km east of Mica Dam, the Sprague Bay wetlands are comprised of a narrow fenland/beaver pond complex extending from 760 to 752m (Figure 9-4). 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.

The Km 88 wetland is a small 25 ha complex of fens and beaver dams located north of Bear Island in Kinbasket Reservoir (Figure 9-6). A series of beaver ponds occur above and step down into the reservoir. The drawdown zone (DDZ) ponds are positioned at 752 and 753m; the reference (REF) pond is positioned approximately 500m upslope at 780 m. Due to seepage, fens and swamps have developed downstream of the ponds. The largest of these extends from 757 to below 750m and is the site of the terrestrial wetland transects (Figure 9-6).

The Bush River wetlands occur downstream of the confluence of the Bush and Valenciennes Rivers (770 m) to the Bush River causeway (752m; Figure 9-8). The DDZ pond and terrestrial wetlands are located adjacent the causeway (753m) and are frequently inundated during spring freshet. Wetlands also occur on the west side of the causeway at 752m and lower and are prone to accumulations of wood debris. The reference wetland occurs 3 km upstream from the Bush River Causeway. 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.

More detailed descriptions of the index sites are provided in Adama et al (2014). Updated maps showing the wetland sampling points are provided in Section 9.3 Appendix C.

4.0 METHODS

4.1 Study Design

Detailed descriptions of the monitoring program are provided in Adama et al. (2013) and Adama and Hawkes (2015). In brief, the potential impacts resulting from Mica Units 5 and 6 will be assessed following three years of before impact sampling (2012 to 2014) and three years of post-impact sampling (2015, 2016, and 2017). The study entails the collection of terrestrial wetland vegetation data, aquatic vegetation data, water physicochemistry data, and weather data. Terrestrial wetlands were stratified by reach, index site, and elevation band.

Aquatic wetlands were stratified by reach, index site, and position (either within the reservoir (DDZ) or above reservoir full pool elevation of 754.4 m (REF); elevations of the reference ponds were between 754.9 and 769.9 m. The study will employ a repeated measures model to compare community composition, productivity, and physiochemical data collected in terrestrial and aquatic wetlands at four index sites over the study period.

After the rapid spring infill that occurred in 2012 (Adama et al 2013), the timing of fieldwork has been scheduled to ensure all field surveys are completed before the reservoir reaches 752 m. Consequently, for some surveys (i.e., terrestrial vegetation and the pond sampling) the effects of inundation on wetlands (if any) are limited to past operations (which may be cumulative) and not the operation of study year. For example, vegetation surveys completed in July 2014 do not account for the prolonged period of inundation that followed from August through to November of that year. Instead the vegetation sampling in 2014 reflects any residual impacts that occurred from 2013 (or earlier if those impacts are cumulative). An exception to this is the water physicochemistry data collected from data loggers, which were retrieved at the end of the season after reservoir levels have dropped below 752 m.

4.2 Terrestrial Wetland Vegetation Sampling

Terrestrial wetland vegetation were sampled using a combination of modified belt transects and circular plots (Figure 4-1). The 20m belt transects were subsampled using ten 2m X 0.5m quadrats to obtain detailed abundance of herbaceous plants and maintain consistence with the sampling methods of CLBMON-10 (Hawkes et al 2010). The larger circular plots (100m^2) were sampled to obtain data on shrub abundance and ground cover.

A minimum of two belt transect/circular plot arrays were established within 1-meter elevation bands between 752 and 755m within Kinbasket Reservoir and in reference wetlands outside the reservoir. An example of this layout is shown in Figure 4-2. As standard procedure, transects are sampled with the surveyors backs to the reservoir to ensure the same side of the transect is sampled every year.

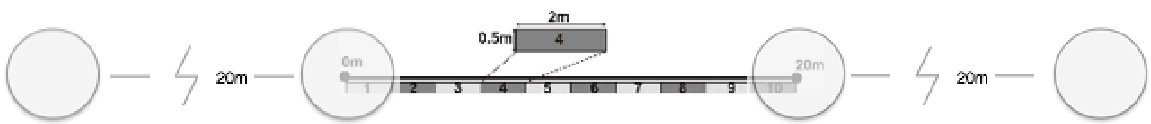

Figure 4-1: Schematic of the belt transect and four 100m^2 circular plots (5.64m radius) used to sample wetland communities in Kinbasket reservoir. Not drawn to scale

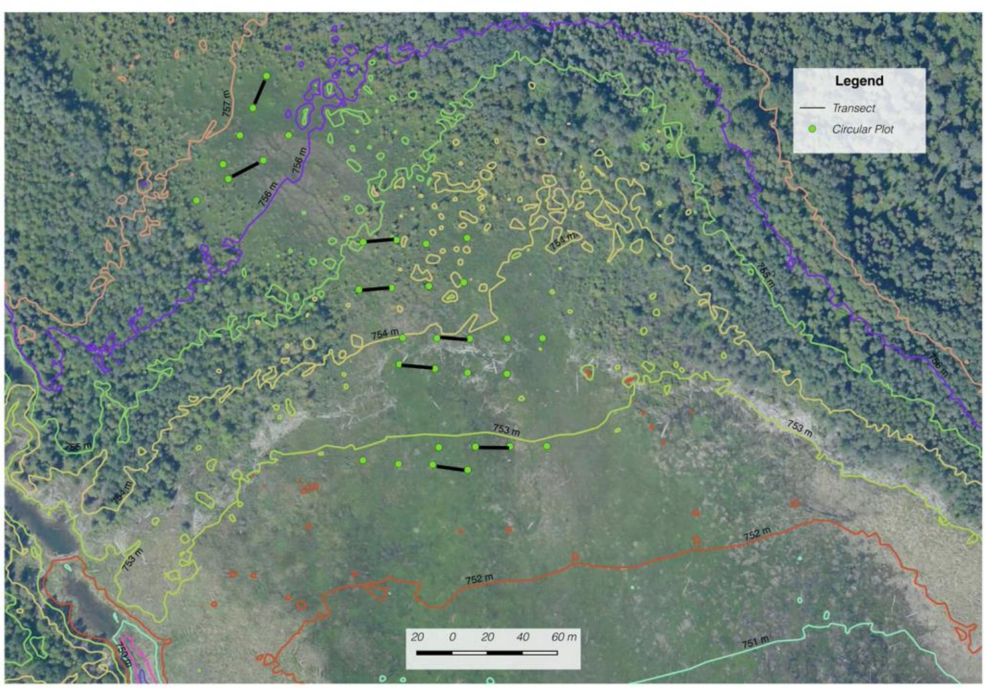

Figure 4-2: Layout of belt transects (black bars) and circular plots (green dots) across the upper four elevation bands 752 to 757 m.

Most transects and plots were established in 2012 and 2013. In 2014, a new Digital Elevation Model (DEM) was provided by LGL Limited (Hawkes et al 2015) from LIDAR data. Comparisons made by Hawkes et al (2015) indicated that the elevation data differed by a mean of 77 cm with some areas differing by up to 180 cm. Upon examining the transects (and plots) established previously, we found that 65 per cent (n=26) of the transects were not located in the correct elevation bands. While most of these transects fell in adjacent elevation bands, some elevations bands were now underrepresented, others were overrepresented, and some were not represented at all. In 2015, 17 transects were reclassified into the correct elevation band and 12 new transects were established to ensure a minimum of 2 transect/circular plot arrays occurred in each elevation band between 752 and 755m (Table 4-1).

Table 4-1: New transects and transect reclassifications required for 2015

Location	Site	New Transects	Reclassified
Valemount Peatland	VP1	3 (1 UC, 2 REF)	6
Valemount Peatland	VP2	4 (2 TGT, 2 REF)	8
Sprague Bay	SB	3	2
Km88	Km88	1 (1 UC)	0
Bush River	Bush River	2 (2 UC)	1
Total		12	17

* LC = Lower Control (752-753m): TGT = Target (753-754): UC = Upper Control (753-754m), REF (> 755).

Rebar fitted with orange safety caps has been installed at the transect endpoints and plot centers. The location of each transect endpoint (0m and 20 m) and circular plot centers were georeferenced using a handheld GPS.

Herbaceous and shrubby vegetation within each quadrat were identified to species, or in some cases, to genus, and the cover was estimated to the nearest 1 per cent. At circular plots, only shrubs and ground cover were collected. Taxonomies used to identify plant species included Douglas et al (1998) and Brayshaw (2000).

Cover estimates were stratified into the following vegetation layers:

B1: Tall Shrubs (woody plants 2m to 10m tall)
B2: Low Shrubs (woody plants less than 2m tall)
C: Herbs (forbs and graminoids)
D: Moss, lichen, and seedlings

Ground surfaces were categorized as either bare soil (mineral, sand, or fines), coarse woody debris, rock, dead organic material, live organic material, or water, and the cover of each surface type was recorded for the quadrats and plots. Photographs were taken along the length of each transect to provide a visual portrayal of vegetation conditions and included close-ups of plant species and general views of each transect. Digital photographs were taken at each circular plot centre in the four cardinal directions (N, E, S, W).

4.3 Terrestrial Wetland Community Classification

In 2013, community descriptions for terrestrial wetlands were provided across the 752 to 755+ m elevation bands for each index site. These descriptions were based on an old digital elevation model that resulted in positioning two-thirds of the transects in the wrong elevation band. Consequently, over half of the community descriptions were incorrect. Using vegetation data from 2015 and the new digital elevation model, community descriptions and classifications were classified following the classifications of Hawkes et al. (2007, and 2010) or MacKenzie and Moran (2004). Communities were assigned to the closet wetland vegetation association based on the presence and abundance of plant species and the position of the community on the edatopic grid (Figure 4-3) using information on soil moisture, pH, hydrodynamic index, and soil nutrient regime.

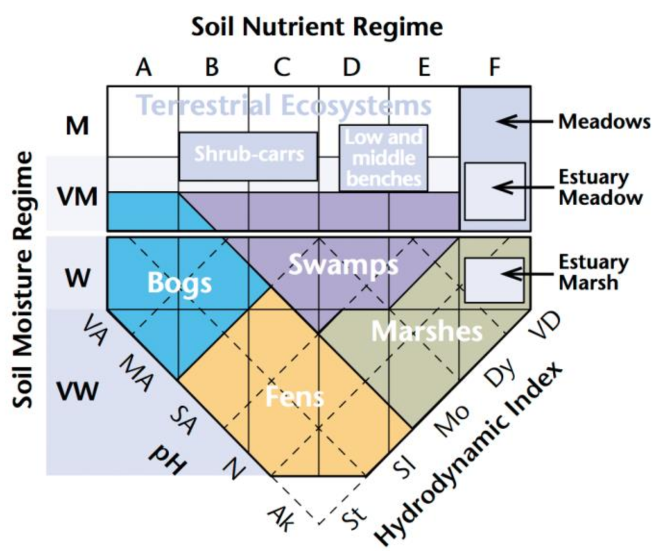

Figure 4-3: The position of wetland classes on the Edatopic grid (MacKenzie and Moran 2004). The four axes of the grid describe the following specific site characteristics: (1) Soil Moisture Regime including moist (M), very moist (VM), very wet (VW), (2) Soil Nutrient Regime from very poor (A) to hyper rich (F),(3) Soil pH from very acid (VA; < 4.5 pH), Neutral (N; 7 pH) to alkaline (Ak; >7.4 pH), and (4) Hydrodynamic Index ranging from stagnant (St), sluggish (Sl), Mobile (Mo), dynamic (Dy), to very dynamic (VD).

In conjunction with classifying the wetland communities, successional pathways were identified to describe the potential impact of water level changes on the communities. These progressions were identified using the species composition and site data collected from the vegetation transects. Where possible, we followed the progressions described in MacKenzie and Moran (2004) and, if required, adapted them based on the hydrology and climate at each index site.

4.4 Aquatic Wetland Monitoring

Wetland physicochemistry and aquatic macrophyte data were collected in each wetland (pond) at permanent sample stations established along a grid pattern using a small 2.5m inflatable boat. Table 4-2 provides the sampling intensity at each pond and the sample dates for 2013 to 2015.

Table 4-2: Sampling intensity and dates of ponds, 2013 to 2015.

Reach Site - Position	Elev. (m)	Area (m^2)	# Sample Stations	2013	2014	2015
Bush River						
Reference Pond	760.0	40,735	5	July 16	July 12	July 13
DDZ Pond	752.8	4,361	3	July 17	July 10	July 17
Km88						
Reference Pond	769.8	2,807	5	July 16	July 11	July 13
DDZ Pond	752.5/ 751.1	897	5	July 15	July 10	July 13
Sprague Bay						
Reference Pond	757	9,251	5	July 10	July 04	July 07
DDZ Pond	754.1	378	6	July 10	July 04	July 07
Valemount Peatlands						
Reference Pond	757.9	830	5	June 26	June 24	June 29
DDZ Pond	752.2	8340	8	June 24	June 23	June 28

4.4.1 Water Physicochemistry

Monitoring wetland physicochemistry is essential for assessing changes in wetland integrity and provides valuable information for interpreting biological data, verifying wetland classification, and diagnosing potential stressors (Finlayson and Davidson 1999; Mitsch and Gosselink 2007; US EPA 2008). At each station, point samples of water temperature, dissolved oxygen, conductivity, and pH were recorded at a depth 30 cm below the surface of the water using multi-metric meters (YSI Model 85 and 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 was recorded as "> 120 cm". Depths were measured using a weighted tape measure and recorded to the nearest centimeter. Organic muck depth was estimated by pushing a D-net handle into the sediment as a probe until met with stiff resistance. The presence of wood debris in benthic substrate was determined by probing the surface of the substrate and recording whether the probe struck wood. Sediments were characterized as muck (OM), wood (LWD), coarse organic matter (CO), or mineral sediment (MS). Photographs were taken at each sampling station in each cardinal direction (N, S, E, W).

Conductivity (Onset HOBO U24-001), depth (Onset Hobo U20L-01), and dissolved oxygen (PME MiniDOT) data loggers were installed to collect continuous data for monitoring changes in water physicochemistry and aquatic metabolism. Data loggers were installed in each pond between June 15 and June 23 of each year and were affixed to ¾" rebar using a pipe clamp in the middle of the water column at approximately 50 cm depth. The DO data loggers were programmed to record data every 10 minutes and calibrated by the manufacturer prior to installation. Depth and conductivity loggers were programmed to record data every 4 hours. Data loggers were retrieved in fall or winter as reservoir levels receded to 751m. Data were downloaded using the manufacture's software (Onset Hoboware and PME miniDOT software). The PME miniDOT DO loggers were recalibrated by the manufacture prior to installation.

Onset weather stations were deployed to collect wind speed, barometric pressure, air temperature, photosynthetic active radio (PAR). These data are required to estimate atmosphere diffusion of dissolved oxygen to calculate pond metabolism (NEP, R, GPP). Weather stations took samples every minute, which were averaged and recorded at ten minute intervals.

4.4.2 Aquatic Macrophytes

Aquatic macrophyte biomass samples were collected as an index of primary productivity. Samples were obtained using the macrophyte grapnel (Figure 4-4). The grapnel (constricted from two rakes bound together to make a double headed rake) was tossed from the boat and allowed to settle on the bottom of the pond. Once on the bottom, the grapnel was dragged to the boat capturing submergent vegetation within the tines of the rakes. Upon hauling the grapnel into the boat, overall vegetation abundance was estimated in cover based on amount of vegetation that passed across the plane of the rake tines. Samples were then bagged and labeled for biomass measurement. We attempted to collect two grapnel samples at each station. Samples with no macrophytes were recorded as 0 biomass and unsuccessful samples were not included in the analysis. Unsuccessful samples were due to water depths that were too deep to sample. Biomass samples were stored in an ice cooler until the end of the field day and then transferred to a freezer. In the lab, the samples were dried at 75 °C for 72 hours. Dry weight (g) of each sample were obtained from a digital balance.

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

4.5 Data Management

Field data were recorded on forms printed on water resistant paper (Adama et al 2015). Sample location were georeferenced using a handheld Garmin® GPSMap 62 handheld receiver. At the end of each field day data forms were sorted and placed in the project binder and all digital data (GPS, and digital images) were transferred to a laptop. At the end of each field session, data sheets were digitally scanned and copied to the server at the LGL Office in Sidney.

Several databases have been developed for this project. In 2015, an SQLite database was developed to store continuous water physicochemistry data and a Filemaker Pro (v15) database was developed for the terrestrial vegetation data. Point source water physicochemistry data and macrophyte data were entered and stored as Microsoft Excel spreadsheets. Data entry commenced following the completion of the summer field session and continued as additional information was obtained (i.e., continuous water physicochemistry data). To prevent data loss, computers are backed up daily using a real-time computer syncing to external hard drives and offsite servers.

4.6 Data Analyses

4.6.1 General Analyses

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 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.

4.6.2 Shrub and Herb Layer Cover

Vegetation abundance was assessed from total shrub and herb cover data collected from circular plots (shrubs) and transects (herbs). Data from 2015 was analyzed to describe the general pattern of vegetation abundance across elevation bands and index sites. To assess trends over time, only circular plots and transects established in 2013 and sampled in all three years (2013, 2014, and 2015) were included in our analyses.

Linear Mixed Effects (LME) modelling (Zuur et al. 2009) was used to model differences as a function of elevation band, location (Index Site), and time (2012 to 2015). Mixed modelling was performed using the JMP (2016). Restricted Maximum Likelihood estimation (REML) was used to obtain the test statistics and estimates. One-way ANOVAs were performed to identify differences.

4.6.3 Aquatic Macrophyte Biomass

Box plots and summary statistics were generated to assess the structure and symmetry of the data. A data transformation of log (1 + biomass) was applied to address the asymmetry and the large number of zero data. LME (Zuur et al. 2009) was used to model differences as a function of pond position (within or above the reservoir: Drawdown [DDZ] and Reference [REF], site (index site), and time (2012 to 2015). Mixed modelling was performed using JMP (2016). Restricted Maximum Likelihood estimation (REML) was used to obtain the test statistics and estimates.

4.6.4 Pond Metabolism

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 data logger 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}.

Data from the weather stations and data loggers were imported into a SQLite database and prepared for analysis. Data preparation included converting timestamps to UTC, aligning timestamps across datasets, and trimming superfluous data. Data were then imported into R for analysis.

Values for NEP, GPP, and R from 2014 and 2015 were estimated with the *metab.kalman* model using the LakeMetabolizer package in R (Winslow et al 2016a). Data from 2013 were not included in our analyses as the associated weather data were not available. The *metab.kalman* model implements observation and process error dynamic linear regression models to estimate metabolism by finding the parameter set that corresponds to the maximum likelihood of the model given the data. The model also employs a Kalman filter that smooths the DO time series (Winslow et al 2016b). Equations for calculating NEP, GPP, and R are provided in Staehr et al (2010).

As defined, negative GPP and positive R are ecologically impossible, but unfortunately, metabolism estimates using free-water oxygen often return negative GPP and positive R. These impossible results are often from days when physical processes (e.g., wind mixing) dominate the DO signal and overwhelm the biological signal by other sources of DO variability (Rose et al. 2014). To constrain the model, all values estimates with negative GPP and positive R where removed from the data set prior to analyses. In total, almost a third of the values were removed for analysis.

Box plots and Violin plots of NEP, GPP, and R values were generated for the DDZ and REF ponds from June 21 to August 31st for data collected in 2014 and 2015. Violin plots display the distribution and shape of the data and its probability density (Hintze and Nelson, 1998). Wider sections of the violin plot represent a higher probability that members of the population will take on the given value; the skinnier sections represent a lower probability. Variation among metabolic values across sites, position, and years was assessed using Brown-Forsythe test for equal variance.

LME modelling was used to model differences as a function of pond position (within or above the reservoir: Drawdown [DDZ] and Reference [REF], and time (2014 to 2015). Restricted Maximum Likelihood estimation (REML) was used to obtain the test statistics and estimates. To address asymmetry in their distributions towards zero, GPP was log transformed and the absolute value of R was log (1 + R) transformed. LME was performed using JMP (2016). To examine the potential effects of inundation on metabolic variables, GPP, R, and NEP were calculated for two 25-day periods in July and August (which correspond to pre-and post-inundation period that occurred in 2014); LME modelling was used to model differences between position and inundation period.

5.0 RESULTS

5.1 Reservoir Levels

The operating regime of Kinbasket Reservoir begins at low levels in early spring, refilling during spring and summer, and then discharging through the fall and winter. The hydrograph in Figure 5-1 illustrates this sequence and provides a graphic interpretation of the water levels since 2010, of the historical maximum and minimum, and of the 1987–2006 baseline mean.

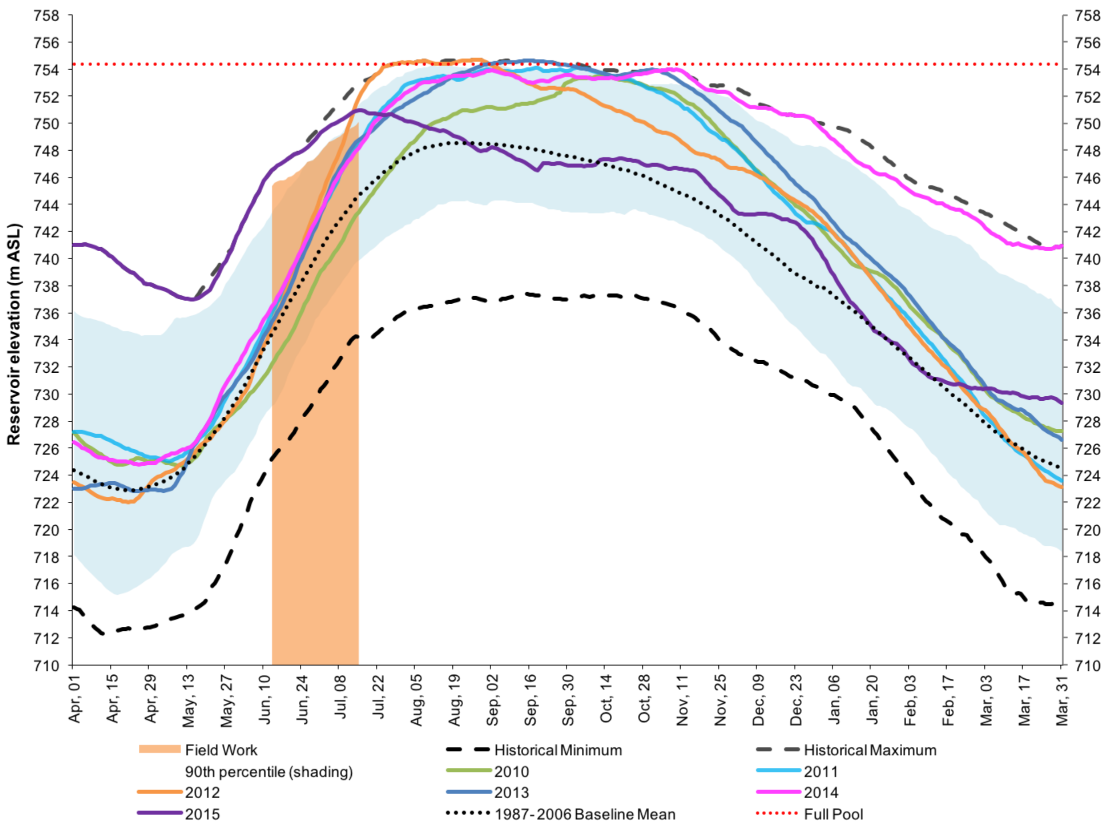

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Figure 5-1: Kinbasket Reservoir hydrograph for the period 2011 through 2015. The black dotted line represents the 1987 to 2006 mean baseline operating regime. The red dotted line represents the normal operating maximum (754.28 m) termed “full pool. Orange shading indicates the timing of fieldwork in 2014 and 2015. Blue shaded area represents the 10th and 90th.

In 2014, water levels were normal during the early part of the year. The reservoir peaked at 753.89 m in early September 2014 and remained historically high until November 2014, where it peaked again at 753.98m. Reservoir levels were historically high through the winter and early spring of 2015 but peaked only at 751.0 on July 19th, 2015. Since 2011, the reservoir has been operating at levels much higher than the 1987 - 2006 historical norm (Table 5-1).

Table 5-1: The number of days that the elevation bands of 752 m and above were inundated in 2010 to 2015 compared against the 1987 - 2006 historical average.

Year	Elevation Band (m)			Max. Elevation
	752-753	753-754	754 -755	
2010	46	35	0	753.48
2011	100	83	9	754.08
2012	84	58	43	754.68
2013	104	87	41	754.63
2014	122	99	0	753.98
2015	0	0	0	750.95
1987 – 2006	24.65	16.5	6.7	749.70

5.2 Weather Conditions

Seasonal weather conditions affect water physicochemistry parameters including water temperature, pH, conductivity, and dissolved oxygen, as well as wetland productivity and the growth of wetland plants. Large fluctuations in daily air temperature and rainfall were observed in 2014 and 2015 and were attributed to summer storms and periods of unsettled weather. Air temperatures in April, May, and June were significantly warmer in 2015 than in 2014 ($p = 0.03$, $p = 0.002$, and $p < 0.001$, respectively), similar in July ($p = 0.035$), and cooler in August and September ($p = 0.05$ and $p < 0.001$, respectively; Figure 5-3). Between 2012 and 2015, mean daily air temperatures increased during May and June ($p = 0.006$ and $p < 0.001$) and decreased in September ($p = 0.03$; Figure 5-3). Mean daily temperatures increased in May and June by 0.67 to 1.05 °C annually from 2012 to 2015.

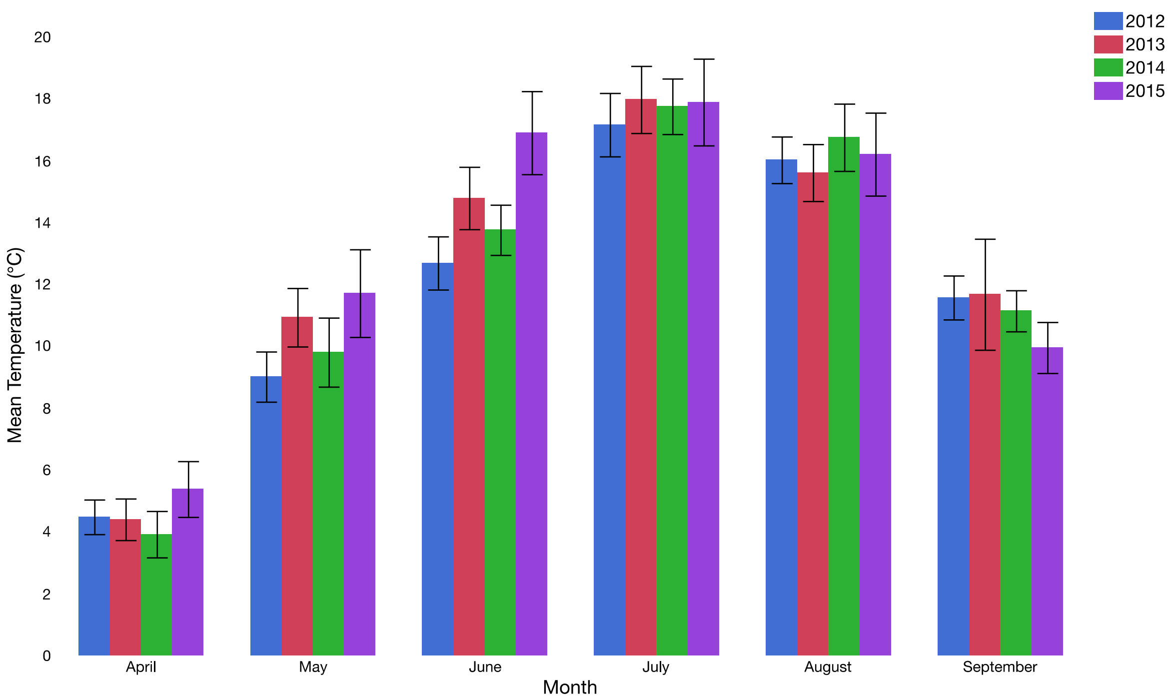

Figure 5-2: Daily mean air temperature at Mica Creek during the growing season (April through September) 2012 to 2015. Bars indicate 95% confidence intervals around the mean.

5.3 Sampling Effort

In 2014, fieldwork commenced on June 18, 2014 beginning with the installation of weather stations and data loggers, which was completed on June 24, 2014. Field surveys occurred from June 25 through to July 13th, 2015 (Figure 5-1). High reservoir levels in the fall and early winter of 2014 delayed the retrieval of data loggers until January 2015. Due to high snowfall, a helicopter was required to access the Km88 sites.

In 2015, fieldwork commenced on June 17, 2015 and weather stations and water physicochemistry data loggers were installed between June 17 and June 23, 2015. Field surveys occurred between June 24 and July 17, 2015 (Figure 5-1). Facilitated by normal water levels, the weather stations and water data loggers were retrieved in October 2015. The number of terrestrial and aquatic vegetation plots and transects sampled from 2012 to 2015 are presented in Table 5-2.

Table 5-2: Sampling effort at the four index sites in the Kinbasket reservoir by year and elevation band. In 2015, additional plots and transects were added to balance the study design against the 2014 digital elevation model. T = Transect; CP = Circular Plot.

Wetland Type	Index Site	Elevation Strata	2012		2013		2014		2015	
			T	CP	T	CP	T	CP	T	CP
Terrestrial	Valemount Peatland	Below 752			5	10	5	19	4	14
		LC (752-753)	1	2	5	10	5	22	5	22
		Target (753 –754)	1	2	3	6	3	12	5	20
		UC (754 –755)			3	6	3	12	4	16
		REF (755 +)							4	16
	Sprague Bay	Below 752							2	8
		LC (752-753)			2	4	2	8	2	8
		Target (753 –754)							2	8
		UC (754 –755)			4	8	4	16	2	8
		REF (755 +)			2	4	2	8	2	8
	Km 88	LC (752-753)	1	2	2	4	2	8	2	8
		Target (753 –754)	1	2	2	4	2	8	2	9
		UC (754 –755)	1	2	2	4	2	8	2	8
		REF (755 +)			2	4	2	8	2	8
	Bush River	LC (752-753)			3	6	3	12	3	16
		Target (753 –754)			3	6	3	12	4	12
		UC (754 –755)							2	8
		REF (755 +)			2	4	2	8	2	8
		Total Transects/Plots Sampled	**5**	**10**	**40**	**80**	**40**	**160**	**52**	**8**
Aquatic	Valemount Peatland	DDZ	8		8		8		8	
		REF	5		5		5		5	
	Sprague Bay	DDZ	7		5		5		5	
		REF	6		5		5		5	
	Km 88	DDZ	3		5		5		5	
		REF	**0**		5		5		5	
	Bush River	DDZ	**0**		3		3		3	
		REF	**0**		5		5		5	
		Total Sites Sampled	**5**		**8**		**8**		**8**	
		Total Stations Sampled	**29**		**41**		**41**		**41**	

5.4 Terrestrial Wetlands

5.4.1 Terrestrial Wetland Community Classification

Table 5-3 provides a revised list of the terrestrial wetland vegetation associations identified in transects established in the four index sites. The descriptions incorporate the 2014 digital elevation model (DEM) and new transects and circular plots established in 2015. Detailed descriptions of the vegetation associations along with representative images are provided in Appendix C (Section 8.3).

Table 5-3: Terrestrial vegetation associations corresponding to the sites and elevation strata sampled in Kinbasket Reservoir, 2015. Vegetation associations that changed between 2013 and 2015 are in bold.

Site	Stratum	Wetland Vegetation Associations	Source
Valemount Peatland	**750 –752**	**Swamp Horsetail (SH)** **Swamp horsetail –Beaked sedge (Wm02)**	**Hawkes *et al.* 2007** **MacKenzie and Moran 2004**
	752 –753	Swamp Horsetail (SH) Swamp horsetail –Beaked sedge (Wm02)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
	753 –754	**Driftwood (DR)** **Transitional between Swamp horsetail –Beaked sedge (Wm02) and Scrub birch–Buckbean–Shoresedge (Wf07)**	**Hawkes et al. 2007** **MacKenzie and Moran 2004**
	754 –755	**Willow–Sedge (WS)** **Transitional between Scrub birch–Buckbean–Shoresedge (Wf07) and the Scrub birch –Water sedge (Wf02)**	**Hawkes et al. 2007** **MacKenzie and Moran 2004**
	755 +	**Black spruce –Water sedge –Peat-moss (Wb05)**	**MacKenzie and Moran 2004**
Sprague Bay	**752 –753**	**Reed Canary Grass (RC) / Beaked Sedge Water Sedge (Wm01)**	**Hawkes et al. 2007** **MacKenzie and Moran 2004**
	753 –754	**Pink spirea –Sitka sedge (Ws50)**	**MacKenzie and Moran 2004**
	754 –755	Scheuchzeria-Peat-moss (Wb12)	MacKenzie and Moran 2004
	755 +	Scheuchzeria-Peat-moss (Wb12) Slender Sedge–Buckbean (Wf06)	MacKenzie and Moran 2004
Km 88	752 –753	Buckbean–Slender Sedge (BS) Slender Sedge–Buckbean (Wf06)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
	753 –754	Willow–Sedge (WS) Scrub birch–Buckbean–Shoresedge (Wf07)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
	754 –755	Willow–Sedge (WS) Scrub Birch–Water Sedge (Wf02) Western redcedar –Spruce –Skunk cabbage (Ws10)	Hawkes *et al.* 2007 MacKenzie and Moran 2004
	755 +	Tufted clubrush –Star moss (Wf11)	MacKenzie and Moran 2004
Bush River	752 –753	**Driftwood (DR)** **Unspecified Willow Flood community (Fl01)** **Beaked sedge –Water sedge (Wm01)**	**Hawkes *et al.* 2007** **MacKenzie and Moran 2004**
	753 –754	Willow–Sedge (WS) Transitional Willow Sedge Swamp/Flood community (Ws/Fl)	**Hawkes et al. 2007** **MacKenzie and Moran 2004**
	754 –755	**Unspecified Willow Flood community (Fl02)**	**MacKenzie and Moran 2004**
	755 +	Slender sedge –Common hook-moss (Wf05)	MacKenzie and Moran 2004

5.4.1.1 Valemount Peatland

Changes in the DEM affected 14 of the 16 transects previously established in the Valemount Peatland. On average, transects were shifted down in elevation by 120 cm. To balance the study deign, seven new transects were established including four in the reference elevation (755m+) (Figure 9-2). This significantly affected the classification of terrestrial wetland communities (Adama et al 2014).

Four transects originally assigned to the 752–753m elevation band and one transect in the 753–754m elevation band were shifted to 750–752m (Figure 9-2). The vegetation community at the elevation band was sparse (15 per cent) with very little shrub cover (< 1 per cent), mainly sprigs of young *Salix*. Dominated by *Equisetum fluviatile*, *Menyanthes trifoliate, Comarum palustre,* and *Scirpus microcarpus,* this community was again classified as Swamp Horsetail (SH) of Hawkes et al (2007), which corresponds to the Swamp horsetail –Beaked sedge (Wm02) vegetation association of MacKenzie and Moran (2004). The vegetation community between 752–753m was also assigned to the SH and Wm02 associations. Herb abundance in these transects increased to 29.3 per cent and shrubs (e.g., *Salix pedicularis* and *S. planifolia*) were starting to become established but were still less than one per cent.

The vegetation community at 753–754m was classified as a DR (driftwood) of Hawkes et al (2007) reflecting the large amount of wood that has accumulated in this elevation band. Based on species composition*),* this community was classified the community as transitional between a Wm02 marsh and fen (Wf07) (MacKenzie and Moran 2004). The dominant shrub cover (*Betula nana*, *Salix pedicellaris*, and *S. planifolia*

The wetland association at 754–755m was assigned as a transitional Scrub birch–Buckbean–Shoresedge (Wf07) and the Scrub birch–Water sedge (Wf02) associations. The Wf07 association occurred at the lower and wetter edge of the elevation band and Wf02 at the upper end of the elevation band above the influence of the reservoir (754.4 m). Although somewhat similar, these associations differ in their hydrodynamic index and soil development (MacKenzie and Moran 2004). Wf02 is dryer with deeper soils that are often hummocked.

The wetland association in the adjacent reference elevation band (755m+) was not sampled previously. It was classified as a developing Black spruce–Water sedge–Peat-moss (Wb05) association. This elevation band had hummock vegetation like the Wf02 associations. MacKenzie and Moran (2004) suggests that the Wb05 represents a successional intermediate between sedge fens and “true” bog.

Wetland communities in the Valemount Peatland appear to progress from a Swamp Horsetail Marsh below 753 m to a Scrub Birch dominated Fen from 753 –754 to a developing Black Spruce Peat bog above the reservoir (Figure 5-3). Wood debris accumulation and reservoir levels appear to be important site modifiers.

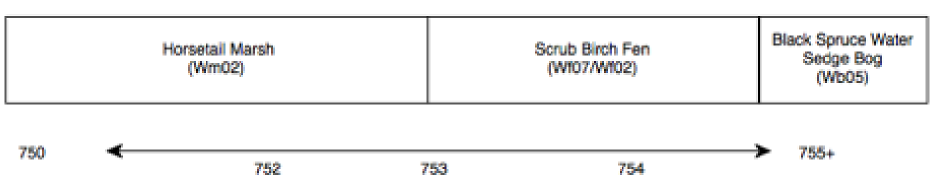

Figure 5-3: The progression of wetland communities in the Valemount Peatland (750 to above 755 m).

5.4.1.2 Sprague Bay

Changes to the elevation bands affected half (4 of 8) of the transects located at Sprague Bay. Two new transects were established in the 752–753m elevation band, and two transects, which were formerly thought to be in the 752–753m elevation band, were assigned to the 753–754m elevation band (Figure 9-4). This altered the wetland classification at these elevations.

The new transects established at 752–753m were assigned to the Reed Canary Grass and Wm02 associations. The presence of stumps indicates that the site was formerly an upland forest and not a natural wetland. The abundance of wetland vegetation (e.g., *Calamogrostis canadensis, Carex aquatalis*, *Comarum palustre*, and *Equisetum fluviatile,* and *Scirpus microcarpus*) was due to local seepage.

The vegetation community at 753 –754m elevation band was classified as Pink spirea–Sitka sedge (Ws50) association of MacKenzie and Moran (2004), which is consistent with our previous classification. This transect is dominated by *Spirea douglasii*, which is characteristic of disturbed water receiving sites and fluctuating water tables (Klinka et al. 1989). Hawkes et al (2007) mapped this community as a Willow Shrub but the species composition was too dissimilar.

The wetland community at 754–755m was classified as developing Scheuchzeria-Peat-moss (Wb12) bog. Soils were Sphagnum peat (*Sphagnum angustifolium, S. subsecundum, S. squarrosum,* and *S. warnstorfii*) on floating mats that extended over open water. This association requires permanent saturation but does not tolerate flooding of more than several centimetres. Adjacent open water is stagnant, slightly acidic (pH = 6.1), and was low in DO (3.0 mg l^{-1} O_2), which are indicators of developing bog wetlands. The vegetation community at 755m+ also included the Wb12 association along with Wf06. The Wf06 association occurred on poorly developed floating mats covered by a considerable amount of standing water (24 to 48 per cent).

Wetland transects at Sprague Bay progressed from disturbed Reed Canarygrass herb and shrub communities between 752–754m to developing bog communities above the influence of the reservoir (Figure 5-4).

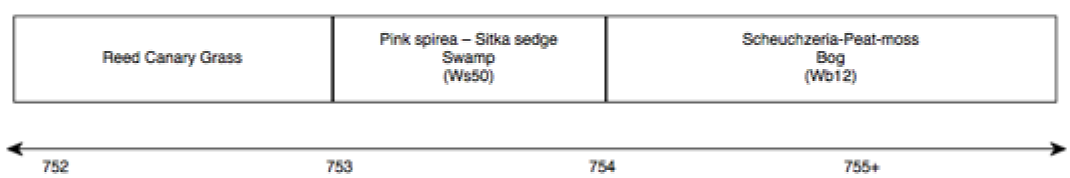

Figure 5-4: The progression of wetland communities at Sprague Bay (752 to above 755 m).

5.4.1.3 Km88

Transects at Km88 were influenced the least by the new DEM and only a single transect was affected (Figure 9-6). The vegetation community at 752–753m was classified as the Buckbean–Slender Sedge (BS) of Hawkes et al (2007) and Slender Sedge–Buckbean (Wf06) of MacKenzie and Moran (2004). As per its namesake, this wetland association is dominated by buckbean (*Menyanthes trifoliate*) and Slender Sedge (*Carex lasiocarpa*). With increased flooding, this community may shift towards either a Wm01 or Wm02 community.

The vegetation community at 753–754m was classified as the Willow–Sedge (WS) association of Hawkes et al (2007) and the Scrub birch–Buckbean–Shoresedge (Wf07) association of MacKenzie and Moran (2004). The vegetation included a diverse (34 species) mix of marsh and fen species (e.g., *Menyanthes trifoliata, Betula pumila, Typha latifolia, Equisetum palustre*) reflecting a range hydrological of characteristics at this elevation band. As in the lower elevation band, reservoir operations likely play a role in determining the plant community; it may shift towards Wm01/Wm02 following several years of high reservoir levels and to possibly Wf02 during lower years.

The vegetation community at 754–755m was classified as the WS community of Hawkes et al (2007) but was difficult to classify following MacKenzie and Moran (2004). It is corresponded most closely to a Scrub Birch–Water Sedge (Wf02) / Western redcedar–Spruce–Skunk cabbage (Ws10) transition community. With 68 species, it is the most diverse community sampled under CLBMON61. *Betula pumila* and *Thuja plicate* were the dominate shrubs and the understory was dominated by *Menyanthes trifoliate.* The red listed orchid, *Liparis loeselii* occurred in this elevation band.

The wetland sampled at 755m+ was an open fen meadow transitioning to swamp communities at the periphery. This community was classified as a developing Hudson Bay clubrush–Red hook-moss (Wf10) association surrounded by a Western redcedar–Spruce – Skunk cabbage (Ws10) association. This community was dominated by *Menyanthes trifoliate, Trichophorum cespitosum*, *Eleocharis elliptica, Equisetum palustre,* and *Trichophorum alpinum* and is associated with alkaline soils, which are typical of Bush Arm. This community was classified with Hawkes et al (2007).

Overall, the progression of wetland communities at Km88 appeared to be Marsh (Wm01/02) to Fen (Wf05/Wf06) to Wf02/07 to Wf10 with the upper elevations intergrading to Swamp (Ws10).

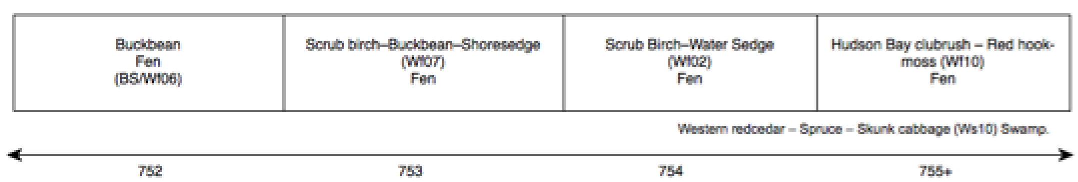

Figure 5-5: The progression of wetland communities at Km88 (752 to above 755 m).

5.4.1.4 Bush River

At Bush River, the only elevation band unaffected by the new elevation data was 755+ (Figure 9-8). Two new transects were established in the 754–755m elevation band and three transects shifted to adjacent elevation bands. An additional transect was also established at 752–753m to replace a transect that was identified for wood debris removal.

Three different wetland communities occurred in the 752–753m elevation band. One transect was in a Driftwood (DR) association (Hawkes et al. (2007), two transects were in an unspecified Flood association (Fl) of MacKenzie and Moran (2004). A fourth transect was located is a developing Water sedge–Beaked sedge fen (Wm01). Vegetation cover less than 10 per cent in the DR and FL association and the main difference between them was substrate cover. Substrate of the DR association was 90 per cent wood debris whereas the substrate of FL association was 83 per cent exposed mineral soil. Substrate of the Wm01 association was 89 per cent moss. Vegetation cover of Wm01 was 29 per cent and dominated by *Carex utriculata, C. aqualitis, Equisetum palustre*, and *E. fluviatile*.

The vegetation community in transects at 753–754m had a diverse willow community and corresponded to the Willow Shrub (WS) associations of Hawkes et al. (2007). Under MacKenzie and Moran (2004), it was classified as transitional between the lower flood community at 752–753m to a developing Willow Swamp community that occurs along the margin of the floodplain.

The community at 754–755m was sparsely vegetated and did not fit the classification of Hawkes et al. (2007) or MacKenzie and Moran (2004). It is most similar to the *Dryas drummondii* (Yellow Mountain Avens) communities described by Kembel (2000). The community is located on an silt/gravel bar in the Bush River floodplain and is prone to seasonal flooding, sediment deposition, and erosion, suppressing the vegetation communities from developing further.

The reference transects (755m +) were in a fen meadow adjacent the Bush River, 3km upstream from the lower transects. This community corresponded most closely to the Slender sedge –Common hook-moss (Wf05) association of MacKenzie and Moran (2004). It was similar to the Wm01 community at 752–753m but it was more developed with higher species diversity and abundance.

The progression of wetland communities at the Bush River Index site is complex (Figure 5-6). The Bush River causeway bisects the index site at 752–753m. The DR and Fl community occurred on the downstream side of the causeway and are exposed scouring by the reservoir and to wood accumulation in high fill years. The Wm01 association occurred on the upstream side of the causeway. The causeway appears to be attenuating the effects of inundation, protecting the site from scouring and wood debris accumulation.

The general pattern of wetland progression appears to be Flood community (FL) to Willow Shrub (WS) to Forest (FO) –along the main river channel, and Water sedge –Beaked sedge Fen (Wm01) to Slender sedge – Common hook-moss (Wf05) along protected side channels.

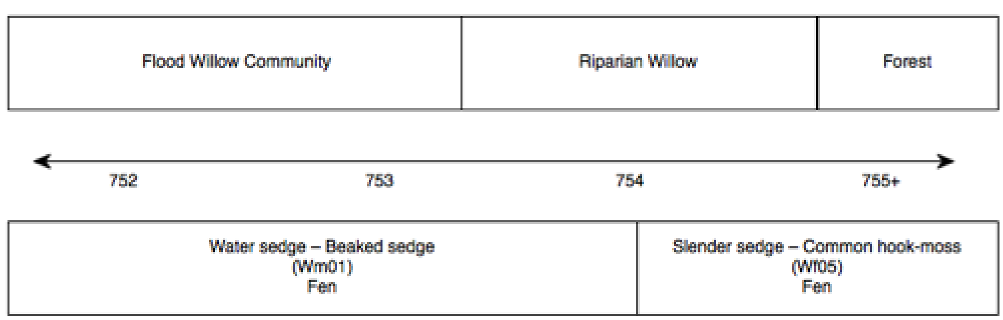

Figure 5-6: The progression of wetland communities at Bush River (752 to above 755 m). The upper progression occurs along the main channel at water shedding sites. The lower progression occurs in side channels at water receiving sites.

5.4.2 Vegetation Cover

5.4.2.1 Shrub Cover

Box plots show the variability in shrub cover among elevations, sites, and across years (Figure 5-7). Shrub cover did not differ among years (LME, F= 0.17, p=0.98) and there was no interaction between Year and Elevation band (F = 0.3, p=0.98). This indicates that total shrub cover did not statistically increase or decrease within the elevation bands over the three-year period (2013 and 2015).

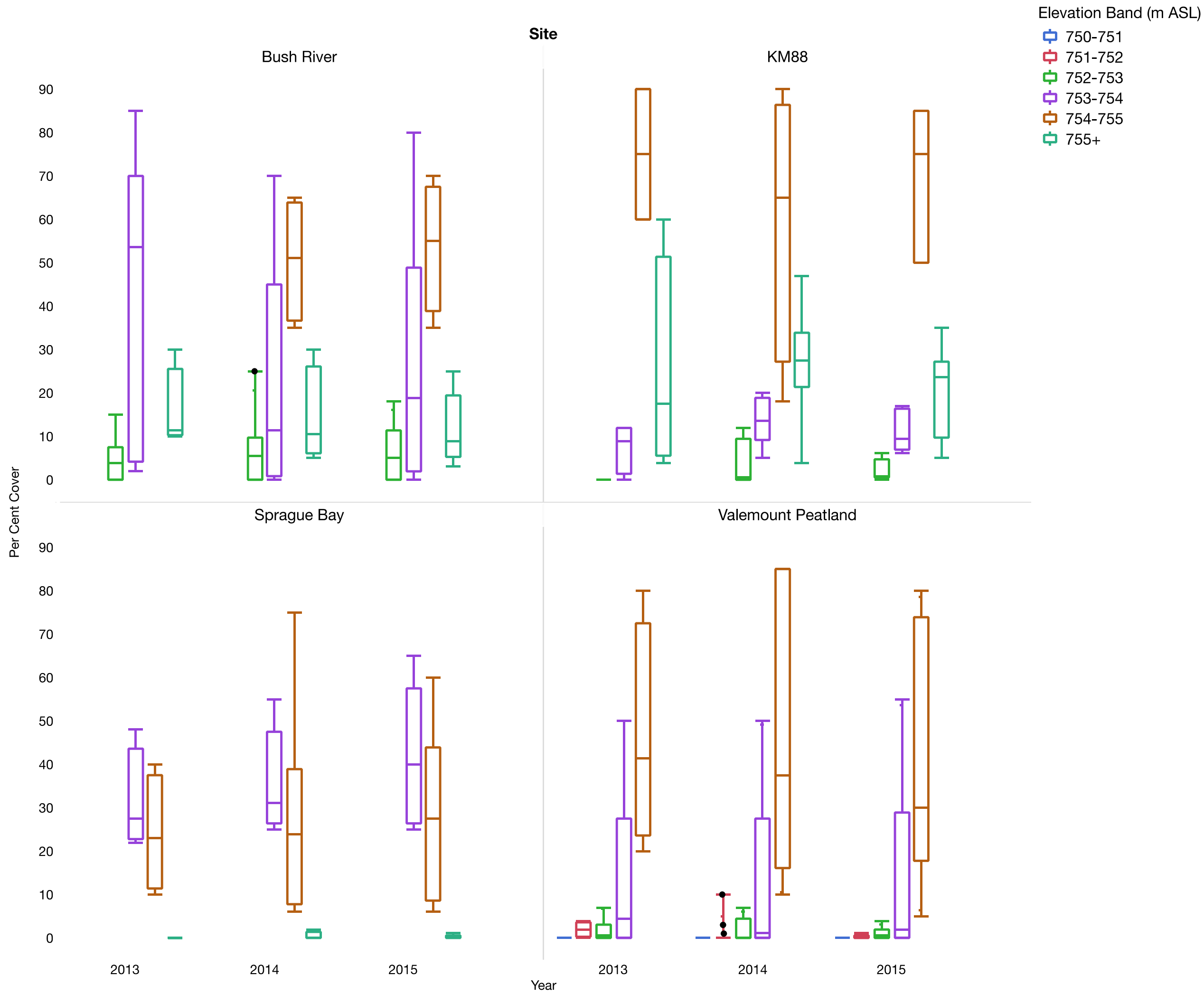

Figure 5-7: Box plots of shrub cover (per cent) from 2013 to 2015 across 1 m elevation bands at the 4 wetland index sites in Kinbasket Reservoir.

Shrub cover from circular plot data collected in 2015 differed significantly across elevation bands (F = 29.5, p< 0.001; all sites pooled; Figure 5-8). In general, cover increased from less than 1 per cent at 751m to 19.3 per cent ± 5.4 at 753–754m, and to 44.7 per cent ± 5.6 at 754–755m. In reference transects, shrub cover generally lower at 13.0 per cent ± 6.38 except for the Valemount peatland where cover values exceeded 50 per cent. These values reflect the progression of wetland communities across the elevation gradient from low shrub cover as described in degraded marsh and flood communities to high shrub abundance in the shrub birch and willow communities at 753 and 754 m.

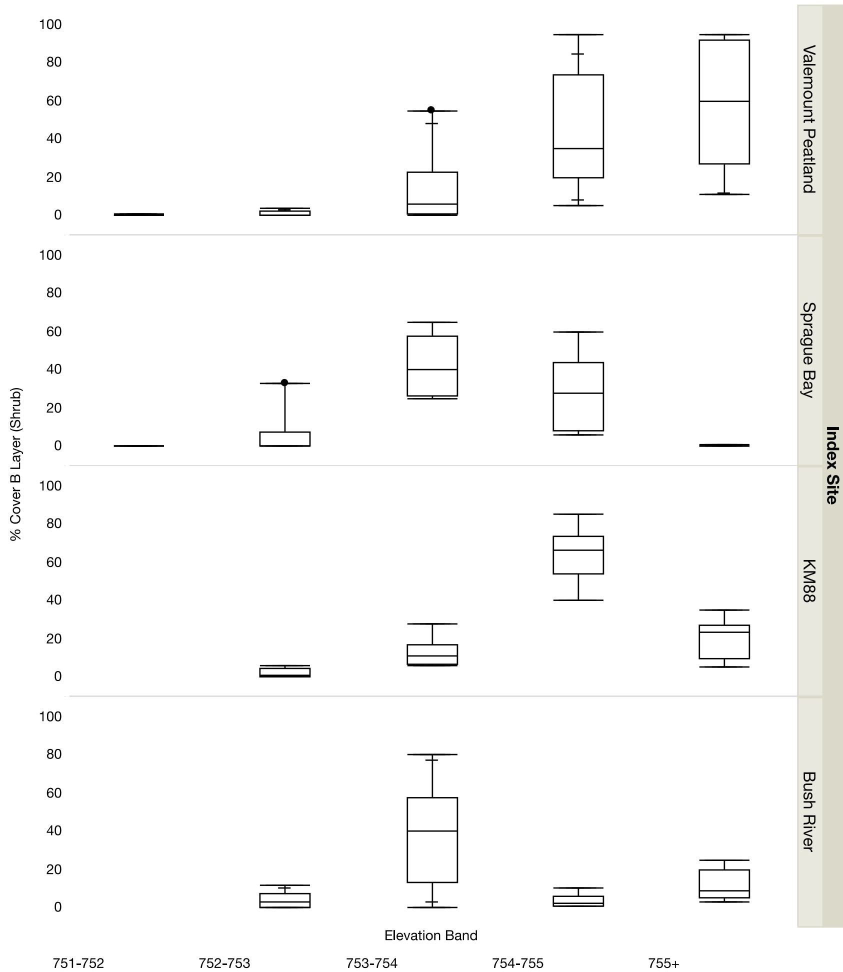

Figure 5-8: Box plots of shrub per cent cover in 2015 across 1m elevation bands in Kinbasket Reservoir.

5.4.2.2 Herb Cover

Box plots in **Figure 5-9** show the variability in herb cover data among elevations, sites, and across years. LME modeling of the 2013 to 2015 data (excluding the new transects established in 2015), indicated that herb cover differed across sites ($F = 53.5$ per cent, $p < 0.001$) and elevation bands ($F = 6.6$, $p < 0.001$), but not among years ($F = 0.7$, $p=0.49$): the interaction between Year *Elevation Band was not significant ($F = 1.3$, $p=0.24$). This indicates that while there may have been differences in herb cover across sites and elevations bands, which is expected, herb cover within the elevation bands did not differ statistically over the three-year period (2013 and 2015).

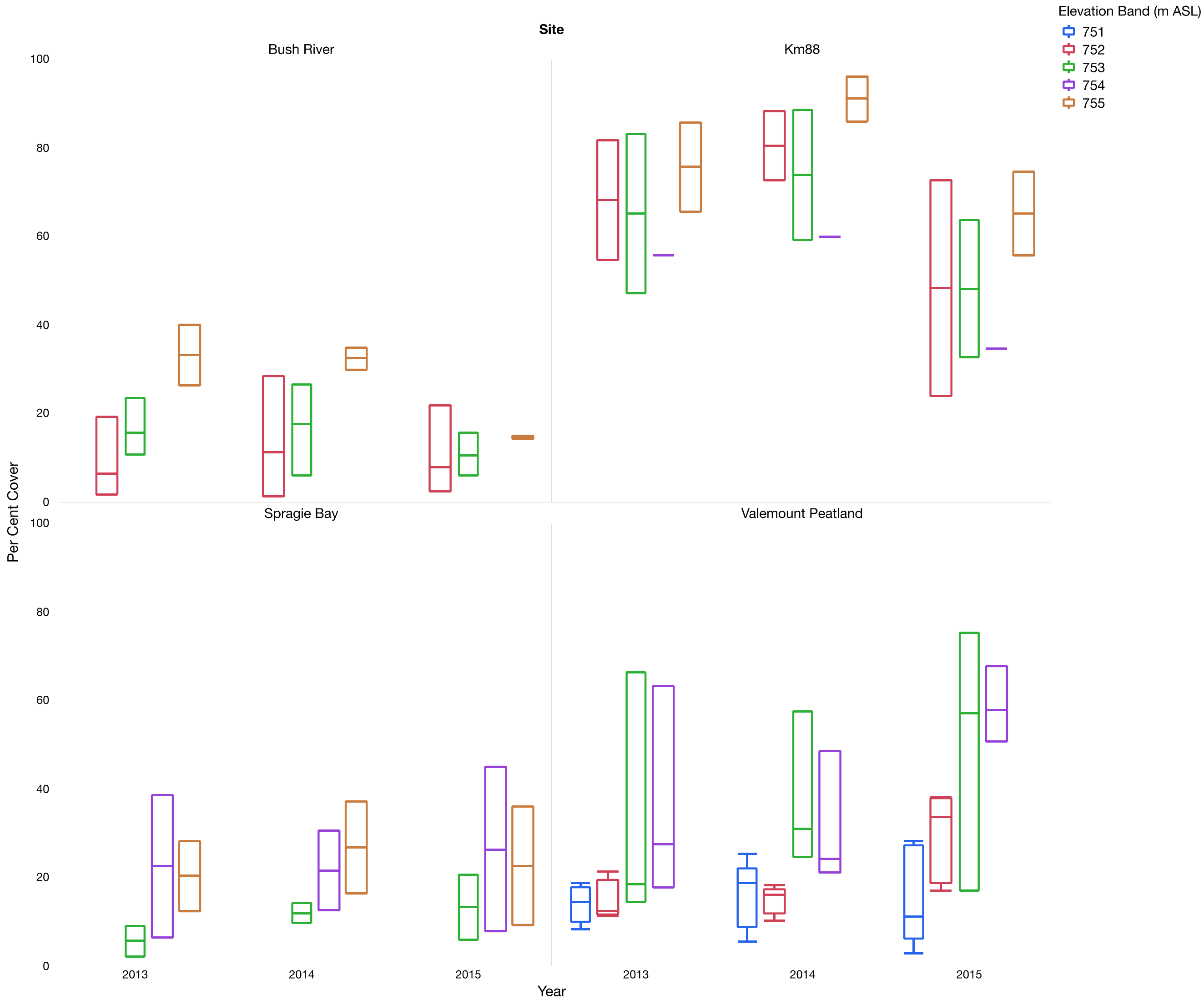

Figure 5-9: Box plots of herb cover data from 2013 to 2015 across 1 m elevation bands at the 4 wetland index sites in Kinbasket Reservoir.

Box plots show the variability of shrub cover among elevations and sites in 2015 (Figure 5-10). Total herb cover (per cent) was highest at Km88 (51.9 per cent ± 24.3), moderate at the Valemount Peatland (40.4 per cent ± 32.0) and Sprague Bay (30.6 per cent ± 7.3), and lowest at Bush River (11.1 per cent ± 10.1). In contrast with shrub cover, differences in herb cover across elevation bands were not as apparent (F= 2.5, p= 0.05) and differences were only significant across elevation bands in the Valemount Peatland (one-way ANOVAs, F= 6.5, p= 0.0022; Figure 5-10).

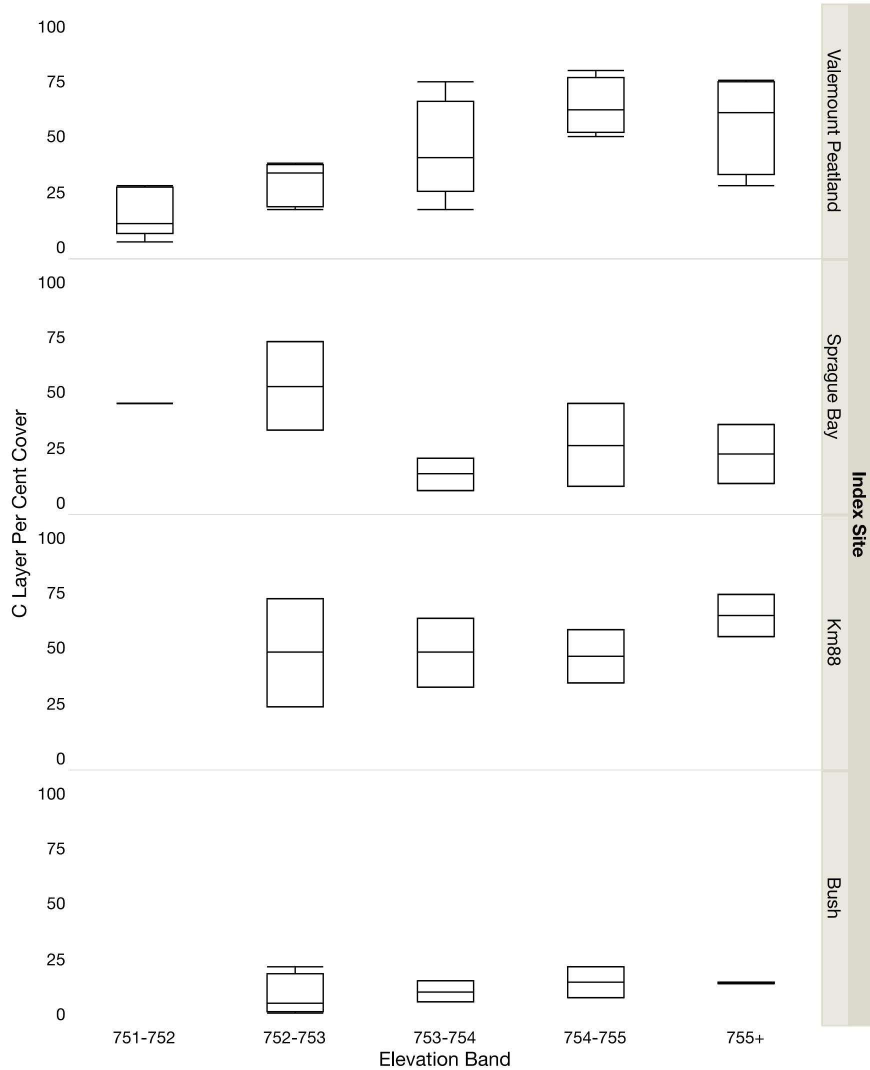

Figure 5-10: Box plots of herb per cent cover across 1m elevation bands in Kinbasket Reservoir. Data from 2015.

5.5 Aquatic Wetlands

5.5.1 Water Physicochemistry

Temperature, pH, conductivity, DO, and water depth measurements were obtained from point samples in the eight-study ponds in late-June to mid-July of 2014 and 2015. Mean and standard deviations of the measurements, along with data collected in 2013, are provided in the Table 5-4; box plots (Figure 9-10 Section 9.4 Appendix) provide a graphic interpretation of their distribution. One-way ANOVAs revealed that many of these values differ significantly among years. As differences for some comparisons can be explained by sampling error, time of day, weather (e.g., rainfall, air temperature) or seasonal variability, only important patterns are discussed.

Table 5-4: Mean values of water physicochemistry data collected in ponds in Kinbasket reservoir. Red values indicate notable changes among years.

Site	Pond Position	Year	Temp (°C) Mean	Temp (°C) SD	pH Mean	pH SD	Conductivity (µS/cm) Mean	Conductivity (µS/cm) SD	DO (mg l-1) Mean	DO (mg l-1) SD	Depth (cm) Mean	Depth (cm) SD
Bush River	DDZ	2013	19.4	1.0	8.8	0.6	228.1	104.4	9.1	1.2	44.7	16.8
		2014	27.3	1.4	9.7	0.7	249.1	20.9	16.0	4.2	31.0	7.2
	n = 3	2015	21.5	2.5	9.4	0.8	184.8	76.6	14.8	2.8	40.5	18.8
	REF	2013	15.0	1.5	8.7	0.1	129.7	7.1	10.6	0.6	67.4	19.1
		2014	13.5	0.3	8.8	0.1	124.0	1.5	9.8	0.7	67.0	26.8
	n = 5	2015	17.8	2.7	9.0	0.1	133.6	16.2	12.6	0.8	41.0	22.9
Km88	DDZ	2013	17.4	0.4	8.4	0.1	331.8	28.2	9.6	0.2	114.6	57.2
		2014	19.0	1.3	8.1	0.1	353.1	9.9	8.8	0.8	114.4	52.4
	n = 5	2015	18.6	0.7	8.5	0.0	312.0	6.5	10.7	0.4	115.0	39.7
	REF	2013	14.3	0.6	8.1	0.0	308.6	4.9	7.0	0.2	120.2	54.8
		2014	19.6	0.1	8.1	0.1	371.1	6.2	7.6	0.5	104.6	55.2
	n = 5	2015	17.5	0.4	8.3	0.1	308.8	2.3	9.4	1.3	32.4	40.4
Sprague	DDZ	2013	26.6	0.1	6.4	0.0	39.3	0.6	4.5	0.3	40.5	-
		2014	24.3	1.6	6.2	0.1	53.2	1.2	3.0	0.3	92.0	12.7
	n = 2	2015	26.5	0.6	6.3	0.3	37.1	4.4	5.3	1.8	93.3	42.5
	REF	2013	23.6	1.3	6.2	0.2	29.0	3.7	3.7	1.5	141.2	50.0
		2014	24.4	1.4	6.0	0.1	31.3	6.1	3.3	0.6	149.1	40.6
	n = 10	2015	25.7	0.6	6.1	0.2	29.5	1.7	3.4	0.7	165.0	37.5
Valemount Peatland	DDZ	2013	19.2	0.3	7.7	0.1	109.7	3.6	8.5	0.3	80.2	31.9
		2014	21.5	1.5	8.1	0.3	112.7	9.3	8.4	0.2	74.6	34.2
	n = 8	2015	26.4	1.7	8.6	0.7	129.8	19.1	10.3	1.7	70.4	36.1
	REF	2013	9.7	0.3	7.1	0.0	43.5	7.4	8.2	0.4	85.7	14.6
		2014	12.5	0.4	7.1	0.1	53.4	0.5	7.4	0.4	115.8	31.0
	n = 5	2015	17.8	0.3	6.8	0.1	59.8	1.1	4.3	0.2	125.4	25.5

pH

pH values varied across the study area and were highest in the Bush River ponds (pH 9.3 ± 0.6), lowest at Sprague Bay, (pH 6.1 ± 0.1) and slightly basic at the Valemount Peatland (7.7 ± 0.2). pH was also higher in the Valemount Peatland DDZ pond than in the paired REF pond and there was a significant Year*Position interaction ($p < 0.001$), indicating an increasing trend over time. pH values in the Valemount Peatland DDZ pond increased from pH 7.7 in 2013 to pH 8.6 in 2015. In contrast, pH in the Valemount Peatland REF pond decreased from pH 7.1 to pH 6.8, which is within the bounds of sampling error. The increase in pH in the DDZ pond may be a result of several factors such as changes in the nutrient cycle, increased respiration and decomposition, dust and sand particles, or recent inundation from the reservoir, which is slightly alkaline pH (Bray 2016).

Conductivity

With the exception of the Bush River DDZ pond, conductivity in ponds was generally consistent over time. The lower value observed Bush River DDZ pond in 2015 was likely due to heavy rainfall that occurred prior to the sampling (July 17, 2015, DOY = 198. The other ponds were sampled on July 13 or earlier and were not significantly affected by the weather system than dominated the late half of July 2015 (DOY 192 to 210; July 11 to July).

Dissolved Oxygen

As in previous years, mean DO concentrations were highest in the Bush River ponds (12.2 mg l^{-1} ± 1.6) and lowest at Sprague Bay (3.9 mg l^{-1} ± 1.7). Moderate DO values were observed in the ponds at Km88 (8.8 mg l^{-1} ± 1.6) and in the Valemount Peatland (7.8 mg l^{-1} ± 1.6). DO concentrations are influenced temperature, barometric pressure, wind turbulence, wave action, photosynthesis, light availability, and time of day which likely account for difference in individual ponds among years.

Water depth

Changes in water depth in the Sprague Bay Pond DDZ (2013) and the Km88 REF pond (2015) were a due to breached beaver dams (Table 5-4). In 2013, a beaver dam retaining the water in the Sprague Bay Pond DDZ collapsed and the pond drained almost completely; in 2014, an alternate pond was found at a slightly higher elevation. A similar situation happened in 2015 at Km88 REF pond. Investigation of the Km88 REF pond prior to sampling in 2016 will be required to determine if it is still a suitable reference pond. The increase in water depth in the Valemount Peatland REF pond in 2014 and 2015 was due to an industrious beaver.

Comparisons between DDZ and REF ponds

Two sample t-tests indicated that pH, temperature, dissolved oxygen, and conductivity were generally higher in the DDZ ponds than in the adjacent REF ponds with the exception of Km88 where temperature and conductivity values did not differ. This indicate that the position within the reservoir may be influencing the water chemistry. For example, prolonged inundation from the reservoir in recent years (2011 to 2014) may explain the higher pH and conductivity values in the DDZ ponds at Sprague Bay and the Valemount Peatland.

5.5.2 Macrophyte Biomass

Mean dry weight of biomass samples collected from 2013 to 2015 ranged from trace amounts of 0.01 to 142.7 grams (Table 9-2; Figure 5-11). LME modeling indicated that macrophyte abundance was influenced by location (index site), position, and year (Table 9-3). Site was the most significant effect term underscoring the geographical differences across the index sites. The greatest amount of biomass came from ponds in Bush River and Km88 (Table 9-2) where mean biomass was over 800 per cent greater than in Sprague Bay or the Valemount Peatland.

Macrophyte biomass was higher in DDZ ponds than in REF ponds in almost all paired samples; exceptions were Km88 in 2013 and Sprague Bay in 2015. In general, mean biomass values were over three times greater in DDZ ponds than in REF ponds (43.6 g ±16.7 versus 12.63g ± 16.7). Macrophyte biomass increased between 2012 and 2015; however, the interaction between Position*Year was not significant (Table 9-3) indicating that the rate of increase in ponds did not differ significantly between DDZ and REF ponds. The profile plot in Figure 5-11 displays the increasing trend in macrophyte abundance in both REF and DDZ ponds.

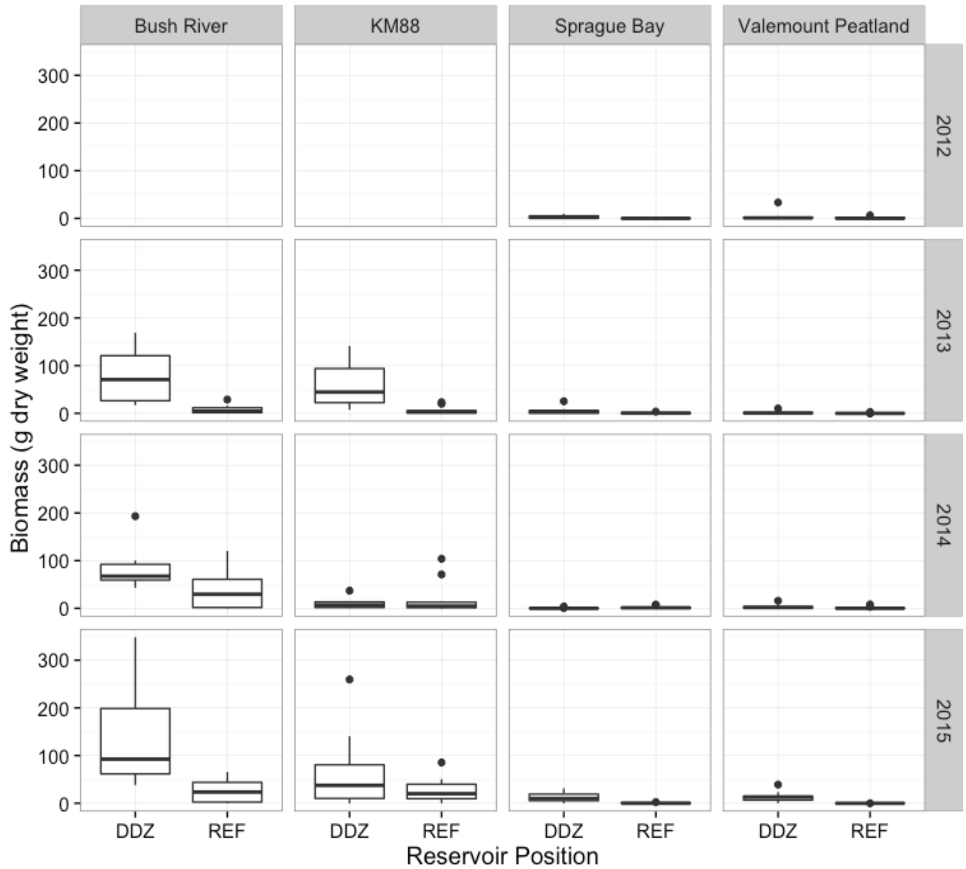

Figure 5-11: Box plots of aquatic macrophyte biomass (grams) from paired ponds at four index sites in Kinbasket Reservoir, 2015. Pond where either within the reservoir (DDZ) and above the reservoir (REF.

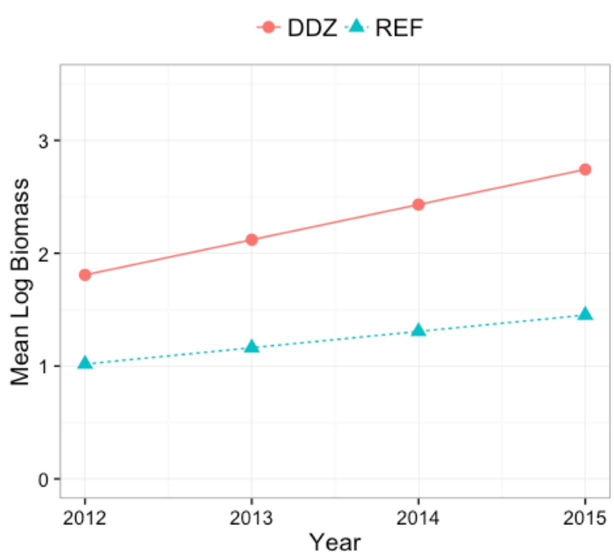

Figure 5-12: Profile (interaction) plot of the mean log biomass (g) by year and reservoir position, all sites pooled. The parallel lines indicate that the increase in macrophyte biomass in DDZ and REF ponds over time was similar (not significant).

5.5.3 Pond metabolism

Dissolved oxygen

Mean daily DO values obtained from data loggers (Table 5-5) were consistent with the point sample measurements (Table 5-4). Mean daily DO ranged from 0.5 to 12.4 mg l^{-1} in the eight study ponds. Dissolved oxygen values as high as 21.5 mg l^{-1} were observed (Bush River DDZ pond); DO values below 1 mg l^{-1} were observed in most ponds in 2014 or 2015 with the exception of the Bush River and Km88 REF ponds (minima were 4.4 and 3.1 mg l^{-1}, respectively).

Table 5-5: Mean daily DO, GPP, R, and NEP (constrained) with standard deviations (SD) from paired DDZ and REF ponds in Kinbasket Reservoir. Dates were from June 21 to August 31, 2014 and 2015. Blue bold text denotes ponds exhibiting autotrophy (positive NEP); red italicized text denotes ponds exhibiting heterotrophy (negative NEP).

Site	Position	Year	n days	DO (mg l^{-1}) Mean	SD	GPP (O_2 mg l^{-1} day^{-1}) Mean	SD	R (O_2 mg l^{-1} day^{-1}) Mean	SD	NEP O_2 mg l^{-1} day^{-1}) Mean	SD
Bush River	DDZ	2014	60	10.3	4.0	8.6	5.7	-7.8	5.5	**0.8**	**3.1**
		2015	58	9.9	4.2	5.0	2.2	-3.6	2.2	**1.4**	**2.0**
	REF	2014	11	12.4	1.6	1.7	0.7	-1.5	1.4	**0.2**	**1.4**
		2015	53	11.9	1.7	5.3	2.4	-4.2	2.6	**1.1**	**1.2**
Km88	DDZ	2014	60	8.5	2.0	4.0	2.8	-4.5	3.0	*-0.4*	*1.6*
		2015	72	9.6	1.7	7.7	3.2	-7.3	3.2	**0.4**	**1.0**
	REF	2014	57	8.2	1.3	2.7	2.6	-3.2	2.5	*-0.5*	*0.9*
		2015	65	8.7	0.6	1.3	1.4	-1.6	1.8	*-0.3*	*0.7*
Sprague Bay	DDZ	2014	19	0.5	0.7	0.8	1.2	-4.8	1.1	*-4.0*	*1.0*
		2015	43	2.6	1.2	0.8	0.6	-3.7	0.8	*-2.9*	*0.7*
	REF	2014	39	1.5	1.1	0.6	0.5	-4.2	0.9	*-3.6*	*0.8*
		2015	41	4.4	1.4	0.8	0.5	-2.6	0.9	*-1.8*	*0.7*
Valemount Peatland	DDZ	2014	46	9.2	3.4	2.8	2.3	-2.9	1.9	*-0.1*	*2.1*
		2015	43	10.2	1.7	2.9	2.6	-2.4	2.2	**0.5**	**1.5**
	REF	2014	32	2.9	3.2	1.8	1.8	-4.6	1.7	*-2.8*	*1.6*
		2015	38	1.5	2.2	1.5	1.1	-5.3	1.5	*-3.8*	*1.4*

Metabolic estimates

Daily GPP, R, and NEP ranged from 0.01 to 25.8, -24.1 to -0.0, and -8.7 to 6.0 O_2 mg l^{-1} day^{-1} (respectively) in the eight study ponds over the June through August sampling windows in 2014 and 2015 (Figure 5-13). GPP and NEP rates were highest in ponds in Bush Arm (Bush River and Km88) and the lowest in ponds at Sprague Bay. These ponds tended to have the greatest and least variability GPP, R, and NEP, respectively. With the exception of the Sprague ponds, GPP and NEP were higher in DDZ ponds than in paired REF ponds. At Sprague Bay, GPP, R, and NEP were similar between the DDZ and REF ponds.

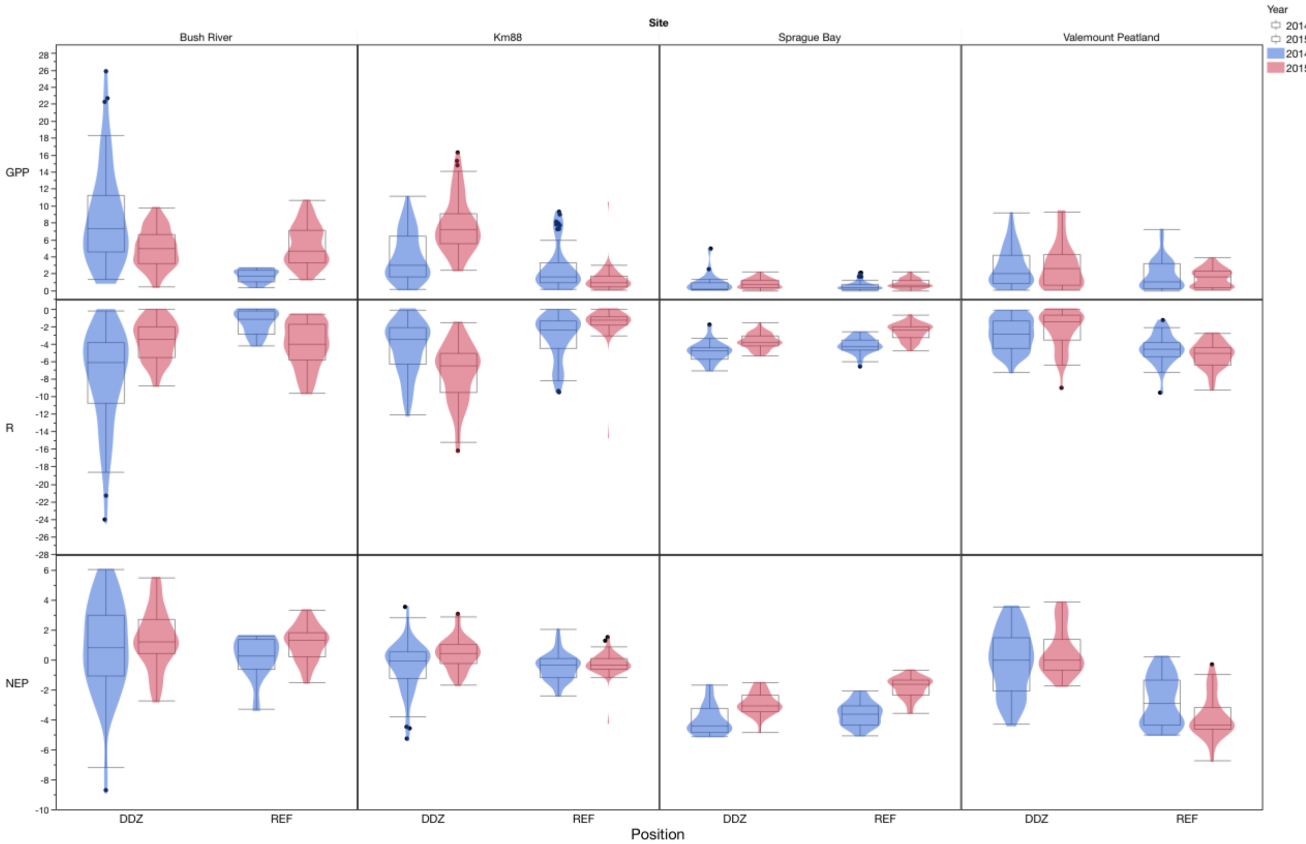

Figure 5-13: Box plots overlain on violin plots showing the variability of daily GPP, R, and NEP (O2 mg l-1 day-1) in drawdown zone and reference ponds at the four index sites over the period June 21 to August 31 in 2014 and 2015.

Significant results for the interaction of Position*Site*Year in LME models for GPP, R, and NEP indicates that differences in the metabolic variables between 2014 and 2015 were influenced by both Site and Position (Table 9-4). In treating Site as an random factor (Table 9-4), the term Year and interaction between Year *Position for GPP were significant. One way ANOVA's indicated that GPP in three of the four DDZ pond differed significantly from 2014 and 2015 as did the Bush River Ref Pond ($p < 0.001$). GPP in the Km88 and Valemount Peatland DDZ pond both increased ($p < 0.001$ and $p = 0.01$, respectively) but decreased in the Bush River DDZ pond ($p < 0.001$). No terms were significant for R or NEP in the second LME model.

Results from LME models (with Site and Pond treated as random factors) for metabolic variables indicated that responses to inundation were influenced by position and year (Table 9-5). One-way ANOVA's of logGPP, log (R + 1), and NEP indicated significant differences in metabolic variables before (July) and after (August) inundation (Table 9-6). Changes in GPP and NEP in Bush River and Km88 DDZ ponds were not observed in the paired REF ponds suggesting a possible inundation effect in these ponds; however, no differences were observed in the Sprague Bay and similar responses (an increase in R and decrease in NEP) were observed in the Valemount Peatland DDZ and REF ponds.

Both GPP and NEP corresponded positively with macrophyte biomass ($R^2 = 0.45$, $p = 0.005$ and $R^2 = 0.40$, $p = 0.008$, respectively; Figure 5-15). The Bush River and Km88 REF and DDZ ponds typically had the highest macrophyte biomass (Table 9-2) and highest rates of GPP (Table 5-5). NEP in these ponds tended toward autotrophy and had rates near or above 0.0 O_2 mg l^{-1} day^{-1}. The Valemount Peatland REF pond and the Sprague Bay ponds had among the lowest macrophyte biomass and lowest rates of GPP. These ponds were heterotrophic and were characterized with low DO concentrations and NEP rates below 0.0 O_2 mg l^{-1} day^{-1} (Table 5-5).

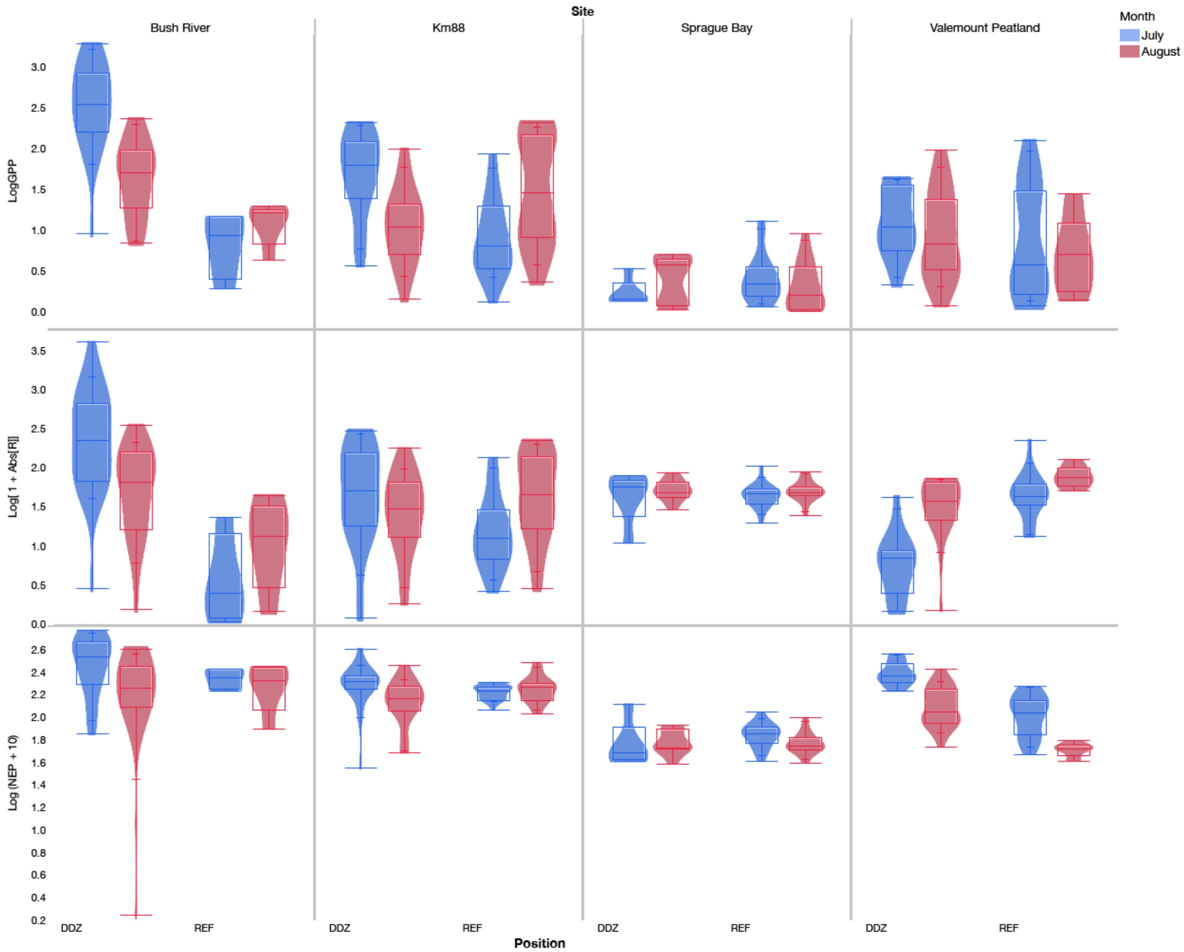

Figure 5-14: Box plots overlain on violin plots showing GPP, R, and NEP (O2 mg l-1 day-1) in drawdown zone (DDZ) and reference (REF) ponds prior to inundation (July) and during inundation (August) in 2014.

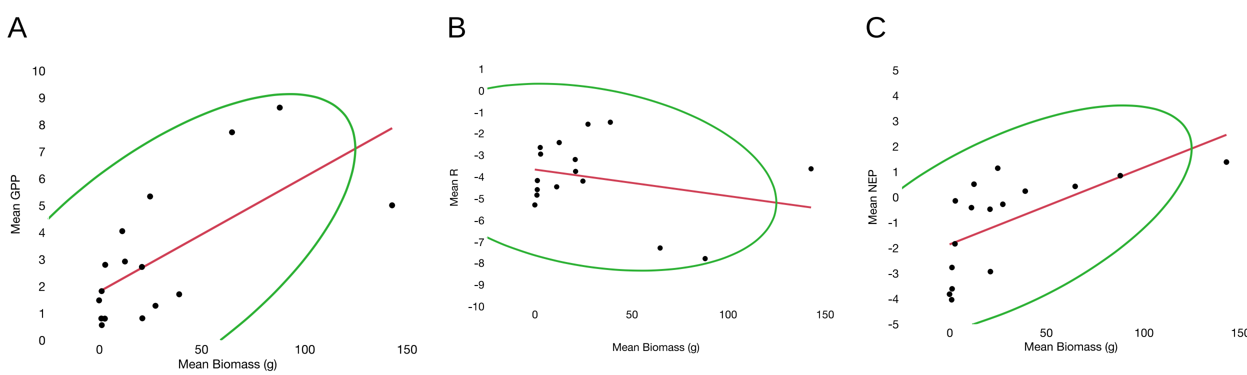

Figure 5-15: Scatter plots macrophyte biomass (g) against (a) GPP, (b) R, (C) NEP (mg l-1 day-1) with 95% density ellipse. R^2_{GPP}= 0.45, p < 0.005 ; R^2_R = 0.07, p = 0.31; R^2_{NEP}= 0.40, p < 0.008.

6.0 DISCUSSION

CLBMON61 is a Before-After-Control-Impact study (BACI) designed to assess the impacts of water level changes associated with the operation on Mica 5 and 6 on wetlands in Kinbasket Reservoir. The before-impact period for the BACI design commenced in 2012 and extended to 2014. The after-impact period began in 2015 and will include 2016 and 2017.

Previous reports were provided for data collected in 2012 and 2013 (Adama et al 2013, 2014) and this report summarizes the results of data collected in 2014 and 2015 – the last before-impact year and the first after-impact year. As in 2012 and 2013, reservoir levels in 2014 were higher than average and the 753-754 m elevation band was inundated for 99 days, six-times greater than the historical average. Data collected in 2015 was the first after-impact monitoring year; however, reservoir levels did not exceed 751 m. Consequently, the operation of Mica Unit 5 did not impact wetlands in the 753 m to 754 m elevation band in the first year of operation. As such, the following discussion summarizes the general trends and observations from the wetland data collected to date. Where possible we have endeavored to examine the effects that high reservoir levels from previous year may have had on wetlands to provide insight into the potential impacts of Mica 5 and 6.

6.1 Terrestrial Wetlands

6.1.1 Terrestrial Wetland Communities

In 2013, community descriptions for terrestrial wetlands were provided across the 752 to 755 m elevation band and in an adjacent reference wetland for each index site (Adama et al 2014). Unfortunately, these descriptions were developed using an inaccurate digital elevation model that resulted in positioning two-thirds of the CLBMON61 transects in the wrong elevation band. Consequently, almost half of the community descriptions for each elevation were incorrect. Using vegetation data from 2015, the community classifications and descriptions were revised to reflect the new digital elevation model and the new data collected (Section 5.4.1; Appendix C Section 9.3).

Using vegetation data collected in 2014 and 2015, eighteen different wetland associations were described that occur across the elevation gradient of the four index sites. Most of these communities were keyed using MacKenzie and Moran (2004); however, several communities from Hawkes et al (2007) also applied including the Swamp Horsetail, Driftwood, Willow Shrub, Reed Canary Grass, and Buck Bean Sedge. In general, the patterns described in Adama et al (2013, 2014) of increasing complexity from sparsely vegetated associations at lower elevations to more diverse shrub –herb associations in the upper elevations were supported. In correcting the transect elevations and with the addition of new transects, we gained a better understanding of how the communities progress across the elevation gradient.

The five terrestrial vegetation community progressions described at the four index sites are summarized as follows:

1. Valemount Peatland: Swamp Horsetail Marsh (SH/Wm02) below 753m to a Scrub Birch dominated Fen (Wf07/Wf02) from 753 to 754 to a developing Black Spruce Peat bog (Wb05) above the reservoir.
2. Sprague Bay: Reed Canarygrass/Water Sedge (RC/Wm02) below 753m to Pink Spirea–Sitka sedge (Ws50) from 753 –754m and shrub communities between 752

–754m to a developing Scheuchzeria-Peat-moss bog (Wb12).

3. Km88: Buckbean Fen at 752m to scrub birch communities (Wf07 and Wf02) between 753 to 755m to a Hudson Bay Clubrush –Red hook-moss (Wf10) above the reservoir.
4. Bush River
   a. Willow flood community at 752m to a riparian shrub community above 753m and to upland forest above the reservoir.
   b. Water sedge –Beaked sedge Fen (Wm01) at 752m to a Slender sedge – Common hook-moss (Wf05) above 754m.

At most sites, wetlands progressed from marsh to fens and then to either bogs (i.e. Valemount peatland and at Sprague Bay), swamps (i.e. Km88), or riparian flood communities (i.e. Bush River) and the nuances of each successional pathway were influenced by site characteristics including hydrology, geology, topography, soil, organic accumulations, nutrient availability, climate, and disturbance regime (inundation from the reservoir, riverine flooding [e.g., Bush River], and beaver activity [e.g., Valemount Peatland, Km88, and Sprague Bay]). These communities will be reassessed for changes in the final study year.

6.1.2 Terrestrial Vegetation Cover

The analyses of transect and circular plot data at the four index sites indicated that total herb cover and total shrub cover within the various elevation bands did not differ from 2013 to 2015. This is not unexpected as the reservoir has operated at high levels since at least 2011 and wetland vegetation below 754 m is likely at a state of equilibrium and dominated by flood tolerant species and annuals. Hawkes and Gibeau (2015) reported that the spatial extent of vegetation communities in the Kinbasket Reservoir in 2014 was largely unchanged from 2010 but differed significantly from data collected in 2007. They further reported a die-off of shrubs and a marked decrease in species richness and diversity since 2007, which they attribute to high water events in 2007. Species intolerant of prolonged periods of inundation and soil anoxia were likely eliminated prior to the start of CLBMON-61 while species adapted to permanently saturated soils (e.g. *Carex aquatilis, Equisetum fluviatile, Menyanthes trifoliate, Calamagrostis canadensis and Utricularia spp.)* continue to thrive. Consequently, it may be difficult to detect any further changes in wetland vegetation abundance except in years of lower reservoir levels when flood intolerant species can establish. Because of the low reservoir levels observed in 2015, it is anticipated that both vegetation abundance and species diversity will increase in the upper elevation bands of the reservoir in 2016.

6.2 Aquatic Wetlands

6.2.1 Macrophyte Biomass

Differences in macrophyte biomass across index sites were likely a function of mineral availability in ground and surface water. Dissolved minerals from the surrounding geology provide an important source of nutrients such as calcium, which can greatly influence the composition macrophyte communities and overall macrophyte abundance. For example, the macroalgae *Chara* spp., which is an obligate calciphile, occurred only in the Km88 and Bush River ponds and comprised up to 100 per cent of the samples. The surrounding geology of the Bush Arm, where the ponds occur, is rich in calcium carbonate and accounts for their higher conductivity and pH values (Wittneben and Lacelle 1986; Adama et al 2012).

Differences in macrophyte biomass between DDZ and REF ponds were likely due warmer water temperatures in the DDZ ponds although nutrient and light availability may also be important. DDZ ponds were up to 9.5 °C warmer than paired REF ponds. Warmer temperatures also account for increase in macrophyte growth observed in both DDZ and REF ponds from 2012 to 2015. During this period, mean daily temperatures in the study area increased from 0.67 to 1.05 °C annually, Temperature is a key factor that determines the growth of aquatic and warmer temperatures facilitate faster colonization, deeper colonization, faster growth rates, and higher macrophyte biomass (Rojo et al 2015; Lacoul and Freedman 2006; Mckee et al 2002.; Rooney and Kalff 2000; Barko et al 1982).

In Adama et al (2014), we predicted that an increase in the frequency and duration of inundation associated with operation of Mica 5 and 6 may negatively affect macrophyte biomass various mechanism including reduced growing season, reduced light availability, and increased wave action. However, despite successive years of prolonged inundation of ponds, macrophyte biomass increased in DDZ ponds, which we attribute to an increase early season temperatures. These results suggest that impacts on macrophyte growth from higher reservoir levels may be negligible or short-lived although we have not yet assessed changes in species composition.

6.3 Pond Metabolism

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 and long-term changes in ecosystem metabolism in wetlands can affect the ecological services they provide (Allen and Castillo 2007; Mitsch and Gosselink 2007; van der Valk 2012). Disturbances that result in changes to hydrology or to the nutrient regime can result in trophic changes that affect water quality, fish and wildlife habitat, and the sequestration and storage of carbon (Mitsch and Gosselink 2007).

NEP can be used to infer the trophic nature of ecosystems where 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 (Odum 1956). Autotrophy (positive NEP) was observed in the Bush River ponds and in the DDZ ponds of Km88 and the Valemount Peatland. Wetlands with high rates of GPP (i.e. Bush River and Km88 Ponds) were characterized macrophyte biomass, which occurs under high light and nutrient regimes. Conversely, heavily shaded and dystrophic wetlands had low GPP, low macrophyte abundance, and low DO, characteristic of the tannin rich ponds of Sprague Bay.

The range of values obtained for GPP, R, and NEP in 2014 and 2015 were comparable with the values reported for wetlands elsewhere (Español et al 2013; Hoellin et al 2013; Reeder 2011; Lauster et al 2006; McKenna 2003). Metabolic values were also within the range of observed in 2013 (Adama et al 2014); however, in 2013, pond metabolism was not corrected for atmospheric diffusion resulting in over-estimating GPP and R (disproportionally) and underestimating NEP. GPP and NEP were higher in DDZ ponds than the REF ponds with the exception of Sprague Bay. Temperature again is likely the driving factor; however, light and nutrient availability may also be important.

In 2014, inundation of ponds by the Kinbasket Reservoir resulted in an immediate decline in GPP and NEP rates in both autotrophic and mildly heterotrophic ponds. These results were consistent with our observations in 2013 and with those reported in the literature. Ballantyne et al (2013) reported a decrease in both GPP and R in peat dominated wetlands following a 10 cm increase water levels and Cooper et al (2013) reported that R and GPP tended to be lower in wetlands with increased hydrologic exposure (wave action, water depth, and current). The metabolic variables in the Valemount Peatland and Sprague Bay responded differently to inundation than the Bush River and Km88, which was also consistent with our observations in 2013. These ponds were either oligotrophic or dystrophic and have deep organic sediments, which may account for the differential response (Lauster et al 2006).

These results suggest that increases in the frequency, magnitude, and duration of inundation as result of operational changes associated with Mica 5 and 6 will likely result in a decrease in aquatic GPP and NEP, although the impacts will likely vary based on the trophic state of the wetland and local water physicochemistry.

7.0 CONCLUSION and RECOMMENDATIONS

7.1 Conclusion

This report largely focused on the potential impacts of reservoir levels on vegetation abundance and wetland productivity. In terrestrial wetlands, neither herb nor shrub cover differed within elevation bands over the study period. These results were not unexpected as Kinbasket reservoir has been operated at consistently high levels since at least 2011 and it is likely that vegetation below 754 m was already at a state of equilibrium prior to the start of the study.

In contrast, increases in macrophyte biomass in both DDZ and REF ponds from 2012 and 2015 were unexpected. Previously, we had predicted that frequent and prolonged inundation would likely suppress macrophyte growth; however, these effects, if they occur, where overwhelmed by a concurrent increase in temperature in May and June that likely promoted macrophyte growth and biomass accumulation.

The short-term effects of inundation on aquatic metabolism were explored by comparing pre- and post-inundation periods in 2014. Our results indicate that increases in flooding may result in a decrease in primary productivity (GPP and NEP); however, longer term impacts have not been examined.

In future years, we will focus more on species composition in both terrestrial and aquatic wetlands.

7.2 Recommendations

For redundancy, we recommend deploying two DO loggers in each pond. Currently, only a single DO logger is deployed in each pond and the loss of data from a single DO unit failing is significant. Installing a second DO logger in each pond will also reduce the noise in the data (negative GPP and positive R values).

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9.0 APPENDIX

9.1 APPENDIX A: 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 2m 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; **DDZ**) or outside/above (> 754.4 m; **REF**).

Target Site/Target Wetland –wetlands or sites within the 753 to 754m elevation band (Figure 9-1).

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

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

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

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

Index Site (Site) –Sites that were identified in Year 1 that:

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.

Wetland Type:

Table 9-1: 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)**

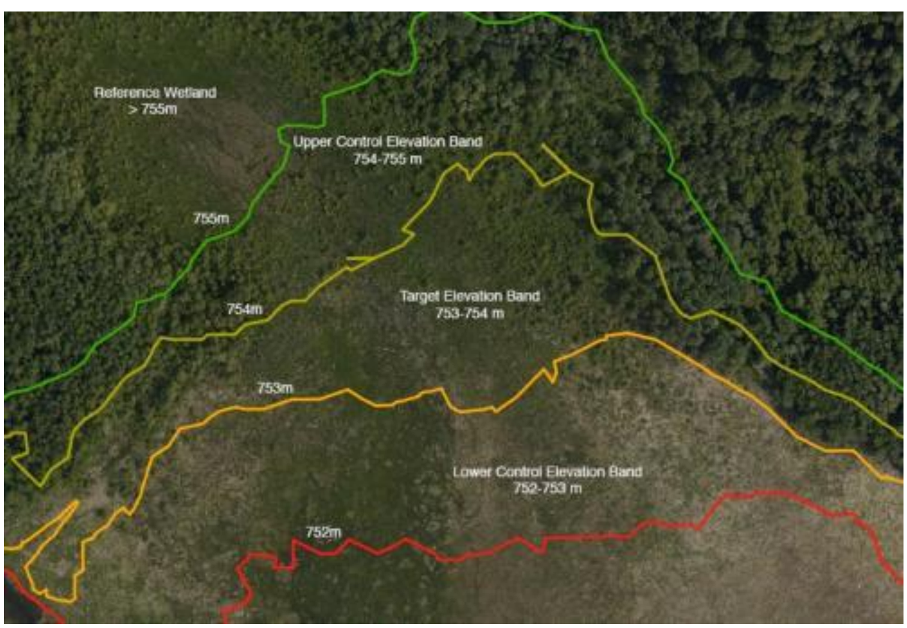


Figure 9-1: Sampling strata of an index monitoring site showing target, control, and reference elevation bands

Vegetation Community/association –plant assemblages characterized by similar species composition and 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 historical 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 cover, 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 impact and control sites and temporal replication with measurements before and after an impact application or impact. Under CLBMON-61, "target" sites can be thought of as "impact sites" for the purposed of the BACI study design.

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

Before impact –the time period prior to when 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.

9.2 Appendix B. Work completed in previous years

9.2.1 Work completed in 2012

CLBMON61 began 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 755m elevations with 34.1 hectares occurring in the target elevation band (753–754m). 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).

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 755m and at reference sites above the reservoir (> 755m). 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 755m. 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 –753m).

- 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 75of 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.

9.2.2 Work completed in 2013

In 2013, a second year of data were collected for CLBMON-61. The objectives for Year 2 were to:

a. summarize the state of the wetland index sites identified in Year 1;

b. provide preliminary insight into the expected changes to vegetation composition or wetland productivity.

c. assess 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, 753 to 754 (target) m, 754 to 755m (upper control), and reference wetlands outside the reservoir above 755m. 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 753m and reference ponds between 756 and 780 m. 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 753m elevation band. Where possible, results were compared against 2012 data.

Key findings from Year 2:

- Similar to 2012, reservoir levels greatly exceeded the 1987 to 2006 historical operating regime, and the 753m 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 (752m) to more diverse shrub-herb communities in the upper elevations (754m 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.

o Shrub cover increased by 200with each 1–metre elevation band from 752 to 754m

o Shrub species richness also increased by over 150across each elevation band from 10 species at 752 to 35 species at 754m and diversity measures (Shannon and Simpson) differed significantly ($\alpha = 0.10$) across the elevation gradient.

o 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 753m 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 753m 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 in an attempt 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 (Hawkes et al 2012).

- Diel changes in dissolved oxygen were used to estimate Net Ecosystem Production (NEP), Gross Primary Production (GPP), and Ecosystem Restoration (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 data logger technology permit the calculation of reliable metabolic rates (NEP, GPP, and R) from diel fluctuations in dissolved oxygen. Dissolved oxygen data loggers 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:

i. 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.

ii. Prolonged inundation (87 days) of the 753m 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.

iii. 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.

9.2.3 Work Completed in 2014

The focus of work in Year 3 (2014) was to collect an additional year of baseline data following methods established in Year 1 and 2 prior to Mica turbine 5 coming online. High reservoir levels limited the sampling in 2012 resulting in an incomplete data set for Year 1 and there was a high risk that prolonged inundation of the 753m elevation band would confound the results in 2015, the next planned year for data collection. Neither data analyses nor an annual report were requested for Year 3 (2014). Sampling followed the methodology established in 2012 and 2013 with the following exceptions 1) the collection of pelagic invertebrate was not collected and 2) additional instrumentation was deployed (wind, temperature, PAR, barometric, and water depth) to acquire atmospheric data to estimate the diffusion of oxygen into the atmosphere for metabolic rate (NEP, GPP, and R) calculations.

Key findings from Year 3:

- For the third consecutive year, reservoir levels exceeded forecast levels resulting in prolonged inundation of upper elevation bands. During 2014, the 752 and 753m elevation bands were inundated for 122 and 100 days. For the before impact baseline period (2012 to 2014), the 752, 753, and 754m elevation bands have been inundated for an average 103.3, 81.7, and 28 days per year. This greatly exceeds the number of days inundated historically (1987 to 2006): 26.7, 17.5 and 6.9 days, respectively as well as elevations predicted from the GOM model with Units 5 and 6 operational: 30.2, 19.1, and 10.5 days, respectively. Hence the before impact period is not representative of the historical baseline and it will not be possible to adhere to BACI study prescribed in the Terms of Reference.

- In 2014, a new DEM was produced by LGL Limited (Hawkes et al, 2015) for key areas of interest in Kinbasket using LIDAR (Laser Illuminated Detection And Ranging) data. LIDAR provides highly accurate elevation data in comparison to most other methods. Comparing the elevation of sample plots from 2014 (n =150), the old DEM and the LIDAR data differed by a mean of 77 cm with some areas differing by up to 180 cm. Differences were statistically significant ($t < 0.001$) and were largest in the Valemount Peatland (μ =121 cm) and the least at Km88 (μ = 21 cm). Sprague Bay and the Bush River differed by 40 and 49 cm respectively. As a consequence of the new elevation data, a number of transects and circular plots are not positioned in the correct elevation band. Of the 40 transects established and monitored in 2013 and 2014, only fourteen are located in the correct elevation. In order to comply with the balanced study design, eleven of the transects need to be reclassified into the proper elevation band and five require minor adjustment to their position to ensure the entire transect lays in correct elevation band. Ten new transects need to be established at elevations bands that are not being sampled. Despite this somewhat dire assessment, the budgetary impacts required for establishing and monitoring the new transects in future years is anticipated to be moderately low. If these changes are not incorporated into the study design, the power of the study design and statistical analyses will be reduced.

- The new elevation data also puts into question some of the results and conclusions made in previous years. Fortunately, this applies only to the terrestrial wetland data and not to the aquatic data, as the aquatic sampling was stratified simply as either within (DDZ) or above the reservoir (REF). All results and conclusion regarding the classification of the terrestrial wetlands, composition and abundance of herbaceous and shrubby species, and the distribution of woody debris must now be considered as unreliable. A revised copy of the 2013 report was provided with sections struck out that are no longer valid. To address this, we recommended that either the 2013 report be updated or a new report be prepared incorporating the new elevation data along with the additional year of data collected in 2014.

9.3 APPENDIX C: Updated Vegetation Associations

The descriptions of the terrestrial communities were updated from data obtained in 2015 using the 2014 digital elevation model to delineate the elevation bands. Cover values are provided for substrate composition and dominate vegetation. Description of the aquatic wetlands and macrophyte communities can be found in Adama et al (2014) as position of the aquatic wetlands were not affected by the 2014 DEM.

9.3.1 Valemount Peatland, Canoe Reach

750 –752m:

- Substrate dominated by water (40.1 per cent) accompanied by moss (52 per cent), wood, (6.1 per cent), and fine organic dead matter (3.9 per cent).
- Vegetation cover was low (15.6herb and < 1shrub). Prominent herbs included *Comarum palustre* (3.9 per cent), *Equisetum fluviatile* (1.6 per cent). *Menyanthes trifoliate* (6.0 per cent), *Scirpus microcarpus* (1.2 per cent), and *Utricularia intermedia* (< 1 per cent). *Salix pedicularis* was the most prevalent shrub although *S. planifolia* and *S. prolixa* were all observed; all species occurring at < 1 per cover.
- This community corresponded most closely to the Swamp Horsetail (SH) association of Hawkes et al. (2007) and a depauperate Swamp horsetail –Beaked sedge (Wm02) association (MacKenzie and Moran 2004).

Lower Control (752 –753m):

- Substrate dominated by water (40.1 per cent), wood (22.5 per cent), moss (19.6 per cent), and dead organics (18.3 per cent) such as decomposing peat and organic fines; wood cover was 17.0 per cent.
- Vegetation cover was moderately low (29.3 per cent herb and 1.0 per cent shrub): dominate herbs included *Equisetum fluviatile* (1.3 per cent), *Comarum palustre* (8.9 per cent), and *Menyanthes trifoliate* (6.0 per cent), *Carex aquatilis* (3.2 per cent). *Carex limosa* (1.4 per cent) and *Scirpus microcarpus* (< 1occurred less frequently. *Salix pedicularis* and *S. planifolia* occurred infrequently typically at < 1shrub. However, values in some quadrats *S. pedicularis* cover was 30indicating increasing suitability for *Salix spp.* over lower elevations.
- This community corresponded most closely to the Swamp Horsetail (SH) association of Hawkes et al. (2007) and the Swamp horsetail –Beaked sedge (Wm02) association MacKenzie and Moran (2004).

Target (753 –754m):

- Substrate dominated by wood (67.7 per cent), accompanied by moss (21.9 per cent), water (3.3 per cent), and decaying organic matter (7.1 per cent).
- Vegetation cover was moderate (44.8 per cent herb and 13.15 per cent shrub): dominate herbs include *Equisetum fluviatile* (1.0 per cent), *Comarum palustre* (14.1 per cent), *Menyanthes trifoliate* (16.8 per cent), *Calamagrostis canadensis* (2.3 per cent), *and C. aquatilis* (4.8 per cent).
- *Salix pedicularis* (< 1 per cent), *Myrica gale* (5.0 per cent), *S. planifolia* (1.5 per cent) were the most prevalent shrubs followed by *Alnus incana* (< 1 per cent) and *Betula pumila* (5.4 per cent).
- Classification of the community corresponded most closely to the SH and Driftwood associations of Hawkes et al. (2007). Due to the influence of reservoir did not fit

well into the classification MacKenzie and Moran (2004). It is likely transitional between the Equisetum fluviatile –Carex utriculata (Wm02) and the Scrub birch–Buckbean–Shoresedge (Wf07) associations.

Upper control (754 –755m)

- Substrate dominated by moss (64.5 per cent) accompanied by water (20.2 per cent), and organic dead matter (13.95 per cent). Wood cover was < 1 per cent.
- Vegetation composition and cover of the upper control was distinctly different from the communities within the reservoir. Herb and shrub cover were much higher in reference community (64.0and 44.2 respectively):
- Dominate herbs included *Menyanthes trifoliate* (58.2 per cent), *Carex lasiocarpa* (4.1 per cent), *C. interior* (1.0 per cent), *C. aquatilis* (1.6 per cent) and *Equisetum fluviatile* (< 1 per cent).
- Prominent shrubs included *Picea mariana* (5.9 per cent), *Myrica gale* (15.5 per cent) and *Betula pumila* (21.3 per cent), accompanied by *Rhododendron groenlandicum* (7.6 per cent), *Vaccinium oxycoccos* (< 1 per cent), and *Salix pedicellaris* (1.2 per cent).
- Moss included *Sphagnum angustifolium, S. capillifolium, S. compactum* and *S. subsecundum.*
- The vegetation community fits the board Willow-Sedge (WS) community of Hawkes et al. (2007) and corresponded most closely with the Betula nana –Carex aquatilis (Wf02) association of MacKenzie and Moran (2004).

Reference (755m+)

- The reference transects were positioned immediately above the reservoir within the floodplain. Herb and shrub cover were moderately high (56.5and 59.0respectively). Substrate was dominated by moss (57.5 per cent) and dead organics (41.4 per cent). Wood and water cover were both less than 1.0
- *Menyanthes trifoliate, Comarum palustre, and Calamagrostis Canadensis*, were the dominant herbs (31.8, 7.8, 7.1, respectively), accompanied by *Equisetum fluviatile* (1.4 per cent) and *Carex lasiocarpa,* and *C. interior* both occurring at less than 1.0 per cent. Prominent shrubs included *Alnus incana* (13.4 per cent), *Betula pumila* (22.5 per cent), *Rhododendron groenlandicum* (21.3 per cent), *Salix planifolia* (5.4 per cent), *Vaccinium oxycoccus* (5.9 per cent) and *Salix pedicellaris* (<1.0 per cent).
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely with the Picea mariana –Carex aquatilis –Sphagnum (Wb05). Further assessment of the moss species may aid with the classification.

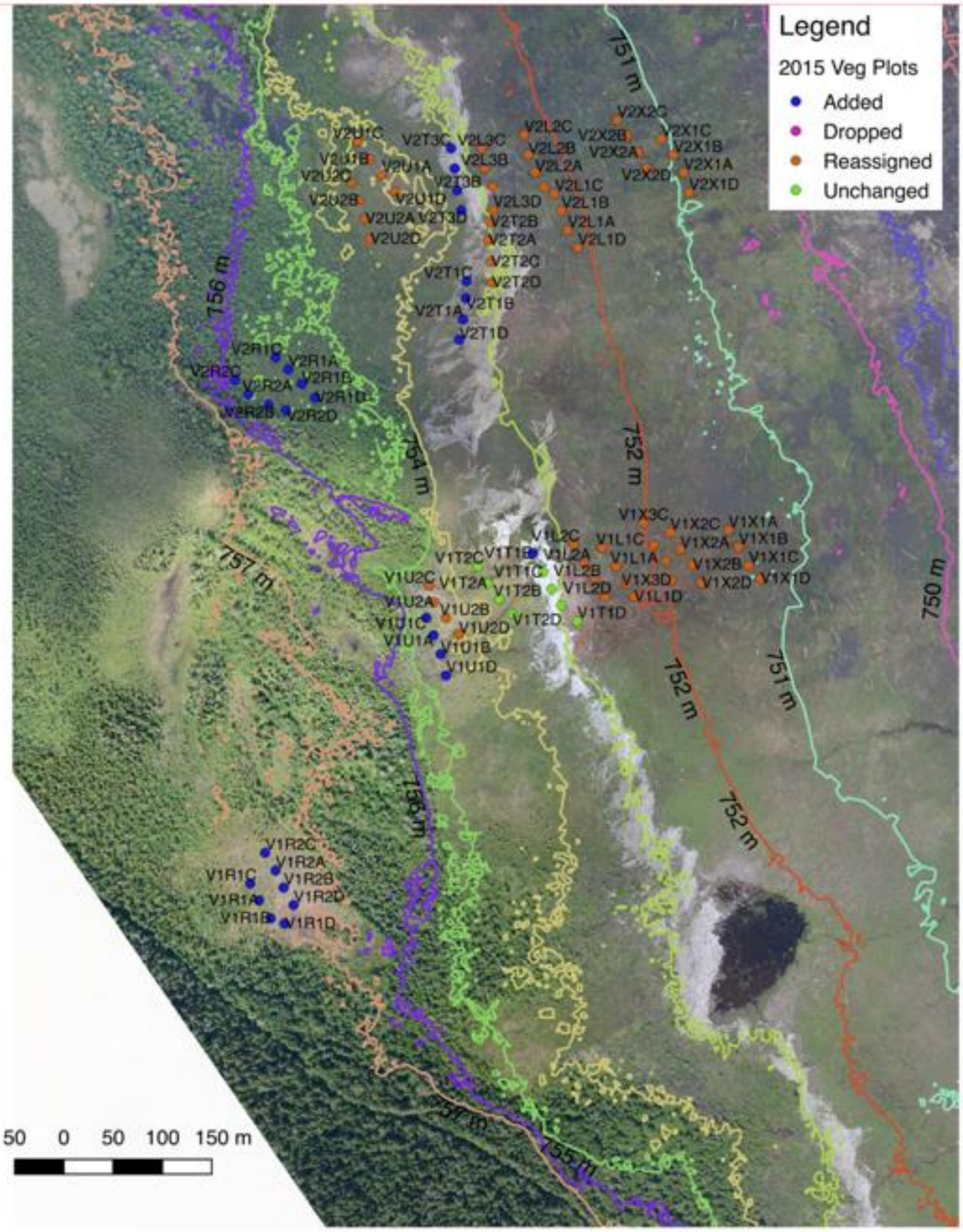


Figure 9-2: Location of terrestrial wetland sampling sites in the Valemount Peatland (Canoe Reach), 2015. Colour indicates whether the point was added, dropped, reassigned, or is unchanged with the 2014 Digital Elevation Model.

Lower Control 752m

Target Elevation Band 753m

Upper Control 754m

Reference Community 755m

Figure 9-3: Representative images of terrestrial wetland transects in the Valemount Peatland.

9.3.2 Sprague Bay Wetlands, Mica Arm

Lower control (752 –753m):

- Vegetation cover in the lower control was moderate (53.1herb cover, shrub cover 5.5 per cent). Substrate was dominated by mineral soils overlain with organic dead matter (52.4 per cent) and wood (45.6 per cent).
- Dominate herbs include *Equisetum fluviatile* (1.7 per cent) and *Calamagrostis Canadensis* (1.3 per cent). *Carex aquatilis* (24.8 per cent), *Comarum palustre* (12.1 per cent) *and Scirpus microcarpus* (2.0 per cent), *Equisetum arvense* (1.0 per cent) occurred in half of the plots. *Spiraea douglasii* was the only shrub observed (5.5 per cent).
- The presence of stumps suggest that this was formerly an upland forest and not a natural wetland. It was most similar to the Reed Canary Grass associations of Hawkes et al (2007) and MacKenzie and Moran (2004). The presence of wetland vegetation was due to local seepage.

Target (753 –754m):

- Vegetation cover in the target elevation band was high (13.2herb and 45shrub): substrate was dominated by dead organic overlain on mineral soil and organic muck (80.5 per cent), accompanied by wood (7.7 per cent) and water (10 per cent).
- *Scirpus microcarpus* (5.4 per cent), *Equisetum palustre* (< 1 per cent), and *Galium triflorum* were the most common herbs (< 1 per cent). *Comarum palustre* (2.1 per cent), *Calamagrostis Canadensis* (< 1 per cent), and *Equisetum fluviatile* (< 1 per cent) occurred in 60of the quadrats. *Spiraea douglasii* was the most prevalent shrub (44.4 per cent) followed by *Alnus incana* at less than 1 per cent.
- The vegetation community did not fit into the classification of Hawkes et al. (2007). Although the transects were within the WS (Willow Shrub) community mapped by Hawkes et al. (2007), the vegetation composition was dissimilar. The vegetation community corresponded most closely to the Pink spiraea-Carex sitchensis (Ws50) association of MacKenzie and Moran (2004).

Upper control (754 –755m)

- Vegetation cover was moderately low (26.3herb and 28shrub): substrate was dominated by moss (88.3 per cent) accompanied by water (7.9 per cent), and organic dead matter (3.2 per cent).
- *Comarum* palustre (8.4 per cent) was the dominant herb accompanied by *Scheuchzeria palustris*, *Equisetum palustre*, *Lycopus americanus, Platanthera dilatata*, *C. lasiocarpa, Trientalis europaea*, *Eriophorum angustifolia*, *Equisetum fluviatile,* and *Drosera rotundifolia*, all less than 1 per cent. *Carex interior* (2.5 per cent), *Platanthera dilatata* (1.8 per cent), *Triglochin maritime* (< 1 per cent), *Carex lasiocarpa* (3.2 per cent), *Calamagrostis Canadensis* (6.4 per cent), *Lysichiton americanus* (3.1 per cent), *Cicuta douglasii* (< 1 per cent), and *Viola fluviatile* (< 1 per cent) occurred in between 25 and 50of the quadrats. Spiraea *douglasii* was the dominant shrub (22.1 per cent) accompanied by *Alnus incana* (4.6 per cent) and *Vaccinium oxycoccos* (< 1 per cent). *Sphagnum spp.* composed much of the ground cover (> 60 per cent).
- 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).

Reference (755m+):

- Vegetation cover was moderately low with little shrub cover (22.5herb and < 1shrub): substrate was dominated by moss (57.8 per cent) accompanied by water (23.4 per cent) overlaying moss, organic dead matter (4.8 per cent). Some wood was present (10.1 per cent) due to forestry and road building activity on the east edge of the wetland.
- *Menyanthes trifoliate* was the dominant herb (14.5 per cent) accompanied by *Carex magellanica* (4.7 per cent), *Scheuchzeria palustris* (2.0 per cent), and *C. utriculata* (1.7 per cent).
- The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely with the Carex limosa –Menyanthes trifoliata – Sphagnum (Wb13) association of MacKenzie and Moran (2004).

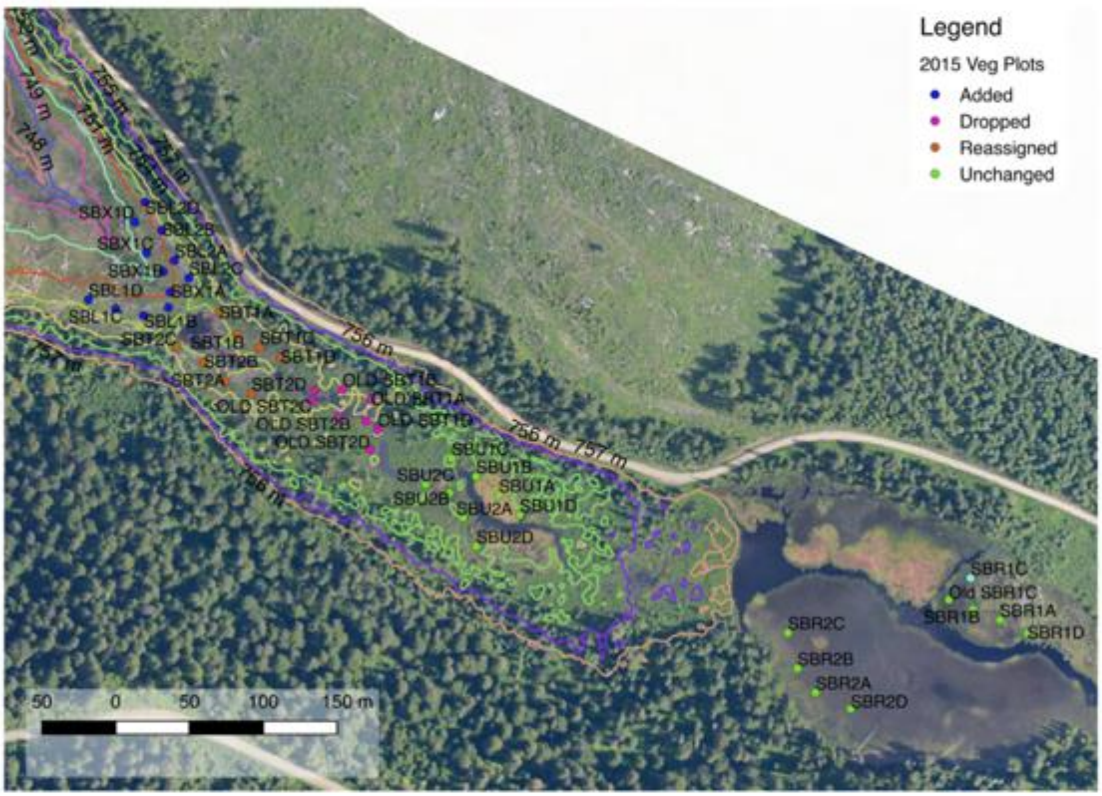


Figure 9-4: Location of terrestrial wetland sampling sites at Sprague Bay, 2015. Colour indicates whether the point was added, dropped, reassigned, or is unchanged with the 2014 Digital Elevation Model.

Lower Control 752m

Target Elevation Band 753m

Upper Control 754m

Reference Community 755m

Figure 9-5: Representative images of the terrestrial wetland transects at Sprague Bay.

9.3.3 Km 88 Wetlands, Bush Arm

Lower control (752 –753m):

- Vegetation in the lower elevation band was high in herb cover (48.2 per cent) and low in shrub cover (1.9 per cent): substrate was dominated by moss (83.7 per cent), accompanied by organics (9.5 per cent), water (4.6 per cent), and wood (2.1 per cent).
- The vegetation cover in this elevation band reflected the hydric site conditions. *Menyanthes trifoliate* (38.5 per cent) was the dominant herb accompanied by *Equisetum palustre* (2.6 per cent), *Carex lasiocarpa* (1.4 per cent), *Utricularia intermedia* (1.6 per cent), *Eleocharis elliptica* (< 1 per cent), *Typha latifolia* (< 1 per cent), *Cicuta douglasii* (<1 per cent), and *Phalaris arundinacea* (< 1 per cent). *Salix pedicellaris* (<1 per cent) was the most prevalent shrub; however, several other *Salix* sp. were also present in low abundance.
- The vegetation community corresponded to the Buckbean–Slender Sedge association of Hawkes et al. (2007) and the Slender Sedge–Buckbean association (Wf06) of MacKenzie and Moran (2004).

Target (753 –754m):

- Vegetation cover in the target elevation band was similar to the lower control band but with more shrub (48.1herb and 12.8shrub). Substrate cover was also similar to the lower control band and was dominated by moss (91.2 per cent), accompanied by wood (5.5 per cent), water (2.1 per cent), and organics (1.2 per cent).
- 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 (38to 90 per cent) accompanied by *Typha latifolia* (2.2 per cent), *Equisetum palustre* (2.5 per cent), *Carex interior* (2.5 per cent), and *Typha latifolia* (2.2 per cent). *Carex aquatalis*, *Equisetum fluviatile*, *Eleocharis elliptica, Juncus nodosus* were also common occurring at less than 1 per cent. *Betula pumila* (9.1 per cent) and *Salix pedicellaris* (1.4 per cent) the most prevalent shrub. Several other species of *Salix* were also observed all at less than 1 per cent.
- The vegetation community corresponded most to the WS association of Hawkes et al. (2007) band the Scrub birch–Buckbean–Shoresedge (Wf07) association of MacKenzie and Moran (2004).

Upper control (754 –755m)

- Vegetation cover in the upper control elevation band was moderately high. Herb cover was similar to the lower control and target elevation bands (46.5 per cent) but shrub cover was much higher (64.6 per cent). Substrate cover was dominated by moss (90.6 per cent) with organic (7.5 per cent) and wood (1.7 per cent).
- *Menyanthes trifoliate* (26.7 per cent), *Petasites frigidus var. sagittatus* (3.2 per cent), *Equisetum palustre* (2.8 per cent), *Maianthemum stellatum* (2.1 per cent), and *Carex interior* (1.6 per cent) were the dominant herbs accompanied by *Maianthemum stellatum* (2.1 per cent), *Lysichiton americanus* (2.7 per cent), *Parnassia palustris* (<1 per cent), and *Zigadenus elegans* (<1 per cent). *Carex flava, Equisetum fluviatile, Symphyotrichum boreale, Petasites frigidus var. palmatus, Eleocharis elliptica* were common occurring at less than 1 per cent. The red listed orchid *Liparis loeselii* was recorded in this elevation band.
- A diverse shrub layer was dominated by *Betula pumila* (29.1 per cent), *Thuja*

plicata (10.5 per cent), and *Alnus incana* (6.0 per cent) accompanied by *Rhododendron groenlandicum* (6.0 per cent), *Picea engelmannii x glauca* (3.0 per cent), *Salix pedicellaris* (3.3 per cent), and *Cornus stolonifera* (< 1 per cent), and

- The vegetation community corresponded to the WS association of Hawkes et al. (2007) and loosely corresponded to the Scrub Birch–Water Sedge (Wf02) and) / Western redcedar –Spruce –Skunk cabbage (Ws10) associations of MacKenzie and Moran (2004).

Reference (755 –757 m):

- Vegetation cover in the reference community (a fen meadow) was high and was dominated by herbs (65 per cent): shrub cover was only 3.0 per cent. Substrate cover was dominated by moss (85.6 per cent) accompanied by dead organic (12.9 per cent) and water (1.2 per cent).
- *Menyanthes trifoliate* was the dominant herb (47.0 per cent) accompanied by *Trichophorum alpinum* (5.3 per cent), *Equisetum palustre* (4.0 per cent), *Eleocharis elliptica* (3.4 per cent), *Cicuta douglasii* (1.8 per cent), *Zigadenus* elegans (<1), *Symphyotrichum boreale* (<1), and *Carex interior* (<1 per cent). *Eriophorum viridicarinatum*, *Trichophorum cespitosum*, *Parnassia palustris*, *Juncus nodosus*, *Mimulus guttatus*, *Deschampsia danthonioides*, *Drosera anglica, Carex flava, Platanthera dilatata, Platanthera aquilonis*, and *Epilobium palustre* were common (25 to 50of the plots) but were less than 1 per cent.
- *Betula pumila* (4.7 per cent), *Abies lasiocarpa* were the most prominent shrubs. *Cornus stolonifera*, *Picea engelmannii x glauca*, and *Thuja plicata* occurred in 25the plots were less than 1 per cent. The vegetation community did not fit into the classification of Hawkes et al. (2007) but corresponded most closely to a low elevation equivalent to a developing it as low elevation equivalent to a developing Tufted clubrush –Star moss (Wf11) association and was bordered by a Western redcedar –Spruce –Skunk cabbage (Ws10) association.

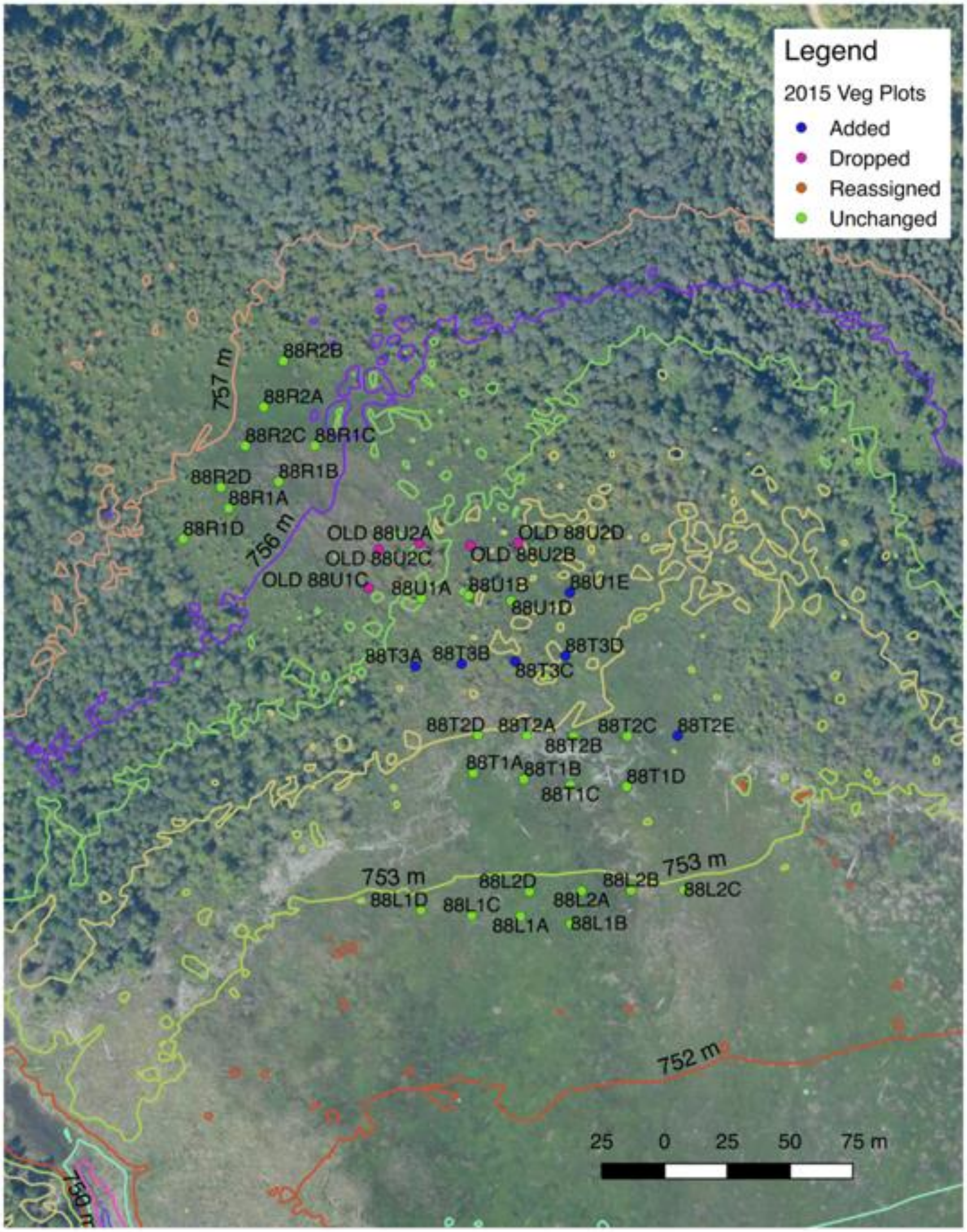


Figure 9-6: Location of terrestrial wetland sampling sites at Km88, 2015. Colour indicates whether the point was added, dropped, reassigned, or is unchanged with the 2014 Digital Elevation Model.

Lower Control 752m

Target Elevation Band 753m

Upper Control 754m

Reference Community 755m

Figure 9-7: Representative images of terrestrial wetland transects at Km 88, 2013

9.3.4 Bush River Wetlands, Bush Arm

Lower control (752 –753m):

- Three different wetland communities occurred at between 752 –753m. One transect was located in a Driftwood (DR) association (Hawkes et al. (2007), two transects were located in an unspecified Flood association (Fl) of MacKenzie and Moran (2004), and a fourth transect was located is a developing Water sedge –Beaked sedge Fen (Wm01).
- Wood Debris
- Flood Association
- The ground cover varied across this elevation band with two transects dominated by mineral soil (83.2 per cent) and wood debris (5.5 per cent), one transect dominated by wood (94 per cent) 4
- Herb cover and shrub cover was very low cross all transects (8.2and 1.5, respectively) with sparse amounts *Carex viridula*, *Deschampsia cespitosa*, *Equisetum variegatum*, *Packera plattensis*, and *Calamagrostis stricta*.

Target (753 –754m):

- Vegetation cover in this elevation band was low herb (10.6 per cent) and moderate shrub (38.7 per cent): substrate was dominated by moss (52.7 per cent) and mineral soil (45.5 per cent).
- *Carex viridula* (2.9 per cent) and *Equisetum palustre* (3.4 per cent) were the dominant herbs. *Salix brachycarpa* (6.7 per cent), *Salix farriae* (5.2 per cent), *Salix commutata* (4.1 per cent) were the dominant shrubs occurring at even higher abundance at the upper end of the elevation band.
- The vegetation community in the target elevation band corresponded to the Willow Shrub (WS) associations of Hawkes et al. (2007). Under MacKenzie and Moran (2004) this community was considered a transitional community between the lower flood community at 752 –753m to a developing Willow Swamp community at the margin of the floodplain.

Upper control (754 –755m)

- This elevation band was sparely vegetated with 8.2herb and 3.6shrub. Substrate was dominated by mineral soil (45.5 per cent) and a thin layer of decomposing organics (53.8 per cent) over a cumulic regisol of gravel and silt.
- *Danthonia spicata* (2.7 per cent), *Triantha glutinosa* (1.1 per cent), *Lobelia kalmia* (*1.1 per cent), Dryas drummondii* (1.0 per cent) and *Antennaria pulcherrima* (1 per cent) were the dominant herbs. *Picea engelmannii x glauca* was the most abundant shrub (1 per cent) followed by *Salix melanopsis* and *Salix brachycarpa* both less than 1 per cent.
- The vegetation community did not correspond to the classifications of Hawkes et al. (2007). It was identified as undescribed flood association (Fl) of MacKenzie and Moran (2004). It is more similar to the *Dryas drummondii* communities described by Kembel (2000) but less well developed.

Reference (755m+)

- Vegetation cover in this elevation band was moderate (33.1herb and 8.6shrub): substrate was dominated by dead and live organic matter (55.0and 42.3 per cent) accompanied by moss (8.8 per cent) and wood (2.3 per cent).
- *Carex utriculata* was the dominant herb (5 per cent) accompanied by *C. aquatalis, C. flava, C. lasiocarpa* between 1 and 4 per cent. *Salix farriae* (4.8 per cent) was the most

prominent shrub. The red listed orchid *Liparis loeselii* was also present

- The terrestrial vegetation community did not correspond to the classifications of Hawkes et al. (2007). It is corresponded most closely to the Wm01 association of MacKenzie and Moran (2004).

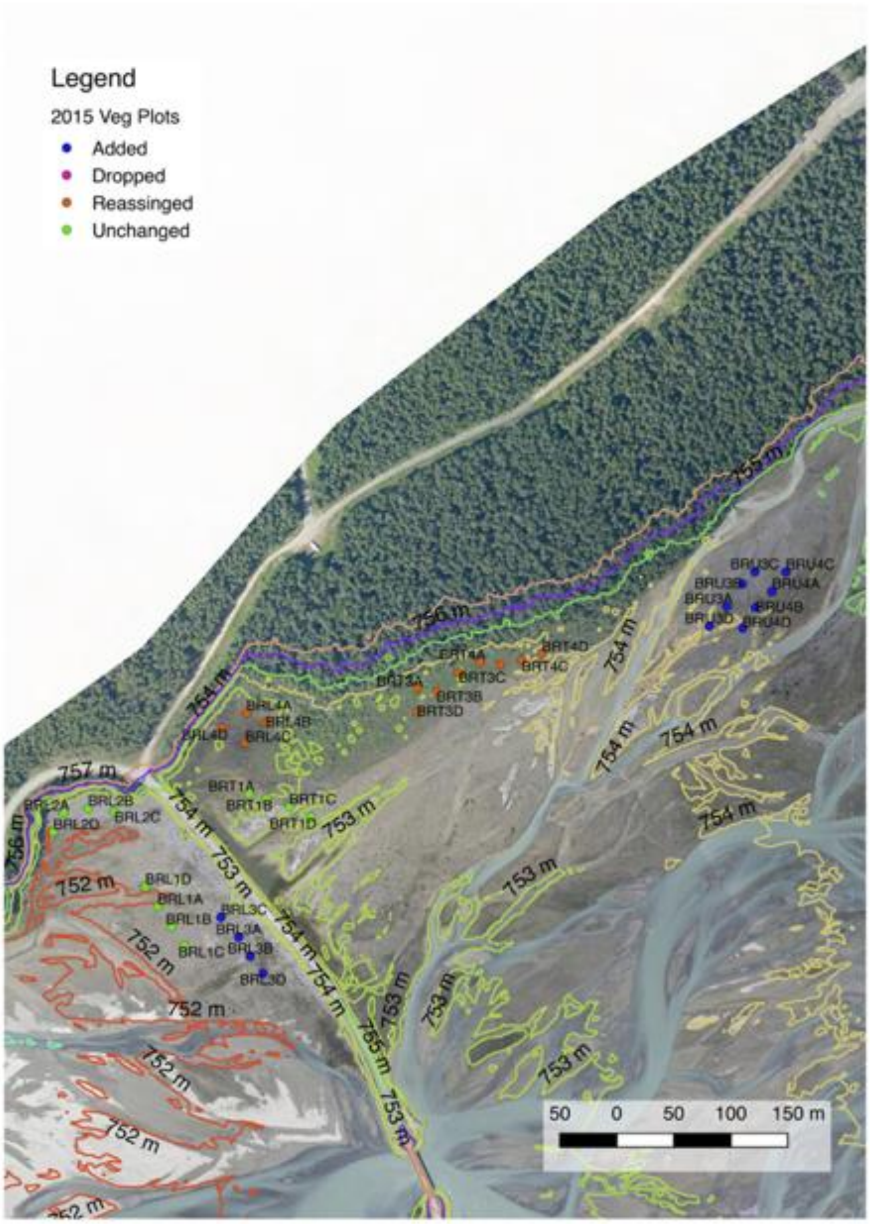
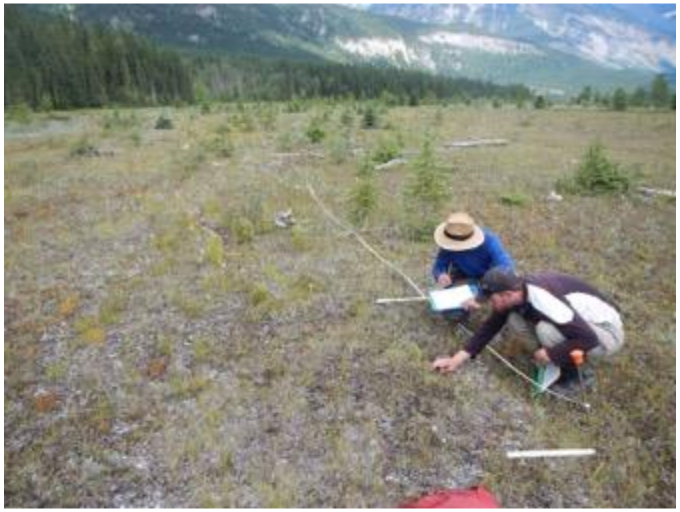


Figure 9-8: Location of terrestrial wetland sampling sites along the Bush River, 2015. Colour indicates whether the point was added, dropped, reassigned, or is unchanged with the 2014 Digital Elevation Model. Reference sites (not shown) were unchanged from previous years.

Lower Control 752m

Target Elevation Band 753m



Upper Control 754m

Reference Community 755m

Figure 9-9: Representative images of the terrestrial wetland transects along the Bush River, 2015

9.4 APPENDIX D: Supplementary results for the analyses of water physicochemistry data

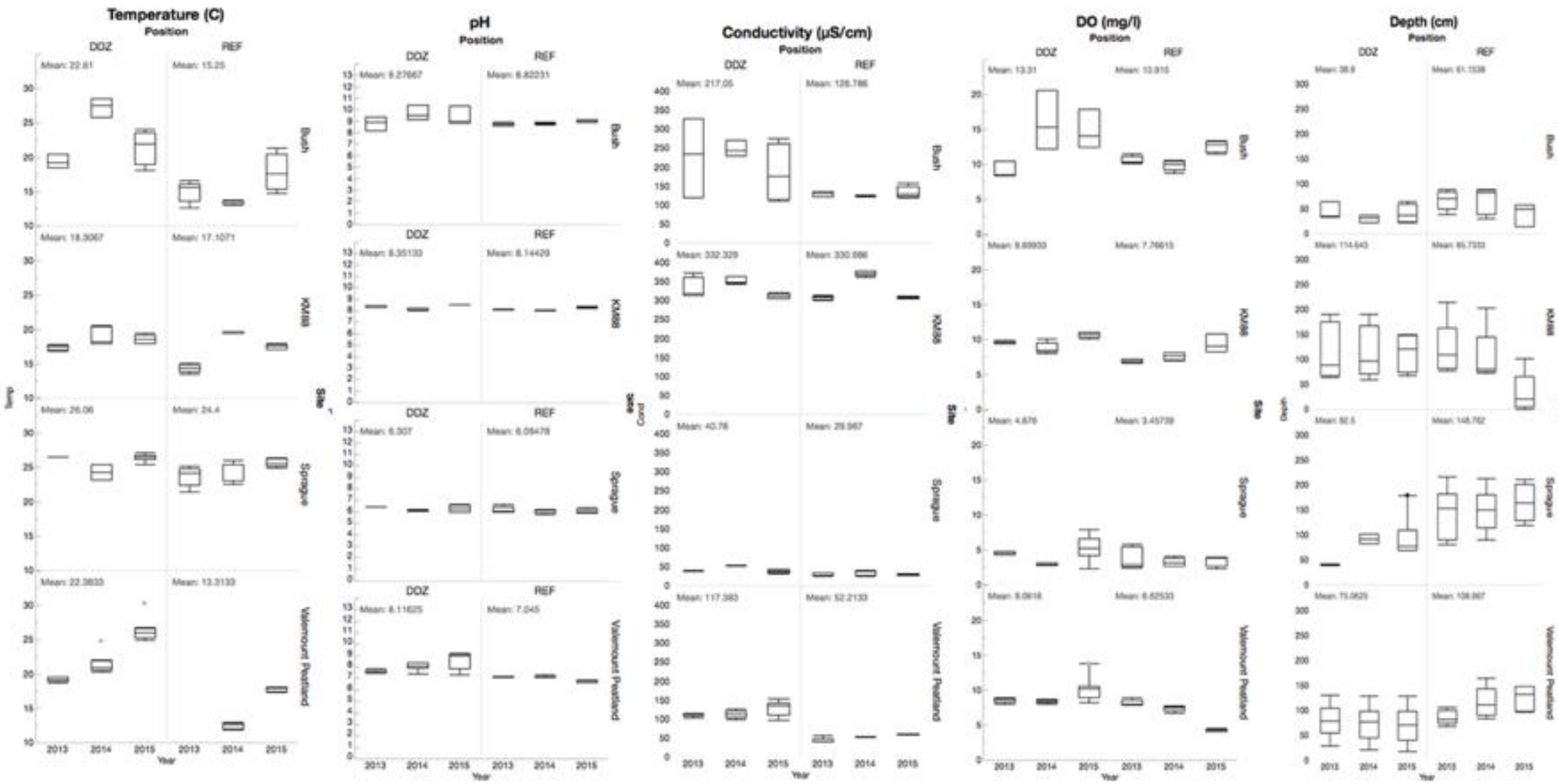


Figure 9-10: Box plots of water physicochemistry parameters collect in REF and DDZ ponds in Kinbasket Reservoir, 2013 to 2015. Means are pooled over time.

9.5 APPENDIX E: Supplementary data and result for the analyses of macrophyte biomass

Table 9-2. Mean macrophyte biomass by index site, pond position, and year.

Location	Position	Year	N	Dry Weight (g) Mean	SD
Bush River	**DDZ**	**2013**	3	79.51	61.77
		2014	3	87.99	51.58
		2015	3	142.72	127.30
	REF	**2013**	5	7.71	7.96
		2014	5	39.01	43.42
		2015	5	24.83	21.42
Km88	**DDZ**	**2012**	3	3.85	3.80
		2013	5	58.56	27.74
		2014	5	11.29	9.05
		2015	5	64.74	69.57
	REF	**2013**	5	6.29	6.11
		2014	5	20.86	37.46
		2015	5	27.46	19.80
Sprague Bay	**DDZ**	**2012**	4	5.44	2.73
		2013	2	1.49	2.08
		2014	3	1.09	1.47
		2015	2	21.06	2.62
	REF	**2012**	8	0.17	0.21
		2013	8	3.66	4.87
		2014	9	1.32	1.83
		2015	9	2.80	4.69
Valemount Peatland	**DDZ**	**2012**	8	3.32	5.60
		2013	8	1.47	1.74
		2014	8	2.98	3.75
		2015	8	12.60	9.21
	REF	**2012**	3	1.45	1.59
		2013	6	0.24	0.55
		2014	5	1.25	2.48
		2015	5	0.01	0.02

Table 9-3. Results from the Linear Mixed Effects model (LME) for macrophyte biomass.

Model where: *Log (1+Biomass) = Position + Year + Position * Year + Site + Plot (R) + Plot * Subplot (R)*

(R) indicates random effect.

Source	Nparm	DF	DFDen	F Ratio	Prob > F
Year	1	1	284.4	7.9580	0.0051*
Site	3	3	288.2	61.4775	< 0.001*
Position	1	1	289.3	55.1059	< 0.001*
Position*Year	1	1	285.8	1.7036	0.1929

9.6 APPENDIX F: Supplementary results for the analyses of aquatic metabolism.

Table 9-4. Results from the Linear Mixed Effects (LME) modeling of ponds metabolism from June 21 to August 31 in 2014 and 2015.

Effects Tests for LME Model*: variable = Position + Year + Site + Position * Year + Site*Year + Site*Position*Year + Pond (R).*

Variable	Source	N	DF	F Ratio	p value*
Log GPP	Year	1	1	21.05	<0.001
	Position	1	1	104.48	<0.001
	Site	3	3	90.71	<0.001
	Position*Site	3	3	21.31	<0.001
	Position*Site*Year	3	3	24.95	<0.001
	Site*Year	3	3	3.03	0.03
	Position*Year	1	1	8.50	0.004
Log (1 + R)	Year	1	1	3.59	0.06
	Position	1	1	34.35	<0.001
	Site	3	3	3.98	0.01
	Position*Site	3	3	61.57	<0.001
	Position*Site*Year	3	3	24.65	<0.001
	Site*Year	3	3	8.90	<0.001
	Position*Year	1	1	3.44	0.06
NEP	Year	1	1	3.54	0.06
	Position	1	1	81.15	<0.001
	Site	3	3	323.12	<0.001
	Position*Site	3	3	99.62	<0.001
	Position*Site*Year	3	3	4.91	0.002
	Site*Year	3	3	7.73	<0.001
	Position*Year	1	1	1.50	0.22

Effects Tests for LME Model*: variable = Position + Year + Position * Year + Site (R) + Pond (R).*

Variable	Source	N	DF	F Ratio	p value
Log GPP	Year	1	1	18.24	<.0001
	Position	1	1	5.01	0.11
	Position*Year	1	1	7.57	0.006
Log (1 + R)	Year	1	1	3.04	0.08
	Position	1	1	0.55	0.51
	Position*Year	1	1	3.05	0.08
NEP	Year	1	1	3.56	*0.06*
	Position	1	1	0.80	0.44
	Position*Year	1	1	1.26	0.26

DF= degrees of freedom.

***** red text includes statistically significance where α > 0.05.

Table 9-5. Results from the Linear Mixed Effects (LME) modeling of ponds between July and August 2014 and 2015.

Effects Tests *Models:*

variable = Position + Year + Month + Position * Year + Month*Year + Position*Year + Site*Position*Year + Pond (R) + Site (R).

Variable	Source	N	DF	F Ratio	p value*
Log GPP	Year	1	1	1.16	0.281
	Month	1	1	33.69	<0.001
	Position	1	1	6.43	0.082
	Position*Month	1	1	2.14	0.144
	Year*Position	1	1	2.57	0.110
	Month*Year	1	1	5.36	0.021
	Position*Month*Year	1	1	10.09	0.001
Log (1 + R)	Year	1	1	14.34	<0.001
	Month	1	1	2.76	0.097
	Position	1	1	0.17	0.706
	Position*Month	1	1	5.56	0.019
	Year*Position	1	1	0.91	0.340
	Month*Year	1	1	15.48	<0.001
	Position*Month*Year	1	1	4.47	0.035
NEP	Year	1	1	30.50	<0.001
	Month	1	1	36.07	<0.001
	Position	1	1	1.19	0.356
	Position*Month	1	1	11.01	0.001
	Year*Position	1	1	3.59	0.059
	Month*Year	1	1	5.21	0.023
	Position*Month*Year	1	1	0.86	0.355

Revised model results. Non-significant terms dropped.

Variable	Source	N	DF	F Ratio	p value
Log GPP	Month	1	1	45.73	<0.001
	Year*Month	1	1	4.85	0.028
	Year*Position*Month	1	1	9.90	0.002
Log (1 + R)	Month*Year	1	1	16.86	<0.001
	Year*Position*Month	1	1	5.04	0.025
	Year	1	1	13.74	<0.001
	Month*Position	1	1	6.25	0.013
NEP	Month*Year	1	1	6.17	0.013
	Year	1	1	33.62	<0.001
	Month*Position	1	1	10.22	0.002
	Month	1	1	36.96	<0.001

Table 9-6. Gross Primary Production (GPP), Net Ecosystem Production (NEP), and Respiration (R) before (July) and after (August) inundation in 2014. DDZ = ponds within Kinbasket Reservoir at 753 m ASL; REF = reference ponds Test statistics between paired DDZ and PER ponds prior to inundation are also provided. Bold text indicates statistically significant test results; red text indicates marginally non-significant results.

Site	Position	Period	GPP $g\ O_2\ m^{-3}\ d^{-1}$		R $g\ O_2\ m^{-3}\ d^{-1}$		NEP $g\ O_2\ m^{-3}\ d^{-1}$		Trend
			Mean	SD	Mean	SD	Mean	SD	
Bush River	DDZ	July	13.12	6.64	-11.60	8.01	1.51	3.82	**GPP⇓, R⇓, NEP⇓**
		August	4.70	2.50	-5.24	3.06	-0.54	3.02	
		p value	**< 0.001**		**< 0.001**		**0.04**		
	REF	July	1.46	0.91	-1.00	1.32	0.46	0.95	**GPP=, R=, NEP=**
		August	2.03	0.71	-2.14	1.56	-0.11	2.01	
		p value	0.32		0.27		0.62		
Km 88	DDZ	July	5.05	2.63	-5.11	3.27	-0.05	1.73	**GPP⇓, R=, NEP⇓**
		August	2.20	1.58	-3.61	2.16	-1.41	1.64	
			0.001		0.21		**0.01**		
	REF	July	1.81	1.54	-2.59	1.87	-0.78	0.61	**GPP⇑, R⇑, NEP=**
		August	4.36	3.22	-4.82	2.88	-0.46	1.14	
		p value	**0.003**		**0.007**		0.28		
Sprague Bay	DDZ	July	0.28	0.24	-4.39	1.48	-4.11	1.38	**GPP=, R=, NEP=**
		August	0.54	0.43	-4.59	0.78	-4.05	0.67	
		p value	0.28		0.59		0.92		
	REF	July	0.59	0.56	-4.25	0.88	-3.66	0.70	**GPP=, R=, NEP=**
		August	0.43	0.50	-4.52	0.77	-4.09	0.64	
		p value	0.36		0.38		0.11		
Valemount Peatland	DDZ	July	2.21	1.32	-1.33	1.06	0.88	1.18	**GPP=, R⇑, NEP⇓**
		August	1.98	1.72	-3.76	1.45	-1.78	1.46	
		p value	0.44		**< 0.001**		**< 0.001**		
	REF	July	2.06	2.21	-4.42	1.76	-2.36	1.37	**GPP=, R⇑, NEP⇓**
		August	1.22	1.08	-5.66	0.92	-4.45	0.34	
		p value	0.47		**0.03**		**< 0.001**		