Columbia River Project Water Use Plan

KINBASKET RESERVOIR

Reference: CLBMON-11A

Wildlife Effectiveness Monitoring of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir

Study Period: 2014 - 2018

Okanagan Nation Alliance, Westbank, BC

and

LGL Limited environmental research associates

Sidney, BC

November 22, 2019

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KINBASKET RESERVOIR

Monitoring Program No. CLBMON-11A

Wildlife Effectiveness of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir




Final Report

Prepared for

BC Hydro Generation

Water Licence Requirements
6911 Southpoint Drive
Burnaby, BC

Prepared by

Okanagan Nation Alliance

and

[1]LGL Limited environmental research associates

Technical Contact: Virgil C. Hawkes, M.Sc., R.P. Bio.

vhawkes@lgl.com; 1.250.656.0127

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Cover photos:

From left to right:

Large wood debris accumulated at the upper elevation of the drawdown zone at Valemount Peatland North, *Habronattus jucundus* jumping spider in the drawdown zone at Hope Creek, Savannah Sparrow perched at the Bush Arm Causeway South, and the wetland known as 'Pond 12' located in the south portion of Valemount Peatland, prior to treatment application. Photos © Charlene Wood.

EXECUTIVE SUMMARY

The goal of CLBMON-11A is to monitor and assess the efficacy of revegetation efforts and physical works trials (i.e., CLBWORKS-1 and CLBWORKS-16) to increase the suitability of wildlife habitats in the drawdown zone of Kinbasket Reservoir. CLBMON-11A was initiated in 2008 and conducted annually from 2008 to 2012 by Cooper Beauchesne and Associates Ltd. The Okanagan Nations Alliance (ONA), in partnership with LGL Limited environmental research associates, has continued monitoring since 2013, with 2018 representing the final sampling year.

This program has undergone several experimental design changes since its inception (see Table 7–1 in Appendix 1). Monitoring years 2008 to 2013 assessed the effectiveness of revegetation of the drawdown zone applied under CLBWORKS-1. By 2013, approximately 69 ha of the drawdown zone had been treated with sedge plugs, live stakes, shrub seedlings, seed mixtures, and/or fertilizer treatments. The stated objectives of CLBWORKS-1 were: (1) to maximize plant species cover in the drawdown zone; (2) to increase plant species diversity in the drawdown zone; (3) to improve littoral productivity through increased plant diversity; (4) to improve shoreline stability; and (5) to protect known archaeological sites.

The vegetation effectiveness monitoring study (CLBMON-9) concluded that transplants had fared poorly, with survivorship of sedge seedling plugs declining to < 10 per cent on average three or more years after planting. Virtually no deciduous stakes survived over this time frame. Most transplants were unable to cope with the combination of inundation timing, frequency, duration and depth, or with the by-products of these factors such as erosion, wood debris scouring, and drought conditions (Hawkes et al. 2013).

Under CLBMON-11A, several wildlife taxa (ungulates, songbirds, small mammals, and terrestrial and aerial arthropods) were monitored across revegetated areas, non-revegetated drawdown zone controls, and upland, non-reservoir upland reference sites from 2008 to 2013 (Table 7-1). However, as revegetation efforts in the Kinbasket Reservoir had been ineffective (BC Hydro 2017), wildlife effectiveness monitoring did not reveal any differences between treatment and control areas. Thus, it was not possible to address the management questions as originally stated.

Beginning in 2014, the ONA and LGL adapted the wildlife monitoring for CLBMON-11A to monitor the effectiveness of wood debris removal and physical works trials to enhance wetlands and promote the establishment of vegetation in the drawdown zone. Physical works trials included installation of log debris booms to exclude woody debris from accumulating following high water events. Debris mounds were also installed above the full-pool level of the reservoir to reduce inundation-related vegetation mortality and increase within site topographic heterogeneity. These prescriptions, alone or in combination, may function to promote the establishment and development of vegetation in the drawdown zone. Sampling at all physical works locations occurred after the treatments were applied (i.e., there was no pre-treatment sampling). This was primarily a result of the timing associated with wood debris removal (either late winter/early spring or fall) which often occurred before or after sampling associated with CLBMON-11A.

We conducted three years of post-treatment monitoring at the Bush Arm Causeway (BAC-S, BAC-N) and post-treatment monitoring at the Valemount Peatland (VP-N; four years) and Yellowjacket Creek (YJ; five years). In 2018 we also assessed the effects of wood removal at Pond 12 in Canoe Reach and the effects of sedge plug revegetation at Bush Arm KM88 (planted in 2013). The focal taxa selected to study the efficacy of these

prescriptions were spiders, beetles, and birds (songbirds, grouse, waterfowl, shorebirds, etc.).

Amphibian breeding was noted immediately following wood clearing from the wetland areas at VP-N and BAC-N. We suggest that Pond 12 (Canoe Reach) be monitored for amphibian responses to wood removal, since there has not been an opportunity to monitor breeding activity at this pond since the 2018 treatment application.

There has been only limited, site-specific evidence supporting an increase of drawdown use utilization by birds due to revegetation or physical works prescriptions. In most cases there was no observable effect of treatment. An exception to this was KM88, where the treatment transect had twice as many bird observations as the control transect. However, we did not find a greater number of arthropods at the KM88 revegetation treatment, rather arthropod abundance was greater in the control.

Similarly, there was no consistent pattern in how treatment prescriptions affected arthropod abundance, richness, and diversity, and the effect often varied site-by-site. While in most cases arthropod abundance increased yearly since treatment application, it did so in both treatment and control transects. In cases where treatment affected arthropod diversity, the effect was positive for carabid beetle species diversity but negative for spider species diversity.

While there were some differences in arthropod species composition between treatment and control areas, these differences were not significant when the effect of year was controlled for and likely reflected pre-treatment differences in species assemblages (implying that the effect was not treatment driven) or showed convergence over the years.

We are unable to draw conclusions on the effects of inundation on the treatment areas or the likelihood that treatments will endure regular inundation cycles, as full pool has not been reached since 2013. This includes the constructed mounds, windrows, and log boom exclosures. For the same reason, vegetation and wildlife responses to treatment inundation are also unknown. Likewise, the biophysical responses of the cleared ponds/wetlands to inundation remain untested and unknown. Thus, we recommend that future monitoring after higher inundation cycles would contribute to the evaluation of physical works effectiveness.

The experimental design of this program was challenged by the lack of pre-treatment data for paired control and treatment polygons at VP-N, YJ, KM88, and Pond 12. These data are essential for evaluating treatment effects. In absence of pre-treatment data, it is unknown to what extent within-site differences confound wildlife response measures.

There were marked inter-annual differences in response measures (abundance and richness) in upland reference areas. Thus, it is likely that this inter-annual variation influenced the drawdown zone communities also. In addition, the lack of coordination between this monitoring program and other programs resulted in several experimental controls and treatments being disturbed by heavy machinery from 2015-2018. With such a brief time scheduled for post-treatment monitoring of physical works trials (2014-2018), the inadvertent clearing of controls and re-clearing of experimental treatments compromised our ability to decipher effects over the short duration of this program since physical works were implemented.

The final status of CLBMON-11A is summarized in table form below. The revised terms of reference (BC Hydro 2017) specified that the words 'revegetation prescriptions' refer to specific works implemented to foster revegetation (e.g., log booms and wood mounds), thus our answers will focus mainly on the results of physical works trials monitored from 2014 to 2018.

KEYWORDS: Kinbasket Reservoir; revegetation; physical works; diversity; arthropods; songbirds; effectiveness monitoring; drawdown zone; hydro.

Management Question (MQ)	Summary of Key Result
1: How effective are the revegetation prescriptions[1] at enhancing and increasing the drawdown zone habitat use by wildlife such as birds and amphibians?	<u>Summary Findings</u> We assessed the species richness and relative abundance and distribution of breeding songbirds and shorebirds relative to revegetation and physical works trials. No consistent pattern was revealed in how treatment affected bird utilization of drawdown zones. The most abundant bird species were in relatively equal proportions in both control and treatment areas, suggesting no effect of treatment type. The exception was a possible increase in Savannah Sparrows following revegetation treatment in KM88. A number of factors (see sources of uncertainty below and discussions in report) limit the utility of using bird data to assess the effectiveness of revegetation prescriptions at the scale of revegetation implemented. Clearing ponds of wood debris was effective at enhancing breeding habitat suitability for amphibians in the drawdown zone (e.g., Western Toad; Hawkes 2017). <u>Sources of Uncertainty/Limitations</u> Data are limited to short-term responses in the absence of full-pool reservoir events. We are unable to attribute changes in wildlife responses to a treatment effect. Vegetation and wildlife use in post-treatment years (2014-2017) may be related to lower maximum reservoir levels (<754 m) than in pre-treatment years (2012 and 2013; >754.6 m). Lack of pre-treatment sampling at paired treatment and control areas within VP-N, YJ, Pond 12, and KM88, thus, we cannot rule out pre-treatment differences between control and treatment areas that would confound assessments of wildlife measures. Due to low reservoir levels (below treatment elevations), we have had no opportunity to test the efficacy of log booms for exclusion of wood debris. Nor have we had the opportunity to assess mounds and windrows following inundation. Likewise, there has been no assessment of cleared wetlands under reservoir operations that cause seasonal inundation. The uncoordinated wood removal that has occurred for CLBWORKS-16 in experimental plots since initial wood removal has hindered the study design of this program. The loss of site replicates at Packsaddle Creek North, Packsaddle Creek South, disturbance of control plot at Valemount Peatland North, and re-treatment of the Yellowjacket treatment plot, challenged our ability to fully answer this management question. Overall the study has low statistical power to detect changes in bird use of the drawdown zone due to the small size of plots and limited number of observations in each survey. <u>Comments</u> Follow-up monitoring is recommended to assess the persistence of revegetation/physical works treatments and long-term effects on wildlife use. Debris mounds have the potential to increase wildlife populations that are not a current focus of study (e.g., small mammals). A longer time series of data is required to address this question completely. To capture the longer-term successional trajectories and better determine the success of treatment areas, it is recommended that further sampling be undertaken at select sites.

Management Question (MQ)	Summary of Key Result
2: To what extent does revegetation**Error! Bookmark not defined.** increase the availability of invertebrate prey (e.g. arthropods) in the food chain for birds and amphibians?	<u>Summary Findings</u> We assessed the relative abundance of ground-dwelling spiders and beetles in response to revegetation and physical works trials overtime. Abundance patterns varied between years, sites, and treatments and were unrelated to revegetation or physical works trials. In most cases, relative abundance of ground dwelling spiders and beetles increased slightly in both treatment and control areas after physical works and/or revegetation prescription implementation. Thus, we found no support for the increased availability of these taxa due to treatment application. Data collection focused on ground-dwelling spiders and beetles because of their known habitat specificity and small-scale, short-term response to changes in vegetation cover. Other arthropod prey groups (e.g., aerial insects, caterpillars, grasshoppers) may respond differently to treatment types over the long term. Given their high abundances, spiders and beetles may also provide an important role in the food chain for wildlife. <u>Sources of Uncertainty/Limitations</u> Post-treatment monitoring was limited to only 1-4 years, due to short timeline after the program focus changed to monitor physical works trials. The timeline was shortened by disturbance of treatment plots by wood removal crews from 2015-2018. Longer duration of data collection would help clarify treatment effects on response measures. Controls and treatments exhibited identical changes in arthropod relative abundance. Arthropod catches may be related to inter-annual changes in climatic conditions, reservoir operations, and/or other factors, rather than a specific treatment effect The unexpected wood removal that has occurred for CLBWORKS-16 in experimental plots since our experimental plots were setup hindered the study design of this program. Due to site disturbance from heavy machinery, all treatment and control areas at Packsaddle Creek North, Packsaddle Creek South were completely cleared of wood. The loss of controls and heavy disturbance of recovering treatment areas caused us to no longer be able to experimentally monitor these two sites in Canoe Reach. Further ground disturbance of control plot at Valemount Peatland North, and re-treatment of the Yellowjacket treatment plot, challenged our ability to address fully this management question because ground disturbance influences the soil-dwelling invertebrate communities. A direct assessment of the availability and abundance/biomass of prey (e.g., for birds, amphibians) has not occurred, which would require a more focused study on the foraging dynamics and prey preferences of specific predator species.

Management Question (MQ)	Summary of Key Result
3: How do revegetationError! Bookmark not defined. prescriptions affect the diversity and abundance of arthropods, amphibians and birds?	<u>Summary Findings</u> Treatment methods achieved mixed success at promoting arthropod abundance (see MQ2), diversity, and richness. Spider guild composition showed a decrease in dominance of ground-runners over time in VP-N and YJ wood removal treatments. The proportional abundance of ambushers, sheet/funnel-weavers, and space-web builders increased since wood removal in YJ treatment. Further, the increase in diversity of spider guilds in the VP-N treatment overtime is an indication of increased niche availability within this site. In particular, the appearance of fishing spiders in 2018 is a reflection of the developing wetland function since treatment application. In cases where treatment affected arthropod diversity or richness, the effect was always negative for spiders (species diversity and/or richness were greater in control than treatment areas) and positive for carabid beetles (species diversity and/or richness were greater in treatment than control areas). Evidence suggests that amphibians continually use both treated and untreated drawdown zone habitats (CLBMON-37), however, enhanced breeding activity was observed in cleared wetlands at VP-N and BAC-N. The abundance and diversity of amphibians increased immediately after ponds were cleared of wood debris at these two study sites. There was no consistent pattern in how treatment type affected bird diversity and richness. In most cases there was no observable effect. At VP-N there were more species in the treatment transect, but the control transect had higher abundances. KM88 showed a potentially positive effect of treatment, where the total bird species was similar between transect types, but treatment had twice as many observations. <u>Sources of Uncertainty/Limitations</u> The sources of uncertainty/limitations for MQ1 and MQ2, above, also apply to this MQ. <u>Comments</u> Follow-up monitoring is recommended to assess the persistence of revegetation/physical works treatments and long-term effects on wildlife use. A longer time series of data is required to answer this question. It is recommended that further sampling be undertaken at select sites to capture the longer-term successional trajectories and better determine the success of treatments. Assessments of amphibian use of Pond 12 post-treatment should be prioritized for future wildlife effectiveness monitoring. Follow-up monitoring of cleared wetland habitats at VP-N and BAC-N should focus on productivity and wetland function in the context of seasonal reservoir inundation.

Management Question (MQ)	Summary of Key Result
4: Which revegetation Error! Bookmark not defined. method is most effective at enhancing or increasing the utilization of wildlife habitat in the drawdown zone?	<u>Summary Findings</u> There were different restoration methods (treatments) to enhance the utilization of wildlife habitat in the drawdown zone. The most successful treatment for enhancing bird use of the drawdown zone was the revegetation prescription applied at KM88 in 2013. Planting at this site was conducted in the spring of 2013 (Adama 2015), with both Kellogg's and Columbia sedge plugs. These plugs were noted to be larger than the previous stock planted during the earlier components of the revegetation program, which may have played a role in their successful establishment. While only ~35% of the transplants had survived by 2018, we measured greater bird use of treatment relative to adjacent control polygons. While three bird species were observed at this site, only Savannah Sparrows were detected in all treatment transects, with twice as many detections as the adjacent control transects. This result was not found for arthropods, which had greater abundance at the KM88 controls. Note: the KM88 revegetation site only had one year of monitoring (2018). The most successful treatment for enhancing amphibian use of the drawdown zone was when wood removal was performed at wetland and pond locations. We immediately observed an increase in amphibian use of the habitat and increased breeding activity at ponds cleared of wood debris, which provides strong support for this technique. Whether wood mound creation will translate to habitat enhancement is yet to be seen. Results of wood mounds and wood removal treatments were mixed. There was no compelling evidence that either method increased utilization by birds or arthropods. Revegetation prescriptions monitored prior to 2014 were largely unsuccessful (low survival), except for the successful sedge plug treatments documented from KM88 (discussed above). <u>Sources of Uncertainty/Limitations</u> The response of certain taxa (e.g., small mammals) to increased topographic heterogeneity (mounds) in the drawdown zone is not currently being monitored but should be considered for future study as they are known to be enhanced by similar wood debris mounds (Sullivan et al. 2017). In some instances, several techniques were applied in the same area (e.g., BAC-N: wood clearing, mound creation, planting, log boom installation) making it difficult to separate the effectiveness of different treatments. In other instances, sites were repeatedly cleared of wood debris (YJ, PS-S, PS-N) or controls were treated (VP-N), hampering our efforts to assess the response of wildlife to clearing in these areas. The sources of uncertainty/limitations for the above MQs, also apply to this MQ. <u>Comments</u> The general comments for MQ1 and MQ3, above, also apply to this MQ.

[1] Revegetation' refers to all methods intended to enhance vegetation in the drawdown zone of Kinbasket Reservoir (e.g., planting prescriptions, wood removal, log booms, mounds).

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1.0 INTRODUCTION

Kinbasket Lake Reservoir is located between the towns of Donald and Valemount in eastern British Columbia. The reservoir was created by the construction of the Mica Dam, which was operational March 29, 1973. Mica is the primary storage reservoir for power generation on the Columbia River drainage system. A Water Use Plan (WUP) was developed in 2007 as a result of a multi-stakeholder consultative process to determine how to best operate BC Hydro's facilities on the Columbia River to balance environmental values, recreation, power generation, culture/heritage, navigation, and flood control (BC Hydro 2007). The WUP consultative committee supported a reservoir wide revegetation program (CLBWORKS-1), which was initiated in 2007 to improve vegetation growth in the reservoir drawdown zone.

Vegetation in the upper elevations of the reservoir is negatively impacted by the operation of Kinbasket Reservoir for power generation which erodes the shoreline habitat (BC Hydro 2005). Revegetation goals include offsetting operational impacts and improving productivity, wildlife habitat, erosion control, archaeological site protection, and shoreline aesthetics. Recognizing the need to assess the effectiveness of this program, the consultative committee also recommended additional studies to monitor and audit the effectiveness of planting efforts on vegetation communities and wildlife habitat use. This recommendation resulted in the creation of CLBMON-11A, an 11-year monitoring program to assess the effectiveness of revegetation efforts at improving habitat for wildlife in the drawdown zone of Kinbasket Reservoir. The history and timeline of CLBMON-11A from 2008 to 2018, including terms of reference revisions, focal taxa and monitoring sites, and treatments monitored, is detailed in Appendix 1.

The key water use planning decision affected by the results of this monitoring program is whether revegetation is effective at enhancing wildlife habitat and reducing any negative effect of reservoir operations on wildlife in lieu of changes to reservoir operations. Results from this study will support more informed decision making with respect to the need to balance the requirements of wildlife species dependent on riparian areas with other values such as recreational opportunities, flood control, and power generation.

Wildlife monitoring was conducted annually from 2008 to 2012 by CBA (CBA 2009, 2010, 2011; MacInnis et al. 2011, 2012), and by the Okanagan Nation Alliance and LGL Limited from 2013 to present. Starting in 2014, an assessment of the effectiveness of woody debris removal to promote the establishment and development of vegetation in the drawdown zone was initiated; as were the efficacy of log booms to prevent the accumulation of woody debris, and debris mounds to reduce inundation-related vegetation mortality and enhance the drawdown zone habitat. These prescriptions, alone or in combination, may function to promote the establishment and development of vegetation in the drawdown zone.

In 2018, monitoring continued at the same locations as in 2017 (see Wood et al. 2018), with two additional sites Pond 12 (wood removal) and KM88 (sedge plug). The focal taxa selected to study the efficacy of these prescriptions were spiders, ground beetles, and birds (includes songbirds, grouse, waterfowl, shorebirds, etc.). Vegetation data were collected and assessed under CLBMON-9.

Amphibians and reptiles are only monitored through incidental observations as they are the focus of two separate studies in the same area (CLBMON-37 and CLBMON-58).

This report is the first to follow the revised terms of reference (BC Hydro 2017) and is a comprehensive assessment of data collected since 2014. Monitoring of revegetation efforts prior to 2014 were concluded to be largely ineffective and wildlife monitoring efforts were unable to adequately address the management hypotheses (Hawkes et al. 2014). Thus, these data are not discussed in this report.

1.1 Objectives

The objectives of CLBMON-11A (BC Hydro 2017) are to:

1. Assess whether the revegetation prescriptions[1] in the drawdown zone of Kinbasket Reservoir improve habitat for wildlife.
2. Report and provide recommendations in Year 10 (2018) on the effectiveness of the revegetation prescriptions on improving habitat for wildlife in the drawdown zone.

2.0 STUDY AREA

The Columbia Basin in southeastern British Columbia is bordered by the Rocky, Selkirk, Columbia, and Monashee Mountains. The headwaters of the Columbia River begin at Columbia Lake in the Rocky Mountain Trench, and the river flows northwest along the trench for about 250 km before it empties into Kinbasket Reservoir behind Mica Dam (BC Hydro 2007). From Mica Dam, the river continues southward for about 130 km to Revelstoke Dam, and then flows almost immediately into Arrow Lakes Reservoir behind Hugh Keenleyside Dam. The entire drainage area upstream of Hugh Keenleyside Dam is approximately 36,500 km^2.

The Columbia Basin is characterized by steep valley side slopes and short tributary streams that flow into Columbia River from all directions. The Columbia River valley floor elevation ranges from approximately 800 m near Columbia Lake to 420 m near Castlegar. Approximately 40 percent of the drainage area within the Columbia Basin is above 2,000 m elevation. Permanent snowfields and glaciers predominate in the northern high mountain areas above 2,500 m elevation. About 10 percent of the Columbia River drainage area above Mica Dam exceeds this elevation.

Precipitation in the basin is produced by the flow of moist, low-pressure weather systems from the Pacific Ocean that move eastward through the region. More than two-thirds of the precipitation in the basin falls as winter snow. Snow packs often accumulate above 2,000 m elevation through the month of May and continue to contribute runoff long after the snow pack has melted at lower elevations. Summer snowmelt is reinforced by rain from frontal storm systems and local convective storms. Runoff begins to increase in April or May and usually peaks in June to early July, when approximately 45 percent of the runoff

[1] ‘Revegetation’ refers to physical works trials in addition to revegetation treatments.

occurs. The mean annual local inflow for the Mica, Revelstoke, and Hugh Keenleyside projects is 577 m^3/s, 236 m^3/s and 355 m^3/s, respectively.

Air temperatures across the basin tend to be more uniform than precipitation. The summer climate is usually warm and dry, with the average daily maximum temperature for June and July ranging from 20–32°C.

2.1 Kinbasket Reservoir

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

Kinbasket Reservoir is lowest during April to mid-May, fills throughout late spring and early summer, and is typically full by mid- to late-summer (Figure 2-1). Notably, in 2012 and 2013 Kinbasket Reservoir was filled beyond the normal operating maximum (i.e., > 754.38 m ASL) for the first time since 1997. Since September 2013, water levels have been kept below the operating maximum.

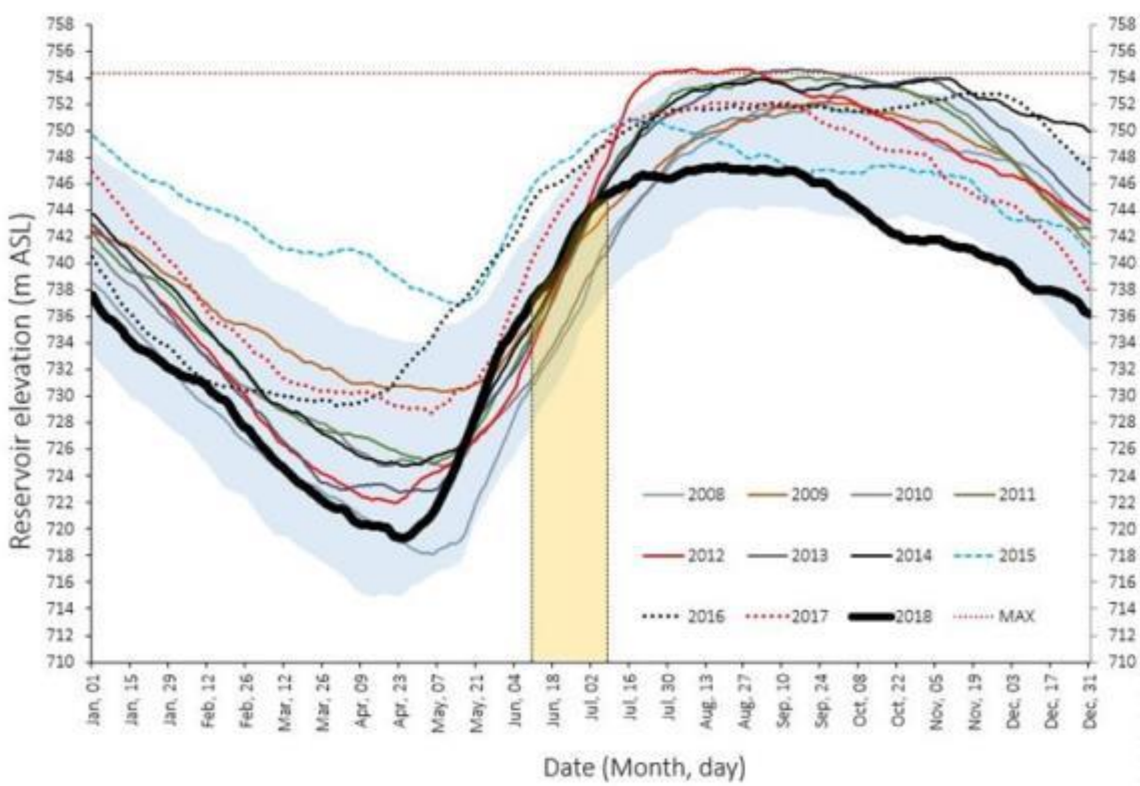


Figure 2-1: Kinbasket Reservoir hydrograph for the period 2008 through 2018. The blue shaded area represents the 10th and 90th percentile for the period 1976 through 2018; the dashed red line is the normal operating maximum; the yellow shaded region indicates the period which field monitoring was conducted in 2018.

The reservoir is located predominately within the Interior Cedar-Hemlock (ICH) Biogeoclimatic (BEC) zone and is represented by four subzone/variants (Table 2-1). The ICH occurs along the valley bottoms and is typified by cool, wet winters and warm dry summers. A small portion of the reservoir extends into the Sub-Boreal Spruce (SBS) BEC zone dh1 variant near Valemount. The climate of the SBS is continental and characterized by moderate annual precipitation and seasonal extremes of temperature that include severe, snowy winters and relatively warm, moist, and short summers.

Table 2-1: Biogeoclimatic zones, subzones, and variants occurring in Kinbasket Reservoir study area.

Subzone	Zone Name	Subzone/Variant Description
ICHmm	Interior Cedar – Hemlock	mm: Moist Mild
ICHwk1	Interior Cedar – Hemlock	mk1: Wells Gray Wet Cool
ICHmw1	Interior Cedar – Hemlock	mw1: Golden Moist Warm
ICHvk1	Interior Cedar – Hemlock	vk1: Mica Very Wet Cool
ICHmk1	Interior Cedar – Hemlock	mk1: Kootenay Moist Cool
SBSdh1	Sub-Boreal Spruce	dh1: McLennan Dry Hot

2.2 Study Sites

The southern end of the reservoir includes Bush Arm and the Columbia Reach. Bush Arm is characterized by flat or gently sloping terrain that was created by fluvial deposition from Bush River and other inflowing streams. These features are often protected from wind and wave action by the islands and peninsulas that protrude along the shoreline. This combination creates the largest variety of valuable wildlife habitat in the entire reservoir. Extensive fens and other wetlands have been identified, and a high diversity of plants is supported (Hawkes et al. 2007).

The extensive Valemount Peatland at the northern end of the reservoir supports the greatest diversity and abundance of wildlife in Canoe Reach. Historically, this peatland was likely a combination of sedge and horsetail fen and a swampy forest dominated by spruce (Ham and Menezes 2008). The wildlife habitat in the peatland varies from highly productive riparian and wetland habitat, to highly eroded sand and cobble parent material. Large areas are virtually devoid of vegetation and portions of the peatland are covered by deposits of wood chips from the breakdown of floating logs (Hawkes et al. 2007). Other notable habitats in the northern end of Kinbasket reservoir include wetlands and ponds on the gently sloping banks along the reservoir's eastern side.

Since 2008, several study sites with a variety of treatments have been monitored under CLBMON-11A (see Appendix 1 and previous annual reports). In 2018, surveys were conducted at six main study sites (Figure 2-2). Four of these sites were the focus of monitoring for 2017. An additional two sites were added in 2018 to monitor the initial baseline treatment condition at these locations (Pond 12 and KM88). Site names, descriptions, and codes are listed in Table 2-2. In addition, the upland forest at Goodfellow Creek is used as a reference sample for comparison with arthropods at Bush Arm Causeway. This site is located approximately 144 meters southwest of the Bush Arm Causeway South control plot.

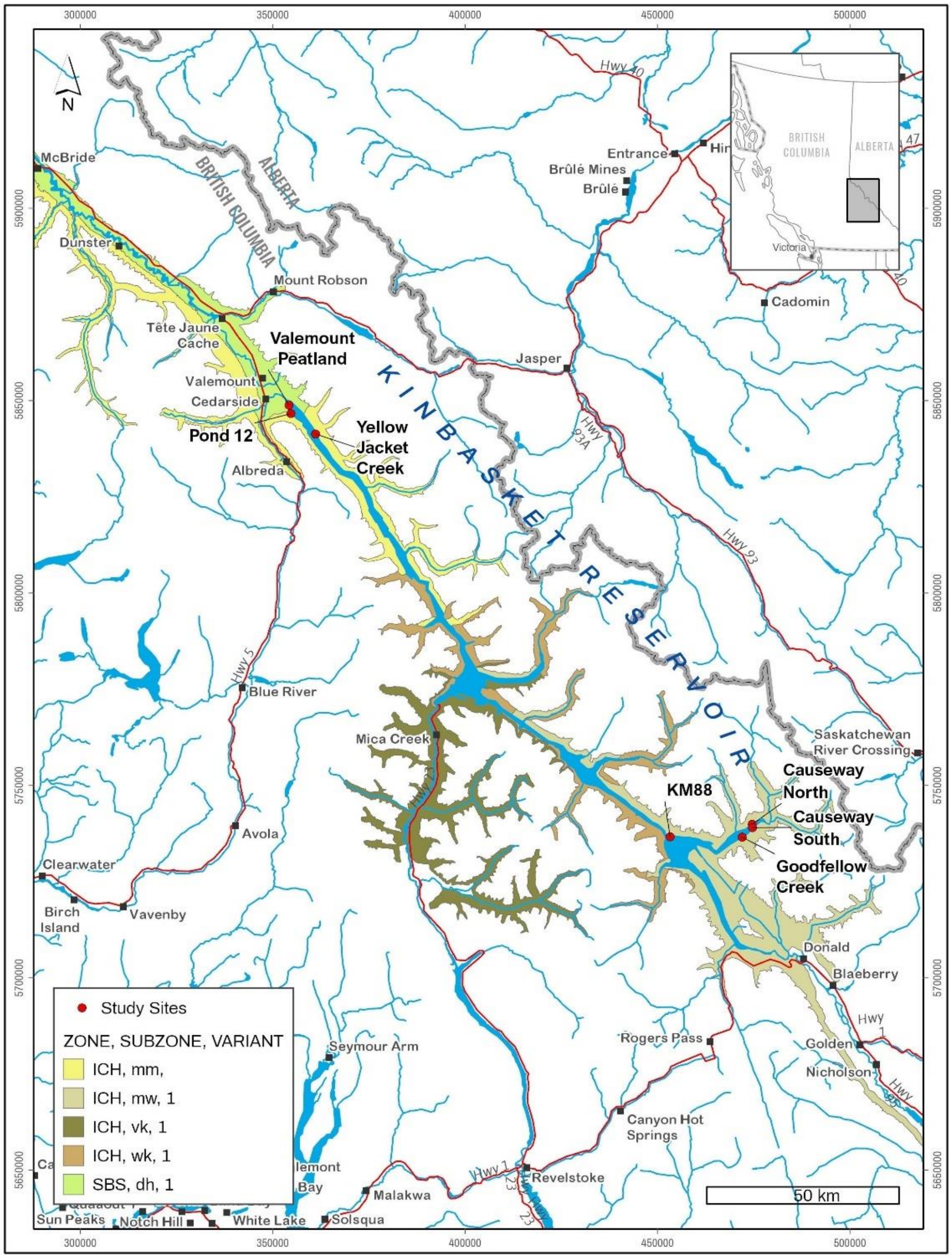


Figure 2-2: Location of Kinbasket Reservoir in British Columbia (insert, upper-right) and locations sampled for CLBMON-11A in 2018 (red points). Refer to Table 2-1 for descriptions of Biogeoclimatic (BEC) zones.

Table 2-2: Study sites established at Canoe Reach and Bush Arm of Kinbasket Reservoir for the 2018 monitoring year. Habitat type: treatment (T), control (C), and reference (R); DDZ = drawdown zone, UPL = upland forest, *treated control plot.

Reach	Site	Plot	Description	2018 Surveys
CANOE REACH	Yellowjacket Creek (YJ)	T	DDZ – wood debris removal (2014 & 2017)	Arthropods, Birds
		C	DDZ – wood debris accumulation	Arthropods, Birds
		R	UPL – upland forest	Arthropods
	Valemount Peatland North (VP-N)	T	DDZ – wood debris removal (2014) & log boom	Arthropods, Birds
		C*	DDZ – wood debris removal (2018)	Arthropods, Birds
		R	UPL – upland forest	Arthropods
	Pond 12	T	DDZ – wood debris removal (2018)	Arthropods, Birds
		C	DDZ – wood debris accumulation	Arthropods, Birds
		R	UPL – upland forest	Arthropods
BUSH ARM	Bush Arm Causeway Northwest (BAC-N)	T	DDZ – wood removal, pond clearing, mound/windrow creation, revegetation, & log boom (2015)	Arthropods, Birds
		C	DDZ – unaltered	Arthropods, Birds
	Bush Arm Causeway Southwest (BAC-S)	T	DDZ – wood removal, mound/windrow creation, revegetation (2015)	Arthropods, Birds
		C	DDZ – unaltered	Arthropods, Birds
	KM88 (Big Bend)	T	DDZ – revegetation with sedge plugs (2013)	Arthropods, Birds
		C	DDZ – unaltered	Arthropods, Birds
		R	UPL – upland forest	Arthropods
	Goodfellow Creek (GDF)	R	UPL – upland forest	Arthropods

2.3 Physical Works Treatments

2.3.1 Wood Removal

Wood removal treatments (CLBWORKS-16) have been monitored in six sites at Canoe Reach from 2014 to 2018. At Valemount Peatland North (VP-N), a shallow wetland and adjacent terrestrial habitat was cleared of wood in 2014. A log boom was also installed at VP-N as a trial to exclude wood accumulation following high reservoir flow events to allow vegetation to naturally regenerate in this area. Wood removal was also applied at Yellowjacket Creek (YJ) in 2014. These two study areas were monitored from 2014 to 2018. In 2018, the surface of a large pond (Pond 12) and adjacent drawdown zone was cleared of wood accumulation and monitored in the same year after treatment (see details below: Pond 12). In addition to the three sites cleared of wood that were studied in 2018 (i.e., those included in Table 2-2: VP-N, YJ, and Pond 12), there were other treatment areas under study in previous monitoring years at Canoe Reach. These were removed from monitoring due to compromised experimental design. In 2012, Valemount Peatland South (VP-S) was cleared of wood, however, no control area was available at this site rendering it unfit for determining treatment effects. This site was included in monitoring for 2014 only. In 2014, Packsaddle Creek North (PS-N) and Packsaddle Creek South (PS-S) were cleared of wood debris at Canoe Reach, yet the treatment and control plots at these sites were heavily disturbed by machinery and subsequently cleared of wood. The newly regenerated vegetation was removed in the process of this unplanned debris manipulation. The elimination of experimental controls hindered our ability to test for treatment effects in these two sites, and as a result, those sites were dropped from monitoring in 2016.

Pond 12. In 2018, this additional site at Canoe Reach was cleared of coarse woody debris. Pond 12 is located approximately 1.6 km south of the study area at Valemount Peatland-North. This area comprises vegetation communities that reflect both the historic fen complex that existed prior to inundation and the elevation gradient within the reservoir (Moody and Carr 2003, Hawkes et al. 2010). Peat is the dominant substrate, however, wood debris and wood fragments previously covered portions of the remnant fenland (Hawkes et al. 2010). Wood debris has accumulated in Pond 12 due to annual reservoir inundation cycles covering a large portion of the pond area with wood (Hawkes 2016; Figure 2-3 and Figure 2-4). This wetland is a particularly diverse location for pond breeding amphibians and was highlighted for wood removal treatments by Hawkes (2016).

In 2018, wood removal was conducted along the terrestrial habitat north of the pond, in which large woody debris were chipped and spread into a mat, which formed the treatment area (Figure 2-5). This treatment differed from the wood removal studied in other sites within Canoe Reach, in that much more fine wood fragments were retained in the treatment plot, blanketing the vegetation existing in the treatment plot north of Pond 12. The pond itself was also cleared of all debris (Figure 2-3). The remaining wood covered area south of Pond 12 served as a control for sampling that occurred in 2018 (see Map 7-1 for details on sampling points).

Figure 2-3: Overhead images of Pond 12 with wood cover over the pond surface and adjacent terrestrial habitat in 2016 (left) and in 2018 post-wood removal (right). See Map 7-1 for location of sampling units.

Figure 2-4: Pond 12 in 2014 showing wood cover over the far end of the pond prior to wood removal.

Figure 2-5: The terrestrial treatment (left) and control (right) areas located on either side of Pond 12, surveyed in 2018.

Lack of coordination with wood-removal crews compromised the experimental design of this program at Canoe Reach. We reported the dismantling of control plots and re-treatment of treatment plots in previous years (Wood et al. 2017, 2018; Appendix 1). In 2018, another study site was impacted by uncoordinated wood removal. The control plot at Valemount Peatland North was treated and all coarse woody material was removed prior to surveys (Figure 2-6). The loss of experimental integrity throughout this monitoring program continuously challenged our ability to assess effectiveness of treatments.

Figure 2-6: Photos of the control plot at Valemount Peatland North (VP-N C) during field surveys in 2017 (above) showing natural levels of wood accumulation and photos from VP-N C in 2018 (bottom) showing the area recently treated by wood-removal.

2.3.2 Wildlife Physical Works

Wildlife physical works occurred in two sites at Bush Arm and were proposed at an additional three sites at Bush Arm under CLBWORKS-1 (Hawkes 2016, 2017). All five of these sites had one year of pre-treatment monitoring in 2015 (Wood et al. 2016). Physical Works trials to construct mounds and wind rows and clear ponds of wood debris in the drawdown zone of Kinbasket Reservoir were implemented at Bush Arm Causeway North and South (BAC-N, BAC-S) in Fall

2015 (Hawkes 2016, 2017). These two study sites have been the focus of treatment monitoring in Bush Arm in 2016, 2017, and 2018.

The 2015 project resulted in the construction of seven mounds in two locations, the cleaning of three previously wood-choked ponds in one location, and the removal of wood debris from the surrounding drawdown zone areas. Additionally, these trials were aimed at increasing the topographic heterogeneity of the upper portion of the drawdown zone (i.e., making the flat and uniform surface conditions of the drawdown zone rough and more diverse). This method was proposed to create a diversity of current physical conditions and result in establishment of a diversity of plant species and thus increase site productivity (Polster 2011; Loreau 2010). To protect areas cleared from wood debris at BAC-N (particularly the cleared wetlands), a log boom was installed in June 2016. Additional work focused on the planting of live stakes at the mounds at the BAC-S site.

2.4 Revegetation Treatments

Monitoring from 2008 to 2013 focused on revegetation treatments (Appendix 1). However, Hawkes et al (2013) concluded that sedge plug and live stake plantings conducted from 2008 to 2011 were largely unsuccessful. Sedge plug survivorship declined from approximately 40 percent in the two years following planting, to < 10 percent three years post-planting, to less than five percent four to five years post-planting. Live stakes of deciduous shrubs (willows, alder, and cottonwood) fared worse, with none found surviving five years after planting.

KM88. One notable exception to the widespread failure of revegetation efforts was the 2013 sedge planting conducted at the KM88 ('Big Bend') site. The KM88 study site sits on an old lake bed on the east side of the Kinbasket Reservoir, 7.5 km northwest of Bush Harbour, and 1.0 km north of Bear Island. The site is bisected by the old 'Big Bend' section of the Trans-Canada Highway that followed the Columbia River from Golden to Revelstoke prior to the creation of the reservoir (visible in Map 7-2). The site has a warm south aspect and is positioned on a bench of glacial lacustrine fines. Soils are fine-textured, silty clay loams with little to no sand or rock, and of glacial-lacustrine origin.

Revegetation was implemented at KM88 in three polygons under CLBWORKS-1 in 2013. Polygons 1 and 3 were planted with Kellogg's sedge and polygon 5 was planted with Columbia and Kellogg's sedge. The planting objectives were to: i) plant at a site that had the greatest likelihood of success for establishment, ii) increase the extent of the Kellogg's sedge (KS) community down to 746 m ASL and, iii) increase the overall abundance of sedges in the proposed planting areas. The planting density was 20,000 plugs per hectare across the three treatment subunits (0.5, 0.82, and 1.95 ha; Adama 2015). In 2015, survival of sedge plugs ranged from 43 to 100 percent (Hawkes and Miller 2016). By 2018, average estimated surviving plug densities (per ha) had declined to approximately 7190, 9310, and 8440 in the three treatment polygons (TU-1, TU-3, and TU-5, respectively; Miller and Hawkes 2019). While survival averaged only ~35% of initial planting densities in 2018, this site is one of the better examples of revegetation success. Elevation ranged from 746 to 750.5 m ASL for treatment polygons. An example of the vegetation found in treatment and control polygons is given in Figure 2-7.

Figure 2-7: Representative photos of KM88 treatment units (top, left to right: TU1, TU3, and TU5) and control units (bottom, left to right: CU1, CU2, and CU3) with varying sedge densities in 2018. TU: treatment unit, CU: control unit.

3.0 METHODS

3.1 Overview

The focal taxa selected for study were ground-dwelling spiders and beetles and breeding birds.

Species of ground-dwelling ('epigaeic') spiders (Araneae) and ground beetles (Coleoptera: Carabidae) are effective focal taxa for monitoring changes in terrestrial habitats. These taxa are easily and simultaneously sampled using pitfall traps (Marshall et al. 1994), comprise a large proportion of epigaeic arthropod abundance and diversity, occur in almost all terrestrial habitats, include both specialist and generalist species (Niemelä et al. 1993), can be studied across any gradient of habitat change, and respond to both fine-scale and landscape-scale environmental changes. Arthropods are also useful for monitoring small areas, since pitfall collections can be made with approximately 10 m spacing between traps (Samu and Lövei 1995; Bess et al. 2002). Pitfall traps also collect many other arthropod taxa, amphibians, and small mammals, though to a much lesser extent.

Birds are model organisms for monitoring studies and can be strong indicators of environmental condition (Bibby et al. 2000). There are several characteristics that make them well-suited as a group to studying ecological processes; notably their widespread distribution, breadth of habitat use, ease of detectability (highly visible and/or highly vocal), extensive pre-existing literature on life-history characteristics, habitat associations, demographic rates, and public appeal (Bibby et al. 2000; Ralph et al. 1995). Their relatively high diversity and niche partitioning by habitat or foraging guilds is also beneficial when comparing different habitat types within limited geographic areas. Bird populations are responsive to environmental changes and can thus be used as indicators of the ecological condition of an area (Furness and Greenwood 1993; Morrison 1986).

The focal taxa and general methods align with those monitored under CLBMON-11A since 2014 (Wood et al. 2015, 2016, 2017, and 2018), with the exception that songbird point counts were not performed in upland reference habitats in 2018.

3.2 Terrestrial Arthropods

Terrestrial arthropods (spiders and beetles) were sampled using pitfall traps and the methods are outlined in previous reports (Wood et al. 2015, 2016, 2017, 2018). Methods were consistent with those described by the Resources Inventory Committee (1998b) and Biological Survey of Canada (Marshall et al. 1994).

3.2.1 Sampling Period

Terrestrial arthropods were sampled in two collection periods at Canoe Reach and Bush Arm (Table 3-1). The collection periods were run with a short period of trap closure between trapping sessions, with a similar sampling period for each site. The date and time (hh:mm) of setup and collection were recorded for each trap so that trap-hours could be calculated to standardize abundance. Trap disturbance resulting in loss of sample (e.g., reservoir inundation or animal disturbance) was recorded to account for the reduced sampling effort in data standardizations. Disturbance causing loss of replicates is noted in Table 4-1.

Table 3-1: Sampling period duration for terrestrial arthropods in 2018. Number of days (24-hour period) is given for pitfall sampling at each location (not adjusted for functional trap time). Site codes are listed in Table 2-2.

Reach	Site	Collection Period	Session Dates	Session Duration (days)
Bush Arm	BAC-N	1	14Jun - 19Jun	5.0
		2	28Jun - 2Jul	4.0
	BAC-S	1	14Jun - 19Jun	5.2
		2	28Jun - 2Jul	4.2
	GDF	1	14Jun - 19Jun	5.2
		2	28Jun - 2Jul	4.1
	KM88	1	15Jun - 19Jun	3.9
		2	27Jun - 2Jul	5.0
Canoe Reach	P12	1	11Jun - 17Jun	5.7
		2	25Jun - 30Jun	5.0
	VP-N	1	11Jun - 17Jun	6.0
		2	25Jun - 30Jun	4.9
	YJ	1	12Jun - 17Jun	5.0
		2	25Jun - 30Jun	4.9

3.2.2 Sampling Methodology

Sampling methods aligned with previous years, where each pitfall trap array contained three pitfall traps spaced ~ 1 m apart, which were pooled in the field as one sample (Figure 3-1). Polygons at each Canoe Reach site (VP-N, YJ, and P12) were sampled with nine pitfall trap arrays in each treatment and control polygon (see Map 7-1 for experimental design at P12; maps for VP-N and YJ are available in Wood et al. 2018). Polygons at each Bush Arm site were sampled by five randomly located pitfall trap arrays. Arrays were spaced no closer than 10 m such that they could be considered independent samples (Samu and Lövei 1995; and Bess et al. 2002).

Figure 3-1: Left: close-up of a functioning pitfall trap equipped with propylene glycol. Right: typical array containing three pitfall trap subsamples (indicated by yellow dashed arrows).

3.2.3 Taxonomy and Natural History

Spider specimens were identified to species, where possible, by a local expert (Dr. Robb Bennett, Ph.D., Research Associate and Darren Copley, Royal British Columbia Museum). Beetle identifications were provided by Charlene Wood (LGL Limited). Dissections of spider and beetle specimens were often necessary to examine traits in genitalia and determine species. Beetle classification was based on numerous taxonomic works, including, but not limited to: Arnett and Thomas (2001), Goulet (1983), Lindroth (1961-1969), and Pearson et al. (2006). The entomology collection at the Royal B.C. Museum (RBCM) in Victoria, British Columbia, was used as a reference for species identifications. In rare cases where a beetle species could not confidently be determined to species, a morphospecies number was assigned. Taxonomies are based on the most recent revision available. However, we would like to acknowledge that species concepts for *Loricera decempunctata* and *L. pilicornis* need revision (Dr. David Kavanaugh, California Academy of Science, pers.comm. to Charlene Wood, Feb 4, 2019). Despite this, both were retained as valid species detected in our study area. Spider and beetle specimens were curated according to museum standards, and a reference collection was deposited at the RBCM.

We classified spider feeding guilds (Uetz et al. 1999), spider adventive status (world spider catalogue), beetle adventive status (Bousquet et al. 2013), and spider and beetle habitat associations (Carcamo et al. 2014 Larochelle and Larivière. 2001; Larochelle and Larivière. 2003).

3.3 Breeding Birds

Breeding birds in treatment and control areas of the drawdown zone were sampled using line transect surveys and nest searching. Surveys followed methods consistent since 2015 (Wood et al. 2016, 2017, 2018). Methods were consistent with standards described by the Resources Inventory Committee (1999) and other protocols (i.e., Bibby et al. 2000; Ralph et al. 1995).

3.3.1 Sampling Period

Songbirds and other breeding birds (e.g., shorebirds) were surveyed within the regional nesting period identified by Environment Canada (EC 2019), and in concordance with provincial standards (RIC 1999). The regional nesting period identifies the time of year with the highest expected number of breeding bird species for a region, which occurs between mid-late May to mid-July for our study area. Line transect data was collected between May 28 and July 10, with most data collected from mid-June to early July. Each transect was visited two (2015, 2018) or three (2016, 2017) times per year. Surveys began at sunrise and ended within about four hours, to capture the most stable song period (Ralph et al. 1995). Surveys only occurred under favourable conditions (i.e., no heavy wind or precipitation; RIC 1999) to minimize variability in bird behaviour and detection rates related to weather.

3.3.2 Sampling Methodology

Line transects were placed within treatment and control areas of the drawdown zone, located relatively close to the shoreline and generally oriented parallel to the reservoir. All line transect surveys were conducted in a straight line between predetermined start and end locations, spaced 100 m apart. The observer traveled the length of the 100 m transect at a speed close to 1.2 km/h, which translated into a five-minute survey (Bibby et al. 2000). All birds detected were recorded and assigned two associated distances: the distance travelled along the transect (0-100 m), and the distance band perpendicular to the transect centreline (0-10 m, 11-25 m, 26-50 m, >50). At each transect the station details and current environmental conditions (e.g., temperature, wind speed) were recorded. Associated bird data recorded included the species, age, sex, location, and detection details (e.g., song, call, flyover).

At both Canoe Reach and Bush Arm, all treatment and control plots were searched for nesting evidence over the same dates as line transect surveys. Information on discovered nests included species, behaviour, nest stage, nest substrate, number of eggs/offspring, and UTM coordinates. Nests were flagged from a minimum of 10 m away and the distance, bearing and nest substrate was written on the flag (Thomas et al. 1997). Active nests were revisited upon subsequent surveys when possible to assess nest status (success/failure). As CLBMON-11A was not designed as a nest productivity study, our results were supplemented with data provided by Cooper, Beauchesne and Associates, Ltd. Bird nest data provided by CBA were collected under CLBMON-36 (e.g., van Oort 2016; see Wood et al. 2018).

3.4 Environmental Conditions

Cover data were compiled to associate substrate and vegetation differences with arthropod communities and assess changes over time. Data were collected during the period arthropod pitfall trapping occurred in 2014 to 2018. In 2014, substrate data were collected from CLBMON-9 (Hawkes and Miller 2016). In 2015 to 2018, covers were estimated with similar methods in three 1 m x 1 m quadrats within each study polygon. Percent cover of the following were recorded at each quadrat: canopy cover, live organic matter (LOM), litter, coarse woody debris, fine woody debris, rock, mineral soil, mixed soil, peat, lichen, moss, and water.

Temperature and Relative Humidity data were collected during arthropod sampling to assess changes in microclimate of treatments overtime. Onset® HOBO® data loggers (U23-002 HOBO Pro v2 External T/RH) were used to measure percent relative humidity and temperature over the period encompassing arthropod surveys. One logger was deployed at the approximate center of each plot in Canoe Reach and Bush Arm. Data loggers were held in place at the surface of the soil by attaching the base to a pin flag. Locations of all deployed data loggers are shown in maps within previous reports and Appendix 8.

4.0 DATA SETS

Below, we provide a summary of the sampling design including temporal replication, number of sites surveyed, and number of samples or sampling points that are comprised in our overall datasets. Despite the wood removal that was conducted in the VP-N control plot in 2018 (Figure 2-6), this sampling location was retained as a “control” in all analyses of songbird and arthropod data. The treatment of this control plot will be considered during discussion of results for 2018 at VP-N C.

We exclude mention of sites and samples that have been removed from study prior to 2018. For example, we do not comment on sites that were dropped from monitoring due to lack of revegetation success (Appendix 1: all locations monitored prior to 2014), lack of treatment application (i.e., Chatter Creek, Hope Creek, Goodfellow Creek T/C), or with severely compromised study design due to lack of controls (i.e., Valemount Peatland South, Packsaddle Creek North, and Packsaddle Creek South). Nor do we retain data from non-paired drawdown zone areas (previously “DDZ”). Please refer to previous annual reports and Appendix 1 for further details.

4.1 Terrestrial Arthropods

Data Set 1 – Pitfall trap data

This data set was created to assess ground-dwelling arthropod abundance, richness, and composition among sites, habitat types, and years. The data set includes abundance data from pitfall traps collected in years 2014 to 2018, comprising 658 trap samples from 6 sites (Table 4-1). Replication varied by reach, site, habitat type, and year, due to disturbance and minor annual adjustments to the pitfall trapping study design (see Wood et al. 2015, 2016, 2017, 2018). The active trap time (in days) were calculated from trap setup and trap collection dates and times to standardize arthropod abundance for effort. Species-level identifications were provided for all mature adult Araneae and adult ground beetles (family Carabidae). Family level identifications were provided for all adult Coleoptera. Immature spiders and beetles are counted in the data set but were excluded for all analyses and data summaries.

Table 4-1: Number of pitfall trap samples included in data set 1 for terrestrial arthropod analyses. One sample was collected from each pitfall trap array (3 trap subsamples). Samples are listed for collection period 1 (typically early to mid-June) and collection period 2 (typically mid-June to early July). Habitat codes: T = treatment, C = control, R = reference. Site codes are listed in Table 2-2. '-' indicates no sample; *disturbance**.**

Reach	Site	Year	Period 1 T	Period 1 C	Period 1 R	Period 2 T	Period 2 C	Period 2 R	Total samples
Bush Arm	BAC-N	2015	3	3	-	3	3	-	12
		2016	8	5	-	5	5	-	23
		2017	5	5	-	5	5	-	20
		2018	5	5	-	5	5	-	20
	BAC-S	2015	3	3	-	3	3	-	12
		2016	8	5	-	5	5	-	23
		2017	5	5	-	5	5	-	20
		2018	5	5	-	5	5	-	20
	GDF	2015	-	-	3	-	-	3	6
		2016	-	-	5	-	-	4*	9
		2017	-	-	3	-	-	3	6
		2018	-	-	3	-	-	3	6
	KM88	2018	15	15	2*	15	15	3	65
Canoe Reach	VP-N	2014	-	-	-	9	9	9	27
		2016	9	9	9	9	9	9	54
		2017	9	9	3	9	9	3	42
		2018	9	9	3	9	9	3	42
	YJ	2014	9	9	9	9	9	9	54
		2015	9	9	9	9	9	9	54
		2016	9	9	-	9	9	-	36
		2017	4*	9	3	1*	7*	3	27
		2018	8	9	3	8*	9	1*	38
	P12	2018	9	9	3	9	9	3	42
Total samples			132	132	58	132	139	65	658

Note: sampling at KM88 occurred in three treatment polygons (TU1, TU3, TU5) and three control polygons (CU1, CU2, CU3) each equipped with 5 pitfall trap arrays (see Map 7-2).

4.2 Breeding Birds

Data Set 2 – Songbird Point Counts

No songbird point count surveys were conducted at reference plots in 2018 as information on the reference condition had already been documented (e.g., Wood et al. 2018) and shown to be distinct from drawdown zone areas. Surveys in 2018 focused on treatment vs. control conditions within the drawdown zone. Please refer to Wood et al. (2018) for details of past surveys.

Data Set 3 – Bird Line Transects

This dataset was created to assess breeding bird abundance, richness, diversity and composition among sites, habitat types, and years. The dataset includes abundance data from line transects between 2015 and 2018. Line transects were completed at 12 sites. Not all sites were surveyed in each year, and for this report only those six sites which had annual sampling to 2018, or sampling only in 2018, are included (Table 4-2). Each site had one transect within a control area, and one transect within a revegetation or physical works treatment area. The only exception to this was KM88 that had three control polygons and three treatment areas, each with one line transect (Table 4-2).

Line transects were repeated one to three times within a year. This resulted in 95 rows of data, each consisting of one visit to a line transect in a given year for a given site. Note that due to a large exodus of toadlets at the treatment transect at VP-N in 2015 (visit 2), no survey was conducted for animal welfare reasons. The data were further subdivided as described in Appendix 5.

Table 4-2: Number of line transects included in data set 2 for bird analyses. Surveys are listed for each visit. Habitat codes: T = treatment, C = control. Site codes are listed in Table 2-2. '-' indicates no sample.

			Visit 1		Visit 2		Visit 3		Total
Reach	**Site**	**Year**	**T**	**C**	**T**	**C**	**T**	**C**	
Bush Arm	BAC-N	2015	1	1	1	1	-	-	4
		2016	1	1	1	1	1	1	6
		2017	1	1	1	1	1	1	6
		2018	1	1	1	1	-	-	4
	BAC-S	2015	1	1	1	1	-	-	4
		2016	1	1	1	1	1	1	6
		2017	1	1	1	1	1	1	6
		2018	1	1	1	1	-	-	4
	KM88	2018	3	3	3	3	-	-	12
Canoe Reach	VP-N	2015	1	1	-*	1	-	-	3
		2016	1	1	1	1	1	1	6
		2017	1	1	1	1	1	1	6
		2018	1	1	1	1	-	-	4
	YJ	2015	1	1	1	1	-	-	4
		2016	1	1	1	1	1	1	6
		2017	1	1	1	1	1	1	6
		2018	1	1	1	1	-	-	4
	P12	2018	1	1	1	1	-	-	4
Total Line Transects			***20***	***20***	***19***	***20***	***8***	***8***	***95***

Data Set 4 – Bird Nesting

This dataset was created to assess nesting locations for study sites in which line transect data was analysed. The dataset includes nest data from searches completed between 2015 and 2018 by LGL Limited (n = 23) and supplemented by point locations for nests located by Cooper, Beauchesne and Associates between 2008 and 2017 (n = 77). Nest searching activity by LGL Limited was focused, but not constrained, by general polygon boundaries denoting treatment and control areas. Nest searching by CBA was completed under CLBMON-36 and was irrespective of the CLBMON-11A study areas (see Wood et al. 2018). We thus constrained CBA nest data to those within 100 m of our study polygons. Nest data are presented qualitatively as the nest search effort by year and area (e.g., elevation bands targeted for search effort) are not known.

4.3 Environmental Conditions

Data Set 5 – Soil surface Temperature and Relative Humidity

Hourly temperature and relative humidity (RH) data were compiled to associate with arthropod communities and assess changes over time. Data were collected during the period arthropod pitfall trapping occurred in 2014 to 2018. One logger was deployed in each polygon within each site as outlined in Table 4-3. There were 24625 individual data points for each of temperature and RH.

Table 4-3: Number of hourly data points recorded from Relative Humidity and Temperature data loggers in each Reach, Site, Year, and Habitat type. T= Treatment, C= Control, R= Reference (non-reservoir control).

REACH	SITE	Year	T	C	R	*Total*
Bush Arm	BAC-N	2015	402	402	402	*1206*
		2016	312	312	312	*936*
		2017	421	421	421	*1263*
		2018	432	432	430	*1294*
	BAC-S	2016	312	312		*624*
		2017	417	417		*834*
		2018	438	436		*874*
	GDF	2015			402	*402*
		2016			312	*312*
		2017			417	*417*
		2018			436	*436*
	KM88	2018	406	406	405	*1217*
Canoe Reach	P12	2018	432	432	433	*1297*
	VP-N	2014	828	828	828	*2484*
		2015	361	361	361	*1083*
		2016	328	328	328	*984*
		2017	421	421	421	*1263*
		2018	457	456	456	*1369*
	YJ	2014	828	828	828	*2484*
		2015	361	361	361	*1083*
		2016	312	312		*624*
		2017		420	420	*840*
		2018	435	432	432	*1299*
Total			*7903*	*8317*	*8405*	*24625*

Data Set 6 – Live Organic Matter, Substrate, and Canopy Cover

Live organic matter and soil substrates was classified within the quadrats in vegetation transects by estimating per cent cover of various substrate classes. These estimates were collected under CLBMON-9 for the 2014 monitoring period (Hawkes and Miller 2016). In 2015 to 2018, covers were estimated with consistent methods in three 1 m x 1 m quadrats within each study polygon. Percent cover of the following substrate classes were derived: live organic matter (LOM), dead organic matter (DOM; litter), wood (coarse and fine woody debris), rock, mineral soil, moss, and water. Canopy cover was estimated at three locations in each polygon, by estimating the overhead cover of vegetation (trees) intercepting the sky. A total of 150 plots were conducted to produce this data (three plots in each polygon; Table 4-4).

Table 4-4: Number of canopy cover, vegetation cover, and substrate cover plots conducted in each year of arthropod monitoring by site and habitat type.

Reach	Site	YEAR	T	C	R	*Total*
Bush Arm	BAC-N	2015	3	3		***6***
		2016	3	3		***6***
		2017	3	3		***6***
		2018	3	3		***6***
	BAC-S	2015	3	3		***6***
		2016	3	3		***6***
		2017	3	3		***6***
		2018	3	3		***6***
	GDF	2015			3	***3***
		2016			3	***3***
		2017			3	***3***
		2018			3	***3***
	KM88	2018	9	9	3	***21***
Canoe Reach	P12	2018	3	3	3	***9***
	VP-N	2016	3	3	3	***9***
		2017	3	3	3	***9***
		2018	3	3	3	***9***
	YJ	2015	3	3	3	***9***
		2016	3	3		***6***
		2017	3	3	3	***9***
		2018	3	3	3	***9***
Total			***57***	***57***	***36***	***150***

5.0 MANAGEMENT QUESTIONS

We summarize below our ability to address each of the management questions (MQ) as per the revised Terms of Reference (TOR; BC Hydro 2017). Cumulative data analysis conducted after the final monitoring year (2018) is primarily used to support each management question. These original analyses are appended to this report as requested in a template provided by BC Hydro.

In our response to answering the management questions, the meaning of the word “revegetation” is extended to include physical works trials as well as revegetation prescriptions (to better align with the current focus of CLBMON-11A).

In addition to reporting relevant knowledge to address each MQ, methodological challenges, associated knowledge gaps, and opportunities for future monitoring are discussed.

MQ1: How effective are the revegetation prescriptions at enhancing and increasing the drawdown zone habitat use of wildlife such as birds and amphibians?

Habitat Enhancement (Vegetation)

All physical works and wood removal treatment plots increased in vegetation cover since treatment.

Valemount Peatland (VP; pond clearing, wood removal, and log boom) was notable in rapid recovery of vegetation and has steadily increased in vegetation cover and diversity since treatment application (Hawkes and Miller 2016; Miller and Hawkes 2019; Figure 5-1).

The constructed mounds at Bush Arm Causeway (BAC-S and BAC-N) have been colonized by a variety of plant species and have ~46% of live stakes surviving since planting in 2014 (Miller and Hawkes 2019; Figure 5-2).

The three Bush Causeway ponds (BAC-N) that were cleared of wood debris and enclosed with a log-boom are exhibiting vigorous growth of both riparian and aquatic vegetation since 2015 (Figure 5-3).

Figure 5-1: VP-N treatment photos showing vegetation growth over time from the initial wood clearing year (2014) through 2018 (photos not available for 2015).

Figure 5-2: Examples of transplant growth and naturally regenerating vegetation on the mounds created at BAC-S (top) and BAC-N (bottom).

Figure 5-3: Photo documentation of pre-treatment wood cover over the wetland at BAC-N in 2015 (top) and post-treatment cleared pond in 2017 (bottom), with vegetation growth along the pond perimeter.

Breeding Birds

There has been only limited, site-specific evidence supporting an increase in drawdown utilization by birds due to revegetation or physical works prescriptions. Overall, there was no consistent pattern in how treatment type affected bird utilization.

Results were often conflicting; for example, the total number of bird species was higher in the treatments, but abundances were higher in controls. Main bird species utilizing the drawdown zone included open-country, ground- or shrub-nesting passerines (e.g., Savannah Sparrow, Lincoln's Sparrow, Common Yellowthroat, Chipping Sparrow and Clay-colored Sparrow), and ground-nesting shorebirds (mainly Spotted Sandpiper, but Killdeer in some sites). These species were most abundant in both control and treatment areas, and often in relatively equal proportions, suggesting no effect of the treatment, regardless of the treatment type (i.e., revegetation, woody debris removal, or debris mounding). Likewise, bird nest locations showed no trend relative to control and treatment areas, and few nests were found each year.

The one site where revegetation may have improved bird use was at KM88 in Bush Arm. At this site, Savannah Sparrow was three times more abundant in treatment than control areas. As Savannah Sparrows were recorded from all treatment transects at the site, this result was not due to a single, highly productive line transect location. However, overall the bird community at that site was depauperate (only three species recorded in total), and monitoring was limited to one year (2018). Due to these limitations, and the possibility that the trend was spurious, we consider this result tentative.

Amphibians

Amphibians were monitored (under CLBMON-37 and CLBMON-58) at two locations associated with physical works: Valemount Peatland and the Bush Arm Causeway. The comprehensive report for CLBMON-37 is forthcoming in 2019, which will summarize amphibian and reptile life history and habitat use for Kinbasket and Arrow Lakes Reservoirs.

As reported by Hawkes (2017), there is evidence that clearing ponds of wood debris in the drawdown zone improves breeding habitat suitability for amphibians.

The cleared wetland at Valemount Peatland North was the second most active breeding site in Canoe Reach, after Pond 12 in May 2014. Western Toad (*Anaxyrus boreas*) and Columbia Spotted Frog (*Rana luteiventris*) were both observed mating and laying eggs during day and night surveys at this pond (Figure 5-4). Similarly, two of the three ponds northwest of Bush Arm Causeway that were choked with wood and devoid of amphibians prior to clearing, were used by Western Toad as breeding habitat immediately the next spring (Figure 5-5).

Figure 5-4: Photo of the treatment area at VP-N in 2014 with western toad tadpoles and metamorphs developing in the cleared wetland (initial post-treatment breeding season).

Figure 5-5: Photo documentation of western toad tadpoles in cleared pond habitat at BAC-N in June 2016 (one year post-treatment).

Challenges and Opportunities

This management question cannot be fully addressed. This is due to few years of post-treatment monitoring and lower reservoir elevations during the post-treatment monitoring period relative to pre-treatment years. Kinbasket Reservoir has not been filled since the physical works were completed. From 2014 to 2018, the reservoir has operated under its maximum. This prevents us from fully assessing how certain treatment techniques, such as log boom installation, might operate to preserve habitat integrity and species richness or diversity in an area following maximum inundation.

While evidence of enhanced pond breeding habitat for amphibians has been reported, our wildlife monitoring has been limited to one to four years of post-treatment data. While promising, it is unclear whether the removal of wood from ponds in the drawdown zone will result in long-term amphibian habitat creation/restoration in the drawdown zone.

The short time since application and short duration of post-treatment monitoring may be insufficient for assessing vegetation and arthropod responses which birds could exploit in a way that produces measurable differences. For some species, like Savannah Sparrow, increased use is expected to occur with increased vegetation cover and area devoid of wood debris. However, we do not know if areas cleared of wood will develop vegetation communities that will benefit bird populations in the long-term. It is also possible that creating suitable habitat for birds in the drawdown zone could create an ecological trap due to inundation from reservoir operations, though results on this from the Arrow Lakes Reservoir are mixed (van Oort et al. 2015; Hepp et al. 2018).

Site-specific differences in ground substrate (moisture and amount of organic matter), vegetation types present (shrubs, herbs), proximity to forest edge, time since treatment, treatment type (log boom or mounding), and inadvertent re-treatment or treating of controls confound our ability to test for effects. Because study areas required independent assessments, our statistical power for detecting differences in birds was also limited. Inter-annual variation in species richness and diversity (e.g., due to inter-year differences in weather, reservoir operations, disturbances, predation, etc.) may further mask trends related to revegetation prescription effectiveness, but more importantly the lack of replication, small areas of revegetation prescriptions, and low bird density in the drawdown zone limits our ability to make inferences.

The lack of consistent differences in treatment and control areas are additionally confounded by a spatial scale mismatch between birds and the extent of revegetation and other habitat prescriptions. The treatment areas as applied are, in many cases, smaller than the home range size of species that may utilize them [e.g., Savannah Sparrow can have breeding territories >1 ha (Wheelwright and Rising 2008)]. The small sizes of the revegetation and physical works prescriptions, lack of replication and stratified treatments, short time scale of pre- and post-physical works monitoring, and inconsistencies in the CLBWORKS-1, CLBWORKS-2, and CLBWORKS-16 methodology make it difficult to achieve more than speculation regarding the program's effect on bird utilization of those habitats.

It is too early to make any conclusions about the treatment application at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. We expect the wildlife to respond as this area becomes established with vegetation and changes over time. Further years of monitoring are required.

In addition, some treatments implemented may be of benefit to wildlife not monitored under CLBMON-11A since 2014. For example, debris mounds have the potential to increase local mammal populations (e.g., small mammal and mesocarnivore populations; Sullivan et al. 2017). We have consistently collected small mammals as bycatch in our pitfall trap samples in study sites (Table 7-17), which shows that these species are using the treatment and control plots in the drawdown zone. The debris mounds at Bush Arm Causeway North and South may provide a benefit to these species.

MQ2: To what extent does revegetation increase the availability of invertebrate prey (e.g. arthropods) in the food chain for birds and amphibians?

Arthropod Relative Abundance

Overall there are no trends indicating that relative abundance of ground-dwelling spiders and beetles were consistently greater in treated areas than in control areas among the sites sampled (Appendix 2). In most cases, relative abundance of these taxa increased slightly in both treatment and control areas after physical works and/or revegetation prescription implementation.

Challenges and Opportunities

It should be noted that we did not monitor all arthropod taxa that might contribute to the diet of wildlife (e.g., aerial insects, caterpillars, grasshoppers) and we are not testing the consumption of arthropods or the diet preferences of birds, amphibians, and small mammals. We monitored ground-dwelling spiders and beetles because they are effective focal taxa for monitoring changes in terrestrial habitats. They are easily and simultaneously sampled, comprise a large proportion of arthropod abundance and diversity, occur in almost all terrestrial habitats, include both specialist and generalist species, can be studied across any gradient of habitat change, and respond to both fine-scale and landscape-scale environmental changes, including changes to vegetation cover, structure, and composition.

Our data of ground-dwelling arthropods (spiders and beetles) show that abundance patterns varied between years, sites, and treatments and seemed unrelated to treatment. Similar patterns were often observed in controls or explained by inter-annual variation. Inter-annual variability in the relative abundance of spiders and beetles were determined from upland reference samples (Figure 7-6, Figure 7-11, Figure 7-14). The large amount of variation in these samples suggests that changes in arthropod abundance in the drawdown zone may be further obscured by other factors impossible to control between years (see next paragraph). Furthermore, the lack of pre-treatment sampling at most study sites limits our ability to infer any connection between arthropod responses and a treatment effect, especially considering prominent within-site ecosite or soil composition differences.

While there was an increase in vegetation cover in all physical works (wood removal) areas, a similar increase was also found in control plots. Thus, relationships between arthropod abundance and physical works trials may be obscured by (1) the effect of low reservoir elevations (since 2013), allowing for vegetation recovery in the upper elevation bands of the drawdown zone and (2) ecosite differences (soil, hydrology, topography) between treatment and control areas at each site, providing differences in growing conditions for vegetation regeneration and establishment (e.g., Yellowjacket Creek treatment substrate has greater mineral soil content and lower soil moisture relative to the Yellowjacket Creek control).

It is too early to make any conclusions about the effectiveness of treatment at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. We expect wildlife to respond as vegetation becomes established and changes over time. Further years of monitoring would be required to asses the effectiveness of the prescription.

MQ3: How do revegetation prescriptions affect the diversity and abundance of arthropods, amphibians and birds?

Arthropod Diversity and Abundance

Because of the varied nature of our results, short duration of monitoring after treatment application, and the lack of pre-treatment data at VP-N, YJ, KM88, and P12, we can not make any conclusions about the success of treatment prescriptions for promoting arthropod diversity and richness. We discussed how treatment affected arthropod abundance in the previous section (see MQ2).

There was no consistent pattern in how treatment affected arthropod diversity and richness (Table 7-2 in Appendix 7). There were several cases in which treatment had no observable effect on arthropod diversity. In the cases where it did affect diversity or richness, the effect was always negative for spiders (species diversity and/or richness were greater in control than treatment areas) and positive for carabid beetles (species diversity and/or richness were greater in treatment than control areas). See Appendix 3 for analyses and more detailed discussion.

We found that windrow or mound treatments (BAC-S and BAC-N) had a largely negative effect on spider species diversity and richness. VP-N and P12 showed a potentially positive effect of treatment on carabid beetle diversity. There was no effect of treatment on spider diversity or richness at those sites, but there was a negative effect of treatment on spider diversity at YJ (where wood removal was repeated, increasing site disturbance). Finally, the revegetated site (KM88) showed no effect of revegetation on carabid species richness or diversity, and a negative effect of revegetation treatment on spider species richness and diversity.

Wood et al. (2016) predicted a positive effect of revegetation on select spider guilds. An increase in dominance of ambush hunters, sheet/funnel-weavers, and space-web weavers was expected as herb and shrub cover increased. We noted increased proportional abundance of ambushers in VP-N treatment overtime and an increase in guild diversity at this plot. This included the detection of fishing spiders (e.g., *Dolomedes triton*) in 2018, which is reflective of the ecological niches available at this site in 2018, relative to 2014. YJ treatment likewise steadily increased in ambushers, sheet/funnel-weavers, and space-web weavers from 2014 to 2016, but following retreatment of this site in 2017, the guild composition was simplified to nearly all ground-running spiders (which are indicative of low vegetation structure).

It is too early to make any conclusions about the treatment application at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. As this area becomes established with vegetation and changes over time, we expect the ground-dwelling arthropod communities to reflect changes in vegetation structure and cover (e.g., species turnover with decreased dominance of bare ground species; spider guild and species composition changes). Further years of monitoring would be required.

Amphibian Diversity and Abundance

Evidence from CLBMON-37 and Hawkes et al. (2017) suggests that amphibians continually use the drawdown zone habitats (i.e., both treatment and control areas). Wetlands cleared of wood debris (i.e., at VP-N and BAC-N treatment areas) were effective for supporting increased amphibian breeding activity (CLBWORKS-1; Hawkes 2017). Incidental bycatch of amphibians in pitfall traps confirms amphibian presence at many of our sites (Appendix 9). See MQ1 for a more in-depth explanation of amphibian use of treatment areas.

Bird Diversity and Abundance

There was no consistent pattern in how treatment type affected bird diversity and abundance (see Appendix 5 and Appendix 6). In most cases there was no observable effect of treatment. In general, if revegetation was successful, we expected treatments to have greater species richness, relative abundances, and nesting opportunities. While some sites had a greater number of species and/or individuals in treatment areas, the magnitude of such change was often very low (e.g., 12 observations vs. 10 observation), or inconsistent across treatments (e.g., VP-N had a greater number of species in the treatment transect but fewer individuals). There was also no indication that treatments were more likely than controls to have nesting birds. Limited data made interpretation for some sites difficult.

There was no clear effect of mound treatments on bird species richness or diversity at BAC-N and BAC-S. Results were similarly non-significant for sites with woody debris removed. VP-N showed a higher species count but lower number of individual counts in treatment transects. Treatment and control transects at P12 appeared similar overall, and at YJ there was an overall decline in richness and diversity. In most cases differences in species could be explained by outside factors, such as location of transect and proximity to adjacent habitat, as well as the substrate and vegetation associated with the site and transect.

Revegetation at KM88 had the most promising results. Bird species (both the type and number of species) were similar between the two transect types, but treatments had twice as many observations. However, these data are limited to a single year of post-sampling, and the bird community was depauperate.

Challenges and Opportunities

Differences in ground substrate (moisture and amount of organic matter), vegetation types present (shrubs, herbs), proximity to forest edge, time since treatment, and treatment type (log boom or mounding) confound the assessment of differences between control and treatment plots both between sites (when considered in aggregate), as well as within sites (by obscuring the direct effects of treatments).

A limitation in interpreting results from P12, KM88, and (to a lesser extent) YJ is the short timescale since treatment (or retreatment) of study plots. KM88 was treated in 2013, but 2018 is the first year of post-treatment wildlife sampling.

Re-treatment of YJ in 2017 and treatment of P12 in 2018 also limit post-treatment sampling at both sites. This makes it difficult to understand whether the response of wildlife using the habitat was due to the type of treatment, or if it was a more general response to habitat disturbance by the initial treatment application. It also limits our ability to control for yearly fluctuations in local communities, which there was evidence of in reference samples at other sites.

Treatment effectiveness should be considered in the context of reservoir levels as inundation may influence establishment of vegetation, wave erosion, and stability of physical works features (e.g., mounds, log booms). The treatment and control plots are all situated in the uppermost extent of the reservoir drawdown zone (~751.6 to 754 m ASL) and are inundated when reservoir levels reach or exceed these elevations. The two years prior to wood removal at Canoe Reach (2012-2013) had the highest reservoir levels since 1997 (Figure 2-1), which likely reduced vegetation in these areas. During the post-treatment monitoring period (2014-2018) reservoir levels have been considerably lower and have not inundated the treatment areas. Enns et al. (2009) suggested that vegetation would increase in the drawdown zone given inundation-free periods in the spring and fall. These conditions were met for the post-treatment monitoring period, and in turn, vegetation cover increased in both treatment and control drawdown zone plots. Given the low reservoir levels favoured natural revegetation of these drawdown zone areas, it is difficult to ascertain the extent to which treatment application enhanced the establishment and growth of vegetation.

Because the intent of log booms (e.g., at VP-N and BAC-N) is to exclude wood debris during inundation events, the lack of such events during this study period prevents us from fully understanding how log booms operate to preserve habitat integrity and species richness or diversity during peak reservoir conditions. Likewise, the function of mounds to increase topographic heterogeneity and increase establishment of a diversity of plant species depends on the permanence of these structures in the drawdown zone treatment areas (BAC-N and BAC-S). However, without inundation of these treatment areas during the monitoring period, we are unable to evaluate their function and stability under reservoir elevations attaining the normal operating maximum. It would be informative to gain an understanding of how reservoir inundation causes these structures to settle/shift and assess whether the vigor and survival of vegetation on mounds overtime.

MQ4: Which revegetation method is the most effective at enhancing or increasing the utilization of wildlife habitat in the drawdown zone?

Habitat Enhancement and Utilization

No single treatment stood out as being the most effective at enhancing or increasing the utilization of wildlife habitat in the drawdown zone. Of the sites monitored in 2018, KM88 (revegetated with sedge plugs) showed the most potential in terms of increased wildlife use (i.e. bird presence, see MQ1 and MQ3), but this was not related to arthropod abundance at that site. These results are tentative given the limited nature of the data (KM88 was only sampled once post-treatment), and revegetation efforts were not similarly successful at encouraging invertebrate prey in the area.

Results of mounds (BAC-S, BAC-N) and wood removal (VP-N, YJ, P12, BAC-N and BAC-S) treatments were mixed. There was no convincing evidence that either method increased bird utilization of the sites they were applied at (see MQ1). In some sites there was an increase in one arthropod group in treatment areas (i.e. carabid beetles, see MQ2 and MQ3). However, the trend was reversed for spiders. The site-by-site nature of the different response to treatment types emphasizes how site-specific factors (such as substrate characteristics, environmental conditions, and/or proximity to other habitat not measured in the study) may play a more critical role in determining wildlife use than treatment method alone.

While we did not specifically address amphibian use of enhanced areas in this study, the comprehensive 2019 report on amphibian habitat use in the Kinbasket and Arrow Lakes Reservoirs (CLBMON-37) may elucidate any amphibian preference for certain habitats. Incidental observations of increased amphibian activity in the first breeding period after wood removal from drawdown zone ponds at VP-N and BAC-N suggest at least short-term benefits of this treatment for amphibians (Wood et al. 2018).

Based on the results obtained thus far for CLBMON-11A, it appears that conventional methods of revegetation were ineffective at enhancing and increasing the utilization of wildlife habitat in the drawdown zone. As found in CLBMON-9 (based on four years of results), only the sedge plug revegetation treatment had any establishment success (live stake treatments did not survive), but even then only in very limited areas (e.g., KM88; Hawkes et al. 2013).

Woody debris removal has the potential to enhance and increase the utilization of wildlife habitat in the drawdown zone, but more years are needed to determine the effectiveness of this approach. Many treatment sites were rapidly and naturally recolonized by plant species following wood debris removal. In addition, there was an increase in wetland vegetation in ponds that were previously devoid of macrophytes after being cleared of wood debris (Hawkes 2016).

Based on the results obtained thus far for CLBMON-11A, it appears that woody debris removal has the potential to enhance and increase the utilization of wildlife habitat in the drawdown zone, particularly when treatment plots include wood-covered wetlands. Further protection is likely offered when these treated areas are fitted with an enclosure (e.g., log boom) to exclude further wood deposition. Results from vegetation surveys (CLBMON-9) suggest that treatment sites are rapidly and naturally recolonized by plant species. The longevity of vegetation on these plots is precarious due to the inevitable re-accumulation of wood each year in sites not protected by log boom installation.

In addition, we have not been able to assess whether vegetation will be sustained in years where the reservoir reaches full pool, as all monitoring years since wood removal, mound creation, and log boom installation have not been monitored after reservoir levels at the maximum. Thus, any positive effects observed in early years post-treatment may be short-lived, given long-term uncertainty in wood accumulation and reservoir impacts on vegetation in the upper elevation bands under study.

Challenges and Opportunities

Challenges to understanding the effectiveness of habitat enhancement are similar to those listed for MQ1, MQ2, MQ3, and discussion sections in each of the data chapter appendices. These include confounding effects of within-site variables, low reservoir elevations during the post-treatment monitoring period, the spatial mis-match between certain focal taxa (such as birds) and the size of treated areas, and the lack of pre-treatment data for most study areas.

An especially important consideration in the context of habitat enhancement effectiveness is the temporal limitation of this study- the duration of monitoring since implementation was still relatively short-term (1-5 years) for most sites. It is likely that some habitat prescriptions will be more effective long-term as vegetation, arthropods, birds, and amphibians are given time to respond to changes.

In particular, the large wetland in the southern portion of Valemount Peatland (“Pond 12”) was cleared of a large amount of wood debris in 2018 and has only had a single monitoring period (see Section: Pond 12). This wetland is a hotspot of amphibian diversity and breeding activity in Kinbasket Reservoir (Hawkes and Wood 2014, Figure 5-22) and thus, the restoration of this habitat through wood removal may be significant to note in terms of amphibian productivity. Further monitoring should be a priority to document changes in wildlife use at this site.

If Kinbasket Reservoir attains full pool in future years, it would provide an excellent opportunity to assess the success of log-boom exclosures (VP-N, BAC-N) and wood mound/windrow treatments (BAC-N, BAC-S). These management questions cannot be addressed without an assessment of the impacts of maximal inundation on physical works at Bush Arm Causeway (BAC-N and BAC-S) and

Valemount Peatland sites (VP-N). During the subsequent growing seasons, follow-up monitoring should be conducted to document covers of plant species growing within log-boom exclosures and on mounds.

6.0 LITERATURE CITED

Adama, D., V.C. Hawkes, M.T. Miller, and J. Sharkey. 2013. CLBMON-61 Kinbasket Reservoir Wetlands Monitoring Program. Annual Report – 2012. LGL Project EA3398. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generations, Water License Requirements, Burnaby, BC. 75 pp + Appendices.

Adama, D. 2015. CLBWORKS-01 Kinbasket Reservoir Revegetation Program, 2014 Post-Planting Report. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generations, Water License Requirements, Burnaby, BC. 20 pp + Appendices.

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26: 32–46.

Arnett, R.H. and M.C. Thomas. 2001. American Beetles, Volume 1. CRC Press. 464 pp.

BC Hydro. 2005. Consultative Committee report: Columbia River water use plan. Volumes 1 and 2. BC Hydro Power Corporation, Burnaby, BC.

BC Hydro. 2007. Columbia River project water use plan. BC Hydro Generation, Burnaby BC.

BC Hydro. 2008. Columbia River Project Water Use Plan, Monitoring Program Terms of Reference: CLBMON-11A Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir. BC Hydro Water License Requirements, Burnaby, BC.

BC Hydro. 2017. Columbia River Project Water Use Plan, Monitoring Program Terms of Reference: CLBMON-11A Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir. BC Hydro Water License Requirements, Burnaby, BC.

Bess, E.C., R.P. Parmenter, S. McCoy, and M.C. Molles. 2002. Responses of a riparian forest-floor arthropod community to wildfire in the Middle Rio Grande Valley, New Mexico. Environmental Entomology 31: 774–784.

Bibby C.J., N.D. Burgess, D.A. Hill and S.H. Mustoe. 2000. Bird census techniques. 2nd ed. New York, NY. Academic Press.

Bousquet, Y., P. Bouchard, A.E. Davies, and D.S. Sikes. 2013. Checklist of beetles (Coleoptera) of Canada and Alaska. Second edition. Zookeys 360:1–44.

Cárcamo H., J. Pinzón, R. Leech, and J. Spence. 2014. Spiders (Arachnida: Araneae) of the Canadian Prairies. In Arthropods of Canadian Grasslands (Volume 3): Biodiversity and Systematics Part 1. Edited by H. A. Cárcamo and D. J. Giberson. Biological Survey of Canada. pp. 75-137.

Chen, H. 2015. VennDiagram: Generate High-Resolution Venn and Euler Plots. R package version 1.6.16. https://CRAN.R-project.org/package=VennDiagram

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18: 117-143.

Cooper Beauchesne and Associates Ltd (CBA). 2009. Monitoring Program No. CLBMON-11A Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir, Draft Technical Report (Vers 2) – 2008. Unpublished report by Cooper Beauchesne and Associates Ltd., Prince George, BC, for BC Hydro, Water License Requirements, Golden, BC. 67 pp. + Appendices

Cooper Beauchesne and Associates Ltd (CBA). 2010. Monitoring Program CLBMON-11A: Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir Final Technical Report – Year 2 (2009). Unpublished report by Cooper Beauchesne and Associates Ltd., Errington, BC, for BC Hydro, Water License Requirements, Golden, BC. 90 pp. + Appendices.

Cooper Beauchesne and Associates Ltd (CBA). 2011. CLBMON-11A: Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir Final Technical Report – Year 3 (2010). Unpublished report by Cooper Beauchesne and Associates Ltd., Errington, BC, for BC Hydro, Water License Requirements, Castlegar, BC. 101 pp. + Appendices.

Dondale, C.D. and M.R. Binns. 1977. Effect of weather factors on spiders (Araeneida) in an Ontario meadow. Canadian Journal of Zoology 55: 1336-1341.

Enns, K., P. Gibeau and B. Enns. 2009. CLBMON-12 Arrow Lakes Reservoir Monitoring of Revegetation Efforts and Vegetation Composition Analysis: 2009 Final Report. Report prepared by Delphinium Holdings Inc. for BC Hydro, 99 pp. plus appendices

Environment Canada. 2017. General Nesting Periods of Migratory Birds in Canada. https://www.canada.ca/en/environment-climate-change/services/avoiding-harm-migratory-birds/general-nesting-periods.html#_01 (Accessed 24 January 2018).

Frampton, G.K., P.J. van den Brink, and P.J.L. Gould. 2000. Effects of spring drought and irrigation on farmland arthropods in southern Britain. Journal of Applied Ecology 37: 865-883.

Furness, R.W. and J.J.D. Greenwood (Eds.) 1993. Birds as Monitors of Environmental Change. Chapman & Hall, London.

Goulet, H. 1983. The genera of Holarctic Elaphrini and species of *Elaphrus* Fabricius (Coleoptera: Carabidae): Classification, phylogeny and zoogeography. Quaestiones Entomologicae 19: 219-482.

Ham, D.G. and C. Menezes. 2008. Kinbasket Reservoir Monitoring of the Valemount Peatland. Report prepared by Northwest Hydraulic Consultants for BC Hydro. 23 pp.

Hawkes, V.C. 2016. CLBWORKS-1 Kinbasket Reservoir revegetation program: year 7 – 2015. Debris mound and wind row construction pilot program. Annual Report. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generations, Water License Requirements, Burnaby, BC. 35 pp.

Hawkes, V.C. 2017. CLBWORKS-1 Kinbasket Reservoir revegetation program: year 8 – 2016. Debris mound and wind row construction pilot program. Fall 2016 Update. Annual Report. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generations, Water License Requirements, Burnaby, BC. 33 pp.

Hawkes, V.C. and D.B. Adama. 2018. CLBMON-35 Arrow Lakes and Kinbasket Reservoirs plant response to inundation. Revegetation prescription catalogue: Kinbasket Reservoir. Draft. Unpublished report by LGL Limited, Sidney, BC, for BC Hydro, Burnaby BC. 73 pp.

Hawkes, V.C. and M.T. Miller. 2016. CLBMON-09 Kinbasket Reservoir monitoring of revegetation efforts and vegetation composition analysis: year 5 – 2015. Annual Report. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 41 pp + appendices.

Hawkes, V.C. and C. Wood. 2014. CLBMON-58. Kinbasket Reservoir: Monitoring of Impacts on Amphibians and Reptiles from Mica Units 5 and 6 in Kinbasket Reservoir. Year 2 Annual Report – 2013. LGL Report EA3452. Unpublished report by Okanagan Nation Alliance and LGL Limited environmental research associates, Sidney, B.C., for BC Hydro Generations, Water License Requirements, Burnaby, B.C. 68 pp + Appendices.

Hawkes, V.C. N. Hentze, J. Muir, J. Sharkey, J. Gatten, B. McKinnon, and P. Gibeau. 2014. CLBMON-11A. Kinbasket and Arrow Lakes Reservoirs: Wildlife Effectiveness Monitoring and Enhancement Area Identification for Lower and Mid-Arrow Lakes Reservoir. Annual Report – 2013. LGL Report EA3450. Unpublished report by Okanagan Nation Alliance, Westbank, BC, and LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 62 pp + Appendices.

Hawkes, V.C., C. Houwers, J.D. Fenneman, and J.E. Muir. 2007. CLBMON-10 Kinbasket Reservoir Inventory of Vegetation Resources. Annual Report – 2007. LGL Report EA1986. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation by LGL Consultants Limited, Sidney. 82 pp.

Hawkes, V.C., J. Sharkey, and P. Gibeau. 2010. Kinbasket and Arrow Lakes Reservoir Monitoring Program No. CBLMON-11 Wildlife Effectiveness Monitoring and Enhancement Area Identification for Lower and Mid-Arrow Lakes Reservoir. LGL Report EA3164. Unpublished report by LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generations, Water License Requirements Burnaby, BC. 97 pp + Appendices.

Hawkes, V.C., M.T. Miller, J.E. Muir, and P. Gibeau. 2013. CLBMON-9 Kinbasket Reservoir Monitoring of Revegetation Efforts and Vegetation Composition Analysis. Annual Report – 2013. LGL Report EA3453. Unpublished report by LGL Limited, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Castlegar, BC. 70 pp. + Appendices.

Hawkes, V.C., B. McKinnon, and C. Wood. 2017. CLBMON-37. Kinbasket and Arrow Lakes Reservoirs: Amphibian and Reptile Life History and Habitat Use Assessment. Year 6 Annual Report – 2016. LGL Report EA3533D. Unpublished report by LGL Limited environmental research associates, Sidney, BC and Okanagan Nation Alliance, Westbank, BC, for BC Hydro Generations, Water License Requirements, Burnaby, BC. 79 pp + Appendices.

Larochelle, A. and M.-C. Larivière. 2001. Natural History of the tiger beetles of North America north of Mexico. Cicindela 33(3-4):41–162.

Larochelle, A. and M.-C. Larivière. 2003. A Natural History of the Ground-Beetles (Coleoptera: Carabidae) of America north of Mexico. Faunistica No 27. Pensoft Publishers, Sofia, Moscow. 583 pp.

Legendre, P. and E.G. Gallagher 2001 Ecologically meaningful transformations for ordination of species data. Oecologia 129(2):271–280.

Lindroth, C.H. 1961-1969. The ground-beetles (Carabidae, excl. Cicindelinae) of Canada and Alaska, parts 1-6. Opuscula Entomologica Supplementa XX, XXIV, XXIX, XXXIII, XXXIV, XXXV.

Loreau, M. 2010. From Populations to Ecosystems: theoretical foundations for a new ecological synthesis. Monographs in Population Biology, No. 46. Princeton University Press. Princeton, NJ. 297 pp.

MacInnis A.J., V. Prigmore, R.A. Gill, and J.M. Cooper. 2011. Monitoring Program CLBMON-11A: Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir, Data Summary Report – Year 4 (2011). Unpublished report by Cooper Beauchesne and Associates Ltd., Errington, BC, for BC Hydro Generation, Water Licence Requirements, Castlegar, BC. 19 pp. + Appendices.

MacInnis A.J., V. Prigmore, and J.M. Cooper. 2012. Monitoring Program CLBMON-11A: Wildlife Effectiveness Monitoring of Revegetation in Kinbasket Reservoir, Data Summary Report – Year 5 (2012). Unpublished report by Cooper Beauchesne and Associates Ltd., Errington, BC, for BC Hydro Generation, Environmental Risk Management, Columbia Water License Requirements, Burnaby, BC. v + 12 pp. + Appendices.

Marshall, S.A., R.S. Anderson, R.E. Roughley, V. Behan-Pelletier, and H.V. Danks. 1994. Terrestrial arthropod biodiversity: planning a study and recommended sampling techniques. A brief. Bulletin of the Entomological Society of Canada 26(1), Supplement, 33 pp.

Miller, M.T and V.C. Hawkes. 2019. CLBMON-09 Kinbasket Reservoir monitoring of revegetation efforts and vegetation composition analysis: year 6 (part 1) – 2018. Interim Report. Unpublished report by LGL Limited, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 26 pp.

Moody, A. and W. Carr. 2003. Mica—Revelstoke—Keenleyside Water Use Plan: Potential Areas for Vegetation Establishment in Kinbasket Reservoir. Prepared for BC Hydro. 39 p.

Morrison M.L. 1986. Bird Populations as Indicators of Environmental Change. *In*: Johnston R.F. (ed.) Current Ornithology. Current Ornithology, vol 3. Springer, Boston, MA

Niemela, J., D.W. Langor., and J.R. Spence. 1993. Effects of clear-cut harvesting on boreal ground-beetle assemblages (Coleoptera: Carabidae) in western Canada. Conservation Biology 7:551–561.

Oksanen, J., F.G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P.R. Minchin, R.B. O'Hara, G.L. Simpson, P. Solymos, M.H.H. Stevens, E. Szoecs, and H. Wagner. 2018. vegan: Community Ecology Package. R package version 2.4-6. https://CRAN.R-project.org/package=vegan.

Pearson, D.L., C.B. Knisley, and C.J. Kazilek. 2006. A field guide to the tiger beetles of the United States and Canada: identification, natural history, and distribution of the Cicindelidae. Oxford University Press, New York. 227 pp. + plates.

Polster, D. 2011. Natural Processes: Restoration of Drastically Disturbed Sites Manual. Polster Environmental Services Ltd. August 2011.

R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: https://www.R-project.org/.

Ralph, C. J., J. R. Sauer, and S. Droege. 1995. Monitoring bird populations by point counts. Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Albany, CA.

Rao, C.R .1995. A review of canonical coordinates and an alternative to correspondence analysis using Hellinger distance. Qüestiió 19:23–63.

Resources Inventory Committee. 1998b. Inventory methods for terrestrial arthropods. Version 2. Page 42. Standards for Components of British Columbia's Biodiversity, No. 40. Ministry of Environment, Lands, and Parks, Victoria, BC.

Resources Inventory Committee. 1999. Inventory Methods for Forest and Grassland Songbirds. Standards for Components of British Columbia's Biodiversity No. 15. Ministry of Environment, Lands and Parks, Victoria, BC. 37 pp.

Rushton, S.P., C.J. Topping, and M.D. Eyre. 1987. The habitat preferences of grassland spiders as identified using Detrended Correspondence Analysis (DECORANA). Bulletin of the British Arachnological Society 7: 165-170.

Samu, F. and G.L. Lövei. 1995. Species richness of a spider community: extrapolation from simulated increasing sampling effort. European Journal of Entomology 92: 633–638.

Sullivan, T.P., D.S. Sullivan, and J.H. Sullivan. 2017. Mammalian responses to windrows of woody debris on clearcuts: Abundance and diversity of forest-floor small mammals and presence of small mustelids. Forest Ecology and Management 399:143–154.

Uetz, G.W., J. Halaj, and A.B. Cady. 1999. Guild Structure of Spiders in Major Crops. The Journal of Arachnology 27:270–280.

Wheelwright, N.T. and J.D. Rising. 2008. Savannah Sparrow (*Passerculus sandwichensis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/045.

Wood, C.M., N. Hentze, and V.C. Hawkes. 2015. CLBMON-11A. Kinbasket and Arrow Lakes Reservoirs: Wildlife Effectiveness Monitoring of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir. Annual Report – 2014. LGL Report EA3451A. Unpublished report by Okanagan Nation Alliance, Westbank, BC, and LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 65 pp. + Appendices.

Wood, C.M., N. Hentze, and V.C. Hawkes. 2016. CLBMON-11A. Kinbasket and Arrow Lakes Reservoirs: Wildlife Effectiveness Monitoring of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir. Annual Report – 2015. LGL Report EA3451B. Unpublished report by Okanagan Nation Alliance, Westbank, BC, and LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 65 pp. + Appendices.

Wood, C.M., N. Johnston, and V.C. Hawkes. 2017. CLBMON-11A. Kinbasket and Arrow Lakes Reservoirs: Wildlife Effectiveness Monitoring of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir. Annual Report – 2016. LGL Report

EA3451C. Unpublished report by Okanagan Nation Alliance, Westbank, BC, and LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 53 pp. + Appendices.

Wood, C.M., Hentze, N., V.C. Hawkes, and N. Johnston. 2018. CLBMON-11A. Kinbasket and Arrow Lakes Reservoirs: Wildlife Effectiveness Monitoring of Revegetation Efforts and Physical Works Trials in Kinbasket Reservoir. Annual Report – 2017. LGL Report EA3451D. Unpublished report by Okanagan Nation Alliance, Westbank, BC, and LGL Limited environmental research associates, Sidney, BC, for BC Hydro Generation, Water Licence Requirements, Burnaby, BC. 74 pp. + Appendices.

7.0 APPENDICES

A timeline of the CLBMON-11A monitoring program and all 2018 data analyses are included in the following appendices. These data chapters are presented as individual reports for each response measure under assessment. Where possible, data from years 2014 to 2018 are used for comprehensive evaluation of wood removal and physical works treatment effects.

Appendix 1: History of CLBMON-11A

Since its inception, CLBMON-11A has been revised considerably in almost every aspect of the program, including focal taxa, terms of reference, management questions, study sites, and treatments of interest. Following is an overview of the evolution of the program from initial planning through final monitoring year.

During the Columbia River Water Use (WUP) planning process, the WUP Consultative Committee (WUP CC) recognized the value of vegetation for improving aesthetic quality, controlling dust, protecting cultural heritage sites from erosion and human access, and enhancing littoral productivity and wildlife habitat. As part of the WUP, a study was initiated in 2001 to identify areas with the highest potential for successful vegetation establishment (Moody and Carr 2003). In 2005, the WUP CC supported a reservoir-wide revegetation program for Kinbasket Reservoir that was compatible with the current operating regime (BC Hydro 2005). The Kinbasket Reservoir Revegetation Program (CLBWORKS-1) was initiated with a field reconnaissance in 2007 and focused on areas identified as having adequate suitability for enhancement.

In 2008, CLBMON-11A, an 11-year program to conduct monitoring of wildlife habitat utilization in response to revegetation efforts in Kinbasket Reservoir, was initiated. Wildlife monitoring during the first five implementation years of CLBMON-11A were administered by Cooper Beauchesne and Associates Ltd (CBA 2009, 2010, 2011, and MacInnis et al. 2011, 2012). This program followed the original Terms of Reference (TOR; BC Hydro 2008), which included the following four management questions in addition to several management hypothesis:

1. How effective is the revegetation program at enhancing and increasing the utilization of habitat in the drawdown zone by wildlife such as amphibians, birds, small mammals, and ungulates?
2. To what extent does revegetation increase the availability of invertebrate prey (e.g. arthropods) in the food chain for birds, amphibians and small mammals?
3. Are revegetation efforts negatively impacting wildlife in the drawdown zone? For example, does revegetation increase the incidence of nest mortality in birds or create sink habitat for amphibians?
4. Which methods of revegetation are most effective at enhancing and increasing the utilization of wildlife habitat in the drawdown zone?

Several focal taxa (ungulates, songbirds, small mammals, and terrestrial and aerial arthropods) were monitored across revegetated areas, adjacent drawdown zone controls, and upland, non-reservoir reference sites Table 7–1). The Okanagan Nation Alliance (ONA), in partnership with LGL Limited environmental research associates, continued monitoring in 2013 with methods consistent with the preceding monitoring years. Based on the conclusions of Hawkes et al (2014), BC Hydro agreed that the methods were not well suited to answering the management questions associated with CLBMON-11A. For example, the wrong species of small mammal were being targeted, the productivity (i.e., seed load) of plants that would be consumed by granivorous small mammals had not been assessed, songbirds had not been considered as focal taxa, and the size of the revegetation prescriptions applied in the drawdown zone were likely of little benefit to ungulates given the proximity and spatial extent of suitable habitat adjacent to the drawdown zone. Overall, there did not appear to have been a connection made between the types of plants used in the revegetation program (CLBWORKS-1) and how the use of those species would benefit wildlife using the drawdown zone of Kinbasket Reservoir. In addition, a technical review workshop comprising representatives from BC Hydro, First Nations, contractors, and other agencies met in December 2014 to discuss past and potential future approaches to revegetation. They concluded that most revegetation efforts in the Kinbasket Reservoir have been ineffective to date (BC Hydro 2017), and thus it was not possible to address the management questions as originally stated.

An outcome of the technical review was to monitor wood removal conducted at Canoe Reach (under CLBWORKS-16: woody-debris removal program) as an alternative to traditional revegetation, enhancing natural vegetation establishment in upper elevation bands of the drawdown zone. The ONA and LGL adapted the wildlife monitoring for CLBMON-11A to monitor the effectiveness of wood removal treatments at Canoe Reach in 2014 (Wood et al. 2015). Five sites were selected within Canoe Reach for monitoring wood removal treatment areas (Valemount Peatland North, Valemount Peatland South, Yellowjacket Creek, Packsaddle Creek North, and Packsaddle Creek South). Control plots were established adjacent to treatment areas[2] for the purpose of monitor the changes in treatment areas relative to controls over the remaining study years.

Another novel approach to revegetation and habitat enhancement was the construction of wood debris structures (mounds and windrows) at Bush Arm Causeway North and South as a pilot project under CLBWORKS-1 (Debris Mound and Wind Row Construction Pilot Program; Hawkes 2016). As part of this initiative, five sites in Bush Arm were identified as potential locations for mound and windrow construction (Hawkes 2016). The five sites were Bush Causeway North, Bush Arm Causeway South, Goodfellow Creek, Hope Creek, and Chatter Creek. In June 2015, CLBMON-11A monitoring was conducted at these five proposed wildlife physical works sites in Bush Arm to assess baseline, pre-treatment conditions (Wood et al. 2016).

In the fall of 2015, the two sites at Bush Arm Causeway (BAC-N and BAC-S) were treated. Locally available wood debris and substrates were used to construct mounds to a height exceeding the maximum operating elevation of the reservoir, with the aim of creating a series of small non-inundated islands and peninsulas where vegetation could establish, and which could eventually provide added habitat value for wildlife. A total of seven mounds were constructed in the two locations, along with windrows at one location. This work uncovered three previously wood-choked ponds at Bush Arm Causeway North. Live stakes (black cottonwood and red-osier dogwood) were planted in the mounds, and locally salvaged sedge plugs were transplanted into suitable substrates at the base of some of the mounds (Hawkes 2016).

Post-treatment monitoring of wood clearing at Canoe Reach and the wood mound and wind rows at the Bush Arm Causeway sites continued through 2018 (Wood et al. 2017, Wood et al. 2018). Focal taxa were streamlined to species of terrestrial arthropods and songbirds in efforts to increase our ability to detect changes in treatment areas and answer management questions. The three other

[2] Exception: Valemount Peatland South (VP-S) site was completely cleared of wood debris, thus, we were unable to establish a control plot at this site for monitoring that occurred in 2014.

sites at Bush Arm have not yet been treated and were therefore not the focus of wildlife monitoring in recent years.

In 2017, the TOR for CLBMON-11A were revised to reflect these changes in the program, including removal of ungulates as a focal taxon. Management Question 3 was omitted and replaced by another one focusing on diversity and abundance of arthropods, amphibians, and birds. All management hypotheses were removed, and the requirement of formal null hypothesis significance testing was omitted.

In 2018, wildlife effectiveness monitoring was expanded to include surveys at an additional site of wood removal in Canoe Reach (Pond 12) and to assess revegetation effectiveness of three revegetation polygons at KM88. These revegetation areas were treated under CLBWORKS-1 in 2013, and were larger in spatial extent, planted with more mature sedge plugs, and had a higher planting density than previously treated areas.

The study sites and surveys conducted in all years of CLBMON 11A are summarized in Table 7-1A major change in study sites occurred in 2014, with the shift toward monitoring physical works trials rather than monitoring failed revegetation treatments. Minor changes have been made to the annual selection of study sites since 2014. Those changes are largely due to the repeat dismantling of experimental sites by unsystematic treatment application, resulting in wood removal within controls and re-treatment of treatments. In 2016, the drawdown zone treatment at Packsaddle Creek North (PS-N) and South (PS-S) were re-cleared of wood debris. Wood debris was completely removed from the control plots at PS-N and PS-S (reported in Wood et al. 2017) causing further detriment to our experimental design. Similarly, the treatment site at Yellowjacket Creek was re-cleared of wood debris immediately prior to wildlife monitoring in 2017 (reported in Wood et al. 2018). In 2018, the control plot at Valemount Peatland North was treated and all coarse woody material was removed prior to surveys (Figure 2-6). With the loss of experimental integrity, some of these sites at Canoe Reach were dropped from monitoring as it would not be possible to infer any treatment effect without a suitable control.

Table 7-1: **Summary of wildlife monitoring under CLBMON-11A from 2008 to 2018.** Information compiled from previous reports (e.g., CBA 2009, 2010; Hawkes et al. 2014) and the Kinbasket Reservoir Revegetation Catalogue (Hawkes and Adama 2018). Survey type: A = arthropod trapping, PC = bird point counts, LT = bird line transects, SM = small mammal trapping, U = ungulate pellet plots.

Study area	Plot	Treatment Type	Treatment Application	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
CANOE REACH:														
Canoe River Mouth	2	Revegetation	Graminoid seedling, shrub stakes, shrub seedlings, and seed mix 2009	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U						
Valemount Peatland	8	Revegetation	Graminoid seedling, shrub stakes, shrub seedlings, and seed mix 2009	A,PCSM,U	A,PC,SM,U	A,PC,SM,U	U	U	A,PC,SM,U					
	VP-N T	Physical Works	Wood removal 2014							A,PC,U	LT	A,LT	A, LT	A,LT
	VP-N C	Wood accumulation control	Wood removal 2018*							A,PC,U	LT	A,LT	A, LT	A,LT
	VP-N DDZ	Drawdown zone control	None							PC	LT	A,LT	LT	LT
	VP-N R	Upland forest control	None							A,PC,U	PC	A,PC	A, PC	A
	VP-S T	Physical Works	Wood removal 2012							A,PC,U	LT	LT	LT	
	VP-S R	Upland forest control	None							A,PC,U	PC	PC	PC	
Pond 12	P12 T	Physical Works	Wood removal 2018											A,LT
	P12 C	Wood accumulation control	None											A,LT
	P12 R	Upland forest control	None											A
Packsaddle Creek	PS-N T	Physical Works	Wood removal 2014, 2016*							A,PC,U	A,LT	A		
	PS-N C	Wood accumulation control	Wood removal 2016*							A,PC,U	A,LT	A		
	PS-N R	Upland forest control	None							A,PC,U	A,PC	A		
	PS-S T	Physical Works	Wood removal 2014, 2016*							A,PC,U	A,LT			
	PS-S C	Wood accumulation control	Wood removal 2016*							A,PC,U	A,LT			
	PS-S R	Upland forest control	None							A,PC,U	A,PC			
Dave Henry Creek	9	Revegetation	Graminoid seedling and seed mix 2009	A,PC,SM,U	A,PC,SM,U									
	12	Revegetation	Graminoid seedling and seed mix 2009	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U	A,PC,SM,U					
Yellowjacket Creek	15	Revegetation	Graminoid seedling, shrub seedlings, seed mix, and fertilizer 2009	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U	A,PC,SM,U					
	YJ-T	Physical Works	Wood removal 2014, 2017*							A,PC,U	A,LT	A,LT	A, LT	A,LT
	YJ-C	Wood accumulation control	None							A,PC,U	A,LT	A,LT	A, LT	A,LT
	YJ-R	Upland forest control	None							A,PC,U	A,PC	PC	A, PC	A
Ptarmigan Creek	25	Revegetation	Graminoid seedling and shrub stake 2009	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U						
	32		None	A,PC,SM,U										
Windfall Creek	33	Revegetation	Graminoid seedlings and seed mix 2009	A,PC,SM,U	A,PC,SM,U									
	34	Revegetation	Graminoid seedlings and seed mix 2009		A,PC,SM,U									
BUSH ARM:														
KM 88	80	Revegetation	Graminoid seedling 2013						A,PC,SM,U					

Study area	Plot	Treatment Type	Treatment Application	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Big Bend	TU1	Revegetation	Graminoid seedling (Kellogg's sedge) 2013											A,LT
	TU3	Revegetation	Graminoid seedling (Kellogg's sedge) 2013											A,LT
	TU5	Revegetation	Graminoid seedling (Columbia sedge) 2013											A,LT
	CU1	Control	None											A,LT
	CU2	Control	None											A,LT
	CU3	Control	None											A,LT
	KM 88 R	Upland forest control	None											A
KM 79	83	Revegetation	Graminoid seedling, shrub seedling, shrub stake, graminoid seed, and seed mix 2008	A,PC,SM,U		A,PC,SM,U	U	U						
KM 77	84	Revegetation	Seed mix and shrub stakes 2008; graminoid seedling 2010	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U						
Chatter Creek	85	Revegetation	Graminoid seedling and shrub stake 2008		A,PC,SM,U									
	CHT T†	Pre-treatment	None								LT	LT		LT
	CHT C	Control	None								LT	LT		LT
Hope Creek	87	Revegetation	Shrub stake and seed mix 2008; graminoid seedling and shrub seedling 2010; and graminoid seedling 2011	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U	A,PC,SM,U					
	HOPE T†	Pre-treatment	None								A,LT	LT		
	HOPE C	Control	None								A,LT	LT		
	HOPE R	Upland forest control	None								A,PC	A,PC		
Goodfellow Creek	88	Revegetation	Shrub stake and seed mix 2008; shrub seedling and shrub stake 2010	A,PC,SM,U	A,PC,SM,U	A,PC,SM,U	U	U	A,PC,SM,U					
	91	Control	None	A,PC,SM,U	A,PC,SM,U									
	GDF T†	Pre-treatment	None								A,LT	LT		
	GDF C	Control	None								A,LT	LT		
	GDF R	Upland forest control	None								A,PC	A,PC		
Bush Arm Causeway	121	Naturally revegetated reference site	None			A,PC,SM,U	U	U	A,PC,SM,U					
	BAC-N T	Physical Works	Wood removal, pond clearing, mounds, revegetation, & log boom (2015)									A,LT	A, LT	A,LT
	BAC-N C		None									A,LT	A, LT	A,LT
	BAC-N R		None								PC	PC	PC	PC
	BAC-S T	Physical Works	Wood removal, mounds, & revegetation (2015)								A,LT	A,LT	A, LT	A,LT
	BAC-S C		None								A,LT	A,LT	A, LT	A,LT
	BAC-S R	Upland forest control	None								PC	PC	PC	PC

† = pre-treatment (physical works) sampling.
* = uncoordinated removal of wood from control or treatment plot.

Appendix 2: Analysis of Arthropod Relative Abundance

Introduction

This monitoring program focuses on how revegetation and physical works prescriptions influence the abundance and diversity of arthropods (MQ2 and MQ3). As arthropods are a fundamental component of the food chain, particularly for small mammals (e.g., shrew, bats), birds and amphibians, an increase in arthropod abundance may relate to increases in other local wildlife. Establishing vegetation cover may also provide additional habitat for species with unique life history requirements, resulting in increased local abundance and species diversity.

Methods

Arthropod abundance is assessed as the relative abundance of ground-dwelling spiders and beetles standardized by trapping effort, which were sampled using pitfall traps (see Sampling Methodology section). Arthropod relative abundance was assessed by habitat type (Treatment, Control, and Reference), site, and year. Each study site is presented as a case study, since treatment types were not replicated.

Data Set

Data Set 1 was used to summarize results of arthropod CPUE (relevant to MQ2 and MQ3). Immature specimens were omitted from all analyses to avoid inflation of abundance from species with aggregated spiderlings or larvae.

Spider and beetle data had the same number of samples (replicates) included in data summaries, as they were both derived from the same pitfall trap data (Table 4-1). The below number of replicates was also the same for all richness and diversity analyses in Appendix 3.

Overall mean spider and beetle CPUE was examined for each treatment and control by site, using data from all years (excluding pre-treatment data and reference data). The number of replicates included for those overall assessments are as follows:

- BAC-N: Treatment (n = 33), Control (n = 30)
- BAC-S: Treatment (n = 33), Control (n = 30)
- KM88: Treatment (n = 30), Control (n = 30)
- VP-N: Treatment (n = 63), Control (n = 60)

YJ: Treatment (n = 75), Control (n = 80)

Below is a summary of replicates included for boxplots of CPUE generated at each study site:

- BAC-N: one year of pre-treatment data (2015) and three years of post-treatment data (2016-2018).
    - 2015: Treatment (n = 6), Control (n = 6)
    - 2016: Treatment (n = 13), Control (n = 10)
    - 2017: Treatment (n = 10), Control (n = 10)

    - 2018: Treatment (n = 10), Control (n = 10)

- BAC-S: one year of pre-treatment data (2015) and three years of post-treatment data (2016-2018).
    - 2015: Treatment (n = 6), Control (n = 6)
    - 2016: Treatment (n = 13), Control (n = 10)
    - 2017: Treatment (n = 10), Control (n = 10)
    - 2018: Treatment (n = 10), Control (n = 10)
- GDF: four years of upland reference data for assessing the non-reservoir inter-annual variation in arthropod abundance (to compare with BAC-N and BAC-S).
    - 2015: Reference (n = 6)
    - 2016: Reference (n = 9)
    - 2017: Reference (n = 6)
    - 2018: Reference (n = 6)
- VP-N: four years of data, all representing post-treatment sampling. Note: 2014 sampling included only one collection period due to timing of wood removal; 2015 data are lacking due to dispersal of Western Toad metamorphs that prevented trap deployment. There was also an unscheduled treating of the control area by wood removal prior to 2018 sampling (Figure 2-6).
    - 2014: Treatment (n = 9), Control (n = 9), Reference (n = 9)
    - 2016: Treatment (n = 18), Control (n = 18), Reference (n = 18)
    - 2017: Treatment (n = 18), Control (n = 18), Reference (n = 6)
    - 2018: Treatment (n = 18), Control (n = 18), Reference (n = 6)
- YJ: 5 years of data, note that there was an unscheduled re-treatment of the treatment plot just prior to 2017 sampling. Trap disturbance by deer reduced functional replicates in recent years (Table 4-1).
    - 2014: Treatment (n = 18), Control (n = 18), Reference (n = 18)
    - 2015: Treatment (n = 18), Control (n = 18), Reference (n = 18)
    - 2016: Treatment (n = 18), Control (n = 18), Reference (N/A)
    - 2017: Treatment (n = 5), Control (n = 16), Reference (n = 6)
    - 2018: Treatment (n = 16), Control (n = 18), Reference (n = 4)
- KM88: 1 year of arthropod sampling (post-revegetation monitoring in 2018). Samples were replicated within three treatment polygons (TU1, TU3, and TU5) and three control polygons (CU1, CU2, and CU3), with additional samples derived from the adjacent upland Reference area (Map 7-2).
    - 2018 Treatment: TU1 (n = 10), TU3 (n = 10), TU5 (n = 10)
        - Total Treatment: n = 30

- 2018 Control: CU1 (n = 10), CU2 (n = 10), CU3 (n = 10)
  - Total Control: n = 30
- 2018 Reference: n = 5

- POND 12: 1 year of arthropod sampling (post-wood removal).
  - 2018: Treatment (n = 18), Control (n = 18), Reference (n = 6).

Analysis

Catch per unit effort (CPUE) was calculated as the number of individuals per trap per day (24-hour period), accounting for uneven survey effort and trap disturbance by wildlife. Mean CPUE was calculated for all sites (pooling data from all years, excluding pre-treatment data). Mean CPUE was plotted in bar graphs with 90% confidence intervals for interpretation of overall differences between treatment and control samples. In addition, we examine CPUE of adult spiders and CPUE of adult beetles in treatment, control, and reference samples in each site and year of study. CPUE of reference sites were also plotted to assess degree of inter-annual variation in arthropod abundance.

Relative abundance trends were examined through boxplots or bar plots. To aid the reader in interpreting boxplot graphs, the boxes represent between 25 percent and 75 percent of the ranked 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 are drawn from the top of the box to the largest observation within 1.5 interquartile range of the top, and from the bottom of the box to the smallest observation within 1.5 interquartile range of the bottom of the box. Boxplots display the differences between groups of data without making any assumptions about their underlying statistical distributions and show their dispersion and skewness. For this reason, they are ideal in displaying ecological data. Strip plots are overlaid for each replicate (jittering points) to allow for interpretation of sample sizes and spread in the data.

Spider species were assigned to one of nine ecological function guilds based on family (Uetz et al. 1999). The relative proportion of each guild represented in treatment, control and references areas were illustrated with stacked bar graphs to assess changes in guild structure in treatment plots over time. Stacked bar plots are suggested for graphical presentation of proportional data (Carl Schwarz, pers. comm. to Charlene Wood, Oct 18, 2017). All figures were generated in R v. 3.5.0 (R Core Team 2018).

Results

<u>Overall Relative Abundance</u>

Differences in relative abundance (CPUE) depended on habitat type (treatment vs. control), taxon (spiders vs. beetles), and study site (Figure 7-1). No difference in overall CPUE of beetles and spiders was found between treatment and control plots for BAC-N, BAC-S, and Pond 12 (see Table 2-2 for treatment types). The revegetation treatment at KM88 had significantly lower CPUE of spiders and beetles than the control. The woody debris removal treatment at YJ had significantly lower CPUE of spiders, but similar beetle CPUE, compared to the control. Conversely, VP-N treatment (cleared wetland and terrestrial habitat, and log boom) had significantly higher spider CPUE, but similar beetle CPUE, compared to the control.

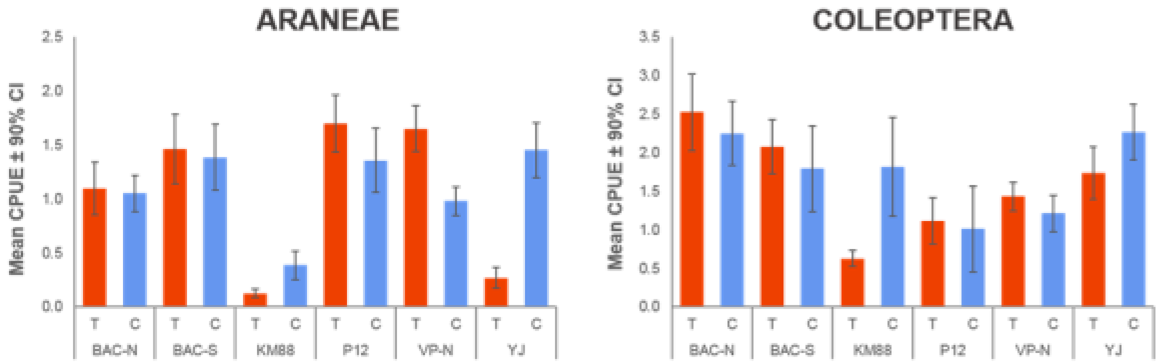


Figure 7-1: Catch per unit effort (adults/trap-day) of spiders (left) and beetles (right) in treatment (T, red), control (C, blue) samples from each study site, combining data from 2014-2018. Note: pre-treatment data were omitted (BAC-N and BAC-S 2015).

As site-specific differences have been noted, below, each site was presented as a separate case study due to different habitat features and types of physical works and/or revegetation treatments employed.

<u>Bush Arm Causeway North (BAC-N; wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom)</u>

Spider abundance tended to increase in both treatment and control plots over the study period, however, beetle abundance remained at similar levels (Figure 7-2). Both drawdown zone control and treatment samples were primarily comprised of ground-runners and wandering-sheet/tangle weavers (Figure 7-3). While proportions of spider guilds varied by habitat and year, the only unique guild documented was the "foliage runners", which was present in the pre-wood removal Treatment plot in 2015 (one individual, *Clubiona kastoni*).

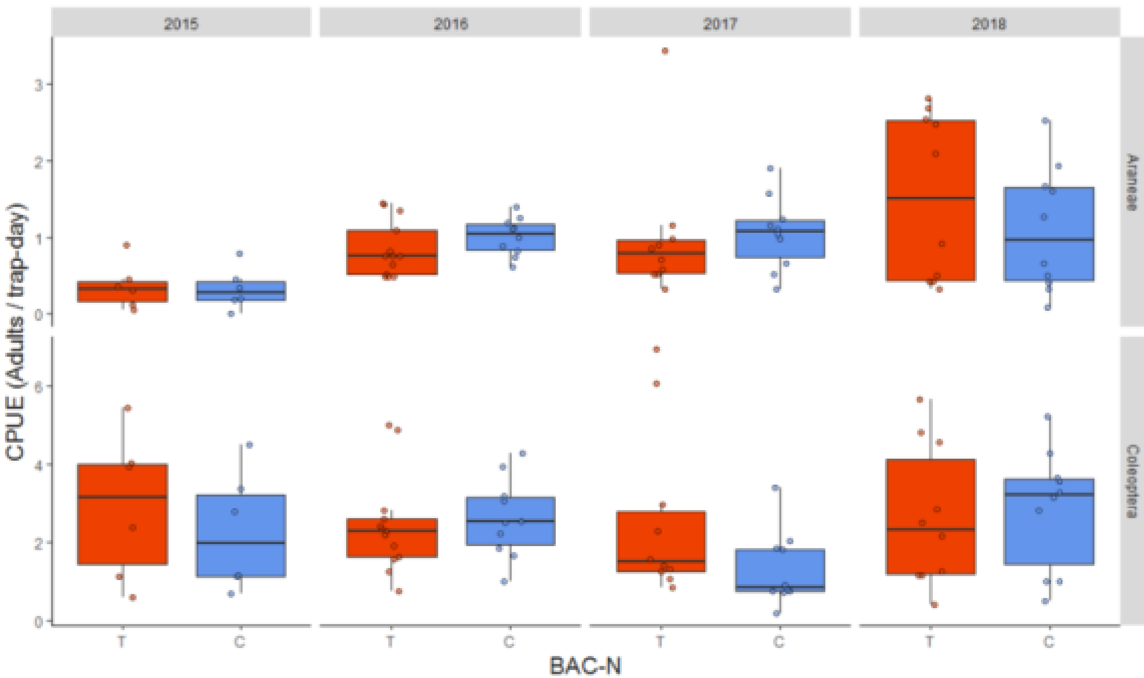


Figure 7-2: **Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and adult beetles (bottom) in treatment (T) and control (C) areas across sampling years at BAC-N.** Note: Year 2015 shows pre-treatment data

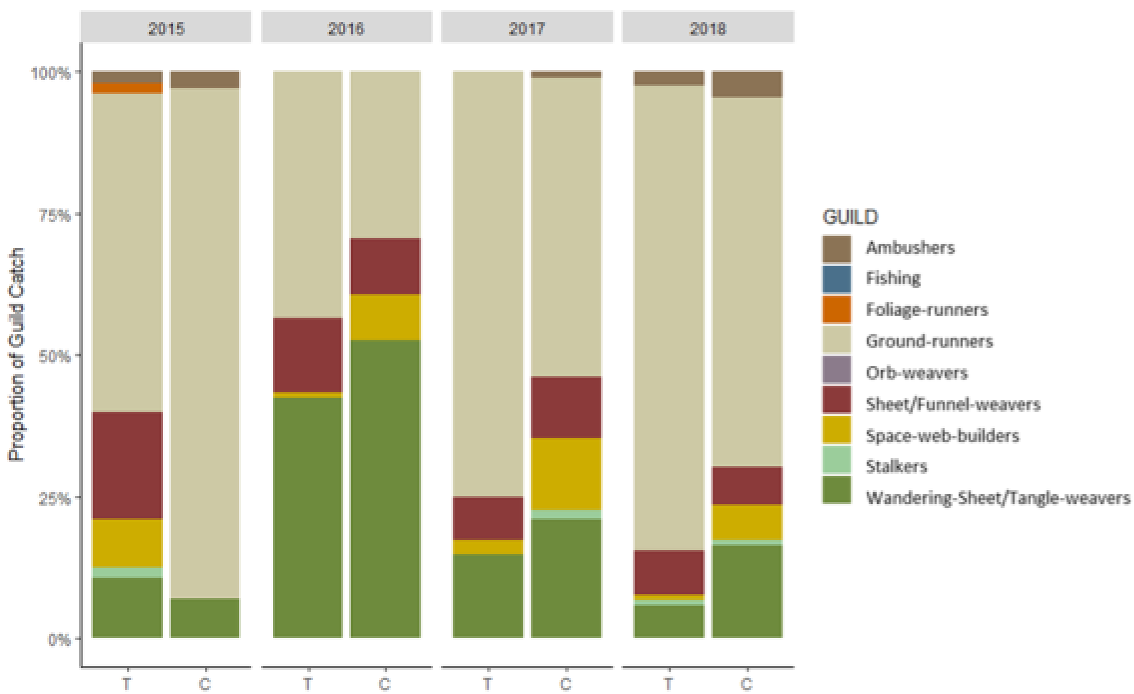


Figure 7-3: **Relative proportion of spider functional guild catch in treatment (T) and control (C) areas across sampling years at BAC-N.** Note: Year 2015 shows pre-treatment data

Bush Arm Causeway South (BAC-S; wood clearing from terrestrial habitat, mounds, revegetation, and log boom)

Spider relative abundance (CPUE) tended to increase in both treatment and control plots over the study period, but also increased in variation overtime (Figure 7-4). This trend in increasing relative abundance was not evident for beetle relative abundance. Beetle CPUE was generally greater for treatment samples in both pre-treatment sampling through 2017 but was similar between treatment and control in 2018 (Figure 7-4). Both drawdown zone control and treatment samples were dominated by ground-runners, and remaining portions primarily consisting of sheet/funnel weavers and wandering-sheet/tangle weavers (Figure 7-5).

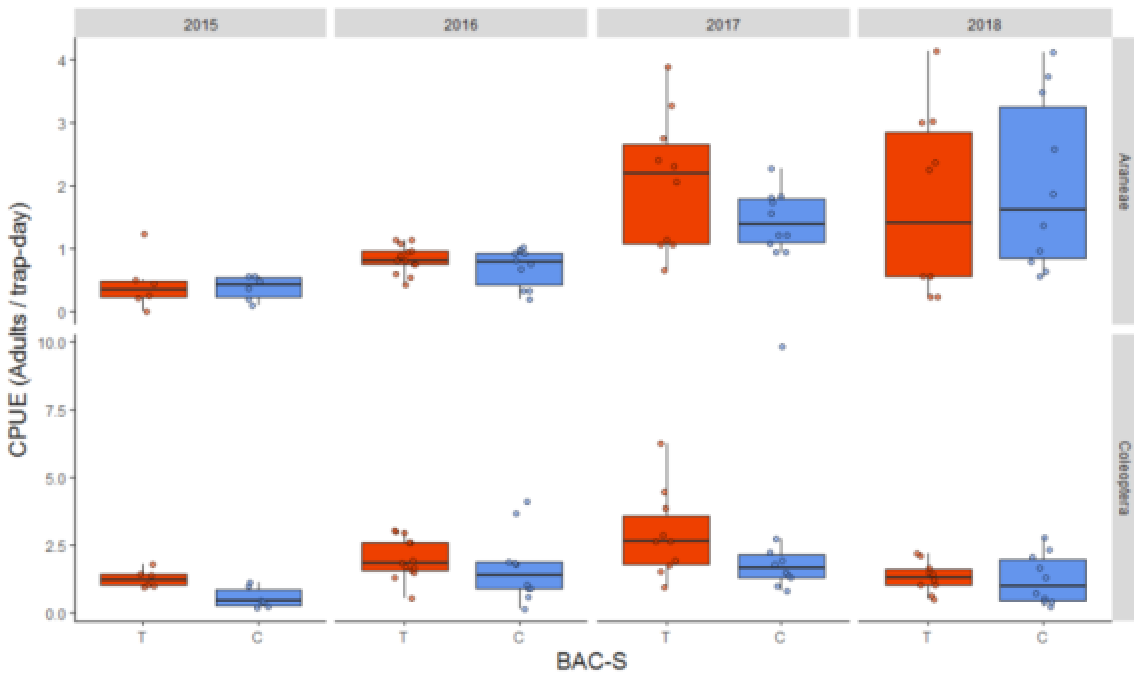


Figure 7-4: Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and beetles (bottom) in treatment (T) and control (C) areas across sampling years at BAC-S. Note: Year 2015 shows pre-treatment data

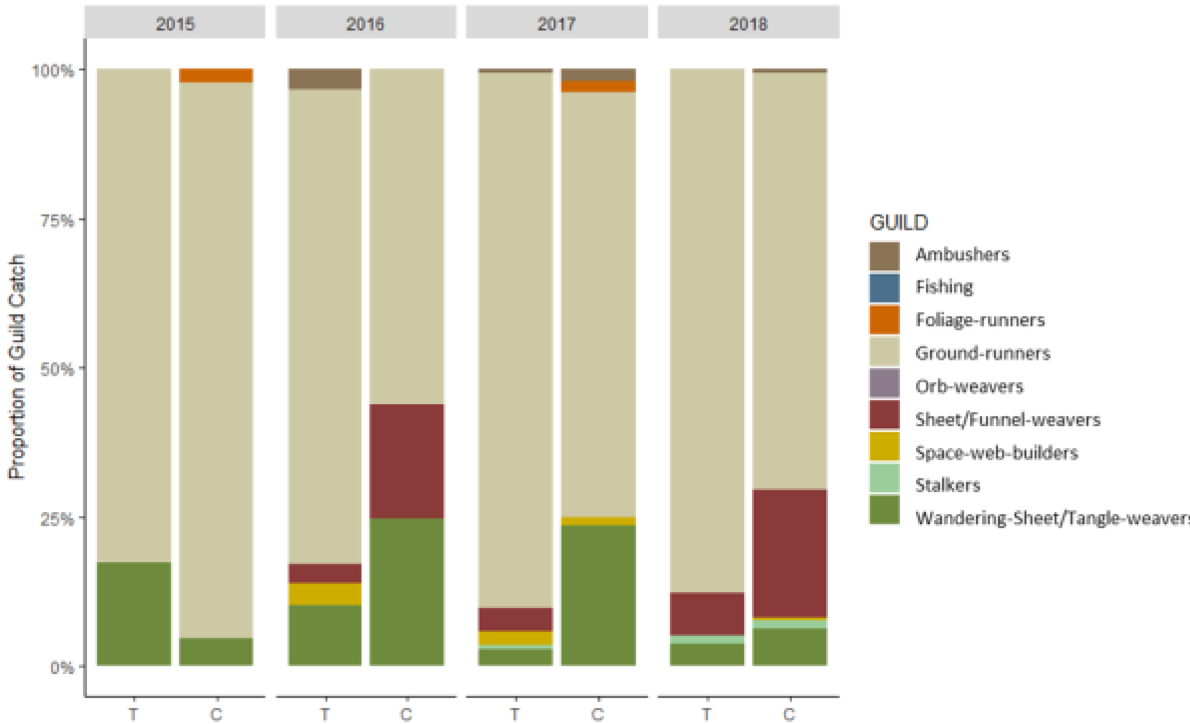


Figure 7-5: Relative proportion of spider functional guild catch in treatment (T) and control (C) areas across sampling years at BAC-S. Note: Year 2015 shows pre-treatment data

<u>Goodfellow Creek (GDF; upland reference, no treatment application)</u>

Relative abundance of spiders and beetles varied on an annual basis in upland reference areas, with 2017 having greater relative abundance than 2015 and 2016 (Figure 7-6). Inter-annual variation in CPUE of beetles and spiders was not always consistent (Figure 7-6). Ground running spiders were much less dominant in the upland reference forest (Figure 7-7) than adjacent drawdown zone areas at BAC-N and BAC-S (Figure 7-3 and Figure 7-5). A large portion of the ground-dwelling spiders in reference habitats comprised the 'wandering sheet/tangle weavers' guild (Figure 7-7).

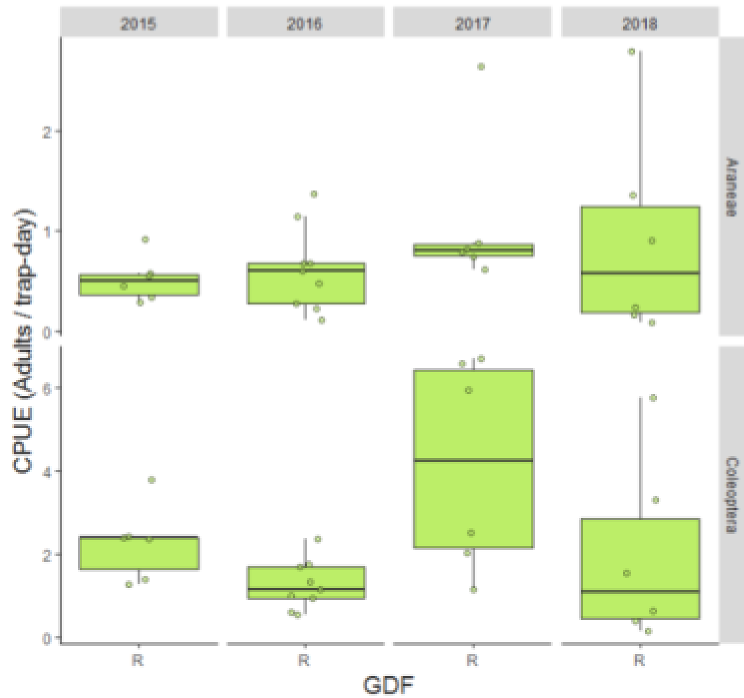


Figure 7-6: Catch per unit effort (abundance/24-hour trap period) of adult spiders in reference (R) areas across sampling years at GDF. This reference area is shared by proximal BAC-N and BAC-S.

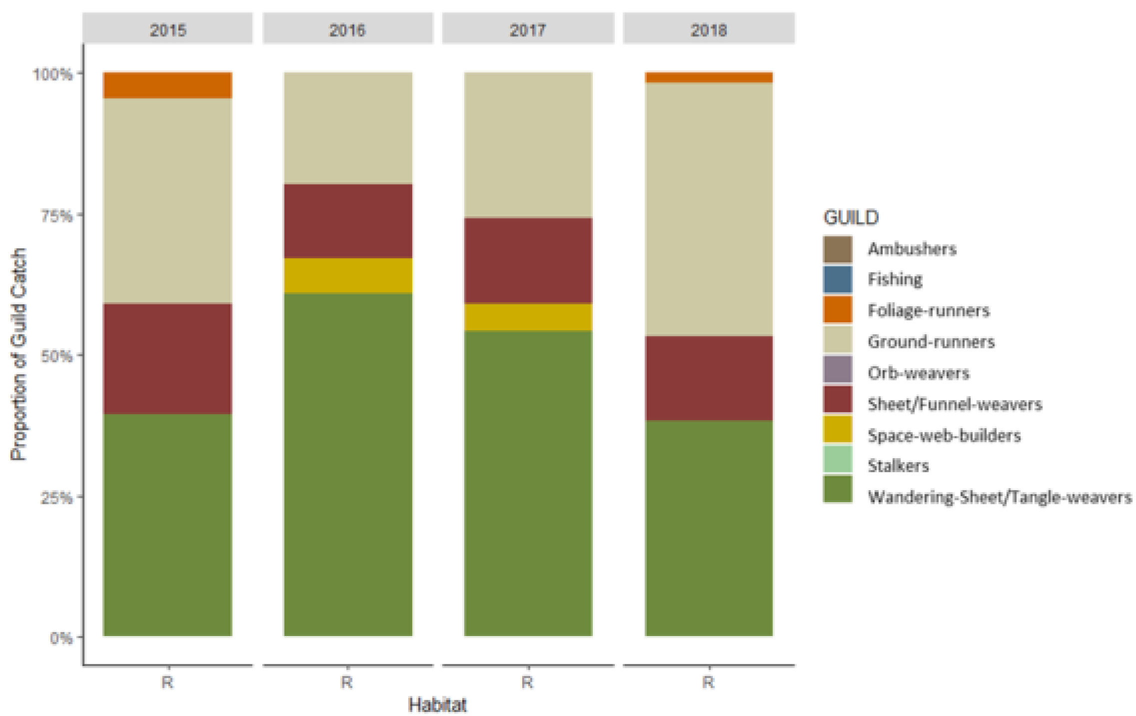


Figure 7-7: Relative proportion of spider functional guild catch in upland reference (R) area across sampling years at GDF. This reference area is shared by proximal BAC-N and BAC-S.

Big Bend (KM88; sedge plug revegetation)

Arthropods were sampled at KM88 in 2018 for the first time since revegetation in 2013. Spider and beetle (Figure 7-8) CPUE was similar among most treatment and control areas, with the exception of polygon CU3, which was more similar to the abundance of the upland reference forest (Figure 7-8). Spider guilds in treatment and control plots both consisted primarily of ground-running species and wandering-sheet/tangle-weavers, whereas upland reference areas comprised a larger portion of sheet/funnel-web weavers and space-web builders (Figure 7-9).

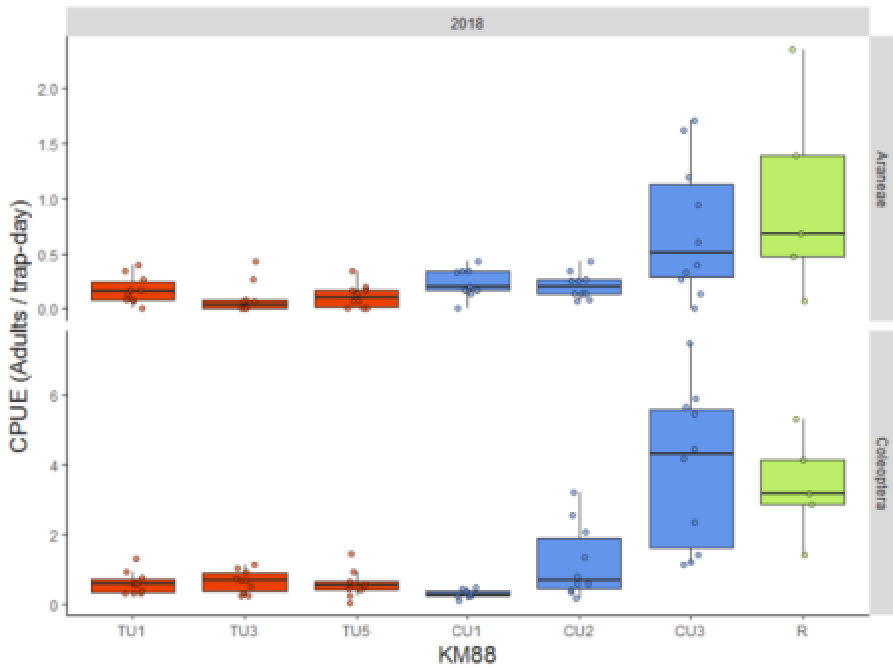


Figure 7-8: Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and adult beetles (bottom) in treatment (T) and control (C), and reference (R) samples at KM88, 2018.

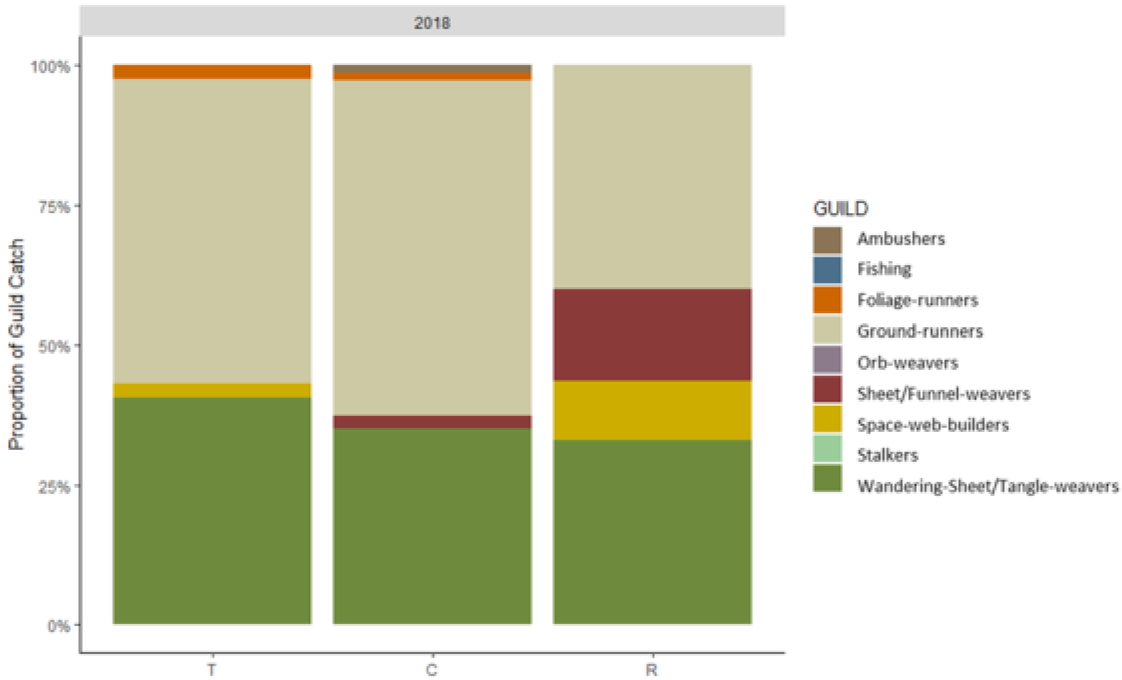


Figure 7-9: Relative proportion of spider functional guild catch in treatment (T), control (C), and reference (R) areas at KM88, 2018.

Valemount Peatland North (VP-N; wood clearing from wetland and terrestrial habitat and log boom)

Spider CPUE was generally greater in treatment than control samples, except in 2018, where treatment and control CPUE was comparable (Figure 7-10). CPUE of beetles in treatment and control areas show a trend towards reduced abundance over time in both T and C, with CPUE becoming progressively more similar between plots over time (Figure 7-10). However, spider and beetle abundance was highly variable on an annual basis. In the upland reference samples, CPUE of spiders and beetles was higher in 2017, then dropped notably in 2018 (Figure 7-11).

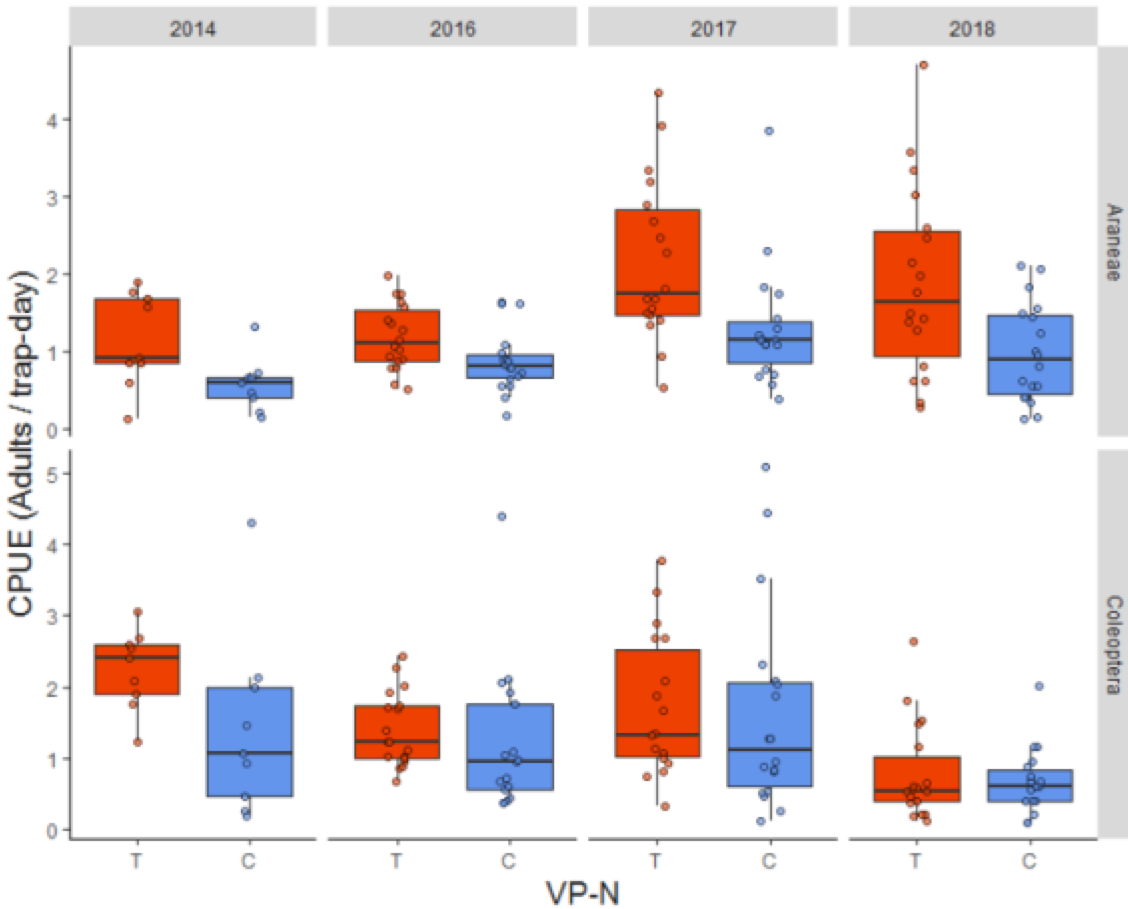


Figure 7-10: **Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and adult beetles (bottom) in treatment (T) and control (C) areas across sampling years at VP-N.**

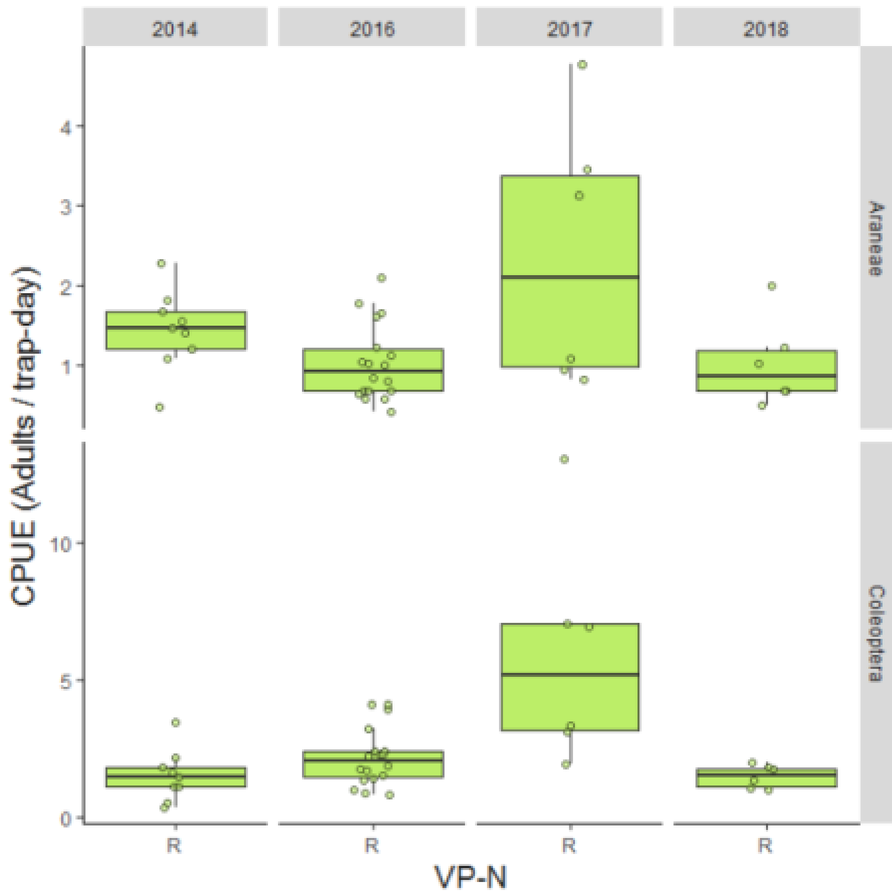


Figure 7-11: Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and beetles (bottom) in reference (R) areas across sampling years at VP-N.

Ground-running spiders dominated treatment samples in all years (Figure 7-12). The increase in number of functional guilds over time in the treatment area is noteworthy. In 2014, only three spider guilds were present: ground-runners, sheet/funnel-weavers, and wandering-sheet/tangle weavers. By 2018, the number of functional guilds of spiders more than doubled (2018 = 7 guilds) with additions of fishing spiders, ambushers, foliage-runners, and space-web builders (Figure 7-12). Stalkers were only recorded in reference samples, whereas orb-weavers were detected in both reference and control samples (but not treatment).

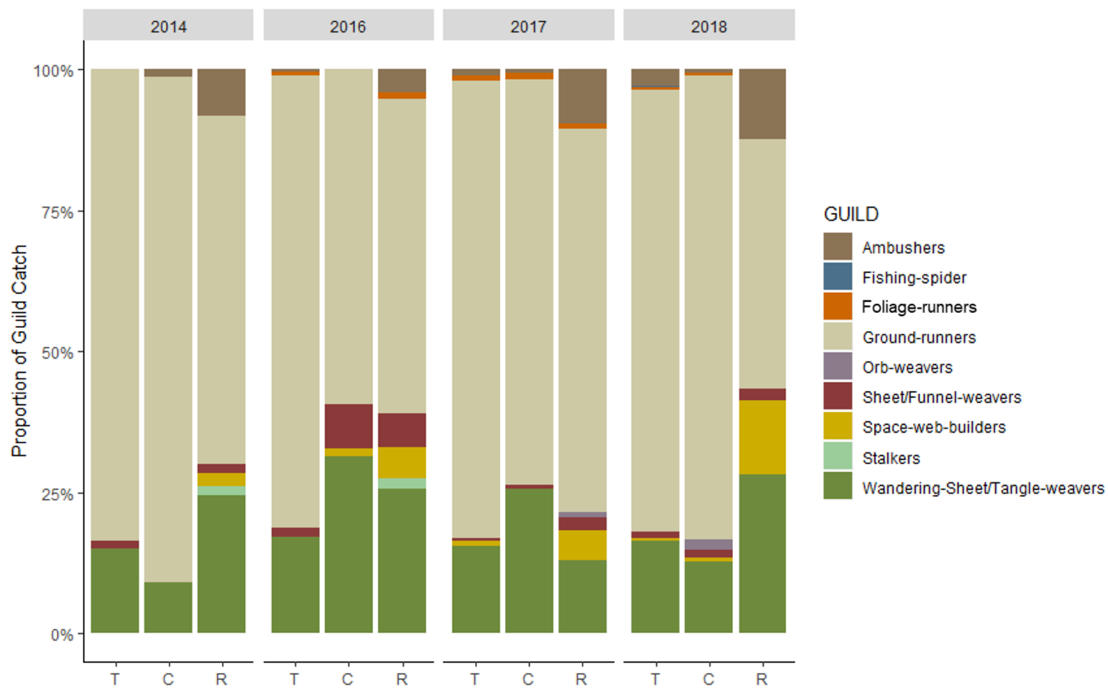


Figure 7-12: Relative proportion of spider functional guild catch in treatment (T), control (C), and reference (R) areas across sampling years at VP-N.

Yellowjacket Creek (YJ; wood clearing from terrestrial habitat)

Spider abundance remained consistently lower in the treatment area compared to the control (Figure 7-13), where inter-annual variation in CPUE were similar to the forested reference sample (Figure 7-14). Beetle relative abundance was variable over time, being greater than control abundance in 2017 (re-treatment year), but lower in all other years (Figure 7-13). Treatment and control areas at YJ housed a diverse number of spider guilds from 2014-2016 (6 guilds on average), with the proportion of ground-runners being increasingly replaced by other guilds. After the second wood removal event in 2017, guild composition in the treated area was greatly simplified to only two guild types and 90% of abundance was comprised of ground-runners. The unaltered control area retained more guild types in this year and had a smaller proportion of ground-runners (Figure 7-15).

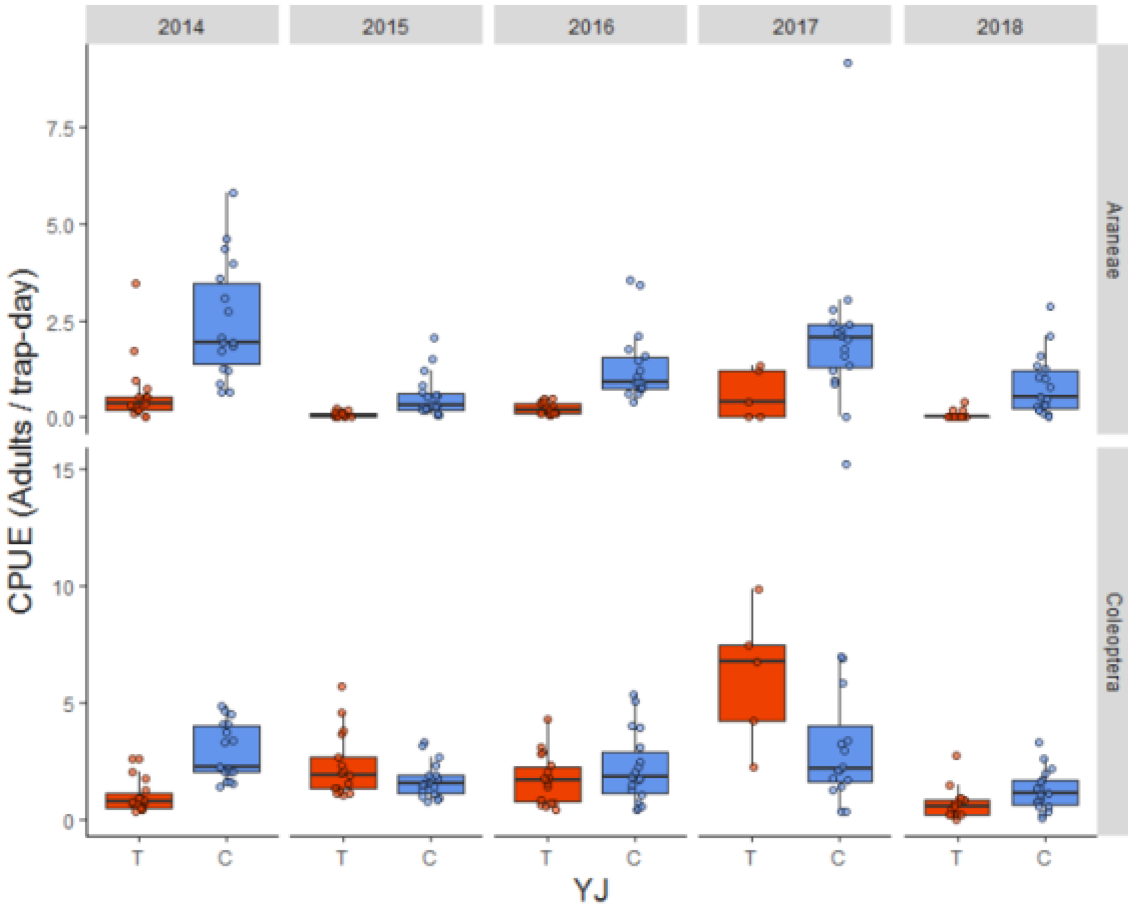


Figure 7-13: Catch per unit effort (abundance/24-hour trap period) of adult spiders (top) and beetles (bottom) in treatment (T) and control (C) areas across sampling years at YJ. Note that unscheduled wood removal occurred just prior to 2017 sampling.

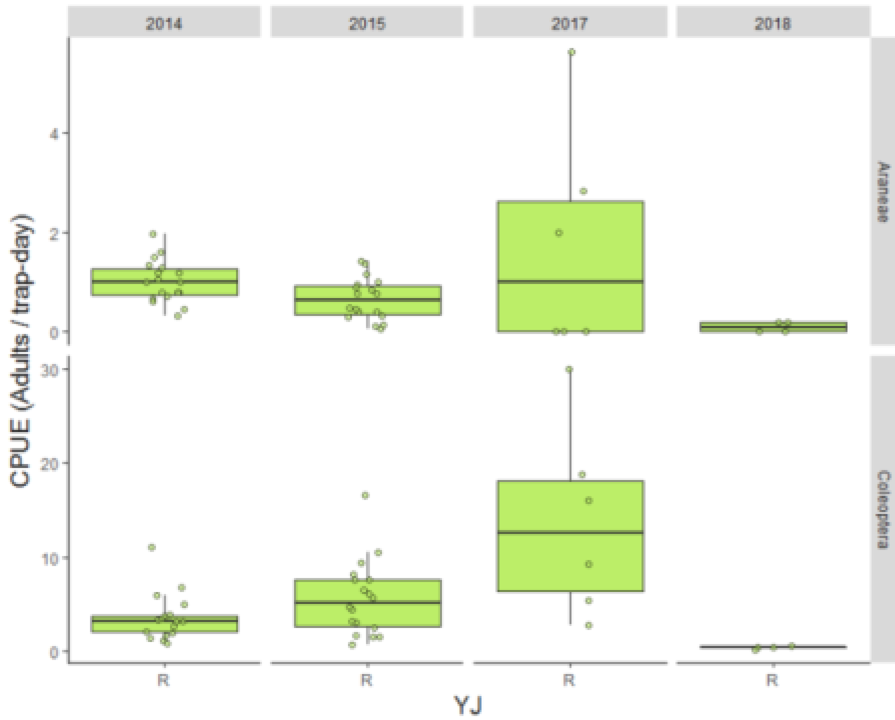


Figure 7-14: Catch per unit effort (abundance/24-hour trap period) of adult spiders in reference (R) areas across sampling years YJ.

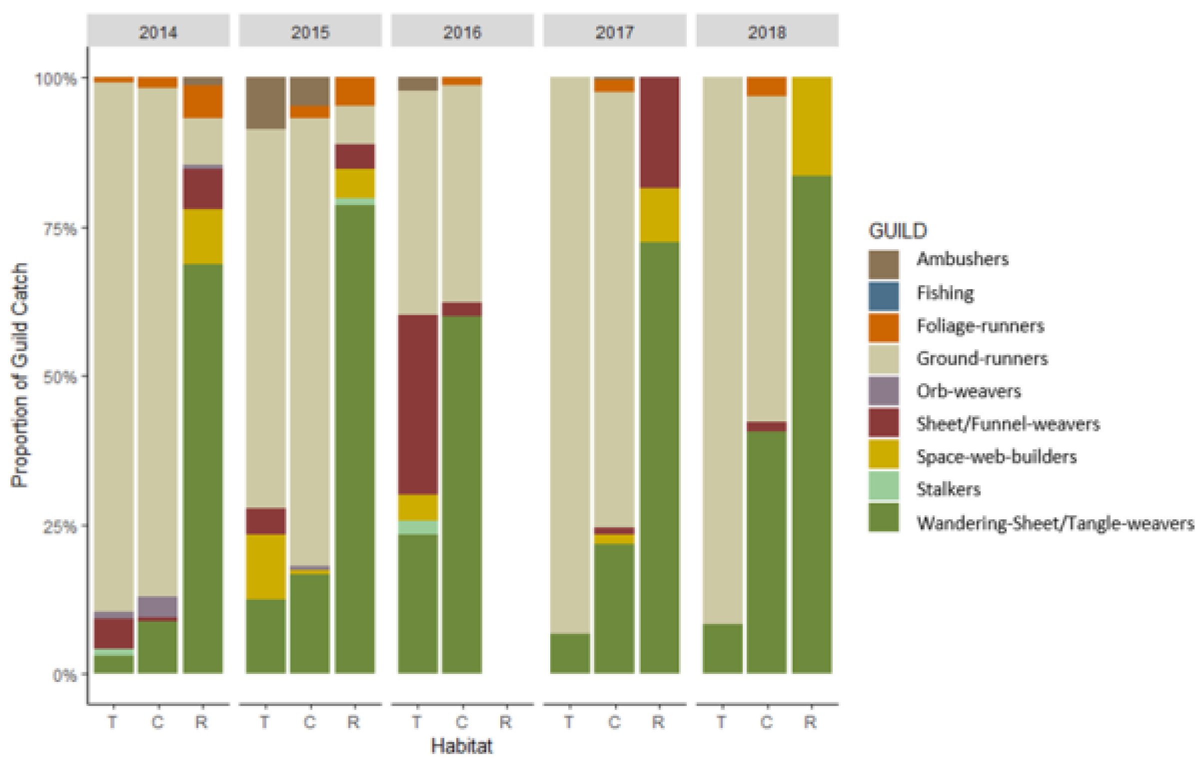


Figure 7-15: Relative proportion of spider functional guild catch in treatment (T), control (C), and upland reference (R) areas across sampling years at YJ. Reference area was not sampled in 2016.

<u>Pond 12 (P12; wood clearing from wetland and terrestrial habitat)</u>

Arthropods were sampled from Pond 12 for the first time in 2018, following large wood debris removal. Spider and beetle CPUE was similar in treatment and control samples (Figure 7-16). Reference samples had a much higher CPUE of beetles than spiders. Treatment and control plots were primarily dominated by ground-running spiders, followed by wandering-sheet/tangle-weavers (Figure 7-17). Reference samples also contained these guilds, yet the proportion of ground-running spiders was much lower than in the drawdown zone, with an increase in wandering-sheet/tangle-weavers, foliage runners, and sheet/funnel weavers (Figure 7-17)

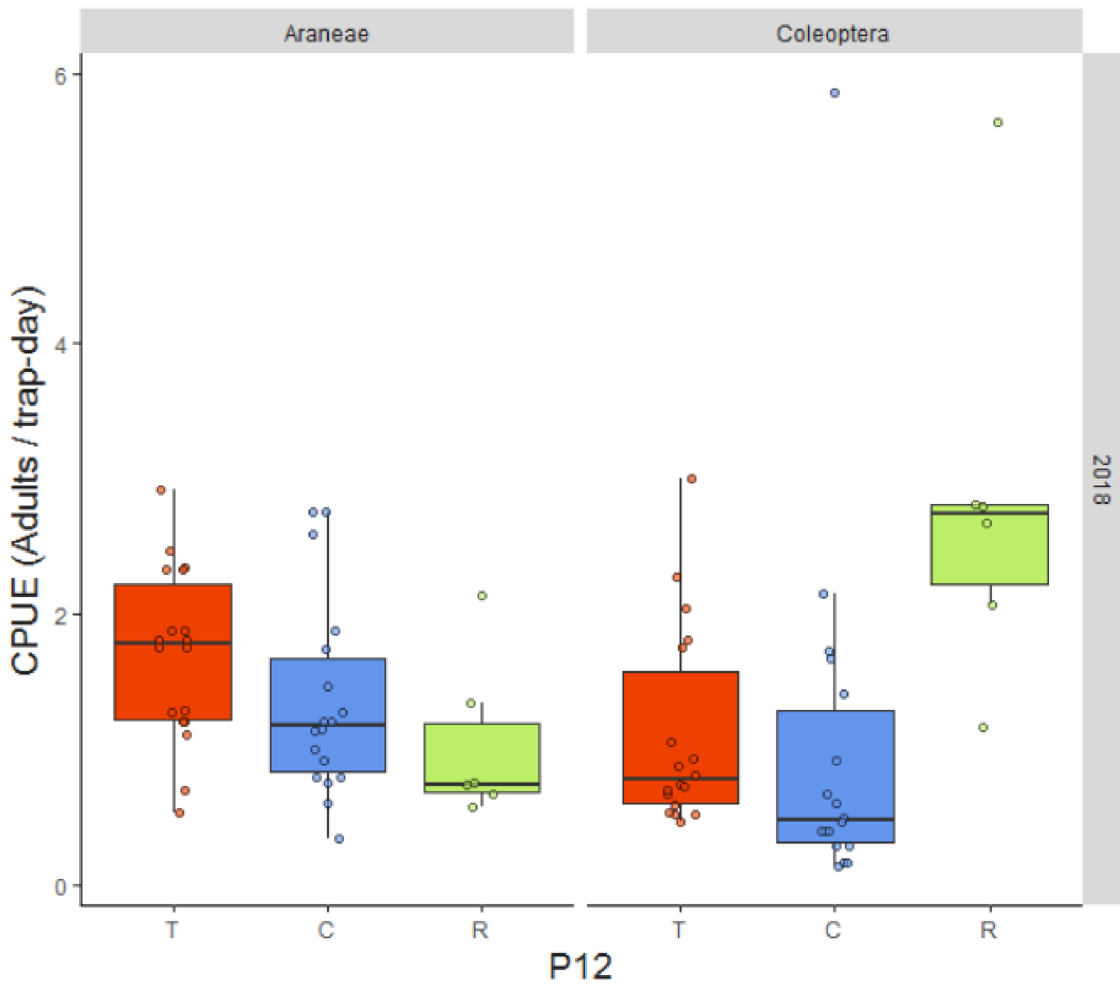


Figure 7-16: Catch per unit effort (abundance/24-hour trap period) of adult spiders (left) and adult beetles (right) in treatment (T), control (C), and reference (R) samples at Pond 12 (P12) in 2018.

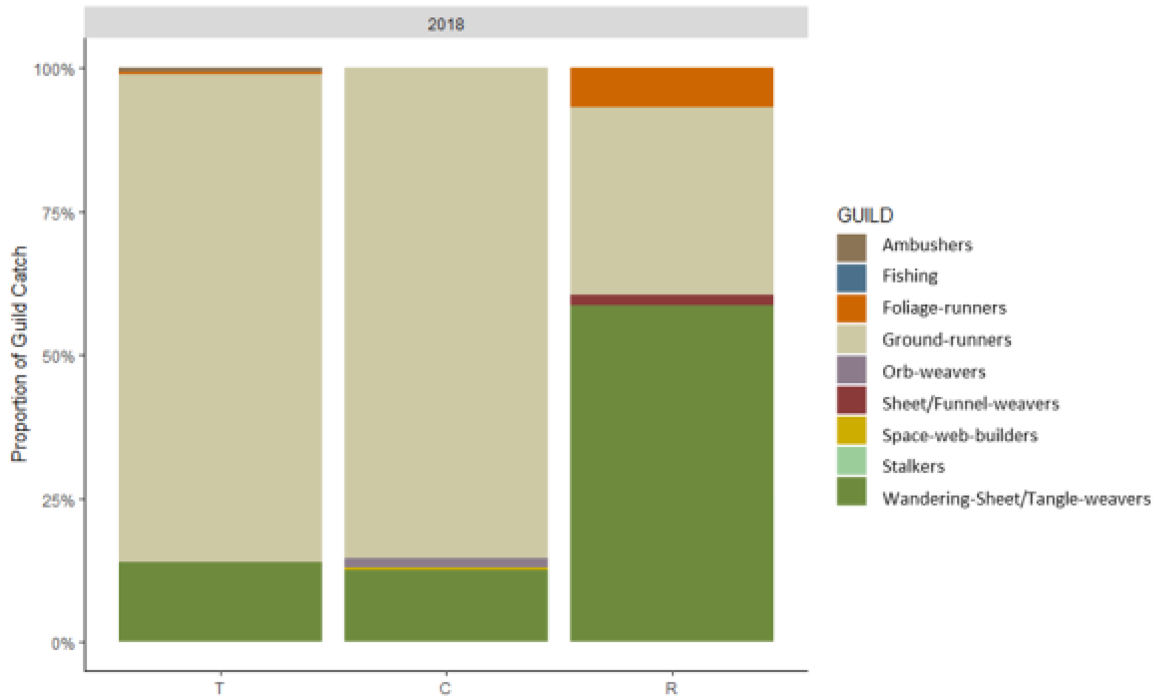


Figure 7-17: Relative proportion of spider functional guild catch in treatment (T), control (C), and upland reference (R) area at P12, 2018.

Discussion

Overall, there were no clear treatment-specific increases in beetle and spider relative abundances across all sites sampled (Figure 7-1). Given the different treatment application, site history, and conditions at each site, this is expected. It is likely that pre-existing differences between control and treatment areas within certain sites explains much of the variability in arthropod abundances. As well, relative abundance of arthropods at associated upland reference sites varied considerably among years, suggesting that inter-annual variation may explain some of the variation in arthropod abundances of the drawdown zone plots.

Arthropod relative abundance (CPUE) was similar between treatment and control areas at sites BAC-N, BAC-S, and P12 (see Table 2-2 for treatment types). At site VP-N (with woody debris removal and log boom treatments), spider relative abundance was initially greater in the area cleared of wood compared to the control plot (2014) but became similar over time (Figure 7-10). It is not known whether the abundance became similar between plots in 2018 due to the unplanned wood removal that was performed on the control plot (Figure 2-6), or if the beetle and spider abundance were gradually becoming more similar due to the recovery of vegetation that was observed for both controls and treatments over the study period.

Other sites, YJ (woody debris removal) and KM88 (revegetation), showed higher arthropod relative abundances in the control areas compared to treatment. At YJ, relative abundance of spiders not only appeared higher in the control area, but treatment CPUE did not increase at all in the 4 years post wood removal. While these results contradict assumptions regarding the benefits of wood removal, they are not surprising considering the unscheduled second re-treatment immediately prior to sampling in 2017. This repeated wood removal effectively limited our post-treatment monitoring duration at this site into two periods: 2014-2016 and 2017-2018, which likely is too limited a timeline to observe recovery of vegetation and arthropod responses.

Similarly, the first year of sampling data from KM88 indicates that control areas had higher arthropod abundances than planted areas. In this case however, mean CPUE may have been skewed by one control polygon (CU-3) which was the most upland polygon sampled, and appeared to house a dense, vigorous cover of naturally established sedges (Figure 2-7). CU-3 was also most comparable to the forested reference site in terms of relative abundance. Though planted sedge plug survival was high, vegetation density in the lower elevations of the drawdown zone (containing treatment and some control plots) remains very low and several more years of growth may be required for sufficient vegetation cover to support more abundant and more diverse communities. In particular, the portion of funnel-web and space-web builders in the drawdown zone are expected to increase as the drawdown zone vegetation structure increases (e.g., greater cover of shrubs or increased structural heterogeneity of vegetation). Due to the differences in quality of treatment and control polygons, any future analyses should not pool polygon units for comparisons.

Ground-hunting spiders, such as Wolf spiders, were much more abundant in the drawdown zone (control and treatment) than in reference sites. Conversely, Space-web and Sheet-web weaving spiders were more abundant at higher elevations in the upland reference sites. The lack of web-building spiders in the drawdown zone is likely due to their requirement of vegetation structure. The second most-represented guild in the drawdown zone was Wandering sheet/Tangle weavers. These spiders belong to the family Linyphiidae and may wander frequently to forage off their web (Uetz et al. 1999). Thus, while they use vegetation structure for prey capture, they are not reliant on their webs. This guild is expected to increase in dominance in drawdown zone plots where vegetation has successfully established.

Wolf spiders in the genus *Pardosa* are among the most dominant of the ground-dwelling spiders. In Wood et al. (2016) predictions were made for these spiders to decline overtime with vegetation establishment (see Table 7-1 in the 2015 annual report). Over the post-treatment period, we found evidence for this at Yellowjacket Creek. At YJ proportional abundance of ground-runners declined from 2014-2016 and increased in dominance in 2017 following re-treatment of that plot (Figure 7-15; ~30% ground-runners in 2016 to ~90% ground-runners in 2017).

At other sites, results were mixed. Ground running spiders at BAC-N increased in proportional abundance since treatment; at BAC-S remained similar; and at VP-N decreased minorly with increased diversification of guilds over time. The site conditions in the drawdown zone are still quite open at this stage in restoration, thus, it is not surprising that the ground-running spider guild is still quite dominant at most sites. It is expected that certain bare-ground-associated species within this guild (e.g., *Pardosa moesta*) will decrease overtime as vegetation establishes, being replaced by ground-runners that are less drought-tolerant.

Wood et al. (2016) also predicted a positive effect on ambush hunters (e.g., *Xysticus* spp.), sheet/funnel-web (e.g., *Agelenopsis* spp., *Agyneta* spp.) and spaceweb weavers (e.g., *Euryopis* spp.) as herb and shrub cover increased.

Funnel-web weavers expected to increase with grass and low-lying vegetation cover over previous bare ground. We noted increased proportional abundance of ambushers in VP-N T overtime and an increase in guild diversity. YJ T steadily increased in ambushers, sheet/funnel-weavers, and space-web weavers from 2014 to 2016, but following retreatment of this site in 2017, the guild composition was simplified to nearly all ground-running spiders.

Likewise, adult long-lipped tiger beetles, *Cicindela longilabris*, are xerophilous and tend to occur in bare areas (Larochelle and Larivière 2001). We found this species was quite dominant in initial wood-removal monitoring, where vegetation was lacking on treatment plots (Wood et al. 2015). At YJ treatment, *C. longilabris* was most abundant following initial wood removal in 2014 and following the retreatment of this plot. This species was replaced in dominance by the western tiger beetle, *Cicindela oregona,* in years 2015-2016, which was absent in from treatment in 2018. A similar pattern exists for these species at BAC-N treatment, which had *C. longilabris* only in the initial post-treatment sampling (2016) followed by increasing standardized abundance of *Cicindela oregona* from 2016-2018. These patterns are consistent with the predictions made in Table 7-1 from Wood et al. (2016).

It is too early to make any conclusions about the treatment application at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. As this area becomes established with vegetation and changes over time, we expect the ground-dwelling arthropod communities to likewise respond. Further years of monitoring are required.

A common environmental condition across all study sites has been the relatively low maximum reservoir levels since 2013, which have allowed vegetation in uppermost elevation bands of the drawdown zone (regardless of treatment application) to establish more successfully in the absence of annual inundation. While this has been beneficial to the end goal of revegetating the drawdown zone, it may conceal smaller-scale effects resulting from the physical works and planting projects, potentially obscuring the detection of treatment effects.

Appendix 3: Analysis of Arthropod Diversity

Introduction

We examined species richness and diversity to understand how arthropod communities responded to the different treatment applications. We focused our analyses on species of spiders (Order: Araneae) and carabid beetles (Family: Carabidae) collected from pitfall traps (see section 3.0). This section assists in answering MQ3 (see section 5.0).

Methods

Spiders and beetles were sampled in the field using pitfall trap arrays. Arthropods were assessed by habitat type (Treatment, Control, and Reference), site, and year. See section 3.0 for more detail on collection methods and Table 2-2 for information on treatment types.

Data Set

Data Set 1 was subset by site and by arthropod taxa (Araneae or Carabidae), as outlined above in Appendix 2. All samples were included as replicates.

Analysis

We evaluated how treatment affected standardized species richness and diversity of adult spiders (Order: Araneae) and ground beetles (Family: Carabidae) for each year of monitoring. Richness and diversity were standardized for each trap per 24-hours of active trapping time (trap-day). Overall mean standardized richness was calculated for control and treatment samples in each site and plotted in bar graphs with 90% confidence intervals for interpretation of overall differences (pooling samples from all years; number of replicates (n) were given in Appendix 2 Overall mean spider and beetle CPUE).

In addition, we considered each site separately in our comprehensive diversity analyses. We compared between 'treatment' and control areas at each site (see Table 2-2 for description of treatment application). Data from reference sites were included in multi-year figures as a visual aid and for inference about yearly variation in arthropod samples (excepting Araneae diversity and richness at Yellowjacket Creek, where large variation in reference data precluded its incorporation into figures). Reference data was excluded from statistical tests to limit comparisons directly to treatment versus control, which is most relevant to answering MQ3.

We conducted our analyses with the statistics program R v. 3.5.2 (R Core Team 2018). We considered the response of species diversity and richness to habitat type and (for sites with multi-year data) year using an ANCOVA with Gaussian error distribution. Response variables were checked for normal distributions. We calculated diversity (Shannon-Wiener Index) and richness (rarefied to a sample size of two to allow for comparison) using the R package 'vegan' (Oksanen et al. 2018). P-values less then or equal to 0.1 were considered significant. We used boxplots to display diversity and richness results (see Methods section of Appendix 2).

Results

<u>Overall Species Richness</u>

The post-treatment sampling at YJ (2014-2018), VP-N (2014, 2016-2018), BAC-n (2016-2018), BAC-S (2016-2018), KM88 (2018), and P12 (2018) yielded 192 species of arthropods from treatments and 193 species from controls. A substantial portion of these species (76%) were found in both treatment and control plots (147 species shared between T and C).

Differences in species richness (species standardized per trap-day) depended on habitat type (treatment vs. control), taxon (spiders vs. beetles), and study site. Overall spider richness was significantly lower in treatment relative to control at BAC-N, BAC-S, KM88, and YJ (Figure 7-18, left). Spider richness was similar between treatment and control for Pond 12 and VP-N. In contrast, overall beetle richness was greater in the treatment at BAC-S, Pond 12, and VP-N, relative to control richness (Figure 7-18, right), while being similar among remaining sites.

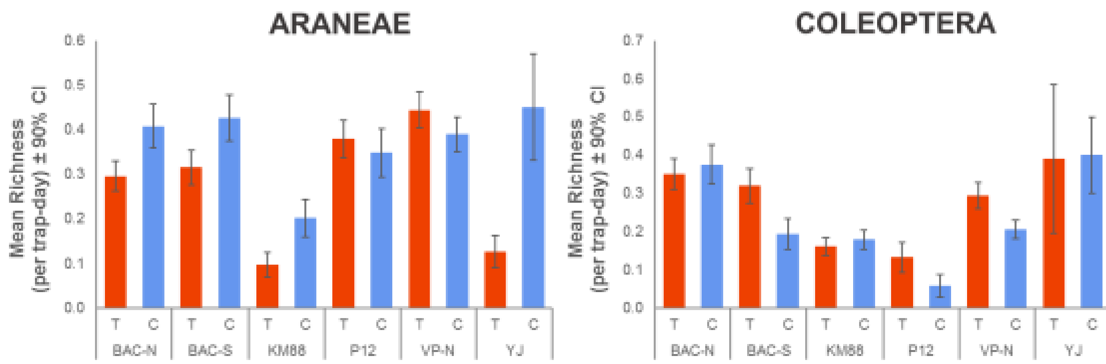


Figure 7-18: Standardized richness (species per trap-day) of spiders (left) and ground beetles (right) in treatment (T), control (C) samples from each study site, combining data from 2014-2018. Note: pre-treatment data omitted.

As site-specific differences have been noted, below, each site was presented as a separate case study due to different habitat features and types of physical works and/or revegetation treatments employed.

BAC-N (wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom)

Overall, there was a lower standardized spider species diversity at BAC-N in treatment than control plots ($F_{1,72}$=4.79, p=0.03). Year had no effect on diversity ($F_{1,72}$=0.38, p=0.54). Species diversity in control plots seemed more aligned with to the yearly fluctuations displayed in reference plots than did species diversity in treatment plots (Figure 7-19). We found no effect of habitat type ($F_{1,72}$=1.70, p=0.20) or year ($F_{1,72}$=1.74, p=0.19) on standardized spider species richness at BAC-N. Rarefied species richness at treatment and control sites seemed relatively consistent with trends in reference sites (Figure 7-19).

Habitat type did not affect standardized carabid diversity ($F_{1,72}$=1.99, p=0.16) or rarefied richness ($F_{1,72}$=1.13, p=0.29). Carabid species diversity decreased marginally over the four-year period ($F_{1,72}$=3.46, p=0.07; β=-0.007 ± 0.004), but carabid species richness did not significantly change over the years ($F_{1,72}$=1.24, p=0.27).

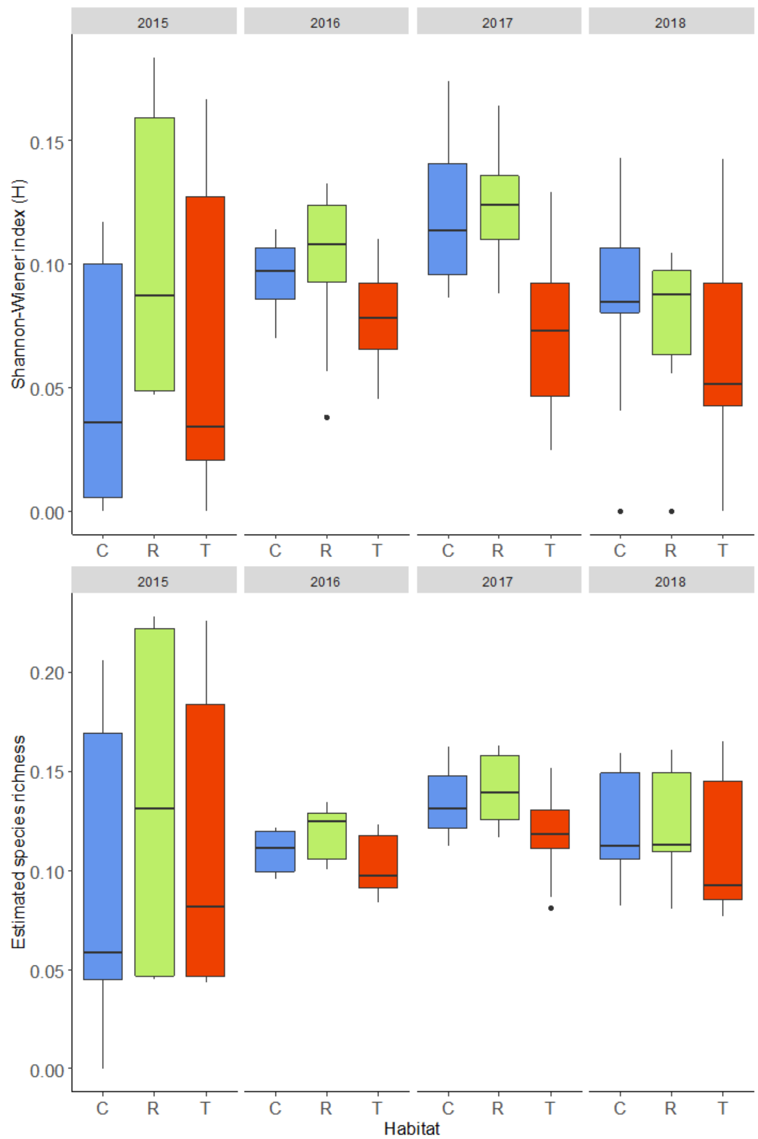


Figure 7-19: Spider species diversity (top) and rarefied spider species richness (bottom) per 24-hour period as a response to habitat type and year at BAC-N. Habitat types include treatment (T), control (C), and reference (R).

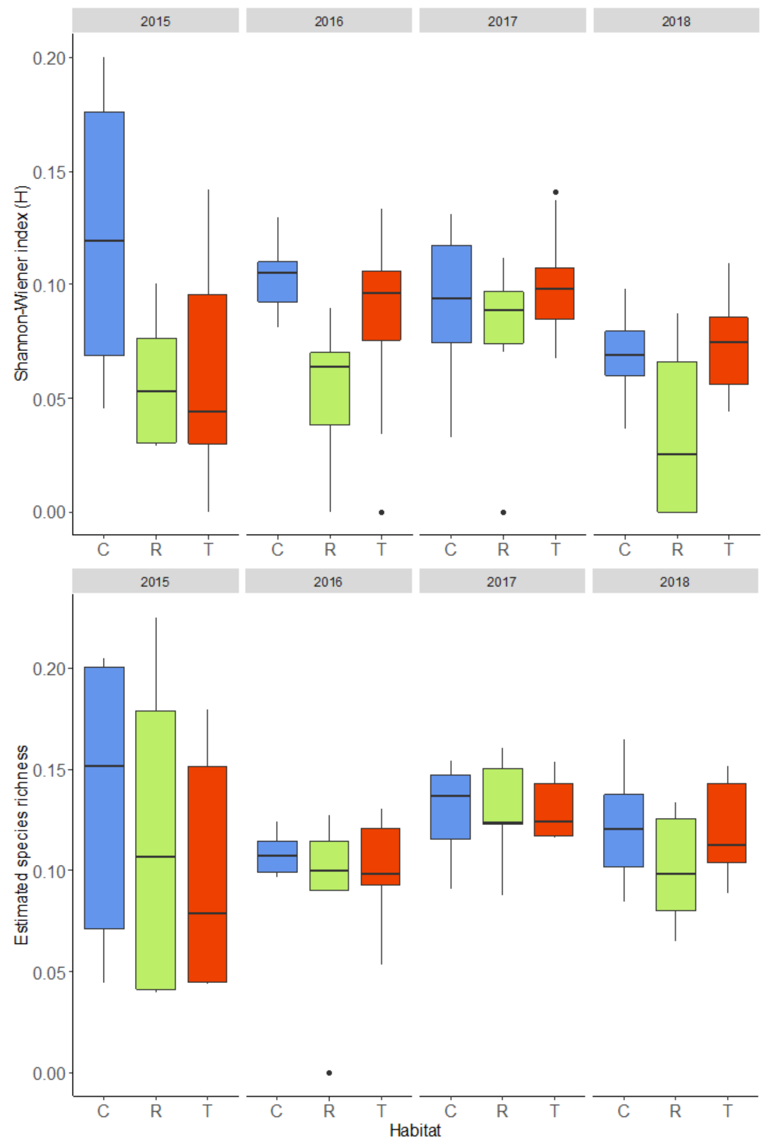


Figure 7-20: Carabid species diversity (top) and rarefied carabid species richness (bottom) per 24-hour period as a response to habitat type and year at BAC-N. Habitat types include treatment (T), control (C), and reference (R).

<u>BAC-S (wood clearing from terrestrial habitat, mounds, and revegetation)</u>

Spider species diversity (standardized) was higher in control plots versus treatment plots at BAC-S ($F_{1,72}$=10.01, p=0.002), while year had no impact on diversity ($F_{1,72}$=1.54, p=0.22). There was an apparent reduction in spider species diversity in control and reference plots in 2018 from the previous year, which was not reflected in treatment plots (Figure 7-21). Overall, treatment plots showed a lower standardized spider richness than controls ($F_{1,72}$=4.48, p=0.04). Spider species richness increased marginally over the four-year period ($F_{1,72}$=5.23, p=0.03; β=0.01±0.03). Yearly fluctuations in species richness in the reference areas were reflected by trends in control and, to a lesser extent, treatment areas (Figure 7-21).

Standardized carabid species diversity ($F_{1,72}$=14.74, p<0.001) and richness ($F_{1,72}$=6.34, p=0.01) were significantly higher in treatment plots than in control plots (Figure 7-22). Year did not significantly affect carabid diversity ($F_{1,72}$=0.16, p=0.69) or richness ($F_{1,72}$=0.03, p=0.87).

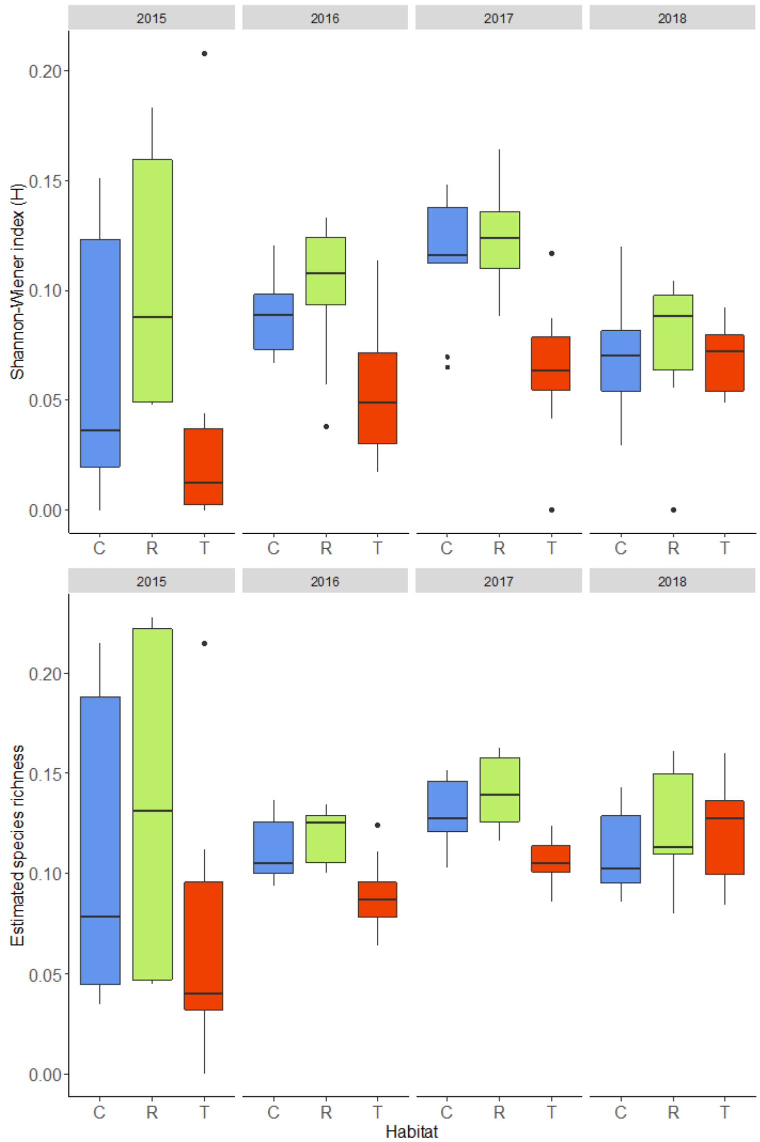


Figure 7-21: Spider species diversity (top) and rarefied spider species richness (bottom) per 24-hour period as a response to habitat type and year at BAC-S. Habitat types include treatment (T), control (C), and reference (R).

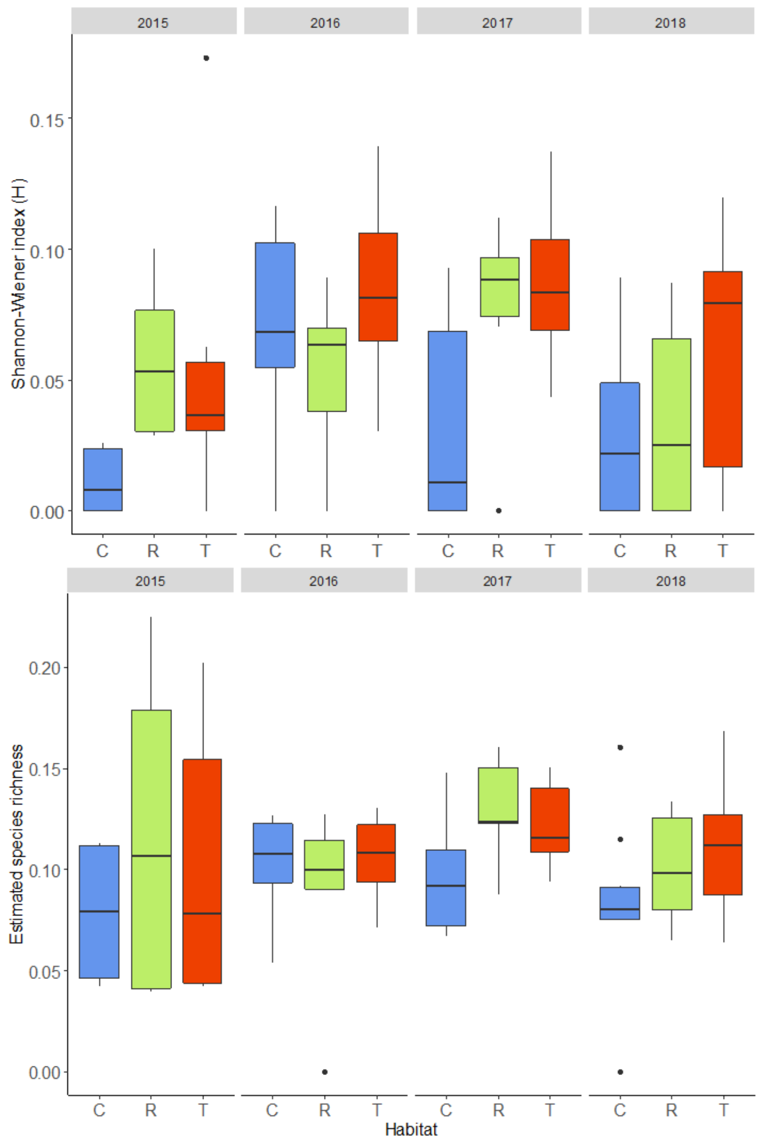


Figure 7-22: Carabid species diversity (top) and rarefied carabid species richness (bottom) per 24-hour period as a response to habitat type and year at BAC-S. Habitat types include treatment (T), control (C), and reference (R).

KM88 (sedge plug revegetation)

Standardized spider species diversity ($F_{1,58}$=9.57, p=0.003) and richness ($F_{1,58}$=8.03, p=0.01) were greater in control plots than in treatment plots (Figure 7-23). However, treatment did not significantly affect carabid species diversity ($F_{1,58}$=0.17, p=0.68) or richness ($F_{1,58}$=0.74, p=0.39) (Figure 7-24).

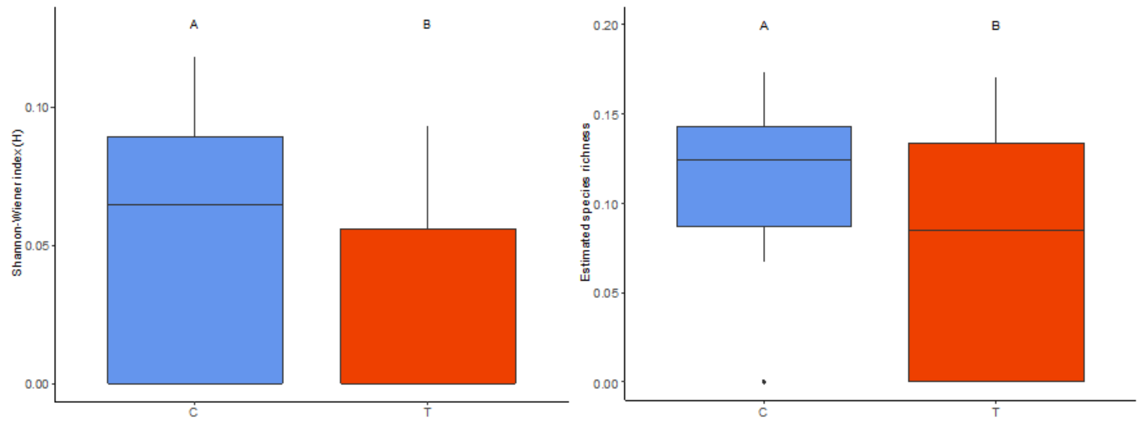


Figure 7-23: Spider species diversity (left) and rarefied spider species richness (right) per 24-hour period as a response to habitat type and year at KM88. Habitat types include treatment (T) and control (C). Letters denote significant differences. All samples are from 2018.

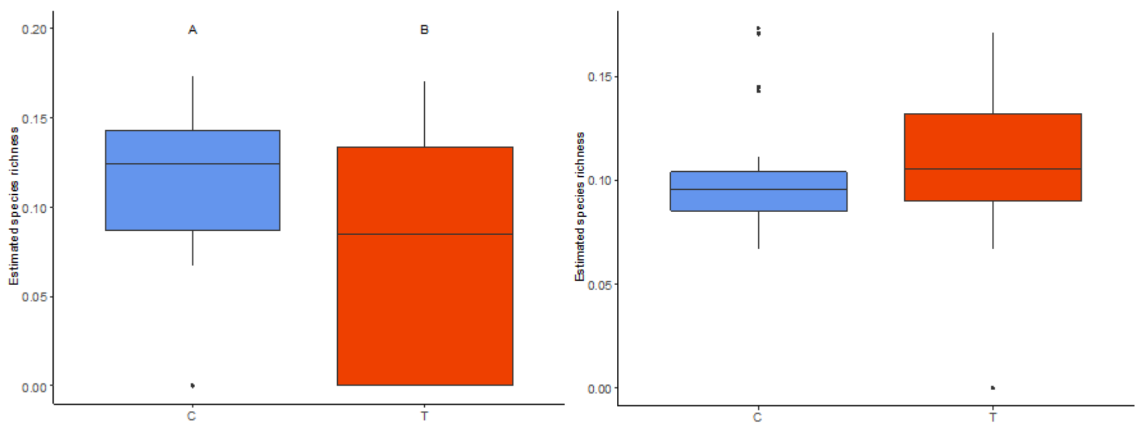


Figure 7-24: Carabid species diversity (left) and rarefied carabid species richness (right) per 24-hour period as a response to habitat type and year at KM88. Habitat types include treatment (T) and control (C). All samples are from 2018.

<u>VP-N (wood clearing from wetland and terrestrial habitat, and log boom)</u>

Habitat type had no effect on spider species diversity ($F_{1,123}$=0.13, p=0.29) or spider species richness ($F_{1,123}$=0.001, p=0.97) at VP-N. Similarly, there was no effect of year on diversity ($F_{1,123}$=1.18, p=0.28) or richness ($F_{1,123}$=1.12, p=0.29). Yearly variation in spider species diversity and richness evident in reference sampling was reflected to some degree in control and treatment samples (Figure 7-25).

Carabid species diversity was significantly higher in treatment than in control plots ($F_{1,123}$=5.99, p=0.02), but there was no significant effect of habitat type on carabid species richness ($F_{1,123}$=1.03, p=0.31) (Figure 7-26). Likewise, carabid species diversity declined somewhat over the years ($F_{1,123}$=10.32, p=0.002; β=-0.007 ± 0.002), but there was no effect of year on carabid species richness ($F_{1,123}$=1.03, p=0.31).

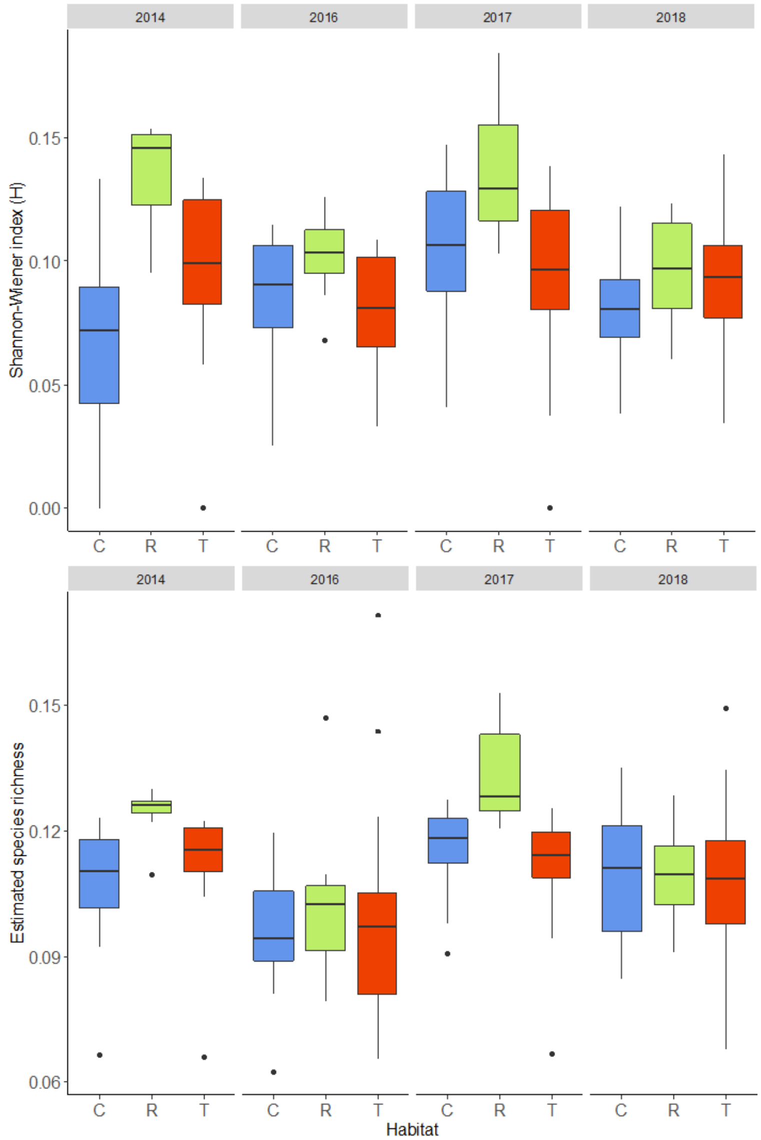


Figure 7-25: Spider species diversity (top) and rarefied spider species richness (bottom) per 24-hour period as a response to habitat type and year at VP-N. Habitat types include treatment (T), control (C), and reference (R).

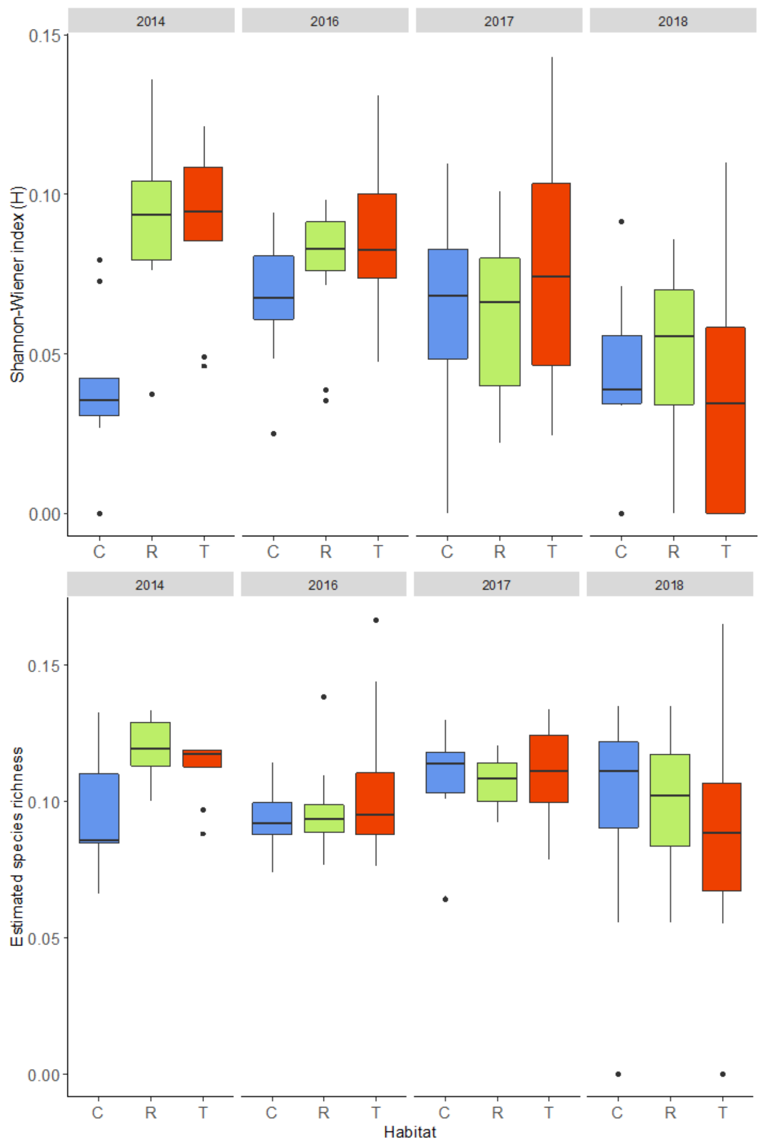


Figure 7-26: Carabid species diversity (top) and rarefied carabid species richness (bottom) per 24-hour period as a response to habitat type and year at VP-N. Habitat types include treatment (T), control (C), and reference (R).

<u>YJ (wood clearing from terrestrial habitat)</u>

Spider species diversity was higher in control areas than in treatment areas ($F_{1,160}$=10.97, p=0.001). Year did not significantly impact diversity ($F_{1,160}$=2.31, p=0.13). Habitat type did not significantly impact spider species richness ($F_{1,160}$=1.90, p=0.17). However, richness increased over time ($F_{1,160}$=2.78, p=0.10; β=0.02 ± 0.01). Following this re-treatment, spider species diversity and richness plummeted in treatment (but not control) areas.

Neither carabid species diversity ($F_{1,160}$=0.02, p=0.89) nor richness ($F_{1,160}$=0.86, p=0.36) were affected by treatment (Figure 7-28). Carabid species richness increased over the years ($F_{1,160}$=5.93, p=0.02; β=0.05 ± 0.02), while there was no overall affect of time on diversity ($F_{1,160}$=1.28, p=0.26).

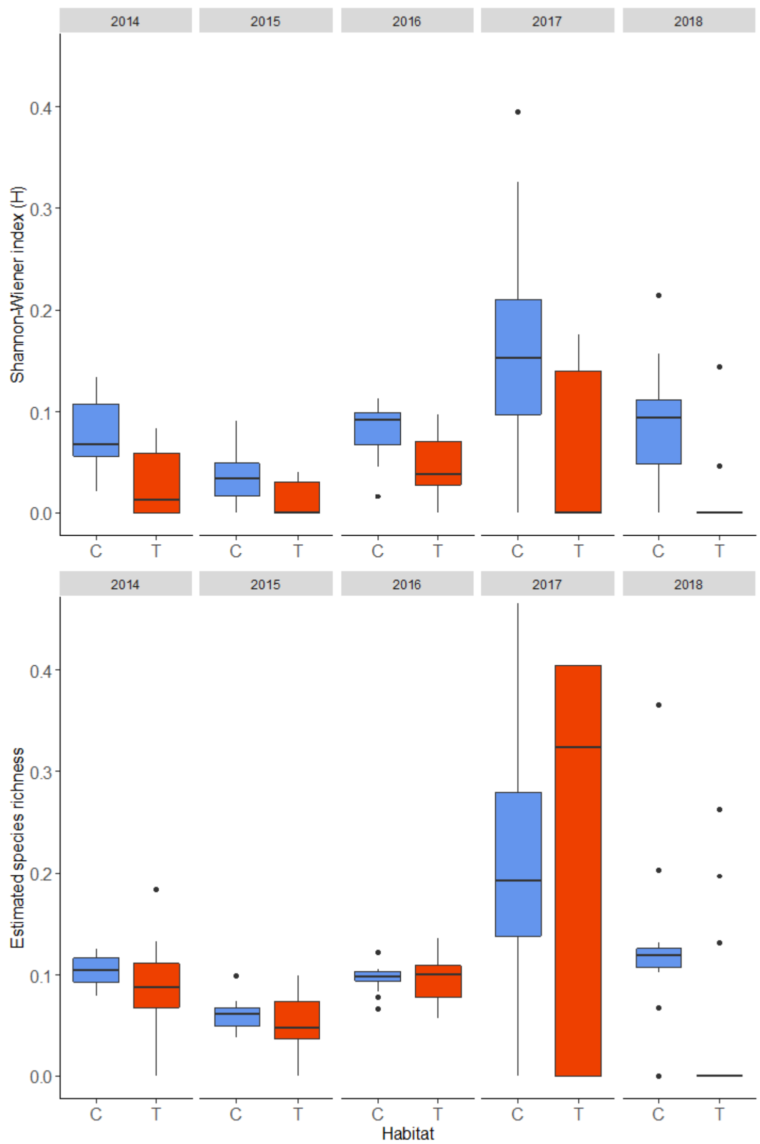


Figure 7-27: Spider species diversity (top) and rarefied spider species richness (bottom) per 24-hour period as a response to habitat type and year at YJ. Habitat types include treatment (T), control (C), and reference (R). Note: reference data for spiders was not plotted due to large variation which precluded its incorporation into figures.

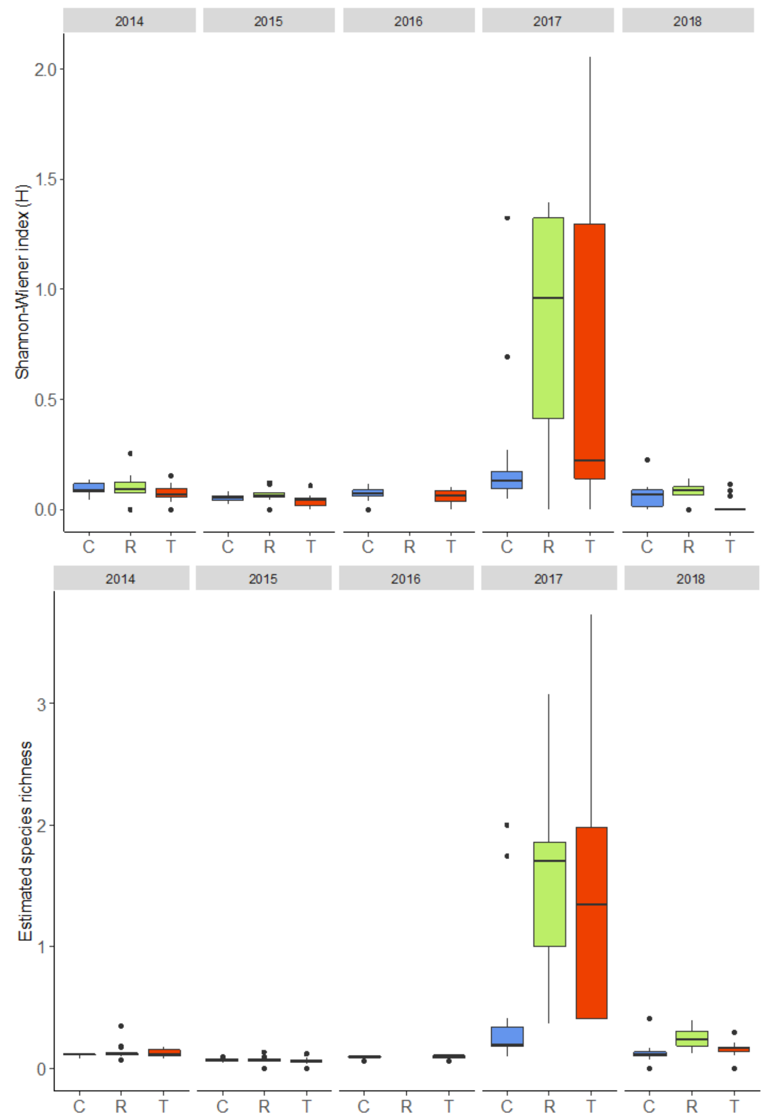


Figure 7-28: Carabid species diversity (top) and rarefied carabid species richness (bottom) per 24-hour period as a response to habitat type and year at YJ. Habitat types include treatment (T), control (C), and reference (R).

<u>P12 (wood clearing from wetland and terrestrial habitat)</u>

There was no effect of treatment on spider species diversity ($F_{1,34}$=0.09, p=0.77) or richness ($F_{1,34}$=0.005, p=0.95) (Figure 7-29). However, there was a higher carabid species diversity ($F_{1,34}$=6.82, p=0.01) and richness ($F_{1,34}$=7.30, p=0.01) in treatment plots compared to control plots (Figure 7-30).

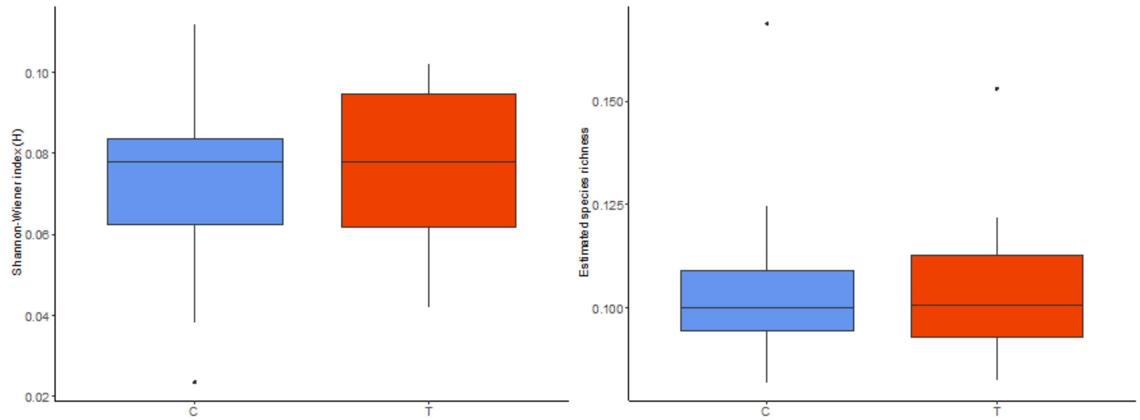


Figure 7-29: Spider species diversity (left) and rarefied spider species richness (right) per 24-hour period as a response to habitat type and year at P12. Habitat types include treatment (T) and control (C).

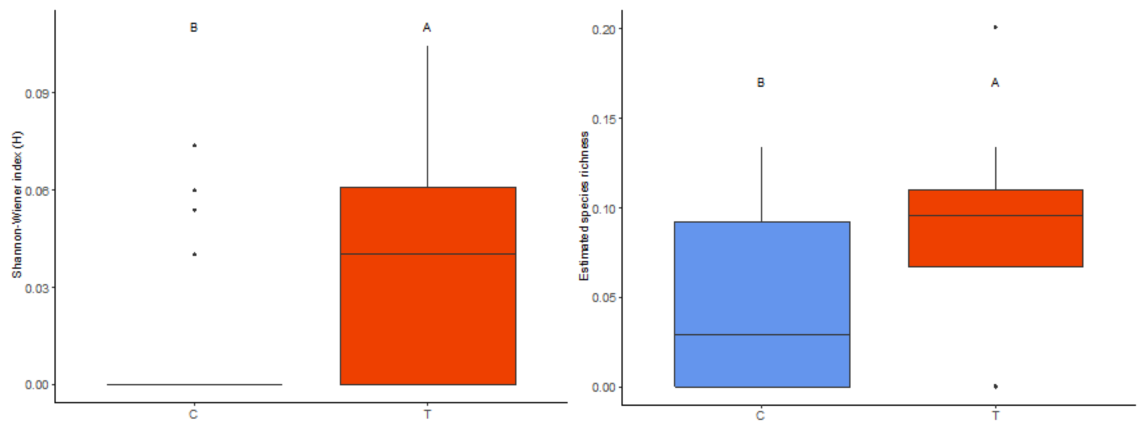


Figure 7-30: Carabid species diversity (right) and rarefied carabid species richness (left) per 24-hour period as a response to habitat type and year at P12. Habitat types include treatment (T) and control (C). Letters denote significant differences.

Discussion

In summary, the effect of treatment type on spider and carabid beetle richness and diversity were mixed (Table 7-2). Overall, there was no consistent pattern in how treatment affected arthropod diversity and richness. In many cases there was no observable effect of treatment. In cases where arthropod richness and diversity were affected by treatment, carabid beetle species always responded positively, and spiders always responded negatively.

Only one site (BAC-S), whose treatments included woody debris removal, revegetation, and mound/windrow creation, showed a response of both spider and carabid beetle diversity and richness to treatment type (negative for spiders, positive for carabid beetles). This result is consistent with pre-treatment assessments of spider and ground beetle richness reported for 2015 data (Wood et al. 2016, Figure 6-10) and is thus related to pre-existing differences between the treatment and control areas rather than a treatment effect. This demonstrates the importance of using pre-treatment data to evaluate post-treatment results. In the absence of this information, the greater beetle richness of treatment samples might wrongly be attributed to a treatment effect. In this case, BAC-S treatment inherently housed more carabid species than the adjacent control, regardless of treatment application (Wood et al. 2016, Figure 6-10). Unfortunately, the lack of pre-treatment data for most of our study sites (VP-N, YJ, KM88, and P12) makes for problematic interpretation of treatment effects. Observed differences between these treatments and controls are not able to be ascribed to a treatment effect. Experimental designs should always consider pre-treatment sampling to control for pre-existing differences between paired treatment and control areas.

Table 7-2: Summary of the effects of treatment on species diversity and richness of spiders (Araneae) and ground beetles (Carabidae) at each study site. Positive effects are indicated by the symbol '+', negative effects are indicated by the symbol '-', and non-significant effects were left blank. Sites include Bush Arm Causeway Northwest (BAC-N) and Southwest (BAC-S), Big Bend (KM88), Yellowjacket Creek (YJ), Valemount Peatland North (VP-N), and Pond 12 (P12).

Reach	Site	Prescription type	Taxon	Diversity measure	Predicted effect	Effect
BUSH ARM	**BAC-N**	Wood removal, pond clearing, mound/windrow creation, revegetation, log boom (2015)	Araneae	Diversity	+	-
				Richness	+	
			Carabidae	Diversity	+	
				Richness	+	
	BAC-S	Wood removal, mound/windrow creation, revegetation (2015)	Araneae	Diversity	+	-
				Richness	+	-
			Carabidae	Diversity	+	+
				Richness	+	+
	KM88	Revegetation with sedge plugs (2013)	Araneae	Diversity	+	-
				Richness	+	-
			Carabidae	Diversity	+	
				Richness	+	
CANOE REACH	**VP-N**	Woody debris removal and log boom installation (2014)	Araneae	Diversity	+	
				Richness	+	
			Carabidae	Diversity	+	+
				Richness	+	
	YJ	Woody debris removal (2014 & 2017)	Araneae	Diversity	+	-
				Richness	+	
			Carabidae	Diversity	+	
				Richness	+	
	P12	Woody debris removal (2018)	Araneae	Diversity	+	
				Richness	+	
			Carabidae	Diversity	+	+
				Richness	+	+

One possible contribution to the negative effect of year on species diversity at BAC-N (see Table 7-2 for treatments at this site) was a decline in carabid diversity from 2017 to 2018, which was reflected in treatment, control, and reference samples (Figure 7-20). This decline may therefore be reflective of overall fluctuations in arthropod communities in the greater study area, rather than any within-site or treatment-specific effects. Yearly patterns of carabid species richness were largely consistent between control, treatment, and reference samples (Figure 7-20) and were likewise observed at BAC-S (Figure 7-21).

There were several types of treatments applied at different sites. While some sites had similar treatments (such as wood removal), a lack of similarity between sites in factors such as treatment applications, substrate composition, and site history prevented us from aggregating treatment effect across sites.

The differential responses of spiders and ground beetles to treatment type may speak to the life history and morphology of the two arthropod groups. The removal or mounding of wood creates more open habitat in the area sampled by pitfall traps. Spiders are particularly sensitive to desiccation, and open habitats would select for only those few species that are particularly robust against exposed environments (e.g., *Pardosa moesta*). This would explain the lower species diversity or richness of spiders in treatment areas compared to control areas. A greater diversity of adult carabid beetles may be able to inhabit open environments than the ground-dwelling spiders, thus their diversity may not have been impacted as severely by treatment. However, if vegetation cover increases in cleared areas over time, spider richness and diversity may increase.

Revegetation (at KM88) had no effect on carabid beetle species diversity or richness, and negatively affected spider diversity and richness. However, these results are limited by a lack of both temporal and spatial treatment replication. In addition, pre-treatment comparisons between polygons are lacking, thus it is not known whether the treatment areas were ecologically disadvantaged relative to the controls. Despite the initial findings that revegetation may not improve arthropod diversity or richness, these restraints make it difficult to make any real inferences.

It is too early to make any conclusions about the treatment application at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. As this area becomes established with vegetation and changes over time, we expect the ground-dwelling arthropod communities to likewise respond. Further years of monitoring are required.

An important consideration is that treatment effectiveness should be considered in the context of reservoir levels. Historically there has been a large amount of variation in reservoir levels, including years where the reservoir exceeded its normal operating maximum (such as in 2012). However, in the duration of time that the CLBMON-11A sampling has taken place (from 2014 to 2018), the reservoir has operated under its maximum. This may prevent us from fully understanding how certain treatment techniques, such as log boom installation, could operate to preserve habitat integrity and species richness or diversity in an area during peak reservoir conditions.

Appendix 4: Analysis of Arthropod Composition

Introduction

Current approaches in community ecology focus less on species richness, and increasingly on the processes governing the variation in species assemblages among sites (or samples). Community analyses were performed to assess the variation in species assemblages within and between habitat types.

Methods

Spiders and beetles were sampled in the field using pitfall trap arrays. See section 3.0 for more detail on collection methods.

Data Set

Data Set 1, Data Set 5, and Data Set 6 were compiled into a sample x species matrix. P12 and KM88 were excluded from these analyses due to data limitation. Species CPUE were averaged for each Site x Habitat x Year combination prior to Hellinger transformation (square root of relative abundance).

Explanatory data were also averaged for each Site x Habitat x Year. From the main data sets, we included the following: Live Organic Material (LOM = plant matter), Canopy Cover (CC), water, mineral soil, mixed soils, Dead Organic Material (DOM), Rock, Wood, Moss, mean T, variation in T, min T, max T, mean RH, and variation in RH. Variance in temperature and relative humidity was computed as the sum of the difference between each value and the mean of all values in the sample (squared), divided by one less than the sample size.

Thus, the final community matrix contained 52 sampling unit rows (Site x Habitat x Year combinations), 18 site/environmental explanatory variables (Site, Habitat, Year, and environmental variables), and 279 response variables (CPUE of 178 spider spp. and 101 carabid spp.). Sites included BAC-N (n = 8), BAC-S (n = 8), GDF (n = 4), VP-N (n = 12), and YJ (n = 14). A schematic of the community matrix is provided in Figure 7-31.

A separate community matrix was created for KM88, that included average CPUE data for each spider and ground beetle species in columns (29 spider species and 19 beetle species) by 7 rows (one for each polygon: TU1, TU3, TU5, CU1, CU2, CU3, R). These data were derived from the same number of samples outlined in Appendix 2. Environmental data were not examined at this site as sampling was limited to only 1 year. A simple summary of the number of shared species between treatment and control are given for KM88 and Pond 12.

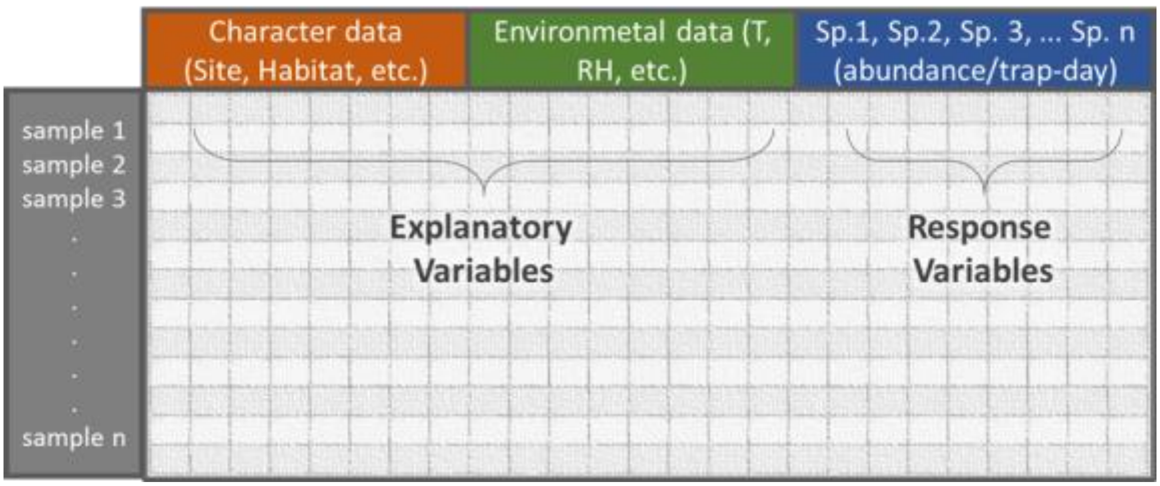


Figure 7-31: Diagram of the matrix format used in community analyses, which presents species abundance data in columns for each species x sample rows. Explanatory variables are optionally included for association with species assemblages.

Analysis

Species composition was assessed using Hellinger Distance ('D17'; Rao 1995). D17 is insensitive to double-zeros and is variance-stabilising. This metric gives less weight to species with low abundance (rare species) than abundant (common) species. Hellinger distance is highly recommended for ordination of species abundance data (Legendre and Gallagher 2001; Rao 1995). This metric involves calculating Euclidean distance on Hellinger-transformed species-abundance data (square-root of relative abundance), between two samples (1, 2) as follows:

$$D_{17\,(x_1,x_2)} = \sqrt{\sum_{i=1}^{p}\left[\sqrt{\frac{y_{1i}}{y_{1+}}} - \sqrt{\frac{y_{2i}}{y_{2+}}}\right]^2},$$

where the abundance of each species i in each sample unit $(1, 2)$ is relativized by the total abundance of all species in the sample (y_{1+}, y_{2+}) to provide frequencies of species in each sample, prior to square-root transformation.

Thus, in this manner, pairwise distances are calculated between samples which measure assemblage differences and translate into spatial distance on ordination plots. Samples sharing many species in common (with similar relative abundances) have a low value of D17 and are plotted close together in ordination space. Samples with few shared species (and vastly different relative abundance) have a high value of D17 and are plotted far apart in ordination space.

We performed non-metric multidimensional scaling ordinations (NMDS) to determine the between and within-treatment compositional differences in species assemblages. By overlaying species centroids on the ordinations, we were able to further examine associations between sites and species. The identification of species assemblages allows for investigation into the ecological requirements that are common to the group, rather than evaluating the ecological needs of each species individually. Unlike other ordination techniques that attempt to maximise the variance between samples, NMDS represents, as closely as possible, the pairwise dissimilarity between samples in two or three axes. NMDS is also a non-metric approach which is based on the ranks of distance coefficients, which makes it more flexible for analysis of non-normal, non-linear, heterogeneous, and zero-inflated datasets, which are common in ecology (Clarke 1993). While the magnitude of distance is lost, ranks are especially useful to resolve ecological patterns in community composition. Spider and beetle species were analysed separately, as were study areas. BAC-N, BAC-S, and GDF were examined together due to their proximity, similarity of treatment prescription history, and experimental design. The composition of each polygon at KM88 was plotted separately for interpretation of differences between spider and beetle assemblages within this site.

To test for differences in Hellinger distance between control and treatment samples, removing the effect of year, Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson 2001) tests were performed on each spider and carabid beetle distance matrix, specifying year as a blocking variable, and running 9999 permutations.

Pond 12 and KM88 assemblages were described by proportion of species shared and unique to each habitat type. These were assessed with Venn diagrams using the package 'VennDiagram' in R (Chen 2015). These graphically display the number of unique species in each habitat and the number of species that were shared between habitats. The area of each ellipse is drawn approximately proportional to the total number of species observed for that treatment type, allowing for comparisons of compositional similarity.

NMDS ordinations and PERMANOVAs were performed in the vegan Community Ecology package (Oksanen et al. 2018) in the R language (R Core Team 2018).

Results

<u>BAC-N (wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom), BAC-S (wood clearing from terrestrial habitat, mounds, and revegetation), and GDF (untreated, upland reference)</u>

Composition differences were greatest between upland reference and drawdown zone communities; however, treatment and control plots were also found to contain discrete species assemblages. Spider composition in the drawdown zone samples at Bush Arm Causeway sites differed from the 2015 pre-treatment for both treatments and controls (Figure 7-32, top). At BAC-N, there appears to be a linear effect of year on both treatment and control composition along the y-axis. This pattern was less consistent for BAC-S and the upland reference (GDF).

BAC-S had similar spider species composition between treatment and control for pre-treatment sampling conducted in 2015, however, after wood removal and mound creation at BAC-S, spider composition diverged between treatment and control areas (i.e., 90% confidence regions are not overlapping in top-left ordination plot). Spider composition appears to converge from 2016 to 2018, with a trend towards becoming more similar between T and C. By 2018, the BAC-S T and C had 61.5% of their spider species in common. These spider assemblages were most strongly associated with canopy cover (CC: $R^2 = 0.80$, $p = 0.0002$), mineral soil (MINERAL: $R^2 = 0.66$, $p = 0.0006$), dead organic matter (DOM: $R^2 = 0.55$, $p = 0.003$), moss cover (MOSS: $R^2 = 0.55$, $p = 0.002$), daily minimum temperature (min T: $R^2 = 0.31$, $p = 0.06$), live organic matter cover (LOM: $R^2 = 0.27$, $p = 0.09$), and daily average relative humidity (mean RH: $R^2 = 0.26$, $p = 0.09$). Ground beetle assemblages were most strongly associated with canopy cover (CC: $R^2 = 0.80$, $p = 0.0002$), mineral soil (MINERAL: $R^2 = 0.74$, $p = 0.0002$), moss cover (MOSS: $R^2 = 0.62$, $p = 0.0007$), dead organic matter (DOM: $R^2 = 0.56$, $p = 0.0025$), and variation in temperature (var T: $R^2 = 0.62$, $p = 0.0006$). LOM cover was not significantly related to beetle species assemblages ($R^2 = 0.26$, $p = 0.11$).

When the effect of year was controlled for, neither spider nor ground beetle species composition differed between treatment and control plots at BAC-N and BAC-S (Araneae BAC-N: $F_{1,7} = 1.13$, $p = 0.13$; BAC-S: $F_{1,7} = 1.13$, $p = 0.13$; Carabidae BAC-N: $F_{1,7} = 1.40$, $p = 0.25$; BAC-S: $F_{1,7} = 1.48$, $p = 0.13$, blocked by year).

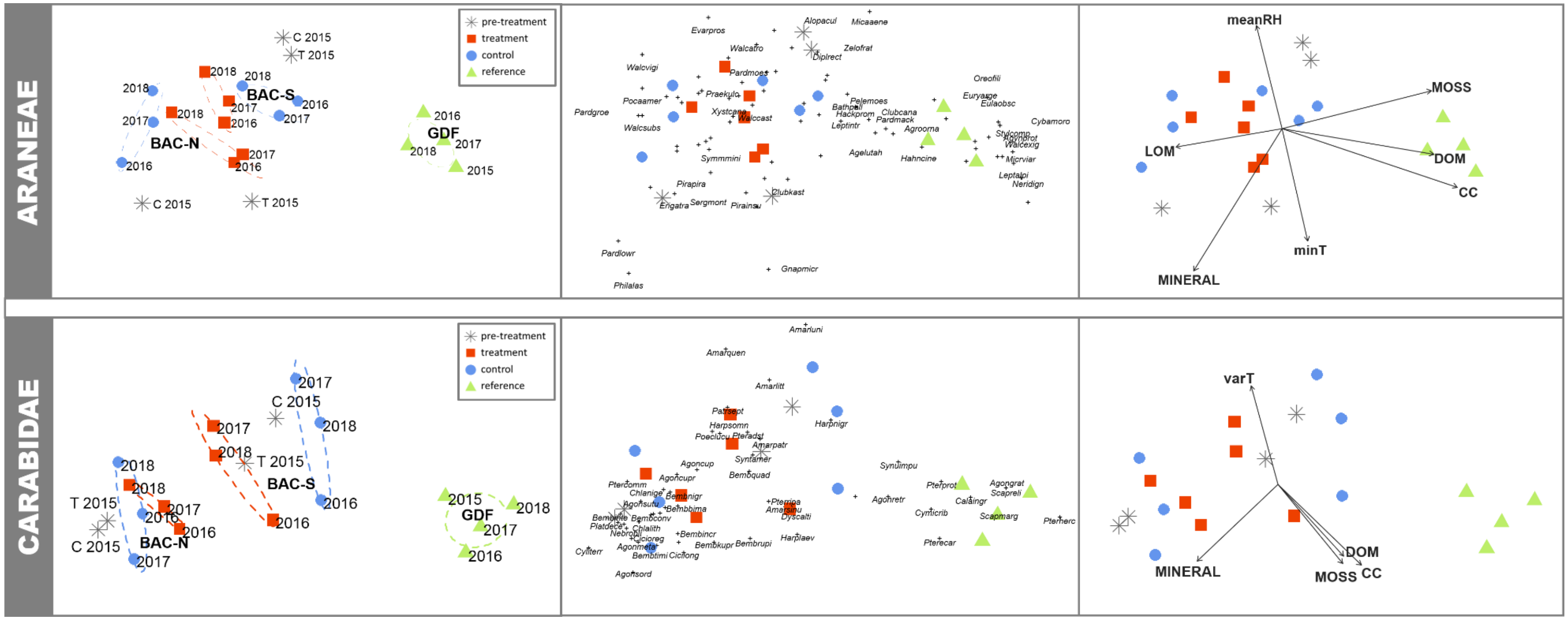


Figure 7-32: NMDS ordination of Bush Arm Causeway (North and South) spider species (top) and ground beetle species (bottom) with 90% confidence regions for Site x Habitat (left), species centroids (centre), and significant ($p \leq 0.1$) relationships with environmental variables (right) overlaid. Habitat types: pre-treatment, treatment (T), control (C), and upland reference (R; located nearby at Goodfellow Creek). Overlapping species not shown for clarity. Species codes defined in Table 7-14.

KM88 (sedge plug revegetation)

Based on only one year of sampling, spider and beetle species composition appears to be similar between control and treatment polygons at KM88 (Figure 7-33). The treatment contained 28 species, while the control housed 30 species of spiders and beetles. Overall, a large portion of the treatment species were shared with either control or reference samples, leaving only five species not found in other habitats (i.e., unique to the treatment samples; Figure 7-34).

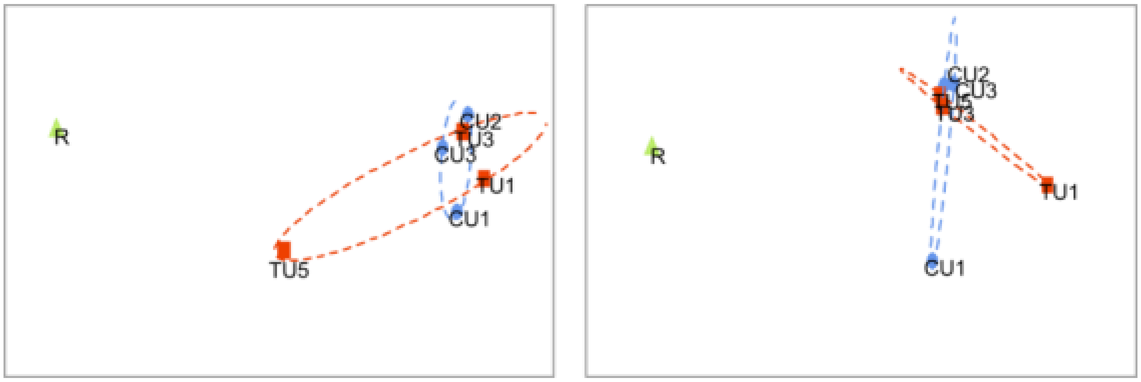


Figure 7-33: NMDS ordination of spider species (left) and ground beetle species (right) with 90% confidence regions for habitat overlaid. Habitat types: treatment (T), control (C), and upland reference (R). Each point represents one polygon sampled in 2018.

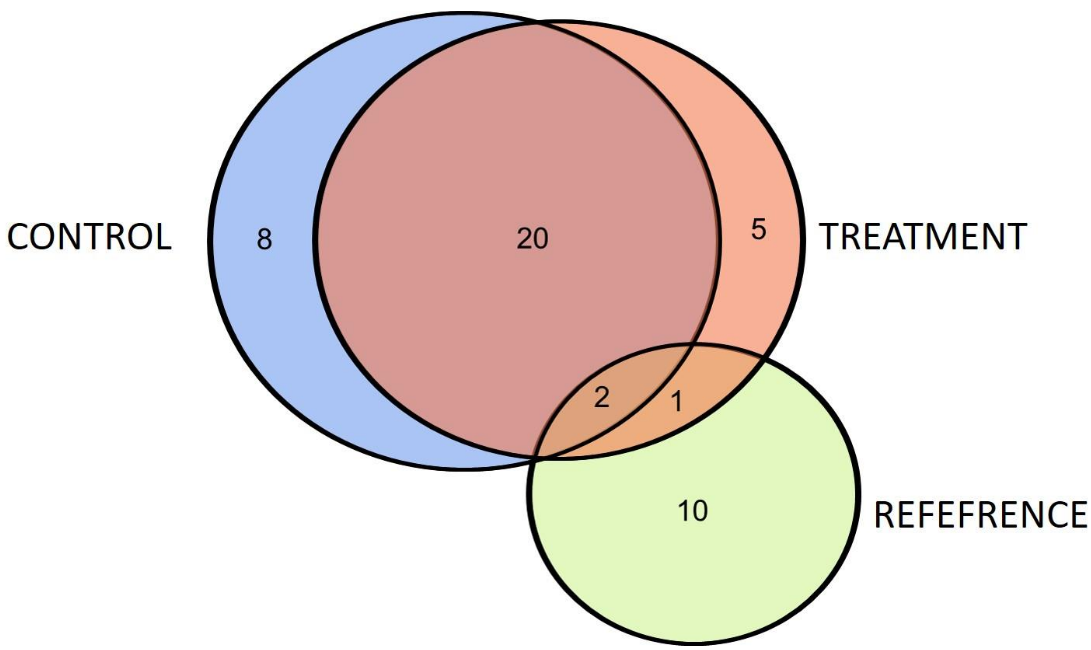


Figure 7-34: Venn diagram showing the number of arthropod species at KM88 unique to control samples (blue area), treatment samples (red area), reference samples (green area), and shared between the habitats (overlapping ellipses). Circle areas are approximately proportional to the number of observed species. Includes both spiders and ground beetles.

VP-N (wood clearing from wetland and terrestrial habitat, and log boom)

Composition differences were greatest between upland reference and drawdown zone communities. Treatment and control plots contained spider and ground beetle species assemblages that were not significantly different (Figure 7-35; i.e., 90% confidence ellipses are overlapping for T and C). Spider assemblages of reference samples were most strongly associated with canopy cover (CC: $R^2 = 0.81$, $p = 0.0008$) and moss cover (MOSS: $R^2 = 0.89$, $p = 0.0007$), and wood debris (WOOD: $R^2 = 0.61$, $p = 0.012$), and rock cover (ROCK: $R^2 = 0.45$, $p = 0.06$) explained spider assemblages in the drawdown zone. Ground beetle assemblages were also strongly associated with canopy cover (CC: $R^2 = 0.74$, $p = 0.002$) and moss cover (MOSS: $R^2 = 0.79$, $p = 0.002$) in the reference samples, and wood cover (WOOD: $R^2 = 0.52$, $p = 0.03$) in the drawdown zone samples.

When the effect of year was controlled for, neither spider nor ground beetle species composition differed between treatment and control plots at VP-N (Araneae $F_{1,7} = 1.13$, $p = 0.138$; Carabidae: $F_{1,7} = 1.57$, $p = 0.125$).

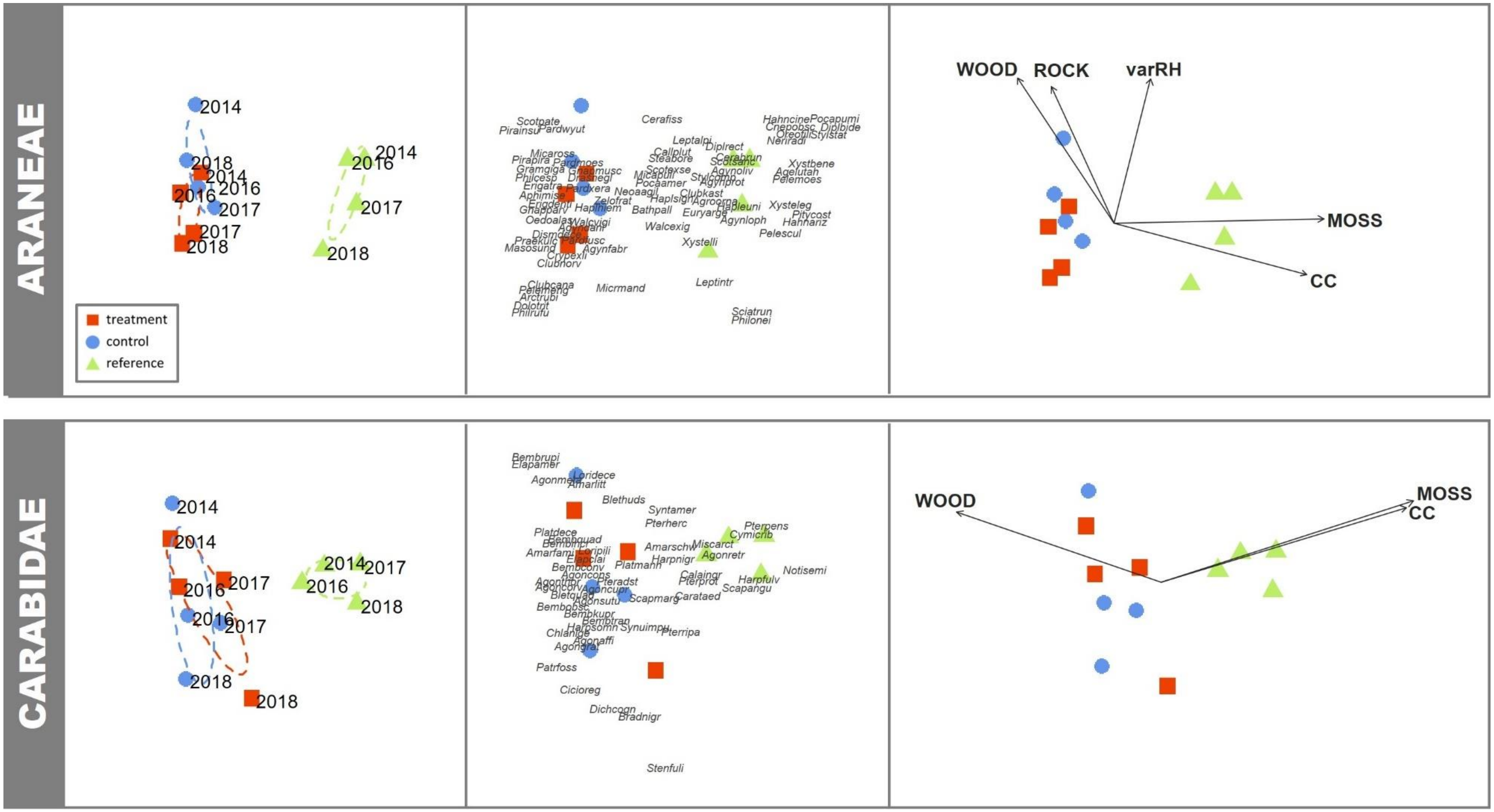


Figure 7-35: NMDS ordination of Valemount Peatland North spider species (top) and ground beetle species (bottom) with 90% confidence regions for Site x Habitat (left), species centroids (centre), and significant (p ≤ 0.1) relationships with environmental variables (right) overlaid. Habitat types: pre-treatment, treatment (T), control (C), and upland reference (R). Overlapping species not shown for clarity. Species codes defined in Table 7-14.

YJ (wood clearing from terrestrial habitat)

Treatment and control plots contained spider and ground beetle species assemblages that were significantly different (Figure 7-36). Spider assemblages were much more heterogeneous between years in the treatment plot compared to the control (relative area of ellipses) but were similar in variability observed for the upland reference spiders. Beetle communities of the reference samples were much less variable than the between years and showed more similar variation between treatment and control samples.

Both spider and beetle assemblages of reference samples were associated with canopy cover (CC: $R^2 = 0.85$, $p = 0.001$, $R^2 = 0.78$, $p = 0.003$, for spiders and beetles, respectively), dead organic matter (DOM: $R^2 = 0.79$, $p = 0.0005$, $R^2 = 0.68$, $p = 0.005$, respectively), and moss cover (MOSS: $R^2 = 0.76$, $p = 0.002$, $R^2 = 0.61$, $p = 0.008$, respectively). Spiders of the drawdown zone were associated with greater cover of mineral soil (MINERAL: $R^2 = 0.62$, $p = 0.08$) and higher variation in Temperature (varT: $R^2 = 0.50$, $p = 0.03$). Carabid species of control samples were associated with greater wood cover (WOOD: $R^2 = 0.50$, $p = 0.02$) and greater mean humidity (meanRH: $R^2 = 0.57$, $p = 0.02$), while those of treatment samples were related to higher rock cover (ROCK: $R^2 = 0.65$, $p = 0.01$), higher variation in relative humidity and temperature (varRH: $R^2 = 0.46$, $p = 0.05$; varT: $R^2 = 0.40$, $p = 0.08$), and greater cover of mineral soils (MINERAL: $R^2 = 0.74$, $p = 0.003$).

Spider and ground beetle species composition differed between treatment and control plots at YJ, when the effect of year was controlled for (Araneae $F_{1,7} = 3.24$, $p = 0.06$; Carabidae: $F_{1,7} = 3.27$, $p = 0.06$).

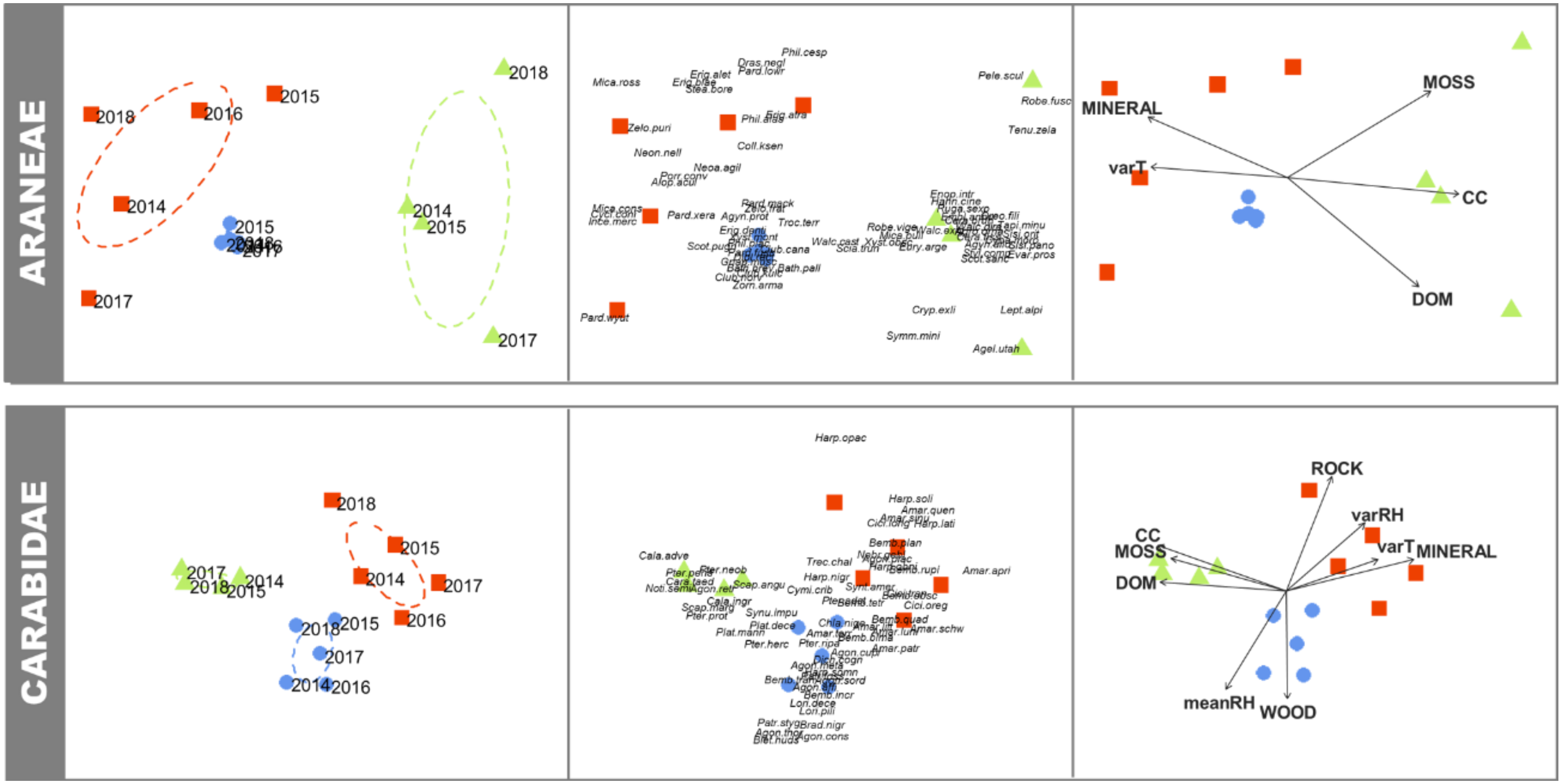


Figure 7-36: NMDS ordination of Yellowjacket Creek spider species (top) and ground beetle species (bottom) with 90% confidence regions for Site x Habitat (left), species centroids (centre), and significant ($p \leq 0.1$) relationships with environmental variables (right) overlaid. Habitat types: pre-treatment, treatment (T), control (C), and upland reference (R). Overlapping species not shown for clarity. Species codes defined in Table 7-14.

<u>P12 (wood clearing from wetland and terrestrial habitat)</u>

Based on only one year of sampling, we collected 69 species of spider and ground beetle at Pond 12. Forty-one of these species were unique to one habitat type (Figure 7-37). Reference and treatment samples contained the same total number of species. Treatment samples contained a total of 39 species, 25 of which were also contained in control and/or reference samples. Nine species were present in all habitat types (reference, control, and treatment).

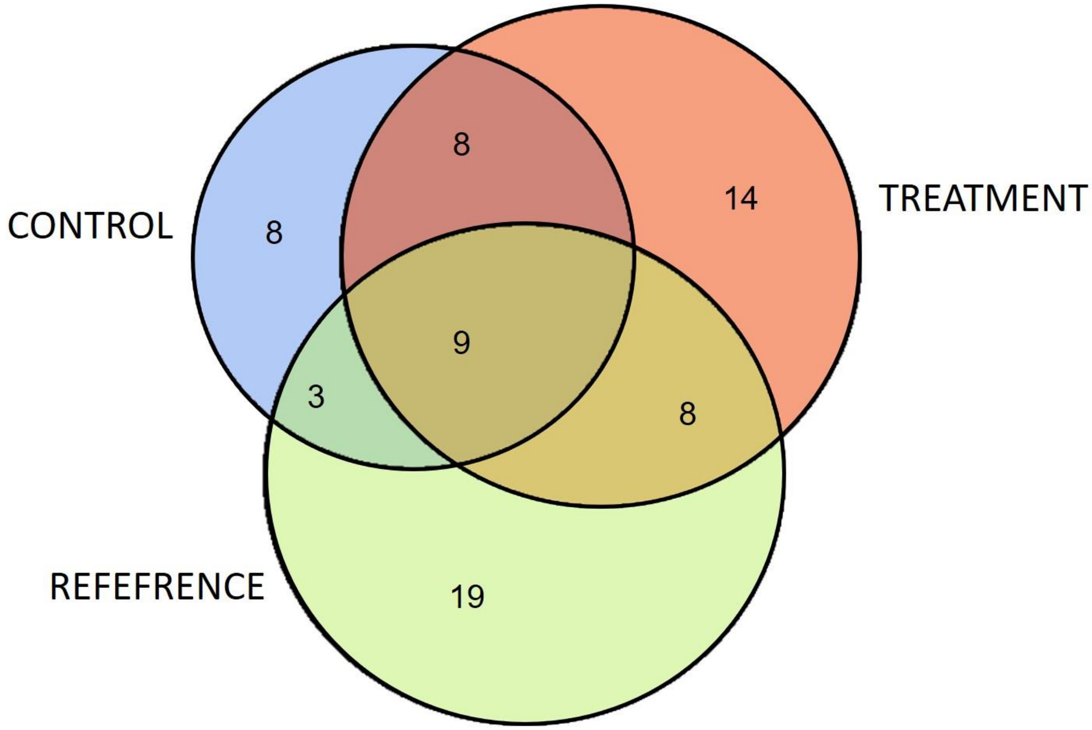


Figure 7-37: Venn diagram showing the number of arthropod species at Pond 12 unique to control samples (blue area), treatment samples (red area), reference samples (green area), and shared between the habitats (overlapping ellipses). Circle areas are approximately proportional to the number of observed species. Includes both spiders and ground beetles.

Discussion

Overall data from post-treatment monitoring among all sites showed that a substantial portion of arthropod species are found in both treatment and control plots (147 species shared of the 192 species from treatments and 193 species from controls). This result suggests that over 76% of ground dwelling spider and beetle species are utilizing both control and treatment areas in the drawdown zone.

Only one study area showed evidence of treatment and control composition differences, when the effect of year was controlled for (Yellowjacket Creek). At this site, both spider and ground beetle species composition differed between treatment and control plots (Figure 7-36). We are unable to determine if compositional differences observed at YJ are in any part due to a treatment effect, since pre-treatment sampling was not conducted at this site. There are notable underlying ecological differences between the paired treatment and control, such as soil texture, composition, and moisture (Figure 7-38), which would figure prominently in structuring the arthropod assemblages from these areas. The assemblages of YJ treatment were strongly related to the higher percent cover of mineral soil and greater range in variation of temperature (Figure 7-36, right). Beetle assemblages at YJ control appear strongly related to the higher relative humidity measured at this plot (Figure 7-36, lower-right). Thus, it is likely that compositional differences at YJ are a result of pre-existing differences in conditions within the site, rather than an effect of treatment.

We also observed greater heterogeneity in treatment spider assemblages at this site compared to the control assemblages. This could be the result of repeated treatment applications, as it is expected for arthropod species to respond to ground disturbance. Spiders are particularly sensitive to differences in site temperature and moisture (Dondale and Binns 1977; Rushton et al. 1987; Frampton et al. 2000), which would be affected by the soil and vegetation disturbance observed during wood removal events (see Wood et al. 2018, figure 6-14).

Figure 7-38: Photos of Yellowjacket Creek treatment (left) and control (right) in 2014, showing vegetation and substrate composition at the centre of each plot.

When the effect of year was controlled for, neither spider nor ground beetle species composition differed between treatment and control plots at BAC-N, BAC-S, or VP-N. Therefore, composition did not seem to be influenced by treatment application, at least for the short duration of post-treatment monitoring. These results were consistent with findings reported in 2016 (Wood et al. 2017), in which VP-N treatment and control assemblages grouped together in ordination space (similar composition), whereas YJ assemblages were distinct. It is possible that the time since treatment application is not sufficient to detect changes in composition, as we have only three (BAC-N, BAC-S) to four years (VP-N) of post-treatment data available. Species of beetles and spiders do respond to changes in vegetation structure, which may be more evident over longer timescales. Unfortunately, the monitoring of physical works treatments under CLBMON-11A became a focus only after 7 years of monitoring failed revegetation treatments. This left little time to detect responses to treatments within the original 11-year program timeline. Follow-up sampling at VP-N, BAC-N, and BAC-S would be warranted if vegetation recovery is determined for these sites.

At KM88, few species were unique to treatment samples (Figure 7-34) and species assemblages from treatment and control samples grouped together on the ordination plot (Figure 7-33), suggesting that the arthropod communities are not distinct between treatment and control at this site. Beetle assemblages were dominated by *Agonum cupreum* and *Amara patruelis*, which are often found in open habitats with sparse vegetation (Larochelle and Larivière 2003). This comparison is limited to only one year of data, which would not provide a thorough inventory of ground beetle and spider species. However, the number of traps deployed in KM88 in 2018 was large compared to other sites, as there were three treatment and three control polygons, thus replication was likely sufficient to reveal major patterns within this site.

While treatment samples at Pond 12 contained more species than control, and just as many as reference (Figure 7-37), data are too limited (one year of sampling, 18 samples in each habitat) and this pattern may not hold given further sampling. While *Agonum affine* was dominant in treatment samples and absent from control, in this initial post-treatment year, there is no reason based on the species natural history for it to not use the control area. This species is generally associated with eutrophic marshes and ponds, swamps, mossy bogs, marshy borders of brooks, and drainage channels, with soil covered with dense vegetation (Larochelle and Larivière 2003).

While interannual variation is implied through the assemblage heterogeneity of reference communities, we are also aware that the reservoir operating regime over the study period may impart temporal variation in drawdown zone communities. Our study plots are located in the uppermost elevation bands of the reservoir drawdown zone, which has a history of seasonal inundation that depends on the annual operating regime of Kinbasket Reservoir. As shown in Figure 2-1, the reservoir has not inundated these sites (VP-N, YJ, BAC-N, BAC-S), since 2013. Further, in the two years preceding our wildlife monitoring at these sites (2012 and 2013), the reservoir was surcharged above the normal operating maximum. We do not know the impact the duration and extent of flooding may have imposed on site vegetation and arthropod communities, and likewise, we do not fully understand how the vegetation and arthropods have responded to the lack of inundation from 2014-2018. We expect that the recovery of these groups in both control and treatment areas over the course of this study could confound or undermine our ability to detect treatment differences.

It is too early to make any conclusions about the treatment application at Pond 12. The treatment area at this site was blanketed by a dense cover of wood chips from the removal work in 2018. As this area becomes established with vegetation and changes over time, we expect the ground-dwelling arthropod communities to likewise respond. Further years of monitoring are required.

Appendix 5: Analysis of Bird Richness and Diversity

Introduction

We examined bird richness and diversity to understand how bird communities responded to the different treatment applications. We focused our analyses on songbird and shorebird species surveyed by line transects (see section 3.0). This section assists in answering Management Questions 1, 3, and 4 (see section 5.0).

Methods

Birds were sampled in the field using standardized line transect surveys. See sampling methodology for Breeding Birds for more details. Bird richness and diversity is assessed by habitat type (i.e., Treatment, Control), site, and year. Each study site is presented as a case study, since treatment types were not replicated.

Data Set

Data set 3 (see DATA SETS: Breeding Birds) was used to assess richness (number of species) and diversity (Shannon-Wiener index) of bird communities in control and treatment transects within the drawdown zone. All birds were recorded during line transect surveys. However, to limit bird data to the specific habitat of interest, we constrained data to 50 m on either side of the line transect. Further, observations of birds other than songbirds, hummingbirds, and shorebirds were excluded as their detection could not be standardized between treatment and control (e.g., a raptor perched briefly before flying into forest) and/or were considered spurious (i.e., a result of random encounter rather than habitat preference). Flyovers of all species except for swallows and hummingbirds were also excluded from analyses. Thus, the analysis dataset was the data-constrained one.

Only sites which were sampled in 2018 (e.g., Pond 12), or were sampled in consecutive years up to and including 2018 were included in dataset 3. The sampling unit was the line transect. Each transect had two or three visits in a year, and each site had one control and one treatment transect, except at KM88, which had three of each transect type (pooling treatment polygons). In total there were 95 datapoints (rows of data), collectively including 41 bird species.

Each site was investigated separately and could be considered subsets of the larger dataset. Details on these subsets are as follows:

- BAC-N: 4 years of bird sampling. 1 treatment line transect, 1 control line transect, 1 year pre-treatment (2015), 3 years post-treatment (2016-2018).
    - 2015: Treatment (n = 2), Control (n = 2)
    - 2016: Treatment (n = 3), Control (n = 3)
    - 2017: Treatment (n = 3), Control (n = 3)
    - 2018: Treatment (n = 2), Control (n = 2)
        - Total: Treatment (n = 10), Control (n = 10)

- BAC-S: 4 years of bird sampling. 1 treatment line transect, 1 control line transect, 1 year pre-treatment (2015), 3 years post-treatment (2016-2018).
    - 2015: Treatment (n = 2), Control (n = 2)
    - 2016: Treatment (n = 3), Control (n = 3)
    - 2017: Treatment (n = 3), Control (n = 3)
    - 2018: Treatment (n = 2), Control (n = 2)
        - Total: Treatment (n = 10), Control (n = 10)
- KM88: 1 year of bird sampling. 3 treatment line transect, 3 control line transect, 1 year post-treatment (2018)
    - 2018 Treatment (n = 6), Control (n = 6)
        - Total: Treatment (n = 6), Control (n = 6)
- VP-N: 4 years of bird sampling. 1 treatment line transect, 1 control line transect, 4 years post-treatment (2015-2018).
    - 2015: Treatment (n = 1), Control (n = 2)
    - 2016: Treatment (n = 3), Control (n = 3)
    - 2017: Treatment (n = 3), Control (n = 3)
    - 2018: Treatment (n = 2), Control (n = 2)
        - Total: Treatment (n = 9), Control (n = 10)
- YJ: 4 years of bird sampling. 1 treatment line transect, 1 control line transect, 2 years initial post-treatment (2015-2016), 2 years re-application post-treatment (2017-2018).
    - 2015: Treatment (n = 2), Control (n = 2)
    - 2016: Treatment (n = 3), Control (n = 3)
    - 2017: Treatment (n = 3), Control (n = 3)
    - 2018: Treatment (n = 2), Control (n = 2)
        - Total: Treatment (n = 10), Control (n = 10)
- POND 12: 1 year of bird sampling. 1 treatment line transect, 1 control line transect, 1 year post-treatment (2018)
    - 2018 Treatment (n = 2), Control (n = 2)
        - Total: Treatment (n = 2), Control (n = 2)

Analysis

We evaluated how treatment affected bird species richness and diversity. Richness and diversity were calculated using the R package 'vegan' (Oksanen et al. 2018). We considered each site separately as conditions vary among sites preventing an overarching analysis. We compared between 'treatment' (see Table 2-2 for more details) and control areas at each site.

Relative richness and diversity at the transect level between treatment and control transects were examined through boxplots. To aid the reader in interpreting boxplot graphs, the boxes represent between 25 percent and 75 percent of the ranked 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 are drawn from the top of the box to the largest observation within 1.5 interquartile range of the top, and from the bottom of the box to the smallest observation within 1.5 interquartile range of the bottom of the box. Boxplots display the differences between groups of data without making any assumptions about their underlying statistical distributions and show their dispersion and skewness. For this reason, they are ideal in displaying ecological data. All boxplots were created using R (R Core Team 2018).

Qualitative assessments on the number of species and observations are also presented using raw data summarizations. When sampling was equal between periods, or when no pre-treatment sampling occurred, the raw values are presented. When sampling was unequal (e.g., only one year of pre-treatment data) the yearly totals (for number of species and number of observations) were calculated, and then averaged over the number of years in that sample (i.e., at BAC-N and BAC-S).

Results

<u>BAC-N (wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom)</u>

Woody debris removal, mounding, and log-boom installation occurred during 2015, after bird surveys had completed for the year. Thus, 2015 data are the pre-treatment phase, while 2016 through 2018 is post-treatment monitoring. Comparisons between pre and post-treatment periods are difficult, owing to the limited replication pre-treatment. Post-treatment, control and treatment transects have similar richness and diversity values (Figure 7-39). The average annual number of species and observations post-treatment is very similar between treatment and control transects (Table 7-3).

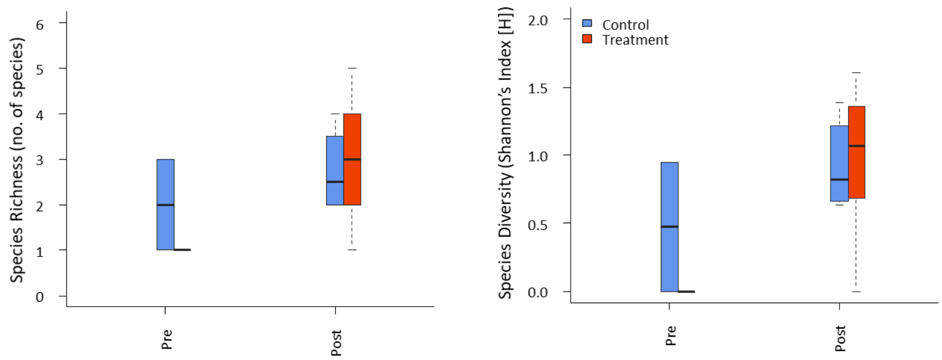


Figure 7-39: Boxplots showing bird species richness (left) and Shannon's diversity index (right) for control (blue boxes) and treatment (red boxes) transects at Bush Arm Causeway NW over three years of surveying. Initial woody debris removal, mounding, and log-boom installation occurred in 2015 after bird surveys had finished (black vertical line).

Table 7-3: Number of bird species and observations in control and treatment transects during pre-treatment application (2015 only) and post-treatment application (2016-2018) at BAC-N. Post-treatment values are derived by determining the number of bird species and observations during each year, and taking the average of those values.

Transect Type	Pre-treatment Sampling (2015 only)		Post-treatment Sampling (2016-2018) (Values are averaged yearly totals)	
	No. of Species	No. of Observations	No. of Species	No. of Observations
Control	4	6	5	9
Treatment	2	2	5.3	9.3

BAC-S (wood clearing from terrestrial habitat, mounds, and revegetation)

Woody debris removal and mounding occurred during 2015, after bird surveys had completed for the year. Thus, 2015 data are the pre-treatment phase, while 2016 through 2018 is post-treatment monitoring. Comparisons between pre and post-treatment periods are difficult, owing to the limited replication pre-treatment. Post-treatment, control and treatment transects have similar richness and diversity values (Figure 7-40). During the single pre-treatment year, there were very few birds encountered in the control transect, for unknown reasons. Following the physical works, richness, and diversity of both treatment and control transects increased. Post-treatment, control and treatment transects have similar richness and diversity values (Figure 7-40). The average annual number of species and observations post-treatment are similar between treatment and control transects (Table 7-4).

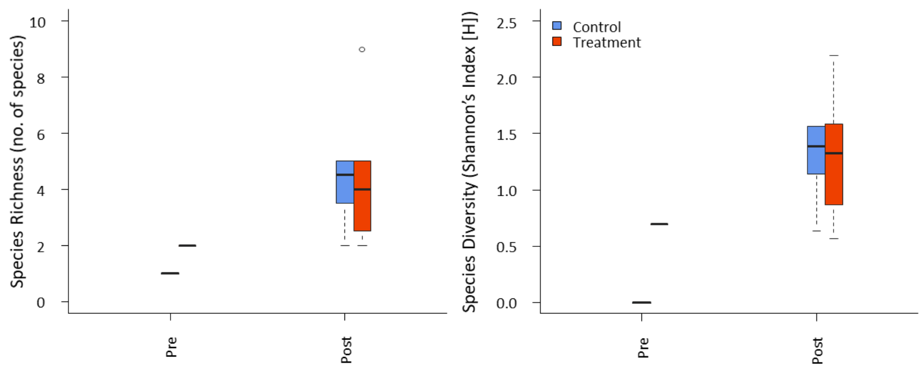


Figure 7-40: Boxplots showing bird species richness (left) and Shannon's diversity index (right) for control (blue boxes) and treatment (red boxes) transects at Bush Arm Causeway SW over four years of surveying. Initial woody debris removal, mounding, and log-boom installation occurred in 2015 after bird surveys had finished (black vertical line).

Table 7-4: Number of bird species and observations in control and treatment transects during pre-treatment application (2015 only) and post-treatment application (2016-2018) at BAC-S. Post-treatment values are derived by determining the number of bird species and observations during each year, and taking the average of those values.

Transect Type	Pre-treatment Sampling (2015 only)		Post-treatment Sampling (2016-2018) (Values are averaged yearly totals)	
	No. of Species	No. of Observations	No. of Species	No. of Observations
Control	2	2	7.3	13.0
Treatment	3	4	8.7	13.3

<u>KM88 (sedge plug revegetation)</u>

Sedge-plug planting occurred during 2013. Thus, 2018 data refers to a single, post-treatment year. As such, analyses and inferences are limited and preliminary. During this single post-treatment year, there were very few birds encountered in either control or treatment transects, for unknown reasons. Treatment transects appear to have higher richness and diversity (Figure 7-41). This was mostly due to twice as many observations of one species in the treatment transect, and one observation of a single species only detected in the treatment transect (Table 7-5).

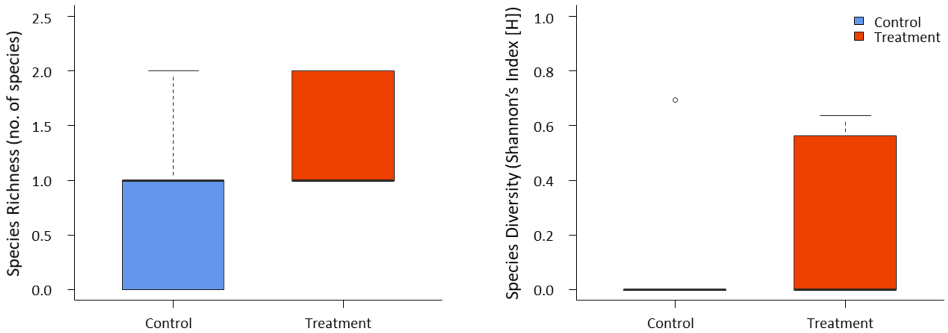


Figure 7-41: Boxplots showing bird species richness (left) and Shannon's diversity index (right) for control (blue boxes) and treatment (red boxes) transects at KM88 in 2018. All sampling was post-revegetation.

Table 7-5: Number of bird species and observations in control and treatment transects during 2018 (post-treatment) sampling at Pond 12.

Transect Type	Post-treatment Sampling (2018)	
	No. of Species	No. of Observations
Control	2	6
Treatment	3	12

VP-N (wood clearing from wetland and terrestrial habitat, and log boom)

Wood debris removal and log-boom installation occurred during 2014. Thus, all sampling years represent post-treatment monitoring. While variance is wide, both richness and diversity appear similar between control and treatment transects, though average higher in the control. Even though richness and diversity are higher for controls at the transect level, when all data are pooled more species were actually documented from treatment transects (though observations still remained higher in the control) (Table 7-6).

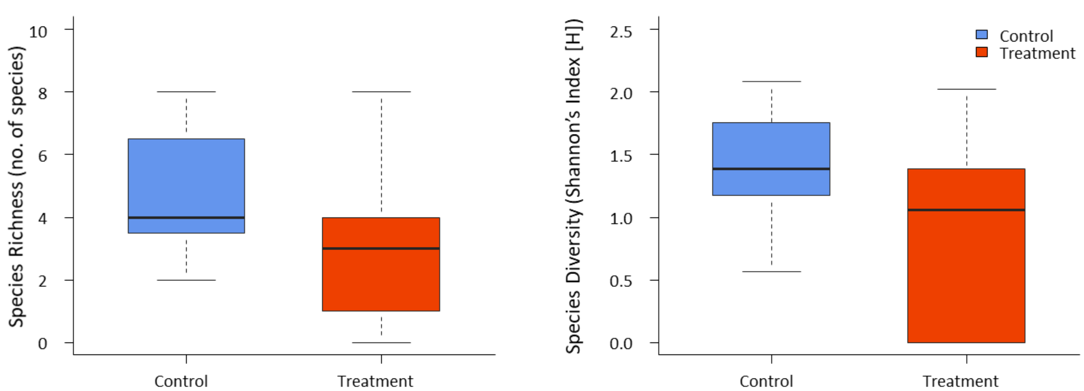


Figure 7-42: Boxplots showing bird species richness (left) and Shannon's diversity index (right) for control (blue boxes) and treatment (red boxes) transects at Valemount Peatland North (VP-N) over four years of surveying. Initial woody debris removal and log-boom installation occurred in 2014.

Table 7-6: Number of bird species and observations in control and treatment transects during post-treatment surveys (2015 to 2018) at VP-N.

Transect Type	Post-treatment Sampling (2015-2018)	
	No. of Species	No. of Observations
Control	14	61
Treatment	20	37

YJ (wood clearing from terrestrial habitat)

The assessment of treatment and control transects at Yellowjacket Creek is confounded by the differences in soil type/moisture regime. They are further confounded by a re-application of woody debris removal in the treatment plot immediately preceding bird surveys in 2017 (the original application occurred in 2014). During the first two years of post-treatment monitoring (2015 and 2016), the control had significantly greater richness and diversity than the treatment (Figure 7-43). The two years following the re-application of woody debris removal had a similar trend (control having greater richness and diversity than treatment). Richness and diversity were lower in the control transect following the re-application of woody debris removal than they were following the initial application. Following the initial treatment twice as many species and nearly triple the number of observations were recorded from the control transect overall (Table 7-7). Following the re-application of woody debris removal, the overall number of species and observations were similar between the transects (Table 7-7).

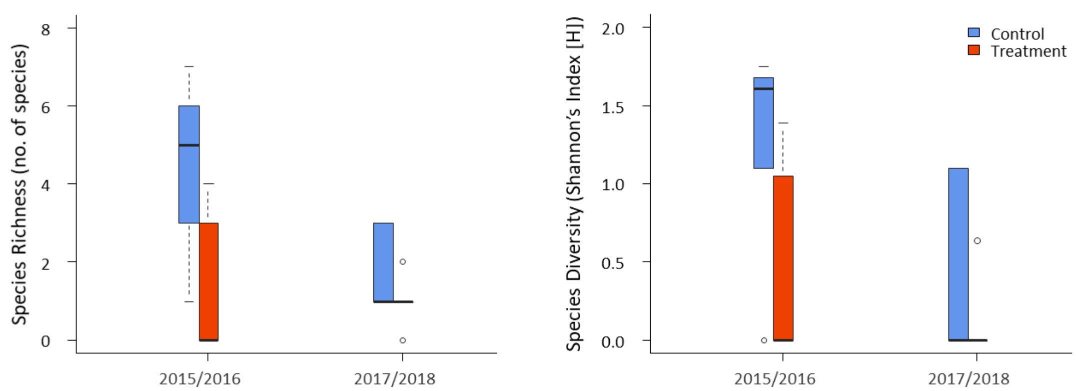


Figure 7-43: Boxplots showing bird species richness (left) and Shannon's diversity index (right) for control (blue boxes) and treatment (red boxes) transects at Yellowjacket Creek over four years of surveying. Initial woody debris removal and log-boom installation occurred in 2014, with a re-application of woody debris removal in 2017 prior to surveys.

Table 7-7: Number of bird species and observations in control and treatment transects following treatment application (2015 and 2016) and following the re-application of treatment (2017 and 2018) at YJ.

Transect Type	Initial Post-treatment Sampling (2015 and 2016)		Secondary Post-treatment Sampling (2017 and 2018)	
	No. of Species	No. of Observations	No. of Species	No. of Observations
Control	11	25	6	9
Treatment	6	9	5	6

P12 (wood clearing from wetland and terrestrial habitat)

Woody debris removal occurred at Pond 12 in 2018. One control transect and one treatment transect were each surveyed twice in 2018. Not enough data are available to generate statistics or boxplots at this site. Comparing only number of species and number of observations recorded in each transect type, as a proxy for richness and diversity, reveals that both transects are overall similar (Table 7-8).

Table 7-8: Number of bird species and observations in control and treatment transects during 2018 (post-treatment) sampling at Pond 12.

Transect Type	Post-treatment Sampling (2018)	
	No. of Species	No. of Observations
Control	10	18
Treatment	12	20

Discussion

Overall, there was no consistent pattern in how treatment type affected bird diversity and richness. In most cases there was no observable effect of treatment. Site-specific differences in ground substrate (moisture and amount of organic matter), vegetation types present (shrubs, herbs), proximity to forest edge, time since treatment, and treatment type (log boom or mounding), confounds the assessment of differences between control and treatment plots when sites are considered in aggregate. For this reason, sites were assessed on an individual basis. Inter-annual variability in measured species richness and diversity due to factors independent of treatments (e.g., weather) may mask trends related to revegetation prescription effectiveness, but more importantly the lack of replication (due to small areas of revegetation prescriptions) and low bird density in the drawdown zone limits our ability to make inferences.

In both sites where mounds were installed (BAC-N and BAC-S), we saw no effects of treatment on bird species richness or diversity. While the post-physical works period had a higher number of species and observations in the treatment transect, the same trend was also observed for control suggesting that this result was not due to a treatment effect. Incorporating windrows or mounds may not be beneficial for bird species richness and diversity, though this habitat feature was not predicted to be utilized by most drawdown bird species.

Sites with woody debris removed (VP-N, YJ, P12) showed similarly non-significant results. At VP-N controls had slightly higher median richness and diversity values. Comparing the total number of species, the treatment area had six more species detected than did control, though control had more individual observations. At YJ, woody debris removal initially done in 2014 was repeated immediately prior to surveys in 2017. Surveys in 2017 and 2018 yielded lower richness and diversity than the 2015/2016 period, but this was noted in both control and treatment transects suggesting the decline was not linked to the woody debris removal itself. The control transect appeared richer and more diverse in the 2015/2016 period. This may have been due to differences in soil substrate and moisture. The non-treated area had moist to wet organic soils located underneath the woody debris, owing to natural seepage in this location, whereas the treated area substrate was drier, rockier (gravel-cobble), lower in organic content, and relatively unproductive. Pond 12 was only surveyed in 2018. Although data were limited to this single season, treatment and controls appeared overall similar in bird richness and diversity. As was the case with the mound treatments, there is no strong evidence that woody debris removal had a positive (or any) effect on bird communities at these sites.

Revegetation (at KM88) had a possible effect on bird species richness and diversity. However, results are limited by a brief sampling period (one year), and a depauperate bird community. Total bird species were similar between the two transect types, but treatment had twice as many observations.

The lack of consistent differences in treatment and control areas may be due to a spatial scale mismatch between birds as focal taxa and the extent of revegetation and other habitat prescriptions. The treatment areas are, in many cases, smaller than the home range size of species that may utilize them [e.g., Savannah Sparrow can have breeding territories >1 ha (Wheelwright and Rising 2008)]. The small sizes of the revegetation and physical works prescriptions, lack of replication and stratified treatments, short time scale of pre- and post-physical works monitoring, and inconsistencies of the CLBWORKS-1, CLBWORKS-2, and CLBWORKS-16 methodology make it difficult to achieve more than speculation regarding the program's effect on bird richness and diversity of those habitats. Given these limitations, we suggest that future studies consider greater investigation into focal taxa which may respond to smaller scale changes, such as arthropods, and that birds (e.g., nesting locations) be considered supplementary.

An important consideration is that treatment effectiveness should be considered in the context of reservoir levels. While there have been years in which reservoir levels have reached the upper elevation bands of our study area (such as in 2012), in more recent years the reservoir has operated under its maximum (full pool). This includes the duration of time that the CLBMON-11A line transect sampling has taken place (2015 to 2018). This may prevent us from fully understanding how certain treatment techniques, such as log boom installation, could operate to preserve habitat integrity and species richness or diversity in an area. It also makes the comparison between control and treatment sites more difficult, especially those sites where log booms have been installed.

Appendix 6: Analysis of Bird Composition and Abundance

Introduction

We examined bird abundance and composition to understand how bird communities responded to the different treatment applications. We focused our analyses on songbird and shorebird species surveyed by line transects (see section 3.0). This section assists in answering Management Questions 1, 3, and 4 (see section 5.0).

Methods

Birds were sampled in the field using standardized line transect surveys. See Breeding Birds methods for more details. Bird abundance and composition is assessed by habitat type (i.e., Treatment, Control), site, and year. Each study site is presented as a case study, since treatment types were not replicated.

Data Set

Data set 3 (see DATA SETS: Breeding Birds) was used to assess bird abundance and composition in control and treatment transects within the drawdown zone. This analysis utilizes Data Set 3, with no further derivation. Please refer to Appendix 5 for a description of the data set and site replication.

Analysis

We evaluated how treatment affected bird composition and abundance. We considered each site separately as conditions vary among sites preventing an overarching analysis. We compared between 'treatment' (see Table 2-2 for more details) and control areas at each site.

Similarity of species assemblages were assessed with Venn diagrams using the package 'VennDiagram' in R (Chen 2015). These graphically display the number of unique species in treatment and control plots and the number of species that were shared between plots. The area of each ellipse is proportional to the total number of species observed for that treatment type, allowing for both comparisons of bird composition (proportion of shared vs unique species).

Qualitative assessments on the composition and abundance of species are also presented using raw data summarizations. When sampling was equal between periods, or when no pre-treatment sampling occurred, the raw values are presented. When sampling was unequal due to only having one year of pre-treatment data, totals were based on the three years of post-treatment data (i.e., at BAC-N and BAC-S).

Results

<u>BAC-N (wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom)</u>

There were 64 observations of 17 species over all years of surveys. Most of the species have been detected from the treatment transect, which also had the highest number of unique species (Figure 7-44). Comparing bird abundance for the three post-treatment years (2016-2018), Savannah Sparrow was the most commonly detected species (7 and 10 observations in treatment and control transects respectively) (Figure 7-45). Spotted Sandpiper had five detections from the control transect and one from the treatment. Killdeer were detected twice in each transect. The majority of species were detected from only a single year, and no species was detected in all four survey years (Table 7-9). The three most abundant species (Savannah Sparrow, Lincoln's Sparrow, and Spotted Sandpipers) were the only ones to be detected in all post-treatment years.

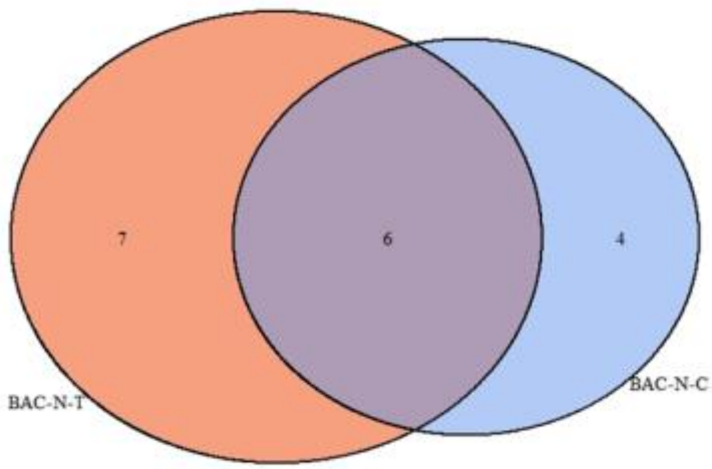


Figure 7-44: Venn diagram showing the number of bird species in the constrained dataset observed at Bush Arm Causeway NW that were unique to control transects (blue area), treatment transects (red area), and shared between the transects (overlapping mauve area) during post-treatment counts (2016-2018) only. Circles are proportional to the number of observed species.

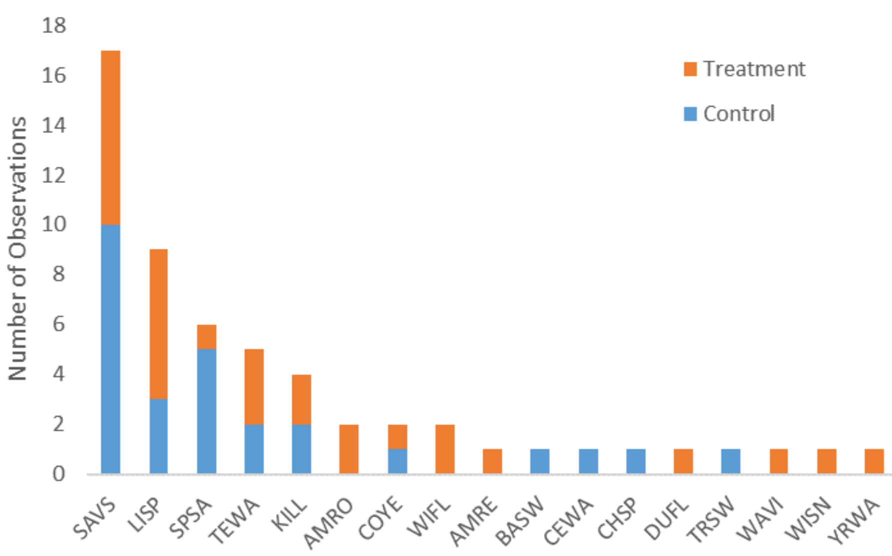


Figure 7-45: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at BAC-N. To assist in treatment vs. control comparisons, data are only presented for the three post-treatment years (2016-2018).

Table 7-9: Number of bird species that were detected at BAC-N in all (4) years or some (1 to 3) years. Species detected in multiple years are more indicative of site conditions or suitability overall.

Bird Species Detected Only In				Total
1 Year	2 Years	3 Years	4 Years	
10	4	3	0	17

BAC-S (wood clearing from terrestrial habitat, mounds, and revegetation)

There were 86 observations of 21 species over all years of surveys. Most of the species have been detected from the treatment transect, which also had the highest number of unique species (Figure 7-46). Comparing bird abundance for the three post-treatment years (2016-2018), Lincoln's Sparrow was the most commonly detected species in these transects, with roughly equal number of observations in each transect (10 and 11 in control and treatment respectively) (Figure 7-47). Savannah Sparrow had the second highest number of observations (10) and was also relatively evenly split between treatment and control (Figure 7-47). Most species were detected from only a single year (Table 7-10). The three species detected in all four surveys years were the among the top four most detected species, while the most common species (Lincoln's Sparrow) was detected in all years post-treatment.

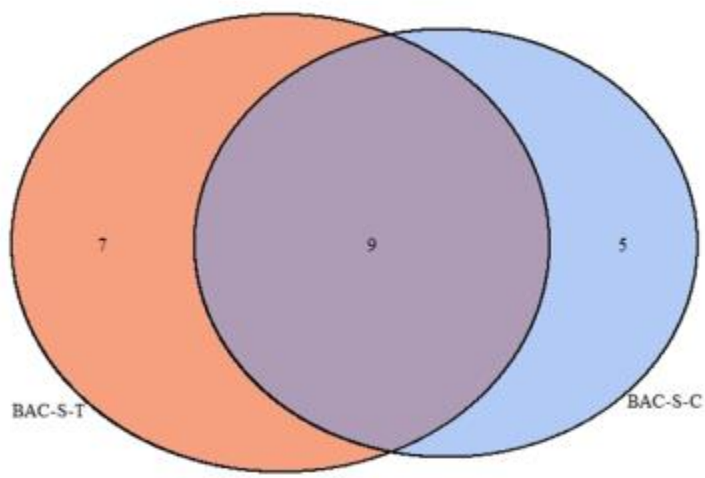


Figure 7-46: Venn diagram showing the number of bird species in the constrained dataset observed at Bush Arm Causeway SW that were unique to control transects (blue area), treatment transects (red area), and shared between the transects (overlapping mauve area) during post-treatment counts (2016-2018) only. Circles are proportional to the number of observed species.

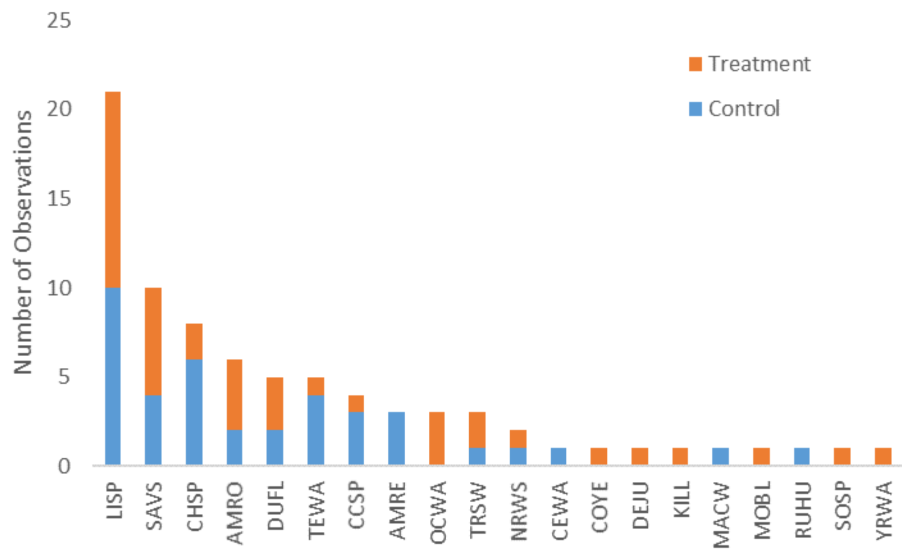


Figure 7-47: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at BAC-S. To assist in treatment vs. control comparisons, data are only presented for the three post-treatment years (2016-2018).

Table 7-10: Number of bird species that were detected at BAC-S in all (4) years or some (1 to 3) years. Species detected in multiple years are more indicative of site conditions or suitability overall.

Bird Species Detected Only In				Total
1 Year	2 Years	3 Years	4 Years	
10	5	2	3	20

KM88 (sedge plug revegetation)

There were 18 observations of 3 species from 2018 surveys. Only Savannah Sparrows were frequently detected, with most observations (10 out of 15) from treatment transects (Figure 7-48). Savannah Sparrows were detected in all three treatment transects.

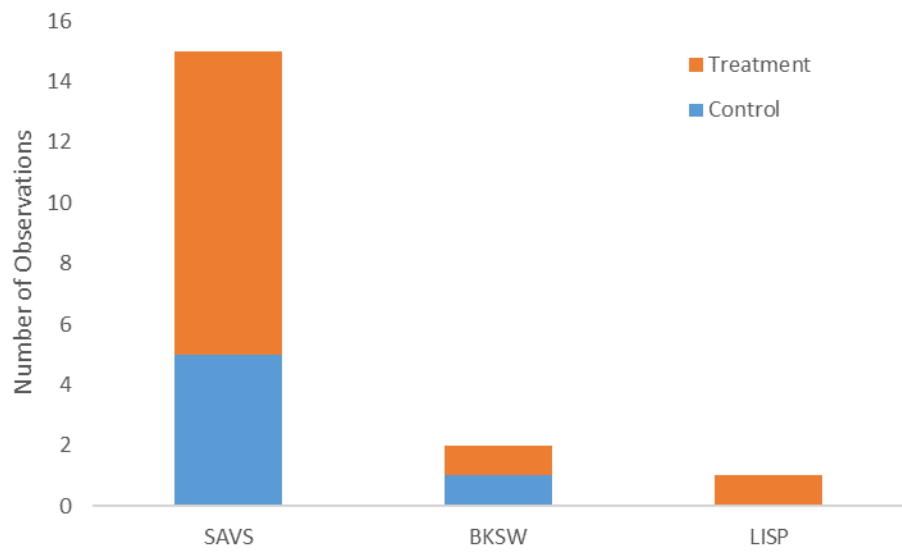


Figure 7-48: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at KM88 in 2018.

VP-N (wood clearing from wetland and terrestrial habitat, and log boom)

There were 98 observations of 24 species over all years of surveys. All but four species have been detected from the treatment transect, which also had the highest number of unique species (Figure 7-49). Almost all species detected at both transects were more abundant in the control (Figure 7-50). Most species were detected from only a single year (Table 7-11). The most common species (Savannah Sparrow) was the only one detected in all four survey years, while the five species detected in three surveys years were also among the most frequently detected species.

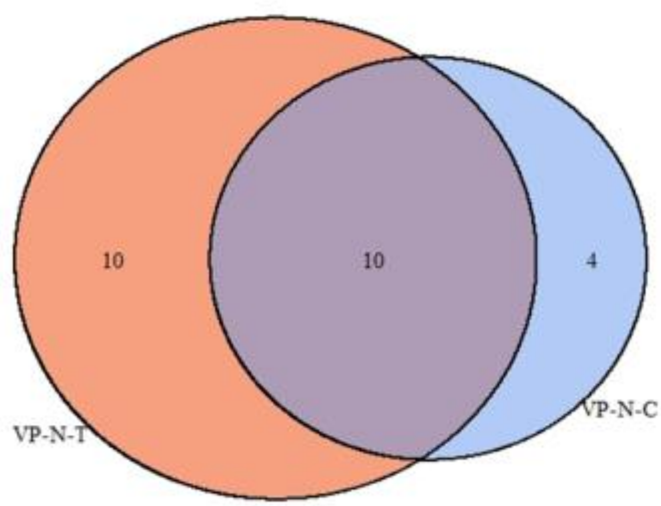


Figure 7-49: Venn diagram showing the number of bird species in the constrained dataset observed at Bush Arm Causeway SW that were unique to control transects (blue area), treatment transects (red area), and shared between the transects (overlapping mauve area) during post-treatment counts (2016-2018) only. Circles are proportional to the number of observed species.

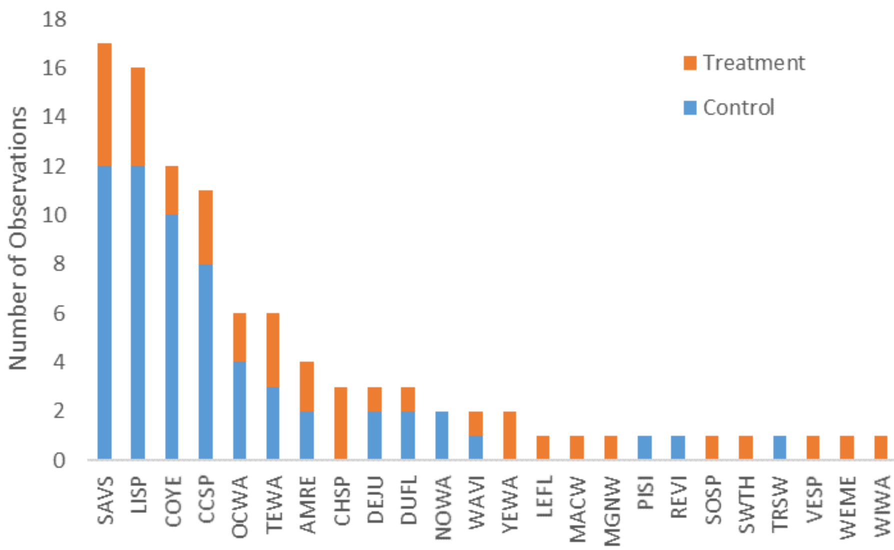


Figure 7-50: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at VP-N in all years (2015-2018).

Table 7-11: Number of bird species that were detected at VP-N in all (4) years or some (1 to 3) years. Species detected in multiple years are more indicative of site conditions or suitability overall.

Bird Species Detected Only In				Total
1 Year	2 Years	3 Years	4 Years	
16	2	5	1	24

YJ (wood clearing from terrestrial habitat)

There were 49 observations of 14 species over all years of surveys. All but two species have been detected from the control transect, which also had the highest number of unique species (Figure 7-51). The control transect accounted for four of the five Common Yellowthroat observations, and nine of the twelve Lincoln's Sparrows. Relatively few species and observations were detected in the treatment transect. Spotted Sandpiper was the only species with >1 observation to be detected solely from the treatment transect. Overall, Lincoln's Sparrow was the most commonly observed species in these transects, with most other species having one to several observations each (Figure 7-52). Half of the species were detected from only a single year (Table 7-12). The most common species (Lincoln's Sparrow) was the only one detected in all four survey years, while the second most common species was detected in three of the four years (Table 7-12).

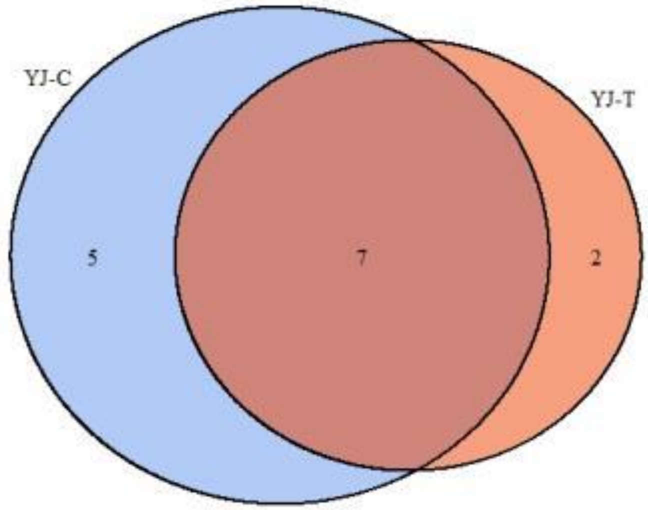


Figure 7-51: Venn diagram showing the number of bird species in the constrained dataset observed at Yellowjacket Creek that were unique to control transects (blue area), treatment transects (red area), and shared between the transects (overlapping mauve area) during years 2015-2018. Circles are proportional to the number of observed species.

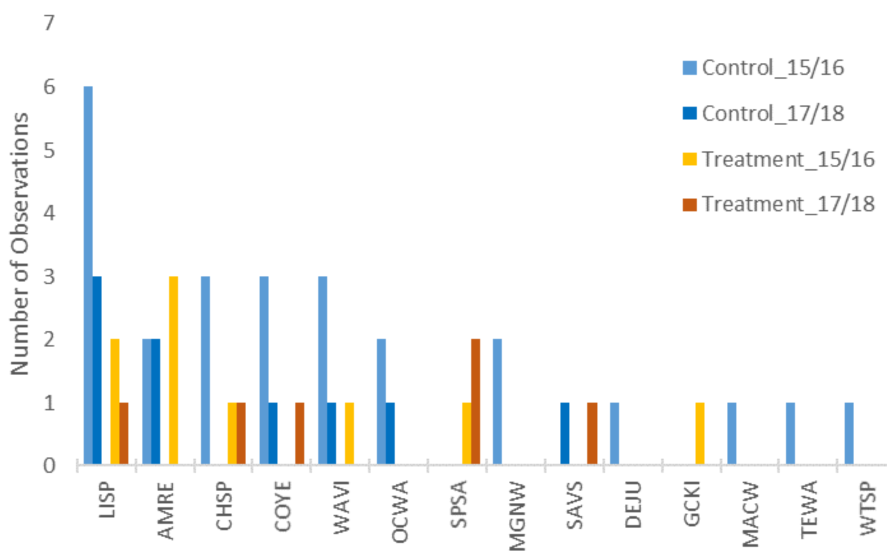


Figure 7-52: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at YJ in all years (2015-2018).

Table 7-12: Number of bird species that were detected at YJ in all (4) years or some (1 to 3) years. Species detected in multiple years are more indicative of site conditions or suitability overall.

Bird Species Detected Only In				Total
1 Year	2 Years	3 Years	4 Years	
7	4	2	1	14

P12 (wood clearing from wetland and terrestrial habitat)

There were 38 observations of 17 species from 2018 surveys. Lincoln's Sparrow was frequently detected, with similar observations (10 out of 15) from treatment and control transects (Figure 7-54). There were more unique bird species detected in the treatment transect than control (Figure 7-53).

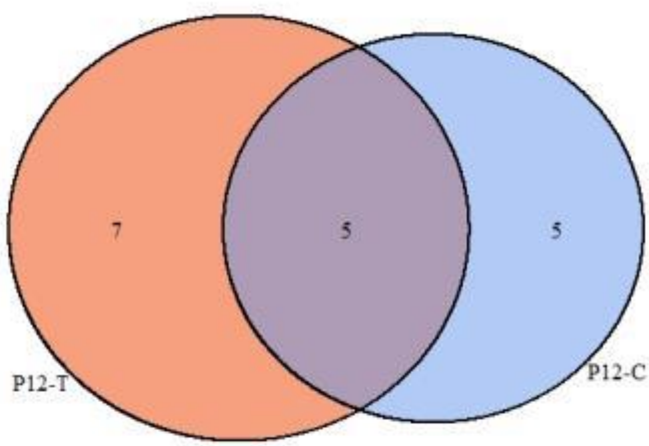


Figure 7-53: Venn diagram showing the number of bird species in the constrained dataset observed at Pond 12 that were unique to control transects (blue area), treatment transects (red area), and shared between the transects (overlapping mauve area) during 2018 surveys. Circles are proportional to the number of observed species.

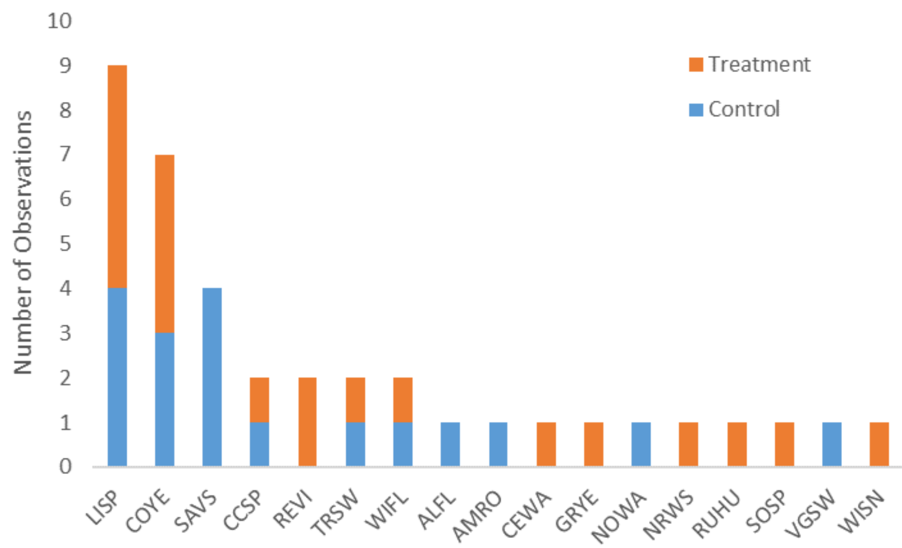


Figure 7-54: Barplot showing the total number of observations for each species in both treatment (orange) and control (blue) transects at Pond 12 in 2018.

Discussion

Overall, there was no consistent pattern in how treatment type affected bird composition and abundance. In most cases there was no observable effect of treatment. Most sites had a greater number of species in treatment than in control areas, though usually only by a few species. This was most evident at Valemount Peatland North. However, the control at VP-N had greater bird abundances. In most cases the differences in species can be explained by the location of the transect, and its proximity to adjacent habitat types (e.g., upland forest). A greater number of species along one transect was typically due to the inclusion of additional single-observation sightings, For this reason the abundance (number of observations) of species is important for understanding bird use of the treatment types. In some cases where the number of species is higher in treatment, the number of observations are significantly less. This discrepancy is greatest at VP-N. At the two Bush Arm Causeway sites, the number of observations were virtually equal, suggesting that there are no treatment effects on the most abundant species. Overall, Savannah Sparrow and Lincoln's Sparrow were the most abundant species, with at least one of those species being the most abundant species at each site. This is due to their preference for open, grassy and/or moist habitats. It is important to remember, that as with richness and diversity comparisons, site-specific differences in ground substrate (moisture and amount of organic matter), vegetation types present (shrubs, herbs), proximity to forest edge, time since treatment, and treatment type (log boom or mounding), confounds the assessment of differences between control and treatment plots. Inter-annual variation in species abundance due to independent factors (e.g., weather) may mask trends related to revegetation prescription effectiveness, but more importantly the lack of replication (due to small areas of revegetation prescriptions) and low bird density in the drawdown zone limits our ability to make strong inferences. Specifics on species results and treatment effects are shown below for each study site:

BAC-N (pond clearing, wood removal, mounds, revegetation, and log boom)

- Many of the unique species in the treatment transect, such as Willow Flycatcher and Warbling Vireo, are reflective of proximity to shrubby and treed habitats at its northern end. Many of the shared species are open-country species that are widespread throughout the Kinbasket Reservoir drawdown zone. These include Savannah Sparrow, Spotted Sandpiper, and Killdeer.
- The open-country species are found in both habitat types. While Savannah Sparrow occurs in both treatment and control relatively equally, Lincoln's Sparrow shows a slight preference for treatment transects. It is not known if this is due to pre-existing conditions or a treatment effect. In contrast, Spotted Sandpiper is notably more abundant in the control.

BAC-S (wood removal, mounds, and revegetation)

- Many of the unique species in the treatment transect, such as Song Sparrow, Mountain Bluebird, Killdeer, and Common Yellowthroat are open-country and/or shrub-preferring species that may benefit from the physical works, though there is no evidence that they have benefited to date.
- The two most abundant species (Lincoln's and Savannah Sparrows) were both found relatively equally between treatment and control. The third most abundant (Chipping Sparrow) was found much more in control, likely owing to the greater shrub cover, proximity to forest, and nesting territories.

KM88 (sedge plug revegetation)

- There were few data to interpret based on one year of surveys. Only 18 observations of 3 species were recorded. One species, an aerial insectivore, was recorded once from each habitat type, and a second, the Lincoln's Sparrow had only one observation. That Savannah Sparrows had three times as many observations in treatment areas and were detected from all three treatment transects, suggests that the habitat within treatment plots was more suitable for that species at this location.

VP-N (pond clearing, wood removal, and log boom)

- Many of the species detected within both the treatment and control transects are woodland species detected in the ecotone between drawdown zone and forest. However, both transects also had species more typical of the drawdown zone, either as unique or shared species. For example, the open-country Vesper Sparrow and Western Meadowlark were only detected from the treatment transect (though both only once). Clay-colored Sparrow and Savannah Sparrow are also open-country species, detected from both the control and treatment transects.
- The much greater number of observations at the control transect may be due to the greater shrub cover along and near that transect, serving as potential nesting habitat, as well as conspicuous perches which may increase detection.

YJ (wood removal)

- Many of the species detected within the control transect are forest species and were detected in the ecotone between drawdown zone and forest, though several (e.g., Lincoln's Sparrow, Common Yellowthroat) would be attracted to the wetter, more vegetated conditions present in that transect.
- Relatively few species were detected in the treatment transect potentially owing to its drier, gravel/cobble substrate with lower vegetation cover.

P12 (pond clearing and wood removal)

- Overall both composition and abundance of species was similar between treatment and controls. In the treatment transect, the slightly higher number of species overall, and observations of both Lincoln's Sparrow and Common Yellowthroat, may indicate that the treatment was successful, but with only a single year of data remains inconclusive.

The lack of consistent differences in treatment and control areas may be due to a spatial scale mismatch between birds as focal taxa and the extent of revegetation and other habitat prescriptions. The treatment areas are, in many cases, smaller than the home range size of species that may utilize them [e.g., Savannah Sparrow can have breeding territories >1 ha (Wheelwright and Rising 2008)].

The small sizes of the revegetation and physical works prescriptions, lack of replication and stratified treatments, short time scale of pre- and post-physical works monitoring, and inconsistencies in the CLBWORKS-1, CLBWORKS-2, and CLBWORKS-16 methodology make it difficult to achieve more than speculation in regard to the program's effect on bird abundance and composition of those habitats.

An important consideration is that treatment effectiveness should be considered in the context of reservoir levels. The study objective to assess whether the revegetation prescriptions in the drawdown zone improve habitat for wildlife can only be properly tested after the prescriptions are exposed to water inundation. While there has been noticeable reservoir inundation activity into the upper reservoir elevation bands where newer prescriptions (e.g., mounds) have been applied over the past decade (such as in 2012), in more recent years the reservoir has operated under its maximum (full pool). This includes the duration of time that the CLBMON-11A line transect sampling has taken place (2015 to 2018). This may prevent us from fully understanding how certain treatment techniques, such as log boom installation, could operate to preserve habitat integrity and species abundance and composition in an area. For example, the treatment areas are not exposed to inundation and the impacts of it (e.g., log deposition), and so it is not possible to test the efficacy of the log-excluding booms that have been installed in the control plots.

Appendix 7: Analysis of Bird Nesting Data

Introduction

We examined nesting bird species and location to understand if treatment applications were successful in increasing nesting opportunities for ground- and/or shrub-nesting birds. We focused our analysis on songbird and shorebird species surveyed by line transects (see section 3.0). This section assists in answering Management Questions 1, 3, and 4 (see section 5.0).

Methods

Nest searches were done at each site, following and on the same date as line transect surveys. Nest searching was a combination of behavioural and systematic searches. Effort was not consistently recorded and varied based on the amount of habitat at each site, so information is presented here qualitatively based on the locations of found nests. Data were supplemented with nest locations provided by CBA Ltd. CLBMON-11A was not designed as a dedicated nesting productivity survey.

Data Set

This dataset was created to assess nesting locations for study sites in which line transect data was analysed. The dataset includes nest data from searches completed between 2015 and 2018 by LGL Limited (n = 23) and supplemented by point locations for nests located by Cooper, Beauchesne and Associates between 2008 and 2017 (n = 77). Nest searching activity by LGL Limited was focused on, but not constrained by, general polygon boundaries denoting treatment and control areas. Nest searching by CBA was completed under CLBMON-36 and was irrespective of the CLBMON-11A study areas (see Wood et al. 2018). We thus constrained CBA nest data to those within 100 m of our study polygons. Nest data are presented qualitatively as the nest search effort by year and area (e.g., elevation bands targeted for search effort) are not known.

Analysis

We evaluated how treatment affected bird nesting qualitatively. Bird nesting data collected under CLBMON-11A by LGL Limited was combined with nest location data provided by CBA Ltd. Nests were mapped in QGIS v. 3.0.2. We considered each site separately as conditions vary among sites preventing an overarching analysis. We compared between 'treatment' (see Table 2-2 for more details) and control areas at each site.

Results

<u>BAC-N (wood clearing from wetland and terrestrial habitat, mounds, revegetation, and log boom)</u>

Several nests were located in the Bush Arm Causeway NW area. These include eight Savannah Sparrow nests and seven Spotted Sandpiper nests (Figure 7-55). Nine nests have been documented in the control polygon, most being of Savannah Sparrow (n = 4) and Spotted Sandpiper (n = 3). Only three nests were documented from the treatment polygon, including one each of Savannah Sparrow, Spotted Sandpiper, and Lincoln's Sparrow. The Savannah Sparrow and Spotted Sandpiper nests in the treatment polygon were both located in 2017.

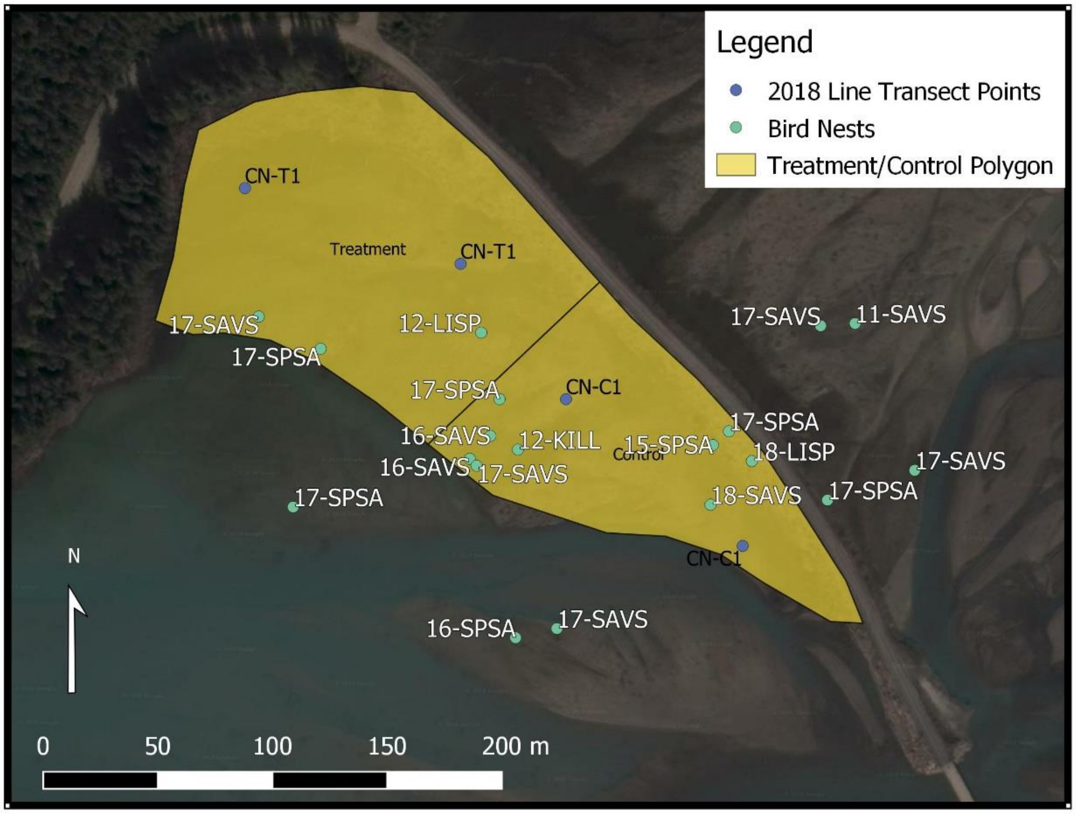


Figure 7-55: Locations of treatment and control and polygons and detected bird nests at Bush Arm Causeway NW at Bush Arm, Kinbasket reservoir. Note: first two numbers in the nest prefix denote the year of detection. Years prior to 2016 were pre-treatment.

<u>BAC-S (wood clearing from terrestrial habitat, mounds, and revegetation)</u>

Few nests were discovered in the BAC-S treatment polygon. One Northern Flicker was found nesting within an old stump during pre-treatment monitoring (2015), and a Spotted Sandpiper nest was discovered in 2018 (Figure 7-56). Both Killdeer and Spotted Sandpiper nests have been found in proximity to the polygon. The only nests within the treatment polygon were a Chipping Sparrow in 2015 (pre-treatment) and Savannah Sparrow in 2018. Multiple nests in proximity, especially at the northern end of the polygon, belonged to Chipping Sparrows. Multiple Cedar Waxwing nests immediately west of the polygon also refer to greater tree and shrub cover near this polygon. Multiple sparrow nests have been found close to the control polygon in grassy areas.

Figure 7-56: Locations of treatment, control and forest reference polygons and detected bird nests at Bush Arm Causeway SW at Bush Arm, Kinbasket reservoir. Note: first two numbers in the nest prefix denote the year of detection. Years prior to 2016 were pre-treatment.

<u>KM88 (sedge plug revegetation)</u>

Only two nests were located at KM88 in 2018. Both were Savannah Sparrow, one near the treatment (transect TU-5-T1) and one near the control (transect CU-3-T1) (Figure 7-57).

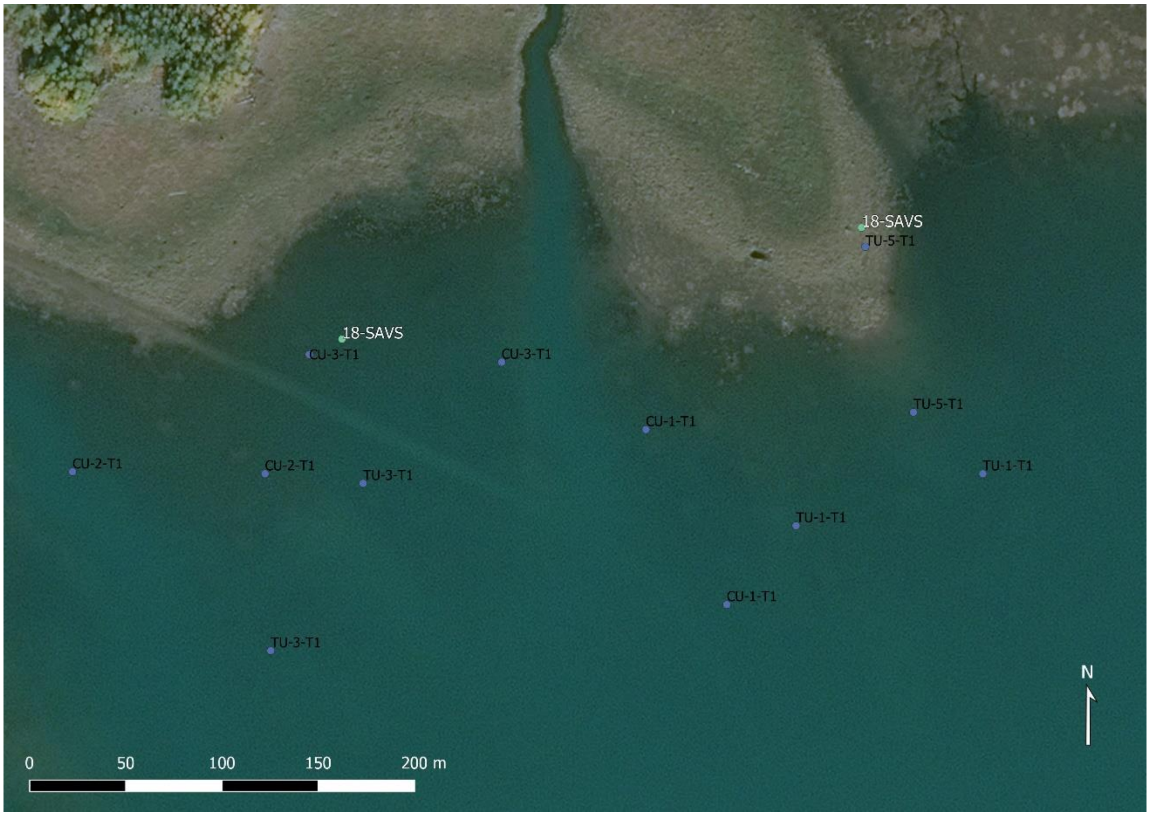


Figure 7-57: Locations of two bird nests at KM88 in Kinbasket reservoir. Note: bird survey line transect points are also overlaid (blue points).

VP-N (wood clearing from wetland and terrestrial habitat, and log boom)

Valemount Peatland North represents one of the main nesting areas for birds in our study areas. A total of 52 nests were in this region, the largest number of any of our study areas (Figure 7-58). These nests were predominantly from sparrows, with Savannah Sparrow (24 nests), Clay-colored Sparrow (12 nests), and Lincoln's Sparrow (8 nests) comprising the majority. Savannah Sparrows, while the most abundant breeding species by number of nests in the area, appear to nest slightly farther from the edge of the upland habitats. Sandpiper breeding activity is sparse in this area, though Killdeer nests (two) have been documented near the treatment area, in what were rockier, less vegetated sections. The upland edge also supports breeding species not characteristic of the drawdown zone (e.g., Dusky Flycatcher, American Redstart, and Cedar Waxwing).

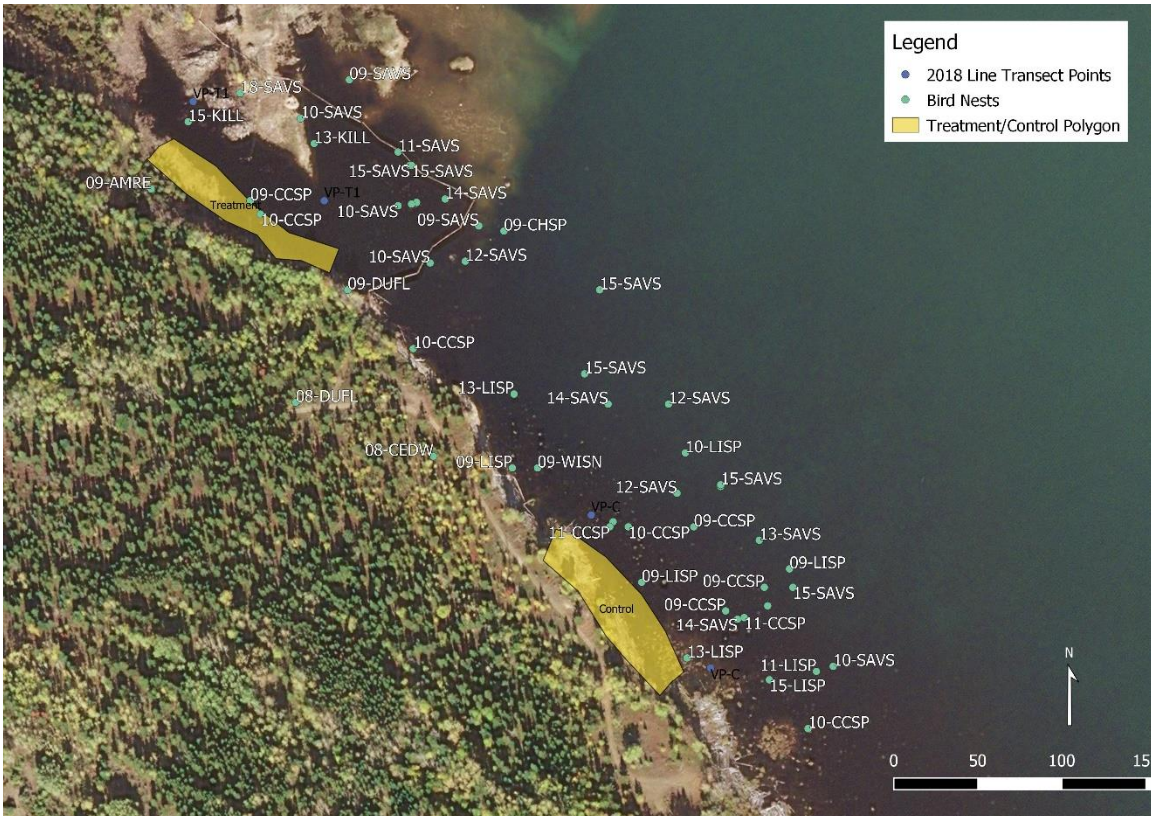


Figure 7-58: Locations of treatment, control and forest reference polygons and detected bird nests at Valemount Peatland North in Canoe Reach, Kinbasket reservoir. The first two numbers of each nest code signify the year of nest detection (2008-2017).

<u>YJ (wood clearing from terrestrial habitat)</u>

Few bird nests have been discovered at Yellowjacket Creek during 2015-2018 surveys (Figure 7-59). One Spotted Sandpiper was located almost mid-way between the control and treatment plots (and thus in neither) in 2017. In 2015, very recently fledged Spotted Sandpiper chicks (Spotted Sandpipers are precocial and leave the nest about 24 hours after hatching) were discovered in virtually the same location. The only other physical nest was a Cedar Waxwing discovered near the edge of the control plot, about 3 m high in a willow.

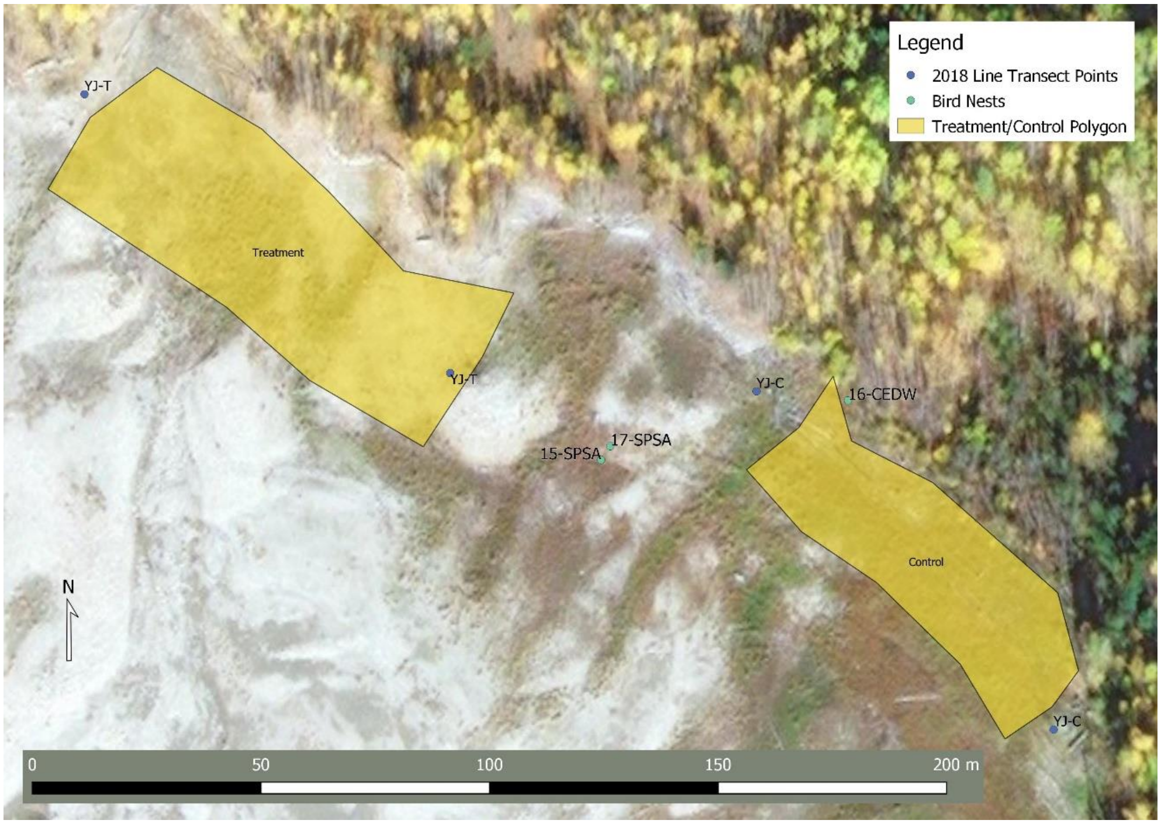


Figure 7-59: Locations of treatment, control and forest reference polygons and detected bird nests at Yellowjacket Creek in Canoe Reach, Kinbasket reservoir.

P12 (wood clearing from wetland and terrestrial habitat)

No nests were discovered near Pond 12 in 2018, the only year of surveys.

Discussion

As per previous years, evidence of nesting was generally low in all study plots, which may reflect the small size of the plots relative to territory requirements of many breeding bird species. However, certain areas, such as Valemount Peatland North, support greater numbers of breeding birds due to larger vegetated areas in proximity. If vegetation establishes on treatment plots, the number of territories and nests of bird species is expected to increase, though differences may be small given the size of prescription areas. Birds are nesting within the drawdown zone, both in heavily vegetated (e.g., Savannah Sparrow) and more open (e.g., Spotted Sandpiper) areas. Revegetation prescriptions that encourage the development of grasses, sedges, and/or shrubs are also predicted to increase nesting opportunities for most ground-nesting species that utilize the drawdown zone, as these species typically do not require much vertical vegetation growth for nesting. It is also possible that suitability has not yet increased to date but will do so as future change occurs. However, to date we have not documented any difference in use of treatment areas relative to their controls. Some site-specific comments on nesting trends are discussed below:

BAC-N

- Both Savannah Sparrow and Spotted Sandpipers are expected to nest at this location based on habitat availability. Both species could be evidence of treatment success, with Spotted Sandpipers potentially occurring in more open, gravelly substrates, and Savannah sparrows preferring areas of denser grasses and sedges with some shrubs. However, the greater number of nests of both species in the control area suggests that pre-existing conditions limit nesting within the treatment area, or that the treatment has not been successful at increasing nest productivity within that area. That both Savannah Sparrow and Spotted Sandpiper nested in 2017 indicates that nesting habitat for these species exists within the treatment area.

BAC-S

- The drawdown zone is more limited at Bush Arm Causeway SW compared to Bush Arm Causeway NW and Valemount Peatland North, potentially limiting nesting opportunities for open-country species at this site.
- The control polygon at this site is characterized by large amounts of woody debris, but also greater shrub cover. The negative impacts of woody debris that may limit nesting by open-country species may be outweighed by the suitability of habitat by shrub-nesting species (e.g., Chipping Sparrow). Multiple sparrow nests have been found close to the control polygon in grassy areas. This indicates, along with the multiple Savannah Sparrow nesting occurrences at Bush Arm Causeway NW, that Savannah Sparrows may breed in this area if suitable conditions (e.g., woody debris removal, revegetation success) are created/persist.

KM88

- Suitable habitat appears to exist for nesting by songbirds (i.e., sparrows) or shorebirds within both treatment and control areas. While Savannah Sparrows were more abundant within treatment transects than controls, one Savannah Sparrow nest each was found near the two treatment types. Only one year of data exist, limiting evidence of trends.

VP-N

- Nests of songbirds and shorebirds were expected in both treatment and control areas based on available habitat. Most of the discovered nests were beyond the actual treatment and control polygons, and many of these nests were detected prior to treatment application in both treatment and control areas (note: search effort may have varied by year and location). There are sparrow nests (Savannah and Clay-colored) from within or immediately adjacent to both the treatment and control areas.
- This area supports relatively high breeding bird activity compared with the other study areas. Sandpiper breeding activity is sparse in this area, though Killdeer nests (two) have been documented near the treatment area, in what were rockier, less vegetated sections. The upland edge also supports several breeding species not characteristic of the drawdown zone (e.g., Dusky Flycatcher, American Redstart, and Cedar Waxwing).

These upland species, while unlikely to nest, may occasionally utilize areas in or near the drawdown zone to forage, especially where shrubbier vegetation exists.

YJ

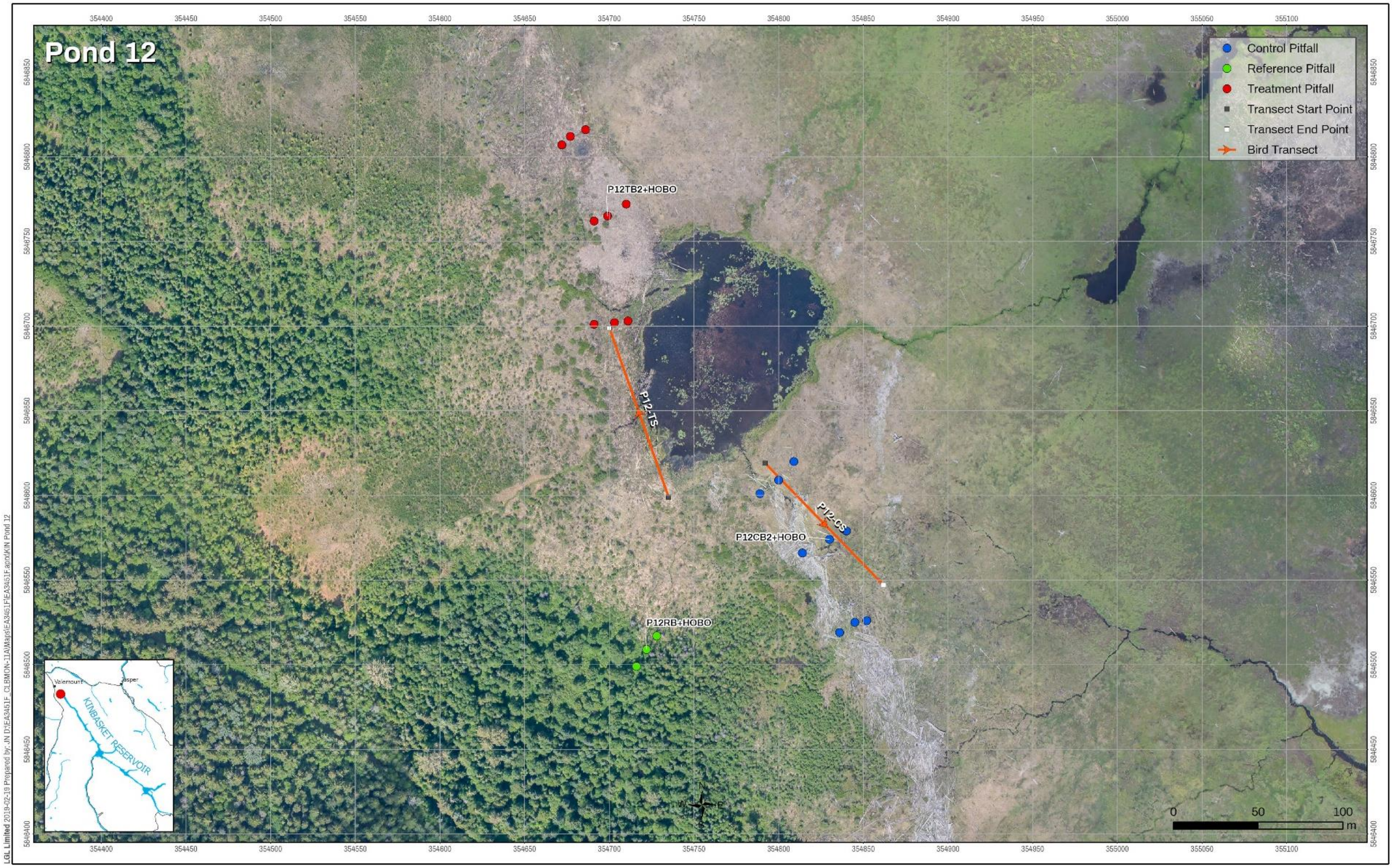
- Very few nests were found in this area. The shrubby vegetation along the edge of the control plot seems likely to provide some nesting opportunities for certain species, or at least foraging opportunities for species that have nested in nearby upland locations, while more open habitat provides nesting potential for species such as Spotted Sandpiper at this location. For example, a recently fledged Tennessee Warbler chick was observed being fed by an adult in a willow along the edge of the drawdown zone.

P12

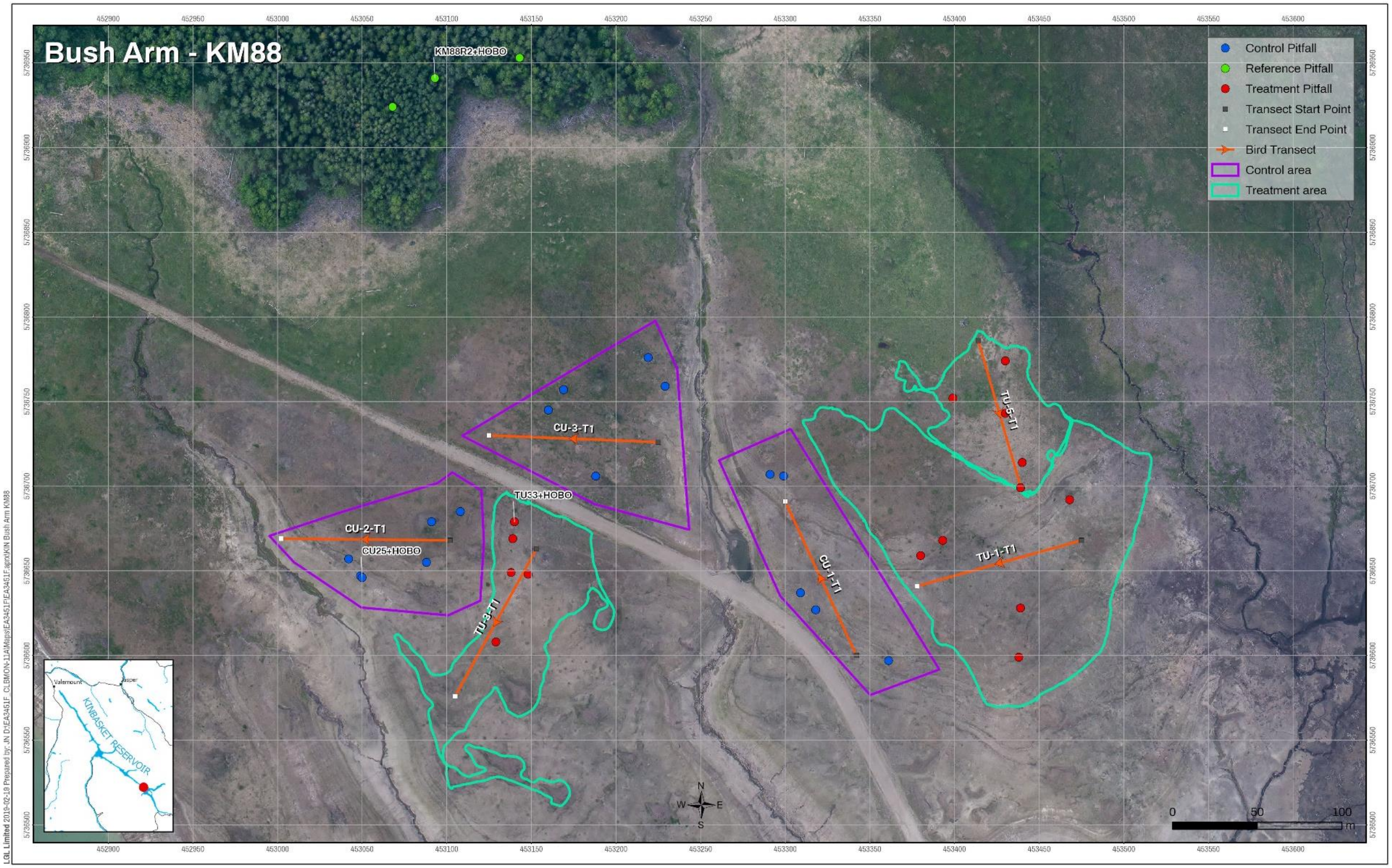
- No nests were discovered in 2018, but nesting of sparrows or shorebirds would be expected, based on habitat availability, with additional years of data collection or increased nest searching effort.

Appendix 8: Maps of sampling points for 2018 monitoring.

Two new sampling areas were added in 2018: KM88 and Pond 12. All other sampling points were equivalent to those presented in Wood et al. 2018 (except that reference songbird point counts were not conducted in 2018 monitoring).



Map 7-1: Sampling locations at Pond 12 in Canoe Reach in 2018.



Map 7-2: **Sampling locations at KM88 (Big Bend) in Bush Arm in 2018.**

Appendix 9: Species lists.

Table 7-13: List of spider (Araneae) species identified in samples for 2018 including adult abundance.

Family	Scientific Name	Authority	Species Code	Adults
Amaurobiidae	*Cybaeopsis euopla*	(Bishop & Crosby)	Cyba.euop	1
Clubionidae	*Clubiona canadensis*	Emerton	Club.cana	5
	Clubiona kulczynskii	Lessert	Club.kulc	5
	Clubiona norvegica	Strand	Club.norv	6
Cybaeidae	*Cryphoeca exlineae*	Roth	Cryp.exli	13
Dictynidae	*Argenna obesa*	Emerton	Arge.obes	8
Gnaphosidae	*Callilepis pluto*	Banks	Call.plut	2
	Drassodes neglectus	(Keyserling)	Dras.negl	3
	Gnaphosa microps	Holm	Gnap.micr	3
	Gnaphosa muscorum	(L. Koch)	Gnap.musc	8
	Gnaphosa parvula	Banks	Gnap.parv	60
	Haplodrassus hiemalis	(Emerton)	Hapl.hiem	6
	Haplodrassus signifer	(C.L. Koch)	Hapl.sign	2
	Micaria aenea	Thorell	Mica.aene	10
	Micaria pulicaria	(Sundevall)	Mica.puli	16
	Micaria rossica	Thorell	Mica.ross	71
	Zelotes fratris	Chamberlin	Zelo.frat	11
	Zelotes puritanus	Chamberlin	Zelo.puri	2
Hahniidae	*Hahnia cinerea*	Emerton	Hahn.cine	1
	Neoantistea agilis	(Keyserling)	Neoa.agil	79
	Neoantistea magna	(Keyserling)	Neoa.magn	21
Linyphiidae	*Agyneta fabra*	(Keyserling)	Agyn.fabr	4
	Agyneta lophophor	(Chamberlin & Ivie)	Agyn.loph	3
	Agyneta protrudens	(Chamberlin & Ivie)	Agyn.prot	1
	Aphileta microtarsa	(Emerton)	Aphi.micr	1
	Aphileta misera	(O. Pickard-Cambridge)	Aphi.mise	3
	Bathyphantes brevipes	(Emerton)	Bath.brev	47
	Bathyphantes pallidus	(Banks)	Bath.pall	37
	Caviphantes saxetorum	(Hull)	Cavi.saxe	4
	Ceraticelus emertoni	(O. Pickard-Cambridge)	Cera.emer	2
	Ceraticelus fissiceps	(O. Pickard-Cambridge)	Cera.fiss	5
	Collinsia ksenia	(Crosby & Bishop)	Coll.ksen	8
	Diplocentria bidentata	(Emerton)	Dipl.bide	1
	Dismodicus decemoculatus	(Emerton)	Dism.dece	27
	Erigone aletris	Crosby & Bishop	Erig.alet	4
	Erigone blaesa	Crosby & Bishop	Erig.blae	8
	Erigone cristatopalpus	Simon	Erig.cris	1
	Erigone dentigera	O. Pickard-Cambridge	Erig.denti	42
	Erigone dentosa	O. Pickard-Cambridge	Erig.dento	13
	Eulaira arctoa	Holm	Eula.arct	2

Family	Scientific Name	Authority	Species Code	Adults
Linyphiidae (continued)	*Gnathonarium taczanowskii*	(O. Pickard-Cambridge)	Gnat.tacz	10
	Grammonota gigas	(Banks)	Gram.giga	3
	Hypomma marxi	(Keyserling)	Hypo.marx	1
	Hypselistes florens	(O. Pickard-Cambridge)	Hyps.flor	1
	Islandiana flaveola	(Banks)	Isla.flav	1
	Kaestneria pullata	(O. Pickard-Cambridge)	Kaes.pull	25
	Lepthyphantes alpinus	(Emerton)	Lept.alpi	6
	Lepthyphantes intricatus	(Emerton)	Lept.intr	3
	Lepthyphantes turbatrix	(O. Pickard-Cambridge)	Lept.turb	2
	Maso sundevalli	(Westring)	Maso.sund	15
	Mermessus trilobatus	(Emerton)	Merm.tril	34
	Microlinyphia mandibulata	(Emerton)	Micr.mand	2
	Neriene litigiosa	(Keyserling)	Neri.liti	1
	Oedothorax alascensis	(Banks)	Oedo.alas	31
	Oreonetides filicatus	(Crosby)	Oreo.fili	1
	Pelecopsis mengei	(Simon)	Pele.meng	91
	Pelecopsis moesta	(Banks)	Pele.moes	1
	Pelecopsis sculpta	(Emerton)	Pele.scul	13
	Pocadicnemis americana	Millidge	Poca.amer	9
	Porrhomma convexum	(Westring)	Porr.conv	1
	Praestigia kulczynskii	Eskov	Prae.kulc	35
	Saaristoa sammamish	(Levi & Levi)	Saar.samm	9
	Sciastes truncatus	(Emerton)	Scia.trun	2
	Scotinotylus exsectoides	Millidge	Scot.exse	1
	Scotinotylus patellatus	(Emerton)	Scot.pate	1
	Sisicottus montanus	(Emerton)	Sisi.mont	1
	Sisicottus nesides	(Chamberlin)	Sisi.nesi	2
	Symmigma minimum	(Emerton)	Symm.mini	1
	Tapinocyba minuta	(Emerton)	Tapi.minu	3
	Tennesseelum formica	(Emerton)	Tenn.form	3
	Tenuiphantes zelatus	(Zorsch)	Tenu.zela	3
	Tunagyna debilis	(Banks)	Tuna.debi	4
	Walckenaeria atrotibialis	(O. Pickard-Cambridge)	Walc.atro	9
	Walckenaeria auranticeps	(Emerton)	Walc.aura	1
	Walckenaeria directa	(O. Pickard-Cambridge)	Walc.dire	1
	Walckenaeria exigua	Millidge	Walc.exig	18
	Walckenaeria vigilax	(Blackwall)	Walc.vigi	2
Liocranidae	*Agroeca ornata*	Banks	Agro.orna	7

Family	Scientific Name	Authority	Species Code	Adults
Lycosidae	*Alopecosa aculeata*	(Clerck)	Alop.acul	40
	Arctosa raptor	(Kulczyński)	Arct.rapt	3
	Arctosa rubicunda	(Keyserling)	Arct.rubi	20
	Pardosa dorsuncata	Lowrie & Dondale	Pard.dors	1
	Pardosa fuscula	(Thorell)	Pard.fusc	679
	Pardosa groenlandica	(Thorell)	Pard.groe	4
	Pardosa mackenziana	(Keyserling)	Pard.mack	114
	Pardosa moesta	Banks	Pard.moes	568
	Pardosa tesquorum	(Odenwall)	Pard.tesq	17
	Pardosa wyuta	Gertsch	Pard.wyut	11
	Pardosa xerampelina	(Keyserling)	Pard.xera	819
	Pirata piraticus	(Clerck)	Pira.pira	134
	Piratula insularis	(Emerton)	Pira.insu	4
	Trochosa terricola	Thorell	Troc.terr	17
Philodromidae	*Philodromus cespitum*	(Walckenaer)	Phil.cesp	1
	Philodromus oneida	Levi	Phil.onei	1
	Philodromus rufus	Walckenaer	Phil.rufu	1
	Thanatus striatus	C.L. Koch	Than.stri	1
	Tibellus oblongus	(Walckenaer)	Tibe.oblo	2
Phrurolithidae	*Phrurotimpus borealis*	(Emerton)	Phru.bore	1
	Scotinella pugnata	(Emerton)	Scot.pugn	1
Pisauridae	*Dolomedes triton*	(Walckenaer)	Dolo.trit	1
Salticidae	*Eris militaris*	(Hentz)	Eris.mili	1
	Evarcha proszynskii	Marusik & Logunov	Evar.pros	1
	Habronattus decorus	(Blackwall)	Habr.deco	4
Tetragnathidae	*Pachygnatha clercki*	Sundevall	Pach.cler	6
Theridiidae	*Enoplognatha marmorata*	(Hentz)	Enop.marm	1
	Euryopis argentea	Emerton	Eury.arge	9
	Robertus fuscus	(Emerton)	Robe.fusc	1
	Robertus vigerens	(Chamberlin & Ivie)	Robe.vige	5
Thomisidae	*Ozyptila sincera*	Kulczynski	Ozyp.sinc	1
	Xysticus discursans	Keyserling	Xyst.disc	11
	Xysticus elegans	Keyserling	Xyst.eleg	1
	Xysticus ellipticus	Turnbull et al.	Xyst.elli	17

Table 7-14: List of ground beetle (Coleoptera: Carabidae) species identified in samples for 2018 including adult abundance.

Species Code	Scientific Name	Authority	Adults
Agon.affi	*Agonum affine*	Kirby, 1837	34
Agon.cons	*Agonum consimile*	(Gyllenhal, 1810)	3
Agon.corv	*Agonum corvus*	(LeConte, 1860)	2
Agon.cup	*Agonum cupreum*	Dejean, 1831	624
Agon.cupr	*Agonum cupripenne*	(Say, 1823)	310
Agon.grat	*Agonum gratiosum*	(Mannerheim, 1853)	2
Agon.meta	*Agonum metallescens*	(LeConte, 1854)	8
Agon.muel	*Agonum muelleri*	(Herbst, 1784)	1
Agon.plac	*Agonum placidum*	(Say, 1823)	8
Agon.retr	*Agonum retractum*	LeConte, 1846	8
Agon.sord	*Agonum sordens*	Kirby, 1837	5
Agon.sutu	*Agonum suturale*	(Say, 1830)	51
Amar.litt	*Amara littoralis*	Dejean, 1828	18
Amar.luni	*Amara lunicollis*	Schiødte, 1837	5
Amar.patr	*Amara patruelis*	Dejean, 1831	36
Amar.sinu	*Amara sinuosa*	(Casey, 1918)	10
Badi.neop	*Badister neopulchellus*	Lindroth, 1954	2
Bemb.bima	*Bembidion bimaculatum*	(Kirby, 1837)	5
Bemb.incr	*Bembidion incrematum*	LeConte, 1860	2
Bemb.inte	*Bembidion interventor*	Lindroth, 1963	69
Bemb.kupr	*Bembidion kuprianovii*	Mannerheim, 1843	1
Bemb.nigr	*Bembidion nigripes*	(Kirby, 1837)	5
Bemb.plan	*Bembidion planatum*	(LeConte, 1847)	1
Bemb.quad	*Bembidion quadrimaculatum*	(LeConte, 1852)	3
Bemb.tran	*Bembidion transparens*	(Gebler, 1830)	16
Blet.huds	*Blethisa hudsonica*	Casey, 1924	1
Blet.quad	*Blethisa quadricollis*	Haldeman, 1847	2
Brad.nigr	*Bradycellus nigrinus*	(Dejean, 1829)	6
Cala.ingr	*Calathus ingratus*	Dejean, 1828	32
Cara.taed	*Carabus taedatus*	LeConte, 1850	4
Chla.lith	*Chlaenius lithophilus*	Say, 1823	5
Chla.nige	*Chlaenius niger*	Randall, 1838	21
Cici.long	*Cicindela longilabris*	Say, 1824	1
Cici.oreg	*Cicindela oregona*	LeConte, 1856	8
Cymi.crib	*Cymindis cribricollis*	Dejean, 1831	1
Dich.cogn	*Dicheirotrichus cognatus*	(Gyllenhal, 1827)	13
Elap.clai	*Elaphrus clairvillei*	Kirby, 1837	3

Species Code	Scientific Name	Authority	Adults
Harp.affi	*Harpalus affinis*	(Schrank, 1781)	2
Harp.fulv	*Harpalus fulvilabris*	Mannerheim, 1853	2
Harp.lati	*Harpalus laticeps*	LeConte, 1850	1
Harp.nigr	*Harpalus nigritarsis*	C.R. Sahlberg, 1827	3
Harp.opac	*Harpalus opacipennis*	(Haldeman, 1843)	1
Harp.somn	*Harpalus somnulentus*	Dejean, 1829	13
Lori.dece	*Loricera decempunctata*	Eschscholtz, 1833	2
Nebr.obli	*Nebria obliqua*	LeConte, 1867	15
Noti.semi	*Notiophilus semistriatus*	Say, 1823	1
Patr.foss	*Patrobus fossifrons*	(Eschscholtz, 1823)	9
Patr.sept	*Patrobus septentrionis*	Dejean, 1828	1
Plat.dece	*Platynus decentis*	(Say, 1823)	16
Plat.mann	*Platynus mannerheimii*	(Dejean, 1828)	17
Poec.lucu	*Poecilus lucublandus*	(Say, 1823)	14
Pter.adst	*Pterostichus adstrictus*	Eschscholtz, 1823	391
Pter.comm	*Pterostichus commutabilis*	(Motschulsky, 1866)	1
Pter.herc	*Pterostichus herculaneus*	Mannerheim, 1843	10
Pter.mela	*Pterostichus melanarius*	(Illiger, 1798)	4
Pter.neob	*Pterostichus neobrunneus*	Lindroth, 1966	10
Pter.patr	*Pterostichus patruelis*	(Dejean, 1831)	1
Pter.pens	*Pterostichus pensylvanicus*	LeConte, 1873	16
Pter.prot	*Pterostichus protractus*	LeConte, 1860	12
Pter.ripa	*Pterostichus riparius*	(Dejean, 1828)	20
Scap.angu	*Scaphinotus angusticollis*	(Mannerheim, 1823)	15
Scap.marg	*Scaphinotus marginatus*	(Fischer von Waldheim, 1820)	55
Sten.fuli	*Stenolophus fuliginosus*	Dejean, 1829	6
Synt.amer	*Syntomus americanus*	(Dejean, 1831)	5
Synu.impu	*Synuchus impunctatus*	(Say, 1823)	5
Trec.chal	*Trechus chalybeus*	Dejean, 1831	2

Table 7-15: Number of observations of birds by site and habitat type. Data constrained by species and distance (see Dataset 5). Data presented in alphabetical order by species code.

		Site and Habitat														
		VP-N		YJ		BAC-N		BAC-S		Pond 12		KM88		Total		
Spp Code	Common Name	C	T	C	T	C	T	C	T	C	T	C	T	C	T	Total
ALFL	Alder Flycatcher									1				1	0	1
AMRE	American Redstart	2	2	4	3		1	3						9	6	15
AMRO	American Robin					1	2	3	4	1				5	6	11
BAEA	Bald Eagle													0	0	0
BASW	Barn Swallow					2	1							2	1	3
BKSW	Bank Swallow											1	1	1	1	2
CCSP	Clay-colored Sparrow	8	3					3	1	1	1			12	5	17
CEWA	Cedar Waxwing					1		1			1			2	1	3
CHSP	Chipping Sparrow		3	3	2	4	1	7	3					14	9	23
COYE	Common Yellowthroat	10	2	4	1	1	1		1	3	4			18	9	27
DEJU	Dark-eyed Junco	2	1	1					1					3	2	5
DUFL	Dusky Flycatcher	2	1				1	2	3					4	5	9
GCKI	Golden-crowned Kinglet				1									0	1	1
GRYE	Greater Yellowlegs										1			0	1	1
HAFL	Hammond's Flycatcher													0	0	0
HAWO	Hairy Woodpecker													0	0	0
KILL	Killdeer					2	2		2					2	4	6
LEFL	Least Flycatcher		1											0	1	1
LISP	Lincoln's Sparrow	12	4	9	3	3	6	10	11	4	5		1	38	30	68
MACW	MacGillivray's Warbler		1	1				1						2	1	3
MGNW	Magnolia Warbler		1	2										2	1	3
MOBL	Mountain Bluebird								1					0	1	1
NOWA	Northern Waterthrush	2								1				3	0	3
NRWS	Northern Rough-winged Swallow							1	1		1			1	2	3
OCWA	Orange-crowned Warbler	4	2	3					3					7	5	12
PISI	Pine Siskin	1												1	0	1
REVI	Red-eyed Vireo	1									2			1	2	3
RUHU	Rufous Hummingbird							1			1			1	1	2
SAVS	Savannah Sparrow	12	5	1	1	10	7	4	8	4		5	10	36	31	67
SOSP	Song Sparrow		1						1		1			0	3	3
SPSA	Spotted Sandpiper				3	6	1							6	4	10
SWTH	Swainson's Thrush		1											0	1	1
TEWA	Tennessee Warbler	3	3	1		2	3	4	1					10	7	17
TRSW	Tree Swallow	1				1		1	2	1	1			4	3	7
VESP	Vesper Sparrow		1											0	1	1
VGSW	Violet-green Swallow									1				1	0	1
WAVI	Warbling Vireo	1	1	4	1		1							5	3	8
WEME	Western Meadowlark		1											0	1	1
WIFL	Willow Flycatcher						2			1	1			1	3	4
WISN	Wilson's Snipe						1				1			0	2	2
WIWA	Wilson's Warbler		1											0	1	1
WTSP	White-throated Sparrow			1										1	0	1
YEWA	Yellow Warbler		2											0	2	2
YRWA	Yellow-rumped Warbler						1		1					0	2	2
	Total	*61*	*37*	*34*	*15*	*33*	*31*	*41*	*44*	*18*	*20*	*6*	*12*	*193*	*159*	*352*

Table 7-16: Comprehensive list of bird species found to utilize the drawdown zone and adjacent margins from all detections from 2015 to 2018. List presented in taxonomic order.

No.	Code	Common Name	Scientific Name	BC Status	COSEWIC	No. Obs
Waterfowl						
1	CAGO	Canada Goose	*Branta canadensis*	Yellow	.	2
2	BWTE	Blue-winged Teal	*Spatula discors*	Yellow	.	1
Upland Game Birds						
3	RUGR	Ruffed Grouse	*Bonasa umbellus*	Yellow	.	2
Swifts and Hummingbirds						
4	BLSW	Black Swift	*Cypseloides niger*	Blue	Endangered	2
5	VASW	Vaux's Swift	*Chaetura vauxi*	Yellow	.	4
6	RUHU	Rufous Hummingbird	*Selasphorus rufus*	Yellow	.	6
Shorebirds, Gulls, Auks, and Allies						
7	KILL	Killdeer	*Charadrius vociferus*	Yellow	.	17
8	WISN	Wilson's Snipe	*Gallinago delicata*	Yellow	.	3
9	SPSA	Spotted Sandpiper	*Actitis macularius*	Yellow	.	34
10	GRYE	Greater Yellowlegs	*Tringa melanoleuca*	Yellow	.	2
11	RBGU	Ring-billed Gull	*Larus delawarensis*	Yellow	.	2
Loons						
12	COLO	Common Loon	*Gavia immer*	Yellow	Not at Risk	2
Hawks, Eagles, and Allies						
13	OSPR	Osprey	*Pandion haliaetus*	Yellow	.	2
14	SSHA	Sharp-shinned Hawk	*Accipiter striatus*	Yellow	Not at Risk	1
15	BAEA	Bald Eagle	*Haliaeetus leucocephalus*	Yellow	Not at Risk	3
16	RTHA	Red-tailed Hawk	*Buteo jamaicensis*	Yellow	Not at Risk	1
Kingfishers and Allies						
17	BEKI	Belted Kingfisher	*Megaceryle alcyon*	Yellow	.	4
Woodpeckers and Allies						
18	HAWO	Hairy Woodpecker	*Dryobates villosus*	Yellow	.	2
19	NOFL	Northern Flicker	*Colaptes auratus*	Yellow	.	2
20	PIWO	Pileated Woodpecker	*Dryocopus pileatus*	Yellow	.	2
21	EAKI	Eastern Kingbird	*Tyrannus tyrannus*	Yellow	.	2
Songbirds						
22	WWPE	Western Wood-Pewee	*Contopus sordidulus*	Yellow	.	1
23	ALFL	Alder Flycatcher	*Empidonax alnorum*	Yellow	.	3
24	WIFL	Willow Flycatcher	*Empidonax traillii*	Yellow	.	9
25	LEFL	Least Flycatcher	*Empidonax minimus*	Yellow	.	5
26	HAFL	Hammond's Flycatcher	*Empidonax hammondii*	Yellow	.	3
27	DUFL	Dusky Flycatcher	*Empidonax oberholseri*	Yellow	.	26
28	WAVI	Warbling Vireo	*Vireo gilvus*	Yellow	.	30
29	REVI	Red-eyed Vireo	*Vireo olivaceus*	Yellow	.	10
30	AMCR	American Crow	*Corvus brachyrhynchos*	Yellow	.	1
31	CORA	Common Raven	*Corvus corax*	Yellow	.	4
32	TRSW	Tree Swallow	*Tachycineta bicolor*	Yellow	.	12
33	VGSW	Violet-green Swallow	*Tachycineta thalassina*	Yellow	.	1
34	NRWS	Northern Rough-winged Swallow	*Stelgidopteryx serripennis*	Yellow	.	12
35	BKSW	Bank Swallow	*Riparia riparia*	Yellow	Threatened	2
36	BASW	Barn Swallow	*Hirundo rustica*	Blue	Threatened	11
37	BCCH	Black-capped Chickadee	*Poecile atricapillus*	Yellow	.	2
38	RBNU	Red-breasted Nuthatch	*Sitta canadensis*	Yellow	.	2
39	PAWR	Pacific Wren	*Troglodytes pacificus*	Yellow	.	1
40	GCKI	Golden-crowned Kinglet	*Regulus satrapa*	Yellow	.	2
41	MOBL	Mountain Bluebird	*Sialia currucoides*	Yellow	.	2
42	SWTH	Swainson's Thrush	*Catharus ustulatus*	Yellow	.	16
43	HETH	Hermit Thrush	*Catharus guttatus*	Yellow	.	2
44	AMRO	American Robin	*Turdus migratorius*	Yellow	.	29
45	CEWA	Cedar Waxwing	*Bombycilla cedrorum*	Yellow	.	17
46	EVGR	Evening Grosbeak	*Coccothraustes vespertinus*	Yellow	Special Concern	2
47	RECR	Red Crossbill	*Loxia curvirostra*	Yellow	.	1
48	PISI	Pine Siskin	*Spinus pinus*	Yellow	.	19
49	CHSP	Chipping Sparrow	*Spizella passerina*	Yellow	.	48
50	CCSP	Clay-colored Sparrow	*Spizella pallida*	Yellow	.	22
51	VESP	Vesper Sparrow	*Pooecetes gramineus*	Yellow	.	2
52	SAVS	Savannah Sparrow	*Passerculus sandwichensis*	Yellow	.	93
53	SOSP	Song Sparrow	*Melospiza melodia*	Yellow	.	3
54	LISP	Lincoln's Sparrow	*Melospiza lincolnii*	Yellow	.	86
55	WTSP	White-throated Sparrow	*Zonotrichia albicollis*	Yellow	.	4
56	DEJU	Dark-eyed Junco	*Junco hyemalis*	Yellow	.	9
57	WEME	Western Meadowlark	*Sturnella neglecta*	Yellow	.	1
58	COGR	Common Grackle	*Quiscalus quiscula*	Yellow	.	1
59	NOWA	Northern Waterthrush	*Parkesia noveboracensis*	Yellow	.	3
60	TEWA	Tennessee Warbler	*Oreothlypis peregrina*	Yellow	.	31
61	OCWA	Orange-crowned Warbler	*Oreothlypis celata*	Yellow	.	13
62	MACW	MacGillivray's Warbler	*Geothlypis tolmiei*	Yellow	.	10
63	COYE	Common Yellowthroat	*Geothlypis trichas*	Yellow	.	29
64	AMRE	American Redstart	*Setophaga ruticilla*	Yellow	.	36
65	MGNW	Magnolia Warbler	*Setophaga magnolia*	Yellow	.	7
66	YEWA	Yellow Warbler	*Setophaga petechia*	Yellow	.	2
67	YRWA	Yellow-rumped Warbler	*Setophaga coronata*	Yellow	.	10
68	WIWA	Wilson's Warbler	*Cardellina pusilla*	Yellow	.	4
69	WETA	Western Tanager	*Piranga ludoviciana*	Yellow	.	4
70	LZBU	Lazuli Bunting	*Passerina amoena*	Yellow	.	1

Table 7-17: List of vertebrate by-catch by reach, site, and year.

			Canoe Reach											Bush Arm								
			VP-N					YJ					P12	BAC-N				BAC-S				KM88
Taxon Group	Scientific Name	Common Name	2014	2015	2016	2017	2018	2014	2015	2016	2017	2018	2018	2015	2016	2017	2018	2015	2016	2017	2018	2018
Small Mammal	*Sorex cinereus*	Masked Shrew	12			3	5	18	4				3	1					1	1	2	
Small Mammal	*Sorex hoyi*	American Pygmy Shrew	3			2																
Small Mammal	*Sorex palustris*	American Water Shrew						1														
Small Mammal	*Sorex vagrans*	Vagrant Shrew	2			4	1	3	1				1			2		2	2	1		
Small Mammal	*Microtus pennsylvanicus*	Meadow Vole	5				3	12														
Small Mammal	*Microtus* sp.	Vole species (unknown)							4													
Amphibian	*Ambystoma macrodactylum*	Long-toed Salamander	8	1			4															
Amphibian	*Anaxyrus boreas*	Western Toad																5				
Amphibian	*Rana luteiventris*	Columbia Spotted Frog											1									