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REGENERATION OF *SALICACEAE* RIPARIAN FORESTS IN THE NORTHERN HEMISPHERE: A NEW FRAMEWORK AND MANAGEMENT TOOL

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Abstract

Human activities on floodplains have severely disrupted the regeneration of foundation riparian shrub and tree species of the *Salicaceae* family (*Populus* and *Salix* spp.) throughout the Northern Hemisphere. Restoration ecologists initially tackled this problem from a terrestrial perspective that emphasized planting. More recently, floodplain restoration activities have embraced an aquatic perspective, inspired by the expanding practice of managing river flows to improve river health (environmental flows). However, riparian *Salicaceae* species occupy floodplain and riparian areas, which lie at the interface of both terrestrial and aquatic ecosystems along watercourses. Thus, their regeneration depends on a complex interaction of hydrologic and geomorphic processes that have shaped key life-cycle requirements for seedling establishment. Ultimately, restoration needs to integrate these concepts to succeed. However, while regeneration of *Salicaceae* is now reasonably well-understood, the literature reporting restoration actions on *Salicaceae* regeneration is sparse, and a specific theoretical framework is still missing. Here, we have reviewed 105 peer-reviewed published experiences in restoration of *Salicaceae* forests, including 91 projects in 10 world regions, to construct a decision tree to inform restoration planning through explicit links between the well-studied biophysical requirements of *Salicaceae* regeneration and 17 specific restoration actions, the most popular being planting (in 55% of the projects), land contouring (30%), removal of competing vegetation (30%), site selection (26%), and irrigation (24%). We also identified research gaps related to *Salicaceae* forest restoration and discuss alternative, innovative and feasible approaches that incorporate the human component.

Keywords

Cottonwood; Decision tree; Environmental flow; Poplar; Riparian forest; Willow.

1 Introduction

In the Northern Hemisphere, most riparian forests have been historically dominated by foundation species in two genera of the *Salicaceae* family: *Populus* (cottonwoods/poplars) and *Salix* (willows). *Salicaceae*-dominated riparian forests ("*Salicaceae* forests" hereafter) provide important ecosystem services such as habitat for diverse wildlife, organic matter and shade for aquatic life, and an environment for human recreation and aesthetic enjoyment (Naiman et al., 2005). Riparian *Salicaceae* are pioneer species that depend on the hydrologic regime of rivers and associated geomorphic adjustments to complete their life cycle (Karrenberg et al., 2002). Recruitment of new individuals or stands ("regeneration" hereafter) in particular may result from various fluvial processes (Scott et al., 1996, 1997; Gom and Rood, 1999; Cooper et al., 2003), but the conditions for seedling establishment are naturally so restrictive that decades may pass without effective large-scale regeneration (Mahoney and Rood, 1998; Stromberg, 1998). As a result, *Salicaceae* forests are commonly composed of mosaics of relatively even-aged cohorts that established in different years (Johnson et al., 1976). In some regions *Salicaceae* species are highly dominant (e.g., Southwestern U.S.: Stromberg, 1993; Mediterranean and Central Europe: González et al., 2010; Klimo and Hager, 2001), whereas in others they may be a component of a more diverse mix of woody and herbaceous taxa (e.g., Scandinavia, Nilsson et al., 2015; Southern U.S., Simmons et al., 2012; north-western U.S., Naiman et al., 1998).

Salicaceae forests globally are impacted in various ways by human activities (e.g., Rood and Mahoney, 1990; Rood et al., 1995; Johnson, 1992, 1994; 1998; Shafroth et al., 2002; Dufour et al., 2007; Stromberg et al., 2007a; González et al., 2010; Dixon et al., 2012; Scott et al., 2013; Garófano-Gómez et al., 2013, González del Tánago et al., 2016; and many others). The most common dysfunction of *Salicaceae* forests is the severe decrease of fluvial disturbance-dependent regeneration. In virtually all human-impacted rivers, hydrogeomorphic processes are simplified and homogenized, causing regeneration to be limited to a less diverse set of smaller size geomorphically-active landforms, such as abandoned channels, channel margins, alluvial bars and instream areas, compared to unregulated, free-flowing rivers. The problem of reduced regeneration may be overlooked in some rivers because recruitment may continue for years after geomorphic dynamism has ceased, as vegetation colonizes bare areas (e.g., former channels) that experienced a reduction in flooding disturbance (Johnson, 1994, 1998; Shafroth et al., 2002; Stromberg et al., 2010; Stella et al., 2011; Coble and Kolb, 2013). Meanwhile, however, remnant *Salicaceae* forests in the disconnected floodplain experience a sharp decline in regeneration, while established populations age and are replaced by later successional vegetation. The latter includes shade-tolerant trees in wet regions and grasslands and shrublands of drought-tolerant taxa in dry regions, frequently including exotic species (Friedman et al., 1995; Glaeser and Wulf, 2009; González et al., 2010; Merritt and Cooper, 2000; Dixon et al., 2012; Garófano-Gómez et al., 2013; Martínez-Fernández et al., 2017a).

There are hundreds of field- (e.g., Mahoney and Rood, 1998; Johnson, 2000), mesocosm- (e.g., Stella et al., 2010; Guilloy et al., 2011) and modeling-based (e.g., Dixon and Turner, 2006; Harper et al., 2011; Benjankar et al., 2014) studies on the biophysical requirements of riparian *Salicaceae* regeneration, particularly for *Populus* spp.; extensive work on how regeneration has been impacted by human activities (e.g., Cooper et al., 1999; Shafroth et al., 2002); and recommendations for minimizing those impacts (e.g., Hughes and Rood, 2003; González et al., 2010). However, the scientific literature reporting results of management actions to promote *Salicaceae* regeneration is less abundant and particularly scattered: traditionally, restoration of *Salicaceae* regeneration has focused on plantings, influenced by a terrestrial approach from forestry, with uncertain results (Briggs et al., 1994; Stromberg, 2001). Inspired by key advances in river ecology (River Continuum, Vannote et al., 1980; Flood Pulse, Junk et al., 1989; Natural Flow Regime; Poff et al., 1997), controlled releases from dams were applied during the 1990s and provided optimism for effectively restoring *Salicaceae* regeneration, extensively and at a low cost (Shafroth et al., 1998; Hill and Platts, 1998; Rood et al., 2003, 2005). Although legitimate and effective, very few projects reported using this restoration technique alone (e.g., Shafroth et al., 2010; Hall et al., 2011; Foster and Rood, 2017), mainly due to technical and socio-political constraints (Glenn et al., 2017). Other approaches have been attempted with mixed success within a gradient of interventionism: from the abandonment of human activities in the floodplain followed by different degrees of assisted regeneration (Roelle and Gladwin, 1999; Bunting et al., 2011; González et al., 2017a), to land contouring and removal of competing vegetation (Friedman et al., 1995; Taylor et al., 1999; Sher et al., 2002; Cooper and Andersen, 2012; Shafroth et al., 2017), or local controlled flooding using irrigation structures (Sprenger et al., 2002; Bhattacharjee et al., 2008). These have proven to be effective alternatives in some cases to implement alone or in combination with revegetation and dam operations to promote the regeneration of *Salicaceae* forests.

Despite the great variety of restoration approaches, few works have explicitly summarized and/or compared *Salicaceae* forest regeneration attempts across different rivers or river segments (e.g., Briggs et al., 1994; Briggs and Cornelius, 1998; Rood et al., 2005; Bay and Sher, 2008; González et al., 2017a; Glenn et al., 2017). More importantly, even fewer articles have discussed the rationale behind the selection of specific strategies for restoration (Stromberg, 2001). A notable exception is Shafroth et al. (2017), who developed a decision tree to inform restoration actions related to *Salicaceae* establishment in a specific restoration project in the Colorado River delta, including water releases from a reservoir and land contouring. However, that decision tree did not include other widely-used restoration actions, such as planting (Simmons et al., 2012; Caplan et al., 2013; González et al., 2017a), and levee manipulation (Florsheim and Mount, 2002; Rohde et al., 2005; González et al., 2017a; Martínez-Fernández et al., 2017b). Another related decision tree included such actions, but it was created to address restoration in the context of biological control of *Tamarix* spp. and did not provide a comprehensive review of the literature, nor did it focus exclusively on regeneration of *Salicaceae* (Bloodworth et al., 2016).

Given that most of the world's regulated rivers are highly unlikely to recover the level of hydrogeomorphic dynamism necessary for historical rates of *Salicaceae* regeneration, understanding the rationale for implementing and in some cases combining restoration approaches is important to guide land managers in efforts to regenerate *Salicaceae* forests. Here, we have reviewed experiences in restoration of *Salicaceae* forests published in the scientific literature to construct a new decision tree to inform restoration planning in any river in the world where regeneration of *Salicaceae* is impaired. The decision tree explicitly links the well-studied biophysical requirements of *Salicaceae* regeneration to specific restoration actions. Our review also serves to identify research gaps in the restoration of *Salicaceae* forests and suggests how it can be improved in the future with alternative, more innovative and feasible approaches which take into account the human component to manage this ecosystem type.

2 Materials and methods

2.1 Organization of the decision tree

As the ultimate goal of this work is to provide resource managers with a tool to help them plan the regeneration of *Salicaceae* forests, we have organized the text of the article following the branches of the decision tree presented in **Figure 1**. The tree follows the establishment requirements of seedlings and planted individuals ordered chronologically by the plant life-cycle. If a requirement is not met, specific restoration actions that could provide the missing requirement are suggested. The actions, however, are not mutually exclusive and may be combined according to specific project needs. Each restoration action (e.g., vegetation removal) could be implemented through a family of restoration techniques (e.g., using bulldozers, or herbicides, or root rakes, etc.). Our primary aim in this article is to provide guidance for determining which actions to take under different sets of conditions during different steps of establishment; further details regarding techniques and implementation can be found in the Supplementary Data. For each establishment requirement, we first describe how it relates to the life-cycle processes and then illustrate the restoration actions with examples of restoration projects and pilot field-based studies published in the peer-reviewed literature. In the Supplementary Data, we also refer to approaches for assessing whether establishment requirements are met.

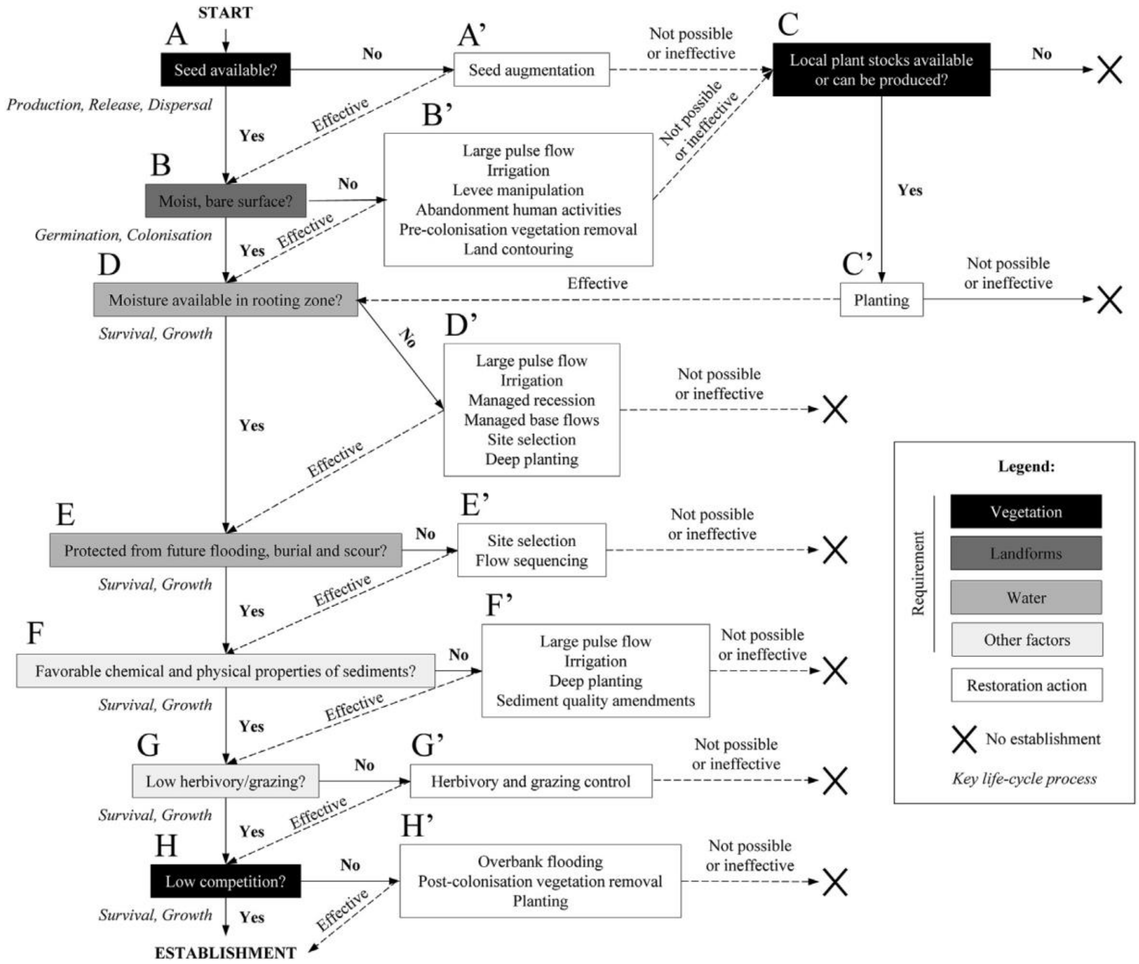

Figure 1. A stepwise dichotomous decision tree for restoring the regeneration capacity of *Salicaceae* forests. Boxes in the tree represent establishment requirements of seedlings and planted individuals. They are formulated as Yes/No questions, and are presented sequentially, following a chronological order related to the life-cycle of the plants associated with characteristics of vegetation, landforms, water and other factors. If a requirement is met ("Yes"), then the key life-cycle processes (in italics) occur and the next requirement is examined (downward facing arrows). "No" answers in the tree lead to white boxes that list restoration actions that would help meet the given requirement (actions appear underlined in the main text). If a chain of requirements is not met ("No"), then no establishment is likely (dead-end indicated by a cross). Dam and weir removal can address many of the requirements for *Salicaceae* recruitment and are increasingly being considered as a river restoration action (O'Connor et al., 2015). However, we did not find any article that reported *Salicaceae* recruitment following dam removal and therefore it was not included in the restoration actions of the decision tree.

We define *establishment* as the recruitment of new *Salicaceae* individuals (and stands, consequently), either by seeds arriving from local populations, artificially sown, or planted poles or rooted saplings. Establishment can be seen as the final step in a seedling life-cycle, after the key processes of seed production, release, dispersal, germination, seedling colonisation, survival and growth. Most authors consider seedlings to have established if they survive the first year (Roelle et al., 2001; Shafroth et al., 2017). However, other authors have considered a longer time-frame: e.g., two (Rood et al., 1998), three (Cooper et al., 1999; Rood et al., 2007), four (Rood et al., 2016) or five (Rood and Mahoney, 2000) growing seasons, as the mechanical and physiological resistance of recruits increases non-linearly and differently among species and site conditions influencing growth rates during the first few years (Corenblit et al., 2016).

Perhaps a more important consideration of establishment is how it manifests at larger spatial and longer temporal scales; that is, whether or not forest stands are created. Arguably, an important measure of success is whether or not the rate of creation of new forest stands is sufficient to compensate for losses by mortality at a minimum spatial scale that reflects the shifting steady state mosaic nature of riparian ecosystems (Johnson et al., 1976; Bormann and Likens, 2012). Determining this requires evaluation over a multidecadal time scale, as natural recruitment is episodic (Mahoney and Rood, 1998; Stromberg, 1998). This evaluation is possible with historical analyses of vegetation dynamics in the study area (e.g., vegetation mapping, dendrochronology) and long-term monitoring of the restoration works, but it is out of the scope of this study. Being able to recreate a new shifting steady state mosaic in degraded rivers, not only promoting regeneration, is a major challenge for long-term success of *Salicaceae* forest restoration.

2.2 Selection of articles

The articles we used to illustrate the restoration actions developed in the decision tree were found systematically based on a literature search done in ISI Web of Science on August 24th, 2017 using the following chain of keywords: "(riparian or floodplain or river or stream) and (resto* or rehabilit* or recover* or remov* or reforest* or planting) and (*Populus* or cottonwood* or poplar* or *Salix* or willow*)". This search yielded 1392 articles, which we evaluated for the following criteria for inclusion in this review. To be used, the project covered in the article had to: i) include the promotion of regeneration of at least one *Salicaceae* species, ii) be an actual completed or ongoing restoration project or, alternatively, a field experiment specifically designed to improve knowledge of restoration actions and techniques, iii) have occurred on freshwater courses, and iv) have been written in English and published in SCI indexed journals or in *Ecological Restoration*, a non-SCI indexed journal of the Society for Ecological Restoration. We limited our search to the scientific literature because screening unindexed technical reports from around the world was not feasible. We acknowledge, however, that many restoration projects have likely been reported only in the grey literature.

We excluded numerous papers that claimed to restore *Salicaceae* forests but only focused on the conservation or maintenance of mature, existing populations without promoting regeneration, even though we acknowledge their importance and value as alternatives for management when promoting *Salicaceae* regeneration is not possible (dead-ends represented by crosses in **Figure 1**). Many of these excluded articles included restoration actions listed in the decision tree, such as water releases from reservoirs and diversion channels to manage base flows. Water releases may help to replenish aquifers, increase groundwater levels and promote growth and survival of existing *Salicaceae* forests but not to promote establishment if, for example, they are not timed with seed dispersal nor create new recruitment sites through channel migration, as was the case for the Tarim and Ejina water conveyance projects in China (Zhu et al., 2016; but see Aishan et al., 2013, 2015). Some of the retained papers, however, included both actions to promote the regeneration and conservation or maintenance of mature populations, as many restoration actions are multi-purpose (e.g., Shafroth et al., 1998, 2010; Hall et al., 2011; Foster and Rood, 2017).

We also excluded articles reporting projects with actions that unintentionally promoted *Salicaceae* regeneration while reducing flood risk. This group of articles included flood releases from dams to evacuate excess water (Zamora-Arroyo et al., 2001; Nagler et al., 2005), operations to improve water conveyance in river channels, such as vegetation removal and mechanical alteration of floodplain and river channels (Geerling et al., 2008), and flood pulses from dams to restore in-channel habitats and scour vegetation on gravel and sandbars (e.g., Kearsley and Ayers, 1999; Stevens et al., 2001). Fifty-nine articles met all these criteria and were retained. Twenty-two more articles that were not found by the automatic search but were cited in at least one of the 59 articles were added to the selected literature because they fit the aforementioned criteria. Twenty-four additional articles were also added based on our professional judgment of their fit with the goals of this study. Thus, a total of 105 articles were ultimately included in the review (Appendix S1).

2.3 Other considerations

We acknowledge that promoting the establishment of *Salicaceae* forests is not always desirable from either an ecological, or socio-political (e.g., *Salicaceae* establishment can increase flood risk), standpoint. Although cultural and aesthetic preferences, as well as indicators used for monitoring ecosystem health, usually favor *Salicaceae* forests over other vegetation types, riverine, non-woody wetlands also can be disfavored by river regulation and human impacts (Stromberg, 2001; Weisberg et al., 2013). In some cases, *Salicaceae* forests may be expanding beyond their historical "natural" limits, displacing other vegetation communities, as a result of human impacts (Johnson, 1994). Before applying our decision tree, it is necessary to determine if the restoration of *Salicaceae* forests is ecologically justified, which is beyond the scope of this study.

It is also important to note that although our focus here is on active planting and sexual regeneration, asexual reproduction is an effective alternative for the spread of *Salicaceae*. In fact, clonal growth can be a much more efficient way of regeneration of *Salicaceae* forests in some species (e.g., *P. trichocarpa*, *S. exigua*), despite being largely overlooked in riparian studies (but see Gom and Rood, 1999; Barsoum, 2002; Barsoum et al., 2004; Douhovnikoff et al., 2005; Moggridge and Gurnell, 2009). Nevertheless, vegetative reproduction alone would not allow for the necessary genetic exchange to adapt to environmental change and sustain *Salicaceae* populations in the long run (Rood et al., 2007; Tiedemann and Rood, 2015). Therefore, the decision tree presented here is focused on establishment by seed and planted materials, and for best practices assumes that production in nurseries considers the genetic local variability (Landis et al., 2006; Zalesny et al., 2014). This does not exclude the possibility that some of the restoration actions proposed in the decision tree can also serve to promote clonal propagation (e.g., fencing clonal sprouts to prevent grazing and clearing competing vegetation).

We have treated *Salicaceae* as a group and only given prescriptions at the genus or species level when considered especially relevant. However, the requirements and strategies for regeneration may greatly vary between the two genera and across species. Within *Populus*, there are also important differences among sections (Aigeiros, Populus, Tacamahaca), particularly in the degree of vegetative reproduction (Gom and Rood, 1999; Rood et al., 2007). It follows that some restoration actions may be more efficient for one of the two genera or for certain species than for others. Artificial irrigation in abandoned farmlands along the Colorado River floodplain, for example, resulted in establishment of *P. fremontii* but not *S. gooddingii* and *S. exigua* (Bunting et al., 2011; Grabau et al., 2011). The requirements for seedling establishment and restoration actions must be calibrated by species and river conditions, including soil characteristics, but this is also beyond the scope of this article. Actions to restore other taxa were not included in this review but may also be useful for *Salicaceae* recruitment. For example, flow prescriptions for fish populations also promoted *Salicaceae* recruitment in the Owens River of California, USA (Hill and Platts, 1998), in the Truckee River of Nevada, USA (Rood et al., 2003) and in the Bridge River of British Columbia, Canada (Hall et al., 2011).

3 A stepwise dichotomous decision tree for restoring *Salicaceae* forests (Figure 1)

*3.1 Seed availability (**Figure 1A**)*

Seed production, release and dispersal determine seed availability, which is the first requirement for the regeneration of riparian *Salicaceae* (**Figure 1A**). Riparian *Salicaceae* have an r-selected reproductive strategy: female cottonwoods and willows (these genera are dioecious) annually produce thousands to millions of tiny, short-lived seeds (Bessey, 1904; Karrenberg and Suter, 2003) that are released during spring and early summer, coinciding with the period of higher flood occurrence (Karrenberg et al., 2002).

Although seed availability varies across species, populations, space and time, both over the dispersal season and across years (Cooper et al., 1999; Guilloy-Froget et al., 2002; Gage and Cooper, 2005; González et al., 2016), seeds are rarely limiting in natural conditions (Lytle and Merritt, 2004; Harper et al., 2011; Morrison and Stone, 2015). However, a lack of seed source may be possible due to premature mortality and consequent scarcity of parental trees (articles 25, 78, 81 and 85 Appendix S1). Regulation may also alter sex ratios, disfavoring females, which are more flood tolerant but more sensitive to water scarcity (Hughes et al., 2010; Nielsen et al., 2010; Rood et al., 2013). Females are sometimes removed to reduce production of *pappus* (cotton) that has been claimed to harm livestock and pets (although we have not found any scientific evidence supporting this) and produce human allergies (Storms, 1984). Different techniques have been used to assess whether the seed availability requirement is met (see Appendix S2).

3.1.1 Restoring seed availability (**Figure 1A'**)

<u>Seed augmentation</u> (techniques in Appendix S1) is recommended when seed availability has been identified as limiting (e.g., article 85 Appendix S1) or to ensure an adequate number of germinants in restoration plots (e.g., articles 13 and 95 Appendix S1) (**Figure 1A'**). Seed augmentation has substantially improved establishment in

some cases (article 81 Appendix S1), but more for *Populus* than *Salix* (article 41 Appendix S1), as seed quality (i.e., longevity, germinability, vigor, early survival) is usually higher in *Populus* (Van Splunder et al., 1995; González et al., 2016). In other cases, no increase in seedling establishment has been reported following seed augmentation (articles 35 and 85 Appendix S1).

*3.2 Moist, bare surface (**Figure 1B**)*

Once released and dispersed in the air, seeds can be deposited on water or land and be further dispersed by flowing river water. Seeds germinate within 24 h in high proportions (>90%) following contact with flowing water, rain or soil moisture, but they need to do so soon after release as they lose viability in a few weeks or even days in field conditions (Karrenberg et al., 2002). For successful germination and colonisation, seeds also need bare, competition-free (understory or overstory) surfaces ("safe-sites" hereafter). Different fluvial processes such as channel abandonment and narrowing, channel meandering, and flood deposition may also create safe-sites, which include gravel-, sand-bars and other flood deposits (Scott et al., 1996; Cooper et al., 2003; Stella et al., 2011).

In regulated rivers with typically reduced and truncated flood peaks and stabilized low flows, pulses of establishment occur during the river adjustment to the new fluvial regime, as safe-sites are left behind with the reduction of flooding disturbance (Johnson, 1994, 1998; Shafroth et al., 2002; Stromberg et al., 2010; Coble and Kolb, 2013). Once these safe-sites are colonized, new safe-sites may still be regularly created, but usually as narrow fringes of the main channel or in-channel areas (Cordes et al., 1997; González et al., 2010; Dixon et al., 2012). The key to detect if the requirement is met therefore relies upon analyzing whether the fluvial processes responsible for the creation of safe-sites will be active at spatial and temporal scales sufficient to maintain the shifting steady state mosaic, or, on the contrary, will be reduced or suppressed (see Appendix S2 for details).

3.2.1 Restoring moist, bare surfaces (**Figure 1B'**)

Releasing large pulse flows from dams and water reservoirs may be a cost-effective solution to reactivate the creation of safe-sites (**Figure 1B'**). However, we are unaware of any river whose impaired capacity to create safe-sites was fully restored by prescribed floods. This is because floods able to do geomorphic work are usually of high magnitude, and controlling large floods to avoid damage to human settlements and infrastructures is a common purpose of dams. The amount of water devoted to environmental flows (sensu Arthington, 2012) is usually what is remaining once human needs are satisfied (articles 87, 88 and 105 Appendix S1; Acreman, 2016). Especially in arid and semi-arid regions, water is a precious resource and ecological restoration is still not seen as a top priority for management in most rivers (e.g., Colorado River, Glenn et al., 2013). Also, dams are typically built with limited capacity for flood releases. Consequently, prescribed large floods are usually much smaller than pre-regulation floods and rarely have the capacity to do significant geomorphic work (articles 25 and 38 Appendix S1). A water release from Glen Canyon Dam in the Colorado River, for example, buried ground-covering herbaceous vegetation but resulted in minimal scouring of woody invasive *Tamarix* spp. and was only able to slightly re-configure channel margins and sand-bars (Stevens et al., 2001). Water releases from reservoirs and diversion channels implemented in the Chinese rivers Tarim and Ejina (Zhu et al., 2016) and in the North American Lower Colorado River (articles 85 and 88 Appendix S1) have contributed to replenish aquifers, increased groundwater levels and reinvigorated *Salicaceae* forests, but large-scale creation of safe-sites has not been reported (Zhu et al., 2016; Mueller et al., 2017). Moreover, even if enough water could be dedicated to environmental flows, recent studies have shown that in rivers that have suffered from the effects of regulation for several decades, floods of magnitude similar to the ones occurring during pre-dam conditions can be ineffective in creating safe-sites for seedling establishment due to bank hardening effects of vegetation encroachment (Green River, Colorado, U.S.; article 25 Appendix S1; Rio Grande, Texas, U.S.; Dean and Schmidt, 2011), or if they are created, they are limited to the active channel only: a 500-yr return period flood in the Missouri River, South Dakota, U.S., scoured instream landforms but did not produce significant channel migration (Dixon et al., 2015; Johnson et al., 2015).

The limitations of large pulse flows to reactivate geomorphic dynamism are not only due to the effects of insufficient flood magnitude but also due to alterations to sediment load and type, which may have dramatically changed (article 80 Appendix S1; Scott et al., 1997; Johnson, 1998; Cooper et al., 1999). Reservoirs tend to trap and accumulate sediments, inducing downstream sediment deficits that may ultimately affect the potential of prescribed floods to induce geomorphic dynamism (article 46 Appendix S1; Wohl et al., 2015). It follows that rivers with more non-cohesive sediments available would be more responsive to environmental flows. In the Bill Williams River in Arizona (USA), for example, geomorphic work of sufficient magnitude to promote *Salicaceae* recruitment resulted only from environmental flows (articles 87, 88 and 105 Appendix S1), although the scale of new floodplain creation is still relatively small (Kui et al., 2017). As part of flood prescriptions, sediment bypass structures may be added to dams (Stromberg, 2001) and sediment releases in addition to water releases should be part of environmental flows (Wohl et al., 2015). For instance, sediment releases from a Japanese reservoir, timed with seed release, promoted recruitment of *S. gilgiana* (article 3 Appendix S1). More details on techniques to implement large pulse flows are available in Appendix S3.

Although controlled pulse flows may not always create significant areas of bare ground, they may serve to disperse seeds (**Figure 1A**) and to moisten bare surfaces that were previously created by another process such as other fluvial events (articles 78 and 79 Appendix S1); as a product of human impacts, such as water abstraction and river regulation (articles 43 and 46 Appendix S1), or by other restoration actions (e.g., vegetation removal and land contouring: articles 85, 90, 98 and 100 Appendix S1; see later in this section). In fact, some authors have suggested and showed that managed pulse flows, despite being of lower magnitude, may lead to positive restoration outcomes in the form of downsized, narrower, but still functional *Salicaceae* forests (articles 38 and 43 Appendix S1).

Moistening bare surfaces can also be achieved through <u>irrigation</u> (see techniques in Appendix S3). However, there are a few common problems associated with irrigation: first, it can be expensive to operate and maintain; second, it can be impractical to implement in remote areas; third, it may be needed until plants develop the root structure to acquire water resources by themselves; and fourth, continuous irrigation may lead to relatively shallow root systems that do not reach the alluvial aquifer (the requirement of moisture in the rooting zone will be discussed in more depth below in section D) (articles 18, 22, 30 and 44 Appendix S1). In agricultural settings, irrigation is often associated with the accumulation of salts in the soil, depending on the water source and drainage, but to our knowledge, increased salinity from irrigation has not been documented in a restoration context. It is recommended that irrigation only be applied when other more ambitious restoration actions such as environmental flows have proven ineffective or infeasible and the only alternative is to concentrate efforts locally (article 38 Appendix S1, for example combined with planting, see below section 3.3.1).

Very often, the main impediment for creating new safe-sites for *Salicaceae* recruitment is the existence of artificial levees, dikes and rip-rap that limit channel migration (Van Looy et al., 2003; Bombino et al., 2007; Dufour et al., 2007). <u>Levee manipulation</u> (techniques in Appendix S3) can be a cost-effective approach to promote channel widening in constrained rivers, creating safe-sites, and ultimately inducing recruitment of riparian *Salicaceae*. However, most reports of this restoration action have noted that the extent of channel widening has been too limited to restore the mosaic of habitats typical of pre-regulation *Salicaceae* forests, as channel migration is still limited and succession is recurrently reset in the safe-sites that are created (articles 40, 59 and 76 Appendix S1).

If safe-sites are not available or cannot be created by restored fluvial processes via large pulse flows or levee manipulation, active site preparation is likely to be needed. In some cases, the direct occupation of the floodplain by human economic activities such as agriculture and mining is the primary cause of lack of safe-sites and the simple <u>abandonment of human activities</u> may be the most cost-effective form of site preparation (see techniques in Appendix S3).

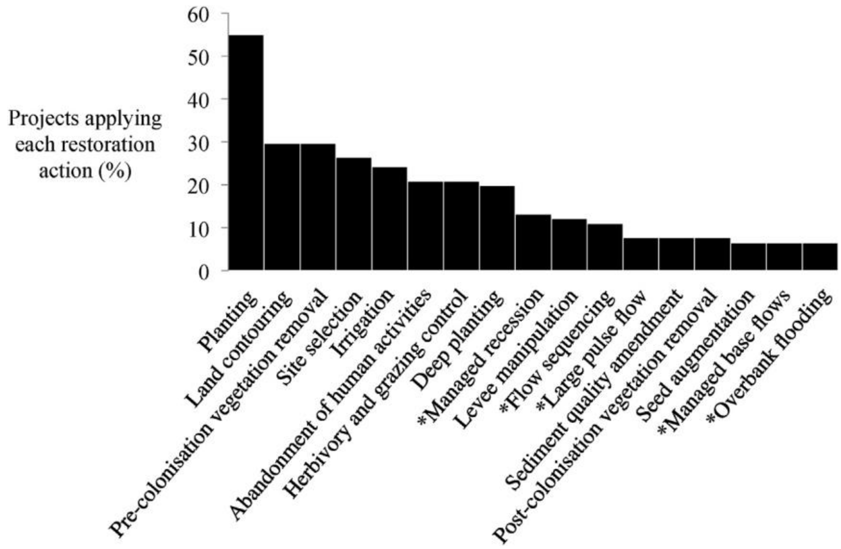

Figure 2. Frequency of occurrence (%) of each restoration action in the 91 projects included in the 105 reviewed articles (Appendix S1). *Can be part of environmental flows (15% of the projects).

Figure 3. Images of actions to restore the regeneration of *Salicaceae* forests. Ordered by frequency of implementation (see **Figure 2**), from upper left to bottom right: 1: Plantation of cottonwoods in an excavated floodplain in the Ebro River, NE Spain, photo by E González; 2: Riverbank reconfiguration in the Genil River, S Spain, V Garófano-Gómez; 3: Precolonisation vegetation removal in the Green River at Dinosaur National Monument, Utah, Southwestern U.S., B Sánchez; 4: Irrigation of planted cuttings in the French Alps, A Matringe; 5: Abandonment of hybrid poplar plantation in the Garonne River, SW France, E González; 6: Individual protections from herbivory in planted cottonwoods in the Jarama River, Central Spain, V Garófano-Gómez; 7: Deep planting in a tributary of the Arkansas River, Colorado, Great Plains of the U.S., A Sher; 8: Environmental flows from Alamo Dam in the Bill Williams River, Arizona, Southwestern U.S., J Evelyn, *includes actions of large pulse flow, managed recession, managed base flows, flow sequencing and overbank flooding; 9: Set-back of a longitudinal defense in the Órbigo River, NW Spain, D García de Jalón; 10: Sediment quality amendment in Moyie Lake, southeastern British Columbia, Canada, P Raymond, Terra Erosion Control Ltd; 11: Post-colonisation vegetation removal of invasive tamarisks in a tributary of the Arkansas River, Great Plains of the U.S., A Sher; 12: Experimental *Populus* and *Salix* seed augmentation with hydroseeding, Cibola National Wildlife Refuge, Arizona-California border, Southwestern U.S., M Grabau. Note that there is no image for site selection.

Once lands are available for restoration, site preparation is frequently achieved by vegetation (and litter) removal (see techniques in Appendix S3). This has been a very common restoration action to restore *Salicaceae* forests (the second most frequent, **Figure 2** and **Figure 3**), particularly within the context of invasive species control (e.g., in Southwestern U.S. rivers where non-native *Tamarix* spp. have invaded many watersheds and replaced native cottonwoods and willows; Friedman et al., 2005; Merritt and Poff, 2010; Sher, 2013). Restoration at the Bosque del Apache Wildlife Refuge on the Rio Grande in New Mexico, for example, was accomplished by creating space for seedling establishment by mechanically clearing exotic *Tamarix*. This replaced the role of large magnitude floods, which no longer occur, for scouring vegetation and opening sites for recruitment (Dello Russo, 2013). Releases from the Cochiti Dam ("large pulse flows" action) were then timed to correspond with seed dispersal of *Salicaceae*, helping to disperse seeds and moisten the created surfaces. Although *Tamarix* establishment was also promoted, the native cottonwoods and willows were able to outcompete them and become well-established (articles 90, 98 and 100 Appendix S1). Vegetation removal may facilitate seedling establishment by producing soil disturbance (article 95 Appendix S1). However, bare surfaces that vegetation removal leaves behind may be not suitable for *Salicaceae* recruitment, for example, if surfaces are too elevated from the water level and cannot be artificially flooded. In these situations, land contouring (see techniques in Appendix S3) may be necessary. Land contouring is probably the most costly restoration action, but it can be highly effective, and it has also been very frequently applied (**Figure 2** and **Figure 3**). Land contouring necessarily involves removal of competing vegetation (e.g., articles 74 and 85 Appendix S1).

*3.3 Local plant stocks available or can be produced (**Figure 1C**)*

If the requirements of "seed availability" (**Figure 1A**) and "moist, bare surfaces" (**Figure 1B**) are not met and cannot be restored through the proposed restoration actions, planting may be considered if plant material from local sources is available or can be produced (articles 37, 64 and 97 Appendix S1; Landis et al., 2006; Zalesny et al., 2014) (**Figure 1C**).

3.3.1 Planting as a strategy to bypass the requirements of seed availability and moist, bare surfaces (**Figure 1C'**)

Planting poles or whole root seedlings or saplings (see techniques in Appendix S3) was the most popular restoration action found in the literature review, as was found in 55% of the 91 projects reported by the 105 articles (**Figure 2**). We believe that even this high proportion underestimates the actual number of planting-centered restoration projects because mortality in plantations is generally high (Stromberg, 2001, article 18 Appendix S1) and there is a bias to publishing positive results in scientific literature in general and restoration literature in particular (González et al., 2015). Plantings have been most successful where moisture is available in the rooting zone (article 21 Appendix S1, see section 3.4), and, conversely, have been most likely to fail where there is insufficient soil moisture to sustain the plantings (article 18 Appendix S1). Even where successful, planting poles and saplings will likely result in lower tree densities and cover compared to natural forest patches because it is not possible to plant at the high densities achieved by natural recruitment, and inevitable mortality of some planted individuals will lead to gaps (González, field observations). However where conditions allow, prolific natural recruitment can fill these in to make plantings indistinguishable (articles 18 and 19 Appendix S1).

Primary rationales for planting include improving ecosystem properties such as providing shelter and habitat for wildlife (article 18, 28 and 91 Appendix S1), accelerating the successional process for the establishment of herbaceous and late successional woody species, increasing plant biodiversity (articles 47, 53, 56, 60, 91, 92 and 96 Appendix S1), stabilizing river banks to avoid soil erosion and channel incision (articles 8, 18, 30, 44, 53, 56, 63, 64, 65, 84, 91 and 104 Appendix S1 e many of these refer to bioengineering, Evette et al., 2009), improving in-channel aquatic habitats by shading streams and increasing input of organic matter (articles 44 and 91 Appendix S1), controlling exotic species (articles 33, 50, 55 and 101 Appendix S1), and occasionally simply for intrinsic ecological and aesthetic values of *Salicaceae* forests (article 40 Appendix S1). Plantings can be also implemented as environmental compensation measures for floodplain development projects. However, contracting requirements often specify the number of plants or hectares to be planted, but do not require assessing survival. Managers may be also attracted to quick results of planting. Paradoxically, deficient plant establishment by seed and lack of safe-sites have rarely been reported as the only motivation for planting. As planting *Salicaceae* was seen as a means to reach other goals rather than restoring *Salicaceae* regeneration per se, projects may have overlooked the underlying causes of seedling establishment failure and applied planting much more often than it would have been desirable.

Another characteristic of planting projects is that they have typically been implemented at a small spatial extent compared to other restoration actions, such as those derived from dam operations, and were very frequently applied on small streams (e.g., articles 53, 56, 63, 64, 65 and 91 Appendix S1) and not on large floodplains (but see article 40 Appendix S1 for the Ebro River, NW Spain; articles 18 and 19 Appendix S1 for the Lower Colorado, SW U.S.; Alpert et al., 1999 for the Sacramento River, California, U.S.). One reason that planting occurs at small scales is the low cost-effectiveness of this restoration method. The cost of restoring *Salicaceae* populations by means of planting at the necessary large spatial scales to have a mosaic of shifting habitats typical of healthy floodplains would be prohibitive (article 79 Appendix S1). For example, Dixon et al. (2012) suggested that in the Missouri River (Great Plains in northern U.S.), where flow and sediment management is insufficient to promote cottonwood recruitment, 435 ha of new plantings per year would be necessary to compensate for area lost due to mortality and successional change along 1127 river kilometers. Considering that the cost of pole planting has been estimated to be ca. $1700 in 1998 per ha (article 97 Appendix S1) - equivalent to $2500 in 2017 -, more than $1 million would be necessary annually to restore the *Salicaceae* forests in the Missouri River (and these estimates do not take into account the purchase or easements of private lands). Even though planting is not very cost-effective, some authors still recommend planting (combined with irrigation) over other actions such as environmental flows, when the application of the latter is not effective or feasible and the restoration efforts must be concentrated locally (e.g., article 38 Appendix S1). This may temporarily contribute to maintain some ecosystem functions locally (e.g., improve bird habitat, Paxton et al., 2011) but rarely will the mosaic of patches and their dynamics be restored following this logic.

*3.4 Moisture availability in the rooting zone (**Figure 1D**)*

Once seedlings and planted fragments have become initially established, to survive and grow, their roots must obtain water from moist sediment, typically provided by the water table and associated capillary fringe, both gradually receding after flood waters decline (article 86 Appendix S1; Johnson, 2000; Harper et al., 2011). Precipitation and the water holding capacity of sediments along the sediment column can also provide the necessary moisture in the rooting zone (articles 13 and 95 Appendix S1; Cooper et al., 1999; Rood et al., 2011) but, in general, seedlings that establish at high topographic positions will die from desiccation (article 77 Appendix S1). Techniques available to assess whether the requirement of moisture availability in the rooting zone is met are detailed in Appendix S2.

3.4.1 Restoring moisture availability in the rooting zone (**Figure 1D'**)

Large pulse flows and irrigation may not only moisten safe-sites for germination and colonisation (Section 3.2.1), but can also recharge the sediment profile with water and provide necessary moisture to the rooting zone for continued survival and growth. Special attention needs to be paid to the rate of groundwater recession after natural or induced large pulse flows and after irrigation by controlled flooding, thus roots of recently established seedlings and planted fragments can keep pace with receding water levels. Managed recessions have been included as a fundamental part of prescriptions for environmental flows (e.g., articles 25, 34, 38, 43, 46, 95, 98 and 105 Appendix S1; see techniques in Appendix S3). Once floodwaters have receded, managed base flows (e.g., articles 34, 38, 43, 78, 85, 86, 87, 88 and 105 Appendix S1; see techniques in Appendix S3) are also important because these usually determine the water table level at the time of higher drought stress during the summer. Site selection also helps determine the locations that will have the appropriate elevation, sediment texture and stratigraphy to provide plants with moisture (e.g., articles 35, 74, 85 and 98 Appendix S1; Appendix S3 for more details). In cases when moisture in the rooting zone is insufficient and cannot be provided artificially, deep planting (see techniques in Appendix S3) to reach the groundwater may be the only restoration alternative (e.g., articles 18, 30, 44, 45 and 97 Appendix S1).

*3.5 Protection from future flooding, burial and scour (**Figure 1E**)*

Seedlings and planted fragments established at very low topographic positions will have a higher risk of dying from flooding, burial and scour during subsequent floods and ice jams in the northern latitudes (articles 84 and 86 Appendix S1; Auble and Scott, 1998; Mahoney and Rood, 1998; Cooper et al., 1999, Johnson, 2000; Rood et al., 2007; Harper et al., 2011). See Appendix S2 for techniques to assess whether this requirement is met.

3.5.1 Restoring protection from future flooding, burial and scour (**Figure 1E'**)

Site selection can help ensure that restored sites are within the range of topographic positions to avoid death by desiccation (upper elevational limit, Section 3.4), or alternatively by flooding, burial or scour (lower elevational limit), as exemplified by the Recruitment Box Model (Mahoney and Rood, 1998; Amlin and Rood, 2002; Rood et al., 2008; details in Appendix S3). Topographic position (local elevation) is an important determinant of survival and growth of seedlings (e.g., articles 35, 74, 75 and 105 Appendix S1; Auble and Scott, 1998) or planted cuttings (e.g., article 84 Appendix S1) in restoration projects. Negative effects of base flows and subsequent floods that potentially flood, bury or scour established seedlings and planted fragments can be partially averted by flow sequencing, such as managing aspects of discharge (e.g., flood frequency, magnitude) over several years following establishment (details in Appendix S3).

*3.6 Favorable chemical and physical properties of sediments (**Figure 1F**)*

Sediment properties other than moisture and texture (see Sections 3.2 and 3.4) can influence seedling establishment. Sediment salinity in floodplains may increase as a result of human impacts (Jolly et al., 1993) and is known to reduce germination rates in *Salicaceae* (Shafroth et al., 1995; Glenn and Nagler, 2005), and negatively affect survival and growth (Rowland et al., 2004; Vandersande et al., 2001) (for examples of salinity assessments, see Appendix S2). Nutrient availability may also affect seedling survival and growth (Marler et al., 2001; Adair and Binkley, 2002), even though riparian *Salicaceae* tolerate poor nutrient levels in the substrate. Mycorrhizal associations are also important for *Salicaceae* (Corenblit et al., 2018), and due to previous degradation may be a limiting factor in riparian zones (Meinhardt and Gehring, 2013).

3.6.1 Restoring favorable chemical and physical properties of sediments (**Figure 1F'**)

Large pulse flows from dams and diversion channels and irrigation can flush salts that have accumulated in floodplain soils (articles 18 and 85 Appendix S1), especially when flooding is repeated over multiple years (articles 13 and 22 Appendix S1; Ohrtman et al., 2012). Sediment quality amendments have been occasionally applied to improve these properties (see Appendix S3) but it has never been reported that any of those practices improved *Salicaceae* performance.

*3.7 Low herbivory and grazing (**Figure 1G**)*

Most riparian *Salicaceae* are highly palatable for both wild ungulates and livestock (articles 19, 20, 25, 45, 48, 81 and 101 Appendix S1; Andersen and Cooper, 2000). Moreover, cattle tend to occupy riparian areas due to easier access to water, lush vegetation, gentler slopes (article 4 Appendix S1) and shade (article 77 Appendix S1). Immediately after germination and during the first days and weeks of life, the main cause of animal-induced seedling mortality is by trampling and uprooting (articles 77, 78 and 81 Appendix S1). Rabbits and rodents such as beavers and nutria can also damage juvenile *Salicaceae* trees (articles 17, 25, 33, 63 and 88 Appendix S1). However, riparian *Salicaceae* tolerate and are resilient to high levels of disturbance; therefore, herbivory and grazing (particularly from beaver) do not necessarily lead to *Salicaceae* mortality but can alter plant architecture, growth patterns and even promote vegetative reproduction (articles 11, 48 and 45 Appendix S1). The effects of grazing on survival and growth of riparian *Salicaceae* vary with the local flooding regime (articles 16, 57 and 82 Appendix S1 and De Jager et al., 2013).

3.7.1 Controlling herbivory and grazing (**Figure 1G'**)

Herbivory and grazing control has been possible directly by installing fencing and other exclosures; and indirectly by introducing herbivore predators that have cascading effects in food webs (Appendix S3). Grazing control can be highly effective (Fitch and Adams, 1998) but is generally undertaken by individual landowners who are neither interested in, nor familiar with scientific reporting; thus, it may be under-represented in our review (**Figure 2**).

*3.8 Low competition (**Figure 1H**)*

After initial colonisation of seedlings or plantings, *Salicaceae* species need to compete with co-occurring vegetation for physical space, light, nutrients and water acquisition and may be affected by allelopathic effects (see examples in Appendix S2). Although *Salicaceae* seedlings and saplings have been found to be highly competitive (Sher et al., 2000, articles 22 and 90 Appendix S1), this ability is dependent on favorable growth conditions (Sher and Marshall, 2003).

3.8.1 Restoring low competition (**Figure 1H'**)

Overbank flooding may be applied to restoration sites to kill competing species, which may be less tolerant of anoxic conditions, burial, or scour than co-occurring *Salicaceae* species (see Appendix S3). In most other cases, (post-colonisation) vegetation removal by mechanical or chemical means is necessary (Appendix S3).

4 Geography of restoration approaches

Restoration of *Salicaceae* forests has been much more often reported in the U.S. (76% of the 91 projects included in the 105 articles) than in Europe (19%) (**Figure 4**). The broadest array of actions has been applied in North America. There was a notable gap of literature in Asia (only 5 projects reported), where most of the articles reporting restoration actions on *Salicaceae* forests came from two restoration projects in China (González et al., 2015; Glenn et al., 2017; but only two were included in the systematic review, articles 1 and 2 Appendix S1 as most focused on conservation of mature populations only). The frequency of application of restoration actions also greatly differed across world regions. Planting and land contouring were not only the most frequently applied actions but they were also widespread globally. Vegetation removal was particularly concentrated in the American West. Many of the projects there dealt with control of invasive species, mainly *Tamarix*, and were not focused on directly promoting *Salicaceae* recruitment. By only addressing removal of *Tamarix*, control projects generally overlooked the cause of river degradation (article 18 Appendix S1, Briggs et al., 1994; Stromberg, 2001; Stromberg et al., 2007b; González et al., 2017b). Actions related to manipulation of the flow regime such as large pulse flows, managed recessions, base flows and flow sequencing were concentrated in regions with lower population density, such as the Great Plains of North America, the Southwestern U.S. and Central Asia. Levee manipulation was most popular in Europe, particularly in Central Europe, and herbivory and grazing control in the Northwestern Forested Mountains of North America.

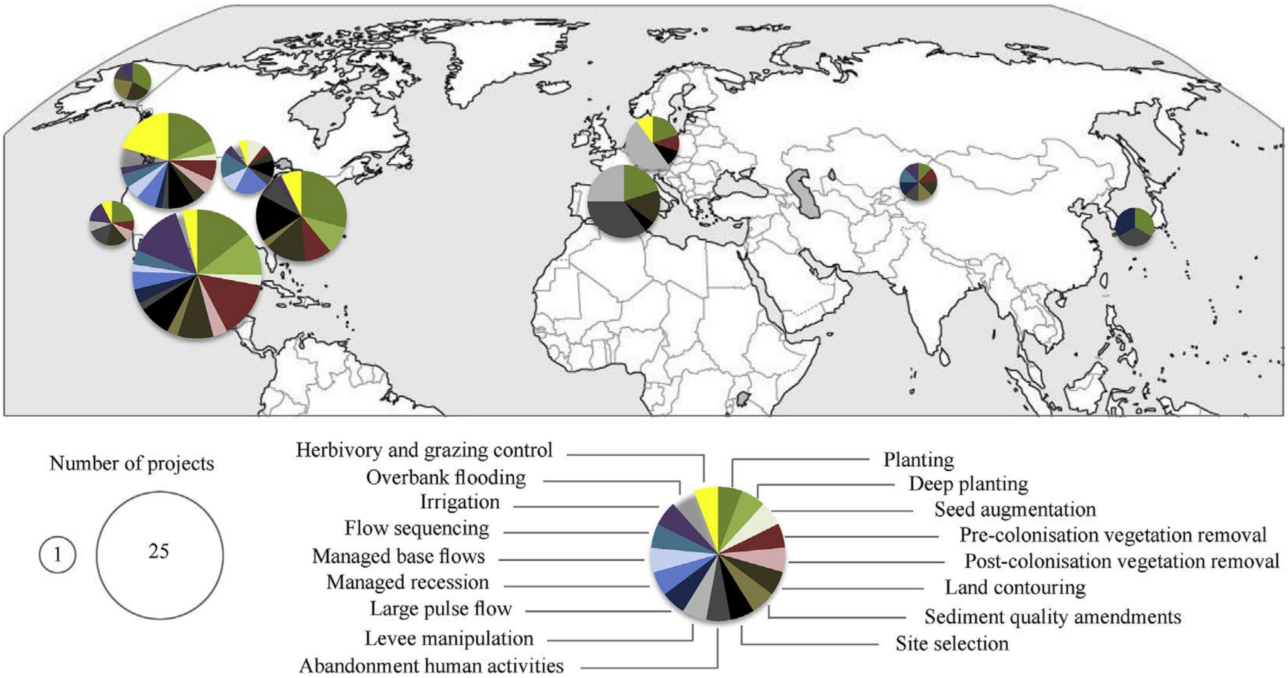

Figure 4. Relative importance of restoration actions by regions of the Northern Hemisphere. Size of pie charts and of the pie "slices" is proportional to the number of projects. North America was divided in the following ecoregions: Alaska (3 projects), Eastern Temperate Forests (including projects in the U.S. States of LA, MD, MS, NC, TN, TX and WI) (14), Great Plains (Canadian province of AB, U.S. State of CO) (5), Mediterranean California (4), Northwestern Forested Mountains (Canadian province of BC, U.S. States of CA, CO, OR, UT and WA) (16), and Southwestern U.S. (Mexico states of Baja California and Sonora, U.S. States of AZ, CA, ID, NM, NV, CO and UT) (27). Europe was divided in Central (E France, Switzerland) (7) and Southern (Spain, S France and Greece) (10). Asia was divided in Central (China and Uzbekistan) (2) and Coastal (Japan and South Korea) (3). Note that different shades of the same color were used for related restoration actions (e.g., actions related to water management are depicted in different intensities of the blue color). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

5 The human component

Social, cultural, historical, legal and political circumstances and legacies may either constrain or produce positive synergies with restoration actions and must be taken into account in an integrative manner with the biophysical aspects considered in this article (González et al., 2017c). A thorough evaluation of such human factors prior to engaging in riparian restoration is recommended as a part of the planning process (Shafroth et al., 2008). For example, large pulse flows (**Figure 1B'**, **Figure 1D'** and **Figure 1F'**) are more appropriate where human population density and land use intensity in the floodplain are low (article 87 Appendix S1; Hughes and Rood, 2003). Planting (**Figure 1C'**, **Figure 1D'**, **Figure 1F'** and **Figure 1H'**), despite its relatively high cost and low effectiveness, may have great value as an environmental education tool and as a means to engage local communities in conservation through volunteer programs (González del Tánago et al., 2012). This can ultimately engender a favorable socio-political context for implementing future, more cost-effective restoration actions that may not be socially acceptable at present (González et al., 2017c).

Economic aspects, however, often represent the most limiting factor for restoration projects. The abandonment of economic activities in the floodplain (**Figure 1B'**) may be incentivized by purchasing ownership or easements on lands that are not economically productive, but landowners may be more open to negotiate or to yield their lands if they share the goals of the project and are engaged in the community (articles 19 and 42 Appendix S1; Ollero, 2010; González et al., 2017c). Purchasing water rights may also facilitate water releases for environmental purposes (Richter et al., 2003, 2006). Restoration of *Salicaceae* forests could be promoted by carbon sequestration credits (Matzek et al., 2015). Briggs et al. (1994) noted that access to rivers by recreationists may be another source of degradation, and thus limiting recreational access, particularly with motor vehicles, to riparian zones might be a restoration action to consider (but not in **Figure 1**). Recreational opportunities could also be incentives or even sources of funding for restoration.

6 Alternative, innovative solutions

To guarantee the sustainability of *Salicaceae* forests, there is an urgent need for innovative, large-scale, original, and integrative solutions (Dixon et al., 2012; González et al., 2017c). We suggest that the following themes include opportunities for improving *Salicaceae* restoration in the future:

Novel ecosystems: Although human activities have destroyed and degraded many *Salicaceae* forests, they have also created novel ecosystems that represent new opportunities for *Salicaceae* regeneration. Examples include urban riparian zones supported by leakage, outflows and urban effluents (article 5 Appendix S1) and naturalized, abandoned crops and hybrid poplar plantations (articles 39, 40, 42 and 103 Appendix S1). Others such as reservoir deltas (Johnson, 2002; Dixon et al., 2012, 2015; Volke et al., 2015) still remain largely unexplored. New building developments can include better integration of urban and riparian corridors through nature-based solutions and green infrastructures (González et al., 2017c). Planting, for example, is often a fundamental part of bioengineering works, which fulfills human interests such as protection of infrastructures (roads, railways, housing) and erosion control while reproducing *Salicaceae* stands locally (Evette et al., 2009). Recognizing the value of novel ecosystems does not exclude the need to conserve existing *Salicaceae* forests or promote their regeneration in degraded rivers and, of course, it does not justify further degradation.

Emerging restoration approaches: Some restoration strategies have been applied too few times or too recently to evaluate. For example, alternative sources of water for restoration such as wastewater have been suggested (e.g., Marler et al., 2001) but only applied in urban rivers (article 5 Appendix S1). Also, many dams have become obsolete in the past few years and their removal can open opportunities for re-establishment of natural river dynamics and *Salicaceae* recruitment (East et al., 2015; O'Connor et al., 2015). The experience gained with small channel widenings by levee manipulation (e.g., articles 40, 59 and 76 Appendix S1) may help design more ambitious projects in the future.

Adaptive management: Monitoring and evaluation of restoration projects has only begun to receive attention in the last few decades, and this information is being used for adaptive management. However, *long-term* monitoring (>5-6 years) is still a pending task in most cases (González et al., 2015). Taylor et al. (2006) for example found that the density of planted species 10 years later changed dramatically from the surveys immediately following restoration. Glenn et al. (2017) suggested that the positive effects of environmental flows are only visible when implemented in an adaptive management framework for more than one decade. Long-term monitoring is especially important, given the episodic nature of *Salicaceae* recruitment. Such monitoring will help inform restoration actions in the future to achieve a shifting steady state mosaic of patches of different ages.

7 Conclusions

Existing challenges for achieving sustainable *Salicaceae* regeneration in most rivers of industrialized countries will likely be exacerbated in the future, because human impacts on rivers and riparian zones are expected to increase due to urban and economic development, the effects of climate change, and the spread of well-established and emerging plant invasions (Richardson et al., 2007; Palmer et al., 2008; Rood et al., 2008; Perry et al., 2015). Being mainly seen as water bodies, management of rivers has historically been approached from the perspective of aquatic sciences, and restoration actions focused on the water component (e.g., environmental flows) have received most recent attention as alternatives to planting, the classic restoration approach from forestry. *Salicaceae* forests, however, occupy riparian zones that lie at the interface between aquatic and terrestrial systems. We have shown here that traditional plantings and the manipulation of the flow regime are just two among many alternatives to restore impaired regeneration. Using the life-cycle of the seedlings as a guide, we have proposed a set of restoration actions linked to other ecosystems components besides water (i.e., fluvial landforms, competing vegetation, herbivory, etc.) to establish a new theoretical framework and a decision-making tool for the restoration of *Salicaceae* regeneration. Following the current pragmatic view of restoration of ecological processes and ecosystem functions (and services) rather than returning to a pre-disturbance historic reference (Dufour and Piégay, 2009; Rohwer and Marris, 2016), we hope our decision tree helps achieve the emerging objective of managing *Salicaceae* forests of river systems to better fit their new hydrologic and fluvial geomorphic situation (articles 43 and 80 Appendix S1).

Author contributions

EG, AAS and PBS conceived and designed the research and developed the decision tree; EG and VMF did the systematic review of literature; EG wrote and edited the manuscript with help from all others.

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Appendix. Supplementary Data

Appendix S1. a) table of restoration actions discussed in each of the articles included in the review.

		Seed augmentation	Large pulse flow	Irrigation	Levee manipulation	Abandonment human activities	Pre-colonisation vegetation removal	Land contouring	Planting	Managed recession	Managed base flows	Site selection	Deep planting	Flow sequencing	Sediment quality amendments	Herbivory and grazing control	Overbank flooding	Post-colonisation vegetation removal
1	Aishan et al., 2013		x											x				
2	Aishan et al., 2015		x											x				
3	Asaeda et al., 2015		x															
4	Batchelor et al., 2015								x							x		
5	Bateman et al., 2015			x			x	x	x			x						
6	Battaglia et al., 2002					x		x				x						
7	Bay and Sher, 2008						x		x				x					
8	Beauchamp et al., 2015							x	x									
9	Beschta and Ripple, 2010															x		
10	Beschta and Ripple, 2015															x		
11	Beschta and Ripple, 2016															x		
12	Beyer et al., 2007															x		
13	Bhattacharjee et al., 2006	x		x			x			x				x	x			
14	Bhattacharjee et al., 2008	x		x			x			x				x	x			
15	Bhattacharjee et al., 2009			x			x			x								
16	Bilyeu et al., 2008															x		
17	Breton et al., 2014								x							x		
18	Briggs et al., 1994			x			x	x	x				x			x		x
19	Briggs and Cornelius, 1998			x		x	x	x	x			x	x		x	x		x

20	Brookshire et al., 2002															x		
21	Bunting et al., 2011	x		x		x		x				x		x				
22	Bunting et al., 2013	x		x		x								x				
23	Caplan et al., 2013							x	x									
24	Conroy and Svejcar, 1991								x			x				x		
25	Cooper and Andersen, 2012		x				x	x		x		x						
26	Cooper et al., 2017							x	x									
27	Densmore and Zasada, 1978								x									
28	Densmore et al., 1987			x		x		x	x						x			
29	Densmore et al., 2009							x	x						x			
30	Dreesen and Fenchel, 2008			x			x		x				x			x		
31	Efthimiou et al., 2017					x			x			x						
32	Florsheim and Mount, 2002				x			x										
33	Foster and Wetzel, 2005			x			x		x			x						
34	Foster and Rood, 2017									x	x			x				
35	Friedman et al., 1995	x		x			x	x				x						
36	Gage and Cooper, 2004								x			x				x		
37	Gladwin and Roelle, 1998	x		x		x		x									x	x
38	Glenn et al., 2017		x						x	x	x			x			x	
39	González et al., 2016					x												
40	González et al., 2017				x	x		x	x						x			
41	Grabau et al., 2011	x		x		x												x
42	Gumiero et al., 2013				x	x		x										
43	Hall et al., 2011a									x	x			x				
44	Hall et al., 2011b								x				x			x		
45	Hall et al., 2015								x				x			x		
46	Hill and Platts, 1998		x							x	x			x				
47	Holl and Crone, 2004			x		x			x									x
48	Holland et al., 2005															x		
49	Hough-Snee et al., 2014															x		
50	Hovick and Reinartz, 2007						x	x	x									
51	Jenkins et al., 2008									x		x					x	
52	Johnson 1965						x	x										
53	Kaase and Katz, 2012								x									
54	Lee et al., 2002					x												
55	Lee et al., 2010								x									
56	Lennox et al., 2011								x							x		
57	Marshall et al., 2013															x		
58	Marshall et al., 2014															x		
59	Martínez-Fernández et al., 2017				x													

60	McClain et al., 2011			x		x	x	x	x									
61	Miller et al., 2008						x		x								x	x
62	Pasquale et al., 2011				x							x						
63	Pezeshki and Shields, 2006								x			x	x					
64	Pezeshki et al., 2007								x			x	x			x		
65	Rey and Labonne, 2015								x									
66	Ripple et al., 2001															x		
67	Ripple and Beschta, 2003															x		
68	Ripple and Beschta, 2004a															x		
69	Ripple and Beschta, 2004b															x		
70	Ripple and Beschta, 2005															x		
71	Ripple and Beschta, 2006															x		
72	Ripple and Beschta, 2007															x		
73	Ripple and Beschta, 2012															x		
74	Roelle and Gladwin, 1999			x		x	x	x		x		x					x	
75	Roelle et al., 2001			x		x	x	x		x		x					x	
76	Rohde et al., 2005				x		x											
77	Rood et al., 1998									x		x		x		x		
78	Rood and Mahoney, 2000									x	x							
79	Rood et al., 2003									x				x				
80	Rood et al., 2005							x		x	x			x				
81	Rood et al., 2016	x								x				x				
82	Rose and Cooper, 2017															x		
83	Schachtsiek et al., 2014			x		x	x	x	x						x			
84	Schaff et al., 2003								x			x						
85	Schlatter et al., 2017	x	x				x			x	x	x		x	x			
86	Shafroth et al., 1998		x							x	x			x				
87	Shafroth et al., 2010		x							x	x			x			x	
88	Shafroth et al., 2017		x				x	x		x	x	x		x				
89	Sheffels et al., 2014					x			x							x		
90	Sher et al., 2002		x				x					x						
91	Shields et al., 1995								x				x					
92	Simmons et al., 2012					x		x	x						x			
93	Smith et al., 2002			x			x											x
94	Sprenger et al., 2001			x			x					x					x	
95	Sprenger et al., 2002	x		x			x			x								
96	Stanturf et al., 2009					x		x	x			x						
97	Taylor and McDaniel, 1998			x			x	x	x			x	x					x
98	Taylor et al., 1999		x				x			x		x						
99	Taylor and McDaniel, 2004						x	x	x			x						x

100	Taylor et al., 2006		x				x					x		x				
101	Thomsen et al., 2012						x		x							x		
102	Tiedemann and Rood, 2015							x										
103	Vasilopoulos et al., 2007					x												
104	Watson et al., 1997								x			x	x			x		
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Appendix S1. b) bibliographic citations for reviewed articles.

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Appendix S2. Techniques to assess if the requirements for seedling establishment are met. Uppercase letters preceding establishment requirements follow the order of the boxes in Figure 1. The list of techniques is not exhaustive. Underlined articles were selected by the systematic review and include restoration actions.

Establishment requirement	**Techniques**
A. Seed availability	Seed availability has been assessed by visually monitoring seed release from the trees (Friedman et al., 1995; Shafroth et al., 1998; Roelle and Gladwin, 1999; Guilloy-Froget et al., 2002; Stella et al., 2006) or seed rain on the surface using Tanglefoot© coated traps (Cooper et al., 1999; Gage and Cooper 2005; González et al., 2010a; 2016; Cooper and Andersen, 2012; Schlatter et al., 2017; Shafroth et al., 2017).
B. Moist, bare surface	Surface and sub-surface sediment moisture have been assessed by direct measurements in the field (Conroy and Svejcar, 1991; Friedman et al., 1995; Taylor et al., 1999; Bhattacharjee et al., 2006; 2008; Bunting et al., 2011; Pasquale et al., 2011; Cooper and Andersen, 2012; Schachtsiek et al., 2014; Schlatter et al. 2017; and many others). Topographic data or digital elevation models and monitoring of the water level in the river and groundwater using wells and piezometers (Conroy and Svejcar, 1991; Taylor and McDaniel, 1998; Shafroth et al., 1998; 2017; Sprenger et al., 2002; Schaff et al., 2003; Foster and Wetzel, 2005; Taylor et al., 2006; Bhattacharjee et al., 2006; Pezeshki et al., 2007; Hall et al., 2011a; 2015; Bunting et al., 2011; Aishan et al., 2013; 2015; Schachtsiek et al., 2014; Schlatter et al., 2017; and many others) and surficial and surface-groundwater models (Shafroth et al., 2010; Pasquale et al., 2011) have been used to estimate the flooding frequency and duration, and indirectly, sediment moisture, at restoration sites. Sediment moisture is also highly influenced by sediment texture (Cooper et al., 1999), and therefore, the latter has been combined with or used as proxy for the former in many restoration assessments (Shafroth et al., 1998; 2017; Taylor et al., 1999; 2006; Sprenger et al., 2002; Sher et al., 2002; Schaff et al., 2003; Pezeshki and Shields, 2006; Pezeshki et al., 2007; Bhattacharjee et al., 2008; Caplan et al., 2013). With increasing spatial resolution and accuracy, remote sensing technology (e.g., Global Navigation Satellite System –GNSS– reflectometry; Jim and Komjathy, 2010) is expected to be an alternative to determine soil moisture in the near future. The creation of bare surfaces needs to be assessed at spatial and temporal scales sufficient to maintain the shifting steady state mosaic (Johnson et al., 1976; Décamps, 1996; Bormann and Likens, 2012). For spatio-temporal analyses of vegetation and river morphology dynamics, the most frequent method is the analysis of stream-flow historical records combined with vegetation and channel mapping obtained from archive maps, aerial photographs, satellite images and remote sensing (e.g., Muller et al., 2002; Friedman and Lee, 2002; Stromberg et al., 2010; González et al., 2010b; Cadol et al. 2011; Stella et al., 2011; Diaz-Redondo et al., 2017; Beller et al., 2016; Foster and Rood, 2017; and many others). Dendrochronological and demographic analyses that relate the different cohorts to hydrogeomorphic events may also help understand if the current *Salicaceae* populations are relicts of fluvial processes that no longer occur (e.g., Bradley and Smith, 1986; Friedman et al., 1996; Scott et al., 1997; Stromberg, 1998; Friedman and Lee, 2002; Cooper et al., 2003; Braatne et al., 2007; Hall et al., 2011b; Stella et al., 2011; Scott and Miller, 2017; and many others).
C. Local plant stock available or can be produced	Genetic diversity can be promoted by producing plant stock from seeds collected in the field or from seeds produced in nurseries by cuttings collected from different locations in the field, grown to flowering and then cross-pollinated at the nursery (Landis et al., 2006). Even if originated from native trees, plants obtained from nurseries might be less adapted to local conditions (Friedman et al., 1995; Rowland et al., 2004; Bhattacharjee et al., 2006).

D. Moisture available in the rooting zone	River water levels obtained from river-gauge stations and groundwater levels from wells and piezometers can be used to determine the rate of recession following floods and the base flows that provide the necessary moisture levels (Mahoney and Rood, 1998; Shafroth et al., 1998, Cooper et al., 1999, Johnson, 2000; Harper et al., 2011). Empirical observations in the field (Rood et al., 1998; 2003; 2005; Taylor et al., 1999; Rood and Mahoney, 2000; Sprenger et al., 2002; Bhattacharjee et al., 2006; Foster and Rood, 2017) and in mesocosms (e.g., for American species: Mahoney and Rood, 1991; 1992; Segelquist et al., 1993; Kranjcec et al., 1998; Horton and Clark, 2001; Amlin and Rood, 2002; Stella et al., 2010; for European species: Barsoum and Hughes, 1998; Francis et al., 2005; González et al., 2010a) have determined that the optimum water table decline following floods ranges between 1 for *Salix* spp. and 2.5 cm per day for *Populus* spp. However, seedlings and cuttings can survive faster drawdown rates, depending on precipitation (Bhattacharjee et al., 2006) and sediment texture and stratigraphy, with coarser soils draining water faster and finer soils having a higher water-holding capacity (Mahoney and Rood, 1992; Bhattacharjee et al., 2008; González et al., 2010a). Typical groundwater depths at base flow levels that guarantee survival of *Salicaceae* mature trees may also depend on the capillary fringe and soil texture and do not exceed 1-2 m for recently established seedlings and planted fragments (Conroy and Svejcar, 1991; Gage and Cooper, 2004; Caplan et al., 2013; Cooper et al., 2017; Shafroth et al., 2017; Schlatter et al., 2017) and 2-3 m for mature trees (Shafroth et al., 2000; Horton et al., 2001; Lite and Stromberg, 2005; González et al., 2012). Deviations from water table conditions under which roots systems developed and abrupt water table declines are detrimental for young seedlings (Guilloy et al., 2011), saplings (Shafroth et al., 2000) and mature individuals (Scott et al., 1999; 2000).
E. Protection from future flooding, burial and scour	Effects of anoxia due to soil saturation and flooding on *Salix* spp. and *Populus* spp. have been investigated through experiments using mesocosms (Tallent-Halsell and Walker, 2002; Kuzovkina et al., 2004; Francis et al., 2005) and field observations (Sprenger et al., 2001). Resistance to burial and scour stress, including biomechanical plant traits, has been less studied (but see for American species: Levine and Stromberg, 2001; Wilcox and Shafroth, 2013; Kui et al., 2016 for burial and Stone et al., 2013; Bywater-Reyes et al., 2015 for scour; for European species: Barsoum, 2000 for burial; Beismann et al., 2000; Karrenberg et al., 2003 for scour; Garófano-Gómez et al., 2016 for burial and scour). An indirect way to assess whether there are mortality risks related to flooding, burial and scour is to study the magnitude and duration of floods subsequent to the recruitment event, combined with monitoring of the fate of seedlings over time (e.g., Johnson, 2000). Dendrochronological and demographic analyses combined with analyses of historical flow regimes and aerial photographs can also help determine if the causes of seedling mortality were flooding, burial and scour (e.g., Polzin and Rood, 2006). Hydrodynamic models can also be used to estimate the shear stress exerted on plants (Pasquale et al., 2011). Ice jams are an important scouring factor in northern latitudes (Scott et al., 1997; Auble and Scott, 1998; Johnson et al., 2000; Rood et al., 2007)
F. Favorable chemical and physical properties of sediments	Moisture of the sediments can be directly measured in the field, or in the lab from sediment samples, using different types of moisture sensors (e.g., Friedman et al., 1995; Conroy and Svejcar, 1991; Taylor et al., 1999; Bhattacharjee et al., 2006; 2008; Pasquale et al., 2011; Schlatter et al. 2017). Sediment moisture is also highly determined by sediment texture (Cooper et al., 1999), and therefore, the latter has been combined with or used as proxy for the former in some restoration assessments (Taylor et al., 1999; 2006; Sprenger et al., 2002; Sher et al., 2002; Bhattacharjee et al., 2008; Caplan et al., 2013). Salinity is frequently assessed by measuring soil electrical conductivity (EC; e.g., Sprenger et al., 2002; Sher et al., 2002; Bhattacharjee et al., 2006; 2008; Bunting et al., 2011; Schachtsiek et al., 2014; Shafroth et al., 1998; 2017; Schlatter et al., 2017).
G. Low herbivory and grazing	Herbivory and grazing are usually monitored by their effects on vegetation, using metrics such as plant architecture, health condition and damage, mortality, cover, height, density, growth and recruitment (Beschta and Ripple, 2016 and references therein). Intensity of herbivory and grazing from livestock can also be indirectly estimated by determining the number and density of the animals through inventories

	(Batchelor et al., 2015; Beschta and Ripple, 2016 and references therein), the density of excrements, and by interviewing landowners and managers (González et al., 2017a).
H. Low competition	Surveys of plant composition, cover or density can be used to identify whether competitors of seedlings and juveniles of *Salicaceae* have emerged following restoration actions (e.g., secondary invasions of noxious weeds following *Tamarix* spp. control, González et al., 2017b; Sher et al., 2018) or resprouted after initial treatment (e.g., Bay and Sher, 2008; Sher et al., 2018). Native and exotic macrophytes with rhizomatous growth (e.g., *Ambrosia trifida*, *Phragmites australis*, *Arundo donax*, *Phalaris arundinacea*: Miller et al., 2008; Lee et al., 2010; Dean and Schmidt, 2011; Thomsen et al., 2012; Shafroth et al., 2017) and other grasses and forbs (e.g., *Fallopia japonica* and *Buddleja daviddii*: Tokarska-Guzik et al., 2006; Bottollier-Curtet et al., 2013; *Echinochloa crus-galli*: Bhattacharjee et al., 2008; *Melilotus* spp.: Taylor et al., 1999; Rood and Mahoney, 2000; Cooper and Andersen, 2012; *Equisetum arvense*: Rood et al., 2016; *Cynodon dactylon*: Grabau et al., 2011) have been identified as competitors of *Salicaceae* seedlings and juveniles. In North America, *Salicaceae* might be outcompeted by co-establishing invasive trees, such as *Elaeagnus angustifolia* and *Tamarix* spp. However, Sher et al. (2000; 2002; 2003) and Bhattacharjee et al. (2009) showed that *Salicaceae* are superior competitors as seedlings, with the strongest effects when hydrological conditions were favorable, and that in no instance did *Tamarix* spp. exclude *Salicaceae*. Allelopathic effects of *Fallopia japonica* on *Populus nigra*, *Salix atrocinerea* and *S. viminali* cuttings were reported by Dommanget et al. (2014); and of *Sapium sebiferum* on *S. nigra* by Conway et al. (2002).

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Appendix S3. Techniques to implement restoration actions found in the literature review. Uppercase letters in parenthesis after establishment requirements follow the order of the boxes in Figure 1. The list of techniques is not exhaustive.

Restoration action	**Establishment requirements targeted**	**Restoration techniques**
Seed augmentation	Seed available (A)	Branches with partially open, ripe catkins containing hundreds of seeds can be collected from local populations and be shaken (Gladwin and Roelle, 1998) or inserted in the ground at the restoration sites (Friedman et al., 1995, Sprenger et al., 2002; Bhattacharjee et al., 2006; Rood et al., 2016). Catkins and seeds can be also manually collected from living branches (Friedman et al., 1995; Rood et al., 2016) and be cold stored and remain viable for a few weeks (e.g., 4 ºC; Friedman et al., 1995; Grabau et al., 2011) or be spread immediately after collection over the restoration site by drill, broadcast (Grabau et al., 2011) and hydroseeding (Grabau et al., 2011; Rood et al., 2016; Schlatter et al. 2017), or using other devices (Friedman et al., 1995). Seeds can be planted at densities ranging from less than 10 seeds per m^2 (Schlatter et al., 2017) to hundreds of seeds per m^2 (Friedman et al., 1995; Grabau et al., 2011), although in natural conditions seed rain can even reach thousands of seeds per m^2 (González et al., 2016a). Some spreading methods (e.g., broadcasting) require that seeds were separated from the surrounding *pappus* (cotton) in a "cleaning" with an air nozzle and a series of sieves (Friedman et al., 1995; Gladwin and Roelle, 1998; Rowland et al., 2004); while others such as hydroseeding can be done without cleaning. The use of uncleaned seeds is preferred in large restoration sites as cottony hairs improve the anemochory (wind dispersal) and buoyancy capacity of seeds, enhancing dispersal and germination in wetter microsites more suitable for establishment (Seiwa et al., 2008). It is also much cheaper. Especially following storage, seed viability tests are recommended before seeding (Friedman et al., 1995; Rood et al., 2016; Schlatter et al., 2017).
Large pulse flow	Seed available (A); Moist, bare surface (B); Moisture available in the rooting zone (D); Favorable chemical and physical properties of sediments (F)	Floods able to do geomorphic work (significant erosion, deposition, channel change) aimed at promoting the creation of moist and bare surfaces adequate for seedling establishment should be ideally timed with seed release (Shafroth et al., 1998; 2010; Wilcox and Shafroth, 2013). High flows of lower magnitude could be artificially promoted to moisten bare surfaces previously created naturally or artificially and to increase moisture availability in the rooting zone (Rood and Mahoney, 2000; Hall et al., 2011a; Glenn et al., 2017). Large pulse flows can also help flush salts that have accumulated in floodplain soils (Briggs and Cornelius, 1998; Schlatter et al., 2017), especially when flooding is repeated over multiple years (Bhattacharjee et al., 2006; Bunting et al., 2013). Sediment releases from a Japanese reservoir provided inorganic nitrogen that favored *S. gilgiana* recruitment (Asaeda et al., 2015).
Irrigation	Moist, bare surface (B); Moisture available in the rooting zone (D); Favorable chemical and physical properties of sediments (F)	Restoration sites can be irrigated by sprinklers (Friedman et al., 1995; Bunting et al., 2013) or by local controlled flooding using irrigation infrastructure (Gladwin and Roelle, 1998; Roelle and Gladwin, 1999; Roelle et al., 2001; Sprenger et al., 2001; 2002; Bhattacharjee et al., 2006; 2008; 2009; Grabau et al., 2011). Irrigation water is usually taken from the stream channel (Friedman et al., 1995), from diversion channels and acequias (Sprenger et al., 2001; 2002; Bhattacharjee et al., 2006) and agricultural return flows (Briggs and Cornelius, 1998; Bhattacharjee et al., 2006). If water is limiting, other sources may be explored: for example, urban storm drains and treated wastewater effluents (Briggs and Cornelius, 1998; White and Stromberg, 2011; Marler et al., 2011). Bateman et al. (2015) showed how runoff from storm drains and effluent drains accidentally revitalized wetland and riparian vegetation including *Salicaceae* trees in urban reaches of the Salt River in the city of Phoenix, Arizona (USA), even though regeneration was not explicitly addressed.

Levee manipulation	Moist, bare surface (B)	Dismantling, breaching, and setting longitudinal structures, such as artificial levees, dikes and rip-raps, back into the floodplain have been reported as successful restoration measures to promote channel widening in constrained rivers, spontaneously re-creating safe-sites, and ultimately inducing recruitment of riparian *Salicaceae* in California, USA (Florsheim and Mount, 2002), Switzerland (Rohde et al., 2005; Pasquale et al., 2011) and Spain (Gumiero et al., 2013; González et al., 2017a; Martínez-Fernández et al., 2017).
Abandonment of human activities	Moist, bare surface (B)	*Salix* and *Populus* species have been reported to regenerate during transient stages of secondary succession without human assistance in abandoned agricultural fields of central Korea (Lee et al., 2002) and south-eastern USA (Battaglia et al., 2002). Abandonment of hybrid poplar plantations, with or without harvesting, can lead to semi-natural poplar groves as shown in Greece (Vasilopoulos et al., 2007), southern France (González et al., 2016b) and northern Spain (González et al., 2017a), but sexual regeneration of native *Salicaceae* in this scenario is extremely rare. In addition, naturalized *Populus* hybrids may increase the risk of hybridization and introgression with native *Populus* spp. (Paffetti et al., 2018). Recolonisation of *Salicaceae* in abandoned agricultural fields and areas formerly exploited for gravel extraction is also possible, especially if those remained hydrologically connected to the river and were regularly flooded (González et al., 2017a). Sexual regeneration in abandoned gravel pits may be more likely as these activities involve floodplain excavation and therefore unintentionally create moist, bare sites and, by lowering elevation, improve access to groundwater (Booth and Loheide, 2012; González et al., 2017a). Others have shown that abandonment of economic activities is the first step to attempt restoration, but then it has to be combined with additional restoration actions. For example, a restoration project in Colorado (USA) was initiated by the abandonment of a gravel extraction pit (Gladwin and Roelle, 1998; Roelle and Gladwin, 1999; Roelle et al., 2001). In the California-Arizona border (USA), agricultural fields were abandoned and replaced with *Salicaceae* forests (Briggs and Cornelius, 1998; Grabau et al., 2011; Bunting et al., 2011; 2013). More active site preparation by vegetation removal and land contouring, and irrigation by controlled flooding was necessary in all cases. Abandoning agricultural fields can be also combined with removal of flood defences (Gumiero et al., 2013) and planting (Holl and Crone 2004, Stanturf et al., 2009; McClain et al., 2011; Schachtsiek et al., 2014; Efthimiou et al., 2017). Likewise, planting has been carried out in abandoned borrow pits (Simmons et al., 2012) and gravel mines (Sheffels et al., 2014).
Pre-colonisation vegetation removal	Moist, bare surface (B)	Mechanical, chemical, burning and biocontrol treatments of invasive *Tamarix* spp. (Taylor et al., 1999; Sprenger et al., 2002; Smith et al., 2002; Bhattacharjee et al., 2009; Bay and Sher, 2008; Briggs et al., 1994; Briggs and Cornelius, 1998; Shafroth et al., 2017; Schlatter et al., 2017; Sher et al., 2018) and other competing weeds (Friedman et al., 1995; Miller et al., 2008; Cooper and Andersen, 2012) have been reported as a first step to promote *Salicaceae* establishment in North American rivers. It is recommended to synchronize vegetation clearing with the seed dispersal period and the start of large pulse flow releases (if applied) so that regrowth of competing vegetation would not occur before *Salicaceae* colonisation (Schlatter et al. 2017). Mature *Salicaceae* trees may also compete with *Salicaceae* seedlings (Cooper et al., 1999) and their removal could provide safe-sites for recruitment. However, we are unaware of any restoration project that removed mature *Salicaceae* trees to create safe-sites for new cohorts, probably because of the common perception that mature, dense *Salicaceae* forests are more valuable than young, sparse stands or unvegetated, potential recruitment areas (Le Lay et al., 2013). However, paradoxically, in projects aiming at improving water conveyance in river channels, *Salicaceae* stands are recurrently removed from the main channel, opening opportunities for recruitment (Geerling et al., 2008). Antagonistic management goals: avoid flood risk and promote *Salicaceae* regeneration (Floods and Habitats Directives in Europe: Gumiero et al., 2013; González et al., 2017b) may be reconciled in this way, provided that some channel migration is guaranteed and there is time for the *Salicaceae* patches to develop and grow to produce a shifting mosaic of changing habitats typical of healthy riparian forests (Johnson et al., 1976; Décamps, 1996). We suggest that mechanical removal of vegetation, including mature *Salicaceae* if necessary, and channel widening by levee manipulation may be combined to restore *Salicaceae* forests while improving channel conveyance, but none of the articles that we found in the literature reported such a combination.

Land contouring	Moist, bare surface (B)	Land contouring may go from relatively simple actions such as soil disturbance by ploughing, disking and bulldozing the surficial soil layer (Johnson, 1965; Friedman et al., 1995; Battaglia et al., 2002; Taylor and McDaniel, 2004; Stanturf et al., 2009), to the creation of microtopography irregularities to enhance trapping seeds and retaining water (Roelle and Gladwin, 1999; Simmons et al., 2012), surface leveling and smoothing (Taylor and McDaniel, 1998; Schachtsiek et al., 2014; Shafroth et al., 2017), re-activation of secondary channels (Rood et al., 2005; Shafroth et al., 2017), lowering floodplain elevation by excavation (Florsheim and Mount, 2002; González et al., 2017a), floodplain infilling to reverse mining effects (Roelle and Gladwin, 1999; Densmore and Karle, 2009; González et al., 2017a), reducing steepness of bank slopes (Roelle and Gladwin, 1999; Schlatter et al., 2017), and resculpting and constructing meanders, pools, ox-bow lakes and water channels (Simmons et al., 2012; Tiedermann and Rood, 2015; Beauchamp et al., 2015; Cooper et al., 2017).
Planting	Local plant stocks available or can be produced (C); Low competition (H)	Cuttings may resist flood scour better than seedlings (Friedman et al., 1995) and thus may be the preferred option at exposed areas, for controlling erosion and stabilizing streambanks (Shields et al., 1995; Watson et al., 1997; Rey and Labonne, 2015). Whole plant-pole plantings may be more appropriate for surfaces with competing vegetation in the understory (González et al., 2017a). Deep pole planting may be preferred in situations of low soil moisture, high soil salinity, deeper groundwater and no irrigation being possible (Dreesen and Fenchel, 2008; Hall et al., 2011b), but the groundwater must be reached by the poles to be successful (Dreesen and Fenchel, 2010; Sher et al., 2010; see sections D' and F'). Attention needs to be paid to introducing both females and males as *Salicaceae* species are dioecious (Landis et al., 2003), with more females than males being preferable to increase seed production. In arid and semi-arid rivers, planting genotypes with higher drought-tolerance (Sher et al., 2010; Grady et al., 2011) and flowering phenology synchronized with new periods of high flows might be advisable in the current context of global climate change (Stella et al., 2006; Rood et al., 2008). Plantings are usually combined with other restoration actions such as irrigation by controlled flooding (Taylor and McDaniel, 1998), sprinklers (Holl and Crone, 2004) or individual-irrigation systems (e.g., drip irrigation; Briggs and Cornelius, 1998; Holl and Crone, 2004; Schachtsiek et al., 2014; hand irrigation; Foster and Wetzel, 2005; see Section B' and D'); removal of competing vegetation, pre- and post-colonisation, both understory and overstory (Briggs et al., 1994; Briggs and Cornelius, 1998; Bay and Sher, 2008, Thomsen et al., 2012; Foster and Wetzel, 2005; Hovick and Reinartz, 2007; Miller et al., 2008; see sections B' and H'), and browsing exclosure through fencing (Briggs and Cornelius, 1998) or plastic or metallic individual containers to avoid consumption by wild ungulates and livestock, and beaver damage (Briggs and Cornelius, 1998; Pezeshki and Shields, 2006; Pezeshki et al., 2007; Hall et al., 2011b; 2015; Sheffels et al., 2014; see section G'). *Salicaceae* naturally grow in dense patches and we cannot expect planting to lead to the same structure (i.e., cover, density, vertical structure) as patches established by seed.
Managed recession	Moisture available in the rooting zone (D)	Managed recessions have been applied as part of environmental flows: Rood et al. (2003) applied managed recessions and associated alluvial water table drawdowns from 1 to 5 cm day^{-1} to restore local populations of *Populus fremontii* in the Truckee river in Nevada, USA. Water levels were brought down at ca. 2 cm day^{-1} following water releases from a reservoir in the Middle Rio Grande (Taylor et al., 1999; 2006), and at less than 1 m^3 s^{-1} day^{-1} in the Bill Williams (Shafroth et al., 2010). In the Colorado River Delta, Shafroth et al. (2017) reported that managed recessions of $<$ 4 cm day^{-1} did not limit seedling establishment. The river stage recession following a natural flood was kept between 2.5 and 5 cm day^{-1} in the Oldman River, Alberta, Canada (Rood et al., 1998), and between 2.5 and 4 cm day^{-1} in the St. Mary River, Alberta, Canada (Rood et al., 2016). Managed recessions have also been applied using former irrigation structures during controlled flooding. For example, in the Cache La Poudre former gravel pit restoration project drawdowns of 1 cm day^{-1} were applied using a system of drain culvert equipped with a screw gate (Roelle and Gladwin, 1999); in the Bosque del Apache National Wildlife, water-control sluice gates were allowed to divert water from an irrigation canal and manage drawdowns of from 2 to 10 cm day^{-1} (Sprenger et al., 2002; Bhattacharjee et al., 2006; 2008).

Managed base flows	Moisture available in the rooting zone (D)	Base flows have been included as a fundamental part of environmental flows prescriptions as well, for example in the Owens River (Hills and Platts, 1998), Bill Williams (Shafroth et al., 2010; Wilcox and Shafroth, 2013), in the Colorado River Delta (Shafroth et al., 2017; Schlatter et al., 2017) and in St. Mary River (Rood and Mahoney, 2000). A rule of thumb for environmental flows is to implement large pulse flows in wet years, when excess water from human uses may be available for ecological purposes, and focus on manage base flows during dry years (Rood and Mahoney, 2000; Rood et al., 2003; 2005; Shafroth et al., 2010; 2017). Managing base flows does not necessarily mean keeping summer stream flow levels abnormally high, as this can benefit non-target species, such as macrophytes and beaver (Wilcox and Shafroth, 2013), which can that can compete with or damage native *Salicaceae*.
Site selection	Moisture available in the rooting zone (D); Protected from future flooding, burial and scour (E)	An upper elevational limit for seedling establishment can be theoretically estimated for each river reach and species (for example 75 and 150 cm for *Salix* and *Populus* spp.; Recruitment Box Model; Mahoney and Rood, 1998; Amlin and Rood, 2002), based on moisture availability in the rooting zone determined by sediment texture, stratigraphy, precipitation and the flooding frequency and duration. A lower elevational limit should guarantee protection from future flooding, burial and scour. The Recruitment Box Model suggested 25 and 50 cm as lower limits of *Salix* and *Populus* spp. (Mahoney and Rood, 1998; Amlin and Rood, 2002). Lower limits are provided for *Salix* spp. given that they are generally more vulnerable to drought stress (e.g., Guilloy et al., 2011) but tolerate soil anoxia better than *Populus* spp. (Glenz et al., 2006). The elevational limits of the Recruitment Box Model are, of course, merely orientative: they need to be calibrated for each species, population, region and study sites. Moreover, they may be susceptible to change with climate change (Rood et al., 2008).
Deep planting	Moisture available in the rooting zone (D)	Even if deep-planted poles tap the groundwater, it is recommended that deep planting be combined with irrigation to improve moisture in the rooting zone whenever possible (Briggs et al., 1994). Some irrigation techniques such as embedded watering tubes are specifically designed to provide deep soil moisture, thus optimizing water use (Dreesen and Friedsen, 2008).
Flow sequencing	Protected from future flooding, burial and scour (E)	Unfortunately, few restoration projects have implemented restoration actions for periods longer than three years and flow sequencing has been infrequently applied. Experience has shown, however, that environmental flows are only successful when implemented as long-term programs over multiple years (e.g., Hill and Platts, 1998; Taylor et al., 2006; Shafroth et al., 2010; Wilcox and Shafroth, 2013; Aishan et al., 2013; 2015; Glenn et al., 2017; Foster and Rood, 2017), rather than isolated events such as one-time water releases (Shafroth et al., 2017, Schlatter et al., 2017). Repeated flooding may help to decrease soil salinity (Bhattacharjee et al., 2006; Bunting et al., 2013), helping to fulfill the requirement of high sediment quality (Figure 1F).
Sediment quality amendments	Favorable chemical and physical properties of sediments (F)	Amendments may include salt leaching (Sher et al., 2010; Schachtsiek et al., 2014), adding mulch and fertilizers (Densmore et al., 1987; Briggs and Cornelius, 1998; Densmore and Karle, 2009; Sher et al., 2010; Simmons et al., 2012), and organic matter (Simmons et al., 2012; González et al., 2017a). If sediments are transferred to the restoration site, attention needs to be paid on propagules of competitors (e.g., seed bank).
Herbivory and grazing control	Low herbivory/grazing (G)	Herbivory can be controlled by installing exclosures, such as fencing (Rood et al., 1998; Andersen and Cooper, 2000; Brookshire et al., 2002; Holland et al., 2005; Thomsen et al., 2012). Planted cuttings and poles can be protected individually using plastic or metallic stem collars and fence cages (Pezeshki et al., 2007; Miller et al., 2008; Lennox et al., 2011; Hall et al., 2011; 2015; Sheffels et al., 2014). Another way to control herbivory is by reducing the populations of herbivores and/or inducing changes in their browsing habits. One alternative for this is introducing herbivore predators. The wolf reintroduction in Yellowstone National Park (USA) was the most emblematic example of this practice. Wolf introduction reduced the population of ungulates and changed their browsing habits, and ultimately promoted regeneration of *Salicaceae* species whose regeneration had been impaired since wolves disappeared from the park (Beschta and Ripple, 2016 and references therein). Rodents such as beaver and nutria can be trapped and removed from restored rivers (Sheffels et al., 2014). Insect control may also

		be seen as restoration action as some insects (e.g., cottonwood defoliating beetle) may feed on *Salicaceae* leaves or use them to complete their life cycles (Watson et al., 1997; Dreesen and Fenchel, 2008).
Overbank flooding	Low competition (H)	Overbank flooding late in the growing season (i.e., fall floods; Roelle and Gladwin, 1998; Gladwin and Roelle, 1999; Roelle et al., 2001) or in the years following water releases (Shafroth et al., 2010; Wilcox and Shafroth, 2013) was shown to be successful in reducing *Tamarix* spp. establishment in several southwestern USA rivers, as *Tamarix* spp. have slower initial growth rates and are able to disperse seeds later in the growing season than *Salicaceae* and therefore seedlings are usually younger and smaller, thus more vulnerable to mortality by flooding than those of *Salicaceae* species (but see Sprenger et al., 2001). Well timed managed flooding also suppressed *Phalaris arundinacea* while increasing the cover of native *S. lucida* in a natural reserve of Oregon, USA (Jenkins et al., 2008).
Post-colonisation vegetation removal	Low competition (H)	In most cases, mechanical, chemical and biocontrol removal of competing vegetation following initial *Salicaceae* colonisation or planting may be necessary (e.g., weed control, Briggs et al., 1994; Briggs and Cornelius, 1998; Grabau et al., 2011; disking of co-occurring *Tamarix* spp. seedlings, Smith et al., 2002; Taylor and McDaniel, 2004). When chemical treatment to remove competing vegetation takes place, tree protectors can be used to minimize herbicide effects on *Salicaceae* (e.g., *P. arundinacea* treatment, Miller et al., 2008).

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