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Evaluation of instream flow management for riparian restoration along the St. Mary River, Alberta

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EVALUATION OF INSTREAM FLOW MANAGEMENT FOR RIPARIAN RESTORATION ALONG THE ST. MARY RIVER, ALBERTA

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ABSTRACT

River damming and offstream water diversion have led to the severe collapse of riparian cottonwoods along the St. Mary River. To promote cottonwood recovery, a systemic restoration approach was applied by improving critical components of the flow regime, and initial assessments showed successful seedling recruitment. This research provides a longer-term assessment by comparing sapling distribution and growth among six regulated and free-flowing river reaches. Sapling patches along the lower St. Mary were very sparse (14% of reach) compared to those along the similarly-regulated Waterton River (82%). This disparity highlighted different conceptual models of degradation and restoration trajectories. Reach mean sapling diameters were strongly positively related to average spring river levels ($R^2 = 0.987$), indicating that growth along the lower St. Mary may be hindered by low flows. However, more variation in sapling size occurred at smaller spatial scales indicating that site-level factors like density are also important determinants of growth.

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Study reaches:

USM	upper St. Mary River
LSM	lower St. Mary River
LSM/P	lower St. Mary River downstream of Pothole
UW	upper Waterton River
LW	lower Waterton River
CSL	Castle River

$7Q_{10}$	the average minimum flow for seven consecutive days with a 10 year return interval
H/D	sapling height-to-diameter ratio; a unitless number
H_{aug}	mean August stage above base stage
H_{sp}	spring peak stage above base stage
IF	instream flow
IFN	instream flow needs
M	mortality coefficient; describes ramping rate favourability with lower values representing a more favourable streamflow pattern
$nQ_d<2.75$	number of days from May-October when the mean daily discharge was less than 2.75 m^3/s
Q	discharge / flow / streamflow; measured in cubic meters per second (m^3/s)
Q_{max}	annual maximum flow
T	recurrence interval; pertaining to flood frequency

CHAPTER 1
Introduction to Riparian Cottonwoods and the St. Mary River

Like many rivers worldwide, the St. Mary River is heavily impacted, with stressors including, but not limited to, climate change and melting glaciers, an interbasin transfer of water that is the subject of international dispute, invasion by non-native species, gravel mining, cattle grazing, and most importantly – irrigation. The St. Mary River has a long history of use for irrigation dating back to 1898. Although St. Mary water provides regional economic benefits and supports Canada's largest irrigation project, ecosystem impacts have been severe. Since completion of the St. Mary Dam in 1951, the downstream cottonwood population has collapsed (Rood & Heinze-Milne 1989; Rood et al. 1995).

Cottonwoods, or poplars (*Populus* spp.), are intimately linked to a river's natural flow regime (Braatne et al. 1996). They are phreatophytes with their roots obtaining water from the capillary fringe situated above the saturated water table, which in semi-arid climates is recharged by the near-horizontal infiltration of river water (Rood et al. 1995). In arid and semi-arid regions, cottonwoods are thus restricted to riparian areas along the river valley bottoms. Their decline, however, imposes an enormous ecological impact.

Southern Alberta is unique in that it supports four *Populus* species that overlap and hybridize, resulting in rich and diverse cottonwood woodlands (Martinsen & Whitham 1994; Whitham et al. 1999; Floate 2004). Cottonwoods are the only native trees in this prairie landscape and provide habitat for many birds and other animals (Knopf et al. 1988; Rood et al. 2003a Trainor et al. 2007). Cottonwood decline not only means a loss

in terrestrial biodiversity, but could also result in a loss of instream habitat and aquatic ecosystem functions, such as the interception of pollutants (Sweeney et al. 2004).

River damming is one of the greatest human impacts on fresh water environments around the world (Nillson et al. 2005), and the St. Mary Dam is only one of numerous dams in southern Alberta and worldwide that have affected riverine landscapes. Loss of riparian woodlands due to damming and diversion has also been reported across North America along the Missouri River in North Dakota (Johnson et al. 1976), the creeks draining into Mono Lake in eastern California (Stine et al. 1984), the Arkansas River in Colorado (Snyder & Miller 1991), the lower Colorado River in Arizona (Busch & Smith 1995), the Yakima and Kootenay Rivers within the Columbia River Basin (Braatne & Jamieson 2001), the Big Lost River in Idaho (Rood et al. 2003b), and the Green River in Colorado (Breck et al. 2003). Other studies found declines in riparian tree growth (Johnson et al. 1976; Reily and Johnson 1982; Stromberg & Patten 1991, 1992), or a lack of younger age classes (Howe & Knopf 1991, Johnson 1992, Rood et al. 1996).

Among these degraded rivers, the St. Mary is a prominent case study (Annear et al. 2004; Rood et al. 2005; Naiman et al. 2005) that has been documented in a trilogy of papers by Rood and others (Rood & Heinze-Milne 1989, Rood et al. 1995, Rood & Mahoney 2000). The final paper in this series progressed further than documenting riparian ecosystem decline, and provided an assessment of seedling recruitment following a novel flow restoration strategy. That strategy involved operating the St. Mary Dam to produce favourable streamflow patterns for seedling establishment. A decade has now passed since the recruitment event, which allows us to ask the question, "was restoration successful?" Before presenting more detailed objectives of this research, the historical

context relative to irrigation, hydrology, and past research on the St. Mary will be introduced.

1.1 Irrigation and the St. Mary River

The St. Mary is a transboundary river originating in Glacier National Park, Montana, and flowing northeastward into Alberta. It is thus subject to sharing by the United States and Canada. An international water sharing agreement and the creation of a governing body called the International Joint Commission (IJC) was laid out in Article VI of the 1909 *Boundary Waters Treaty* with apportionment to each country set according to the IJC's 1921 Order (Task Force 2006). Briefly, the agreement defines the international sharing of both the St. Mary River and the Milk River, an adjacent transboundary river. Because the St. Mary is a larger river and flows away from Montana, water is transferred from the St. Mary into the Milk so that it can return to Montana.

The diversion infrastructure to support this interbasin transfer was completed by 1917 and consists of the U.S. St. Mary Canal and Sherburne Dam and Reservoir on Swiftcurrent Creek (Figure 1-1). Near the outflow of Lower St. Mary Lake, a weir diverts water from the St. Mary River into the U.S. St. Mary Canal, which consists of a series of canals and inverted siphons, for delivery into the North Fork of the Milk River. Sherburne Dam is regulated to release water in coordination with water intake into the canal. Therefore, although the upper St. Mary is considered a regulated river and has experienced flow decreases after 1917 (Figure 1-2), releases from Sherburne reservoir

compensate for some of the water removal so that the river has retained a relatively natural flow pattern.

The St. Mary-Milk River Project is not the only water management project on the river. In 1898, the first irrigation structure, a weir, was constructed along the upper St. Mary River at Kimball, south of Cardston, Alberta (Figure 1-1). Water was diverted offstream and conveyed through a series of man-made and natural channels, first reaching Lethbridge by 1900. Irrigation was used to promote settlement of the area and demand for water progressively grew. Demand exceeded supply in some years and a dam with a storage reservoir was deemed necessary (Gilpin 2000). The St. Mary Dam and Reservoir were completed by 1951 and allowed for further irrigation expansion such that the St. Mary Irrigation District now supplies water to over 370,000 acres (150,000 ha) extending from the Lethbridge area to Medicine Hat (SMRID 2007).

The St. Mary Project is also supported by water diverted from the adjacent Belly and Waterton Rivers. The Waterton Dam was constructed by 1964 and is used to store water for diversion into the Belly River and from the Belly River to the St. Mary Reservoir. Demand for water is so great that 118% of the median annual flow of the St. Mary is currently allocated (Alberta Environment 2003), meaning that in one out of two years there is not enough water for all users to take their full allocation.

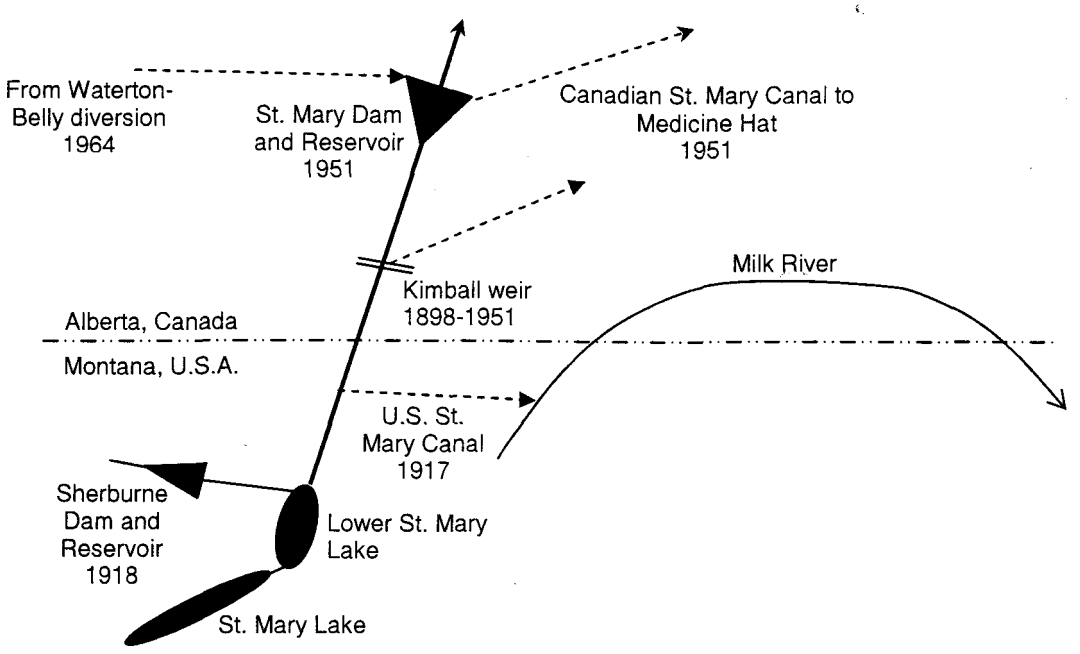

Figure 1-1. Schematic representation of irrigation infrastructure affecting the St. Mary River.

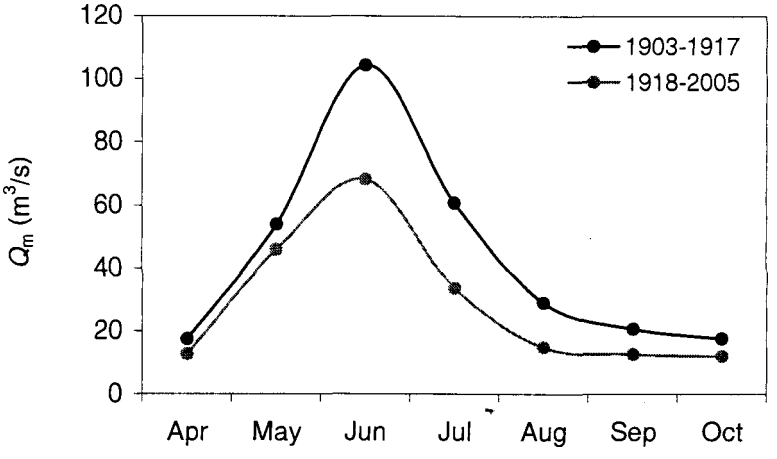

Figure 1-2. Mean monthly discharge during the irrigation season on the upper St. Mary River (05AE027) from before and after the completion of the St. Mary-Milk diversion. The interval from 1903-1917 was in a cooling phase of the Pacific Decadal Oscillation (Mantua and Hare 2002), which generally brings greater streamflows and accounts for some of the difference between the average pre- and post-diversion river flows.

1.2 Past riparian research on the St. Mary River

In 1989, Rood and Heinze-Milne first reported cottonwood woodland decline on the lower St. Mary River. By comparing the lineal distance covered by cottonwoods in aerial photographs from 1961 and 1981 (ten and thirty years post-dam), a 48% reduction was discovered in the 40-km stretch downstream from the dam, compared to only a 5% reduction upstream. In 1995, a second paper (Rood et. al. 1995) extended that data set to include analysis of air photos from 1951 and 1985, revealing an even greater 68% decline. Moreover, field surveys from 1985-1994 indicated further decline and a deficiency of seedlings and saplings.

To determine causes, Rood and Heinze-Milne (1989) first noted that the collapse was not symptomatic of a regional pattern since there were no declines along the undammed upper St. Mary and Waterton Rivers, and an increase in cottonwood abundance along the neighbouring Belly River. Second, by studying the water table elevation at two sites, Rood et al. (1995) confirmed a close link between river stage and groundwater level along the lower St. Mary River. The riparian water table increased and decreased in co-ordination with river stage, with a gradual down slope in water table level extending away from the channel. This means that phreatophyte reliance on the groundwater table is ultimately a reliance on streamflow.

Both studies involved hydrological analyses using historic streamflow records. They found that (1) summer flows were very low in many years, and that (2) while flood magnitudes had not substantially decreased downstream of the dam, post-flood recession rates were abrupt compared to the gradual decline upstream of the dam. Flood peaks, which did not change, are important for scouring, mobilizing and rejuvenating point bars,

creating ideal sites for seed germination (Braatne et al. 1996; Trush et al. 2000). Flood recession rates, which did change, are important for cottonwood seedlings whose quickly growing roots need to maintain contact with the declining water table (Mahoney & Rood 1998). Observations on sediment character downstream of the dam ruled out loss of fine sediments as the primary cause of decline. Therefore, decline of mature trees was attributed to drought stress caused by low summer flows, and lack of seedlings and saplings was attributed to abrupt flood recession rates that did not allow for recruitment.

Concern over cottonwood decline, and controversy surrounding the new Oldman River Dam that also brought issues on its southern tributaries into focus, prompted changes in water management on the St. Mary River. In 2000, Rood and Mahoney reported on two changes in the operation of the St. Mary Dam that were implemented to improve instream flows, and thus promote restoration of the cottonwood population. The first change was implemented in 1991 when the Alberta Government established a minimum instream flow of 2.75 m^3/s (*Water Act*, Alb.Reg. 307/91). There was no prior formal minimum criterion, but generally 0.93 m^3/s (30 cfs) was considered as an operating objective (Mahoney & Rood 2000).

The second change in dam operations occurred following the flood of 1995, the third largest flood on record, with a return interval of approximately 50 years. Large floods often produce the biggest and most successful cohorts of cottonwoods because they scour and mobilize a greater area of the floodplain providing extensive barren sites for seed germination (Whited et al. 2007). In 1995, not only was the magnitude of the flood favourable, but dam operations produced post-flood flows that were gradually reduced according to a more natural pattern (ramped) at a rate of approximately 2.5 to 5

cm/day. This rate was determined in a prior greenhouse experiment to be sufficient for elongating seedling roots to maintain contact with the water table (Mahoney & Rood 1991). Ramping flows were successfully implemented and promoted a major recruitment event in that year (Rood & Mahoney 2000). The physical disturbance provided by the flood also helped promote establishment in subsequent years. Because the flood was a regional occurrence and ramping flows were also implemented on the Waterton and Oldman dams, the seedling recruitment event was regionally widespread (Rood et al. 1998; Kalischuk et al. 2001).

In the analysis of Rood and Mahoney (2000), the lower St. Mary was divided into three segments based on channel characteristics such as floodplain width, sinuosity, slope, and meander amplitude and wavelength. Eighteen transects were established along the three segments to assess seedling density and heights. On the upper segment, known as Box Canyon, only one site had seedlings, at a density of about 30 per m^2. On the middle reach, seedling densities were generally sparse, but did reach 340 per m^2 at one site. Lack of seed source on these upper two reaches was likely a factor in the limited extent of recruitment. The third reach had a wider floodplain with more abundant mature woodlands and was located downstream from the confluence with Pothole Creek – a tributary that can provide on average approximately 13% of the St. Mary's August discharge (Appendix E). This segment provided areas that were suitable for a large recruitment event at four of the six sites, with densities as high as 144 seedlings per m^2 in 1996. A 95% reduction in the initial seedling density was observed in 1999 as the seedlings increased in height. The St. Mary River case study demonstrated the importance of understanding historic hydrology.

1.3 Historic Hydrology of the St. Mary River

The average annual hydrographs for both the upper and lower St. Mary River show a typical flow regime for rivers of this region with low flows throughout the winter, steeply increasing in May leading to a period of higher flows in the late spring and early summer associated with snowmelt and rain events (Figure 1-3). Flows then gradually decline to baseflow, by late summer. The flood peak naturally occurs between late May and mid June. However, river damming has increased the seasonal variation in the day of the annual maximum along the lower St. Mary such that peak flows occasionally occur in winter or in October when the reservoir level is reduced before freeze-up (Figure 1-4).

The magnitudes of flood peaks are also highly variable from year to year (Figure 1-5), ranging from 27 m^3/s in 1941 to 793 m^3/s in 1908 along the upper St. Mary. Along the lower St. Mary, flood peaks ranged from 2.8 m^3/s in 1961 to 702 m^3/s in 1974 (period of record commenced in 1912). As indicated in section 1.2, the St. Mary Dam has not attenuated the magnitude of large flood peaks because the reservoir has a relatively small capacity (Figure 1-5). Five of the six largest floods on record have occurred post-damming. Flood recurrence analysis was presented in Rood et al. (1995), and I have updated that analysis to include the most recent data, including the flood of 2005. Along the upper St. Mary, a 100-year flood has a magnitude of 499 m^3/s, and along the lower St. Mary, 560 m^3/s. Although flood magnitudes have not decreased, and are often larger on the lower St. Mary than the upper St. Mary as seen in 2005 (Figure 1-5), the overall volume of flow during the spring peak has been greatly reduced due to reservoir filling. Mean May and June flows decreased from 54 m^3/s in the pre-dam period to 33 m^3/s in the post-dam period. Also, a comparison of mean recorded versus mean naturalized flow

during the post-dam period showed greatly reduced flows during the spring freshet (Figure 1-6).

Reduced May and June flows due to reservoir filling contrast with increased flows due to reservoir lowering in the late summer. Although low summer flows have been cited as the cause of mortality of mature trees along the St. Mary River, and there were many years where flows were very low (Rood et al. 1995), mean August discharge has increased from 6.82 m^3/s in the pre-dam period to 8.79 m^3/s in the post-dam period (and see Figure 1-7), although this increase was not significant (t-test: $t = 0.797$, $df = 76$, $p = 0.428$). Prior to damming, late summer water diversions at Kimball would come from late summer flow, but now, summer irrigation supply comes from reservoir storage enabling somewhat supplemented flows. It is thus likely that the ecosystem experienced some stress even before construction of the St. Mary Dam in 1951.

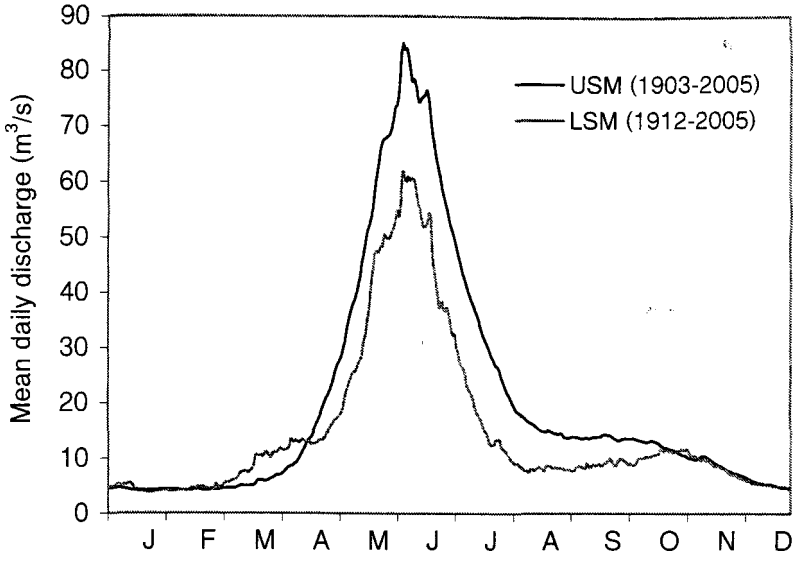

Figure 1-3. Average hydrographs for the period of record for the upper St. Mary (USM; 05AE027) and lower St. Mary (LSM; 05AE006). Hydrographs show the typical snowmelt-dominated pattern for rivers of this region.

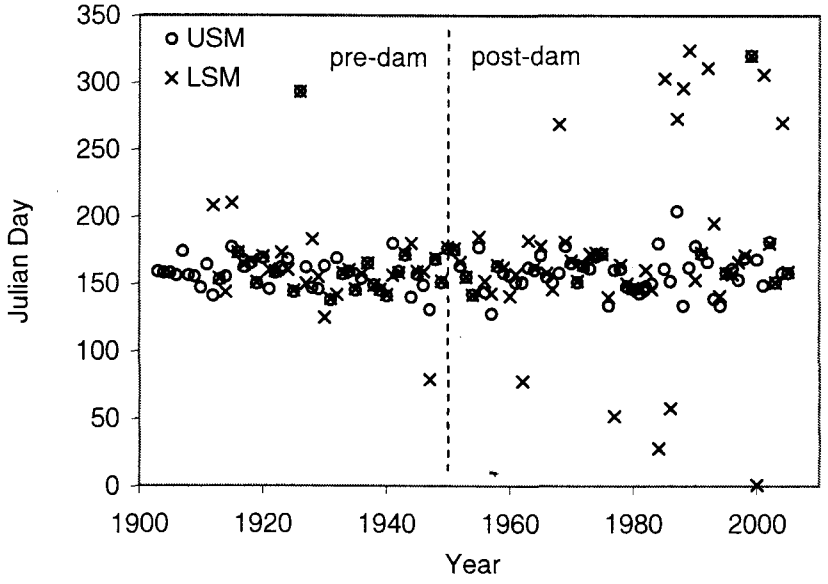

Figure 1-4. Julian day of the annual maxima (Q_{max}) for the upper and lower St. Mary (USM and LSM respectively). Most Q_{max} occur between mid-May and the end of June (JD 135-180), but river damming along the lower St. Mary has altered the timing so that flood peaks occasionally occur in winter or in October when the reservoir level is brought down before freeze-up.

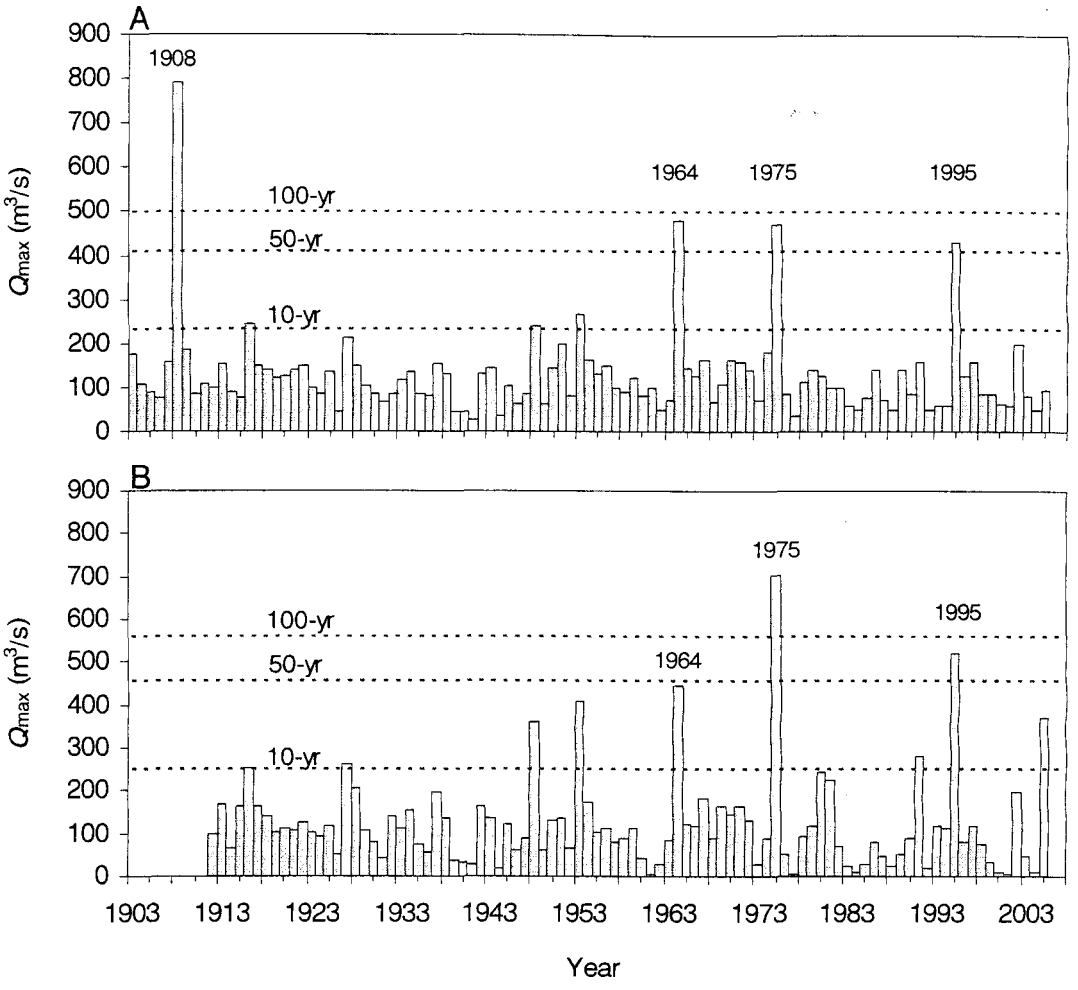

Figure 1-5. Historic annual maximum flood peaks for the period of record along the upper St. Mary (A) and the lower St. Mary (B). Dashed lines represent return intervals calculated with a Log Pearson Type III (A), and a 2 Parameter Log Normal (B) distributions in DISTRIB 2.2.

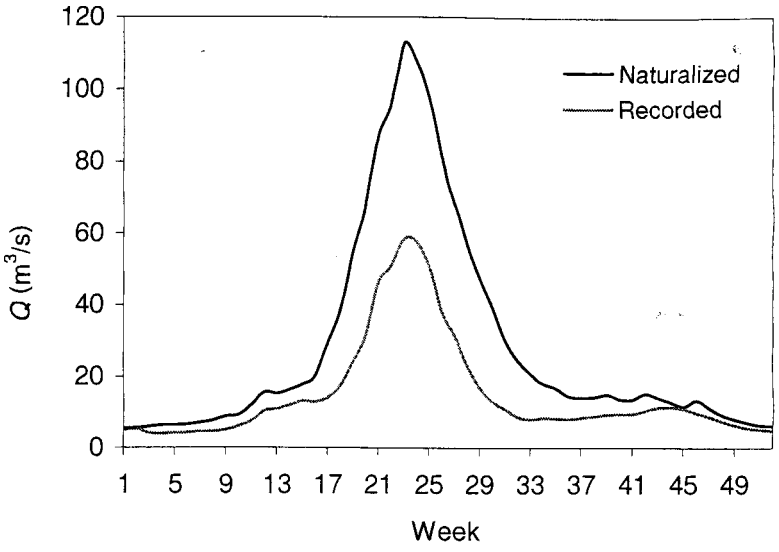

Figure 1-6. Comparison of average weekly discharge hydrographs from recorded flow data (05AE006) versus naturalized flow data (05AE006B) along the lower St. Mary for the post-dam period (1952-2005). Naturalized flows represent what the flow would be without regulation and were obtained from Alberta Environment (2004). The advantage of using naturalized flows is that a comparison can be done in the same time period, avoiding the effects of natural climatic variation that occur throughout the period of record.

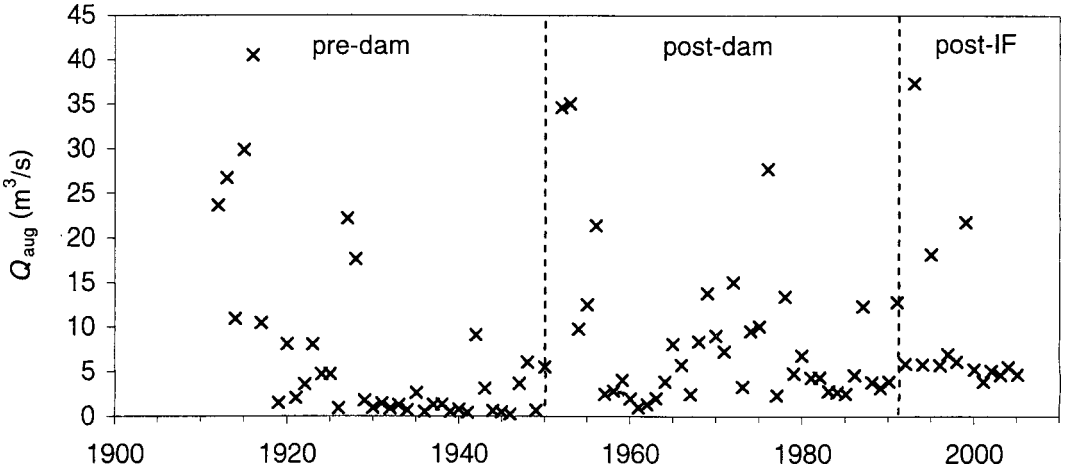

Figure 1-7. Mean August discharge for the period of record along the lower St. Mary (05AE006). There were many years with very low flow during the pre-dam period due to diversion at Kimball. In the post-dam period, increased storage capacity has allowed for increased late summer flows and in the post-IF (instream flow) period flows are consistently greater.

1.4 Hydrology as the key variable: Instream flows for cottonwoods

Hydrology is a key variable driving riverine ecosystems and affects geomorphic and biological components, water quality, and the overall connectivity of the system (Annear et al. 2004). Cottonwoods depend on hydrology for both survival of established trees, and for recruitment of new seedlings, as seen along the St. Mary River. There is much further evidence for cottonwood-streamflow relationships in each of the broad categories of survival and growth, and recruitment.

Although the exact requirements for survival and growth are not fully understood, many studies have shown a link between these characteristics and streamflow. Declines in cottonwood survival, vigor, and growth have been reported along numerous rivers whose hydrology has been altered by damming and diversion (Stromberg & Patten 1990; Rood et al. 2003a; Rood et al. 2003b). Additional studies have shown decreased vigor and growth of cottonwoods following declines in the water table due to channel excavation for gravel mining (Amlin & Rood 2003; Scott et al. 1999). Declines in cottonwood health, such as precocious leaf senescence, branch sacrifice, and crown die-back, have also been observed following naturally or artificially low flows (Rood et al. 2000). Willms et al. (1998) found that annual branch growth increments were positively correlated with streamflow in the early spring when stem elongation occurs and high flows recharge the water table. Andersen (2005) has also shown a correspondence between sapling height growth and peak streamflow or the maximum streamflow for 30 days during the growing season. More specifically, Stromberg and Patten (1991) have correlated annual growth rings with streamflow and found that 40-60% of the natural flow was required to maintain a healthy canopy in mature black cottonwoods (*Populus*

trichocarpa). Physiological responses to dewatering that could impact growth were reviewed in Rood et al. (2003b) and include stomatal closure, reduced transpiration and photosynthesis, reduced water potential, and xylem cavitation.

In common with the St. Mary River, other regulated rivers show missing younger age classes from floodplain woodlands due to the prevention of recruitment (Howe & Knopf 1991; Merigliano 1998; Battacharjee et al. 2006). For seedling recruitment, flow patterns are important, and those patterns are described by the Recruitment Box Model (Mahoney & Rood 1998). This model integrates the seedlings' ecophysiological constraints with the timing and magnitude of the spring peak and the post-flood rate of recession. Two recent papers have built upon the Recruitment Box Model further developing and quantifying the instream flow needs of cottonwood recruitment (Dixon & Turner 2006; Braatne et al. 2007). In Braatne et al. (2007), we built a step-wise model by developing criteria for each of the important hydrograph components. We considered not only the current year's hydrograph, but the effects of the years before and after the establishment year that would influence floodplain condition and seedling survival.

In these recruitment models, two of the aspects of the flow regime that are important for seedling recruitment are flood flows and gradual post-flood recession rates. Floods are needed to scour away existing vegetation and mobilize sediment creating moist and barren substrate zones. These suitable nursery sites are required by cottonwood seedlings, which have limited nutrient reserves and are shade-intolerant (Braatne et al. 1996). The post-flood recession that exposes the new seed beds must coincide with the timing of seed dispersal, and occur at a rate that allows the roots of the newly germinated seedlings to maintain contact with the declining moisture zone. By

placing seedlings in rhizopods, a type of growth chamber that allows for the controlled manipulation of water level, a decline of about 2.5 cm/day was found to be sufficient (Mahoney & Rood 1991).

The recognition of the importance of such aspects of hydrology is reflected in the concept of instream flow needs (IFN), which are generally defined as the flow regime that would adequately meet a desired level of ecosystem function (Gordon et al. 2005). With the recognition of declining ecosystems, instream flow management has been used to promote conservation and restoration.

The concept of instream flow needs, or environmental flows, emerged in the late 1960s to 1970s and has led to the development of about three dozen methods (Annear et al. 2004), including the popular Tennant method (Tennant 1976) and Instream Flow Incremental Methodology (Stalnaker 1995). Determination of minimum flows and single species objectives (usually sport fish) are common themes among many of these methods, but more recently, holistic approaches have been suggested for conservation and restoration. Junk et al. (1989) developed a 'Flood-pulse concept' that recognized the multiple ecosystem benefits of over-bank flooding. Poff et al. (1997) suggested that a 'Natural Flow Regime' would be beneficial to the ecosystem as a whole and not just a few fish species, but they recognized that determining what 'natural' means is challenging and may not be possible in heavily altered systems. Similarly, Hughes and Rood (2001) suggested a "systemic restoration" approach aimed at restoring natural flow dynamics, thus benefiting the entire ecosystem, rather than site-specific artificial measures such as vegetation plantings or fish stocking. Trush et al. (2000) acknowledged

that all components of the annual hydrograph are needed to maintain geomorphic and ecological functions of healthy alluvial rivers.

Clipperton et al. (2003) recognized the principles of the 'Natural Flow Regime' and its intra- and inter-annual variability and compiled a report outlining the instream flow needs of four ecosystem components in the Oldman River Basin: water quality, fish habitat, riparian vegetation, and channel maintenance. They integrated the results into an overall instream recommendation for ecosystem protection, but the flow prescription has yet to be implemented.

Hauer and Lorang (2004) supplied dam operators with plans for more favourable instream flows that they developed based on an intensive study of floodplain dynamics on the Snake River. They hypothesized that their flow prescriptions would, among other aspects, promote cottonwood woodland regeneration by restoring floodplain dynamics and connectivity. Studies where instream flows have already been implemented for restoration include the successful restoration of the Truckee River in Nevada (Rood et al. 2003a), and successful seedling establishment following simulated or prescribed flooding along Boulder Creek in Colorado (Friedman et al. 1995), the Middle Rio Grande in New Mexico (Taylor et al. 1999; Sprenger et al. 2002) and throughout the Oldman River Basin in Alberta (Rood et al. 1998), including the St. Mary River.

Although assessment or long-term monitoring is recognized as an important part of any management project (Kondolf & Micheli 1995; Gordon et al. 2005; Woolsey et al. 2007), few of these studies have observed the longer-term response of these restoration attempts, with the exception of that of Taylor et al. (2006), which evaluated the Middle

Rio Grande seedlings ten years after establishment. Like Taylor et al. (2006), I will be assessing the longer-term response of flow restoration on the St. Mary River.

1.5 Objectives

The primary objective of this research on the St. Mary River is to answer the question, "Was restoration successful?" More specifically, have changes in dam operation implemented over a decade ago along the lower St. Mary River, including increased minimum flows in 1991, and ramping flows in 1995, promoted recovery of the cottonwood population?

Initial success occurred as ramping flows promoted seedling recruitment, especially along the reach downstream of Pothole Creek (Rood & Mahoney 2000). That initial response to flow management was important, but for restoration to be considered successful, continued survival and growth are necessary. If the seedlings that established as a result of the flood of 1995 survived, they would now be juveniles or saplings.

The main chapter of this thesis, Chapter 2, will present a study on the extent and size of saplings along the lower St. Mary River with comparisons to other regional river reaches. The study system involved both regulated and free-flowing reaches with both similar and different hydrologic regimes for contrasts and comparisons. Because hydrology was the factor that was used to promote restoration, I will determine if differences in sapling size are attributable to differences in streamflow.

The third chapter, like the initial restoration assessment in Rood and Mahoney (2000), will focus on recruitment. However, this chapter will present a hydrological

analysis using streamflow data from 1995-2005 to determine if recent flows are favourable for recruitment.

Together, the growth and the recruitment analyses will provide an overall assessment of restoration success and relate cottonwood life history components to streamflow. This research will provide continued and longer-term evaluation of management practices, which is an often forgotten but critical part of any restoration program. Further, this research will increase our general understanding of riparian ecosystem water needs in a region where there is an ever present human demand for increased water use.

CHAPTER 2
Assessment of flow restoration for riparian cottonwoods along the St. Mary River with comparison to other regional rivers

2.1 INTRODUCTION

River damming is one of the greatest human impacts on riverine ecosystems in the world (Nilsson et al. 2005). Dams alter natural streamflow patterns, resulting in the decline of streamflow-dependent riparian ecosystems (e.g. Johnson et al. 1976; Snyder & Miller 1991; Busch & Smith 1995). In addition to damming, riparian zones tend to be subjected to many other impacts such as cattle grazing and land clearing (Howe & Knopf 1991), resulting in widespread decline (Noss et al. 1995).

In response to ecosystem declines, river restoration has become a prominent field with more than 37,000 restoration projects implemented in the United States (Bernhardt et al. 2005). However, most restoration projects are site specific, often involving vegetation plantings or invasive species removal. Landscape-scale or systemic approaches that restore ecosystem processes necessary for the function of biological communities are closer to the goals of restoration ecology (Hobbs & Norton 1996; Hughes & Rood 2001). In a river degraded by damming or water diversion, landscape-scale methods include restoring a more natural flow regime with intra- and inter-annual variation (Poff et al. 1997; Richter et al. 1997; Stromberg 2001).

By restoring streamflow patterns, it follows that organisms such as riparian cottonwoods, whose life history patterns have evolved in coordination with streamflow patterns, would also be restored (Braatne et al. 1996). For example, on the dammed and dewatered Truckee River in Nevada, flows were restored for the endangered Cui-ui sucker, but these more natural flows also promoted cottonwood and willow recruitment,

resulting in restoration of the riparian woodland, which in turn, prompted the return of many bird species (Rood et al. 2003a).

Like the Truckee River, the St. Mary River has also been dammed and extensively dewatered, and provides a prominent case study for the effects of water management on riparian ecosystem decline. In the 40-km stretch downstream from the St. Mary Dam, most of the cottonwood groves that were once naturally-sparse have died, and throughout the lower St. Mary, younger age classes became absent (Rood & Heinze-Milne 1989; Rood et al. 1995). The cause of the decline was attributed to low summer flows causing mortality of mature trees, and abrupt post-flood flow declines that prevented seedling replenishment (Rood et al. 1995).

Flow restoration measures along the lower St. Mary River commenced 16 years ago with a tripling of minimum flows in 1991, and a novel management strategy for recruitment involving gradual flow recession (ramping) following the flood peak in 1995. The 1-in-50 year flood in 1995 created barren colonization sites, and was followed by ramping flows that satisfied seedling requirements as described by the Recruitment Box Model (Mahoney & Rood 1998). A large recruitment event occurred resulting in initial restoration success (Rood & Mahoney 2000). In 1996, seedling densities ranged from 30 to 340 seedlings per m^2 at various sites along the river.

Assessment or long-term monitoring is recognized as an important part of any management project (Kondolf & Micheli 1995; Gordon et al. 2005; Woolsey et al. 2007), but few studies have documented the longer-term response of riparian restoration attempts. Taylor et al. (2006) provide an exception in their evaluation of cottonwoods along the Middle Rio Grande River ten years after seedlings established as a result of

prescribed flooding and invasive species removal. In common with the approach of Taylor et al. (2006), I have assessed the longer-term response of cottonwoods to flow restoration of the lower St. Mary River.

The seedlings that established about a decade ago as a result of the 1995 flood would now be saplings, or juvenile trees. The primary objective of this chapter was to assess the distribution and growth of the sapling population to assess the success of instream flow management.

A complexity in restoration is determining an appropriate target, or a reference system from which to judge success (White & Walker 1997). For this research, other regional river reaches were studied to provide comparisons and contrasts to the lower St. Mary. Study sites were established along the slightly regulated upper St. Mary above the St. Mary Dam, and along the free-flowing upper Waterton and Castle rivers. Sites were also established along the lower Waterton River below the Waterton Dam – a reach that shares many hydrologic and geomorphic characteristics with the lower St. Mary. The lower St. Mary was divided into two reaches with one immediately downstream of the St. Mary Dam and the other below an inflowing tributary. All of these river reaches would have a similar cohort of saplings because they also experienced a major recruitment event in, or after 1995 due to the widespread flooding, and flow ramping from the dams (Rood et al. 1998; Mahoney & Rood 2000).

My objective was to determine if saplings along the lower St. Mary differed in size from those along other reaches, and to determine if differences were attributable to streamflow, as streamflow was the factor that was changed to promote restoration. Not only did this research provide a longer-term assessment of restoration efforts, but it

provided information on cottonwood sapling survival and growth, which is a subject that deserved further study.

Few studies have looked at cottonwood saplings, an important life stage that represents post-establishment survival. The study of cottonwood saplings along the Green and Yampa rivers in Colorado (Andersen 2005) provides an exception. Andersen (2005) stressed the need to study saplings because flow models for optimizing seedling recruitment have been developed, and even verified (Shafroth 1998; Rood et al. 2003a; Dixon & Turner 2006), but flow prescriptions to optimize growth and survivorship of saplings are lacking.

The first step in this research was a hydrological analysis to determine whether the minimum flow criterion, one of the flow restoration measures, was being implemented. The next step was to establish if the initial seedlings had survived and grown into saplings. Then, size versus streamflow relationships were investigated. This is potentially quite complex. First, due to the spatial and temporal heterogeneity that is characteristic of riparian areas (Naiman et al. 2005); and second, due to the nature of observational field studies and the differences in environmental factors.

Environmental factors include edaphic factors such as substrate texture and nutrients; climatic factors such as temperature, precipitation, humidity, and length of the growing season; topographic factors such as altitude, slope, and aspect; or biotic factors such as the density of other trees and vegetation (Husch et al. 2003). Andersen (2005) also recognized that a sapling's biotic environment (herbivores), its physical environment (soil properties and topography), and its physical state (root distribution) influence its response to a hydrological event. Further, foresters have long recognized the potential

for different growth rates at different sites by developing a range of growth curves dependent on site quality (Husch et al. 2003; Pederson et al. 1996).

If one views the overall growth of a sapling as a summation of the effects of all the factors, then the contribution of any one factor could be quite small, with a few major factors having more effect. I hypothesized that streamflow, as a key variable, is one of those major factors. Therefore, I predicted that saplings growing along river reaches that differ in hydrology could be of different sizes with a positive association between water supply and growth.

Other factors that might predominantly affect sapling size, such as beaver herbivory, growth form, position on the floodplain (i.e. elevation above river stage at baseflow), density, and geomorphic context were also investigated for their effect on sapling size. Sapling excavations to determine precise age of establishment were carried out to confirm that the saplings present were from the 1995 cohort.

By analyzing the streamflow regime and the presence and growth of saplings along the St. Mary compared to other reaches, the success of flow management was assessed. Using this study system as an example, conceptual models of river degradation and restoration trajectories are then discussed.

2.2 METHODS

2.2.1 Study Area - River Courses and Descriptions

The three rivers involved in this study – the St. Mary, the Waterton, and the Castle – drain the east slopes of the Rocky Mountains and are all southern tributaries of the Oldman River (Figure 2-1). They are located in the Oldman River Sub Basin of the South Saskatchewan River Basin, which drains into Hudson Bay. All rivers start in the alpine and subalpine ecoregion, and flow through to the fescue grassland ecoregion. While the Castle joins the Oldman River, the St. Mary and Waterton Rivers continue flowing through to the moist mixed grass ecoregion before contributing to the Oldman River (Alberta Agriculture and Food 2005).

The St. Mary River begins in the Lewis Range of the Rocky Mountains in Glacier National Park, Montana and flows into St. Mary Lake and then Lower St. Mary Lake. The river flows northward into Alberta, and then from the international border the St. Mary flows for 48 km into the St. Mary Reservoir. The reach upstream of the reservoir will be referred to as the upper St. Mary (USM) and is partly regulated because the tributary, Swiftcurrent Creek, is dammed by the Sherburne Dam, and because of offstream diversion into the U.S. St. Mary Canal (Figure 1-1). But the upper St. Mary has retained a natural flow pattern, and thus the term 'partly regulated' is used when describing this reach. Downstream of the international border, and near the town of Kimball, the St. Mary River passes through Coal Canyon and St. Mary Canyon, which are segments with steep sandstone cliffs that create a very narrow floodplain and confine meandering (Figure A-1). Lee Creek, an almost free- flowing tributary, joins the St. Mary River near Cardston.

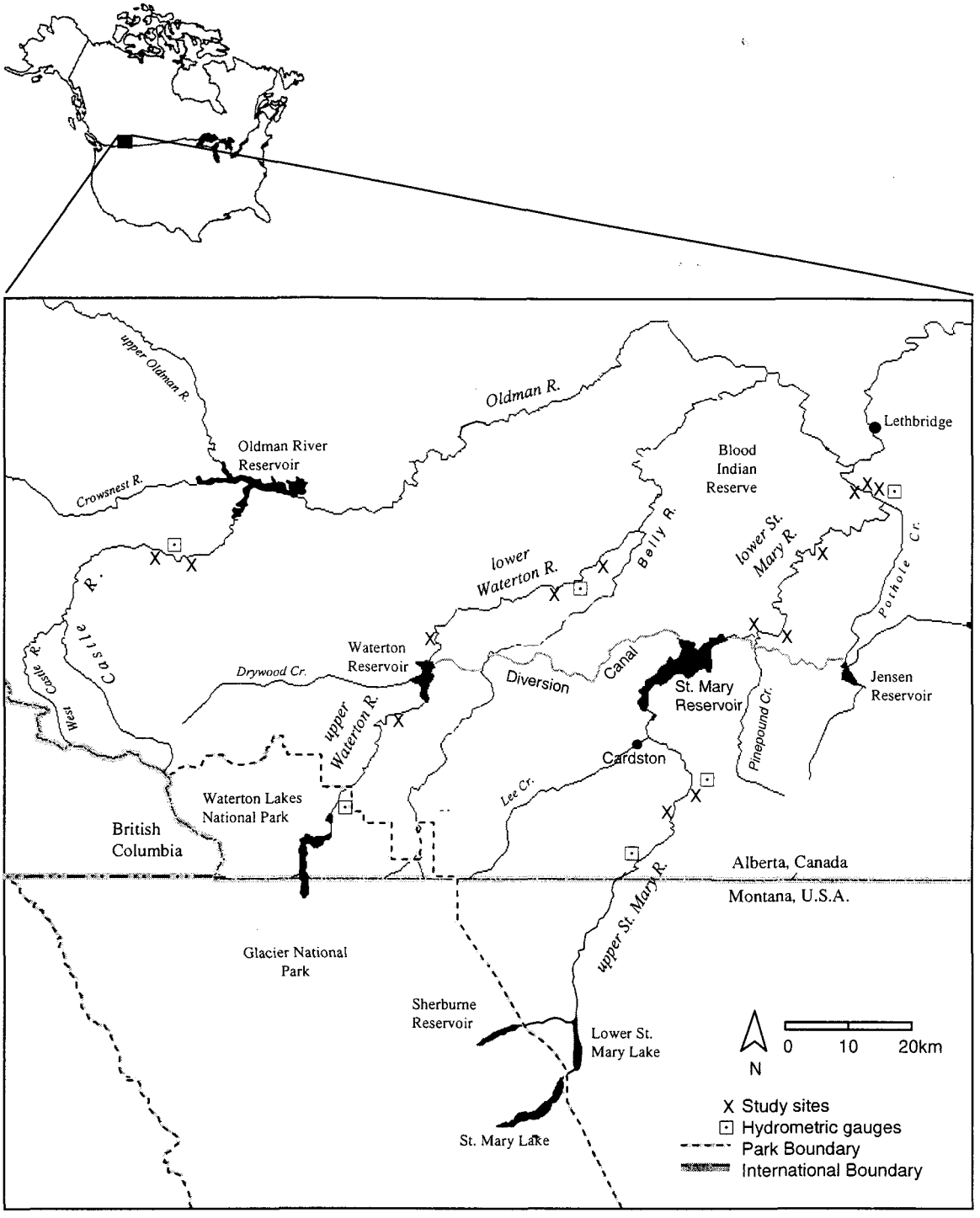

Figure 2-1. Map of the study area in southern Alberta showing study rivers, study site locations, and hydrometric gauging stations.

Streamflow downstream of the reservoir is regulated by the St. Mary Dam and this reach will be referred to as the as the lower St. Mary (LSM). Approximately 14 km downstream from the Dam, the LSM enters Box Canyon (Figure A-2). Like the canyons of the upstream reach, Box Canyon is composed of sandstone cliffs that partially confine the river. This reach is highly sinuous, but the meanders are deeply entrenched by the resistant and steep cliffs. Eighty-three kilometers downstream from the St. Mary Dam, the river is joined by Pothole Creek, a small creek that also delivers irrigation return flows. The St. Mary River from this point to the confluence with the Oldman River, 14 km downstream, widens and is less confined (Figure A-3). This final reach will be referred to as the lower St. Mary downstream of Pothole Creek (LSM/P).

The Waterton River parallels the St. Mary River in its northeastward journey from the Rocky Mountains to the Oldman River. Numerous alpine streams contribute to the large Waterton Lakes complex. These are large lakes, and consequently flood flows are attenuated and sediments settle, leaving the reach below the lakes sediment depleted. From the lakes, the Waterton River flows for 32 km (Buhrman & Young 1982) until it reaches the Waterton Reservoir. This reach will be defined as the upper Waterton (UW). Flow into the Waterton reservoir is augmented by several creeks including Drywood Creek, but flow is also diverted from the reservoir into the Belly River and eventually into the St. Mary Reservoir through the Blood Indian Reserve (Figure 2-1).

Downstream from the Waterton Dam, the Waterton River flows for another 39 km until its confluence with the Belly River. This reach from the Waterton Dam to the Belly River will be referred to as the lower Waterton (LW). The lower Waterton River,

like the St. Mary River also has sections where the river is confined by steep cliffs (termed 1st Canyon and 2nd Canyon by Buhrman and Young (1982); Figure A-5).

The Castle River starts close to the Continental Divide just north of Waterton Lakes National Park. It flows northward for many kilometers before it is joined by the West Castle River that also begins close to the continental divide. The river continues to flow northward and then eastward for about 50 kilometers into the south arm of the Oldman River Reservoir, passing through a bedrock-confined section called the Castle Canyon (Figure A-7; distances according to Buhrman and Young (1982)).

These three rivers that encompass six river reaches provided both similarities and comparisons for my sapling study. Several attributes of these reaches are summarized in Table 2-1. The higher elevation reaches (USM, UW, and CSL) are partly regulated or free-flowing, have larger mean annual flows and greater precipitation, are cooler, and provide fewer degree days than the lower reaches, which are regulated by dams.

Table 2-1. Hydrologic and climatic attributes of the six river reaches studied. Mean annual discharge (Q_a) from 1996-2005 is presented. Total May-August precipitation, mean July temperature, and annual total degree days (DD) are climate normals for 1971-2000 (Alberta Agriculture and Food 2005). Climate values represent upland conditions and could be different than the adjacent river valley, but are provided as indicators of differences among reaches.

Reach	Reach ID	Status	Q_a (m^3/s)	May-Aug ppt (mm)	Mean July temp (°C)	Annual total DD (> 5° C)
upper St. Mary	USM	slightly regulated	17.41	250-300	15-16	1350-1500
lower St. Mary	LSM	regulated	9.90	225-275	16-17	1500-1650
lower St. Mary d/s of Pothole	LSM/P	regulated	10.35	225-250	16-17	1500-1650
upper Waterton	UW	free-flowing	10.42	275-300	15-16	1350-1500
lower Waterton	LW	regulated	10.62	250-300	15-17	1350-1650
Castle	CSL	free-flowing	14.60	250-275	14-15	1200-1350

2.2.2 Hydrology

Each river reach has an associated hydrometric gauging station that measures river stage (Figure 2-1; Table 2-3). River stage is converted to discharge using a stage-discharge rating curve, and discharge values are published in Water Survey of Canada's hydrological database (HYDAT; http://scitech.pyr.ec.gc.ca/ waterweb/main.asp). Although Pothole Creek was not a study reach, its gauging station was listed in Table 2-2 because data from this gauge were used to calculate a more accurate discharge for the lower St. Mary (LSM) reach by subtracting the Pothole flow from the St. Mary near Lethbridge flow. Data from this gauge were also used to determine the importance of Pothole Creek flow contribution to the St. Mary River.

Table 2-2. Hydrometric gauges used for hydrological analyses. Gauge locations are marked on the map of the study region (Figure 2-1).

Station Name	Reach Association	Station No.	Period of Record
St. Mary River at International Boundary	USM	05AE027	1902 - present
St. Mary River at Hwy #501	USM	05AE043	1998 - present
St. Mary River near Lethbridge	LSM, LSM/P	05AE006	1911 - present
Pothole Creek at Russell's Ranch	LSM	05AE016	1919 - 1956; 1972 - present (seasonal)
Waterton River near Waterton Park	UW	05AD003	1908 - present
Waterton River near Glenwood	LW	05AD028	1966 - present
Waterton River near Stand Off	LW	05AD008	1915 -1966
Castle River near Beaver Mines	CSL	05AA022	1945 - present

Minimum criteria

To determine if the minimum flow criteria set in 1991 were being met along the St. Mary and Waterton rivers, I calculated the average number of days during the growing season (May-Oct) when the daily mean discharge was less than the criteria -

2.75 m^3/s for the St. Mary and 2.27 m^3/s for the Waterton. Three time periods (pre-dam, post-dam, and post-IFN) and upstream versus downstream comparisons were made. The pre-dam time period for the St. Mary River was from 1912-1950 and from 1916-1963 for the Waterton River. There were data gaps, with 1931 to 1934 missing along the LW and 1931 to 1947 missing along UW, when the gauges were not maintained. For the LW, I corrected the 'Waterton River near Standoff' gauge to the 'Waterton River near Glenwood' gauge with an equation derived by linear regression ($R^2 = 0.978$), using data from the period of overlap. I used the corrected data for the interval prior to 1966 when the Glenwood gauge became operational.

To provide another perspective on low flows, I compared $7Q_{10}$ values among the time periods and reaches. This metric represents the average minimum flow for seven consecutive days with a 10 year return interval. Return intervals were calculated in DISTRIB 2.20 (Eaglin 1999) using Log Pearson Type III distributions, which provided the best fit. The flow that had a 0.10 probability of occurring was the $7Q_{10}$.

Streamflow within a cottonwood sapling's life

For these analyses, I considered the ten years of streamflow data from 1996-2005 as these were the flow conditions that the post-1995 cottonwood cohort would have experienced prior to the field measurements. Mean monthly discharge hydrographs for the growing season were compared across the reaches. Daily discharge values were back-converted to stage values using rating curves provided by Alberta Environment, and the base stage was subtracted to produce mean monthly stage hydrographs. In addition, the percentage of natural flow experienced by the regulated reaches was calculated using

Alberta Environment's Natural Flow Database (Alberta Environment 2004). For the percentage of natural flow determination, the average percent difference between weekly recorded flow and weekly natural flow from 1996 to 2001 was calculated.

2.2.3 Sapling Assessment and Measurement

Sapling extent

To accomplish the first objective of this study and determine if the post-1995 cohort of seedlings had survived to the sapling stage, the riparian zones along the lower St. Mary River were extensively surveyed for the presence of saplings. Between 2005 and 2007, the entire 95 km section of the lower St. Mary downstream from the Dam was floated by a combination of kayak, raft, and canoe, with many sections visited more than once. Along the upper St. Mary a 12 km section through Coal Canyon was floated in 2005. Along the lower Waterton, a section representing approximately one-third of the reach length was floated in 2006, and along the Castle River, preliminary measurements were taken along a 7 km section that was floated in 2005.

In addition, a low-altitude photography flight was undertaken on August 2, 2007 with photographs taken almost continuously along the entire lower St. Mary and Waterton reaches. Photograph locations were identified using topographic maps, and 1:30,000 aerial photos from 1999 for the St. Mary, and 1:10,000 aerial photos from 2005 for the Waterton. From the bottom of the spillways to the confluences, points were marked at one kilometer intervals along both rivers. At each point, a photograph was assessed for the presence of saplings and mature trees, and for the possibility that it could provide habitat for cottonwood saplings. A point could potentially provide habitat if it

was not a valley wall, a cut bank, or severely impacted by land clearing. For the LSM and LSM/P, flight photographs were available for 57 out of the 83 kilometer points, and 13 out of the 14 points respectively. For the LW, photographs were available for 51 out of 60 points.

Study sites

To further assess and quantify sapling characteristics, fourteen study sites were established along the six river reaches. I selected sites based on accessibility and presence and abundance of saplings. Along the LSM, there were only three locations that supported abundant sapling patches, therefore, these locations were designated as study sites. Sapling measurements at these sites are not representative of reach-wide characteristics, but rather represent the potential for restoration.

Along the upper St. Mary and lower Waterton, partially confined canyon sites were selected for comparison to the geomorphically similar sites on the lower St. Mary. One of the only locations with saplings in the canyons on the upper St. Mary was selected in addition to a second site further downstream with more abundant saplings. Along the lower Waterton, a site was established in the 1st Canyon, with two additional sites further downstream, including one that had been used in a prior seedling establishment study (Kalishcuk et al. 2001). The site along the upper Waterton was provided by a willing landowner, as was one of the sites on the Castle. The other site on the Castle was used in the prior seedling establishment study (Kalischuk et al. 2001). Site locations are shown in Figure 2-1 with specific coordinates listed in Table 2-3. More detailed maps showing the locations of each study site are found in Appendix A. Characteristics of each site are listed in Table 2-3.

Sapling data collection

Excavations: At each study site, up to ten saplings were arbitrarily selected for excavation. For each sapling, the height of the above-ground stem was measured and then cut off at the substrate surface. A hole was dug down around the stem to expose the roots. Holes varied in depth from shallow to over 0.5 m. Once the root was exposed, a section containing the stem to root transition was cut out with a handsaw, and brought to the lab for aging. Multiple cross-sectional discs from each sample were cut throughout the stem-root transition so that the maximum number of annual growth rings could be counted. The discs were sanded with progressively finer sandpaper down to a 400 grit paper to reveal annual rings and to determine if pith was present.

Roots do not have a central pith, a mass of spongy parenchyma tissue inside the vascular cylinder, whereas shoots do (Esau 1977). The sanded discs were aged under a dissecting microscope with up to four-times magnification. If the root crown, the point where the stem and root meet, was not obtained, a '+' was added to the apparent year of establishment indicating that the sapling could have established in a prior year(s). The diameter of the disc with the maximum number of rings was measured with calipers along two different axes, and an average diameter was calculated.

Table 2-3. Some attributes of the study sites selected for sapling measurements. Study sites identification tags include the reach abbreviation, followed by a sequential upstream to downstream number. Coordinates are from Google Earth (2007) and were taken from the approximate middle of the site.

Study site ID	Coordinates		Altitude (m.a.s.l.)	Dist. from gauge (km)	Landform type	Main substrate	Major disturbances
upper St. Mary							
USM-1	49°03'58.56"	113°13'34.59"	1196	5 u/s	island	large cobble / silt	cattle
USM-2	49°05'05.43"	113°13'29.45"	1180	1 u/s	island	silt	beaver / some browsing
lower St. Mary							
LSM-1	49°21'44.71"	113°03'30.50"	1052	67 u/s (6 to dam)	bar	cobble	beaver, cattle
LSM-2	49°22'06.64"	113°01'30.35"	1029	60 u/s	island	silt	beaver, cattle, deposition
LSM-3	49°30'35.21"	112°56'42.35"	910	35 u/s	island	silt	deposition
lower St. Mary d/s Pothole							
LSM/P-1	49°35'43.29"	112°53'40.82"	850	6 d/s	bar	silt	deposition
LSM/P-2	49°36'09.37"	112°54'13.19"	850	7 d/s	bar	silt	browsing, deposition
LSM/P-3	49°36'16.99"	112°54'24.73"	850	7.5 d/s	bar	silt	browsing, beaver
upper Waterton							
UW-1	49°14'39.27"	113°44'48.23"	1219	24 d/s	island	cobble / silt	-
lower Waterton							
LW-1	49°21'20.77"	113°39'20.72"	1128	25.5 u/s	island	cobble/gravel/sand	beaver, leaf galls
LW-2	49°25'39.43"	113°30'13.52"	1052	4 u/s	bar	silt/sand	deposition
LW-3	49°30'11.50"	113°19'27.27"	983	22 d/s	bar	silt	some deposition
Castle							
CSL-1	49°30'17.02"	114°11'23.19"	1219	5 u/s	island	small cobble / sand	-
CSL-2	49°29'10.67"	114°07'21.40"	1183	3/5 d/s	bar	organic litter / silt / cobble	heavily browsed (moose or elk)

Transects: Two or three belt transects were established at each study site for a total of 33 transects. Originally, three transects were established at each study site, but two transects provided sufficient numbers of saplings. Transects were positioned through sapling zones and spaced fairly evenly throughout the site. Transects ran from the water's edge, towards the back of the floodplain, perpendicular to the river channel, and stopped where there were no more saplings. Transect lengths were thus variable and so were transect widths. Depending on sapling density, transect widths varied from 1 m to 6 m.

Along each transect, floodplain elevations, sapling densities, sapling heights and diameters were measured. Sapling growth forms were noted, species were categorized and evidence of damage was recorded.

First, elevations were surveyed to the nearest 0.5 cm every one to four meters along the transect using a transit and staff gauge. When floodplain surfaces were highly variable, one meter intervals were used and when change was gradual, intervals up to four meters were used. Linear interpolation was used to calculate elevations for points along the transect that were not directly measured, but supported saplings. I converted the elevation measurements to elevation above river stage at baseflow using river stage data obtained from real-time hydrometric stations available from either Alberta Environment (http://www3.gov.ab.ca/env/water/basins/ BasinForm.cfm) or Water Survey of Canada (http://scitech.pyr.ec.gc.ca/waterweb/ main.asp). An average water level around the time when the measurements were taken was obtained, accounting for a time lag based on the distance between the gauge and the study site and an estimated water velocity of 1 m/s. My definition of baseflow was derived from Leopold's (1994)

definition of a low flow, such that low flow was the flow that was exceeded 95% of the time for the last ten years of data.

Next, the number of saplings in each of the continuous 1 m long quadrats along the belt transect were counted. The count of each quadrat was divided by the width of the belt transect to determine saplings per m^2, which was then averaged with the value from the preceding and proceeding quadrats to provide a three meter moving-average density.

Subsequently, the height of each sapling encountered along the transect was measured with a meter stick or staff gauge to the nearest centimeter. Height was measured as the length from the ground surface to the 2005 terminal bud scar, and thus saplings that were measured early or late in the 2006 growing season were comparable. Stem diameter at ground level of every sapling was measured with calipers to the nearest 0.5 mm. Thus, for every sapling there was a pair of height-diameter measurements.

Saplings were classified as single stems, branched single stems, clumps, or sprouts based on apparent clonal structure and origin. Single stems and branched single stems had only one point of emergence from the substrate, but branched single stems had more than one dominant stem. They were grouped with the single stems for some of the analyses. Clumps had multiple stems emerging from the substrate and were assumed to be joined below the surface. For clumps, height and diameter measurements were taken on the tallest stem. A sprout was defined as adventitious shoots (suckers) from a larger base such as a stump, and so was of non-seedling origin.

For species categorization, leaf shapes were visually assessed with classification into *Populus angustifolia* (POan), *P. trichocarpa* (POtr), *P. deltoides* (POde), and hybrid categories (POan-x-POtr, POtr-x-POan, POan-x-POde, POde-x-POan). Classification

was based on a simplified form of that in Gom and Rood (1999). The bidirectional introgression between *Tacamahaca* species, and heterophylly of preformed versus neoformed leaves, adventitious shoots (Floate 2004), and long versus short shoots (Dickmann et al. 2001), produce a complete continuum of leaf morphologies. Consequently, species and hybrids were only tentatively identified.

Finally, any damage to the sapling was noted. Evidence of beaver and other browsing, if the tip of the main stem was missing, if the sapling was covered in flood debris or was prostrate, and any other health problems were recorded.

2.2.4 Sapling Data Analysis

Sapling size variables

Both height and diameter variables were indicators of total growth occurring over the life of the sapling, and a third size variable that combined height and diameter was also calculated. Height versus diameter relationships are commonly used in forest mensuration and in studies of allometric growth (Husch et al. 2003; Niklas 1995; Huang et al. 1992), and they provide information on whether the saplings were tall and thin or short and wide. For each sapling with a pair of height-diameter measurements, a height-to-diameter ratio (H/D) was calculated.

Statistical test overview

Statistical analyses were performed in SPSS (version 13.0 for Windows, 2004) with the exception of the nested ANOVA, which was performed in JMP (version 7.0, SAS Institute Inc. 2007). When possible, both parametric and non-parametric tests were

carried out because the residuals of the size variables were not normally distributed (Figure D-1, D-2; Table D-1).

Excavated saplings

Establishment patterns in relation to the flood year were explored with relative frequency histograms. Saplings in which the precise age was not obtained (e.g. 1999+) contributed to the maximum year obtained, the prior year, and the year before that, with a weighting of 0.4 to 0.4 to 0.2 (e.g. 1999: 0.4, 1998: 0.4, 1997: 0.2). Sapling height and diameter versus age relationships were tested with linear regression to determine if an equation could be derived to assign ages to saplings measured along the transects. Also, differences in mean size of saplings that established from 1995 to 1998, including saplings identified as 1999+, were tested among the reaches using ANOVA and Kruskal-Wallis tests. This comparison helped eliminate age as a confounding factor. Levene's tests for equality of variances were carried out and if variances were unequal, Tamhane's post-hoc tests for unequal variance were performed.

Patterns in the sapling transect data

Initially, all saplings measured along the transects were summarized in data graphs that illustrated aspects including sapling heights, densities, and spatial distribution with distances from the river and floodplain elevation. This data set included saplings that were damaged or beaver-browsed, sprouts, and saplings from the different cottonwood species and their hybrids. As these factors would influence sapling size, a sapling whose growth form was classified as a sprout, whose main stem had been

damaged by browsing or other causes, or whose diameter at ground level included beaver-browsed stems or only represented one of two attached stems (i.e. a branched single stem) were removed from the sample for subsequent size comparisons.

Saplings identified as *P. deltoides* or their hybrids were also removed from the sample (Appendix C), but all of the saplings within the *Tacamahaca* section were analyzed together as there did not appear to be species-related size differences within this section (Appendix C). Clumped and single stemmed saplings were also analyzed together, after an initial analysis indicated that there were no size differences between these two growth forms (Appendix D).

Size distribution: To compare the ranges of sapling sizes, differences in sapling size distributions among the transects within a site, and among the sites were tested using two-sample Kolmogorov-Smirnov tests.

Mean sapling size: Differences in mean sapling size were tested using a nested ANOVA in JMP in which transects were nested in sites, nested in reaches, and sites were nested in reaches. This test determines if there is a significant difference among groups of interest (i.e. reaches), when there is no link among sampling units from one reach to other. It also indicates whether variation among the reaches is greater than the variation that is to be expected among the sampling units. Student's t post hoc tests were performed at the site and reach levels.

Relationships between sapling size and streamflow: As the rivers studied were of different sizes, an analysis with discharge values would be confounded, therefore river stage above base stage values were used to determine relationships between size and hydrology. This analysis was done at the reach level as this was the level on which hydrological variables differed. Linear regression was used to determine relationships between adjusted reach mean sapling sizes and streamflow. Adjusted means are the means of means, and thus weight transects within a site equally, and sites within a reach equally.

Relationships between sapling size and other factors:

Beaver-browsing: The proportion of beaver-browsed saplings was compared among the reaches using a χ^2 test to determine if beaver-browsing was greater along the LSM. The expected values used in the χ^2 test were derived from the data, and were calculated based on an equal distribution among the reaches. Although beaver-browsed saplings were removed from the data set for sapling size analyses, the size of browsed and non-browsed saplings were compared using t-tests and Mann-Whitney tests to determine if beaver-browsing was contributing to the observation of short saplings along the lower St. Mary.

Elevation: The relationships between transect mean elevation and transect mean heights, diameters, and H/D were tested using linear regression as floodplain position was an edaphic factor that could potentially affect size.

Density: The ecological theory of the effect of density on plant performance is well-developed (i.e. -3/2 power law of self-thinning, density-dependent growth or intraspecific competition), whereby decreasing density results in increased biomass of

individual plants (Smith 1996). Primary literature on the effects of density on height and diameter growth is less prevalent. Studies in loblolly pine plantations and in young stands of oak and yellow-poplar (*Liriodendron sp.*) have indicated that height growth is less affected than diameter growth by thinning (Allen and Marquis 1970; Harrington 2001). Other silviculture studies on coastal stands of black cottonwood have shown an increase in both height and diameter growth with early stand thinning (Peterson et al. 1996). Because of the potential for different growth rates at different levels of competition, the effect of density on the sapling size variables was explored at the transect, site, and reach levels with both linear regression and power curve functions. However, this was undertaken with the understanding that the analysis was confounded with the effect from different age classes and that the level of density at which competition occurs would vary with resource limitation, which would vary across the sites.

Geomorphic context: ArcMap 9.0 was used to measure valley widths, which provided an indication of geomorphic context. Using a contour interval shape file, I measured the distance perpendicular to the river between the first contour intervals on either side of the river. Measurements were taken at five points within a 1 km area surrounding each study site. The relationship between mean sapling size and valley widths was then tested using linear regression. The impact of geomorphic context was explored because Willms et al. (2006) found that cottonwood growth was greater in more confined reaches, possibly due to decreased density. However, this pattern only applied to mature trees and was not observed during the juvenile growth phase. Stromberg and

Patten (1996) also observed differences in cottonwood growth with geomorphic context, possibly due to differences in streamflow-groundwater connections.

2.3 RESULTS

2.3.1 Hydrology

Minimum criteria

Even before the St. Mary Dam was completed, the lower St. Mary River (LSM) experienced many more low flow days (days < 2.75 m^3/s) than the upper reach (Table 2-4), primarily due to the Kimball diversion. After the construction of the Dam, the number of days when the criterion was not met along the LSM remained high, compared to the USM (Table 2-4). When the minimum flow criterion was implemented in 1991 the number of days when discharge was less than 2.75 m^3/s dramatically decreased to about zero days, while it increased on the upper reach (Table 2-4). Further evidence for successful implementation was provided by substantially greater $7Q_{10}$ values that were equivalent to values of the USM (Figure 2-2).

Compliance was also generally successful on the lower Waterton River. There were no major irrigation projects on the Waterton prior to the construction of the Waterton Dam in 1964, and there were no days when the flow was less than 2.27 m^3/s (Table 2-4). The Waterton Dam resulted in a large increase in low flow days, but after 1991 that number was greatly reduced. There were about five days per growing season when the criterion was not being met, similar to the number of low flow days on the free-flowing upper reach, indicating that the natural upstream water supply hindered compliance. Prior to damming on the Waterton, the lower reach had a greater $7Q_{10}$ than

the upper reach due to the input from several small tributaries. The Dam greatly reduced $7Q_{10}$, but the 1991 criterion helped to improve low flows. The post-1991 period was also a drier period than the previous intervals as indicated by the lower $7Q_{10}$ value for the free-flowing upper reach.

Table 2-4. Mean number of days during the growing season (±SE) when average daily streamflow did not meet the minimum flow criteria along the upper and lower St. Mary River and the upper and lower Waterton River, among three time periods. The pre-dam period was from 1912-1950 and 1916-1963 for the St. Mary and Waterton respectively. The post dam – pre-IF (instream flow) period was from 1952-1990 and 1965-1990, and the post-IF period was from 1991-2005 along both rivers.

	St. Mary		Waterton	
	upper	lower	upper	lower
pre-dam	0.5 ± 0.3	57.7 ± 6.8	0.1 ± 0.1	0.0 ± 0.0
post-dam - pre-IF	0.0 ± 0.0	55.7 ± 8.3	1.0 ± 0.8	53.5 ± 10.6
post-IF	1.5 ± 1.4	0.5 ± 0.3	4.7 ± 2.8	4.7 ± 2.0

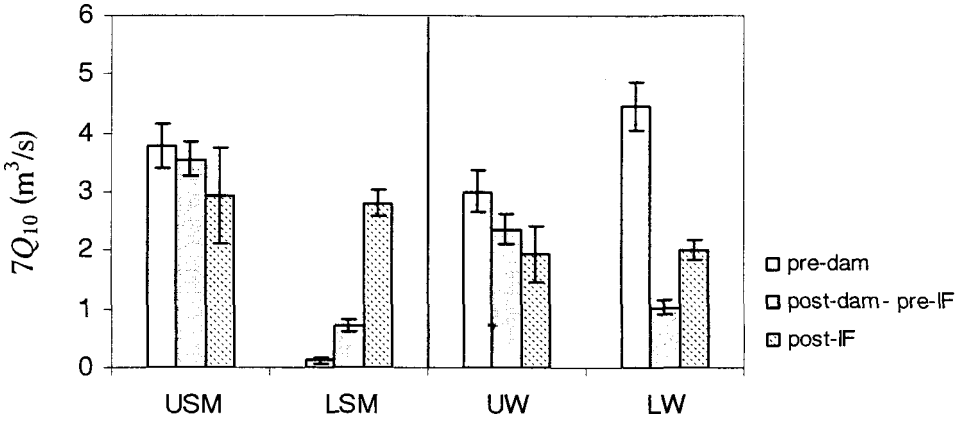

Figure 2-2. Comparison of the $7Q_{10}$ (±SD), the average minimum flow for seven consecutive days with a 10 year return interval, along the upper and lower St. Mary River (USM, LSM) and the upper and lower Waterton River (UW, LW), among three time periods. The pre-dam period along the LSM was regulated by the Kimball diversion, while the pre-dam period on the LW was free-flowing. For both rivers, the minimum instream flows (IF) were implemented in 1991.

Streamflow within a sapling's life

Discharge along the lower St. Mary reach was slightly lower than discharge along the lower St. Mary downstream of Pothole reach because of the minor contribution from Pothole Creek. The historic contribution of Pothole Creek to St. Mary River near Lethbridge flow is further explored in Appendix E.

Over a sapling's life to date, estimated to be from 1996–2005, the higher altitude partly regulated and free-flowing reaches (USM, UW, CSL) had the largest May and June discharges (Figure 2-3A). June flows on LW were also relatively high whereas the other regulated reach, the LSM, had the lowest June flows. The USM continued to have the highest flows throughout the rest of the months in the growing season, whereas the CSL, the LW and LSM had lower and relatively comparable flows. The CSL and LSM thus provide an interesting comparison since both a free-flowing and a regulated reach have low summer flows.

When river stage above baseflow was examined, CSL had the highest spring (May-June) stage, and LSM and LW had low spring stage (Figure 2-3B). LSM had comparable late summer stage to the other reaches while LW had very low summer stage.

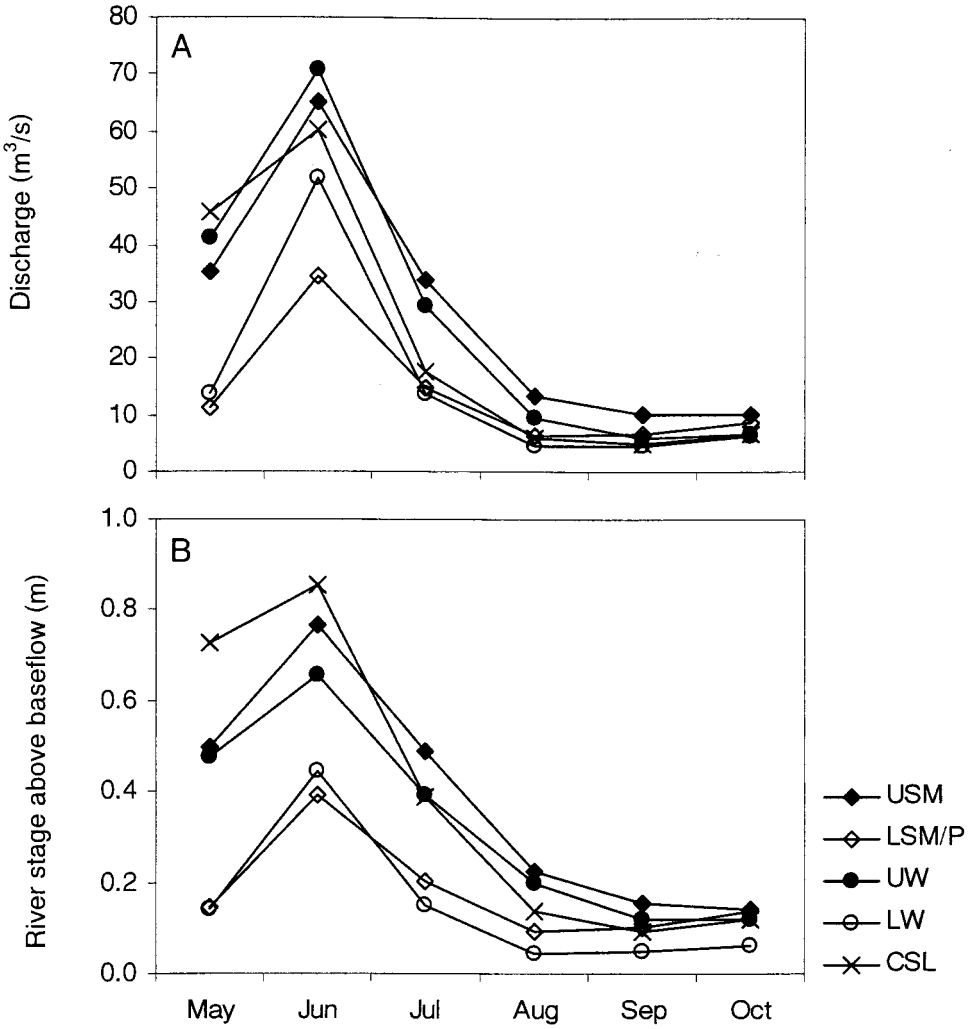

Figure 2-3. Reach comparisons of mean monthly discharge (A) and river stage above baseflow (B) during the growing season within a sapling's life (1996-2005).

2.3.2 Sapling extent

From field observations made between 2005 and 2007, and from photographs taken during the 2007 low-altitude flight along the entire length of the lower St. Mary River, we found cottonwood saplings distributed in sparse patches indicating successful post-establishment survival. However, the saplings appeared small and stunted given their likely establishment approximately ten years ago. Saplings were generally absent along most of the LSM reach from the dam to the confluence with Pothole Creek, with only 8 out of 57 of the kilometer points supporting saplings (Figure 2-4). In many locations, apparently suitable sites were not colonized and in other areas, especially in Box Canyon, sandbar willow (*Salix exigua*), but not cottonwood, was abundant. In one location, saplings that had been over 2 m tall in 2002 were no longer there due to beaver-browsing (Figure 2-5). This location was Site II-4 from the prior study of seedling recruitment (Rood & Mahoney 2000).

Below the confluence with Pothole Creek, mature woodlands were more abundant and so were saplings, with 8 out of the 13 kilometer points supporting saplings (Figure 2-4). However, there were point bars and other apparently suitable locations with woodlands, but not saplings. One of those locations, Site III-3 from the seedling study, had a documented seedling density of about 106 individuals per m^2 in 1996, and about 5 individuals per m^2 with a mean height of 50 cm in 1999 (Rood & Mahoney 2000). However, in 2006 there were no saplings present at that site, only weedy herbaceous species (Figure 2-6). This could have been due to cattle grazing as cattle were observed numerous times in the vicinity.

In contrast to the St. Mary with its sparse patches of saplings, the entire section of the Waterton River that was floated in July 2006 supported abundant saplings with a range of vertical structure (Figure 2-7). The abundance and extensive distribution of saplings in this segment, representing approximately one third of the reach length and starting immediately downstream from the dam, indicated successful post-establishment survival along the lower Waterton River. Photographs of the entire reach from the low-altitude flight confirmed that sapling distribution was extensive with 42 out of 51 kilometer points supporting saplings, and all points supporting mature trees (Figure 2-4).

The proportion of points that supported trees and saplings differed significantly from the expectation that saplings and trees were distributed evenly among the reaches (trees: $\chi^2 = 26.8$, df = 2, $p < 0.001$; saplings: $\chi^2 = 26.8$, df = 2, $p < 0.001$). The LSM had fewer trees and saplings than expected. However, when only sites that supported trees were considered, there was no significant difference among the reaches in the proportion of points that supported saplings ($\chi^2 = 1.86$, df = 2, $p = 0.395$).

There was also no significant difference among the reaches in the proportion of points that could have provided suitable sites for saplings ($\chi^2 = 2.96$, df = 2, $p = 0.227$).

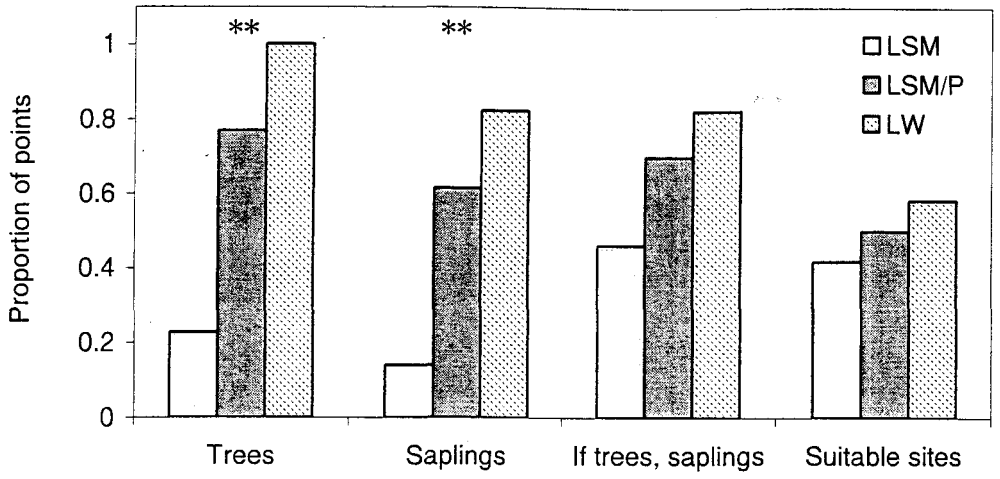

Figure 2-4. The proportion of kilometer points supporting cottonwood trees and/or saplings along the lower St. Mary, lower St. Mary downstream of Pothole, and lower Waterton reaches. Chi-square tests indicated that significantly less trees and saplings were observed along the LSM (**). However, among the reaches, if a site had trees, there was no significant difference in the probability that it would support saplings. There was also no difference in the distribution of apparently suitable sites for supporting saplings.

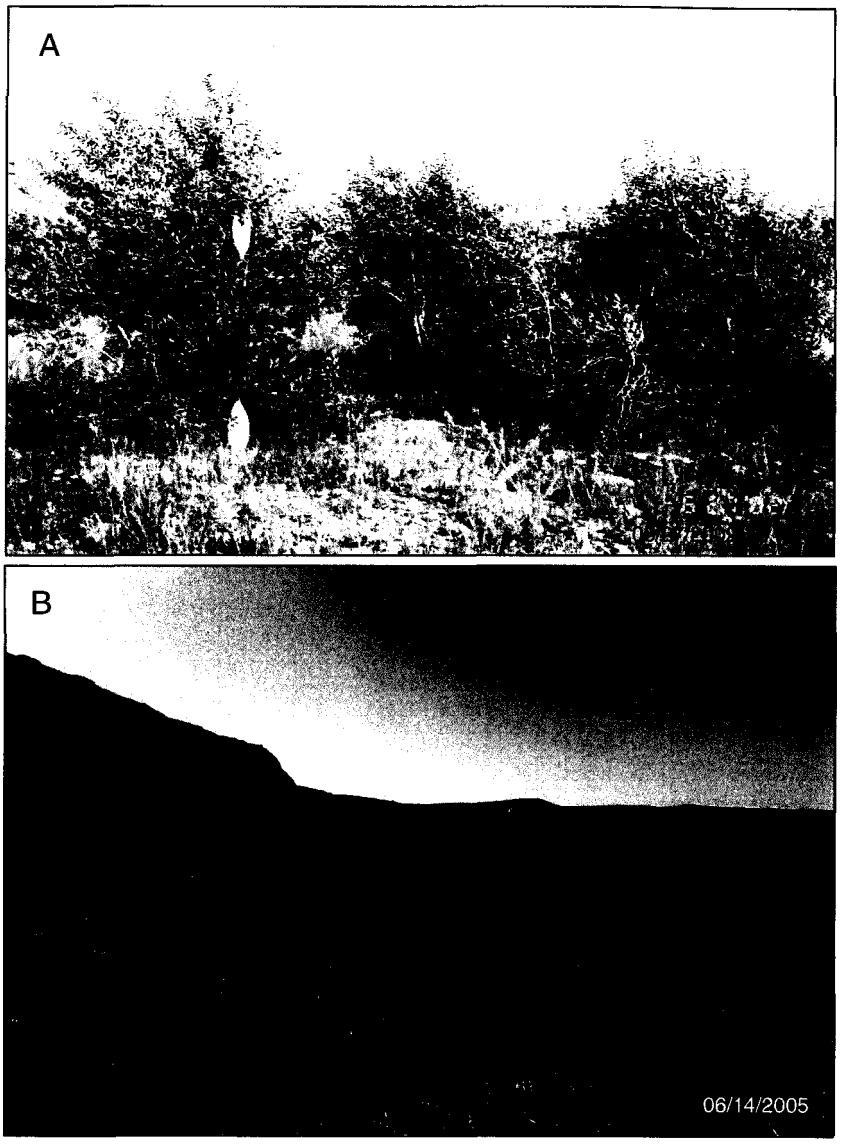

Figure 2-5. Beaver-browsing at Site II-4 from the previous seedling recruitment study on the lower St. Mary River (49°30'14" 112°57'25"). This site is about 2 km upstream of the current study's LSM-3 site. (A) In 2002, approximately 7-yr-old saplings stood over 2 m tall. (B) In 2005 those saplings were no longer present as they had been harvested by beaver. Beaver-cut stems with coppice sprouts remained.

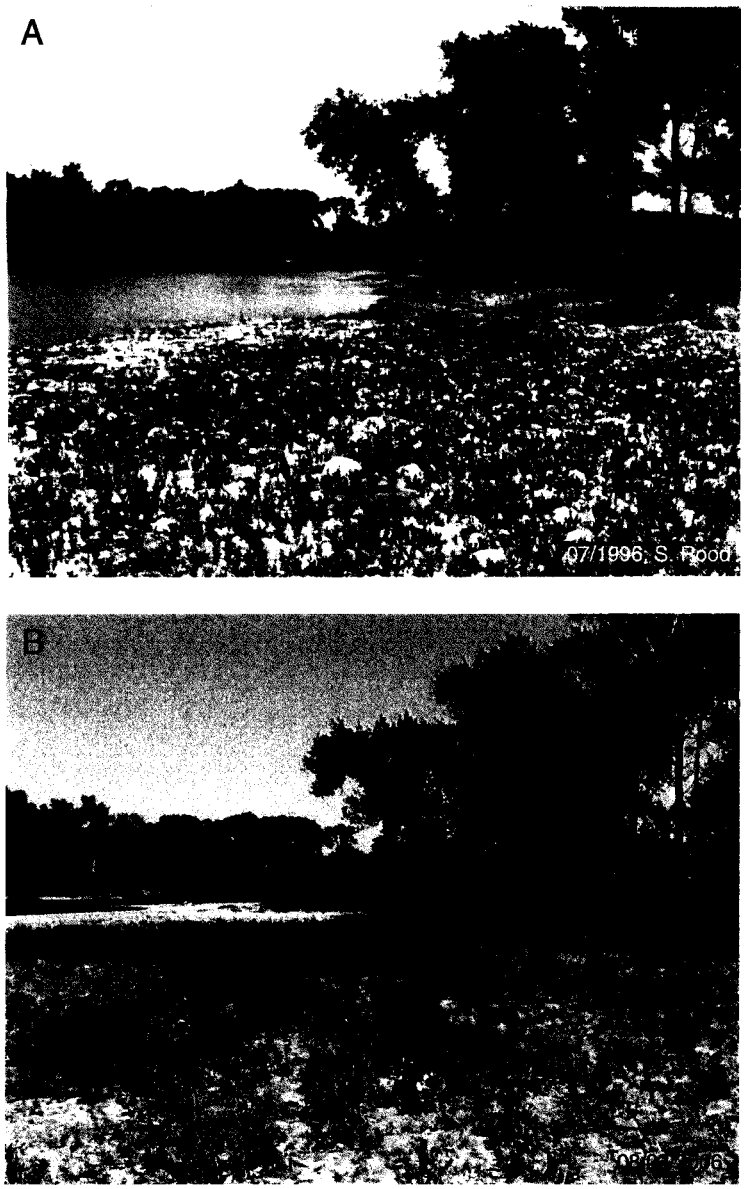

Figure 2-6. Comparison of Site III-3 (49°36'47", 112°54'26") on the lower St. Mary River downstream of Pothole Creek in 1996 (A) and 2006 (B). In 1996, this location supported an extensive seedling lawn as a result of the large 1995 flood. In 2006, no saplings were present; the plants in the photograph are herbaceous weedy species. The top photo is a scan of a slide and was originally published in Rood and Mahoney (2000).

Figure 2-7. Extensive and dense saplings from multiple age classes at a location along the lower Waterton River. This location is on the left bank of the river (photo is taken facing upstream), just downstream of the Waterton Dam (49°20'52", 113°39'27").

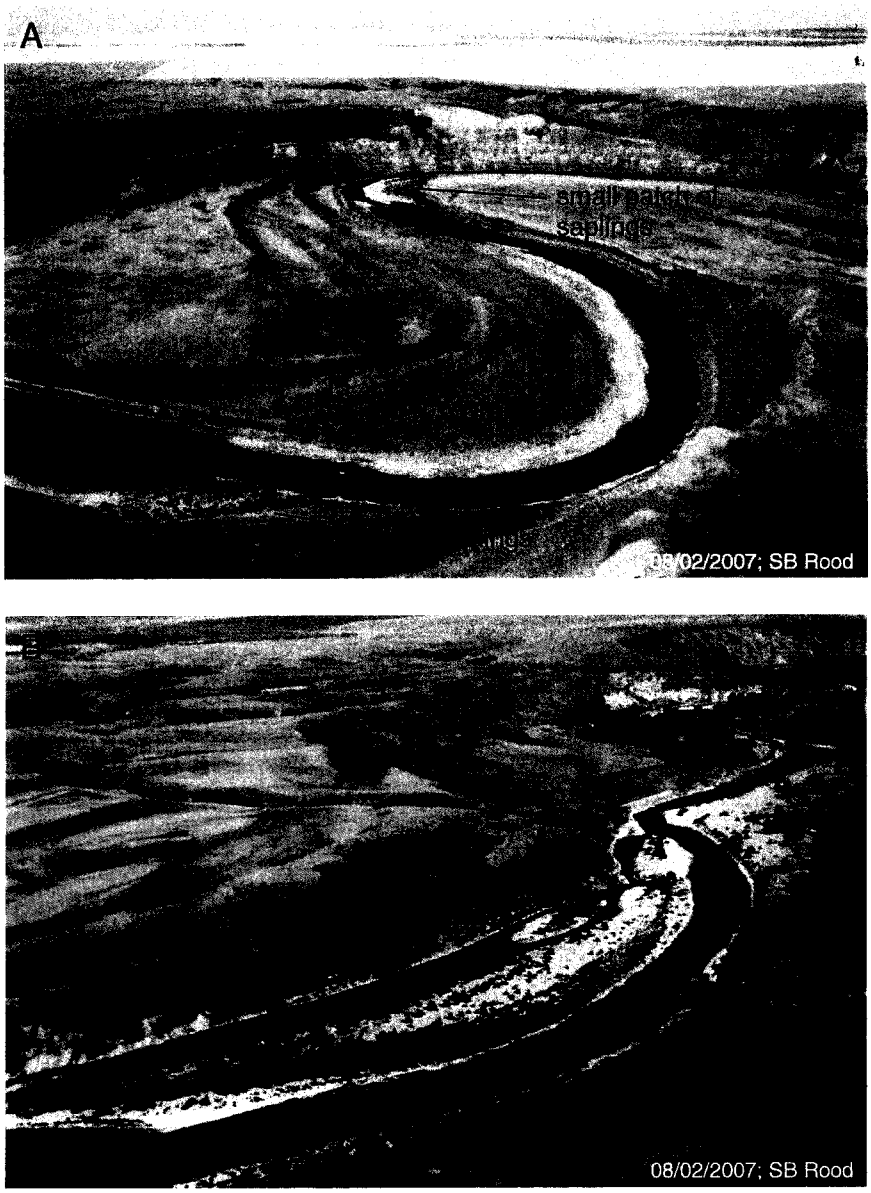

Figure 2-8. Photographs showing sapling extent along the lower St. Mary (A; 49°30'48", 112°56'57") and lower Waterton (B; 49°21'57", 113°39'18") from the 2007 low-altitude flight. Sapling were more extensive along the Waterton, and at points where saplings were present along the St. Mary, they were less abundant.

2.3.3 Sapling Analysis from Excavated Saplings

Most saplings became established in the years following the 1995 flood, with a few saplings establishing later in 1999 to 2001 and a few before 1995, especially along the Castle River (Figure 2-9). There were large size ranges for saplings that established in the same year, and consequently there were only weak relationships between sapling age and size for some study reaches (Figure 2-10, 2-11). For example, saplings that established in 1997 along the LSM/P ranged in height from about 100 to 300 cm. There were also a few saplings that had established in the early 1990s or late 1980s, and these were often in the same size range as the saplings that established after the 1995 flood.

The reaches with the larger sample sizes, the LW and LSM/P, displayed positive height or diameter versus age trends (Figure 2-10, 2-11; height vs. age: LSM/P: R^2 = 0.139, df = 27, F = 4.19, p = 0.051; LW: R^2 = 0.142, df = 23, F = 3.64, p = 0.070; diameter vs. age: LSM/P: R^2 = 0.132, df = 26, F = 3.27, p = 0.083; LW: R^2 = 0.132, df = 26, F = 3.79, p = 0.063). With only weak associations, it was not possible to assign ages to the saplings that were measured along the transects.

Among the post-1995 cohort, the LSM supported some of the shortest saplings, but they were only significantly smaller than those along the CSL reach (Figure 2-12A; one-way ANOVA: F = 3.76, df = 73, p = 0.005; Tamhane: LSM-CSL: p = 0.030; Kruskal-Wallis: χ^2 = 17.3, df = 5, p = 0.004). Significant differences in diameters were not detected among the reaches (Figure 2-12B; one-way ANOVA: F = 1.082, df = 76, p = 0.378; Kruskal-Wallis: χ^2 = 5.550, df = 5, p = 0.352), but Tamhane's post-hoc test detected a difference between LSM and UW at α = 0.10 (p = 0.079). The height-to-diameter ratios were significantly different, with the LSM reach supporting shorter, wider

saplings than the UW, LW and LSM/P reaches (Figure 2-12C; one-way ANOVA: $F = 4.44$, $df = 69$, $p = 0.003$; Kruskal-Wallis: $\chi^2 = 18.3$, $df = 5$, $p = 0.003$; Tamhane: LSM-LSM/P: $p = 0.002$, LSM-UW: $p = 0.015$, LSM-LW: $p = 0.018$). The CSL reach had the tallest, narrowest saplings, but they were not significantly different than the LSM (Tamhane: LSM-CSL: $p = 0.211$). The post-hoc groupings seemed inaccurate, possibly due to not meeting the assumptions of the ANOVA. In summary, the excavated saplings growing along the regulated reaches did not substantially vary from the saplings growing along the free-flowing reaches. The excavated sapling data set displayed large variation in size, and consequently the data collected along the transects were explored for relationships to streamflow regime.

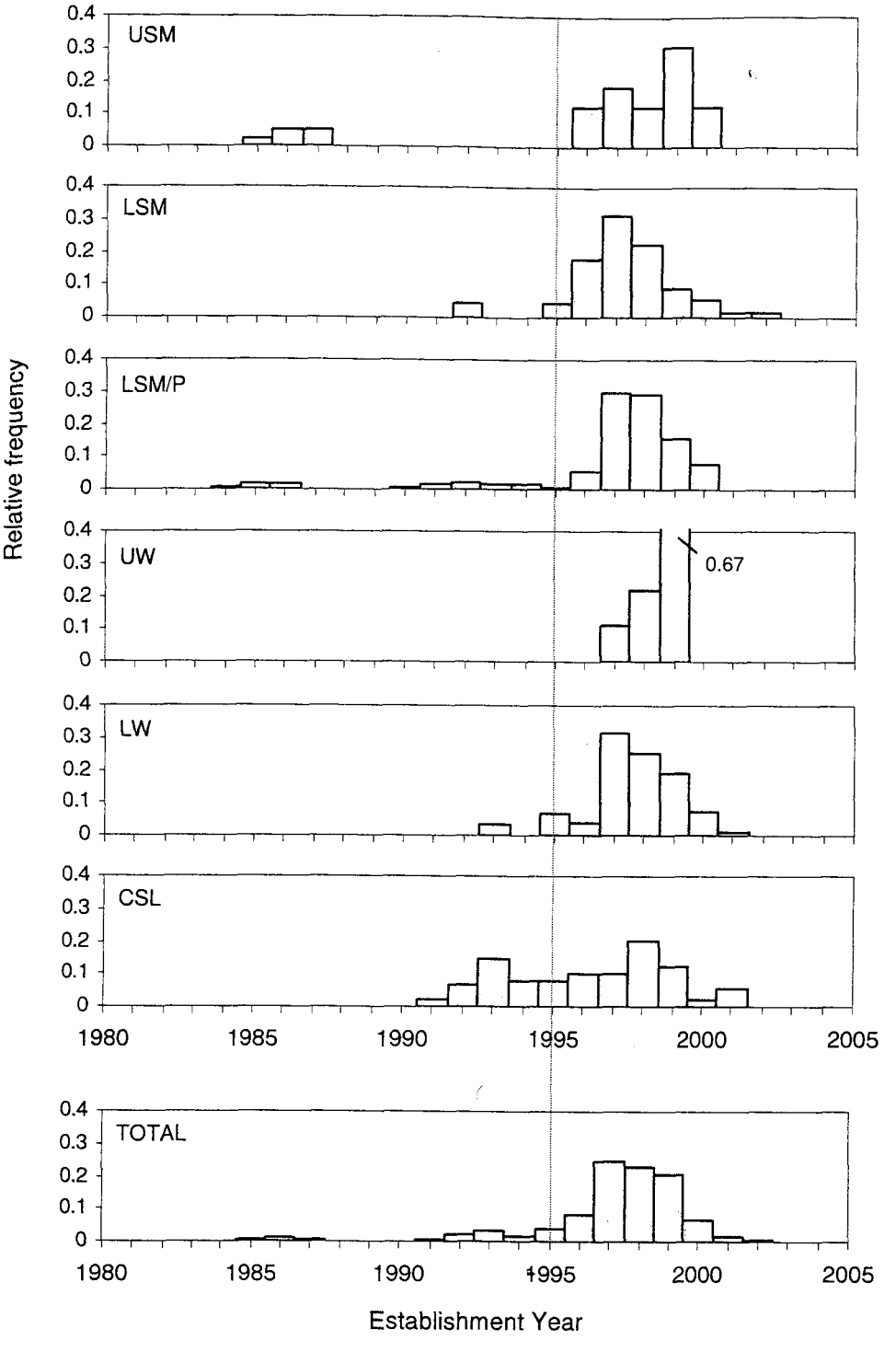

Figure 2-9. Relative frequency age distributions of saplings by reach and for the all the reaches combined. The vertical line represents the flood year.

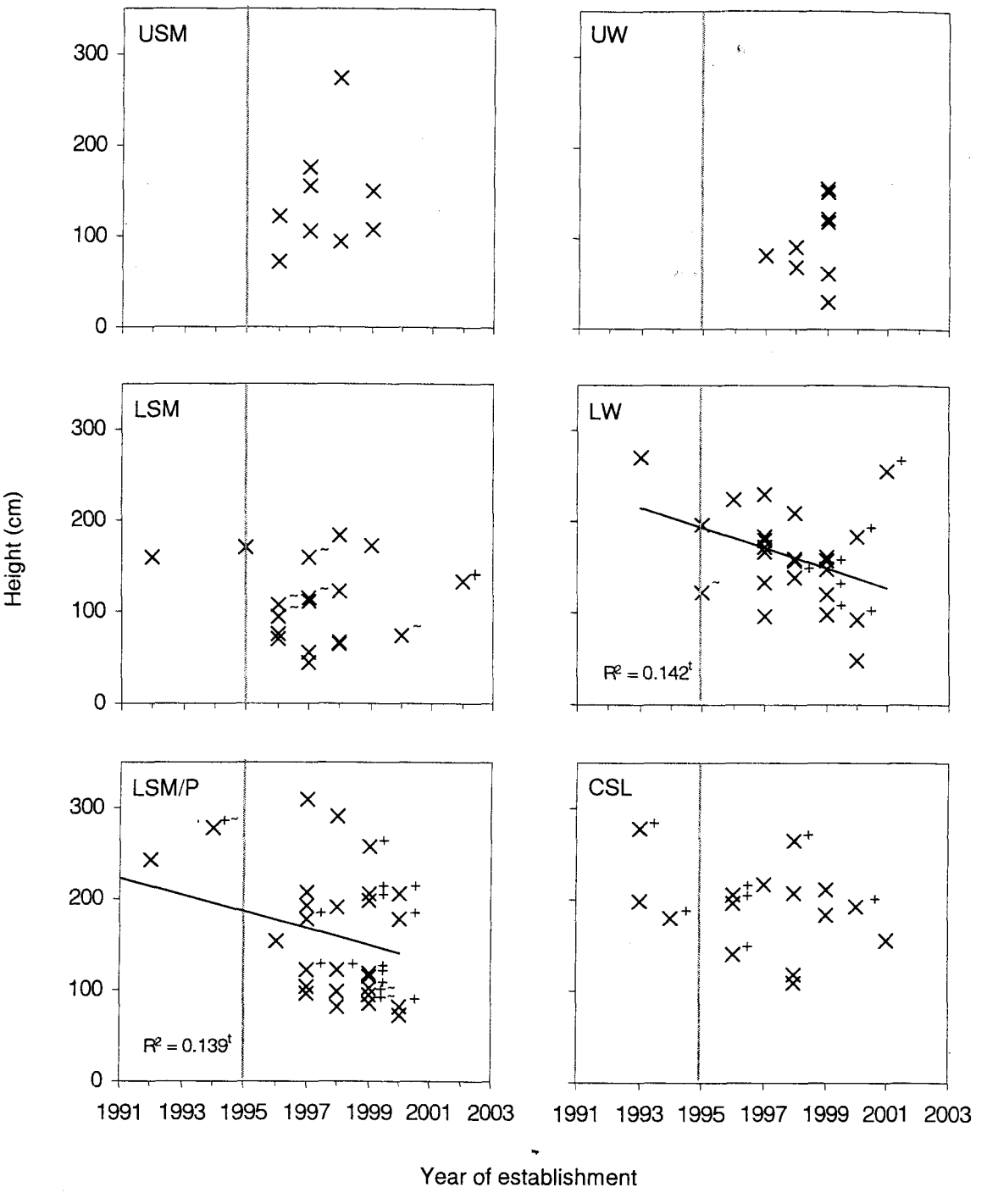

Figure 2-10. Relationship between height and year of establishment for saplings excavated along each reach. Relationship trends were detected for LSM/P and LW. The vertical line represents the flood year. (+:establishment year is earlier than or equal to the value presented; ~: height is an approximation; t: $p < 0.10$)

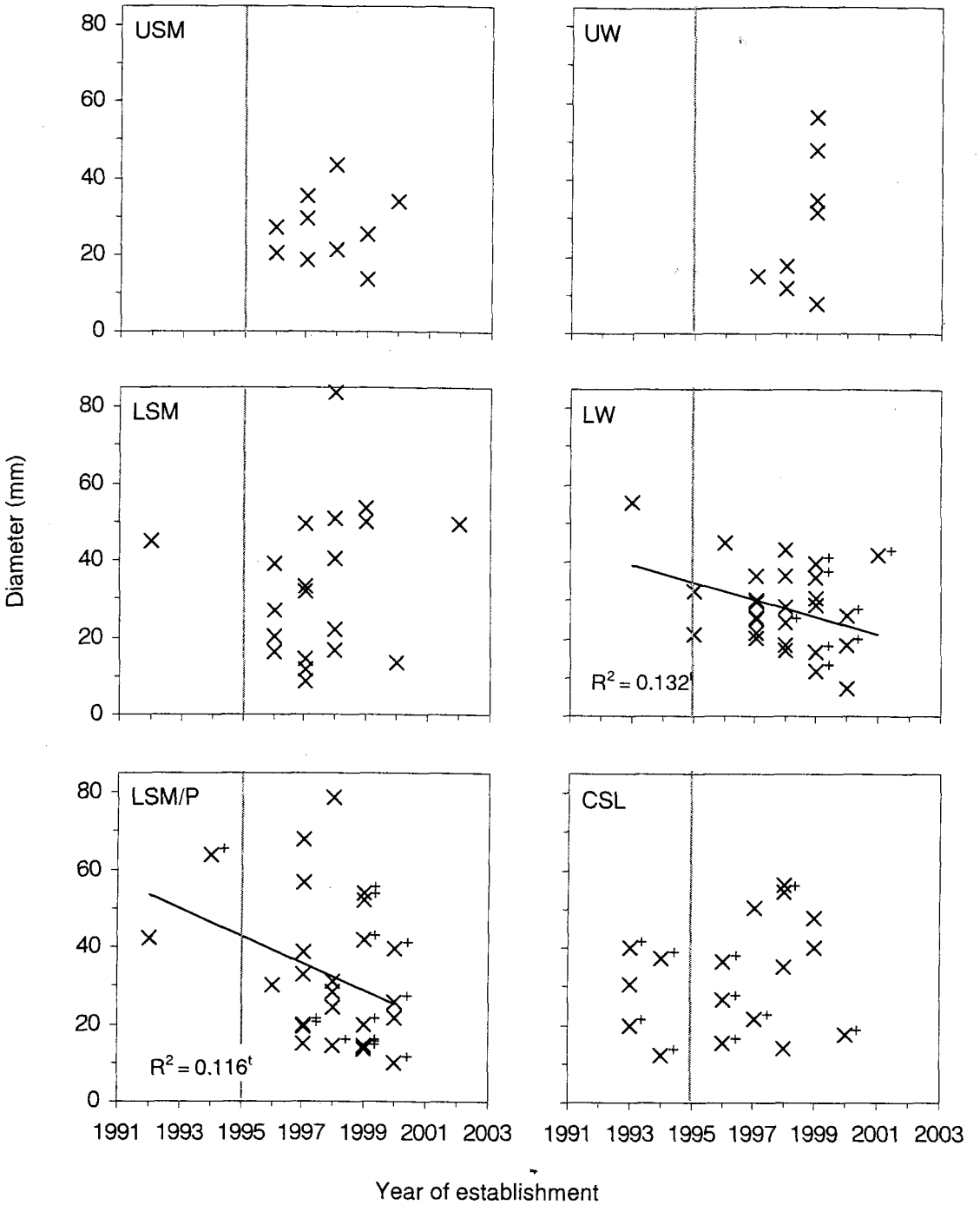

Figure 2-11. Relationship between diameter at point of maximum age and year of establishment (age) for saplings excavated along each reach. Relationship trends (t) occurred for the LSM/P and LW reaches, the reaches with the highest sample sizes. The vertical lines represent the flood year. (+: establishment year is earlier than or equal to the value presented; t: $p < 0.10$)

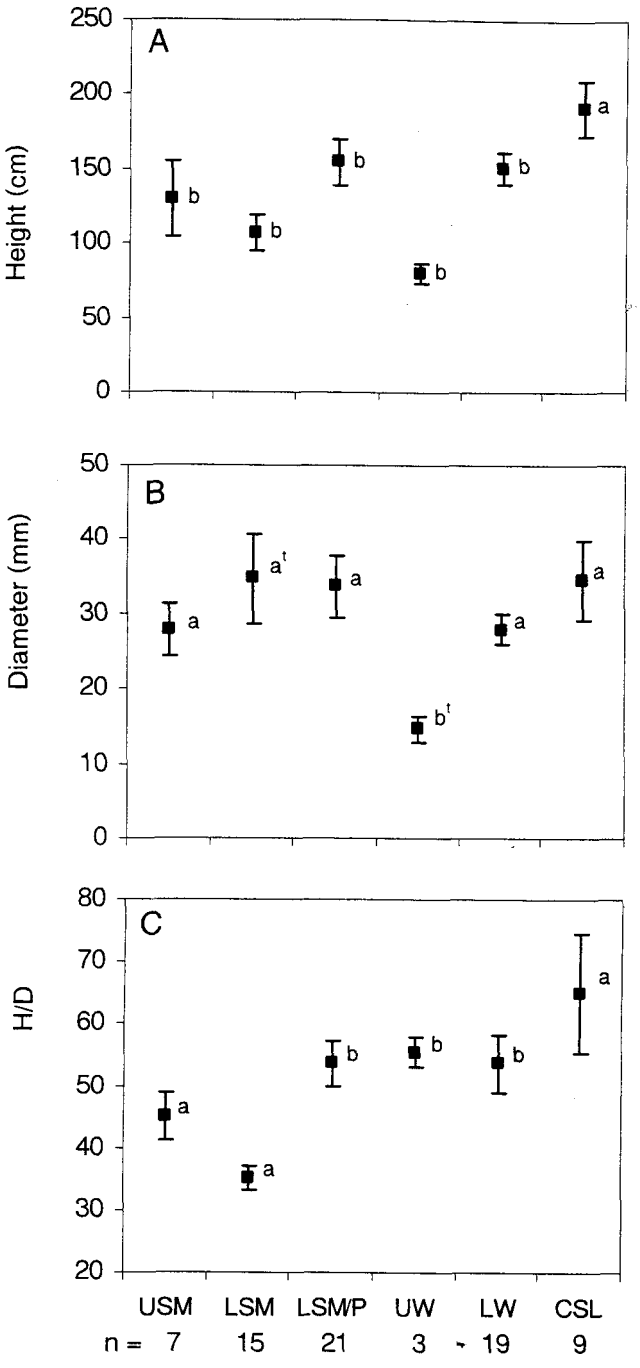

Figure 2-12. Reach mean heights (A), diameters (B), and height-to-diameter ratios (C) (±SE) for saplings that established from 1995-1998. Letters ('a', 'b') indicate groupings of similarly-sized saplings only in relation to LSM (due to the pairwise nature of the post-hoc tests) according to a one-way ANOVA with Tamhane post-hoc tests for unequal variances. (t: $p < 0.10$).

2.3.4 Sapling characteristics from transect data

Patterns in the transect sapling data

There was apparent substantial variation in sapling locations, densities, and heights among transects, even those within a single site (Figure C-1). Sapling heights did not necessarily get larger with increasing elevation or distance from the river, and the oldest saplings weren't always towards the back of the floodplain. Along some transects, saplings were found in disjunct patches (e.g. LSM-2 T1) whereas along other transects, sapling distribution was more extensive, with continuous sapling bands (e.g. LSM/P-2).

Size distribution

Figure C-1 demonstrated that the range or distribution of sapling sizes varied between the transects. Along some transects, all saplings were about the same size (e.g. LSM-3 T1), while along other transects, there were wider ranges of sapling sizes (e.g. LW-3 T1). Distributions of sapling heights along transects within a site weren't necessarily similar. For example, transects one and two at USM-2 had significantly different height distributions (Table D-2). Other sites where the transect height distributions were also significantly different were LSM-3, LSM/P-1, LSM/P-2, LSM/P-3, LW-2, LW-3, and CSL-1 (Table D-2).

Site-specific height distributions were also variable with some sites showing a Poisson-like distribution with many shorter saplings, while other sites had a more even or a normal distribution (Figure 2-13). Sites within the same reach were not necessarily similar (Table D-3). For example, LSM-1 was similar to a site on the lower Waterton and on the Castle, but not similar to LSM-2, which was only seven kilometers downstream.

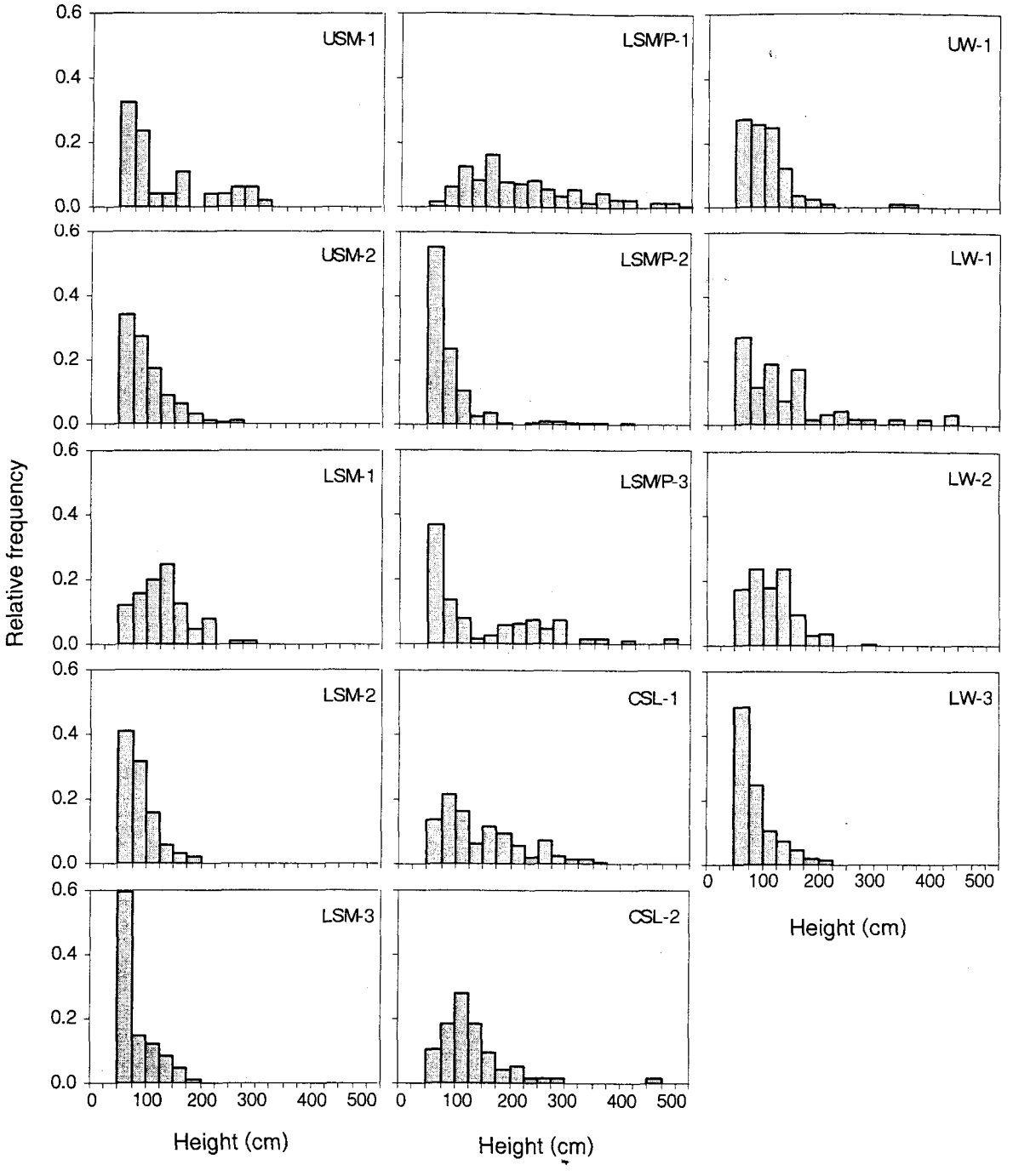

Figure 2-13. Comparison of relative frequency distributions for sapling height among study sites. LW-3 was an example of a site that had a Poisson-like distribution with many shorter saplings. Heights at LSM/P-1 were more evenly distributed with saplings in all height classes. CSL-2 had a distribution that was approaching normal.

Mean sapling size

A nested ANOVA with transects designated as a random factor, and sites and reaches as fixed factors showed that differences in sapling heights occurred among the transects nested in sites, nested in reaches (Figure 2-14C; $F = 11.2$, $df = 19$, $p < 0.001$), among the sites nested in reaches (Figure 2-14B; $F = 4.99$, $df = 8$, $p = 0.002$), and among the reaches (Figure 2-14A; $F = 2.83$, $df = 5$, $p = 0.043$). Overall, the model accounted for 35% of the variation in sapling heights, with reaches accounting for about 4% of the variation, sites for 14%, and transects for 8%. At the reach level, LSM saplings were significantly shorter than LSM/P and CSL saplings, but at the site level, saplings at LSM sites were not significantly shorter than saplings at sites on all the other reaches (Figure 2-14).

Differences in sapling diameters were also detected at all three levels (Figure 2-15; transects: $F = 4.54$, $df = 19$, $p < 0.001$; sites: $F = 10.4$, $df = 8$, $p < 0.001$; reaches: $F = 4.75$, $df = 5$, $p = 0.004$), with the model accounting for 25% of the variation. Most of that variation was accounted for at the site level (12%), while the reach and transect levels each accounted for about 3%. LSM, LSM/P, and LW had saplings with the smallest diameters, and saplings along those reaches were significantly narrower than USM and CSL saplings (Figure 2-15). At the site level, there was a lot of variation between the sites within a reach. For example, USM-1 and LW-1 supported some of the widest saplings, and USM-2 and LW-3 supported some of the narrowest. Sites along the LSM covered the complete range of sapling diameter groupings.

The height-to-diameter ratio was also significantly different among the transects, sites, and reaches (Figure 2-16; transect: $F = 1.97$, $df = 19$, $p = 0.008$; site: $F = 11.5$, $df =$

8, $p < 0.001$; reach: $F = 44.0$, $df = 5$, $p < 0.001$). This model accounted for 25% of the variation in sapling size ratio, but, in contrast to the height and diameter models, most of the variation was explained at the reach level (16%), with reduced variation occurring at the site and transect levels (7% and 2% respectively). The USM and LSM, and the LW and CSL had similar mean H/D ratios, LSM/P supported taller or narrower saplings, and UW supported shorter or wider saplings. While the other sites within a reach supported a wider range of H/D ratios, LSM sites were more similar.

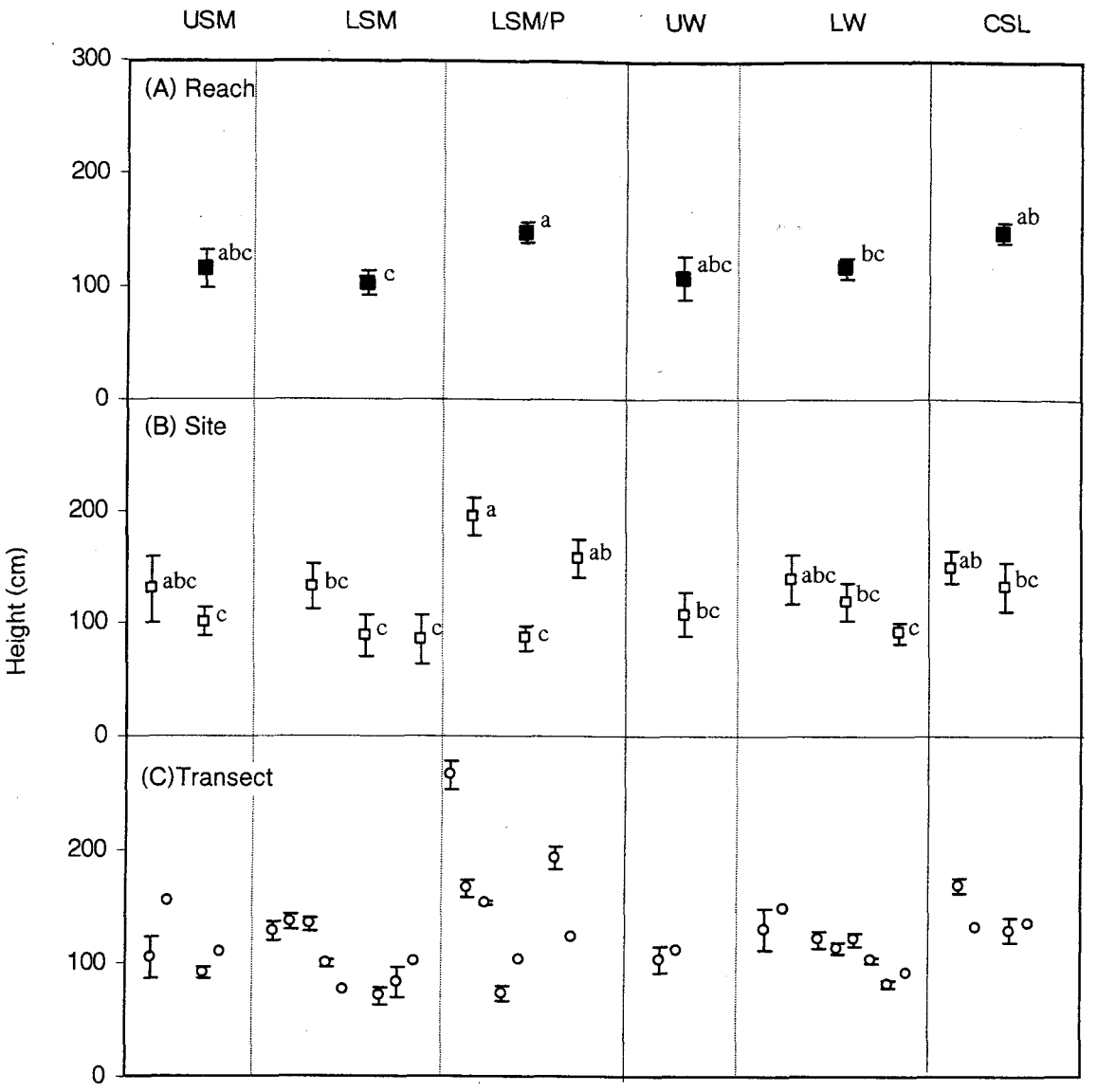

Figure 2-14. Mean (±SE) sapling heights for transects (C), which were nested in sites (B), which were nested in reaches (A). Adjusted means (±SE) are presented for sites and reaches. For reaches and sites, letters indicate groupings that were not significantly different according to Student's t post-hoc tests.

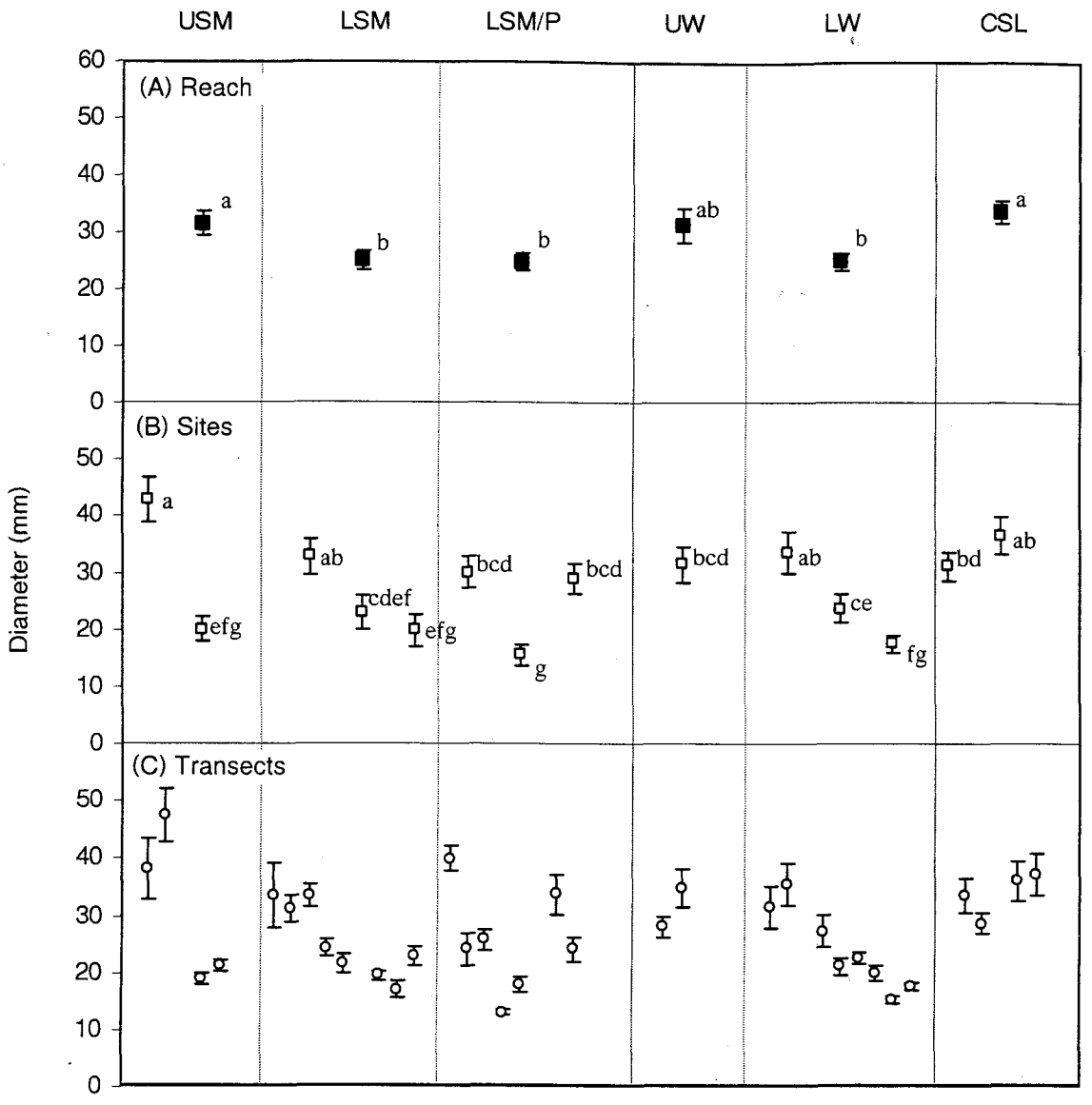

Figure 2-15. Mean (±SE) sapling diameters for transects (C), which were nested in sites (B), which were nested in reaches (A). Adjusted means (±SE) are presented for sites and reaches. Letters indicate groupings that were not significantly different according to Student's t post-hoc tests.

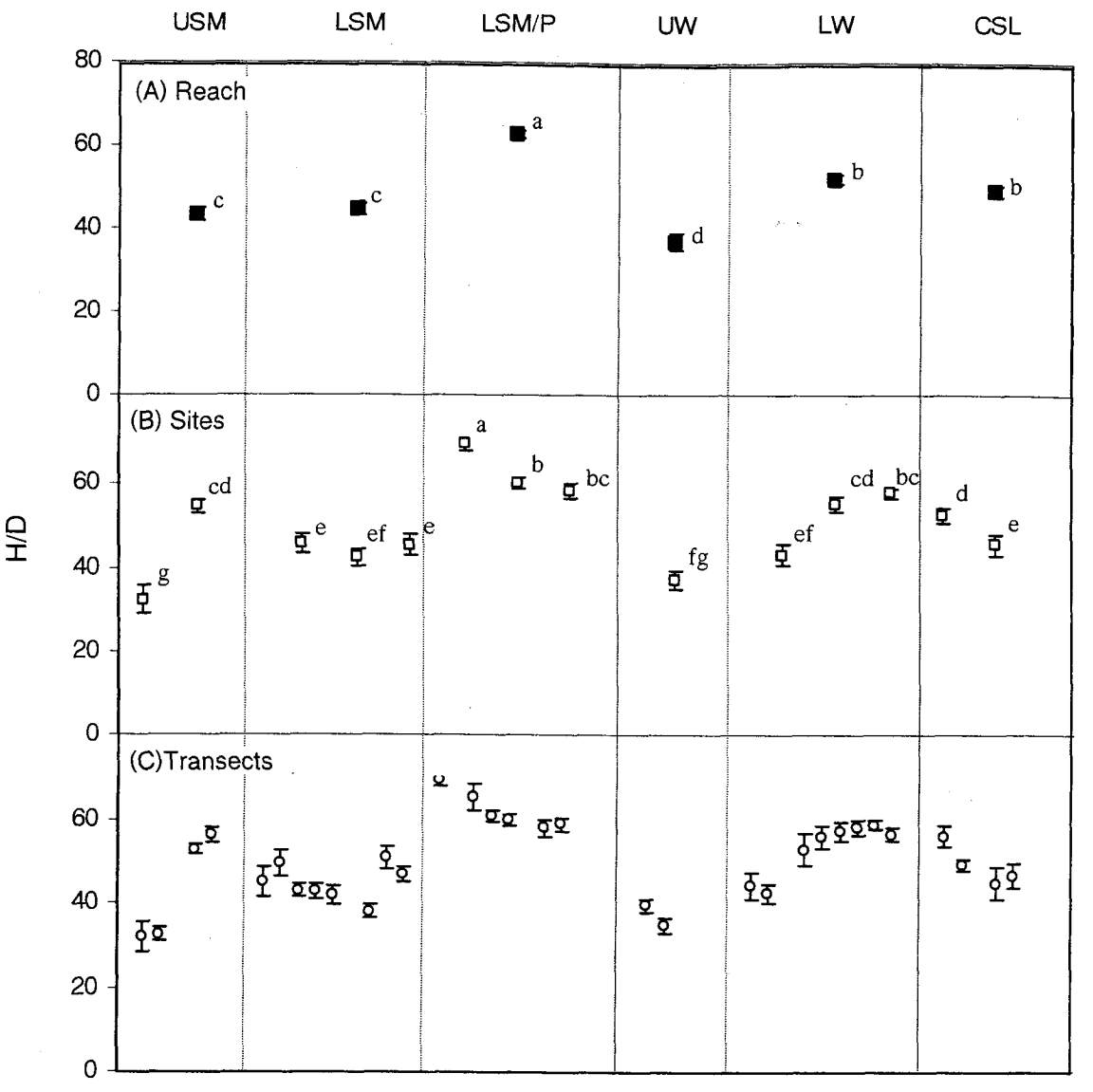

Figure 2-16. Mean (±SE) sapling height-to-diameter ratio for transects (C), which were nested in sites (B), which were nested in reaches (A). Adjusted means (±SE) are presented for sites and reaches. Letters ('a','b', etc.) indicate groupings that were not significantly different according to Student's-t post-hoc tests performed in JMP.

Relationships between sapling size and streamflow

At the reach level, the level on which streamflow varied, there was little variation in sapling size among the reaches. Heights varied within a range of about 40 cm, while diameters varied within about 10 mm. At the reach level, one of the few significant differences detected in mean sapling height by the nested ANOVA was between LSM and LSM/P (Figure 2-14). These reaches had very similar streamflow values. Consequently, there was no significant relationships between reach mean sapling height and the river stage above base stage in spring (May-June) or summer (July-August) (Figure 2-17A; spring: $R^2 = 0.064$, $F = 0.275$, $df = 5$, $p = 0.628$; summer: $R^2 = 0.002$, $F = 0.008$, $df = 5$, $p = 0.933$).

For diameter, relationships with streamflow were significant, with a positive association between reach mean diameter and the river stage above baseflow in spring and summer (Figure 2-17B; spring: $R^2 = 0.978$, $F = 198$, $df = 5$, $p < 0.001$; summer: $R^2 = 0.754$, $F = 11.5$, $df = 5$, $p = 0.028$).

There were no significant relationships between H/D and river stage (Figure 2-17C; spring: $R^2 = 0.198$, $F = 0.987$, $df = 5$, $p = 0.377$; summer: $R^2 = 0.334$, $F = 2.01$, $df = 5$, $p = 0.230$). Other factors that could have influenced sapling sizes were explored in subsequent analyses.

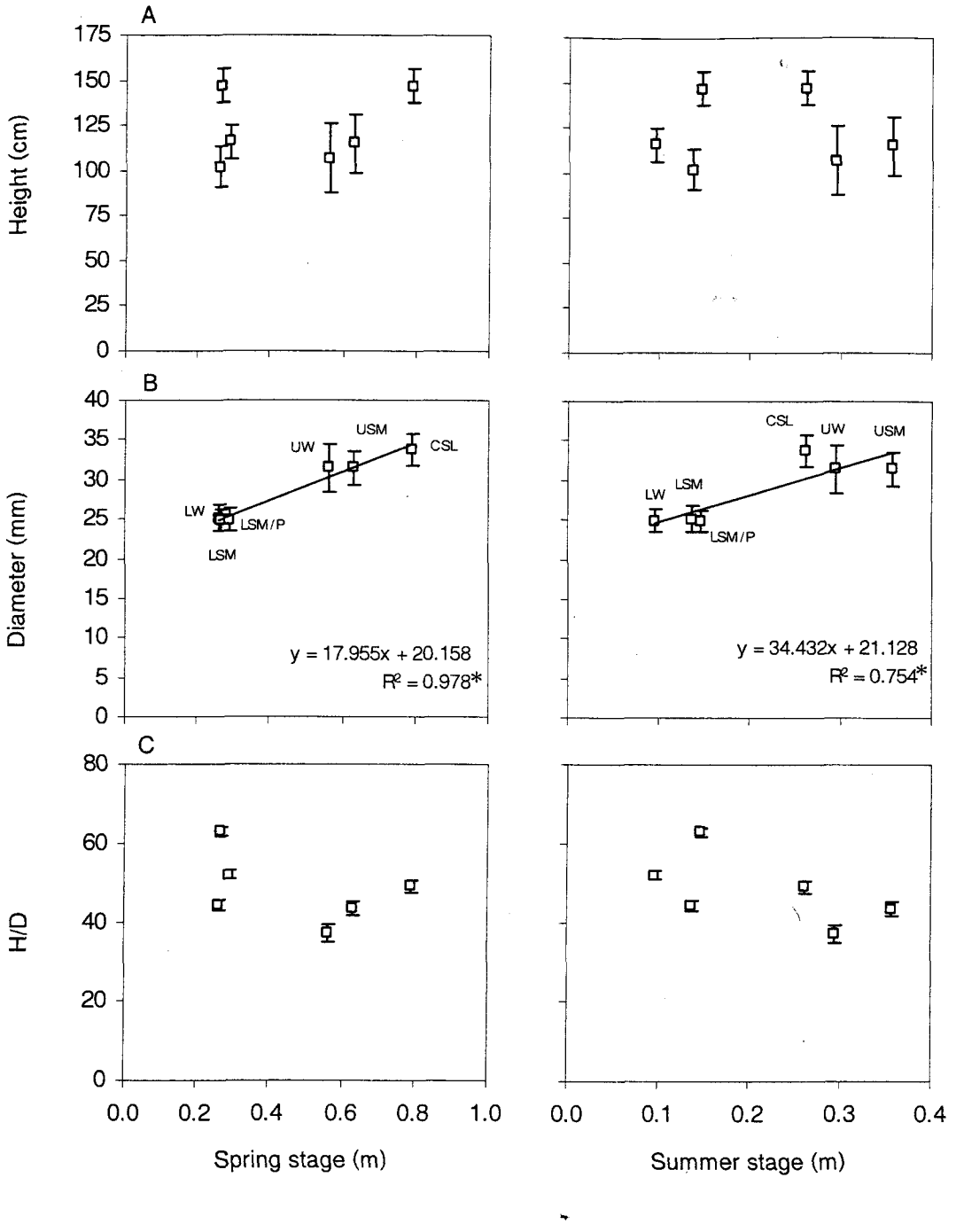

Figure 2-17. Relationships between adjusted reach mean sapling height (A), diameter (B) and H/D (C) (±SE), and the river stage above baseflow occurring in the spring (May-June) and summer (July-August). (*: $p < 0.05$)

Beaver-browsing

Field observations indicated that the lower St. Mary reach was heavily beaver-browsed, with browsing resulting in complete harvesting at one location (Figure 2-5). Subsequent data collection confirmed that LSM had been more heavily beaver-browsed than the other river reaches (Figure 2-18; $\chi^2 = 234$ df = 5, $p < 0.001$). This has likely resulted in the higher proportion of clumped and sprout saplings along this reach (Figure 2-19; $\chi^2 = 92.0$, df = 5, $p < 0.001$).

It was hypothesized that the higher occurrence of beaver-browsing was partially responsible for the small post-1995 saplings along the lower St. Mary. The mean heights of non-browsed (103 ± 3) and browsed saplings (109 ± 4) along the LSM were either not significantly different according to a t-test, or the browsed saplings were significantly taller according to a Mann-Whitney test (t-test with unequal variances: t = -1.29, df = 173, p = 0.200; Mann-Whitney: Z = -2.40, total N = 391, p = 0.016). Conversely, the mean diameter of non-browsed saplings (25.0 ± 0.8) was significantly larger than the mean diameter of browsed saplings (18.6 ± 1.0) (t-test with unequal variances: t = 5.006, df = 179, $p < 0.001$; Mann-Whitney: Z = -4.523, total N = 384, $p < 0.001$).

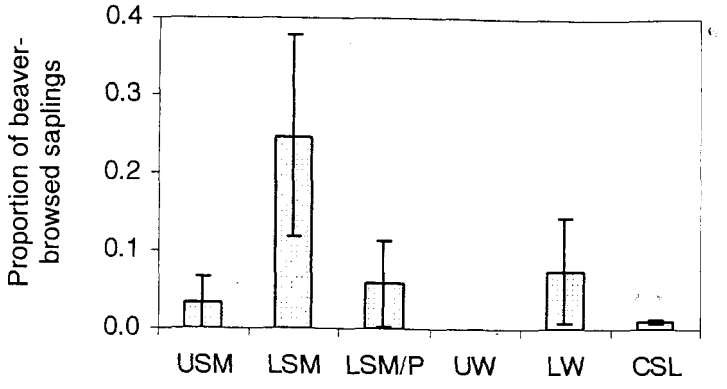

Figure 2-18. Mean proportion of beaver-browsed saplings by reach (±SE), calculated as an average of mean site proportions.

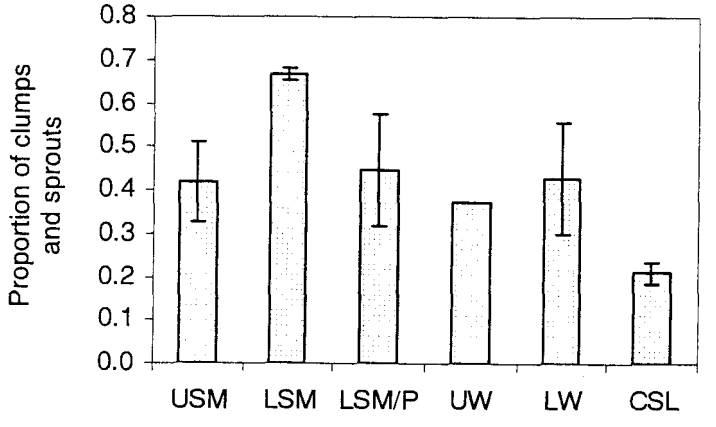

Figure 2-19. Mean proportion of clumped and sprout saplings by reach (±SE), calculated as an average of mean site proportions. The remaining proportion of saplings were classified as single stemmed.

Elevation

All saplings were found from 0.45 m to 3 m above river stage at baseflow, with 90% of the saplings found between 0.75 m and 2.25 m (Figure 2-20). Most saplings were found at 1.10 m, and the average elevation (±SE) was 1.44 ± 0.01 m.

Within this narrow range of sapling occurrence, elevational position did not affect sapling size as there was no relationship between reach mean elevation and reach mean sapling height, diameter, or height-to-diameter ratio (linear regression: height: $F = 0.043$, $df = 5$, $p = 0.845$; diameter: $F = 2.47$, $df = 5$, $p = 0.191$; H/D: $F = 1.16$, $df = 5$, $p = 0.342$).

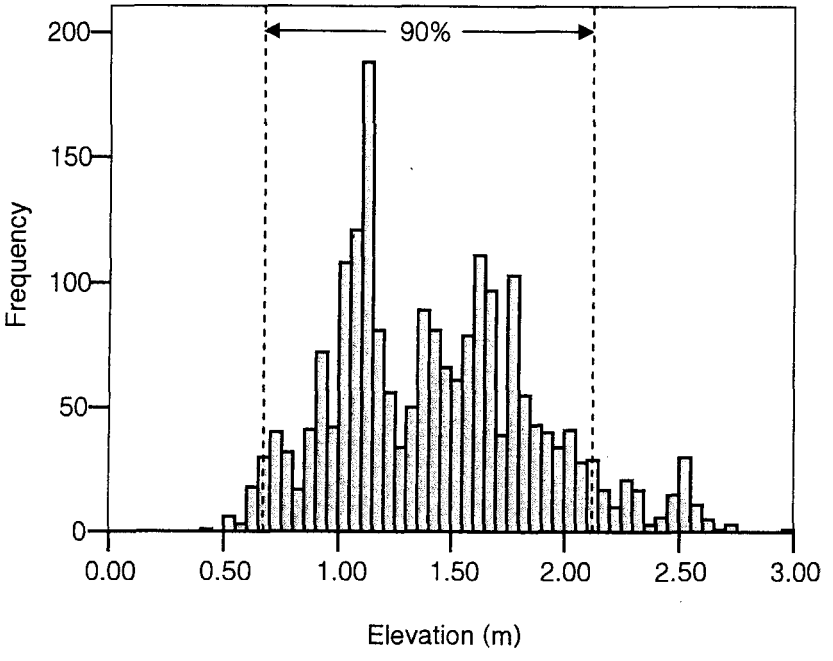

Figure 2-20. Elevational position of all saplings measured. While all saplings were found from 0.45 to 3 m, most saplings (90%) were found from 0.75 m to 2.25 m.

Density

Different reaches supported a wide range of sapling densities (Figure 2-21). Most saplings along the lower St. Mary were found in low densities (less than 1.5 saplings per m^2), similar to the distribution along the upper Waterton. In contrast, USM and LSM/P had a wide range of densities, extending to values as high as $9/m^2$. LW and CSL had low to intermediate densities. Not only were densities significantly different among the reaches, they were also different at the site and transect levels, with most of the variation occurring at the site level (Table D-4).

The relationship between sapling size and density varied dependent upon the spatial scale (Figure 2-22). At the transect level, there were significant negative power relationships between height, or diameter, and density, but the height-density relationship was not as prominent as that for diameter, or the height-to-diameter ratio (curve fit: height: $R^2 = 0.089$, $F = 3.04$, $df = 32$, $p = 0.091$; diameter: $R^2 = 0.465$, $F = 27.0$, $df = 32$, $p < 0.001$; H/D: $R^2 = 0.394$, $F = 20.1$, $df = 32$, $p < 0.001$). At the site level, the height versus density negative power relationship was not significant ($R^2 = 0.088$, $F = 1.163$, $df = 13$, $p = 0.302$), but the diameter and H/D relationships were (diameter: $R^2 = 0.541$, $F = 14.124$, $df = 13$, $p = 0.003$; H/D: $R2 = 0.443$, $F = 9.536$, $df = 13$, $p = 0.009$). At the reach level, there was a positive linear relationship between height or H/D and density (linear regression: height: $R^2 = 0.605$, $F = 6.13$, $df = 5$, $p = 0.069$; H/D: $R^2 = 0.873$, $F = 27.5$, $df = 5$, $p = 0.006$), and no relationship between diameter and density ($R^2 = 0.140$, $F = 0.650$, $df = 5$, $p = 0.465$).

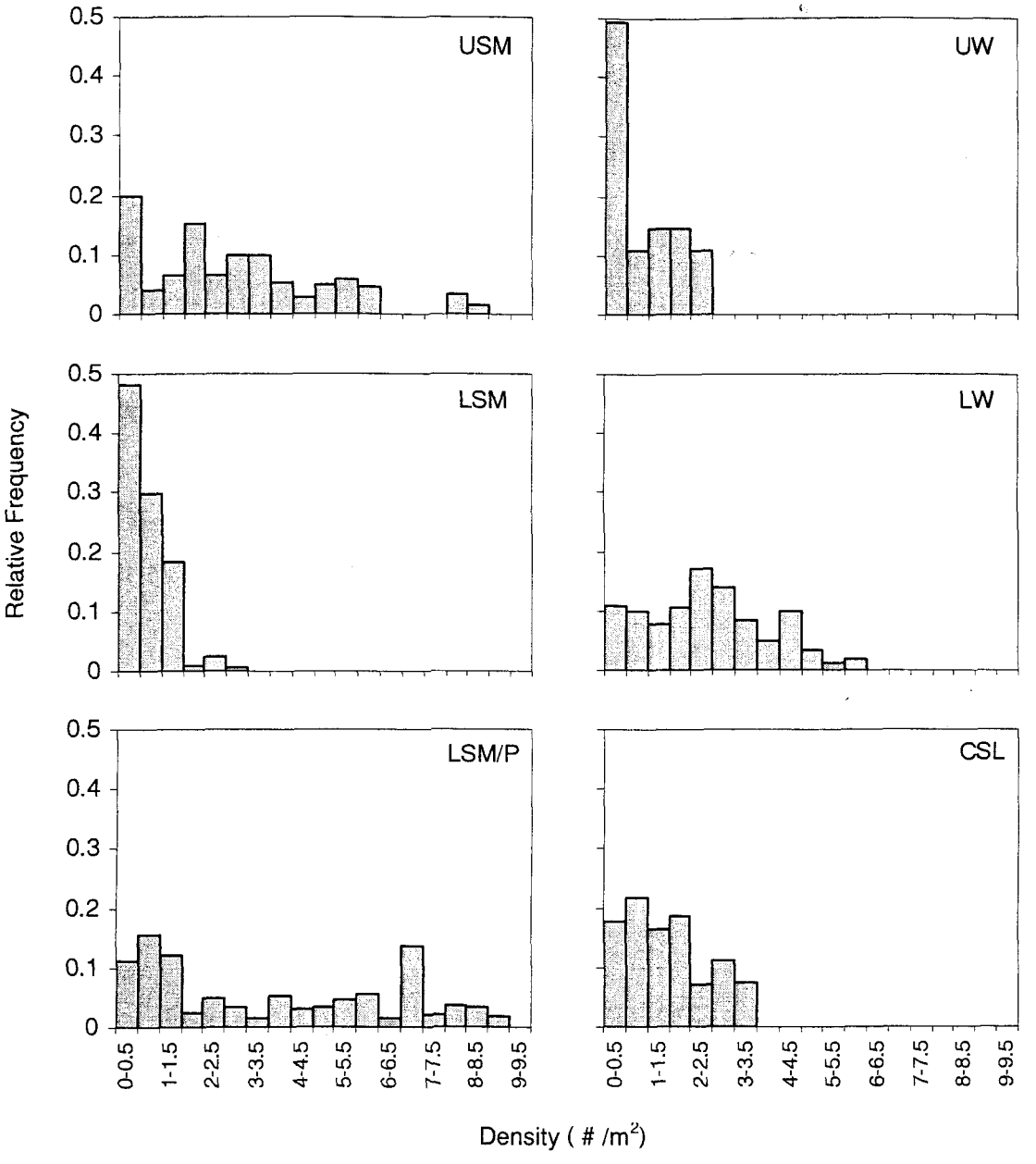

Figure 2-21. Relative frequency histograms of sapling density for each reach. LSM was similar to UW in that both reaches had mostly low sapling densities, with no high density classes. USM and LSM/P had a wide range of densities extending to some very high density classes, and LW and CSL had intermediate levels of density.

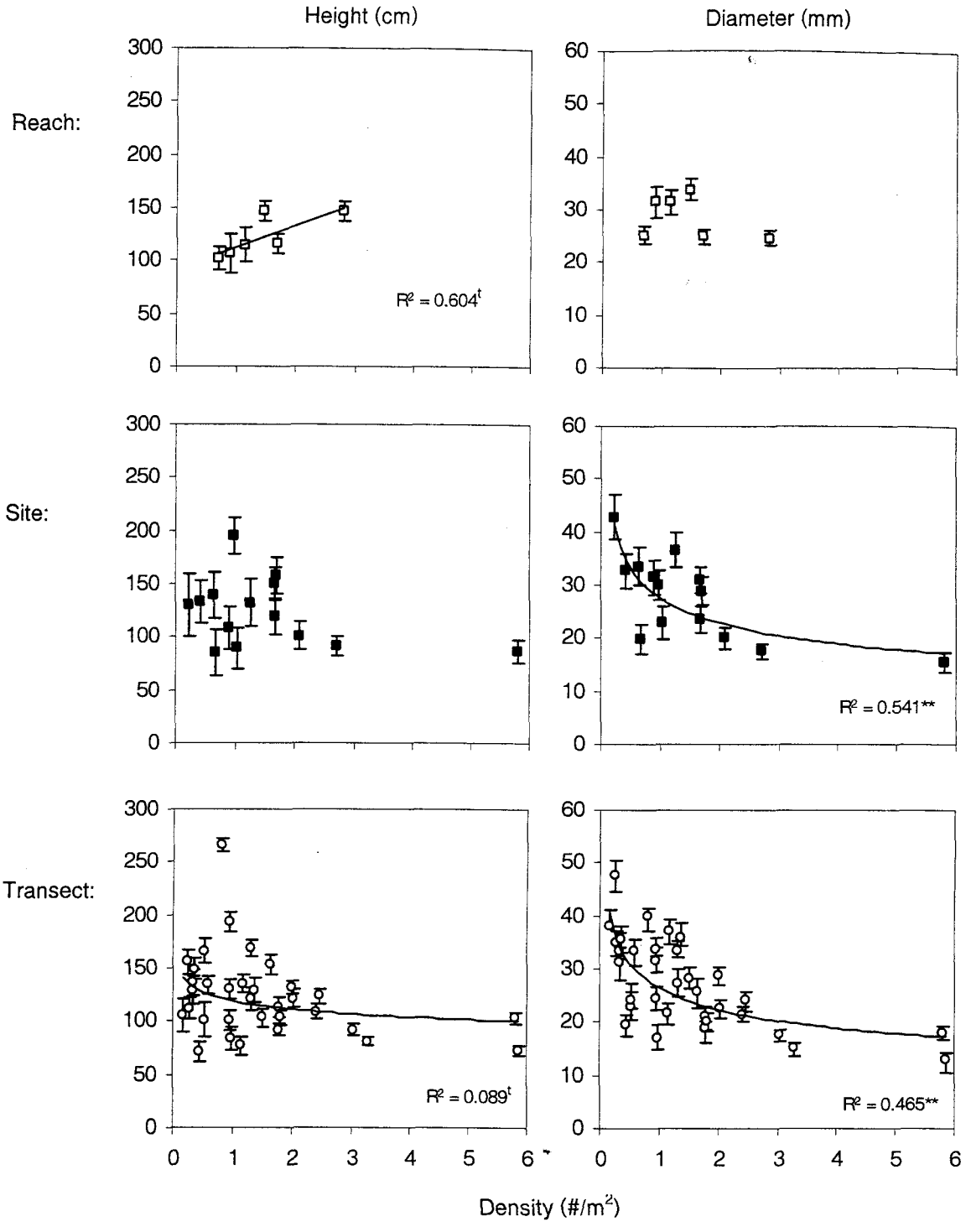

Figure 2-22. Relationships between mean sapling height and diameter (±SE) and mean density at the reach, site, and transect levels. (t: $p < 0.10$; **: $p < 0.001$)

Floodplain width

The sites along the more alluvial LSM/P reach had much larger floodplain widths than the partially confined sites on the LSM (Table 2-5). USM and LW sites were a mix of narrow and wide sites, the UW site was intermediate, and the CSL sites were relatively wide. CSL-2 was an anomaly because it had an extremely large standard deviation and the value measured does not seem to fit the appearance of the site, and consequently it was removed from subsequent analyses.

Table 2-5. Site floodplain widths as calculated in ArcMap. Floodplain width was defined as the distance perpendicular to the river between the first contour intervals on either side of the river.

Site ID	Floodplain width (m ± SD)
USM-1	110 ± 26
USM-2	693 ± 101
LSM-1	349 ± 36
LSM-2	374 ± 32
LSM-3	136 ± 11
LSM/P-1	949 ± 291
LSM/P-2	856 ± 38
LSM/P-3	721 ± 191
UW-1	452 ± 63
LW-1	153 ± 8
LW-2	379 ± 17
LW-3	725 ± 63
CSL-1	741 ± 47
CSL-2	1315 ± 551

No relationship was detected between adjusted site mean height or diameter and floodplain width (linear regression: height: $R^2 = 0.080$, $F = 0.956$, $p = 0.349$; diameter: $R^2 = 0.165$, $F = 2.17$, $p = 0.169$). There was a significant relationship between adjusted site

mean height-to-diameter ratio and floodplain width (Figure 2-23; linear regression: R^2 = 0.708, F = 26.7, df = 12, $p < 0.001$). The height versus floodplain width relationship, although not significant, was apparently positive, and the diameter versus width relationship was negative, resulting in the strong H/D versus floodplain width relationship. Therefore, sites with wider floodplains were more likely to have proportionally taller saplings.

Because geomorphic context was found to have an influence on cottonwood growth through differences in population density (Willms et al. 2006), and the sapling sizes in this study were found to be related to density (Figure 2-24), the relationship between sapling density and floodplain width was investigated. A significant positive relationship was detected (Figure 2-24). Because the H/D was significantly related to density at the site level, the relationship between H/D and floodplain width may be due to differences in density.

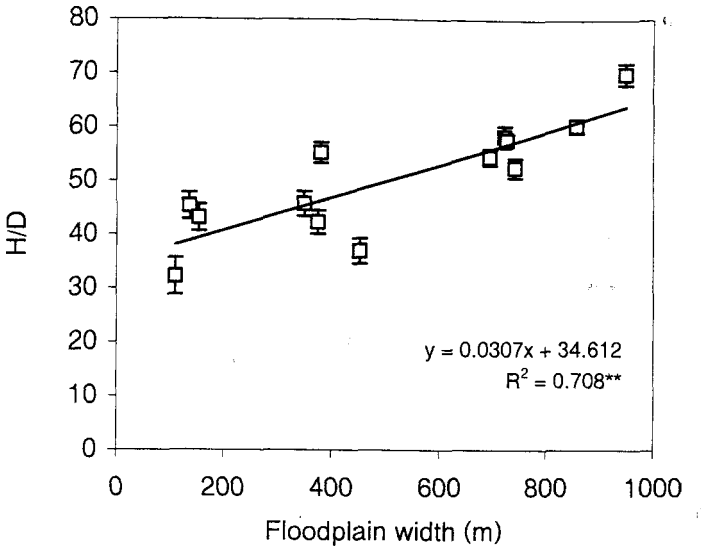

Figure 2-23. Positive relationship between site mean sapling height-to-diameter ratio (±SE) and floodplain width. (**: $p < 0.001$)

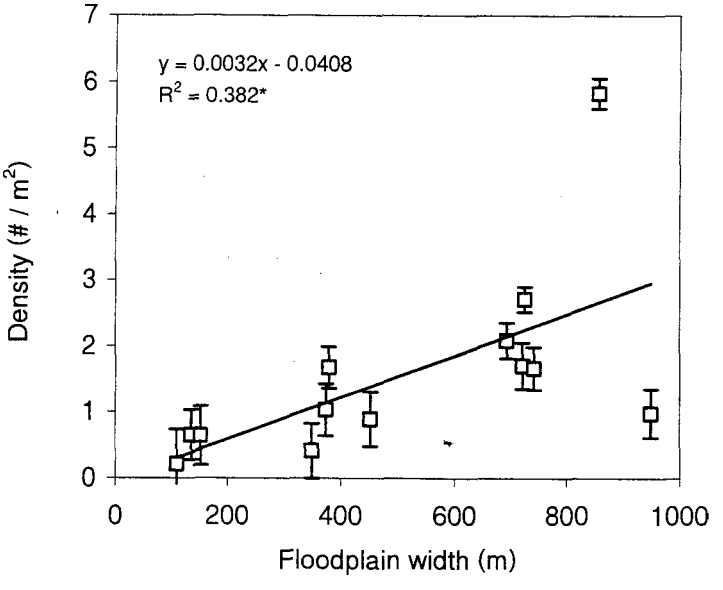

Figure 2-24. Positive relationship between adjusted mean site density (±SE) and floodplain width. Sites with wider floodplains were more likely to have increased sapling density (linear regression: $F = 6.81$, $df = 12$, $p = 0.024$).

2.4 DISCUSSION

Based on comparisons of cottonwood sapling extent and size, the outcome of instream flow restoration along the lower St. Mary River (LSM) was partially successful. Some characteristics indicated success, while others indicated failure. In terms of restoration success, the implementation of the 1991 minimum instream flow has resulted in greatly increased low flows (Table 2-4; Figure 2-2). Those low flows have probably helped to promote the observed seedling survival by increasing late summer water supply. Ten years after a seedling recruitment event that resulted from a 1-in-50-year flood, followed by managed ramping flows, saplings were present. In addition, mean sapling heights, both of excavated saplings established in 1995 to 1998, and saplings measured along the transects, were no different than mean sapling heights along other partly regulated and free-flowing river reaches (Figure 2-12, 2-14). Therefore, there was relatively consistent height growth of saplings across a range of streamflow conditions. Stem diameter may be a better indicator of growth in response to environmental variables, and it displayed a positive response to increasing river levels.

In terms of restoration failure, sapling extent along the LSM was limited. This was evident when compared to the extent of saplings along the lower Waterton River (Figure 2-4; Figure 2-8). However, the lower St. Mary River downstream of Pothole reach (LSM/P) had more extensive saplings even though it experienced very similar discharge to the LSM. The LSM, LSM/P, and LW study system provided similarities and contrasts in recent hydrologic regimes, seed source, geomorphic context, and management history that will provide the foundation for a discussion of conceptual ecological models of restoration pathways and processes.

2.4.1. Restoration models based on sapling extent data

Ecological restoration is considered the 'acid test' of ecological theory (Bradshaw 1987). We confirm that we understand the underlying ecosystem processes if we can restore ecosystem health after degradation or damage. Operating the St. Mary Dam to provide ramping flows, for example, was essentially a test of the Recruitment Box Model. Conversely, ecological restoration can advance ecological theory with restoration projects acting as experimental systems (SER 2004). Comparison of sapling and tree extent along the three regulated reaches in our study provides the basis for developing models of restoration trajectories.

Sarr (2002) has developed three relevant conceptual models of degradation and restoration trajectories for ecosystems that were degraded by livestock grazing and restored with livestock exclusion. Lake et al. (2007) adapted these models and discussed them in relation to any stressor. These models include the 'rubber band', 'broken leg' or 'hysteresis', and 'Humpty Dumpty' models. In the rubber band model, once the stressor is removed, a resilient ecosystem rapidly recovers, following a trajectory that is the reverse of degradation. This model might be especially applicable to sites in which factors supporting vegetation, such as soils and channel form, have not been degraded (Sarr 2002). The 'broken leg' model, renamed the 'hysteresis' model by Lake et al. (2007), describes recovery as a lengthy non-linear process that follows a different pathway than the reverse of degradation. Successional processes, assembly rules, lag times, and feedback mechanisms are all possible causes for the different trajectories (Sarr 2002, Lane et al. 2007). The 'Humpty Dumpty' model, describes a situation when the ecosystem cannot be restored after the stressor is removed. Explanations for non-

recovery include invasion by non-native species, or a change to an alternative stable state (Sarr 2002).

Two important terms for the discussion of ecological trajectories include the closely related terms of 'resilience' and 'resistance'. In the ecological literature, resilience has two meanings (Gunderson 2000). The term was first introduced in the 1970s, and was defined as "a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables" (Holling 1973). In contrast, it has also been used to describe "the speed at which a system returns to its former state after it has been perturbed or displaced from that state" (Suding et al. 2004). The latter definition is more consistent with the non-scientific definition of resilience: "the ability to bounce or spring back into shape, position, etc." ("Resilience" 1986), and the former definition has the same meaning as the term 'resistance', which is used in the restoration ecology literature. Resistance is defined as "the ability to maintain its structural and functional attributes in the face of stress and disturbance" by the Society for Ecological Restoration (SER 2004), and similarly by Suding et al. (2004). The definitions that I will use for the remainder of the study are as follows: resilience is the ability to readily recover, and resistance is the capacity to withstand stress.

A degradation trajectory is represented by a curve that shows that the ecosystem is able to withstand some stress and still remain healthy up until a certain threshold is reached, after which collapse occurs rapidly. Threshold responses have been observed in *Populus fremontii-Salix gooddingii* woodlands along the San Pedro River in Arizona in relation to surface flow permanence and depth to groundwater (Lite & Stromberg 2005).

While Sarr (2002) and Lake et al. (2007) show identical degradation pathways in all of their models, I propose that the degradation pathway varies with ecosystem sensitivity or resistance. This concept is similar to a model presented in Gordon et al. (2005) in which pathways of ecological disturbance differ between robust and fragile ecosystems. A prior study in the Oldman River Basin showed that riparian cottonwood recruitment and growth had different sensitivities to flow regulation dependent on their geomorphic context (Willms 2006).

The LSM and LSM/P differ in geomorphic context, but have experienced very similar levels of flow regulation over the same time period. The narrower floodplain of the LSM provides less habitat for riparian cottonwoods than the wider floodplains of the LSM/P, and this is shown in their historic cottonwood distribution. Prior to any river regulation, the LSM was sparsely wooded, while woodlands were more abundant downstream of Pothole Creek (Dawson 1884). Fewer trees and a more sensitive geomorphic context have resulted in a steeper degradation curve along the LSM than the LSM/P. Also, the impact of losing one tree in a system that only has one hundred trees is much greater than the impact of losing one tree in a system that has 10,000 trees. The steeper degradation pathway of LSM ultimately resulted in a more degraded state than the condition along the LSM/P (Figure 2-25).

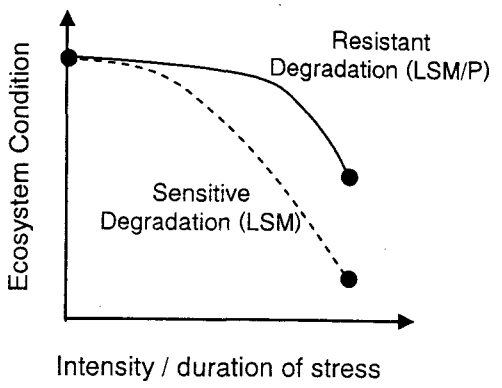

Figure 2-25. Differences in degradation trajectories based on ecosystem resistance or sensitivity. The lower St. Mary downstream of Pothole reach (LSM/P) was more resistant to stress than the sensitive lower St. Mary reach (LSM).

The starting point of degradation began with off-stream diversion in the late 1890s which resulted in streamflow decline, and cottonwoods became stressed. When the St. Mary Dam was constructed in 1951, streamflows remained low, stress persisted and the mature trees died. The combined effects of other stressors such as beaver activity (Figure 2-18), cattle grazing (Figure 2-6) and possibly herbicide spraying for the invasive leafy spurge, *Euphorbia esula* (J. Bevers, pers.com.) may have also contributed to declines. A century of river regulation that prevented recruitment means that an entire population of cottonwoods could have died, and along the LSM most of the population has disappeared, with the exception of 13 out of 57 locations where sparse remnant mature trees persist. The alluvial LSM/P was more resistant to the stress, and so recruitment was not entirely excluded and population degradation has not been nearly as severe, with 10 out of 13 locations supporting mature trees.

While summer streamflows along the lower St. Mary River were reduced to 30% of the natural flow prior to the implementation of the minimum flow criterion, streamflows along the lower Waterton River were reduced to only 44%. In addition, the LW experienced unfavourable river regulation for about 30 years, a period much shorter than the typical lifespan of a cottonwood tree, and as such, all 52 locations observed in 2007 through aerial photographs still supported mature trees. Drought-stressed trees and some dead trees were observed in the late 1980s (S. Rood, pers. comm.), but decline in areal extent as measured from air photographs was not severe (Rood et al. 1995). Therefore, the LW had not progressed very far along the degradation trajectory, and had not yet reached the threshold of collapse prior to the implementation of the minimum criterion.

With only small to moderate degradation, recovery along the LW can be represented by a reversal in the degradation pathway, and thus follows the rubber band model. We consider that the LW currently supports a healthy riparian ecosystem as there are extensive saplings from many size classes that cover 82% of the locations surveyed with photographs from the low-elevation flight. Mature trees, and thus seed sources, were still plentiful, and therefore, restoration was able to occur to a state where the ecosystem can be considered healthy.

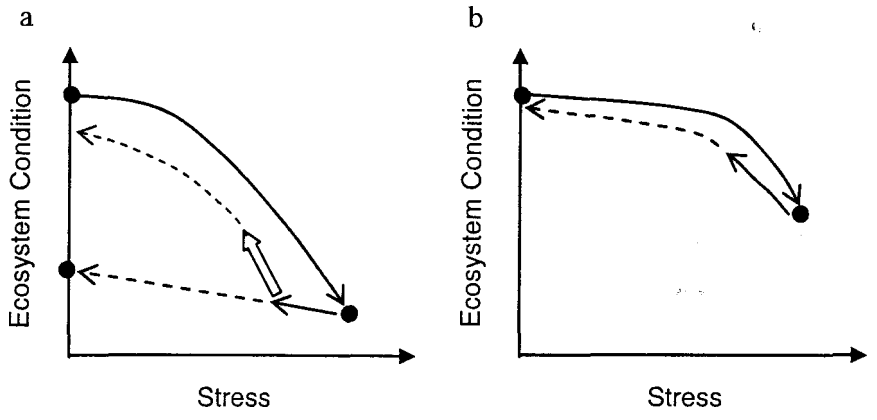

Figure 2-26. Conceptual models of cottonwood restoration trajectories along the lower St. Mary (a), and lower St. Mary downstream of Pothole, or lower Waterton (b). The LSM follows the Humpty Dumpty model whereby instream flow restoration reduced stress and ecosystem condition improved marginally (solid arrow), but lack of seed source means that given time (dashed arrow), the ecosystem would never return to the desired condition. The existing cottonwood population may expand slowly resulting in further improvements of ecosystem condition. With increased management effort in the form of seed addition (block arrow), the ecosystem could be put on a different path of restoration that could improve the cottonwood condition. The LSM/P and LW follow the rubber band model in which less severe degradation enables rapid improvement of ecosystem condition following stress reduction. Adapted from Sarr (2002) and Suding et al. (2004).

However, can the healthy state along the lower Waterton be attributed to flow restoration? Even with the implementation of the minimum flow criterion, the proportion of natural flow during the growing season actually decreased after 1991, and late summer flows only increased from 44% of natural flows to 47%. Further, ramping flows were only moderately favourable since their implementation in 1995 (see Chapter 3). Low flow indicators (nQ_d<2.27 and $7Q_{10}$) showed improvements in low flows (Table 2-4; Figure 2-2), but on a monthly time scale, these differences were minor or negligible. Differences in the health of the cottonwood population could also be influenced by long-

term climate regimes like the Pacific Decadal Oscillation (PDO). There was a warming phase from 1977 to the mid-1990s, and currently there is a possibility of a shift into a cool phase (Mantua and Hare 2002).

Restoration along the LSM/P may also be following the rubber band model as the return of ramping flows and summer flows that increased from 31% to 56% of natural flows promoted seedling recruitment such that saplings now occur at 62% of locations surveyed, and it is likely that saplings would have occurred more extensively if cattle had been excluded (Figure 2-6).

Like the degradation pathways, restoration pathways between LSM/P and LSM also differ with the LSM following the Humpty Dumpty model (Figure 2-26). These reaches currently have similar flow regimes, and while hydrology is considered a key variable structuring riverine ecosystems (NRC 2002; Annear et al. 2004), the reaches differ in the extent of the seed source. Historically sparse woodlands, coupled with more severe degradation, have left the LSM with few mature trees that can act as seed sources. With little seed source, recruitment has been limited, and only 8 out of 57 locations supported saplings in 2007. Proximity to seed source has been recognized as an important factor in the initial establishment of seedlings along the St. Mary River (Rood & Mahoney 2000). The reaches that had more plentiful seed source had higher proportions of locations with saplings, and therefore, lack of seed source is at least a partial cause for the limited restoration along the LSM (Figure 2-4).

Seeds are not the only source of propagules for cottonwoods. Cottonwoods can also reproduce asexually through root suckering or branch fragmentation, and these forms of reproduction are important determinants of woodland structure (Gom & Rood 1999).

With few mature trees as sources for asexual propagules, these forms of reproduction would also be minimal, further limiting population growth and expansion.

If seed source had been more plentiful, restoration along the LSM would probably have been more successful. When considering only locations that supported mature trees, and thus a seed source, there was no difference among the reaches in the proportion of locations that supported saplings (Figure 2-4). Sites that supported mature trees might also be sites that are naturally more favourable for saplings, and thus the hypothesis that seed source is the limiting factor might be confounded by environmental or site favourability.

In general, seed banks and propagule dispersal are important considerations in restoration ecology (Suding et al. 2004; Young et al. 2005). The 'efficient-community' hypothesis of recolonization proposed by Galatowitsch and vanderValk (1996), or passive restoration approaches (Kaufmann et al. 1997), rely on the presence of a seed bank or seed dispersal for recolonization. Cottonwood restoration is somewhat unusual in this respect, since cottonwood seeds have a very short period of viability (Braatne et al. 1996), and do not form seed banks. The cottony seeds do have the potential for long-range dispersal via wind and water, but no studies have formally documented dispersal distances. Some sources indicate that long distance dispersal might not be prevalent as large numbers of seeds fall within a stand, and most of the seeds are deposited within a few hundred meters of the parental tree (Zasada & Phipps 1990; Braatne et al. 1996). Hydrochory also has the potential for to disperse seeds long distances, but can only supply seeds to downstream sites. Dams such as the St. Mary Dam interrupt the longitudinal connectivity which would have naturally brought seeds from the upper St.

Mary to the downstream reach. Seed concentrations in the water column along downstream fragmented river reaches can be 70-94% lower than in the upstream, free-flowing reach (Merritt & Wohl 2006). Long-distance dispersal may have resulted in two sapling patches along the lower St. Mary at locations with no nearby mature trees. Or else, mature trees may have been present at those locations when the seedlings established.

A mature cottonwood may produce 25 million seeds (Van Haverbeke 1990), and thus cottonwoods are not normally limited by fecundity. However, out of the millions of seeds produced, only a portion would reach suitable germination sites. Out of those, only a portion of seeds would germinate, and then only a portion of those seedlings would survive. Therefore, although the few trees at some of the LSM locations may produce many seeds, the abundance of those seeds might not be sufficient to overcome the very low odds of successful recruitment.

In a study of prairie pothole wetland restoration in northern Idaho, the restoration approach was natural revegetation after restoration of hydrology (Galatowitsch & vanderValk 1996), similar to the St. Mary River. However, the restored wetlands had fewer plant species and fewer species represented in the seed bank than in natural sites (Galatowitsch & vanderValk 1996). The authors concluded that passive restoration cannot be the only method undertaken in this type of habitat. A study on riparian willow restoration in Yellowstone National Park also found that simply decreasing the level of the stressor – large elk populations due to predator extirpation – would not bring back the desired willow-beaver state, and that further management effort to restore the hydrologic regime would be needed (Wolf et al. 2007). In these cases, and in the St. Mary case, the

ecosystem cannot cross a threshold or revert to the desired alternative stable state without increased management effort (Hobbs & Norton 1996; Suding et al. 2004). Given only time, the ecosystem condition may improve somewhat (Figure 2-28a), but if the threshold is not overcome, than the desired state cannot be reached.

For the St. Mary River, the additional management effort might take the form of seed addition. While restoration ecologists strive for landscape-scale or systemic restoration methods such as a return to a more natural flow regime, for the St. Mary River and other severely altered systems, removing the stressor by returning the necessary hydrology is not sufficient.

2.4.2 The nature of variation in sapling growth

Among the different analyses that I performed on cottonwood sapling size, more variation occurred on smaller spatial scales than at the reach level. I found differences in sapling height distributions between sites within a reach, and between transects within a site (Table D-2; Figure 2-13). There were also large size ranges for saplings that were the same age along the same reach (Figures 2-10, 2-11), and there were no differences in heights or diameters among the reaches for saplings that established from 1995 to1998, with the exception of the smaller UW saplings. These saplings could have been smaller due to chance as the sample size was smaller than along the other reaches, or false rings may have indicated the saplings were older than they actually were (Everitt 1968), as these saplings were harvested from a patch that predominantly established in 1999.

In addition to the other analyses that indicated greater smaller-scale variation, the nested analyses of variance showed more variation in height and diameter at the transect

and site level than at the reach level (Figures 2-14, 2-15). Also, the variation that was not accounted for by the nested models would include variation found within a transect, a spatial scale smaller than the transect level.

The importance of spatial scale has been recognized in riparian ecology (Baker 1989; Bendix 1994; Dixon et al. 2002) and forms the basis for the field of landscape ecology (Naiman et al. 2005). I expected reach level differences to be greatest, reflecting differences in hydrologic regime since this is considered the key variable (NRC 2002; Annear et al. 2004), and because cottonwoods are highly sensitive to changes in water supply (Stromberg & Patten 1991; Willms et al. 1998; Scott et al. 1999; DiSalvo & Hart 2002; Horton et al. 2001). However, less sensitive complacent growth can occur when water levels are sufficient (Willms 2005; Braatne et al. 2007).

There are several possible explanations why lower variation occurred at the reach level. First, although the hydrological variables such as the river stage above base stage in spring and in summer differed only at the reach scale, streamflow interacts with each site differently, perhaps dependent upon factors such as substrate, floodplain elevation, geomorphic context, and vegetation. This relates to hierarchy theory which states that processes at one scale exercise at least partial control over processes at smaller scales (Bendix 1994). For example, field observations suggested that substrates at some sites were highly variable, with patches of silt adjacent to cobble. The silt patches may wick-up moisture more readily from the water table creating a large capillary fringe that provides more moisture to a sapling than the adjacent cobble patch. Therefore, the discharge that is consistent along the reach provides very different levels of moisture availability dependent on finer-scale processes of substrate texture distribution.

A second reason for complacent growth across the reaches may be due to adaptation of root morphology. The distribution of cottonwood roots is apparently adapted to maximize uptake from predictable water sources (Williams & Cooper 2005; Andersen 2005). Lower water levels, such as along the lower St. Mary River may result in saplings that invest more energy in deeper roots. Along reaches with higher water levels, more investment may occur in lateral roots.

The high variability in sapling growth also provides support for, and may reflect the high spatial heterogeneity that characterizes riparian zones (Naiman et al. 2005). Moreover, differences in microhabitats, defined as the part of the general habitat used by an organism (Smith 1996), may be accentuated because of the inherent pattern in the cottonwood growth curve. A typical cottonwood growth curve, as presented in Willms et al. (2006) and Berg et al. (2007), commences with a slow growing 'establishment' phase that may last up to seven years. Once the seedling root system is established, energy is redirected into the shoot system and the juvenile grows rapidly. This growth levels off as the tree reaches about its twentieth year. If a sapling is located in a favourable microhabitat it may be released from the establishment phase earlier than the other members of its cohort that are located in less favourable microhabitats. A transition from the slow-growing establishment phase to the 'growth acceleration' phase could emphasize differences in sizes of saplings of the same age.

Factors such as floodplain elevation and density are part of a sapling's microhabitat, but the effect of elevation on sapling size was not detected. The lack of an elevational effect could be due to: the narrow range of elevations in which saplings

occurred, confounding factors such as age variation, or a tradeoff between living at low versus high positions on the floodplain.

Most saplings were found growing within a relatively narrow range of elevations between 0.75 and 2.25 m above the river stage at baseflow (Figure 2-20). They occurred within a zone that provides further support for the elevation parameters of 0.6 to 2 m that were initially utilized in the Recruitment Box Model (Mahoney & Rood 1998). However, following establishment, sediment deposition and scour could have altered the floodplain surface elevation.

It was expected that sapling size should increase with elevation, due to the formation of arcuate bands with older saplings further back on the floodplain. However, this type of establishment mainly occurs on point bar formations in reaches with progressive meander migration (Scott et al. 1996). Many of the landforms that I sampled from were attached islands or other types of bars, or in reaches where active meandering is confined by steep valley walls. In addition, saplings from multiple establishment years were often found intermixed within sapling zones (Figure C-1).

A tradeoff between low and high elevation habitats may occur because the roots of low elevation seedlings would reach water more quickly, and subsequently more energy could be put into shoot growth. But more prolonged inundation, or more severe scour and deposition could reduce shoot growth as well as result in mortality. Higher elevation saplings would have to expend more energy in root growth to reach the water table and would be more prone to drought, but they would be more protected from geomorphic processes (Dixon et al. 2002; Polzin & Rood 2006).

Although floodplain elevation was a microhabitat factor that did not affect sapling size, density was a factor that did. Sapling heights were not related to density at the transect and site levels (Figure 2-22), and this is consistent with a prior study (Andersen 2005). In contrast, Taylor et al. (2006) found that saplings growing in less dense plots increased quickly in height. At the reach level, there was a positive trend between height and density, but because there were no height-density relationships at the smaller spatial scales, this weak relationship may be due to chance rather than reflecting causal associations.

Transects and sites that had lower densities supported saplings with larger mean diameters (Figure 2-22). In low density zones, there is less competition and potentially more resources available for greater sapling growth. This diameter-density relationship is consistent with the results of another study of ten-year old *Populus deltoides* saplings (Taylor et al. 2006).

For diameter at the reach level, the lack of relationship with density may be due to confounding effects from the diameter versus streamflow relationship. Given the decreasing diameter versus density relationships apparent at the smaller spatial levels, and the low density of saplings along the LSM, it would be expected that those saplings would have a larger diameter than the other reaches. If the LSM diameter had been larger, it appears from Figure 2-22, that there would have been a decreasing relationship between diameter versus density. However, the diameter versus streamflow relationships were significant, indicating that at the reach level streamflow has more effect on diameter than density. What may be happening is that saplings along the lower St. Mary, although not significantly thinner than saplings along the LSM/P, LW, and UW, should be larger

in diameter due to their low density, but their growth was being inhibited by low streamflows. Low densities imply a lack of competition, but the density at which competition occurs can vary due to differences in resource availability. If resources are limiting, competition will occur at lower densities than when resources are plentiful. Lower leaf litter-derived nutrients such as nitrogen from the few remaining mature trees (Naiman et al. 2005), as well as lower water levels may be contributing to lower resource levels along the LSM.

The low density along the LSM may also be due in part to the limited seed supply, increased mortality of established seedlings and fewer years in which favourable hydrologic conditions promoted seedling establishment. In confined reaches, it is often only large floods that produce the conditions necessary for successful seedling recruitment, and thus extensive recruitment may naturally occur only once in several decades (Scott et al. 1996).

While density did not significantly account for variation in reach mean sapling diameter, the average spring water level and the average summer water level accounted for most of the variation (Figure 2-17). However, in this study the differences among the reaches only explained 3% of the overall variation in diameter. An increase in spring stage from 0.2 to 0.6 m, about the magnitude of difference between the regulated reaches and the free-flowing and moderately regulated USM, would increase diameter about 9 mm over all the years of growth, an increase of 37%. An increase in summer stage from 0.1 to 0.3 m, about the magnitude of difference between the regulated and the free-flowing reaches, would increase diameter about 7 mm, an increase of 28%. Either flow variable can account for much of the variation because spring and summer water levels

are strongly positively correlated. However, this does not reveal whether spring flows or summer flows are more important for radial growth. Studies of annual branch or stem increments reveal that for stem elongation, spring flows are more important (Willms et al. 1998; Andersen 2005), as branch elongation occurs from early in the growing season to mid-July in this region (Phelan 2007). The seasonal pattern of radial growth is unknown but may occur over a longer period of time with limited growth in the early and later parts of the growing season, and faster growth rate in the middle (Kort 2005). Streamflows extending into the late summer are probably more important for diameter growth than height growth.

Whereas streamflow is an abiotic factor affecting saplings, beaver-browsing is a biotic factor. Beaver are sometimes considered beneficial and necessary for the maintenance of a riparian ecosystem. For example, in Yellowstone National Park they play a key role in a hydrologic regime that is necessary for riparian willows (Wolf et al. 2007). However, beavers can severely damage other riparian plants. Barnes (1985) found that cottonwoods at his study site along the Chippewa River in Wisconsin were so severely damaged by recurring beaver cutting that their future was uncertain. Similar to Barnes' (1985) site, beaver-browsing along the LSM resulted in many cottonwoods with a shrubby growth form (Figure 2-21). In my study, beaver-browsing was more prevalent along the LSM than the other reaches (Figure 2-18). Comparing the extent of trees and saplings among the three regulated reaches (Figure 2-4), and the density of patches that were present (Figure 2-21) shows that the woodland resource available for beavers along the LSM was sparse.

Beavers are a natural disturbance along rivers in the Oldman Basin, but when their effect is combined with the effects of river regulation, the cumulative impact may be greater than the sum of the individual influences. Studying the effects of multiple stressors is one of the foremost challenges in ecosystem science and especially important for restoration and management (Breitburg et al. 1998). A 'stressor' may be defined as a perturbation applied to a system at an excessive level, whether it is foreign or natural to that system (Barrett et al. 1976). Along the LSM, river regulation represents an anthropogenic stressor, beaver-browsing represents a natural stressor, and their cumulative effect on the cottonwood population has been severe. River damming and off-stream diversion have reduced streamflows, causing a decline in the cottonwood population and many of the remaining trees have been harvested by beavers. Beavers by themselves might not impose such a serious threat, but their actions on a population that has already been stressed are proportionally greater than along a healthy river with numerous trees. Breck et al. (2003) reported the same phenomenon along the regulated Green River in Colorado whereby river regulation magnified the impacts of beaver on cottonwoods due to decreased spatial distribution and density.

2.4.3 Conclusions

Assessing and isolating the effects of restoration is challenging (Klein et al. 2007). In our study, non-linear degradation and restoration trajectories, thresholds, multiple impacts, and extensive natural variation at small spatial scales added to the complexity. The contrast in the ecological condition along two adjacent regulated rivers -

the lower St. Mary and the lower Waterton - was striking. The lower St. Mary supported few sapling patches and the lower Waterton supported abundant saplings with a range of vertical structure. The differences between these two rivers highlighted different models of degradation and restoration trajectories. The lower Waterton River has been subjected to 30 years of regulation by the Waterton Dam. Vigor of mature trees suffered, but their areal extent had not declined severely (Rood et al. 1995), and some recruitment persisted. The degradation pathway had not yet reached the threshold of collapse, but ecosystem resistance may have been compromised. If further diversions, rather than instream flow restoration and a moratorium on additional water allocations, were allowed to occur, ecosystem condition may have quickly advanced along the degradation pathway. Minimum flows have been successfully implemented, reducing the number of low flow days, and ramping flows following the 1995 flood led to a successful recruitment event. Thus, instead of progressing further along the degradation pathway, cottonwood condition has moved along a trajectory of rapid restoration. We now consider the lower Waterton River as supporting a healthy population of riparian cottonwoods.

In contrast, the lower St. Mary reach has been subjected to 100 years of river regulation, resulting in collapse of the cottonwood population. Flows have been more severely reduced within a geomorphic context and historic cottonwood abundance that was more sensitive to change. The threshold of collapse along the degradation pathway was surpassed. When increased minimum flows and ramping flows were implemented, the ecosystem commenced along its restoration pathway. Seedlings successfully established and survived to the sapling stage, but their distribution is extremely limited. The landscape-scale, or systemic approach of restoring elements of the flow regime is the

approach recommended by restoration ecologists. However, with a lack of propagules and limited long-distance dispersal, the restoration pathway will never lead back to the desired ecosystem condition along the LSM. Increased management effort using more active and site-specific restoration measures such as seed addition may need to be undertaken. Future studies along the St. Mary River could thus include seed addition experiments.

In the patches of saplings that were present along the lower St. Mary reach, height growth was not significantly different from that of other regional river reaches. Diameter growth was significantly different among the reaches, and differences in river levels explained most of that variation. However, much more variation in diameter occurred at spatial scales smaller that the reach level. While hydrologic variables differed at the reach level and hydrology is considered a 'master variable' (NRC 2002), more variation occurred among the sites and transects. Microhabitat factors or interaction of hydrology with those microhabitat factors may have produced the variation at the smaller spatial scale. In addition, root adaptation to different hydrologic regimes may have limited the variation due to reach level hydrological variables.

In conclusion, restoration has commenced along the lower St. Mary as indicated by sapling survival at some locations and consistent sapling growth at those locations in comparison to other regional river reaches. However, for more widespread reach-level benefits, increased intervention, and particularly seed addition, may be needed to overcome the threshold along the restoration trajectory.

CHAPTER 3
Unfavourable Streamflows for Cottonwood Recruitment along Regulated Rivers - a Model of Recruitment Likelihood

3.1 INTRODUCTION

Regulation of streamflow by damming and dewatering is responsible for unsuccessful cottonwood reproduction along many rivers, and has resulted in decadent populations with missing younger age classes (Howe and Knopf 1991, Johnson 1992, Rood et al. 1995). The dependency of seedling recruitment on specific streamflow patterns is now well understood and is summarized by the Recruitment Box Model (Mahoney and Rood 1998). This model integrates the seedlings' ecophysiological constraints with the timing and magnitude of the spring peak and the post-flood rate of recession (Figure 3-1).

First, a flood event is needed to scour existing vegetation and deposit sediments, creating moist and barren substrates. These suitable nursery sites are required by cottonwood seedlings, which have limited nutrient reserves and are shade-intolerant (Braatne et al. 1996). Second, the post-flood recession that exposes the new seed beds must coincide with the timing of seed dispersal, which occurs from June to July, usually in coordination with the spring flood peak (Stella et al. 2006).

It has been observed that seedling survival generally occurs in an elevational band of 0.6 to 2 m above the base stage (late summer, low water elevation) with these limits being defined by loss through scour or deposition in lower elevations, and drought-induced mortality in higher elevations. Thus, moist seed beds between these elevations should be exposed during the period of seed dispersal. By placing seedlings in rhizopods, a type of growth chamber that allowed for the controlled manipulation of water level, a

decline of about 2.5 cm/day was found to be beneficial (Mahoney & Rood 1991). With this gradual recession, seedling root growth is able to maintain contact with the declining moisture zone, preventing drought. By the end of the growing season, at low water elevation, the combination of 60-90 cm of seedling root growth, 30-60 cm of capillary fringe above the saturated water table, and residual soil moisture should be able to supply seedlings with sufficient water.

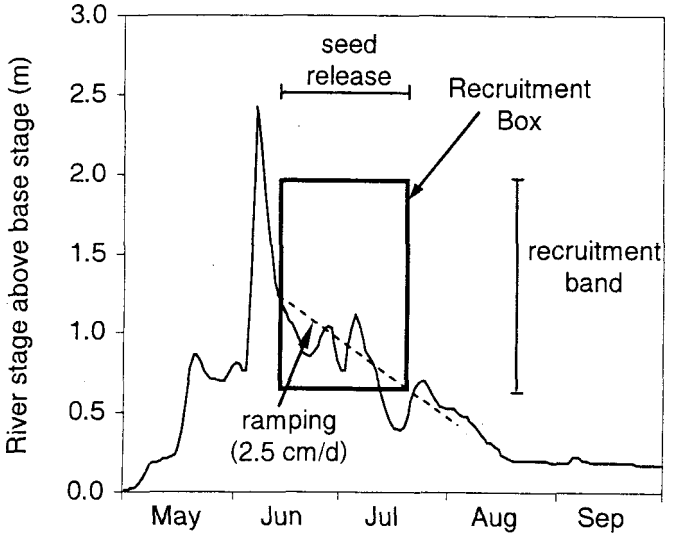

Figure 3-1. The Recruitment Box Model. The recruitment box defines an area in time (seed release) and space (recruitment band) in which cottonwood seedlings are likely to become established if streamflow patterns (hydrograph peak and ramping) are favourable. The stage hydrograph shown is from the lower St. Mary River (05AE006) in 1995, and represents a 3-day moving average. (Modified from Mahoney and Rood 1998).

The underlying concepts of the Recruitment Box and especially ramping flows were applied to dam operations along several rivers and were successful at promoting seedling recruitment, providing support for the validity of the model (Kalischuk et al.

1998; Rood and Mahoney 2000; Rood et al. 2003a). These studies show that the Recruitment Box Model is a valuable tool for river conservation and restoration. However, this tool can be further developed and quantified with an analysis similar to the one we developed for the Yakima River in Braatne et al. (2007), helping us to further understand streamflow requirements for recruitment.

By analyzing streamflow data from the last decade (1995-2005) on each of the river reaches studied in the previous chapter, not only can streamflow requirements for recruitment be further understood through the further quantification of a conceptual model, but recent flow management practices can be assessed, and the likelihood of recruitment along free-flowing and regulated reaches can be compared. The model can be somewhat tested by comparing the model outcome with the establishment years obtained from the excavated saplings analyzed in Chapter 2.

3.2 METHODS

This analysis involved a three step process of: developing model criteria, calculating values from historic data, and rating the historic values using the criteria. For the first step, criteria were developed based on the Recruitment Box Model and other hydrological requirements established in prior studies (Mahoney & Rood 1998 and references therein). Developing the model involved determining which components of the annual hydrograph are important for seedling establishment, and what values of those components are necessary.

The second step involved obtaining and calculating the historic values from the streamflow record. Streamflow data were obtained from the Water Survey of Canada's

hydrological database (HYDAT) for each of the river reaches analyzed (Table 3-1). Because stage data were needed for certain criteria, discharge data were converted to stage data based on the stage-discharge rating curves provided by Alberta Environment. The version of the rating curve used is also presented in Table 3-1. To convert discharge values positioned between the points provided by the rating curve, linear interpolation, a method also used by Alberta Environment, was used. The 'forecast' function in Excel was useful for this task.

Table 3-1. Water Survey of Canada's streamflow gauges used in the recruitment analysis with associated stage-rating curves provided by Alberta Environment.

Gauge Name	Reach	Gauge #	Stage-rating curve #
St. Mary River at International Boundary	USM	05AE027	29
St. Mary River near Lethbridge	LSM/P	05AE006	12
Waterton River near Waterton Park	UW	05AD003	16
Waterton River near Glenwood	LW	05AD028	15
Castle River near Beaver Mines	CSL	05AA022	16

The third step involved rating the historic values according to the set criteria. An overall recruitment score considering each of the criteria was then calculated, and indicated the likelihood of successful seedling establishment in any particular year.

A final step involved verifying the model outcome by comparing the generated recruitment score to the observed years of establishment obtained from the sapling excavations presented in Chapter 2.

3.3 RESULTS

3.3.1 Development of model criteria

Five hydrograph components provided the foundation for the recruitment model: the magnitude of the disturbance flow, the height (= stage) of the spring peak above baseflow, the timing of the spring peak, the post-spring peak ramping rate, and the late summer stage. The criteria and ratings given to each of the components are described below and are summarized in Table 3-2.

Disturbance flow (Q_{max}): Highly favourable floods had a return interval of ten years or greater (Scott et al. 1997; Stromberg 1998; Auble and Scott 1998; Whithed et al. 2007), and the barren substrates produced by those large floods were considered to have lasted for three years before vegetation encroachment. Moderately favourable floods had a recurrence interval of five to ten years, with effects lasting up to two years later. The difference between this analysis and the Recruitment Box Model was that in this model the disturbance flow, which creates barren nursery sites, was uncoupled from the spring peak flow that creates moist nursery sites. A disturbance flow, as the precursor of recruitment, was considered an important aspect in the recruitment process and was thus given a rating out of two.

Spring peak timing: Favourable timing occurred from the end of May to mid June (May 24 – June 15), coinciding with the period preceding potential seed release, which generally occurs from June to July in this region, depending on species and location (DeBell 1990, Van Haverbeke 1990). Because spring peak timing is fairly consistent

throughout the period of record (Figure 1-4), this criteria was given a rating out of one. It was included in the model, because regulation has the potential to alter this naturally consistent phenomenon.

Height of spring peak above base stage (H_{sp}): The spring peak occurring in the year of interest had to reach a stage of at least 0.6 m above baseflow, or suitable germination sites were considered unavailable. As outlined in the Recruitment Box Model, this lower elevation occurs because of loss from scour below this point. The higher the floodplain surface that is wetted (up until 2 m) the more area that would be available for seed germination. Therefore, a minimum rating out of three would be given if only part of the recruitment band was wetted and a maximum rating would be given if the entire recruitment band up to 2 m was moistened.

Ramping: A mortality coefficient (M) to describe ramping rate favourability was developed in Braatne et al. (2007), with lower values representing more favourable post-flood stage declines. Mortality coefficients for all reaches in this analysis were lower than the coefficients for the Yakima River possibly due to differences in streamflow supply, the different character of the regions, or different stream sizes. Therefore, the value of the coefficient that would describe a favourable ramping rate was altered in this analysis from < 20 on the Yakima, to < 5 in the Oldman Basin. The $M < 5$ criterion was selected because it was known to represent a favourable ramping rate for this region. Ramping rates have been determined to be an important factor for seedling survival, but the variation in capillary fringe depth due to differences in substrate, and the possibility

of moisture sources other than streamflow (i.e. precipitation) that could also prevent drought-induced mortality, mean that it could be possible for seedlings to survive without favourable ramping. For example, in 1995 along the LSM/P reach, successful seedling recruitment occurred (Rood & Mahoney 2000) despite the unfavourable mortality coefficient (M = 21). The summer of 1995 was relatively wet with seven days of precipitation greater than 2 mm in July (total July precipitation = 59.6 mm) that would not only supply moisture, but also reduce vapor pressure demand. For this reason, the ramping rate was not given a rating as high as H_{sp}, but rather given a rating out of two.

Late summer stage (H_{aug}): Post-colonization drought survival in the first year of the seedling's life was considered by rating mean August discharge. Various methods of determining an appropriate August discharge were considered. One method was based on a set of assumptions to determine what the water table would need to be to support a seedling based on root growth of 1 cm/d, a specific number of days for root growth to occur that is in turn based on water table declines, and a capillary fringe of 40 cm. The values of 1 cm/d and 40 cm were taken from the literature (Mahoney and Rood 1998). If a seedling established at the maximum elevation of 2 m on June 15, by August 15 (average of August), after 65 days of root growth the roots would be at 1.35 m above base stage. The capillary fringe of 0.40 m would allow the water table to be at 0.95 m. If a seedling established at the minimum elevation of 0.6 m at a later date (July 15) due to that surface just recently becoming available, and roots grew for 30 days, then the roots would be at 0.30 m above base stage. With a capillary fringe of 0.40 m, the water level could be 0.10 m *below* base stage. This method was not used because it resulted in a

large range of possible values, it would have been a difficult analysis to carry out for each individual year with the variation that occurs in the width of the elevational recruitment band, and in the timing, and it is based on assumptions of root growth and fringe depth that could vary with site and elevational position.

The method used was based on a simpler set of assumptions and produced a universal value. If a seedling established at 0.6 m at around mid-July, by August 1 an optimal and proven ramping rate of 2.5 cm/d would produce a water level of 0.225 m above base stage [0.6m – 15d*2.5cm/d]. By the end of the summer, flows are naturally around baseflow, and so on average August flows should be 0.11 m above base stage [(0.225 + 0) / 2]. Because many years met the 0.11 m criterion, because precipitation can also influence drought survival, and because seedlings may not rely on saturated soil water in their first few years (Cooper et al. 1999), this component was only given a rating out of one (/ +).

3.3.2 Calculation of historic values

Annual peak flow (Q_{max}): Flood recurrence intervals were calculated using streamflow data for the periods of record and by using 2-Parameter Log Normal distributions in DISTRIB 2.20 (Eaglin 1999). Distribution fit was assessed with DISTRIB's K-S statistic, which provided a measure of the difference between the actual and the hypothetical distribution. When considering all the reaches, the 2-Parameter Log Normal distribution provided the best fit (better than the typical Log Pearson Type III, although results were very similar). The exception was the Upper Waterton reach that had a compound function with a break at about the 20-yr return interval probably due to

attenuation by the large Waterton Lakes. The two functions for the upper Waterton were calculated in Excel using regression.

Height of spring peak above base (H_{sp}): To find the maximum spring (May-July) stage, discharge data were converted to stage data, and base stages were subtracted from each value. Baseflow was calculated as the flow during the growing season that was exceeded 95% of the time over the last ten years. The baseflow values were then converted to stage values.

Ramping: The converted stage data above base stage were assessed from the day of the spring peak to the day when stage change became minimal. Minimal stage change was defined as a change less than 1 cm/d for seven consecutive days. Three-day moving averages were calculated, and then daily stage changes were calculated by subtracting the current day's level from the previous day's level. Stage declines were classified as sufficient (< 5 cm/d), stressful (5-10 cm/d) or lethal (>10 cm/d) and proportions of each category were used to calculate a differentially weighted mortality coefficient (M) such that M = (prop. of lethal days * 3 + prop. of stressful days) / 3.

Originally a set date (July 31) was used as the endpoint of the analysis because it eliminated the process of determining when stage leveled out, but this resulted in an artificially high number of sufficient days, which diluted the proportion of stressful and lethal days. For example, along the lower St. Mary, 1964 was a notoriously bad year for ramping where flows were abruptly dropped and kept at baseflow for the rest of the summer (Rood et al. 1995). With the method described above, the mortality coefficient

was 65. But with the set endpoint of July 31, the constant baseflow increased the number of favourable days and resulted in a mortality coefficient equal to 1995, a year when ramping flows were implemented fairly successfully.

Late summer stage (H_{aug}): Mean monthly discharge values for August published in HYDAT were back converted to stage data, and then base stage was subtracted.

Table 3-2. Rating criteria for the recruitment analysis. The following criteria were developed and applied to historic values to produce an overall recruitment score indicating the likelihood of recruitment in any particular year.

Disturbance (Q_{max}):

++	large disturbance (>10 y return interval (T)) within 3 y
+	moderate disturbance (5-10 y) within 2 y
-	small disturbance (<5 y)

Spring peak timing (H_{sp} date):

+	May 24 - Jun 15
-	outside limits

Spring peak stage above base stage (H_{sp}):

+++	> 2 m
++	1.3 - 2 m
+	0.6 - 1.3 m
-	< 0.6 m

Ramping (M):

++	M < 5
+	M = 5-10
-	M > 10 or $H_{sp/b}$ too low

Drought survival (H_{aug}):

+	H_{aug} > 0.11 m above base stage
-	H_{aug} < 0.11 m above base stage

/9

Proceeding pages:

Table 3-3. Historic values used in the recruitment analysis. Recurrence intervals (T) were calculated from flood magnitudes (Q_{max}), mortality coefficients (M) were calculated from the rate of post-flood flow decline.

Table 3-4. Likelihood of recruitment from 1995-2005 along the five river reaches. Ratings (+/-) were based on how well the historic values met the rating criteria. The number of '+' ratings was summed for an overall indicator of success regarding each hydrograph component.

		disturbance		timing	spring peak	ramping	survival
		Q_{max} (m^3/s)	T	H_{sp} (date)	H_{sp} (m)	M	H_{aug} (m)
USM:	1995	337	25	7-Jun	1.94	9	0.48
	1996	126	2-3	10-Jun	1.18	2	0.26
	1997	157	3-5	2-Jun	1.32	3	0.33
	1998	84	< 2	18-Jun	0.94	2	0.18
	1999	88	< 2	20-Jun	0.85	2	0.35
	2000	61	< 2	17-Jun	0.78	0	0.19
	2001	58	< 2	29-May	0.75	0	0.15
	2002	197	6-7	30-Jun	1.49	10	0.25
	2003	80	< 2	31-May	0.92	1	0.13
	2004	50	< 2	7-Jun	0.69	2	0.29
	2005	95	< 2	7-Jun	1.01	3	0.15
LSM/P:	1995	522	80-90	7-Jun	2.58	21	0.34
	1996	81	< 2	6-Jun	0.9	9	0.07
	1997	117	2-3	15-Jun	1.11	18	0.11
	1998	76	< 2	20-Jun	0.87	9	0.08
	1999	31	< 2	18-Jul	0.43	N/S	0.39
	2000	11	< 2	14-May	0.08	N/S	0.06
	2001	6	< 2	4-Jun	0.08	N/S	0.01
	2002	196	6-7	29-Jun	1.48	33	0.06
	2003	45	< 2	31-May	0.63	6	0.04
	2004	11	< 2	28-Jul	0.11	N/S	0.07
	2005	370	20-30	8-Jun	2.11	29	0.04
UW:	1995	449	20	7-Jun	1.96	6	0.26
	1996	127	3	9-Jun	0.93	2	0.25
	1997	154	5	2-Jun	0.99	1	0.26
	1998	105	< 2	17-Jun	0.83	0	0.19
	1999	101	< 2	19-Jun	0.81	0	0.30
	2000	64	< 2	24-May	0.63	0	0.17
	2001	101	< 2	5-Jun	0.81	0	0.11
	2002	180	8	18-Jun	1.11	3	0.22
	2003	94	< 2	30-May	0.78	0	0.13
	2004	69	< 2	7-Jun	0.66	0	0.23
	2005	115	2-3	7-Jun	0.86	0	0.17
LW:	1995	902	~300	7-Jun	4.47	17	0.10
	1996	106	< 2	6-Jun	0.88	6	-0.02
	1997	156	2-3	3-Jun	1.18	15	-0.04
	1998	126	< 2	19-Jun	1	14	0.03
	1999	66	< 2	7-Jun	0.58	N/S	0.12
	2000	8	< 2	13-May	0.04	N/S	-0.05
	2001	30	< 2	6-Jun	0.29	N/S	-0.06
	2002	300	5-10	10-Jun	1.94	13	0.02
	2003	67	< 2	4-Jun	0.59	N/S	-0.05
	2004	31	< 2	11-Jun	0.29	N/S	-0.02
	2005	245	5	7-Jun	1.09	16	-0.02
CSL:	1995	812	300-500	7-Jun	3.45	8	0.34
	1996	125	2-3	5-Jun	1.3	4	0.16
	1997	167	3-5	1-Jun	1.47	7	0.19
	1998	178	3-5	27-May	1.51	7	0.11
	1999	104	< 2	26-May	1.19	6	0.21
	2000	67	< 2	23-May	0.95	1	0.08
	2001	90	< 2	5-Jun	1.11	5	0.05
	2002	225	5-6	17-Jun	1.68	9	0.16
	2003	101	< 2	26-May	1.17	1	0.06
	2004	56	< 2	7-Jun	0.87	1	0.21
	2005	150	2-3	7-Jun	1.41	4	0.17

		disturbance (/ ++)	H_{sp} timing (/ +)	H_{sp} (/+++)	ramping (/ ++)	survival (/ +)	Total / 9
USM:	1995	++	+	++	+	+	7
	1996	++	+	+	++	+	7
	1997	++	+	+	++	+	7
	1998	-	-	+	++	+	4
	1999	-	-	+	++	+	4
	2000	-	-	+	++	+	4
	2001	-	+	+	++	+	5
	2002	+	-	+	+	+	4
	2003	+	+	+	++	+	6
	2004	-	+	+	++	+	5
	2005	-	+	+	++	+	5
	Overall	8/22	7/11	12/33	20/22	11/11	
LSM/P:	1995	++	+	+++	-	+	7
	1996	++	+	+	+	-	5
	1997	++	+	+	-	-	4
	1998	-	-	+	+	-	1
	1999	-	-	-	-	+	1
	2000	-	-	-	-	-	0
	2001	-	+	-	-	-	1
	2002	+	-	+	-	-	2
	2003	+	+	+	+	-	4
	2004	-	-	-	-	-	0
	2005	++	+	+++	-	-	6
	Overall	10/22	6/11	11/33	3/22	2/11	
UW:	1995	++	+	++	+	+	7
	1996	++	+	+	++	+	7
	1997	++	+	+	++	+	7
	1998	-	-	+	++	+	4
	1999	-	-	+	++	+	4
	2000	-	-	+	++	+	4
	2001	-	+	+	++	-	4
	2002	+	-	+	++	+	5
	2003	+	+	+	++	+	6
	2004	-	+	+	++	+	5
	2005	-	+	+	++	+	5
	Overall	8/22	7/11	12/33	21/22	10/11	
LW:	1995	++	+	+++	-	-	6
	1996	++	+	+	+	-	5
	1997	++	+	+	-	-	4
	1998	-	-	+	-	-	1
	1999	-	+	-	-	+	2
	2000	-	-	-	-	-	0
	2001	-	+	-	-	-	1
	2002	+	+	++	-	-	4
	2003	+	+	-	-	-	2
	2004	-	+	-	-	-	1
	2005	+	+	+	-	-	3
	Overall	9/22	9/11	9/33	1/22	1/11	
CSL:	1995	++	+	+++	+	+	8
	1996	++	+	+	++	+	7
	1997	++	+	+	+	+	6
	1998	-	+	++	+	-	4
	1999	-	+	+	+	+	4
	2000	-	-	+	++	-	3
	2001	-	+	+	++	-	4
	2002	+	-	++	+	+	5
	2003	+	+	+	++	-	5
	2004	-	+	+	++	+	5
	2005	-	+	+	++	+	5
	Overall	8/22	9/11	15/33	17/22	7/11	

3.3.3 Application of the model to historic values

In the decade of streamflow analyzed, over the five reaches, nearly a complete range of establishment scores from zero to eight out of a possible nine were observed (Table 3-4). The most favourable years on all reaches were the years following the flood of 1995. This flood provided two extra points for each of 1995, 1996, and 1997 that were not given for any other year (with the exception of 2005 for LSM/P). Ramping and summer drought survival were less favourable on the regulated LSM/P and LW in those years, but the other hydrograph characteristics were favourable enough to give a relatively high overall recruitment score.

Disturbance flows were approximately equal across the reaches throughout the period analyzed, and this is consistent with the observation that river damming has not attenuated flood flows along the St. Mary and Waterton rivers (Figure 1-5). The timing of those floods has slightly changed, however. Along the St. Mary, timing was slightly less favourable along the regulated reach, whereas along the Waterton, the lower reach had slightly more favourable spring peak timing (Table 3-4).

There was also a difference between the free-flowing reaches and the regulated reaches in the number of years with favourable spring peaks. In years with smaller floods (< 2 yr return interval) such as 1999-2001 and 2004, the free-flowing reaches still had spring peaks that were greater than the minimum elevation where seedlings establish, but the LSM and LW did not. Annual maximum discharge in those years on those reaches often occurred at a time other than spring.

Ramping rates were much less favourable along the regulated reaches, as the time period when ramping should occur coincides with reservoir filling, reducing the water supply available for environmental functions. The USM and UW had favourable ramping in almost every year, while the LSM/P and especially the LW had highly unfavourable ramping. The CSL had favourable ramping in most years, with moderately favourable ramping in the remaining years. The Castle River provided an interesting intermediate between the upper and lower reaches of the St. Mary and Waterton. Similar to the upper reaches, it is free-flowing; however it is not attenuated by any lakes, which results in a flashier flow regime.

A similar phenomenon occurred with the drought survival ratings. Along the LW and LSM/P, only one or two years, respectively, out of the eleven years analyzed had sufficient August discharge (Table 3-4). The upper reaches were favourable in almost all years, and the CSL reach was again intermediate.

After 1995-1997, there was reduced likelihood of recruitment; however, a score of four might still result in some recruitment. Years that scored a four, were missing some aspect of the establishment process, but might have sufficient ratings in the rest of the aspects to enable some colonization. For example in 1998-2001 on the USM and the UW, there was no disturbance flow, but some of the recruitment band would have been wetted, and subsequent ramping and drought survival flows were favourable.

Years that scored a 0-3 would likely not support recruitment as they were missing several critical aspects of the hydrograph, and the aspects that they did have were only marginally favourable. Years that scored a five, might support limited recruitment, and years with a score of 6 or above likely had fairly extensive recruitment events.

3.3.4 Verification of the model with excavated sapling data

The excavated saplings from Chapter 2 provided a way of verifying the model's output from the earlier years (i.e. 1995-2000). The association should be interpreted with caution as not all saplings were randomly sampled, sample sizes were small, and the excavation data represents recruitment into the sapling phase rather than seedling recruitment.

For all reaches, the associations between the model predictions and the observed sapling recruitment revealed an apparent lag between the prediction and the outcome (Figure 3-2). While the model predicted that most recruitment should have occurred in 1995, 1996, and 1997, the sapling data showed that most establishment occurred in 1997, 1998, and 1999.

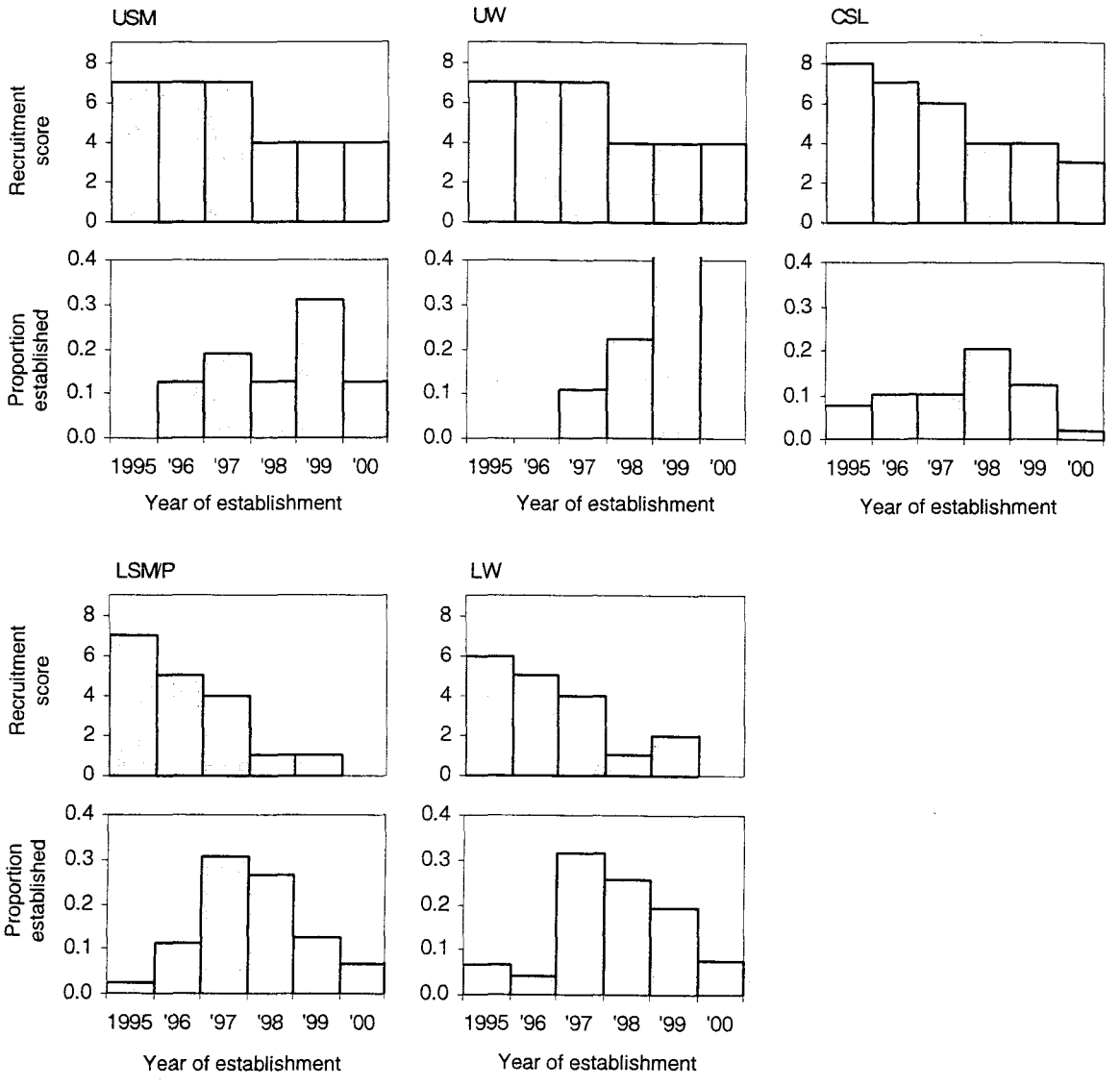

Figure 3-2. Comparison of recruitment model predictions (top) and the observed frequency of saplings (bottom) for each river reach analyzed from 1995 to 2000. Values plotted in the top figure for each reach are the overall recruitment score out of nine generated by the model. Values plotted in the bottom figures are the relative frequencies of excavated saplings that established in that year.

3.4 DISCUSSION

A model is defined as an abstraction or simplification of a natural phenomenon (Smith 1996). In riparian systems, which have been described as "the epitome of heterogeneity" (Naiman et al. 2005), the simplification of a process such as recruitment should be undertaken with caution. In addition to the five hydrograph components considered in my model, there are many other factors that could affect seedling recruitment such as climate, seed source, or the availability of suitable recruitment sites.

For example, favourable ramping flows and sufficient August flows are going to be more important in a hot and dry year when water demand is greater. It is also possible that climatic variables would play different roles depending on location in the basin. The more western higher elevation reaches, which are cooler and receive more precipitation (Table 2-1), might rely less on streamflow than the eastern lower elevation reaches. Another climatic factor that could affect model outcome, and that was observed in 2005, was a late season frost that caused abortion of catkins resulting in low seed production. Despite somewhat favourable flows in 2005, field observations indicated minimal establishment.

The exclusion of climatic factors may be one of the reasons that the model did not accurately predict the observed sapling age distributions (Figure 3-2). Conversely, another variable may need to be added to the model to account for post-establishment mortality. A scour survival component was included in the model developed in Braatne et al. (2007), but that was included to account for the common winter storm events that could flush away newly established seedlings. As we do not commonly receive winter floods in this region, this variable was not included in the model. A variable that

represents ice scour, or next year's spring flood scour potential would be more suitable for this region, and may improve model predictions.

Although the model did not predict sapling frequency accurately, it may be more accurate in predicting initial seedling establishment, which was the intended purpose of the model. The high recruitment scores in 1995, 1996, and 1997 are consistent with the observation of a wide spread recruitment event in the Oldman River Basin (Kalishchuk et al. 1998; Rood & Mahoney 2000).

Differences in recruitment likelihood between free-flowing and regulated reaches were apparent. Along the free-flowing reaches, most years were quite favourable, which is contradictory to the observation that recruitment is naturally episodic (Scott et al. 1997; Samuelson & Rood 2004). However, my model focused on establishment or colonization and first year survival, which may occur more frequently than recruitment (survival to the 3rd year) if mortality occurs after the first year (Rood et al. 2007). My model incorporates some aspect of longer term survival, and thus recruitment, by ensuring that establishment occurs within an elevational band in which seedlings usually survive.

Other factors such as cattle grazing and pugging (breakdown of substrate surface from cattle trampling) can affect recruitment. There were quite a few years in which recruitment seemed likely on the USM, but from field observations on river trips, few saplings were observed, but evidence of cattle was. Seedling study sites have been lost to cattle grazing along the Oldman River (Kalischuk et al. 2001) and along the Elk River (Polzin & Rood 2006).

The generalizations and assumptions that the model relied on may not apply to all site-specific conditions. The recruitment band was set from 0.6 to 2 m above base stage

with sufficient stage recession rate of 5 cm/d. However, other studies have found recruitment both above and below these boundaries (Scott et al. 1997; Rood et al. 1998). The upper boundary is dependent on the amount of precipitation usually available and both the upper boundary and ramping rate could be variable dependent on the substrate present, with finer substrate allowing more moisture retention. The lower boundary depends partly on ice scour, which may be more prevalent along some reaches than others (Rood et al. 2007), and subsequent flood scour.

Another assumption was that the stage rating curve, which is developed at one cross-section along the entire reach, is representative of that reach. In a segment with a steeper or narrower channel compared to the cross-section measured, the same amount of flow would result in a higher stage. However, site specific stage-discharge patterns have been closely associated with the gauging station's stage-discharge pattern along some rivers (Polzin & Rood 2006).

The area of floodplain surface encompassed by the recruitment band is also dependent on channel shape with larger areas available on gradually sloping, broad surfaces. This might mean that a similar recruitment score on two reaches that differ in geomorphology could result in different amounts of recruitment. Further, the interaction of streamflow with geomorphology means that a streamflow criterion applicable to an alluvial reach might not apply to a partially confined reach. For example, along partially confined channel sections along the upper Missouri River, recruitment occurs only at high flows, whereas in sections with lateral migration, recruitment occurs at both moderate and high flows (Scott et al. 1996).

The lower Waterton and lower St. Mary had very similar recruitment scores, but the lower Waterton has extensive zones of saplings with multiple age classes, while the lower St. Mary has few sapling patches (Section 2.3.2). This is partly due to a lack of seed supply on the St. Mary, but despite the low recruitment scores, the Waterton has still supported recent recruitment.

In conclusion, the overall recruitment score produced by the model showed a co-ordination with initial seedling establishment documented in the literature (Rood et al. 1998; Rood & Mahoney 2000), but not seedling recruitment (i.e. survival to the third year). This was shown by the disparity between model predictions and the observed sapling frequencies. The model did consider one aspect of post-establishment survival by ensuring that seedling establishment occurred within the recruitment band–an elevational band that generally favours survival–but even with this consideration, the model was more accurate at predicting establishment, rather than recruitment. Refining the model with the addition of another hydrograph component that represents post-establishment scour from the following years flood events would likely improve predictions of recruitment, and thus co-ordination with observed sapling frequencies.

CHAPTER 4
Implications for Management and Future Research

To restore the degraded cottonwood population along the lower St. Mary River, a systemic, or landscape-scale restoration approach was undertaken. Critical components of the flow regime were changed to restore the abiotic processes needed for the function of the biological community. This can also be considered a passive restoration approach in which the stress causing ecosystem degradation is removed, and should be the first step taken for ecological restoration (Kauffman et al. 1997; SER 2004). With current instream flow management practices, ecosystem condition along the lower St. Mary reach (LSM) has displayed some improvements, but will never be restored to its pre-degraded condition (Figure 2-26). Two prospective strategies to promote further restoration are increased management effort, or a more active restoration approach, and the improvement of instream flow management.

4.1 Increased management effort

In systems where removal of the stressor does not prompt re-initiation of autogenic ecosystem processes, a solution that has been suggested is increased management effort (Hobbs & Norton 1996; Suding et al. 2004). Increased management alters the restoration trajectory and allows the system to overcome the threshold that is preventing recovery. For the lower St. Mary reach, a management strategy that will likely promote further recovery is seed addition. Alternatively, vegetative shoot cuttings could be planted. There are advantages with either strategy, and consideration should be given to experimental, genetic, and practical aspects.

Vegetative cuttings versus seed addition

The advantage of using vegetative shoot cuttings is that they contain more energy reserves, and thus have a high probability of survival (Rood et al. 2003c). Also, cuttings grow quickly, and provide a more advanced stage of recovery. One of the large river restoration projects in North America–the Sacramento River Project in California–used cuttings to re-establish riparian vegetation (Alpert et al. 1999). In addition, the use of cuttings has been extensively developed for the establishment of poplar plantations (Stanturf et al. 2001), and could provide many lessons for natural stand recovery. However, as vegetative plantings for restoration have not yet been carried out in southern Alberta, the collection, storage and preparation of cuttings may present a challenge.

The advantage of the seed addition strategy is that seed bearing branches could be easily cut, bagged, and placed on the floodplain surface when hydrological conditions are favourable for recruitment. Hydrological conditions are generally favourable in high flow years when irrigation demands are lower, and more water is available for beneficial manipulation by Alberta Environment Dam managers. The placement of seed-bearing branches was a strategy that was used along the Middle Rio Grande River (Bhattacharjee et al. 2006). Alternatively, seeds could be collected, cleaned, and then sprinkled onto the floodplain surface in known densities (Friedman et al. 1995). A disadvantage of seed addition is that seedlings typically have high mortality rates (Schweitzer et al. 2002; Dixon 2003). However, this could be ameliorated by adding seeds to floodplain surfaces at favourable elevations as determined by the Recruitment Box Model. Large mammal exclosures or fencing that prevent cattle, deer, and beaver from browsing or disturbing

seedlings could also enhance survival. Another advantage of seed addition is that it provides an opportunity to test the hypothesis that seed source is limiting.

Experimental considerations

The correspondence between mature tree and sapling patch distributions provides evidence for seed source as a limiting factor (Figure 2-4), but this interpretation could be confounded by site favorability. Sites that supported mature trees might also be sites that are naturally more favourable for saplings. To further explore the hypothesis that seed source is limiting, observations of natural seed fall should be undertaken to determine whether sites without trees receive seeds from regional sources. Then, to assess site favourability, equal densities of seeds could be added to sites with trees and to sites without trees, and subsequent seedling establishment and survival would be monitored.

Another aspect of site suitability that could be further assessed is the apparent equal distribution of suitable sites along the lower St. Mary, lower St. Mary downstream of Pothole, and lower Waterton reaches. Subjective assessments from aerial photos suggest that suitable sites are not limiting, but more quantitative assessment of whether these sites are suitable or not could be determined by adding seeds and observing if they survive and grow.

Seed addition over many years could also provide an opportunity to calibrate the recruitment model developed in Chapter 3. A known density of seeds could be added to a range of floodplain surfaces and elevations. The responses of seedling survival and density to each subsequent aspect of the hydrograph could then be monitored.

Genetic considerations

Although collection of seeds and shoots from remnant trees along the LSM may provide propagules that are well-adapted to local conditions, genetic variation may be limited. A range of regional sources should be used for seed or shoot collection, because genetic variation is important for the future success of the reproductive population.

Many studies have shown that most of the genetic variation in *Populus* is found within a population, and that gene flow over large regions has resulted in little variation among populations (Farmer 1996). The range of variation within a population can produce a wide range of flowering and seed dispersal times, which can take advantage of interannual variation in climatic conditions or the flow regime, and provide seeds throughout the period when flood waters are gradually receding and exposing new substrate. Therefore, genetic variation is important for the future reproductive success of the population.

Beyond the population level, recent studies have shown that genotypes of keystone or foundation species like poplar can have effects at the community and ecosystem levels (Whitham et al. 2006). For example, some genotypes have high tannin levels, and are less palatable to many invertebrates, and so affect the community composition. Further, different community compositions affect ecosystem processes like nitrogen mineralization (Whitham et al. 2006).

4.2 Implications for instream flow management

Government managers of the St. Mary Dam have indicated that it would be challenging to obtain more flows for environmental functions (J. Mahoney, pers. comm.)

Should opportunities emerge, however, flows could be managed differently by increasing spring river levels or improving ramping flows.

This study provides some evidence that increased spring flows may promote diameter growth (Figure 2-17). Providing support for a causative association between spring flows and diameter, a preliminary study indicates that seasonal radial growth at a site along the Oldman River occurs in the period when flows are highest, from May to mid-July (C. Phelan, unpublished). These results should be interpreted with caution, however. The diameter-flow relationship I detected may not be causative, but rather explained by differences in altitude and climate between the upper free-flowing and lower regulated reaches (Table 2-1). The diameter-flow relationship may also have been confounded from mixed age classes. Looking at the results from only the excavated saplings that established from 1995 to 1998, there was no significant difference in diameter between the reaches, with the exception of the Castle reach that supported wider saplings (Figure 2-12). There is also a possibility that cottonwoods less than 50 cm in height that were excluded from data collection were not seedlings, but rather stunted older saplings. If there were more stunted saplings along the LSM, this would mean that sapling sizes were smaller than the results indicated.

The recruitment model suggests that improvements to ramping flows could be made, as the regulated reaches were not as favourable as the free-flowing reaches in relation to the mortality coefficient (Table 3-4). Improved ramping flows would also provide increased water in May and June that would be available for sapling growth. In addition, favourable ramping would aid in the establishment of seedlings or cuttings from the seed addition and shoot planting experiments.

In conclusion, this research provides some evidence that increased spring flows, or improved ramping flows could further recover ecosystem condition. Also, seed addition or vegetative shoot plantings could not only aid in the recovery of the cottonwood population, but could provide interesting experimental opportunities. This study provides an impetus to accompany the landscape-scale passive restoration approach that has already been implemented, in which abiotic factors were restored for biotic components, with a site-specific active restoration approach, in which biotic components are directly restored.

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APPENDIX A
MAPS AND PHOTOGRAPHS OF STUDY SITES

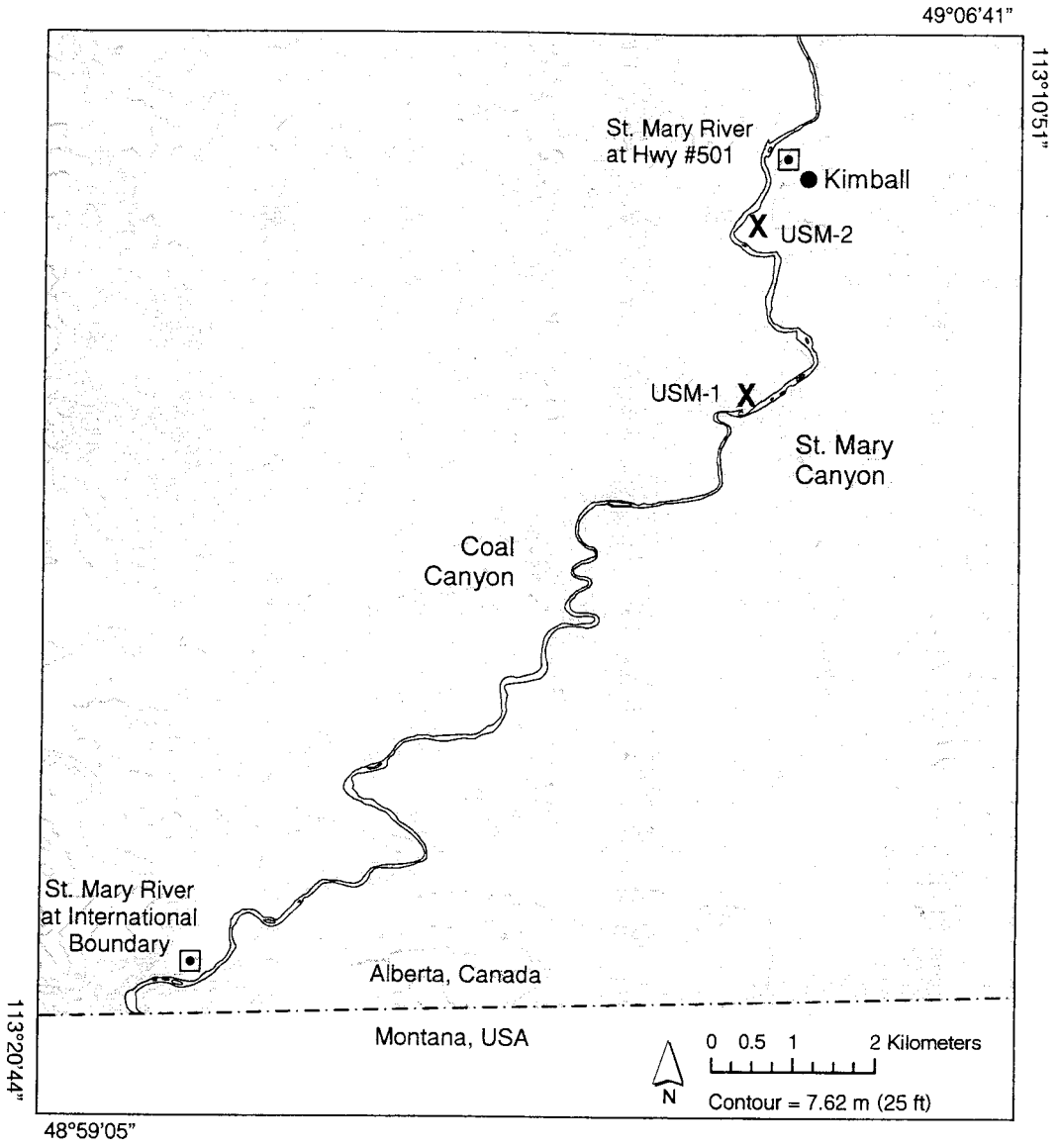


Figure A-1. Location of study sites (X) and hydrometric gauges (⊡; 05AE027 and 05AE043) along the upper St. Mary reach.

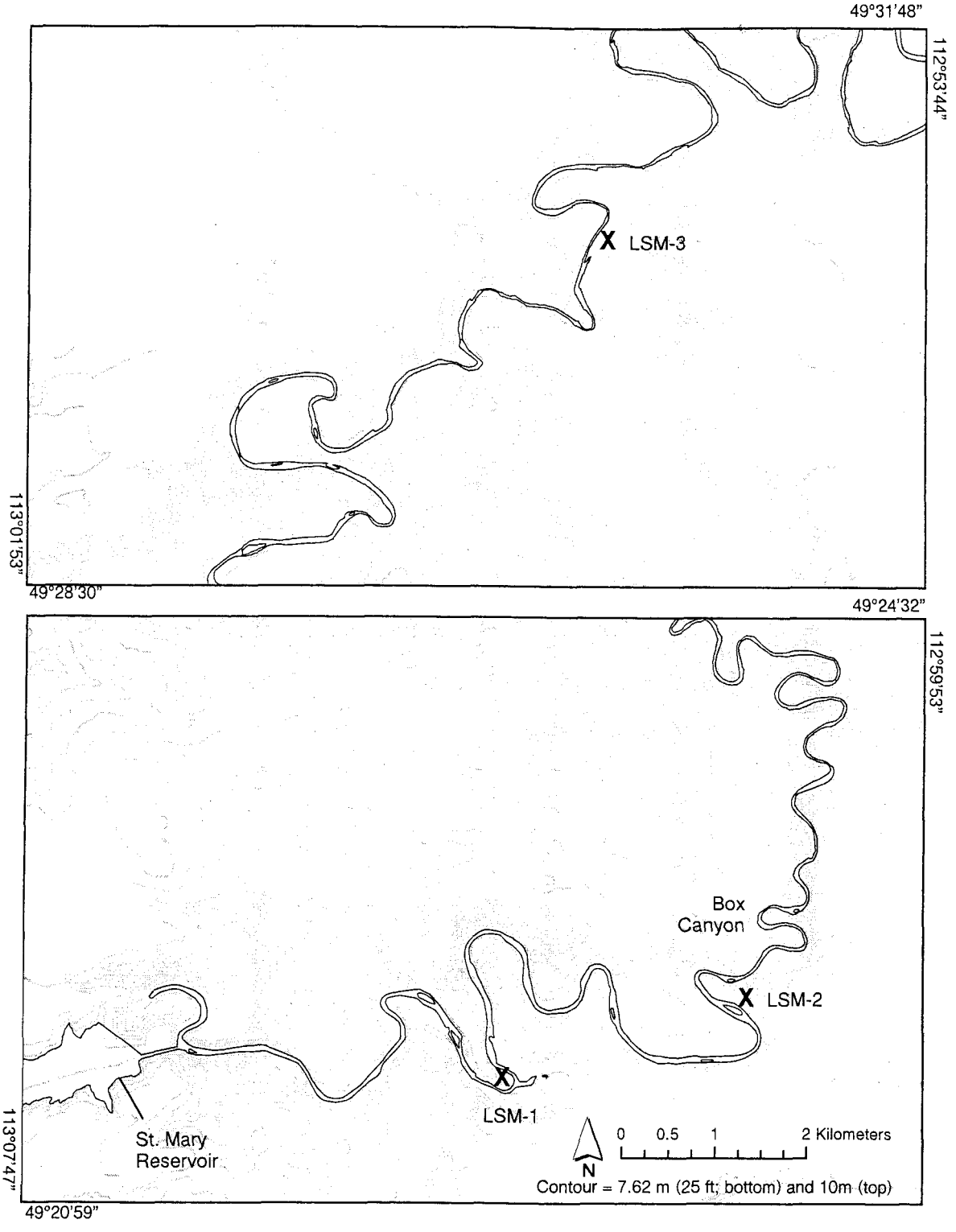


Figure A-2. Location of study sites (X) along the lower St. Mary reach downstream from the Dam and Reservoir. Box Canyon displays closely spaced contour intervals. The start of the river segment shown in the top map is 13 km (river distance) downstream from the end of the river segment shown in lower map. Both maps have the same scale.

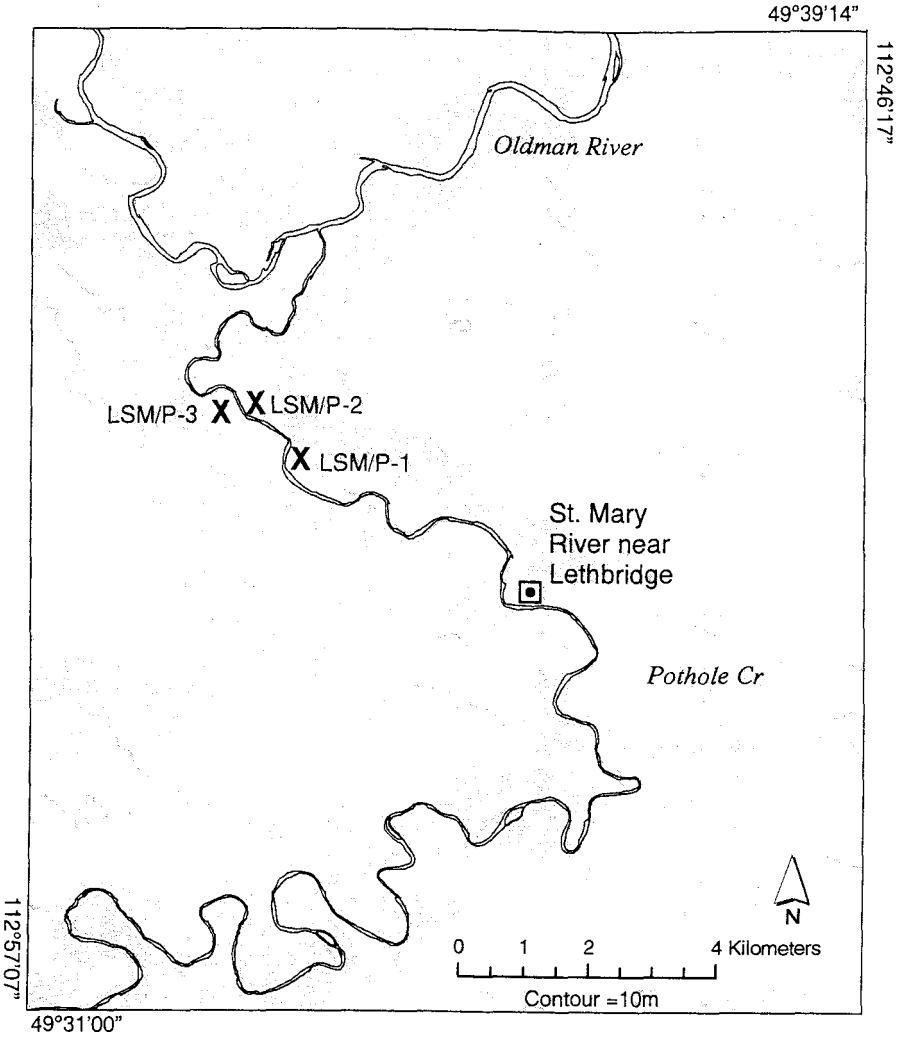


Figure A-3. Location of study sites (X) and streamflow gauge (⊡; 05AE006) along the lower St. Mary downstream of Pothole reach.

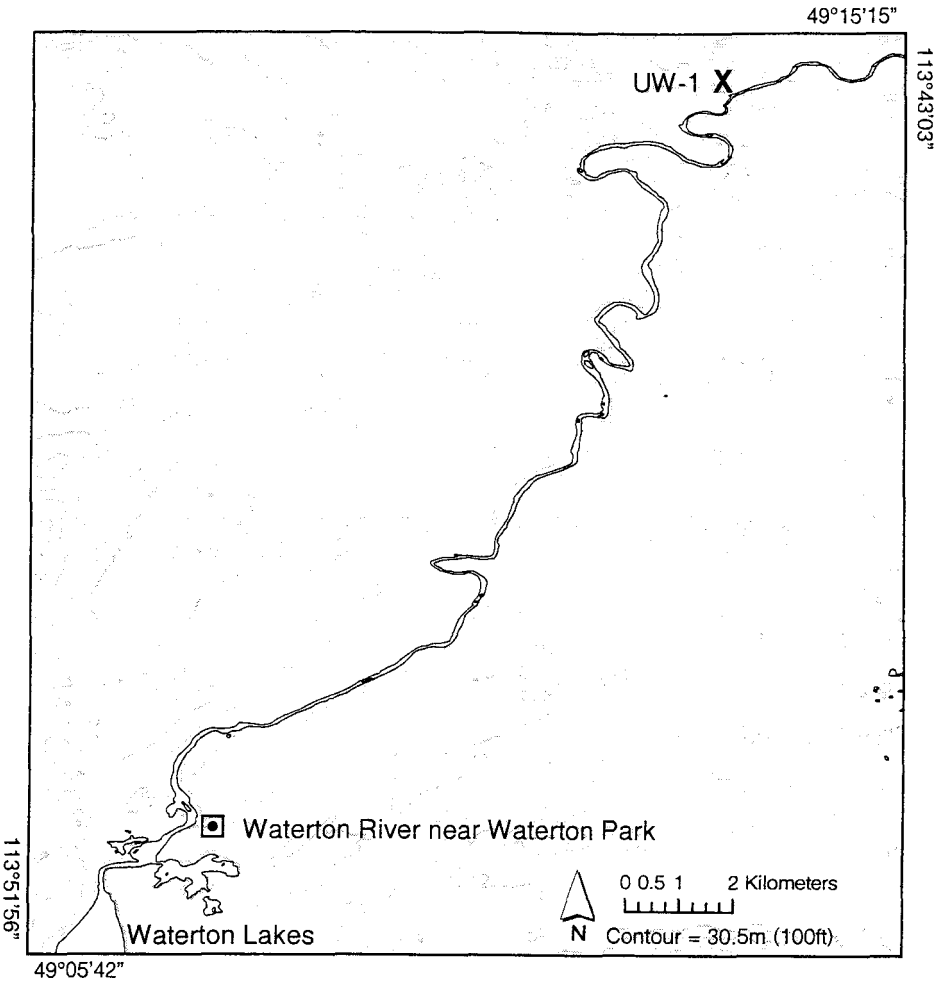


Figure A-4. Location of the study site UW-1 (X) and the hydrometric gauge (⊡) on the upper Waterton reach.

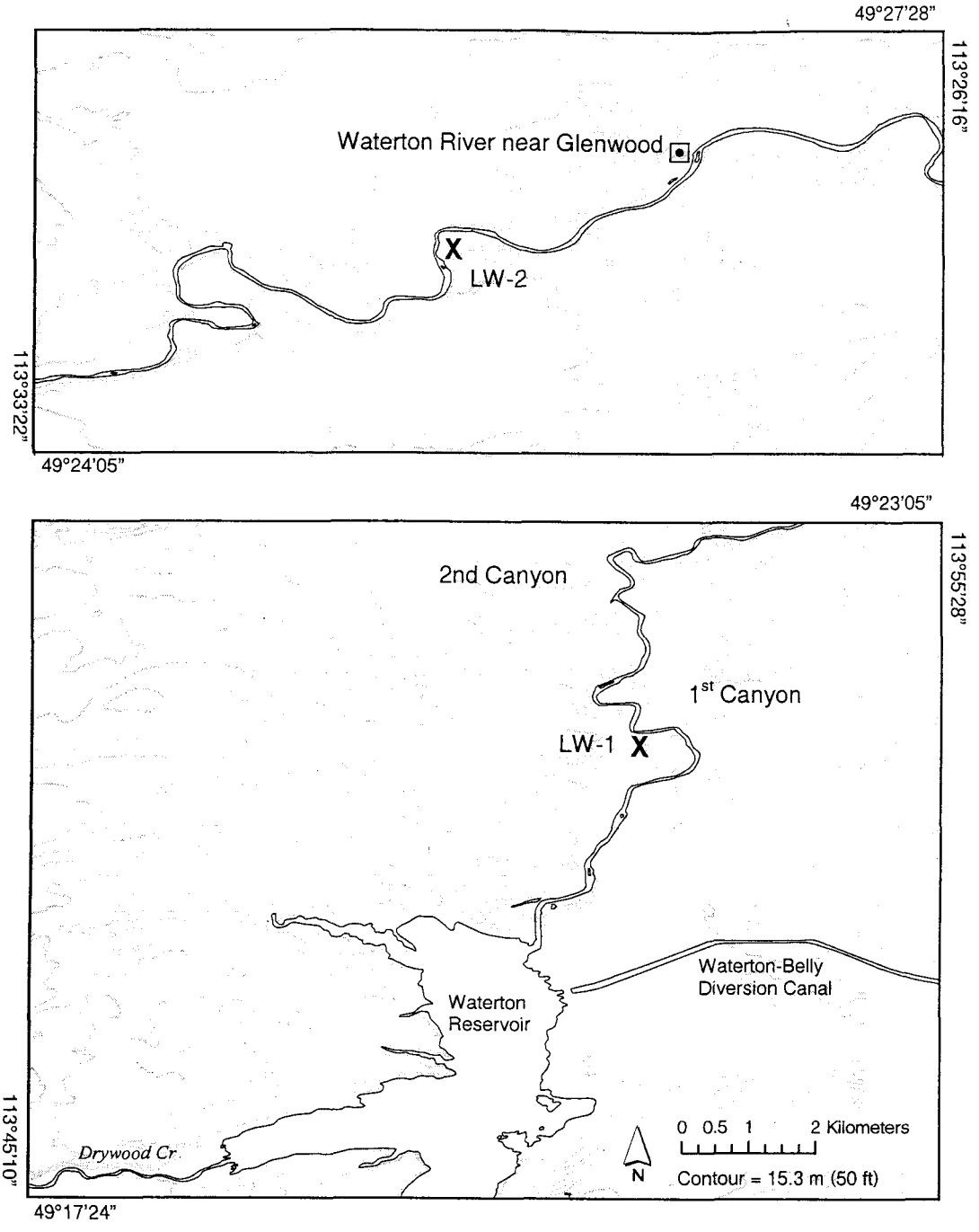


Figure A-5. Location of study sites LW-1 and LW-2 (X) and current hydrometric gauge (⊡; 05AD028) along the lower Waterton reach. The start of the river segment in the top map is approximately 10 km downstream (river distance) from the end of the segment in the bottom map. Both maps have the same scale.

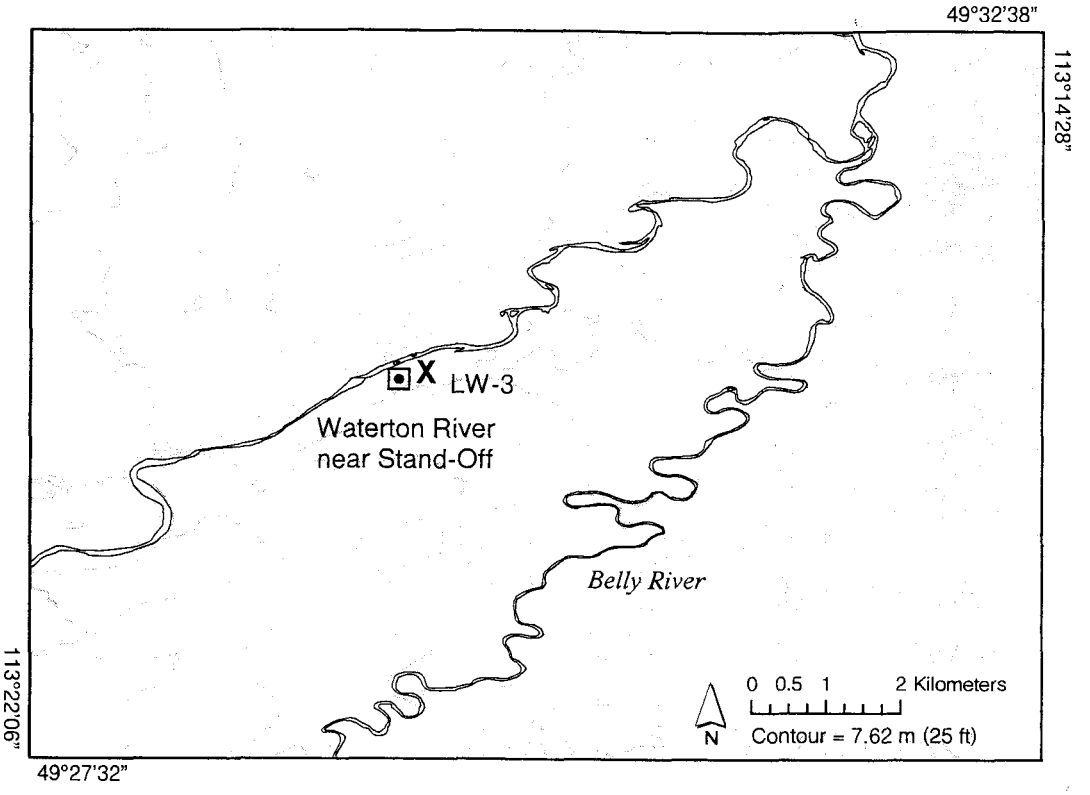


Figure A-6. Location of study site LW-3 (X) and hydrometric gauge (⊡) that is no longer operational (05AD008) along the lower Waterton reach.

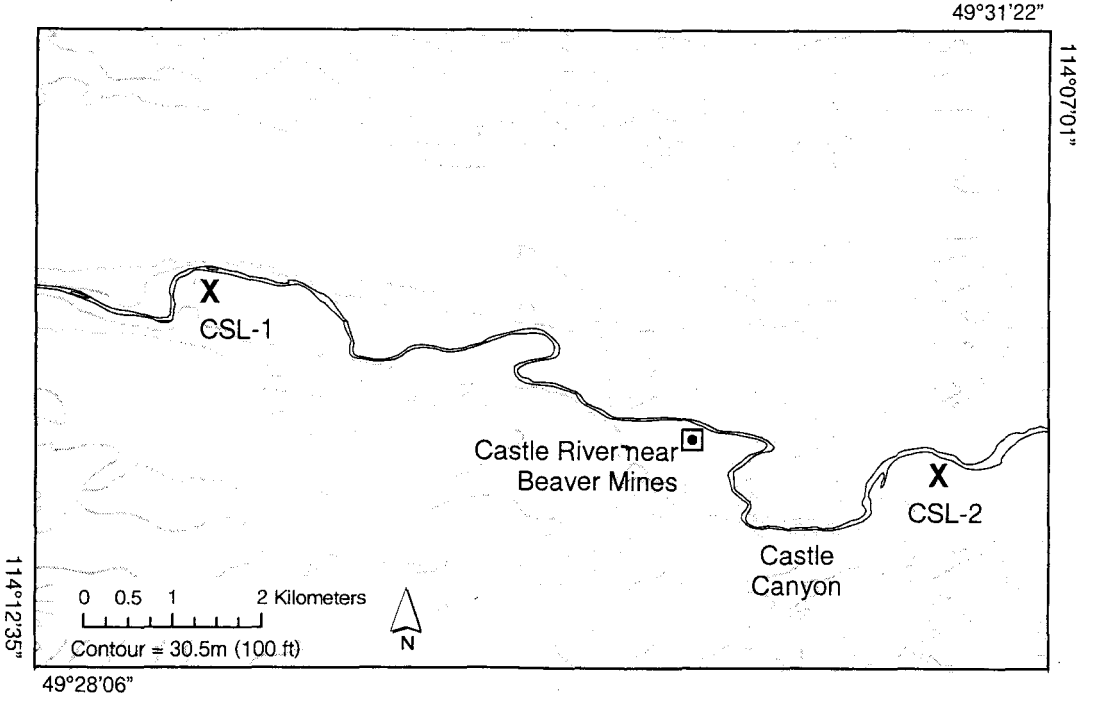


Figure A-7. Location of study sites (X) and hydrometric gauge (⊡; 05AA022) on the Castle reach.

Figure A-8a. Dense band of saplings at USM-2 with mature trees in the background. The photograph was taken in the approximate middle of the site and the main channel is out of the photo on the right. Note: Photograph of USM-1 not available.

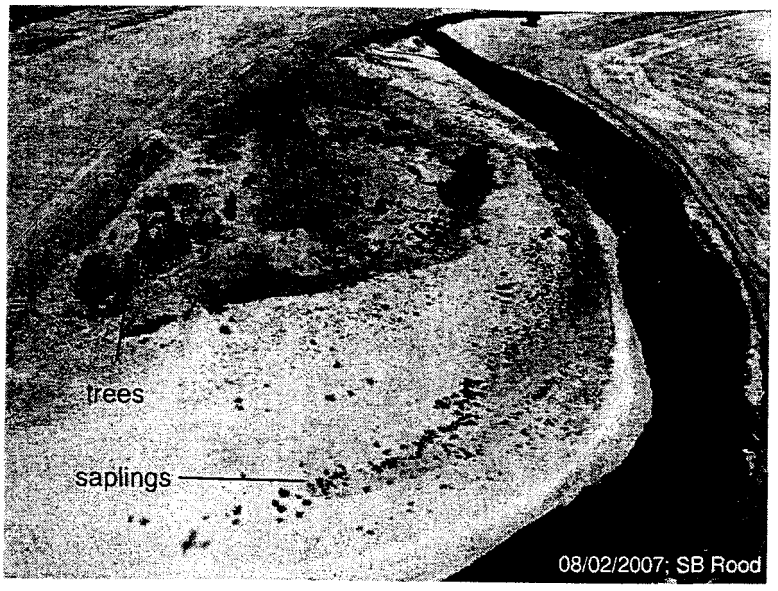


Figure A-8b. Overview of sapling band and patches with some remnant mature trees towards the back of the floodplain on a gradually sloping cobble bar at LSM-1.

Figure A-8c. Dispersed and shrubby saplings at the upstream end, and in the back channel of the attached island at LSM-2.

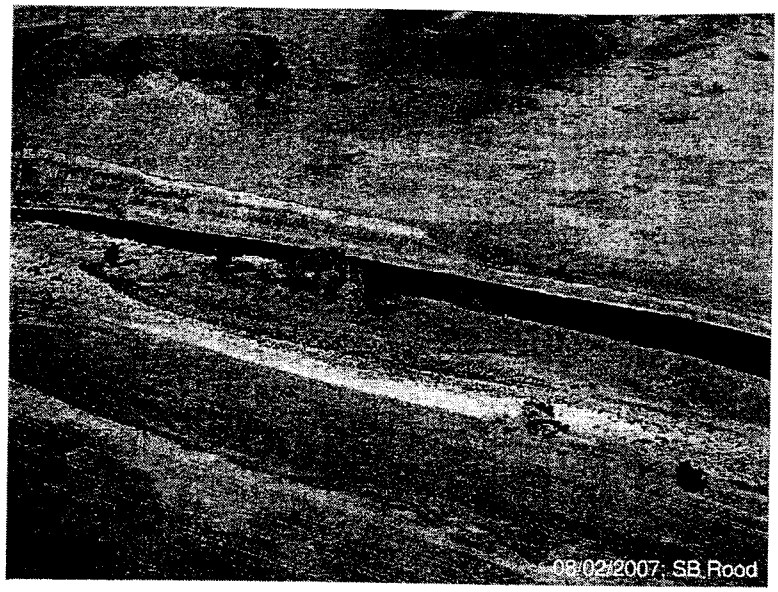


Figure A-8d. Overview of sapling band along the back channel, and mature trees on the attached island at LSM-3.

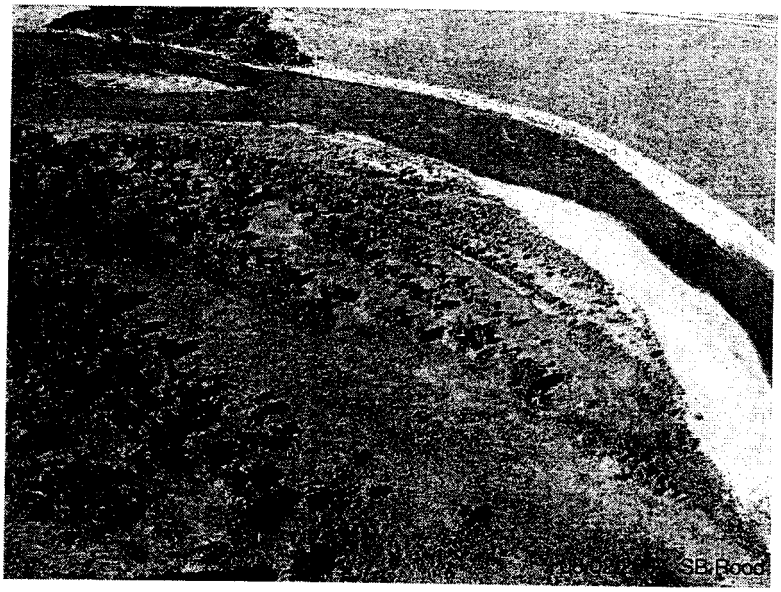


Figure A-8e. Overview of LSM/P-1 from low-altitude flight. The river is flowing from left to right.

Figure A-8f. Dense saplings intermixed with willows at LSM/P-2. The photograph was taken towards the upstream end of the site and a small portion of river is visible on the left.

Figure A-8g. Overview of LSM/P-3 showing multiple sapling size classes with mature trees as seen from a low-altitude flight. River is flowing from left to right, and the study site is located on the point bar in the middle of the photo.

Figure A-8h. Small saplings with pole-size trees at UW-1. The photograph was taken from near the upstream end of the site looking downstream and the river is visible on the right.

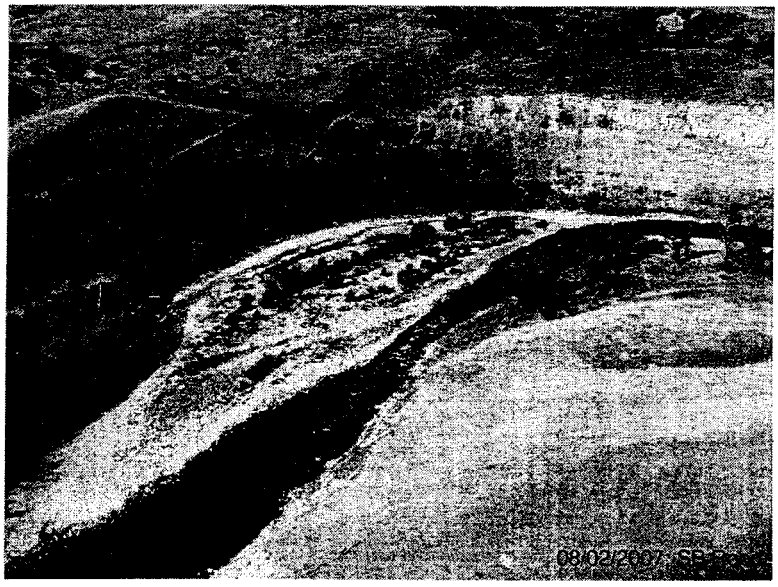


Figure A-8i. Seedlings, saplings, and poles on the island, and mature trees along the valley wall at LW-1.

Figure A-8j. Narrow sapling band with terrace vegetation in the foreground at LW-2. The photograph was taken from the downstream end of the site looking upstream.

Figure A-8k. Dense and wide sapling band at LW-3. The photograph was taken from close to the river channel looking away from the channel.

Figure A-8l. Small sapling band at CSL-1. The photograph was taken at the downstream end of the site looking downstream. The back channel is visible on the right and the bare middle of the island is visible on the left. Another sapling band was located on the other bank of the island.

Figure A-8m. Small sapling band with wolf willow and other shrubs in the foreground at CSL-2. The photograph was taken from close to the river channel.

APPENDIX B
HISTORIC HYDROLOGY OF POTHOLE CREEK

The confluence of Pothole Creek creates a slight change in the hydrologic regime of the St. Mary River. Because the flow of the lower St. Mary can be quite small at times, any contribution to its flow could be important for the downstream ecosystem. Also, as the only gage on the lower St. Mary is downstream from the confluence, it was important to determine how much Pothole Creek contributed to the flow recorded at that gage so that an indication of flow for the reach above the gage can be obtained.

Pothole Creek is a natural channel that was used as part of the irrigation canal system and thus some of the flow represents irrigation return water. It was thus expected that any major change in irrigation infrastructure such as the St. Mary Dam would lead to changes in Pothole hydrology. Flow during the growing season was examined over the period of record, and selected results from spring, late summer and the overall growing season are presented.

From the pre- to the post-dam period (1919-1950 and 1953-1956; 1972-2006 respectively) early spring flows represented by Q_{may} decreased from an average of 2.41 ± 0.29 m^3/s to 0.80 ± 0.15 m^3/s (Figure B-1A). This could have been due to a change in the storage capacity of the irrigation system. With early irrigation infrastructure (i.e. the Kimball weir), there would have been minimal storage and the high flows in spring would have to be returned to the river. After the reservoir was built, spring flows could now be stored for use later in the season. Despite the decline in Pothole discharge during May, its contribution to the St. Mary slightly increased (pre-dam: 7.50 ± 1.15 %; post-dam: 7.77 ± 1.34 %). The increased storage provided by the dam also caused declines in May and June flows on the St. Mary, therefore, the proportional contribution of Pothole remained relatively similar.

In contrast to spring flows, late summer flows increased due to a more consistent discharge from year to year (Figure B-1C; Q_{aug}: pre-dam: 0.63 ± 0.14 m^3/s; post-dam: 0.89 ± 0.14 m^3/s). The highly variable discharge in the pre-dam period meant that contributions to the St. Mary ranged from 0 – 38% with an average of 17.01 ± 2.48%. The more consistent discharges in the post-dam period on average contributed much less

to the St. Mary (12.59 ± 1.37%). The construction of the large storage reservoir providing a more constant and dependable water supply into the late summer and increased irrigation efficiency perhaps lead to the change in flow character. Contribution declined because the August flow on the St. Mary increased from the pre-dam to post-dam period (6.7m^3/s to 8.3 m^3/s), likely due to stored water being released later in the season.

Over the entire growing season, Pothole discharge and its contribution to the St. Mary (Figure B-1C) slightly declined from 1.67 ± 0.15 m^3/s to 1.00 ± 0.12 m^3/s, and 8.99 ± 1.09% to 7.84 ± 1.00%. This means that the LSM reach (above Pothole) experienced around 8-9% less water than the reach below Pothole throughout the period of record. When analyzing historic hydrology of the lower St. Mary above Pothole this reduction in flow should be considered, and when analyzing changes at a smaller time scale (i.e. monthly flows), removing Pothole Creek discharge may provide a more accurate description of flows that that reach experienced.

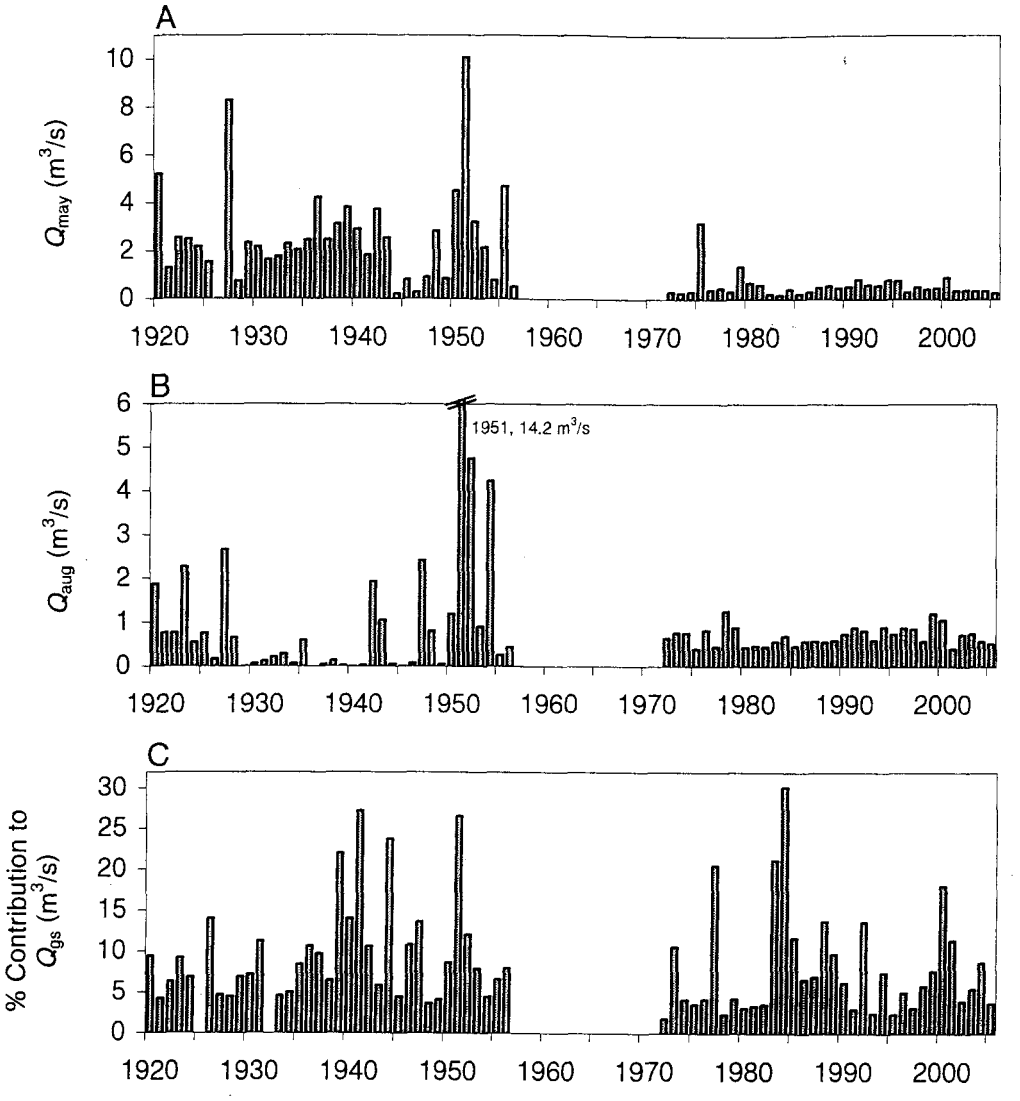


Figure B-1. Historic hydrology of Pothole Creek. Mean monthly discharge in May (A) was much greater during the pre-dam period (1920-1950) than the post-dam period (1953-1956; 1972-2006). Mean August discharge (B) was more consistent from year to year and thus overall, in contrast to spring flows, it was greater in the post-dam period. Despite changes in hydrology between the pre-dam and post-dam periods, the proportional contribution of Pothole Creek to the St. Mary River (C) has not changed during the growing season.

APPENDIX C
TRANSECT REPRESENTATIONS

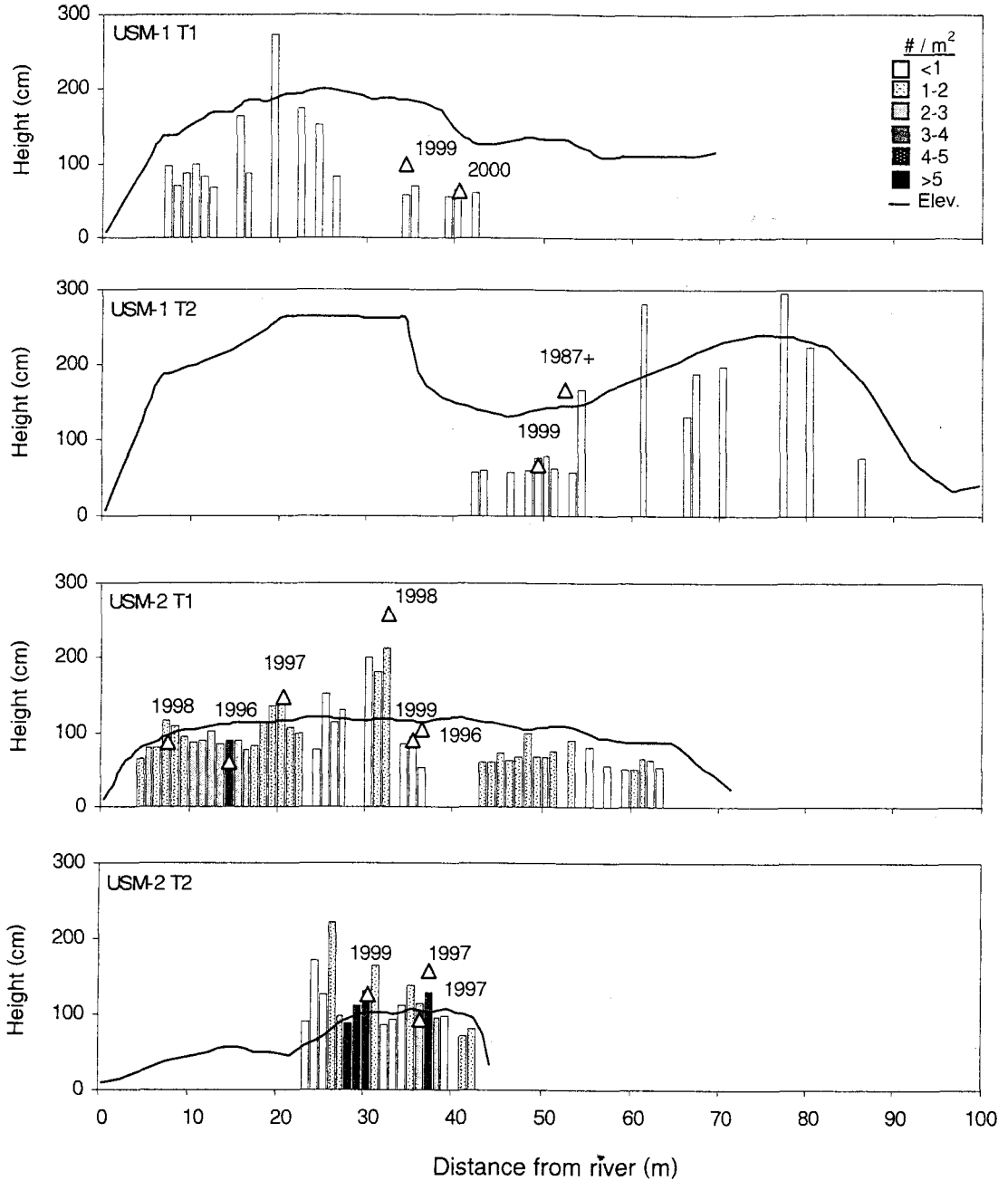
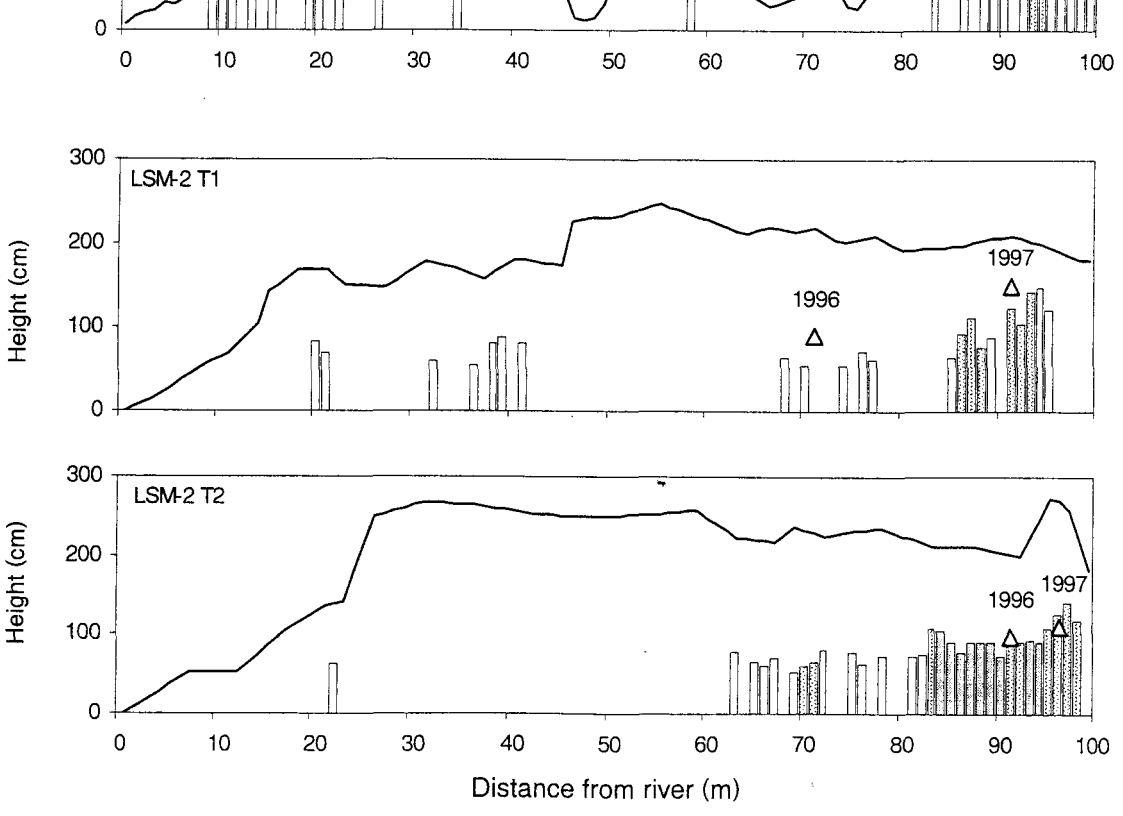
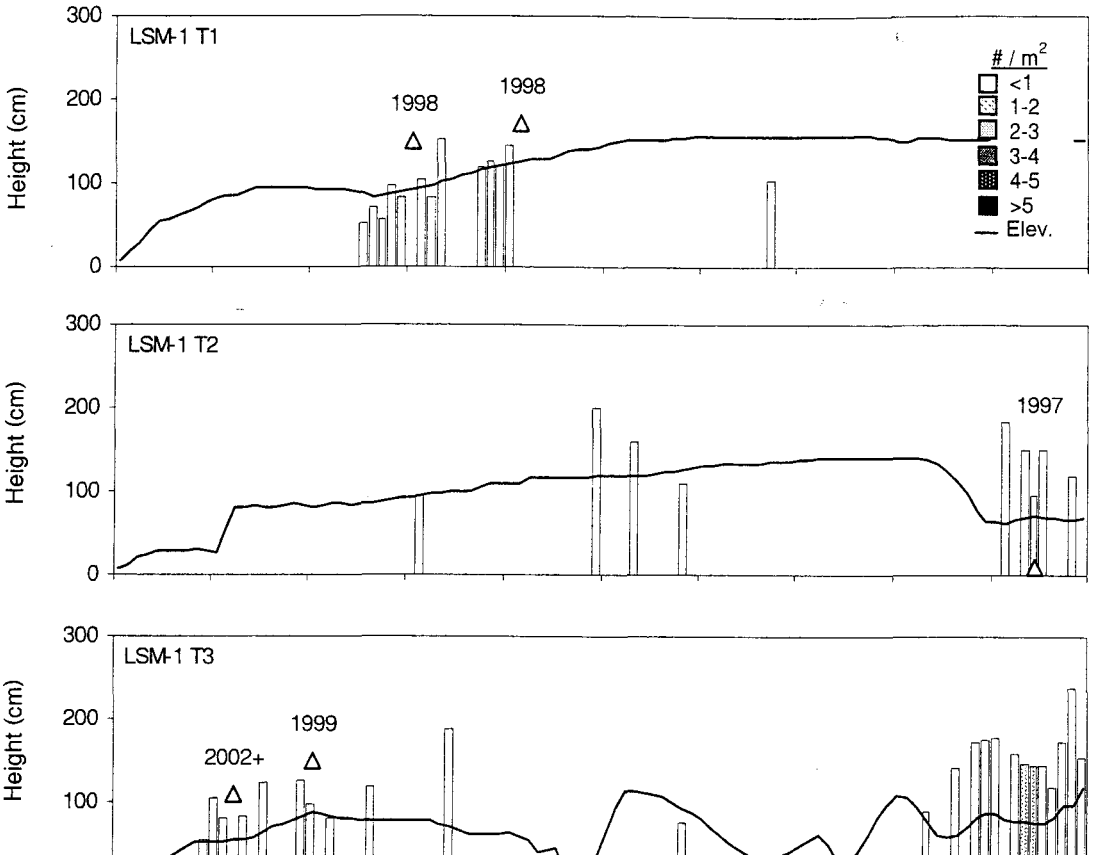
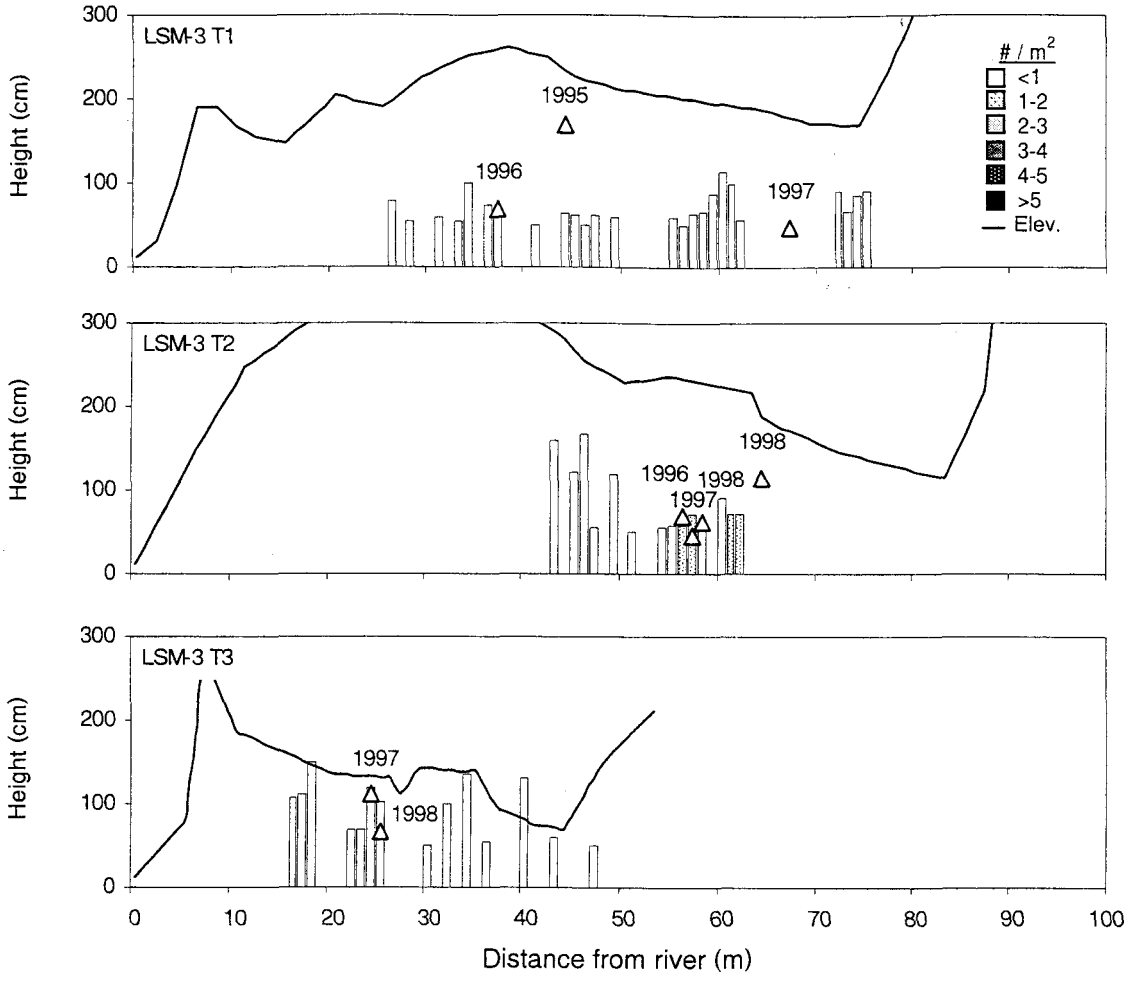
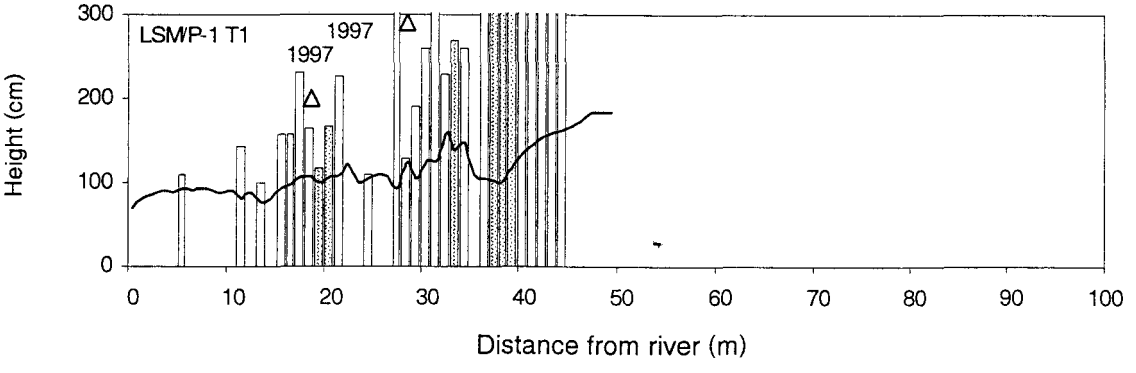


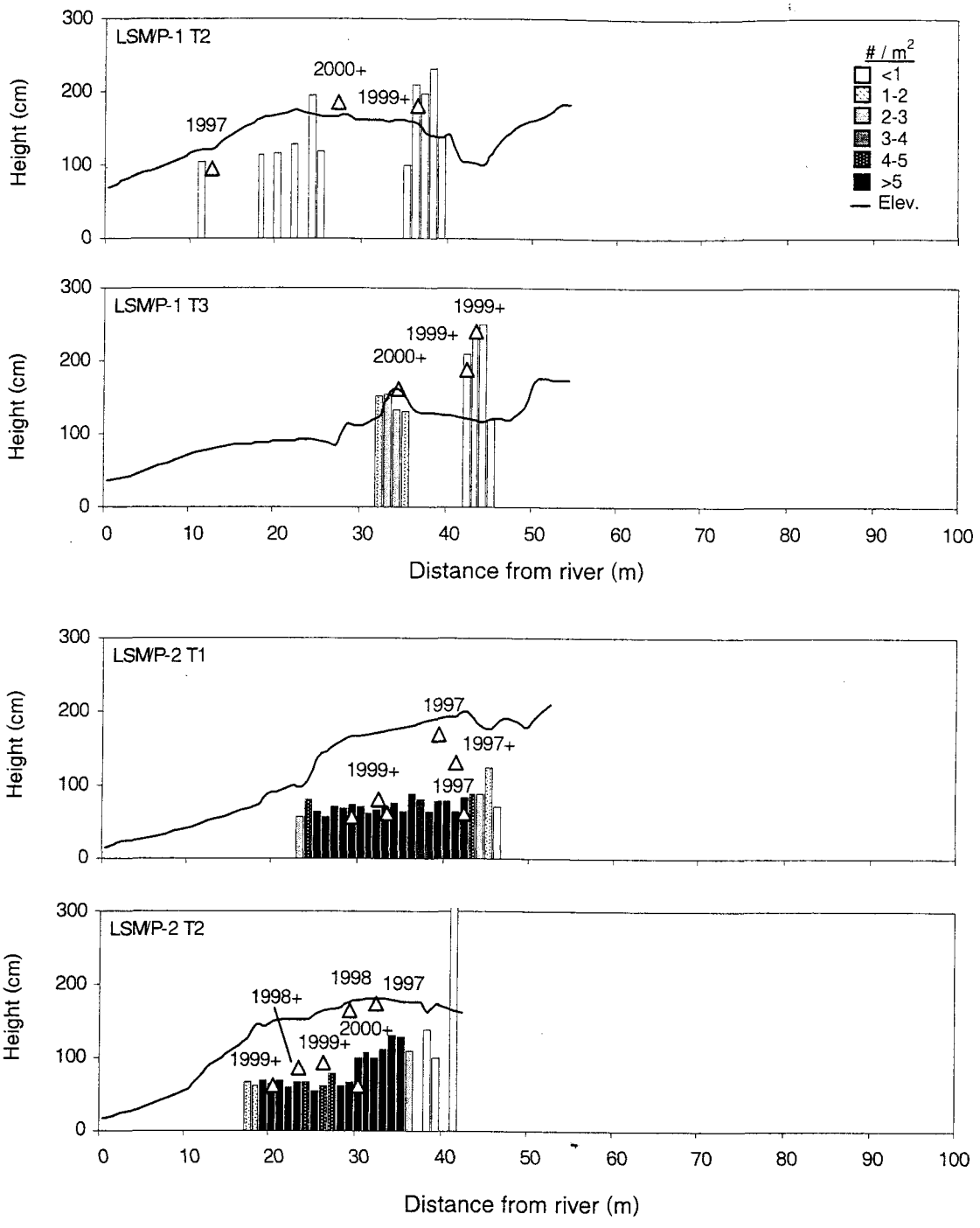
Figure C-1. Sapling height and density in relation to floodplain elevation above baseflow and distance from the river along each of the 33 transects. Bars represent mean height of saplings per quadrat and are categorized by density. Triangles (Δ) represent the above-ground height of an excavated sapling that was aged, and are thus associated with an establishment year. The '+' denotes that the maximum age was not obtained. If the heights of the excavated saplings were not available, they were plotted along the x-axis. Transect identification labels are in the top left corner (e.g. "USM-1 T1" is transect #1 at site #1 along the upper St. Mary). This figure continues on the next 7 pages.

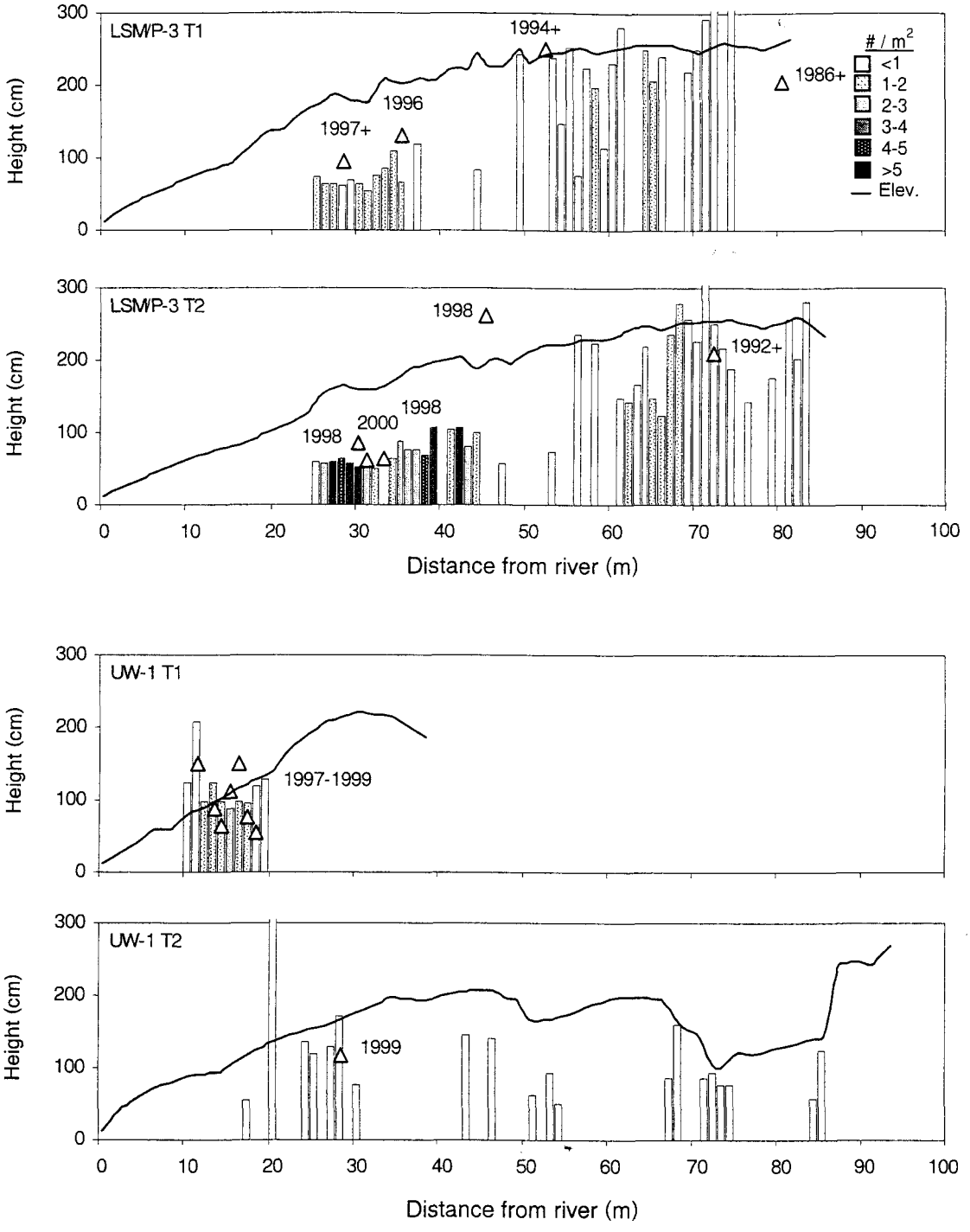

LSM-2 T1
1996
1997
LSM-2 T2
1996
1997
Height (cm)
Distance from river (m)

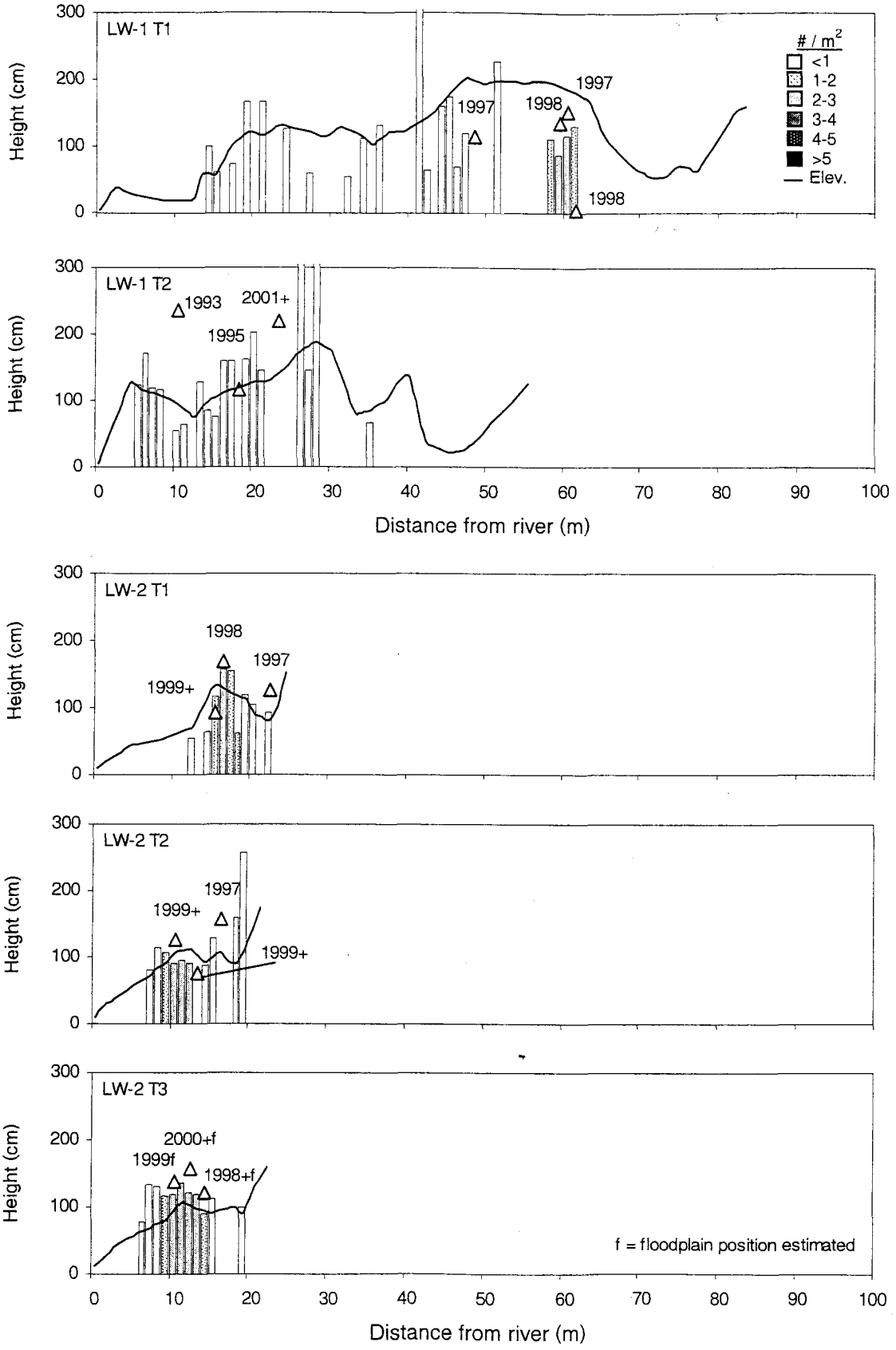

LSM-1 T1
Height (cm)
300
200
100
0
1998
1998
/ m2
<1
1-2
2-3
3-4
4-5
>5
Elev.
LSM-1 T2
1997
LSM-1 T3
2002+
1999

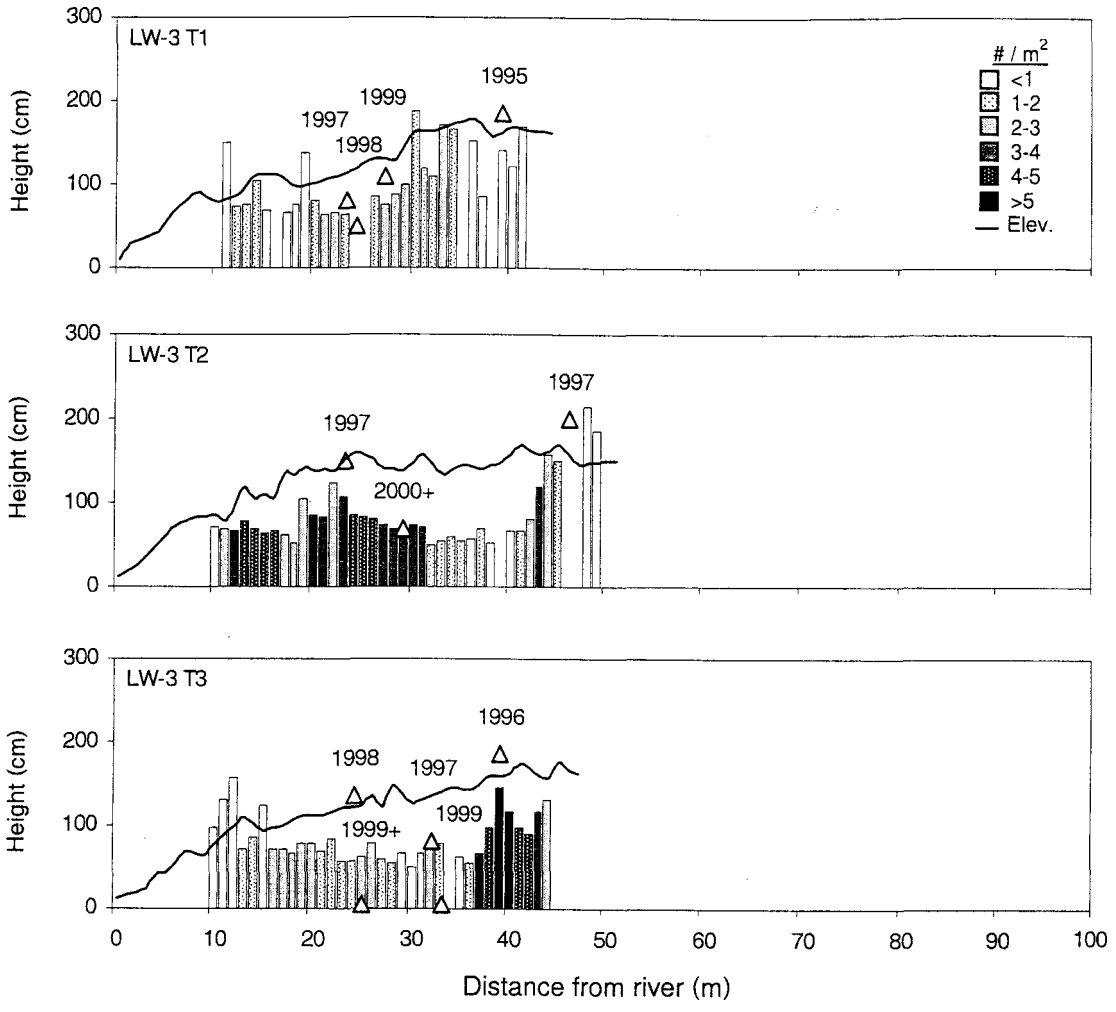

LSM-3 T1
1995
1996
1997
/ m²
<1
1-2
2-3
3-4
4-5
>5
Elev.
LSM-3 T2
1998
1996
1998
1997
LSM-3 T3
1997
1998
Height (cm)
0
100
200
300
0
10
20
30
40
50
60
70
80
90
100
Distance from river (m)

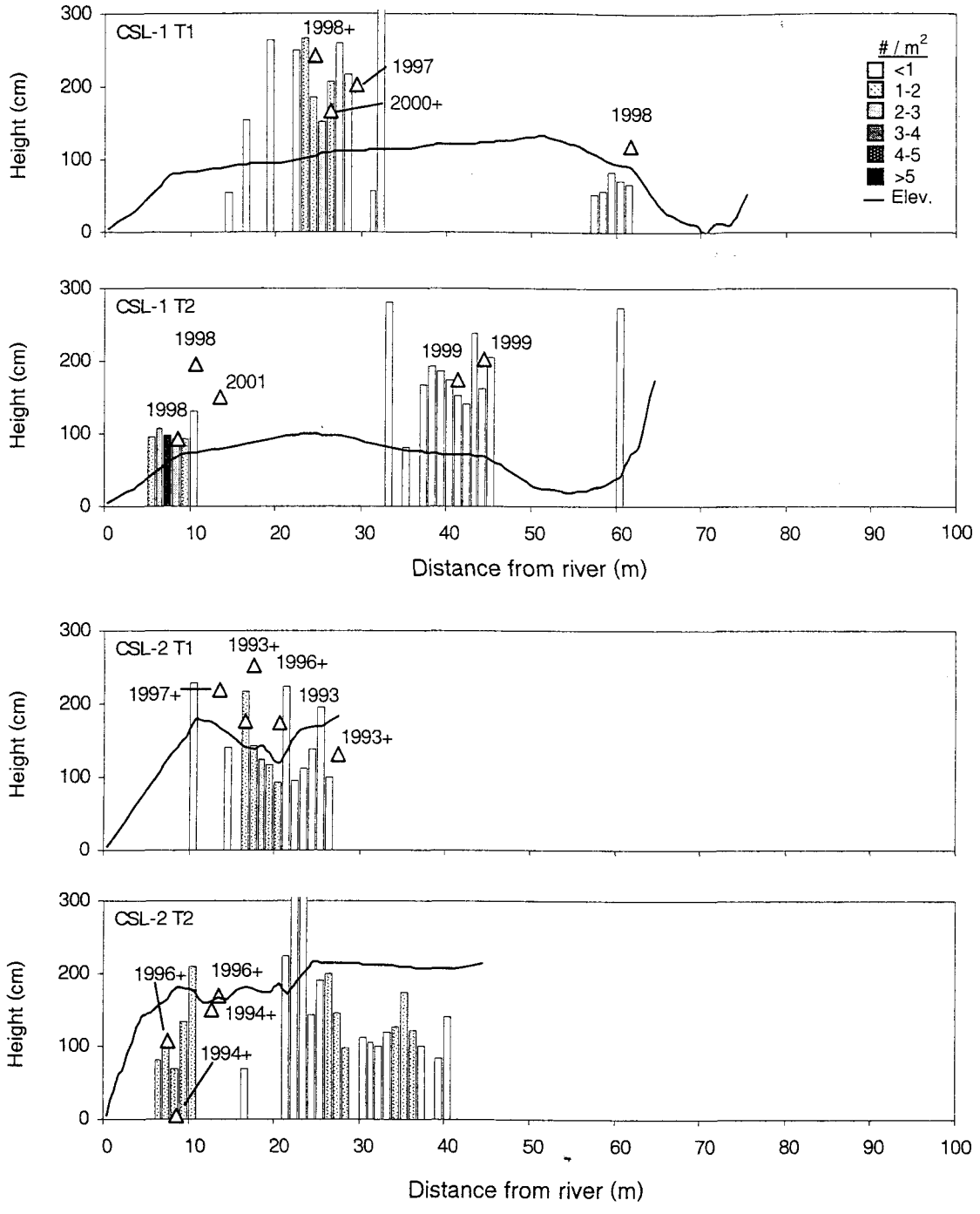

LSMP-1 T1
1997
1997
Height (cm)
0
100
200
300
0
10
20
30
40
50
60
70
80
90
100
Distance from river (m)


LSM/P-1 T2
2000+
1999+
1997
/ m2
<1
1-2
2-3
3-4
4-5
>5
Elev.
Height (cm)
LSM/P-1 T3
1999+
1999+
2000+
Height (cm)
0
10
20
30
40
50
60
70
80
90
100
Distance from river (m)
LSM/P-2 T1
1997
1997+
1999+
1997
Height (cm)
LSM/P-2 T2
1998
1997
1998+
2000+
1999+
1999+
Height (cm)
Distance from river (m)


LSMP-3 T1
1994+
1986+
1996
1997+
/ m2
<1
1-2
2-3
3-4
4-5
>5
Elev.
LSMP-3 T2
1998
1992+
1998
2000
1998
UW-1 T1
1997-1999
UW-1 T2
1999
Height (cm)
Distance from river (m)


LW-1 T1
/ m2
<1
1-2
2-3
3-4
4-5
>5
Elev.
1997
1998
1997
1998
LW-1 T2
1993
2001+
1995
Height (cm)
Distance from river (m)
LW-2 T1
1998
1997
1999+
LW-2 T2
1997
1999+
1999+
LW-2 T3
2000+f
1999f
1998+f
f = floodplain position estimated


LW-3 T1
LW-3 T2
LW-3 T3
Height (cm)
Distance from river (m)
/ m2
<1
1-2
2-3
3-4
4-5
>5
Elev.
1995
1997
1998
1999
2000+
1996
1999+



...End of Figure C-1

APPENDIX D
ADDITIONAL STATISTICS

Table D-1. Tests of normality for height (top) and diameter (bottom) residuals. Residuals (calculated as the difference between measured values and site means) were not normally distributed. Histograms showing the lack of normality are found in Figures D-1 and D-2.

	Kolmogorov-Smirnov			Shapiro-Wilk		
	Statistic	df	p	Statistic	df	p
USM-1	0.223	46	<0.001	0.834	46	<0.001
USM-2	0.125	207	<0.001	0.886	207	<0.001
LSM-1	0.084	101	0.078	0.966	101	0.010
LSM-2	0.133	88	0.001	0.886	88	<0.001
LSM-3	0.188	107	<0.001	0.837	107	<0.001
LSM/P-1	0.144	140	<0.001	0.928	140	<0.001
LSM/P-2	0.238	248	<0.001	0.614	248	<0.001
LSM/P-3	0.201	123	<0.001	0.844	123	<0.001
UW-1	0.158	81	<0.001	0.755	81	<0.001
LW-1	0.165	69	<0.001	0.810	69	<0.001
LW-2	0.084	137	0.020	0.945	137	<0.001
LW-3	0.145	428	<0.001	0.856	428	<0.001
CSL-1	0.144	140	<0.001	0.917	140	<0.001
CSL-2	0.169	75	<0.001	0.778	75	<0.001

	Kolmogorov-Smirnov			Shapiro-Wilk		
	Statistic	df	p	Statistic	df	p
USM-1	0.160	51	0.002	0.895	51	<0.001
USM-2	0.124	214	<0.001	0.903	214	<0.001
LSM-1	0.097	103	0.019	0.922	103	<0.001
LSM-2	0.122	88	0.003	0.908	88	<0.001
LSM-3	0.107	106	0.004	0.941	106	<0.001
LSM/P-1	0.175	139	<0.001	0.900	139	<0.001
LSM/P-2	0.238	265	<0.001	0.600	265	<0.001
LSM/P-3	0.206	126	<0.001	0.835	126	<0.001
UW-1	0.168	83	<0.001	0.809	83	<0.001
LW-1	0.135	71	0.003	0.863	71	<0.001
LW-2	0.153	158	<0.001	0.875	158	<0.001
LW-3	0.183	438	<0.001	0.759	438	<0.001
CSL-1	0.144	140	<0.001	0.849	140	<0.001
CSL-2	0.178	82	<0.001	0.836	82	<0.001

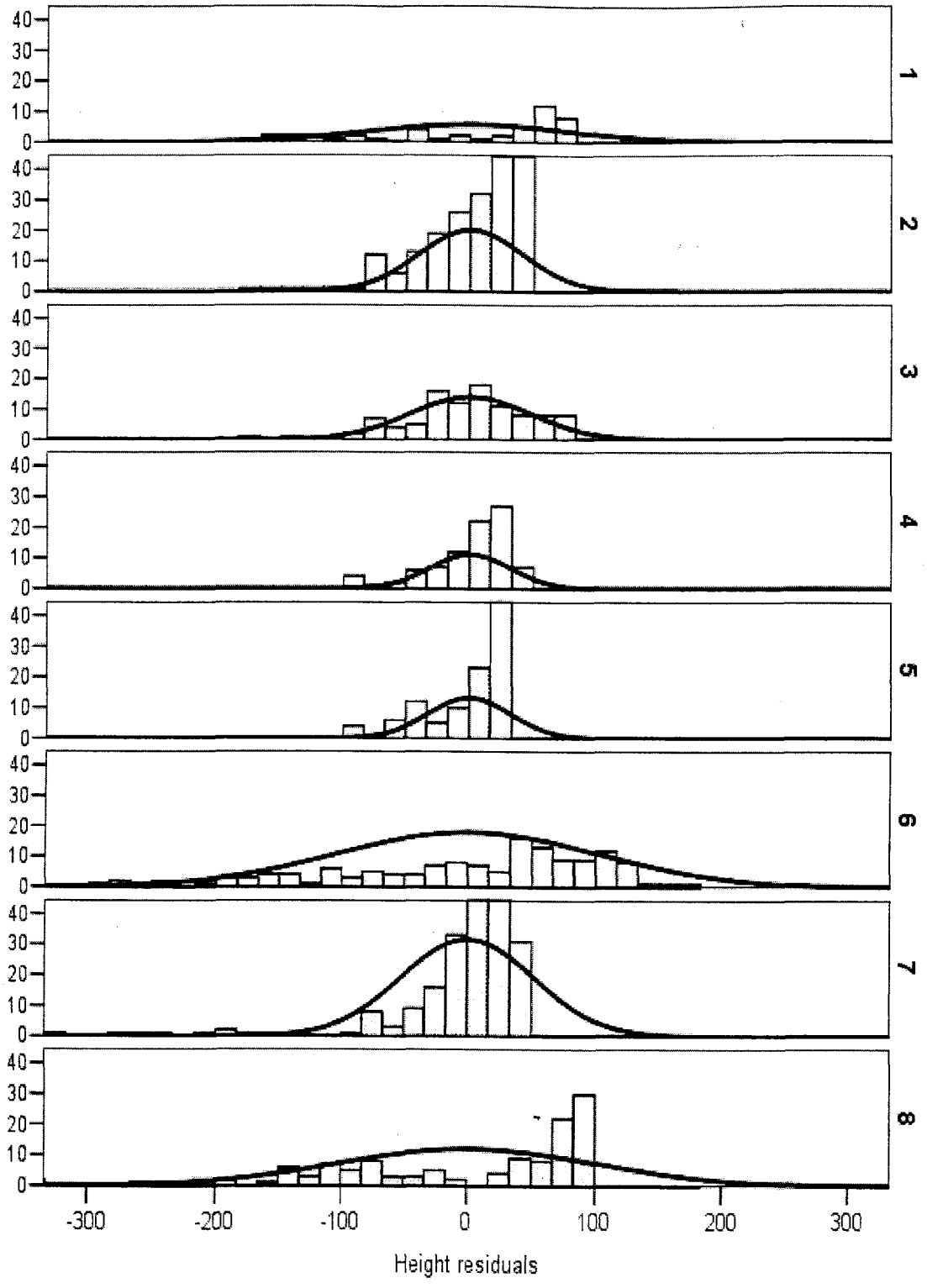
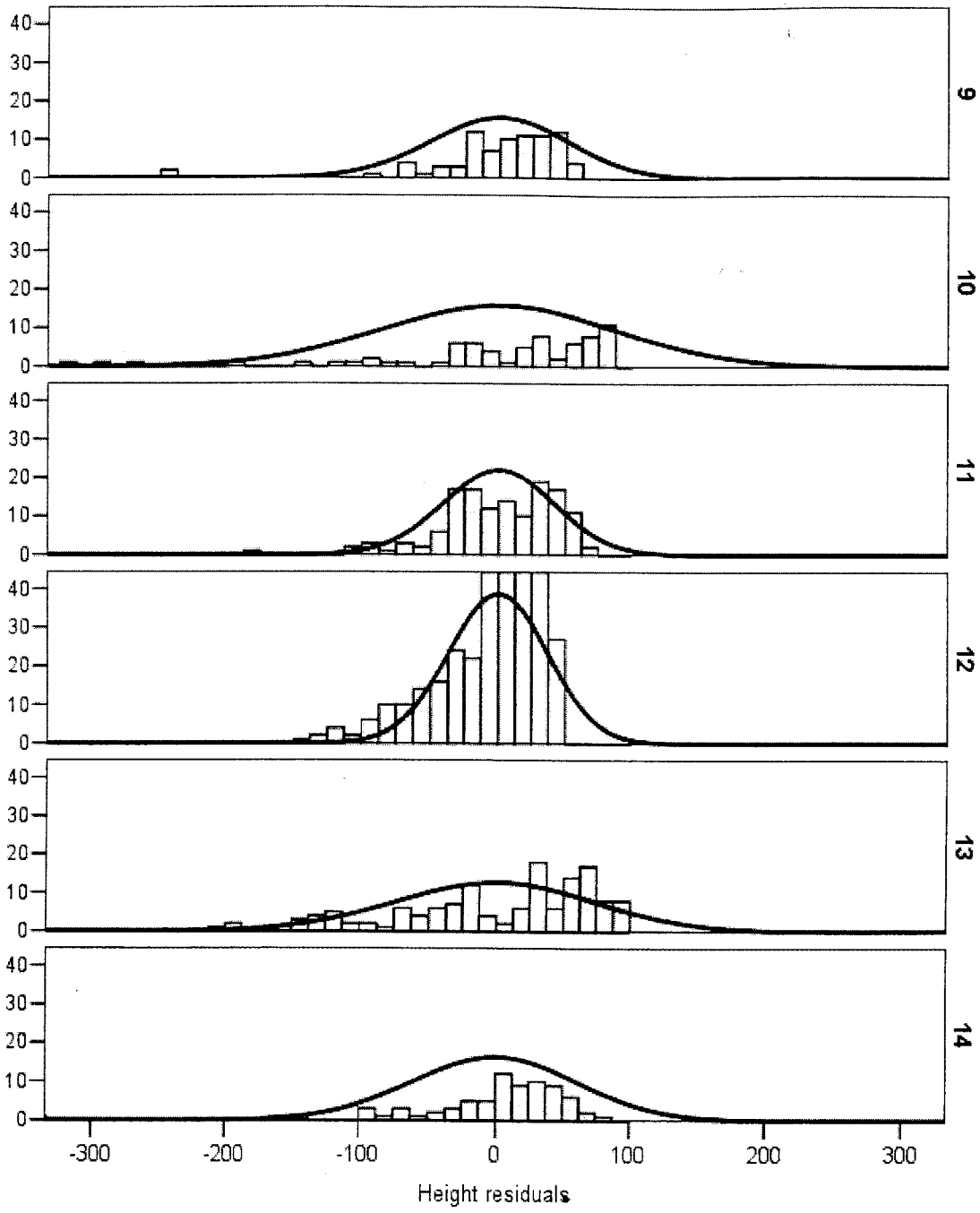


Figure D-1. Histogram of height residuals with normal distribution plotted for each study site. Most residuals were not normally distributed. Figure continues on next page.


40
30
20
10
0
9
10
11
12
13
14
-300
-200
-100
0
100
200
300
Height residuals

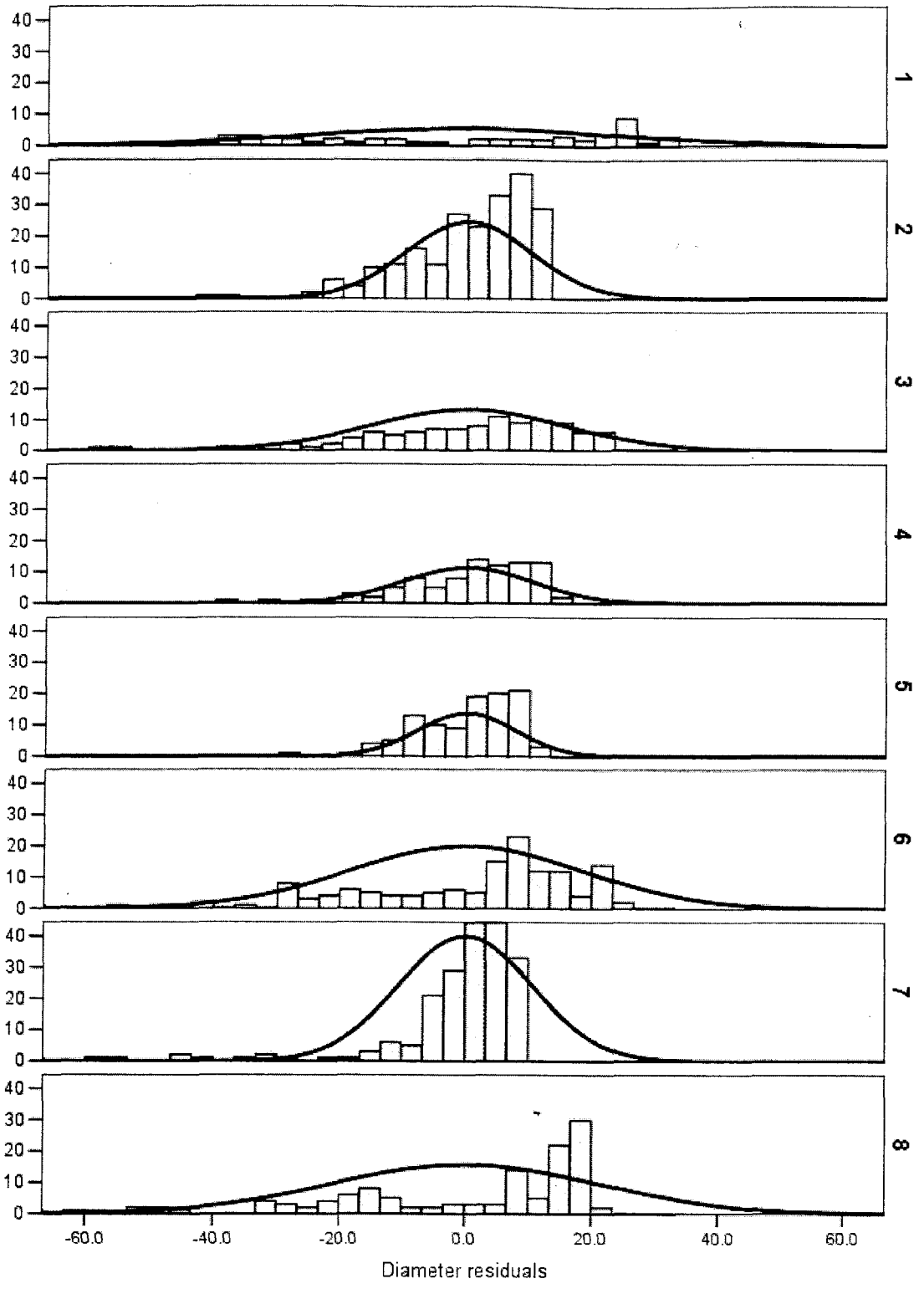


Figure D-2. Histogram of diameter residuals with normal distribution plotted for each study site. Residuals were not normally distributed. Figure continues on next page.

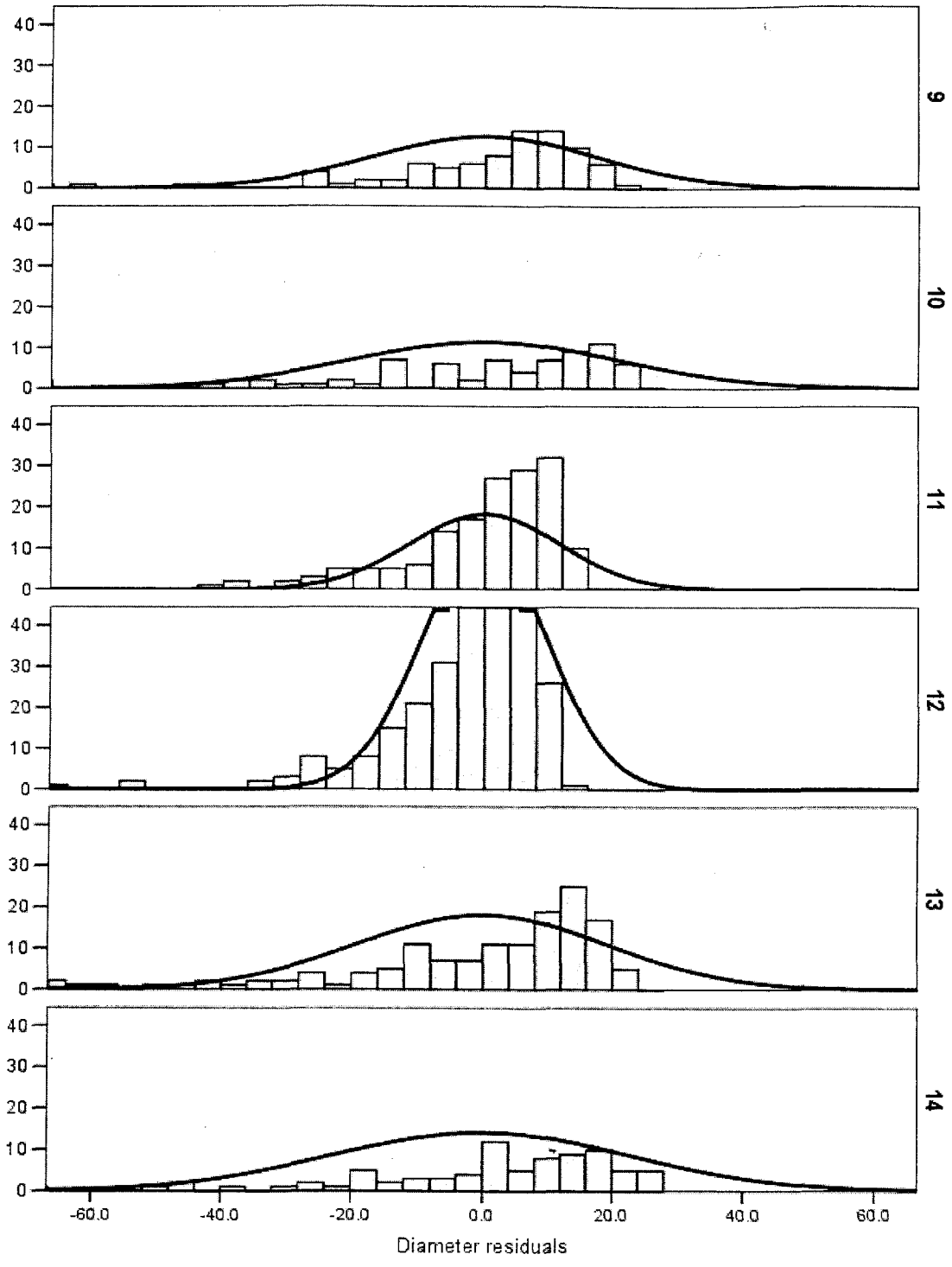

40
30
20
10
0
9
10
11
12
13
14
-60.0
-40.0
-20.0
0.0
20.0
40.0
60.0
Diameter residuals

Table D-2. Variation in sapling height distribution among transects within a site according to two-sample Kolmogorov-Smirnov tests. For sites with three transects, pair wise comparisons were done between each combination of transects. Total sample size (N), the test statistic (Z), and significance value (p) are presented.

Site	Transects		N	Z	p	Transects similar
USM-1	T1	T2	53	1.099	0.179	yes
USM-2	T1	T2	221	1.637	0.009	no
LSM-1	T1	T2	65	0.928	0.356	yes
	T1	T3	107	0.898	0.396	yes
	T2	T3	108	0.470	0.980	yes
LSM-2	T1	T2	157	0.906	0.384	yes
LSM-3	T1	T2	83	0.883	0.417	yes
	T1	T3	80	1.664	0.008	no
	T2	T3	59	1.392	0.042	no
LSM/P-1	T1	T2	106	1.921	0.001	no
	T1	T3	120	2.759	<0.001	no
	T2	T3	60	0.863	0.446	yes
LSM/P-2	T1	T2	266	2.250	<0.001	no
LSM/P-3	T1	T2	149	1.638	0.009	no
UW-1	T1	T2	83	0.699	0.713	yes
LW-1	T1	T2	90	0.632	0.819	yes
LW-2	T1	T2	95	1.399	0.040	no
	T1	T3	103	1.123	0.161	yes
	T2	T3	130	1.386	0.043	no
LW-3	T1	T2	222	1.273	0.078	yes
	T1	T3	164	0.958	0.317	yes
	T2	T3	278	1.571	0.014	no
CSL-1	T1	T2	141	1.795	0.003	no
CSL-2	T1	T2	86	0.522	0.948	yes

Table D-3. Differences in sapling height distribution among sites according to two-sample Kolmogorov-Smirnov tests. Tests were run between all sites, but only those results where significant differences were not detected are presented.

Site	Similar distribution to:	Z (test statistic)	P (sig. value)
USM-1	LSM/P-3	0.780	0.577
	LW-1	0.914	0.374
USM-2	LSM-2	0.891	0.405
	UW-1	1.109	0.183
LSM-1	LW-1	1.249	0.089
	CSL-2	1.016	0.253
LSM-2	LW-3	0.720	0.677
	USM-2	0.891	0.405
LSM-3	LSM/P-2	0.843	0.476
LSM/P-1	-		
LSM/P-2	LSM-3	0.843	0.476
	LW-3	1.329	0.058
LSM/P-3	LW-1	1.330	0.058
	USM-1	0.780	0.577
UW-1	USM-2	1.109	0.183
LW-1	CSL-1	1.095	0.182
	LSM/P-3	1.330	0.058
	LSM-1	1.249	0.089
	USM-1	0.914	0.374
LW-2	CSL-2	1.251	0.088
LW-3	LSM-2	1.016	0.253
	LSM/P-2	1.329	0.058
CSL-1	LW-1	1.095	0.182
CSL-2	LSM-1	1.016	0.253
	LW-2	1.251	0.088

Table D-4. Nested ANOVA testing for differences in sapling density at the transect, site, and reach levels, followed by student's-t post hoc tests for differences among reaches. Letters indicate reaches along which sapling densities were not significantly different.

Source	DF	SS	MS	F	p
Model	32	5473.4859	171.046	142.139	
river	5	900.106		14.2221	<0.001
site[river]	8	2853.06		25.6507	<0.001
transect[river,site]	19	304.619		13.323	<0.001
Error	2001	2407.9514	1.203		
Total	2033	7881.4373			

Reach	group
USM	bc
LSM	c
LSM/P	a
UW	bc
LW	b
CSL	b

APPENDIX E
CONSISTENT SAPLING GROWTH IN SECTION *TACAMAHACA* COTTONWOODS

Species effects were tested because seedlings of the different *Populus* species can have different growth rates both in the field and in a controlled laboratory setting (Kalishcuk et al. 2001, Kranjcec et al. 1998). Also, poplar hybrids have the potential for heterosis, although this phenomenon did not occur in 3-yr-old cuttings of the interspecific hybrids of southern Alberta (Campbell et al. 1993). That same study however, did show slower growth of *Populus angustifolia* compared to *P. trichocarpa* and *P. deltoides*, consistent with the other studies (Kalischuk et al. 2001, Kranjcec et al. 1998).

The most abundant species measured was *P. angustifolia*, followed by *P. trichocarpa* (Table E-1). *P. deltoides* and their hybrids were not numerous, had limited distribution, and were observed to have larger annual stem growth than their *P. angustifolia* neighbours, and so were eliminated from the sapling size data set. To compare *P. angustifolia*, *P. trichocarpa*, and their hybrids, USM-2 was the only site that had sufficient samples of each of the species (Table E-1) and was thus used for a size comparison. One site was selected to minimize the influence of environmental factors that would vary among sites. Because transects within a site can vary (Figure E-1), the USM-2 sample was further narrowed to an elevational range of 1.00 to 1.10 m in which all species groups were found.

Within the subset of saplings, there were no significant differences in height, diameter, or height to diameter ratio between the species categories (Figure E-1; ANOVA: height: $F = 0.512$, $df = 84$, $p = 0.675$; diameter: $F = 1.037$, $df = 94$, $p = 0.380$; H/D: $F = 1.575$, $df = 84$, $p = 0.202$). This means that all saplings in the section *Tacamahaca* (i.e. POan, POtr, and intrasectional hybrids) were used in the sapling size analyses.

Table E-1. Number of saplings in each species category by site. *Populus angustifolia* is the most dominant species at all sites except for the two sites on the Castle reach. USM-2 (in bold) was used for size comparison between *P. angustifolia*, *P. trichocarpa*, and their hybrids.

	Number of saplings per species per site							
	POan	POan-x-POtr	POan-x-POde	POtr	POtr-x-POan	POde	POde-x-POan	POan-x-?
USM-1	12	30		6	5			
USM-2	**135**	**39**		**21**	**20**			
LSM-1	108			1	2			
LSM-2	97	4						
LSM-3	101	5		1	3			
LSM/P-1	127	6	2		4		3	2
LSM/P-2	216		18		1	15	5	11
LSM/P-3	55		66			6	5	
UW-1	53	7		3	20			
LW-1	73	5			2			
LW-2	151			1				11
LW-3	427	6		4	1			
CSL-1		3		138				
CSL-2	1			84				
proportion	0.73	0.05	0.04	0.12	0.03	0.01	0.01	0.01

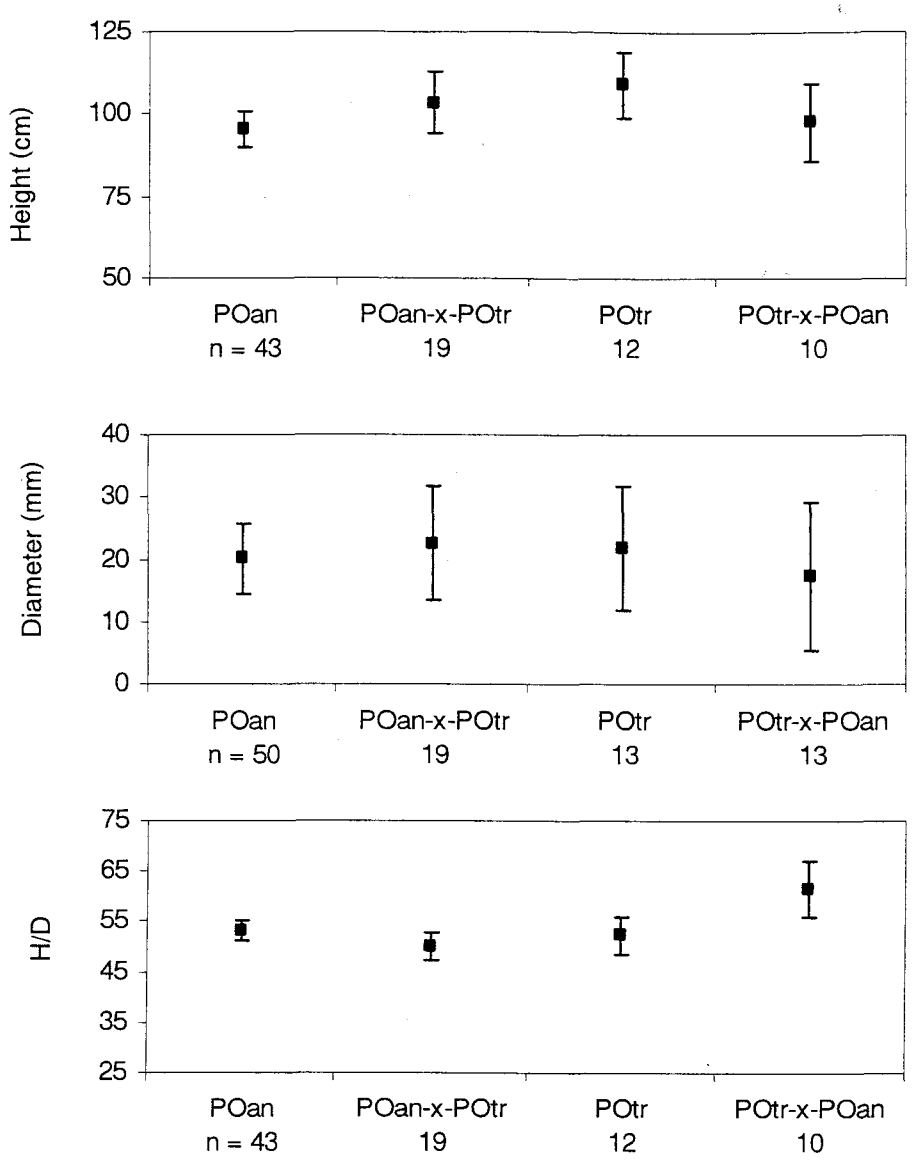


Figure E-1. Mean sapling size (± SE) comparison among the cottonwood species found in an elevation band of 1.00 to 1.10 m at USM-2. There were no significant differences in height, diameter, or height to diameter ratio among saplings identified as *Populus angustifolia* (POan), *P. trichocarpa* (POtr), and their hybrids.

APPENDIX F
GROWTH FORM DIFFERENCES IN SAPLING SIZE

The mean diameter and H/D of clumps was not significantly different than single stems (Table F-1; diameter: $t = -0.471$, $df = 2052$, $p = 0.683$; H/D: $t = -0.809$, $df = 1867$, $p = 0.418$), but the mean height of clumps was significantly larger than single stems at $\alpha = 0.10$ (Table F-1; $t = -1.924$, $df = 1877$, $p = 0.055$). However, at some study sites, single stems were larger and at other sites, the heights were approximately the same (Figure F-1). Therefore, saplings of both growth forms were pooled for the analyses.

Table F-1. Mean size (±SE) of single stemmed and clumped saplings. Heights of the two different growth forms were significantly different at $\alpha = 0.10$, but diameter and H/D were not significantly different.

	Single stems	Clumps
Height[t]	113 ± 2	120 ± 3
Diameter	23.8 ± 0.4	24.1 ± 0.6
H/D	53.5 ± 0.5	54.1 ± 0.7

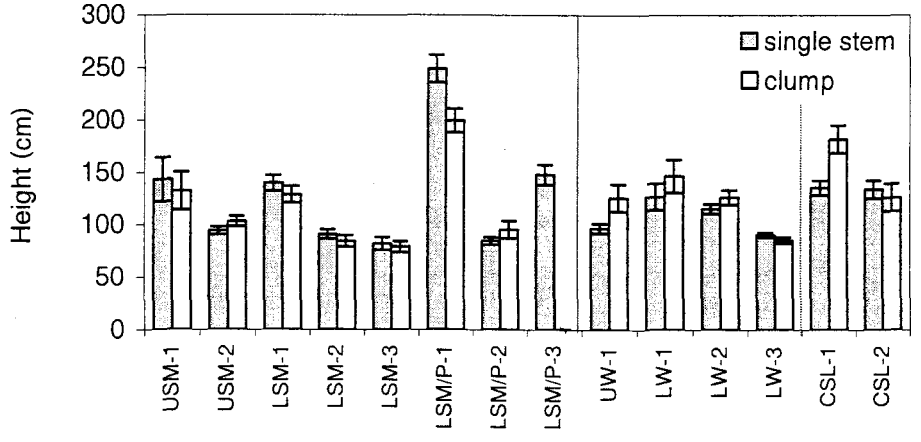


Figure F-1. Comparison of site mean heights between single stemmed and clumped saplings. Although clump heights were significantly larger than single stems overall, at some sites single stems were taller or equivalent to the clumped saplings.