Geomorphology 102 (2008) 445–459

Contents lists available at ScienceDirect

Geomorphology

journal homepage: www.elsevier.com/locate/geomorph

Riparian reforestation and channel change: A case study of two small tributaries to Sleepers River, northeastern Vermont, USA

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ARTICLE INFO

Article history:
Received 28 August 2007
Received in revised form 3 May 2008
Accepted 5 May 2008
Available online 16 May 2008

Keywords:
Riparian reforestation
Stream widening
Channel morphology
Large woody debris
Conceptual model

ABSTRACT

Measurements of two small streams in northeastern Vermont, collected in 1966 and 2004–2005, document considerable change in channel width following a period of passive reforestation. Channel widths of several tributaries to Sleepers River in Danville, VT, USA, were previously measured in 1966 when the area had a diverse patchwork of forested and nonforested riparian vegetation. Nearly 40 years later, we remeasured bed widths and surveyed large woody debris (LWD) in two of these tributaries, along 500 m of upper Pope Brook and along nearly the entire length (3 km) of an unnamed tributary (W12). Following the longitudinal survey, we collected detailed channel and riparian information for nine reaches along the same two streams. Four reaches had reforested since 1966; two reaches remained nonforested. The other three reaches have been forested since at least the 1940s. Results show that reforested reaches were significantly wider than as measured in 1966, and they are more incised than all other forested and nonforested reaches. Visual observations, cross-sectional surveys, and LWD characteristics indicate that reforested reaches continue to change in response to riparian reforestation. The three reaches with the oldest forest were widest for a given drainage area, and the nonforested reaches were substantially narrower. Our observations culminated in a conceptual model that describes a multiphase process of incision, widening, and recovery following riparian reforestation of nonforested areas. Results from this case study may help inform stream restoration efforts by providing insight into potentially unanticipated changes in channel size associated with the replanting of forested riparian buffers adjacent to small streams.

1. Introduction

Restoration of streamside forests is a major focus of watershed initiatives throughout the United States (National Reseach Council, 1992; U.S. Environmental Protection Agency, 1999) and commonly accompanies in-stream restoration efforts. One example is the Chesapeake Bay Program's initiative to restore riparian forests along 3234 km (2010 mi) of stream by the year 2010 (Palone and Todd, 1997). When the Bay Program achieved this goal ahead of schedule in 2002, the goal was increased to 16,093 km (10,000 mi) by 2010 (Chesapeake Bay Program, 2007). Similarly, the federal Conservation Reserve Enhancement Program (CREP), a voluntary agricultural land retirement program, has provided incentives leading to over 3200 km^2 of replanted riparian buffers nationwide since 1997 (U.S. Department of Agriculture, 2007). Stream restoration projects aimed at reconfiguring channels, protecting streambanks and infrastructure, or in-stream habitat modifications often include riparian plantings to supplement these activities (Bernhardt et al., 2005). Indeed, riparian forest restoration has been identified as a successful tool for improving stream ecosystems through filtering of pollutants (Herson-Jones et al., 1995; Berg et al., 2003); regulating nutrients, light, and temperature; and providing both physical habitat and the food/energy base (Gregory et al., 1991; Sweeney, 1992; Berg et al., 2003; Sweeney et al., 2004). Despite these benefits, the best treatment for streambanks remains unresolved because of the uncertain role of riparian vegetation in affecting streambank erosion rates and processes (Montgomery, 1997; Trimble, 1997; Lyons et al., 2000).

Riparian vegetation exerts important influences on stream channel morphology (Thorne, 1990; Montgomery, 1997; Anderson et al., 2004). Riparian vegetation is highly varied, hindering simple quantification; however, many studies have shown distinct differences in morphology resulting from two broad types of vegetation: forests and grasslands. Most studies from widely different geographic locations indicate that stream reaches with riparian forests are wider than those with adjacent grassy vegetation (Zimmerman et al., 1967; Murgatroyd and Ternan, 1983; Clifton, 1989; Sweeney, 1992; Peterson, 1993; Davies-Colley, 1997; Trimble, 1997; Scarsbrook and Halliday, 1999; Hession et al., 2000, 2003; Sweeney et al., 2004; Allmendinger et al., 2005; Roy et al., 2005). In contrast, a few other studies suggest that widths of streams through grassland are generally greater than those through forest (Charlton et al.,

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doi:10.1016/j.geomorph.2008.05.008

1978; Hey and Thorne, 1986; Gregory and Gurnell, 1988; Rosgen, 1996). The aforementioned studies have categorized riparian vegetation in unique ways, so it can be challenging to make generalized conclusions. These apparently contrary findings may be partially explained by a scale-dependent effect of riparian vegetation. Small forested streams are wider than small nonforested streams; whereas in large streams with drainages greater than 10 to 100 km^2, an opposite effect is observed such that channel widths are narrower when thick woody vegetation is present (Anderson et al., 2004).

Ideas about the geomorphic processes responsible for the observed channel width differences are varied and remain untested. Zimmerman et al. (1967) attributed the wider channels in forested stream reaches to local scouring around LWD, debris dams, or at streamside tree-throw locations. They surmised that grass roots in nonforested reaches reinforced streambanks and encroached more rapidly on the channel during periods of low flow than roots from trees or understory plants in the forest (Zimmerman et al., 1967). Subsequently, Murgatroyd and Ternan (1983) found more active bank erosion in forested reaches than in nonforested reaches, which they attributed primarily to the loss of a thick grass turf and secondly to scour around log jams and debris dams. Davies-Colley (1997) hypothesized that when a riparian forest is cleared, grasses are able to grow on the gravel bars in the channel, making the bars more stable and narrowing the channel. In the same year, Trimble (1997) proposed that wider forested channels were unstable and that narrower nonforested channels more efficiently trapped sediment causing channel narrowing. The influence of riparian vegetation on streambank stability is complex, highly variable, and a broader topic with its own body of literature (Gray and MacDonald, 1989; Thorne, 1990; Dunaway et al., 1994; Montgomery, 1997; Stott, 1997; Abernethy and Rutherfurd, 1998; Simon and Collison, 2002; Pollen et al., 2004; Rutherfurd and Grove, 2004; Wynn and Mostaghimi, 2006). Recently, Allmendinger et al. (2005) found higher rates of deposition and lateral migration in nonforested reaches than in forested reaches, and they suggested that the differences in width between forested and nonforested reaches were related to a balance between rates of erosion and deposition on active floodplains. In addition, results from a flume study suggest that near-bank turbulence during overbank flows may be an important process in channel widening (McBride et al., 2007).

Clear evidence for channel widening or narrowing with a change in riparian vegetation is difficult to obtain as such processes likely operate on a timescale greater than the length of a typical research study, thus limiting field-based research opportunities. Long-term channel change in response to riparian vegetation change in small streams has been documented in a few cases (Clifton, 1989; Parkyn et al., 2003; McBride et al., 2005). Most studies must rely on the space-for-time substitution and attempt to find paired sites that have similar background characteristics (Davies-Colley, 1997; Huang and Nanson, 1997; Stott, 1997; Trimble, 1997; Scarsbrook and Halliday, 1999; Hession et al., 2003; Parkyn et al., 2003; Sweeney et al., 2004; Allmendinger et al., 2005; McBride et al., 2005).

In this paper we expand on a preliminary effort to compare historic and current channel-size data from the Sleepers River Research Watershed (SRRW) in Danville, VT (McBride et al., 2005). One of the first studies to document the influence of riparian forests on channel form was completed within the SRRW by Zimmerman et al. (1967). We revisited two of the streams described in the Zimmerman et al. (1967) study to assess potential changes in channel dimensions and LWD characteristics in response to riparian reforestation. Our objectives were (i) to evaluate the extent of riparian reforestation along the two study streams; (ii) to determine differences in bed widths since 1967; and (iii) to investigate differences in channel dimensions and LWD as related to current riparian vegetation. Based on our results, we present a conceptual model that describes the process of channel adjustment following the introduction of forested riparian vegetation over time.

2. Study area

The SRRW is one of the longest-running, cold-region research watersheds in the United States (Fig. 1); hydrologic data have been collected continuously since 1958 (Pionke et al., 1986). The site is currently administered by the U.S. Geological Survey (USGS). Sleepers River drains 111 km^2 that is predominantly forested (~67%); agriculture and rural residences are the other common land uses (Shanley et al., 1995). The SRRW contains rolling topography at elevations between 201 to 780 m, and it is underlain with 1 to 20 m of calcareous till atop the Waits River Formation bedrock, a metamorphosed limestone (Shanley et al., 1995). The mean annual temperature is 6 °C, and mean annual precipitation is 90 cm (Shanley et al., 1995). We studied two streams: (i) stream W12 is an unnamed tributary to Sleepers River that drains a 2.1-km^2 mixed land-use drainage; and (ii) upper Pope Brook, another tributary, is a headwater stream that drains a 1.1-km^2 forested area within State of Vermont forest lands. Forests are predominantly mixed with both coniferous and deciduous trees. The most common species are northern white cedar (*Thuja occidentalis*), yellow birch (*Betula alleghaniensis*), white spruce (*Picea glauca*), balsam fir (*Abies balsamea*), and sugar maple (*Acer saccharum*).

3. Methods

3.1. Field methods

We conducted a longitudinal survey of W12 and portions of upper Pope Brook, revisiting the field sites of Zimmerman et al. (1967) during the summer of 2004. We excluded two segments of W12 where cattle had access to the stream and riparian areas. We measured bed widths every 8 m to replicate the measurements made by Zimmerman et al. (1967). In that study, widths were measured “between breaks-in-slope between bed and bank” (Zimmerman et al., 1967); likewise, we measured widths at the edge of the bed, essentially capturing the active channel width, or bed width. We also collected information on LWD during the longitudinal survey. For each piece of LWD > 10 cm in diameter and 1 m in length (Montgomery et al., 1995; Jackson and Sturm, 2002), we recorded its location, diameter, length, and decay class. Decay class describes the age of the LWD on a scale from 1 to 6, where 1 is a recently recruited piece of LWD with leaves or needles still present, and 6 is an old, weathered piece of LWD (Martin and Benda, 2001). Debris dams were also located and tallied.

In the summer of 2005, we conducted detailed channel and riparian surveys at nine reaches within the stream sections previously surveyed. Seven of the reaches were in W12: two nonforested sites (NF1 and NF2); one forested site (F1); and four reforested sites (R1–R4). Reforested sites were located in segments of W12 that were identified as nonforested in 1966 but have reforested in the last 40 years. The other two forested reaches were in upper Pope Brook (F2 and F3). All reaches were 75 m long, a length equivalent to approximately 20 bankfull widths. In each reach we completed a detailed cross-sectional survey of the channel at a characteristic riffle feature, extending to the floodplain or valley slope adjacent to each streambank. Bankfull elevations were determined considering a combination of the presence or absence of perennial vegetation, topographic breaks in the bank, and a change in sediment texture or size (Dunne and Leopold, 1978). We randomly chose a 10-m^2 plot, as recommended by Kent and Coker (1992), along either the right or left bank of each reach, within which we measured the diameter at breast height (DBH) of every tree stem > 1.0 cm in diameter. Within the plot, we also tallied and measured the diameter and length of each woody debris piece on the floodplain using the same size criteria as the in-stream LWD. To estimate forest age at reaches F1, F2, and F3, we cored the largest tree within the riparian zone of the reach with an increment tree borer.

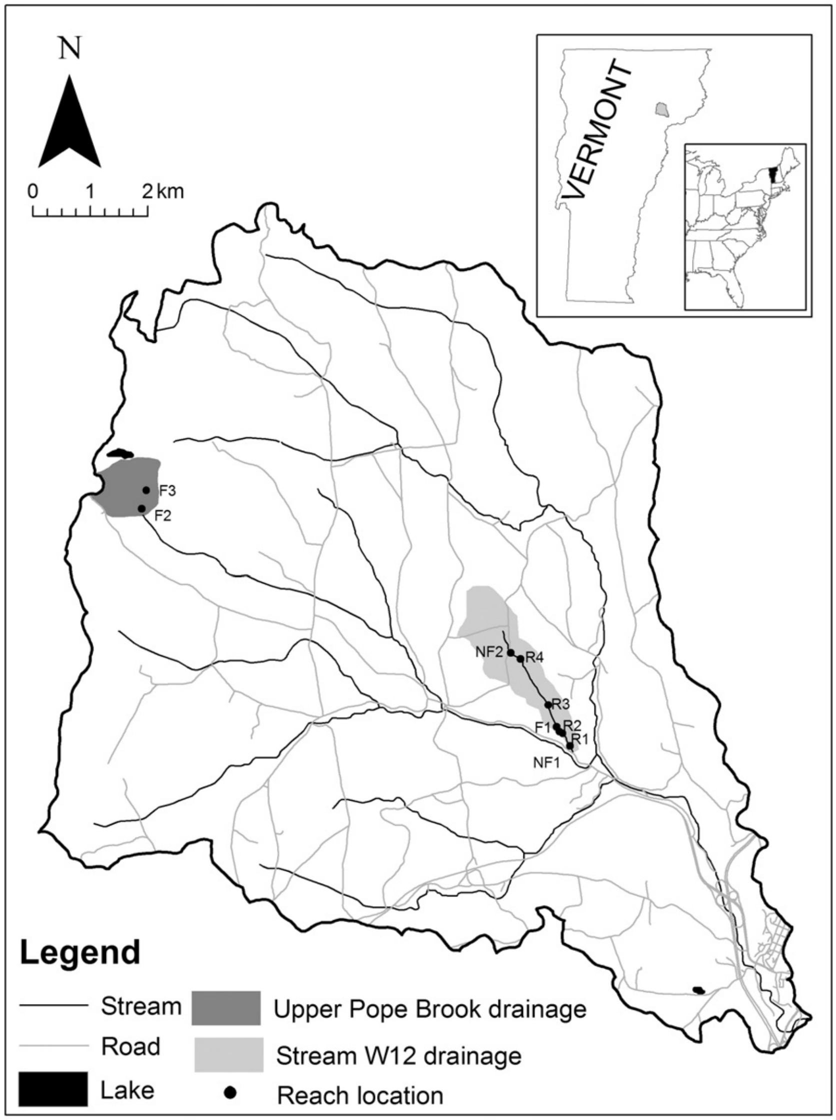

Fig. 1. Location of the SRRW showing drainages and reaches of W12 and upper Pope Brook.

3.2. Spatial data methods

To assess quantitatively the change in riparian vegetation, we assembled, scanned, and georeferenced two historic aerial photographs. The earliest aerial photograph available was from 1943 (1:1000). An aerial photograph from 1965 (1:10,000) was nearly concurrent with the Zimmerman et al. (1967) data. The 1943 and 1965 aerial photographs were georeferenced to the most current orthophotograph (1999) using ArcMap software (ESRI, Redlands, CA). Once georeferenced, we overlaid the aerial photographs to determine areas of reforestation.

3.3. Analytical methods

Data collected in the Zimmerman et al. (1967) study included channel dimensions (bed widths and occasional depths) and riparian vegetation type for approximately five different tributaries to Sleepers River. Individual width measurements could be determined from graphs in the Zimmerman et al. (1967) paper for two of the study streams: a 500-m segment of upper Pope Brook and the entire length (3 km) of stream W12. Zimmerman et al. (1967) specifically selected stream segments to avoid confounding disturbances such as backwater from weirs, trampling from cattle, and other human-caused alterations. The Zimmerman et al. (1967) study classified the riparian condition of different sections of W12 as either sod (i.e., grass), thicket, or forest. Data categorized as “thicket” were excluded from our analysis because this vegetation type was not well described.

As a precursor to statistical testing of the bed-width data, we checked the distribution and created a testable subset of the data. We created a subset of the past and current bed-width measurements by using the bed-width data points falling within the nine detailed reaches of this study. This resulted in ~10 data points per reach for a total of 79 data points for 1967 and 87 data points for 2004. One reach (NF1) was increased to 140 m in length to include a sufficient number of data points from the Zimmerman et al. (1967) data. We found the bed-width data (past and present) to be approximately normal by assessing normal probability plots, which allowed for parametric testing (Zar, 1999).

Differences in mean bed width over nearly 40 years of time (between 1966 and 2004) were tested using a series of two-sample t tests and assessed using a percent difference. A total of nine t tests were performed on the seven W12 reaches and the two upper Pope Brook reaches. Homogeneity of variance was tested in all cases using Levene's test (SAS, 2004). In one case where homogeneity of variance

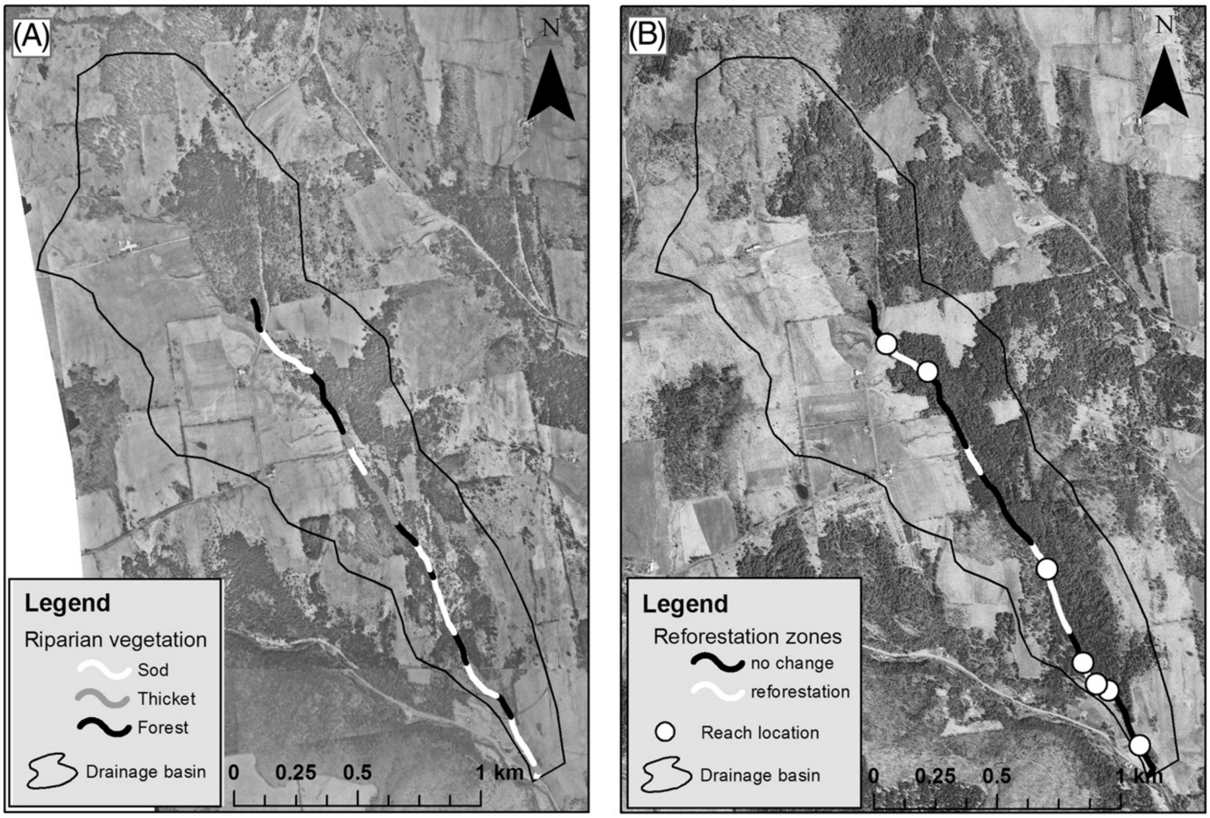

Fig. 2. Aerial photographs of W12 in 1965 (A) with the riparian classifications of Zimmerman et al. (1967) and in 1999 (B) with reforested segments highlighted and sample reach locations identified.

was rejected (reach R4), we used the Satterthwaite version of the two-sample t test (SAS, 2004). Percent differences (PD) between 1966 and 2004 data were determined as

$$PD = (W_{2004} - W_{1966}) / W_{1966} \times 100, \tag{1}$$

where W_{2004} is the mean bed width as measured in 2004, and W_{1966} is the mean bed width as measured by Zimmerman et al. (1967). We tested whether the PD of reforested reaches were greater than other reaches using the nonparametric, one-tailed Mann–Whitney test (Zar, 1999).

Key parameters were identified from the LWD data, cross-sectional surveys, and riparian vegetation plots. To describe the LWD characteristics of each reach, we determined total counts and volumes. The decay classes were combined for the three forested reaches and the four reforested reaches to assess the frequency distributions. Bankfull channel width, cross-sectional area, mean depth, and width-to-depth ratio were determined for each reach. For each riparian plot, we determined summary statistics for the trees and debris measured, including counts, mean DBH, and maximum DBH. Forest ages of the riparian zones of F1, F2, and F3 were estimated by counting the tree rings present in the increment bores.

Channel dimension data from previously published, paired forested and nonforested studies were also compiled to produce hydraulic geometry relationships between bankfull width, bankfull depth, and drainage area (Davies-Colley, 1997; Hession et al., 2003). Because the largest drainage area of our study was 2.06 km^2, we selected data from previous studies for reaches that had drainage areas smaller than 3 km^2. The drainages from the previous studies had land cover that was similar to the SRRW with mixed agricultural and forest land cover (Davies-Colley, 1997; Hession et al., 2003). Climate data indicates that the SRRW is colder and dryer than the drainages in the mid-Atlantic states (Hession et al., 2003) or in northern New

Fig. 3. Photographs of nonforested reaches NF1 (left) and NF2 (right) of W12.

Fig. 4. Photographs of reforested reaches R1 (A), R2 (B), R3 (C), and R4 (D) of W12.

Zealand (Davies-Colley, 1997). The Hession et al. (2003) drainages have a mean annual precipitation of 117 cm; and in nearby Wilmington, DE, the mean annual temperature is 12.4 °C (NCDC, 2006). The Davies-Colley (1997) drainages are closest to the city of Hamilton, which receives 119 cm of annual precipitation and the mean annual temperature is 13.7 °C (National Institute of Water & Atmospheric Research (NIWA), 2007). Eighteen reaches from the two studies were used to create the hydraulic geometry relationship between bankfull width and drainage area. Only seven reaches from the Hession et al. (2003) study were similarly used for the bankfull depth relationship, as channel depths were not collected by Davies-Colley (1997).

4. Results

4.1. Riparian reforestation

Substantial riparian reforestation has taken place adjacent to W12 since the Zimmerman et al. (1967) study (Fig. 2). In 1966, W12 had ~50% of its total stream length bordered by sod or nonforested vegetation (Zimmerman et al., 1967), while in 1999 only 22% of the stream remained bordered by nonforested vegetation. Almost 1 km of W12's riparian zone has reforested within the last 40 years. Only two sections of W12 remain nonforested (Fig. 3) because of mowing practices of the landowners, and in these sections nearby trees were at least 5 m from the stream's edge. Interpretation of the aerial photographs indicated that riparian forests at reaches F1, F2, and F3 were established prior to 1943, indicating a forest age of at least 60 years.

The riparian forest at upper Pope Brook (F2 and F3) is likely older than the riparian forest surrounding reach F1 along W12 based on the tree-coring results. A yellow birch with a 50-cm DBH in reach F2 was dated as ~85 years old, and a balsam fir with a 35-cm DBH in reach F3 was ~60 years old. The largest tree near reach F1 was a white pine (*Pinus strobus*) with a 50-cm DBH, but the estimated age was only ~50 years.

Visual observations and riparian plot measurements highlight differences between the forest vegetation at reach locations. All reforested reaches appeared to have immature forested vegetation both from our visual observations (Fig. 4) and the plot measurements. Forest vegetation in reforested reaches appeared more homogeneous than forested reaches, and this observation was confirmed by the lower standard deviations of the DBH measurements (Table 1). Commonly, we observed "wolf" trees in the reforested reaches, providing evidence for the previously nonforested riparian zones. A "wolf" tree differentiates itself from other trees by its large size and horizontal branches, because it once grew in an open area without competition from other trees (Spirn, 1998). The characteristics of the reforested reaches were variable and unique to each reach: reach R1's floodplain was densely populated with moderate-sized trees with very little woody debris on the floodplain; reaches R2 and R3 had the smallest trees and woody debris of all reaches; and reach R4 had few

Table 1
Riparian vegetation plot measurements

Reach	Stream	Stem count	DBH (cm)	DBH_{max} (cm)	Debris count	Debris size (cm)	Debris $size_{max}$ (cm)
NF1	W12	0	na	na	0	na	na
NF2	W12	0	na	na	0	na	na
R1	W12	56	11.8 (8.5)	36.0	0	na	na
R2	W12	30	9.0 (7.6)	32.6	4	4.4 (3.6)	5.2
R3	W12	28	8.6 (6.1)	27.9	16	4.7 (1.3)	10.5
R4	W12	6	10.7 (6.3)	16.2	8	24.9 (6.9)	42.0
F1	W12	33	11.3 (9.4)	34.5	12	12.2 (3.0)	42.0
F2	Pope	15	17.4 (10.5)	32.5	7	17.8 (10.5)	33.0
F3	Pope	31	12.1 (8.9)	27.0	6	12.1 (12.9)	37.6

DBH: mean diameter at breast height of all tree stems, standard deviation in parentheses; DBH_{max}: maximum diameter at breast height; debris size: mean diameter of woody debris at the midpoint, standard deviation in parentheses; debris $size_{max}$: maximum diameter of woody debris.

Fig. 5. Photographs of forested reaches F1 (A) of W12, and F2 (B) and F3 (C) of upper Pope Brook.

trees, but large-sized woody debris on the floodplain. We observed grassy, herbaceous vegetation along portions of the streambanks of R2, R3, and R4, but mosses were the predominant ground cover in R1. The three forested reaches (Fig. 5) where the forest vegetation was >60 years old had trees with greater mean DBH (13.6 cm) than the more recently reforested reaches (10.0 cm); however, this difference was not statistically significant.

4.2. Bed width

The reforested reaches of W12 exhibited the greatest increase in mean bed width between 1966 and 2004; however, all reaches, except F2, were significantly wider in 2004 than in 1966 (Table 2; Fig. 6). Among all reaches evaluated, the increase in mean bed width ranged from 9% to 376%. Three of the four reforested reaches in W12 (reaches R1, R2, and R4) experienced the greatest change in mean bed width, as shown by the three highest PD (231%, 263%, and 376%, respectively). When the reforested reaches were compared against all other reaches, they had significantly greater PD (p=0.05). Mean bed widths of F2 in upper Pope Brook exhibited the least change in the last 40 years, and current mean values (2004) were not significantly different from the mean values derived from the Zimmerman et al. (1967) estimates. In general, bed widths were more variable in 2004 than those measured in 1966; the standard deviation of bed widths in most reaches is greater for the 2004 data.

Table 2
Current and past bed-width measurements

Reach	Stream	n		Mean bed width (m)		Percent difference
		1967	2004	1967	2004	
NF1	W12	6	10	**0.81** (0.22)	**1.19** (0.23)	47%
NF2	W12	6	7	**0.67** (0.12)	**1.11** (0.31)	66%
R1	W12	9	10	**0.70** (0.38)	**2.32** (0.33)	231%
R2	W12	10	10	**0.65** (0.30)	**2.36** (0.42)	263%
R3	W12	9	10	**1.24** (0.34)	**2.06** (0.68)	66%
R4	W12	8	10	**0.51** (0.27)	**2.43** (0.74)	376%
F1	W12	11	10	**1.23** (0.58)	**2.69** (0.70)	119%
F2	Pope	10	10	1.70 (0.77)	1.86 (0.41)	9%
F3	Pope	10	10	**1.42** (0.33)	**2.09** (0.43)	47%

Bold mean values are significantly different with α=0.01. Standard deviations are in parentheses. Reach name indicates riparian vegetation type.

4.3. Large woody debris

LWD was more abundant in the forested reaches than the reforested reaches and was virtually absent from the nonforested reaches (Table 3). Forested reaches had 10.7 pieces of LWD per 75-m reach on average, while reforested reaches had only 6.8 pieces. In contrast, average LWD volume was not significantly different between forested and reforested reaches, highlighting that LWD pieces in reforested reaches were larger. Forested and reforested reaches were also similar in the number of debris dams and the volume of LWD in debris dams. LWD pieces in forested reaches were classified in the oldest decay classes, but LWD in reforested reaches were fairly evenly distributed throughout the six decay classes (Fig. 7). In general, reforested reaches had LWD that was more recently recruited to the stream.

4.4. Current bankfull channel dimensions

Bankfull channel dimensions were a function both of drainage area and riparian vegetation type. Although the drainage areas of the reaches only ranged from 0.55 km^2 to 2.06 km^2, the range in bankfull widths was more than an order of magnitude (Table 4). Differences in channel size and undercut banks in nonforested, reforested, and forested reaches are displayed in Figs. 8, 9, and 10, respectively. Bankfull widths and bankfull depths were each plotted with hydraulic geometry relationships developed for forested and nonforested reaches

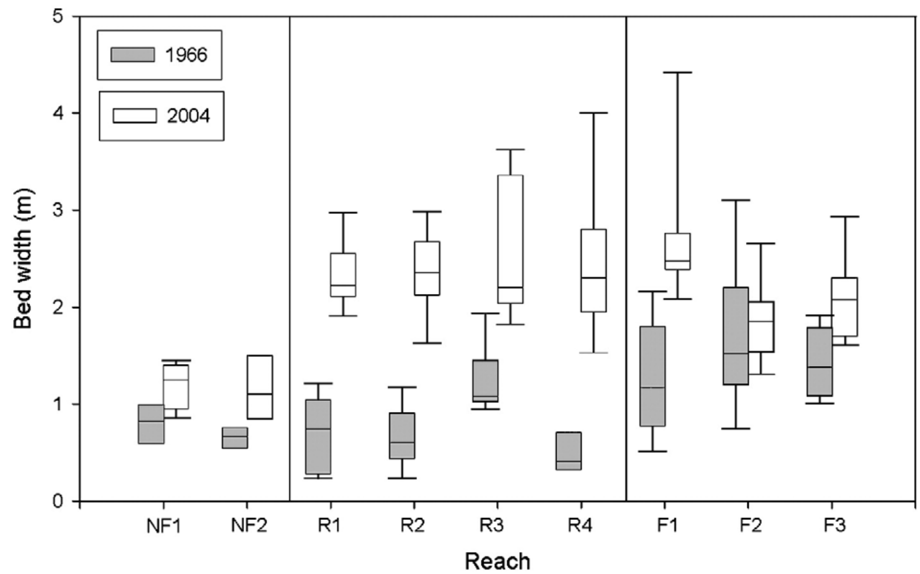

Fig. 6. Box plots of bed width data from 1966 and 2004 showing median values (mid-line), 25th and 75th percentiles (box), and 10th and 90th percentiles (whiskers).

of previous studies (Figs. 11 and 12). Although hydraulic geometry relationships are generally specific by region, our data corresponded well with data from Pennsylvania (Hession et al., 2003) and New Zealand (Davies-Colley, 1997). Forested reaches were substantially wider than nonforested reaches for a given drainage area, and they generally align with the hydraulic geometry of forested reaches. Widths of reforested reaches were narrower than the forested reaches of this study and smaller than the widths predicted by the hydraulic geometry relationship for other forested reaches. Reforested reaches were wider than the nonforested reaches in this study, but they aligned with the widths expected from nonforested reaches from other studies (Davies-Colley, 1997; Hession et al., 2003). In general, widths from our study may be relatively smaller than widths from Pennsylvania and New Zealand because the northeastern Vermont climate is dryer and cooler.

The range in mean bankfull depths was less pronounced than the range in bankfull widths. Mean bankfull depths for forested and nonforested reaches corresponded well with the hydraulic geometry relationship derived from previously published data (Fig. 12), where forested and nonforested sites were combined because Hession et al. (2003) found no differences in depth. Reforested reaches tended to have mean bankfull depths that were at least 10 cm deeper than the mean bankfull depth expected for a given forested or nonforested reach with the same drainage area. A paired t test of measured and expected bankfull depths for reforested reaches revealed a significant difference ($p=0.004$), but measured and expected bankfull depths for the five other reaches were not significantly different.

4.5. Hydrology

Although much of the historical hydrologic record from Sleepers River gages is missing, we were able to compare the annual mean and annual peak flows of Pope Brook in order to identify potential differences between historic and current stream flow that might affect channel morphology. Stream gage records for Pope Brook (USGS gage 01135150) from 1992 to 2005 are available and indicate an annual mean discharge of 0.17 cm. Annual peak flows for the same time period range from 2.61 to 7.05 cm, and the mean annual peak flow is 4.47 cm. Zimmerman et al. (1967) provided similar statistics for the same gaging station for a period of record from 1960 to 1966, where the mean annual discharge was 0.15 cm and the maximum peak discharge on the record was 4.47 cm. From this limited comparison of hydrologic summary statistics, significant differences are not apparent in the background hydrology over the period of interest.

5. Discussion

5.1. Reforestation and land-use change

Much of the state of Vermont has reforested passively as agricultural lands were abandoned. In fact, ~30% of late nineteenth-century Vermont was forested, while over 70% of Vermont is forested at present (Albers, 2000). The reforestation of portions of the riparian zone of W12 was a result of changes in land-use practices. Nonforested areas, maintained by local landowners as pastures or fields, reverted back to forest once maintenance declined. Portions of W12's riparian zone are still used as pasture for cattle, but these areas were excluded from the analysis because of known impacts to channel dimensions from cattle grazing (Clifton, 1989; Belsky et al., 1999).

The jumbled history of deforestation and reforestation in Vermont, and undoubtedly in other locations as well, sets a complex context for geomorphic inquiries. Within this context the response of a stream or river to a single anthropogenic or natural change may be difficult to discern, such as the riparian reforestation impacts. The widespread deforestation following European settlement changed the hydrologic regime and sediment loading to streams and rivers; several studies in Vermont using sediment coring techniques and alluvial fan trenching determined that upland erosion rates increased following European settlement (Bierman et al., 1997; Noren et al., 2002; Jennings et al., 2003). When hillslopes were cleared, they were primed for erosion,

Table 3
Number and volumes of individual LWD and debris dams by reach

Reach	Stream	LWD count	LWD volume (m^3)	Debris dam count	Debris dam volume (m^3)
NF1	W12	0	0.00	0	0
NF2	W12	1	0.03	1	0.03
R1	W12	10	0.40	2	0.20
R2	W12	8	0.55	2	0.39
R3	W12	3	0.20	1	0.08
R4	W12	6	0.33	2	0.31
F1	W12	14	0.76	2	0.50
F2	Pope	10	0.19	3	0.11
F3	Pope	8	0.23	1	0.02

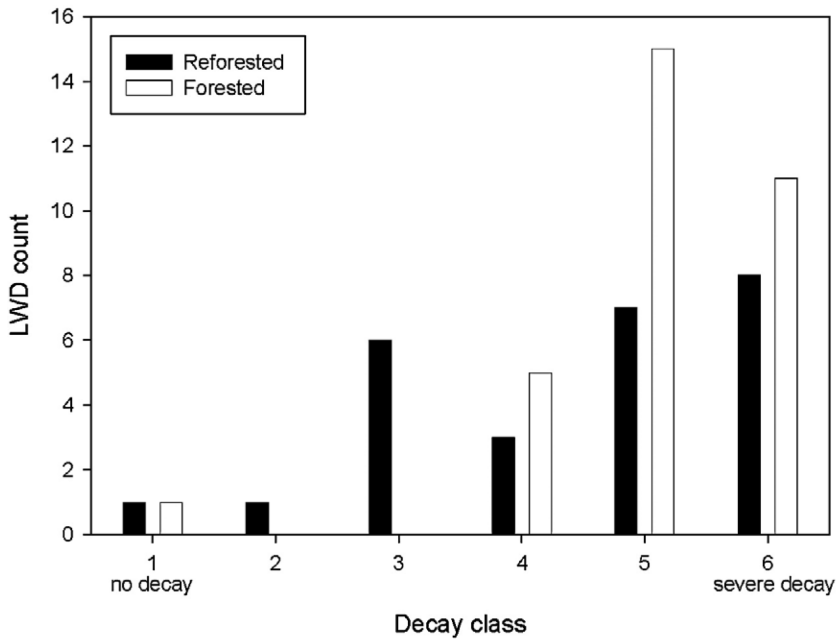

Fig. 7. Histogram of decay classes for LWD in forested and reforested reaches (class 1 had green leaves or needles present, class 2 had twigs present, class 3 had secondary branches present, class 4 had primary branches present, class 5 had no branches, and class 6 was severely decayed).

landslides, and mass wasting. Streams and rivers aggraded in response to the increased sediment load, evidenced by aggradation in floodplain deposits (Costa, 1975) and in the expansion of the Winooski River delta in Lake Champlain, VT (Bierman et al., 1997). Streams and rivers may be working through these legacy sediments. Indeed, conditions may not remain stable long enough for many streams and rivers to reach an equilibrium (Knighton, 1998). Although our study site may still be "recovering" from the historic deforestation centuries ago and although the site may continue to evolve in response to new input conditions (e.g., climate change), the decadal change in riparian vegetation has created a strong and discernable imprint on the channel morphology.

Similarly, other conditions, including channelization, impoundments, beaver activity, and grazing can complicate the geomorphic history. To the best of our knowledge, our study reaches were located in stream segments with limited historic and current impacts. Historic maps from the mid-1800 s of the two study streams did not show any impoundments or channel straightening; however, two homesteads and a former road crossing were located on W12 between reaches R3 and R4. Beaver are present in the region, and during our field data collection in 2004 and 2005 they maintained a dam in the headwaters of W12. We assume that whatever effects the beaver dam may have had on the sediment or hydrologic regime of W12 that the study reaches are equivalently influenced by this background condition. Portions of W12 (between R3 and R4 and between NF1 and R1) are currently accessible by cattle, but these segments were excluded from our analysis.

Table 4
Current bankfull dimensions at representative riffle feature

Reach	Stream	Drainage area (km^2)	Width (m)	Mean depth (m)	Cross-sectional area (m^2)	W:D[a]
NF1	W12	2.06	1.57	0.22	0.35	7.1
NF2	W12	0.91	0.33	0.28	0.09	1.2
R1	W12	1.98	2.68	0.47	1.25	5.7
R2	W12	1.95	2.12	0.50	1.07	4.2
R3	W12	1.72	2.06	0.45	0.93	4.6
R4	W12	1.02	2.04	0.37	0.76	5.5
F1	W12	1.91	3.84	0.33	1.28	11.6
F2	Pope	0.55	2.84	0.13	0.38	21.1
F3	Pope	1.06	2.27	0.29	0.65	7.9

[a] W:D: bankfull width to mean bankfull depth ratio.

5.2. Riparian vegetation and channel size

We found that forested reaches were significantly wider than reaches with nonforested riparian vegetation, confirming the findings of Zimmerman et al. (1967) and other studies (Murgatroyd and Ternan, 1983; Sweeney, 1992; Peterson, 1993; Davies-Colley, 1997; Trimble, 1997; Scarsbrook and Halliday, 1999; Hession et al., 2003; Sweeney et al., 2004; Allmendinger et al., 2005). As Zimmerman et al. (1967) described, in small streams, riparian vegetation exerts more influence on channel size than drainage area. For example, reach NF1 had the largest drainage area of all W12 reaches, but was roughly half as wide as the other forested or reforested reaches. Reach F1 was the widest of all reaches, even though reaches R1 and R2 had larger drainage areas than F1.

Our results suggest that reforested reaches may not be as wide as forested reaches, and thus we infer that reforested reaches will continue to widen for some unknown period of time. Reach F1 had the largest mean bed width among all seven reaches in W12 (Table 1), and this reach has been forested for at least 60 years. Reaches that have reforested within the last 40 years (reaches R1, R2, R3, and R4) had mean bed widths that were larger than the nonforested reaches, but smaller than the mean bed width of reach F1; however, an analysis of covariance, using drainage area as the covariate, found no significant difference between the bed widths of forested and reforested reaches. Although we did not have sufficient data to perform statistical testing on the bankfull widths, we found that the bankfull widths of the four reforested reaches aligned more closely with previously published nonforested hydraulic geometry relationships (Davies-Colley, 1997; Hession et al., 2003) than the forested ones (Fig. 11).

Another major difference between reforested reaches and the other reaches was the deeper channel depth. The mean bankfull depths of the three forested and two nonforested reaches corresponded well with the hydraulic relationship between mean bankfull depth and drainage area (Fig. 12). Similarly, other studies have found no difference in either the hydraulic geometry of depth and discharge

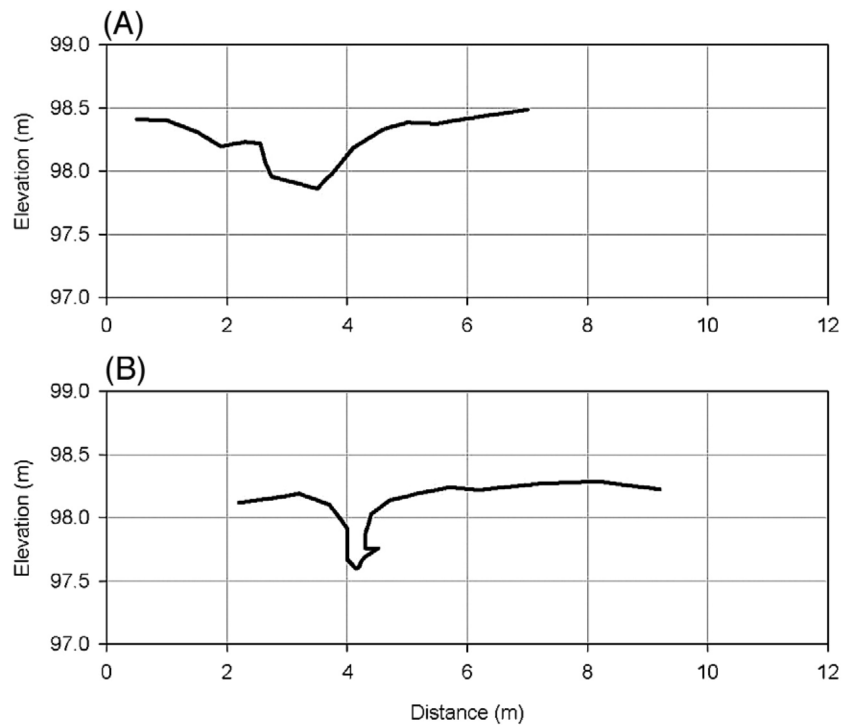

Fig. 8. Cross sections of nonforested reaches NF1 (A) and NF2 (B) of W12.

or depth and drainage area with different riparian vegetation types (Murgatroyd and Ternan, 1983; Andrews, 1984; Hey and Thorne, 1986; Trimble, 1997; Hession et al., 2003). The four reforested reaches were significantly deeper than the other forested or nonforested reaches for a given drainage area, which suggests that they may be incised at this stage of transition between a nonforested morphology and a forested morphology.

Although few studies have monitored channel change with reforestation, close inspection of previous literature suggests that incision may be an active process following reforestation. We found mention of slumping banks in two studies, where presumably an incised channel made banks increasingly high and steep enough to cause slumping failures (Murgatroyd and Ternan, 1983; Davies-Colley, 1997). In addition, during a 5-year period, Murgatroyd and Ternan (1983) observed neither overbank nor bankfull flows in the reaches with 50-year-old forest vegetation, while upstream in nonforested reaches, overbank flow occurred on average two to three times per year. A third study focusing on the water quality of two headwater streams after nine years of riparian reforestation described the reaches as incised and found an adverse riparian forest effect on water quality (Smith, 1992). We suspect that although reforested reaches may widen and contribute a greater sediment load during a transitional period, reforested reaches will eventually have slower lateral migration (Allmendinger et al., 2005) and additional sediment storage associated with woody debris (Keller and Swanson, 1979; Montgomery, 1997; Bilby and Bisson, 1998) that might compensate for a temporary increase in sediment load.

5.3. Channel widening

Our results suggest that riparian reforestation led to stream widening in W12 because the greatest difference in bed widths occurred in reaches that have partially or completely reforested since 1966. Reforested reaches are likely still adjusting to an equilibrium channel form. We commonly found features indicating recent channel change in reforested reaches, such as undercut, eroding banks, and avulsions. Davies-Colley (1997) speculated that the “streambank recession phase” following reforestation might continue for decades, while the channel recreates a forested equilibrium morphology. Parkyn et al. (2003) found little evidence for channel widening in reaches with forest vegetation planted 2 to 24 years prior to their study and suggested that the plantings were too young for widening to have begun. However, their finding was based on comparisons of nine forested reaches to upstream or nearby control reaches that were unfenced and actively grazed (Parkyn et al., 2003), and grazed reaches can be widened because of streamside trampling from livestock (Clifton, 1989). The time needed for reforested stream reaches to attain equilibrium is unclear. Our results suggest that this process may take longer than a few decades and perhaps last as long as a century, as the reforested reaches appeared to be roughly half-way between the nonforested and older forested reaches in terms of channel width (Fig. 11). Additionally, the forested reaches (F2, F3, and especially F1) may still be adjusting to earlier forest-clearing episodes, as those forest parcels are likely secondary or tertiary growth.

Although reforested reaches experienced widening that was significantly greater than the other reach types, both nonforested and forested reaches also had greater mean bed widths than when measured in 1966. Some possible explanations for this finding include (i) measurement error, (ii) long-term channel evolution, and (iii) a recent disturbance. First, we might not have measured widths at the same feature as Zimmerman et al. (1967); however, given the small stream size, measuring widths at different features would unlikely create threefold and larger differences. Second, results may indicate a long-term response to the historic land-use change accompanying European settlement centuries ago. The sediment supplied by the uplands of these drainages likely peaked during the large-scale land clearing of colonial times and has likely declined continually since that time. Throughout the watersheds of this region, streams may be widening in response to an overall reduction in upland sediment supply and may be slowly eroding legacy sediments previously trapped in floodplains, channel margins, or within the channel (Walter and Merritts, 2008). Third, natural or anthropogenic events may have caused larger

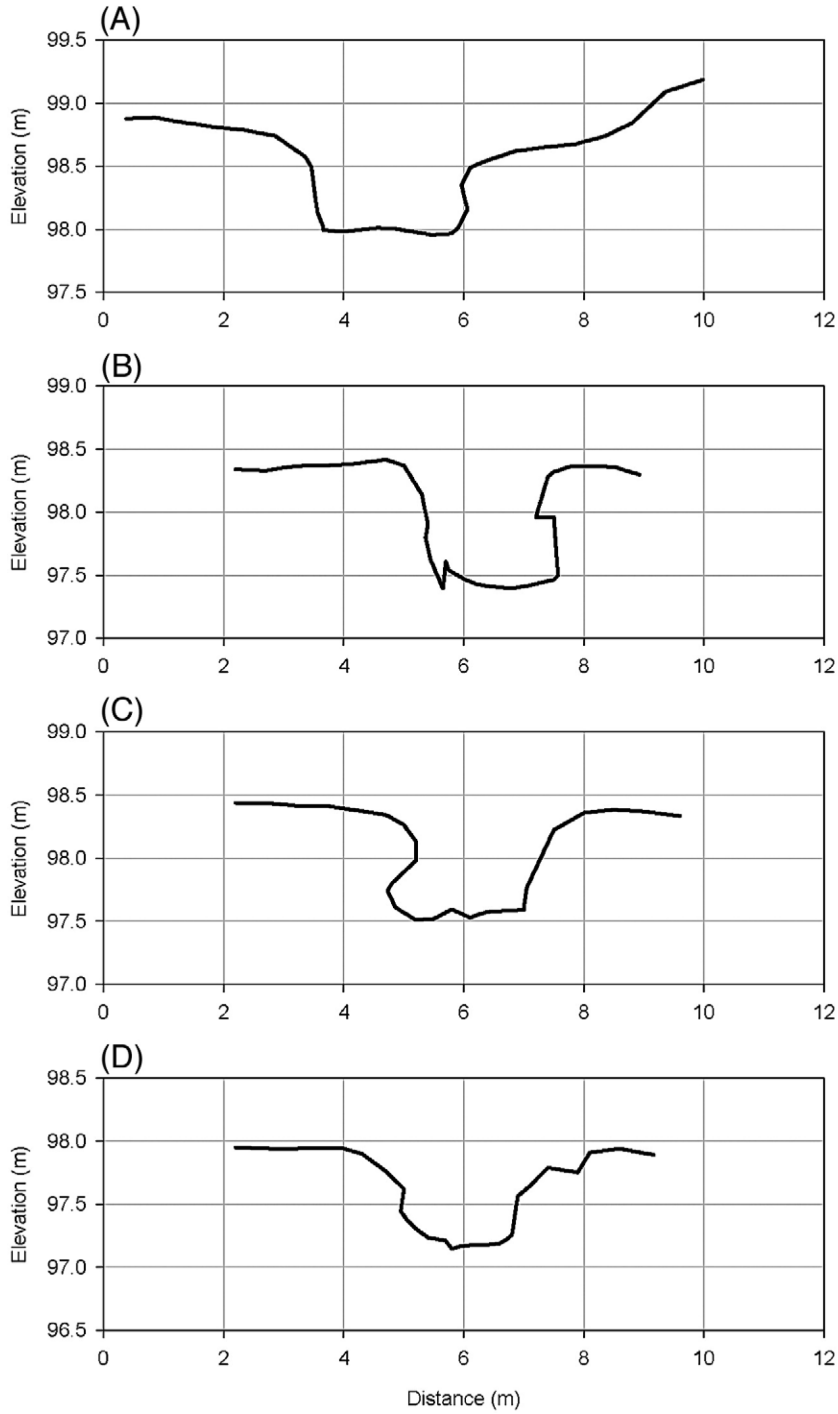

Fig. 9. Cross sections of reforested reaches R1 (A), R2 (B), R3 (C), and R4 (D) of W12.

bed widths overall (e.g., unusually large storm events, beaver dam breaching, road construction, or other developments). Despite these complicating factors, the effect of reforestation on channel width outweighs all other background effects, as the change in channel width was significantly greater for reforested reaches than the forested and nonforested reaches. Although all study reaches may be on different paths of adjustment to watershed, riparian, or climatic changes, channel widening following riparian reforestation is considerable.

The frequently discussed driving mechanisms for channel widening (i.e., either bank weakness from canopy shading of grassy vegetation or scour around LWD) did not appear to be the primary drivers for channel widening given our results; however, this study was not intended to specifically test for these mechanisms. First, the absence or presence of grassy bank vegetation did not appear to modify the amount of widening. Reach R1 was nearly completely shaded with mossy bank vegetation, while reaches R2, R3, and R4 had sufficient canopy opening to allow for ample grassy vegetation on their banks, yet all four reaches had widened to a similar degree (Figs. 4 and 11). Second, the amount of LWD did not appear to affect the extent of widening. Reforested reaches had widely ranging amounts of LWD (Table 3), but bankfull width did not appear to correspond with that parameter. Likewise, LWD counts and volumes were not significantly different between reforested and forested reaches. Furthermore, we did not observe greater streambank scouring adjacent to individual

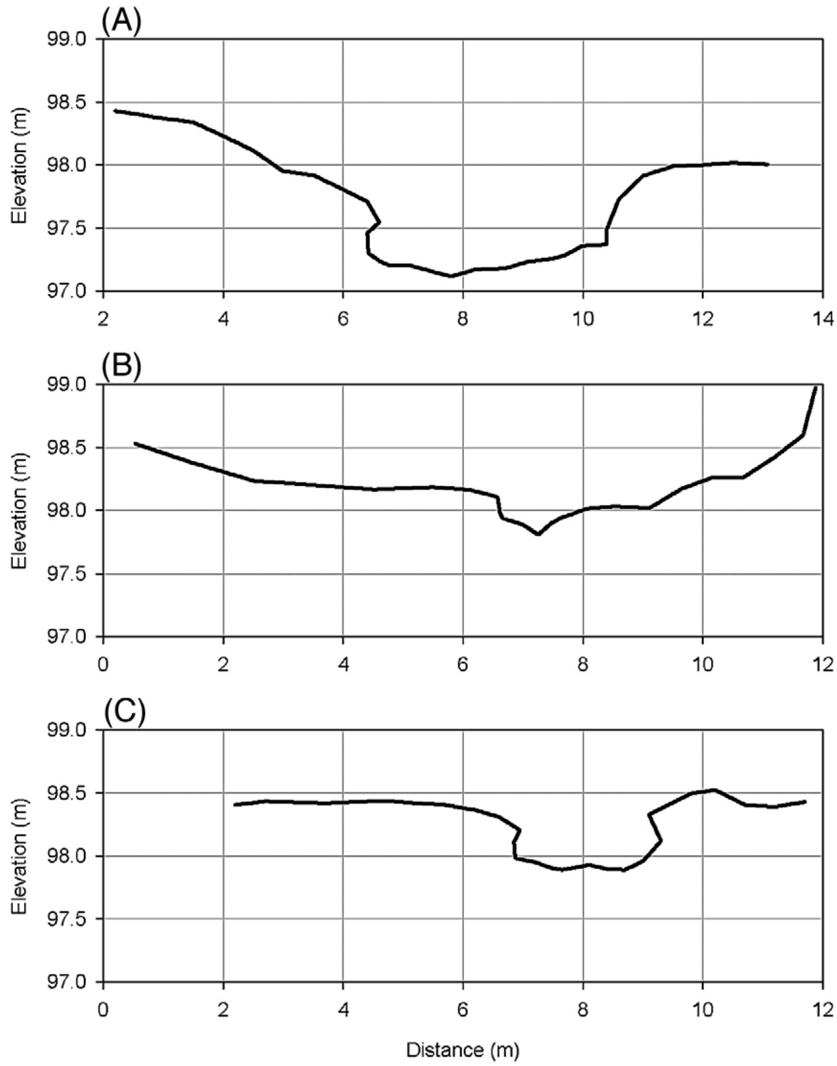

Fig. 10. Cross sections of forested reaches F1 (A) of W12, and F2 (B) and F3 (C) of upper Pope Brook.

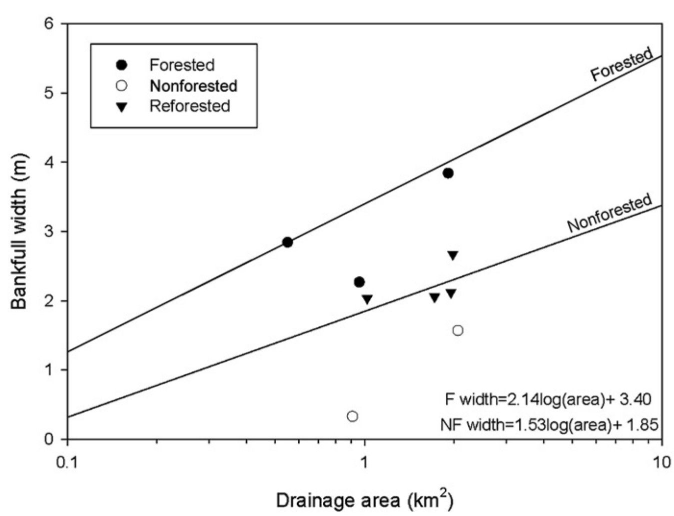

Fig. 11. Bankfull width hydraulic geometry for forested and nonforested reaches from Davies-Colley (1997) and Hession et al. (2003) plotted with individual bankfull widths of nine study reaches.

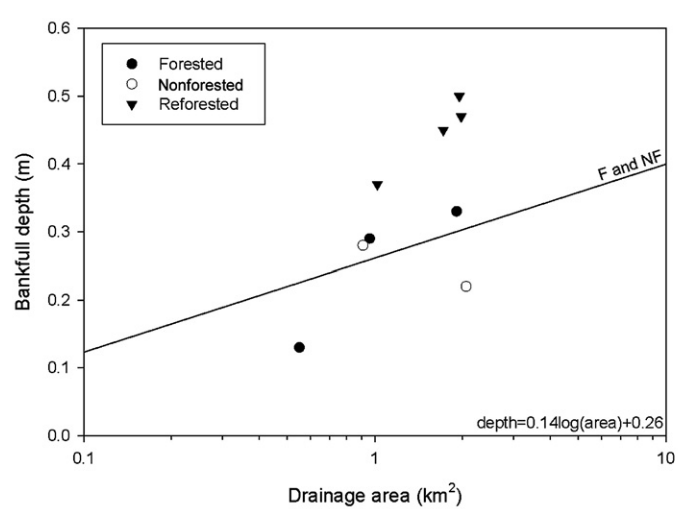

Fig. 12. Bankfull depth hydraulic geometry for combined forested and nonforested reaches from Hession et al. (2003) data plotted with bankfull depths of nine study reaches.

LWD pieces or debris dams. Bed widths measured near LWD or debris dams were not significantly wider than bed widths measured elsewhere in either forested or reforested reaches.

Differences found between the LWD characteristics of reforested and forested reaches may reveal aspects of the forest recovery process. In-stream LWD and woody debris on the floodplain were not significantly more plentiful in forested reaches. Because these are small streams where LWD is not in transport, the quantity of in-stream LWD was expected to be a function of the age of the nearby forest. Studies have found that older riparian forests contribute more LWD than younger forests (Evans et al., 1993; Ralph et al., 1994; Bilby and Bisson, 1998) and that recovery of LWD levels following a disturbance may take at least 40 to 60 years (Gregory et al., 1987). Although LWD was abundant in reforested reaches, LWD was likely younger and more recently recruited. In reforested reaches, LWD pieces were larger, had collected in fewer debris dams, and exhibited fewer signs of decay. The comparatively abundant amount of recently recruited LWD in reforested reaches, in spite of the age of the forest, may be a result of channel widening where young trees that grew along the edge of the once narrow stream have fallen prematurely because of the incision and widening (Hupp, 1999).

5.4. Conceptual model for channel widening

Based on our findings and previous work, we formulated a conceptual model to explain channel change over time at Sleepers River tributaries. Reforestation of riparian buffers appears to cause a multiphase channel adjustment where the channel first incises and then widens in response to changes in the local hydraulics and bank-resisting forces. Our conceptual model is similar to other models that have been proposed for channel evolution following other disturbances such as channelization and urbanization (Schumm et al., 1984; Booth, 1990; Hupp and Simon, 1991), but our model does not explain how either forested or nonforested reaches maintain a dynamic equilibrium (e.g., Allmendinger et al., 2005). The Schumm et al. (1984) model described a systematic evolution along the longitudinal profile

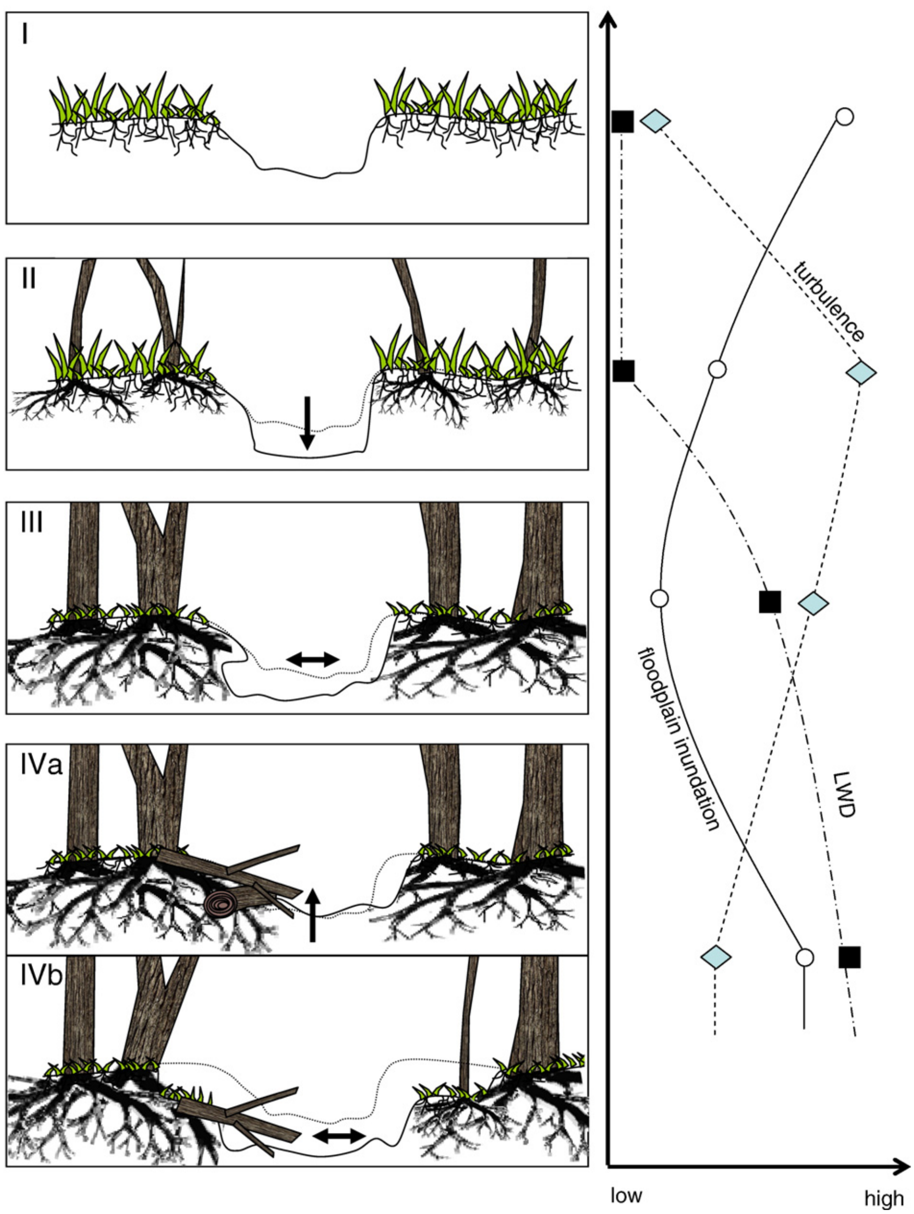

Fig. 13. Sketch of conceptual model with original channel profile repeated throughout. Stage I — equilibrium nonforested; stage II — reforestation begins and channel incises; stage III — incision slows, channel widens, forest matures; stage IV — channel reaches forested equilibrium by either (a) aggradation or (b) creating an inset floodplain. Relative levels of in-stream LWD, near-bank turbulence, and floodplain inundation are graphed adjacently.

of incised channels within an impacted drainage. In a downstream direction from an active nickpoint, reaches are first incised, then overwidened, and finally the most distant reaches aggrade back to a quasi-equilibrium state (Schumm et al., 1984). Although we recognize that reach-scale changes in riparian vegetation may provoke channel network-scale responses, our conceptual model describes at-a-station morphologic change. A network-scale response was exemplified by Clifton (1989) where a formerly grazed meadow reach of Wickiup Creek aggraded by 1 m, creating a “bulge” in the valley and channel bed profiles over a 50-year period following the exclusion of livestock. Another possible network-scale response to riparian vegetation change is the sediment supply regime. Although we recognize that sediment supply plays an interactive role in the processes described below, for lack of data and for simplicity we assume that the sediment supply is essentially constant through time.

Our conceptual model can be described in four basic steps, recognizing that this likely oversimplifies some of the complexities of the process (Fig. 13). The first stage of the model (stage I; Fig. 13) represents the nonforested condition as observed in reaches NF1 and NF2. The channel size is relatively small, providing for frequent floodplain inundation. LWD is largely absent within the channel and on the floodplain. Given the absence of LWD and the flexibility of the grassy vegetation, turbulence during overbank flood events is expected to be relatively low. The transition process begins when forest species colonize this nonforested floodplain and grow along the streambanks (stage II; Fig. 13).

In the second stage, the small woody plants have established along the stream's edge and on the floodplain. At this time, the trees are too young to shade out grassy vegetation or to provide any LWD to the stream; therefore, minimal change is expected in either the bank resistance or the in-stream roughness. The key change at this stage is the increased roughness on the floodplain that alters the local hydraulics during overbank flows (McBride et al., 2007). The additional roughness from the rigid woody stems reduces velocities on the floodplain and creates a larger velocity differential between the floodplain and in-stream flows. This velocity differential generates turbulence at the interface between the floodplain and the channel. McBride et al. (2007) found that turbulent kinetic energy was doubled in a narrow zone along the bank's edge when rigid woody stems were added to a physical model of a small nonforested stream. When increased turbulence is coupled with high downstream velocities, erosion is likely amplified (Thompson, 2004). In keeping with our observations of reforested reaches, we surmise that the channel responds to the increased turbulent energy by scouring at the base of the banks. This scouring will promote degradation across the channel bottom, especially if the grassy cohesive streambanks are more resistant than the bed material. The amount of vertical and lateral adjustment to the increased turbulence will be site specific and will depend upon many factors such as bed substrate and armoring, bank cohesion, rooting depth, and upstream sediment supply.

The third stage of the model represents the current conditions at the reforested reaches (R1–R4). In stage III (Fig. 13), the channel is deeper and wider than stage I or II. Assuming the hydrologic regime is constant throughout the stages of the model, the floodplain inundation frequency will have decreased as the channel has incised and widened. Records of peak stages in reaches NF1 and R1 showed that a greater proportion of the active channel was filled in reach NF1 than in reach R1 during several different peak flow events (McBride, 2007). The near-bank turbulence phenomenon is likely less active because of the reduced floodplain inundation in this larger channel and because of the increased roughness within the active channel from recently recruited LWD. LWD has been recruited to the stream at this stage as banks collapse and the forest vegetation ages. The forest vegetation has likely matured at this stage to provide more shade to limit the growth of grassy vegetation, potentially weakening streambanks, as speculated in previous studies (Davies-Colley, 1997; Trimble, 1997). Streambanks may be additionally weakened by the weight of young trees along the edge of the stream channel (Thorne, 1990; Simon and Collison, 2002).

In the final stage, the channel reaches a new “forested” equilibrium. Reaches F1, F2, and F3 are likely still approaching this final stage of the model. We suspect that the channel will return to a normal depth either by aggrading back to the original bed level (stage IVa; Fig. 13) or by creating a new lower floodplain surface (stage IVb; Fig. 13), depending on the background discharge and sediment regimes. If the sediment supply is great, we expect that LWD will trap sediment and that the floodplain will be reactivated, at least partially. Forested floodplains of small streams may be best described as a complex patchwork of different elevations that have responded to in-stream debris dams, which can locally increase the frequency and extent of overbank flows (Jeffries et al., 2003). Ultimately, the new “forested” channel reaches an equilibrium size such that the channel has adequate capacity for the discharge and sediment regimes, given the added roughness of both in-stream LWD and the forested floodplain. Near-bank turbulence is greater than in stage I but lower than in stages II or III because of the increased channel size and the increased in-stream complexity. Compared to the original stage I channel, the channel is wider with lower stream velocities (Sweeney et al., 2004), higher in-stream roughness (Sweeney et al., 2004), slower migration rates (Allmendinger et al., 2005), and greater benthic habitat (Sweeney et al., 2004).

In summary, the conceptual model may provide guidance for future research efforts. Ideally, a next step would be to monitor stream reaches that have recently reforested or that have been planted as a restoration effort over a long time span and document change over time. A more plausible option might be to conduct a field investigation with a broader geographic scope where stream reaches typical of the stages described in the model could be monitored over a shorter time span. Either of these options might shed light on whether our conceptual model of incision-widening-recovery is widely applicable or site specific.

6. Conclusions

Reforested reaches were shown to have widened by a comparison of measurements collected in 1966 and in 2004–2005. Several characteristics of the reforested reaches indicated that these reaches were not in equilibrium and will likely continue to widen and adjust to the change in riparian vegetation. Similar to many other studies of small streams, we found that nonforested reaches were considerably narrower than forested reaches. Our observations did not provide strong evidence that previously hypothesized mechanisms were primarily responsible for differences in channel width, namely that widening is a result of either scour around LWD or bank weakness from the suppression of grassy bank vegetation. Alternatively, we present a conceptual model describing a process of incision, widening, and recovery primarily instigated by a change in stream hydraulics. Although we have documented channel widening associated with reforestation, this study was not intended to investigate the effect of riparian vegetation on bank stability. Additionally, our small sample size and limited geographic extent restrict generalized conclusions; however, we hope results from this study will provoke future investigations into the timing of channel widening associated with reforestation and the effects of widening on sediment delivery, sediment transport, inundation frequency, and stream habitat. Our results provide valuable information for stream restoration efforts that involve the conversion of nonforested riparian vegetation to forests. Replanting forested buffers on small streams may invoke unanticipated channel widening with yet unknown consequences for sediment delivery to downstream water bodies. With an awareness of the channel widening response, perhaps stream restoration efforts could be modified to accommodate or mitigate for possible undesirable short-term outcomes. Undoubtedly, further investigations are needed to determine the effects of riparian reforestation to sediment supply, habitat complexity, and ecosystem structure and function.

Acknowledgements

Research supported in part by the National Science Foundation Graduate Fellowship Program and a graduate research assistantship from the Vermont Experimental Program to Stimulate Competitive Research (VT EPSCoR, EPS0236976). We thank T. Owen, A. Pearce, J. Clark, and T. Clason for their helpful field assistance. We thank J. Shanley of the U.S. Geological Survey for his support of our research. We are especially grateful to K. Dwyer, R. Boyle, and S. Boyle for allowing us access to research sites on their property. This paper improved significantly thanks to the insightful reviews of D. Thompson, P. Bierman, three anonymous reviewers, and the editor.

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