THE INFLUENCE OF HISTORICAL FORESTRY PRACTICES AND CLIMATE ON THE SEDIMENT RETENTION FUNCTION OF WETLANDS

by

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Abstract

Wetlands provide beneficial functions and services (e.g. sediment retention, nutrient sequestration) to downstream aquatic environments. The resiliency of these functions under disturbance conditions is, however, not fully understood. Two wetland–lake systems (Boswell and Viewland) in the central interior of British Columbia whose contributing catchments have historically been impacted by forestry practices were selected to examine how wetland sediment retention responds to disturbance. Core chronologies and sedimentation rates were calculated from unsupported ^{210}Pb measurements using the Constant Rate of Supply (CRS) model, and sediment source contributions were determined using a multivariate unmixing model, for both wetlands and their downstream lakes. Sedimentation rates did not significantly change post-logging in either lake; however, the dominant source to Viewland Lake changed from channel bank material to subsurface material. The increase in the proportion subsurface material consistent with increases in dry density and magnetic susceptibility, and decreases in median grain size and C:N. The bordering wetland was not found to contain any material other than channel bank material. The ephemeral nature of the wetland channel, as well as the length of the channel and the significant decrease in median grain size are thought to have prevented sediment deposition, or increased the potential for resuspension and further transport. Sedimentation rates were greatest near the inflow of Boswell wetland, however, the strongest responses to forestry practices were observed near the wetland outflow. Similarly, significantly lower median grain sizes could have limited deposition in the upstream areas of the wetland. Increases in precipitation as snow and stream discharge in addition to effects associated with forestry practices are thought to have been responsible for driving sedimentation rates in both catchments; however, changes in source contributions were likely the result of active forestry practices.

Contents

List of Tables

List of Figures

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Chapter 1

Literature Review

1.1 Introduction

Wetlands regulate the flow of materials in many landscapes from terrestrial surfaces to aquatic systems (Johnston, 1991). This ecosystem service is thought to be disproportionately large compared to the actual wetland area (Hemond, 1988; Johnston et al., 1990). As a result, the loss and degradation of wetlands can have significant repercussions for downstream water quality and wetland habitat quality. For example, the sediment retention function also regulates the flow of limiting nutrients, such as phosphorus which is typically bound to the surface of sediment particles. Furthermore, the highest concentrations of phosphorus are associated with the smallest size fractions of sediment (Owens & Walling, 2002) which require low energy environments (such as those found in wetlands) to be deposited. When present in high concentrations, phosphorus has been shown to produce eutrophic conditions (Schindler, 1977), often resulting in depleted oxygen concentrations. Thus, the loss of upstream wetlands and their functions could result in the increased delivery of phosphorus to lakes, which might otherwise have been deposited within upstream wetlands.

Wetland losses have been driven primarily by conversions to other land use activities, such as agriculture, urban development, flooding for hydroelectric production, and draining for pest management. Increasing densities of human settlements have been linked to increased fragmentation of wetland area and greater distances between individual wetlands (Gibbs, 2000). With increasing recognition of the important role that they play in the watershed, wetlands and their functions have been receiving more attention. Attempts to protect wetlands have included designating wetland areas for conservation purposes, identifying ecologically significant wetlands, and placing economic value on wetlands and those functions that benefit society (Davis & Froend, 1999). Canada, for instance, has lost approximately 70% of its total wetland area. In an attempt to preserve the remaining wetland areas, it has been estimated that based on wetland services (e.g. migratory bird habitat and water quality improvement), Canada's wetlands have a value of approximately $19,580 per hectare per year (British Columbia Ministry of Environment, 2010).

Wetlands are commonly defined with respect to their hydrological (i.e. water table location) and biological features (i.e. vegetation), with the biological features ultimately driven by the hydrological regime. In a review on sediment storage in fluvial wetlands, Phillips (1989) redefined wetlands in a geomorphic context. The author stated that "their presence and extent is both a reflection and a determinant of the magnitude of sediment storage (or remobilization) within a drainage basin". This definition suggests that any major hydrologic or geomorphic changes in the drainage basin could have significant impacts on downstream wetlands, and has been corroborated by the results of other studies. In a forested wetland in West Tennessee, USA, Hupp & Bazemore (1993) observed that the channelization of streams in the upstream drainage basin resulted in less sediment deposition than in the unchannelized streams. Flow constriction caused by channelization increased stream velocities preventing sedimentation in the wetland.

Land use activities do not necessarily result in the loss of wetland area, but they do disturb the surrounding area and have been found to alter hydrological conditions. Forestry practices, for instance, have been linked to surface compaction, increased runoff, and increased sediment production (Church & Eaton, 2001). Little information is currently available regarding the long-term variability of the sediment storage function of wetlands, how this function is impacted by forestry practices (which have been linked to increased suspended sediment concentrations), and the subsequent impact on downstream water quality (Zedler & Kercher, 2005). Additionally, few studies have focused on the status of wetland functions pre- and post-disturbance. Long-term studies are therefore needed to address the capacity of wetlands to act as buffers under upstream disturbance conditions. This requires an adequate assessment of baseline functions, as well as monitoring during and after land use activities.

Due to high costs and limited resources, it is unreasonable to think that long-term monitoring data would be available for all wetlands and lakes residing in logged catchments. Other techniques are available that enable the development of a long-term historical dataset. Specifically, paleoenvironmental reconstruction attempts to reconstruct past environmental conditions using sediment records. This requires the measurement of "proxy indicators" which allow inferences to be made about historical environmental conditions based on current knowledge of these indicators, and the processes that control their behaviour in a particular environment (Smol, 2008, 2010). This type of approach provides insight into baseline environmental conditions, as well as responses to natural and/or anthropogenic disturbances.

The aim of this thesis is to use paleoenvironmental reconstruction techniques to evaluate the sediment retention function of two wetlands in the central interior of British Columbia, and how they have responded to changing hydrologic and geomorphic conditions as a result of historical forestry practices. The following sections will provide an overview of literature that has been published on wetland characteristics and the sediment storage function of wetlands, sediment transport and storage, disturbance response regimes, impacts of forestry practices and climate on sediment yields, as well as key analytical techniques that have been used to complete the present study. Research questions and objectives are also provided at the end of the literature review.

1.2 Wetlands

1.2.1 Features and functions

A general definition of a wetland includes three main characteristics: all wetlands are temporarily or permanently inundated, possess hydric (i.e. oxygen deprived) soils, and are inhabited by rooted vegetation that is adapted to these conditions. Categories of wetlands often rely on hydrology, vegetation type, and pH to define their boundaries. Marshes are typically inhabited by herbaceous vegetation, while swamps are capable of supporting woody species. Peatlands exist where the soil is rich in organic matter and the water table is at or below the ground surface. Acidic and alkaline conditions belong to bogs and fens, respectively. "True bogs" are more specifically bogs with no defined inflow or outflow, and receive water only through precipitation which is lost only by evaporation. Floodplains and riparian wetlands are situated along stream and river banks where they are able to intercept lateral runoff from uplands. Floodplains also receive overbank flood water when rivers exceed bankfull levels (Mitsch & Gosselink, 2000). These definitions provide a brief overview of the diversity of wetlands that exist. Sub-types of wetlands lie within each of these categories whose definitions are a function of dominant vegetation type, ultimately driven by climate.

Wetlands are commonly viewed as transitional environments between terrestrial and aquatic ecosystems, filtering or buffering downstream ecosystems and improving water quality. Wetlands also act as carbon sinks storing organic matter, support a diverse array of biota, and mitigate flood events (Hemond, 1988; Zedler & Kercher, 2005). Their ability to enhance sedimentation, trap nutrients, metals and contaminants, and improve water quality has also been recognized as an important tool for the management of wastewater (Srivastava et al., 2008) and agricultural runoff (Owens et al., 2007). However, the function(s) that a wetland is able to support is dependent on wetland type, and more importantly, its position in a watershed (Johnston et al., 1990).

1.2.2 Water flow and sediment storage

The movement, or advective-dispersive transport, of particulate matter through wetlands is largely a function of water flow, and the presence (or absence) of aquatic vegetation (Huang et al., 2008). Wetlands that are characterized by inundation or ponding typically promote a sediment trapping or buffering function (Johnston, 1991; Hupp & Bazemore, 1993) as these low-lying regions significantly reduce water velocity and facilitate sedimentation (Hemond, 1988). An urban wetland studied by Brown (1985) was found to reduce peak discharge at its outlet by 12-70% compared to the inlet. In the same study, the author also observed that sedimentation rates were enhanced during peak discharge events, especially those associated with early-May to late-June storms when 56-70% reductions in peak discharge were recorded. This is consistent with the findings of Johnston et al. (1990) who observed that wetlands were associated with higher concentrations of suspended solids during periods of high flow.

1.2.3 Hydrophytic macrophytes

According to Manning's equation, mean flow velocity (in main channels with low slopes) is inversely proportional to surface roughness (also known as Manning's roughness coefficient), and is largely controlled by the presence or absence of vegetation, which is also influenced by slope and substrate type. Hydrophytic macrophytes, or flood-tolerant plants, play an important role in water flow dynamics and sedimentation in wetlands (Clarke, 2002). Water flow reduction and sediment accumulation by macrophytes are dependent on vegetation density and type (Dawson, 1978). Petticrew & Kalff (1992) observed that as leaf area index (LAI) increased, water velocity near the lake bed decreased. Leaf pattern (Clarke, 2002; Huang et al., 2008), plant morphology, as well as shoot movements and flexibility (Sand-Jensen & Pedersen, 1999) can also result in small-scale velocity variations which can be accompanied by a decrease in turbulence. In addition to significantly reducing water velocity (Sand-Jensen & Pedersen, 1999), dense stands of vegetation prevent bed scouring and sediment resuspension (Sand-Jensen & Mebus, 1996; Braskerud, 2001). Although these studies focused on lake environments, it is likely that hydrophytic macrophytes would have the same effect on water velocity, and sediment trapping in wetlands.

Flow modification as a result of wetland vegetation is also linked to the trapping of fine organic and inorganic particles (Clarke, 2002). LAI was found to explain 74% of the variation in the percent of clay accumulating below macrophyte stands in lake bottoms (Petticrew & Kalff, 1992). This suggests that an increase in the density of macrophytes would facilitate the deposition and retention of fine sediments. The reconstruction of historical sedimentation rates in several lakes revealed that lakes containing macrophytes had greater accumulations of sediment over time compared to those which were relatively macrophyte-free (Brenner et al., 1999). However, the role of macrophyte stands in sediment retention is often temporary, as their predominant function is to stabilize the sediment bed through the binding effects of their roots (Sand-Jensen, 1998). Similarly, Phillips (2003) suggested that most wetlands are temporary storage sites of sediment to buffer the output to downstream waterbodies.

1.3 Sediment transfer

Sediment transfer is a two stage process that involves sediment production or mobilization, and subsequently sediment transport by a medium capable of entraining the sediment particles. Firstly, sediment production requires material to be eroded from a terrestrial surface. Surface erosion may occur as a result of bank erosion, raindrop erosion, sheetwash, soil creep, or rapid mass movement (Pye, 1994; Church & Eaton, 2001). Several local and regional factors moderate hillslope erosion, including lithology, vegetation cover, availability of rock and soil, slope length, steepness and roughness (Pye, 1994). Second, sediment transport typically occurs during storm events when a sufficient volume of water is present to overcome the shear stress acting on a particle (Pye, 1994). In an undisturbed forested environment, surface erosion due to running water is rare because of interception by vegetation and subsequent infiltration (Lehre, 1982; Swanson et al., 1982). However, the intensity and frequency of weather events also plays an important role in controlling rates of erosion. Blais et al. (1998) found that lake sedimentation rates decreased an average of 80% during a year when annual runoff experienced a 63% reduction.

Mobilized sediment eventually enters either dispersive or channelized pathways (Bracken & Croke, 2007). The ability of these pathways to transport sediment is a function of stream power (i.e. stream volume and velocity), and particle size. Dispersive pathways are associated with overland flow and are characterized by diffuse connectivity. These pathways exhibit a branching structure and tend to lose volume and power as they travel down a hillslope. As a result, they lose the ability to transport greater volumes of sediment, and larger particles such as sand and gravel. Furthermore, due to their systematic branching and limited power, there is a low probability of these pathways reaching streams via overland flow (Church & Eaton, 2001). On the other hand, channelized pathways are typically longer (Croke et al., 2005), accumulate water with distance, and are more likely to be directly connected to the fluvial system (Church & Eaton, 2001).

The "sediment delivery ratio" relates the amount of sediment delivered to the catchment outlet or the sediment yield (t km^{-2} y^{-1}) to the gross erosion in the basin (t km^{-2} y^{-1}) (Walling, 1983). In general, sediment yield has a positive relationship with slope angle as the degree of inclination provides the potential energy for runoff (Pye, 1994). For smaller catchment sizes the relationship between slope angle and delivery is significant, however, it tends to change with increasing catchment size. Larger basins provide more opportunities for temporary sediment storage as compared to smaller, less complex catchments. Furthermore, hillslopes in larger basins have been found to be decoupled from the fluvial network which again interrupts sediment delivery to the catchment outlet (Phillips, 1995). Other factors affecting this relationship include sediment source characteristics, drainage patterns, channel conditions, vegetation cover, and land use (for a comprehensive review see de Vente, 2007).

1.4 Sediment deposition and storage

The deposition and storage of sediment in aquatic environments depends on the properties of both the depositional environment and the material being transported. More specifically, Stokes' Law states that the deposition of a sediment particle strongly depends on the size of the sediment particle and the viscosity of the transport medium (Pye, 1994); however, this relationship only holds true for low Reynolds numbers (i.e. low turbulence). As the size of a sediment particle increases, the energy required to keep it suspended in the water column also increases. Therefore, as the energy of the system decreases (i.e. velocity decreases), its ability to support larger particles will diminish resulting in sediment sorting (Powell, 1998).

This relationship assumes that sediment particles are transported as discrete particles and does not account for the behaviour of aggregated particles in a water column. Aggregated or flocculated particles (flocs) are comprised of organic and inorganic material which are bound together by surface adhesion. Flocs have varied sizes, shapes and densities, and as a result, different settling velocities from their discrete counterparts (Droppo et al., 1997). Although the formation and transportation of flocs have been found to have a significant impact on the deposition of both organic and inorganic materials, they will not be considered in the present study. Compaction and degradation of the material in the sediment cores would likely not provide an accurate representation of the material at the time of deposition.

With respect to the characteristics of a depositional environment, sediment storage occurs when the energy of the system is low enough to facilitate deposition, and where there is minimal re-suspension as a result of wave action and bed scouring. Lakes typically redistribute sediment from shallower areas towards the deepest point of the basin (Davis, 1968, 1973), also known as "sediment focusing". Consequently, lakes often provide excellent environments for reconstructing historical sedimentation rates and sediment yields. Wetlands have also been recognized as areas of sediment deposition (Johnston, 1991), however, storage in these systems is often temporary. Wetlands regulate the movement of sediment through the watershed and buffer downstream environments against environmental change (Phillips, 1989, 2003).

1.5 Disturbance-response regimes

The processes of sediment transport and delivery under natural and disturbance (e.g. forestry practices) conditions have been reviewed above. As well, the quantity of sediment delivered to the catchment outlet was considered in terms of the sediment delivery ratio, and catchment size and complexity. However, the timing of sediment delivery, and the concept of equilibrium states, have not yet been discussed. Viles et al. (2008) proposed a conceptual model for a generalized geomorphological disturbance response of a system (Fig. 1.1). The model illustrates that there is often a lag between the timing of the disturbance, or "forcing", and the observed response. Lags in the response have been attributed by Viles et al. (2008) to be the result of stabilizing effects, or characteristics and/or processes in the catchment that limit erosion and/or increase sediment storage.

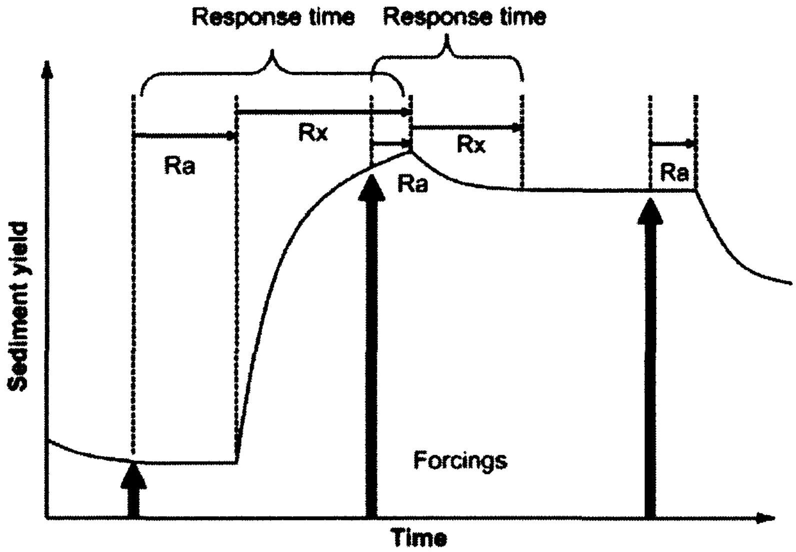


Figure 1.1: Conceptual diagram of a geomorphological disturbance, where: Ra = reaction time; and, Rx = relaxation time. The sum of Ra and Rx equals the response time. Diagram from Viles et al. (2008).

In the conceptual model by Viles et al. (2008) (Fig. 1.1), forcings are intended to represent any disturbance, including storm events, climate change, and human activities. Figure 1.2 (originally from Knox, 1972) has been broken down into several components to illustrate a possible biogeomorphological response to a climate forcing (i.e. fluctuating precipitation). The increased growth of vegetation, as a result of increased precipitation, has an inverse relationship with erosion potential as vegetation growth provides a stabilizing effect by maintaining a strong soil structure through the binding effects of the root network. This biological response thus translates into a negative feedback on sediment delivery by mitigating hillslope erosion. A similar response has been observed in field studies where despite the presence of active logging, water yield did not increase, and was attributed to the rapid reestablishment of vegetation (Paterson et al., 1998).

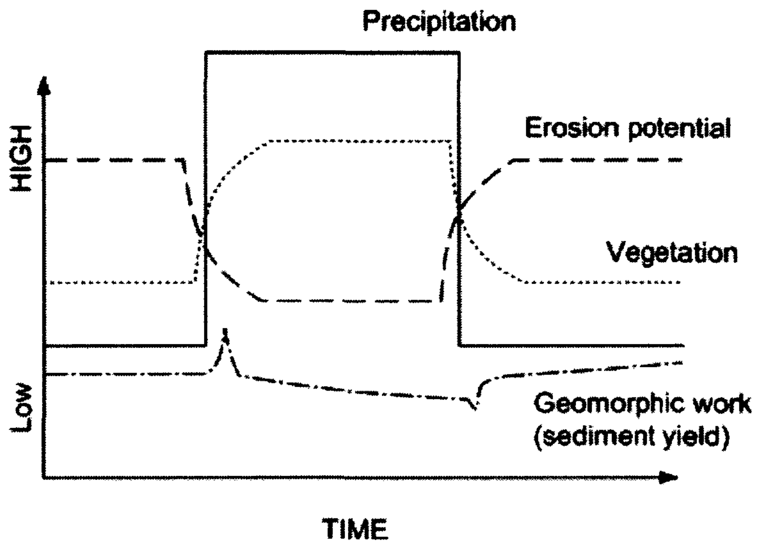


Figure 1.2: Conceptual diagram of a biogeomorphological response to precipitation. Diagram from Viles et al. (2008) (adapted from original by Knox, 1972).

The previous example has shown how stabilizing effects can produce either a lag or dampen the response to a catchment disturbance. However, their ability to do so also depends on the climate regime, the magnitude of the disturbance, and the cumulative impacts of destabilizing effects (i.e. processes that promote erosion). Forcings can also produce alternate stable states wherein the relaxation phase does not return the system to its previous condition and a new level of equilibrium is obtained (Owens et al., 2010). State changes in terms of sediment delivery can relate to either the quantity or quality of the sediment, or both. While the quality of the sediment (i.e. presence of contaminants or elevated concentrations of metals) is typically influenced by human activities (e.g. mining), sediment quantity can be influenced by both human activities and hydroclimatic processes. In a review comparing the effects of landscape disturbance and climate change on erosion, Slaymaker (2001) argued that, with the exception of the polar regions, human land use activities have a much greater impact on global erosion rates than climate change. Bracken & Croke (2007), however, suggested that local climate conditions and storm events provide the conditions required to generate runoff, which in turn mobilize and deliver sediments to downstream waterbodies.

1.6 Forestry practices

The goal of the present study is to evaluate the sediment trapping function in wetlands using historical forestry practices as an indicator of disturbance. Therefore, it is important to understand how forestry practices impact forest hydrologic and geomorphic processes. The following provides a brief overview of forestry practices and their effects on water yield and sediment production.

Undisturbed forests are vital elements in the water cycle as they intercept rainfall, promote infiltration, and contribute water vapour to the atmosphere through evapotranspiration. Spittlehouse (2006a) estimated that 60 to 65% less rainfall reaches river systems when a tree canopy is effective in intercepting rainfall. It has been well documented that the removal of forest cover is strongly related to increases in water yield (Harr et al., 1982; Keppeler & Ziemer, 1990; Stednick, 1996), although this relationship tends to be seasonal and is strongly influenced by precipitation (Bosch & Hewlett, 1982).

The term "forestry practices" is used here to include forest harvest as well as other associated activities. Forestry roads are known to cause soil compaction and reduced water infiltration (Croke et al., 1999), both of which alter the volume and distribution of overland flow (Pike & Scherer, 2003), and change the magnitude and timing of peak flows following storm events. Jones & Grant (1996) found that a 25% patch-cut watershed in the western Cascades, Oregon, USA, with 6% road cover increased peak flows to the same extent as a clearcut watershed. Additionally, peak flows remained 25% greater than before logging and road construction over the following 25 years.

Several studies have identified forestry roads as a major source of fine-grained sediment from logged catchments (Reid & Dunne, 1984). A detailed geochemical analysis of lake sediment in central British Columbia by Christie & Fletcher (1999) revealed that the sediment did not originate from cut blocks, but instead from forestry roads and culverts. Cut blocks have also been reported to increase sediment transfer, however, significant contributions of sediment are more likely to occur as a result of subsequent mass wasting events. Generally, these occur several years post-logging when root networks and other organic debris have decomposed. Furthermore, the impact that logging and roads have on fine-grained sediment production is influenced by landscape position (Tague & Band, 2001), the degree of connectedness to the stream network, and the importance of the road within the overall road network (Sheridan & Noske, 2007).

Ditches and culverts installed under roads are common features in a forest road network and are used to prevent the flooding and erosion of roadways, and to channel the flow of existing streams. As a result, drainage systems also provide direct connections between the road network and fluvial pathways (Croke & Mockler, 2001). Moreover, road construction coupled with the presence of culverts tends to produce channelized pathways which increases the drainage density of the catchment (Wemple & Jones, 1996). This suggests that ditches and culverts would also increase the amount of fine-grained sediment mobilized, and the probability of it reaching the fluvial system.

1.7 Climate

In addition to anthropogenic land disturbances, climate has been shown to influence sediment erosion and delivery (Walling, 1999). Therefore, it is important to consider the effects of varying climatic conditions on sediment retention in wetlands in addition to the effects of historical forestry practices. Event-based processes, such as rainfall and spring freshet can increase the potential for sediment erosion and delivery to occur. During stream monitoring of Fitzsimmons Creek in southwestern British Columbia, Canada, Menounos et al. (2006) found that the highest measured suspended sediment concentration occurred during a rainfall-driven bank-full discharge event. From a paleoenvironmental reconstruction perspective, intense rainfall events or a fast spring freshet could increase the deposition of sediment in downstream wetlands and lakes resulting in greater sedimentation rates for that time period. Alternatively, large decreases in rainfall can also have an impact on sedimentation rates which have been shown to drastically decline during years of little rainfall (Blais et al., 1998).

The magnitude and frequency of precipitation events is often the result of larger scale processes, such as the El Niño/La Niña-Southern Oscillation (ENSO). These processes typically have a longer periodicity and affect a larger area or region. For example, ENSO is a quasiperiodic climate process that occurs approximately every five years. Throughout most of North America, El Niño results in warmer and drier winters and summers, while La Niña produces cooler and wetter winters and summers (Ropelewski & Halpert, 1986). With respect to sediment erosion and delivery, larger scale climatic processes can enhance or subdue the intensity and/or the timing of rainfall events or the spring freshet. Déry et al. (2009) found that a phase change to a cool phase in Arctic Oscillation resulted in increased stream discharge in North American rivers. An intensification of the hydrological regime could provide the necessary energy to increase sediment erosion on the landscape, as well as increase the delivery of sediment to the stream network.

1.8 Research questions and objectives

The following research questions and objectives focus on evaluating the sediment retention function of two wetland buffers in the central interior of British Columbia. To evaluate this wetland function, wetland sedimentation rates, as well as the sedimentation rates of their downstream lakes, were determined using paleoenvironmental reconstruction techniques. The goal of this thesis was to establish if the selected wetland buffers provided the downstream lake with a sediment buffering function, and whether or not that function was compromised by the disturbance caused by upstream historical forestry activities.

1.8.1 Research question 1

1.8.1.1 Wetland and lake sedimentation rates

The first set of research questions relate to sediment storage in wetland buffers. Do wetland sedimentation rates increase after the onset of forestry activities? Also, are sedimentation rates in downstream lakes affected by forestry activities?

These questions address the sediment buffering function of the wetland and its capacity to function "normally" when its contributing catchment area is disturbed by forestry activities. Assuming that the majority of sediment-bearing runoff that reaches the lake must pass through the wetland first, then any increase in lake sedimentation rates would suggest that either the wetland does not perform a sediment buffering function, or that the capacity of that function has been exceeded. The objective was then to calculate sedimentation rates for both the wetland and the lake to characterize baseline sedimentation rates against which post-harvest rates could be compared.

1.8.1.2 Paleoenvironmental reconstruction

One aim of paleoenvironmental reconstruction is to develop a historical dataset consisting of several lines of evidence which guide the interpretation of a system and its changes over time. One of the advantages of using this approach is that a long-term record can be produced for a system for which no monitoring data exist. The natural variability of the system (e.g. lake, wetland) along with responses to catchment disturbances can thus be reconstructed.

Oldfield (1977) developed a conceptual model identifying lakes as ideal environments for reconstructing past conditions. While lakes are not closed environments, they have been shown to focus sediment toward the deepest areas, continuously creating an archive of the material delivered from hillslopes, river channels and the atmosphere, and of the material and organisms produced *in situ*. Retrieval of an intact sediment profile from the deepest point of a lake should therefore provide a long-term record of the conditions in the surrounding catchment area and the lake itself. Other environments, such as wetlands and floodplains, have since been recognized as being able to provide a similar record of long-term change; however, other features and processes (e.g. vegetation, water level) affect the temporal and spatial distribution of sediment in these systems.

The collection and interpretation of the long-term dataset require the measurement of proxy indicators which, as mentioned earlier, allow inferences to be made about historical environmental conditions based on current knowledge of these indicators, and the processes that control their behaviour in a particular environment (Smol, 2008, 2010). The selection of a set of proxy indicators should be driven by the research questions and objectives. Commonly used proxies include: bulk physical characteristics; mineral magnetic properties; grain size and shape; geochemical and nutrient concentrations; radionuclides; isotopic tracers; and, remnants of organisms which are not susceptible to physical breakage or chemical dissolution (Smol, 2008). In order to understand the changes of these measurements in a temporal context, core chronologies must be developed. Core chronologies can be developed using radionuclide activities found in the sediment; most frequently used is the unsupported component of ^{210}Pb.

The development of a dataset using paleoenvironmental reconstruction techniques is a powerful tool that can provide information on long-term environmental change, as well the impacts of human activities on natural processes (e.g. sediment delivery), and environmental quality. A long-term perspective is especially important when attempting to establish guidelines for restoration projects (Foster et al., 2011), as guidelines based on recent monitoring data may be inaccurate due to recent disturbances, cumulative effects, and the establishment of alternate stable states (Owens et al., 2010).

1.8.2 Research question 2

1.8.2.1 Changes in sediment provenance

The second research question pertains to the source of the sediment being delivered to the wetlands and lakes. Do the relative contributions of sediment source materials identified within each catchment area change as a result of forestry activities?

If sedimentation rates in either the wetland, lake, or both are altered, then it is important to evaluate a suite of alternate, potential drivers of that change, since it may not be related to forestry practices. Climatic factors, such as precipitation, are known to play a significant role in sediment delivery, sometimes having a larger impact on sedimentation rates than landscape disturbances (Blais et al., 1998). One would suspect that if forestry practices were driving increased sedimentation rates, then an increase in the relative proportion of subsurface soil material would occur as a result of erosion of surface soil exposing underlying subsurface materials (Thompson et al., 1975). Similarly, the construction and use of roads would increase the delivery of subsurface sediment. By identifying and characterizing the sediment sources in the catchment, it is then possible to determine the relative contribution of each to the wetland and lake, and how they change over time relative to the timing of the disturbance. However, this assumes that the fluxes of sediment from sites of intermediate storage do not change.

1.8.2.2 Sediment source tracing

Sediment source tracing has been used in many studies to identify the impact of different land use types on contemporary suspended sediment loads (Walling & Woodward, 1995; Collins et al., 1998). Long-term studies have also been carried out which used sediment stored within depositional environments, such as floodplains (Owens et al., 1999), to reconstruct historical changes in sediment sources over time. Figure 1.3 illustrates the main principles on which the sediment source tracing process is founded. The diagram has been adapted to match the research questions raised by this study.

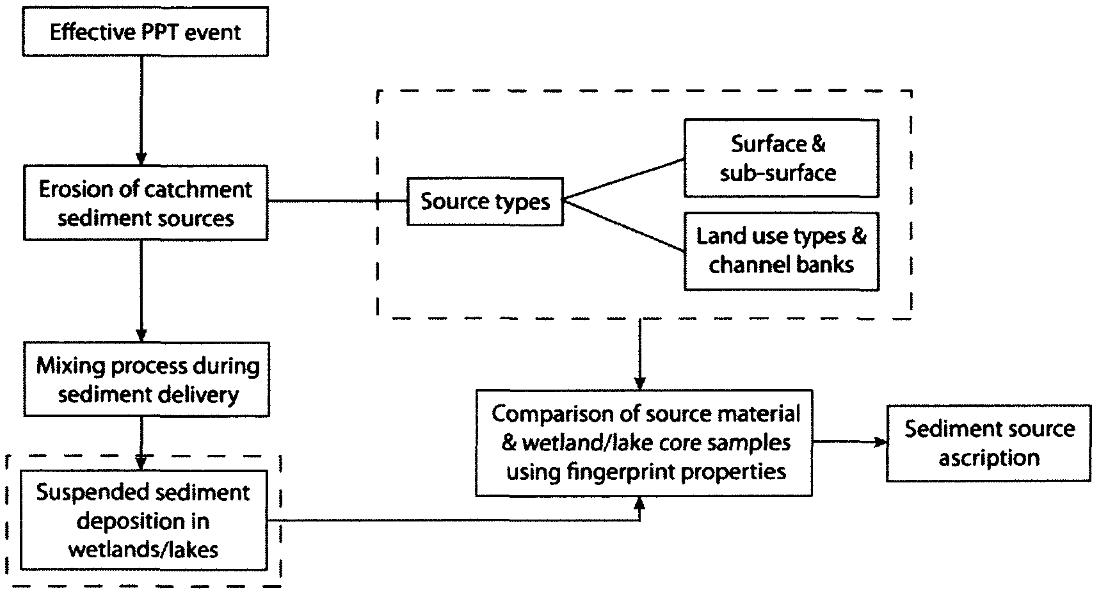


Figure 1.3: A conceptual diagram outlining the sediment fingerprinting approach. Adapted from Collins & Walling (2002) to this wetland focus.

The process of sediment source tracing requires that sediment source types can be differentiated according to their fingerprint properties. A simple, and commonly used set of source groups consists of: surface soil material; subsurface soil material; and, channel bank material. Where well-defined land use activities are present in the catchment (e.g. agriculture, mining, urban areas and roads), many studies rely upon an *a priori* selection of source groups. Others have used statistical methods to verify the accuracy of the selected source groups with respect to the selected fingerprint properties (Hatfield & Maher, 2009). It is also important to consider the underlying bedrock, and if source areas extend beyond a single bedrock type as this could impact the geochemical and mineralogical composition of the eroded sediment (Walling & Woodward, 1995).

Various fingerprint properties have been used to characterize source groups, including soil geochemistry (Foster, 1994), radionuclides (Walling et al., 1993), nutrients (Walling et al., 2008), mineral magnetic properties (Yu & Oldfield, 1989), and colour (Martínez-Carreras et al., 2010). The selection of a set of properties should be driven by the characteristics of the catchment(s) being studied, surrounding land use types, and ultimately the research questions being asked. A composite fingerprint is then statistically selected from the full set of fingerprint properties. Ideally, the final composite fingerprint consists of several parameters from more than one property type (Collins & Walling, 2002). Once an appropriate composite fingerprint has been been identified, a multivariate unmixing model can be used to calculate relative contributions of each source material.

If a paleoenvironmental approach is being taken then the nature of the properties must also be considered in concert with the characteristics of the depositional environment. The challenge with reconstructing sediment source contributions over time is that fingerprint properties behave differently after deposition than when suspended in a water column (Owens et al., 1999). Post-depositional changes (or diagenesis) such as decomposition, physical mixing and bioturbation alter the nature of the fingerprint properties and their vertical distribution in the sediment profile. It is therefore necessary to select properties which behave conservatively not only during transport, but also in a depositional environment (Motha et al., 2002). Physical, mineral magnetic, radionuclide and geochemical properties tend to be conservative in a sedimentary environment, and have been widely used in both contemporary and historical studies. However, due to sorting effects during transport differences in grain size need to be accounted for by targeting a specific size fraction (e.g. <63 μm) for analysis (Carter et al., 2003; Foster et al., 2008), and by including a particle size correction in the unmixing model (Collins et al., 1997).

1.9 Thesis organization

The chapters of this thesis on the effects of historical forestry practices and climate on the sediment retention function of two wetlands in the central interior of British Columbia have been organized according to a traditional thesis style which follows the scientific method. Chapter two describes the study sites and methodology used to address the research questions. Chapters three and four summarize the results from the paleoenvironmental reconstruction and sediment source tracing procedures, respectively. Chapter five provides a discussion on each of the study sites by interpreting the results in chapters three and four simultaneously. Study limitations and future research directions are given at the end of chapter five. Finally, a conclusion is given in chapter six, along with management implications and final remarks.

Chapter 2

Methodology

2.1 Study area

Located in the Cariboo Mountains (mean elevation of 1,375 m above sea level) in the central interior of British Columbia (Fig. 2.1), the Quesnel River Basin is composed of three watersheds: the Quesnel River Watershed, the Cariboo River Watershed, and the Horsefly River Watershed. All three watersheds have a combined area of approximately 12,000 km^2. The land area within the basin has been used historically and at present for various resource extraction activities including forestry, agriculture, ranching and mining.

The two selected study wetlands were chosen not only because their catchments have a history of forestry practices, but also because they border another depositional environment (i.e. a lake). Since the inflows of these lakes are surrounded by wetlands, they may have been provided with a buffering function which would have influenced the amount of sediment delivered to them. Since the project aims to establish the ability of the two study wetlands to promote sedimentation under disturbance conditions, it was necessary to also evaluate lake sedimentation rates over time.

2.1.1 Boswell Lake catchment

The Boswell Lake catchment (52°32'25"N, 121°27'6"W; see Figure 2.2) is situated in the Quesnel River Watershed, and has an area of 2.1 km^2. According to the Biogeoclimatic Ecosytem Classification (BEC) System, the catchment is located in an interior cedar hemlock zone characterized by a wet and cool climate (British Columbia Ministry of Forests and Range, 2008). Mean annual temperature for this BEC zone ranges from 2 to 8.7 °C, mean

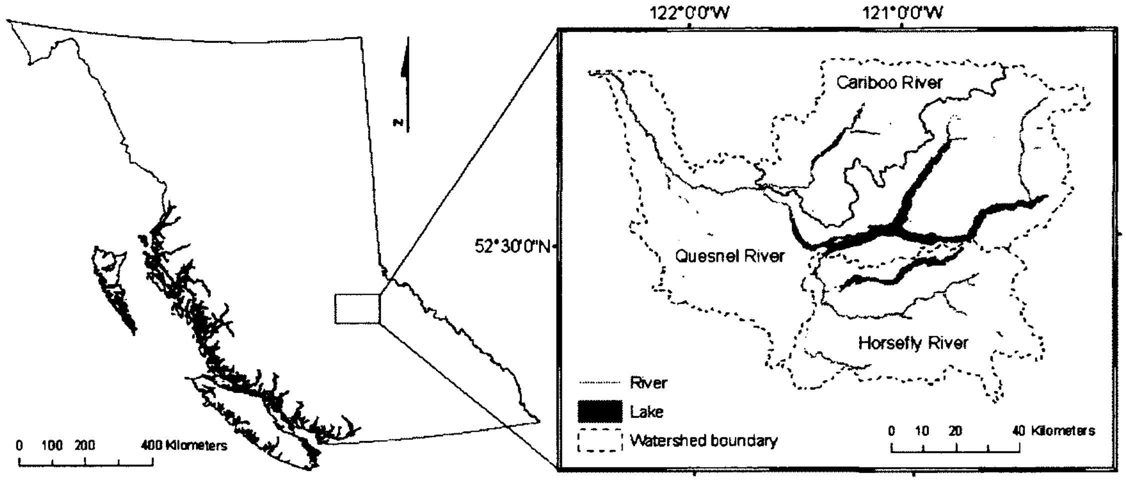


Figure 2.1: Map of the province of British Columbia. The rectangle indicates the approximate location of the Quesnel River Basin which is composed of three watersheds: the Cariboo River Watershed; the Quesnel River Watershed; and, the Horsefly River Watershed.

annual precipitation is 500-1200 mm, 25-50% of which falls as snow (Ketcheson et al., n.d.). The local bedrock geology consists of basaltic volcanic rocks from the Upper Triassic period (Massey et al., 2004). The maximum and average slopes of the catchment are 38° and 12°, respectively.

The wetland bordering Boswell Lake (herein referred to as Boswell wetland) is situated at the inflow of the lake, and has a surface area of 0.020 km^2 and a maximum width of approximately 85 m (measured from the wetland-land border to the lake edge). Four channels cross the wetland border flowing in a south to north direction from the deforested areas to Boswell Lake, and meandering through the wetland is low. Two of the four channels were identified as "major" channels, and the other two were labelled as "minor", based on their size, degree of inundation, and connectivity to the logged slopes. The two major channels were flowing during visits to the lake in the late spring, summer and fall, while flow in the minor channels was only observed in the spring and not during any other visit to the study site. Water depths in the major channels ranged from 10 to 65 cm with increasing depths downstream. The dominant vegetation types in the wetland and the wetland channels are sedges (*Carex spp.*) and Yellow Water Lilies (*Nuphar variegata*). Sedges form the bulk of the vegetation in the wetland and were primarily observed closest to the lake edge where the wetland channels became diffuse. Based on these characteristics the Boswell wetland has been identified as a fen (MacKenzie & Moran, 2004).

Forestry activities occurred in the catchment during two separate time periods. During the first period from 1960 to 1975, approximately 42% (0.873 km^2) of the catchment was clearcut. From 1982 until 2008 clearcut logging affected another 15% (0.324 km^2) of the catchment. Currently a 1.4 km active gravel road crosses through the catchment on the north side of Boswell Lake. A 14.4 km network of deactivated dirt roads associated with the first logging period (1960-1975) also exists on the south side of Boswell Lake. Field observations confirmed that these dirt roads are no longer in use as there is substantial vegetative growth along these roadways.

2.1.2 Viewland Lake catchment

The Viewland Lake catchment (52°25'44"N, 121°6'57"W; see Figure 2.3) is located in the Horsefly River Watershed, and has an area of 2.5 km^2. It is located in an interior cedar hemlock zone having a wet and cool climate (British Columbia Ministry of Forests and Range, 2008). Mean annual temperature ranges from 2 to 8.7 °C, mean annual precipitation is 500-1200 mm, 25-50% of which falls as snow (Ketcheson et al., n.d.). The local bedrock geology is composed of two groups; sedimentary rocks from the mid-to-Upper Triassic period, and basaltic volcanic rocks from the Upper Triassic period (Massey et al., 2004). Maximum and average slopes of the catchment are 35 ° and 7 °, respectively. The Viewland Lake catchment area drains into three lakes all connected by a single channel running between them from north to south. The top two lakes are each fed by a channel that originates from the cutblock.

The wetland bordering Viewland Lake (herein referred to as Viewland wetland) runs along the entire east side of the lake chain, and has a surface area of 0.074 km^2 and a width of approximately 30 m (measured from the wetland-land border to the lake edge). During site visits in late summer and autumn it was observed that the channel through the Viewland wetland was not flowing. It is currently unknown whether or not this channel experiences any degree of flooding during the year. Channel meandering was also found to be relatively low. Sedges were observed to be the dominant vegetation type in the wetland channel. Similar to Boswell wetland, sedges were densest near the lake edge. Yellow water lilies were also present, however, they only occurred near the lake edge where flooding was present. Based on these characteristics the Viewland wetland has been identified as a fen (MacKenzie & Moran, 2004).

Forestry practices in the Viewland Lake catchment occurred only in 1983 resulting in deforestation of 58% of the catchment. A deactivated unpaved road is present just east of Viewland Lake which crosses over the channel that flows into the lake. As the road has not yet been decommissioned, a culvert from the inital construction of the road still remains in place. Total road length in the Viewland Lake catchment is approximately 6.9 km.

Morphometric characteristics of both Boswell Lake and Viewland Lake can be found in Table 2.1. Their bathymetric maps are presented in Appendix A.

Table 2.1: Morphometric characteristics of Boswell Lake and Viewland Lake. Information for Boswell Lake was taken from The Angler's Atlas (2010).

Measurement	Boswell Lake	Viewland Lake
Catchment:Lake Area	0.06	0.03
Surface Area (km^2)	0.128	0.073
Volume (m^3)	148,000	219,030
Mean depth (m)	1.2	3.0
Maximum depth (m)	2.5	8.2
Perimeter (m)	1850	1468

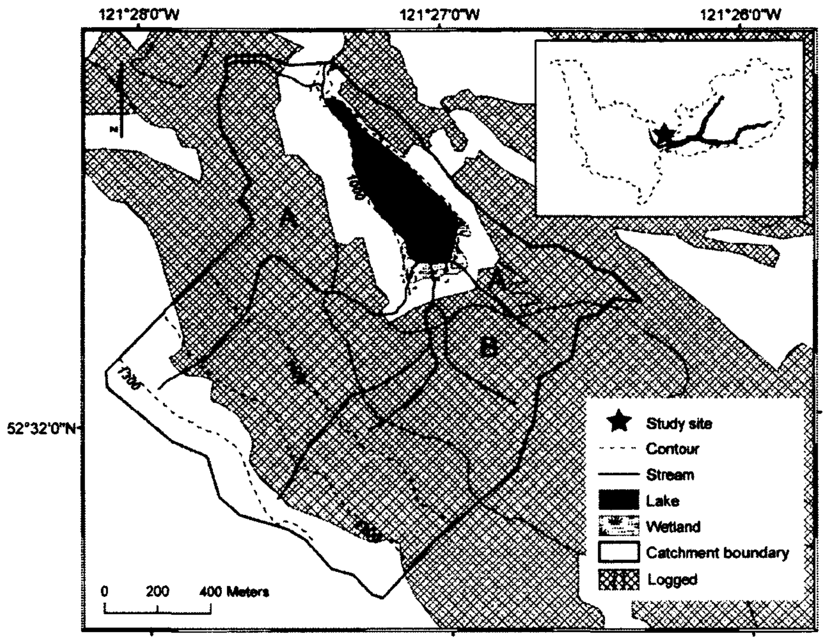


Figure 2.2: Map of the Boswell Lake catchment. Forestry practices were active in the catchment during two time periods: A) 1960-1975; and B) 1982-2008. Inset: Outline of the Quesnel River Watershed. The star represents the approximate location of the Boswell Lake catchment in the watershed.

2.2 Sample collection and preparation

2.2.1 Wetland coring

Initially Boswell Lake was intended to be the primary study site with Viewland Lake acting as a secondary study site in the event that the cores taken from the primary site did not produce a useful core chronology. As a result, a more detailed sampling campaign was undertaken at Boswell Lake and wetland, and only one of the three lakes in the Viewland catchment was selected for coring. The middle lake was chosen because its stream drains a larger area that was impacted by forestry practices (see Fig. 2.3), and it was thought that a stronger logging signal may be observed in the middle lake.

Six sampling locations were identified for Boswell wetland and one for Viewland wetland from which a single core was taken (Fig. 2.4(a)). Since wetlands typically do not possess a single deepest point where sediment focusing will occur (unlike many lakes), it was necessary

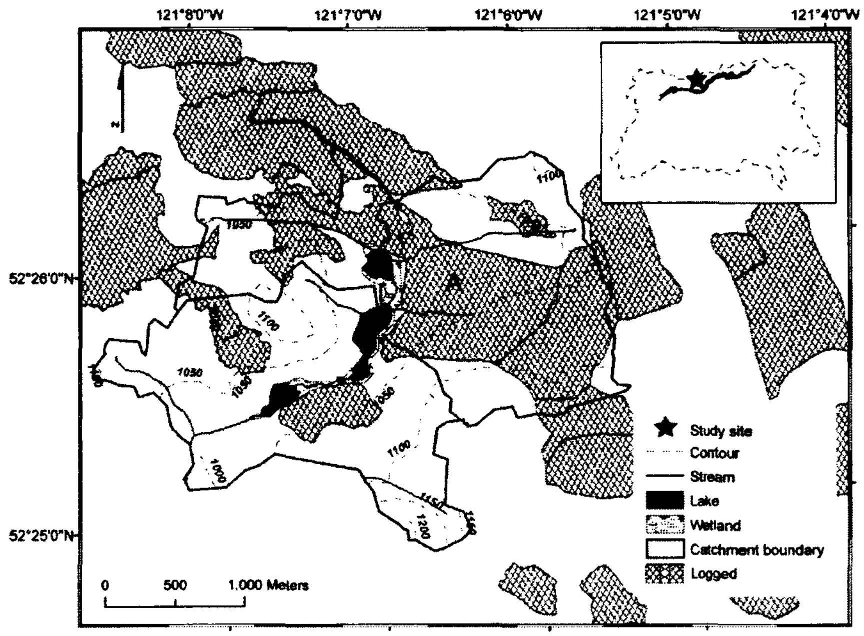


Figure 2.3: Map of the Viewland Lake catchment. Forestry practices were active in the catchment in 1983 (A). Inset: Outline of the Horsefly River Watershed. The star represents the approximate location of the Viewland Lake catchment in the watershed.

to identify areas in the wetland which, based on physical characteristics such as inundation and channelization, likely experienced the greatest sedimentation rates. Four channels were identified in Boswell wetland, and one in Viewland wetland, from which the sediment cores were taken. As both wetlands were not completely inundated, sediment transport and deposition was assumed to have occurred primarily along these pathways. This assumption is consistent with the observations of Craft & Casey (2000) who found that "open" (e.g. riparian and floodplain) wetlands had greater sediment accumulation rates than "closed" (e.g. depressional) wetlands; where open and closed refer to the degree of connectivity to the hillslope and surface water bodies.

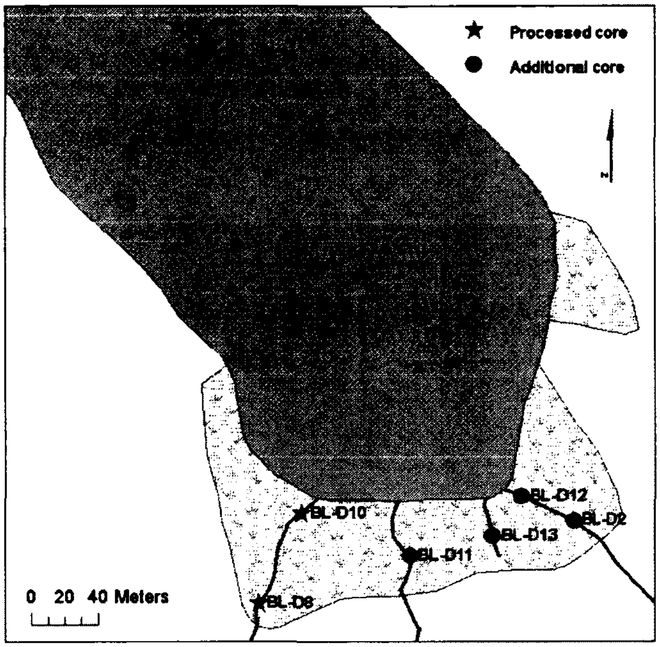
At Boswell wetland, sampling areas were selected near the wetland inflow and outflow to characterize the sedimentation rates along each of the major channels (i.e. four sampling locations in total). Based on the characteristics of the two minor channels, described above, these pathways were considered to be relatively less important for sediment delivery and these two sampling locations were not analyzed in the current study. In the case of Viewland wetland, only one sampling location was selected as only one channel exists between the deforested area and the middle lake (Fig. 2.4(b)). Core lengths were dependent on the characteristics of the sediment at each site and ranged between 0.25 and 1.0 m. Refer to Figure 2.4 for a map of the sampling locations at both sites. All wetland coring took place during July and August 2009.

As this study is primarily concerned with contemporary sedimentation rates it was crucial to obtain profiles with intact upper sediment layers. It was decided that the open-barrel coring method was therefore more appropriate than other methods (i.e. Russian Peat Corer) as it minimally disturbs the top of the sediment profile (Glew, 2001). Using 2 m lengths of PVC piping, 7.6 cm in diameter, two 3.2 cm holes were drilled approximately 2.5 cm from the top of the PVC pipe. A metal rod 2.5 cm in diameter and 50 cm long was fit through the top holes to provide a handle to assist in core removal (Fig. 2.5). This corer design was adapted from Reinhardt et al. (2000).

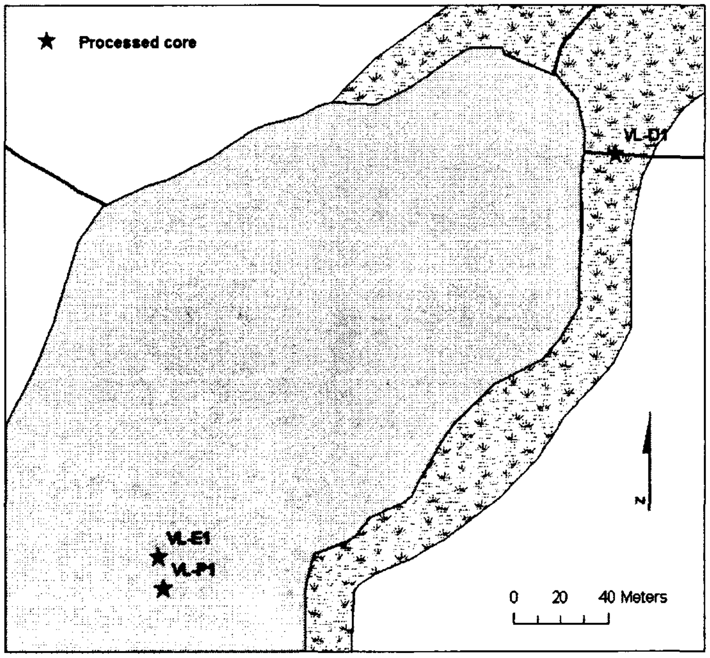
2.2.2 Lake coring

For each of the two study sites, one core was retrieved from the deepest point of the lake using a percussion corer (Reasoner, 1993). A core catcher constructed from stove pipe metal was fixed in the bottom of the core tube to prevent captured sediments from being lost during retrieval. An additional short core was taken using an Ekman dredge to ensure that an undisturbed sample of the water-surface interface was taken. Coring at Boswell Lake occurred in October 2009. Since Viewland Lake does not have direct vehicle access, cores were retrieved in March 2010 when there was sufficient ice cover to provide a stable coring platform.

All lake and wetland cores were transported back to UNBC where they were stored at a temperature of approximately 4 °C to prevent decomposition of organic matter and any changes that may be associated with exposing anoxic soil to oxidizing conditions. Cores



(a) Boswell Lake and wetland



(b) Viewland Lake and wetland

Figure 2.4: (a) Boswell Lake and (b) Viewland Lake and wetland coring locations. Codes containing a 'D' indicate an open barrel core, 'P' refers to a percussion core, and those with an 'E' denote a core taken with an Ekman dredge. Note: The location of the stream containing Boswell wetland core BL-D13 was not shown in original spatial dataset. This line feature was created by extracting point locations from a Google Earth image of the catchment.

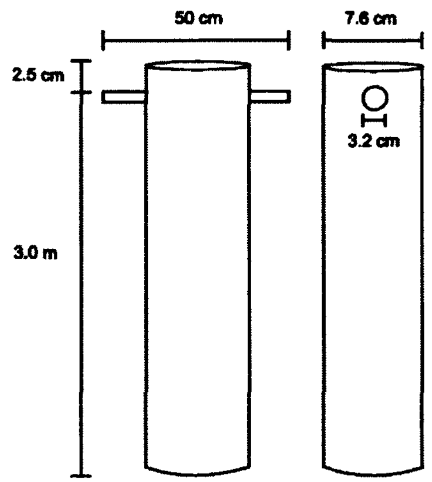


Figure 2.5: Diagram of the open barrel corer used to retrieve wetland sediment cores.

that were selected for further analysis were cut length-wise, photographed and logged prior to slicing the sediment at 1 cm intervals.

2.2.3 Source materials

To assess if forestry practices resulted in altered sediment source contributions and changes in the dominant sediment sources, sediment samples were collected from six source types: harvested surface soil material; harvested subsurface soil material; forested surface soil material; forested subsurface soil material; road surface soil material; and channel bank material. Approximately 5-8 samples were taken for each source type and a GPS coordinate was recorded for each sampling location. Samples were taken with a stainless steel trowel which was rinsed with distilled water and acetone between each sample to minimize cross-contamination. Each sample was itself a composite of 3-5 subsamples collected within an area of approximately 5 m by 5 m to account for any local spatial variability.

All source materials were air dried prior to laboratory analysis. Samples that contained moisture after air drying were placed in an oven at 60 °C until dry. Source materials were then disaggregated and sieved to <63 μm. Analysis of this particle size fraction was intended to minimize the differences in the particle size composition between core sediment and source material (Carter et al., 2003; Motha, 2003) as most of the lake and wetland sediment was <63 μm.

2.3 Radionuclides and core chronology

2.3.1 Origin of lead-210

Reconstructing sediment chronologies over the last 100-150 years requires the use of a radionuclide which has a relatively fast decay rate. Lead-210 (^{210}Pb) has a half-life of 22.26 years and is ubquitous in the environment as a result of the natural decay of ^{238}U in bedrock (Binford, 1990). Following the decay of ^{238}U to ^{226}Ra, ^{226}Ra then decays to ^{222}Rn. ^{222}Rn forms a gas which escapes to the atmosphere, and through several additional decays, becomes ^{210}Pb. In order for ^{210}Pb to fall out of the atmosphere it needs to adsorb onto atmospheric particulates and/or water droplets which are typically delivered to land and water surfaces via precipitation. In the water column ^{210}Pb binds to fine particles and organic material and is deposited on the bottom of the water body (e.g. ocean, lake, river, wetland). This fraction of ^{210}Pb is referred to as *unsupported* ^{210}*Pb*.

^{222}Rn is also produced in the soil which decays through the same decay series resulting in the *in situ* production of ^{210}Pb. This is known as *supported* ^{210}*Pb* (Binford, 1990; Noller, 2000). Unsupported ^{210}Pb is calculated as the difference between the total ^{210}Pb and estimates of the supported component.

2.3.2 Lead-210 dating models

Several models exist that utilize the unsupported fraction of ^{210}Pb to assign chronologies to sediment core profiles. Most commonly used are the Constant Initial Concentration (CIC) and Constant Rate of Supply (CRS) models. The CIC model assumes that the initial concentration of unsupported ^{210}Pb remains constant over the time that unsupported ^{210}Pb is measurable (Turner & Delorme, 1996). As a result, the log transformation of unsupported ^{210}Pb activities should yield a linear decrease over depth. When the CIC model is applied to a non-monotonic decay curve, resultant core chronologies include one or more time inversions. These occur because other processes acting on the sediment profile have impacted unsupported ^{210}Pb activities leading to an imperfect decay curve. For example, Appleby et al. (1988) found that due to organic matter degradation and loss over time, the CIC model was not suitable for peat cores. Sediment dates (t) can be determined with the CIC model using:

$$t = \frac{1}{\lambda} \ln \frac{C_x}{C_o} \tag{2.1}$$

where λ is the radioactive decay constant for ^{210}Pb, C_x (Bq kg^{-1}) is the activity of unsupported ^{210}Pb at depth x; and C_o (Bq kg^{-1}) is the activity at the surface.

The CRS model assumes that the absolute flux rate of ^{210}Pb remains constant, regardless of background sedimentation, such that higher rates of background sedimentation will lead to lower ^{210}Pb concentrations (Appleby & Oldfield, 1978). Unlike the CIC model, it is able to account for fluctuations in unsupported ^{210}Pb sedimentation which may have occurred either in response to climatic variations or anthropogenic disturbance (Brenner et al., 1999; Cohen et al., 2005). Futhermore, inversions in the unsupported ^{210}Pb profile may be better explained by a dilution effect of unsupported ^{210}Pb due to an increase in sedimentation rates. However, one limitation of the CRS model is that it tends to over-estimate sediment ages near the bottom of the profile. Sediment ages (t) based on the CRS model can be calculated by:

$$t = \frac{1}{\lambda} \ln \frac{A_x}{A_o} \tag{2.2}$$

where A_x is the inventory of unsupported ^{210}Pb (Bq m^{-2}) to depth x; and A_o is the total inventory of unsupported ^{210}Pb (Bq m^{-2}).

Other more computationally intensive models have been proposed, such as the Sediment

Isotope Tomography (SIT) model. Unlike the other dating models, the SIT model allows both the absolute flux rate of ^{210}Pb and the sedimentation rate to vary (Carroll et al., 1995; von Gunten et al., 2008). Another difference between the SIT model and other conventional ^{210}Pb dating models is that it reconstructs the unsupported ^{210}Pb activity profile before calculating a core chronology. This is accomplished by modelling nonexponential changes in unsupported ^{210}Pb with a Fourier sine series, while any additonal changes caused by other processes are modelled with a Fourier cosine series. A ^{210}Pb profile is selected when a pre-determined measure of fit (χ^2) is achieved which compares the modelled profile to the original unsupported ^{210}Pb profile (Carroll & Abraham, 1996). See Table 2.2 for a summary of all model assumptions.

Table 2.2: Summary of model assumptions. Adapted from Carroll & Lerche (2003).

Model name	Specific activity	Accumulation rate	Flux of ^{210}Pb
Constant Initial Concentration	constant	variable	variable
Constant Rate of Supply	variable	variable	constant
Sediment Isotope Tomography	variable	variable	variable

2.3.3 Caesium-137

Ideally, paleoenvironmental studies should not rely on a single dating model, and would employ the use of a marker horizon to confirm the constructed chronology. The most commonly used marker horizon is caesium-137 (^{137}Cs). ^{137}Cs is an artificial fallout product of atmospheric bomb testing that began in the early 1950s and ended in the early 1970s. Peak fallout of ^{137}Cs as a result of atmospheric bomb testing occurred in 1963 (Owens et al., 1996), and is often represented in sediment profiles as a peak in down-core measurements. The location of the 1963 ^{137}Cs peak, as well as the onset of increasing ^{137}Cs concentrations (1954), can be used paleolimnological studies to verify the accuracy of core chronologies (von Gunten et al., 2008). Good agreement between these peaks and the location of the modelled dates in the profile (i.e. using unsupported ^{210}Pb) should indicate that the dating model is appropriate for that environment. A secondary ^{137}Cs peak produced by the explosion of the Chernobyl nuclear reactor in 1986 has also been used for the same purpose, but is not detected within western Canada. Post-depositional processes, such as mixing, can impact the ^{137}Cs profile (He & Walling, 1996; Foster et al., 2006). Other studies have used *Ambrosia* pollen (Blais et al., 1995), tephra (Reasoner & Healy, 1986), stable lead (Blais et al., 1998), and other metals and contaminants (Cooke & Abbott, 2008) to mark known historical events and verify core chronologies.

2.3.4 Core chronology

Since several cores were taken from Boswell wetland and not all could be analyzed due to time and financial constraints, it was necessary to select representative for laboratory analysis. Cores BL-D8 and BL-D10 from the far west channel were selected as the key wetland cores (Fig 2.4(a)). Several attempts were required to retrieve cores from the other major stream which likely disturbed and redistributed the top sediments contaminating other coring sites. Cores BL-D8 and BL-D10 were successfully removed on the first attempt minimizing the disturbance and redistribution of top sediments.

Lake cores (BL-P1 and VL-P1) and selected wetland cores (BL-D8, BL-D10 and VL-D1) were analyzed for ^{210}Pb and ^{137}Cs. ^{210}Pb, with a half-life of approximately 22.26 years, was decided to be the most appropriate radionuclide for constructing a core chronology for the last 100-150 years[1]. Core chronologies were calculated using the CRS, CIC, and the SIT models. Details of all models are found in Section 2.3.2. Software for the SIT model was provided by Dr. J. Carroll of the Polar Environmental Centre, Norway. The final core chronology was selected based on the model whose assumptions were satisfied, and produced the smallest date errors (i.e. error bars). As it is assumed that similar processes are acting on these systems, a single dating model was selected for all cores. The results of the ^{210}Pb dating models were also compared to ^{137}Cs activities to verify model accuracy.

[1]Most equipment for measuring radionuclide activities is only able to detect radionuclides up to 4-5 half-lives

Approximately 1-3 g of sediment from each 1 cm core section was packed into a 4 mL plastic vial and left for three weeks to allow equilibrium to be reached between ^{214}Pb and its parent radioisotope ^{226}Ra (Köhler et al., 2000). Measurements of total ^{210}Pb, supported ^{210}Pb and ^{137}Cs activities were undertaken at the Plymouth University Consolidated Radioisotope Facility, England, UK using a EG&G Ortec well (GWL-170-16-S N-type) HPGe Gamma spectrometry system over a period of 24 to 48 hours for each sample. Longer measurement times were necessary to minimize the higher error associated with lower sample masses[2].

2.4 Proxy measurements

Proxy measurements were used to compile information on the physical and chemical characteristics of the sediment trapped by the study wetlands and lakes over the last century. The data provided multiple lines of evidence for understanding the nature of the material captured by both wetland and lake environments and, therefore, the type of material being mobilized from the hillslopes. Dry bulk density and percent water content were also used in conjunction with magnetic susceptibility to match age-equivalent sediment layers in overlapping sections of cores from each of the two lakes (i.e. Ekman and percussion cores) and create contiguous lake sediment profiles (Snowball & Sandgren, 2001). This process is similar to that of core correlation which aims to match cores taken from various coring locations so that chronologies may be extended to undated cores (Foster et al., 1985). The Boswell Lake core is therefore a combination of Ekman (BL-E1) and percussion (BL-P1) cores, however, it will be referred to as BL-P1. Similarly, the Viewland Lake core is a combination of cores VL-E1 and VL-P1, but will be referred to as VL-P1.

[2]The ideal mass for gamma spectrometry is 5 g. However, this mass could not be reached with the material retrieved from any of the sediment cores as they were highly organic and had low clastic contents.

2.4.1 Bulk physical properties

Changes in dry bulk density and percent water content were used to provide information on the type of material being delivered to the lake and wetland. Increases in dry bulk density may be indicative of more minerogenic material which has been previously linked to the mobilization of subsurface soil material (Thompson et al., 1975). Each 1 cm section of sediment was placed in a pre-weighed plastic WhirlPak bag and re-weighed. Bags of sediment were frozen at $-10\,^{\circ}\mathrm{C}$ and subsequently placed in a freeze drier for approximately 72 hours to remove all moisture. The bags were then re-weighed to determine the mass of dry sediment. Dry bulk density and percent water content were calculated according to Equations 2.3 and 2.4, respectively.

$$\text{Dry bulk density} = \frac{\text{Dry mass (g)}}{\text{Volume (cm}^3\text{)}} \tag{2.3}$$

$$\text{Percent water content} = \frac{\text{Wet mass (g) - Dry mass (g)}}{\text{Wet mass (g)}} \cdot 100 \tag{2.4}$$

2.4.2 Magnetic susceptibility

Magnetic susceptibility is a measure of the concentration of magnetic minerals in the sediment, or the clastic content of the sediment. A large positive magnetic susceptibility value indicates that the materials in the sediment maintain a magnetic charge after the sediment has been exposed to a magnetic field. Conversely, low or negative magnetic susceptibility values indicate that the materials in the sediment do not maintain a magnetic charge after exposure to a magnetic field. For example, iron-bearing minerals have a high magnetic susceptibility values while wood and other plant materials have low values (Nowaczyk, 2001). Trends in down-core magnetic susceptibility have previously been linked to the timing of deforestation and erosion of minerogenic soils (Thompson et al., 1975), and were used for the same purpose in the present study. Magnetic susceptibility was measured in triplicate at each

1 cm interval for all wetland and lake cores using a Bartington MS2 Magnetic Susceptibility System at the University of Northern British Columbia. All magnetic susceptibility measurements were normalized by sediment mass to give mass-specific magnetic susceptibility (Sandgren & Snowball, 2001).

2.4.3 Particle size

Particle size analysis was completed for all lake and key wetland cores as well as all source materials of the <63 μm particle size fraction. Variations in element concentrations may be related to grain size and must therefore be taken into account in the mixing model (described further below). Down core changes in particle size distribution have also been linked to historical changes in land cover and human activities (van Hengstum et al., 2007). Analysis of pre-logging conditions will provide background particle size distributions and their natural variations against which periods of forestry practices and post-logging can be compared. Particle size analysis could not be completed for several slices (6, 7, 11-14 cm) of the Viewland wetland core (VL-D1) as not enough inorganic material was present in these 1 cm core slices to reach the recommended degree of obscuration[3] (Sperazza et al., 2004).

Sediment samples were pre-treated with 30% hydrogen peroxide and heated to approximately 70 °C to digest organic material. Approximately 10 mL of a 0.55% sodium hexametaphosphate solution, $(NaPO_3)_6$, was added to each sample to promote dispersal of the individual sediment grains and prevent flocculation (Sperazza et al., 2004). Samples were stirred for approximately 30 seconds prior to particle size analysis to resuspend particles into the water column. Particle size distributions were determined using a Mastersizer 2000 laser diffractometry analyzer in the Department of Earth Sciences laboratory at Simon Fraser University, Burnaby, BC.

[3]Obscuration is a measure of the quantity of sediment added to the analyzer. Between 15 and 20% obscuration has been recommended to minimize variability of results.

2.4.4 Total carbon and nitrogen

As a result of low sample masses for each core section, measurements of total carbon (C) and total nitrogen (N) were used in lieu of organic matter content. Dry sediment samples (ca. 0.05 g) from each 1 cm core section were sent to the Forestry and Technical Services laboratory (Ministry of Forests and Range) in Victoria, BC for analysis of total C and total N content. A C:N ratio was then calculated from the total C and N percentages for each 1 cm core section to provide additional information on the source of organic matter. Typically C:N values between 4 and 10 represent organic matter derived from phytoplankton. Values greater than or equal to 10 are more indicative of vascular terrestrial vegetation (Meyers & Teranes, 2001; Kim, 2003).

2.4.5 Geochemistry

A suite of 34 geochemical properties[4] were measured for all lake and wetland cores (except Boswell wetland core BL-D10), as well as all source materials. BL-D10 was not analyzed for geochemistry due to a miscommunication regarding sample priorities in other analyses and unavoidable time constraints. The geochemical properties then became the fingerprint properties used in the sediment source tracing procedure. Dry sediment samples were prepared for geochemical analysis by adding concentrated acid (5 mL HNO_3 and 1 mL HCl) and further digesting the samples in a microwave digester. Digested samples were diluted with Milli-Q water such that the total volume equalled 50 mL. Samples were analyzed by Inductively Coupled Plasma Atomic Emission Spectrometry (ICP-AES) using a Leeman PS1000-UV to determine element concentrations. Sample preparation and ICP-AES analyses were completed in the Central Equipment Laboratory at the University of Northern British Columbia.

[4]The 34 geochemical properties measured were: lithium; beryllium; sodium; magnesium; aluminum; silicon; phosphorus; potassium; calcium; titanium; vanadium; chromium; manganese; iron; cobalt; nickel; copper; zinc; arsenic; selenium; strontium; zirconium; molybdenum; silver; cadmium; tin; antimony; barium; tungsten; mercury; thallium; lead; bismuth; and, uranium.

2.5 Climate and stream discharge

Since sediment transport typically occurs during large precipitation events or spring snowmelt, historical climate data were compiled to determine whether any fluctuations in lake or wetland sedimentation rates could be explained by variations in weather patterns. A climate modelling program, ClimateBC, has been developed by British Columbia's Ministry of Forests and Range to estimate historical and future climate variables. This program requires a user input of latitude, longitude and elevation for the area of interest (Spittlehouse, 2006b). It calculates climate variables for that location by interpolating between existing weather stations. Although this program was created to produce climate data for areas where weather stations are limited, this in turn has become a limitation for the model itself. ClimateBC has been found to provide poor climate predictions for areas not well-covered by weather stations (Spittlehouse, 2006b). Despite this limitation, data calculated by ClimateBC span the full temporal range of the dated cores, and was thus chosen over the incomplete datasets from Environment Canada[5].

Latitude, longitude and elevation data were extracted from a digital elevation model for every 30 m grid in each catchment. These data were subsequently run in ClimateBC to produce climate data from 1901 to 2002. Each climate variable was then averaged over each catchment for every year that climate data were calculated. A full list of the variables produced by ClimateBC is provided in Appendix B. ClimateBC was downloaded from the University of British Columbia's Centre for Forest Gene Conservation (University of British Columbia, 2010). All mapping and spatial analyses were completed using ArcGIS 9.3.

Although stream discharge data were not collected as a part of this study, data were available for the Quesnel River at Likely, BC (52°36'56" N, 121°34'16" W). These data do

[5]Environment Canada's National Climate Archive was found to have an incomplete dataset for the Likely, BC weather station (1974-1993) which is approximately 9 km from Boswell Lake. The next closest weather stations to Boswell Lake are located in Barkerville (1888-2008) and Williams Lake (1936-2009) which are both located approximately 60 km from Boswell Lake. A weather station situated at Gruhs Lake (1950-2009), approximately 16 km from Viewland Lake, was the closest weather station that could be used to characterize weather conditions around Viewland Lake.

not reflect the same magnitude of water flow in the channels draining the Boswell Lake catchment, however, they provide an estimate of the trends that may have been observed over a similar time frame (1924-2009). As a nearby stream discharge gauge was not found for the Viewland Lake catchment, stream discharge are only presented for the Quesnel River (Boswell Lake catchment). Stream discharge data were downloaded from the Water Survey of Canada website (Environment Canada, 2010).

2.6 Sediment source tracing

2.6.1 Source groups

Source materials were initially classified according to six source types: harvested surface soil material; harvested subsurface soil material; forested surface soil material; forested subsurface soil material; road surface soil material; and, channel bank material. However, since the source materials were all derived from a single bedrock type, and forestry practices do not typically alter the geochemical regime of the disturbed area, it was recognized that the source materials could have similar characteristics and may not be significantly different from one another. To determine if a different set of groups was more appropriate than the *a priori* groupings, a Principle Component Analysis (PCA) was performed on the geochemical data. Visual examination of the biplot of the first two principle components provided an estimate of the number of source groups. A fuzzy k-means clustering analysis of the geochemical data was subsequently used to confirm the number of source groups which informed the multivariate unmixing model. The aim of this statistical test is to establish the number of groups that minimizes Dunn's coefficient which measures the "fuzziness" of the resulting group(s) (Trauwaert, 1988).

2.6.2 Multivariate unmixing model

The multivariate unmixing model described by Collins et al. (1997) was used to estimate the relative proportion of each source material in each 1-cm sediment core slice of all lake and wetland cores. The unmixing model is composed of a set of linear equations, which is subject to the conditions: (a) each source contribution must be greater than zero and less than one; and (b) the sum of the source contributions must equal one. A final solution is found when the sums of squares of the relative errors have been minimized, and all conditions have been met. The optimization routine was carried out using Microsoft Excel Solver (version 2003).

Unlike the original model by Collins et al. (1997), Equation 2.5 does not include an organic matter correction factor. The inclusion of an organic matter correction factor, along with a particle size correction factor, may result in the over-correction of the fingerprint properties (Carter et al., 2003). The unmixing model is given in Equation 2.5:

$$\sum_{i=1}^{n}\left(\frac{C_i-((P_{sf}\cdot S_{if}\cdot Z_f)+(P_{ssub}\cdot S_{sub}\cdot Z_{sub})+(P_{scb}\cdot S_{cb}\cdot Z_{cb}))}{C_i}\right)^2 \tag{2.5}$$

where C_i = concentrations of tracer parameters (i) in each 1 cm sediment core slice, S_i = mean concentration of tracer parameter (i) for each source material, Z = particle size correction factor (ratio of core slice specific surface area to mean specific surface area for each source type), and P_s = percentage contribution from each source type (s = surface; sub = subsurface; cb = channel bank).

2.7 Statistical analysis

2.7.1 Pre- versus post-logging

Two-sample t-tests were used to compare post-logging total (clastic and organic sediment) sedimentation rates to pre-logging rates. The two periods of forestry practices in the Boswell Lake catchment were combined and analyzed as a single "post-logging" period to improve the statistical power since sample sizes were low in individual post-logging periods. Therefore, the post-logging period in the Boswell Lake catchment begins at the onset of logging and includes periods of active logging and the recovery periods (i.e. 1960-2009). A series of one-sample t-tests were also used to compare total sedimentation rates calculated for each post-logging 1-cm core slice to pre-logging rates. These comparisons were useful in identifying peaks or depressions in the post-logging period that represented significant departures from average pre-logging conditions. A Holm correction (sequential Bonferroni correction) was applied to the p-values of the 1-sample t-tests for each proxy within a given core to account for an inflated type I error that may occur as a result of multiple tests on a single group of data (Holm, 1979; Rice, 1989). See Figure 2.6 for a diagram comparing the 1- and 2-sample t-tests used on the lake and wetland core measurements.

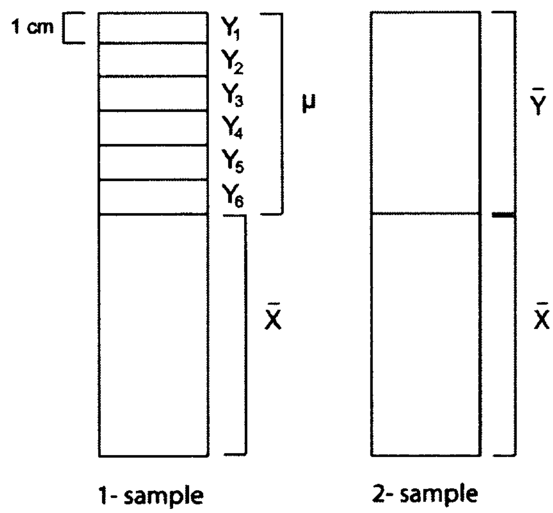


Figure 2.6: Comparison of the 1- and 2-sample t-tests used to evaluate post-logging ($\bar{Y}$) changes in the sediment profiles against average pre-logging ($\bar{X}$) conditions. For 1-sample t-tests, sedimentation rate and proxy values given by individual post-logging 1 cm core slices ($\bar{Y}_i$) represent the null hypothesis (μ).

Where the assumption of normally distributed residuals was not met, non-parametric statistics were applied (e.g. Mann–Whitney U-test, Wilcoxon Rank Sum Test). The same approach was also taken for the analysis of each proxy indicator. It has also been recognized that temporal autocorrelation is often an issue with sediment cores which violates the assumption of independent samples. Temporal autocorrelation can be the result of lags between sediment erosion and delivery, physical mixing of the sediments, and post-depositional mobility of proxy indicators (e.g. radionuclides). Corrections for temporal autocorrelation often requires increasing the temporal separation between samples which can be achieved by combining samples. Not correcting for temporal autocorrelation can result in increased type I error (α); however, due to low sample sizes a correction was not applied.

2.7.2 Climate and stream discharge trends

To account for any potential effects of climate and stream discharge on lake and wetland total sedimentation rates stepwise linear regressions were fit. Variable selection was carried out in both directions (i.e. forward selection and backward elimination) and a final model was selected when the Akaike's Information Criterion (AIC) value was minimized. Predictor variables included climate variables produced by ClimateBC (mean annual precipitation, mean annual temperature, precipitation as snow, beginning and end of frost-free period), and stream discharge. It was necessary to narrow down the climate variables to a smaller set such that the total number of variables entered into the model did not exceed the total degrees of freedom. A factor was also included in the model to identify pre- and post-logging periods. Climate and stream discharge data were averaged for each core to match the time intervals represented by each 1 cm core slice.

Although averaging the climate data was necessary to be included in the stepwise linear regression models, this also reduced the resolution of the climate series and flattened out annual variations. Trends in annual climate and discharge data were assessed using the method outlined by Tomé (2004), which identifies breakpoints and linear trends in time series data. The sign and magnitude of the trends in between sets of breakpoints were examined with the regression lines produced by the model. A single regression line was also calculated for each full time series to observe long-term trends.

2.7.3 Correlations

Pearson product moment correlations were used to relate changes in the contribution of each sediment source to down-core variations of total sedimentation rates, as well as those of each proxy indicator. A change in the dominant sediment source, or the relative proportions of these sources, may be the result of forestry practices; though, it is also possible that changes in source materials are not related to fluctuating total sedimentation rates in either the lake or the wetland.

All statistical results were considered significant at an α level of 0.05. All statistical analyses were conducted using R 2.10.1 (2009).

Chapter 3

Results: Lake and wetland sedimentation rates

This chapter addresses the research questions: do wetland sedimentation rates increase in response to forestry activities?; and, are sedimentation rates in downstream lakes affected by forestry activities? The information presented here adresses the variations in sedimentation rates, as well as the physical and chemical properties of the sediment, before and during/after periods of active logging. Historical changes in climate and their impact on sedimentation rates and sediment properties have also been explored in addition to forestry practices. Finally, variations in bulk physical properties over the last century were compared to changes over the deeper undated profile to understand their importance in the context of the longer-term natural variability.

3.1 Physical descriptions

Material in the Boswell Lake core was predominantly light brown, fine-grained organic sediment (gyttja) which was found throughout the top 80 cm of the core (see Figure 3.6). Below 80 cm alternating bands of dark and light brown fine organic sediment were visible. A light grey tephra layer was found at 56 cm depth. Visual inspection of the tephra under a polarizing light microscope revealed chunky glass shards containing lineated gas vesicles. Based on the observed colour and glass shard morphology (Brian Menounos pers. comm.), it was concluded that the tephra originated from the Bridge River eruption ca. 2,410 calendar years ago (Clague et al., 1995). See Appendix C for a microscope image of the glass shards found in the Boswell Lake core. Other than the tephra there were no other obvious changes in texture along the length of the core.

The Viewland Lake core was primarily composed of light brown, fine-grained organic sediment (gyttja) similar to that found in the Boswell Lake core. A 0.5 cm layer of light grey, fine-grained clastic sediment was found 1.5 cm down-core below which was a thin layer of dark brown, fine-grained organic sediment. Alternating light and dark brown layers of sediment were also seen deeper (ca. 70 cm) in the core. However, the layers of sediment were not as well-defined as those found in the Boswell Lake core. A light grey tephra layer was found at 67 cm down-core that, based on colour and shard morphology, was correlative with the Bridge River event (Brian Menounos pers. comm.).

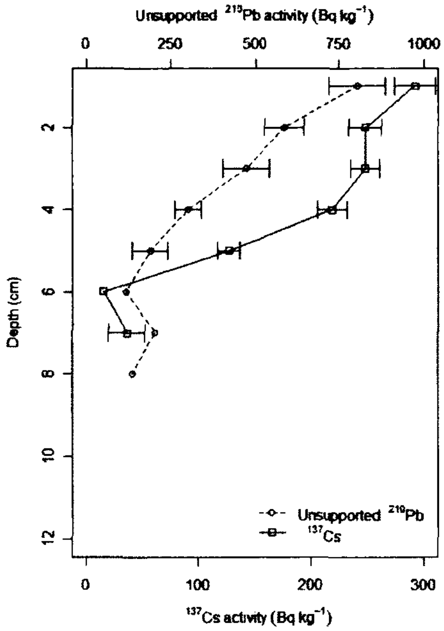
Wetland cores from both sites consisted of dark brown, unsorted, organic-rich sediments. Large pieces of woody debris, roots and twigs were observed throughout all wetland cores in no observable pattern. The top 10 cm of the Viewland wetland core was predominantly composed of twigs and other woody debris.

3.2 Lead-210 profiles and core chronologies

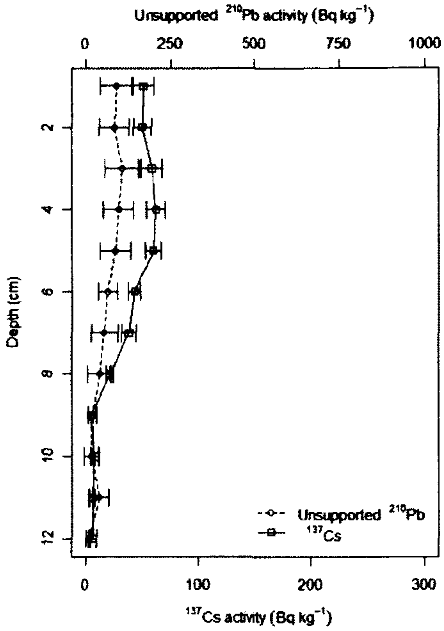
In Boswell Lake, background concentrations of unsupported ^{210}Pb were reached at 9 cm in the lake core (BL-P1), 13 cm in one wetland core (BL-D8), and 12 cm in the other wetland core (BL-D10). Background activities were reached by 19 cm and 16 cm in the Viewland Lake (VL-P1) and the wetland (VL-D1) cores, respectively. Shallower ^{210}Pb profiles in the Bowell Lake core versus either of the wetland cores suggests that less material is accumulating in the lake as compared to the wetland. Alternatively, regular resuspension and transport of material from the lake bottom could lower ^{210}Pb concentrations. The opposite is observed in the Viewland Lake scenario where, according to the ^{210}Pb profile, the lake is accumulating a greater amount of material than the wetland. Since wetland cores were taken from channels, erosion of the channel bottom due to flowing water may have redistributed sediments creating unconformities in the depositional record. The implications of this sampling design are discussed in the study limitations (see Section 5.5).

Before each ^{210}Pb dating model was applied to the unsupported ^{210}Pb activities of each sediment core, the model assumptions were reviewed against the activity profiles of unsupported ^{210}Pb in Figures 3.1 and 3.3. Non-monotonic decreases in unsupported ^{210}Pb activities in all cores confirm that the CIC model is not a suitable dating model. Modelled unsupported ^{210}Pb activities produced by the SIT model were accompanied by relatively large χ^2 values (Table 3.1) which suggests that the model does a poor job reconstructing the original unsupported ^{210}Pb profiles. This is consistent with the limitations of the SIT model as outlined by Carroll & Abraham (1996) which state that large fluctuations in unsupported ^{210}Pb activities may not be accurately modelled by a Fourier sine series. Thus, the final core chronologies were calculated using the CRS model - see Figures 3.2 and 3.4.

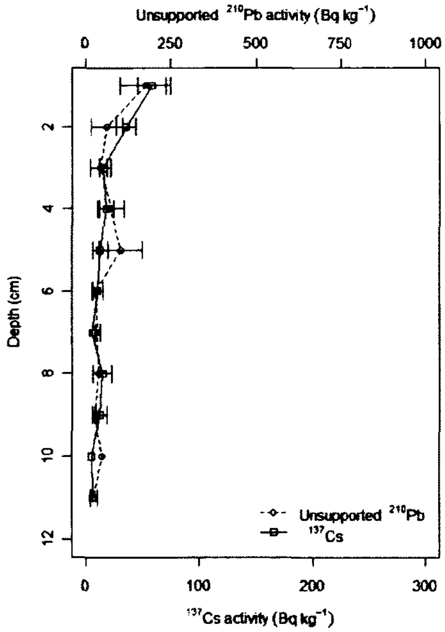
With the exception of the VL-P1 core, well-defined ^{137}Cs peaks are not present in any of the cores which suggests that post-depositional changes (i.e. bioturbation, upward/downward diffusion) have impacted down-core concentrations of ^{137}Cs (Foster et al., 2006). As a result, ^{137}Cs was not used to verify core chronologies in this study. Despite a strong peak, ^{137}Cs data were also disregarded for the VL-P1 core. Based on the resultant core chronologies there is a large discrepancy between the ^{137}Cs peak and the CRS-modelled 1963 date for this core. The ^{137}Cs peak also coincides with a layer of fine-grained silty-clay material which, through the binding effects of clay (Ambers, 2001), likely limited the mobility of ^{137}Cs resulting in increased concentrations. Davis et al. (1984) stated that high mobility of ^{137}Cs in organic-rich sediments is the result of the breakdown of organic material. On the other hand, ^{210}Pb is bound tightly to organic material (Dörr & Münnich, 2006) suggesting that the use of ^{210}Pb to date organic-rich sediments is more reliable than ^{137}Cs (Davis et al., 1984). A similar conclusion was reached by Foster & Lees (1999) who also found ^{137}Cs profiles to be unreliable.



(a) Lake core (BL-P1)

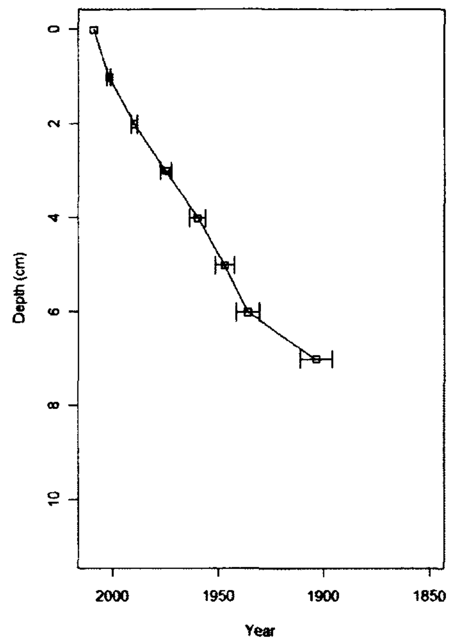


(b) Wetland core (BL-D8)

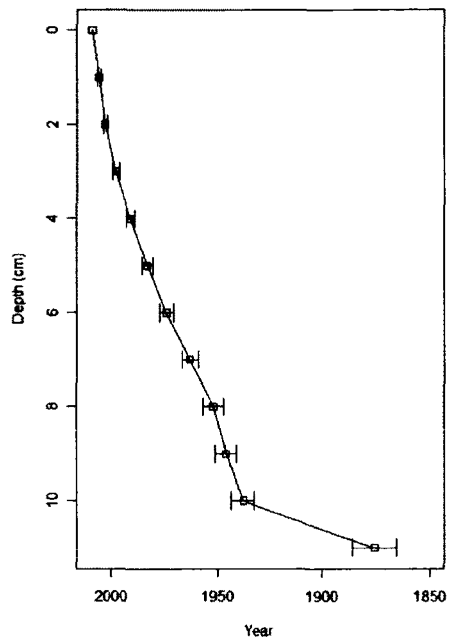


(c) Wetland core (BL-D10)

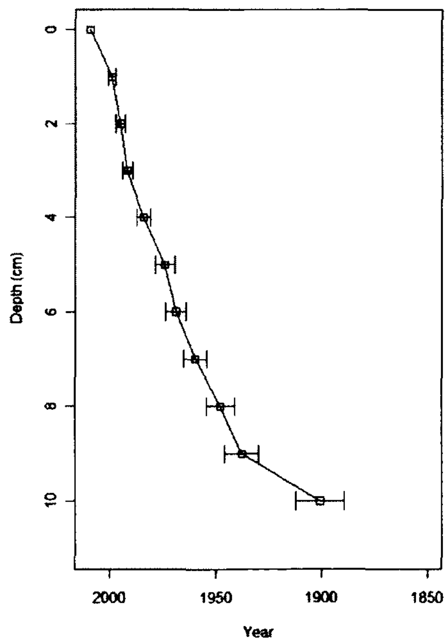
Figure 3.1: Unsupported ^{210}Pb and ^{137}Cs activity depth profiles for Boswell Lake and wetland cores. Unsupported ^{210}Pb error bars represent the sum of the total ^{210}Pb and supported ^{210}Pb errors. Values without errors were measured at the minimum detectable limit of the gamma assay.



(a) Lake core (BL-P1)

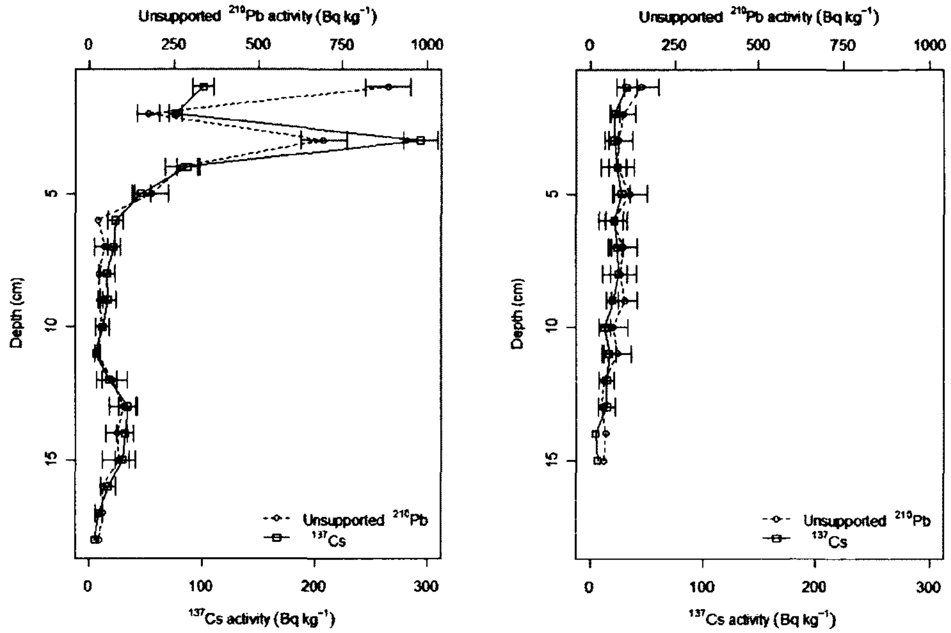


(b) Wetland core (BL-D8)



(c) Wetland core (BL-D10)

Figure 3.2: Core chronologies for (a) Boswell Lake (BL-P1) and wetland cores (b) BL-D8, and (c) BL-D10 produced by the Constant Rate of Supply (CRS) model. Error bars were also calculated using the CRS model and represent the error on each of the calculated dates.



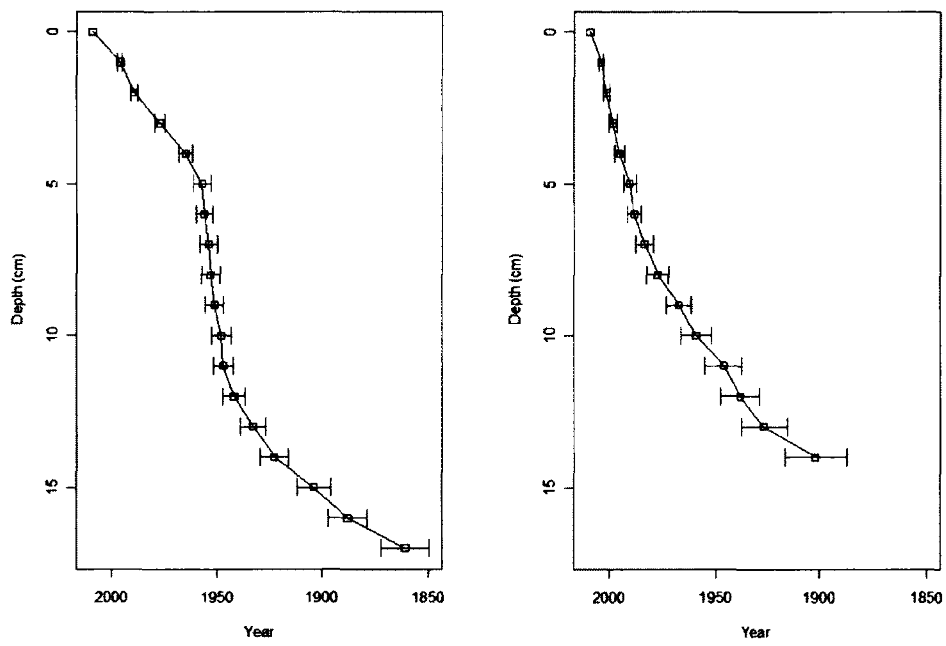
(a) Lake core (VL-P1)

(b) Wetland core (VL-D1)

Figure 3.3: Unsupported ^{210}Pb and ^{137}Cs activity depth profiles for Viewland Lake and wetland cores. Unsupported ^{210}Pb error bars represent the sum of the total ^{210}Pb and supported ^{210}Pb errors. Values without errors were measured at the minimum detectable limit of the gamma assay.

Table 3.1: Summary of the χ^2 values produced by the Sediment Isotope Tomography (SIT) model. These values represent the goodness-of-fit between an observed distribution (measured unsupported ^{210} activities) and a theoretical distribution (modelled unsupported ^{210}Pb activities). For a sample size of 10, two sample distributions would be considered to be not significantly different (p>0.05) if the χ^2 value was <16.9.

Core	χ^2 (Bq kg^{-1})
BL-P1	924
BL-D8	80.3
BL-D10	649
VL-P1	11800
VL-D1	181



(a) Lake core (VL-P1)

(b) Wetland core (VL-D1)

Figure 3.4: Core chronologies for (a) Viewland Lake (VL-P1) and (b) wetland (VL-D1) cores produced by the CRS model. Error bars were also calculated using the CRS model and represent the error on each of the calculated dates.

3.3 Boswell Lake catchment

3.3.1 Total sedimentation rates

Comparison of pre- and post-logging sedimentation rates (Fig. 3.5) using two-sample t-tests showed that the two periods are not significantly different for any of the cores taken from Boswell Lake or the wetland (Table 3.2). Individual one-sample t-tests for each of the 1 cm increments in the post-logging period revealed no significant departures from average pre-logging sedimentation rates in Boswell Lake.

Above 10 cm (ca. 1938), a gradual increase in total sedimentation rates was observed in wetland core (BL-D8) and reached a maximum of 0.0756 g cm^{-2} y^{-1} in the top 1 cm. The second wetland core (BL-D10) had two distinct peaks in sedimentation rates at 3 and 6 cm down-core. Both peaks corresponded to the two logging periods in the Boswell Lake catchment, however, neither was found to be statistically greater than average pre-logging sedimentation rates.

Table 3.2: Summary of the two-sample t-tests results comparing pre- and post-logging total sedimentation rates (g cm^{-2} y^{-1}) in Boswell Lake and wetland cores. In BL-P1, BL-D8, and BL-D10 the post-logging periods are above 4 cm, 7 cm, and 7 cm, respectively. Values in brackets denote sample size.

	BL-P1 (Lake)			BL-D8 (Wetland)			BL-D10 (Wetland)		
Period	Mean	sd	p	Mean	sd	p	Mean	sd	p
Pre	0.0057	0.0035 (3)		0.0342	0.023 (4)		0.0169	0.0109 (3)	
Post	0.0079	0.0013 (4)	0.386	0.0492	0.018 (7)	0.308	0.0466	0.0228 (7)	0.067†

†p was calculated using non-parametric analysis as the assumption of normality was not met.

3.3.2 Proxy indicators

Depth profiles of all proxies are presented in Figure 3.6. Two-sample t-test results are summarized in Table 3.3. Median grain size in Boswell Lake (BL-P1) significantly decreased post-logging, and was the only proxy indicator for which a significant change occurred. One-

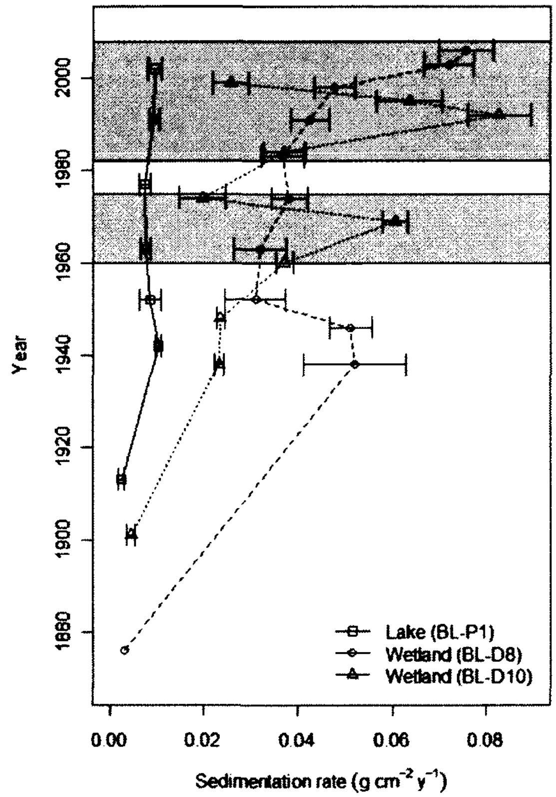


Figure 3.5: Total sedimentation rates (calculated using the CRS model) for Boswell Lake and wetland cores. The highlighted areas represent the periods of time that forestry practices were present in the Boswell Lake catchment. Error bars on the sedimentation rates represent the standard error calculated using the CRS model. An error value could not be calculated for the bottom of the BL-D8 profile.

sample t-tests revealed that significant decreases in median grain size occurred at 1 (t=11.0, p=0.019), 2 (t=1.14, p=0.020) and 4 cm (t=9.70, p=0.019) down-core, all of which were within the post-logging periods. A small post-logging increase was observed in dry bulk density, which was mirrored by a small decrease in percent water content. The start of these changes, however, occurred approximately 20 years (ca. 1940) before the beginning of the first logging period. Maximum values of dry bulk density were reached at 3 cm and were followed by a decrease at 2 cm down-core, with the opposite pattern being observed for water content. At 6 cm depth, minor increases in magnetic susceptibility and C:N were observed along with small decreases in total carbon and total nitrogen.

A significant post-logging decrease in median grain size was observed in wetland core BL-D8. While generally smaller median grain sizes were observed post-logging, a small increase occurred during the first logging period. However, the second logging period did not produce any distinct changes in median grain size. Decreases in total C, total N and C:N occurred at the end of the first logging period. These three proxies continued to fluctuate throughout the second period of logging, but only C:N remained significantly lower than pre-logging values.

On average, magnetic susceptibility values were found to be significantly higher during the post-logging period in wetland core BL-D10; yet, this change only appears at the end of the second logging period. Aside from a small decrease in dry bulk density at the end of the first logging period, no other notable changes were seen in the BL-D10 wetland core.

3.3.3 Long-term changes in bulk physical properties

Although ^{210}Pb is only able to date (with any accuracy) approximately the last 100-150 years, data for the bulk physical properties were still collected beyond the ^{210}Pb-dated region of the sediment cores. Long-term changes of these proxies allow recent changes to be placed in a broader context. Dry bulk density and percent water content were measured to various depths for all three cores (Fig. 3.7). The amount of data available is dependent on the length of core that could be retrieved from individual sampling locations.

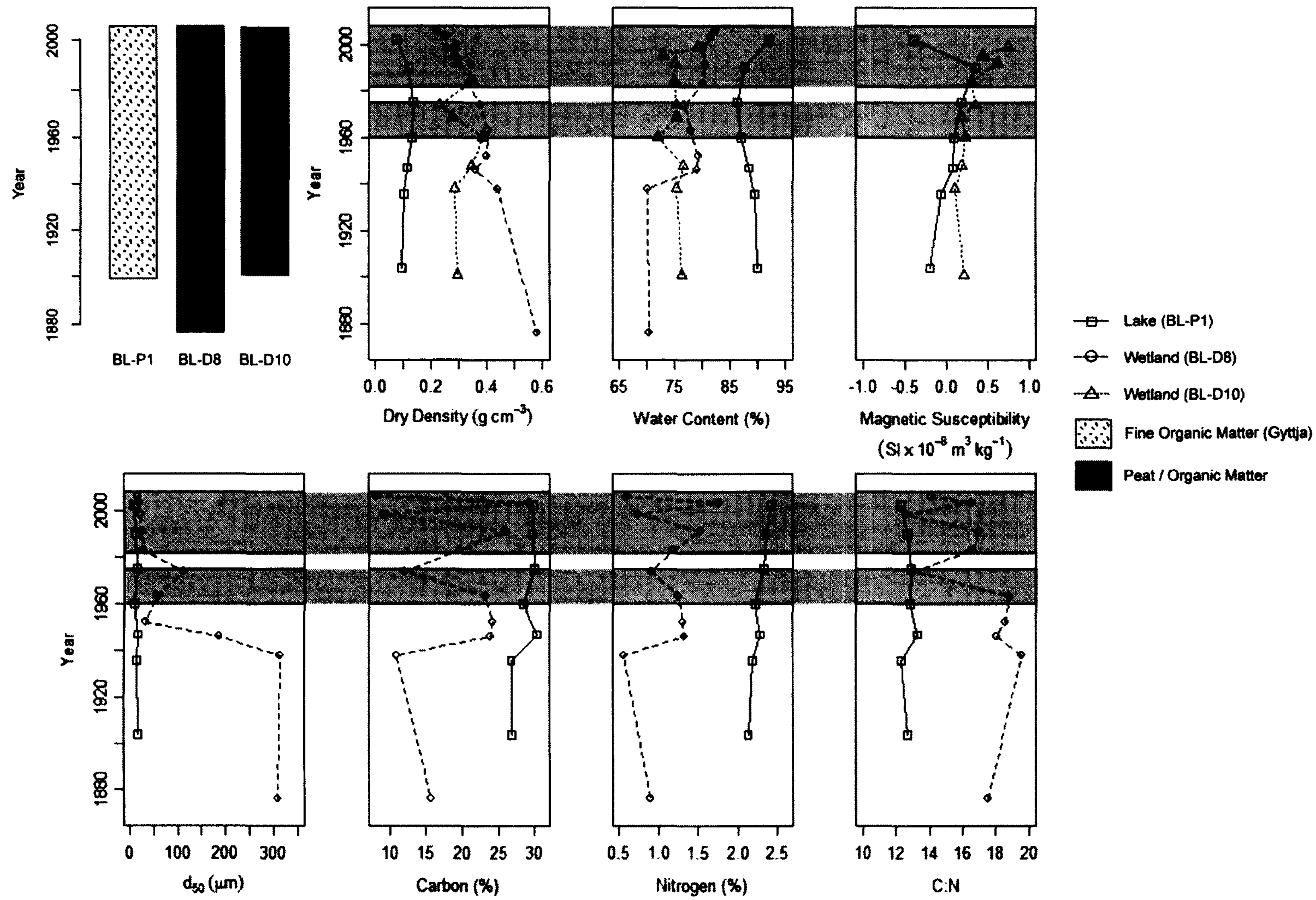


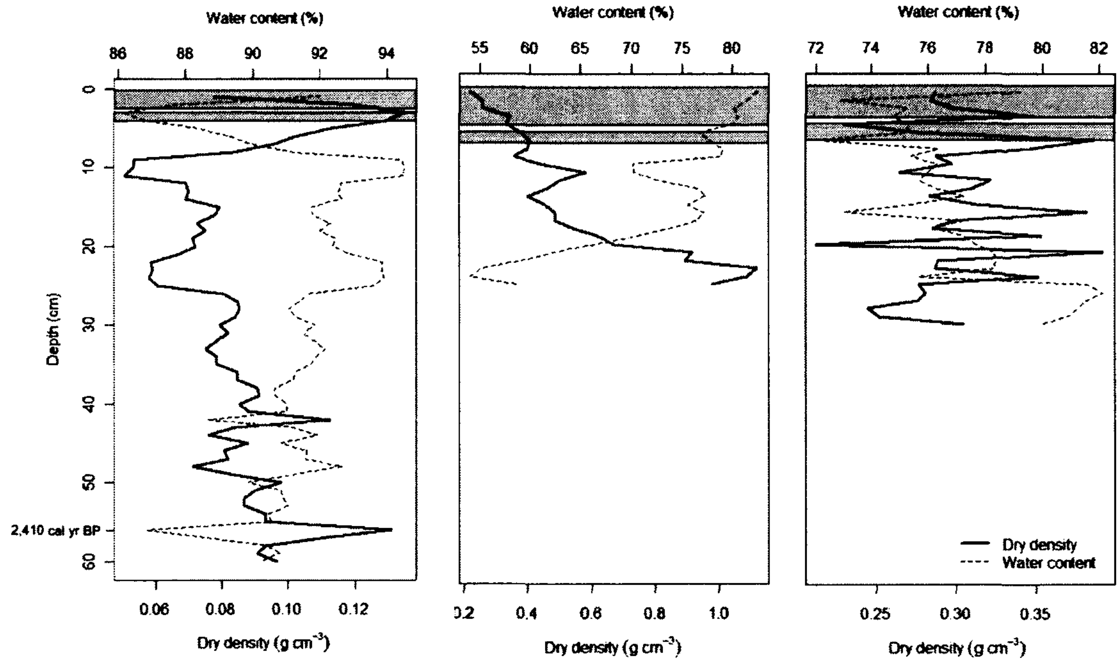
Figure 3.6: The seven proxy indicators (dry bulk density, percent water content, magnetic susceptibility, median particle size, total C, total N, and C:N) are shown over time for the dated portion of each of the Boswell Lake and wetland cores. The highlighted areas represent the years that forestry practices were present in the Boswell Lake catchment. Core logs and general descriptions of the sediment are also provided for each core (top left).

Table 3.3: Summary of two-sample t-tests comparing the means of pre- and post-logging periods for each proxy indicator measured in the Boswell Lake and wetland cores. In BL-P1, BL-D8, and BL-D10 the post-logging periods are above 4 cm, 7 cm, and 7 cm, respectively. Values in brackets denote sample sizes which are consistent across all proxies.

		BL-P1 (Lake)			BL-D8 (Wetland)			BL-D10 (Wetland)		
Proxy	Period	Mean	sd	p	Mean	sd	p	Mean	sd	p
Dry bulk density ($g\ cm^{-3}$)	Pre	0.1038	0.0088 (3)		0.4453	0.0971 (4)		0.3101	0.0317 (3)	
	Post	0.1156	0.0260 (4)	0.451	0.3139	0.0698 (7)	0.065	0.3011	0.0511 (7)	0.747
Water content (%)	Pre	89.24	0.79		74.66	5.09		76.04	0.57	
	Post	88.18	2.63	0.491	79.89	1.98	0.073†	74.97	2.29	0.284
Magnetic susceptibility ($SI\ x\ 10^{-8}\ m^3\ kg^{-1}$)	Pre	-0.1	0.1		0.3	0.1		0.2	0.1	
	Post	0.1	0.3	0.528	0.2	0.3	0.889	0.4	0.2	0.025*
d_{50} (μm)	Pre	14.8	0.8		209.4	131.6				
	Post	10.1	2.6	0.031*	39.1	34.3	0.024*†			
Total carbon (%)	Pre	28.0	2.0		18.6	6.4				
	Post	29.4	0.7	0.348	18.2	8.4	0.922			
Total nitrogen (%)	Pre	2.19	0.08		1.02	0.36				
	Post	2.32	0.09	0.103	1.13	0.42	0.644			
C:N	Pre	12.7	0.5		18.4	0.9				
	Post	12.7	0.3	0.852	15.5	2.3	0.016*			

*Significant at p=0.05

†p was calculated using non-parametric analysis as the assumption of normality was not met.



(a) Boswell Lake (BL-P1) (b) Boswell wetland (BL-D8) (c) Boswell wetland (BL-D10)

Figure 3.7: Long-term depth profiles of dry bulk density and percent water content for (a) Boswell Lake (BL-P1) and wetland cores (b) BL-D8, and (c) BL-D10. Values are presented over depth as they extend beyond the dated region of the sediment cores where ^{210}Pb was not present in measurable concentrations. Highlighted areas represent years that forestry activities were present in the catchment. The date (2,410 yrs BP) provided at 56 cm is the location of the Bridge River tephra layer in the lake core.

Dry bulk density values from 10 to 50 cm in the lake core showed a gradually increasing pattern over increasing depth, which is consistent with sediment de-watering and compaction. Other than sudden increases at 42 and 57 cm, the latter being associated with the Bridge River tephra layer, little variation was observed. Above 10 cm, dry bulk density increased to approximately 0.135 g cm^{-3} after which point it decreased in the top 2 cm of the core. A similar pattern was seen in wetland core BL-D8, with gradually increasing dry bulk density values with increasing depth to a maximum depth of 25 cm. This pattern was not observed in the second wetland core (BL-D10); rather, dry bulk density values did not exhibit a long-term trend and the range of variability was much smaller than that of the other wetland core (BL-D8). Percent water content generally mirrored dry bulk density patterns throughout each of the sediment cores.

3.3.4 Hydrometerological influences and trends

Several climate variables, along with stream discharge data and a before-and-after logging factor were examined to determine whether any changes in the lake and wetland sedimentation rates could be explained by fluctuations in climate in addition to, or instead of, the timing of logging. According to the results of the stepwise linear regression, patterns of sedimentation rates could not be explained for any of the cores using any combination of climate variables, stream discharge data, or the presence/absence of logging.

Based on the trend analysis (Tomé, 2004), a breakpoint in stream discharge values was found at 1944 (Fig. 3.8). Following this breakpoint, average stream discharge values significantly increased from approximately 3.77 km^3 y^{-1} to 4.14 km^3 y^{-1} (t=-2.46, p=0.018). This type of change in climate variables has been referred to a "step change" (Macklin & Lewin, 2003) because it is an abrupt change which produces a new average condition or equilibrium. The breakpoint at 1944 was also associated with a significant increase mean annual precipitation (MAP) which increased from an average value of approximately 724 mm y^{-1} to 787 mm y^{-1} (t=-3.31, p=0.001). An increase in the variability of MAP was also observed after 1944. Prior to 1944, MAP values ranged between approximately 583 mm y^{-1} and 891 mm y^{-1}. After 1944, minimum MAP fell slightly to 580 mm y^{-1}, however, the maximum value rose to 1023 mm y^{-1}. Other climate variables were found to also significantly change after 1944, including: precipitation as snow (increase; t=-2.20, p=0.030), frost-free period (increase; t=-4.72, p<0.001), beginning of frost-free period (decrease; t=3.23, p<0.001) and the end of the frost-free period (increase; t=-4.46, p<0.001).

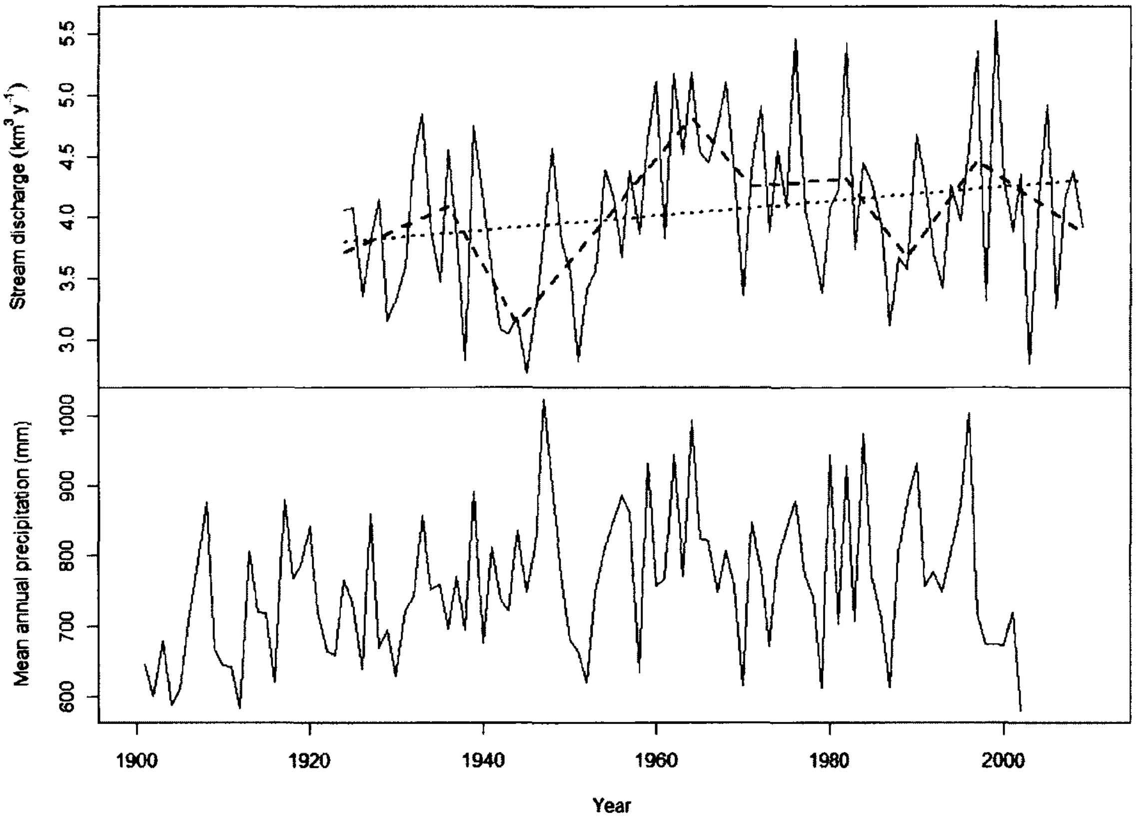


Figure 3.8: Annual stream discharge (1924-2009) and mean annual precipitation (1901-2002) values. Stream discharge values are for Quesnel River at Likely, BC and were taken from the Water Survey of Canada (Environment Canada). Mean annual precipitation measurements are specific to the Boswell Lake catchment and were modelled using ClimateBC. The small dashed line represents the linear regression line for the full time series of stream discharge. The large dashed lines are linear regression lines in between each set of breakpoints.

3.4 Viewland Lake catchment

3.4.1 Total sedimentation rates

Post-logging sedimentation rates were not found to be significantly differently from pre-logging rates in the Viewland Lake core (Table 3.4). The depth profile of Viewland Lake sedimentation rates (Fig. 3.9) shows a post-logging peak at 2 cm depth, however, a one-sample t-test on this core slice revealed it is not significantly greater than pre-logging rates. On the other hand, the decrease found at the top-most layer (0-1 cm) does reveal that sedimentation rates dropped below pre-logging rates. This drop below pre-logging rates is due to the fact that sedimentation rates peak in the late-1940's and do not drop again until the late-1950's.

On average, post-logging sedimentation rates in the wetland (VL-D1) were significantly higher than those before logging occurred in the catchment area (Table 3.4). Figure 3.9 shows gradually increasing sedimentation rates over the pre-logging period. Post-logging sedimentation rates sharply increased at 3 and 6 cm, both of which are significantly greater than pre-logging rates (t=-13.0, p=<0.001; t=-11.7, p=<0.001).

Table 3.4: Summary of the two-sample t-tests results comparing pre- and post-logging sedimentation rates (g cm^{-2} y^{-1}) in Viewland Lake and wetland cores. In VL-P1 and VL-D1, the post-logging periods are above 2 cm and 7 cm, respectively. Values in brackets denote sample size.

	VL-P1 (Lake)			VL-D1 (Wetland)		
Period	Mean	sd	p	Mean	sd	p
Pre	0.0282	0.0259 (15)		0.0163	0.0064 (7)	
Post	0.0206	0.0175 (2)	0.824†	0.0423	0.0101 (7)	<0.001*

*Significant at p=0.05

$^{\dagger}p$ was calculated using non-parametric analysis as the assumptions of normality were not met.

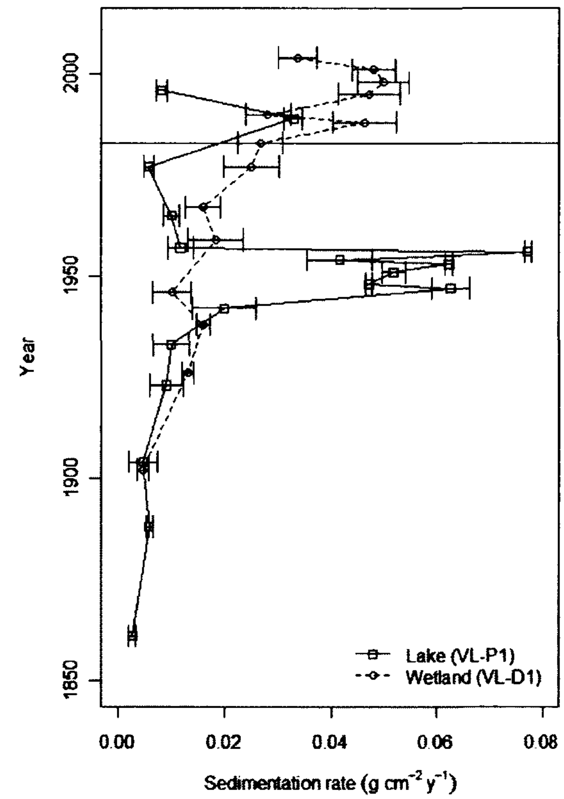


Figure 3.9: Total sedimentation rates (calculated using the CRS model) for Viewland Lake and wetland cores. The horizontal line at 1983 represents the year the Viewland Lake catchment was logged. Error bars on the sedimentation rates were also calculated using the CRS model.

3.4.2 Proxy indicators

Depth profiles of all proxies are found in Figure 3.10. Two-sample t-test results are summarized in Table 3.5. With respect to the Viewland Lake core, except for median grain size, little variation was observed in any of the proxies during the pre-logging period. Median grain size increased sharply at 13 cm and quickly decreased moving up-core. Apart from C:N, a significant post-logging change was evident in all proxies. Sharp increases were observed in dry bulk density and magnetic susceptibility, while percent water content, median grain size, total C and total N all decreased immediately after the catchment was logged in 1983. Although the change was not statistically significant, C:N shows evidence of a post-logging increase. All proxies returned to pre-logging conditions in the 0-1 cm core slice.

Proxy indicators measured for the Viewland wetland core remained relatively consistent in both the pre- and post-logging periods. Magnetic susceptibility increases significantly post-logging from an average value of -0.03 to 0.00. At 2 cm, there is a sharp decrease in dry bulk density and corresponding increase in percent water content, both of which return to pre-logging conditions at 1 cm.

3.4.3 Long-term changes in bulk physical properties

Deeper profiles (30 cm) of dry bulk density and percent water content were also collected for the Viewland Lake and wetland cores (Fig. 3.11). The large increase in dry bulk density and corresponding decrease in percent water content that occurred immediately after the logging event in 1983 were much higher and lower, respectively, than any other changes that have taken place over the deeper profile of the lake core. Dry bulk density values calculated for the Viewland wetland core (VL-D1) increased consistently down-core which may be the result of sediment compaction over time. However, the post-logging decrease in dry bulk density observed at 6 cm appears to extend beyond the range of normal variability. The opposite pattern was observed for percent water content in the wetland core with a strong increase in percent water content at 6 cm depth.

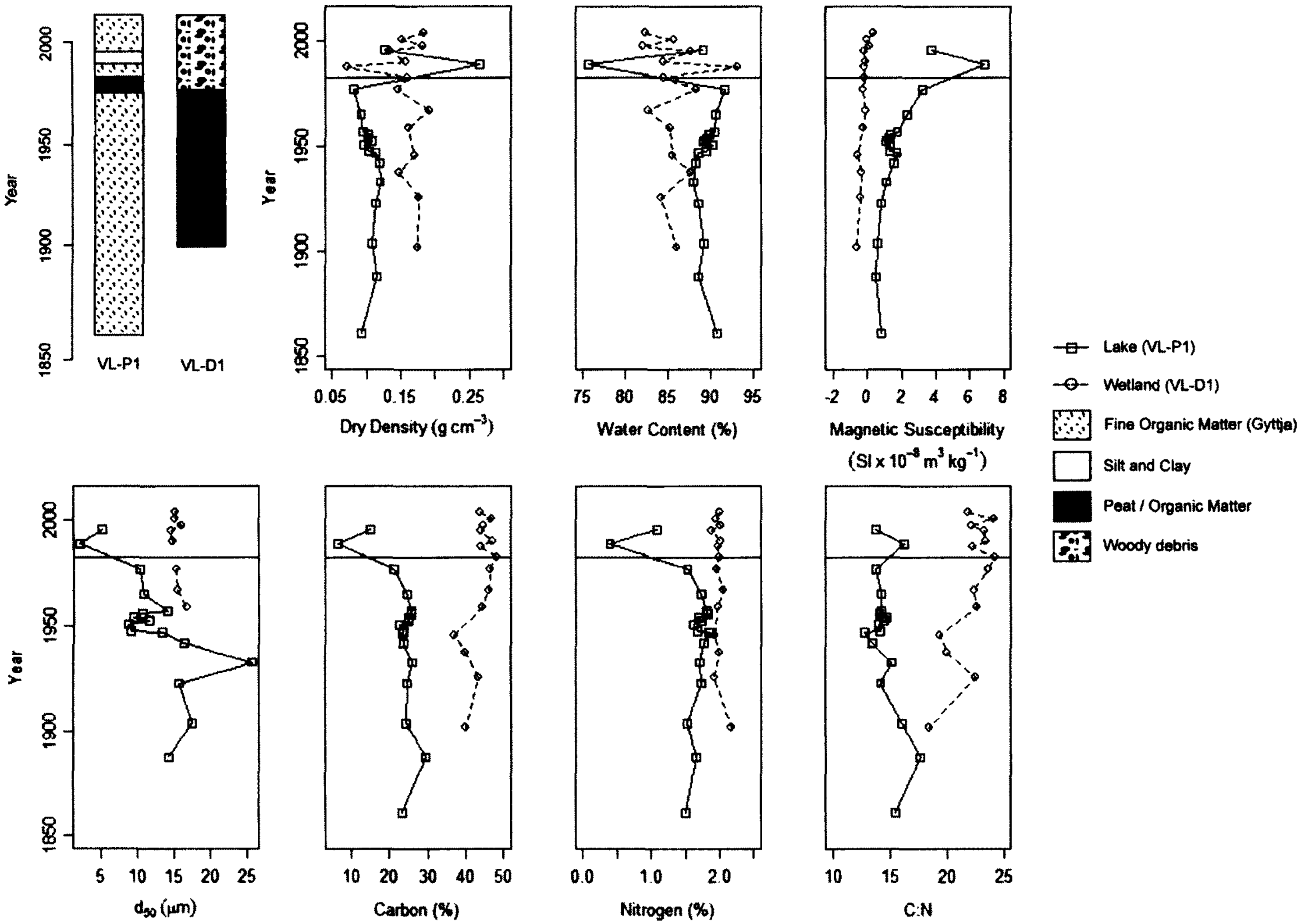


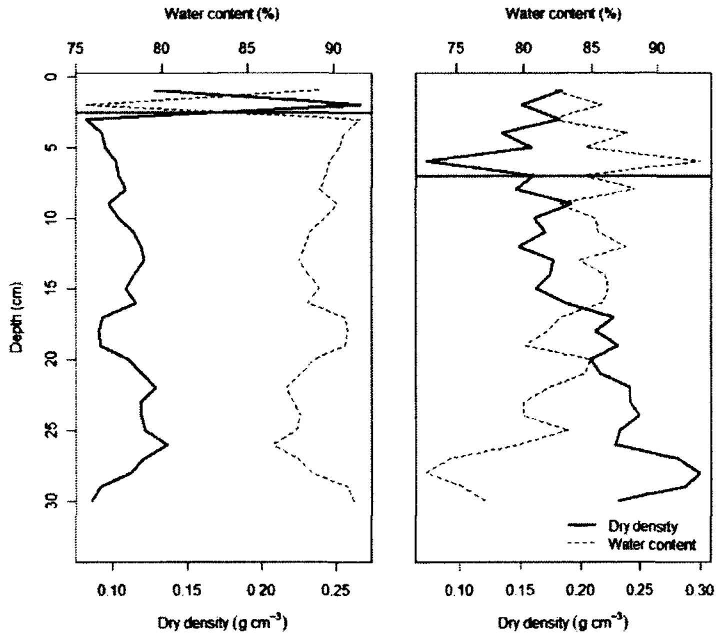
Figure 3.10: The seven proxy indicators (dry bulk density, percent water content, magnetic susceptibility, median particle size, total C, total N, and the C:N) are shown over time for the dated portion of each of the Viewland Lake and wetland cores. The horizontal line represents the year the Viewland Lake catchment was logged (1983). Core logs and general descriptions of the sediment are also provided for each core (top left).

Table 3.5: Summary of the two-sample t-test results comparing the means of pre- and post-logging periods for each proxy indicator measured for the Viewland Lake and wetland cores. In VL-P1 and VL-D1, the post-logging periods are above 2 cm and 7 cm, respectively. Values in brackets denote sample sizes. Total number of samples are given under dry bulk density. Other values are given where sample size was less than the total.

		VL-P1 (Lake)			VL-D1 (Wetland)		
Proxy	Period	Mean	sd	$p^{\dagger}$	Mean	sd	p
Dry bulk density ($g\ cm^{-3}$)	Pre	0.1047	0.0111 (15)		0.1665	0.0162 (7)	
	Post	0.1967	0.0977 (2)	0.015*	0.1469	0.0382 (7)	0.312
Water content (%)	Pre	89.49	1.02		85.48	1.94	
	Post	82.36	9.52	0.177	85.89	3.79	0.827
Magnetic susceptibility	Pre	1.4	0.7		-0.3	0.2	
(SI x $10^{-8}\ m^3\ kg^{-1}$)	Post	5.3	2.2	0.030*	0.0	0.2	0.005*
d_{50} (μm)	Pre	13.4	4.5 (14)		15.9	0.8 (3)	
	Post	3.6	2.1	0.017*	15.2	0.6 (5)	0.249
Total carbon (%)	Pre	24.5	1.8		43.2	3.6	
	Post	10.8	6.1	0.015*	45.1	1.7	0.239
Total nitrogen (%)	Pre	1.69	0.11		2.00	0.09	
	Post	0.75	0.49	0.015*	1.97	0.05	0.444
C:N	Pre	14.5	1.2		21.7	2.0	
	Post	15.0	1.8	0.941	22.9	0.9	0.167

*Significant at p=0.05

$^{\dagger}p$ was calculated using non-parametric analysis as the assumptions of normality were not met.



(a) Viewland Lake (VL-P1)

(b) Viewland wetland (VL-D1)

Figure 3.11: Long-term depth profiles of dry bulk density and percent water content for (a) Viewland Lake and (b) wetland cores. Values are presented over depth as they extend beyond the dated region of the sediment cores where ^{210}Pb is not present in measurable concentrations. Horizontal lines represent the year that logging activities were present in the catchment. Although not show here, the Bridge River tephra layer (2,410 yrs BP) occurred at 67 cm depth in the lake core (VL-P1).

3.4.4 Hydrometerological influences and trends

Based on the results of the stepwise linear regression (see Table 3.6), precipitation as snow (mm) was found to explain 44% ($F_{1,14}$=11.22, p=0.005) of the variation observed in Viewland Lake sedimentation rates (Fig. 3.12). Mean annual temperature, mean annual precipitation and the length of the frost-free period were not included in the final model as that produced models with higher AIC values. The before- and after-logging factor also was not included in the final model.

Table 3.6: Final model produced by the stepwise linear regression for Viewland Lake (VL-P1) sedimentation rates. PAS=precipitation as snow.

Variable	Df	Estimate	Std Error	t	p
Intercept	1	-0.0905	0.0359	-2.518	0.025
PAS	1	0.0005	0.0001	3.350	0.005
Residuals	14		0.0191		

The final model for the wetland core contained only the before- and after-logging factor (see Table 3.7). None of the climate variables (mean annual temperature, mean annual precipitation, precipitation as snow, and length of the frost-free period) were included in the final model produced by the stepwise linear regression. The logging factor accounted for 72% ($F_{1,12}$=31.29, p<0.001) of the variation in wetland sedimentation rates with rates being significantly higher in the post-logging period.

Table 3.7: Final model produced by the stepwise linear regression for Viewland wetland (VL-D1) sedimentation rates.

Variable	Df	Estimate	Std Error	t	p
Intercept	1	0.0400	0.0032	12.53	<0.001
Logging (Before/After)	1	-0.0253	0.0045	-5.593	0.001
Residuals	12		0.0085		

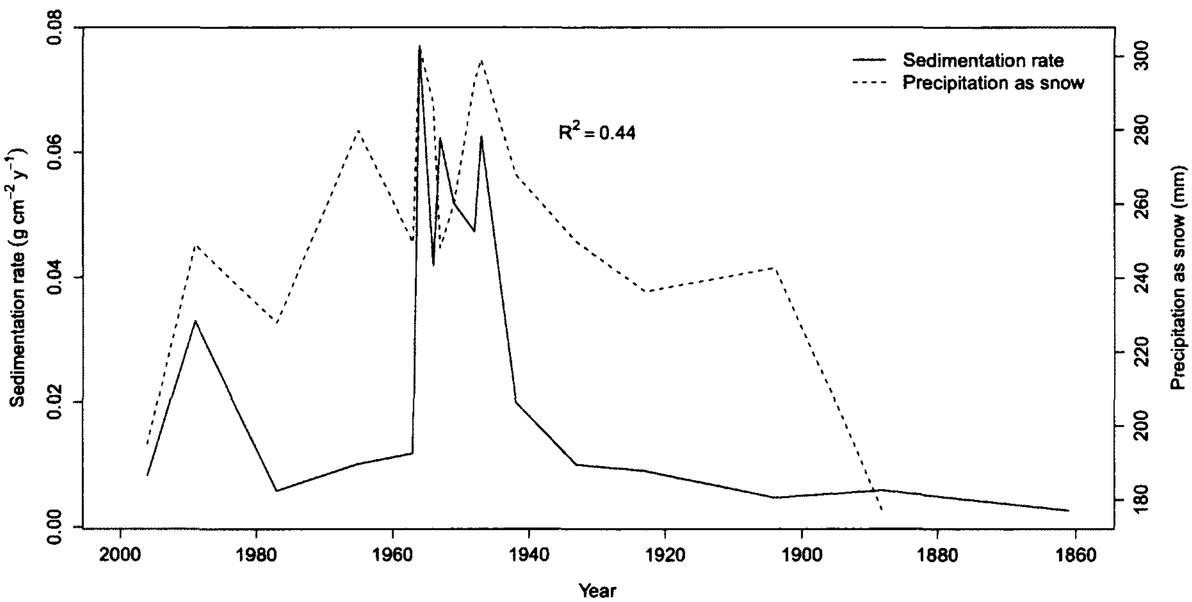


Figure 3.12: Precipitation as snow (mm) and sedimentation rates (g cm^{-2} y^{-1}) over time for the Viewland Lake core. Climate data are specific to the Viewland Lake catchment area and were modelled using ClimateBC. Sedimentation rates were calculated using the CRS model.

Chapter 4

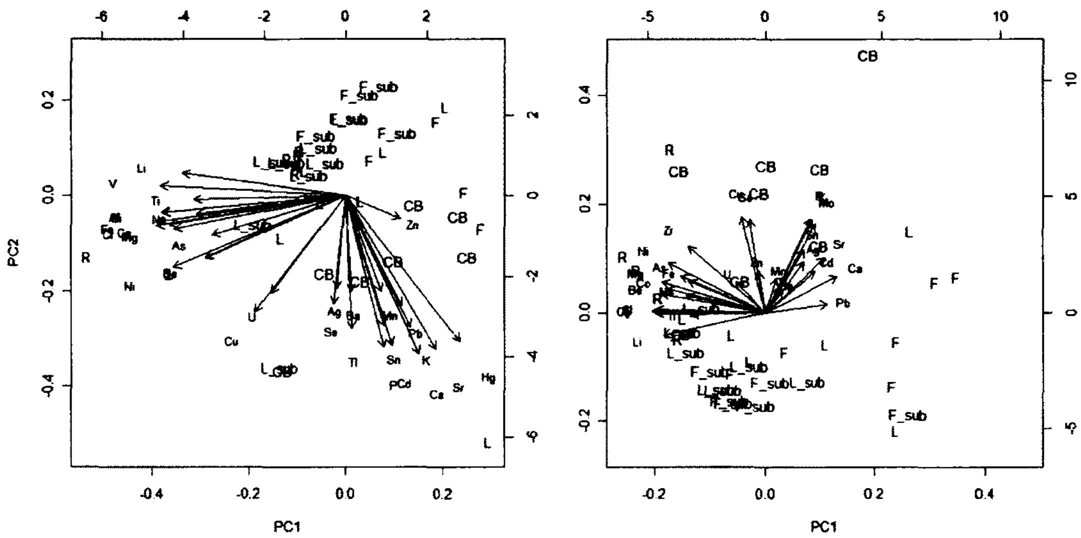
Results: Sediment source tracing

Chapter four focuses on the sediment source tracing procedure outlined by Collins et al. (1997). Sediment source groups have been redefined using visual observations and statistical analysis. Long-term changes in sediment source contributions to both study lakes and wetlands were reconstructed using a multivariate unmixing model. Changes in the proportions of source materials have also been related to the sedimentation rates and proxy indicators measured for their respective cores which were presented in Chapter 3.

4.1 Source groups

The Principle Component Analysis (PCA) of the geochemical properties for the sediment source materials collected throughout each catchment revealed no obvious separations of the original (*a priori*) source categories (Fig. 4.1). Through visual examination of the biplot of the first two principle components it was observed that all natural forest samples (i.e. surface soil and subsurface soil) were interspersed with logged samples. The fuzzy-k clustering analysis confirmed these findings with Dunn's coefficient (see description in Section 2.6.1) being greatest for one group (D=1). A summary of the Dunn's coefficients for all groups (k=1,2,3,4,5) is given in Table 4.1.

In the case of the channel bank samples, they generally fell within the range of the forest and logged samples, however, the channel bank samples tended to loosely cluster together. Similarly, the surface and subsurface samples belonging to each of the forest and logged groups showed a tendency to group together. A Kruskal–Wallis H-test was used to determine whether or not a sufficient number of geochemical properties could distinguish between



(a) Boswell Lake catchment (b) Viewland Lake catchment

Figure 4.1: Results of the Principle Component Analysis (PCA) of the fingerprint properties for (a) Boswell Lake and (b) Viewland Lake sediment source materials. F=forest, F_sub=forest subsoil, L=logged, L_sub=logged subsoil, R=road, CB=channel bank. Biplots represent the first two principle components of the PCA.

Table 4.1: Fuzzy k-means clustering results for Boswell Lake and Viewland Lake source materials.

	Dunn's coefficient	
No. of Groups	Boswell Lake	Viewland Lake
1	1.000	1.000
2	0.648	0.678
3	0.513	0.465
4	0.428	0.399
5	0.364	0.339

surface, subsurface and channel bank samples (i.e. a fairly simple source categorization) to justify carrying out the final steps of the sediment source tracing procedure. For the Boswell Lake catchment, a total of 24 out of 34 geochemical properties were found to have significantly different mean values (corresponding to each of the three source categories; see Table 4.2). Similar results were found for the Viewland Lake catchment with 29 out of 34 properties having significantly different mean values (Table 4.3). Based on these results it was determined that three source categories - surface soil material, subsurface soil material, and channel bank material - would be used in the subsequent sediment source tracing steps.

4.2 Boswell Lake catchment

4.2.1 Composite fingerprint

From the 24 fingerprint properties that were identified by the Kruskal–Wallis H-test to have at least one pair of significantly different means (Table 4.2), a composite fingerprint was developed to correctly label the source group for each sample. Using stepwise multivariate discriminant function analysis (MDFA)[1], Se and Al were found to be the most appropriate combination of properties to use in the multivariate unmixing model as they were able to correctly assign 100% of the source materials to their original groups (Table 4.4). An additional fingerprint property, Ba, was incorporated into the composite fingerprint to increase the discriminatory power of the composite fingerprint. The additional property was selected on the basis of the next property provided by the results of the stepwise MDFA which would result in overall lower Wilks' lambda values.

[1]This procedure selects a combination of fingerprint properties that minimizes Wilks' lambda and is able to distinguish source types within a given catchment. The selected properties then become the composite fingerprint to be used in the multivariate unmixing model.

Table 4.2: Kruskal–Wallis H-test probabilities (p) for distinguishing surface, subsurface and channel bank materials in the Boswell Lake catchment using individual fingerprint properties. Mean concentration values are also given for each fingerprint property for each source type.

Fingerprint property	Surface mean (mg kg^{-1})	Subsurface mean (mg kg^{-1})	Channel bank mean (mg kg^{-1})	p
Li	6.38	16.42	3.71	0.002*
Be	0.15	0.38	0.15	0.010*
Na	141	265	107	0.110
Mg	2654	7227	2505	0.018*
Al	9177	22085	7265	0.006*
Si	5588	13315	4430	0.006*
P	955	890	1154	0.044*
K	187	169	264	0.002*
Ca	14673	10487	39240	<0.001*
Ti	585	1119	346	0.073
V	34.0	84.9	23.1	0.002*
Cr	23.1	58.7	20.7	0.028*
Mn	1316	525	421	0.008*
Fe	11274	27715	9889	0.010*
Co	5.9	13.3	3.9	0.035*
Ni	12.3	31.9	13.8	0.078
Cu	19.7	52.1	58.1	0.003*
Zn	242.8	93.5	50.7	0.203
As	4.0	20.8	8.4	0.004*
Se	0.55	1.21	2.99	<0.001*
Sr	68.8	49.5	129.4	<0.001*
Zr	2.8	6.2	4.4	0.039*
Mo	0.72	0.69	1.09	0.104
Ag	0.37	0.63	0.61	0.039*
Cd	0.58	0.46	1.13	<0.001*
Sn	0.94	0.94	0.74	0.439
Sb	0.13	0.47	0.45	0.163
Ba	126.7	96.3	50.0	0.015*
W	0.06	0.14	0.03	0.592
Hg	0.19	0.07	0.23	<0.001*
Tl	0.21	0.19	0.16	0.207
Pb	11.62	7.55	6.06	0.008*
Bi	0.16	0.26	0.07	0.199
U	0.29	0.77	1.01	0.009*

* Significant at p=0.05

Table 4.3: Kruskal–Wallis H-test probabilities (p) for distinguishing surface, subsurface and channel bank materials in the Viewland Lake catchment using individual fingerprint properties. Mean concentration values are also given for each fingerprint property for each source type.

Fingerprint property	Surface mean (mg kg^{-1})	Subsurface mean (mg kg^{-1})	Channel bank mean (mg kg^{-1})	p
Li	5.03	13.05	5.68	0.002*
Be	0.18	0.43	0.25	0.010*
Na	160	301	204	0.110
Mg	2943	6638	4077	0.018*
Al	8181	18754	10472	0.006*
Si	4926	11272	6358	0.006*
P	1114	1060	1193	0.044*
K	205	191	226	0.002*
Ca	23122	11900	13350	<0.001*
Ti	689	1200	486	0.073
V	40.1	99.6	38.8	0.002*
Cr	31.4	73.4	34.4	0.028*
Mn	1085	805	241	0.008*
Fe	11272	33005	12603	0.010*
Co	5.0	13.1	5.0	0.035*
Ni	13.4	32.6	21.3	0.078
Cu	26.0	55.3	67.8	0.003*
Zn	94.7	109.0	45.9	0.203
As	1.6	4.1	1.5	0.004*
Se	0.65	1.17	1.53	<0.001*
Sr	94.7	80.9	71.6	<0.001*
Zr	3.4	6.4	5.3	0.039*
Mo	2.50	1.84	3.22	0.104
Ag	0.54	0.54	0.55	0.039*
Cd	0.78	0.57	0.42	<0.001*
Sn	0.45	0.49	0.41	0.439
Sb	0.10	0.14	0.19	0.163
Ba	125.8	104.8	65.4	0.015*
W	0.04	0.09	0.04	0.592
Hg	0.20	0.08	0.19	<0.001*
Tl	0.19	0.17	0.15	0.207
Pb	10.31	6.23	3.06	0.008*
Bi	0.03	0.06	0.02	0.199
U	0.59	0.89	0.72	0.009*

* Significant at p=0.05

Table 4.4: Fingerprint properties selected by the stepwise Multivariate Discriminant Function Analysis to distinguish source types in the Boswell Lake catchment.

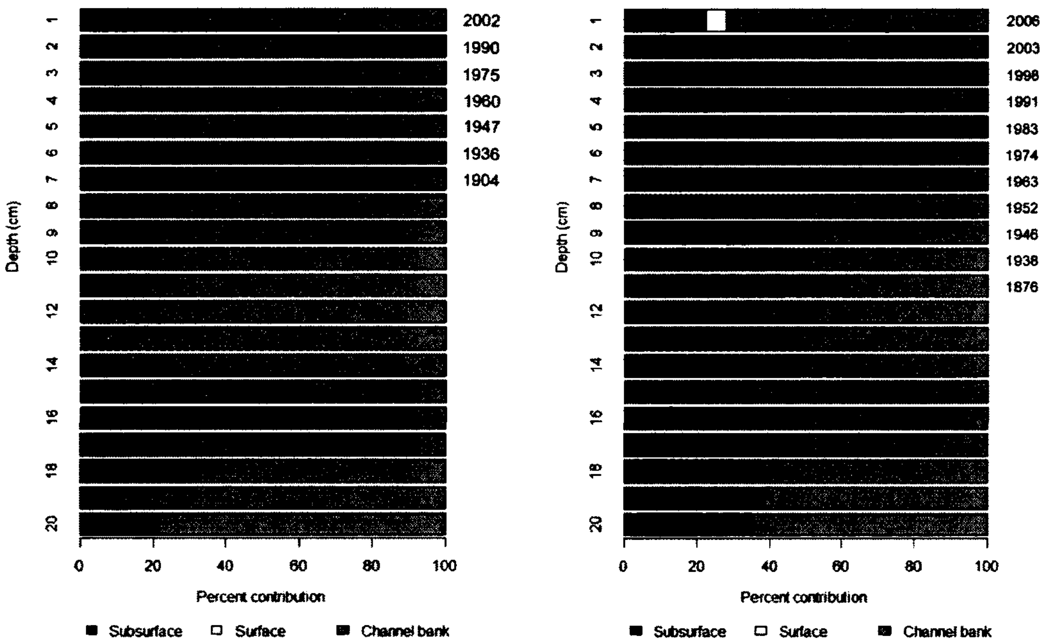
Fingerprint property	Wilks' lambda	Cumulative % source type samples classified correctly
Se	0.427	77.1
Al	0.241	100
Ba	0.130	100

4.2.2 Sediment source contributions

The results of the multivariate unmixing model for Boswell Lake and wetland cores are shown in Figure 4.2. The dominant sediment sources for both cores were channel bank and subsurface material. It is important, however, to remember when interpreting these results that there are errors associated with these percentages. The errors for the Boswell Lake and wetland source tracing results are summarized in Table 4.5. The percent relative errors are considerably higher than values that have been documented by other source tracing studies (Collins et al., 1997; Carter et al., 2003), which report that errors below ±15% provide an accurate interpretation of the source material proportions. Here, errors range from ca. 5-40% in the lake core, and ca. 13-50% in the wetland core. These higher error values likely reflect the high degree of overlap observed among the three source groups with respect to their geochemical characteristics (see Fig 4.1(a)).

At 6 cm down-core (ca. 1936), the lake core was composed of 100% channel bank material and gradually received a greater proportion of subsurface material up-core. A maximum of 26% subsurface material was reached at 2 cm depth. A slight decrease to 23% subsurface material was observed at 1 cm with a corresponding increase in channel bank material. The increase in subsurface material also coincided with the time periods during which logging was present (1960-1975, 1982-2008), approximately 3 to 4 cm depth.

The maximum percentage of subsurface material (62%) occurred at 11 cm depth (ca. 1876) in the wetland core (BL-D8). A general decreasing trend of subsurface material was then observed up-core until a minimum of 16% was reached at 2 cm. Surface material appeared along with subsurface and channel bank material in the top 1 cm, however, it only accounted for 6% of the material in the core. The beginning of the post-logging period, which occurred at approximately 7 cm depth in the wetland core, did not appear to correspond with any significant changes in sediment source contributions.



(a) Boswell Lake core (BL-P1)

(b) Wetland core (BL-D8)

Figure 4.2: Results of the multivariate unmixing model for the (a) Boswell Lake and (b) wetland cores. Values on the secondary y-axis represent the dates calculated using the CRS model for each 1 cm core slice containing detectable concentrations of ^{210}Pb. Each date aligns with the bottom of its respective 1 cm core segment.

4.2.3 Correlations

Table 4.6 provides a summary of all correlation coefficients and significance values. Sedimentation rates were not significantly related to changes in any of the source materials in the lake core. Changes in source materials were, however, highly correlated with other proxy indicators (see Table 4.6). In the Boswell Lake core (BL-P1), median grain size was found to be inversely related to percent subsurface material. Increasing proportions of subsurface

Table 4.5: Percent relative errors and standard errors for the unmixing model calculations for the Boswell Lake (BL-P1) and wetland (BL-D8) cores.

	BL-P1		BL-D8	
Depth (cm)	% Error	Standard error	% Error	Standard error
1	22.0	0.52	23.1	0.01
2	16.4	0.33	24.0	0.08
3	18.5	0.04	15.6	0.12
4	15.9	0.81	13.5	0.11
5	16.7	0.01	13.7	0.21
6	20.7	0.05	20.4	0.64
7	20.2	0.01	17.8	0.54
8	18.3	0.07	25.4	0.43
9	15.8	0.29	19.1	1.07
10	10.2	0.16	23.0	1.78
11	30.2	0.04	21.0	1.66
12	39.1	0.15	19.4	0.47
13	35.7	0.07	31.1	0.92
14	19.8	0.02	44.4	0.21
15	16.2	0.00	41.9	0.26
16	4.5	0.01	49.3	0.40
17	8.0	0.05	41.5	0.65
18	13.7	0.24	38.3	0.43
19	17.4	0.09	45.4	0.68
20	18.0	0.37	41.4	0.57

material were also associated with increases in total nitrogen. The reverse was true of channel bank material which had a positive relationship with median grain size and a negative relationship with total nitrogen.

Sedimentation rates in the Boswell wetland (BL-D8) were negatively correlated with subsurface material, and positively with channel bank material. dry bulk density and median grain size increased with increasing subsurface material, while percent water content was found to decrease with greater proportions of subsurface material. Percent channel bank material had the opposite relationship with these proxy indicators; dry bulk density and median grain size decreased, and percent water content increased with increasing relative contributions of channel bank material.

Table 4.6: Summary of the significant (p<0.05) correlations found between each source material and sedimentation rates, and proxy indicators.

	% Surface		% Subsurface		% Channel bank	
Lake (BL-P1)	r	p	r	p	r	p
Sedimentation rates		NS		NS		NS
dry bulk density		NS		NS		NS
Water content		NS		NS		NS
Magnetic susceptibility		NS		NS		NS
d_{50}		NS	-0.77	0.043	0.77	0.042
Total carbon		NS		NS		NS
Total nitrogen	0.74	0.057	0.82	0.023	-0.83	0.022
C:N		NS		NS		NS
Wetland (BL-D8)						
Sedimentation rates		NS	-0.67	0.024	0.62	0.041
dry bulk density		NS	0.88	<0.001	-0.85	<0.001
Water content		NS	-0.88	<0.001	0.86	<0.001
Magnetic susceptibility		NS		NS		NS
d_{50}		NS		NS	-0.92	<0.001
Total carbon		NS	0.92	<0.001	-0.93	<0.001
Total nitrogen		NS	-0.77	<0.001	0.83	<0.001
C:N		NS		NS		NS

NS = not significant

4.3 Viewland Lake catchment

4.3.1 Composite fingerprint

According to the Kruskal–Wallis H-test, 29 out of the 34 fingerprint properties were appropriate to continue on to the stepwise MDFA analysis (Table 4.3). From these 29 properties, Se and Pb were selected as the composite fingerprint as they were able to correctly identify the original source group for 100% of the source materials (Table 4.7). To ensure that the chosen composite fingerprint was a reliable source discriminator, a third fingerprint property, As, was added. Similar to the selection process for the source materials in the Boswell Lake catchment, As was chosen because it was the next property selected by the stepwise MDFA to lower Wilks' lambda values.

Table 4.7: Fingerprint properties selected by the stepwise Multivariate Discriminant Function Analysis to distinguish source types in the Viewland Lake catchment.

Fingerprint property	Wilks' lambda	Cumulative % source type samples classified correctly
Se	0.225	82.0
Pb	0.129	100
As	0.082	100

4.3.2 Sediment source contributions

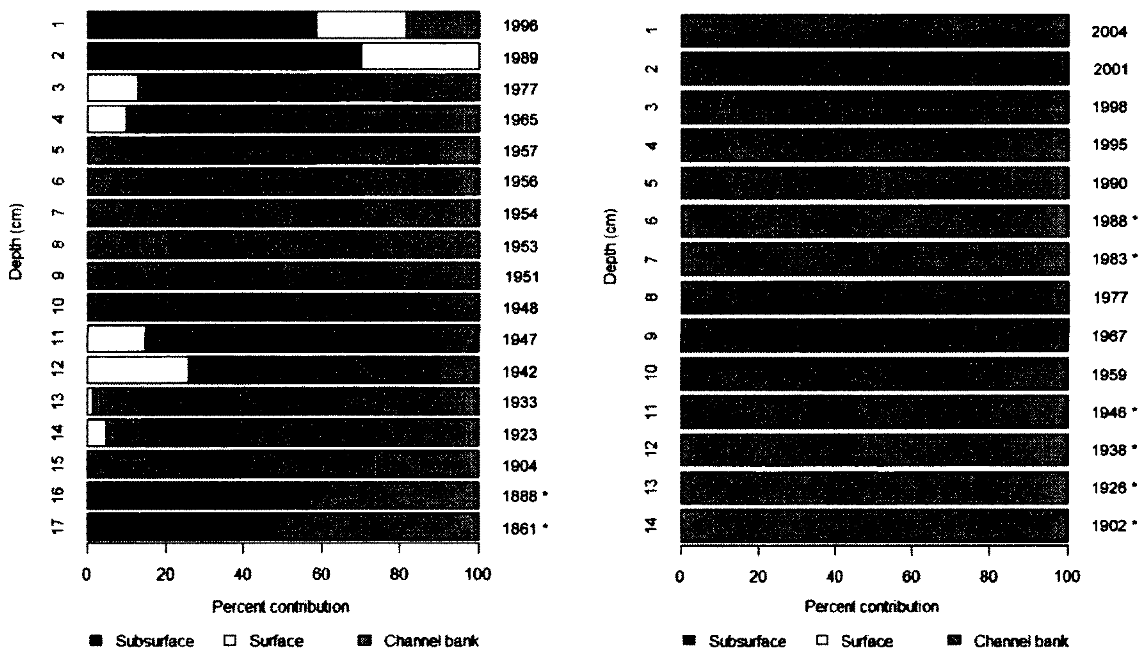
Results of the multivariate unmixing model are provided in Figure 4.3. When interpreting the relative changes in the source type contributions over time it is important to keep in mind the errors associated with these values. The errors for the Viewland Lake core range from approximately 39% to 81%. Errors for the wetland core are considerably lower and range from approximately 12% to 39%. Similar to the unmixing results for the Boswell Lake catchment, these higher errors are likely due to the high overlap observed for the geochemical characteristics of the source types (see Fig. 4.1(b)).

Periodic changes in source type contributions are found throughout the Viewland Lake core. In general, the dominant source contributing to the lake was channel bank material. Subsurface material is present at 16 and 17 cm down-core constituting approximately 57% and 40% of the sediment, respectively. Relatively small amounts of surface materials are present at 11 to 14 cm down-core and do not appear again until 4 cm. Subsurface material is the dominant source (70%) at 2 cm with surface material making up the other 30%. Surface and subsurface materials are still present in similar proportions in the top 1 cm, however, channel bank material is present and accounts for 19% of the material in the lake core. The timing of these changes in the top 2 cm of the lake core align well with the onset of forestry practices. Logging in the Viewland Lake catchment area occurred only in 1983 which coincides with 3 cm depth in the lake core.

The wetland core (VL-D1) did not show any variations in source material composition over time and was composed entirely of channel bank material.

Table 4.8: Percent relative errors and standard errors for the unmixing model calculations for the Viewland Lake (VL-P1) and wetland (VL-D1) cores.

	VL-P1		VL-D1	
Depth (cm)	% Error	Standard error	% Error	Standard error
1	47.0	0.05	14.2	0.09
2	44.8	2.28	12.0	0.10
3	80.5	1.17	12.8	0.09
4	63.2	0.85	13.5	0.11
5	62.5	0.55	14.0	0.06
6	61.3	0.47	13.3	0.11
7	49.1	0.48	23.3	0.17
8	51.4	0.48	28.5	0.27
9	49.8	0.25	30.3	0.34
10	61.0	0.65	31.7	0.40
11	57.6	1.01	28.7	0.25
12	50.7	1.22	39.1	0.62
13	60.2	0.86	45.3	0.91
14	56.4	0.83	38.1	0.52
15	56.2	0.66		
16	38.6	0.60		
17	38.7	0.58		



(a) Viewland Lake core (VL-P1)

(b) Wetland core (VL-D1)

Figure 4.3: Results of the multivariate unmixing model for the (a) Viewland Lake and (b) wetland cores. Values on the secondary y-axis represent the dates calculated using the constant rate of supply model for each 1 cm core slice containing detectable concentrations of ^{210}Pb. Each date aligns with the bottom of its respective 1 cm core segment. The asterisk (*) identifies core slices that were not corrected for particle size due to a lack of material.

4.3.3 Correlations

Correlation analyses were also used to determine if any relationships exist between sediment sources and sedimentation rates. Source material proportions were also compared to all proxy indicators (Table 4.9). Changes in sediment source materials were not found to have a significant relationship with sedimentation rates in Viewland Lake. They were, however, significantly correlated with several proxy indicators. Increasing dry bulk density and magnetic susceptibility, as well as decreasing water content, total carbon and total nitrogen were all associated with increasing percentages of surface materials in the lake core. The same was true of subsurface material, which was also positively correlated with C:N. Increases in the percentage of channel bank material were related to decreasing dry bulk density and magnetic susceptibility, and increasing values of water content, median grain size, total carbon and total nitrogen.

Variations in sediment source contributions were not compared to sedimentation rates in the Viewland wetland since sediment sources did not change over time and were composed entirely of channel bank material.

Table 4.9: Summary of the significant (p<0.05) correlations found between each source material and each proxy indicator for the Viewland Lake core (VL-P1).

	% Surface		% Subsurface		% Channel bank	
Proxy	r	p	r	p	r	p
Sedimentation rates		NS		NS		NS
dry bulk density	0.62	0.007	0.63	0.007	-0.72	0.001
Water content	-0.58	0.016	-0.58	0.015	0.66	0.004
Magnetic susceptibility	0.78	<0.001	0.54	0.024	-0.70	0.002
d_{50}		NS		NS	0.52	0.038
Total carbon	-0.75	<0.001	-0.60	0.011	0.73	<0.001
Total nitrogen	-0.61	0.009	-0.77	<0.001	0.83	<0.001
C:N		NS	0.57	0.016		NS

NS = not significant

Chapter 5

Discussion

The following chapter discusses the results for each study site independently as the characteristics and histories of their catchments are different. Subsequent sections explore the impacts of local versus regional factors, as well as the importance of landscape position. Limitations of the study and future research directions are also provided at the end of the chapter.

5.1 Boswell Lake catchment

5.1.1 Lake sediment

The results of the one and two sample t-tests indicate that lake sedimentation rates did not change significantly at any point during the post-logging period. A weak or absent logging signal in downstream lakes has been reported by other paleolimnological studies. Paterson et al. (1998) found that logging did not have a significant impact on lake chemistry and species composition, and indicated that site-specific characteristics such as buffer strips, or rapid re-growth of vegetation can minimize sediment transfers. However, slight increases in dry bulk density and magnetic susceptibility indicate a possible increase in the delivery of clastic sediment to the lake from allochthonous sources. Additionally, moderately high (>12) C:N values are present throughout the entire lake profile which suggests the majority of the sediment was delivered from a terrestrial source (Meyers & Ishiwatari, 1993), as opposed to an internal lake source. Furthermore, the underlying bedrock contains limestone ($CaCO_3$) which may cause the sediment to be enriched in inorganic carbon leading to high C:N values, although further research would be needed to confirm this.

Based on the source tracing results, the dominant sediment source to Boswell Lake is channel bank material, and the secondary source is subsurface soil material. After approximately 1947, the proportion of subsurface material delivered to the lake increased. During this same period, a shift in each of the proxy indicators occurred all providing evidence for increasing proportions of clastic-rich sediment. This change to increasing proportions of clastic-rich sediment becomes particularly obvious when dry bulk density and percent water content are examined over a longer time period (Fig. 3.7). The peak in dry bulk density at approximately 1976 (3 cm depth) is the highest it has been in over 2,410 years (where 56 cm down-core corresponds to the Bridge River tephra layer). The presence of subsurface material is, however, not sufficient to account for the unprecedented increase in dry bulk density since subsurface material has been observed before without similar increases in dry bulk density (e.g. 9 cm depth).

Based on the location of the Bridge River tephra layer, average sedimentation rates over the last 2,410 years were found to be approximately 1.9 x 10^{-3} g cm^{-2} y^{-1}. Compared to this value, sedimentation rates increased approximately 4-fold at the beginning of the 1940s and remained elevated throughout the remainder of the profile. High sedimentation rates coupled with the erosion of clastic-rich subsurface soil material could account for elevated dry bulk density values. However, since this shift to increasing proportions of clastic-rich sediment began in the 1940s, it cannot be attributed solely to forestry practices which began in 1960.

Statistically, neither variations in Boswell Lake sedimentation rates nor the sediment characteristics could be explained using changes in climate variables; yet, visible changes in several hydrometerological variables should be discussed. A break point and subsequent step increase in average stream discharge for the Quesnel River at Likely, BC occurred in 1944. This step change was accompanied by increased stream discharge variability, increased mean annual precipitation, and an increase in the number of frost-free days. Déry et al. (2009) reported a similar trend in northern Canadian rivers which they described as an intensification of the hydrological cycle. They noted that large-scale climate processes are known to produce step changes or trend reversals in the hydrological regime of North American rivers. With respect to the present study, 1945/46 represents a phase shift in the Pacific Decadal Oscillation (PDO) to a "cool phase". Cool phases in northwestern North America are charcterized by above average October-March precipitation, as well as above average snow pack and spring stream flow (Mantua & Hare, 2002). This cool phase ended in approximately 1976 and was followed by a warm phase which produced the opposite effect (Woo et al., 2006).

Surface erosion in an undisturbed catchment is dependent on climate, soil type, vegetation cover, and water input (Wondzell & King, 2003). Overland flow is not expected to have caused a significant increase in the erosion and delivery of sediment since it is unlikely that, based on this climate regime, rainfall intensity would have exceeded the infiltration capacity of the soil. Therefore, it is reasonable to assume that the majority of sediment erosion and delivery occurred within the channels. Hooke (1979) found that the amount of precipitation, antecedent soil moisture conditions, and peak discharge were strong predictors of channel bank erosion. Significant increases in the amount of precipitation after 1944 may have therefore driven channel bank erosion. Additionally, the magnitude and timing of spring snowmelt can strongly impact stream discharge. This biogeoclimatic zone receives up to 50% of its precipitation as snow. A significant increase in the frost-free period may have resulted in faster snowmelt thereby increasing channel bank erosion, and possibly the redistribution of channel bed sediment.

5.1.2 Wetland buffering function

The objective of this study was to characterize the response of the sediment trapping function of wetlands to a landscape disturbance (i.e. forestry practices). Several cores were taken throughout Boswell wetland, but only two were analyzed in this study. Cores taken from the far west channel were selected for analysis because they were successfully retrieved on the first attempt and surface sediments were minimally disturbed. Additionally, cores were taken from near the wetland inflow and outflow to attempt to characterize the distribution of sediment along the length of the wetland channel.

In general, the majority of sediment accumulation in the Boswell wetland occurred closest to the inflow and rapidly decreased moving toward the wetland outflow. This finding is consistent with the results of other studies that examined wetland sediment retention (Brueske & Barrett, 1994; Cahoon, 1994; French et al., 1995; Reed et al., 1997). Forestry activities did not have a statistically significant impact on sedimentation rates in either wetland core. However, peaks in the sedimentation rates of the wetland outflow core (BL-D10) correspond to both periods of forestry activities. The spatial distribution of sediment in a wetland has previously been attributed to the characteristics of the wetland and channel flow patterns (Hupp & Bazemore, 1993; Harter & Mitsch, 2003). The only noticeable difference between the characteristics of the two coring sites in the Boswell wetland channel was the water depth which increased with increasing distance from the wetland inflow. Harter & Mitsch (2003) observed that deeper areas within two experimental wetlands had higher sedimentation rates than shallower areas of the wetlands. Since sedimentation rates were generally higher near the inflow where the channel is shallower, then the physical characteristics of the sediment may have influenced its spatial distribution in the wetland.

Median grain size near the wetland inflow experienced a large decrease (i.e. sediment became finer) after the 1940s and remained significantly low during the active logging periods. A significant decrease in median grain size may have influenced the conditions necessary to facilitate sediment deposition. Finer material requires less stream power to remain in suspension and can therefore be transported over greater distances than coarser material (Duncan et al., 1987). However, the physical characteristics of the material near the outflow (i.e. reduced dry bulk density and relatively low magnetic susceptibility values) suggest that the sediment is not predominantly clastic. Despite the lack of sediment source tracing data for BL-D10, it could be assumed that a decrease in dry bulk density along with increased sedimentation rates may have been the result of enhanced channel bank erosion (based on the source tracing results of BL-D8). Nevertheless, deeper areas near the wetland outflow may have provided more ideal conditions for the deposition and storage of fine sediment.

Sedimentation rates in both wetland cores also possess the same step increase found in the Boswell Lake core at approximately 1940. Rates of sediment accumulation in the wetland core near the inflow (BL-D8; 5.2 x 10^{-2} g cm^{-2} y^{-1}) are more than double that of the core near the outflow (BL-D10; 2.3 x 10^{-2} g cm^{-2} y^{-1}) which suggests that this area of the wetland is providing a more effective trapping function. The 1940 step change also produced small decreases in dry bulk density and magnetic susceptibility. Although C:N was not greatly affected by the 1940 step change, indicating that the source of the sediment was still terrestrial, total C and total N experienced large increases shortly after 1940. Total C more than doubled, rising from 11% to 24%. Total N increased from 0.56% to 1.3%, almost tripling its pre-1940 value. These changes in the physical and chemical characteristics of the sediment provide strong evidence that this section of the wetland is trapping primarily allochthonous organic material (Meyers & Ishiwatari, 1993). Moreover, the strong positive correlation between sedimentation rates and channel bank material along with a corresponding increase in discharge variability suggests that channel bank erosion likely intensified during this time.

Although sharp increases in the sedimentation rates near the wetland outflow appear to correspond with the timing of logging, other drivers of wetland sedimentation rates should also be recognized. As a result of their sediment and nutrient trapping function, wetlands tend to be areas of high productivity. Sedimentation rates presented in the current study reflect total sediment accumulation over time. Trends in mean annual temperature (MAT) in the Boswell Lake catchment have been found to steadily increase over the last century by approximately 0.085°C y^{-1} (t=2.99, p=0.003). Therefore, it is possible that increases in primary production have contributed to increases in sedimentation rates.

5.2 Viewland Lake catchment

5.2.1 Lake sediment

The presence of active logging in the upstream areas of the Viewland Lake catchment did not have a statistically significant impact on the sedimentation rates found in Viewland Lake. Although a post-logging increase in sedimentation rates was observed, several much larger spikes in lake sedimentation rates occurred in the late 1940s to mid 1950s. The results of the stepwise linear regression for Viewland Lake sedimentation rates indicate that sediment delivery in this catchment is largely governed by regional climatic processes. Precipitation as snow experienced a large increase in the early 1900s, peaking in the late 1940s to mid 1950s, after which it declined until the end of the century. These observations correspond well with lake sedimentation rates (Fig. 3.12), as well as the phase changes in PDO. The shift to a cool phase in 1945/46 until approximately 1976/77 would have increased snow pack and spring freshet leading to greater runoff and sediment delivery (Mantua & Hare, 2002; Woo et al., 2006).

Despite consistently elevated lake sedimentation rates through the 1940s and 1950s, physical and chemical characteristics of the lake sediment did not change. A high background C:N suggests that the sediments are largely allochthonous and in-lake productivity does not make a significant contribution to Viewland Lake. A post-logging decrease in median grain size is consistent with the results of other studies that found logging activities to increase the production of fine-grained sediment (Reid & Dunne, 1984; Tague & Band, 2001). Sharp increases in both dry bulk density and magnetic susceptibility have previously been associated with periods of land clearance (Thompson et al., 1975; Lott et al., 1994). Thompson et al. (1975) also noted a decrease in total carbon content, and concluded that these changes were all indicative of increasing contributions of inorganic allochthonous material. These changes in the physical properties of the sediment are also supported by the source tracing results which show evidence of a change from predominantly channel bank material to a mix of subsurface material and surface material (Fig. 4.3(a)).

The possible driver of the two increases in the proportion of subsurface materials found at the bottom of the core (16 and 17 cm) cannot be identified using the current dataset. Records of landuse activities are not available for the mid to late 1800s. As well, ClimateBC is only able to provide modelled climate data beginning in 1901. It has been recognized that a particle size correction was not available for either of these core slices; however, the inclusion of a particle size correction would not drastically alter the source tracing results such that subsurface material would be excluded from the source tracing results. Based on the changes that have occurred during the last century it is unlikely that subsurface materials would have been transported in such high proportions without a significant disturbance. For example, mining activities would provide the necessary disturbance to increase the erosion and delivery of subsurface material to downstream waterbodies, and was also a prominent land use disturbance during this time (late 1800s). However, when considering the physical and chemical changes of the lake sediment associated with the presence of subsurface material, one would have expected to observe an increase in dry bulk density, magnetic susceptibility and C:N, none of which appear in the bottom 2 cm.

5.2.2 Wetland buffering function

Statistically, the timing of forestry practices was a significant predictor variable of wetland sedimentation rates; however, sedimentation rates began to rise in the early 1900s while logging only occurred in 1983. Although climate variables were not found to explain any additional variation in wetland sedimentation rates, an overall increase in mean annual precipitation of 0.466 mm y^{-1} (t=1.58, p=0.117) may have driven increased sediment transport. A similar line of reasoning was also used by Foster (1995) who could not find a detectable trend in annual precipitation, but suspected that an increase in precipitation provided the energy necessary to increase sediment yield. As was discussed for Boswell wetland, possible increases in primary production should be taken into account as they may have contributed to wetland sedimentation rates. MAT was also found to significantly increase over the last century by approximately 0.0084°C y^{-1} (t=3.01, p=0.003). The steady increase of wetland sedimentation rates could thus be the result of increased MAT driving primary production.

During visual inspection of the core it was found that above 10 cm (mid 1950s) the dominant material changed from dark brown organic sediment to woody debris. This is consistent with low magnetic susceptibility values, and decreases in dry bulk density and high C:N values. Cahoon (1994) found that unmanaged wetlands had higher rates of organic matter accumulation, especially closer to the wetland inflow. However, he also observed an increase in the accumulation of minerogenic material which was not observed in the Viewland wetland. If the wetland was performing a buffering function then evidence of minerogenic material should have been found in the wetland core. Additionally, such high proportions of surface and subsurface material should not have been observed in the lake core. Several explanations for the lack of subsurface material in the wetland core, as well as the presence of subsurface material in the lake core, have been described below.

When wetland coring was being carried out (late summer) it was noted that water was not present in the Viewland wetland channel. In general, wetlands that are permanently inundated provide a more effective sediment trapping function (Johnston, 1991; Hupp & Bazemore, 1993). As well, the ephemeral nature of flooding in this channel would have prevented the growth of wetland vegetation which play an important role in sediment trapping and stabilization of the wetland bottom. A study by Duncan et al. (1987) tested the ability of two ephemeral channels to capture various sediment grain sizes derived from logging roads. They found that throughout periods of active flow, differences in channel length, vegetation density and the amount of woody debris affected the ability of ephemeral channels to reduce the delivery of sediment greater than 63 μm to the mouth of the channel, regardless of stream discharge. However, these characteristics became less important with sediment less than 63 μm. Only during extremely low flow conditions was fine sediment retained in the channels, and slight increases in discharge resulted in the resuspension of previously deposited fine sediment.

The presence of logging roads in the Viewland Lake catchment is thought to be a possible driver of fine-sediment production. Road density in this catchment is approximately 4-times that of the Boswell Lake catchment (2.76 km km^{-1} in the Viewland Lake catchment versus 0.67 km km^{-1} in the Boswell Lake catchment). In general, surface erosion due to overland flow in the Viewland Lake catchment would not have contributed a large amount of sediment to surface waters since it is unlikely that rainfall intensity would not have exceeded infiltration capacity of the soil. However, surface erosion of roads and subsequent sediment transport would have been possible since the infiltration rate of compacted road surface is quickly exceeded during rainfall events. Furthermore, the arrangement of roads relative to streams, and the number of road-stream crossings can modify the direction and magnitude of water flow (Jones et al., 2000). The direct connection between the logging road and the Viewland channel likely increased the rate of sediment delivery from the road surface to the stream network.

The Viewland Lake catchment was originally included in this study as a secondary site, and as a result, only one core was collected from the wetland. An upper section of the lake exists which is also bordered by a wetland with a channelized inflow and outflow (see Fig. 2.3). This upstream area was also impacted by forestry practices in 1983. Assuming low water and sediment residence times for the upper lake, and negligible wetland buffering, sediment could have been transported from another area of the cutblock to Viewland Lake.

Finally, a particle size correction was not available (due to the lack of sediment for analysis) for the wetland core slice that, based on the timing of logging, would have been impacted by logging (6 cm). However, the inclusion of a particle size correction should not have impacted the rank order of the source groups and channel bank material would have remained the dominant source type. Additionally, changes in the physical and chemical characteristics do not provide any evidence of an input of subsurface material.

5.3 Importance of landscape position

The concept of hydrological connectivity is important when considering the position of a disturbance in the landscape and its potential to increase sediment yield. At the patch scale, the factors that have the greatest influence on sediment yield are slope angle, slope length, and whether runoff will enter a dispersive or channelized pathway (Bracken & Croke, 2007). Spicer (1999) used an impact factor in his statistical models to relate likely travel path and path distance to sedimentation rates in lakes in the central interior of British Columbia. The impact factor took into consideration slope angle as well as the path of least resistance down the hillslope. Steeper areas located a shorter distance to a stream or channel were more likely to deliver sediment to the downstream lake. While, due to sample size, it was not possible to incorporate an impact factor into the present linear models, the concept of hydrological connectivity can be used to help explain the spatial distribution of sediment in Boswell wetland, as well as the presence of subsurface material in Viewland Lake.

Although logging activities had no apparent effect on Boswell Lake sedimentation rates (see Table 3.2), a disturbance response was observed in the wetland channel from which cores were collected and analyzed. Notably, the wetland core near the wetland outflow (BL-D10) exhibited the strongest responses to both logging events. These logging events resulted in similar areas of deforestation, and both occurred on slopes of moderate inclination. The main difference between these two events was their locations in the catchment relative to the channel and the wetland. The first logging event was higher in the catchment while the second occurred further downstream. By reducing the path length there would have been less opportunities for in-channel deposition and storage of sediment. Additionally, the decrease in median grain size observed near the wetland inflow (BL-D8) supports the idea that an increase in the length of the wetland channel would have been necessary to encourage sediment deposition, unless extremely low flow conditions were present.

5.4 Local versus regional effects

A conceptual framework developed by Bracken & Croke (2007) identified five components involved in the hydrological connectivity of a catchment. Figure 5.1 shows four of the components surrounded by the fifth component, climate. Water and sediment yield, while they are strongly influenced by landscape position, delivery pathway, runoff potential and lateral buffering, all are driven by climate variables. The results of this study fit well within this framework as sedimentation rates in both Boswell Lake and Viewland Lake appear to be largely driven by regional, medium-term (i.e. decadal) climatic events rather than short-term localized logging events. In terms of forestry practices, other studies have also found that lake sedimentation rates (Blais et al., 1998) and resultant lake conditions (Paterson et al., 1998) were more strongly influenced by regional climatic processes.

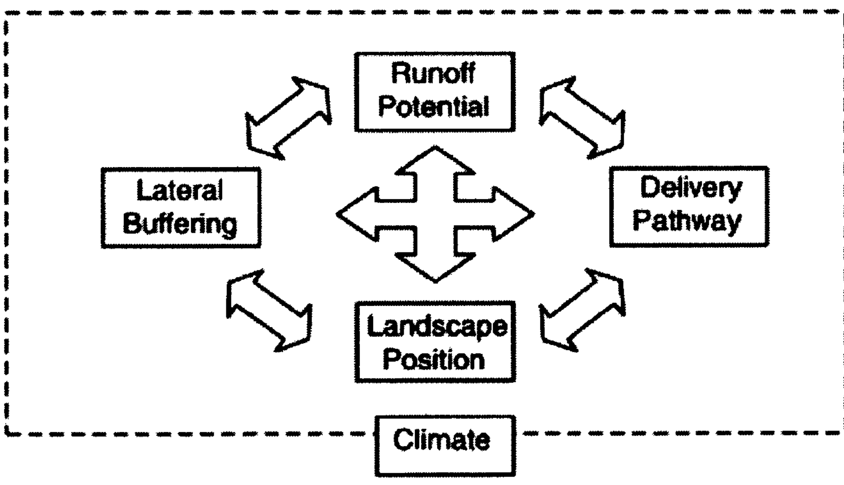


Figure 5.1: The components of catchment connectivity (from Bracken & Croke (2007)).

Although forestry practices produced significant responses in sedimentation rates near the outflow of Boswell wetland (BL-D10), these changes were short-lived and pre-logging conditions were soon re-established (within approximately four years) . Comparatively, the step increase in Boswell Lake and wetland sedimentation rates beginning in the 1940s was sustained throughout the remainder of the sediment profiles and did not return to pre-1940 rates. Similarly, the post-logging change in the dominant sediment source to Viewland

Lake from channel bank material to subsurface and surface materials was episodic, although the system has not yet returned to pre-logging conditions (i.e. dominated by channel bank material). Ambers (2001) suggested that logging practices "enhance the effect of big storms". In otherwords, forestry practices prepare the landscape for erosion and sediment transport, but ultimately suitable hydrological and climatic conditions (i.e. runoff) are required to mobilize and deliver sediment to downstream waterbodies. This is consistent with the idea that sediment delivery is limited by the total sediment supply. Therefore, forestry activities have the potential to increase the amount of sediment on the hillslopes available for transport, but do not necessarily result in the immediate increase in wetland or lake sedimentation rates.

5.5 Study limitations

The interpretation of the results presented is heavily dependent on the accuracy of the core chronologies. Conclusions made in this study have relied upon the changes in sedimentation rates, as well as changes in the physical and chemical characteristics, with respect to the timing of forestry practices in the catchment. Though, given a different core chronology the conclusions drawn from this study may have been different. As discussed in Section 2.3.4, the constant rate of supply (CRS) model was selected because its assumptions were best satisfied given the unsupported ^{210}Pb profiles. However, the CRS model is not without its flaws. The calculated ages at the bottom of the profile tend to be over-estimated, and the model does not take into consideration variable fluxes of ^{210}Pb as in the sediment isotope tomography (SIT) model. Additionally, the assumption of minimal post-depositional changes to supported and unsupported ^{210}Pb was made as core chronologies could not be verified using ^{137}Cs activities.

Post-depositional changes to the sediment record are important to consider as they affect the reconstruction of the core chronology and the interpretation of past environmental conditions. Wetland cores were taken from the channels flowing through the wetlands since it was suspected that the channels are the major delivery pathways for water and sediment moving down the hillslopes. Water flow in these channels was observed to be negligible and would have facilitated sediment deposition. However, any increases in flow may have resulted in the erosion of the channel bottom and the subsequent redistribution of the deposited sediment. Figure 3.3(b) shows the ^{210}Pb and ^{137}Cs profiles for the Viewland wetland core. Compared to the Viewland Lake core (Fig. 3.3(a)), the activities of these radionuclides experience greater fluctuations over time. Erosion of the wetland bottom due to increases in channel flow may have caused these variations. Unconformities or hiatuses in the sediment record as a result of erosion would produce under-estimates of the total sedimentation rates in the wetland channels.

Two important aspects of the wetland were not considered in this study which may have played a critical role in sediment trapping. Firstly, with changing discharge patterns over the last century, the water level in the lake and wetland would have been affected. Dead Black Spruce trees (*Picea mariana*) were found scattered throughout the wetland suggesting that the water level would have been sufficiently low at one time to allow for tree seed germination and tree growth. If channel length and depth are in fact critical for sediment trapping in wetlands, then water level fluctuations would have impacted where sediment deposition would have occurred in the wetland. As seen in Boswell wetland, a sediment trapping response to deforestation was more prominent in the coring site closest to the wetland outflow, presumably due to a fining of the sediment. With respect to sample collection, coring locations were selected based on the present-day wetland boundaries which were determined using spatial data obtained from the British Columbia Ministry of Forests and Range, as well as observed vegetation type. If locations of sediment deposition changed with changing water level, then the cores taken may not be representative of an overall wetland buffering function. Therefore it is possible that the changes in sedimentation rate noted for the wetland core collected at the wetland-lake interface (which coincided with the timing of forestry activities) may be due to changes in lake level caused by increased runoff to the lake as a result of forest harvest activities (i.e. tree removal and increased road network).

Secondly, and also related to the first point, wetland vegetation type was not an integral component of the study. During site investigation and coring it was noted that the dominant species in both wetlands were sedges (*Carex spp.*) and the Yellow Water Lily (*Nuphar variegata*), however, the presence of these species and their spatial distributions could have changed depending on the extent of the water level fluctuations. This second point is not of major concern as it was pointed out by Duncan et al. (1987) that wetland vegetation did not have an impact on the deposition of the finest grain sizes (<63 μm) in channels, and stream discharge was a more important factor for sediment deposition. However, it is important to note that under certain conditions (i.e. ponded wetlands), vegetation type and density are strong determinants of sediment trapping and resuspension.

5.6 Future research directions

Certain aspects of the study design have limited the conclusions on the sediment trapping function to be extended to the full areas of Boswell wetland and Viewland wetland. Only two cores from Boswell wetland, and one from Viewland wetland, were analyzed. Although the assumption of negligible overland flow in the wetland area has been made, an additional three channels exist in each of the study catchments. A better overall assessment of the wetlands' buffering functions could have been attempted had cores from each of those channels been analyzed. Additionally, one of the channels in the Viewland Lake catchment was not affected by forestry practices. Natural temporal variability of these systems was established using a temporal control (i.e. pre-logging conditions), however, analysis of a core from the unaffected channel would have provided an appropriate control throughout both the pre- and post-logging periods. Therefore, future studies attempting to evaluate the sediment buffering function of wetlands should consider selecting a study site which has a combination of impacted and unimpacted areas.

This study, like many others, has selected small catchments to address the research questions and objectives. The advantage of studying a small catchment is that they are generally less complex and offer fewer opportunities for terrestrial sediment storage which increases the lag time between sediment mobilization and delivery to the catchment outlet. Additionally, these catchments were selected because the logged hillslopes were moderately steep and directly connected to the channels and thus the downstream wetlands. However, many catchments do not have such a simple topography, possess only one lake and one wetland, and have a history of only a single land use type at one point in time. These spatial and temporal complexities make cumulative effects of disturbance on sediment delivery and water quality difficult to understand, especially when the disturbance(s) is a non-point source. Similarly, understanding the cumulative effects of wetland functions on sediment quantity and water quality is not a straightforward task, and has been largely unstudied.

One study on the cumulative effects of wetlands on sediment and water quality was conducted by Johnston et al. (1990) who analyzed 33 watershed variables extracted from aerial photographs, along with water quality data, for 15 watersheds in the Minneapolis–St. Paul, Minnesota, USA metropolitan area. This study aimed to identify wetland characteristics which are, statistically, more likely to impact stream water quality (i.e. improve or degrade) and quantity. Johnston et al. (1990) found that wetland proximity was an important factor in determining the water quality of downstream adjacent lakes and streams, and the effect of wetland functions on water quality was not detectable downstream of the wetland. It was recognized that further work needs to be done to determine the distance relationships between wetlands and downstream water quality. Moreover, this thesis has demonstrated that when landscape disturbances (e.g. forestry practices), which change the physical characteristics of the eroded sediment (e.g. grain size), are coupled with hydroclimatic processes which increase sediment delivery (e.g. runoff), sediment retention in wetland buffers may either: a) not occur, or b) be limited due to sediment redistribution from wetland channel bottoms. Therefore, future studies on the cumulative effects of wetlands on water quality need to also consider how landscape disturbances have altered the hydrological connectivity of the watershed and the physical characteristics of the sediment, all in the context of local and regional climatic processes which are in a constant state of change.

Chapter 6

Conclusions and management implications

6.1 Conclusions

The results of this thesis provide a description of how the sediment trapping function of two central interior British Columbia wetlands has changed over time in response to forestry activities and climate processes, in particular precipitation and snowfall. Previous studies have identified the need to understand wetland functions and their contributions to water quality over a long time frame as much of the literature contains primarily contemporary studies. This study has addressed this gap and has also provided information on the origin of the sediment deposited in the wetlands and the lakes that they buffer, once again in a temporal context of the last century. More specifically, this thesis aimed to determine if sedimentation rates and sediment source proportions in two wetlands and their downstream lakes were impacted by upstream forestry practices.

It was demonstrated that forestry practices produced a strong increase in Boswell wetland sedimentation rates which was not observed in Boswell Lake. This suggests that Boswell wetland provides a buffering function which has not been compromised by an increase in sediment delivery. Nonetheless, differences in sediment deposition between the two wetland sampling sites (BL-D8 and BL-D10) suggest that certain areas of the wetland provide a more effective sediment buffering function than others. The effectiveness of the sediment retention function of wetlands has previously been related to several factors intrinsic to the wetland, including percentage of wetland coverage, vegetation type and density, and channel depth and length. Channel depth and length may have been an important factor in the Boswell wetland as stronger post-logging responses were observed near the Boswell wetland outflow where the channel was deeper. However, sedimentation rates were generally higher near the wetland inflow which is consistent with the spatial distribution of sediment deposition reported for previously studied wetlands.

A significant decrease in median grain size near the wetland inflow suggests that differences in the sedimentation rates between these two sampling sites and their reponses to forestry practices may also be related to the characteristics of the sediment. It is well-known that, in addition to the water flow conditions, the properties of the sediment also affect settling rates. Additionally, forestry practices have been reported to increase the production of fine-grained sediment. A fining of the sediment delivered from the hillslopes could have increased the distance necessary for sediment deposition to take place. Similarly, in the Viewland Lake catchment, there was no evidence of wetland storage of minerogenic subsurface material, however, lake sediment was predominantly composed of subsurface material and also experienced a significant decrease in median grain size post-logging. It has been suggested that the ephemeral nature of the wetland channel, and the smaller width of the wetland (ca. 30 m) limited the buffering function of the wetland. Therefore, it is possible that the Viewland wetland did not provide a sufficient "buffering distance" to capture the fine-grained subsurface material observed in the lake.

This study also showed evidence of a strong climatic control on wetland and lake sedimentation rates. An intensification of the hydrological regime, which produced an increase in both the variability and the magnitude of mean annual precipitation and stream discharge in the Boswell Lake catchment, may have been the result of the 1944/45 shift to a cool phase in the Pacific Decadal Oscillation. Likewise, sedimentation rates in Viewland Lake were found to be strongly influenced by snowmelt. While it is unclear as to why these two catchments, which are relatively close in proximity, are impacted by two different climate forcings, these findings are ultimately consistent with those of others who have found that lake sedimentation rates were largely controlled by the amount of runoff generated.

Since Boswell Lake was not significantly impacted by the forestry practices, it was not possible to identify a recovery phase. The sedimentation rates at the outflow of the Boswell wetland have returned to pre-logging rates, however, they appear to have remained within the new climatic regime that began in the early 1940s. A consistent increase in the sedimentation rates near the wetland inflow suggests that sediment production from the hillslopes has not returned to pre-logging rates. Alternatively, increases in sedimentation rates were strongly related to increasing contributions of channel bank material which may have been eroded during periods of increased discharge.

Based on the physical and chemical characteristics of the sediment, recovery to pre-logging conditions has already occurred in Viewland Lake, however, the results of the unmixing model do not entirely support this finding. Subsurface material continued to be the dominant source material in the top 1 cm of the lake core. Since the dominant driver of subsurface sediment mobilization in this catchment has been assumed to be road construction, recovery of this system will depend strongly on the amount of use the road receives and the amount of sediment mobilized during road deactivation (i.e. culvert decommissioning).

Phillips (1989) stated that wetlands offer sites primarily for temporary sediment storage. Johnston (1991) argued that wetlands are more likely to provide permanent storage, but also recognized that the importance of a wetland as a "storage compartment" depends on the flux into the wetland and the duration of retention. Forestry practices in both catchments, as well as local and regional climatic influences, were shown to impact both the amount of sediment, and the dominant sediment sources. Boswell wetland offered a more effective sediment buffering function than Viewland wetland which has been attributed to the depth and length of its channel, and thus the length of the wetland buffer. However, differences in the spatial distribution of sediment along the channel were likely influenced by the position of logging in the catchment relative to the wetland and lake, as well as the characteristics of the sediment produced by natural erosive processes and forestry practices.

Finally, Smol (1991) recognized that the flow of information between "neolimnologists"[1]

[1]Smol (1991) used the term "neolimnologist" to refer to a limnologist working with present-day aquatic systems.

and paleolimnologists needs to be bidirectional. Paleolimnology requires the understanding of present-day processes to interpret long-term findings, while conversely neolimnology should assess present-day processes in a long-term context. The purpose of this exercise would be to understand the importance of "unusual" events on a broader temporal scale. Based on the findings of this research I would extend this recommendation to paleolimnologists whose research focuses on the "medium-term". Changes in sedimentation rates and the characteristics of the sediment which appeared insignificant over the last century, proved to be meaningful over a longer time frame. Furthermore, larger scale processes such as climate forcings have been shown here and in other studies to have a greater and more prolonged influence on hydrological regimes and sediment delivery than forestry practices.

6.2 Management implications

According to the Forest Planning and Practices Regulations under the Forest and Range Practices Act set out by the British Columbia Ministry of Forests and Range, riparian reserve zones[2] are not required for fish-bearing streams with a bank-full width less than 1.5 m or non-fish-bearing streams (FPPR s.47(4)). Riparian reserve zones for wetlands are not required by this same legislation where wetlands are less than 5 ha in size, and not in dry or wetland sensitive bigeoclimatic zones. Similarly, wetland complexes that are not located in dry or wetland sensitive biogeoclimatic environments also do not require reserve zones if they have an aggreagate area less than 5 ha (FPPR s.48).

A Riparian Management Area Guidebook has been established under the regulations to better define the purpose of these areas and how they should be applied to streams and wetlands. The objectives of the riparian management areas fail to recognize the importance of small streams as sediment delivery pathways, and the implications of increased sediment loading via these pathways as a result of forestry practices. This thesis has demonstrated

[2]Riparian reserve zones are defined as zones within the riparian management area that are intended to protect fish, wildlife habitat, biodiversity and the water values of the riparian reserve zone.

that while wetlands can provide a sediment trapping function, this function can be impacted by the both the wetland and stream/channel characteristics, as well as the nature of the mobilized sediment.

Furthermore, one must consider the prevalence of small streams and wetlands in the landscape versus that of larger streams and wetlands. In the case of the Quesnel River Basin, only 12% of the wetlands are greater than or equal to 5 ha while 88% are less than 5 ha. Watershed activities should take into account the cumulative effects on sediment yield at the basin scale and the consequences of not adequately protecting these smaller areas which are much more numerous. However, this then raises the point of, what is "adequate protection"? Foster et al. (2011) recognized that paleolimnology offers the potential for watershed managers to develop site-specific baseline data which can inform water quality guidelines and therefore management decisions. When regarded in association with wetland and stream characteristics and their potential for sediment deposition under applicable climatic and disturbance regimes, a more appropriate and comprehensive management strategy may be achieved.

6.3 Final remarks

Wetlands perform important hydrologic and geomorphic functions including buffering downstream waterbodies from accelerated hillslope erosion. However, landscape disturbances such as forestry practices have been shown to alter the physical characteristics of the sediment which, under certain hydrological conditions (i.e. high discharge) can cause the sediment trapping function of wetlands to be impaired. Therefore, additional research is needed to improve our understanding of wetland functions and their contributions to water quality, and the conditions under which landscape disturbances diminish those functions.

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Appendix A

Bathymetric maps

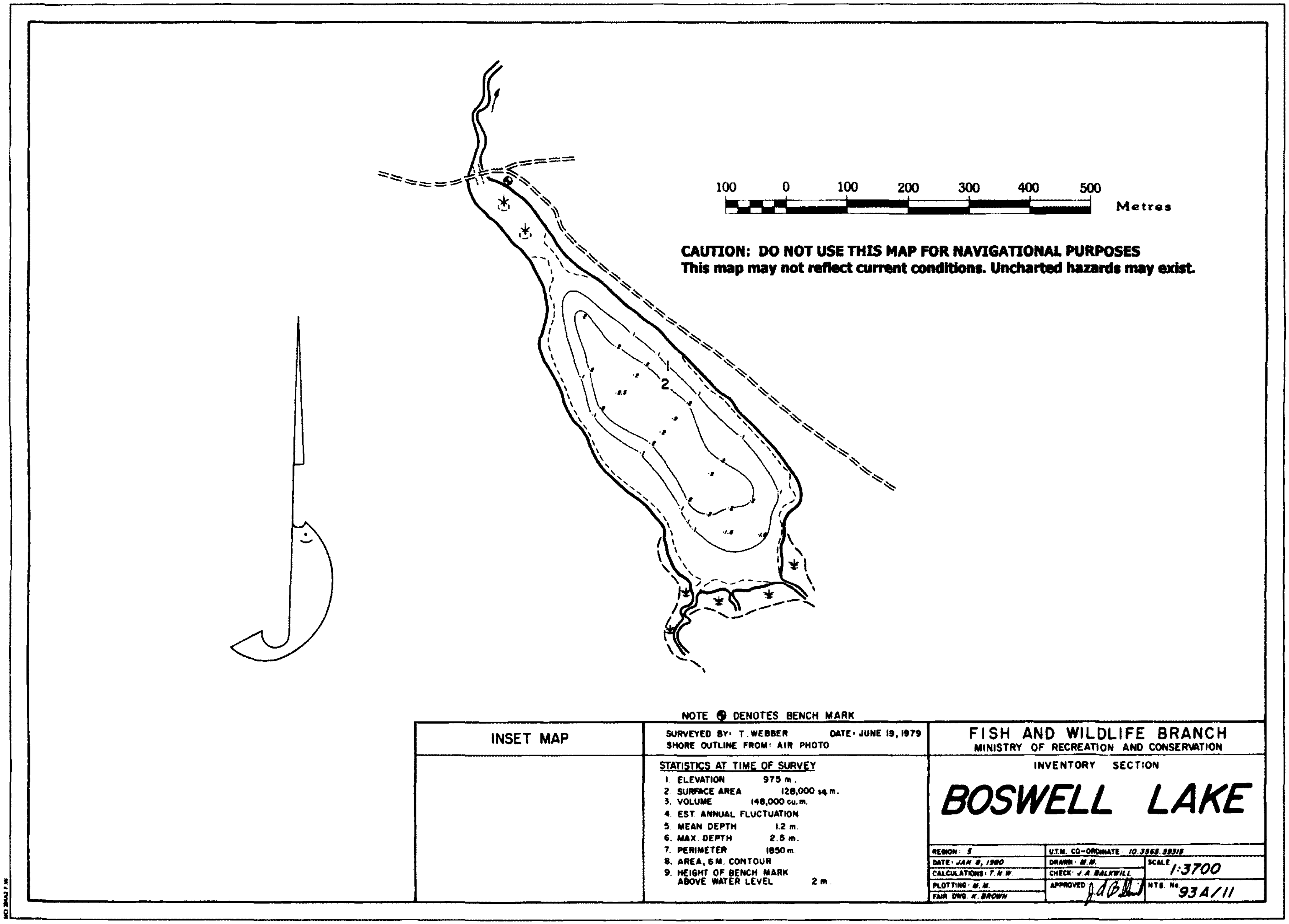


Figure A.1: Bathymetric map for Boswell Lake. Map was obtained online from the Anglers' Atlas.

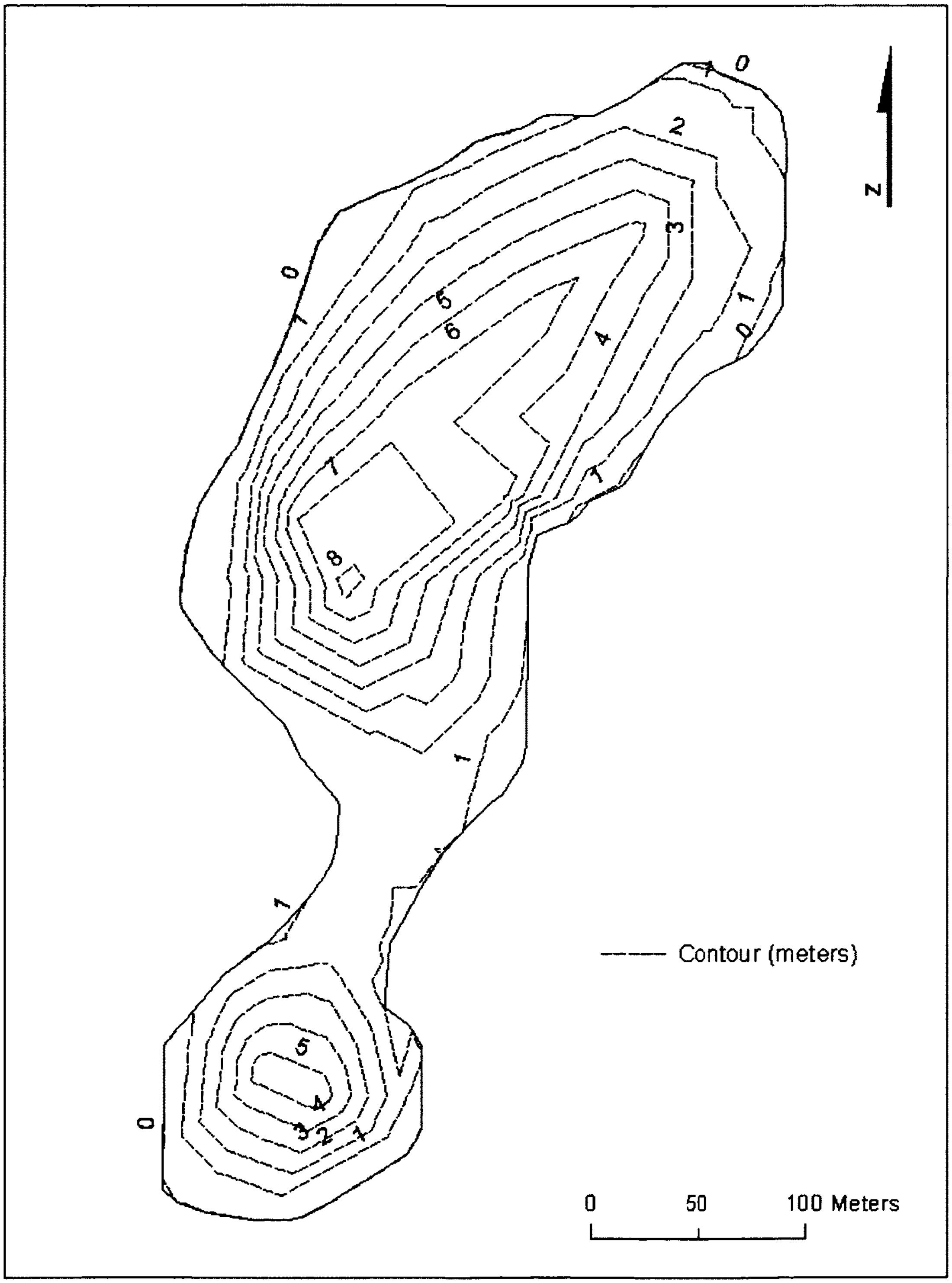


Figure A.2: Bathymetric map for Viewland Lake. Map was created in ArcGIS using latitude-longitude coordinates and water depths obtained during a depth survey of the lake.

Appendix B

ClimateBC variables

Directly calculated variables:

MAT mean annual temperature (°C)

MWMT mean warmest month temperature(°C)

MCMT mean coldest month temperature (°C)

TD temperature difference between MWMT and MCMT (°C)

MAP mean annual precipitation (mm)

MSP mean annual summer (May to September) precipitation (mm)

AH:M annual heat:moisture index ((MAT+10)/(MAP/1000))

SH:M summer heat:moisture index ((MWMT)/(MSP/100))

Derived variables:

DD<0 degree-days below 0 °C, chilling degree-days

DD>5 degree-days above 5 °C, growing degree-days

$DD5_{100}$ the Julian date on which DD>5 reaches 100, the date of budburst for most plants

DD<18 degree-days below 18 °C, heating degree-days

DD>18 degree-days above 18 °C, cooling degree-days

NFFD the number of frost-free days

FFP frost-free period

bFFP the Julian date on which FFP begins

eFFP the Julian date on which FFP ends

PAS precipiation as snow (mm)

EMT extreme minimum temperature over 30 years

Appendix C

Microscope image of tephra

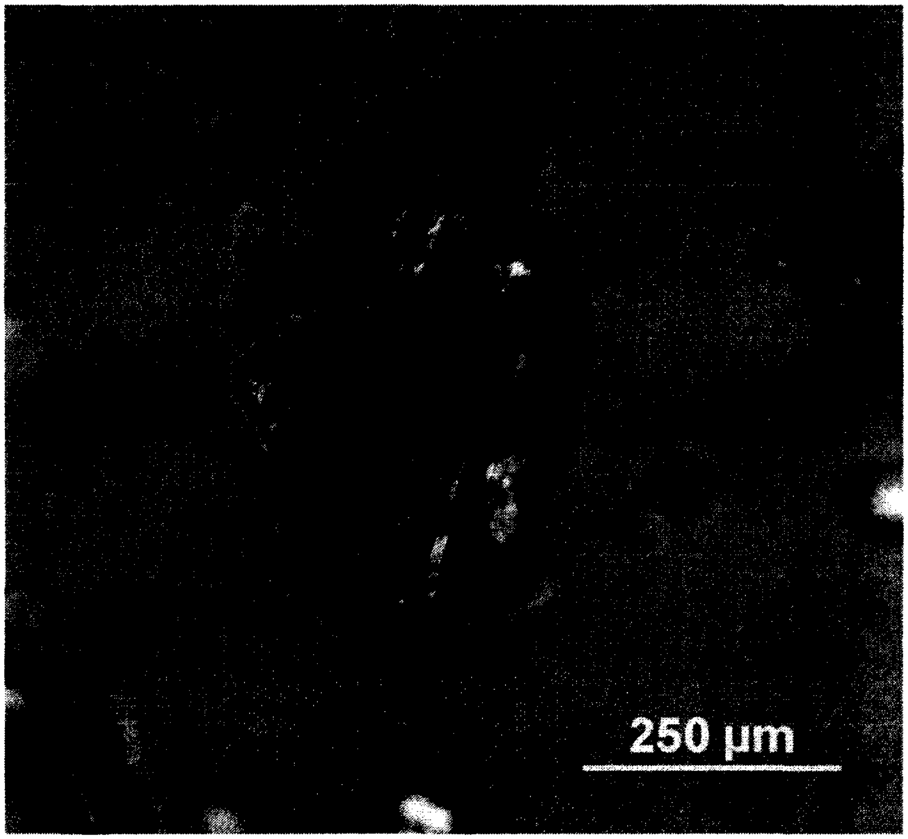
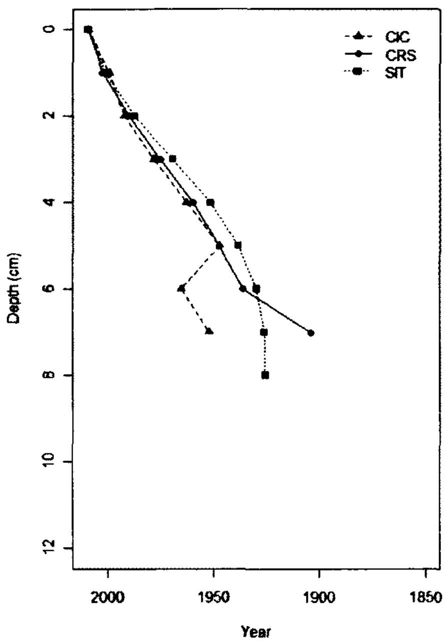


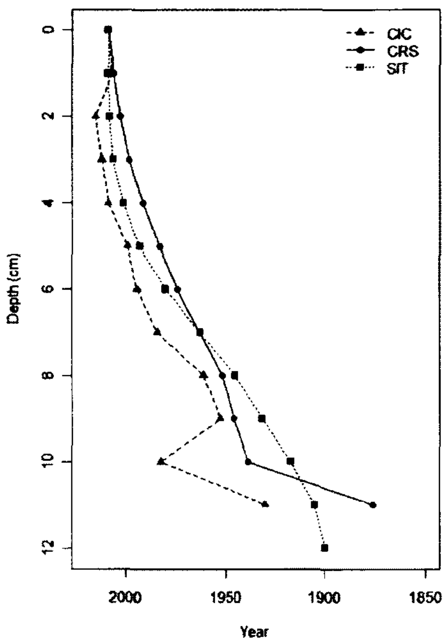
Figure C.1: Microscope image of the tephra found in both the Boswell Lake and Viewland Lake cores. Tephra was identified as having originated from the Bridge River event (ca. 2,410 calendar years BP) based on the glass shard morphology and tephra colour.

Appendix D

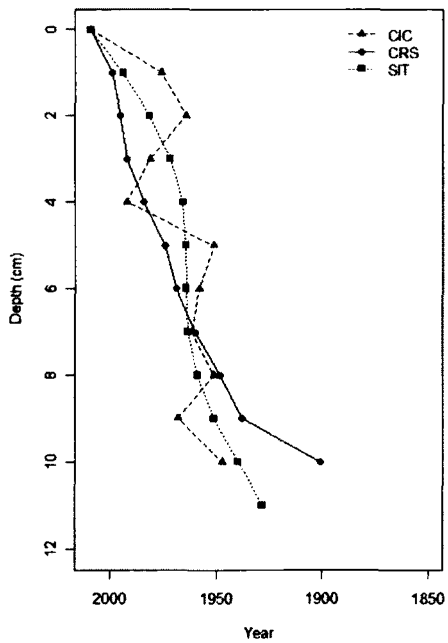
Lead-210 dating models



(a) Boswell Lake core (BL-P1)

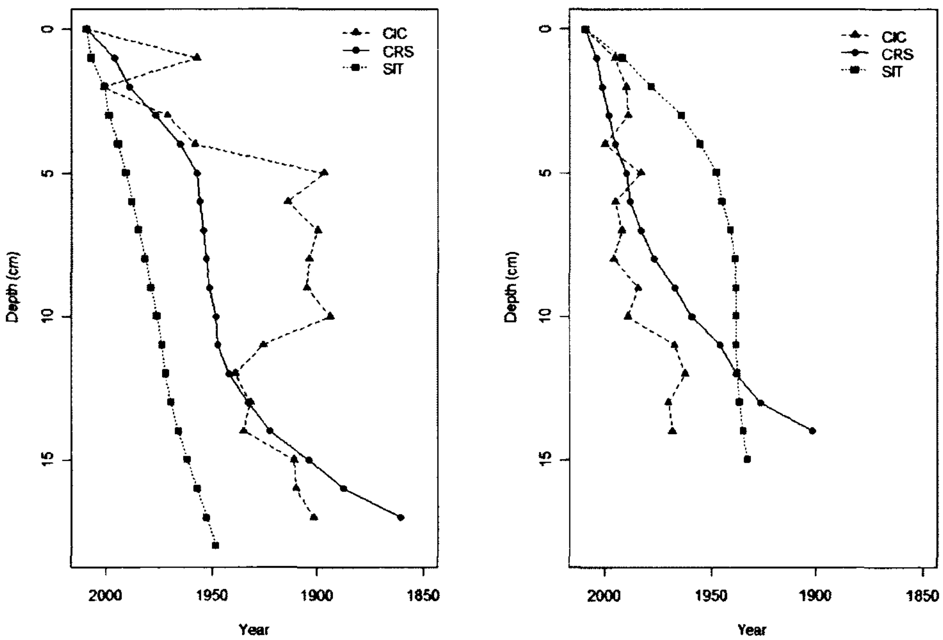


(b) Wetland core (BL-D8)



(c) Wetland core (BL-D10)

Figure D.1: Comparison of the ^{210}Pb-based depth-to-age models (CIC, CRS, SIT) for (a) Boswell Lake and wetland cores (b) BL-D8 and (c) BL-D10. Error bars are not given to enhance the readability of the figure.



(a) Viewland Lake core (VL-P1)

(b) Wetland core (VL-D1)

Figure D.2: Comparison of the ^{210}Pb-based depth-to-age models (CIC, CRS, SIT) for (a) Viewland Lake and (b) wetland cores. Error bars are not given to enhance the readability of the figure.