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Testing the effective-discharge paradigm in gravel-bed river restoration

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A R T I C L E I N F O

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A B S T R A C T

Restoration of an alluvial wet-meadow system was conducted in the late 1990s to reestablish hydraulic interactions between the river, floodplain, and groundwater to support aquatic–riparian ecosystem function. A single-discharge approach sized the bankfull channel dimensions to the effective discharge (Q_e) and three degrees of channel widening relative to the Q_e design were explored to identify which design attained dynamic equilibrium in the shortest time. The three experimental channel designs were implemented with bankfull widths of 96%, 157%, and 191%, respectively, of the Q_e geometry. Response trajectories were documented for channel dimensions, sediment mobility, channel morphology, floodplain connectivity, and riparian vegetation for the three channel designs, and the efficacy of a single-discharge approach for restoration was examined.

Analysis of 20 years of monitoring data and hydraulic modeling revealed that each design responded differently to the imposed initial channel conditions and evolved at substantially different rates. The design with bankfull dimensions most closely approximating Q_e reached dynamic equilibrium within four years of restoration, whereas the moderately over-widened channel (57% larger) exhibited slower responses toward dynamic equilibrium for some metrics and did not fully attain the Q_e design dimensions within the monitoring period. The extremely-over-widened channel (91% larger) mainly induced slow rates of bed deposition that are projected to take nearly 300 years for the bankfull dimensions to narrow to the Q_e width. All reaches had low bed mobility (bankfull Shields stress < 0.03) 14 years after restoration, demonstrating the challenge of reducing the drivers of channel widening, while maintaining sufficient competence for bedload transport and a sustained supply of coarse bed material for salmonid habitat. Restoration that sizes the channel to Q_e can provide rapid dynamic equilibrium, but is a first-order simplification of 1) channel dynamics and 2) the range of flows needed for restoring physical and biological processes in wet-meadow systems.

1. Introduction

Wet-meadow fluvial systems and their associated riparian floodplains provide ecosystem services, including flood attenuation, sediment storage, geochemical cycling, and riverine habitat for a broad range of biota (Hammersmark et al., 2008; Loheide et al., 2009; Lowry et al., 2011; Miller et al., 2012; Dickard et al., 2015; Nilsson et al., 2018). In western North America, valuable habitats in unconfined meadow systems have often been impacted by historic land uses (e.g., fur trapping, grazing, mining, farming) that frequently result in habitat degradation (Keeley et al., 2003; Naiman et al., 2005; Ramirez and Weir, 2010; Heitmeyer et al., 2010; Ramstead et al., 2012; Viers et al., 2013; Castro and Thorne, 2019). For example, channelization for agriculture can initiate a negative feedback process of channel incision, lowered groundwater table, loss of riparian vegetation, and decreased bank stability (Schumm et al., 1984; Shields et al., 2010; Loheide and Booth, 2011; Viers et al., 2013; Cluer and Thorne, 2014; Dickard et al., 2015). In response, many meadow systems on public lands are currently the focus of stream and habitat restoration activities (Bernhardt et al., 2005; Rumps et al., 2007; Viers et al., 2013), with typical goals to restore

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physical and biological processes and initiate a positive trajectory of biophysical interactions (Palmer et al., 2005; England et al., 2007; Roni et al., 2008; Feld et al., 2011; Wohl et al., 2015; Yochum, 2016; Johnson et al., 2020). Successful stream restoration of meadow environments requires reestablishing hydraulic interactions between the river, floodplain, and groundwater, as well as biophysical interactions between the riverine and riparian systems (Tague et al., 2008; Hammersmark et al., 2008, 2010; Loheide and Booth, 2011; Ramstead et al., 2012; Rodriguez et al., 2017; Johnson et al., 2020). However, restoration of active physical and biological processes is usually constrained by a mixture of local anthropogenic activities and the larger watershed-scale processes that determine the appropriate stream type (e.g., single vs. multi-thread) and the potential for achieving restoration goals (Stoddard et al., 2006). Thus, river restoration in meadow systems must rely on a suite of different approaches, ranging from promoting a single-thread meandering channel to a Stage 0 (multi-thread) design depending on local conditions and constraints (e.g., Cluer and Thorne, 2014; Wohl et al., 2015). While the above principles are generally understood, few restoration projects provide a sufficiently long monitoring period to evaluate project success and inform future restoration efforts.

Here, 20 years of monitoring data and hydraulic modeling were used to assess post-restoration evolution of a formerly straightened segment of the Red River, an unconfined meadow system in north-central Idaho, USA. Response trajectories were documented for channel dimensions, sediment mobility, channel morphology, floodplain connectivity, and riparian vegetation for three restoration designs to evaluate which design achieved the restoration goals in the least amount of time. The designs employed the effective discharge (Q_e, that which transports the most sediment over time sensu Wolman and Miller, 1960) and the notion that a single, formative discharge can be used to create a self-sustaining channel in dynamic equilibrium. The approach stems from the hypothesis that the time-averaged channel morphology adjusts to Q_e, which is corroborated by the observation that bankfull geometry in many gravel-bed rivers is coincident with Q_e (e.g., Andrews, 1980; Andrews and Nankervis, 1995; Emmett and Wolman, 2001) and forms the basis for bankfull geometry being used in “natural channel design” (e.g., Rosgen, 1996, 2011). As such, Q_e was used as the design basis for the bankfull channel geometry in the restored reaches at Red River (Klein et al., 2007). The Wolman and Miller (1960) method was used here to determine Q_e from observed flow and transport rates, although it is recognized that Qe is sensitive to the chosen analytical method and local environmental conditions, and that alternative definitions of effectiveness have been proposed (e.g., Nash, 1994; Costa and O'Connor, 1995; Vogel et al., 2003; Goodwin, 2004; Lenzi et al., 2006; Barry et al., 2008; Doyle and Shields, 2008; Hassan et al., 2014; Basso et al., 2015). Furthermore, use of Q_e for stream restoration and landscape evolution modeling has been criticized because the naturally occurring range of discharge and sediment transport events is reduced to a single discharge representing the time-averaged maximum sediment load, rather than considering the total sediment load integrated across all flows (Doyle and Shields, 2008; Croissant et al., 2019). Within this context, this study does not compare different Q_e calculation methods, but examines whether single-discharge designs can produce dynamic equilibrium and whether bankfull dimensions sized closest to Q_e attain dynamic equilibrium in less time than other designs. Dynamic equilibrium is defined here as a channel that actively transports sediment and migrates across its floodplain, while maintaining a steady, time-averaged cross-sectional form and a reach-scale balance of erosion and deposition (e.g., Hack, 1960; Leopold, 1994; Tranmer et al., 2018).

2. Study site

2.1. Historic conditions and human impacts

The Red River is a 45 km-long tributary of the South Fork Clearwater River located in north-central Idaho, USA. The restoration site is in the Lower Red River Meadow, a 5.3 km^2 unconfined, alluvial valley that is 6.2 km long and hosts a seasonal wet-meadow ecosystem (Fig. 1). Elevation of the site is 1280 m, with an upstream drainage area of 294 km^2. Annual precipitation ranges from 760 to 1020 mm, with a snowmelt hydrograph that typically peaks between late April and early June. Peak overbank floods historically supported a native riparian community of alder (*Alnus* spp.), willow (*Salix drummondiana*, *S. geyeriana*, *S. exigua*), sedge (*Carex* spp.), and rush (*Juncus* spp.). Soil horizons consist of fine-textured material ranging from 38 to 137 cm that overlie coarse sand and gravel alluvium (McGeehan, 1995). Extensive paleochannels are present throughout the valley bottom (Fig. 1), indicative of historic avulsion events, as well as meander migration and cutoff processes. Prior to human alteration in the 1940s, the gravel-bed river exhibited a meandering, pool-riffle morphology (Fig. 1A) that was a stronghold for spring Chinook salmon (*Oncorhynchus tshawytscha*), steelhead trout (*O. mykiss*), westslope cutthroat trout (*O. clarki lewisi*), and bull trout (*Salvelinus confluentus*) (USFS, 1998; IDFG, 1999).

Beginning in the 1940s, the meadow was straightened for flood conveyance, dredge mining, cattle grazing, and agricultural activities (BPA, 1996; Klein et al., 2007) (Fig. 1B). Channelization reduced river sinuosity by approximately 25%, thereby increasing channel slope, velocity, and shear stress, resulting in channel incision of 46–61 cm, reduced morphologic complexity, and limited floodplain connectivity (RME, 1995; BPA, 1996). Channel incision and lowering of the near-channel groundwater table contributed to a conversion of riparian vegetation from seasonal native wetland and woody habitats to upland grasses (Brunsfeld et al., 1996; BPA, 2003a). The loss of native riparian vegetation and associated root strength was followed by streambank erosion, channel widening, loss of floodplain connectivity (PWI, 1994), and fine sediment deposition detrimental to salmonid eggs and alevins (e.g., Bjornn and Reiser, 1991).

2.2. Restoration design and construction

In 1993, a 4.2-km reach in the Lower Red River Meadow was identified for restoration to 1) establish a self-sustaining channel in dynamic equilibrium that approximated historic channel conditions (pre-1940s), 2) improve aquatic habitat conditions for salmonids, and 3) increase hydraulic connectivity with the floodplain to support native riparian vegetation (USFS, 1998; BPA, 1996, 2003b). Unconfined valleys like that of Red River generally have wide, meandering streams with pool-riffle morphology that migrate across their floodplains (e.g., Schumm, 1977; Kondolf, 1994; Brierley and Fryirs, 2005; Buffington and Montgomery, 2021). Alternatively, where beaver are present, unconfined valleys may exhibit multi-thread channels, with extensive pond systems, and the predominance of willow species (e.g., Polvi and Wohl, 2012; Castro and Thorne, 2019), which may have been the case for the study site prior to beaver extirpation in the 18th and 19th centuries by French and American fur trappers. However, restoration at Red River focused on recent human disturbance (1940s onward), rather than conditions prior to the arrival of European settlers. Furthermore, while conceptual models were used to interpret the incised and over-widened state of the river prior to restoration (e.g., Schumm et al., 1984), the restoration approach involved re-setting the system to pre-1940s conditions through mechanical intervention and planting of native vegetation, rather than promoting response of the degraded system toward the desired state by manipulating environmental variables within a channel evolution framework (e.g., Brierley and Fryirs, 2005; Rosgen, 2006; Cluer and Thorne, 2014). Instead, the channel was re-set to observed conditions 60 years prior (1930s), assuming that basin inputs of water and sediment had not changed significantly during that time (BPA, 2003a). As such, the pre-disturbance channel sinuosity and slope were determined from 1936 aerial imagery (Fig. 1A) as 2.4 and 0.0017, respectively (RME, 1995). During restoration, channel length was increased by 60%, increasing sinuosity from 1.7 to 2.7 and decreasing channel slope from 0.0025 (1994) to 0.0015 (2000), producing values

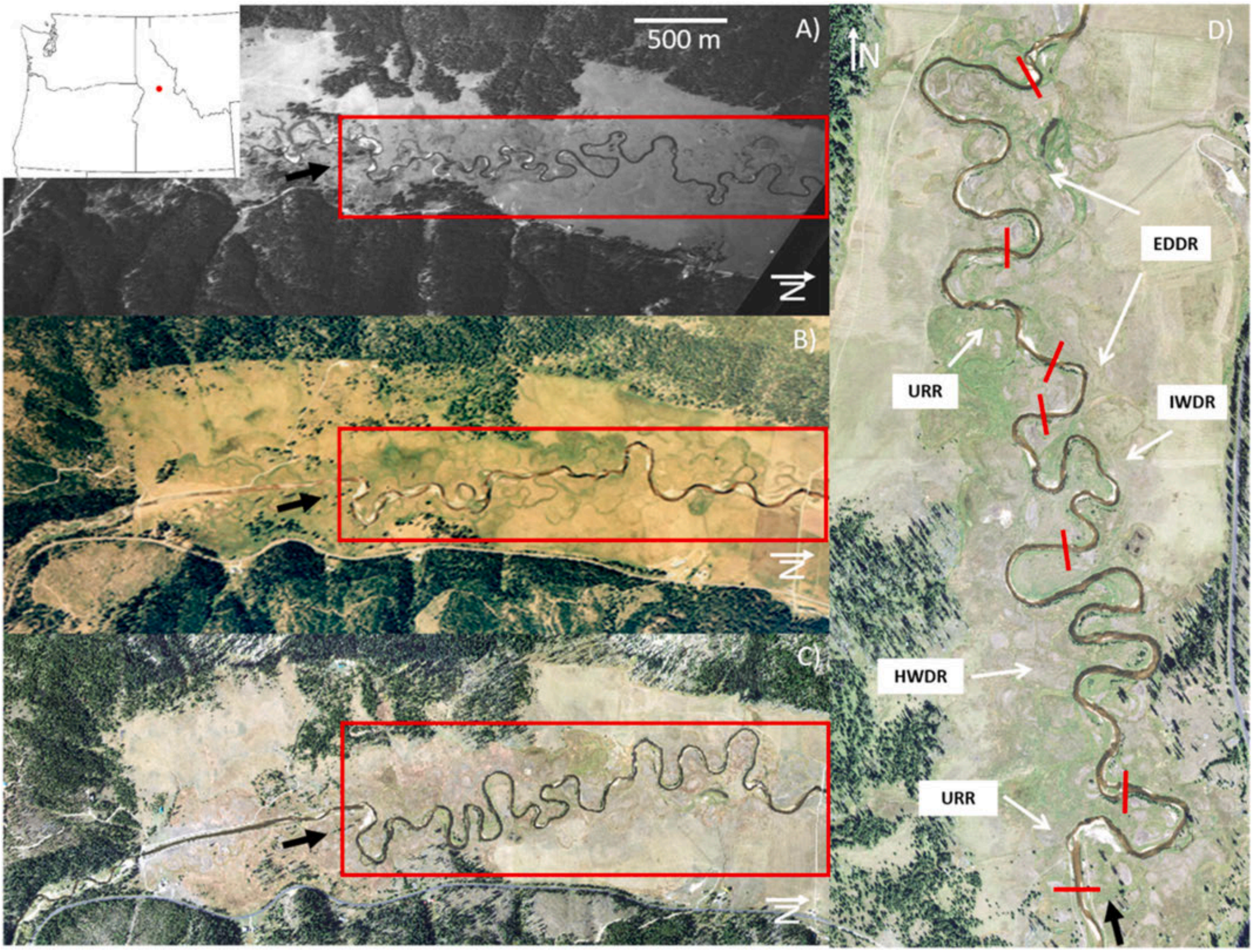

Fig. 1. Aerial photographs of the Lower Red River Meadow in A) 1936 (pre-disturbance of the channel), B) 1992 (post-channel straightening and pre-restoration), C) 2000 (post-restoration), and D) 2018 (post-restoration, showing boundaries (red lines) between design reaches (high-width design reach (HWDR), intermediate-width design reach (IWDR), effective discharge design reach (EDDR) and unrestored reaches (URR))). Red boxes in A–C depict extent of the 4.2 km study site, and black arrows indicate flow direction. Note the orientation in D differs from A–C. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

similar to those in 1936 (Klein et al., 2007). A total of 115,000 native riparian and wetland seedlings were planted on the streambanks and adjacent floodplain to stabilize banks and provide riparian habitat (BPA, 2004). To establish and maintain native vegetation, channel restoration and subsequent evolution were expected to increase floodplain inundation during snowmelt runoff and to reduce the vertical distance between the top of bank and summer low-flow water surface to ≤1 m (Klein et al., 2007). Full restoration objectives are available in Table S1 of the supplementary materials.

The overall restoration goal was to produce a freely-meandering channel that rapidly achieved dynamic equilibrium through meander migration and self-adjustment (BPA, 2003a). Because little information for design strategies existed in the early 1990s, an experimental approach was employed to assess how initial channel conditions influenced rates of evolution and attainment of restoration goals. Three bankfull design approaches of decreasing width/depth ratios were used during restoration (Fig. 1D) with the hypothesis that each channel design would evolve uniquely for similar boundary conditions of water and sediment input from the upstream watershed, affording an opportunity to assess the best approach. The implicit assumption of the experimental design approach was that because each design reach was short (≤1000 m), it would locally adjust to the upstream basin inputs of water and sediment without appreciably modifying the basin signal to the downstream reaches (e.g., Hession et al., 2003). However, subsequent interpretation of post-restoration responses recognized that the study reaches are inherently linked, with local conditions potentially modifying the signal to downstream reaches. Adaptive learning, where continuous monitoring informs subsequent design, was used to refine successive channel designs over a 5-year construction period (1996–2000).

Daily discharge and sediment transport rating curves from U.S. Forest Service gages on the South Fork and mainstem Red River (King et al., 2004) located ~8 km upstream of the study site were used to estimate the effective discharge (Q_e) as that which transports the greatest amount of sediment over time (Wolman and Miller, 1960). Effective discharge was calculated using the sediment transport rating curves at the gages and between 20 and 30 uniform discharge intervals based on the flow frequency distribution of Goodwin (2004). The gages were located in alluvial gravel-bed reaches with bankfull width-to-depth ratios between 15 and 20 (King et al., 2004). Median bed-surface grain sizes (D_{50}) at the gages were similar to those in the restoration reach (D_{50} = 27–165 mm), as were floodplain sediments (D_{50} = 0.14–0.54 mm). Bedload measurements at the two gage sites indicated that the majority of bedload had D_{50} < 2 mm, passing over an armored bed composed of 8–24% sand. Equating the effective and bankfull discharges ($Q_e = Q_b$), the bankfull dimensions for Q_e design were then determined from regional, downstream, bankfull hydraulic geometry relations for tributaries of the Salmon and Clearwater rivers (drainage areas = 1.3–381 km^2, bankfull widths (B_b) = 1.44-32 m, and bankfull depths (h_b) = 0.1-1.12 m), which included two measurement locations on Red River (Whiting, 1998; Whiting et al., 1999). For the effective discharge (Q_e = 18.9 m^3/s), the Q_e design for bankfull geometry had a bankfull width (B_{be}) = 18.3 m, hydraulic depth (h_{be}) = 0.92 m, and bankfull width-to-depth ratio (B/h_{be}) = 20 (BPA, 2003a; Klein et al., 2007).

Because bank erosion and fine sediment input were major management concerns, the initial channel design (constructed in 1996) was conservative and intentionally created a channel that was wider and shallower than the Q_e design geometry (RME, 1995; BPA, 1996, 2003b), with B_b = 35.1 m and h_b = 0.65 m, respectively equivalent to 191% and 71% of the Q_e design geometry. This high-width design reach (HWDR) was intended to reduce outer-bank shear stress in meander bends and increase point bar deposition, thereby encouraging channel narrowing and vegetation establishment over time. The restored reach slope for the HWDR (S_0 = 0.00137) was established through a combination of historic meander reconnection and new channel construction. The HWDR incorporated broad, low-angle, point bars (lateral slope = 50:1) and channel-spanning rock sills (above grade) to prevent bed incision, locally trap sediment, and increase low-flow water-surface elevation. However, monitored changes after construction showed less channel narrowing than anticipated owing to lower than expected upstream sediment supply.

The second restoration approach (constructed in 1997) formed an intermediate-width design reach (IWDR, B_b = 28.8 m, h_b = 0.69 m) that was respectively 157% and 75% of the Q_e design geometry. The IWDR provided a reduced width-to-depth ratio, reconnected historic meanders (S_0 = 0.00138), and added fewer rock sills for the same purpose as the HWDR. Streambanks of the reconnected meanders in the IWDR were intentionally left undisturbed to protect existing vegetation and preserve undercut banks to increase habitat complexity.

After a year of monitoring (1998) and building on lessons learned from the previous designs, the third restoration approach sized the bankfull channel dimensions to convey Q_e. Two effective-discharge design reaches (EDDR; middle and downstream end of the site; Fig. 1D) were constructed in 1999 and 2000 that followed historic meanders (S_0 = 0.00131), did not include rock sills, and created the lowest width-to-depth ratio, with constructed bankfull dimensions of B_b = 17.6 m and h_b = 0.91 m (respectively, 96% and 99% of the bankfull Q_e design geometry).

Two unrestored reaches (URR) at the upstream end and lower third of the site (Fig. 1D) were used as reference reaches for evaluating post-restoration response (S_0 = 0.00212). At the start of the post-restoration period (2000) the imposed channel dimensions of the restored reaches (HWDR, IWDR, EDDR) represented three degrees of channel widening relative to the Q_e design, creating a unique opportunity to assess how different channel designs evolved in alluvial meadow settings relative to the unrestored reaches (URR).

3. Methods

Analysis of post-restoration response involved both direct field observations for each year of measurement and hydraulic model simulations for select years during the restoration period. Partial field surveys were performed some years to address project-specific questions (e.g., riparian vegetation response, morphologic habitat availability, meander migration) and data were only available for specific reaches in those cases. Complete surveys were performed in 1994, 2000, 2004, and 2014 that allowed the channel geometry of reaches across the restoration site to be compared (Table S2 in the supplementary materials has a complete list of field measurements).

3.1. Data collection

Baseline channel conditions (e.g., cross sections, longitudinal profile, water-surface elevation) were surveyed prior to restoration in 1994. Three post-restoration topographic/bathymetric surveys were completed to document channel response relative to expected recovery times (Supplementary Materials) and flow history. After restoration construction (2000), the channel was surveyed to document the as-built conditions. A 2004 survey captured geomorphic changes that occurred since restoration completion, including those resulting from the first overbank flood in 2003 (Q_{peak} = 21.2 m^3/s, 5-year recurrence interval) to document objectives expected to occur within 1–3 years after restoration. A survey in 2014 incorporated changes from three overbank floods in 2008 (Q_{peak} = 28.9 m^3/s, 9-year recurrence interval), 2009 (Q_{peak} = 23.8 m^3/s, 6.8-year recurrence interval), and 2011 (Q_{peak} = 21.2 m^3/s, 5-year recurrence interval) and captured objectives within the expected 15-year period.

Surveys of channel cross sections were repeated at monumented locations during summer low-flow conditions. The pre-restoration survey (1994) was performed using a rod and autolevel, and post-restoration surveys (2000, 2004, 2014) were conducted with RTK GPS. In addition to documenting bankfull morphologic changes, the surveys provided input data for hydraulic models. The relatively homogeneous stream conditions in the pre-restoration period were documented with 23 cross sections (8 in URR, 4 in HWDR, 2 in IWDR, 9 in EDDR). With a longer channel, post-restoration conditions were monitored with 56 cross sections (10 in URR, 15 in HWDR, 16 in IWDR, 15 in EDDR). One high-flow event (18.9 m^3/s, 2.9-year recurrence interval) equal to Q_e was surveyed in May 2011 and associated water-surface elevations were used for calibrating the hydraulic model. Statistical significance of morphologic adjustment was evaluated by non-parametric Mann-Whitney tests, and all analyses were evaluated at an α of 0.05.

Grain-size distributions were monitored with Wolman (1954) pebble counts (100+ grains) in each reach. Because no pre-restoration grain-size data were available, reference samples were taken in 1997 in locations immediately upstream and downstream of the restoration site. Aquatic habitat was classified in terms of the number and areal extent of pools, riffles, and runs/glides following the methods of Bisson et al. (1982). Riparian vegetation was surveyed in each reach during summer low flow to determine percent cover by native vs. invasive species using 20 paired, 100-m transects that respectively (1) paralleled the water's edge and (2) extended perpendicular from the water's edge across the bank and floodplain (USFS, 1992; Cagney, 1993). Additionally, the time to meander cutoff for each reach was estimated as $T_c = R_{eros}/L_{min}$, where R_{eros} is the lateral bank erosion rate (m/yr) and L_{min} is the minimum distance across the neck of the meander (m).

3.2. Hydraulic modeling

One-dimensional hydraulic modeling provided estimates of specific stream power, shear stress, water surface elevation, and inundation extent. HEC-RAS 5.0.3 (USACE, 2016) was used to simulate conditions for four separate scenarios: 1) pre-restoration (1994), 2) immediately after restoration (2000), 3) four years after restoration (2004), and 4) 14 years after restoration (2014). Steady-state flows were modeled for the 7-day low flow (0.48 m^3/s) and several high flows to capture a range of floodplain inundation bracketing that of Q_e. High discharges were modeled for 16 m^3/s, 17.5 m^3/s, 18.9 m^3/s, 20.5 m^3/s, and 22 m^3/s to describe floodplain inundation levels for 2- to 6-year recurrence intervals. To assess the low-flow criterion for supporting riparian vegetation (<1 m between the water surface and top of bank), simulated low-flow conditions were calibrated from surveyed water-surface elevations (July or August) of each modeled year. Calibration occurred for discharges of 0.62 m^3/s (1994), 0.53 m^3/s (2000), 0.83 m^3/s (2004), and 0.63 m^3/s (2014). Manning's n values were spatially adjusted to match the surveyed water-surface elevation, with reach-average values ranging from 0.065–0.08 s/m$^{1/3}$ in low-flow model scenarios. High discharge was calibrated using 2011 measurements at a discharge of 18.9 m^3/s, with a reach-average Manning's *n* of 0.032 s/m$^{1/3}$. Similar Manning's *n* values were used for high-flow conditions in each year. Comparisons of simulated and observed water-surface elevations were performed using the root mean squared error (RMSE) for all available calibration data. RMSE was calculated as $\sqrt{\sum_{i=1}^{N}\frac{(x_{oi}-x_{si})^2}{N}}$, where x_{oi} is the observed water-surface elevation, x_{si} is the simulated water-surface elevation, and N is the number of data points. RMSE values were determined for low-flow conditions in 1994 (0.131 m), 2000 (0.114 m), 2004 (0.115 m), and

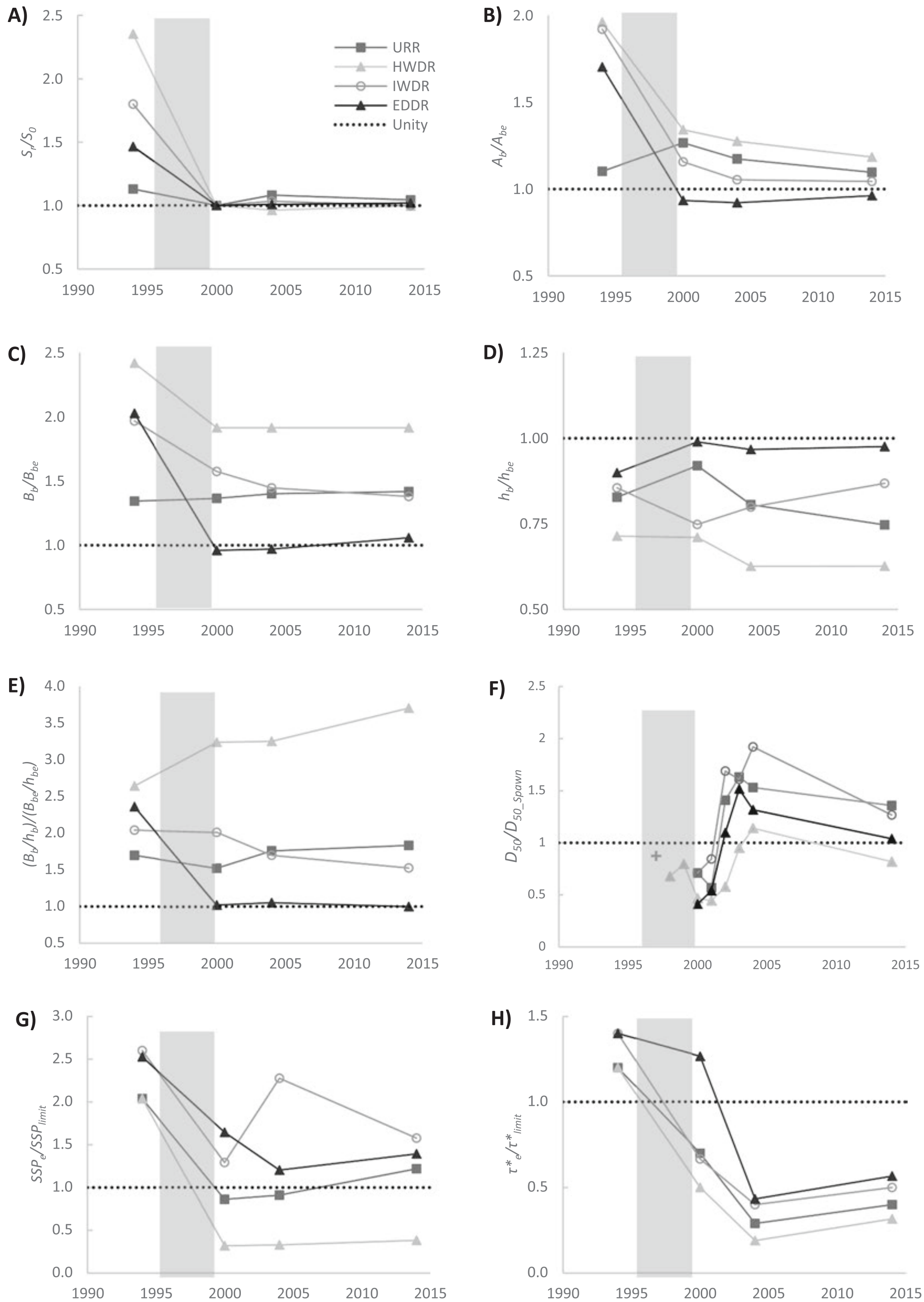

Fig. 2. Time series of normalized median values for A) reach slope (S_r)/initial slope (S_0), B) bankfull cross-sectional area (A_b)/Q_e-design area (A_{be}), C) bankfull width (B_b)/Q_e-design width (B_{be}), D) bankfull hydraulic depth (h_b)/Q_e-design hydraulic depth (h_{be}), E) bankfull width/depth ratio (B_b/h_b)/Q_e-design width/depth ratio (B_{be}/h_{be}), F) median grain size (D_{50})/intermediate spawning size of 45 mm (D_{50_spawn}), G) specific stream power during Q_e (SSP_e)/lower stability limit of 10 W/m^2 (SSP_{limit}), and H) Shields stress during Q_e (τ^*_e)/lower mobility limit of 0.03 (τ^*_{limit}). Shaded area identifies restoration period.

2014 (0.096 m), and for effective discharge in 2014 (0.077 m). Because constructed bankfull channel dimensions differ between each reach, metrics of energy expenditure and sediment mobility were calculated at constant Q_e for specific stream power (SSP_e) and Shields stress (τ^*_e) as:

$$SSP_e = \frac{\rho g Q_e S_e}{B_e} \quad (1)$$

and

$$\tau^*_e = \frac{\tau_e}{(\rho_s - \rho) g D_{50}} \quad (2)$$

where ρ and ρ_s = density of water and sediment (kg/m^3), respectively, g = gravitational constant (m/s^2), S_e = energy slope, B_e = flow width (m), τ_e = reach-average boundary shear stress (N/m^2), and D_{50} = measured, reach-average, median grain size (m). Additionally, the modeled water-surface elevation (*WSE*) for the 7-day low flow was used to assess the riparian vegetation criterion.

4. Results

4.1. Objective 1: establish dynamic equilibrium for the reconstructed channel morphology

4.1.1. Temporal trends in channel stability

Channel construction during restoration directly altered stream sinuosity, slope, and cross-sectional dimensions. In the 14 years following construction (2000–2014), average channel slope in each of the reaches did not change significantly from the as-built condition (Fig. 2A). In contrast, temporal changes in bankfull dimensions varied by design (Fig. 2B–E). One metric of dynamic equilibrium is a stable bankfull cross-sectional area (A_b), which occurred within four years for both the EDDR and IWDR (Fig. 2B), although bankfull width (B_b), depth (h_b), and width-to-depth ratio (B_b/h_b) continued to adjust in the IWDR over the following 10 years (Fig. 2B–E). Bankfull dimensions of the HWDR trended toward equilibrium only in terms of A_b, but remained over-widened and exhibited shallowing compared to the Q_e design geometry, resulting in substantial increases in the bankfull width-to-depth ratio (Fig. 2B–E). The URR continued to widen slightly, but shallowed in response to the lowered slope across the study area, and exhibited a similar pattern of declining cross-sectional area as that of the restored reaches, suggesting that changes in adjacent reaches influenced the URR. Further details of channel response are provided in the Supplementary Materials.

Overall, restoration reduced lateral streambank erosion rates (R_{eros}) at meanders (0.90 m/yr pre-restoration vs. 0.17 m/yr post-restoration) and the consequent input of fine sediment. Lateral mobility was lower in the restored reaches than in the URR, with the EDDR having the highest mobility and the HWDR having the lowest mobility of the restored reaches (Fig. 3A). Rates of absolute change in bankfull width (ΔB_b) were used to assess whether point bar deposition rates matched outer-bank erosion rates in meanders, with the EDDR exhibiting the fastest decline in ΔB_b following restoration (Fig. 3B). A stable cross-sectional area and small values of ΔB_b in the face of active lateral mobility is another indicator of dynamic equilibrium; in this sense, the EDDR showed the greatest degree of dynamic equilibrium.

4.1.2. Reduced hydraulic drivers of erosion

Because restoration construction decreased channel slope and widened cross-sectional dimensions to promote deposition in some reaches, median specific stream power (SSP_e) at Q_e decreased by 61% over the entire restoration site (p = 0.003) relative to pre-restoration conditions. SSP_e varied by channel design, but was within the range of values typically observed in meadow systems (10–30 W/m^2, e.g., Nanson and Croke, 1992), except for the HWDR, which exhibited exceptionally low specific stream power (Figs. 2G, 4A). Similar to SSP_e, the median Shields stress at Q_e (τ^*_e) decreased by 39% across all reaches (p = 0.003) during restoration (1994–2000) due to reduced channel slope (Fig. 2A) and increased grain size (Figs. 2F, H, 4B). However, median τ^*_e values were < 0.03 in most cases, indicating only partial bed mobility during Q_e (Figs. 2H, 4B) and a possible lack of equilibrium transport (supply = transport capacity). Similar trends in SSP_e and τ^*_e were observed in the URR to those in the restored reaches in response to reduced channel slope from adjacent reach restoration.

4.2. Objective 2: improve aquatic habitat conditions for salmonids

4.2.1. Channel morphology

The number of habitat units generally increased following restoration, in part from creation of a longer channel, but then declined during three high-flow events between 2004 and 2014 that reorganized the channel topography (Fig. 5A). Similar trends were observed for the spatial extent of habitat units, although pool area increased following flood reorganization (Fig. 5B). By 2014, the number and spatial extent of pools exceeded pre-restoration values, while the number and spatial extent of riffles and runs/glides were less than or equal to the pre-

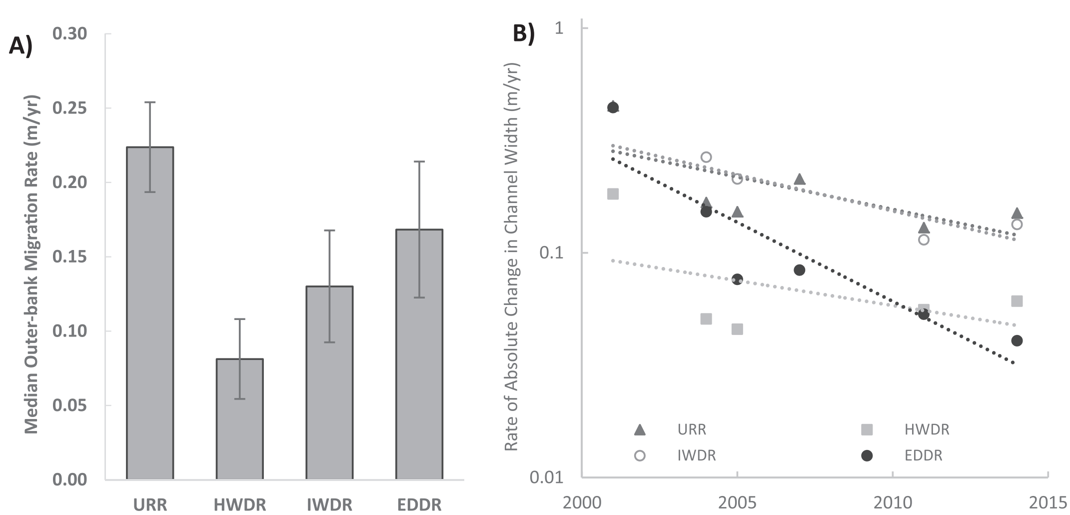

Fig. 3. A) Median lateral migration rate (R_{eros}) of meanders in each reach (2000–2014). Error bars represent standard errors. B) Rates of change in absolute median width (ΔB_b) by reach. Lines depict exponential trends (URR R^2 = 0.55, HWDR R^2 = 0.38, IWDR R^2 = 0.88, EDDR R^2 = 0.83). Some reaches have missing years due to partial surveys.

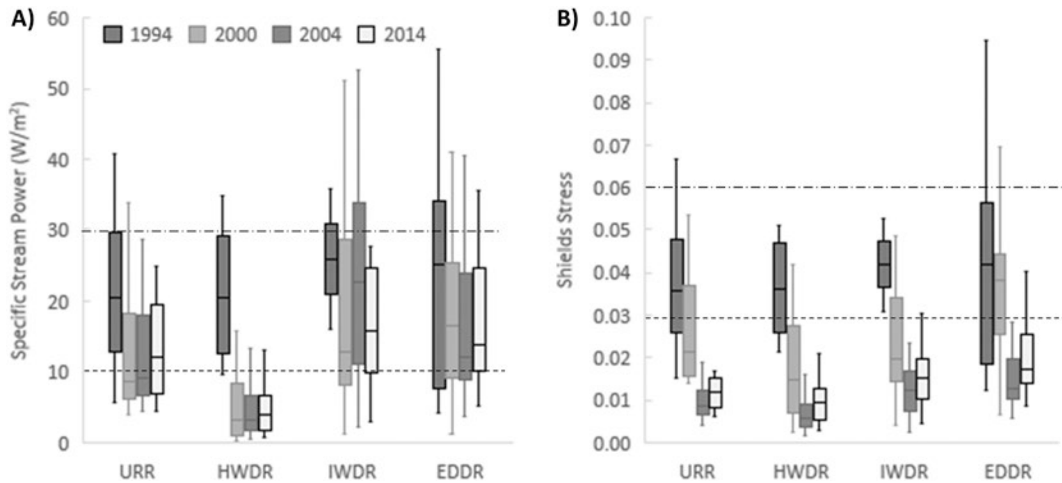

Fig. 4. Boxplots showing pre-restoration (1994) and post-restoration (2000, 2004, 2014) effective discharge values of A) specific stream power SSP_e and B) Shields stress (τ^*_e) for each reach. Horizontal lines indicate the range of *SSP* (10–30 W/m^2) values common in meadow systems in (A) and the threshold for bedload transport τ^* (0.03–0.06) typically observed in gravel-bed rivers in (B).

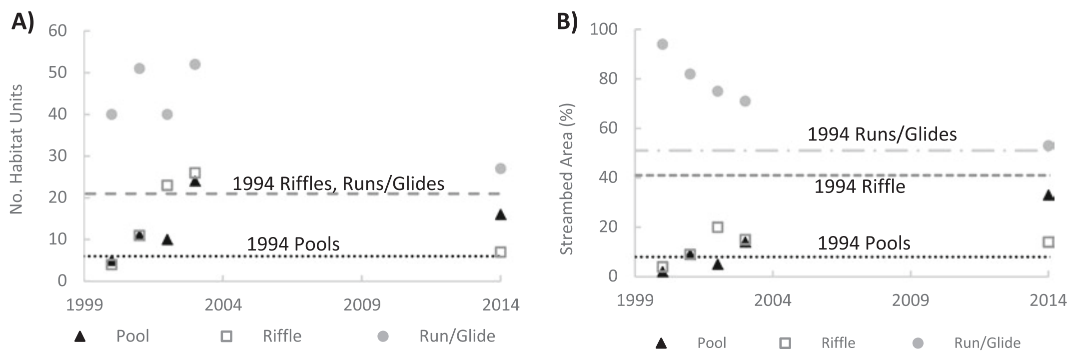

Fig. 5. A) Number of morphologic units within the study site between 2000 and 2014. B) Percent bed area of pools (triangles), riffles (boxes), and runs/glides (circles). Horizontal lines depict pre-restoration conditions (1994).

restoration conditions.

4.2.2. Grain-size trends

For each reach, the percentage of fine sediment ≤1 mm and ≤5.6 mm (detrimental to survival of salmonid embryos incubating within the streambed) generally increased in the first year after construction in response to bed and bank disturbance, as well as decreased channel slope and lower competence, but subsequently declined to stable values at or below the target percentages (Fig. 6A, B). Responses varied between the reaches, with the HWDR taking eight years to attain the 5.6 mm objective and then increasing above the threshold by 2014, while the EDDR met it two years after restoration (Fig. 6B). By 2014 the percentage of fine material roughly tracked the values of SSP_e (Fig. 4A), with the least fines in the IWDR (which maintained the highest SSP_e) and the most fine material in the HWDR (the lowest SSP_e). Reductions in both streambank erosion (Fig. 3A) and the percentage of fine sediment (Fig. 6A, B) over time following restoration contributed to bed-surface coarsening (Fig. 6C, D), with a slight decline in grain size following the high-flow events between 2008 and 2011 that reorganized the channel topography. Furthermore, the percentage of spawning substrate (16–128 mm) generally increased in all reaches following restoration (Fig. 6C), with the median grain size meeting the target value ($D_{50} = 45$ mm) in all but the HWDR (Fig. 2F).

4.3. Objective 3: increase hydraulic connectivity to support native riparian vegetation

4.3.1. Floodplain inundation extent

Simulated flows that bracket the effective discharge (16–22 m^3/s) and span 2–6-year recurrence intervals show that the incised, pre-restoration stream channel (1994) provided limited floodplain connectivity in terms of flows spilling onto the floodplain (Fig. 7A). Increased inundation is predicted for all reaches following restoration (Fig. 7B–D), but remains spatially limited in the URR (2–5% of the cross sections). Modeled inundation extent increased through time due to a combination of decreasing B_b and h_b in the restored reaches (Fig. 2C–E) that elevated water-surface elevations, but was limited in the HWDR due to its enlarged channel area that was capable of containing most high flows (Fig. 2B). Collectively, the results show that restoration activities directly increased total floodplain inundation within the restoration site, which was subsequently increased by evolution of bankfull dimensions, especially in the IWDR and EDDR.

4.3.2. Low-flow water-surface elevations to support riparian vegetation

Floodplain inundation in the spring can recharge the local groundwater within the meadow, but maintenance of soil moisture within the riparian rooting depth late in the growing season is dependent on the river water-surface elevation (*WSE*), an indicator of the local groundwater table. Differences in elevation between the simulated summer low-flow *WSE* and the top of the streambank were computed to evaluate the restoration goal of ≤1 m difference in elevation. Median differences

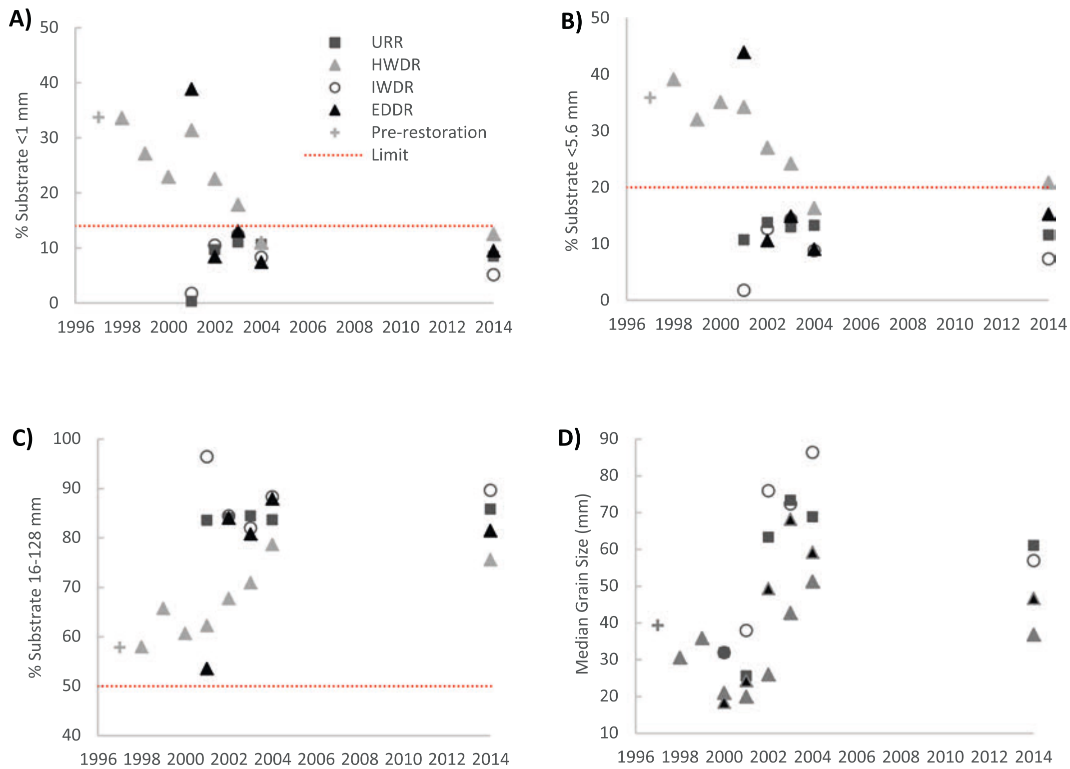

Fig. 6. Time series of percentage of streambed substrate A) ≤1 mm, B) ≤5.6 mm, and C) 16–128 mm in each reach. Points represent the ensemble value of all data in a given design reach. Stippled lines mark the limits for suitable habitat, A) 14% ≤1 mm, B) 20% ≤5.6 mm, and C) 50% 16–128 mm. Panel D) shows temporal changes in median grain size by reach.

in elevation were greater than 1 m in all reaches prior to restoration because of historic channel incision, but systematically declined over time following restoration (Fig. 8). The HWDR effectively satisfied the criterion immediately flowing restoration, whereas the EDDR was the furthest from the threshold in 2000 and had the greatest change in median elevation difference (20%, 0.24 m) over the 14-year monitoring period.

4.3.3. *Riparian vegetation response*

Repeat surveys showed that native vegetation response after restoration varied based on proximity to the river, with a 12% reduction in median native vegetation cover in transects parallel to the water's edge and a 15% increase in transects perpendicular to the channel (Fig. 9A). In contrast, invasive cover increased both on the floodplain (perpendicular transect) and along the water's edge (parallel transect) (Fig. 9A), with the latter mainly due to colonization of reed canary grass (*Phalaris arundinacea*) in areas with laterally migrating banks. Vegetation response varied by reach, but exhibited an overall decline in native cover along the water's edge by 2013 and failed to meet target values of >50% cover, except in the HWDR (Fig. 9B). The perpendicular transects across the floodplain showed greater native cover than the parallel transects, but values remained fairly stable in the restored reaches, with the largest increase occurring in the URR (Fig. 9B). Invasive reed canary grass was the dominant cover (66–84%) in all reaches along the water's edge by 2013, except in the HWDR owing to low lateral migration rates (and consequently low bank erosion) in that reach (Fig. 9C). Invasive vegetation also increased on the floodplain in all reaches, with the greatest increase occurring in the IWDR due to its larger extent of flood inundation (Fig. 7), which promoted invasive colonization.

5. Discussion

Twenty years of monitoring channel evolution at Red River revealed that if no restoration had occurred, the channel would have continued to widen and shallow, have limited hydraulic connectivity with the floodplain, and maintain a degraded riparian ecosystem. Restoration success, in terms of putting the channel on a trajectory toward recovery of the 1930s conditions, depended on the design approach. Of the response variables monitored, channel slope remained closest to the target, as-built condition and had the lowest rate of change (Fig. 2A), which is consistent with previous research (e.g., Leopold and Maddock, 1953; Grams and Schmidt, 2005; Ziliani and Surian, 2012; Naito and Parker, 2020). Over a longer timeframe than the 14-year monitoring period, evolution of individual bends will modify reach-scale channel sinuosity and slope, particularly where meander cutoff processes occur. The estimated time to meander cutoff (T_c) and subsequent change in sinuosity and slope is 271 years on average for the entire study site, with T_c ranging from 144 years in the URR to 403 years in the HWDR. However, large overbank floods may induce more rapid evolution through chute or neck cutoff processes (Constantine et al., 2010; van Dijk et al., 2012, 2014; Harrison et al., 2015).

At the decadal scale monitored here, streambank erosion promoted active meander migration, providing a laterally dynamic channel (Fig. 3A) that affected cross-sectional dimensions (Fig. 2B–E), bed substrate (Fig. 2F), hydraulics (Fig. 2G–H), and ultimately the evolutionary trajectory of each design reach. Results demonstrate that 1) the channel geometry in the EDDR stabilized within approximately 4 years after restoration, 2) the IWDR attained a stable cross-sectional area within 4 years (indicating no net erosion or deposition and, thus, dynamic equilibrium), but was still adjusting its bankfull width and depth 14 years afterward, and 3) the HWDR generally did not evolve toward the desired channel dimensions during the monitoring period. The over-widened design of the HWDR was intended to promote lateral deposition and channel narrowing over time, but surprisingly B_b/h_b values increased (Fig. 2E). This undesired response trajectory did not result from a lack of sediment supply since A_b declined by 12% in the HWDR

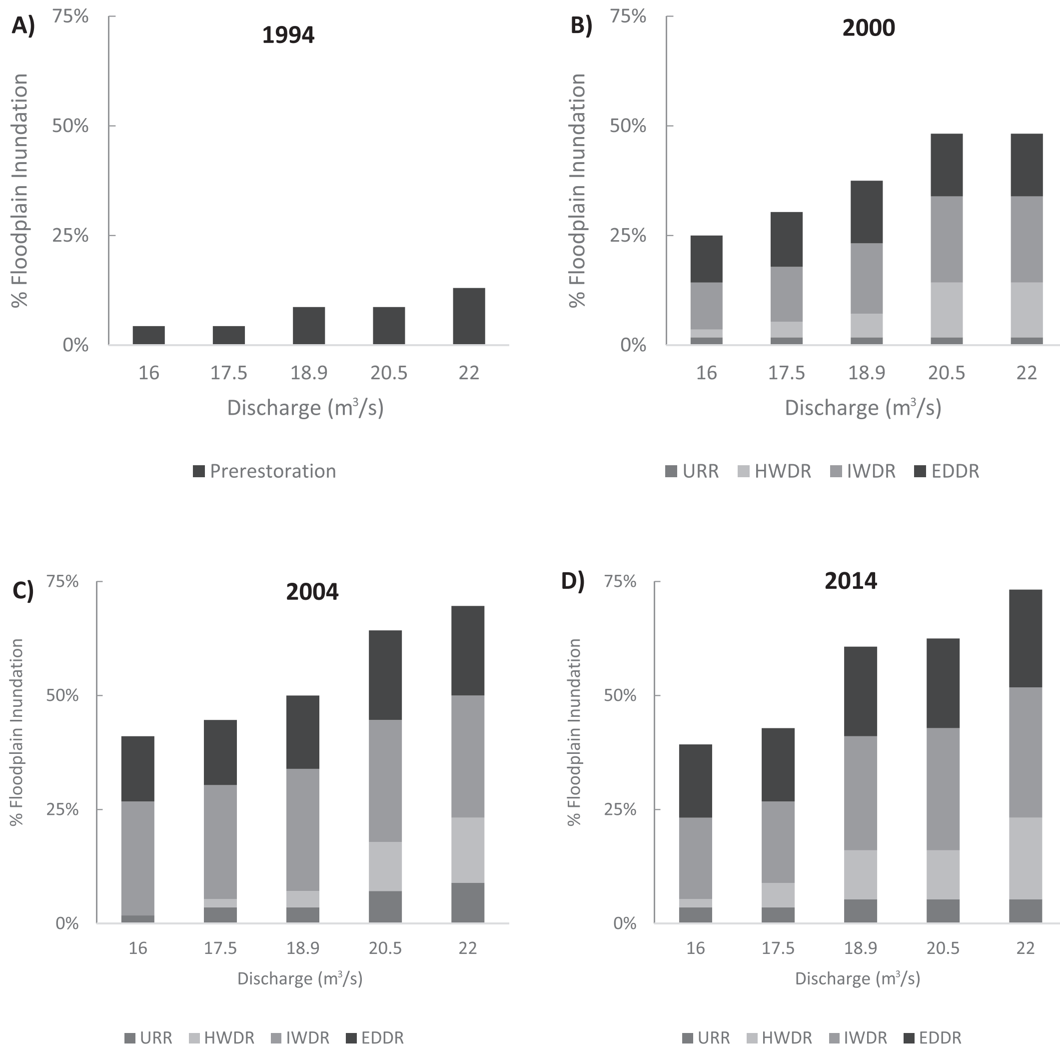

Fig. 7. Percentage of cross sections in each design reach that are predicted to overtop their banks and inundate the floodplain during near-Q_e discharges (16–22 m^3/s) in A) 1994, B) 2000, C) 2004, and D) 2014.

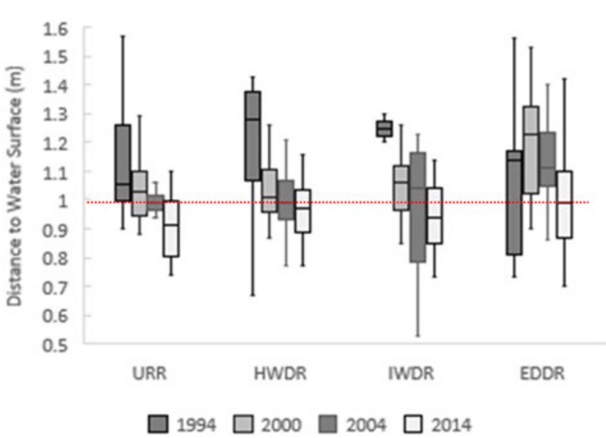

Fig. 8. Distance between simulated low-flow water-surface elevation and top of bank for each reach over time. Dashed line indicates the ≤1 m restoration criterion to support soil moisture levels for riparian vegetation.

following restoration (Fig. 2B). Instead, the over-widened channel led to low values of both SSP_e and τ^*_e (Fig. 2G, H), primarily inducing bed deposition and decreased hydraulic depth (Fig. 2D). Therefore, if restoration involves over-widening to reduce outer-bank erosion rates and promote channel narrowing, the resultant instream deposition may negatively impact the desired evolutionary trajectory of the stream if over-widened too much beyond the Q_e design.

To assess the hydraulic drivers of channel change, specific stream power (*SSP*) can be used as a metric for channel response (McEwen, 1994; Kale, 2008; Tranmer et al., 2015, 2020b). As *SSP* increases above approximately 30–35 W/m^2 channel widening and instability may occur, whereas values below ~10 W/m^2 can lead to fine sediment deposition in meadow settings (Brookes, 1987; Nanson and Croke, 1992; Bizzi and Lerner, 2015). In Red River, the upper quartiles of SSP_e in the pre-restoration channel ranged between 20 and 35 W/m^2 (Fig. 4A), which were likely responsible for the historic bank erosion and widening that occurred. By 2014, the upper quartiles in all reaches were less than 25 W/m^2, well below the threshold of instability. However, peak SSP_e in the EDDR reached 35 W/m^2, which likely drove the elevated lateral channel migration rates in that reach (Fig. 3A). While SSP_e values were initially variable following restoration (2000), 14 years later median SSP_e values in the URR, IWDR, and EDDR converged to approximately 14 W/m^2. However, 47% (URR) and 89% (HWDR) of the SSP_e distribution had low SSP_e values (<10 W/m^2, Fig. 4A), suggesting potential for further bed deposition and sediment fining in these reaches depending on flow regime and upstream inputs of sediment. In terms of sediment mobility, self-formed gravel-bed rivers typically have a near-bankfull threshold for bedload motion (Andrews, 1984; Mueller et al.,

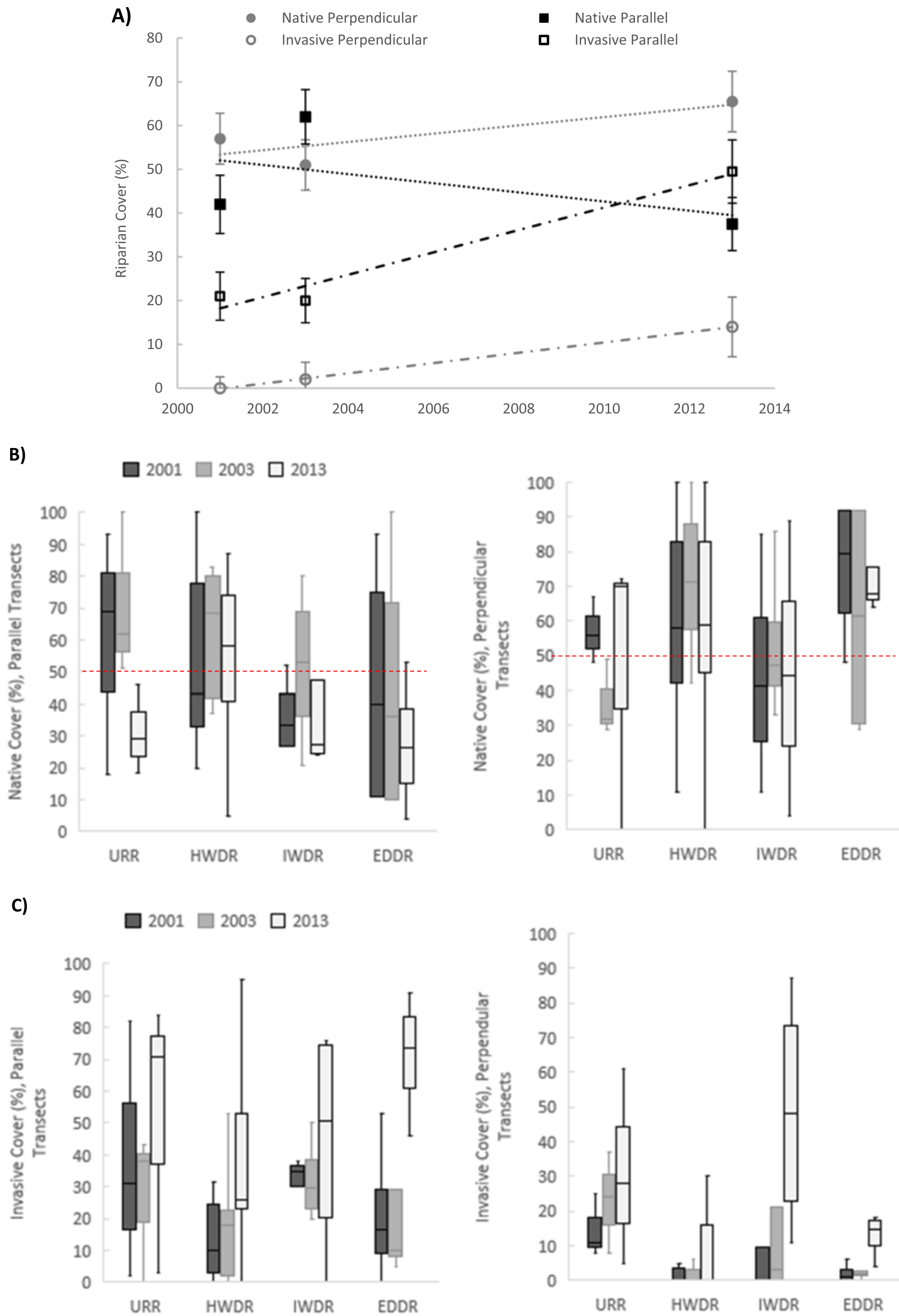

Fig. 9. A) Percent native and invasive riparian vegetation cover across the restoration site. B) Box plots of percent native riparian vegetation in each channel design for transects parallel (left) and perpendicular (right) to the channel. Horizontal lines indicate the restoration goal of >50% native vegetation. C) Same as (B), but for invasive reed canary grass.

2005; Bunte et al., 2010), with Shields stresses (τ^*) between 0.03 and 0.06 for channels with slopes of 0.001–0.01 (Buffington and Montgomery, 1997; Dade and Friend, 1998; García, 2000; Church, 2006; Buffington, 2012), similar to the pre-restoration state (Fig. 4B). However, lower Shields stresses have been reported in other meadow systems at flows above bankfull (McKean and Tonina, 2013). At Red River, post-restoration τ^*_e values were intentionally designed to be low to reduce the potential for erosion, but median τ^*_e values < 0.03 (Fig. 4B) indicate limited coarse bed mobility during effective discharge (e.g., Andrews and Parker, 1987; Lisle, 2005) and highlight the challenge for restoration designs intended to reduce bank erosion, while maintaining sufficient competence for bedload transport. The low mobility of coarse bed sediments through the restoration site supports the interpretation that bank erosion and lateral channel adjustment are the primary modes of geomorphic change through this meadow system. It also should be stressed that 1D models, like that used in this study, only capture average hydraulic conditions, while cross-sectional variability of SSP_e, τ^*_e, and D_{50} could result in greater sediment mobility through portions of the channel (e.g., Ferguson, 2003; May et al., 2009; Monsalve et al., 2020).

At Red River, channel morphology, as a metric of aquatic habitat, showed an overall increase in morphologic complexity after project completion (Fig. 5). However, changes during later high-flow events diminished the initial gains in habitat abundance. In particular, the number of pools declined (Fig. 5A) as smaller features were washed out or merged into larger features that ultimately increased overall available pool area (Fig. 5B). While the available monitoring data did not allow for areal evaluation of micro-scale habitat diversity seen in higher-resolution datasets (e.g., Wyrick et al., 2014; Passalacqua et al., 2015), mesoscale habitats showed continuous improvement over time (Fig. 5). Further, elevated bank erosion rates in freely migrating reaches (like the EDDR) have been shown to increase geomorphic diversity that supports ecosystem function for a variety of aquatic and terrestrial species (Florsheim et al., 2008; Miller et al., 2012; Bellmore and Baxter, 2014; Williams et al., 2020). In contrast, the rapid increase in the percentage of spawning substrate in the restored reaches was driven, in part, by reduced fine material inputs from lower streambank erosion rates (shifting the grain-size distribution toward coarser sizes) and by lower channel competence from reduced SSP_e and τ^*_e (Fig. 2G, H) that may have promoted deposition of coarse sediment inputs from upstream. Collectively, these conditions led to improved bed grain-size distributions for salmonid embryo development within 2–5 years in the restored reaches (Fig. 2F), but response times in other rivers will likely be site-specific based on basin lithology, sediment supply, and available stream energy (Mueller and Pitlick, 2013; Mueller et al., 2016). Therefore, while restoration can facilitate equilibrium (i.e., beneficial) bank erosion to increase the rate of evolution and diversity of aquatic habitats, it must balance the input of fine sediment to maintain the appropriate bed substrate and habitat conditions through meadow systems. The transport processes to maintain these grain-size distributions in the river would require the transport of finer material over the top of a less-mobile coarse bed. In channels with low τ^*, the transport of finer sediments over a relatively coarse stable bed has been observed (e.g., Wilcock and McArdell, 1997; Yager et al., 2012; Grams and Wilcock, 2015) and reported in relatively undisturbed meadow systems that support productive salmonid populations (e.g., McKean and Tonina, 2013; Maturana et al., 2013). This is corroborated by measurements of King et al. (2004) for the two upstream Red River gages that show movement of a sandy bedload over an armored bed surface.

The restoration site was also subject to exogenous sediment supply from the upstream watershed (independent from local restoration activities). The cumulative effects of restoration decreased channel slope (Fig. 2A) and lowered the overall energy expenditure and Shields stress (Fig. 4) by increasing the channel length and sinuosity. The resultant deposition in the over-widened channels (URR, HWDR, and IWDR) led to sediment storage and reduction in bankfull area, whereas the EDDR maintained sediment continuity and stable cross-sectional area (Fig. 2B). At a coarse scale, the contiguous reaches were subject to nearly identical upstream inputs of water and sediment from the basin, however local conditions may have modified the signal to downstream reaches. In this regard, relative slope and energy expenditure through the restoration site can provide a surrogate for sediment transport and indicate how individual reaches will respond to and modify the imposed upstream sediment inputs (e.g., Parker et al., 2015; Gartner et al., 2015; Kasprak et al., 2016). When the slope of a given reach is steeper than the one immediately upstream, the channel is expected to be supply-limited relative to upstream conditions and is likely to narrow, deepen, and/or coarsen its bed (e.g., Schumm, 1985; Dietrich et al., 1989). Relative slopes through the restoration site at Red River indicate that the URR and HWDR were locally transport-limited (less steep than the upstream reach), which may explain why the over-widened design in the HWDR retained high amounts of fine bed material (Fig. 6A) even though the low lateral erosion rates (Fig. 3A) provided less localized fine bank material. Sediment deposition and low lateral erosion rates in the HWDR may have further reduced the fine sediment supply to the narrower and deeper channels in the downstream IWDR and EDDR, promoting coarser bed substrate (Fig. 6) despite low values of SSP_e and τ^*_e in those reaches (Fig. 2G, H). Alternatively, the fact that the IWDR and EDDR were locally supply-limited (steeper than upstream reaches) may have facilitated their rapid response toward the desired conditions, while the URR and HWDR exhibited limited recovery toward the desired Q_e geometry. Collectively, these reach-scale adjustments converged toward the desired cross-sectional area for Q_e (Fig. 2B), albeit at different rates, necessary to maintain long-term sediment continuity through the study area (Fig. 2B), but results suggest that reach-scale adjustments may further modify the basin inputs and future restoration projects should consider the spatial sequence of design reaches to facilitate the desired evolutionary responses.

Successful restoration of aquatic–riparian function requires reestablishing hydraulic connectivity between the river and floodplain, as well as the presence of a suitable seedbank. At Red River, there was a sufficient native riparian seedbank both from existing riparian communities and restoration planting to support hydrochory. Additionally, there was continuous improvement in hydraulic connectivity between the channel and floodplain at both high (Fig. 7) and low flows (Fig. 8) to support pioneer vegetation species, but the initial increases in native riparian vegetation from 2001 to 2003 were subsequently diminished over the entire site (Fig. 9A). The limited success of native species in certain reaches after restoration illustrates that, while soil moisture and wildlife browsing can be critical controls on riparian vegetation (e.g., Busch et al., 1992; Tabacchi et al., 1998; Naiman et al., 2005; Beschta and Ripple, 2012), bank erosion is a further control (e.g., Gurnell et al., 2016; Corenblit et al., 2016). Successful recruitment and maintenance of native riparian species tends to occur along the high/intermediate flood-frequency interface adjacent to the channel that experiences active erosional and depositional processes (Mahoney and Rood, 1998; Corenblit et al., 2007, 2015). Reaches with greater R_{eros} (Fig. 3A), either through channel migration or general widening, led to bare mineral soil and readily accessible soil moisture that was ideal for pioneer vegetation species to colonize (e.g., Gurnell, 2014; Benjankar et al., 2020). However, Tranmer et al. (2020a) found that native riparian vegetation recruitment was more successful in depositional areas than erosional ones. This could explain why the erosional zones near the channel were preferentially colonized by mats of invasive reed canary grass, which have extensive rhizomes that grow rapidly after hydraulic disturbances and outcompete seedlings of native species (e.g., Reinhardt-Adams and Galatowitsch, 2005). The reaches with the highest R_{eros}, like the EDDR and URR, had the highest percentage of reed canary grass cover (Fig. 9C), while the HWDR, with the lowest R_{eros}, was able to maintain a dominant cover of native vegetation (Fig. 9B). This is particularly evident in the period between 2004 and 2014 that had the three highest flood events for driving bank erosion.

The vegetation transects perpendicular to the channel demonstrate greater success of native vegetation on the floodplain due to increased inundation and relatively less erosive environments compared to the channel margins (Fig. 9A). However, while increased seasonal floodplain inundation may recharge groundwater levels, it can also distribute invasive plant propagules like reed canary grass farther from the channel, causing competition with more desirable native plant communities, as was the case for the IWDR that had the greatest increase in floodplain inundation (Fig. 7). These findings suggest that restoring biophysical processes to support native riparian ecosystems may also create conditions that accelerate propagation of invasive species, requiring active intervention until native species mature (Fierke and Kauffman, 2005; Hladyz et al., 2011). Results from Red River indicate that the establishment of native riparian vegetation communities in wet-meadow systems requires at least 14 years of active management when invasive species are present, but the recovery period for other streams will likely depend upon local hydrologic conditions, length of growing season, browsing impacts, and species-specific growth rates (Wohl et al., 2005; Jowett et al., 2009; Feld et al., 2011; Beschta and Ripple, 2012).

In addition to documenting post-restoration response, a primary objective of the current analysis was to test the efficacy of a single-discharge design for stream restoration (i.e., the effective-discharge paradigm). Specifically, testing whether dynamic equilibrium can be achieved from a single-discharge design and whether dynamic equilibrium is attained in less time for reaches with bankfull dimensions that are sized more closely to Q_e. Here, dynamic equilibrium is inferred by the occurrence of stable bankfull dimensions in the face of active lateral migration. Channel dimensions evolved toward stable values in the reach constructed closest to the Q_e design (EDDR), as well as in the moderately-over-widened reach (IWDR), but the latter exhibited slower responses for some metrics and did not fully attain the Q_e design dimensions (Fig. 2A–E). For example, the IWDR achieved a stable bankfull cross-sectional area within 4 years, but its width-to-depth ratio was still adjusting after 14 years. Furthermore, while the attainment of stable channel characteristics in concert with active channel migration across the floodplain (Fig. 3A) demonstrates that single-discharge designs can produce dynamic equilibrium, the metrics of grain size, SSP_e, and τ^*_e fluctuated 14 years after restoration (Fig. 2F–H), indicating that they may be more sensitive to variable upstream inputs of water and sediment than channel geometry. Although SSP_e values were typical for meadow systems in all but the HWDR (Fig. 4), values of $\tau^*_e < 0.03$ indicate low mobility (partial transport) and a possible lack of equilibrium transport (supply = transport capacity). The extremely-over-widened channel (HWDR) showed the slowest response, despite rock sills that induced local sedimentation (Fig. 2D) and raised the low-flow water-surface elevation (Fig. 8). Using the 2014 ΔB_b value for the HWDR, indicates that it will take ~290 years for the bankfull dimensions to narrow to the Q_e width. Interestingly, while the URR continued to show adverse channel widening (Fig. 2C and E), other channel characteristics in the URR followed similar patterns of response to those in the EDDR and IWDR (Fig. 2B, F–H), suggesting that imposed reach-scale changes in channel slope from restoration influenced the neighboring URR.

The restoration community has advanced considerably since the 1990s when the Red River project was designed. The importance of factors such as (1) the variability and timing of both flow and sediment supply (e.g., Poff et al., 1997; Buffington and Parker, 2005), (2) natural conditions prior to human disturbance of the landscape (e.g., Walter and Merritts, 2008; Comiti, 2012; Wohl, 2019), (3) basin context (location within the stream network and associated process domains; e.g., Montgomery, 1999; Brierley and Fryirs, 2005, 2009), (4) restoration potential (defined as the limits of physical and ecological processes given societal constraints and human use of river corridors; e.g., Stoddard et al., 2006; Bernhardt and Palmer, 2007; Beechie et al., 2010), (5) large wood dynamics (e.g., Gurnell et al., 2002; Wohl et al., 2019; Sutfin et al., 2021), (6) value of hydraulically connected lentic-lotic habitats (e.g., Fausch et al., 2002; Pander et al., 2019; Favrot and Jonasson, 2020), (7) positive effects of bank erosion on channel diversity (e.g., Florsheim et al., 2008; Williams et al., 2020), (8) ecosystem engineering by flora and fauna (e.g., Rice et al., 2012; Manners et al., 2015; Castro and Thorne, 2019; Johnson et al., 2020; Rice, 2021), (9) effects of vegetation on bank strength and channel morphology (e.g., Beechie et al., 2006; Eaton and Church, 2007; Politti et al., 2018), (10) regime theory for mechanistically predicting channel conditions and response potential (Eaton et al., 2004; Buffington, 2012; Eaton and Millar, 2017), and (11) conceptual evolutionary response models (e.g., Brierley and Fryirs, 2005; Rosgen, 2006; Cluer and Thorne, 2014) are now more broadly recognized. If designed today, the Red River project would likely employ different restoration strategies, use more advanced measurement and analysis technologies (e.g., LiDAR-derived multi-dimensional analyses), and include a broader range of factors for successful restoration. Additionally, some of the design goals might be accomplished through less intensive construction methods (e.g., Zeedyk and Clothier, 2014; Silverman et al., 2019), but may require longer timeframes to achieve the desired objectives. We briefly discuss the importance of some of the above factors in relation to the 1990s approach used at Red River, particularly because many restoration projects continue to use single-discharge designs despite the above advances in our understanding of fluvial systems and restoration design.

Restoration at Red River employed the effective discharge paradigm by sizing bankfull channel dimensions to a single formative discharge instead of designing for a full range of flows. While sizing the bankfull channel to Q_e may produce rapid dynamic equilibrium, as demonstrated in this analysis, channel morphology over time results from a competition between disturbance and recovery processes (e.g., Wolman and Gerson, 1978; Kasprak et al., 2016; Dethier et al., 2016; Fryirs and Brierley, 2016). In particular, flood flows that exceed bankfull can reset the trajectory of channel and riparian conditions (Stevens et al., 1975; Ward et al., 1999; Fryirs et al., 2012; Brogan et al., 2019). After restoration at Red River, there was a period of low hydraulic disturbance between 2000 and 2004, with several near-bankfull flows and one overbank flood (5-year recurrence interval) that increased the abundance of aquatic habitat units and the extent of native riparian vegetation cover along the streambanks. However, three larger floods (5- to 9-year recurrence intervals) between 2008 and 2011 disrupted this trajectory, causing consolidation of habitat units and disturbance of the near-channel vegetation communities that were subsequently colonized by invasive species (Fig. 9).

Bankfull channel designs based on a single discharge neglect flood variation and its role in channel stability (or lack thereof) and the creation of variable geomorphic and ecological conditions (e.g., Buffington, 2017; Davidson and Eaton, 2018; Pander et al., 2019). Consequently, while bankfull channel designs employing the effective discharge paradigm are a useful starting point, they are a simplification of channel dynamics that, in some cases, may produce spectacular failures. For example, a historically braided gravel-bed river in northern California (Uvas Creek) was re-designed as a single-thread, meandering channel based on bankfull stability analyses, but subsequently reverted to a braided morphology during a low-magnitude, 5-year flood (Kondolf et al., 2001). Subsequent regime analyses demonstrated that Uvas Creek is a transitional stream that can exhibit a stable single-thread morphology during bankfull flows, but will transform to a braided morphology during larger floods (Buffington and Parker, 2005), the transition of Uvas Creek highlights the importance of flood variation in controlling channel morphology, as well as having a clear understanding of historic morphologic conditions prior to restoration (Kondolf et al., 2001). Because sediment transport and morphologic response are impacted by the history and variability of flow (e.g., Stevens et al., 1975; Redolfi et al., 2018; Davidson and Eaton, 2018; Wang et al., 2019), future restoration designs that use a single-discharge approach for sizing bankfull channel dimensions should additionally account for a sufficient range of flows to capture potential geomorphic responses (e.g., Costa and O'Connor, 1995; Buffington and Parker, 2005; Shields et al., 2008; Davidson and Eaton, 2018).

The effects of natural and anthropogenic disturbances on basin hydrology and sediment supply also should be considered for successful restoration design (Lytle and Poff, 2004; Yarnell et al., 2015; Buffington, 2017; Poff, 2017; Tranmer et al., 2020c). Although designs based on Q_e implicitly account for flood variation, they typically assume equilibrium transport (supply = transport capacity), a basin-specific condition that varies in natural river systems. For example, the HWDR may be a successful restoration approach in places like coastal California, where pluvial-driven winter storms and high sediment supply induce rapid channel adjustment (Lavé and Burbank, 2004; Warrick and Mertes, 2009; Warrick et al., 2015), but was less effective at Red River because of a snowmelt hydrologic regime with lower-magnitude, longer-duration inputs of water and sediment. Ultimately, contemporary site-specific supplies of both water and sediment should be considered in restoration design (Wilcock, 1997), as well as recognizing that those inputs may not be stationary in the face of climate change, wildfire, and imposed human impacts to the landscape (e.g., Poff, 2017; Tranmer et al., 2020a).

6. Conclusions

Human disturbance of unconfined alluvial valleys typically results in channel straightening and subsequent incision, widening, and disconnection from floodplain and riparian processes that leads to habitat degradation and fluvial disfunction. Stream restoration can successfully reverse this degradation in relatively short time frames using standard practices, but requires a sufficiently long monitoring period to fully understand response trajectories and the effectiveness of management actions, as demonstrated here. In this study, the most successful design in terms of achieving restoration goals in the shortest time period employed sizing the bankfull channel dimensions to the effective discharge (Q_e) and re-occupying historic channels and meander patterns that may intrinsically account for historic flood variability and aquatic–riparian interactions. While dynamic equilibrium was rapidly attained using the Q_e design (EDDR), the moderately-over-widened channel (57% larger, IWDR) still trended toward dynamic equilibrium, but at a slower rate and did not fully achieve the desired bankfull dimensions for the Q_e design over the 14-year monitoring period. Recovery rates were even slower for the extremely-over-widened channel (91% larger, HWDR), with narrowing of the bankfull width to the Q_e design expected to take hundreds of years. Consequently, excessively-over-widened designs (by ~60–100% or more) should be avoided, as they may lead to low rates of change and reduced interactions between the river, floodplain, and riparian zone.

The single-discharge restoration design is a simplifying approach because channel dynamics and ecological conditions are responsive to a range of flood events that can reset response trajectories as shown by the high flows experienced at Red River during post-restoration evolution. Although the efficacy of sizing bankfull dimensions to the effective discharge is demonstrated by this work and gives credence to its continued use in stream restoration (at least as a first-order approach), advances in the restoration community since the inception of the Red River project in the 1990s highlight a host of other factors that should be considered for successful reestablishment of physical and biological processes in alluvial valleys.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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