Campbell River Project Water Use Plan

Upper and Lower Campbell Lake Reservoirs Shoreline Vegetation Model Validation

Implementation Year 5

Reference: JHTMON-10

Upper and Lower Campbell Lake Reservoirs Shoreline Vegetation Model Validation Year 5 Monitoring Report

Study Period: 2019

Laich-Kwil-Tach Environmental Assessment Ltd. Partnership
Ecofish Research

July 29, 2019

JHTMON-10: Upper and Lower Campbell Lake Reservoirs Shoreline Vegetation Model Validation

Year 5 Monitoring Report

BC Hydro Water Licensing Requirements
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EXECUTIVE SUMMARY

Water Use Plans (WUPs) were developed for most of BC Hydro's hydroelectric facilities through a multi-stakeholder consultative process. WUPs were developed to balance power generation with other water uses such fish, wildlife, and recreation. To address outstanding uncertainties, BC Hydro is undertaking several multi-year environmental monitoring studies, as directed by orders from the provincial Comptroller of Water Rights.

The Campbell River WUP Order established the JHTMON-10 monitoring program to address uncertainty associated with the accuracy of a shoreline vegetation model (SVM). The SVM was developed to predict the response of riparian and emergent shoreline vegetation to operational changes proposed during WUP development. Model predictions of the elevations of boundaries between riparian vegetation communities were used to assess potential changes in the extent and distribution of shoreline vegetation in response to operational changes. Predictions were then used to make inferences about associated effects on wildlife use.

Following revisions to the terms of reference (TOR; BC Hydro 2016), JHTMON-10 focuses exclusively on Upper Campbell Reservoir, which was the subject of vegetation surveys in 2001 (McLennan and Veenstra 2001) that were used to develop the SVM. The JHTMON-10 program aims to address the following five management questions:

1. Does the lacustrine shoreline vegetation model accurately predict the reservoir elevation bands that bound the predefined plant community types?
2. If the model is in error, is the magnitude of the error such that it would warrant a change in the predicted outcome of the WUP?
3. Are there changes to the modelling approach that could improve its accuracy for implementation in future WUP reviews?
4. Is it reasonable to expect that most riparian plant ecosystems require shoreline slopes to have a gradient less than 15% to perpetuate? If this is not reasonable, what is the shoreline slope gradient that is required for plant ecosystem persistence? [Note that the second part of this question was added following the TOR amendment (BC Hydro 2016)].
5. Has the distribution of riparian plant ecosystems changed following implementation of the WUP and if so, can the change be attributed to the WUP operation?

The three hypotheses related to the management questions are:

$H_0$1: Measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model.

$H_0$2: The likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient.

$H_0$3: Plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation.

Year 5 of the JHTMON-10 study was completed during 2018–2019 and included data collection at Upper Campbell Reservoir, which built on vegetation surveys and analysis undertaken in Year 1 (2014-2015) (Ballin *et al.* 2015). This Year 5 Annual Report describes all work completed in Year 5 associated with the revised TOR to address the study management questions and hypotheses. Currently, no further work is scheduled to be completed as part of JHTMON-10.

To answer Management Question 1, the mean elevations of vegetation communities measured around the shoreline of Upper Campbell Reservoir in Year 1 and Year 5 were compared with corresponding predictions from the SVM. From lowest to highest elevations, the six dominant vegetation communities include Lake Mudflat, Spearwort Lakeflat; Hairgrass – Water sedge, short Sitka willow – Water sedge, tall Sitka willow – Water sedge and Upland Forest. In Year 5, SVM predictions for 2018 were developed using a version of the model (SVM_{2018}) that was applied using reservoir elevation data for 2001–2017; i.e., a 17-year period that ended the year prior to field data collection, consistent with the approach taken when the SVM model was originally developed (Bruce 2002). Detailed error statistics were calculated for each plant community boundary and several data analyses were completed to evaluate the accuracy of SVM predictions and test $H_0$1. Specifically, ANOVA was undertaken to compare model predictions with field measurements, a t-test was undertaken to confirm whether the mean model error differed from zero, and a Kolmogorov–Smirnov test was undertaken to evaluate whether SVM parameters that describe vegetation exposure times were consistent among different versions of the SVM. A hectare estimation tool was then used to convert elevation predictions made using the SVM_{2018} to predictions of the area of each vegetation community, based on a digital elevation model (DEM) and assumptions about the critical slope above which shoreline is unvegetated. These predicted areas were then compared with areas of each vegetation community measured using air photo analysis to calculate error in the areal estimates. Based on these analyses, Management Question 1 was answered by stating that the SVM accurately predicted the average elevation bands of *four out of five* vegetation community boundaries, but it failed to accurately predict the mean elevation of the lowest community boundary between Mudflat and Spearwort Lakeflat. The SVM_{2018} predicted a higher elevation boundary between Mudflat and Spearwort Lakeflat than was observed. It was therefore appropriate to retain $H_0$1 that *measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model*, with the qualifier that the hypothesis was rejected for the lowest plant community boundary.

The failure of the SVM to accurately predict the Mudflat and Spearwort Lakeflat boundary is not expected to alter the assessment of the consequences to obligate and facultative aquatic wildlife use from operational change. This is because the Lake Mudflat community is expected to generally have lower wildlife values than Spearwort Lakeflat. The occurrence of a wider band of Spearwort Lakeflat than predicted by the SVM therefore indicates a benefit to wildlife relative to the predictions that were made during WUP development. Based on this, our answer to Management Question 2 is that it is not expected to be necessary to re-evaluate the predicted outcome of the WUP for wildlife use. Nonetheless, it would be appropriate for the details of model performance described in this report to be reviewed in the context of the assessments made by the Wildlife Technical Committee during WUP development. This is because riparian vegetation communities were not always consistently present in the defined sequence as conceptualized in the SVM. Similarly, although the model is not statistically in error, the implications of the error should be assessed relative to current wildlife habitat availability and potential future successional changes. These differences are not expected to affect the JHTMON-10 conclusions; however, there is some uncertainty about how these differences relate to the assessments made by the Wildlife Technical Committee because these original assessments were not provided and thus were not explicitly considered in JHTMON-10.

Changes could be made to the modelling approach to improve the accuracy of the SVM for use in future WUP reviews (Management Question 3). The SVM could be developed further with the aim of improving model accuracy and increasing the potential to apply the SVM to other waterbodies (for which the SVM is currently unsuited). Key areas to focus on to improve the SVM are the assumptions that vegetation communities are equally sensitive to the timing and duration over which water level fluctuations occur. For example, an assumption of the SVM is that communities such as Upland Forest and Spearwort Lakeflat are sensitive to water level fluctuations over the same duration (17 years for SWM_{2018}) and that all communities are insensitive to the timing of water level fluctuations, e.g., whether fluctuations occur in the winter or the growing season. To improve the accuracy of the SVM, we therefore recommend that the SVM assumptions are critically examined from the perspective of the ecological requirements of each vegetation community. The desired outcome of this would be some *a priori* rules to inform a model calibration exercise designed to improve model accuracy. These rules could then be used to set the boundaries for a model calibration exercise to further optimize the SVM parameters, while observing the principle of parsimony; i.e., it is desirable to develop a model that is as simple as necessary.

Three datasets were used to test Management Question 4 and assess whether shoreline gradient affects the distribution of reservoir plant communities: (1) Ground-based slope validation, which involved ground slope measurements collected in the field along 16 vegetation community transects and at 83 additional locations (for a total of 174 slopes analyzed); (2) Slope analysis of mapped polygons, which included a contingency test of the average slope of each vegetation community polygon delineated by air photo interpretation that had 70% or more coverage of vegetated versus non-vegetated communities (385 polygons); and, (3) DEM resampling of the slopes of 7,605 points (pixels) with and without vegetation that were randomly sampled from the shoreline area mapped with air photo interpretation. Each dataset presented complementary lines of evidence to address Management Question 4 and H_02. Following synthesis of results, it was concluded that H_02 can be rejected; i.e., we reject the hypothesis that *the likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient*. Rejection of H_02 reflects that there was a negative relationship between vegetation cover and slope for all plant community types except for

Upland Forest. In areas with a slope greater than 15%, the shoreline is more likely to be unvegetated than vegetated. However, the DEM analysis also identified that substantial areas of vegetated shoreline with cover >50% occur on slopes greater than 15%. Analysis of transect data showed that the critical slope above which shoreline was considered unvegetated (<30% cover) varied among vegetation communities. The critical slope varied from 15% to 40% depending on vegetation community, with a critical slope of 22% identified when all vegetation communities were combined. These vegetation-community-specific slope thresholds were considered more accurate than the single 15% threshold identified using the polygon analysis; accordingly, these thresholds were used when applying a hectare estimation tool to convert SVM predictions to estimates of the area of each vegetation community around the shoreline of Upper Campbell Reservoir.

In answer to Management Question 5, we conclude that the distribution of riparian plant communities has changed following implementation of the WUP and that these changes can be attributed to WUP operations. This conclusion is based on the finding that the area and distribution of riparian plant communities has increased and shifted downwards, respectively, between the pre-WUP (2001) and post-WUP survey years (2014, 2018). Based on this, we reject $H_0$3: *plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation*. The SVM generally predicted the reservoir elevation bands at which shoreline riparian vegetation communities occurred. The changes in vegetation distribution predicted and observed are due to changes in the water level regime, which differed significantly between pre- and post-WUP periods, with water levels generally higher prior to WUP implementation. Overall, the area of mapped vegetated riparian habitat in the reservoir increased from 176 ha to 363 ha between 2001 and 2018. The increase in the area of riparian habitat around the reservoir, particularly of more advanced seral stages (e.g., tall Sitka willow versus Spearwort), indicates that the WUP has increased habitat for wildlife, including obligate and facultative aquatic wildlife, around Upper Campbell Reservoir.

Overall, work completed during Year 5 provided answers for all five management questions (Table 1). Key sources of uncertainty are described; these include differences between how shoreline vegetation communities are conceptualized in the SVM and how communities are distributed in the field, as well as uncertainty with identifying an appropriate threshold of vegetation cover to define "unvegetated" shoreline. At this stage, the SVM has not been applied to develop predictions for other waterbodies in the Campbell River watershed, nor have field measurements (i.e., along vegetation transects) been collected for other waterbodies. Therefore, further work would be necessary to evaluate changes to shoreline vegetation around other waterbodies in the Campbell River watershed, including how these changes relate to predictions made during WUP development.

Table 1. Status of JHTMON-10 objectives, management questions and hypotheses after Year 5 (2018–2019).

Study Objectives	Management Questions	Management Hypotheses	Year 5 (2018-2019) Status
1.Determine if the Shoreline Vegetation Model accurately predicts the elevation of the vegetation communities.	1. Does the lacustrine shoreline vegetation model accurately predict the reservoir elevation bands that bound the predefined plant community types?	$H_0$1. Measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model.	Management Question 1 and Hypothesis $H_0$1 have been addressed for Upper Campbell Reservoir. The SVM does accurately predict the elevation bands of *4 out of 5* of the riparian vegetation community boundaries around the shoreline of Upper Campbell Reservoir, but it failed to consistently predict the mean elevation of the lowest community boundary between Mudflat and Spearwort Lakeflat. Thus, H_01 is retained for all community boundaries except for the MF/SL boundary (Section 4.1).
2. Determine if any errors in model predictions will warrant a change in the predicted outcome of the WUP.	2. If the model is in error, is the magnitude of the error such that it would warrant a change in the predicted outcome of the WUP?	*No associated hypothesis.*	Management Question 2 has been answered for Upper Campbell Reservoir. The SVM only failed to predict the elevation of the lowest vegetation community boundary (SL/MF). Specifically, the SVM overestimated the elevation of the Mudflat Community, which generally has lower wildlife values than the Spearwort Lakeflat community. Thus, this error indicates a benefit to wildlife, relative to the predictions that were made during WUP development. Based on this, we do not expect it is necessary to re-evaluate the predicted outcome of the WUP for wildlife values (Section 4.2).
3. Determine if changes to the modelling approach could improve its accuracy.	3. Are there changes to the modelling approach that could improve its accuracy for implementation in future WUP reviews?	*No associated hypothesis.*	Management Question 3 has been answered for Upper Campbell Reservoir. Changes to the modelling approach could be made to improve its accuracy for Upper Campbell Reservoir. These include: accounting for differences among vegetation communities in the sensitivity to the timing and duration of inundation, as well as considering how to incorporate the influence of factors other than slope that affect vegetation growth (e.g., fetch, aspect; Section 4.3)

4. Determine whether the riparian plant ecosystems in question require a gradient less than 15% to perpetuate.	4. Is it reasonable to expect that most riparian plant ecosystems require shoreline slopes to have a gradient less than 15% to perpetuate? If this is not reasonable, what is the shoreline slope gradient that is required for plant ecosystem persistence?	$H_0$2. The likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient.	Management Question 4 and Hypothesis $H_0$2 have been addressed for Upper Campbell Reservoir. Riparian vegetation community cover decreases with an increase in slope; therefore, *H_02* is rejected. Further, shoreline is more likely to be unvegetated than vegetated on slopes greater than 15%. Based on this, 15% may be considered a reasonable threshold, although the critical slope at which a vegetation community is more likely to be vegetated than unvegetated on average (based on a 30% cover criterion) varies among vegetation community types. The critical slopes identified for the Spearwort Lakeflat and Hairgrass – Water sedge communities were 15%, while the critical slopes were 18–40% for Sitka willow communities. These community-specific critical slopes were therefore used to convert SVM predictions to areal estimates (Section 4.4).
5. Determine whether the distribution of riparian plant ecosystems have changed following implementation of the WUP and determine whether the change is attributed to the WUP.	5. Has the distribution of riparian plant ecosystems changed following implementation of the WUP and if so, can the change be attributed to the WUP operation?	$H_0$3. Plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation.	Management Question 5 and $H_0$3 have been answered for Upper Campbell Reservoir. The distribution of riparian vegetation communities has changed following implementation of the WUP and the change can be attributed to WUP operations. Therefore, $H_0$3 is rejected (Section 4.5). The elevation boundaries of riparian vegetation communities have migrated downwards in response to the change in reservoir operations, resulting in an overall increase in the area occupied by riparian vegetation communities, as well as an increase in the area occupied by each individual community. Therefore, the WUP is expected to have increased wildlife habitat in Upper Campbell Reservoir, including for obligate and facultative aquatic wildlife.

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1. INTRODUCTION

1.1. Background

BC Hydro engages in water use planning to provide a balance between competing uses of water that include fish and wildlife, recreation, and power generation. Water Use Plans (WUPs) were developed for most of BC Hydro's hydroelectric facilities through a consultative process involving local stakeholders, government agencies, First Nations and other interested parties. Under the provision of the BC *Water Act* (2006), the Comptroller of Water Rights issued WUP related Orders that prescribed operational requirements that balance power production with other water uses. The Orders also direct BC Hydro to undertake specific actions including multi-year environmental monitoring studies. The objectives of these monitoring studies are to address outstanding uncertainties in the years following the implementation of a WUP.

The JHTMON-10 program examines potential changes to shoreline riparian vegetation communities as a consequence of water management. Riparian habitats are the interface between aquatic and terrestrial ecosystems (MWLAP 2006). These habitats provide important ecological functions (Richardson *et al.* 2005, Hoover *et al.* 2006) and are typically associated with more valuable wildlife habitat than adjacent upland areas as they are productive and structurally diverse (DFO and MELP 1998, Richardson 2003, MWLAP 2004). Moreover, riparian habitats provide critical migratory, feeding and breeding habitats for amphibians, birds, and other terrestrial wildlife. The focus of this monitoring program is on the potential for reservoir operations to directly and indirectly affect riparian habitats through the alteration of water levels. Excess or insufficient water levels can adversely affect vegetation community composition (Wilcox and Meeker 1991). The effects may differ depending on the temporal nature of the fluctuations and the floristic composition (Riis and Hawes 2002, Nilsson and Keddy 1988, van Eck *et al.* 2006). In addition, riparian communities can be affected by erosion following drawdown and surcharge of reservoirs on soils and establishing communities (BC Hydro 2013).

During development of the WUP, a lacustrine shoreline vegetation model (SVM) was used to predict how shoreline plant ecosystems change in response to operational changes in the Campbell River hydroelectric system (Bruce 2002). Changes in the aerial extent and location of shoreline plant communities predicted by the model were used to assess effects to wildlife. Based on the proposed operational changes, the potential effects on wildlife were predicted to be either benign or positive. Although the Wildlife Technical Committee (WTC) considered the SVM to be technically sound, uncertainty remained as the model was largely untested. Consequently, the WTC accepted the results of the model under the provision that the *Upper and Lower Campbell Lake Reservoirs Shoreline Vegetation Model Validation* monitoring program (JHTMON-10) be executed to validate the model and confirm assumptions that were made (BC Hydro 2013).

The JHTMON-10 program commenced in 2014 (Year 1), when data collection was undertaken in three study waterbodies: Upper Campbell Lake Reservoir (Upper Campbell Reservoir; comprised of

Upper Campbell Lake and Buttle Lake), Lower Campbell Lake Reservoir (Lower Campbell Reservoir) and Brewster Lake (a 'small' diversion lake). Year 1 data collection consisted of sampling shoreline vegetation communities, conducting air photo interpretation, and obtaining water level data, either from BC Hydro's existing gauges in the reservoirs, or from a gauge that was installed in Brewster Lake during Year 1 as part of JHTMON-10. Year 1 data analysis included validating the SVM for Upper Campbell Reservoir and quantifying the relationships between slope and vegetation community cover for all three waterbodies. Year 1 results are described in detail in Ballin *et al.* (2015), which also includes a literature review of the effects of water level fluctuations on lacustrine and riparian plant communities. In summary, the Year 1 analysis showed that SVM predictions for Upper Campbell Reservoir satisfied the two predefined tests of model accuracy, including that the modelled and measured boundary elevations were not significantly different and the mean model error was not significantly different from zero. However, when community boundaries were analyzed separately, there were statistically significant differences between SVM predictions and field measurements for some community boundaries, indicating that model error was higher for some vegetation communities than others. Slope analysis showed that unvegetated areas were more abundant on slopes >15% around Upper Campbell Reservoir; however, data from Lower Campbell Reservoir and Brewster Lake demonstrated only weak and inconclusive relationships between slope and vegetation cover.

During Year 2 to Year 4 of JHTMON-10, water level monitoring and associated gauge maintenance were undertaken on Brewster Lake. Details of this work are presented in annual memos (e.g., Marriner and Wright 2015). This gauge was removed in Year 5 and data will be provided separately to BC Hydro.

In 2016, the JHTMON-10 TOR was revised (BC Hydro 2016) to focus the study only on Upper Campbell Reservoir and the second validation assessment was rescheduled from Year 10 to Year 5, prior to the end of WUP review period. This TOR revision reflected uncertainty and risk identified in Year 1 related to the study design and the ability to answer the management questions. The Year 5 Workplan was subsequently revised to include data collection and analysis activities necessary to address the management questions, based on data collection in the Upper Campbell Reservoir. Accordingly, this Year 5 Annual Report describes all work completed in Year 5 associated with the revised TOR to address the study management questions. Currently, no further work is scheduled to be completed as part of JHTMON-10.

1.2. Study Area

In Year 5, the JHTMON-10 program focused on Upper Campbell Reservoir (Map 1). The reservoir is located approximately 33 km west of the City of Campbell River and is bounded by Strathcona Provincial Park, private managed forest lands, and other private land.

Upper Campbell Reservoir comprises the largest and most southern and western component of the Campbell River hydroelectric system. The largest tributaries are the Thelwood River, entering the system at the south end of Buttle Lake, and the Elk River which enters the west side of Upper Campbell Reservoir.

The Upper Campbell Reservoir is impounded by the Strathcona Dam. The dam also provides primary flow regulation for the Ladore and John Hart dams downstream. The Strathcona Dam was constructed between 1955 and 1958, with a second generating unit installed in 1968. The reservoir's historic operational water elevation has been between 210.0 m and 221.0 m. The current maximum and minimum operating levels are 212.0 m and 220.5 m respectively (as measured at Strathcona Dam), although the "preferred operating zone" varies throughout the year, with higher elevation preferred zone in the summer (217.0–220.5 m) (BC Hydro 2012).

The study area falls within the Very Dry Maritime Coastal Western Hemlock biogeoclimatic subzone western variant (CWHxm2). The CWHxm2 occurs at lower elevations (up to 700 m) on the east side of Vancouver Island and is characterised by warm, dry summers, and moist, mild winters with little snowfall (Green and Klinka 1994). Vegetation growth is constrained by water deficits over the long growing season. Zonal sites are dominated by Douglas-fir (*Pseudotsuga menzeisii*), with western hemlock (*Tsuga heterophylla*) and some western redcedar (*Thuja plicata*). Dominant understory species include salal (*Gaultheria shallon*), dull Oregon-grape (*Mahonia nervosa*), red huckleberry (*Vaccinium parviflorum*), step moss (*Hylocomium splendens*) and Oregon beaked moss (*Kindbergia oregana*).

Project Overview

Legend

Dam

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 2 4 6 8 10 12 km

Scale: 1:300,000

NO.	DATE	REVISION	BY
1		10/02/2015 1230_JHT_ProjectOverview_2015Feb10_LB	CGA
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Date Saved: 10/02/2015

Coordinate System: NAD 1983 UTM Zone 10N

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1.3. Vegetation Communities

Riparian vegetation communities on large lakes in the Campbell River system mostly occur on alluvial fans and other floodplains because of their low gradients, development of fine sediments and persistence of soils. Whereas on steeper shorelines around the reservoir, soils that were present prior to reservoir operations are more likely to have washed away. In general, the shoreline of Upper Campbell Reservoir where alluvial fans/ floodplains are absent, is occupied by steep gradient beaches with a substrate dominated by large gravels to steep rock bluffs. McLennan and Veenstra (2001) described six vegetation communities that occur along the shoreline of large lakes (Table 2). Four of these communities are described briefly below, including Spearwort Lakeflat, Hairgrass – Water sedge, short Sitka willow – Water sedge and tall Sitka willow – Water sedge. Lake Mudflat, which occurs below the Spearwort Lakeflat, is not described because it was most often under water during sampling, no lower boundary is defined in field measurements or the SVM, and therefore its extent could not be accurately compared between years or between modelled and measured data. Upland Forest is also not described because it is not directly dependant on the hydrologic regime of the reservoir, but rather is infilling the elevations that are no longer regularly flooded under the current WUP. Similar to Lake Mudflat, the upper boundary of the Upland Forest is not defined in field measurements or the SVM, thus Upland Forest can be used to define the boundary with Sitka willow -Water sedge but cannot be used to compare modelled area to measured area. Furthermore, Upland Forest itself is not a community, but rather encompasses the various Upland Forest communities that are present around the reservoir, as mapped with air photo interpretation (Section 2.2.3). Another 'type' that is mentioned throughout the report is 'unvegetated' shoreline. These are areas where there is sparse (<10%), to no vegetation cover.

Spearwort Lakeflat (SL)

The Spearwort Lakeflat community is frequently located on the lower elevations fluvial lakeflats and other low gradient shoreline below the Hairgrass – Water sedge community, as observed at 14 of the 16 transects in the Upper Campbell Reservoir. In lower gradient areas the community typically has a high coverage of vegetation dominated by lesser spearwort (*Rannunculus flammula*) and occurs on soils (Figure 1), whereas in steeper or more exposed areas vegetation is relatively sparse and grows between compacted gravels (Figure 2). In some areas, patches of sedges, grasses, and other herbs were present in clumps at the upper extent of the community. In other areas, patches of short (i.e., <0.5 m) willow and/or black cottonwood were dispersed amongst the community at various elevations (Figure 3). The lower extent of the community was often a mosaic of lesser spearwort, and other emergent aquatic vegetation, and exposed sandy or mudflat substrate.

Figure 1. Spearwort Lakeflat at JHT-SVM02 on Buttle Lake September 11, 2018.

Figure 2. Sparse occurrence of Spearwort Lakeflat on exposed gravel beach of JHT-SVM37 on Buttle Lake, November 21, 2018.

Figure 3. Sparse occurrences of short shrubs and graminoids amongst Spearwort Lakeflat at JHT-SVM01 on Buttle Lake, September 11, 2018.

Hairgrass – Water Sedge (HS)

The Hairgrass – Water sedge community typically occurs along the shoreline of Upper Campbell Reservoir in the drawdown zone between the short Sitka willow – Water sedge community and the Spearwort Lakeflat community (Figure 4). On the Upper Campbell Reservoir, it was present along 12 of the 16 transects. The most extensive presence of HS appeared to be near the outflows of streams on alluvial fans, however the community also occurred in a narrow band along low gradient shorelines. Various herbs and grasses were often present at the upper extent of the community. Vigorous occurrences of the community were rare. Often invasive reed canary grass or St. Johns wort comprised some of the herbaceous vegetation in this community. In some areas the Hairgrass – Water sedge community was trending towards a sedge wetland (SW), as differentiated by a thick cover of diverse sedges growing on deep soils and an absence of hairgrass (Figure 5). HS was rarely present in the steeper drawdown zone surrounding most of the lake shoreline. In these areas, HS was replaced by unvegetated shoreline or a sparse HS (<10%) composed of sparsely occurring sedges and grasses on a gravel substrate.

Figure 4. Hairgrass - Water sedge community along a creek outflow at JHT-SVM36 on Upper Campbell Lake, September 13, 2018.

Figure 5. Hairgrass - Water sedge community that is trending towards a Sedge Wetland community at JHT-SVM06 on Buttle Lake, November 21, 2018.

Sitka willow – Water sedge (short and tall stages) (WSt. WSs)

The Sitka willow – Water sedge community typically occurs along the shoreline of larger lakes and provides a transition between the Hairgrass – Water sedge and Upland Forest communities. The community demonstrates two distinctive structural stages: a short (3a) stage (Figure 6) and a tall (3b) stage (Figure 7). The taller stage consistently occupies a higher elevation band than the shorter stage due to inundation regime. The short (3a) structure of this community experiences more frequent and lengthy inundation lending to more suppressed growth.

Vegetation was typically dominated by Sitka willow with components of other willows such as Pacific willow (*Salix lasiandra*), as well as black cottonwood (*Populus trichocarpa*). Red alder (*Alnus rubra*) was sometimes present at the upper extent of the community and sweet gale (*Myrica gale*) was often present in calmer backwaters on deeper soils where the below community was closer to Sedge Wetland than HS. In some drier sites, especially on Upper Campbell Lake, where gravels dominated the substrate, Scotch broom (*Cytisus scoparius*) composed a substantial portion of the shrub layer (Figure 8). The understory vegetation was typically composed of sedges and exotic and native grasses. The upper elevations, where inundation is less frequent, included of a variety of herbs that ranged from drought tolerant strawberry (*Fragaria virginiana*), and trailing blackberry (*Rubus ursinus*), to invasive St. John's wort (*Hyperacum perforatum*) and oxeye daisy (*Leucanthemum vulgare*).

Many of the areas that had been occupied by the tall WS community during previous baseline studies conducted when the communities were regularly flooded (McLennan and Veenstra 2001) were now transitioning to Upland Forest types, and were occupied by approximately 18-year-old red alder or coniferous species. On wetter sites, where alder is dominant, the willow was overtopped by alder and had mostly died off. On drier sites, where conifers are dominant, the tree species were just starting to surpass willows in height (Figure 9).

Figure 6. Short Sitka willow - Water sedge community at JHT-SVM07 on Upper Campbell Lake, September 12, 2018.

Figure 7. Tall Sitka willow - Water sedge community at JHT-SVM04 on Buttle Lake, September 13, 2018.

Figure 8. **Tall Sitka willow – Water sedge community at JHT – SVM09 on Upper Campbell Lake that has a high component of invasive Scotch broom. Note photograph is taken from edge of Upland Forest.**

Figure 9. **Upland Forest community that transitioned from Tall Sitka willow - Water sedge to conifer forest between Year 1 (left) and Year 5 (right), observed at JHT-SVM02 on Buttle Lake, on January 15, 2015 and again on September 11, 2018.**

1.4. Shoreline Vegetation Model

During WUP planning, the SVM was developed to predict the relationship between water levels and the distribution of riparian shoreline vegetation in the Campbell River watershed (BC Hydro 2013, Bruce 2002). This information was used to evaluate potential impacts to wildlife that use shoreline and riparian habitats.

The SVM relates historic water levels (1984 to 2000 inclusive) (Figure 10) to the mean elevation boundaries of the six main vegetation communities that were present on the shore of Upper Campbell Reservoir during surveys conducted in 2001 (McLennan and Veenstra 2001) (Table 2). From lowest to highest elevations, the six dominant vegetation communities include Lake Mudflat (MF; not specifically vegetated), Spearwort Lakeflat (SL); Hairgrass – Water sedge (HS), short Sitka willow – Water sedge (WSs), tall Sitka willow – Water sedge (WSt) and Upland Forest (UF; not necessarily a riparian community, but encompass the variety of forests that may encroach upon the upper drawdown zone upon lowering of water levels). Based on historical water levels, the SVM is designed to predict the elevation boundaries between these six vegetation communities that occur around Upper Campbell Reservoir (Figure 11).

The cumulative frequency distributions of the air exposure times of the six vegetation communities that were surveyed are illustrated in Figure 12. The figure is based on daily water level measured during 1984–2000 inclusive. The premise of the SVM is that each vegetation community is adapted to a specific regime of drying and wetting, as illustrated in Figure 12. If the wetting/drying regime (i.e., annual water level fluctuations) changes, then the upper and lower elevation bounds of vegetation communities are presumed to shift so that the optimum wetting/drying regime for each community is restored. Thus, applying the model to predict the effects of alternative operational regimes involves: 1) constructing a water level record that corresponds to the new regime; 2) optimizing the boundary elevations of each vegetation community to match the cumulative exposure times in Figure 12.

Figure 10. **Daily water level in Upper Campbell Reservoir, 1984–2000.**

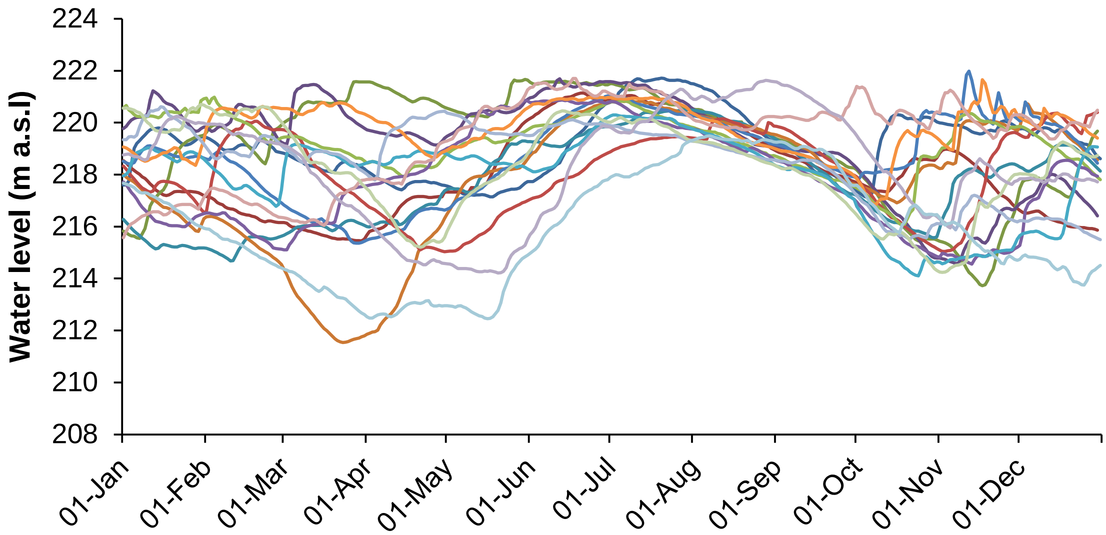


Table 2. **Summary of mean elevation boundaries of vegetation communities surveyed by McLennan and Veenstra (2001). Reproduced from Bruce (2002).**

No.	Vegetation Community Name	Map Code	Boundary Elevation (m)	
			Lower	Upper
1	Lake Mudflat	MF/MFl[1]	211.5	218
2	Spearwort Lakeflat	SL	218	219.2
3	Hairgrass – Water sedge	HS	219.2	219.8
4	Sitka willow - Water sedge (short)	WS_s	219.8	220.5
5	Sitka willow - Water sedge (tall)	WS_t	220.5	221.2
6	Upland Forest	UF	221.2	222

[1]The Lake Mudflat community is denoted as both MF and MFl throughout this document reflective of background sources.

Figure 11. Conceptual diagram of how vegetation communities are represented in the Shoreline Vegetation Model.

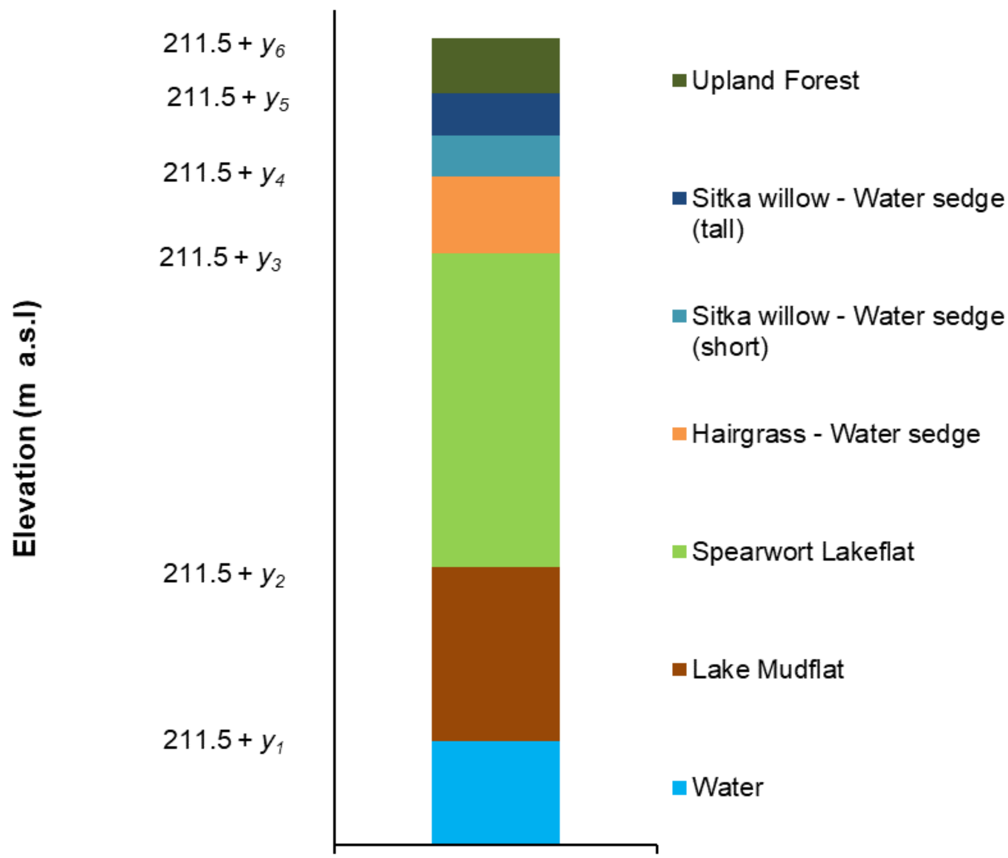
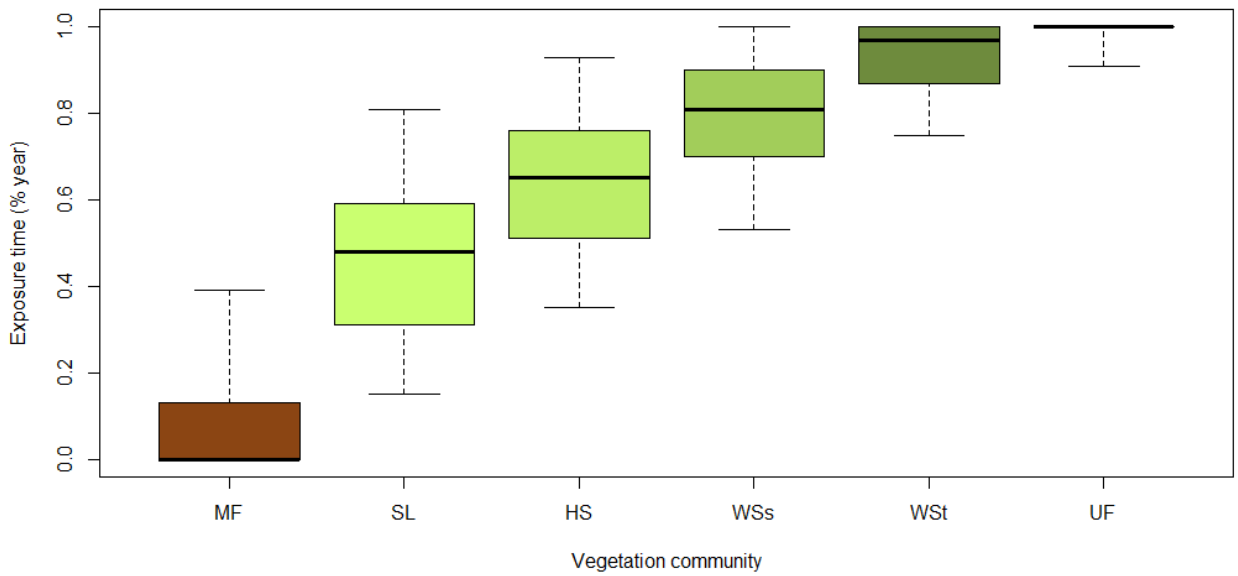


Figure 12. Summary of the air exposure times of the six vegetation communities included in the Shoreline Vegetation Model (Bruce 2002). Thick lines denote the median, boxes denote the interquartile range and whiskers denote the 5th and 95th percentiles.



The SVM is based on several key assumptions:

1. Variability in historic lake water level is the dominant control on the vertical distribution of shoreline vegetation.
2. Vegetation communities surveyed in 2001 by McLennan and Veenstra (2001) were in equilibrium with the water level regime.
3. The shoreline flora of Upper Campbell Reservoir comprises the six main vegetation communities.
4. All six vegetation communities are present along the shoreline, aligned in the same sequence with respect to elevation (see Figure 11).
5. The limits of tolerance to inundation for each vegetation community can be well quantified using water level data for the same duration; e.g., the duration of water level data used to predict the elevation of Upland Forest is the same as that used to predict the elevation of the Hairgrass – Water sedge community.
6. Each vegetation community is assumed to be sensitive to the duration but not the *timing* of inundation. Thus, for each vegetation community, there is equal sensitivity to water level fluctuations in all prior years that are included in the model, e.g., each community that was surveyed in 2001 is assumed to be equally sensitive to changes in water level that occurred

both in the previous year (2000) and 17 years prior to the survey (1984). Similarly, individual communities are assumed to be equally sensitive to inundation, regardless of the season during which it occurs; e.g., the model assumes that inundation during summer has the same effect as inundation during the winter.

7. The variability in the elevation of a vegetation community around each waterbody is minor relative to the variability in the elevations of different vegetation communities at a single site.

Thus, as with all environmental models, the SVM involves a range of assumptions that simplify natural processes. The appropriateness of these assumptions can be questioned; however, the starting point for JHTMON-10 is to evaluate whether the SVM yields satisfactory predictions *despite* these assumptions, rather than to investigate alternative approaches.

1.5. Management Questions and Hypotheses

The JHTMON-10 program aims to address the following five management questions:

1. Does the lacustrine shoreline vegetation model accurately predict the reservoir elevation bands that bound the predefined plant community types?
2. If the model is in error, is the magnitude of the error such that it would warrant a change in the predicted outcome of the WUP?
3. Are there changes to the modelling approach that could improve its accuracy for implementation in future WUP reviews?
4. Is it reasonable to expect that most riparian plant ecosystems require shoreline slopes to have a gradient less than 15% to perpetuate? If this is not reasonable, what is the shoreline slope gradient that is required for plant ecosystem persistence? [Note that the second part of this question was added following the TOR amendment (BC Hydro 2016)].
5. Has the distribution of riparian plant ecosystems changed following implementation of the WUP and if so, can the change be attributed to the WUP operation?

To address these management questions, the three alternate hypotheses that are listed below will be tested. A brief overview is provided of how each hypothesis will be tested in this report, with a summary presented in Table 3 of how each study component supports the management questions and hypotheses.

$H_0$1: Measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model.

This hypothesis will be tested based on data collected along the shoreline of Upper Campbell Reservoir. Failure to reject this hypothesis would suggest the SVM accurately predicts vegetation community boundary elevation bands. Rejection of this hypothesis will lead to an evaluation of model error; first, to determine whether the error is large enough to significantly affect the deliberations and conclusions of the WTC (Management Question 2); and second, to identify shortcomings in the modelling process that could potentially be addressed to improve future model accuracy (Management Question 3).

$H_0$2: The likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient.

The hypothesis will be tested in for vegetation communities around the shoreline of Upper Campbell Reservoir. The relationship will be examined between shoreline gradient and vegetation cover for individual vegetation community, as well as for groups of communities within the drawdown zone. During the WUP, the WTC assumed that a 15% gradient formed a reasonable threshold for plant growth and therefore the analysis will focus on examining whether this assumption is valid (Management Question 4). This will involve analysis of the slopes observed in vegetation transects, as well as geostatistical analysis of plant ecosystem polygons overlaid on a digital elevation model (DEM).

$H_0$3: Plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation.

This hypothesis will be tested in for vegetation communities around the shoreline of Upper Campbell Reservoir. If the SVM proves valid (i.e., $H_0$1 is accepted), then it can be inferred that observed changes (i.e., $H_0$3 is rejected) are likely due to WUP operations (Management Question 5). Addressing this hypothesis will involve extrapolating the model results over the DEM and calculating the aerial extent of each vegetation community. Vegetation community distributions 'before' and 'after' implementation of the WUP will be assessed.

2. METHODS

2.1. Overview

The JHTMON-10 study involves multiple study components, including field data collection and analysis. Table 3 provides a summary of how each study component supports the management questions and hypotheses listed in Section 1.3. Further details about the rationale for each component are provided in the descriptions of the methods for individual study components in the sections below.

Table 3. Summary of how study components contribute to addressing the JHTMON-10 management questions and hypotheses.

Phase	Study Component	Management Question					Hypothesis		
		1	2	3	4	5	1	2	3
Data collection	Vegetation Community	x	x	x	x	x	x	x	x
	Water Elevation	x	x	x		x	x		x
	Air Photo Interpretation				x	x		x	x
Analysis	SVM Validation	x	x	x		x	x		x
	Vegetation Community Gradient				x	x		x	x

2.2. Data Capture

2.2.1. Vegetation Communities

The boundaries of the six shoreline vegetation communities, characteristic of large reservoirs that were recognized by McLennan and Veenstra (2001) (Section 1.3), were identified along transects located around Upper Campbell Reservoir. Sixteen transects were surveyed around the Upper Campbell Reservoir in Year 5, including 10 transects that replicated Year 1 transects (eight of which were located near 2001 transects) and seven new transects (Map 2).

The purpose of collecting vegetation community transect data is twofold. Firstly, the data provide measured elevations of lacustrine riparian vegetation communities that can be used to test the SVM elevation predictions ($H_0$1). Secondly, the data provides gradients for each vegetation community that can be used to analyse the relationship between slope and vegetation ($H_0$2). The results of these two hypotheses are instrumental for detecting if the vegetation community distribution has changed following implementation of the WUP ($H_0$3).

Vegetation community transect data were collected at Upper Campbell Reservoir during two sampling periods: September 10-13 and November 21, 2018. Sample dates aimed to capture the growing season, while the water levels were predicted to be lowest, or occurred during suitable sampling conditions for each site (i.e., some transects required a specific water level so that the transect could be repeated). One transect that was surveyed in Year 1 (JHT-SVM31) was not repeated because the water level was too low to accurately calculate the community boundaries.

For established sites, transects started at benchmarks that were established in previous years. Transects were laid out along the same azimuth as in 2014 (where feasible, as the end of the transect must reach the reservoir) with Eslon transect tapes. New transects were laid out perpendicular to the water from the benchmark to the water's edge. The vegetation community boundaries were defined and temporarily marked, and the distance from the benchmark to each community boundary bisected by the tape recorded. The relative height of each vegetation community boundary was measured from the benchmark at the top of the transect to the water with a laser rotating Futtura level survey station. The data was related to real time water elevation readings obtained from Environment Canada (2018) from BC Hydro's online live transmission and reservoir data site for Strathcona Dam on the Upper Campbell Reservoir.

Additional field data were recorded at each site and along each transect to support data analysis and validation of the SVM, and to assist with recommendations (Table 4). Transect sites were surveyed by a terrestrial ecologist familiar with local vegetation communities and experienced in ecosystem identification in the CWHxm2. Vegetation community boundaries were identified based on transitions in vegetation species composition.

For new sites, benchmarks were installed at the base of trees, approximately 30 cm off the ground, as bedrock was infrequently located at the lower extent of adjacent Upland Forest types. New transects were located in areas that included several vegetation communities, a variety of aspects and slopes, and where transects could be effectively surveyed.

Additional slope and vegetation coverage data (both in %) were collected to increase the dataset for testing what slopes vegetation occurs on (Section 2.3.2.1). Data was opportunistically collected at another 83 locations around the reservoir, chosen to capture a range of slopes and vegetation coverage.

Table 4. Transect and vegetation community occurrence data attributes.

Data Level	Attribute	Description
Transect data	Site name	Transect label following format 'JHT-SVM00'
	Waterbody	Upper Campbell
	UTM start	UTM zone, easting and northing of benchmark
	UTM end	UTM zone, easting and northing of end point
	Benchmark #	Benchmark tag number
	Direction	Transect direction (and aspect of vegetation community) from benchmark to end in degrees from magnetic north
	Slope	Transect slope in percent
	Transect length	Total length of transect
	Photographs	Site overview photographs including benchmark and cardinal directions from top and bottom of transect
	Time at water	Time of gauge reading at which height measurements were taken at water's surface
	Water elevation	Water elevation at time of gauge reading
	Location	Description of site and benchmark location
	Invasive species	List of invasive species observed. Note that no additional effort was applied to identification or searching
	Wildlife	Wildlife observations or sign
	Comments	Other comments included anthropogenic impacts and disturbances
Community occurrence data	Occurrence	Sequential number of communities measured
	Dominant community	Dominant vegetation community
	Sub-dominant community	Subdominant or emergent vegetation community
	Start distance	Nearest vegetation community boundary along transect from benchmark
	End distance	Furthest vegetation community boundary along transect from benchmark
	Start height	Rod height at nearest boundary relative to survey station
	End height	Rod height at furthest boundary relative to survey station
	Start elevation	Elevation at nearest boundary as calculated from water surface elevation
	End elevation	Elevation at furthest boundary as calculated from water surface elevation
	Slope	Slope of individual vegetation community
	Community vigour	Vigour of the community ranked from 0-4 (0=dead, 1=poor, 2=fair, 3=good, 4=very good)[1]
	Vegetation % cover	Estimated percent cover of vegetation
	Photographs	Photo of community from the top down and the bottom up taken from 1 m beyond each boundary and capturing the ground cover starting at a meter from where the photographer was standing

[1] MOFR 2010, RIC 1998

2.2.2. Water Elevation

Water elevation data are required to quantify how waterbody water levels change through time, potentially influencing riparian and lacustrine vegetation distribution. In Year 1, a hydrometric gauging station was installed in Brewster Lake on June 30, 2014 to record lake water levels. Details of the installation are provided in Ballin *et al.* (2015) and subsequent maintenance is described in separate memorandums (e.g., Marriner and Wright 2015). In Year 5, the gauge was removed on October 18, 2018. Data were quality assured and compiled to be submitted separately to BC Hydro. No analysis of these data was undertaken in Year 5 because applying the SVM to Brewster Lake is not a component of JHTMON-10, following revisions to the TOR (BC Hydro 2016).

In Year 5, the existing timeseries of daily average water surface elevation in Upper Campbell Reservoir was updated by obtaining recent data from BC Hydro (Hofer, pers. comm. 2018). Mean daily water level data were used to apply the SVM (Section 2.3.1). Data were also compared among the SVM_{2001} (1984–2000), the SVM_{2014} (2001–2013), and SVM_{2018} (2001–2017) modelling periods to investigate variability in the hydrological regime. Inter-annual and seasonal variability in water levels were evaluated by preparing graphs of water level timeseries and frequency distributions. Further, the three periods were compared using a Kolmogorov-Smirnov goodness-of-fit test. This non-parametric test is used to establish whether the differences between two continuous distribution functions are statistically significantly different from zero (i.e., whether the distributions are the same; Zar 1999). The test is insensitive to differences in the length of the time series among the three periods. The purpose of this test was to examine whether the water level regime was different among the three modelling periods, with the test undertaken with paired timeseries. This test (and all other inferential statistical tests; see below) was undertaken using R (version 3.5.1; R Core Team 2018) with a significance criterion (α) of $p < 0.05$. The *ks.test* function in R was used to calculate the test statistic (D) and associated p-value.

2.2.3. Air Photo Interpretation

Air photo interpretation was conducted to map the aerial extent of the riparian vegetation communities identified by McLennan and Veenstra (2001) within the study area. Ecosystem mapping can be used to record site conditions and provide a framework for monitoring ecosystem response to management (RIC 1998). In JHTMON-10, the air photo interpreted vegetation community mapping is used to answer two of the monitors' hypotheses. Firstly, the average slope of each vegetation community is calculated for each mapped polygon to analyse the relationship between slope and the presence of shoreline vegetation communities ($H_0$2). Secondly, the area of the delineated vegetation community polygons is compared to the area of each vegetation community predicted by the model ($H_0$3), as a means to validate the model. The mapped extent of vegetation communities also provides a contingency method of monitoring and measuring change to riparian vegetation communities in the Campbell River watershed following implementation of the WUP ($H_0$3), as the aerial extent of the vegetation communities mapped each year can be compared.

In Year 5, the air photographs were collected on October 13, 2017 by BC Hydro's photogrammetry department. The flight was scheduled to capture low reservoir water levels and the vegetation growing season. The 1:10,000 digital photographs were orthorectified by BC Hydro and received by Ecofish on June 25th, 2018 (Hofer, pers. comm 2018a).

Riparian vegetation communities were delineated on the orthophotos (i.e., orthorectified air photos) in ArcMAP (v. 10.5) based on the *Standard for Terrestrial Ecosystem Mapping in British Columbia* (TEM) (RIC 1998, RIC 2000), as well as the 2014 polygon delineations, the methods employed in 2001 (McLennan and Veenstra), and other local mapping projects (Green 2009). Vegetation community polygons were delineated by a terrestrial ecologist experienced in air photo interpretation and familiar with ecosystems in the CWHxm2. For baseline mapping, polygons were delineated at a minimum scale of 1:10,000 (McLennan and Veenstra 2001). In Year 1, and again in Year 5, polygons were delineated at a maximum scale of 1:2,000, as this larger scale was deemed necessary to adequately visually observe and delineate vegetation communities on the photographs. Vegetation community delineation focused on those communities that have a soil moisture that is directly associated with the water level of the adjacent waterbody and that were previously identified by McLennan and Veenstra (2001). A sub-sample of Upland Forest types that occur just beyond the upper extent of the current drawdown zone was delineated. Upland Forest mapping focused on young (i.e., tall shrub or pole/ sapling structural stage) communities that occur at elevations that were within the drawdown zone under the previous WUP, and were thus supressed by historic reservoir operations, and now are above the current planned high water mark (BC Hydro 2012), and thus are able to transition from having a soil moisture regime associated with lake levels (often young alder or conifer stands) to Upland Forest types. In addition, a subset of polygons with no or sparse vegetation growth was delineated as a means for testing slope (H_02). Unvegetated polygon delineation aimed to capture a variety of slopes around the reservoir where the growing substrate would be affected by the hydrology of the reservoir (e.g., steep or flat gravel slopes but not bedrock cliffs).

Attributes that were recorded for each polygon delineated through air photo interpretation included: interpretation of decile (proportion of polygon dominated by a site series [vegetation community as defined by TEM]), TEM map code, and structural stage; calculation of mean slopes and areas for each polygon, assignments\ of mapsheet numbers and biogeoclimatic zones, and QA tracking (Table 5). Terrain and soil attributes were not included in the database; however, they are reflected in adoption of the bioterrain approach to polygon delineation.

Table 5. Attribute table associated with air photo interpretation polygon shapefile.

Attribute	Label	Description
FID	FID	Unique polygon identifier
Mapsheet	Mapsheet	1:20,000 mapsheet number
FC_ID	FC_ID	Same as FC_TAG
FC_TAG	FC_TAG	Mapsheet Number and Polygon Number used for unique identification of a polygon under format mapsheet#_polygon#
BEC zone	BEC_ZONE	The first-rank unit in the hierarchical Biogeoclimatic Ecosystem Classification (BGC) system[1]
BEC subzone	BEC_SUBZONE	The second-rank unit in the BGC system[1]
BEC variant	BEC_VRT	A third-rank unit in the BGC system occurring within particular subzones[1]
Decile 1	SDEC_1	The proportion of the polygon covered by Component 1, in deciles. Deciles in components 1–3 must total 10. Decile 1 must be greater or equal to Decile 2, which must be greater or equal to Decile 3
Site series 1	Site_S1	Site series map code[2]. Site series is a vegetation community association based on the sites ability to produce specific climax vegetation within a particular BGC Subzone or Variant
Structural stage 1	STRCT_S1	The structure of the vegetation cover at the time of survey, ranging from 1-7[1]
Structural stage modifier 1	STRCT_M1	Substage of structural stage used for stages 1-3[1]
Decile 2	SDEC_2	See above
Site series 2	Site_S2	See above
Structural stage 2	STRCT_S2	See above
Structural stage modifier 2	STRCT_M2	See above
Decile 3	SDEC_3	See above
Site series 3	Site_S3	See above
Structural stage 3	STRCT_S3	See above
Structural stage modifier 2	STRCT_M3	See above
Comments	COMMENTS	Additional pertinent information regarding the polygon, primarily from ground verification but also as observed on air photographs
Check	CHECK_	Field verified with visual check in 2018 (S), visual check in 2014 (V) or blank
Water body	Water_Body	Study lake or reservoir name
Mean slope	MEAN_SLP%	Average slope of the polygon as calculated from DEM
Area	Area_m	Area in m^2
Decile 1 area	Dec1_area	Relative area of decile 1 (Decile 1*.1*area)
Decile 2 area	Dec2_area	See above
Decile 3 area	Dec3_area	See above

[1]Described in *Field Manual for Describing Ecosystems in the Field* (MOFR 2010) and RIC 1998

[2] MOE 2006

2.2.3.1. Quality Assurance

Provincial standards for ecosystem mapping include guidelines for the percentage and density of polygons ground-truthed and the intensity of the actual ground-truthing, in relation to project scale and objectives. The appropriate survey intensity level depends on the use of the data. The guidelines list a range of survey intensity from 1-5, plus a reconnaissance level (RIC 1998 (i.e., RISC)). RISC methodology recommends that 5-14% of polygons are ground-truthed for ecosystem representation, forest productivity, local resource planning applications and wildlife capability (RIC 1998).

The accuracy and consistency of air photo interpretation was verified through four methods. Firstly, vegetation community polygons were initially delineated after initial field reconnaissance of Upper Campbell Reservoir, surveying the transects and collection of additional slope and vegetation information. Secondly, visual inspections, one of three types of field inspection methods defined in the TEM standards (RIC 1998), were conducted to verify line work and site series assignment. Visual inspections were conducted on a subset of the polygons at predetermined (e.g., targeted to resolve uncertainty, capture the range of vegetation communities at a variety of locations around the reservoir) and opportunistic locations (Table 5). Verification of each polygon included a visual check of the site series, structural stage, % cover of vegetation and slope, and accuracy of a portion of the line work. Visual checks were primarily conducted on October 18, 2018; however, verification did occur during the entire sampling period. Georeferenced photographs were collected with an iPad using GISPro, as well as a camera and GPS. In the office, the shapefile of mapped vegetation community polygons was updated based on the visual check data. In addition, the vegetation communities delineated along transects were used to verify the field-measured ground distance of vegetation community boundaries from the benchmark to those measured on the orthophoto. Thirdly, the delineation of polygon boundaries and site series assignment (vegetation community designation) were compared with those previously delineated by Ballin *et al.* (2015) and McLennan and Veenstra (2001). Fourth, the shapefile and database were reviewed by a GIS analyst for consistency and integrity of line work and attributes.

2.3. Data Analysis

2.3.1. SVM Validation

2.3.1.1. Overview

The SVM developed by Bruce (2002) predicts the elevation of shoreline vegetation communities based on water levels during preceding years, as described in Section 1.3, which also lists the main assumptions of the model. The validation of SVM predictions is central to the JHTMON-10 program and the outcomes of the validation will address the first two management questions (Section 1.3).

In Year 5, the validation involved applying the SVM that was developed using water level data for 1984–2000 (SVM_{2001}), and field measurements of vegetation communities from 2001, to predict the 2018 boundary elevations of vegetation communities along the shores of Upper Campbell Reservoir, based on water level data for 2001–2017 (SVM_{2018}). The SVM validation then involved undertaking quantitative analysis to answer the following questions, consistent with the approach outlined in the TOR (BC Hydro 2016):

1) Are the predicted elevation boundaries statistically significantly different from the field measurements? The answer to this question will directly address H_01 and partly address Management Question 1.
2) Are the values of model error (i.e., measured - predicted) significantly different from zero? If not, it may be concluded that the SVM predictions satisfy a pre-defined test of model accuracy identified in BC Hydro (2016). The answer to this question will partly address Management Question 2.
3) Are the SVM parameters that prescribe vegetation exposure times consistent between different versions of the SVM? If not, this would indicate that the vegetation communities are not in equilibrium with the hydrological regime and/or the SVM parameters do not accurately reflect the tolerances of vegetation communities to hydrological variability (discussed further in Section 2.3.1.3. The answer to this question will help to address Management Question 1 and 2 and may also inform Management Question 3.

2.3.1.2. Year 5 Elevation Boundary Predictions

Applying the SVM

In Year 5, the SVM was coded and applied using R (version 3.5.1). Results for the SVM_{2014} period were compared with results of the model version used for the Year 1 Annual Report (Ballin *et al.* 2015), which was implemented using MS Excel. Results matched when the same input (water level) data were used in both versions, demonstrating that the model was implemented correctly (although see note below regarding adjustments following a sensitivity analysis). This change in software was undertaken because analysis steps are more transparent in R, which also provides greater flexibility, meaning that it will be easier to apply and modify the SVM if later required.

Water level data for Upper Campbell Reservoir were analyzed to derive community-specific cumulative distribution functions (CDFs) of observed air exposure times for the period 1984–2000 (Figure 13), consistent with the original SVM_{2001} developed by Bruce (2002). CDFs were constructed using daily mean water level data for each year as shown in Figure 13. Each curve therefore quantifies the total proportion of the overall year that water level was at or below a given elevation, as shown in the context of the mean elevations of the boundaries between the vegetation communities surveyed by McLennan and Veenstra (2001). For example, the curve for year 2000 labelled in Figure 13 indicates that water levels were unusually low during this year, with the water level being lower than the mean upper boundary of Lake Mudflat (MF) for approximately 75% of the year.

The distributions of the values represented by the curves in Figure 13 for each of the six vegetation communities are summarized in Table 6, which forms the basis of the SVM (see Figure 12 for a graphical representation of Table 6). Thus, Table 6 was produced by first separating the daily calculated CDF values shown in Figure 13 into the six elevation bands. For each vegetation community, values were selected that were greater than (>) the lower elevation boundary and less than or equal (≤) to the upper elevation boundary. Vegetation community percentiles were then calculated from those CDF values that were between the lower and upper boundary of that community. For example, based on Figure 13, the CDF values for SL range from approximately 0.05 to 0.90, with most values clustered between 0.30 and 0.60. Thus, we would expect the 50th percentile of these CDF values to fall between 0.30 and 0.60 and, as expected, Table 6 confirms that the 50th percentile for SL is 0.48. Similarly, values for WSs range from approximately 0.40 to 1.00 (Figure 13), with a 50th percentile of 0.81 (Table 6). While Table 6 forms the basis of the SVM, it can be difficult to interpret. In broad terms, the table presents the percentiles of the annual CDF values that fall within each elevation band (as prescribed by Table 2). For example, the median value for the SL community (see fifth row) is 0.48, which is the median CDF value between the elevations of 218.0 m and 219.6 m (Table 2). This can be interpreted to mean that, on 50% of days, the SL community is dry for 48% or less of the time. Likewise, on 85% of days (eighth row), the SL community is dry for 66% of the time or less.

When applying the SVM, the lower trial boundary for the MF community was set as 210.00 m which is near the lower limit of the historical operating range of Upper Campbell Reservoir and approximate to the water level at which Buttle Lake becomes isolated (BC Hydro 2012). The upper boundary of the Upland Forest community was set as 224.00 m which is 2 m above the Critical Level specified in the WUP (BC Hydro 2012). These assumptions are based on those used in the original SVM (Bruce 2002), although the lowest trial elevation was reduced from 211.5 m to 210.0 m in response to lower water levels.

To check that the SVM had been configured correctly, Table 6 was compared with the corresponding table reported in Bruce (2002). In theory, the two tables should be identical, although there were some minor differences, as quantified in Table 7 which is shaded in proportion to the magnitude of the absolute differences. Most of the percentiles in the two tables match and all differences are minor. These differences likely reflect rounding errors and/or differences in the method used to divide the data between vegetation boundaries, e.g., whether a ‘<’ or a ‘≤’ operator was used to filter water levels less than an upper boundary. Although the data files originally used to prepare Bruce (2002) could not be sourced, the model’s author has reviewed the methods described here and confirmed that the procedure was followed correctly to determine percentiles (Bruce, pers. comm. 2015). Thus, the values presented in Table 6 of this report were used for the SVM validation to ensure that any minor differences in methods were applied consistently.

Model sensitivity analysis undertaken in Year 5 showed that the parameterization of the SVM (i.e., the CDFs in Table 6) was highly sensitive to gaps in the water level records. Specifically, data were missing for ten consecutive days in each of the 1985 and 1986 records. For the Year 1 Annual Report, these gaps were filled by inserting the mean elevation for consecutive days, as described in

Ballin *et al.* (2015). In Year 5, we investigated this issue further and determined that a superior match to Bruce (2002) could be achieved if null values (N/A) were instead inserted for the missing days in the records. Based on this, we used these amended records for all Year 5 analysis, including comparisons with SVM_{2014} results. That is, the "SVM_{2014}" results presented in this report are subtly different from those presented in the Year 1 Annual Report (Ballin *et al.* 2015). This modified approach does not affect the conclusions from Year 1, although it means that the performance of the SVM_{2014} reported here is slightly better than reported in Ballin *et al.* (2015).

Figure 13. Cumulative density functions of mean daily water levels for individual years during 1984–2000. Vertical dashed lines denote mean elevation boundaries between vegetation communities surveyed by McLennan and Veenstra (2001).

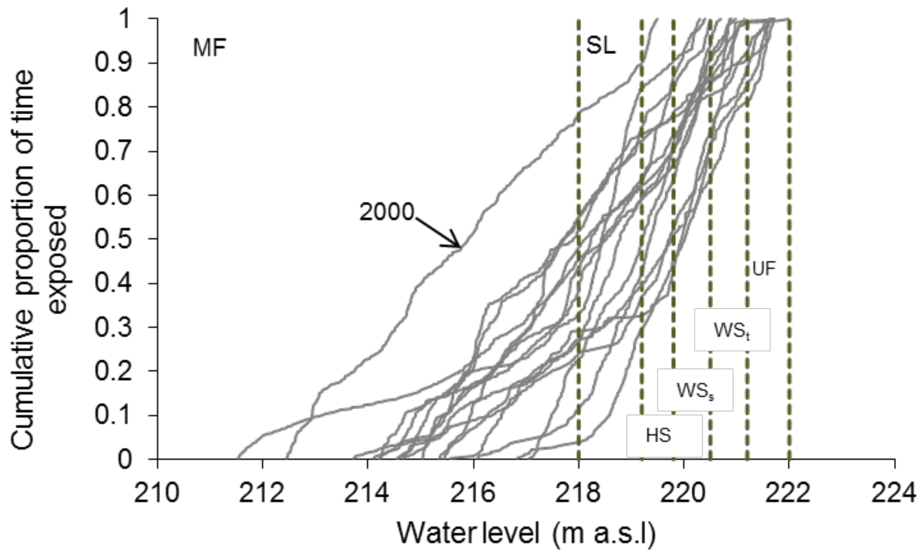


Figure 14. **The distribution of cumulative density function (CDF) values for an example community (HS) for elevations between 219.2 m and 219.8 m. The dashed vertical lines from left to right are the 5th, 15th, 25th, 40th, 50th, 60th, 75th, 85th, and 95th percentiles reported in Table 6.**

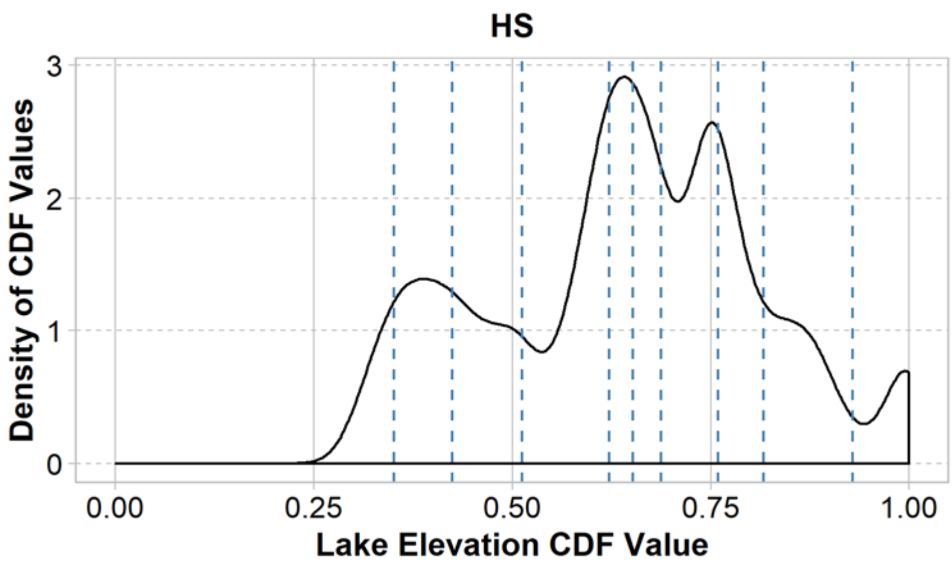


Table 6. **Percentile values of annual cumulative density functions of shoreline air exposure times for 1984–2001 (Figure 13).**

Percentile	Community Specific Percentile Values[1]					
	MF	SL	HS	WS_s	WS_t	UF
0.05	0.00	0.15	0.35	0.53	0.75	0.91
0.15	0.00	0.28	0.42	0.61	0.81	0.99
0.25	0.00	0.31	0.51	0.70	0.87	1.00
0.40	0.00	0.40	0.62	0.78	0.92	1.00
0.50	0.00	0.48	0.65	0.81	0.97	1.00
0.60	0.01	0.54	0.69	0.84	1.00	1.00
0.75	0.13	0.59	0.76	0.90	1.00	1.00
0.85	0.22	0.66	0.82	0.95	1.00	1.00
0.95	0.39	0.81	0.93	1.00	1.00	1.00

[1]Vegetation community codes are defined in Table 1.

Table 7. Absolute differences between values shown in Table 6 and the corresponding table presented in Bruce (2002). Shading denotes magnitude of absolute differences.

Percentile	Community Specific Percentile Values					
	MF	SL	HS	WS_s	WS_t	UF
0.05	0.00	0.01	0.01	0.03	0.01	0.01
0.15	0.00	0.01	0.00	0.01	0.00	0.01
0.25	0.00	0.01	0.00	0.00	0.01	0.01
0.4	0.00	0.00	0.00	0.00	0.01	0.00
0.5	0.00	0.00	0.00	0.00	0.00	0.00
0.6	0.05	0.00	0.00	0.00	0.00	0.00
0.75	0.05	0.01	0.00	0.00	0.00	0.00
0.85	0.03	0.00	0.01	0.00	0.00	0.00
0.95	0.03	0.00	0.03	0.00	0.00	0.00

Predicting Elevation Boundaries

The SVM was used to predict 2018 vegetation community boundary elevations for Upper Campbell Reservoir, based on water level data for 2001 to 2017 inclusive (SVM_{2018} predictions). Water elevation data for 2018, the year of vegetation sampling, was not included in the model to ensure consistency with the approach taken by Bruce (2002); i.e., the SVM_{2001} model was developed based on vegetation surveys conducted in 2001 but it was configured using a water elevation record that ceased during the previous year (2000). The duration of the water level record used for SVM_{2018} predictions (17 years from 2001 to 2017) matched the duration that was used to develop the original SVM_{2001} model
(17 years from 1984 to 2000).

A boundary search procedure, as outlined by Bruce (2002), was used to predict the five elevation boundaries between the six vegetation communities. The procedure involved an iterative process that sought to identify the elevation boundaries that provided the best match to the community-specific air exposure times presented in Table 6. The procedure was used to first predict the lowest elevation boundary between Mudflat and Spearwort Lakeflat. A starting elevation of 210 m was chosen, consistent with Bruce (2002). Trial elevations were then selected in increments of 0.01 m from the starting elevation. For each trial elevation, percentiles were calculated from the CDF values between the starting elevation, and the trial elevation. These percentiles were compared to the values in
Table 6, and a sum of square differences (SS_{MF}) was calculated for that trial elevation to quantify the error between the CDF values corresponding to the trial elevations, and the CDF values in

Table 6 that are assumed to reflect the "optimum" inundation regime for each community (see Section 1.3 for model assumptions). Specifically:

$$SS_i = \sum_{P=0.05}^{P=0.95} \left(P_{trial,P} - P_{measured,P}\right)^2$$

where SSi is the sum of squared differences for vegetation community i, $P_{trial,P}$ is the value of percentile P for the trial elevation, and $P_{measured,\ P}$ is the value of percentile P based on the SVM_{2001} predictions (Table 6). The process was then repeated over different trial elevations to identify the elevation (nearest 0.01 m) that corresponded to the minimum value of SS_{MF}. This elevation was defined as the upper boundary of the MF community and was thus set as the lower bound of the next community (SL). This process was repeated until all five boundaries were defined.

2.3.1.3. Model Error and Validation

Model error was calculated by subtracting SVM_{2018} predicted boundary elevations from elevations that were measured in the field. Mean error, absolute mean error and the square root of the mean of the squared error values (RMSE) were calculated separately for each boundary type and for all boundary types combined. Mean error provides a measure of model bias (i.e., whether predictions are systematically too high or too low), whereas absolute mean error and RMSE provide a measure of model accuracy, with RMSE placing greater weight on errors that are large rather than small.

A series of inferential statistical tests were then conducted to answer the three questions listed in Section 2.3.1.1. The questions are repeated below with descriptions of the associated tests that were undertaken. The tests included the three tests (or variants) that are described in BC Hydro (2016), in addition to one further test (one sample Wilcoxon Signed-Rank test) that was selected to provide further insight into model performance for individual vegetation communities.

1) Are the predicted elevation boundaries statistically significantly different from the field measurements?

First, a one-way analysis of variance (ANOVA) was conducted to test the null hypothesis that mean measured and modelled boundary elevations are the same. ANOVA was undertaken using the *aov* function in R to compare mean and measured elevations, with values pooled for all boundary types (e.g., MF and SL, SL and HS etc.).

Second, a non-parametric one sample Wilcoxon Signed-Rank test was used to test whether there were significant differences between modelled and measured boundary elevations for individual boundary types. The one sample Wilcoxon Signed-Rank test was undertaken using the *wilcox.test* function in R. This extends the first test by examining whether there are differences between SVM_{2018} predictions and measurements for individual vegetation communities, rather than only the pooled set of values. This test is a non-parametric equivalent of a t-test. A non-parametric test was used because the distributions of the measurements for some communities deviated from a normal distribution. The test was conducted separately for each community with no adjustment for multiple comparisons.

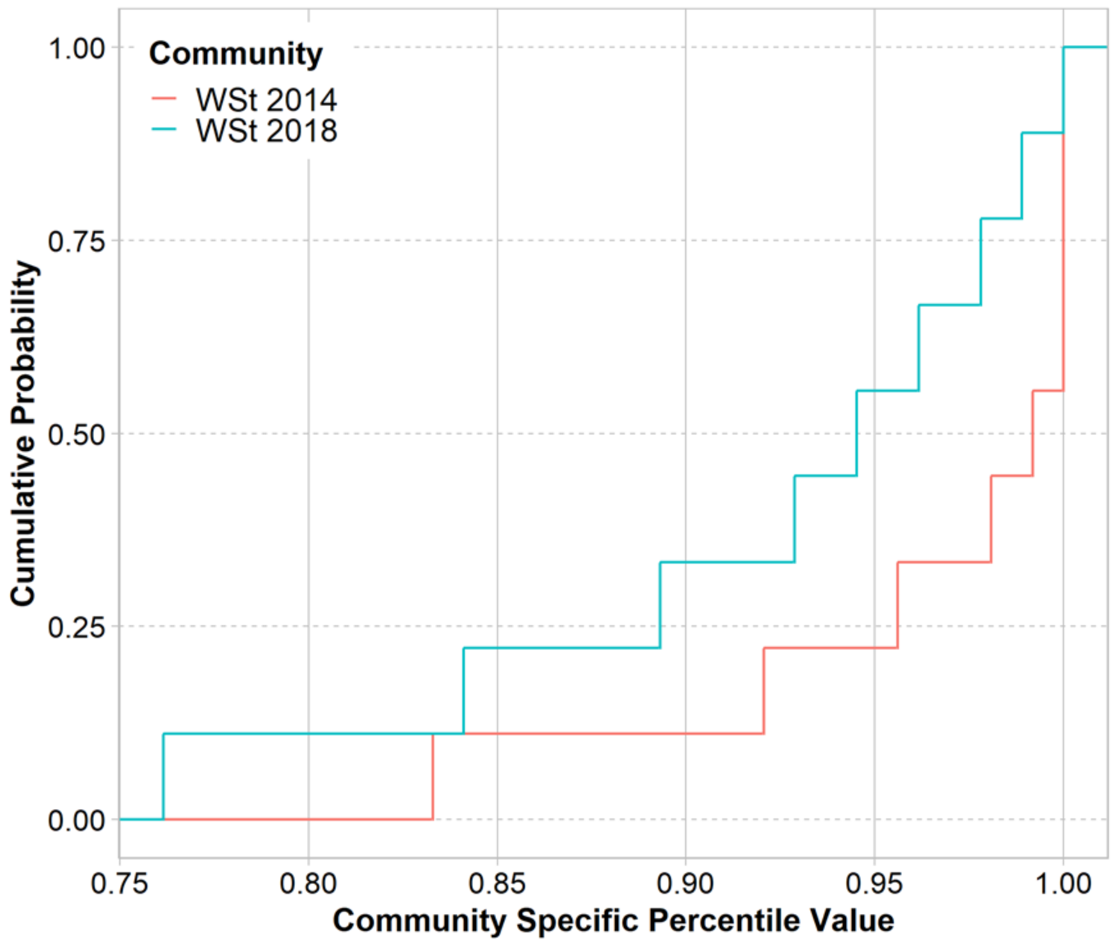
2) Are the values of model error (i.e., measured - predicted) significantly different from zero?

To test the null hypothesis that the mean error was not significantly different from zero, a t-test was conducted using the *t.test* function in R. A z-test has previously been suggested to test this hypothesis (BC Hydro 2016); however, a t-test was deemed more appropriate than because the sample was small and the variance of the population was unknown (Zar 1999), although the two tests are similar.

3) Are the SVM parameters that prescribe vegetation exposure times consistent between different versions of the SVM?

A premise of the SVM is that it has been developed (parameterized) to reflect differences in the tolerances of vegetation communities to water level fluctuations. An assumption of the SVM is that these tolerances remain constant and vegetation communities adapt to environmental change by colonizing habitats that best provide the hydrological conditions that each community is adapted to (Section 1.3). If this assumption is correct, then values of model parameters (i.e., the distributions presented in Table 6) should remain constant over time, even though the elevations of communities may change to reflect changes in hydrological regime. To test this assumption, the SVM parameters were reproduced separately based on Year 1 and Year 5 field data. This involved creating alternate versions of Table 6 that were based on measured community elevations in these two years and the preceding water levels (2001–2013 for SVM_{2014} and 2001–2017 for SVM_{2018}). A non-parametric Kolmogorov-Smirnov goodness-of-fit test was then used to compare the distributions for each vegetation community (i.e., the values in columns in Table 6) among the three periods (see Figure 15 for an example). The *ks.test* function in R was used to calculate the test statistic (D) and associated p-value.

Figure 15. **Comparison between 2014 (Year 1) and 2018 (Year 5) of how the WSt vegetation community was represented (parameterized) in the SVM. An assumption of the SVM is that the tolerance of a vegetation community to water level fluctuations is described correctly by the assigned distribution (represented by the coloured lines) and remains static over time. If this assumption is correct, then the two lines should not be statistically significantly different. A Kolmogorov-Smirnov test was used to test this assumption for each community, using data for three periods.**



2.3.2. Shoreline Gradient and Vegetation Presence

During development of the WUP (BC Hydro 2012), the SVM was applied with the assumption that shoreline gradients of less than 15% are required to allow soils to accumulate or persist in the operational zone, and support plant growth ($H_0$2). However, this assumption was never validated during WUP development, and therefore may be incorrect. This assumption was examined in Year 5 with three different lines of evidence that used the following three datasets: (1) Ground-based slope validation, which involved ground slope measurements collected in the field along vegetation community transects and at additional locations; (2) Slope analysis of mapped polygons, which included a contingency test of the average slope of each vegetation community polygon delineated by air photo interpretation that had 70% or more coverage of vegetated versus non-vegetated communities; and, (3) DEM resampling of the slopes of points (pixels) that were randomly sampled from the shoreline area mapped with air photo interpretation. The field measurements and two types of air photo measurements were considered separately in the analysis because each dataset has different attributes, which support complementary lines of evidence. Specifically, the transect data are collected at a fine scale and are entirely field verified and therefore are expected to be more accurate and precise than the desktop air photo analysis. However, transect data are collected at a relatively small number of locations where transects exist, and therefore may not be fully representative of conditions along the entire shoreline. By contrast, the air photo interpreted polygons are collected at a larger scale, are partially field verified, cover large areas of the reservoir shoreline and are therefore expected provide reasonable representation of conditions. The randomly sampled point data contains the same vegetation data as the air photo interpreted polygons and more detailed slope data. Due to the large number of slope pixels within the entire drawdown zone, this dataset enables a larger sample size, and also sampling of unvegetated areas that were not delineated as polygons.

The definition of 'vegetated' vs. 'unvegetated' was considered different for each of the three lines of evidence examined because of the different types of input data used and analysis conducted. These definitions are included with the description of each respective method.

2.3.2.1. Ground-based Slope Validation

Field data collected along 16 transects (Section 2.2.1) and at 83 additional locations (selected to increase sample size) were analysed to examine the relationship between the vegetation cover (%) and slope (%) of each community. For the vegetation transect line of evidence for testing slope assumptions, an area was considered vegetated if it had ≥30% cover of vegetation. The value of 30% vegetation cover was determined by professional judgement as adequate to provide a meaningful amount of wildlife habitat for all of the vegetation communities assessed (i.e., SL, HS, WSs, WSt, UF) and because it was considered sufficient to provide other riparian habitat functions, such as substrate stabilization, and creation of pockets where succession could potentially take place.

Slope and vegetation cover data were collected for each community along each transect (91 measurements) and at additional locations around the reservoir (83 measurements) over a range of slopes and vegetation covers during field studies (i.e., transect data collection and air photo interpretation data collection). Scatter plots of slope and cover were created for each shoreline vegetation community type. To test for a statistically significant relationship between shoreline slope and vegetation cover, linear regression analysis was applied to data for each community using the *lm* function in R. This quantified the average relationship between vegetation cover and slope, with associated 95% confidence intervals for predictions. Histograms of regression residuals were inspected to confirm that the values were approximately normally distributed.

A statistically significant negative relationship was observed between vegetation cover and slope for some communities, while no relationship was observed for others. For communities that exhibited a negative relationship, the regression model was used to estimate the average slope at which the average vegetation cover was <30%, which was the threshold used to determine unvegetated shoreline (US) (see below) This analysis was completed separately for individual vegetation communities and for all vegetation communities combined.

For communities that exhibited a statistically significant negative relationship between vegetation cover and slope, piecewise linear regression models were trialled to examine whether there was a robust breakpoint in the data that may indicate an appropriate slope threshold; i.e., models were trialled to examine whether there was a slope above which the negative slope of the relationship abruptly increased. Piecewise regression was undertaken using the *segmented* function in R.

2.3.2.2. Slope Analysis of Mapped Polygons

A DEM of the study area was generated in ArcMAP (version 10.5) from LiDAR data collected in October 2017 (Hofer, pers. comm. 2018a). The LiDAR ground point cloud data was interpolated into a DEM with a 0.5 m grid cell size. A slope raster was generated from the DEM and the zonal statistics tool was used to calculate the mean slope for each polygon delineated using air photo interpretation (Section 2.2.3).

For the slope analysis of mapped polygons, a TEM polygon was considered 'vegetated' if the sum of the vegetation community deciles equaled seven or higher, meaning that ≥70% of the delineated polygon was occupied by a vegetation community. Similarly, if a vegetation polygon was occupied by a decile of seven or higher (≥70%) of an unvegetated type then it was considered 'unvegetated'. These thresholds are the same that were used in the Year 1 report (Ballin *et al.* 2015). In only including polygons with ≥70% occupancy of vegetation communities, the analysis excludes polygons that are somewhat occupied by vegetation communities, for example half vegetated and half unvegetated, in order to arrive at a more definite distinction between vegetated and unvegetated types. It is also of value to note that a vegetation community polygon could comprise of a community occurrence with vegetation cover ranging from 10-100% (as sparsely vegetated polygons with cover under 10% generally had a portion attributed to an unvegetated type).

An independent two-group t-test was used to compare the average of mean slope values between polygons dominated by unvegetated shoreline and polygons dominated by vegetation communities.

In total, 291 vegetated and 86 unvegetated polygons were analysed, and 63 were omitted from the analyses.

ANOVA was used to examine whether there was a statistically significant difference in mean polygon slopes between vegetation community types. A posthoc Tukey HSD test (*TukeyHSD* function in R) was used to identify statistically significant differences between the average values of mean polygon slope for individual communities and for unvegetated shoreline. This test was restricted to community types that dominated three or more polygons. Test assumptions were checked by reviewing scatterplots (to confirm normality) and undertaking a Levene's test (to confirm homogeneity of variance). Data were subsequently $\log_{10}$-transformed to ensure that the assumptions were met.

As per the TOR (BC Hydro 2016), a two × two contingency table was constructed to explicitly test whether polygons with slopes >15% are significantly more likely to be unvegetated than polygons with slopes ≤15% ($H_0$2). A table was constructed that quantified the number of vegetated and unvegetated (US) polygons with mean slopes of ≤15% or >15%. A Fisher's exact test was then used to test the null hypothesis that the distribution of vegetated and unvegetated areas is independent of whether slope is greater or less than 15%. The Fisher's exact test was completed using the *fisher.test* function in R.

2.3.2.3. DEM Resampling

The DEM resampling analysis was completed so that the entire population of vegetated and unvegetated areas and slopes in the Upper Campbell Reservoir could be included. This approach minimizes any bias in site selection. All polygons and un-delineated unvegetated areas were included in the analysis, and needed to be assigned a value of 'vegetated' or 'unvegetated'. Thus, a pixel was considered 'vegetated' if the polygon within which it was nested was ≥50% occupied by a vegetation community (i.e., decile of 5 or higher), and 'unvegetated' if the polygon was occupied by <50% of an unvegetated type (i.e., decile of <5).

For this third line of evidence to examine the dependency of vegetation cover on slope, slope values were analysed for randomly selected subsamples of points in vegetated and unvegetated areas. The frequency distributions of these slope values were then compared between vegetated and unvegetated areas to characterise how slopes differ between the two groups. This analysis was undertaken as a complement to the analysis that is described above and prescribed in the TOR (BC Hydro 2016). The purpose of this analysis was to attempt to resolve the uncertainty in the analysis of the mean polygon slopes that is caused by the polygons being of unequal size – note that all polygons in a group (vegetated or unvegetated) contribute equally to the mean slope of that group, despite the fact that polygons vary considerably in area. In Year 1, this uncertainty was examined by calculating area-weighted slopes (Ballin *et al.* 2015); however, a resampling technique was selected as a superior approach in Year 5 because it should theoretically yield larger sub-samples of slope values that are fully representative of the topography of the reservoir shoreline.

Values of slope and vegetation cover were randomly sampled for points within the domain used for air photo analysis. A total of 7605 points were randomly selected at the pixel scale using the *spsample* function in *R* and ArcMap (v. 10.5). To ensure that samples were only collected from within the drawdown zone, and included communities for which a modelled and measured upper and lower bound exists, samples were only collected from elevations between 216.5 m and 220.5 m, which correspond to the lower elevation of measured Spearwort Lakeflat (below which most areas were unvegetated) and the upper elevation of the reservoir drawdown zone (based on the maximum operating level; BC Hydro 2012), respectively.

Point samples were then divided into two groups, vegetated (≥50% occupied by a vegetation community) and unvegetated (<50% cover by an unvegetated type). Frequency distributions of slope values for the two groups were plotted and a t-test was conducted to test for difference in mean slope between the two groups.

2.3.3. Validation of Modelling Approach with Air Photo Interpretation

One of the primary objectives of JHTMON-10 is to understand whether the shoreline vegetation community distribution has changed in relation to reservoir operations prescribed by the WUP ($H_0$3), and if so, whether changes are consistent with predictions from the SVM, which was used during WUP development ($H_0$1). The final step to verify the ability of the SVM to predict the areal extent of shoreline vegetation communities around the reservoir involved extrapolating the shoreline vegetation community elevations modelled by the SVM ($H_0$1) around the DEM of the reservoir to estimate the area of vegetation communities around the reservoir, in consideration of the maximum slope on which vegetation occurs ($H_0$2) (i.e., validation of the hectare estimation tool). The modelled areas were then compared to the areas mapped with air photo interpretation to quantify differences in total area presented.

2.3.3.1. SVM Validation with the Hectare Estimation Tool

A hectare estimation tool was used to estimate the aerial extent of each vegetation community within its respective elevation boundaries, as modeled by the SVM (Table 10) up to an elevation of 221.0 m (historic operating level), and on slopes below a specific threshold. Slope thresholds for each vegetation community were calculated using regression analysis of the transect data, as described in Section 2.3.2.1 above, and as presented in Section 3.2.2.1. Transect data were used to identify these slope thresholds (as opposed to analysis of aerial photographs) because they were considered more accurate measures of conditions present in individual vegetation communities because these data are collected at the finest scale and are entirely comprised of field measurements.

The hectare estimation tool was applied by extrapolating the SVM predicted vegetation community boundaries (see Section 3.2.1.1) onto the 0.5 m resolution DEM of the Upper Campbell Reservoir in ArcGIS, and calculating the area of each community present within the predicted boundary elevations at or below the critical slope, as per the example presented in the TOR (BC Hydro 2016):

$$\text{Total Vegetation Community Area (ha)} = \qquad (1)$$

$$\sum_{i=Lower\ elevation}^{Upper\ elevation}(Area_i|Slope_{<Critical}).$$

where $Area_i$ is the area between the boundary elevations predicted for the vegetation community using the SVM and $Slope_{<Critical}$ is the slope threshold (%) specific to the vegetation community that is assumed to differentiate vegetated and unvegetated areas.

The spatial output was a raster layer that spatially delineates the area occupied by each vegetation community. Some areas were omitted from the raster and thus the summary areas. These included mudflat (MF) because the lower boundary could not be defined, and smaller ponds and lakes that were assumed to not be directly connected to the reservoir. To QA the data, results were visually compared to orthophotos and air photo interpreted polygons (Section 3.1.3), and spatial summaries completed in R were analysed.

Year 5 areas measured with air photo interpretation were compared with modelled areas calculated with the SVM and hectare estimation tool. Modelled and measured areas of individual vegetation communities were plotted and linear regression was used to analyse the relationship between these. Regression statistics (p, r^2, standard error of the estimate) were quantified to evaluate model performance.

2.3.3.2. Temporal Change in Vegetation Communities

Changes in the areal extents of vegetation community types along the shoreline of Upper Campbell Reservoir were compared over time using images from baseline (pre-WUP) (2001; McLennan and Veenstra 2001), Year 1 (Ballin *et al.* 2015) and Year 5 (Hofer, pers. comm. 2018a) periods.

Although methodology was generally consistent between years, there is potential for some error due to slight differences in methods (e.g., images, methods, tools or observers). Baseline (2001) images were interpreted at 1:10,000 scale by one ecologist, whereas the Year 1 and Year 5 images were interpreted at a 1:2,000 scale by another ecologist. Some assumptions were also made to summarize the 2001 data. Several of the 2001 polygons were mapped as a community with a specific structure and other attributes (e.g., WS 3b or SL 1a) but were not assigned any decile (i.e., the attribute was blank), thus it was unknown what proportion of the polygons area should be assigned to the community. First the assumption that these polygons had 100% occupancy of the assigned community was tested. This yielded some unreasonably large areas, especially for SL, which had a mapped area of 1,484.3 ha under this method. Therefore, it is assumed that the blank deciles were assigned because the community only sparsely occurred (i.e., <10% cover) in these areas and thus it was determined that attributing a decile from 1-10 (as expected) was not warranted. In 2014 and 2018, these sparsely vegetated lakeflats were considered partially Unvegetated Shoreline and partially Spearwort Lakeflat, and relative areas (total area * decile of each community) assigned accordingly.

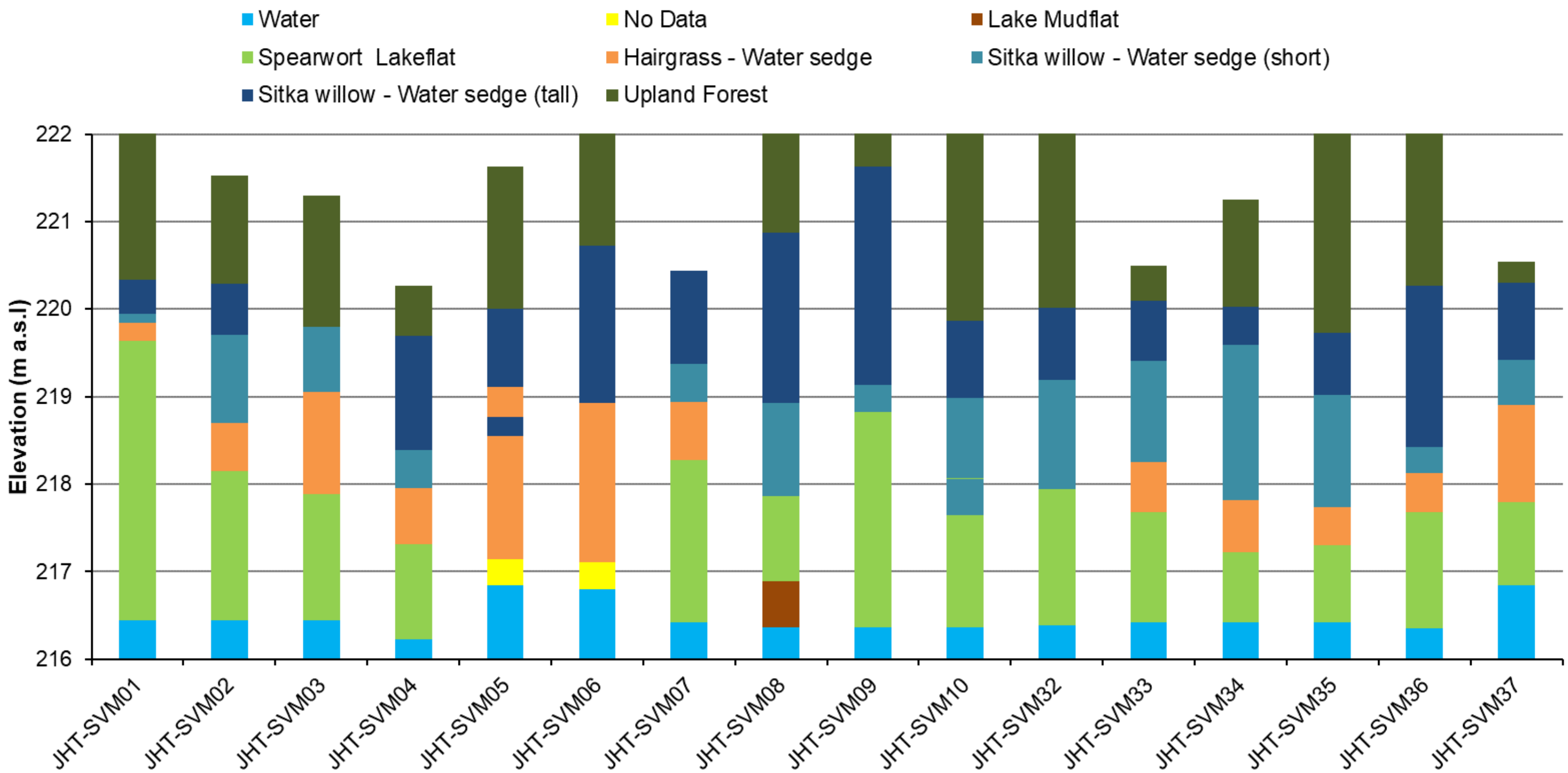
3. RESULTS

3.1. Data Capture

3.1.1. Vegetation Communities

The measured elevations of riparian vegetation communities on Upper Campbell Reservoir differed by transect but generally followed similar elevation distributions and the orders of community occurrences with some exceptions (Figure 16). Transects with no data are such because the transect tape would not reach the water under current reservoir conditions, therefore the transect elevations were measured from the benchmark established in Year 1.

Figure 16. Distribution of vegetation communities by elevation in the Upper Campbell Reservoir in 2018.



3.1.2. Water Elevation

From 2001 to 2017 (period used for SVM_{2018}), the mean daily water elevation (i.e., water level) in Upper Campbell Reservoir ranged from 209.51 m (April 2008) to 221.24 m (December 2014; Figure 17). In general, the annual minimum water elevation typically occurred during spring (~April to June), with a smaller decline often observed in late summer or early fall (September to November), although there was high variability among years in this seasonal pattern (Figure 18). On average, water elevations were lower during the SVM_{2018} period than the SVM_{2001} period 1984–2000 (Figure 19).

Results of the Kolmogorov-Smirnov goodness-of-fit test showed that the distribution of mean daily water levels is different between the periods of 1984 – 2000 (SVM_{2001}) and 2001 – 2017 (SVM_{2018}) ($D = 0.2536$; $p < 0.001$). However, the same test shows that water levels are not significantly different between the periods of 2001–2013 (SVM_{2014}) and 2001–2017 (SVM_{2018}). Frequency distributions for the three period support these results; e.g., Figure 20 shows that relatively high water-levels occurred more frequently prior to 2001 than since that time.

Figure 17. Daily time series of water elevation between 2001 and 2017. The mean elevation for 2001–2017 is marked with the dashed blue line; the mean elevation for 1984–2000 is marked with the dashed red line.

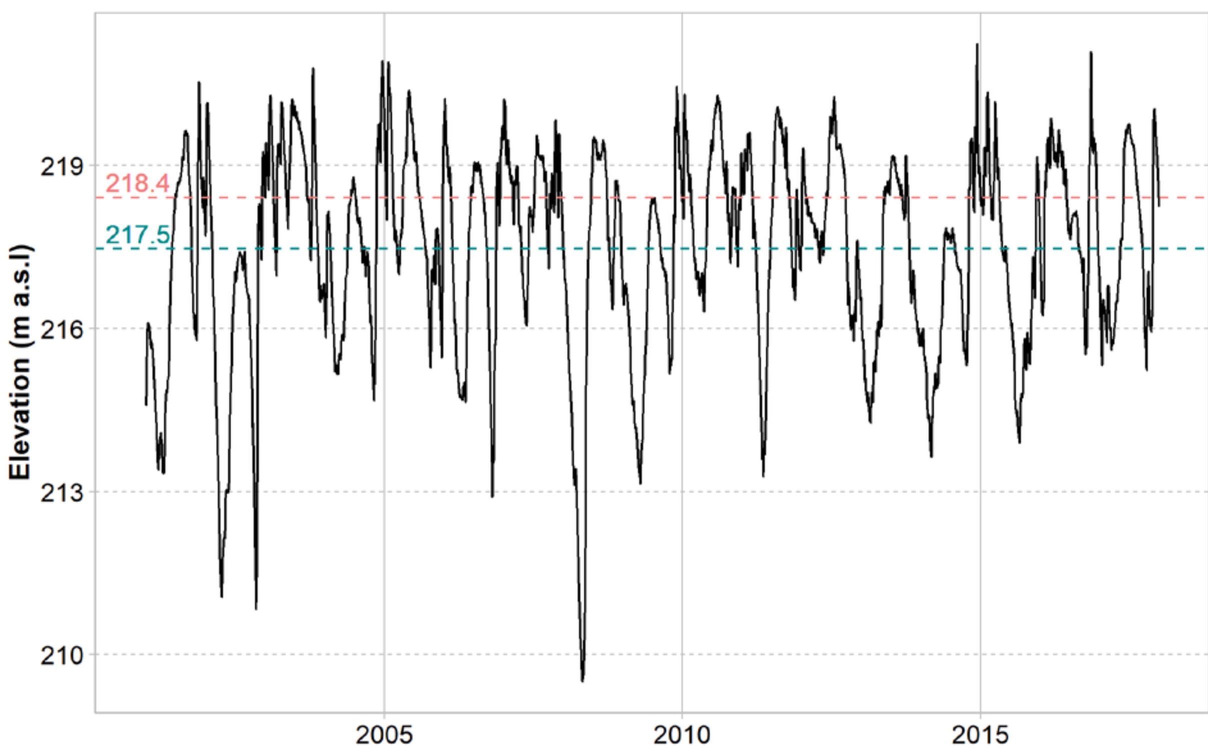


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Figure 18. Daily water elevation in Upper Campbell Reservoir 2001–2017.

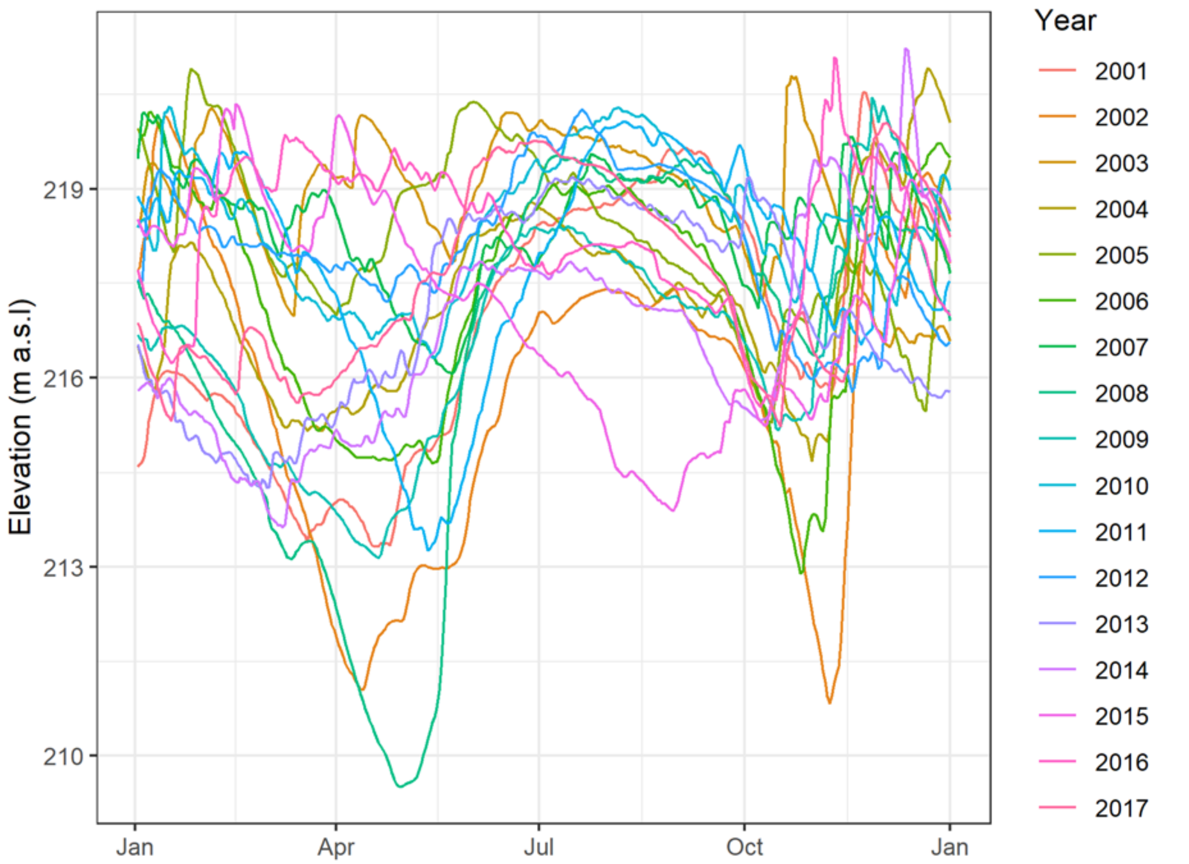


Figure 19. Average daily water elevation in Upper Campbell Reservoir for the periods of 1984–2000 and 2001–2017.

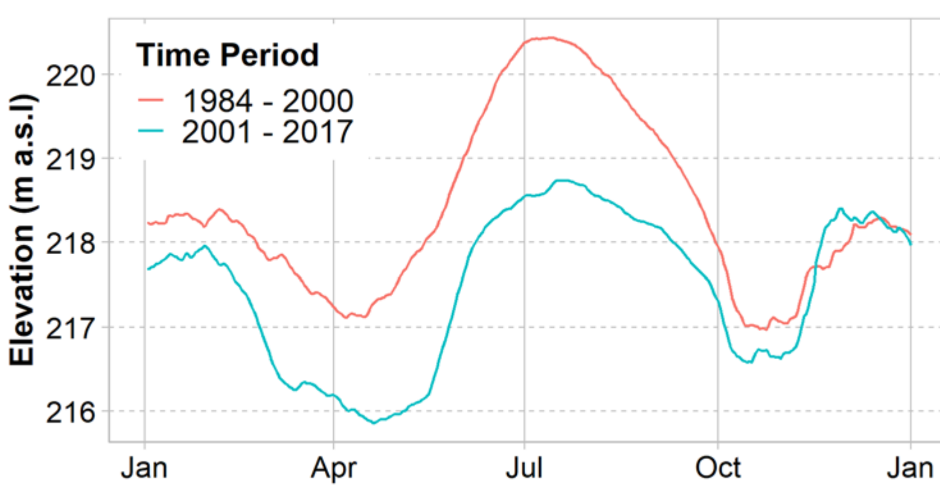
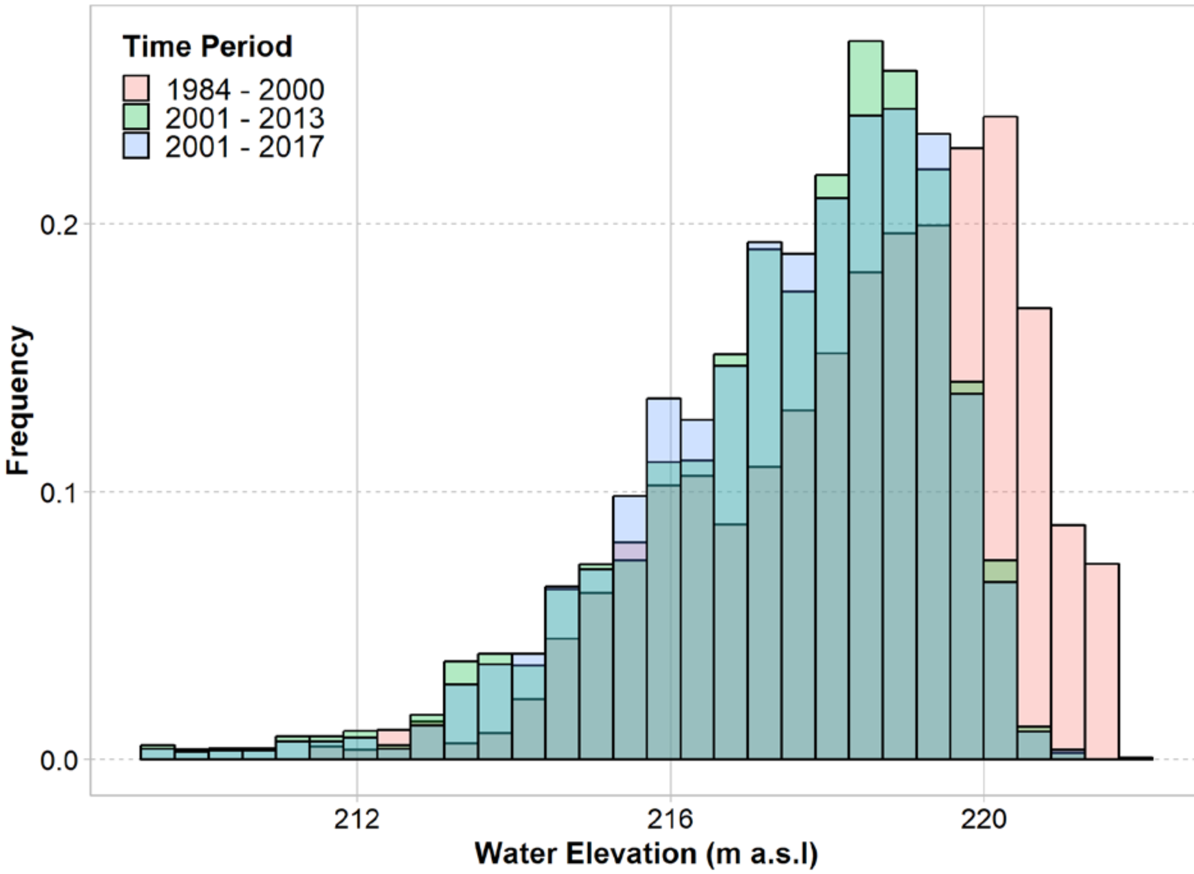


Figure 20. Frequency distributions of daily mean water elevations in Upper Campbell Reservoir for the periods 1984–2000 (red), 2001–2013 (green), and 2001–2017 (blue).



3.1.3. Air Photo Interpretation

In total 820 ha of the Upper Campbell Reservoir was mapped, including vegetated and unvegetated community types (Table 8). Most relevant to this study are the areas of vegetation communities considered in the SVM, i.e., those occurring on the lakeshore of large lakes. These results are presented in the context of change over time and the model in Section 3.2.3.

Table 8. Aerial extent of vegetation communities surrounding the Upper Campbell Reservoir, as mapped with air photo interpretation.

Ecosystem type	Vegetation Community Name	Map Code	Site Series	Area (ha)
Lakeshore - large lakes	Lake Mudflat	MF	00	139.34
	Spearwort Lakeflat	SL	00	166.72
	Hairgrass - Water sedge	HS	00	94.33
	Sitka willow - Water sedge	WS	00	102.41
	Sitka willow - Water sedge Short			*39.93*
	Sitka willow - Water sedge Tall			*62.47*
Lakeshore - small lakes[1]	Sedge Wetland	SW	00	4.49
	Western redcedar/Sitka spruce - Skunk cabbage	RC	12	8.43
Upland Forest	Western hemlock/Douglas-fir - Kindbergia	HK	01	18.25
	Western redcedar - Foamflower	RF	07	16.02
	Western redcedar - Swordfern	RS	05	21.32
Floodplain	Cottonwood - Willow	CW	10	52.09
	Sitka spruce - Salmonberry	SS	08	68.99
Unvegetated	Unvegetated Shoreline	US	00	74.92
	Gravel Bar	GB	00	51.40
	Pond (open water)	PD	00	1.80
Total				***820.51***

[1]Although the study examines Upper Campbell Reservoir only, some areas of the reservoir support vegetation communities that typically occur on small lakes.

Quality Assurance

In 2018, 154 of the 443 delineated vegetation community polygons were field verified with the visual check method (RIC 1998), representing 35% of all polygons (Table 9). Furthermore, an additional 44 polygons were field verified in 2014 by the same terrestrial ecologist that conducted the mapping in 2018. Field verification efforts conducted in 2018, correspond with a survey level intensity of 3. The number of polygons field verified is appropriate for this level of survey, however the survey intensity is typically higher, and included more detailed ecological data collection in full plots and ground inspection plots. A survey level intensity of 3 is appropriate for 1:10,000-1:50,000 mapping projects or a study area of 5,000-50,000 hectares (RIC 1998). This level of survey intensity is also recommended for habitat enhancement prescriptions. Given that only 821 ha of riparian habitat were delineated along the shoreline of the study waterbodies, the survey intensity in practice was estimated at higher than a level 3.

Table 9. Number and percent of each mapped dominant community polygon field verified for the Upper Campbell Reservoir.

Ecosystem type	Vegetation Community Name	Map Code	Field Verified Polygons[1]				Total Mapped Polygons
			2018		2014		
			#	%	#	%	
Lakeshore - large lakes	Lake Mudflat	MF	4	29%		0%	14
	Spearwort Lakeflat	SL	28	38%	6	8%	73
	Hairgrass - Water sedge	HS	30	34%	12	14%	88
	Sitka willow - Water sedge	WS	47	40%	13	11%	117
Lakeshore - small lakes	Western redcedar/Sitka spruce - Skunk cabbage	RC	1	50%	1	50%	2
Upland forest	Western hemlock/Douglas-fir - Kindbergia	HK	7	26%	6	22%	27
	Lodgepole pine - Sphagnum	LS					
	Western redcedar - Foamflower	RF	6	55%		0%	11
	Western redcedar - Swordfern	RS	3	27%	1	9%	11
Floodplain	Cottonwood - Willow	CW	4	50%	2	25%	8
	Sitka spruce - Salmonberry	SS	3	38%	1	13%	8
Unvegetated	Unvegetated Shoreline	US	19	24%	2	3%	79
	Gravel Bar	GB	2	50%		0%	4
	Pond (open water)	PD		0%		0%	1
Total			154	35%	44	10%	443

[1]Several polygons were verified in 2018. In addition, several areas that were not field verified in 2018, were field verified during the 2014 ecological community mapping that was completed by the same surveyor

3.2. Data Analysis

3.2.1. SVM Validation

3.2.1.1. Year 5 Boundary Elevation Predictions

Applying the SVM

Annual cumulative distribution functions of mean daily water level for 2001–2017 (Figure 21) show inter-annual differences in water levels, as described further in Section 3.1.2. Water levels were particularly low during 2008 when the level was less than the SVM lower bound of 210 m for a total of nine days (minimum = 209.51 m). The results of the boundary elevation search procedure used to predict the upper boundaries of the communities in Year 5 are shown in Figure 22, while the predicted upper and lower boundary elevations are presented in Table 10 and discussed further below, where results are compared with field measurements.

As described in Section 2.3.1.2, sensitivity analysis during model development led to changes to how gaps in the water level record (two ten-day gaps in 1985 and 1986) were managed when applying the SVM. This resulted in different boundary predictions from the SVM_{2014}, hereafter referred to as SVM_{2014}* to distinguish it from the version applied in Year 1 (SVM_{2014}; Ballin *et al*. 2015). In particular, the modifications increased the predicted elevation bands of the HS and WSt communities from 0.1 m to 0.47 m and 0.57 m respectively (Figure 23). Model error for SVM_{2014}* was slightly lower than the original SVM_{2014} model.

Figure 21. Cumulative distribution functions of daily water level for individual years during 2001–2017.

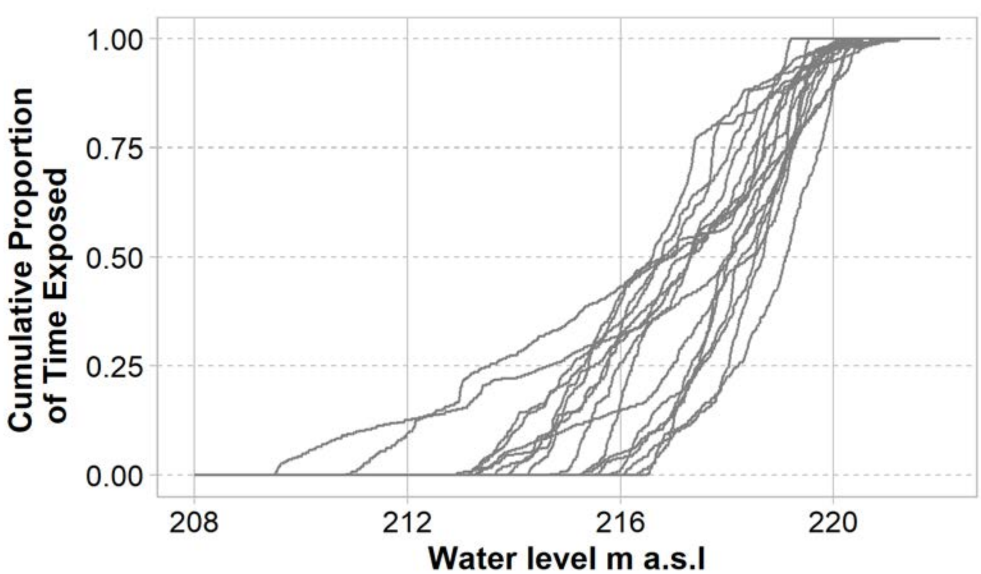


Figure 22. Results of the boundary search procedure to define upper elevation boundaries of vegetation communities. The minimum value of SS (sum of squared differences) represents the boundary.

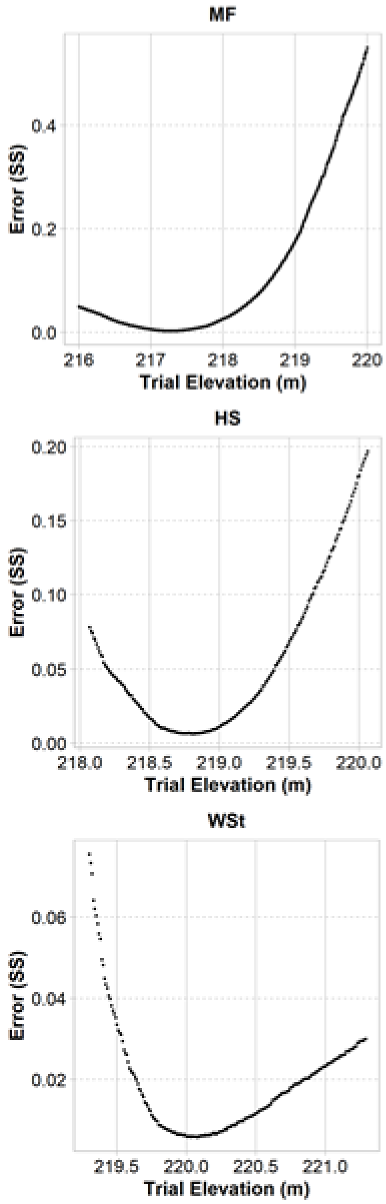
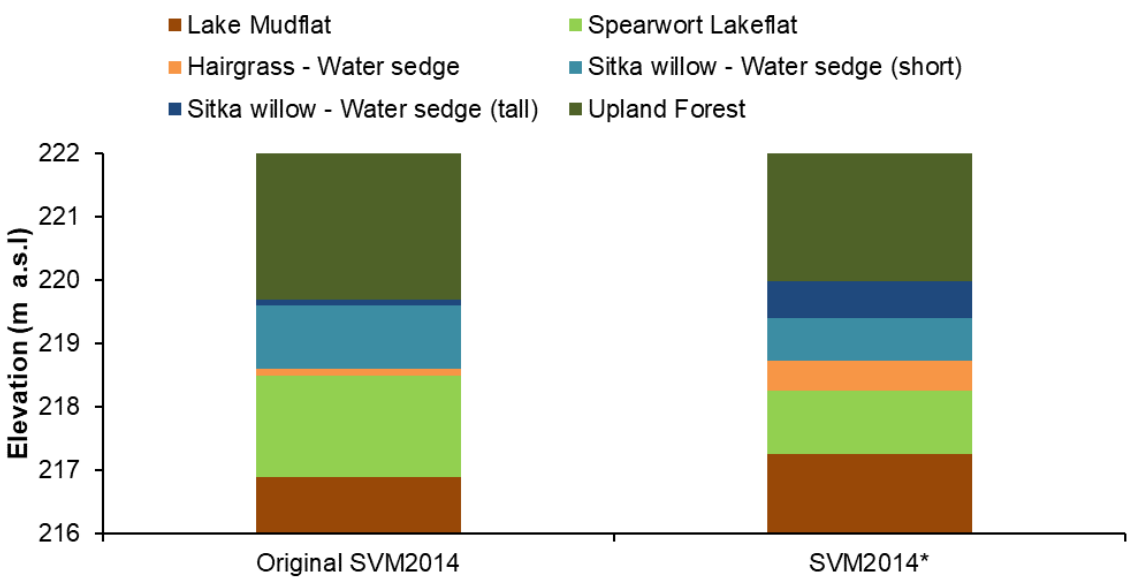


Table 10. Boundary elevations of vegetation communities at Upper Campbell Reservoir in 2018, as predicted by the SVM_{2018}.

Vegetation Community	Map Code	SVM_{2018} boundary elevation predictions (m a.s.l)	
		Lower	Upper
Lake Mudflat	MF	-	217.27
Spearwort Lakeflat	SL	217.27	218.06
Hairgrass – Water Sedge	HS	218.06	218.80
Sitka Willow - Water Sedge (short)	WS_s	218.80	219.29
Sitka Willow - Water Sedge (tall)	WS_t	219.29	220.08
Upland Forest	UF	220.08	-

Figure 23. Comparison of the original SVM_{2014} results presented in the Year 1 Annual Report (right) with the updated version of SVM_{2014}* (left). The differences reflect a change in how small gaps (i.e., <10 days) in the reservoir level records were managed during model application.



Predicting Elevation Boundaries

The vegetation boundary elevations predicted with the SVM_{2018} (Table 10, Figure 22) were similar to those from Year 1, derived using the SVM_{2014}* (Figure 24). Model predictions and field data for both Year 1 and Year 5 show the occurrence of downslope colonization of vegetation communities since the observations by McLennan and Veenstra in 2001 (Figure 24). This period of downslope colonization coincides with the occurrence of generally lower water levels since 2001 (Figure 20).

Field data indicates that Upland Forest has continued to establish and gain height at lower elevations since 2014. As such the measured boundary was closer to the modelled boundary in 2018 than in 2014. This downslope expansion of Upland Forest has resulted in a reduction in the width of the band occupied by tall Sitka Willow – Water sedge. Other vegetation communities sampled in the field in 2018 generally occupy similar elevation bands as in 2014, with a slight downslope shift for short Sitka willow - Water sedge and Spearwort Lakeflat. Unlike Upland Forest, the measured downslope shift in these lower elevation communities was generally not reproduced by the model. The greatest difference between SVM_{2018} predictions and average 2018 measurements was that the SVM_{2018} overestimated the elevation of the Lake Mudflat – Spearwort Lakeflat boundary (discussed further below).

The discussion above of Figure 24 describes differences between the mean measured and predicted elevations. However, there are several qualifiers to consider when evaluating these differences. Specifically:

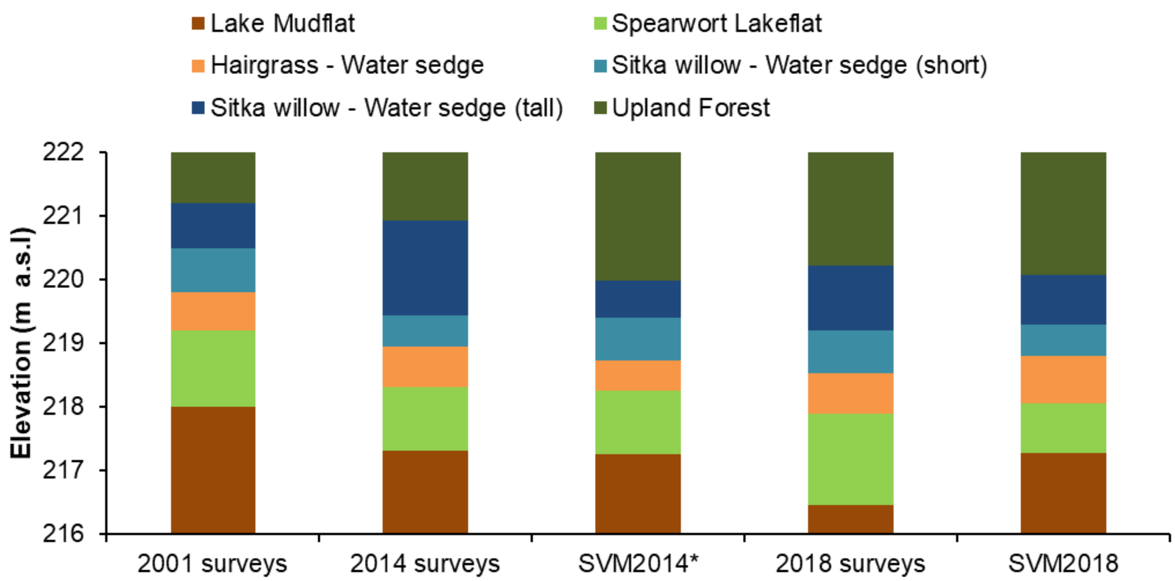
1. The SVM predicts the elevation of six vegetation communities, including Lake Mudflat. These were the six dominant communities observed in the field (i.e., the SVM includes the main communities present in 2018), although one of the 16 vegetation transects (JHT-SVM03) included an area of unvegetated shoreline which is not represented in the SVM. A few occurrences were more similar to other communities that are more frequently observed on small lakes, particularly Sedge Wetland. Sedge Wetland occurs in a similar elevation band as, and is likely a more mature community than HS, thus was classified as HS for the purpose of the transects.
2. The SVM assumes that a particular community is present in only one discrete area along each transect. Two of the 16 transects that were surveyed had 'repeating' communities that occurred in two locations along a single transect (JHT-SVM05 and JHT-SVM10; Figure 16).
3. Most significantly, the SVM assumes that *all* six communities are present on each transect in a defined sequence (Figure 11). Lake Mudflat was only observed on JHT-SVM08 (Figure 16; discussed further below). Otherwise, seven of the remaining 15 transects did not contain all of the remaining five vegetation communities (Figure 16); i.e., some communities were not present along some of the transects.

The differences listed above limit the number of measurements that can be considered during the SVM validation. This is because the validation involves measuring the error between the predicted and measured elevations of the five boundaries; however, the boundaries are not present between all community types for half of the transects. This is typically because the communities were not consistently aligned as the model predicted, e.g., only six of the 16 transects had a Hairgrass-Water sedge community present adjacent to, and upslope of, a Spearwort Lakeflat community, as predicted by the SVM.

In the case of the Lake Mudflat community, the absence of the expected upper boundary (with SL community) in the transect data likely reflects sampling conditions as the ML community may have been submerged or absent, and therefore undetectable at the time of sampling. A MF/SL boundary was only measured at one transect in 2018 (transect JHT-SVM08 in Figure 16). This single measurement of the MF/SL was unlikely to be representative of the elevation of the boundary around the reservoir. However, for multiple transects in Figure 16, the SL community extends to the water surface, below the elevation of the MF/SL boundary measured at JHT-SVM08. This suggests that the MF/SL boundary is generally lower at other sites, but it was not surveyed because it was below the water surface. For SVM_{2018} validation the measured MF/SL boundary was therefore assumed to be equal to the average of the single measured MF/SL boundary, *plus* the additional 13 measurements of the boundary between the SL community and the water surface. This assumed boundary would likely be an overestimate of the "true" elevation of the MF/SL boundary at the reservoir; however, it was lower than the single measured MF/SL boundary at Figure 16; i.e., if this single boundary had been used in the validation, then the magnitude of this over-estimation would likely have been greater.

In total, there were 59 measurements of boundary elevations available for model validation, out of an expected total of 80, due to the three reasons listed above. Thus, validation could be undertaken for 74% of expected boundaries using the model. This was higher than in Year 1, when 62% of potential maximum number of boundaries could be used for validation (Ballin *et al.* 2015), and fewer boundaries were available due to fewer transects being sampled. Also, for transects with 'repeating' communities (i.e. there are two boundaries between two specific communities), as occurred at JHT-SVM05 and JHT-SVM10, only the lower-most boundary was used for the validation.

Figure 24. Comparison of vegetation boundary elevations determined by McLennan and Veenstra (2001) with 2014 survey observations, SVM_{2014}*, 2018 survey observations and SVM_{2018}.



3.2.1.2. Model Error and Validation

Comparison of 2001, 2014, and 2018 Field Data

The standard deviation in field measurements was compared among data collected for baseline (2001; McLennan and Veenstra 2001).[1], Year 1 (Ballin *et al.* 2015), and Year 5 (2018) at Upper Campbell Reservoir (Table 11). This provides insight into whether the variability in the field data that was observed in Year 1 and Year 5 is representative of the variability of the data used to originally construct the SVM. This comparison shows that the boundary elevation measurements were generally more variable in Year 1 (2014) and Year 5 (2018) compared with 2001 (see standard deviation values in Table 11). However, Year 1 (2014) and Year 5 (2018) surveys show a similar level of variability, with Year 5 having slightly less due largely to better agreement between measurements for WS_t/UF boundary. Still, the standard deviation for every boundary was greater in Years 1 and 5 as compared with 2001.

[1] Note that there is some uncertainty regarding the 2001 measurements. Data were transcribed from a document provided by BC Hydro (Appendix 6 of McLennan and Veenstra 2001); however, it is not certain that these are the exact measurements that were used for the SVM development as the mean elevation boundaries do not directly correspond to those presented in Bruce (2002), and no record is present for MF communities.

Table 11. Variability in individual vegetation community boundaries for data collected during 2001, 2014 and 2018 surveys at Upper Campbell Reservoir. σ, standard deviation; CV, coefficient of variation.

Downslope community	Upslope community	2001 surveys (McLennan and Veenstra 2001)					Year 1 surveys					Year 5 surveys				
		# transects with boundary (max = 10)	% of transects with boundary	Elevation range (m)	σ (m)	CV (%)	# transects with boundary (max = 11)	% of transects with boundary	Elevation range (m)	σ (m)	CV (%)	# transects with boundary (max = 16)	% of transects with boundary	Elevation range (m)	σ (m)	CV (%)
MF[1]	SL						4	31	0.70	0.32	0.15	14	88	0.66	0.19	0.09
SL	HS	7	70	1.05	0.36	0.16	8	62	1.60	0.48	0.22	9	56	2.41	0.75	0.34
HS	WS_s	7	70	0.68	0.24	0.11	8	62	2.05	0.67	0.31	10	63	2.10	0.67	0.31
WS_s	WS_t	8	80	0.79	0.26	0.12	10	77	2.06	0.61	0.28	13	81	1.55	0.46	0.21
WS_t	UF	5	50	0.71	0.30	0.14	9	69	4.29	1.28	0.58	14	88	1.93	0.52	0.23

[1] MF community is not recorded for the 2001 transect data provided (Appendix 6 of McLennan and Veenstra 2001)

Summary of Model Error

Table 12 and Table 13 present a summary of the SVM_{2014}* and SVM_{2018} predictions and the applicable measured boundary elevations. With the exception of the MF/SL boundary, the SVM_{2018} predictions were within the range of the field measurements (also the case for SVM_{2014}*). Overall, SVM_{2018} error was highest for the MF/SL boundary (Figure 25) for which mean absolute error was 0.81 m (an over prediction) and RMSE was 0.83 m (Table 13). The predictions were less biased for the other four boundaries, but the model continued to show a bias towards overestimation of boundary elevations, with the exception of the WSt/UF boundary for which the model underestimated the boundary elevation (Figure 25).

Table 14 summarizes how the error in SVM_{2018} predictions of boundary elevations corresponds to error in distances parallel to the ground, based on the mean slope of the downslope communities. The greatest error relates to the MF/SL boundary, indicating that the SVM_{2018} error for this boundary corresponds to a prediction that the SL community was approximately 3.12 m further upslope than observed. (Although see discussion above that indicates that this value is likely conservative as the MF community was only observed at one of sixteen transects, meaning that average elevation of the MF community was likely lower; i.e. underwater at the time of the survey).

Table 12. SVM_{2014}* boundary elevation predictions, Year 1 surveys, and associated model error.

Down-slope community	Up-slope community	SVM_{2014}* boundary elevation prediction (m)	Year 1 surveys				Model error (m)		
			Number of transects with boundary present (max = 11)	Mean elevation (m)	Minimum elevation (m)	Maximum elevation (m)	Mean error	Mean absolute error	RMSE
MF	SL	216.9	4	217.31	217.06	217.75	0.06	0.23	0.28
SL	HS	218.5	7	218.31	217.63	219.23	0.06	0.30	0.45
HS	WS_s	218.6	7	218.95	217.79	219.84	0.23	0.54	0.66
WS_s	WS_t	219.6	8	219.43	218.17	220.23	0.02	0.43	0.57
WS_t	UF	219.7	8	221.00	219.23	223.51	0.95	1.15	1.53
All communities combined		-	-				**0.29**	**0.57**	**0.87**

Table 13. SVM_{2018} boundary elevation predictions, Year 5 surveys, and associated model error.

Down-slope community	Up-slope community	SVM_{2018} boundary elevation prediction (m)	Year 5 surveys				Model error (m)		
			Number of transects with boundary present (max = 16)	Mean elevation (m)	Minimum elevation (m)	Maximum elevation (m)	Mean error	Mean absolute error	RMSE
MF	SL	217.27	14	216.46	216.23	216.89	-0.81	0.81	0.83
SL	HS	218.06	9	217.90	217.23	219.63	-0.16	0.58	0.72
HS	WS_s	218.80	10	218.53	217.74	219.84	-0.27	0.58	0.69
WS_s	WS_t	219.29	13	219.19	218.39	219.95	-0.10	0.36	0.45
WS_t	UF	220.08	14	220.23	219.70	221.63	0.15	0.36	0.52
All communities combined		-					**-0.24**	**0.53**	**0.65**

Figure 25. **SVM_{2018} model error (m) by boundary type. Positive values denote model under-estimates and negative values denote model over-estimates. Points denote data values.**

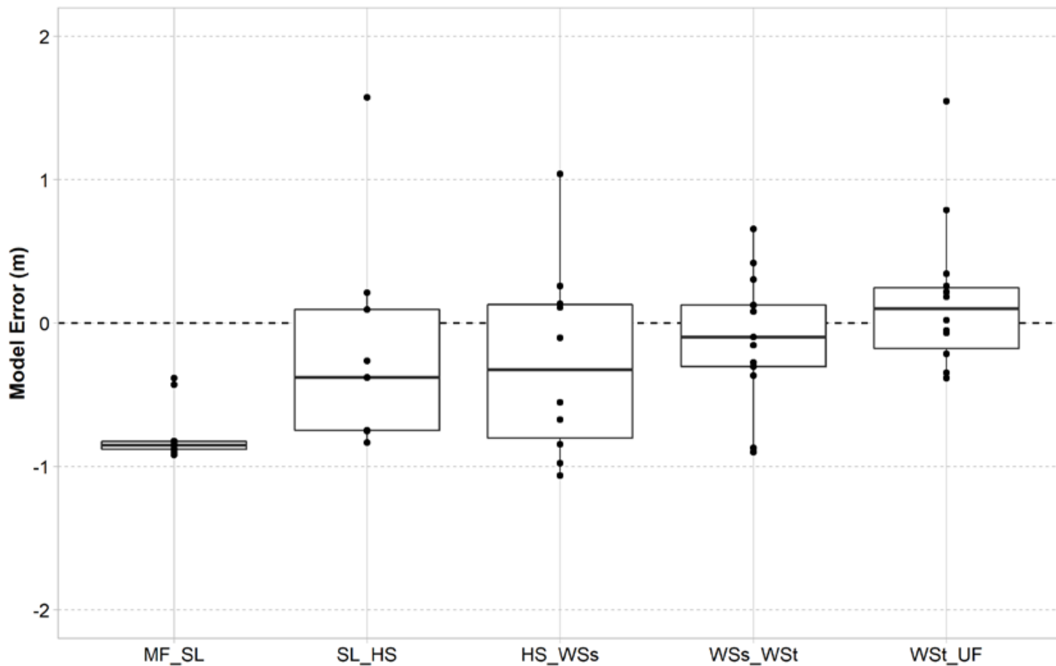


Table 14. **Estimates of how the error in SVM_{2018} predictions of elevation correspond to error in distance on the ground of slope boundaries, based on the mean slope of downslope communities.**

Boundary Type		Mean Community Slope (%)	Mean Model Error (m)	Estimated Mean Error as Distance Parallel to Ground (m)
Downslope Community	Upslope Community			
MF†	SL	26	-0.81	-3.12
SL	HS	10.1	-0.16	-1.58
HS	WS_s	9.3	-0.27	-2.90
WS_s	WS_t	11.3	-0.10	-0.88
WS_t	UF	9.4	0.15	1.60

† MF slope was only measured at one location, JHT-SVM08

Statistical Test Results

Are the predicted elevation boundaries statistically significantly different from the field measurements?

When modelled and measured elevations for all boundaries were pooled, comparison of the elevations for the two groups using one-way ANOVA showed that there was no statistically significant difference between the model predictions and the measurements ($F = 0.096$, $p = 0.757$, $n = 63$).

When the SVM_{2018} predictions of individual vegetation community boundaries were compared with corresponding field measurements, single sample Wilcoxon Signed-Rank tests showed that there were no significant differences between predictions and measurements for all communities, with the exception of the MF/SL boundary (Table 15). This is consistent with the errors shown in Figure 25.

Table 15. Results of a non-parametric Wilcoxon Signed-Rank test to compare modelled and measured vegetation community boundary elevations for individual boundary types. Bold *p*-values denote statistically significant differences ($\alpha = 0.05$). *n*, sample size; *V*, test statistic.

Downslope	Upslope	n	V	*p*
MF	SL	14	0	**0.001**
SL	HS	9	12	0.25
HS	WSs	10	18	0.38
WSs	WSt	13	37	0.59
WSt	UF	14	62	0.58

Are the values of model error (i.e., measured - predicted) significantly different from zero?

When SVM_{2018} errors were pooled, the results of a t-test led to rejection of the null hypotheses that the mean of the error was equal to zero ($t = 2.92$, $df = 58$, p-value = 0.005). This result reflected the general negative bias in the SVM_{2018} errors, which was greatest for the MF/SL boundary (Figure 25) that was only measured at one transect (Figure 16). Omitting the SVM_{2018} errors for this boundary from the input data used for the t-test resulted in the acceptance of the null hypotheses ($t = 0.82$, $df = 45$, $p = 0.42$).

Are the SVM parameters that prescribe vegetation exposure times consistent between different versions of the SVM?

Comparison of the three sets of inundation CDFs (2001, 2014, 2018) using a Kolmogorov–Smirnov test showed that there was no statistically significant difference between any of the sets of CDFs (Table 16). This result validates the conceptual basis of the SVM because it shows that, on the three sampling periods, vegetation communities were present in areas with consistent hydrological regimes, even though the elevations of the communities had changed (Figure 24). This lends support to the assumptions that water level is a dominant control on the vertical distribution of vegetation communities and that communities are in equilibrium with the water level regime (Section 1.3).

Table 16. Results of a Kolmogorov–Smirnov test to compare the SVM parameters among three periods.

Comparison Period	Vegetation Community											
	MF		SL		HS		WSs		WSt		UF	
	D	*p* value	D	*p* value	D	*p* value	D	*p* value	D	*p* value	D	*p* value
2001 v. 2014	0.11	1.00	0.11	1.00	0.11	1.00	0.22	0.99	0.33	0.70	0.33	0.70
2001 v. 2018	0.22	0.98	0.22	0.99	0.22	0.99	0.22	0.99	0.33	0.70	0.22	0.98
2014 v. 2018	0.11	1.00	0.22	0.99	0.33	0.73	0.33	0.73	0.44	0.34	0.22	0.98

3.2.2. Shoreline Gradient and Vegetation Presence

3.2.2.1. Ground-based Slope Validation

Vegetation cover generally decreased as slope of the ground increased for the vegetation communities sampled around Upper Campbell Reservoir (Figure 26). Linear regression results show that the negative relationship between cover and slope was statistically significant when all communities around Upper Campbell Reservoir were analysed (r^2 = 0.32, p<0.001, n = 174, (Table 16). Overall, vegetation cover was <30% (i.e., the assumed threshold used to define unvegetated shoreline) when slope exceeded 22%, based on the regression equation (Table 16). When communities were analyzed individually, the only community that did not show a statistically significant negative relationship between cover and slope was the upland forest community (UF) (i.e., vegetation presence on UF is independent of slope). Vegetation cover for the remaining four communities exhibited a negative relationship with slope (Figure 26). For these communities, the critical slope at which average vegetation cover was <30% ranged from 15% to 40%. It is notable that the critical slope was 15% (i.e., the threshold assumed during WUP development) for two of the four communities (SL and HS).

Visual inspection of the scatterplots (Figure 26) indicates that there is no clear slope threshold above which vegetation cover abruptly declines. This was confirmed based on applying piecewise regression models, which confirmed that there is no apparent "breakpoint" in the regression lines that would indicate the presence of a clear slope threshold that differentiates relatively high and low vegetation cover.

Figure 26. Relationships between vegetation cover and slope for all (a) and dominant vegetation community types (b–f) sampled around Upper Campbell Reservoir. The blue line indicates the linear best fit, grey banding shows the 95% confidence interval.

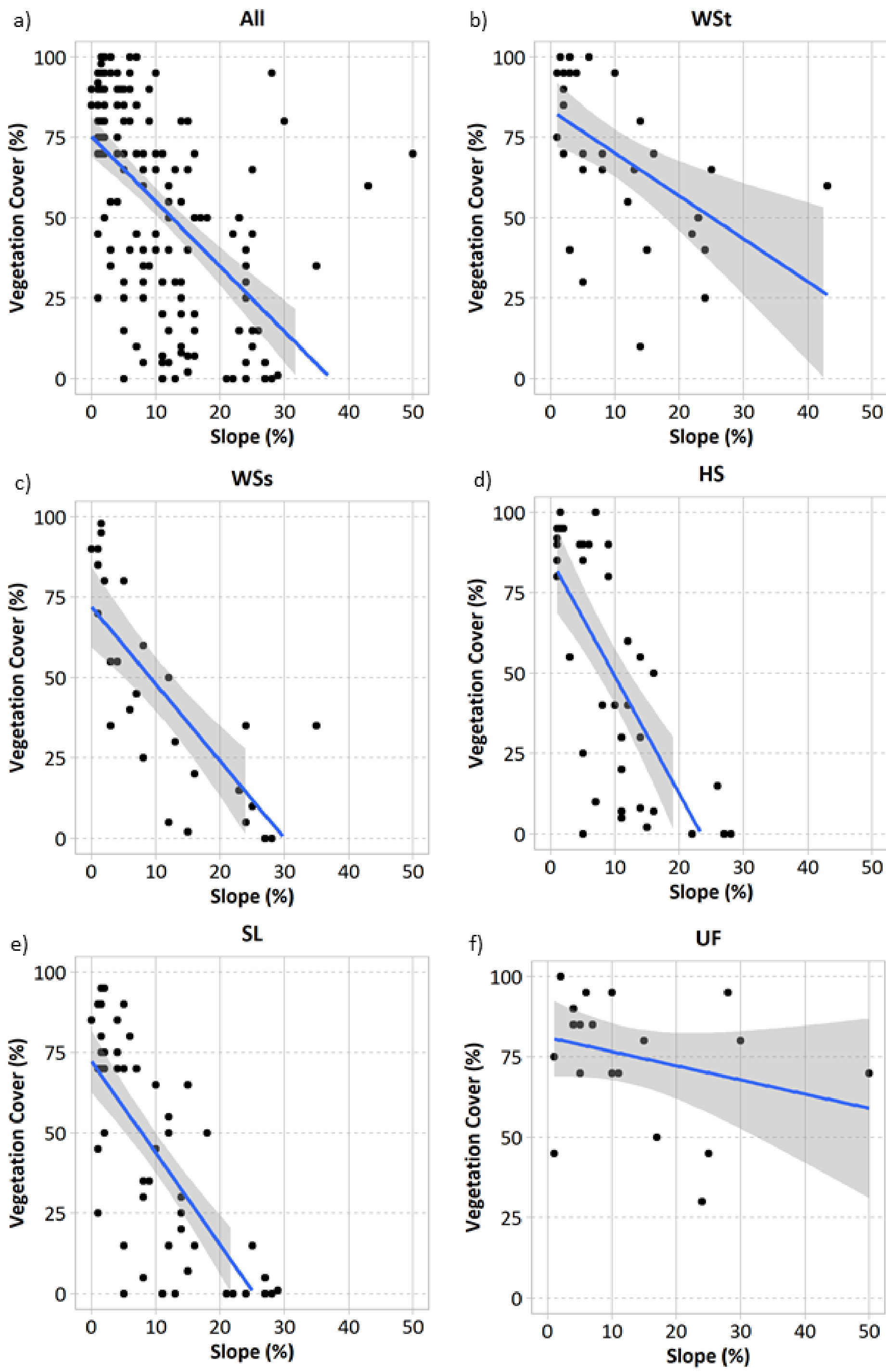


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Table 17. Results of linear regression of vegetation cover on slope for each vegetation community.

Vegetation Community	Critical Slope[1]	n	Regression Slope	Standard Error	R^2	*p* value
All	22	174	-2.0	0.22	0.322	<0.001
UF	n/a	21	n/a	n/a	n/a	0.21
WSt	40	34	-1.3	0.38	0.277	<0.01
WSs	18	27	-2.4	0.40	0.591	<0.001
HS	15	38	-3.6	0.58	0.523	<0.001
SL	15	54	-2.9	0.35	0.556	<0.001

[1] Shoreline slope above which there is <30% vegetation cover on average, assumed to indicate unvegetated conditions

3.2.2.2. Slope Analysis of Mapped Polygons

Slopes of polygons dominated by unvegetated shoreline were generally higher than slopes of polygons dominated by all other vegetation community types. The mean value of the mean polygon slope was 17.8% for polygons dominated by unvegetated shoreline types (US, MF, PD, and GB) and 13.7% for polygons dominated by other vegetation community types (Figure 27, Figure 28). The difference in these mean values is statistically significant based on a t-test ($t = 3.0094$, $df = 108$, $p < 0.01$).

Figure 27. **Boxplot of mean polygon slopes for polygons dominated by unvegetated shoreline (*n* = 76) and polygons dominated by vegetation (all other communities; *n* = 309)**

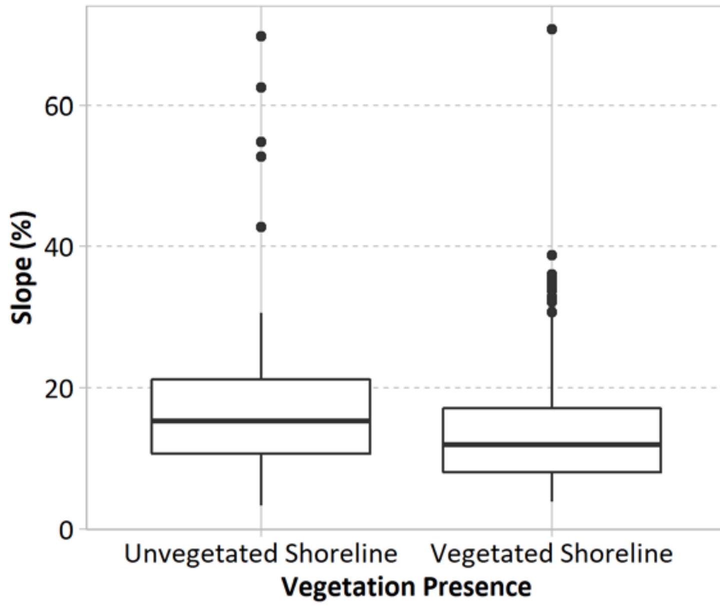
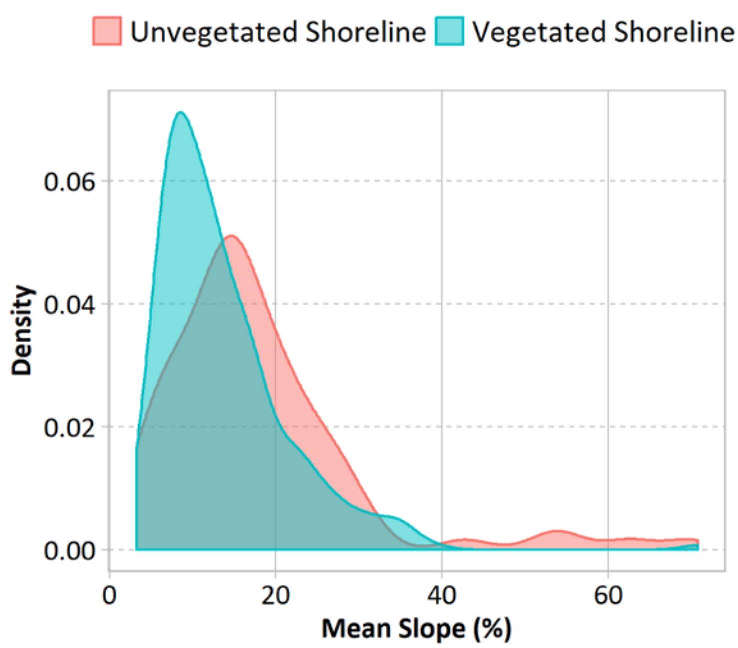


Figure 28. **Frequency distributions of slopes for vegetated (blue) and unvegetated (red) shoreline polygons.**



Mean polygon slopes for individual vegetation communities are summarized in Figure 29 and Table 18. The mean slopes for each vegetated community ranged from 4.3% to 18.4% and were therefore consistently lower than the mean slope of unvegetated shoreline (17.8%). Based on the results of an ANOVA with Tukey's HSD post hoc test (Table 18), the mean slopes of the three of the four shoreline communities were statistically significantly lower than the mean slope of unvegetated shoreline; these communities were: Lake Mudflat (mean slope = 7.6%), Spearwort Lakeflat (mean slope = 9.5%), and Hairgrass – Water sedge (mean slope = 11.4%). Sitka willow did not have a statistically different slope from unvegetated slopes.

Figure 29. Boxplot of mean polygon slopes for dominant vegetation communities.

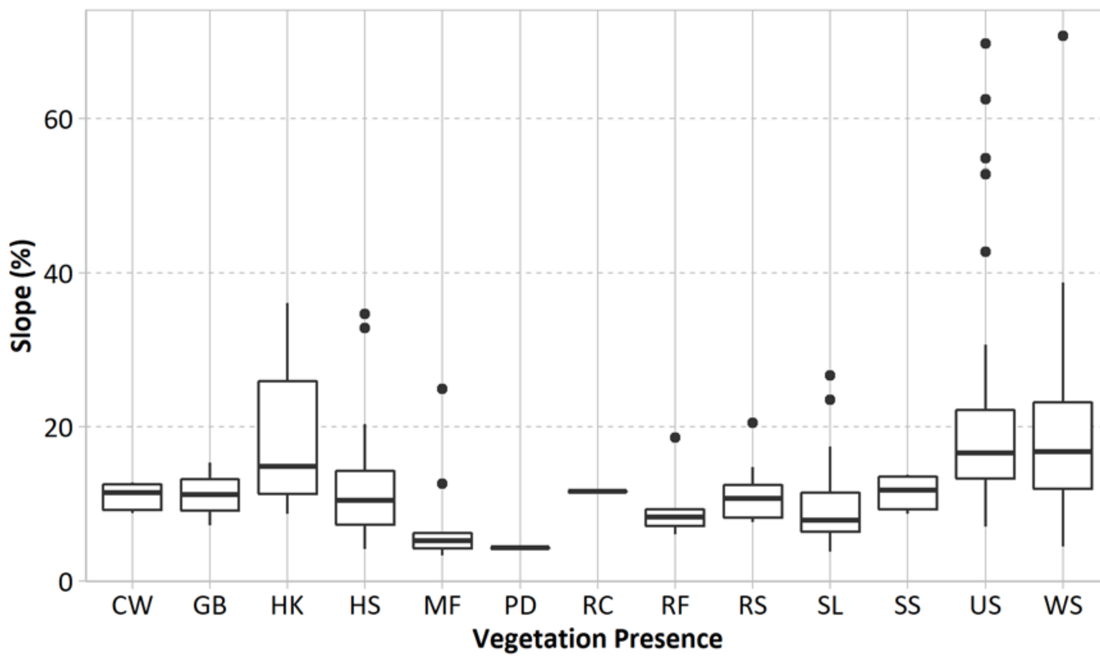


Table 18. Summary of mean polygon slopes by dominant vegetation community. *p* denotes the p-value corresponding to statistical comparison of the average of mean slope values between vegetated community and unvegetated shoreline. Bold denotes statistical significance (see text for further details).

Ecosystem Type	Vegetation Community	Map Code	n	Mean (%)	Minimum (%)	Maximum (%)	Standard Deviation (%)	*p*
Lakeshore - large lakes	Lake Mudflat	MF	10	7.6	3.3	24.8	6.6	**<0.001**
	Spearwort Lakeflat	SL	51	9.5	3.8	26.6	4.7	**<0.001**
	Hairgrass - Water sedge	HS	56	11.4	4.1	34.7	6.0	**<0.001**
	Sitka willow - Water sedge	WS	62	18.4	4.5	70.7	10.5	0.998
Lakeshore - small lakes	Western redcedar/Sitka spruce - Skunk cabbage	RC	1	11.6	11.6	11.6	-	-
Upland Forest	Western hemlock / Douglas fir - Kindbergia	HK	15	18.4	8.7	36.0	8.8	>0.999
	Western redcedar - Foamflower	RF	10	9.0	6.0	18.6	3.6	**<0.001**
	Western redcedar - Swordfern	RS	9	11.5	7.6	20.5	4.1	0.237
Floodplain	Cottonwood - Willow	CW	5	10.9	8.8	12.7	1.8	0.606
	Sitka spruce - Salmonberry	SS	6	11.4	8.7	13.7	2.3	0.610
Non-Vegetated	Unvegetated Shoreline	US	71	19.8	7.1	69.7	12.0	-
	Pond (open water)	PD	1	4.3	4.3	4.3	-	-
	Gravel Bar	GB	4	11.2	7.2	15.3	3.5	0.757

The contingency table for Upper Campbell Reservoir is presented in Table 19 to show the proportion of the total number of polygons that are vegetated or unvegetated with mean slope of either >15% or ≤15%. For vegetated polygons, there are more polygons present in areas with mean slope ≤15% than areas with mean slope >15%, with the opposite trend observed for unvegetated polygons. A Fisher's exact test conducted using the polygon count data (Table 18) shows that the null hypothesis can be rejected for Upper Campbell Reservoir (n=377, p<0.01). Thus, for Upper Campbell Reservoir, the distribution of vegetated and unvegetated areas is dependent on whether slope is greater or less than 15%, with vegetated polygons more likely to be present in areas with slope ≤15% and unvegetated polygons more likely to be present in areas with slope >15%. A qualifier to this statement is that 15% is not a clear threshold to distinguish vegetated and unvegetated areas (e.g., see Figure 26 and DEM resampling analysis described below), as substantial areas of unvegetated shoreline are present on slopes ≤15% and vice versa.

Table 19. Contingency table showing the distribution of vegetated and unvegetated shoreline polygons with mean slopes greater and less than or equal to 15% around Upper Campbell Reservoir. Percentages in parentheses denote proportion of each polygon type in either slope category.

Slope	Vegetated	Unvegetated	Total
>15%	96 (32.9%)	45 (52.3%)	142
≤15%	195 (66.8%)	41 (47.7%)	236
Total	292	86	377

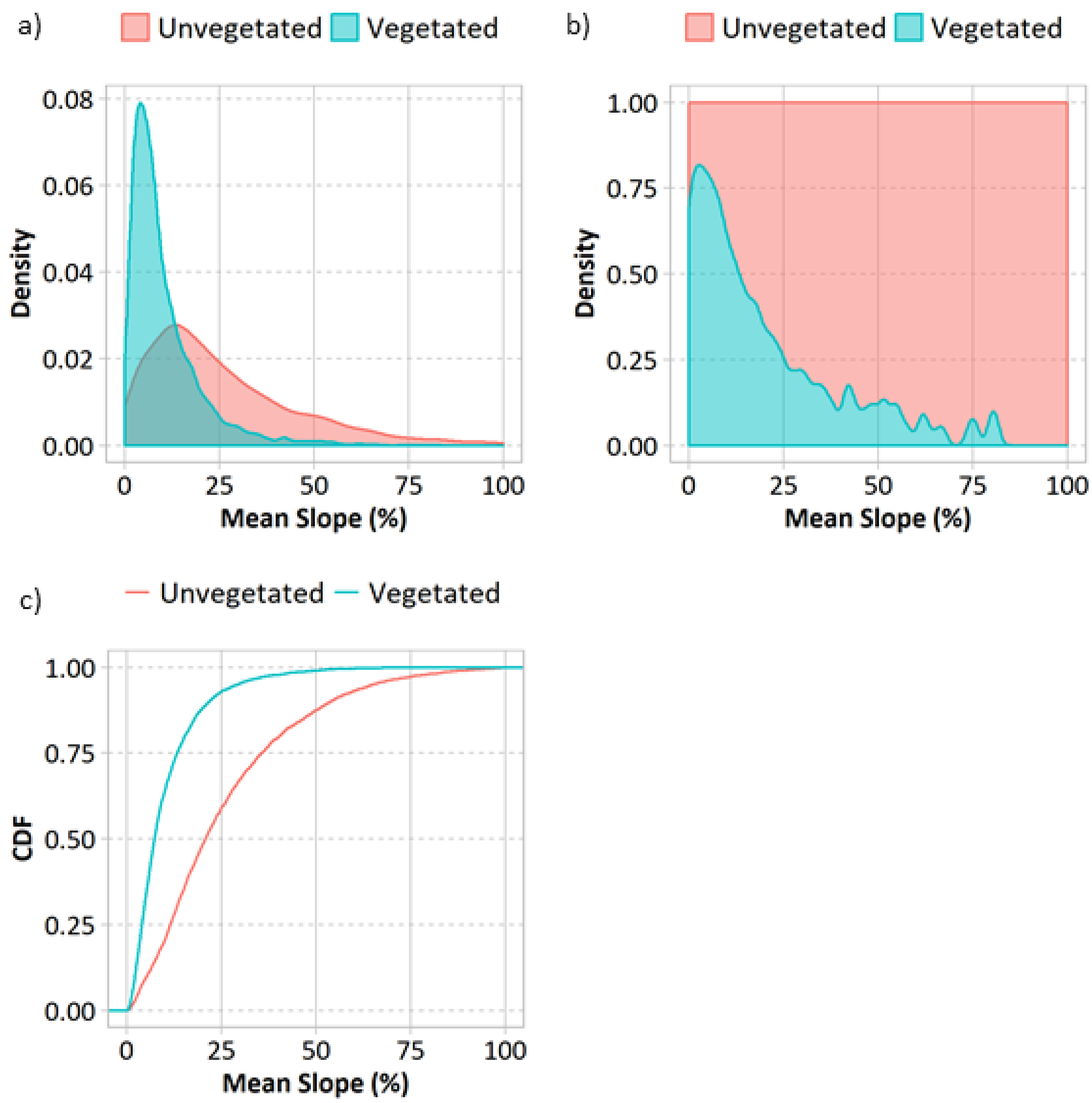
3.2.2.3. DEM Resampling

A total of 7,605 points were randomly generated within the drawdown zone of the Upper Campbell Reservoir shoreline between 216.5 m and 220.5 m. Of these, roughly half (3,827) were within mapped vegetated areas (vegetation coverage ≥ 50%) and the remaining 3,778 were within mapped or unmapped unvegetated areas. The mean slope of the vegetated points was 10.4% while the mean slope of the unvegetated points was 27.6% (Figure 30), with this difference shown to be statistically significant based on a t-test (t = -39.34, df = 4899.8, p <0.001).

Figure 30 demonstrates that the median slopes on which vegetation occurs (7.3%), and vegetation does not occur (21.1%), are somewhat different, but have high overlap; i.e., some steeper slopes are vegetated and some shallow slopes are unvegetated. Key results shown in Figure 30 are:

- half of the vegetated shoreline occurs on slopes ≤7.3% (the median value of the distribution of vegetated land in Figure 30a) but small areas of vegetated land are still present on slopes >30% (approximately the 95th percentile of the distribution of vegetated land in Figure 30a);
- a slope of 15% is approximately the threshold at above which land is more likely to be unvegetated than vegetated (Figure 30b), recognizing that substantial areas of vegetated shoreline occur on slopes greater than this threshold, and;
- unvegetated land is more prevalent on steeper slopes; however, a substantial proportion (~30%; Figure 30c) of unvegetated land occurs on slopes < 15% (the threshold assumed during WUP development).

Figure 30. **Slope and vegetation of randomly sampled points in the drawdown zone of Upper Campbell Reservoir demonstrating: a) the frequency distributions for each vegetation group, b) the relative fraction of each group present at different slopes, and c) the cumulative density function (cumulative frequency distribution) for each vegetation group.**



3.2.3. Validation of Modelling Approach with Airphoto Interpretation

3.2.3.1. SVM Validation with the Hectare Estimation Tool

For the Year 5 results, there was a statistically significant positive relationship between the modelled and measured areas of riparian vegetation communities (Figure 31; r^2 – 0.77, $p = 0.05$, standard error of the estimate = 12.3 ha). The area of riparian vegetation mapped around Upper Campbell Reservoir is higher than the modelled area indicating that the model yields conservative estimates of vegetation presence in the reservoir (Table 20). Specifically, the model predicts that there will be 16%, or 74.3 ha, less vegetated area than that delineated with air photo interpretation. As indicated by the comparison of SVM predictions to transect measurements (Table 14), the areas of SL and HS communities were underpredicted by the model, as compared to the air photograph interpretation. However, contrary to the comparison between the SVM_{2018} and transect data (Table 14), the total areas of WSt and WSs, as determined by air photograph interpretation, are underpredicted by the model by 13% and 8% respectively. This equates to a difference of 5.6 ha and 7.1 ha, respectively. The mapped and modelled areas of young UF were comparable, with the mapped area only 1.6 ha larger than the modelled area.

Visual comparison of the spatial distribution of modelled and measured vegetation communities showed spatial differences in model accuracy, particularly for shoreline areas within the drawdown zone where factors other than water level and slope likely affected vegetation. For example, the model predicted the distribution of vegetation on most alluvial fans fairly well, except for some areas where there was a stream inflow, which in reality, limits vegetation occurring on otherwise habitable slopes and elevations (e.g., Figure 32). In a few areas with relatively steep slopes and other environmental limitations such as a large fetch, model accuracy was lower (Figure 33).

Table 20. Results of the hectare estimation tool (model) and comparison of area measured by air photo interpretation and modelling in Year 5.

Vegetation Community Name	Map/ SVM Code	Areal Extent (ha)		Difference (ha)
		Air Photo Interpretation[1]	Model	
Spearwort Lakeflat	SL	166.72	100.73	65.99
Hairgrass - Water sedge	HS[2]	98.82	79.41	19.41
Sitka willow - Water sedge	WS[3]	123.66	136.33	-12.67
Sitka willow - Water sedge Short	*WSs*	*39.93*	*45.54*	-5.61
Sitka willow - Water sedge Tall	*WSt*[3]	*83.73*	*90.79*	-7.06
Subtotal drawdown zone communities		*389.20*	*316.47*	*72.73*
Upland forest (combined)	UF[4]	64.03	62.48	1.55
Total		**453.23**	**378.95**	**74.28**

[1] Air photographs for Year 5 were collected in October 2017 and field verified in 2018. The difference in vegetation community cover between the end of the growing season in 2017 and 2018 was considered to be negligible

[2]Includes the mapped sedge wetland community that occurs at approximately the same elevation

[3]Includes Cottonwood - Willow fluvial community in tall shrub stage that occurs on large rivers and that was located approximately within the drawdown zone .

[4]Includes all Upland Forest communities and Western redcedar/ Sitka spruce - Skunk cabbage (swamp). Does not include Sitka spruce - Salmonberry that occurs at higher elevations up rivers

Figure 31. Relationship between modelled and measured areas of vegetation communities for Year 5. Statistics relate to linear regression analysis.

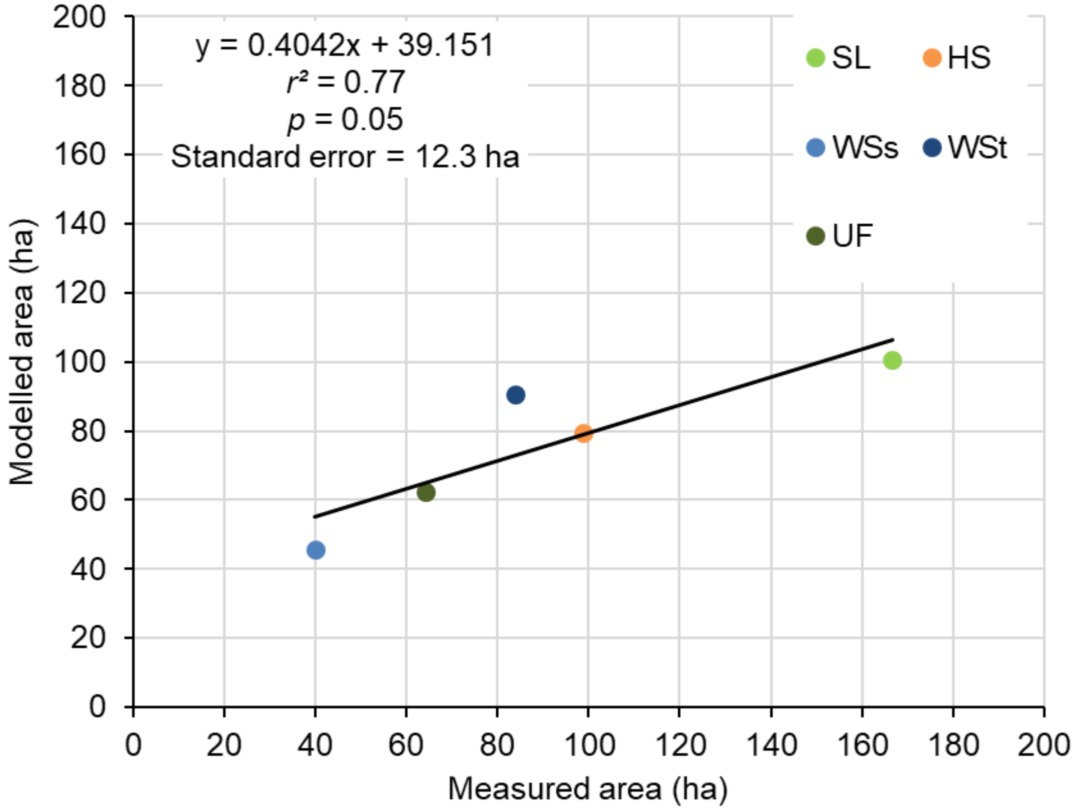


Figure 32. **Comparison of modelled (left) and mapped (right) areas of vegetation communities on alluvial fans on Buttle Lake.**

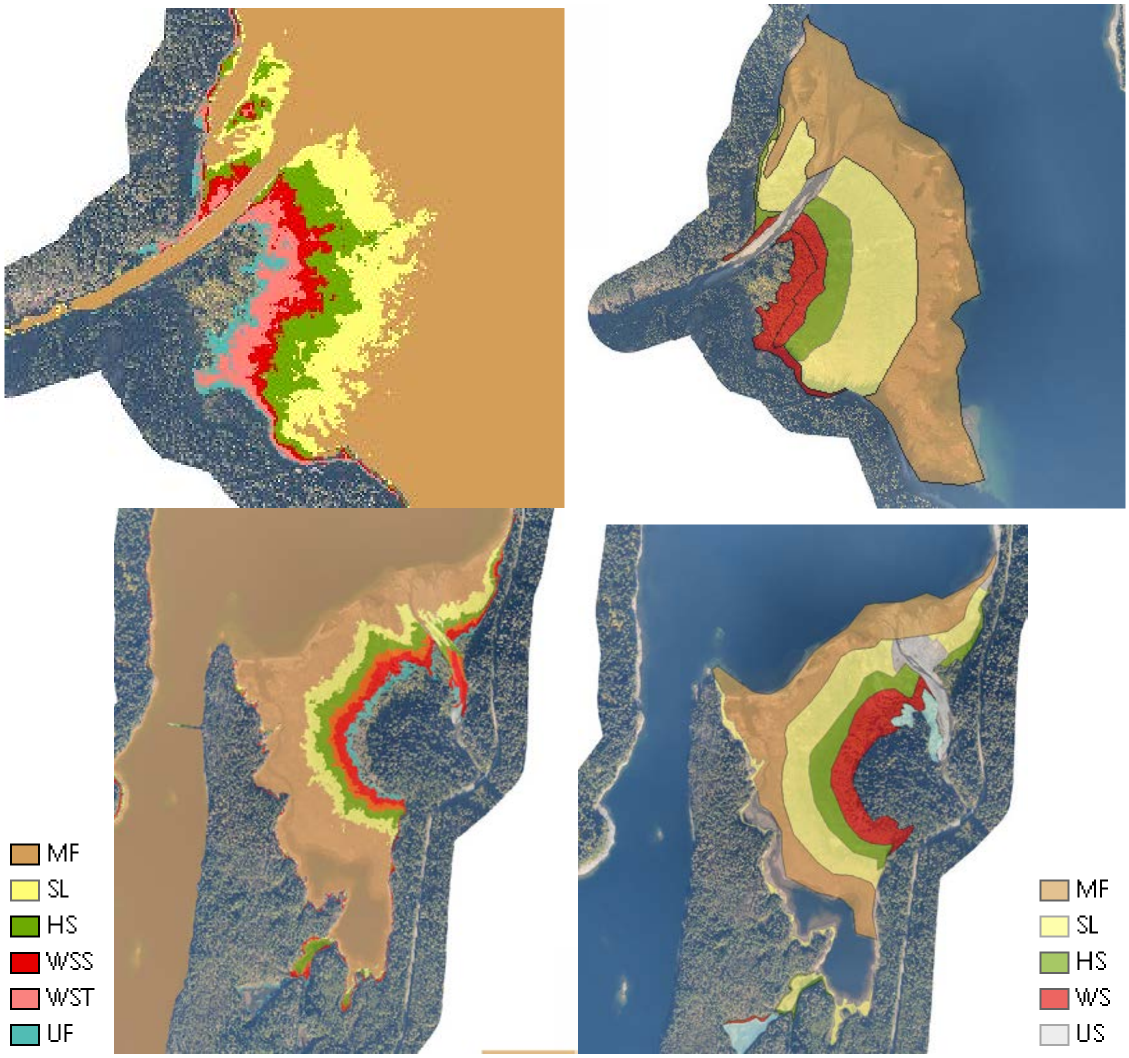


Figure 33. Comparison of modelled (left) and mapped (right) areas of shoreline vegetation communities at Rainbow Island on Buttle Lake for Year 5.

3.2.3.2. Temporal Change in Vegetation Communities

Air photo interpretation can be used to monitor ecosystem change over time. Analysis of air photo interpreted data for Upper Campbell Reservoir from baseline (pre-WUP) (2001; McLennan and Veenstra 2001), Year 1 (Ballin *et al.* 2015) and Year 5 (Hofer 2018a, pers. comm.) images indicate that the total vegetated area of the drawdown zone increased substantially before and after WUP implementation (i.e., 2001 and 2014) from 250.5 ha to 444.5 ha respectively, and then continued to increase after WUP implementation (2018) to 553.1 ha (Table 21). Similarly, the area of all of the individual communities assessed in the SVM (i.e., MF, SL, HS, WSt, WSs, UF) incrementally increased over time, with the exception of MF, of which more was mapped in 2014 than either 2001 or 2018.

Visual comparison of the spatial distribution of the mapped ecological communities supports the numeric data in that the largest difference between the 2001 and 2018 mapped areas is the extent of Spearwort Lakeflat. Large expanses of this community were mapped in 2001 in areas that are now occupied by Lake Mudflat, Hairgrass – Water sedge, and Sitka willow – Water sedge (Figure 34, Figure 35). Furthermore, it appears that for Sitka willow – Water sedge, there is not only an increase in the area of the community overall, but that the proportion of the community in the tall structural stage has increased relative to the short structural stage. Hairgrass – Water sedge and Sitka willow – Water sedge (tall and short) appear to have increased a reasonable amount between baseline and 2018 and 2014 and 2018. The total area of Upland Forest communities increased over time, including between 2014 and 2018, as trees grew and overtopped the Sitka willow – Water sedge community.

A few differences between mapping between years can be observed from visual comparison of the mapping. For example, gravel bars at the outwashes of small to large streams were often mapped as Spearwort Lakeflat in 2001 and were mapped as unvegetated or not mapped in 2014 or 2018 (Figure 31; third figure). Overall more Unvegetated Shoreline was delineated in 2018 than in

previous years. This was intentionally done to increase the sample size for critical slope determinations
(Section 3.2.2).

Table 21. Comparison of total air photo interpreted area over the three assessment years (Year 5, Year 1 and baseline (2001)).

Ecosystem Type	Vegetation Community Name	Map Code	Site Series	Areal Extent (ha)		
				Air Photo Interpretation		
				2018[1]	2014[2]	2001[3]
Lakeshore - large lakes	Lake Mudflat	MF	00	139.34	199.01	16.97
	Spearwort Lakeflat	SL (Ql)	00	166.72	117.92	109.65
	Hairgrass - Water sedge	HS	00	94.33	76.27	35.03
	Sitka willow - Water sedge	WS	00	102.41	77.16	31.42
	Sitka willow - Water sedge Short	-	-	*39.93*	*29.89*	*11.72*
	Sitka willow - Water sedge Tall	-	-	*62.47*	*47.27*	*19.70*
Subtotal vegetated[4]				*363.46*	*271.36*	*176.11*
Lakeshore - small lakes	Sedge Wetland	SW	00	4.49	4.49	8.05
	Hardhack - Labrador tea	HL (HG)	00	-	-	3.83
	Western redcedar/Sitka spruce - Skunk cabbage	RC	12	8.43	14.04	-
Subtotal				*12.92*	*18.53*	*11.89*
Upland Forest	Western hemlock/Douglas-fir - Kindbergia	HK	01	18.25	9.71	-
	Western redcedar - Foamflower	RF	07	16.02	11.42	-
	Western redcedar - Swordfern	RS	05	21.32	16.52	1.91
Subtotal				*55.59*	*37.64*	*1.91*
Floodplain	Cottonwood - Willow	CW	10	52.09	52.09	39.19
	Sitka spruce - Salmonberry	SS	08	68.99	64.90	21.42
Subtotal				*121.08*	*116.99*	*60.61*
Subtotal - all vegetated types				***553.05***	***444.51***	***250.52***
Unvegetated	Unvegetated Shoreline	US	00	74.92	38.35	22.10
	Gravel Bar	GB	00	51.40	21.65	19.93
	Pond (open water)	PD	00	1.80	1.80	16.71
	River	RI	00	-	-	52.90
Other	Unknown[5]			-	-	7.54
Total				***820.51***	***705.32***	***386.67***

[1] It is important to note that the air photographs for Year 5 were collected in October 2017 and field verified in 2018. The difference in vegetation community cover between the end of the growing season in 2017 and 2018 was assessed as being negligible; '-' signifies no data

[2]Ballin *et al.* 2015

[3]McLennan and Veenstra 2001, extracted from raw data, note that all first deciles with a blank value and that had no other deciles, were assumed to have a value of '0', and thus the polygon area was not included. Conversion of blank deciles to '10' was also trialed, which resulted in a drastically different result of 3,040.48 ha.

[4]Excludes Lake Mudflat

[5]Area delineated in 2001 that had erroneous map codes

Figure 34. **Three air photo interpreted alluvial fans on Buttle Lake, 2001 (left) and 2018 (right). Note that the 2001 figures include additional colours to those included in the legend; these are mostly upland forest types.**

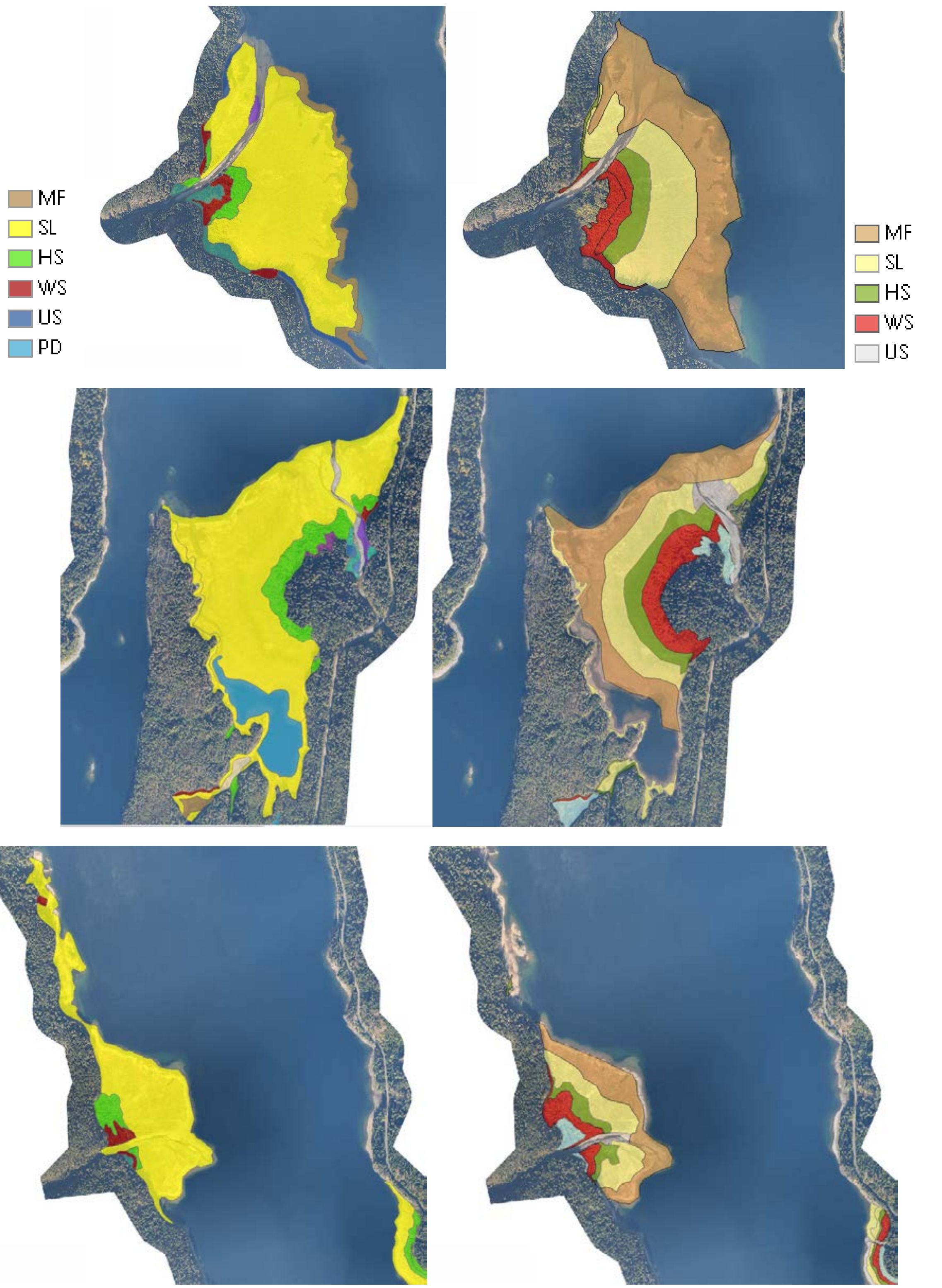
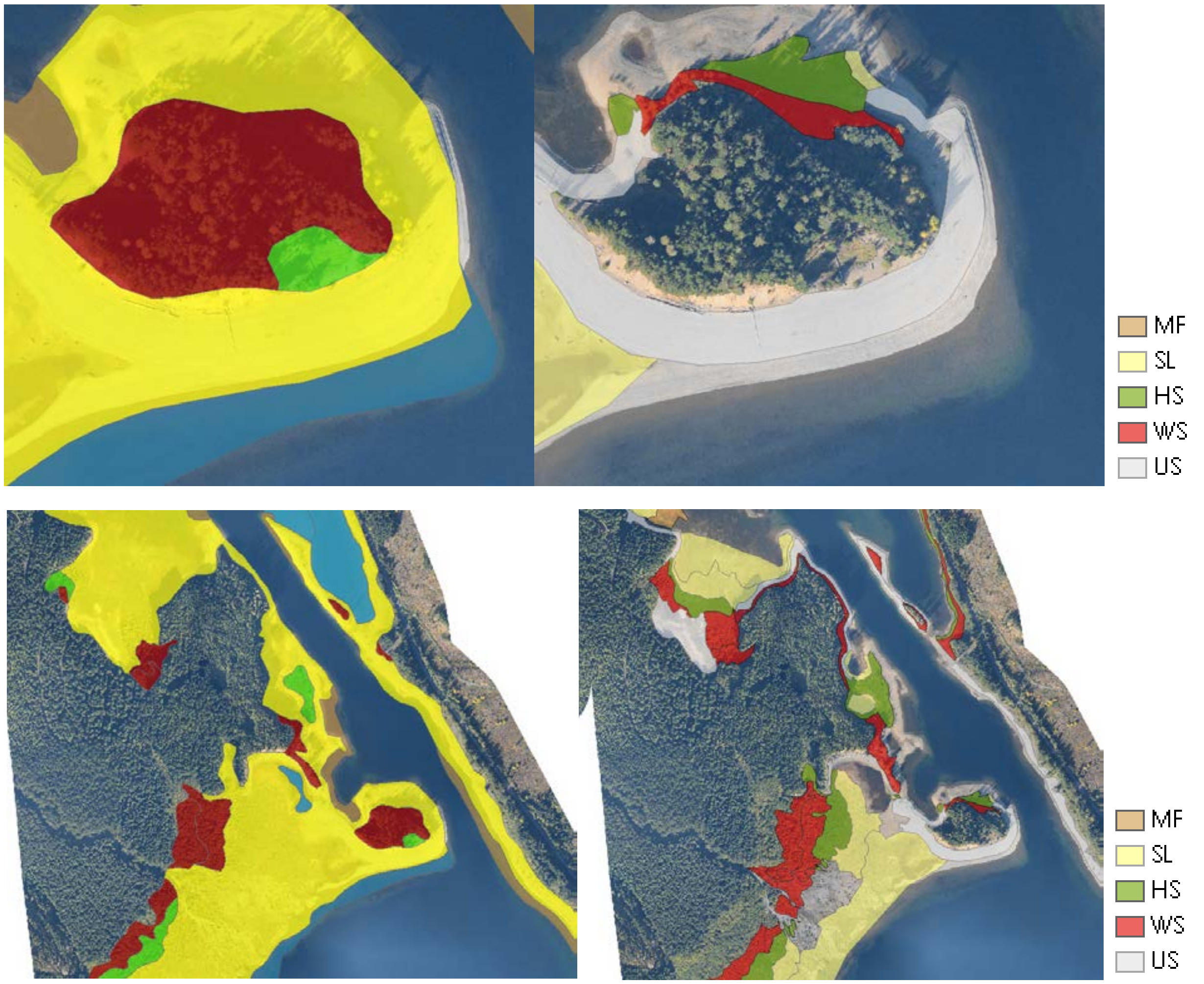


Figure 35. Mapped vegetation around Rainbow Island on Buttle Lake in 2001 (left) and 2018 (right).



4. DISCUSSION

4.1. <u>Does the lacustrine shoreline vegetation model accurately predict the reservoir elevation bands that bound the predefined plant community types?</u>

Measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model ($H_0 1$) may be retained for 4 out of 5 vegetation community boundaries around the shoreline of Upper Campbell Reservoir, but it failed to consistently predict the mean elevation of the lowest community boundary between Lake Mudflat and Spearwort Lakeflat.

To more thoroughly answer this question, the mean elevations of vegetation communities measured around the shoreline of Upper Campbell Reservoir were compared with corresponding predictions from the SVM separately for Year 1 and Year 5. Detailed error statistics were calculated for each community (Table 13, Figure 25) and statistical tests were completed to evaluate the accuracy of SVM predictions and test $H_0 1$ (Section 1.3).

When modelled and measured elevations for plant community boundaries were pooled, one-way ANOVA confirmed that there was no statistically significant difference between the model predictions and the measurements in Year 5 (Section 3.2.1.2), as was also found during validation in Year 1 (Ballin *et al.* 2015). Based on the approach prescribed in BC Hydro (2016), this result led to a second step to test whether the average model error (i.e., measured minus predicted elevations) was significantly different from zero. In contrast to Year 1 (Ballin *et al.* 2015), this test led to rejection of the null hypothesis that the mean of the error in the Year 5 predictions was equal to zero ($p < 0.01$; Section 3.2.1.2). This result was caused by high error in the prediction of the elevations of a single boundary, specifically, the Lake Mudflat-Spearwort Lakeflat boundary; e.g., see Figure 25, which shows markedly higher error for this boundary compared with the other four boundaries. Predictions of the MF/SL boundary were consistently overestimated by the SVM. Note however that there was only one field measurement for MF which limited the ability to test this boundary. Model error was not significantly different from zero when results for the MF/SL boundary were omitted from the test (Section 3.2.1.2).

To further examine model performance, the values of SVM parameters for each vegetation community were compared among the versions of the model developed for each of the three periods (SVM_{2001}, SVM_{2014}, SVM_{2018}). In theory, these parameter values should be consistent if model assumptions (Section 1.3) were correct. Results of this analysis (Section 3.2.1.2) showed that these parameter values were not significantly different, thereby providing support for the conceptual basis of the SVM (see Figure 15 for an illustration of how this analysis was conducted).

Based on these results, Management Question 1 may be answered by stating that the SVM accurately predicts the average elevation bands of *most* vegetation communities but it failed to consistently predict the mean elevation of the lowest community (MF/SL).

Some additional observations can be made about model performance, based on model testing and evaluation completed in Year 1 and Year 5. In general, the finding that model parameter values were consistent among model versions (discussed above) provides some reassurance that the way that the relationship between water level fluctuations and plant distribution is conceptualized in the SVM is generally sound. However, there are some qualifiers to this. First, although comparison of SVM predictions with the averages of field measurements suggests reasonable model performance (Figure 16), many of the individual transects measured in the field (Figure 24) differed substantially from the model predictions, as described in detail in Section 3.2.1.1. This indicates that the SVM is poorly suited to predict the vertical distribution of vegetation communities at an individual site and that other variables not included in the model also have an important influence on vegetation distribution (discussed further in Section 4.3 and 4.4). Second, sensitivity analysis conducted in Year 5 highlighted that the SVM predictions can be particularly sensitive to small changes in the daily water elevation records (see discussion in Section 2.3.1.2). The potential effects of this are illustrated in Figure 23, which shows predictions from slightly different versions of the SVM_{2014}. The two model versions differ only in how a small number of gaps in the water level records were filled when the models were developed (specifically, two ten-day periods in 1985 and 1986). The differences in the model predictions seem disproportionally large, given the minor magnitude of the differences in the water level records, which apply to short periods 21 and 22 years before the field data were collected. This sensitivity potentially indicates that the model is “overfitted”, meaning that it is highly specific to conditions during the model configuration period. In theory, the use of different periods for SVM validation should help to detect this issue; however, it is notable that the neither the mean measurements of community boundaries (Figure 23), nor the preceding water levels (Figure 20) differ drastically among the validation periods. Thus, there would be high uncertainty about the validity of the SVM if there was a more substantial difference in the water level regime between the model configuration period (i.e., before the change in operations) and the validation period (i.e., after the change in operations. These shortcomings do not mean that the SVM is not useful. However, managers should have an appreciation of the limitations of the SVM when using results to inform decisions. These issues also highlight aspects of the SVM that could be improved if changes to the SVM are considered (Section 4.3).

4.2. <u>If the model is in error, is the magnitude of the error such that it would warrant a change in the predicted outcome of the WUP?</u>

The SVM performed well at predicting the boundaries of vegetation communities in Upper Campbell Reservoir with the exception of the MF/SL vegetation community boundary. However, this error in model performance is not expected to result in negative outcomes for wildlife.

The Lake Mudflat community is expected to generally have lower values for obligate and facultative aquatic wildlife than Spearwort Lakeflat, which has similar substrate characteristics, but also supports herbaceous vegetation. Therefore, Spearwort Lakeflat is expected to provide some similar habitat characteristics as Lake Mudflat (e.g., meiofauna invertebrate forage for shorebirds), plus additional characteristics that improve the habitat value overall (e.g., cover for juvenile amphibians[2], forage for waterfowl and mammals). Therefore, the finding that the Spearwort Lakeflat community had colonized lower elevations than predicted (and therefore covered a larger area than would be estimated using the SVM_{2018}) in Year 5 likely indicates a benefit to wildlife, relative to the predictions that were made during WUP development.

Although the only statistically significant difference between measured and modelled community boundaries is between MF/SL, there are non-statistically significant (α=0.05) differences for the other community boundaries. Table 14 provides a useful reference to understand the implications of model error for available wildlife habitat as it quantifies the SVM_{2018} error in terms of distance

[2] Note that impacts to amphibians due to changes in the hydrology of shoreline ponds are being considered separately as part of JHTMON-9.

parallel to the ground (rather than vertical elevation). All of the communities except Upland Forest have shifted downwards along the reservoir shoreline than predicted, with a bias of 0.88–3.12 m when expressed as estimated distance parallel to the ground (Table 14). For large alluvial fans, this distance is minor in comparison to the extent of the riparian area, which may extend up to 300 m from shore. However, on smaller alluvial fans and beaches, this difference may have a larger impact. Habitat values for wildlife vary among each defined vegetation community; for example, tall and short Sitka willow – Water sedge communities provide valuable wildlife habitat including forage and cover for Roosevelt Elk (blue-listed species) and deer, as well as abundant and varied structure for bird nesting, foraging and cover. However, in general, greater downward migration of riparian vegetation communities than predicted is expected to be positive for wildlife.

Therefore, it is reasonable to conclude that any error with the SVM may be positive or benign with respect to wildlife around the shoreline of Upper Campbell Reservoir. This conclusion suggests that it is likely unnecessary to re-evaluate the predicted outcome of the WUP for wildlife values. Nonetheless, it would be appropriate for the details of model performance described in this report to be reviewed in the context of the assessments made by the Wildlife Technical Committee during WUP development. In particular, Table 14 and Figure 31 could support this as they quantify model error in terms of distance parallel to the ground and vegetation community area, respectively.

When the SVM_{2018} is extrapolated over the area of the reservoir, as per slope assumptions, the model predicts that there will be 16%, or 74.3 ha, less vegetated area than measured from air photographs in Year 5 (Table 20). Thus, overall, the SVM predictions are conservative. As indicated by the comparison of SVM predictions to transect measurements (Table 14), the areas of SL and HS communities are underpredicted by the model, as compared to the air photograph interpretation. However, contrary to the comparison between the SVM_{2018} and transect data (Table 14), the total areas of WSt and WSs, as determined by air photograph interpretation, are underpredicted by the model by 13% and 8% respectively. This equates to a difference of 5.6 ha and 7.1 ha, respectively, in the area of these riparian habitats available for wildlife. The mapped and modelled area of young UF was comparable, with the mapped area only 1.6 ha larger than the modelled area. These differences in SVM bias (i.e., over- or underpredictions) for a specific community between which error is expressed as elevations (Table 14) or areas (Table 20) reflect differences in topography (slopes) among the communities and the use of the slope assumption when applying the hectare estimation tool (discussed further in Section 4.4).

4.3. Are there changes to the modelling approach that could improve its accuracy for implementation in future WUP reviews?

It is good practice to examine the potential for model improvement following a model validation exercise (Jakeman *et al.* 2006). Insights to ways to potentially improve the SVM can be gained by examining model performance (Section 3.2.1) and critically evaluating the underlying assumptions of the SVM (Section 1.3).

It is notable that the SVM_{2018} error varied among vegetation community types (i.e., the predictions exhibited heterogeneity of variance), with the SVM_{2018} tending to overpredict the elevation of the lowest communities in Year 5, for which mean absolute error was highest (Table 13, Figure 24, Figure 25). A difference in model error among vegetation community types was also observed in the Year 1 results, although mean absolute error in SVM_{2014} predictions was highest for the highest elevation boundary (i.e., the lower boundary of Upland Forest; Table 12). This variability in the error structure of the SVM predictions suggests that one or more assumptions of the SVM are incorrect. Evaluating the assumptions listed in Section 1.3 from an ecological perspective suggests that a key area to focus on to improve the SVM is the assumption that vegetation communities are sensitive to water level fluctuations over the same duration – note that the duration over which the inundation frequencies are calculated is the same for each community, e.g., 17 years for either the SVM_{2001} predictions (1984 to 2000) or the SVM_{2018} predictions (2001 to 2017). Furthermore, the SVM predictions are not affected by the timing of water level fluctuations during water level records; e.g., SVM_{2018} predictions are insensitive to whether an inundation event occurred in 2001 or 2017, or whether it occurred during winter or the growing season. These assumptions are inconsistent with studies of the effects of shoreline inundation elsewhere, which show that vegetation communities display varying sensitivity to inundation depending on how recently it occurs (e.g., Nilsson and Keddy 1988, Riis and Hawes 2002), and the time of the year at which it occurs (e.g., van Eck *et al.* 2006). Intuitively, these assumptions also seem somewhat unrealistic; e.g., it would be expected that the distribution of the herbaceous Spearwort Lakeflat community, which is a rapid coloniser but sensitive to disturbance, would be particularly sensitive to water level fluctuations in the previous one or two years, but less sensitive to water level fluctuations that occurred several years prior.

Another assumption that may be incorrect is that vegetation communities are in equilibrium with the history of water level fluctuations at a site. For lower elevation and herbaceous communities (e.g., Spearwort Lakeflat), this assumption may be reasonable as such communities are relatively rapid to adapt to changes in water level regime (Riis and Hawes 2002). However, this assumption is not expected to be consistently met for upland forest communities (UF) that may initiate soon after the water level regime change has been implemented but take decades (at least) to grow, through successional processes, into the potential community for that site. It is notable that SVM_{2014} model error was highest for the Upland Forest community in Year 1 (model underpredicted elevation; Table 12; Figure 13); however, since then, the Upland Forest colonized downslope (i.e., coniferous trees had slightly overtopped tall Sitka willow) and model error was lower in Year 5 (Table 12), as demonstrated by the comparison of modelled and measured elevations for this community in Figure 24. This suggests that Upland Forest communities were still equilibrating to the water level regime in Year 1.

If there is desire to improve the SVM, then we recommend that the assumptions discussed above and listed in Section 1.3 are critically examined, with specific reference to the ecological requirements of each vegetation community. The desired outcome of this would be some *a priori* rules to inform a model calibration exercise designed to improve model accuracy. For example, it may be appropriate to set an expectation that Upland Forest is sensitive to water level fluctuations over a duration of 10–30 years, whereas Spearwort Lakeflat communities are sensitive to water level fluctuations over a duration of 2–5 years, with heightened sensitivity during the growing season. These rules, based on existing ecological knowledge could then be used to set the boundaries for model calibration; i.e., bound the parameter space. The purpose of the calibration would be to further optimize the SVM parameters (e.g., the distributions shown in Figure 15) to improve model accuracy. This would also better integrate existing ecological knowledge into the model design, which should increase the extent to which the model can be used to make predictions under different conditions.

Undertaking such a calibration exercise would require programming automatic calibration routines to adjust the SVM parameters to minimize error, based on a pre-determined objective function (Refsgaard *et al*. 2007). To avoid overfitting, it is desirable to use a subset of the data for calibration and retain a separate subset of data for independent model validation. The availability of field measurements for three separate periods (2001, 2014, and 2018) supports the potential for further model development; i.e., as opposed to prior to JHTMON-10, when only one set of field data had been collected. In addition to refining the functions that control the effect of water level regime, another potential line of enquiry to improve the SVM is to further examine causes of the variability among transects (Figure 16). As described in Section 3.2.1.1, most of the transects did not completely conform to how vegetation communities were conceptualized in the SVM (Figure 11). Therefore, it could be useful to try and identify whether other factors aside from water level regime (e.g., aspect, wave exposure, presence of streams) influence vegetation distribution and, if so, attempt to incorporate them into the model. In doing so, the principle of parsimony should be observed; i.e., it is desirable to develop a model that is as simple as necessary (Jakeman *et al*. 2006). Therefore, careful consideration should be given before adding further predictor variables to the model that would increase model complexity and degrees of freedom. Further, there is potential to reduce uncertainty associated with model assumptions regarding the dependency of vegetation on slope; these opportunities are considered separately in Section 4.4, which focuses on the issue of slope.

Following a scope change (BC Hydro 2016), JHTMON-10 has focused only on Upper Campbell Reservoir, including the connected Buttle Lake. Therefore, the conclusions presented here are specific to this waterbody and further work (fieldwork and model development) would be required to develop and validate a model that could be applied with confidence to other waterbodies in the Campbell River watershed. However, even if BC Hydro wishes to focus any future model development only on Upper Campbell Reservoir, it could still be useful to further "stress-test" the model by applying it to other systems. Although the model is not expected to be fully applicable to other systems due to differences in vegetation communities, it may nonetheless be useful to, for example, attempt to validate the model using measurements of the WS_t/UF boundary collected at Lower Campbell Reservoir in Year 1 (note that this boundary was measured at most transects, although other boundaries included in the SVM were generally absent at Lower Campbell Reservoir;

(Ballin *et al.* 2015). If successful, validation using data for another system (following parameterization with field data that reflects the operational regime, vegetation communities and associated elevations of that system) would provide good reassurance that the model configuration accurately reflects the ecological requirements of an individual vegetation community. This would also demonstrate that the SVM has potential to inform how reservoir management operations affect riparian vegetation communities more widely throughout BC.

4.4. Is it reasonable to expect that most riparian plant ecosystems require shoreline slopes to have a gradient less than 15% to perpetuate (presumably because it allows for the accumulation of nutrient rich soil through time)? If this is not reasonable, what is the shoreline slope gradient that is required for plant ecosystem persistence?

In summary, results show that the hypothesis that *the likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient* ($H_0$2) can be rejected for all communities except Upland Forest. Further, 15% is a reasonable slope threshold to differentiate vegetated and unvegetated shoreline, although field data show that this threshold varies among vegetation communities. Accordingly, vegetation-community-specific slope thresholds were used when applying the hectare estimation tool to convert SVM predictions to areal estimates.

The relationship between vegetation presence and slope was examined by evaluating three datasets: 1) Ground-based slope validation, which involved field measurements collected primarily along shoreline transects; (2) Slope analysis of mapped polygons, which included tests of the average slope of each vegetation community polygon delineated by air photo interpretation; and, (3) DEM resampling of slopes for randomly selected points within the drawdown zone. This analysis approach provided complementary lines of evidence to answer this management question, recognizing that each dataset provided unique insights, as described in Section 2.3.2.

Analysis of field and air photo interpreted data through three lines of evidence confirm that there is a negative relationship between vegetation cover and slope, supporting rejection of $H_0$2.

Field data, used to support the ground-based slope validation, generally confirmed that, when 'vegetated' was defined as ≥30% vegetation cover (see Section 3.2.2), there was a negative relationship between vegetation cover and slope. When all vegetation communities were combined, the critical slope was 22% (see critical slopes in Table 17); i.e., slightly higher than the hypothesised threshold of 15%. However, analysis of the relationship between slope and cover for specific communities indicates that 15% is a reasonable slope threshold for the three communities closest to the shoreline (i.e., SL, HS, WSs; Table 17), for which the critical slope ranged from 15% to 18%). However, the critical slope identified for the WSt community (40%) was considerably higher than 15%, and no critical slope was identified for the UF community, as there was not a statistically significant relationship between vegetation cover and slope for this community (Table 17). When all field measurements were combined, slope explained 32% of the variation in vegetation cover (Table 17). When field measurements for individual communities were considered separately, slope accounted for 52–59% of the variability in vegetation cover for the three communities closest to the shoreline, and no relationship existed for the community highest elevation community (UF).

The second line of evidence relied on air photo interpreted vegetation polygons and high-resolution DEM data and a vegetation threshold of ≥70% occupancy of a vegetation community per polygon. This data showed that vegetated and unvegetated communities can be differentiated using a slope threshold of 15%; i.e., unvegetated communities were more likely to be present on slopes above the threshold and vice versa. This result was based on analysis of mean polygon slope and a contingency table (Table 19), which was the main line of evidence identified in BC Hydro (2016) to inform this management question.

The third line of evidence, the DEM resampling analysis (Section 3.2.2.2), also generally the supported the analysis of the contingency table, as it showed that 15% is approximately the slope threshold above which shoreline is more likely to be unvegetated than vegetated, based on vegetation communities comprising 50% or more of the overlying polygon. However, the DEM analysis also identified that substantial areas of vegetated shoreline occur on slopes greater than this threshold, and vice versa (see Figure 30).

Based on synthesizing the three lines of evidence, it may be concluded that 15% is a reasonable threshold to differentiate vegetated and unvegetated shoreline areas. However, a more refined approach is to apply slope thresholds that are specific to vegetation communities. This was therefore undertaken when using the hectare estimation tool (Section 3.2.3.1), which was applied using the critical slope thresholds identified from field data (Table 17).

The results show that slope is an imperfect predictor of the presence or amount of vegetation. This is expected to be in part due to differences in exposure to fetch, soil texture, and aspect. If BC Hydro wish to reduce uncertainty associated with this, a more sophisticated approach would be to use the frequency distributions developed using the DEM resampling analysis (Figure 30) to calculate the proportion of shoreline that is vegetated by each community as a function of slope. These results could then be used to adjust areas predicted using the SVM for Upper Campbell Reservoir. This approach assumes that that the relationship between the occurrence of unvegetated areas and slope does not change through time (as is also an assumption when using a single slope threshold).

A challenge with addressing this management question is that it characterizes vegetation cover using discrete classes (vegetated and unvegetated), whereas vegetation cover is generally present at varying densities along shorelines. From the perspective of providing functional habitat for wildlife (i.e., the focus of JHTMON-10), it is reasonable to assume that vegetation provides negligible value below some threshold of low cover, although this threshold has previously not been defined for this study. It was therefore necessary to define a threshold of vegetation cover to differentiate "vegetated" and "unvegetated" shoreline; when estimating critical slopes (Table 17). For each line of evidence used to test the slope analysis we used different criteria for defining "vegetated", due to differences in data types and analysis methods. For the transect data, which was determined to be best suited to the hectare estimation tool, we selected a threshold of 30% vegetation cover based on reviewing the data from an ecological perspective and our understanding of the requirements of wildlife present in the watershed (see Section 3.2.2 for further rationale for this threshold). We believe that this threshold is appropriate and well-suited to the objectives of the monitor, although we recognize that development of the threshold was partly subjective and that there will be some variability in the appropriateness of this threshold among vegetation communities. Readers should consider this context if conclusions are extended to make inferences about potential effects to wildlife.

4.5. Has the distribution of riparian plant ecosystems changed following implementation of the WUP and if so, can the change be attributed to the WUP operation?

The hypothesis that *plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation"* can be rejected ($H_0$3).

The distribution of riparian plant ecosystems has changed following implementation of the WUP and these changes can be attributed to WUP operations. This conclusion is based on the finding that the area and distribution of riparian vegetation communities has changed between the pre-WUP (2001) and post-WUP survey years (2014, 2018), as quantified in Table 21 and discussed in Section 4.2. Further, as discussed in relation to Management Question 1 (Section 4.1), the SVM generally predicted the reservoir elevation bands at which shoreline riparian vegetation communities occur (with statistical significance) (Figure 24), with the exception of SL/MF, which was estimated to be higher upslope than measured (discussed in Section 4.2). There was a strong positive relationship between the total area of each community mapped and modelled in 2018, further supporting the use of the SVM predictions to estimate changes in the area of shoreline vegetation communities (Section 4.1;
Table 20; Figure 31). These changes between the pre- and post-WUP periods are concomitant with a significant change in water level regime as water levels were generally higher during 1984 – 2000 (pre-WUP SVM_{2001} period) than 2001 – 2017 (post-WUP SVM_{2018} period) (Section 3.1.2).

In general, there has been a downslope expansion of the riparian vegetation communities following the WUP implementation, as shown by the transect data presented in Figure 24 and discussed further in Section 4.2. Examination of the area of riparian vegetation communities around the reservoir based on air photo interpretation provides further insight to changes in the extent and distribution of vegetation communities over time and indicates whether changes can be attributed to WUP operations. Overall, comparison of TEM mapped shoreline vegetation community polygons indicates that there has been an increase in all riparian vegetation communities identified by the SVM (SL, HS, WS, UF) communities between 2001 and 2018, with an approximate doubling in vegetated area between these periods (an increase from 251 ha to 553 ha; Table 21). Visual examination of the mapping indicates that much of the area interpreted to be SL prior to WUP implementation is now assigned to later successional communities (i.e., HS, WS), while some has been attributed to MF. Evaluation of the 2001 shapefile data indicates that many of these sparsely vegetated areas of SL vegetation were mapped without a decile (i.e., proportion of community occupancy of a polygon) so that they would be interpreted as sparsely vegetated and their areas not included in vegetation community totals (Table 21).

Therefore, both transect data and air photo interpreted data indicate that the distribution and area of riparian vegetation communities along the shoreline of the Upper Campbell Reservoir have changed since implementation of the WUP and these changes are generally consistent with the predictions of the validated SVM. Overall, there has been a biologically significant increase in the area of riparian vegetation, indicating that the WUP has increased the area of habitat for obligate and facultative wildlife.

5. CONCLUSION

JHTMON-10 conclusions after Year 5 are summarized in Table 22. Year 5 analysis confirms that, for Upper Campbell Reservoir, the SVM accurately predicted the elevations of riparian vegetation communities (Management Question 1), with the exception of the elevation of the lowermost boundary between Lake Mudflat and Spearwort Lakeflat ($H_0$1 retained for 4 of 5 community boundaries). This elevation was overpredicted by the SVM, based on validation using a single field measurement. The Lake Mudflat community is expected to generally have lower wildlife values than Spearwort Lakeflat and therefore the presence of a wider band of Spearwort Lakeflat than predicted indicates a benefit to wildlife, relative to the predictions that were made during WUP development. Based on this, the SVM error does not warrant a change in the predicted outcome of the WUP (Management Question 2).

Although, the SVM and associated approach to estimating vegetated areas was generally validated, the model could be improved in several ways to improve accuracy (Management Question 3). Most notably, this could be achieved by refining the assumptions that vegetation communities are equally sensitive to the timing and duration over which water level fluctuations occur. Further, the assumption that vegetation does not grow above a critical slope could be re-examined, potentially by considering other factors that affect vegetation distribution such as fetch and aspect. Currently, the model has been validated for Upper Campbell Reservoir and further work would be required to apply the SVM to other waterbodies.

Currently, the use of the SVM to estimate the areas of vegetation communities is based on the assumption that vegetation cover does not occur above a critical slope threshold. Analysis of Year 5 field data confirmed that there was a negative relationship between vegetation cover and slope ($H_0$2 rejected). Further, in areas with a slope greater than 15%, the shoreline was more likely to be unvegetated than vegetated, indicating that a threshold of 15% is reasonable, with a qualifier that analysis of the DEM showed that substantial areas of vegetated shoreline with cover >50% occurred on slopes greater than 15% (Management Question 4). Analysis of transect data showed that the critical slope above which shoreline was considered unvegetated (<30% cover) varied among vegetation communities, with the exception of Upland Forest for which cover was unrelated to slope. This critical slope varied from 15% to 18% for the three communities closest to the shoreline, while the critical slope was 40% for the tall Sitka willow community. These vegetation-community-specific slope thresholds were considered more accurate than the single 15% threshold identified using the polygon analysis and therefore these thresholds were used when applying a hectare estimation tool to convert SVM predictions to areal estimates.

The distribution of riparian plant ecosystems has changed following implementation of the WUP and these changes can be attributed to WUP operations (Management Question 5; $H_0$3 rejected). This conclusion is based on the finding that the area and distribution of riparian vegetation communities has changed between the pre-WUP (2001) and post-WUP survey years (2014, 2018), with changes generally consistent with SVM predictions. Overall, there has been a biologically significant increase in the area of riparian vegetation, with an approximate doubling in vegetated riparian area between 2001 and 2018. This indicates that the WUP has increased the area of wildlife habitat around Upper Campbell Reservoir, including for facultative and obligate aquatic wildlife. Changes at other waterbodies have not been investigated.

Table 22. Status of JHTMON-10 objectives, management questions and hypotheses after Year 5 (2018–2019).

Study Objectives	Management Questions	Management Hypotheses	Year 5 (2018-2019) Status
1. Determine if the Shoreline Vegetation Model accurately predicts the elevation of the vegetation communities.	1. Does the lacustrine shoreline vegetation model accurately predict the reservoir elevation bands that bound the predefined plant community types?	$H_0$1. Measured elevation bands defining the upper and lower extents of each vegetation community type in the area are not significantly different than those predicted by the shoreline vegetation model.	Management Question 1 and Hypothesis $H_0$1 have been addressed for Upper Campbell Reservoir. The SVM does accurately predict the elevation bands of 4 of 5 of the riparian vegetation community boundaries around the shoreline of Upper Campbell Reservoir, but it failed to consistently predict the mean elevation of the lowest community boundary between Mudflat and Spearwort Lakeflat. Thus, H_01 is retained for all community boundaries except for the MF/SL boundary (Section 4.1).
2. Determine if any errors in model predictions will warrant a change in the predicted outcome of the WUP.	2. If the model is in error, is the magnitude of the error such that it would warrant a change in the predicted outcome of the WUP?	*No associated hypothesis.*	Management Question 2 has been answered for Upper Campbell Reservoir. The SVM only failed to predict the elevation of the lowest vegetation community boundary (SL/MF). Specifically, the SVM overestimated the elevation of the Mudflat Community, which generally has lower wildlife values than the Spearwort Lakeflat community. Thus, this error indicates a benefit to wildlife, relative to the predictions that were made during WUP development. Based on this, we do not expect it is necessary to re-evaluate the predicted outcome of the WUP for wildlife values (Section 4.2).
3. Determine if changes to the modelling approach could improve its accuracy.	3. Are there changes to the modelling approach that could improve its accuracy for implementation in future WUP reviews?	*No associated hypothesis.*	Management Question 3 has been answered for Upper Campbell Reservoir. Changes to the modelling approach could be made to improve its accuracy for Upper Campbell Reservoir. These include: accounting for differences among vegetation communities in the sensitivity to the timing and duration of inundation, as well as considering how to incorporate the influence of factors other than slope that supress vegetation growth (e.g., fetch, aspect; Section 4.3)

4. Determine whether the riparian plant ecosystems in question require a gradient less than 15% to perpetuate.	4. Is it reasonable to expect that most riparian plant ecosystems require shoreline slopes to have a gradient less than 15% to perpetuate? If this is not reasonable, what is the shoreline slope gradient that is required for plant ecosystem persistence?	$H_0$2. The likelihood that a particular plant ecosystem type occurs within a predicted reservoir elevation band is not dependent on shoreline gradient.	Management Question 4 and Hypothesis $H_0$2 have been addressed for Upper Campbell Reservoir. Riparian vegetation community cover decreases with an increase in slope; therefore, *$H_0$2* is rejected. Further, shoreline is more likely to be unvegetated than vegetated on slopes greater than 15%. Based on this, 15% may be considered a reasonable threshold, although the critical slope at which a vegetation community is more likely to be vegetated than unvegetated on average (based on a 30% cover criterion) varies among community types. The critical slopes identified for the Spearwort Lakeflat and Hairgrass – Water sedge communities were 15%, while the critical slopes were 18–40% for Sitka willow communities. These community-specific critical slopes were therefore used to convert SVM predictions to areal estimates (Section 4.4).
5. Determine whether the distribution of riparian plant ecosystems have changed following implementation of the WUP, and determine whether the change is attributed to the WUP.	5. Has the distribution of riparian plant ecosystems changed following implementation of the WUP and if so, can the change be attributed to the WUP operation?	$H_0$3. Plant community distribution following implementation of the WUP does not differ significantly from the measured state prior to implementation.	Management Question 5 and $H_0$3 have been answered for Upper Campbell Reservoir. The distribution of riparian vegetation communities has changed following implementation of the WUP and the change can be attributed to WUP operations. Therefore, $H_0$3 is rejected (Section 4.5). The elevation boundaries of riparian vegetation communities have migrated downwards, resulting in an overall increase in the area occupied by riparian vegetation communities, as well as an increase in the area occupied by each individual community. Therefore, the WUP is expected to have increased wildlife habitat in Upper Campbell Reservoir, including for obligate and facultative aquatic wildlife.

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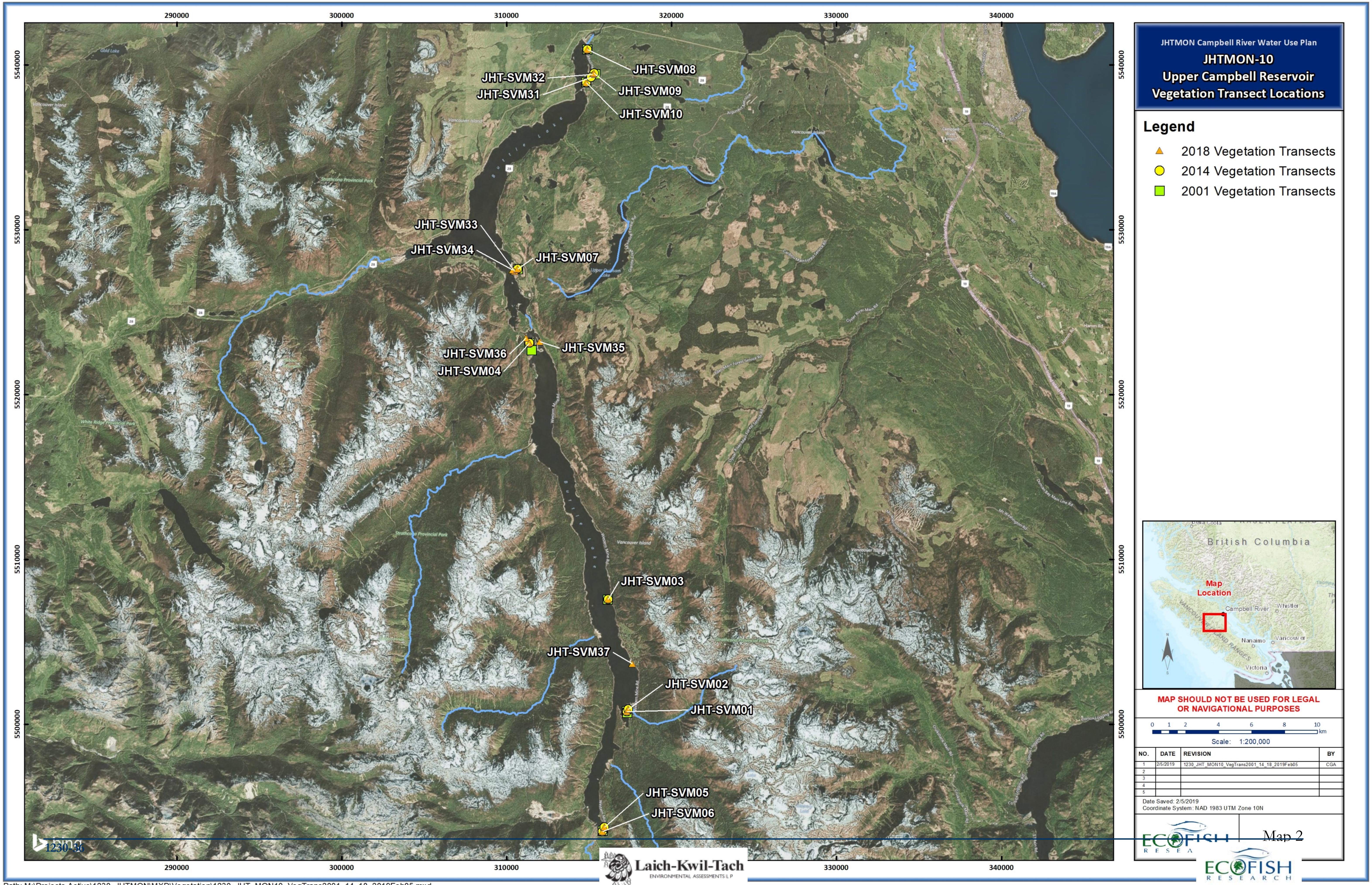
PROJECT MAPS



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