Received Date : 07-Aug-2015
Revised Date : 16-May-2016
Accepted Date : 03-Jun-2016
Article type : Technical Paper

Synthesis of common management concerns associated with dam removal

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Abstract: Managers make decisions regarding if and how to remove dams in spite of uncertainty surrounding physical and ecological responses, and stakeholders often raise concerns about certain negative effects, regardless of whether or not these concerns are warranted at a particular site. We used a dam-removal science database supplemented with other information sources to explore seven frequently-raised concerns, herein Common Management Concerns (CMCs). We investigate the occurrence of these concerns and the contributing biophysical controls. The CMCs addressed are: degree and rate of reservoir sediment erosion, excessive channel incision upstream of reservoirs, downstream sediment aggradation, elevated downstream turbidity, drawdown impacts on local water infrastructure, colonization of reservoir sediments by non-native plants, and expansion of invasive fish. Biophysical controls emerged for some of the concerns, providing managers with information to assess whether a given concern is likely to occur at a site. To fully assess CMC risk, managers should concurrently evaluate site conditions and identify the ecosystem or human uses that will be negatively affected if the biophysical phenomenon producing the CMC occurs. We show how many CMCs have one or more controls in common, facilitating the identification of multiple risks at a site, and demonstrate why CMC risks should be considered in the context of other factors like natural watershed variability and disturbance history.

(**Key Terms:** sediment management, headcut, aggradation, reservoir erosion, reservoir drawdown, wells, turbidity, non-native plants, invasive fish, dam removal, river restoration.)

Introduction

<u>Background</u>

The heterogeneity of implementation strategies, geographies, and characteristics of dam removals (O'Connor et al., 2015) has resulted in incomplete scientific knowledge and predictive models of river responses. Whereas conceptual models inform some elements of physical responses to dam removal (Doyle et al., 2003; Cannatelli and Curran, 2012), broadly applicable conceptual models to inform predictions about biological responses to dam removal and their linkages to physical responses are notably absent. As a result, resource managers often face uncertainty about how quickly and to what extent physical and ecological systems will respond to dam removal. In addition, stakeholders and managers often raise concerns about potential negative effects with each new project, whether or not a given concern is warranted. These sentiments illustrate that negative effects can and do occur at some sites, but can also reflect incomplete understanding of the controlling factors involved when overgeneralized to expect negative effects at every site. Whether or not specific negative impacts manifest appears to be strongly influenced by site conditions, such as the hydrogeomorphic setting, and the method by which the dam is removed (e.g., instantaneous or staged removal, Cannatelli, 2013), which offers promise that the occurrence of the management concerns can eventually be predicted.

Dam-removal science has increased in scope and depth, but frequently fails to provide the insights needed for managers to know if and when to anticipate negative effects. Based on a recent literature review, 586 documents related to the science of dam removal were identified, but only 179 of these were found to contain empirical information on measured responses (Bellmore et al., 2015). This research has largely emphasized field monitoring (e.g. Doyle et al., 2002; Kibler et al., 2011; Major et al., 2012) or numerical modeling (Cui et al., 2006; Cantelli et al., 2007; Wells et al., 2007; Cui and Wilcox, 2008; Downs et al., 2009; Konrad, 2009) to advance understanding of the rates and patterns of erosion and deposition associated with dam removal-induced sediment pulses, or on the impacts of those sediment pulses on fish (e.g. Allen et al., 2016, plus see Doyle et al., 2005 for review) and benthic macroinvertebrates (e.g. Stanley et al., 2002; Renöfält et al., 2013; Tullos et al., 2014). While these studies increase scientific knowledge, they usually do not focus on applied management issues, such as identifying when a dam removal is likely to negatively affect ecosystems or infrastructure, or address regulatory, engineering, or socioeconomic concerns. Furthermore, there is a suite of management concerns not directly related to the study of sediment dynamics or biological responses that are largely unstudied. Dams will continue to be removed and managers will need to make informed scientifically-based decisions about potential negative impacts and how to mitigate them, to which science should directly contribute. Advancing dam-removal science to help answer management questions is therefore imperative, and as this happens it will be equally important for managers to reevaluate applicable regulations and standards of practice to ensure that they are based on current science and are appropriate for a given project.

We investigate some of the common concerns managers face as they design and implement dam removals, in order to: 1) explicitly articulate these concerns and their potential negative consequences, 2) identify where and how commonly these concerns were ultimately valid, and 3) evaluate what conditions control their occurrence. We define these concerns, henceforth referred to as Common Management Concerns (CMCs), as outcomes that may require intervention but are broadly assumed, sometimes incorrectly, to occur at most sites. The CMCs addressed in this review are: 1) the degree and rate of reservoir erosion, 2) prolonged or excessive channel incision upstream of the reservoir pool, 3) downstream sediment aggradation, 4) elevated downstream turbidity,5) impacts of reservoir drawdown on local water infrastructure, 6) non-native plant colonization of former reservoirs, and 7) expansion of non-native fish. Figure 1 schematically depicts these concerns, and their typical geographic occurrence in a watershed.

Identifying common management concerns (CMCs)

As part of the USGS John Wesley Powell Center for Analysis and Synthesis (http://powellcenter.usgs.gov, accessed May 02, 2016) dam removal working group, comprising experts from a broad range of disciplines (e.g., geomorphology, water quality, ecology, engineering) and sectors (e.g., academia, government research institutions, government management agencies, and practitioners), we identified a broad suite of CMCs frequently raised in the dam removal planning process. We narrowed this broad set of CMCs to the seven investigated here that are among the most commonly raised across many geographies and project types (e.g., high-head vs. low-head dam removals), with others (e.g., contaminated sediments) briefly addressed in the Discussion.

Case study approach

For each identified CMC, we evaluated their relevance at dam-removal sites with available data of adequate spatial and/or temporal resolution. Because dam removals are often not thoroughly studied and the results of many dam removal studies are not widely disseminated in publicly-available literature, the analysis of each CMC was based on data or information from a relatively small number of sites. Moreover, because most dam removal studies only monitor a small set of physical or biological responses, we were unable to evaluate all seven CMCs at all of the sites. Random sampling or stratification of sites was not possible given the paucity and inconsistency of data available to assess each CMC.

We identified candidate sites (Figure 2, Table A1), henceforth referred to as case studies, for each CMC by querying a recently published database (Dam-Removal Information Portal, Bellmore et al., 2015) and by searching technical reports and grey literature. The database contains basic information from 179 scientific studies published between 1977 and 2014 that measure physical and/or ecological responses to 130 different dam removals across the United States and abroad. From each of these studies, the database contains information on (1) the physical, water-quality, and biological response metrics measured; (2) the type of experimental design employed, as well as the duration and frequency of sampling; and (3) the characteristics of the dam and its removal (e.g., dam height, location, year of removal). In addition to the Bellmore et al. (2015) database, we also identified candidate sites by consulting with working group members, and informally querying networks of practitioners, engineers, and scientists working in the science, practice, and regulation of dam removals. These impoundments, referred to herein as reservoirs, represent a range of sizes, removal strategies, and locations (Table A1). Despite biases of geography and literature availability in the data, the 65 case studies that were selected reasonably represent (Table A1) the geography of dam removals within the United States, plus 10 international locations. Below we separately present findings for each of the seven CMCs.

Evaluating the CMCs

1) <u>Degree and rate of reservoir sediment erosion</u>

Characterizing the concern

There are two management questions that are central to many dam removals: 1) How much of the sediment impounded within a reservoir will erode? and 2) How quickly will the eroded sediment move through the downstream river corridor (e.g., Downs et al., 2009; Sawaske and Freyberg, 2012; MacBroom and Shiff, 2013)? From a more practical perspective, will reservoir erosion be slow and incomplete, leaving behind exposed impoundment sediment that can be perceived as a “stinking mudflat” from which sediment bleeds out over time, or will erosion be rapid and complete, such that fluvial forms and processes are swiftly reestablished? The answer to these questions informs many other ecological and physical elements of dam removal, including several other CMCs that we examine in this paper. For example, the pace and volume of reservoir erosion influence downstream aggradation and turbidity, reservoir incision dynamics, groundwater impacts, revegetation of former reservoir reaches, and responses of downstream biota. Questions surrounding reservoir erosion also influence public perceptions of project success and inform management of reservoir areas, including the selected dam removal style (gradual versus rapid) and whether sediment excavation or other mitigation measures are implemented to reduce downstream transport, such as stabilizing banks in exposed reservoir reaches (e.g., Downs et al., 2009).

Approach

Rather than compiling findings from dam removal case studies, as we do for other CMCs in this paper, we instead take advantage of and summarize recently published syntheses of reservoir erosion following dam removal, with the intent of laying the groundwork for discussion of subsequent CMCs.

Key findings

Based on their analyses of reservoir erosion results from 12 predominantly low-head (2–14 m high) dam removals from across the northern US, Sawaske and Freyberg (2012) highlighted the influence of reservoir sediment characteristics (grain size, cohesion, spatial variability), and removal method on the evolution of reservoir sediment. They found that as of one-year post removal, the percent of the original reservoir sediment volume eroded ranged from less than 10% at sites where structures were constructed to limit erosion (LaValle Dam, Baraboo River, Wisconsin) to approximately 65% where not sediment management was implemented (Merrimack Village Dam, Souhegan River, New Hampshire). In addition, they found that greater fractions of sediment tended to be retained within reservoirs where deposits were predominantly fine and consolidated or cohesive. Conditions favoring larger (>15%) amounts of retained sediment occur when the ratio of the average width of the reservoir sediment deposit to channel width was > ~2.5, or where dam removal was phased rather than instantaneous. Major et al. (2016) synthesized post-dam removal reservoir erosion findings for twenty cases, including some of those assessed by Sawaske and Freyberg (2012) along with more recent large dam removals. Major et al. (2016) emphasized the role of dam height and removal strategy on reservoir erosion, and found for small and large dams that the rate and magnitude of reduction in base level strongly influences the rate and magnitude of reservoir erosion. Findings generally indicate that post-breach hydrology is of limited influence, at least initially, on the rate of reservoir sediment evacuation because erosional processes are dominated by the lowering of base level following dam removals (Randle et al., 2015; Major et al., 2016), though some exceptions exist where event-based erosion dominated and thus post-removal hydrology drove evacuation rates (e.g., Peck and Kasper, 2013; Harris and Evans, 2014). Post-removal hydrology is of particular significance in achieving the final "equilibrium" extent of lateral reservoir erosion, and can affect both vertical and lateral progression of reservoir erosion in reservoir deposits with predominantly cohesive sediment or ephemeral hydrology.

Condit Dam, on the White Salmon River, Washington, which was breached in 2011 by blasting a hole in the base of the 38-m high dam, illustrates how dam height and removal strategy can trump grain size as an influence on reservoir sediment erosion. Despite a substantial fraction of fine sediment in the reservoir deposit, which Sawaske and Freyberg (2012) suggested as a key potential factor in inhibiting reservoir erosion, erosion by landsliding and mudflows at Condit Dam resulted in evacuation of about one-third of the reservoir sediment by one week after the breach, and about 70 percent of the total reservoir sediment (i.e., 1.3 million m^3 of the original 1.8 million m^3 of stored sediment) as of one year after dam removal (Wilcox et al., 2014). The removal of Elwha and Glines Canyon dams on the Elwha River, Washington, between 2011 and 2014 highlight how, even for large dams, phased removal strategies can initially result in relatively slow rates of reservoir sediment release while the river erodes and redistributes sediment delta deposits that had not yet reached the dam at the time reservoir dewatering began (Randle et al., 2015). However, once the sediment from upstream deltas reached the dam sites, the multi-year phased removal on the Elwha River ultimately released a sediment quantity many times greater than the average annual load and exceeded more than 50% of the reservoir-deposit volume. Although the implications of reservoir erosion may be greatest for dams that store large sediment volumes, reservoir erosion is a concern across the spectrum of dam removals (e.g., MacBroom and Shiff, 2013), highlighting the need for ongoing evaluation and improvement of predictive tools to inform dam removal CMCs.

2) Excessive channel incision upstream of reservoirs

Characterizing the concern

Erosion of reservoir sediments can occur by a variety of mechanisms (e.g., Cantelli et al., 2007) but typically entails some degree of channel incision in the former reservoir that, if "excessive" or prolonged can cause a range of management problems. Here we evaluate CMCs associated with excessive or prolonged channel incision, which we define as: 1) post-removal incision, which may undermine infrastructure such as bridge piers or increase sediment loads; 2) headward incision upstream from the former reservoir pool which may result in bank erosion or unintended impacts to adjacent habitat or property; or 3) prolonged incision that takes years to occur, affecting how quickly dam removal goals are achieved and potentially creating a barrier to aquatic species passage.

Approach

We examined the magnitude, upstream extent, and rate of channel incision within reservoir sediment deposits after dam removal. We identified 38 sites via queries of the Bellmore et al. (2015) database, supplemented with information from technical reports, monitoring data, and discussions with project managers and/or scientists. The monitoring methods commonly used to document channel incision in the reviewed studies (Table A1) were longitudinal profile and/or cross-section surveys supplemented with time-lapse cameras, aerial photography, and field inspections. We use the term "headcutting" to describe the process of upstream progressing channel incision and "headcut" or "knickpoint" to represent the discrete location where there is an abrupt change in slope between the downstream reach that is in the process of adjusting to the new base level control and the upstream reach that has not yet incised (Leopold et al., 1964). For cases where data were available, we computed the dimensionless ratio of the length of headcut progression to the length of reservoir sediment deposition.

Of the 38 cases evaluated for channel incision, 12 were phased dam removals, 13 were considered instantaneous because reservoir drawdown associated with dam removal happened within hours to a few days, and 13 cases involved an unplanned dam failure or breach ("accidental"). The height of dams evaluated (Table A1) range from 1 to 64 m, reservoir sediment volume ranged from a few tens of cubic meters to tens of millions of cubic meters, and sediment distributions were mostly represented by non-cohesive coarse (sands and gravels) sediment. A cohesive sediment deposit was present at Brewster Dam (Straub, 2007), and within reservoir deposits behind Elwha and Glines Canyon dams (Randle et al., 2015). Relative to reservoir capacity, there was a range of cases from very little stored sediment to reservoirs completely full of sediment at time of removal. Three sites had ephemeral flow (Dinner and Maple Creek Dams, OR; Wellington Dam, Australia), whereas the others had perennial discharge. One site had tidal influence but the study area was located beyond the extent of the tidal backwater (Lake Charles VCU – Rice Center Dam, VA).

Key findings

The case studies generally conformed with conceptual stages of channel incision described in the literature for the vertical response to dam removal in the upstream impoundment (Doyle et al., 2002; Cannatelli and Curran, 2012). The magnitude of incision within the reservoir sediment deposit was generally equivalent to the thickness of the deposit above the pre-dam river bed for any given location. The post-removal channel stopped incising when the pre-dam gradient was achieved. This gradient typically coincided with the pre-dam river bed location, composed of an armored river bottom with relatively coarser sediment than the reservoir sediment deposit. Exceptions were documented on the Elwha River, where within a matter of weeks the river incision in Lake Aldwell formed a new channel across the valley from the pre-dam channel. During initial drawdown at Lake Aldwell, the river was flowing on the opposite side of the valley from the pre-dam channel location. As a result, during reservoir drawdown the river incised into a former terrace with a dense network of large tree stumps. During post-removal floods, the river laterally migrated back toward the pre-dam alignment in most locations. Further, the delta was composed of cohesive sediment that initially limited lateral erosion during 8 months of phased drawdown. Two years post-removal when the first large floods (5- to 20-year frequency) occurred, the river began to laterally migrate into the cohesive delta sediment back toward the pre-dam alignment. In total, the river incised 5.6 km upstream of the Elwha Dam, which was 20% farther than the estimated upstream reservoir extent. Upon removal of Dinner Creek Dam and Maple Gulch Dam in Oregon, the initially rapid headcuts (several m/hr) stalled when they encountered a former bedrock valley wall that confined the pre-dam channels (Stewart, 2006). At Dinner Creek Dam, the second flood four months post-removal resulted in channel erosion through an adjacent floodplain forest. However, at Maple Gulch Dam the discharge was intermittent and the channel remained perched above the original river bed at the conclusion of the study.

Few studies provide evidence of incision progressing below the pre-dam river bed. However, this did occur at Union City Dam, Connecticut (Wildman and MacBroom, 2005), where deep incision occurred due to an exposed sanitary sewer pipe with rock riprap that caused local downstream scour. Once the pipe feature failed (5 years post-removal), the headcut progressed upstream about 0.5 m below the original river bed. This resulted in a total incision length of 0.4 km, extending slightly farther upstream of the reservoir sedimentation effects.

Research from 10 of the 38 sites documented that incision progressed upstream beyond initial expectations, while another 10 studies documented that the extent of incision was within the identified reservoir backwater or sedimentation effects. Two of the studies had minimal sediment deposits that did not extend to the upstream end of the reservoir, so determining the headcut location was not applicable. At Condit Dam, project managers noted the knickpoint migrated a considerable distance upstream of the established project boundary, but the expected upstream extent of incision was not estimated before removal (personal communication, PacificCorp, April 2015). The extent and speed of downcutting of the riverbed in the upper reaches of the reservoir following the breach of Condit Dam (Wilcox et al., 2014) exceeded expectations and caused a loss of salmon redds (i.e., nests) during the year of dam removal (Engle et al., 2012). At Elwha Dam, incision exposed the pier foundation of an active highway bridge originally expected to have net aggradation from large sediment loads being released from upstream removal of Glines Canyon Dam (Randle et al., 2015). Eight sites were associated with historical dam failures in Canada where knickpoints continued up to 3 times beyond the boundary of estimated reservoir impoundments. The knickpoints were approximately 0.5 m in height and similar in size to nearby riffle features (Amos, 2008). There were no identified consequences of the headcut migrating beyond the reservoir impoundment at these sites over a period of years to decades.

Five studies stopped monitoring before the incision progressed to the upstream end of the reservoir, and 12 studies did not include monitoring in the upstream-most portion of the reservoir. Possible explanations are lack of safe access, limited monitoring budgets, or lack of perceived consequences. Additionally, it can be technically difficult to determine the upstream extent of reservoir sediment. Furthermore, it can be challenging to determine the exact location of a migrating headcut as it decreases in magnitude and in some cases looks similar to naturally-occurring steep cascades or riffles. Increased sedimentation and vegetation in the delta can also extend the distance of upstream sedimentation over time (Morris and Fan, 1998). Using complementary methods may help technical staff provide or refine sedimentation extent estimates to managers for decision making. These methods may include pre-dam topography (rare for small dams or sites constructed pre-1900), photos, and maps, projecting estimated pre-dam slopes from the dam site upstream, looking at depositional features that appear finer than adjacent sediment features in upstream and downstream alluvial reaches, and sediment probing or drilling investigations.

The case studies illustrate that managers can expect rapid (hours to weeks), initial incision in response to base-level lowering. This is followed by slower (weeks to years) subsequent event-driven incision when high flows are capable of eroding coarser sediments or sediments more distal from the newly-formed channel (Pearson et al., 2011; Major et al., *2016*). The subsequent incision from event-driven periods occurs during higher flow seasons of the year as proposed by Cannatelli and Curran (2012), with limited incision between events. Interestingly, the documentation available did not indicate that delayed or prolonged incision in itself caused any adverse consequences in achieving dam removal goals. One explanation may be that slower rates of sediment evacuation into the downstream channel are desired and considered a benefit. For example, at Brewster Dam the knickpoint took approximately a decade to progress upstream. Straub (2007) noted that it can be beneficial to remove a dam in phases for some settings where phased removal can reduce possible environmental effects by allowing the impounded sediment to slowly move downstream, and a stable stream and re-vegetated floodplain to form upstream.

The rate of upstream progression of channel incision varies with degree of reservoir sedimentation, rate of reservoir drawn down (phased vs. instantaneous), and erodibility of reservoir sediment. The fastest progression of channel incision occurred when removals were instantaneous and reservoir sediments were non-cohesive (Pearson et al., 2011; Major et al., 2012; Bountry et al., 2013; Tullos and Wang, 2014). Channel incision temporarily stalled or slowed when the river encountered newly exposed infrastructure, incised down to resistant pre-dam terraces, or intercepted coarser delta sediment at the upstream end of the reservoir. These instances required management action to remove or modify the infrastructure, a large flood capable of eroding more resistant delta sediment and bed layers, or a channel-widening phase that allowed the river to migrate off the terrace to the original stream bed location. The slowest rates of headcut progression (months to years) occurred at Wellington Dam and Maple Gulch Dam due to ephemeral hydrology, at Brewster Creek and Boulder Creek "upper dam" due to cohesive reservoir sediment, and at five sites where dam removal was phased over a period of months to years (Stewart, 2006; Orr et al., 2006; Straub, 2007; Burroughs et al., 2009; Neave et al., 2009; Randle et al., 2015). On the Elwha River, tributaries within the reservoir experienced lagged channel incision due to less streamflow relative to the main channel or becoming abandoned above the main channel during the incision phase.

Thirty cases documented the upstream extent of incision, which ranged from 0.2 to 5.6 km and was associated with dam heights ranging from 1 to 64 m. Fifteen sites recorded incision extent to be less than 1 km, nine sites had incision that extended 1 to 3 km, while the remaining 12 sites had incision that extended between 3 and 5.6 km upstream of the dam. Four sites had dams greater than 30 m in height, and, as would be expected, these dams (Condit, Barlin, Elwha, and Glines Canyon Dams) were associated with larger incision extents ranging from 2.7 to 5.6 km. However, the extent of incision for dams between 0 to 10 m was variable, with 13 dams extending 0 to 1 km upstream, 6 extending 1 to 3 km, and 5 sites extending 3 to 5 km upstream. The pre-dam slope divided by the dam height was found to be a good way to estimate the extent of upstream incision at some sites, including those that lacked channel obstructions such as bedrock outcrops, but not at all sites. Potential explanations for sites with poor correlations include incorrect assumptions of pre-dam slope or complex pre-dam profiles with slope breaks or bedrock outcrops.

In some cases, post-removal sediment management was performed to slow or stop channel widening or grade-control was placed at the dam site to limit erosion. In the cases reviewed, unanticipated storage and fining of new sediment occurred once the dam was removed because of the replacement of the base level control, which, in one case, impacted the project's ability to meet restoration goals (Greene et al., 2013). Use of a grade control may only be warranted when excessive downstream channel incision has occurred that could propagate upstream or consequences of additional sediment release are not tolerable. Additionally, use of bank stabilization may be warranted to prevent lateral migration where critical infrastructure or property are at risk. However, any such implementation of bank protection soon after dam removal should proceed with caution, because if the river has not had enough time to adjust its slope and width during post-removal high flows, the bank protection may be undercut or outflanked.

3) Downstream sediment aggradation

Characterizing the concern

While the restoration of sediment continuity is a benefit of some dam removals, the deposition of sediment (i.e., aggradation) downstream of dam removals can produce management concerns for aquatic ecosystems and human uses (e.g., infrastructure). Aggradation can influence ecosystems by directly burying organisms(e.g., spawning redds) and altering aquatic and riparian habitats (e.g., habitat homogenization by reducing the variability of bed elevations). Potential effects of aggradation on human uses include increases in the magnitude and frequency of overbank flow (i.e., flooding), as well as adverse impacts to water supply (e.g., if groundwater-surface water exchange is altered or if diversion structures are affected) and recreational use (e.g., if river access points are impacted). From a management perspective, it would be helpful to know how much of the sediment eroded from the reservoir will be transported downstream, where deposition will occur, how much will be delivered to the next waterbody downstream, and for how long deposition will persist. However, quantifying patterns of deposition is complicated by the spatially and temporally dynamic nature of deposition processes. Consequently, understanding physical versus ecological management concerns associated with sediment deposition may require alternative monitoring approaches. For example, cross-section-averaged values of channel aggradation may be relevant to investigating potential changes in stage-discharge relationships and effects on overbank flooding, yet such values may obscure patterns of aggradation at scales relevant to aquatic organisms (e.g., between bars and pools; Zunka et al., 2015).

Approach

Post-removal changes in downstream bed elevation, based on comparisons of pre- and post-removal topography collected using a range of survey methods, have been documented in numerous dam removal studies. The distances downstream of removed dams for which aggradation results have been reported are not standard due to differences in study designs, varying distances to downstream confluences, and variations in downstream morphology and deposition potential. Duration of data collection is also inconsistent, though most data are limited to within one-year post dam-removal. We therefore report findings on downstream aggradation as of one-year post-dam removal for seven dam removals, all except one of which are in the Pacific Northwest (Table A1), recognizing that impacts frequently diminish over time and thus long-term responses will likely vary from those reported below.

Key findings

For some of the dam removals evaluated, the magnitude and duration of downstream sediment deposition following dam removal tended to be most influenced by proximity to the dam, i.e., aggradation was greatest near the dam. Following removal of Savage Rapids Dam from the Rogue River, Oregon, filling of pools immediately downstream of the dam was detected in the year after removal (Bountry et al., 2013; Tullos et al., 2014). Aggradation of downstream riffles was limited, although sediment did accumulate immediately downstream along the intakes of the pumping station constructed to replace the diversion dam. Clogging of the intakes required small volumes (~1500 – 4500 m^3) of sand and gravel to be excavated in the first two springs following dam removal before irrigation season but no action was needed in subsequent years (Bob Hamilton USBR, personal communication).

Likewise, following removal of Marmot Dam on the Sandy River, Oregon, downstream aggradation patterns were strongly influenced by proximity to the dam and by the coarse grain size of the reservoir deposit. Immediately downstream of the dam, the bed aggraded in a sediment wedge that tapered from a maximum thickness of 4 m to zero thickness (i.e., back to the pre-removal channel bed) 1.3 km downstream of the dam site during the first year after removal. This sediment wedge represented about one-third of the sediment eroded from the reservoir in the first year following removal (Major et al., 2012). The remainder of the eroded reservoir sediment traveled downstream of this initial wedge, where the river transitions into a steep and confined reach in which sediment aggradation was generally limited and restricted to pools. Downstream of the gorge (9 km from the dam), in a reach where managers had initially been concerned about potential aggradation impacts on salmon redds, minimal aggradation was observed (Major et al., 2012).

A river's transport capacity, which depends on channel slope, discharge, grain size, and confinement, is also a primary control on the magnitude and duration of downstream sediment deposition and in some cases can trump the effects of proximity to the dam. Removal of Condit Dam exposed a 5.3 km reach between the dam and the Columbia River to sediment deposition and channel aggradation. In a steep and confined reach downstream of the dam, riffles returned to near pre-removal elevation within 15 days of breaching (Wilcox et al., 2014), a condition that persisted as of one year post-removal. Downstream, in a less-confined reach influenced by the backwater effect of the Columbia River, large-magnitude (3–6 m) and persistent (as of 9 months post-breach, and likely beyond) bed-elevation increases occurred following dam removal (Colaiacomo, 2014). From a management perspective, the primary issue raised by aggradation of the lowermost White Salmon River was temporary loss of access to a tribal fishing site associated with filling of a deep pool.

After removal of Milltown Dam on the Clark Fork River, Montana, downstream aggradation showed substantial spatial variability that reflected transport capacity rather than proximity to the dam. Repeat cross section surveys in a high transport-capacity reach extending from 2 to 6 km downstream of the dam indicated minimal topographic change as of one year after dam removal. Sixteen km downstream, in an unconfined reach with lower transport capacity, aggradation on the order of 1 m was estimated on bars within the first year following dam removal. Anecdotally, this aggradation led to filling of irrigation ditches and modified aquatic habitat.

Other dam removals show a mix of controls on downstream aggradation. Removal of Merrimack Village Dam released sediment into a short (~ 0.5 km) reach of the Souhegan River in New Hampshire before it enters the Merrimack River. The Merrimack River influences hydraulics and deposition in this lowest portion of the Souhegan River (Pearson et al., 2011), analogous to the White Salmon River's confluence with the Columbia River. In the first several weeks after dam removal, bed aggradation averaging 2.1 m and as much as 3.2 m occurred in the Souhegan River downstream of the former dam. This aggradation resulted in a steeper river gradient, which increased transport capacity that subsequently incised through the new deposits. As of one-year post-removal, average net aggradation in the ~0.5 km downstream of the dam was 0.24 m (maximum 1.5 m), although in some locations, the channel had incised to a lower elevation than the pre-removal bed (Pearson et al., 2011). Bed elevation changes on the Souhegan River following dam removal reflected a hybrid of sediment transport associated with high-flow events and inter-event sand transport, and deposition associated with backwatering of the Merrimack River (Pearson et al., 2011).

On the Elwha River, where the largest-ever dam-removal volume of reservoir sediment is being released into a downstream river system, downstream of the former Elwha Dam and Lake Aldwell, in the ~7 km "lower reach," one-year post removal aggradation was generally on the order of 0.1-0.5 m in the channel and 0.39 m (±0.43m) in floodplain channels (East et al., 2015). Greater aggradation was observed in the second year of dam removal on the Elwha River, when a large sediment pulse from the upper reservoir, Lake Mills (mobilized by removal of Glines Canyon Dam) moved downstream. The resulting aggradation included widespread bed-elevation increase of ~1 m in the lower reach, accompanied by new bar formation and increased braiding, and greater sediment thicknesses accumulating in former pools (East et al., 2015). This aggradation (and continued bedload transport) in the lower Elwha River temporarily impaired a water supply intake for the city of Port Angeles, which was mitigated by mechanical sediment removal in the vicinity of the intake and construction of additional pumps. Overall effects on water supply infrastructure, however, were limited as a result of pre-removal mitigation efforts (Bountry et al., 2015). Once substantial erosion out of Lake Mills (i.e., past Glines Canyon Dam) began, aggradation levels in the ~14 km "middle reach," between Glines Canyon Dam and Lake Aldwell, were similar to those seen in the lower reach, despite a steeper (0.7-0.8%) gradient in the middle reach. Management implications of this aggradation included flooding of campgrounds and bank erosion, which would be expected in a sediment starved fluvial reach after re-introducing bedload (Bountry et al., 2015). As the sediment wave (largely originating from the upper reservoir, Lake Mills) passed, the middle and lower Elwha River reaches began to undergo incision through the new sediment deposits. Downstream incision was already apparent as of two years into the removal of the Elwha dams (East et al., 2015).

As the cases above suggest (see also Major et al., 2016), downstream sediment deposition following dam removal tends to be influenced by proximity to the dam, the river's transport capacity, and the downstream distance to the next larger river or water body; and, the overall effect can be transient (<1 year), persistent (>1 year), or longer term (>5 years). The ratio of the volume of stored reservoir sediment to the river's average annual sediment load, denoted here as V*, can be predictive of downstream aggradation, where lower or higher V* values are associated with smaller or greater downstream impacts, respectively (BOR, 2006). However, the data needed to estimate V* may be unavailable in many rivers, potentially restricting its use for predicting aggradation potential.

To evaluate if ecologically significant aggradation has occurred after dam removal, measuring bed adjustments in terms of bed relief, defined as the difference in elevation along a cross section between the bottom of a pool and the top of a bar (Zunka et al., 2015), can provide an assessment of habitat variability and homogenization. Bed relief can be normalized by the 90th percentile of observed, pre-dam-removal relief values within a reach, providing a dimensionless metric for assessing aggradation. This type of analysis could be categorized by channel units, to illustrate if only low-energy pools and backwater areas have filled with sediment or additionally higher energy, hydraulic controls have also aggraded that could increase flood stage.

4) <u>Elevated turbidity</u>

Characterizing the concern

Suspended sediment is a naturally occurring and necessary component of many biophysical processes, yet unnaturally elevated concentrations can have deleterious ecological effects and consequences for human uses (USEPA, 2006). For example, fish can suffer a range of direct and indirect effects on both behavioral (e.g., inability to see prey) and physiological (e.g., impaired gills) systems (Kemp et al., 2011). Human-use impacts from elevated turbidity include recreation, aesthetics, and safety, as well as increased drinking-water treatment costs (USEPA, 2006). In addition, 30 of the 32 states that have numeric criteria for regulating sediment in surface waters prescribe criteria for turbidity (USEPA, 2006), and thus dam removal practitioners and regulators are concerned about exceeding state turbidity regulatory standards and the impacts those standards are designed to avoid. At most dam removals with stored sediment, some proportion of the released sediment is fine-grained and will increase the suspended-sediment load downstream, at least temporarily (e.g., Major et al., 2012; Bountry et al., 2013).

Approach

There are a limited number of published dam removal studies that have quantitative analyses of turbidity data (<20) and some of these present data from the same sites (Bellmore et al., 2015). We restricted our analysis to projects with high temporal resolution turbidity data (≤1 hour) upstream and downstream of the removal for at least one month before and after project implementation. Four sites from the Pacific Northwest U.S., one from New England, and two from the Mid-Atlantic U.S. met these criteria (Table 1). At each site, the paired turbidity data were collected by the U.S. Geological Survey using comparable instruments, methods, and sampling frequencies (Chaplin et al., 2005; Major et al., 2012; Magirl et al., 2015; C. Anderson and W. Banks, personal communications). Only three of the seven sites had pre- or post-removal data collected for more than one year, so our analyses evaluate relatively short-term, acute, and likely maximum turbidity impacts from dam removal.

We compared observed turbidities at the seven sites to applicable state turbidity standards, evaluating the occurrence and degree of elevated turbidity pre- and post-removal by measuring estimated state threshold (ST) exceedance magnitude and duration, respectively, as:

$$T_{max}\ (\%) = (\text{maximum peak value/ST}) * 100;$$

$$T_{dur}\ (\%) = (\text{duration of ST exceedance/monitoring duration}) * 100$$

Of the five states where the sites are located, one has absolute numeric criteria and two have numeric criteria indexed to "natural" conditions for the stream (Table 1). Neither of the state standards that are indexed to natural conditions prescribe methods for determining them, so we estimated natural conditions as the range of values below, and including, the 90th percentile value in the upstream turbidity time series (ODEQ, 2014). This approach may be less common than using contemporaneous readings at a control instrument above the project site as the "natural" comparison (e.g., Major et al., 2012; Bountry et al., 2013), but our approach facilitates comparisons between observations at the downstream impact location and the full record of natural variability at the upstream control.

Two sites are in states with no turbidity standards: Conodoguinet Creek (Pennsylvania) and the Penobscot River (Maine). For the Conodoguinet site, we compared observed turbidities to natural conditions defined as above: the 90th percentile value of the upstream turbidity record. At the Penobscot River dam removals, where there is no turbidity record upstream, we estimated natural conditions at the 90th percentile value of the downstream record, benefitting from an extended pre-removal monitoring period. Our estimate may therefore be affected by particle settling in the two upstream reservoirs before the Penobscot dams were removed. However, we believe this effect is negligible because the reservoirs trapped very little suspended sediment as a result of basin physiography, impoundments further upstream, local hydraulics, and reservoir operations (Collins et al., 2012).

Key findings

Not surprisingly, contemporaneous turbidity measurements upstream and downstream of dam removal sites frequently show larger downstream turbidities in the days and months following dam removal (Figure 3). Indeed, the data we examined show that the largest turbidity values downstream of dam removals are commonly at least an order of magnitude higher than the estimated ST (i.e., T_{max} > 1000 %). Yet this is also true during high flows at the gages upstream of these sites, and/or at the downstream gage during high flows in the pre-removal period, indicating that turbidity peaks associated with dam removals are generally within the range observed during storm events (Table 1, Figure 3a, Figure 3b). In addition, the relative magnitudes of T_{max} for upstream and downstream sites during storm events are similar for their respective pre- and post-removal periods, and the directions of change from pre- to post-removal are the same (Table 1). T_{max} values computed for the Elwha River removals are artificially low because the nephelometer instruments used to measure turbidity there have an operational range up to 1,500 FNU (Figure 3c) and values above 1,200 FNU are considered unreliable (Curran et al., 2014). Turbidity durations above estimated ST (T_{dur}) also suggest storm events are a more important influence on river turbidity than dam removals. For most sites, T_{dur} values in respective pre- and post-removal periods for upstream and downstream records correspond, as do directions of change from pre- to post-removal (Table 1). Data from sites we did not quantitatively analyze broadly support our observations that dam removal-induced turbidity events are analogous to events produced naturally by storms in their magnitude and duration (Stewart, 2006; Granata et al., 2008; Gibson et al., 2011; Marion, 2014).

There are two important exceptions among the seven sites we analyzed. The downstream gage at the Elwha River shows a clear post-dam removal shift to prolonged, elevated turbidity that is not manifest in the upstream turbidity record and reflects site-specific conditions and a phased dam removal strategy, as discussed below (Table 1 and Figure 3c; Magirl et al., 2015). The other exception is the Conodoguinet Creek in Pennsylvania, where the downstream gage has prolonged, elevated turbidity pre- *and* post-removal relative to the estimated ST (Figure 3d) and the upstream record (Table 1). Chaplin et al. (2005) suggest that elevated turbidity here may reflect unmeasured inputs from an urbanized tributary and/or disturbance by waterfowl, recreational boaters, and fisherman.

Considering a large proportion of the annual sediment load of many rivers is transported during just a few days of the year when flows are high (Wolman and Miller, 1960), and that many impoundments have low volumes of stored reservoir sediment relative to the river's average annual sediment load (V*), it is reasonable that turbidities caused by dam removal would infrequently exceed those produced naturally by event flows. Moreover, the finer sediment fractions that compose suspended load are often not trapped effectively in the many shallow, riverine impoundments in the United States (Brune, 1953). Thus, even impoundments storing multiple years of bedload may be storing considerably less suspended load, and those materials may not be accessed *en masse* after dam removal (Bountry et al., 2013).

Cases where dam removal-induced turbidity exceeds storm-induced turbidity, either in magnitude or duration, appear to be associated with exceptional situations where large impoundments storing decades of annual sediment load are removed and/or with specific dam removal methods. For example, the two Elwha River reservoirs together stored approximately 100 years of annual sediment load, nearly half of which was fine material (<0.063 mm; Warrick et al., 2012; Magirl et al., 2015; Randle et al., 2015). Large quantities of fine sediments, combined with a multi-year phased removal, resulted in elevated turbidity for durations much longer than observed at other sites we examined (Figure 3c and Table 1). Another example is the Condit Dam removal, where the dam stored approximately 100 years of annual sediment load with a large proportion of fine sediment (35% by volume <0.063 mm) (Wilcox et al., 2014). Estimated turbidity magnitudes downstream in the days and weeks following dam removal were at least an order of magnitude higher than any values documented during storms at an upstream turbidity station over the six months preceding, and the year following, removal (Riverbend Engineering and J.R. Merit, 2012; Kleinfelder et al., 2012). This occurred because of the large quantity of stored fines and an unusual dam removal method, an explosion at the base of the dam that facilitated rapid water and sediment drainage from the impoundment and ultimately resulted in hyperconcentrated flow (Wilcox et al., 2014). These results highlight how reservoirs with a high V* and fine-grained sediment can lead to large post-removal turbidity pulses, which may be mitigated by phased drawdown and revegetation (to stabilize reservoir sediment) or dam removal during low flow conditions in order to minimize sediment suspension.

5) Drawdown impacts on local water infrastructure

Characterizing the concern

Reservoir drawdowns associated with dam removal not only cause erosion of impounded sediments, but also lowering of water tables that may have been elevated because of the dam. As a result, communities that developed around reservoirs and came to rely upon infrastructure around the elevated water table upstream of dams, or the associated groundwater pressure gradient downstream of dams, may be impacted by reservoir drawdown. Reservoir drawdown may have additional effects on septic systems, well-water supply (USBR, 1997), and groundwater-dependent habitats (HDR, 2010). We focus our review on the impacts on wells in near proximity to the reservoir or those located along the depositional zone in the downstream river because stranded wells appear to be the most common concern around local groundwater changes with dam removal.

Approach

At each site, we attempted to identify the number of pumps and wells impacted, if those impacts were anticipated and investigated in advance, the costs of modifications to and replacements of impacted wells and pumps, the distance of the pumps and wells to the dam site, and other potentially relevant factors associated with the impact of the reservoir dewatering on local water infrastructure. However, peer-reviewed literature on local water infrastructure is scarce, which led us to seek information from engineering memos, Environmental Impact Statements (EIS), an unpublished dissertation, and Federal Energy Regulatory Commission (FERC) documents associated with the case studies. In addition, we contacted dam removal project managers to discuss their knowledge of impacts to wells and pumps at dam removal sites. The result is a review of six case studies from the Pacific Northwest, Intermountain West, and Northeast of the U.S.

Key findings

We identified three primary scenarios associated with impacts to local water infrastructure. The first and most common scenario was when the aquifer was hydraulically connected to the reservoir and managers anticipated that wells constructed after the dam was built needed to be deepened or moved in advance of the dam removal. This occurred at Milltown, Gold Ray, and the Elwha and Glines Canyon dam removals (Table A1). The drawdown of the reservoir behind the 12.8 m tall Milltown Dam, Montana impacted groundwater elevations within the reservoir area, as well as 6 km downstream and 2 to 3 km upstream of the dam, with the degree of lowering varying with distance to the reservoir and the direction of groundwater flow (Berthelote, 2013). An analysis of groundwater elevations pre (2006) and post (2010) dam removal indicates that the water table dropped by up to 2.4m, though most locations were in the range of 1.5-1.8m (USEPA, 2011a). Monitoring of groundwater wells also indicated that dam removal modified the direction of groundwater flow near the dam (USEPA, 2011a). The impacts to the groundwater table were mitigated through the construction of 82 new wells, lowering of 20 pumps, and reconfiguration of an unspecified number of additional wells (USEPA, 2011a). At Gold Ray Dam, Oregon, it was anticipated that some of the wells within 0.4 km of the reservoir would be affected by the lowering of the reservoir (NMFS, 2010), though no quantitative analysis was conducted. Following dam removal, two nearby private potable water wells (Jackson Co., personal communication)—both shallow wells that were hydrologically connected to the artificially-elevated slough—went dry, interrupting water supply. At Elwha and Glines Canyon dams, pre-removal projections indicated that wells upstream of the reservoirs would be impacted by the reservoir drawdowns and surface-water intakes downstream of the dams would be impacted by the changes in water stage and clogging from increased suspended sediment (USBR, 1997; URS, 2001). Analyses were based on field exploration of the surficial and subsurface geology and aquifer testing, as well as groundwater modeling ranging from Darcy flow calculations and groundwater budgeting to two-dimensional groundwater modeling (URS, 2001). Anticipated impacts were concentrated on water infrastructure downstream of each former dam and recommendations for mitigation measures included the replacement of domestic and municipal supply wells (USBR, 1997), though not all of the recommendations were implemented. While formal groundwater monitoring post-removal was not funded, landowners anecdotally reported decreased yields from their wells. Key factors controlling impacts to downstream water users appear to be the location of the channel and sediment accumulation in side channels (East et al., 2015).

The second scenario for drawdown impacts on water infrastructure is reflected by a hydraulically-connected reservoir where managers failed to anticipate that wells needed to be deepened or moved in advance of the dam removal. Impacts on local wells from the drawdown of the reservoir behind Condit Dam were not anticipated in the pre-removal EIS analysis (Sandison, 2010), though the FERC surrender order had provided some indication before dam removal that well impacts were anticipated (FERC, 2010). After the removal, at least eight wells were impacted by the lowering of the groundwater table (Learn, 2011). Whether PacifiCorp, the dam owner, was legally obligated to replace or modify the wells was debated, given the seniority of their water rights and because the wells, drilled after dam construction in 1913, accessed groundwater that was artificially elevated by the reservoir. PacifiCorp offered to pay property owners up to $5500 to drill deeper wells, though local landowners argued that the compensation was not adequate (Learn, 2011). Another unanticipated consequence of groundwater response to dam removal occurred on the Elwha River, where some groundwater-fed side channels within the floodplain of the lower reaches dewatered (with a water-table decrease of approximately 1 m) within a year after the start of dam removal, after the lower reservoir had been drained of water. Prior to reservoir dewatering, the water-surface elevation in those same side channels had been constant throughout the previous 6 years of pre-dam-removal monitoring (Draut and Ritchie, 2015). The reduced groundwater contribution to floodplain side channels had ecological consequences because these channels constitute important rearing habitat for young fish in gravel-bed rivers of the Pacific Northwest (Morley et al., 2005), though the impact is not likely to be persistent.

Finally, the third scenario was no hydraulic connection between aquifer, reservoir, and local infrastructure. In the one case we identified representing this scenario, analysis was conducted in advance to evaluate the need for mitigation. Pre-removal investigations of the potential impacts of Great Works and Veazie dams, Maine, on local wells indicated that all examined wells were drawing water at elevations below the expected post-removal water-surface elevation, in many cases much below, and thus were unlikely to be impacted (PRRT, 2009). Many of the wells were drilled into bedrock, and the water source was not hydraulically connected to the reservoir, therefore, the likelihood of negative effects on the wells was low (PRRT, 2009). This prediction was validated when no wells were affected by the project.

The case studies examined herein appear to have certain commonalities around shallow groundwater and primarily drinking-water uses. Anticipating if pump and well impacts will occur and require mitigation is an important element of the dam removal planning process requiring knowledge of the local hydrogeology. The relevant hydrogeologic processes that control if reservoir drawdown will impact local infrastructure are consistent with processes associated with rising local water tables following dam construction (Leopold and Maddock, 1954; Rains et al., 2004; Heilweil et al., 2005), driven primarily by valley confinement and water-table depth (Berthelote, 2013) that impact groundwater recharge and discharge. The hydrogeologic conditions associated with large groundwater table responses to dam removal that can impact water infrastructure appear to be: 1) a large drop in reservoir elevation relative to impacted infrastructure, 2) a high degree of connectivity and groundwater flow directions that provide high rates of exchange between the reservoir, river, and groundwater; and 3) wells drawing from an alluvial, and artificially-elevated, aquifer as opposed to a confined aquifer.

The available tools for evaluating if these conditions exist at a site range from simple to complex. To assess potential impacts on local water infrastructure, project scientists and engineers might: 1) analyze surficial geology information and depth to groundwater, well records (where available), and projected dam-removal hydraulics; 2) develop mathematical, statistical, and/or water budget models of groundwater flows (e.g., Berthelote, 2013; URS, 2001); and/or 3) apply numerical groundwater models to simulate water-table elevations (e.g., Berthelote, 2013). Post-removal monitoring of wells, discharge from springs, and water levels in nearby lakes, wetlands, and alluvial floodplain channels (Shuman, 1995; Draut and Ritchie, 2015) will provide evidence to validate models and advance general understanding of groundwater responses to dam removal.

6) Non-native plant colonization of reservoirs

The introduction and spread of non-native species, including both fish and plants, is considered to be one of the largest threats to ecosystems around the globe (Simberloff et al., 2013). When removed from ecological constraints found in their native environment, non-natives can become “invasive,” which then have the potential to out-compete and hybridize with native species (McLaughlin et al., 2013), restructure food webs (Cross et al., 2013), and undermine ecosystem diversity and productivity (Rahel, 2007).

Characterizing the concern

Colonization and spread of non-native plants on former reservoir sediments is a common concern of resource managers. This CMC applies across various land management objectives, including protection and maintenance of natural ecosystems and species diversity (Chenoweth and Acker, 2009; Woodward et al., 2011), or municipal-park management (Orr and Stanley, 2006). However, the acceptable range of non-native plant colonization or expansion varies with each dam removal case and management entity, though it is likely that most resource managers would find it unacceptable if one or a few non-native species came to dominate a former reservoir site. Also, control and containment of noxious weeds or particularly invasive species may be legally required in some cases.

Approach

We assessed the severity of non-native plant richness (i.e., the number of non-native species present at a site) or abundance (i.e., percent cover or frequency) by comparing values in recently exposed sediments at former reservoirs to values reported for riparian corridors. Studies in South Africa, France, and North America suggest that non-native plants commonly comprise 20-30% of the species present in riparian floras (Hood and Naiman, 2000; Planty-Tabacchi et al., 1996). To assess the extent to which non-native plant species colonize and spread on former reservoir sediments, we compiled datasets from published or grey literature, or unpublished data that contained a high-quality list of species present at a site and/or a quantitative estimate of the abundance of each species present. Further, we excluded sites that were known to have had significant management, such as weed control or active revegetation. This resulted in 25 sites with acceptable species richness data and 18 sites with acceptable abundance data (Figure 4). Because data collection methods were not consistent across sites (i.e., sample plot distribution and size; abundance metrics), we were only able to calculate the relative contribution of non-native species to species richness and vegetation abundance, estimated as the proportion of non-native species in the observed flora for each dam removal site (richness) or the relative abundance of non-native vascular plants. Typical abundance measures, before relativizing, were percent cover or frequency.

Further, we explored relationships between variables known or hypothesized to influence patterns of plant colonization and succession in former reservoirs or similar environments, including two available for all sites (time since dam removal; dam size), and several others which we discuss based on a relatively small number of case studies (landform or topographic position; sediment grain size and chemistry; weed control; active revegetation efforts; non-native propagule pressure).

Key findings

The proportion of non-native species averaged 0.31 and ranged from 0.13 to 0.68 at the 25 study sites for which data were available (Figure 4a). The proportion of non-native taxa at the majority of the sites (15 of 25) was 0.3 or less, similar to many riparian floras around the world (see above). The proportion at five sites was between 0.31 and 0.4, and was >0.4 at five sites. Most sites were sampled from 2-10 years following dam removal, and there was not a significant quadratic or linear relationship between time since dam removal and the proportion of non-native species. Height of the former dam, a proxy for reservoir area, also does not appear to have a clear influence on the contribution of non-native taxa to species richness (Figure 4a).

The relative abundance of non-native plants was very similar to relative species richness, averaging 0.32 and ranging from 0.07 to 0.71 (Figure 4b). Orr and Stanley (2006) noted a high frequency of occurrence of non-native species in their study plots, but their metric was quite different than ours and does not reflect the relative frequency of occurrence, which we calculated from their data (C. Orr, personal communication) and which is much lower. Though highly variable across the full range of sites, there were some instances where non-native species were highly abundant, including some of the small dam removal examples in Wisconsin (Lenhart, 2000; Orr and Stanley, 2006). The most commonly-cited non-native invasive plant in these studies was reed canary grass (*Phalaris arundinacea*), which occurred at 100% of the Wisconsin study sites. *P. arundinacea* is also a common invasive plant in other parts of the U.S. including the Pacific Northwest, where it occurred in post-dam removal vegetation surveys on the Rogue and Sprague Rivers in Oregon (Tullos unpublished data) and the Elwha River (Schuster, 2015). Across all sites examined for this synthesis, only two (Parfrey Glen Dam on the Pine River, Wisconsin and Wonewoc Dam on the Baraboo River, Wisconsin) of 20 exceeded 50% relative abundance of non-native plants. As with relative species richness, neither the height of the former dam nor the time since dam removal appears to have a clear relationship to relative abundance of non-native plants.

A number of variables likely influence non-native species richness and relative abundance in former reservoirs and are relevant to managing reservoir plant colonization. These variables include landform type, sediment texture and chemistry, revegetation and weed control, and propagule pressure (e.g., local seed source) of non-native plants. Time since dam removal was expected to influence the relative richness or abundance of non-native species, as sites with high dominance of *P. arundinacea* or *Urtica dioica* may have the potential to arrest successional change (Lenhart, 2000; Orr and Stanley, 2006). However, consistent with Orr and Stanley (2006) and Auble et al. (2007), we found no clear evidence (Figure 4) of non-native species richness and abundance increasing with time since removal.

Formerly inundated landforms typically include slopes that were uplands or valley walls prior to dam removal, and various surfaces composed of alluvial sediments (Shafroth et al., 2002). The height of the dam and distribution of trapped sediment influence the types and relative abundance of different landforms, including high terraces in some cases. With respect to non-native species, sites closer to the surrounding uplands could receive seeds of non-native taxa that are growing nearby, which can vary depending on adjacent land uses. The initially bare, open nature of all surfaces in former reservoirs would be expected to support weedy, pioneer species, some of which are likely to be non-native. Schuster (2015) examined native and non-native species richness and cover on three landforms (valley wall, terrace, riparian) in two former reservoirs. His results suggest that there may have been slightly higher relative non-native species richness on terraces than valley walls and riparian areas, but little difference in relative cover of non-native species between landforms.

The texture and chemistry (e.g., nutrient status) of sediments can influence the composition and abundance of colonizing vegetation. Some non-native invasive plants perform better in high nutrient soils in various ecosystems (Dukes and Mooney, 1999), and sediments in former reservoirs can have relatively high nutrient levels (Stanley and Doyle, 2002). There is some evidence that exotic species richness and cover may be related to concentrations of different nutrients (e.g., N, P, K) in former reservoirs on the Elwha River (Werner, 2014; Schuster, 2015). However, native species richness and cover are also influenced by nutrient status, and teasing apart the relative differences is challenging. Lenhart (2000) suggested that high nutrient levels in fine sediments of small dam removals in Wisconsin may have contributed to success of invasive species. Finer sediments in the former Elwha reservoirs tend to have higher nutrient levels and water-holding capacity than fluvial sediments outside of the reservoir, and are associated with greater plant cover, growth rates, and species richness (Calimpong, 2013; Schuster, 2015).

Resource managers frequently implement actions, such as controlling non-native plants and/or planting native species immediately following decommissioning and at regular intervals afterwards, in efforts to reduce non-native plant dominance, sometimes as a requirement of the dam removal permit or plan (e.g., U.S. Army Corps of Engineers, 2012). The extent to which these activities have occurred is clear in some (e.g., Orr and Koenig, 2006; Lenhart, 2000; Chenoweth, 2013), but not all of the studies we examined. A few studies report findings of revegetation efforts that allow some inference into the effects on non-native species richness and abundance. Orr and Koenig (2006) reported findings from two small dam removal sites where significant planting of native taxa occurred and where monitoring occurred one and four years following removal. Although planting initially appeared to be having the desired effects of increasing the ratio of native to non-native species and excluding some particularly aggressive non-native taxa, after four years, the proportion of non-native taxa increased at both sites and the authors concluded that the planting efforts were largely unsuccessful (Orr and Koenig, 2006). In the two former reservoirs on the Elwha River, the frequency of non-native plants was slightly lower (by 3.9–5%) in planted areas than in unplanted areas, except on valley walls in one of the former reservoirs, where the frequency of non-native plants was only 0.4% higher in planted, vs. unplanted, areas (J. Chenoweth, personal communication). At the former Milltown Dam site on the Clark Fork River in Montana, seeded and planted species comprised approximately 25% of total cover and about 15% of total species richness (Sacry et al., 2013). In most of these studies, it is difficult to directly assess the extent to which planting native vegetation inhibits non-native species richness; however, it is reasonable to infer that the space occupied by these plants is not available for non-natives and therefore plantings likely limit abundance.

Finally, the abundance and proximity of propagules (e.g., seeds, or existing stands of spreading plants) is a key factor influencing the extent of non-native plant colonization and spread. These propagule sources may vary depending on surrounding land use and interact with time since removal, landform, and the direct management actions discussed above. Through surveys of non-native plant populations in areas upstream and surrounding a dam removal site, managers can evaluate potential non-native plant propagule pressure prior to dam removal to help assess the likelihood of significant invasion (Woodward et al., 2011).

7) Expansion of non-native fish

Characterizing the concern

By modifying riverine conditions, the construction of dams can facilitate the introduction and spread of invasive fish. Many of these introductions have been the result of deliberate management decisions, aimed at creating productive fisheries either in the reservoir itself (Bednarek, 2001; Havel et al., 2005; Rahel and Olden, 2008; Johnson et al., 2008) or in the tailwaters below the dam (Pejchar and Warner, 2001). For example, at Glen Canyon Dam on the Colorado River, Arizona, warm water sport fish have been introduced in the reservoir above the dam and rainbow trout have been introduced to the cold tailwater below (Schmidt et al., 1998). Fish introductions that were not planned by fisheries managers have also occurred, for example as a result of fish releases by self-motivated individuals (Lintermans, 2004). To succeed, however, these introduced fish populations must be predisposed for survival and reproduction in the newly created habitats and conditions afforded by the dams and their operations.

Whereas some dams may facilitate the introduction of non-native species, other dams are serving, purposely or incidentally, to block access of undesired fish species to upstream waters (McLaughlin et al., 2013). Consequently, this raises the concern that the removal of some dams may unintentionally extend the range of invasive species and increase their effects on native species and ecosystems (Hart et al., 2002; Jackson and Pringle, 2010). In some cases barriers have been built specifically to protect native species and ecosystems from potentially harmful invaders, such as those to prevent encroachment by non-native trout species that represent threats for displacement, introgression, or hybridization (Novinger and Rahel, 2003; Fausch et al., 2009). In the Great Lakes region of the U.S., numerous small dams were in place before introduction of the sea lamprey (*Petromyzon marinus)* and serve to suppress their spread (Lavis et al., 2003).Here we ask, based on available case studies: what evidence exists that dam removals have facilitated the spread of invasive fish species, and have these new invasions had undesirable impacts on aquatic communities and ecosystems?

Approach

We queried the Bellmore et al. (2015) database to identify dam removal case studies that explicitly stated a concern for introduced or undesired fish species, and/or for those studies that indicated an attempt to document the spread of non-natives and their impacts on native fish communities. We searched the database using the following keywords: “invasive”, “non-native”, “exotic”, “introduced” and “introduction.” While there were many published works that expressed concern that undesirable introduced fish species would invade the newly opened habitat because of dam removal, few studies actually tracked and documented this response after a dam was removed. Four case studies published in the literature (a total of six dam removals in four watersheds in Wisconsin, South Carolina, and Ohio; Table 2) were combined with three unpublished dam removal case studies in the Pacific Northwest (Table A1) where concerns about the spread of invasive fish exist, but are not yet expressed in published literature. We do not attempt a quantitative analysis with the limited number of case studies, but instead review the responses observed across these case studies and broadly discuss the potential controls on the spread and impact of invasive fishes following dam removal.

Key findings

Of the four case studies found in the published literature, three of these confirmed that invasive fish spread upstream of the former dam site (Gottgens, 2009; Kornis et al., 2014; Marion, 2014), but the fourth did not observe an upstream invasion (Stanley et al., 2007). Marion (2014) found non-native Alabama spotted bass (*Micropterus henshalli)*, which can hybridize and compete with native Redeye bass (*M. coosae)*, upstream of the removed Woodside dams on Twelvemile Creek in South Carolina. However, the magnitude of impact, if any, of the spotted bass invasion was not known. In the case of the removal of Big Spring Dam from Big Spring Creek in Wisconsin, non-native white sucker (*Catostomus commersonii)* and Yellow Perch (*Perca flavescens)* quickly invaded upstream and became a major portion of the fish population, while the native Mottled Sculpin (*Cottus bairdii)* population decreased by more than half (Kornis et al., 2014). In the Ottawa River in Ohio, the removal of Secor Dam coincided with the spread of invasive Round Goby (*Neogobius melanostomus)*, a potentially harmful and difficult-to-contain invasive fish (Corkum et al., 2004). Within two years after this dam removal, Round Goby composed 9% of the fish population at sites upstream of the former dam, though the dam may not have been an effective barrier because the bulkhead ports were opened (Jim Evans, personal communication).

Of the three unpublished case studies in the Pacific Northwest, all are located in the Columbia River basin. Condit Dam and Powerdale Dams were removed from downstream reaches near their junctions with the Columbia River (5 rkm and 7 rkm upstream from the Columbia, respectively). Non-native fish thrive in the Columbia River (Sanderson et al., 2009), including Smallmouth Bass (*M. dolomieu)* and Walleye (*Sander vitreus*) which prey on native salmon and steelhead (anadromous form of Rainbow Trout (*Oncorhynchus mykiss*) and the introduced American Shad (*Alosa sapidissima)*(Petersen et al., 2003). So far, however, none of these introduced species have been observed by moderate to intensive monitoring efforts above the former Condit (Allen and Connolly, 2011) or Powerdale (Rob Regan, ODFW, personal communication) dam sites. The third Pacific Northwest case study involves a removal site well upstream (19 rkm) of the main stem Columbia River (Hemlock Dam on Trout Creek), where it is considered unlikely that Smallmouth Bass, Walleye, or American Shad could reach because of various falls and cascades. Without a barrier, however, Trout Creek is now vulnerable to settlement by non-native spring Chinook Salmon (*O. tshawytscha*), several thousand of which pass the mouth of Trout Creek annually as they home towards their natal fish hatchery 11 rkm upstream. The concern is that these non-native Chinook Salmon may establish a population in Trout Creek and compete with native steelhead, which are the subject of a long-term and expensive restoration effort (Jezorek and Connolly, 2015). To date, spring Chinook Salmon have not established a population in Trout Creek as evident from rigorous annual smolt trapping (Buehrens et al., 2014) and electrofishing (Jezorek and Connolly, 2014). However, a few individual adult Chinook are likely to swim up and beyond the old dam site as suggested by the occasional adult Chinook that were caught in the trap within the former fish ladder of the old dam.

There are two potential explanations for the lack of observed invasions by fish in the case studies we reviewed. First, the duration of post-removal monitoring, where it has occurred at all, may be too short. Biological communities may require several years to stabilize following dam removal (Kornis et al., 2014; but see Tullos et al., 2014), with the response time for fish being longer than that of other biota such as macroinvertebrates (Maloney et al., 2008). While short-term shifts in fish assemblages may be evident, long-term shifts may continue to develop over time (Poulos et al., 2014). Second, the newly opened habitat may be poorly suited to invasive fish. Environmental and biological factors interact in unique combinations across the geographical extremes of species distribution, which create various strengths of resistance to fish invasion (Fausch, 2008). Frequency of disturbance (Leprieur et al., 2006), water temperature, and habitat structure (Jones et al., 2003) can constrain the ability of fish to establish a population in the new habitat. This resistance, however, is subject to change as the environmental (e.g., climate change, Rahel and Olden, 2008) and biological (e.g., food web changes, Jackson et al., 2001) conditions themselves change through time. These observations suggest that the mere presence or proximity of an introduced fish species does not predispose its spread and establishment in newly available habitat. Assessments of dam removal risks will need to examine the environmental (e.g., temperature and flow regimes) and ecological (e.g., food resources, fish assemblage) characteristics of habitats in relation to the biology and life history of potential invaders to evaluate the potential for invasive species to spread. Moreover, these risks will have to be weighed against the potential for the re-establishment of desired native species.

Discussion

The CMCs addressed in this study directly impact the practice of dam removal. These concerns frequently affect project advancement because conversations among dam owners, project managers, and the public are often colored by past case studies that may have been outliers with unique conditions, misinformation, and/or incomplete analyses. Our investigation represents a first attempt to systematically review the available information for these CMCs. This information highlights the issues and misconceptions associated with each CMC, which can be used to guide more focused and transparent pre-removal evaluations. Our review shows that there is substantial knowledge derived from individual dam removals, but also illustrates how it is difficult to identify definitive, generalizable findings for a given CMC.

<u>CMCs: What do we know?</u>

A common thread among CMCs is an incomplete understanding of the physical and/or biological process controls that influence their occurrence at a site. As a result, managers and particularly the public tend to assume that the negative consequences of CMCs will occur at all sites even when their occurrence is unlikely. In our experience, incomplete understanding operates at two levels. First, there are cases where the highly-specialized expertise necessary to correctly assess a given CMC is simply not represented on a project team, though a general understanding of the relevant processes does exist among specialists. Second, even where relevant expertise is available, scientific understanding of controlling processes may be incomplete. Dam-removal science is frequently not readily available to project teams, and studies are not typically framed in reference to management concerns. Despite the many challenges to analyzing CMCs as discussed below, our review allows us to improve understanding at both levels by identifying the specific biophysical phenomena associated with each of the CMCs. We also attempt to identify site conditions that suggest there will be a management implication if a phenomenon associated with the CMC occurs. This is important because there will be no basis for management concern at a site if there is no impact people or regulators care or know about.

Challenges to analyzing CMCs

A number of challenges prevent generalizing about the occurrence of CMCs. First, study designs and methods are often inconsistent across sites and monitoring durations are insufficient to answer questions of interest. Second, projects are often focused on very specific objectives (e.g., FERC requirements) and thus may not generate enough data to say something comprehensive about physical and biological processes relevant to CMCs. Third, access to information about prior dam removals is often limited or hard to find. For example, studies reported in National Environmental Policy Act (NEPA) documents, consultant reports, and other grey literature are difficult to discover and access. Other issues like confidentiality can also impact data discovery, such as finding information about private wells for evaluating impacts to water infrastructure. For some of the more contentious dam removals, managers and dam owners may not be permitted to discuss details of project impacts due to pending litigation. Finally, despite working with the best available information, there may be biases in our information sources towards particular dam sizes, geographies, and management scenarios (Bellmore et al., 2015). Thus, future monitoring that focuses on hypotheses about biophysical processes relevant to dam removal CMCs (e.g., Table 3), conducted with proper designs, monitoring durations, and broad public dissemination, would improve understanding of CMC occurrence and drivers.

Eliminating surprises: conditions controlling CMC occurrence

The oft-stated claim that “every dam is different” (Poff and Hart, 2002) reflects the local variability and nuances between sites that contribute to uncertainty about whether a CMC will occur during dam removal. Even if the physical and/or biological phenomena underlying a given CMC are ubiquitous (e.g., reservoir drawdown, increased connectivity), it is the local infrastructure, politics, social values, and regulatory thresholds that dictate if those phenomena generate unacceptable impacts that require management. Indeed, the human dimensions of dam removal decision making are frequently more complex than the biophysical dimensions (Heinz Center, 2002; Johnson and Graber, 2002; Provencher et al., 2008). To facilitate an evaluation of the interacting human and biophysical dimensions for the CMCs we analyzed, we outline the biophysical processes that control each CMC occurrence and provide examples of site conditions that suggest management implications (Table 3). With this information, managers can assess whether a CMC should be investigated further at their project site or else proceed with confidence that the CMC is unlikely to be problematic.

We suggest a process through which managers can determine both the likelihood of the biophysical phenomena occurring and its intersection with a management implication. This evaluation can be based on a series of decision points based on linking the biophysical phenomena to management outcomes and management implications—i.e., there must be ecological or human uses at stake that people care about. First, project managers can ask if the biophysical process controls for their CMC of interest (Table 3) are likely to be operative at their particular site. For example, if the groundwater table is regionally near the surface and not substantially elevated by the reservoir, managers may proceed with confidence that wells will not be impacted by the reservoir drawdown. Second, project managers should investigate if there is an intersection of the biophysical phenomena with a human use of relevance to stakeholders. For example, is there water infrastructure proximal to the impoundment that will be affected by reservoir drawdown? Third, because CMCs can co-occur, we suggest that managers identify CMCs that have biophysical process controls in common that might indicate co-occurrence. For example, the grain size of stored sediment appears to be an important control on the degree and rate of reservoir erosion, excessive channel incision (headcut migration rate through the reservoir), elevated turbidity, and non-native plant colonization of the reservoirs. Broadly evaluating the biophysical controls shown in Table 3 to identify those that are operative at a site may lead to discovery of unanticipated CMCs. These three separate, but related, evaluations will help managers facilitate science-based discussion among stakeholders, avoid surprises and inform stakeholders of likely outcomes, and allow managers to develop plans for addressing CMCs in a timely manner.

Other relevant management concerns

We attempted to select the most frequent and uncertain dam removal CMCs, but we acknowledge that our analyses are incomplete. There are other management issues that may occur, some of which can be anticipated.

For example, contaminated sediments are an important management concern at some sites (Evans, 2015), most famously at Fort Edward Dam in 1973 (Stanley and Doyle, 2003) and more recently at Milltown Dam (USEPA, 2011b). Reservoir sediment contamination is most likely to occur in reservoirs with high proportions of fine sediment and watershed land use histories that suggest contaminant releases were possible, which is why the likelihood of contaminated sediments and degree of contamination are much higher in some areas (e.g., Major and Warner, 2008) than others. Managers need to identify if sediments are clean, and they face additional challenges including a) a lack of consistency regarding biologically and legally acceptable limits of concentration and duration of contaminants in the sediment released during dam removal, b) limited knowledge of the effectiveness of various management strategies for contaminated sediments (capping, phased drawdown, isolation), and c) a need for better characterization of contaminant risks based on the bioavailability of the contaminant (Evans, 2015).

Disruption of aquatic communities downstream of dam removals is another potentially important management concern. In locations where native freshwater mussels occur, there is typically a concern that increased sediment loads will bury and suffocate mussel beds (Stanley and Doyle, 2003). This concern may apply to other aquatic invertebrates (but see Tullos et al., 2014), but the concern is high for mussels because they are long-lived species with long generation times and limited mobility, such that it may take decades for populations to re-establish after a mass mortality event (Box and Mossa, 1999; Strayer et al., 2004). Other example management concerns include stranding of biota in the reservoir as drawdown occurs, destruction of tribal and cultural resources as the reservoir is dewatered followed by a sediment pulse moving downstream (USBR & CDFG, 2012a; NMFS, 2010), and impacts to water rights (USBR & CDFG, 2012b).

Conclusions

Data for the seven CMCs we investigated are not sufficient to support broadly applicable conclusions about the probability of their occurrence at dam removal sites. Most CMCs have few case studies that are limited to the geography of where dams have been removed, and the study durations are typically short. Nonetheless, the CMC occurrence data we have, combined with established knowledge of relevant processes, reveal important biophysical phenomena that act as controls for the occurrence of CMCs. We propose that managers evaluate CMCs in the context of risk, whereby both the likelihood of the biophysical phenomena contributing to a concern occurring at a site and its intersection with a management implication are evaluated. Knowledge of the biophysical phenomena controlling CMCs enables managers to better determine if further analyses are warranted to assess CMC risk and provide science-based information to stakeholders. Increased knowledge of these biophysical controls will be gained with ongoing, and likely expanded, scientific dam-removal studies.

In addition to assessing CMC risk, it is important to acknowledge and communicate that long-term benefits of dam removal may be accompanied by tradeoffs inherent to dam-removal and to unrelated basin-scale activities (Stanley and Doyle 2003). In most cases, studies indicate that negative impacts associated with dam removal are transient (e.g., Tullos et al., 2014). In addition, even if managers determine that their site is at risk, those potential negative impacts should be weighed against the potential benefits of dam removal. By articulating project goals, developing metrics for evaluating project success, and identifying and monitoring triggers for adaptively managing to minimize negative impacts (Peters et al., 2014), the tradeoffs with dam removal can be effectively managed and communicated. Furthermore, ecosystem disturbances caused by dam removal, such as elevated turbidity, should be contextualized by comparison to natural variability and/or other human disturbances.

Dam removal practitioners thus have the challenge of estimating CMC occurrence at their sites, evaluating the potential risks in light of tradeoffs and other project context, and communicating this information to project stakeholders in a manner that allows communities to make informed decisions. Scientists also have a challenge to advance practice-relevant, dam-removal science to further support practitioner efforts. Our analysis and findings provide a means for managers to assess some of the common management concerns of dam removal, and highlight the need for additional dam-removal science to inform practice.

Supporting Information

Additional supporting information may be found online under the Supporting Information tab for this article: A table providing the names, locations, and some characteristics of the case studies analyzed for each CMC.

Acknowledgments

This paper is derived from discussions and analysis efforts conducted by the working group on *Dam removal: Synthesis of ecological and physical responses* of the U.S. Geological Survey's John Wesley Powell Center for Analysis and Synthesis. In addition, we very gratefully acknowledge Jeff Duda, Amy East, Laura Craig, Melissa Foley, Jim Evans, Chris Magirl for their reviews of an earlier version of this manuscript. We appreciate assistance with graphics from Cindy Gray and Joel Murray at the Bureau of Reclamation.

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Table 1. Turbidity data, state criteria, and occurrence characteristics for seven dam removal sites. Formazin Nephelometric Units (FNU) are similar to Nephelometric Turbidity Units (NTU) to the extent that both measure scattered light at 90 degrees from the incident light beam. However, they are measured using different wavelengths of light to comply with different regulatory standards (ISO 7027 and EPA method 180.1, respectively). For more information see http://or.water.usgs.gov/grapher/fnu.html.

Dam(s), river, state	**USGS gage number**		**Turbidity data**		**State threshold (ST)**		**T_{max}[1] (%)**				**T_{dur}[2] (%)**			
	above	*below*	*unit*	*freq.*[3]	*criteria*[4]	*ST*[5,6]	*above*		*below*		*above*		*below*	
							pre	post	pre	post	pre	post	pre	post
Veazie/Great Works, Penobscot, ME		01036390	FNU	60 m	none	2.5			1160	1240			9.7	10.2
Simkins, Patapsco, MD	01589000	01589025	NTU	15 m	150 NTU	150	15	1047	51	1247	0	1.2	0	1.7
Savage Rapids, Rogue, OR	14361050	14361180	FNU	15 m	nat. + 10%	6.4	188	1724	1207	3292	0.3	12.3	10.9	14.5
Marmot, Sandy, OR	14136500	14137002	FNU	15,30 m	nat.+ 10%	14.3	4336	7483	3497	7483	47.3	8.2	31.5	11.9
Chiloquin, Sprague, OR[7]	11501000	11502500	FNU	15 m	nat. + 10%	17.6	477	341	295	290	22.7	4.4	5.9	1.4
Good Hope Mill, Conodoguinet, PA	01570064	01570078	NTU	15 m	None	2.9	879	1272	3793	4828	7.8	10.6	94.1	69.2
Elwha/Glines Canyon, Elwha, WA[8]	12044900	12046260	FNU	15 m	nat. + 5 NTU[9]	22	205	>6091	532	>7545	6.8	7.4	14.3	96.6

[1] T_{max} = (maximum peak value/ST) * 100

[2] T_{dur} = (duration of ST exceedance/monitoring duration) * 100

[3] Sampling frequency in minutes. The Marmot site had different sampling frequencies for the sensors above and below the dam.

[4] See http://water.epa.gov/scitech/swguidance/standards/wqslibrary/index.cfm. Accessed July 30, 2014.

[5] Based on upstream record for all sites except Penobscot, which only has a downstream record with a long pre-removal monitoring period

[6] "Natural" conditions are estimated by the 90th percentile value; if no state-wide criteria, 90th percentile is used

[7] Downstream gage is on Williamson River below Sprague confluence

[8] Turbidity instruments here have operational ranges up to 1,500 FNU (Fig. 3c); values above 1,200 FNU are considered unreliable (Curran et al., 2014)

[9] When natural is 50 NTU or less (USGS reports FNU at Elwha project gages)

Table 2. Review of studies on fish invasion with dam removal

River, State	Non-native fish of concern: Downstream	Non-native fish of concern: Upstream	Desired species of concerns	Post-removal response	Degree of monitoring	Source
Boulder Cr., WI	Brown trout		Brook Trout	Limited to no effect; Brown Trout not observed to invade two years after removal.	High	Stanley et al. 2007
Twelvemile Cr., SC	Alabama Spotted Bass, Flathead Catfish	Flathead Catfish	Redeye Bass	Alabama Spotted Bass found upstream; unknown if hybridized with Redeye Bass, Flathead Catfish increased.	High	Marion 2014

Big Spring Cr., WI	Largemouth Bass, Yellow Perch, Bluegill, White Sucker		Mottled Sculpin and other native species	Yellow Perch and White Sucker invaded and established populations upstream; Downstream dam still in place provided source for non-native fish.	High	Kornis et al. 2014
Ottawa R., OH	Round Goby		Native fish assemblage in general	Round Goby fist observed in mid-2008; Made up 9% of total catch upstream and 21% downstream by end of 2009.	High	Gottgens 2009
White Salmon R., WA	Walleye, Smallmouth Bass, American Shad		Chinook, Steelhead, Bull Trout, others	No invaders observed.	Low	Allen and Connolly 2011; Allen, B. USGS,, Pers. Comm.
Trout Cr., WA	Spring Chinook		Steelhead	Juvenile Spring Chinook have not been observed.	High	Jezorek and Connolly 2014; Buehrens, T., WDFW,

						Pers. Comm.
Hood R., OR	Walleye, Smallmouth Bass, American Shad		Chinook, Steelhead, Bull Trout, others	No invaders observed.	High	Reagan, R.., ODFW, Pers. Comm.

Table 3. Conditions controlling CMC occurrence. "N/A" for the rate and degree of reservoir erosion reflects the different approach taken for this CMC, where we summarized a number of synthesis papers rather than directly evaluated individual case studies. V* is the ratio of the volume of stored reservoir sediment to the river's annual sediment load, with lower or higher V* values associated with smaller or greater downstream impacts from aggradation, respectively.

CMC	Case studies	Biophysical process controls	Example site conditions suggesting management implications
Degree and rate of reservoir erosion	N/A	• high % of stored fine grained sediments[a] • average sediment deposit width/channel width > ~2.5 • phased removal[b]	stakeholder values; fish passage needs or sensitive habitats
Excessive channel incision upstream of reservoir	38	• reach-scale incision downstream • high % of stored fine grained sediments[a] • phased removal[b] • coarse delta • ephemeral flow	In-stream infrastructure within reservoir deposit; infrastructure or property along reservoir margins at risk for bank erosion; fish passage needs or sensitive habitats
Downstream aggradation	6	• proximal to dam • antecedent channel has low slope/unconfined • high V*[b]	low-lying properties, transportation infrastructure; pump intakes; fish passage needs or sensitive

[a] This process is common across four CMCs.
[b] This process is common across two CMCs.

			habitats
Elevated turbidity	7	• high % of stored fine grained sediments[a] • high V*[b] • rapid reservoir drawdown	Sensitive aquatic organisms; human recreational uses; drinking water treatment intakes
Drawdown impacts on local water infrastructure	5	• large drop in water surface elevation • high degree of connectivity between the reservoir, river, and the groundwater • regionally deep groundwater table	wells or intakes in the reservoir vicinity
Non-native plant colonization of reservoirs	23	• proximity to non-native seed sources • high % of stored fine grained sediments[a] (with high nutrient content) • no planting or weed control	legal requirements for noxious weed and/or invasive species control; stakeholder values
Non-native fish	7	• abundance and proximity of non-native fish • availability of suitable habitat and temperatures for non-native species	state fisheries regulations or management plans; stakeholder values

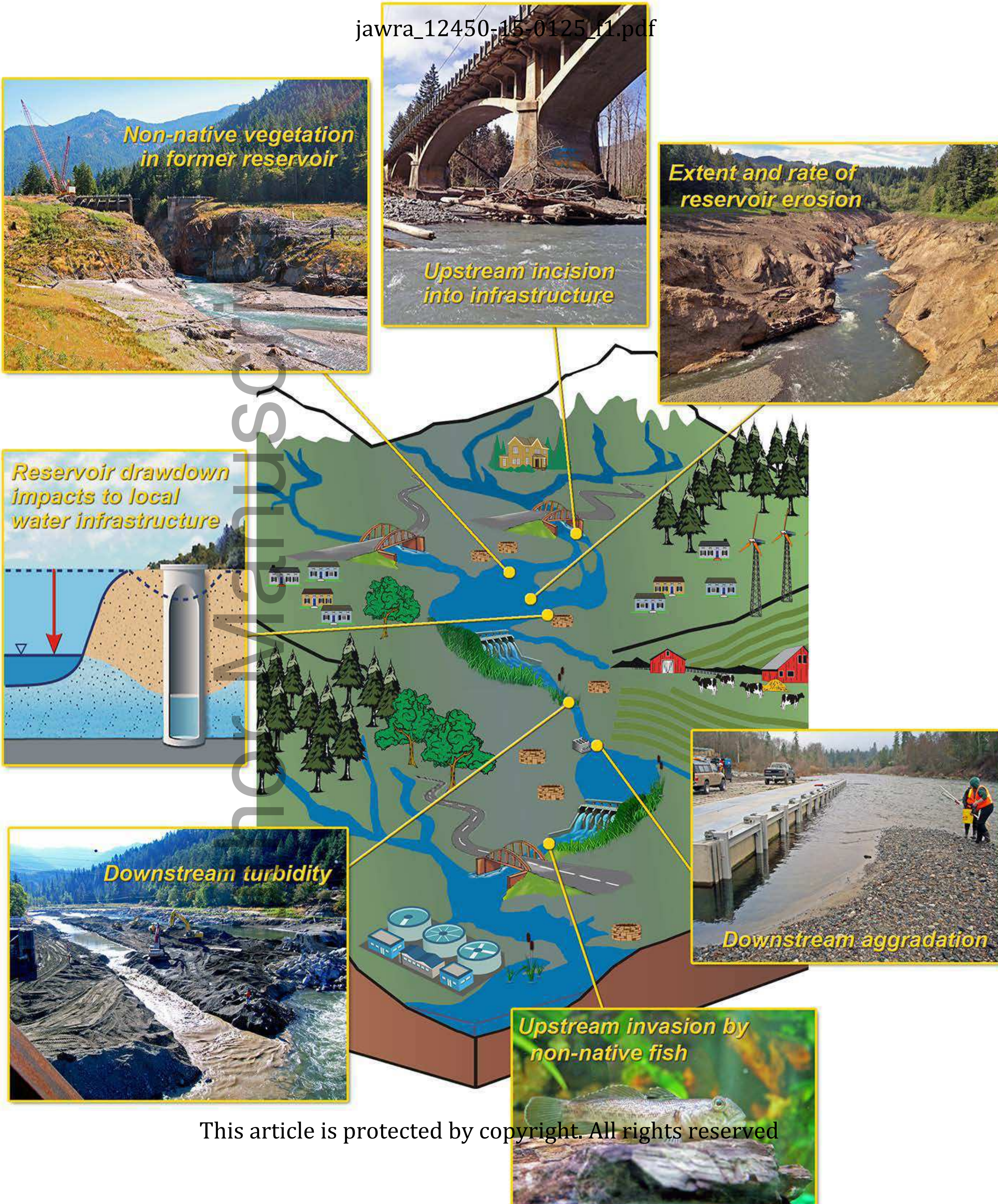
Figure Captions

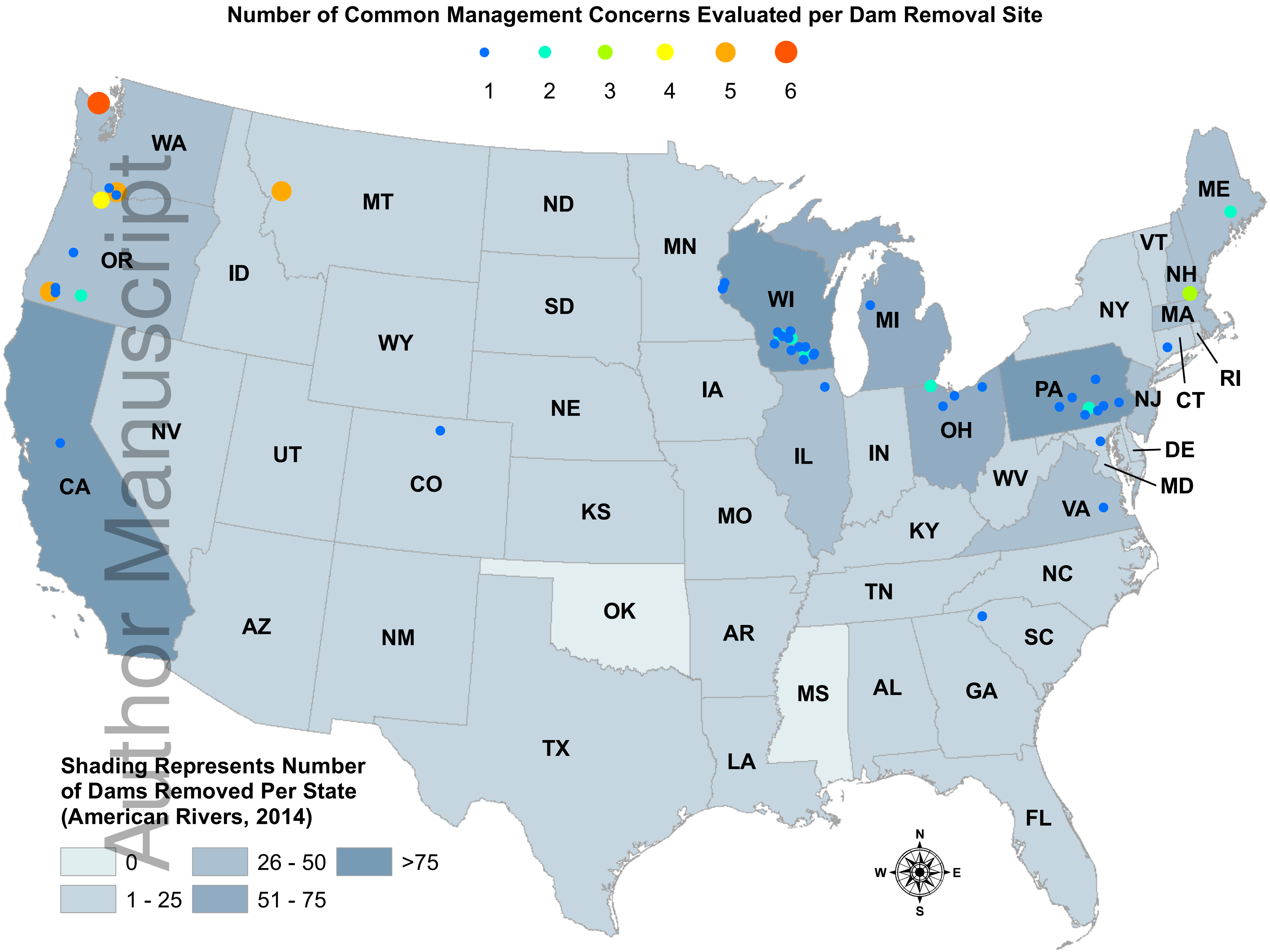
Figure 1. Illustration of CMCs across a catchment.

Figure 2. Distribution of case studies analyzed and numbers of dams removed, by state. In addition to sites shown on map, ten international sites were evaluated for a single CMC (excessive channel incision) including one site in Australia, one site in Taiwan, and eight sites in Canada.

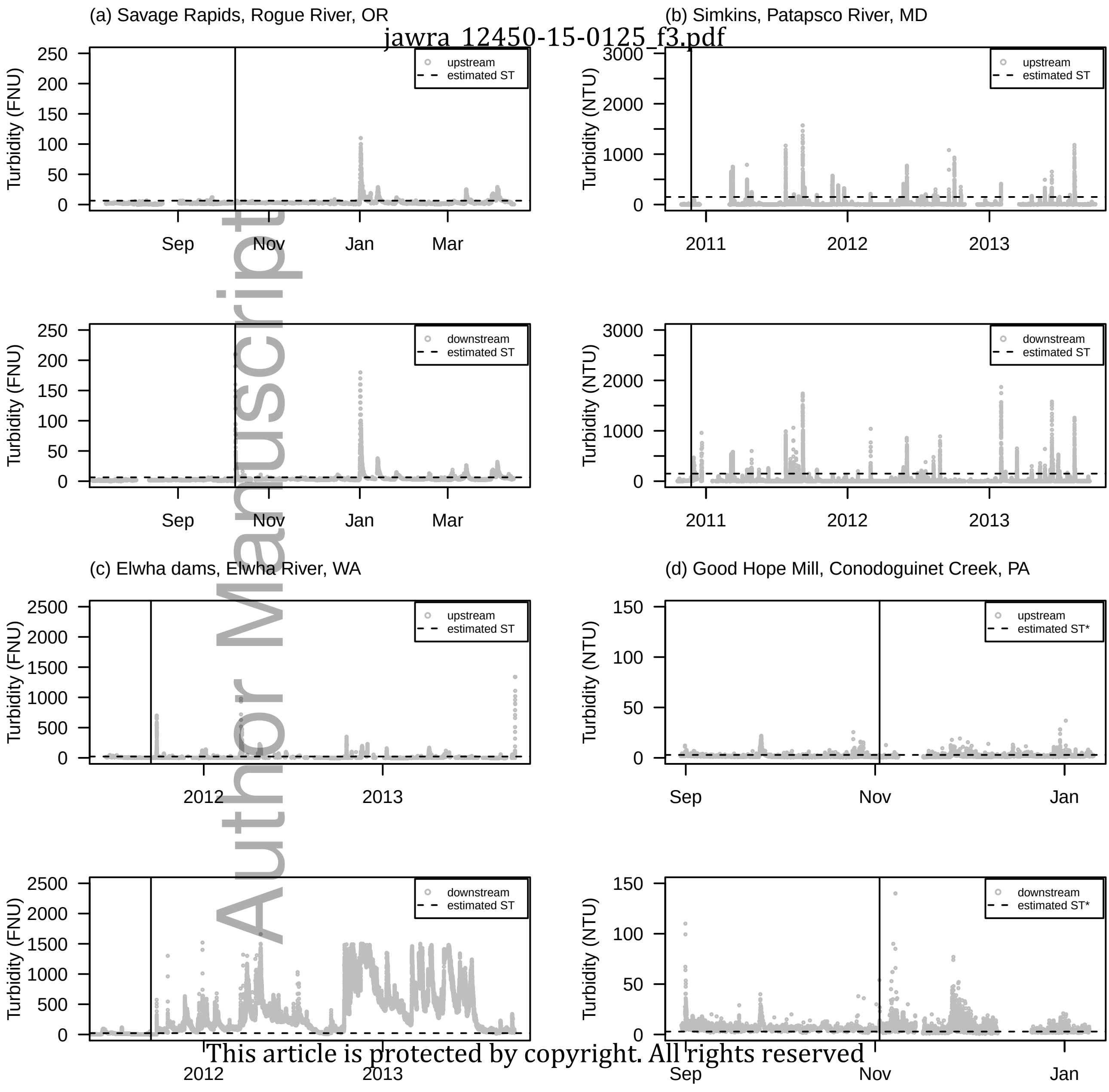
Figure 3. Paired upstream and downstream turbidity time series for four dam removal sites. The vertical line in each panel shows the dam removal start date. Only overlapping periods of record for site stations are plotted. Units and ordinate scales vary across sites. Resuls are from (a) Savage Rapids Dam, (b) Simkins Dam (c) Elwha River Dams, and (d) Good Hope Mill Dam. The downstream record pre- and post-removal may be affected by tributary inputs, bioturbation, and human uses (Chaplin et al., 2005). Formazin Nephelometric Units (FNU) are similar to Nephelometric Turbidity Units (NTU) to the extent that both measure scattered light at 90 degrees from the incident light beam. But they are measured using different wavelengths of light to comply with different regulatory standards (ISO 7027 and EPA method 180.1, respectively). For more information see http://or.water.usgs.gov/grapher/fnu.html.

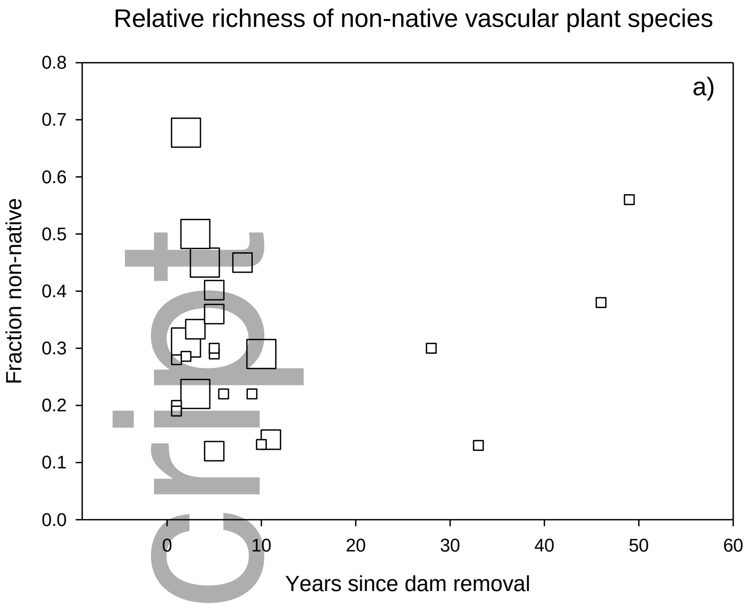
Figure 4. a) Relative species richness of non-native vascular plants in 25 former reservoir areas following dam removal. Values are the fraction of total species richness. b) Relative abundance of non-native vascular plants in 18 former reservoir areas following dam removal. Relative abundance values are the proportional contribution to frequency of occurrence or percent cover, depending on the study. Symbol sizes reflect dam heights: large symbols = dams > 10m tall; medium symbols = dams 4-10m; small symbols = dams < 4m.

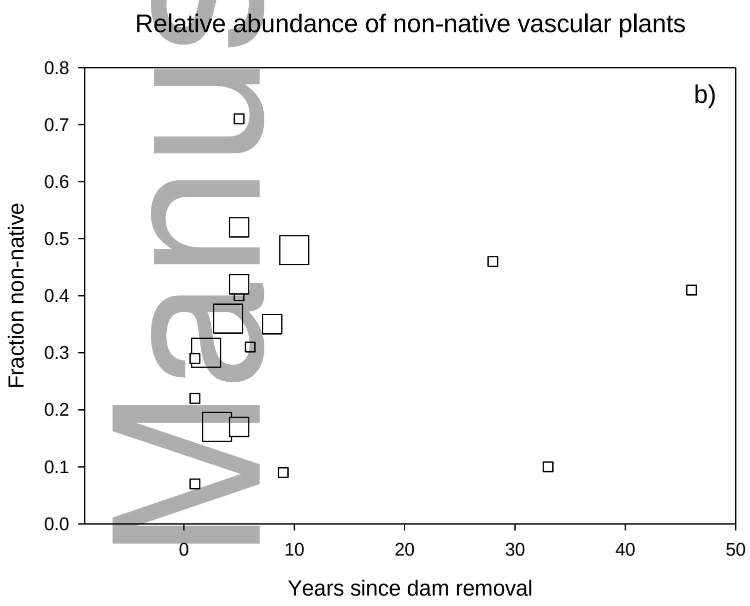

Non-native vegetation in former reservoir
Upstream incision into infrastructure
Extent and rate of reservoir erosion
Reservoir drawdown impacts to local water infrastructure
Downstream turbidity
Downstream aggradation
Upstream invasion by non-native fish



Number of Common Management Concerns Evaluated per Dam Removal Site
1
2
3
4
5
6
WA
OR
CA
ID
NV
MT
WY
UT
AZ
CO
NM
ND
SD
NE
KS
OK
TX
MN
IA
MO
AR
LA
WI
IL
MI
IN
OH
KY
TN
MS
AL
GA
FL
SC
NC
VA
WV
PA
NY
VT
NH
ME
MA
RI
CT
NJ
DE
MD
Shading Represents Number of Dams Removed Per State (American Rivers, 2014)
0
1 - 25
26 - 50
51 - 75
>75
N
W
E
S


(a) Savage Rapids, Rogue River, OR
(b) Simkins, Patapsco River, MD
(c) Elwha dams, Elwha River, WA
(d) Good Hope Mill, Conodoguinet Creek, PA
Turbidity (FNU)
Turbidity (NTU)
upstream
downstream
estimated ST
estimated ST*
Sep
Nov
Jan
Mar
2011
2012
2013
Author Manuscript


Relative richness of non-native vascular plant species
a)
Fraction non-native
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0
10
20
30
40
50
60
Years since dam removal


Relative abundance of non-native vascular plants
b)
Fraction non-native
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0
10
20
30
40
50
Years since dam removal