Making the case for Water Funds:

Evidence and gap assessment

The Nature Conservancy

Global Water

31 Jan 2018

Contents

Contributors

Robin Abell, Jonathan Higgins, Shiteng (Stan) Kang, Nathan Karres, Paulo Petry, Kari Vigerstol

Special thanks to colleagues from TNC and partner organizations who provided feedback to help shape the scope and content of the final report, including: Nigel Asquith, Deborah Bossio, Kate Brauman, Leah Bremer, Alejandro Calvache, Eddie Game, Gena Gammie, Kris Johnson, Timm Kroeger, Alejandro Lopez-Serrano, Juan Lozano, Yuta Masuda, Suzanne Ozment, Keaton Sandeman, Daniel Shemie, Priya Shyamsundar, Adrian Vogl, Cory Zyla.

Note to readers

This document represents a 'working paper' that will be updated and revised as we collect additional evidence and literature relevant to the key research questions. Additionally, we welcome feedback and contributions from readers. Please email Stan Kang (stan.kang@tnc.org) and Nathan Karres (nkarres@tnc.org) with your comments.

Summary

Water fund programs from around the world are keen to build their projects with the best available science, and excited by the prospect of obtaining answers to key science questions related to source water protection. Program managers are generally in agreement about the highest priority science questions—or categories of questions—yet to be adequately assessed, with the greatest interest around the impacts of source water protection on downstream water flows (e.g. dry season flows and floods). Our initial review of the literature—focused on a small number of priority questions, revealed commonalities around a lack of empirical studies at meaningful temporal and spatial scales, and a lack of representativeness across ecosystem types and geographies. In addition, our review suggests that many documented impacts are generalizable only across a limited range of contexts.

The key guidance from our limited investigation is that **many of the proposed benefits resulting from source water protection activities—including those within the context of water fund type projects—cannot be assumed**. Caution is required in communications regarding proposed benefits in order to avoid misrepresenting the likelihood of potential benefits from water fund projects. This review further **suggests the importance of data-driven conservation planning and the need for robust project monitoring and project documentation** in order to augment existing scientific evidence gaps. Adhering to this guidance will help ensure the continued credibility of Water Fund projects and increase efficiency and efficacy of future efforts.

Introduction

Interest in water funds as a source water protection mechanism is growing rapidly. The Nature Conservancy and partners have established 30 operating water funds around the world and some 30 more are in development. The core objective of water funds is the improvement or maintenance of water quality and/or water quantity conditions using land management activities to support water security for people and biodiversity. Some additional objectives relate to climate change mitigation and adaptation, and human health and well-being.

To date, empirical evidence for project-level impacts resulting from on-the ground interventions in many of these benefit areas lacks adequate robustness required to substantiate predicted outcomes or solicit adequate stakeholder support. This is due to a combination of project-specific factors, including: constraints on available resources and expertise; temporal lag for certain impacts at certain scales; insufficient implementation of activities at necessary spatial extents; and scientific knowledge gaps related to source water protection activities and expected results. Without stronger evidence, it will be challenging to secure sustainable financing for existing water funds and expand the adoption of nature-based solutions globally across additional source watersheds.

Key first steps in building the evidence base are to ascertain the current state of empirical evidence, and to identify essential information gaps. The project described here represents an initial effort to rapidly document existing evidence and identify major gaps in evidence, and then lay out a path forward for further, more detailed investigations of the literature and for priority monitoring needs.

Methodology

Our approach consisted of two main steps: (1) Identify and prioritize key science questions, and; (2) conduct rapid literature reviews related to the highest priority questions and summarize the findings.

To identify and prioritize key science questions, a core team drafted an initial set of questions organized into benefit categories (water quality, water quantity, biodiversity, human health and well-being, climate change mitigation and adaptation) and several thematic categories (e.g. modeling). The team then invited approximately thirty experts within and outside TNC to provide feedback to the list, add their own questions, and identify their highest priority questions. These experts included project-level professionals affiliated with specific water funds as well as scientists who have provided guidance to water funds over time.

Based on the feedback we received, we compiled a final list of questions and identified the highest priority questions in each category. We restricted our full list to biophysical and social science questions related to source water protection activities, and excluded questions related to water funds as a source water protection mechanism (e.g. around the benefits that upstream communities accrue via participation incentives, or around factors affecting landowner participation). We then selected four representative questions for a 'deep dive' exploration.

Because a comprehensive systematic literature review of any one question could take as long as several years given our capacity (Collaboration for Environmental Evidence 2013), we undertook a rapid assessment approach to provide an initial starting point for identifying significant and common information gaps. For each question, we developed a set of search terms to query the Web of Science database, which compiles articles from over 33,000 journals. We restricted our search to articles published since 1990 and, for searches that generated large numbers of results, we looked first at systematic reviews and meta-analysis reports. Because Web of Science does not typically capture 'gray' literature, where relevant we also searched AGRICOLA (the U.S. Department of Agriculture's National Agricultural Library database) and, to a lesser extent, the worldwide web. We also screened bibliographies of articles for relevant studies that were not included in initial search results but met the inclusion criteria.

Using a standardized template, we briefly summarized our findings from each question's literature review. We focused on characterizing evidence identifying key information gaps, considering subject matter gaps as well as gaps related to ecosystem type and geographic representation. We also synthesized key findings from each review, and suggested areas for future investigation. Given that many findings from individual studies are highly context-specific, we did not attempt to summarize all study results, but instead focused on big-picture conclusions where those could be drawn. Where we did highlight the results of individual studies, it was typically either to underscore the diversity of findings or to exhibit examples of supporting evidence. Additionally, we did not attempt to review or summarize the methodologies and tools applied in data collection and analysis across studies.

We solicited reviews of each summary from a small number of TNC and external experts, and incorporated that feedback.

Results

Key representative questions

Below is the list of four representative questions that were selected for literature review. The full set of questions from which these were drawn can be found in the *Appendix*.

1) To what extent can different source water protection activities contribute to aquatic ecosystem and biodiversity benefits?
2) To what extent can common source water protection activities affect baseflows, groundwater quantity, and flooding through changes in infiltration and runoff?
3) To what extent can specific agricultural best management practices result in crop productivity gains or cost reductions?
4) What are the relative expected magnitudes of change to water quality and quantity from climate change as compared to land use/land cover change?

Reviews in brief

Highlights of the four rapid reviews are found below. Detailed review commentary and literature citations are in the *Appendix* . Each review has a slightly different structure due to the distinct nature of each question. The cited literature focuses on key papers, reviews, and illustrative studies; the references reviewed and listed are generally extensive but should not be considered comprehensive. Access to the literature libraries assembled as part of each review is available upon request.

Aquatic biodiversity

To what extent can different source water protection activities contribute to aquatic ecosystem and biodiversity benefits?

Summary

The ability to generalize the types and degrees of beneficial freshwater biodiversity responses from source watershed actions that address surface sediment and nutrient loading is not supported by the literature. Studies indicate conflicting outcomes within and across taxa, and there is limited observation of biotic responses to actions addressing only a single type or source of ecosystem stress.

Key findings

There is a wealth of evidence linking excess sediments and nutrients (phosphorus and nitrogen) to negative impacts on ecological conditions and native biodiversity composition in freshwater systems (Allan, 2004; Carpenter et al., 1998; Castro & Reckendorf, 1995; Correll, 1998; Smith, 2003). There is little doubt that the reduction of unnaturally high levels of sediments and/or nutrients in freshwater systems can confer benefits to aquatic ecosystems and native aquatic species. However, rarely will these nonpoint source pollutants be the only stresses on aquatic systems and, in many cases, they may not be the primary stressors (as many aquatic species are concurrently impacted by water abstraction, instream barriers, invasive species, overexploitation, and direct habitat alteration, as well as point source pollution). Nonpoint pollution reduction via source water protection may result in little or no easily measurable impact on aquatic biodiversity, given the predominance of other threats and land use legacies (e.g. Harding, Benfield, Bolstad, Helfman, & Jones, 1998; Maloney et al., 2008; Teels, Rewa, & Myers, 2006). Some source water protection activities aimed at reducing nonpoint source pollution may even have unintended consequences for freshwater systems and species, such as increasing no-till agriculture to reduce soil erosion and sediment loading into freshwater ecosystems but with the increased use of pesticides (Elias, Wang, & Jacinthe, 2018).

Our review finds a dearth of studies measuring the direct impact of source water protection activities on aquatic ecosystems and biodiversity. Further, there is often a lack of consensus among existing studies as to the direction of impact. This result aligns with a recent synthesis of the effectiveness of a variety of conservation interventions on a number of biodiversity elements, which finds an alarmingly small number of available empirical studies, and frequent contradictions among their conclusions (Sutherland, Dicks, Ockendon, & Smith, 2017). For instance, looking at amphibian conservation, that assessment concluded that there was conflicting evidence across conservation interventions including agricultural management and pollution control, unknown effectiveness on amphibian populations resulting from the management of grazing regimes, and no evidence at all related to reduced tillage farming (Smith, Meredith, & Sutherland, 2017).

Conflicting findings, especially when the overall number of studies is small, may be a function of different taxonomic groups, ecological contexts, scales, intervention practices, measurements, seasons, or any number of other parameters. Importantly, lack of published studies should not be interpreted as a lack of on-the-ground impact, as projects often suffer from poor written documentation (Palmer, Allan, Meyer, & Bernhardt, 2007). For the most part, we are left to infer potential impacts of any given tactic based on known relationships (e.g. there is evidence that rural road management can reduce sediment inputs, and reduced sediment inputs should benefit native freshwater species). Riparian zone protection or restoration is an exception, with a greater number of studies linking vegetated (primarily forested) riparian buffer zones with maintained or improved ecological conditions of adjacent and downstream aquatic systems. However, even the specifics of how riparian zones can best be managed to provide species and system benefits are not entirely resolved (e.g. Richardson, 2008).

An important but unsurprising conclusion that we can draw from this review is that, while source water protection activities *can contribute* to aquatic biodiversity conservation, in many cases, such activities may be necessary but not sufficient to achieve conservation objectives on their own.

Source water protection programs may need to invest in comprehensive assessment of impacts to aquatic species and ecosystems of interest to understand how specific source water protection activities alone can contribute, what the expected response based on those actions alone would be, what additional interventions might be required to achieve conservation goals, and whether those interventions fall within the remit of the program.

Key gaps

- Studies that assess impacts from actions that address surface runoff as a sole type and source of impact, since most freshwater ecosystems and species have multiple and significant types and sources of impacts, precluding the ability to observe benefits that may be necessary but insufficient to result in measurable responses

- Studies of beneficial impacts to aquatic biodiversity from agricultural BMPs, agroforestry, and silvopasture outside of North America. Well-designed monitoring programs associated with existing and new source water protection programs in different geographies and environmental contexts could contribute to filling gaps in geographic representation and results from actions in different contexts and with different associated mixes of counterfactuals.

Summary table

	Land protect-ion	Re-veg-tation	Riparian restor-ation	Ag BMPs	Ranch-ing BMPs	Fire risk manage-ment	Wetland restor-ation and creation	Road manage-ment
Level of available scientific information	Low	Low	High	Med.	Med.	Low	Med.	Low
Degree of consensus in findings	Med.	Med.	High	High	Med.	Med.	Med.	Med.

Land management interventions and flow changes

To what extent can source water protection activities affect baseflow, groundwater quantity, and flooding through changes in infiltration and runoff?

Summary

Based on current evidence and in the absence of additional site-specific information, it is not possible to infer the direction or magnitude of changes to hydrologic flux resulting from source water protection activities. Benefits to baseflow or groundwater recharge are site-specific and cannot be generalized. Changes in baseflow, groundwater recharge, and flood mitigation have been measured in some instances, but neither the outcomes nor the significance to people are readily generalizable. In short, beneficial changes to flow-related outcomes resulting from land management changes cannot simply be presumed without additional evidence.

Key findings

On balance, the existing literature suggests that source water protection activities should not be assumed to generate definitive benefits in terms of increased total water yield, baseflow, groundwater recharge, and/or mitigated flood flows—though in some contexts such impacts have been measured and are statistically significant.

Our review finds a wealth of evidence measuring and discussing the impacts on hydrology from (temperate) forest-related activities, including forest protection, natural forest regrowth (passive revegetation), deforestation, reforestation, and afforestation. Nonetheless, evidence is skewed towards streamflow or water yield (as opposed to dry-season flows or groundwater recharge) and tree plantations (instead of native forest restoration).[1] In general, the surveyed literature provides evidence that these activities can alter the balance between soil conditions (infiltration) and forest water use (evapotranspiration), hence affecting water quantity fluxes. The magnitude and direction of hydrologic flux changes, however, is highly context-specific. A recent meta-analysis suggested reforestation generally results in a reduction in annual water yield due to evapotranspiration/infiltration ratios, a lack of consistent results for groundwater recharge, and extended dry-season baseflow. Our review of forest-related activities and water quantity impacts points to the following gaps: (1) a lack of empirical understanding of the interactions of climate and land-use change, and subsequent hydrologic impacts[2]; (2) a relative lack of evidence on long-term and large-scale relationships, particularly how hydrologic responses may change with forest composition and maturity; (3) unequal representation across forest types[3]; (4) a relative paucity of information in the tropics and (semi-) arid tropics; (5) relatively poor documentation of the relationship between natural reforestation or afforestation (not monoculture tree plantations) and soil infiltration; and (6) the continued need for more site-specific information on hydrologic function at varying spatial scales.

For non-forest related source water protection activities, evidence of water quantity impacts is relatively sparse. A small number of high quality studies documented the hydrologic roles of non-forest ecosystems including grassland, wetland, and riparian zones, implying the importance of protecting native habitats. On the other hand, water quantity impacts from restoring these ecosystems have been extensively discussed and potential benefits modeled, but published empirical studies are lacking. For other source water protection activities (e.g. agricultural BMPs, road management, or fire risk management), we found that some studies documented evidence that implementing certain practices such as conservation agriculture or forest road removal *could* improve infiltration and reduce runoff on-site. However, the impacts of these changes on baseflow or groundwater recharge remain mostly theoretical (empirical linkages need to be established).

Further, when statistically significant changes in baseflow, groundwater recharge, and flood mitigation have been documented in the literature, the magnitude of such impacts may not translate into changes that are large enough to be meaningful to society. It is also critical to note the *bases of inference* when assessing water quantity changes (i.e. significant change compared to what?) when assessing the evidence base. Another critical factor to note when assessing hydrologic impacts is the *trajectory of change*—impacts from certain source water protection activities (revegetation in particular) would be expected to vary depending on differences in baseline conditions and the composition and age of the revegetated landscape—particularly with forests.

[1] This lack of evidence is in part due to the increased complexity of measuring and attributing changes to sub-surface flows.

[2] In contrast, scientific understanding of the impacts of climate change and land use change separately is comparatively well studied.

[3] For instance, studies focused more on plantations compared to intact or secondary forests or reforested lands. Of studies on reforestation, evidence is skewed towards forestry and exotic species instead of native species (Filoso et al. 2017).

Given the breadth of this question, and the variability across published studies in terms of hydrologic measures, a more systematic review could examine whether there is greater consensus and generalizability across more specific interventions or biomes.

Key gaps

- For targeted land protection and revegetation activities, literature is skewed towards forests (where studies are biased towards forestry/plantations instead of reforestation using native species), and empirical evidence on the impacts from wetland, grassland, and riparian restoration is sparse.
- For other source water protection activities—agricultural BMPs (e.g., conservation agriculture, including tillage, cover crops/residue retention, and crop rotation, agroforestry, and buffer strips), ranching BMPs (e.g., grazing management, livestock management), forest fuel reduction, and road management—scientific evidence of watershed scale water quantity impacts is sparse.
- A large body of literature across activities (agricultural BMPs in particular) documents *changes in infiltration and/or runoff* at local scales. However, whether and how such changes in runoff or infiltration could accumulate and translate to observable or meaningful flow changes (groundwater recharge, baseflow, or flood reduction) is less clear. Lack of empirical evidence on the direct linkage between infiltration/runoff changes and watershed scale water quantity impacts remains an important gap.
- To demonstrate potential hydrologic benefits, there is a need to evaluate water quantity impacts for people and aquatic ecosystems and biodiversity through the lens of specific stakeholder groups. There is a paucity of information on the extent to which source water protection can deliver hydrologic changes that are meaningful to society.
- There is limited understanding of the impacts resulting from broader climate and land-use interactions relative to potential hydrologic services/benefits from source water protection activities.
- There is a lack of information on how different land use changes, including a mixture of SWP activities, may interact and influence changes in flow regime at the landscape or watershed scale. To make a stronger case for the hydrologic benefits of source water protection, empirical evidence is needed at larger scales and over longer timescales that represent the matrix of land uses and land cover commonly found in source water protection efforts.
- Understanding the time-scale at which impacts will occur across different objectives is also a key gap. For instance, as groundwater processes tend to be much slower than surface water, changes in land use/cover/management may take months, years, or even decades to propagate through groundwater flow systems to impact surface water bodies. The scientific understanding of such processes across extended timescales is limited.

Summary table

	Land protection	Revegetat-ion	Riparian restoration	Ag BMPs	Ranching BMPs	Fire risk manageme nt	Wetland restoration / creation	Road manage-ment
Level of available scientific informati on	High	High	Med. - low	Med. (on changes in runoff or infiltration)	Low	Low	Med.	Low
Degree of consensu s in findings	Mixed	Mixed	Mixed	Mixed	Med.	High (related to consensus in forest hydrology)	Mixed (too context specific)	High

Agricultural practices and crop yield

To what extent can specific agricultural best management practices result in crop yield gains or cost reductions?

Summary

Overall, there is limited agreement on direct benefits of agricultural BMPs in terms of crop yield gains to farmers across geographical regions and farming contexts. Variations are significant depending on site conditions and factors. There is no guarantee of direct benefits to farmers, and it takes time to realize benefits and/or avoid future losses, which may be difficult to appreciate. To improve farmer livelihood, and realize the full potential of Water Funds, there is a need to include additional practices targeting soil quality and farming system improvements.

Key findings

Agricultural best management practices (BMPs) that are commonly adopted or promoted as source water protection activities—such as crop rotation, cover crops, conservation tillage, contour farming, buffer strips, and nutrient management—have been encouraged to reduce erosion risks and improve soil and water conservation, among other benefits (OECD 2016; Liu et al. 2017). A suite of environmental benefits (e.g. improving soil structure, enhancing soil biological activity, nutrient cycling, soil water holding capacity, water infiltration) have been theorized and documented (Brouder & Gomez-Macpherson 2014; OECD 2016). Resulting improvements in crop yield and/or economic benefits to farmers overall (e.g. reducing production costs through decreases in energy consumption) have also been extensively discussed in literature. Our review focused on peer-reviewed studies assessing the crop yield of on-field agricultural BMPs *relevant to source water protection*. Studies on profitability (in terms of production costs) of BMPs were also included in the search, where possible. We focused on those BMPs with a clear and direct relationships to crop yield or cost reduction, setting aside those (such as riparian buffers) which necessitate the removal of land from agricultural production.[4]

In general, our review found a rich body of literature (case studies, reviews, and meta-analyses) across cropping systems and geographic areas. Studies were dominated by conservation agriculture, defined by a combination of conservation tillage, crop rotations, and cover crops (e.g. Ogle et al. 2012; Brouder & Gomez-Macpherson 2014; Giller et al. 2015; Pittelkow et al. 2015b; Zhao et al. 2017), and to a lesser extent, agroforestry. As expected, the effects on crop yield vary based on different farming contexts, including practices employed, climate conditions, crop type, biophysical conditions, and farming resources (smallholder or large-scale). There is a limited ability to generalize the direct benefits of soil protection BMPs in terms of crop yield gains to farmers across geographical regions.

There is evidence suggesting that conservation agriculture has the greatest potential to increase crop yields when implemented as a set of integrated practices in rainfed systems in water-limited or water-stressed regions, including potentially on millions of hectares within Sub-Saharan Africa and South Asia (Brouder & Gomez-Macpherson 2014; Palm et al. 2014; Pittelkow et al. 2015a, 2015b; OECD 2016). Studies also point out that, when combined with residue retention (e.g. cover crops) and crop rotation, declines in yield resulting from no-till practices can by minimized (Rusinamhodzi et al. 2011; Pittelkow et al. 2015a).

For agroforestry (agriculture incorporating the cultivation and conservation of trees), impacts on crop yield tend to be most influenced by the type and objective of the particular agroforestry system. For example, agroforestry supporting soil fertility increased crop yields in Africa (Ajayi et al. 2011), whereas shade grown coffee and cacao agroforestry, targeting other ecological benefits, shows more nuanced effects on yield and income (e.g. Steffan-Dewenter et al. 2007; Tscharntke et al. 2011).

It is often assumed that implementing conservation agriculture might increase farmers' incomes mainly through cost reductions (fuel consumption-related costs in particular) and economies of scale (for large mechanized farms) rather than yield increases. For small farmers in developing countries, economic benefits are limited due to farm size, degree of mechanization, and capital investment capacity (Giller et al. 2015; OECD 2016). Detailed economic studies on how agricultural BMPs might affect production costs or profitability are limited. On the other hand, despite the limited economic provision in terms of cost reduction, there may be economic incentives for farmers to implement agricultural BMPs (e.g., where agroforestry enables a diversification of products to sell), and non-economic impacts may also motivate participation (e.g., climate resiliency benefits as one of the major drivers to adopt agricultural BMPs). On balance, the evidence suggests that agricultural BMPs may not, at least in the short term, pay for themselves, especially for smallholder farmers, and should be carefully designed, managed, and monitored.

[4] These BMPs, including building terraces, planting buffer strips, planting trees etc., do not necessary improve yields, and/or may improve yields in the longer term (not immediately) as soil quality is built up or local water quality is improved, and/or may simply avoid future losses – such benefits are rather hard to appreciate from a farmer livelihood point of view, and/or may actually compete with farm production by taking land out of production. However, we recognize that practices like these may have both costs (in terms of productivity losses or input of labor and materials) and benefits and that these practices could be the subject of a valuable separate review.

At a fundamental level, our review suggests that there is a lack of agreement on how agricultural BMPs deliver direct benefits such as crop yield to farmers. To better answer this question in a more applicable way, we suggest narrowing the scope and focusing on a suite of conditions (e.g. climate, soil, slope, scale, smallholder or large holder, crop type, land uses) that are generally the target for source water protection and conduct a meta-analysis using existing datasets. Furthermore, within the context of source water protection, farm level productivity gains and livelihood improvements may require farming system changes that are more complex to interpret than how a single BMP may or may not affect plot level yields. Therefore, our review calls for the need to look beyond just direct crop yield gains or cost savings, and to assess the holistic societal benefits of agricultural BMPs at a watershed scale.

Key gaps

- Given the range of factors influencing yield outcomes, generalization of potential outcomes via systematic review requires higher quality, more standardized monitoring data that cover a range of explanatory factors, over a longer period of time (Brouder and Gomez-Macpherson 2014).
- Compared to conservation agriculture, fewer studies of yield outcomes were found for agroforestry, irrigation management, and nutrient management.
- For conservation agriculture, studies on large-scale farms in North America and Australia suggest potential profitability through increased crop yield and/or reduced input costs. However, for smallholder farmers in the developing world, empirical evidence is sparse, limiting our ability to generalize how much and under which conditions conservation agriculture can generate benefits.
- In terms of geographies, all major regions were represented in the literature, but fewer studies were found from Southeast Asia and Latin America.

Summary table

	Conservation ag	No-till	Cover crops	Agroforestry	Other BMPs
Level of available scientific information	High	High	Medium	Medium-low	Overall low (depending on the practice)
Degree of consensus in findings	Medium	Medium	Medium	Mixed (agroforestry systems matter)	Difficult to assess given the practice types and numbers of studies reviewed

Impacts of anthropogenic change: climate and land use

What are the expected relative magnitudes of contributions to hydrologic outcomes from anthropogenic climate change as compared to land use/land cover change?

Summary

The direction and comparative impacts of climate change and land use/land cover patterns cannot be generalized across regions or sites from retrospective or forecasting studies. Depending on site-specific conditions, either climate or land use conditions can be a dominant driver. The aggregate results of these studies suggest that changing climate conditions has the potential offset beneficial impacts that might result from water fund interventions. Accordingly, it is imperative to consider future climate conditions during the development of conservation plans for any source water protection project, including water funds.

Key findings

For retrospective studies of historical flow outcomes, the relative impact of climate versus land use change varies. Some studies identified climate as the greater driver of past changes to flow. Others identified land use change as the greater driver. A majority of studies evaluated here indicated that both drivers had impacts on flow changes and were of similar magnitude.

For prospective studies of potential flow outcomes, several studies found climate and land use changes to have similar magnitudes of impact, while others identified climate change as the primary driver of future flow changes. Only one study identified land use change as the primary driver of future flow conditions (Ma et al., 2014).

Beyond these broad patterns, it is difficult to generalize study results towards predicting the direction and magnitude of potential climate and land use change impacts on hydrology, given the importance of local context (climate and land use/cover characteristics and trends) and specific study design choices (time periods and scenario selection). Additionally, the combined effects of climate and land use change vary. In some cases, changes in land use were or would be additive, further amplifying changes in precipitation and evapotranspiration (Bari, Silberstein, & Aryal, 2011; DeBano et al., 2016; Ma et al., 2014; Mishra et al., 2010; Montenegro & Ragab, 2012; Records et al., 2014; Woltemade & Hawkins, 2016; Yang et al., 2015). In other cases, land use changes served or could serve to mitigate climate related impacts (Diaz & Querner, 2005; Kharel, Zheng, & Kirilenko, 2016; Ma et al., 2009; Mishra et al., 2010; Yang et al., 2015).

Considerations of both current and future climate and land use patterns are critical for assessing the potential for interventions to influence watershed outcomes. Projects that do not consider the implications of climate change effects could fail to realize project outcomes even where implementation targets have been achieved.

Key gaps

- The lack of a generalized and comparable approach to prospective assessments of climate and land use change limits the ability to draw comparisons between ecosystem types. Whether this is in and of itself a meaningful shortcoming is debatable. However, the lack of agreement on approach could pose a roadblock for project-focused assessments which could likely benefit from clearer guidance on how to incorporate climate considerations into project planning and prioritization.

- Given large regional differences in the expected hydrologic effects resulting from climate change[5], greater regional representation among studies would be welcome. Latin America, North and Sub-Saharan Africa, Middle East, Southeast Asia, and Oceania, in particular, are underrepresented.
- Comparing the relative impacts of climate and land use on water quality outcomes is a notable gap in the existing literature. As noted below, investigations of flow-related outcomes dominate. In light of the primary role that source water protection can play with regard to water quality outcomes[6], additional assessment of these parameters under future climate and land use change would be a valuable contribution to the evidence needed to support watershed management planning.

Discussion

Conclusions

This rapid literature review to identify critical information gaps related to source water protection within the context of water funds, yielded several findings worth highlighting:

1) There is a fair amount of consensus among water funds experts as to the key science questions that they face in the process of designing and implementing activities and programs of work. Questions around water quantity/flows were prioritized by nearly all experts, which is understandable given that water quantity concerns are growing due to the convergence of increasing water demands, landscape change, and climate change.
2) Knowledge gaps are largely the result of a lack of scientific evidence, as opposed to a lack of information transfer to practitioners. The majority of empirical studies have been conducted in North America and, to a lesser extent, Europe and Oceania. Studies have rarely been conducted over the variety of site-specific situations or spatial or temporal scales that are necessary for substantially informing project-level decisions.
3) The reviews underscore the highly context-specific nature of each question, meaning that few conclusions can be generalized across a wide range of contexts, and even the direction of change can often be different from one study to another.

---

[5] However, we recognize that practices like these may have both costs (in terms of productivity losses or input of labor and materials) and benefits (e.g. improved local water quality) and that these practices could be the subject of a valuable separate review.

[6] Mulligan, M. (2009). The human water quality footprint: agricultural, industrial, and urban impacts on the quality of available water globally and in the Andean region. In Proceedings of the International Conference on Integrated Water Resource Management and Climate Change, Cali, Colombia. Retrieved from http://www.ambiotek.com/publications/CINARA_Industry_and_mining.pdf

4) Practitioners should take great care about presuming the direction or magnitude of impacts that have not been well modeled with locally-appropriate data or, better yet, measured *in situ* at appropriate scales and within similar contexts.

Next steps

There are both potential immediate and longer-term next steps emerging from this effort. In the immediate term, the deep dive summaries would benefit from additional review by recognized external experts. These experts should include individuals with expertise on geographies, taxa, and mechanisms that were poorly represented in our literature reviews to ensure that key studies and reports have not been overlooked. Following review and revision, summaries can be repackaged for one or more targeted audiences, such as senior management within organizations exploring and establishing water funds, and policy makers (e.g. multilaterals) considering whether and how to invest in source water protection programs around the world.

For the highest priority questions, a more extensive and systematic identification of key findings and data gaps would be worthwhile. Already, as an outgrowth of this effort and in collaboration[7] with the Natural Capital Project, we have been awarded a SNAPP (Science for Nature and People Partnership) to further investigate components of this work. Through the SNAPP project, we will be convening leading scientists and practitioners to answer the question of whether, and how, source water protection activities can generate meaningful water quantity impacts for different upstream and downstream stakeholder groups, and we will provide guidance on this issue to key decision makers.

We see this initial effort as ultimately leading to development of a shared collaborative research agenda with one or more organizations or research institutions. There is a clear appetite and need for strengthening the science behind water funds in the service of supporting improved efficiency and effectiveness of actions, accelerating the development of new projects, and building a track record of credibility in performance—namely, achieving measurable and meaningful benefits for people and nature.

[7] Our formal implementing partners for the proposal are Forest Trends and the Alliance for Global Water Adaptation (whose members include institutions like the World Bank), along with TNC and NatCap.

Appendix

List of key questions

Below is the full list of key biophysical and social science questions identified that are related to source water protection, within the context of water funds. Highest priority questions are in **bold**.

Water quality

1. Effectiveness

   1.1. What are the most effective source water protection activities in different biophysical contexts to address surface and subsurface runoff loading of pollutants into waterways?

   1.2. What are the most cost-efficient source water protection activities in different biophysical contexts and geographies?

2. Groundwater

   2.1. Which source water protection activities can improve groundwater quality, and which components of water quality can they affect?

3. Activities

   3.1. How much can improved forest management actually reduce the risk of catastrophic fires? For example, what is the reduction in expected severity and/or extent and or occurrence over a given time period?

4. Time scale

   4.1. What is the lag-time between the onset of land use change and measurable differences in water quality, and what is the temporal profile for achieving full functionality of interventions (i.e., maximum effectiveness)?

   4.2. What is the rate of change (i.e., how do benefits change over time; are they immediate/remain constant over time)?

5. Spatial scale

   5.1. How much is enough? How much area for different actions do you need to intervene in, relative to the size of the overall catchment, to have a measurable impact on water quality? Are there general guidelines/rules of thumb for areas of natural lands and interventions across different typologies of ecoregions/development impact type and extent/water quality concerns?

   5.2. What are the key variables that affect attenuation of water quality impacts between intervention site and beneficiary locations?

6. Context

6.1. What are the most important site characteristics that affect the impact of specific activities at a given site for a given pollutant type (e.g., soil type and depth, slope, vegetation type and cover, precipitation type, intensity and temporal distribution)?

6.2. To what extent do different environmental settings influence the types of appropriate activities and expectations for water quality results in a watershed? To what extent are empirical water quality results from one biome or major habitat type applicable to another?

Water quantity

1. Water yield

1.1. To what extent can common source water protection activities affect baseflows, groundwater quantity, and flooding through changes in infiltration and runoff?

1.1.1. What forest management activities are ideal to maximize baseflow/increase dry season water yields, and with what levels of confidence?

1.1.2. What are the infiltration benefits that result from agricultural BMPs?

1.1.3. In what settings/regions/climatic conditions do natural ecosystems increase water yield in streams through infiltration, versus settings where they decrease water yield in streams due to evaporation or storage? In which regions of the world is the science more certain vs. too uncertain to inform decisions?

1.1.4. To what extent can flood peaks be reduced through different source water protection activities, and how does the reduction in flood peaks differ for different recurrence intervals (5, 10, 50, 100 year)? In what landscape contexts can flood peaks be most reduced?

1.1.5. To what extent can source water protection activities help replenish groundwater systems in a range of land cover and soil conditions? What are the dynamics of groundwater recharge zones, and how do they influence the design and effectiveness of source water protection activities?

1.2. What is the lag-time between the onset of land use change and measurable differences in baseflows resulting from restoring natural hydrologic modifiers? What activities would be expected to result in relatively shorter or longer response times?

1.3. To what extent can source water protection activities increase fog capture as a form of water yield?

1.4. How does reforestation affect water yield to streams and aquifers over time? How do results change over the maturity of forest cover and growth?

1.5. To what extent can improved forest management (and subsequent reduced risk of fire) contribute to seasonal water regulation?

1.6. What are the trade-offs in implementing activities for different outcomes? E.g., increasing vegetative cover to reduce landslides may have important trade-offs on water yield

2. Context

   2.1. To what extent do different environmental settings influence the types of appropriate activities and expectations for water quantity results in a watershed? To what extent are empirical water quantity results from one biome or major habitat type applicable to another?

Biodiversity/habitat outcomes

1. Evidence

   1.1. To what extent can different source water protection activities contribute to terrestrial ecosystem and biodiversity benefits?

      1.1.1. Which agricultural BMPs, individually or in combination, produce the best terrestrial biodiversity outcomes?

   1.2. To what extent can different source water protection activities contribute to aquatic ecosystem and biodiversity benefits?

      1.2.1. Which agricultural BMPs, individually or in combination, produce the best aquatic biodiversity outcomes?

2. Trade-offs

   2.1. What are the trade-offs and risks associated with source water protection activities from a biodiversity conservation perspective?

Climate change mitigation

1. Forests

   1.1. How much can improved forest management and restoration activities actually contribute to carbon storage? What criteria most highly affect carbons storage outcomes?

   1.2. What scales of work are required to make a significant difference in carbon stocks?

Health and well-being

1. Health

   1.1. Under what circumstances can source water protection activities reduce the risk of water-borne diseases?

      1.1.1. Which activities are most effective at reducing the risk of water-borne diseases?

   1.2. Under what circumstances can source water protection activities reduce the risk of vector-borne diseases?

      1.2.1. Which activities are most effective at reducing the risk of vector-borne diseases?

   1.3. Who are the direct beneficiaries of disease risk reductions?

2. Livelihoods

   2.1. When/for what changes in land management are negative livelihood impacts likely?

2.2. What are the most appropriate and low-cost measures to minimize and compensate for negative livelihood impacts?

Agricultural best management practices

1. Livelihood benefits

   1.1. To what extent can specific agricultural best management practices result in crop yield gains or cost reductions?

      1.1.1. Under what particular contextual parameters are productivity gains predicted? How much variability can be expected with regard to productivity outcomes?

   1.2. Which BMPs are likely to have the highest net benefits to farmers and to other stakeholders?

2. Trade-offs

   2.1. In what situations are the intended co-benefit outcomes of agricultural BMPs in potential conflict with each other?

Watershed scale questions

1. Watershed-scale land use/land cover

   1.1. What is known about how different proportions and positioning of land use/land cover and BMPs in a watershed affect water quality, water quantity, and biodiversity?

Modeling

1. Uncertainty/future change

   1.1. How do evaluate the extent to which natural infrastructure planned today will perform into the future under uncertain climate changes?

      1.1.1. What are the relative expected magnitudes of contributions to water quality and quantity outcomes from climate change as compared to land use/land cover change?

2. Validation

   2.1. To what extent have the benefits predicted by water quality and quantity models been validated by real-world results?

   2.2. How do we design studies that can ground-truth models?

3. Assumptions

   3.1. In water fund planning, how can we a priori account for the difference in outcomes between optimized (modeled) activity portfolios and actual implemented portfolios?

   3.2. What costs assumptions are modelers using? Do these assumptions include transaction costs? How many water funds calibrate their models? In what context are uncalibrated results credible?

   3.3. How do we frame or establish sound assumptions for results or data we're not sure about?

4. Best practice
   4.1. How important is spatial resolution? In what context are lower res sufficient? In what context is higher res a must?
   4.2. What are the preferred models for predicting particular watershed hydrologic responses (quality, flow regulation etc.) to water fund interventions, and what does this depend on?
   4.3. When are global data sets enough to answers questions and when do we need more specific information?

Detailed reviews

Aquatic biodiversity

To what extent can different source water protection activities contribute to aquatic ecosystem and biodiversity benefits?

Targeted land protection

Protecting intact, native terrestrial systems should rarely if ever produce negative consequences for aquatic biodiversity, but benefits are not assured. While protected terrestrial systems themselves are beneficial, the size and locations of them in a watershed can affect their efficacy for freshwater conservation. The aquatic biodiversity benefits of formal protected areas (PAs) have been little studied, in part because so few protected areas have been designated based on freshwater objectives (Abell, Allan, & Lehner, 2007; Adams, Setterfield, Douglas, Kennard, & Ferdinands, 2015; Chessman, 2013; Chu, Ellis, & de Kerckhove, 2017; Mancini et al., 2005). Existing studies have pointed to factors influencing impact and effectiveness:

<u>Spatial scale</u> – Small protected areas may have little impact, though if they drain to small headwater streams the impact may be greater than if they are located downstream (Thieme et al., 2016). While there is no universal threshold for how much of a catchment should be protected to conserve freshwater biodiversity, some studies (e.g. Death & Collier, 2010) have attempted to identify thresholds for particular settings.

<u>Spatial scale</u> – Small protected areas may have little impact, though if they drain to or comprise small headwater streams the impact may be proportionately greater than if they are located further downstream in river networks (Thieme et al., 2016). While there is no universal threshold for how much of a catchment should be protected to conserve freshwater biodiversity, some studies (e.g. Death & Collier, 2010) have attempted to identify thresholds for particular settings.

<u>Location</u> – Some sub-watersheds or even stream reaches may support higher priority freshwater biodiversity elements than others, but most terrestrial protected areas have not been sited with these elements in mind (Abell et al., 2007; Herbert, Mcintyre, Doran, Allan, & Abell, 2010).

Upland vs. riparian – Protecting riparian buffers is almost universally considered an effective conservation activity, though larger catchment influences may swamp riparian benefits (see Riparian Restoration below).

Type and level of protection – Even well-protected land cannot provide comprehensive protection for aquatic species if there are direct threats like overfishing, instream barriers, and water withdrawals (Chessman, 2013). As well, the designation of a terrestrial protected area does not in itself ensure the existence and maintenance of natural land cover (Mancini et al., 2005), and the legacy of past land uses may continue to produce stresses on local and downstream aquatic communities via water quality and flow impacts (Harding, Benfield, Bolstad, Helfman, & Jones, 1998).

Exogenous threats – Freshwater systems sit at the lowest point on the landscape and are subject to hydrologically-mediated threats traveling downslope and downstream. As a result, from a freshwater perspective especially, no protected area is an island, and threats outside a PA's borders (e.g. dams, water withdrawals, agriculture, mining, forestry, urbanization) can impinge upon the species and systems inside (Rodríguez-Jorquera et al., 2017; Thieme et al., 2016). As well, many aquatic species are highly mobile, and they may encounter threats outside a PA that reduce their viability inside it. Finally, climate change is already manifesting itself largely through hydrologic impacts; protected areas may be able to build resilience for freshwater species (e.g. through providing thermal refugia or serving as dispersal corridors) but there will be limits to their threat mitigation potential (Kingsford, 2011).

Formally designated PAs are only one form of targeted land protection, and 'other effective area-based conservation measures' (OECMs) may be equally if not more relevant to source water protection programs. A formal designation does not ensure effective management, and other land protection approaches (e.g. through indigenous or community-managed areas) may produce beneficial aquatic biodiversity outcomes as well.

Revegetation

Afforestation. Many but not all studies of afforestation – the planting of stands of trees where there were none previously – focus on plantation forestry, looking primarily at the impacts of afforestation with non-native (typically conifer) species on stream chemistry (Friberg, Rebsdorf, & Larsen, 1998). There are a small number of studies that generally confirm the negative impacts of afforestation on native aquatic biota (Sievers, Hale, & Morrongiello, 2017; Tierney, Kelly-Quinn, & Bracken, 1998), though that response is not uniform (Quinn, Croker, Smith, & Bellingham, 2009; Tierney et al., 1998). Most afforestation studies have taken place in Europe and New Zealand, with a focus on riparian zones.

Reforestation. Reforestation studies are largely focused on riparian zones (see below). Despite the prevalence of reforestation (active and passive) in some geographies (e.g. in North America), we found virtually no studies that evaluate the impacts of upland reforestation on aquatic biodiversity. While we can assume that on balance the impacts of reforestation using native species should be positive for freshwater biodiversity, there are few empirical data to confirm that assumption. One modeling study of re-establishing old-growth forests suggested potentially negative impacts on freshwater fisheries productivity, at least in the near term (Nislow, 2005), underscoring the possibility of unintended consequences of more natural conditions on native species composition and abundance. The legacy of past land use (e.g. agriculture, mining) within a catchment may be a stronger indicator of present-day ecological status than current land use, suggesting that upland or riparian reforestation can in some circumstances have limited benefits in terms of native species recovery (Brady, 2016; Harding et al., 1998; Ogden, Crouch, Stallard, & Hall, 2013; Scott, 2006). As well, the biotic impacts of reforestation may be manifested over longer time scales than water quality impacts (Yeung, Lecerf, & Richardson, 2017).

Grassland restoration. There is virtually no literature exploring the impacts of grassland restoration on aquatic biodiversity. However, depending on the classification some grassland types may be considered wetlands (Dixon, Faber-Langendoen, Josse, Morrison, & Loucks, 2014), and as noted below (see Wetland Restoration and Creation) there is a richer literature on wetland restoration. Some grasslands will need to be maintained by natural disturbance processes like fire or grazing, and re-created disturbances may have negative impacts on aquatic systems if not carefully designed and implemented (see Ranching BMP below).

Riparian restoration

The role of riparian zones in sustaining healthy aquatic systems has received ample research attention (e.g. Gregory, Swanson, McKee, & Cummins, 1991; Naiman & Decamps, 1997). However, more research is focused on physical habitat attributes than on biotic indicators, and long-term studies of the biotic impacts of riparian restoration (as opposed to management, often as part of forestry practices) are relatively sparse (e.g. Roni, Hanson, & Beechie, 2008; Roni et al., 2002; Sievers, Hale, & Morrongiello, 2017). Measures of benefits to aquatic systems have often focused on representative species and system attributes – often measured with indexes of biotic integrity or ecosystem function – and the response of those attributes to riparian land use (sometimes in comparison to upland land use). In other cases – generally in North America and Europe – restoration has been driven by one or more endangered fish species, such as salmon in the Pacific Northwest (Palmer et al., 2007; Sievers, Hale, & Morrongiello, 2017). Our review focused primarily on riparian restoration within agricultural contexts, recognizing that the abundance of literature has focused on riparian management in the context of forestry practices.

There is consensus that vegetated riparian zones can serve as filters for nutrients and sediment, thereby improving water quality; can contribute important woody debris and other organic materials that are essential to sustaining aquatic systems and species; can regulate stream temperature (especially in the case of forests); can contribute aquatic and terrestrial invertebrates and detritus important for sustaining downstream systems; and can provide habitat for diverse assemblages of riparian-dependent species (Naiman & Decamps, 1997). Importantly, these generalizations apply to restoration, rather than to afforestation. Despite largely positive results, empirical studies of the biotic impacts of riparian restoration have not been uniform in their findings for riparian-dependent and aquatic biota (Marczak et al., 2010; Quinn et al., 2009; Sievers, Hale, & Morrongiello, 2017), and time lags on the order of a decade or more may occur before native biota are restored (Becker & Robson, 2009; Orzetti, Jones, & Murphy, 2010). Mixed results have occurred for a variety of reasons, including but not limited to an inability to recreate natural ecosystem composition, structure and function through plantings or passive restoration (e.g. Faulkner, Barrow, Keeland, Walls, & Telesco, 2011). The dependencies of riparian zones on natural flow patterns for appropriate seasonal inundation can also determine the potential for riparian restoration in systems with altered flows, and flow management may be the primary tactic for riparian restoration (e.g. Rood et al., 2003; Rood, Braatne, & Hughes, 2003; Rood et al., 2005).

As with all source water protection activities, the degree to which these benefits can be realized is context-specific, and in the case of extensive natural land cover conversion and/or development, catchment-scale and in-stream conditions may swamp the services that riparian zones can provide (Death & Collier, 2010; Houlahan & Findlay, 2004; Lennox et al., 2011; Teels et al., 2006; Walsh, Waller, Gehling, & Mac Nally, 2007).

Despite a rich literature, most studies of riparian restoration have been focused on North America and Oceania, with a smaller number from Europe. We found few relevant studies from Latin America, Africa, or Asia.

Agricultural BMPs

While there is a wealth of literature on the impacts of land use/land cover on aquatic ecosystem and biotic conditions, there are few empirical studies of the results of agricultural best management practices on them, other than riparian buffer zone management. As with other source water protection tactics detailed above, there is a presumption that reductions in sediment and nutrients will benefit aquatic ecosystems and species, but few studies have documented the impact (Holmes, Armanini, & Yates, 2016). Measures of ecological impact in the context of agricultural land use are frequently based on indicator species, indices of biotic integrity, or habitat measures, which specify overall habitat quality but by definition give no information about particular species of concern; these species may fail to show improvement due to a range of factors outside the influence of spatially-limited BMPs (e.g. Wang, Lyons, & Kanehl, 2006).

Of the existing literature on agricultural BMP impacts on aquatic biodiversity and environmental conditions, many show positive (though sometimes limited) impacts (e.g. Barton & Farmer, 1997; Sallenave & Day, 1991). Other studies find that, as with other source water protection activities, observed impacts – or lack thereof -- may result from broader catchment-wide land use as well as the legacy of past practices and additional threats (Maret, MacCoy, & Carlisle, 2008; Nerbonne & Vondracek, 2001; Pearce & Yates, 2015). Thresholds of areas of BMP implementation resulting in benefits to biodiversity may exist (Marshall, Fayram, Panuska, Baumann, & Hennessy, 2008; Yates, Bailey, & Schwindt, 2007), and BMPs applied only in riparian zones may be necessary but insufficient in scope (Wang, Lyons, & Kanehl, 2002).

We were unable to find literature exploring the impacts on aquatic biodiversity of agroforestry systems, though there is a relatively abundant literature on agroforestry in general.

Ranching BMPs

The relationship between animal grazing and aquatic ecosystems is complicated. Because grasslands can require natural disturbance or active management to prevent succession to forest, grazing at certain levels and times of the year may be an appropriate and necessary activity in some systems and can contribute to maintaining native aquatic biodiversity (Bloom, Howerter, Emery, & Armstrong, 2013; Burton, Gray, Schmutzer, & Miller, 2009; Marty, 2015; Mester, Szalai, Mero, Puky, & Lengyel, 2015;

Schaich & Barthelmes, 2012). However, overgrazing can have impacts on sediment and nutrient delivery, as well as on hydrology. Furthermore, livestock congregating in riparian zones and entering streams in order to cross or access water can have direct impacts on stream banks and can introduce bacterial and nutrient contaminants into water, as well as increase sediments (Fitch & Adams, 1998).

The literature examining livestock or ranching BMP impacts on biodiversity is focused primarily in three areas: 1) impacts on grassland plant, bird, and amphibian species from changes in grazing intensity or timing, with an emphasis on wet grasslands and wetland birds; 2) impacts on native riparian plants, birds and mammals from livestock exclusion (via fencing); and 3) impacts on stream ecosystem condition from livestock exclusion. Studies generally confirm the value of excluding livestock from streams, reducing grazing intensity, and providing livestock with alternative water sources (Ellison, Skinner, & Hicks, 2009; Jansen & Robertson, 2001a, 2001b; Kauffman, Thorpe, & Brookshire, 2004; Line, Harman, Jennings, Thompson, & Osmond, 2000; Sievers et al., 2017); however, the degree and scope of benefits may be localized and overwhelmed by larger catchment land uses (Magierowski, Davies, Read, & Horrigan, 2012; Ranganath, Hession, & Wynn, 2009; Raymond & Vondracek, 2011), and exclusion may also facilitate the establishment and spread of invasive plant species (Loo, Mac Nally, O'Dowd, & Lake, 2009). The effects of intensive grazing may also persist over many years after the removal of livestock, and recovery times for species will depend on a variety of factors (like dispersal ability) (Homyack & Giuliano, 2002). Some other ranching best management practices (e.g. rotational grazing) may be appropriate in different contexts (Magner, Vondracek, & Brooks, 2008), though there is little information to support a significant positive impact on aquatic biodiversity and other factors (e.g. riparian buffer function) may be more important (Brand, Vondracek, & Jordan, 2015; Sovell, Vondracek, Frost, & Mumford, 2000). A main discriminator may be whether a system historically experienced natural grazing pressure or not. In the absence of native grazers, the creation and maintenance of wetlands (e.g. vernal pools) may depend on cattle grazing, with benefits for native and even endemic aquatic species (Marty, 2005; Mero, Lontay, & Lengyel, 2015; Pelinson, Garey, & Rossa-Feres, 2016).

Most studies of grazing impacts have examined North American systems, though a smaller number have examined grazing and ranching in Europe and South America (e.g. the Pantanal). We were unable to find literature exploring the impacts on aquatic biodiversity of silvopastoral systems.

Fire risk management

The impacts of fire risk management on aquatic systems and species are complicated and in general have received relatively little attention (Bisson et al., 2003; Dwire & Kauffman, 2003; Pilliod, Bury, Hyde, Pearl, & Corn, 2003). The question of whether and how severe wildfire itself is damaging to aquatic systems and species is context-specific. Fires mobilize nutrients, sediments and debris, increase runoff and river discharge, and have impacts on turbidity, light and temperature, and organic inputs. The types and time-lags of biotic responses are influenced by the scope of fires and the pre- and post-fire patterns of droughts and floods. Biotic responses vary by life cycles, habitat specificity, dispersal abilities, and the availability and distribution of refugia (See Bixby et al., (2015) for a summary of impacts of fires on patterns, processes and biological responses.)

A meta-analysis of studies of amphibian responses to severe wildfires found both positive and negative impacts and noted that natural recovery of species populations affected by fire may be hampered by land use change and fragmentation (Hossack & Pilliod, 2011). A review of the impacts of forest fires on fishes in North America found that although some populations might be extirpated by severe fires, recolonization by more mobile species is relatively rapid (Gresswell, 1999; Howell, 2006). For species with low mobility (either naturally or due to habitat fragmentation), high habitat specificity, or for those species with highly reduced population numbers, the impacts may be greater and more long-lasting (Dunham, Young, Gresswell, & Rieman, 2003; Gresswell, 1999).

Activities designed to lessen the severity and/or risk of fires can impact stream biota. Of those studies that have addressed the question of how fire risk management activities affect aquatic species and systems, virtually all have examined North American coniferous systems. Typically, studies have investigated combinations of prescribed burning and tree thinning. Some aquatic species are adapted to systems that experience lower-level, periodic fires so in principle might benefit from prescribed burns, especially in conjunction with mitigation of other threats (Whitney, Gido, Pilger, Propst, & Turner, 2016). However, already vulnerable species with low population viability may be an important exception (Driscoll & Roberts, 1997). Stream ecosystems flowing through forested landscapes are also typically reliant on terrestrial inputs of coarse woody materials, and forest thinning in the service of reducing the risk of catastrophic fires may deprive streams of that material if the cut trees are removed (Rieman et al., 2003). Additionally, forest thinning activities may involve the construction of logging roads, with their concomitant chronic impacts (Rieman et al., 2003). The importance of retaining riparian buffer zones is a common theme across many studies (Olson, Leirness, Cunningham, & Steel, 2014), as is the importance of being aware of unintended consequences (Rieman & Clayton, 1997). Based on the limited literature, the impacts of forest fuel reduction on aquatic species are mixed and highly context-specific.

Wetland restoration and creation

There is a reasonably good literature on wetland restoration and creation impacts on biodiversity. In general, restored or rehabilitated wetlands are found to provide greater aquatic biodiversity and ecosystem function benefits than created wetlands, yet less than natural wetlands (Meli, Rey Benayas, Balvanera, & Martinez Ramos, 2014; Sebastián-González & Green, 2016; Spadafora et al., 2016). Some studies examining wetland birds, invertebrates, and amphibians have found evidence of positive change in terms of population abundance or species presence, though these changes are not necessarily sustained over time (Brown, Smith, & Batzer, 1997; Hapner et al., 2011; Locky, Davies, & Warner, 2005; Ruhi et al., 2012; Walls, Waddle, & Faulkner, 2014). Restored peatlands have been shown to support aquatic communities comparable to those in natural wetlands (Brown, Ramchunder, Beadle, & Holden, 2016). Created compensatory wetlands often fail to achieve the same ecosystem functions as restored/rehabilitated natural wetlands (Brown & Smith, 1998; Brown & Veneman, 2001; Español, Gallardo, Comín, & Pino, 2015; Spadafora et al., 2016; Wan, Qin, Li, & Liu, 2001; Whigham, 1999). 'Dual purpose ponds' or multi-objective wetlands on agricultural landscapes, aimed at biodiversity conservation as well as nutrient retention and/or flood abatement, result in generally positive impacts but there is also a recognition that no single wetland can indefinitely provide all these services (Zedler, 2003). Wetland restoration for biodiversity conservation may be most effective when a density of wetlands is achieved within a landscape, both to facilitate dispersal among species and because wetlands undergo successional processes (Thiere et al., 2009).

It is important to note that these studies have looked at the impacts of biodiversity within the created or restored wetlands, rather than examining downstream aquatic biodiversity, which presumably would benefit to some extent from improved water quality. Many of the studied wetlands are not connected directly to river systems and so there would be minimal exchange of obligate aquatic species among them.

Studies of the biodiversity and ecosystem integrity impacts of wetland restoration and creation are almost exclusively focused on North America and Europe, with a small number examining wetlands in Asia.

Road management

Roads, especially but not exclusively those that are unpaved, are widely acknowledged to be substantial sources of sediment to streams, though even that conventional wisdom has been empirically challenged (Al-Chokhachy et al., 2016). Literature on the impacts of road management, including road removal, on aquatic species is sparse (Angermeier, Wheeler, & Rosenberger, 2004; McCaffery, Switalski, & Eby, 2007; Roni et al., 2008; Switalski, Bissonette, DeLuca, Luce, & Madej, 2004), with much of it focused on forested landscapes in the Pacific Northwest of North America (Jones, Swanson, Wemple, & Snyder, 2000; Roni et al., 2002). Many studies of the impacts of roads on aquatic biodiversity focus on the barriers that road-stream crossings pose to species movement. Others look at stormwater-related impacts of urban roads, but these are often modeled (Roni et al., 2008). There is some literature providing guidance on managing unpaved roads (typically associated with forestry operations) to reduce sediment impacts, but the literature is largely focused in the United States (Roni et al., 2008). At the same time, the sediment contributions of roads at basin scales may be dwarfed by other contributors, such as from wildfire (Goode, Luce, & Buffington, 2012), underscoring the multi-faceted nature of threats to freshwater systems. A rare study in the Amazon found that reduced-impact logging, which includes minimizing construction of access roads, can reduce negative impacts to aquatic communities (Dias, Magnusson, & Zuanon, 2010).

While there is a dearth of literature on the measured aquatic biodiversity impacts of road management (other than removing instream barriers), there seems to be a reasonably clear pathway between road management that successfully reduces sediment loading to streams, and benefits to aquatic species (with the caveat that many species will be affected by multiple threats, of which sediment will be only one).

Literature reviewed

Abell, R., Allan, J. D., & Lehner, B. (2007). Unlocking the potential of protected areas for freshwaters. *Biological Conservation*, *134*(1), 48–63. https://doi.org/10.1016/j.biocon.2006.08.017

Adams, V. M., Setterfield, S. A., Douglas, M. M., Kennard, M. J., & Ferdinands, K. (2015). Measuring benefits of protected area management: trends across realms and research gaps for freshwater systems. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*(1681). https://doi.org/10.1098/rstb.2014.0274

Al-Chokhachy, R., Black, T. A., Thomas, C., Luce, C. H., Rieman, B., Cissel, R., … Kershner, J. L. (2016). Linkages between unpaved forest roads and streambed sediment: why context matters in directing road restoration: Unpaved roads and streambed sediment linkages. *Restoration Ecology*, *24*(5), 589–598. https://doi.org/10.1111/rec.12365

Allan, J. D. (2004). Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, *35*(1), 257–284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122

Angermeier, P., Wheeler, A., & Rosenberger, A. (2004). A conceptual framework for assessing impacts of roads on aquatic biota. *Fisheries*, *29*(12), 19–29. https://doi.org/10.1577/1548-8446(2004)29[19:ACFFAI]2.0.CO;2

Barton, D., & Farmer, M. (1997). The effects of conservation tillage practices on benthic invertebrate communities in headwater streams in southwestern Ontario, Canada. *Environmental Pollution*, *96*(2), 207–215. https://doi.org/10.1016/S0269-7491(97)00020-1

Becker, A., & Robson, B. J. (2009). Riverine macroinvertebrate assemblages up to 8 years after riparian restoration in a semi-rural catchment in Victoria, Australia. *Marine and Freshwater Research*, *60*(12), 1309–1316. https://doi.org/10.1071/MF08350

Bisson, P., Rieman, B., Luce, C., Hessburg, P., Lee, D., Kershner, J., … Gresswell, R. (2003). Fire and aquatic ecosystems of the western USA: current knowledge and key questions. *Forest Ecology and Management*, *178*(1–2), 213–229. https://doi.org/10.1016/S0378-1127(03)00063-X

Bixby, R. J., Cooper, S. D., Gresswell, R. E., Brown, L. E., Dahm, C. N., & Dwire, K. A. (2015). Fire effects on aquatic ecosystems: an assessment of the current state of the science. *Freshwater Science*, *34*(4), 1340–1350. https://doi.org/10.1086/684073

Bloom, P. M., Howerter, D. W., Emery, R. B., & Armstrong, L. M. (2013). Relationships between grazing and waterfowl production in the Canadian prairies. *Journal of Wildlife Management*, *77*(3), 534–544. https://doi.org/10.1002/jwmg.497

Brady, J. K. (2016). Salamander diversity of reforested abandoned surface coal mines in the Appalachian Region, USA. *Restoration Ecology*, *24*(3), 398–405. https://doi.org/10.1111/rec.12314

Brand, G., Vondracek, B., & Jordan, N. R. (2015). Influence of grazing and land use on stream-channel characteristics among small dairy farms in the Eastern United States. *Renewable Agriculture and Food Systems*, *30*(6), 524–536. https://doi.org/10.1017/S1742170514000362

Brown, L. E., Ramchunder, S. J., Beadle, J. M., & Holden, J. (2016). Macroinvertebrate community assembly in pools created during peatland restoration. *Science of The Total Environment*, *569*, 361–372. https://doi.org/10.1016/j.scitotenv.2016.06.169

Brown, S. C., Smith, K., & Batzer, D. (1997). Macroinvertebrate responses to wetland restoration in northern New York. *Environmental Entomology*, *26*(5), 1016–1024. https://doi.org/10.1093/ee/26.5.1016

Brown, S., & Smith, C. (1998). Breeding season bird use of recently restored versus natural wetlands in New York. *JOURNAL OF WILDLIFE MANAGEMENT*, *62*(4), 1480–1491. https://doi.org/10.2307/3802014

Brown, S., & Veneman, P. (2001). Effectiveness of compensatory wetland mitigation in Massachusetts, USA. *Wetlands*, *21*(4), 508–518. https://doi.org/10.1672/0277-5212(2001)021[0508:EOCWMI]2.0.CO;2

Burton, E. C., Gray, M. J., Schmutzer, A. C., & Miller, D. L. (2009). Differential Responses of Postmetamorphic Amphibians to Cattle Grazing in Wetlands. *Journal of Wildlife Management*, *73*(2), 269–277. https://doi.org/10.2193/2007-562

Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., & Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorous and nitrogen. *Ecological Applications*, *8*(3), 559–568. https://doi.org/10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2

Castro, J., & Reckendorf, F. (1995). *Effects of sediment on the aquatic environment* (Potential NRCS Actions to Improve Aquatic Habitat - Working Paper No. 6 No. 6). Natural Resources Conservation Service. Retrieved from https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/?cid=nrcs143_014201

Chessman, B. C. (2013). Do protected areas benefit freshwater species? A broad-scale assessment for fish in Australia's Murray-Darling Basin. *Journal of Applied Ecology*, *50*(4), 969–976. https://doi.org/10.1111/1365-2664.12104

Chu, C., Ellis, L., & de Kerckhove, D. T. (2017). The effectiveness of terrestrial protected areas for lake fish community conservation. *Conservation Biology*. https://doi.org/10.1111/cobi.13034

Correll, D. L. (1998). The Role of Phosphorus in the Eutrophication of Receiving Waters: A Review. *Journal of Environment Quality*, *27*(2), 261. https://doi.org/10.2134/jeq1998.00472425002700020004x

Death, R. G., & Collier, K. J. (2010). Measuring stream macroinvertebrate responses to gradients of vegetation cover: when is enough enough?: Macroinvertebrate responses to gradients of vegetation cover. *Freshwater Biology*, *55*(7), 1447–1464. https://doi.org/10.1111/j.1365-2427.2009.02233.x

Dias, M. S., Magnusson, W. E., & Zuanon, J. (2010). Effects of Reduced-Impact Logging on Fish Assemblages in Central Amazonia. *Conservation Biology*, *24*(1), 278–286. https://doi.org/10.1111/j.1523-1739.2009.01299.x

Dixon, A. P., Faber-Langendoen, D., Josse, C., Morrison, J., & Loucks, C. J. (2014). Distribution mapping of world grassland types. *Journal of Biogeography*, *41*(11), 2003–2019. https://doi.org/10.1111/jbi.12381

Driscoll, D. A., & Roberts, J. D. (1997). Impact of fuel-reduction burning on the frog Geocrinia lutea in southwest Western Australia. *Austral Ecology*, *22*(3), 334–339. https://doi.org/10.1111/j.1442-9993.1997.tb00679.x

Dunham, J., Young, M., Gresswell, R., & Rieman, B. (2003). Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. *Forest Ecology and Management*, *178*(1–2), 183–196. https://doi.org/10.1016/S0378-1127(03)00061-6

Dwire, K., & Kauffman, J. (2003). Fire and riparian ecosystems in landscapes of the western USA. *Forest Ecology and Management*, *178*(1–2), 61–74. https://doi.org/10.1016/S0378-1127(03)00053-7

Elias, D., Wang, L., & Jacinthe, P.-A. (2018). A meta-analysis of pesticide loss in runoff under conventional tillage and no-till management. *Environmental Monitoring and Assessment*, *190*(2). https://doi.org/10.1007/s10661-017-6441-1

Ellison, C. A., Skinner, Q. D., & Hicks, L. S. (2009). Assessment of Best-Management Practice Effects on Rangeland Stream Water Quality Using Multivariate Statistical Techniques. *Rangeland Ecology & Management*, *62*(4), 371–386. https://doi.org/10.2111/08-026.1

Español, C., Gallardo, B., Comín, F. A., & Pino, M. R. (2015). Constructed wetlands increase the taxonomic and functional diversity of a degraded floodplain. *Aquatic Sciences*, *77*(1), 27–44. https://doi.org/10.1007/s00027-014-0375-2

Faulkner, S., Barrow, W., Keeland, B., Walls, S., & Telesco, D. (2011). Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. *Ecological Applications*, *21*(sp1), S31–S48. https://doi.org/10.1890/10-0592.1

Fitch, L., & Adams, B. (1998). Can cows and fish co-exist? *Canadian Journal of Plant Science*, *78*(2), 191–198.

Friberg, N., Rebsdorf, A., & Larsen, S. (1998). Effects of afforestation on acidity and invertebrates in Danish streams and implications for freshwater communities in Denmark. *Water, Air, and Soil Pollution*, *101*(1–4), 235–256. https://doi.org/10.1023/A:1004949203686

Goode, J. R., Luce, C. H., & Buffington, J. M. (2012). Enhanced sediment delivery in a changing climate in semi-arid mountain basins: Implications for water resource management and aquatic habitat in the northern Rocky Mountains. *Geomorphology*, *139*, 1–15. https://doi.org/10.1016/j.geomorph.2011.06.021

Gregory, S. V., Swanson, F. J., McKee, W. A., & Cummins, K. W. (1991). An Ecosystem Perspective of Riparian Zones: Focus on links between land and water. *BioScience*, *41*(8), 540–551.

Gresswell, R. E. (1999). Fire and Aquatic Ecosystems in Forested Biomes of North America. *Transactions of the American Fisheries Society*, *128*(2), 193–221. https://doi.org/10.1577/1548-8659(1999)128<0193:FAAEIF>2.0.CO;2

Hapner, J. A., Reinartz, J. A., Fredlund, G. G., Leithoff, K. G., Cutright, N. J., & Mueller, W. P. (2011). Avian Succession in Small Created and Restored Wetlands. *Wetlands*, *31*(6), 1089–1102. https://doi.org/10.1007/s13157-011-0220-5

Harding, J. S., Benfield, E. F., Bolstad, P. V., Helfman, G. S., & Jones, E. B. (1998). Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Sciences of the United States of America*, *95*(25), 14843–14847.

Herbert, M. E., Mcintyre, P. B., Doran, P. J., Allan, J. D., & Abell, R. (2010). Terrestrial Reserve Networks Do Not Adequately Represent Aquatic Ecosystems. *Conservation Biology*, *24*(4), 1002–1011. https://doi.org/10.1111/j.1523-1739.2010.01460.x

Holmes, R., Armanini, D. G., & Yates, A. G. (2016). Effects of Best Management Practice on Ecological Condition: Does Location Matter? *Environmental Management*, *57*(5), 1062–1076. https://doi.org/10.1007/s00267-016-0662-x

Homyack, J., & Giuliano, W. (2002). Effect of streambank fencing on herpetofauna in pasture stream zones. *Wildlife Society Bulletin*, *30*(2), 361–369.

Hossack, B. R., & Pilliod, D. S. (2011). Amphibian Responses to Wildfire in the Western United States: Emerging Patterns from Short-Term Studies. *Fire Ecology*, *7*(2), 129–144. https://doi.org/10.4996/fireecology.0702129

Houlahan, J., & Findlay, C. (2004). Estimating the `critical' distance at which adjacent land-use degrades wetland water and sediment quality. *Landscape Ecology*, *19*(6), 677–690. https://doi.org/10.1023/B:LAND.0000042912.87067.35

Howell, P. J. (2006). Effects of wildfire and subsequent hydrologic events on fish distribution and abundance in tributaries of north fork John Day River. *North American Journal of Fisheries Management*, *26*(4), 983–994. https://doi.org/10.1577/M05-114.1

Jansen, A., & Robertson, A. I. (2001a). Relationships between livestock management and the ecological condition of riparian habitats along an Australian floodplain river. *Journal of Applied Ecology*, *38*(1), 63–75. https://doi.org/10.1046/j.1365-2664.2001.00557.x

Jansen, A., & Robertson, A. I. (2001b). Riparian bird communities in relation to land management practices in floodplain woodlands of south-eastern Australia. *Biological Conservation*, *100*(2), 173–185. https://doi.org/10.1016/S0006-3207(00)00235-4

Jones, J. A., Swanson, F. J., Wemple, B. C., & Snyder, K. U. (2000). Effects of Roads on Hydrology, Geomorphology, and Disturbance Patches in Stream Networks. *Conservation Biology*, *14*(1), 76–85. https://doi.org/10.1046/j.1523-1739.2000.99083.x

Kauffman, J., Thorpe, A., & Brookshire, E. (2004). Livestock exclusion and belowground ecosystem responses in riparian meadows of Eastern Oregon. *Ecological Applications*, *14*(6), 1671–1679. https://doi.org/10.1890/03-5083

Kingsford, R. T. (2011). Conservation management of rivers and wetlands under climate change - a synthesis. *Marine and Freshwater Research*, *62*(3), 217–222. https://doi.org/10.1071/MF11029

Lennox, M. S., Lewis, D. J., Jackson, R. D., Harper, J., Larson, S., & Tate, K. W. (2011). Development of Vegetation and Aquatic Habitat in Restored Riparian Sites of California's North Coast Rangelands. *Restoration Ecology*, *19*(2), 225–233. https://doi.org/10.1111/j.1526-100X.2009.00558.x

Line, D. E., Harman, W. A., Jennings, G. D., Thompson, E. J., & Osmond, D. L. (2000). Nonpoint-Source Pollutant Load Reductions Associated with Livestock Exclusion. *Journal of Environment Quality*, *29*(6), 1882. https://doi.org/10.2134/jeq2000.00472425002900060022x

Locky, D. A., Davies, J. C., & Warner, B. G. (2005). Effects of wetland creation on breeding season bird use in boreal eastern Ontario. *The Canadian Field-Naturalist*, *119*(1), 64–75. https://doi.org/10.22621/cfn.v119i1.82

Loo, S. E., Mac Nally, R., O'Dowd, D. J., & Lake, P. S. (2009). Secondary Invasions: Implications of Riparian Restoration for In-Stream Invasion by an Aquatic Grass. *Restoration Ecology*, *17*(3), 378–385. https://doi.org/10.1111/j.1526-100X.2008.00378.x

Magierowski, R. H., Davies, P. E., Read, S. M., & Horrigan, N. (2012). Impacts of land use on the structure of river macroinvertebrate communities across Tasmania, Australia: spatial scales and thresholds. *Marine and Freshwater Research*, *63*(9), 762–776. https://doi.org/10.1071/MF11267

Magner, J. A., Vondracek, B., & Brooks, K. N. (2008). Grazed riparian management and stream channel response in southeastern Minnesota (USA) streams. *Environmental Management*, *42*(3), 377–390. https://doi.org/10.1007/s00267-008-9132-4

Maloney, K. O., Feminella, J. W., Mitchell, R. M., Miller, S. A., Mulholland, P. J., & Houser, J. N. (2008). Landuse legacies and small streams: identifying relationships between historical land use and contemporary stream conditions. *Journal of the North American Benthological Society*, *27*(2), 280–294. https://doi.org/10.1899/07-070.1

Mancini, L., Formichetti, P., Anselmo, A., Tancioni, L., Marchini, S., & Sorace, A. (2005). Biological quality of running waters in protected areas: the influence of size and land use. *Biodiversity and Conservation*, *14*(2), 351–364. https://doi.org/10.1007/s10531-004-5055-8

Marczak, L. B., Sakamaki, T., Turvey, S. L., Deguise, I., Wood, S. L. R., & Richardson, J. S. (2010). Are forested buffers an effective conservation strategy for riparian fauna? An assessment using meta-analysis. *Ecological Applications*, *20*(1), 126–134. https://doi.org/10.1890/08-2064.1

Maret, T. R., MacCoy, D. E., & Carlisle, D. M. (2008). Long-term water quality and biological responses to multiple best management practices in Rock Creek, Idaho. *Journal of the American Water Resources Association*, *44*(5), 1248–1269. https://doi.org/10.1111/j.1752-1688.2008.00221.x

Marshall, D. W., Fayram, A. H., Panuska, J. C., Baumann, J., & Hennessy, J. (2008). Positive Effects of Agricultural Land Use Changes on Coldwater Fish Communities in Southwest Wisconsin Streams. *North American Journal of Fisheries Management*, *28*(3), 944–953. https://doi.org/10.1577/M06-139.1

Marty, J. (2005). Effects of cattle grazing on diversity in ephemeral wetlands. *Conservation Biology*, *19*(5), 1626–1632. https://doi.org/10.1111/j.1523-1739.2005.00198.x

Marty, J. T. (2015). Loss of biodiversity and hydrologic function in seasonal wetlands persists over 10 years of livestock grazing removal. *Restoration Ecology*, *23*(5), 548–554. https://doi.org/10.1111/rec.12226

McCaffery, M., Switalski, T. A., & Eby, L. (2007). Effects of road decommissioning on stream habitat characteristics in the South Fork Flathead River, Montana. *Transactions of the American Fisheries Society*, *136*(3), 553–561. https://doi.org/10.1577/T06-134.1

Meli, P., Rey Benayas, J. M., Balvanera, P., & Martinez Ramos, M. (2014). Restoration Enhances Wetland Biodiversity and Ecosystem Service Supply, but Results Are Context-Dependent: A Meta-Analysis. *PLOS ONE*, *9*(4), e93507. https://doi.org/10.1371/journal.pone.0093507

Mero, T. O., Lontay, L., & Lengyel, S. (2015). Habitat management varying in space and time: the effects of grazing and fire management on marshland birds. *Journal of Ornithology*, *156*(3), 579–590. https://doi.org/10.1007/s10336-015-1202-9

Mester, B., Szalai, M., Mero, T. O., Puky, M., & Lengyel, S. (2015). Spatiotemporally variable management by grazing and burning increases marsh diversity and benefits amphibians: A field experiment. *Biological Conservation*, *192*, 237–246. https://doi.org/10.1016/j.biocon.2015.09.030

Naiman, R., & Decamps, H. (1997). The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics*, *28*, 621–658. https://doi.org/10.1146/annurev.ecolsys.28.1.621

Nerbonne, B. A., & Vondracek, B. (2001). Effects of Local Land Use on Physical Habitat, Benthic Macroinvertebrates, and Fish in the Whitewater River, Minnesota, USA. *Environmental Management*, *28*(1), 87–99. https://doi.org/10.1007/s002670010209

Nislow, K. H. (2005). Forest change and stream fish habitat: lessons from `Olde’ and New England. *Journal of Fish Biology*, *67*(B), 186–204. https://doi.org/10.1111/j.0022-1112.2005.00913.x

Ogden, F. L., Crouch, T. D., Stallard, R. F., & Hall, J. S. (2013). Effect of land cover and use on dry season river runoff, runoff efficiency, and peak storm runoff in the seasonal tropics of Central Panama. *Water Resources Research*, *49*(12), 8443–8462. https://doi.org/10.1002/2013WR013956

Olson, D. H., Leirness, J. B., Cunningham, P. G., & Steel, E. A. (2014). Riparian buffers and forest thinning: Effects on headwater vertebrates 10 years after thinning. *Forest Ecology and Management*, *321*(SI), 81–93. https://doi.org/10.1016/j.foreco.2013.06.013

Orzetti, L. L., Jones, R. C., & Murphy, R. F. (2010). Stream Condition in Piedmont Streams with Restored Riparian Buffers in the Chesapeake Bay Watershed. *JAWRA Journal of the American Water Resources Association*, *46*(3), 473–485. https://doi.org/10.1111/j.1752-1688.2009.00414.x

Palmer, M., Allan, J. D., Meyer, J., & Bernhardt, E. S. (2007). River Restoration in the Twenty-First Century: Data and Experiential Knowledge to Inform Future Efforts. *Restoration Ecology*, *15*(3), 472–481. https://doi.org/10.1111/j.1526-100X.2007.00243.x

Pearce, N. J. T., & Yates, A. G. (2015). Agricultural Best Management Practice Abundance and Location does not Influence Stream Ecosystem Function or Water Quality in the Summer Season. *Water*, *7*(12), 6861–6876. https://doi.org/10.3390/w7126661

Pelinson, R. M., Garey, M. V., & Rossa-Feres, D. C. (2016). Effects of Grazing Management and Cattle on Aquatic Habitat Use by the Anuran Pseudopaludicola mystacalis in Agro-Savannah Landscapes. *PLOS ONE*, *11*(9), e0163094. https://doi.org/10.1371/journal.pone.0163094

Pilliod, D., Bury, R., Hyde, E., Pearl, C., & Corn, P. (2003). Fire and amphibians in North America. *Forest Ecology and Management*, *178*(1–2), 163–181. https://doi.org/10.1016/S0378-1127(03)00060-4

Quinn, J. M., Croker, G. F., Smith, B. J., & Bellingham, M. A. (2009). Integrated catchment management effects on flow, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research*, *43*(3), 775–802. https://doi.org/10.1080/00288330909510041

Ranganath, S. C., Hession, W. C., & Wynn, T. M. (2009). Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. *Journal of Soil and Water Conservation*, *64*(1), 33–42. https://doi.org/10.2489/jswc.64.1.33

Raymond, K. L., & Vondracek, B. (2011). Relationships among rotational and conventional grazing systems, stream channels, and macroinvertebrates. *Hydrobiologia*, *669*(1), 105–117. https://doi.org/10.1007/s10750-011-0653-0

Richardson, J. S. (2008). Aquatic arthropods and forestry: effects of large-scale land use on aquatic systems in Nearctic temperate regions. *The Canadian Entomologist*, *140*(4), 495–509. https://doi.org/10.4039/n07-LS04

Rieman, B., & Clayton, J. (1997). Wildfire and Native Fish: Issues of Forest Health and Conservation of Sensitive Species. *Fisheries*, *22*(11), 6–15. https://doi.org/10.1577/1548-8446(1997)022<0006:WANFIO>2.0.CO;2

Rieman, B., Lee, D., Burns, D., Gresswell, R., Young, M., Stowell, R., … Howell, P. (2003). Status of native fishes in the western United States and issues for fire and fuels management. *Forest Ecology and Management*, *178*(1–2), 197–211. https://doi.org/10.1016/S0378-1127(03)00062-8

Rodríguez-Jorquera, I. A., Siroski, P., Espejo, W., Nimptsch, J., Choueri, P. G., Choueri, R. B., … Toor, G. S. (2017). Latin American Protected Areas: Protected from Chemical Pollution? *Integrated Environmental Assessment and Management*, *13*(2), 360–370. https://doi.org/10.1002/ieam.1839

Roni, P., Beechie, T. J., Bilby, R. E., Leonetti, F. E., Pollock, M. M., & Pess, G. R. (2002). A Review of Stream Restoration Techniques and a Hierarchical Strategy for Prioritizing Restoration in Pacific Northwest Watersheds. *North American Journal of Fisheries Management*, *22*(1), 1–20. https://doi.org/10.1577/1548-8675(2002)022<0001:AROSRT>2.0.CO;2

Roni, P., Hanson, K., & Beechie, T. (2008). Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. *North American Journal of Fisheries Management*, *28*(3), 856–890. https://doi.org/10.1577/M06-169.1

Rood, S. B., Braatne, J. H., & Hughes, F. M. R. (2003). Ecophysiology of riparian cottonwoods: stream flow dependency, water relations and restoration. *Tree Physiology*, *23*(16), 1113–1124. https://doi.org/10.1093/treephys/23.16.1113

Rood, S. B., Samuelson, G. M., Braatne, J. H., Gourley, C. R., Hughes, F. M., & Mahoney, J. M. (2005). Managing river flows to restore floodplain forests. *Frontiers in Ecology and the Environment*, *3*(4), 193–201. https://doi.org/10.1890/1540-9295(2005)003[0193:MRFTRF]2.0.CO;2

Rood, S., Gourley, C., Ammon, E., Heki, L., Klotz, J., Morrison, M., … Wagner, P. (2003). Flows for floodplain forests: A successful riparian restoration. *BioScience*, *53*(7), 647–656. https://doi.org/10.1641/0006-3568(2003)053[0647:FFFFAS]2.0.CO;2

Ruhi, A., San Sebastian, O., Feo, C., Franch, M., Gascon, S., Richter-Boix, A., … Llorente, G. (2012). Man-made Mediterranean temporary ponds as a tool for amphibian conservation. *Annales de Limnologie - International Journal of Limnology*, *48*(1), 81–93. https://doi.org/10.1051/limn/2011059

Sallenave, R. M., & Day, K. E. (1991). Secondary production of benthic stream invertebrates in agricultural watersheds with different land management practices. *Chemosphere*, *23*(1), 57–76. https://doi.org/10.1016/0045-6535(91)90116-U

Schaich, H., & Barthelmes, B. (2012). Management of grasslands in rewetted floodplains: effects of grazing and cutting on vegetation development. *Tuexenia*, (32), 207–231.

Scott, M. C. (2006). Winners and losers among stream fishes in relation to land use legacies and urban development in the southeastern US. *Biological Conservation*, *127*(3), 301–309. https://doi.org/10.1016/j.biocon.2005.07.020

Sebastián-González, E., & Green, A. J. (2016). Reduction of avian diversity in created versus natural and restored wetlands. *Ecography*, *39*(12), 1176–1184. https://doi.org/10.1111/ecog.01736

Sievers, M., Hale, R., & Morrongiello, J. R. (2017). Do trout respond to riparian change? A meta-analysis with implications for restoration and management. *Freshwater Biology*, *62*(3), 445–457. https://doi.org/10.1111/fwb.12888

Smith, R. K., Meredith, H., & Sutherland, W. J. (2017). Amphibian Conservation. In W. J. Sutherland, L. V. Dicks, N. Ockendon, & R. K. Smith (Eds.), *What Works in Conservation 2017*. Cambridge, UK: Open Book Publishers.

Smith, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research*, *10*(2), 126–139. https://doi.org/10.1065/espr2002.12.142

Sovell, L. A., Vondracek, B., Frost, J. A., & Mumford, K. G. (2000). Impacts of rotational grazing and riparian buffers on physicochemical and biological characteristics of southeastern Minnesota, USA, streams. *Environmental Management*, *26*(6), 629–641. https://doi.org/10.1007/s002670010121

Spadafora, E., Leslie, A. W., Culler, L. E., Smith, R. F., Staver, K. W., & Lamp, W. O. (2016). Macroinvertebrate community convergence between natural, rehabilitated, and created wetlands. *Restoration Ecology*, *24*(4), 463–470. https://doi.org/10.1111/rec.12352

Sutherland, W. J., Dicks, L. V., Ockendon, N., & Smith, R. K. (2017). What Works in Conservation 2017. Open Book Publishers. Retrieved from http://dx.doi.org/10.11647/OBP.0109

Switalski, T., Bissonette, J., DeLuca, T., Luce, C., & Madej, M. (2004). Benefits and impacts of road removal. *Frontiers in Ecology and the Environment*, *2*(1), 21–28. https://doi.org/10.2307/3868291

Teels, B. M., Rewa, C. A., & Myers, J. (2006). Aquatic condition response to riparian buffer establishment. *Wildlife Society Bulletin*, *34*(4), 927–935. https://doi.org/10.2193/0091-7648(2006)34[927:ACRTRB]2.0.CO;2

Thieme, M. L., Sindorf, N., Higgins, J., Abell, R., Takats, J. A., Naidoo, R., & Barnett, A. (2016). Freshwater conservation potential of protected areas in the Tennessee and Cumberland River Basins, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *26*(1), 60–77. https://doi.org/10.1002/aqc.2644

Thiere, G., Milenkovski, S., Lindgren, P.-E., Sahlén, G., Berglund, O., & Weisner, S. E. B. (2009). Wetland creation in agricultural landscapes: Biodiversity benefits on local and regional scales. *Biological Conservation*, *142*(5), 964–973. https://doi.org/10.1016/j.biocon.2009.01.006

Tierney, D., Kelly-Quinn, M., & Bracken, J. (1998). The faunal communities of upland streams in the eastern region of Ireland with reference to afforestation impacts. *Hydrobiologia*, *389*(1–3), 115–130. https://doi.org/10.1023/A:1003510016292

Walls, S. C., Waddle, J. H., & Faulkner, S. P. (2014). Wetland Reserve Program Enhances Site Occupancy and Species Richness in Assemblages of Anuran Amphibians in the Mississippi Alluvial Valley, USA. *Wetlands*, *34*(1), 197–207. https://doi.org/10.1007/s13157-013-0498-6

Walsh, C. J., Waller, K. A., Gehling, J., & Mac Nally, R. (2007). Riverine invertebrate assemblages are degraded more by catchment urbanisation than by riparian deforestation. *Freshwater Biology*, *52*(3), 574–587. https://doi.org/10.1111/j.1365-2427.2006.01706.x

Wan, S., Qin, P., Li, Y., & Liu, X. (2001). Wetland creation for rare waterfowl conservation: A project designed according to the principles of ecological succession. *Ecological Engineering*, *18*(1), 115–120. https://doi.org/10.1016/S0925-8574(01)00062-3

Wang, L., Lyons, J., & Kanehl, P. (2002). Effects of watershed best management practices on habitat and fish in Wisconsin streams. *Journal of the American Water Resources Association*, *38*(3), 663–680. https://doi.org/10.1111/j.1752-1688.2002.tb00988.x

Wang, L., Lyons, J., & Kanehl, P. (2006). Habitat and fish responses to multiple agricultural best management practices in a warm water stream. *Journal of the American Water Resources Association*, *42*(4), 1047–1062. https://doi.org/10.1111/j.1752-1688.2006.tb04513.x

Whigham, D. (1999). Ecological issues related to wetland preservation, restoration, creation and assessment. *Science of The Total Environment*, *240*(1–3), 31–40. https://doi.org/10.1016/S0048-9697(99)00321-6

Whitney, J. E., Gido, K. B., Pilger, T. J., Propst, D. L., & Turner, T. F. (2016). Metapopulation analysis indicates native and non-native fishes respond differently to effects of wildfire on desert streams. *Ecology of Freshwater Fish*, *25*(3), 376–392. https://doi.org/10.1111/eff.12217

Yates, A. G., Bailey, R. C., & Schwindt, J. A. (2007). Effectiveness of best management practices in improving stream ecosystem quality. *Hydrobiologia*, *583*, 331–344. https://doi.org/10.1007/s10750-007-0619-4

Yeung, A. C. Y., Lecerf, A., & Richardson, J. S. (2017). Assessing the long-term ecological effects of riparian management practices on headwater streams in a coastal temperate rainforest. *Forest Ecology and Management*, *384*, 100–109. https://doi.org/10.1016/j.foreco.2016.10.044

Zedler, J. B. (2003). Wetlands at your service: reducing impacts of agriculture at the watershed scale. *Frontiers in Ecology and the Environment*, *1*(2), 65–72. https://doi.org/10.1890/1540-9295(2003)001[0065:WAYSRI]2.0.CO;2

Land management interventions and flow changes

To what extent can source water protection activities affect baseflow, groundwater quantity, and flooding through changes in infiltration and runoff?

Targeted land protection and revegetation

In general, protecting intact, native terrestrial ecosystems and preserving ecosystem integrity are key to maintaining provisioning ecosystem services and the hydrologic status quo (Dudley et al. 2003; Bruijnzeel 2016; Ellison et al. 2017). However, the interplay of changing land use and climate, and complex ecohydrologic responses to those changes at various scales, creates significant uncertainty regarding the provision of water quantity services. Ecosystem restoration via revegetation is even more complex than protection and may result in ambiguous or unintended consequences such as reduced water yield or baseflow.

Scientists argue that full consideration of the roles of soil and surface degradation are oftentimes missing from discussions of hydrologic impacts (Bruijnzeel 2004). Other factors that affect the effectiveness of land protection and restoration activities on water quantity include ecosystem type, location (e.g. upstream vs. downstream), watershed size, timing, context (e.g. soil characteristics, moisture status), and interactions with other land use types and landscape linkages.

Forest protection and restoration/afforestation

A wealth of literature on forest hydrology in different geographic contexts exists, and the impacts of primary forest deforestation, afforestation and/or reforestation on hydrology, at local and catchment scales, are reasonably well-documented and reviewed. Several comprehensive review articles and reports summarize ongoing discussions around the impacts of forests on water quantity, including streamflow, flooding, and dry-season flows (Smakhtin 2001; Dudley et al. 2003; Andréassian 2004; Bruijnzeel 2004; Scott et al. 2005; Brown et al. 2005, 2013; Farley et al. 2005; Ilstedt et al. 2007; Bradshaw et al. 2007; Ellison et al. 2012, 2017; Beck et al. 2013; Garcia-Chevesich et al. 2017; Filoso et al. 2017). For instance, **Ellison et al. (2017)** provided an extensive review of research on how forests regulate water supplies, namely through: fog and cloud water capture, infiltration and groundwater recharge, and flood moderation/mitigation. Below we summarize the major gaps identified using a similar categorization:

- <u>Cloud and fog capture</u>: Several studies were found regarding the hydrometeorology and hydrology of cloud forests, highlighting (1) the importance of tropical montane cloud forest conservation both in ecological and hydrologic terms, (2) the need for site-specific information on hydrologic functioning at spatial scales and across cloud forest types, and (3) the lack of understanding on the hydrologic impacts of climate and land-use change, and their interactions (Bruijnzeel 2004; Bruijnzeel et al. 2010, 2011; Muñoz-Villers et al. 2015; Ellison et al. 2017). Evidence also points out that cloud forest conversion to agriculture, pasture, or agroforestry showed mixed results in terms of streamflow/downstream water yield, and that the direction and magnitude of changes should be examined in a context- and site-specific manner. For example, **Bruijnzeel et al. (2010)** presented cases showing that converting cloud forests to pasture can have either positive (defined as increased flow, as in eastern Mexico), negative (defined as decreased flows, as in Pacific Northwest of US) or neutral (as under particularly wet conditions in Costa Rica) hydrologic impacts. For cloud forest regeneration and restoration, our review did not find documented studies on water quantity impacts. In a recent systematic review about the impacts of forest restoration on water yields, the authors recorded 5 cases from Central America showing positive water yield results from forest expansion (suggesting implications for cloud forest restoration), but also emphasized the general lack of information in the humid tropics (Filoso et al. 2017). Variation among cloud forest types also suggests caution when extrapolating hydrologic functions of one forest type to another.

- Infiltration, baseflow, and groundwater recharge: A variety of case studies documented changes in *annual* streamflow following forest clearing, forest thinning (as a forestry practice), afforestation (mostly plantations) or reforestation, adopting an underlying paradigm of trade-off between forest cover and streamflow. Some scientists argue that this paradigm is biased and factors such as soil characteristics are insufficiently considered (Bruijnzeel 2004; Scott et al. 2005; Ellison et al. 2012).[8] Without a better understanding of what happens underground, simply planting trees (especially using plantations as an afforestation approach) won't necessarily generate beneficial outcomes (Garcia-Chevesich et al. 2017). The following gaps were identified: (1) evidence focuses on total annual streamflow or water yield, but studies on dry-season baseflows and groundwater recharge dynamics are lacking; (2) long-term and large scale relationships are missing; (3) studies are skewed towards young, fast-growing plantations; (4) evidence from the tropics and (semi-) arid tropics is scarce; (5) how reforestation or afforestation may rebuild soil infiltration capacity is poorly documented (Bruijnzeel 2016; Ellison et al. 2017).
- Flood mitigation: A smaller body of literature exists around the role of forests in flood risk reduction (Bradshaw et al. 2007; van Dijk et al. 2009; Wahren et al. 2012; van Noordwijk et al. 2017a, 2017b), with a greater focus on riverine flooding and urban stormwater management than watershed impacts. While it is commonly assumed that deforestation and removal of

[8] One prominent school of thought, for example, demonstrates that the trade-off should be between positive impacts (e.g. increases in water yield or reduction in runoff peak) due to enhanced soil water recharge and infiltration on the one hand, and negative impacts (e.g. reduced baseflow or decreases in soil water reserves) on the other hand due to the higher water use of trees compared to crops, pasture, or scrubs. Put simply, there is a balance between soil infiltration and forest water use (evapotranspiration) (Bruijnzeel 2016). Soil degradation due to land use change (e.g. conversion to pastures or agricultural lands) results in reduced soil infiltration and water retention capacity, and consequently reduce groundwater reserves critical for maintaining dry-season baseflows. Without a better understanding of what happens underground, simply planting trees (especially using plantations as an afforestation approach) won't necessarily generate beneficial outcomes. Unless infiltration capacity is restored, reforestation may even reduce water yield or flow, as shown in many case studies. In other words, while reforestation may increase dry season baseflows via restored soil infiltration and water holding capacity, such effects should not be taken as universal. Furthermore, hydrologic change due to soil degradation may be slow, and largely underrated in case studies that measure various other hydrologic parameters. How reforestation or afforestation may rebuild the infiltration capacity is also poorly documented (Bruijnzeel 2016). One study in the seasonally dry tropics proposed an optimum tree cover theory, suggesting that intermediate tree densities on degraded lands may maximize groundwater recharge via improved soil hydraulic properties (Ilstedt et al. 2016).

forest may increase flood risks, and restoring forests or planting trees could mitigate floods, the available literature indicates such statements are debatable. In addition to the aforementioned factors that affect forest hydrologic function (e.g. changes to soil infiltration and conductivity), event-specific conditions also matter for flood risk reduction benefits. For example, antecedent soil moisture conditions can significantly affect the timing and magnitude of runoff with implications for the responsiveness to upstream land use conditions (Wahren et al. 2012; Dadson et al. 2017). Climate change induced changes in precipitation patterns, including rainfall intensity and locations, may also limit forest flood reduction potential when available land for afforestation or reforestation is limited. Tree species also need to be taken into consideration when examining the role of reforestation/afforestation on flood mitigation. Ellison et al. (2017) suggests that different tree species ought to be chosen based on the specific geography: in water-rich areas, fast growing, high water-consuming tree species might reduce, but not eliminate, flood risks; whereas in water-limited areas, slow growing, low water-consuming tree species might increase infiltration and help moderate flooding. To summarize, these factors illustrate that the forest-flood relationship is another highly debated and complex topic, and deserves its own literature review.

An overarching set of messages from these reviews is that (1) temporal and spatial scales matter, but long-term and large-scale studies are relatively scarce, (2) conclusions drawn from one practice (e.g. reforestation) cannot necessarily be applied to another (e.g. agroforestry, forest plantations), (3) what happens belowground is as if not more important than what happens aboveground ('not seeing the soil for the trees,' per Bruijnzeel 2004), and (4) limited understanding on the hydrologic impacts of the interactions between climate and land-use change.

Grassland protection and restoration

The literature on the hydrologic impacts of grassland protection and restoration is sparse, though a small number of individual grassland types have been well-studied. Relatively speaking, there is a body of high quality literature on the hydrology of Andean páramos in South America, both in natural conditions and under anthropogenic land use changes (Buytaert et al. 2006, 2007; Celleri & Feyen 2009; Crespo et al. 2011; Mosquera et al. 2015; Guzmán et al. 2015; Ochoa-Tocachi et al. 2016). These studies and reviews emphasize the importance of protecting páramos as they play a key role in hydrology, and they identify the need to better understand hydrologic impacts under combinations of climate change and human disturbances. Studies from the Argentine pampas focused on grassland conversion and also shed light on the value of protecting native grasslands, suggesting that afforestation on grasslands may compromise soil quality and water availability in predictable ways based on water use, climate, and soil texture (Jobbágy and Jackson 2004; Nosetto et al. 2005; Nosetto et al. 2008).

If we expand our search to include rangelands (typically located in dryland biomes such as savannas, grasslands, shrublands, and deserts), we find a wealth of literature looking at water quantity as a supporting service of rangelands (Havstad et al. 2007) and the hydrologic implications of rangeland degradation (Wilcox 2007), woody plant encroachment (Wilcox et al. 2005; Wilcox & Thurow 2006; Wilcox & Huang 2010), shrub control (Wilcox 2002; Afinowicz et al. 2005), and prescribed fire and aspen removal (Flerchinger et al. 2016). While these studies and reviews are mostly limited to the United

States, and Texas in particular, they identify similar knowledge gaps as the páramo ecosystem, such as limited long-term data, the difficulty of scaling up from small scale data, and the need to evaluate impacts from multiple spatial scales, including watershed and basin perspectives (e.g. Wilcox et al. 2008; Berg et al. 2016).

Regarding restoration, we identified only one recent case study examining the impact of large-scale grassland restoration (headwater region of China's Yangtze River). That study found that grassland restoration improves local soil water conditions but reduces streamflow (via increased evapotranspiration and enhanced soil water retention) (Li et al. 2017).

We note here the importance of terminology regarding the scope of literature reviewed. Depending on the classification of grassland and wetland (for example, "prairie wetland" or "wet meadow"), more literature on wetland restoration may be relevant. Additional search review could use more detailed search terms to include specific grassland ecosystems and biomes, if necessary.

Wetland protection and restoration

In our review, we excluded wetlands created or installed for urban stormwater management due to the more distant relationship to source water protection. Overall, we found fewer studies on the hydrologic function of wetlands as compared to water quality mediation (especially for treatment wetlands). In particular, empirical studies of hydrologic impacts resulting from wetland restoration are missing. Geographically, studies on the impacts of wetland protection and restoration on water quantity outcomes are still largely focused on North America and Western Europe. These studies also demonstrate variability in outcomes due in part to the variety of wetland types and contexts.

It is clear from the relatively abundant literature that there is no simple and generalizable relationship between wetland types and their hydrologic function (e.g. Zedler 2003; Bullock & Acreman 2003). Bullock & Acreman (2003) reviewed 169 quantitative studies worldwide (however, with more than 50% of cases from North America, and 20% from Europe) and provided a collation of scientific evidence among hydrologic measures and wetland types. The review, considered to be comprehensive at the time, challenged the conventional generalization that wetlands *always* reduce floods, enhance groundwater recharge, and/or regulate river flows, while pointing out greater consistency depending on the on wetland types. For example, they found that 41% of relevant studies suggested headwater wetlands enhance flooding, while 80% of studies showed floodplain wetlands reduce flooding (Bullock & Acreman 2003). They also revealed a dearth of studies of Asian and South American wetland hydrology. We identified several more recent case studies addressing the hydrologic role of different wetland types (e.g. Grayson et al. 2010; Morley et al. 2011; McLaughlin & Cohen 2013; Mosquera et al. 2015; Li & Shi 2015). It could be informative to update the Bullock & Acreman (2003) dataset and analyze trends and gaps via a more systematic approach.

In another review, Acreman & Holden (2013) specifically focused on upland rain-fed wetlands and floodplain river-fed wetlands, and demonstrated differences in flood function within and between wetland types. While showing some degree of consistency in the role of upstream and downstream wetlands, this review indicated that the influences of wetlands on floods can be both positive and negative—largely determined by landscape location and configuration, topography, soil characteristics and moisture status, and land management. The authors also noted the limited effect, if any, wetlands have on major catastrophic floods (Acreman & Holden 2013).

Compared to wetland protection, empirical studies on the hydrologic impacts of wetland restoration are largely missing in our review. Restoration is even more complex since (1) the effectiveness of restoration is influenced by the antecedent hydrologic processes and conditions that resulted in wetland degradation and destruction, (2) restorable hydrologic services differ by wetland types (Zedler 2003; Zedler & Kercher 2005; Acreman et al. 2007), and (3) wetland-landscape linkage is another key aspect to consider when assessing the water quantity impacts of wetland restoration, but have rarely been addressed in restoration projects (Bedford 1999; Verhoeven et al. 2008). Zedler (2003) reviewed literature on the potential flood abatement benefits of wetland restoration in the Midwest United States, but the studies cited were largely theoretical. Our review also found a variety of modeling studies that predicted the hydrologic outcomes of wetland restoration (e.g. Zhang & Mitsch 2005; Boswell & Olyphant 2007; Staes et al. 2009; Yang et al. 2010; Wang et al. 2010; Martinez-Martinez et al. 2014; Javaheri & Babbar-Sebens 2014), but they are almost exclusively focused in North America, and to a lesser extent, Europe. Overall, despite existing modeling efforts, this review found a lack of empirical evidence in regard to wetland restoration and its hydrologic benefits.

Riparian protection and restoration

The range of ecosystem and socioeconomic functions and services that riparian zones provide are well documented (Gregory et al. 1991; Beechie et al. 2013; Capon et al. 2013). Protecting undisturbed riparian systems would rarely have negative outcomes on water quantity, especially in regard to maintaining vital spatial connectivity and environmental heterogeneity (Capon et al. 2013; Acreman et al. 2014). However, the focus of most literature reviewed related to riparian zones (both protection and restoration) is on theoretical discussion instead of empirical evaluation. While there is a vast literature on riparian zones in the context of river restoration and environmental flows, we excluded those papers from our review since they are typically driven by objectives other than source water protection. Excluding buffer strips or filter strips that are designed for stormwater management within urban landscapes or water quality control within agricultural lands—which are reviewed under *Agricultural BMPs* below—our review found few empirical studies that consider the hydrology of riparian zones (McGlynn & McDonnell 2003; Rassam 2005; Angier et al. 2005; Mallik et al. 2011; Kolka et al. 2011; von Freyberg et al. 2014; Inserillo et al. 2017).

Many studies of riparian restoration or riparian forest expansion impacts on hydrology have looked at restoring native riparian vegetation as a way to increase water yield—not necessarily within the context source water protection outcomes. Martinet et al. (2009) monitored the influence of understory vegetation removal as a restoration measure on riparian groundwater fluctuations along Rio Grande in central New Mexico. Wine & Zou (2012) measured changes in long-term streamflow in relation to riparian gallery forest expansion. A series of studies in the western United States examined the impact of non-native species control as one riparian restoration measure to increase water yield, and found mixed and context-specific outcomes (Shafroth et al. 2005, 2008; Moore & Owens 2012). Salemi et al. (2012) reviewed the negative effect that riparian forests, forest plantations or regeneration have on water yield on a daily to annual basis using a small sample size. Other modeling studies predict the potential benefit of riparian forest restoration to flood risk reduction in the UK (e.g. Dixon et al. 2016).

A number of reviews call for improved documentation and monitoring to evaluate successful restoration programs (Palmer et al. 2005, 2007, 2008; Seavy et al. 2009; Capon et al. 2013; Perry et al. 2015). While we can anticipate context-dependency of hydrologic impacts from riparian restoration activities, the lack of empirical evidence or long-term monitoring data is indeed a major gap. Extensive land use/cover change and interactions with other drivers at landscape scales may also "dilute" the benefits that riparian zones can provide.

Agricultural BMPs

Hydrologic services, such as increasing baseflows, groundwater recharge, or flooding mitigation, are rarely the primary focus of implementing agricultural best management practices (BMPs). In our review, we found that the impacts of conservation agriculture (including tillage, cover crops/residue retention, and crop rotation) and other agricultural best management practices (contour farming, terracing, etc.) on hydrologic *processes* are relatively well-documented, with modest geographic representation (with the exception of Latin America, where we did not find studies on conservation agriculture). The direction of impacts vary by crop systems, climate and soil conditions, scale, and other context-specific factors (e.g. Bjorneberg et al. 1996; Chow et al. 1999; Kramer et al. 1999; McGarry et al. 2000; Shipitalo et al. 2000; Leys et al. 2007; Valentin et al. 2008; Araya et al. 2012, 2015; Baumhardt et al. 2017). The majority of these studies measured changes in infiltration capacity or runoff volume, which are used as parameters to evaluate the effectiveness of BMPs on water quality rather than water quantity. For example, Rittenburg et al. (2015) reviewed published studies on the impacts and effectiveness of BMPs on hydrologic flow paths (including overland flow velocity and infiltration) for different scales and land/soil types in relation to pollution management. Liu et al. (2017) reviewed current knowledge about the short- and long-term effectiveness and efficiencies of BMPs in both agricultural and urban areas on water quality and quantity, and identified gaps, but again empirical evidence on water quantity summarized in this review is limited to reduced runoff volumes. In general, the literature suggests that conservation agriculture could increase infiltration and reduce runoff. However, empirical evidence of consequent water quantity impacts (flood reduction, baseflow maintenance, or groundwater recharge) at larger scales is largely missing (Smakhtin 2001; O'Connell et al. 2007).

Our review found very limited literature on direct linkages between agricultural BMPs and water quantity outcomes.[9] Overall, there remains a major knowledge gap in empirical evidence as to how and

---

[9] For example, Potter (1991) showed a decrease in flood peaks and an increase in the contribution of winter/spring snowmelt due to soil conservation practices in Wisconsin, US; Daniel (1999) compared the impact of tillage practices on shallow groundwater recharge in Oklahoma, US, and showed no-till limited the amount of surface water percolating into the water table by 10-fold compared to moldboard or native grass; O'Connell et al. (2007) reviewed literature in the UK on the relationship between agricultural land use and flooding, but found little evidence that local scale changes in runoff generation will propagate downstream to create impacts at the catchment scale; Deasy et al. (2014) empirically compared the impacts of different BMPs on local-scaled size, duration and timing of flood peaks in UK, and suggested that, while practices can affect local-scale runoff generation, treatment effects were not significant and the benefits to mitigating downstream flood risk remain

to what extent changes in local-scale soil hydrologic properties influence water quantity outcomes in watersheds at different scales. Similar to conservation agriculture, impacts on water quantity resulting from agroforestry are context-specific and scale-dependent. Variations in agroforestry systems (e.g. coffee agroforests in Latin America, agroforest parkland in Africa, or eucalyptus-pasture agroforestry in Australia) add yet another layer of complexity for generalizing linkages between agroforestry and water quantity outcomes. For example, studies on tropical coffee and cocoa agroforests in Latin America showed improvement in soil properties and reduction in surface runoff (Benegas et al. 2014), as well as potential impacts of streamflow regulation and aquifer recharge (Gomez-Delgado et al. 2011). In dryland landscapes in Africa, agroforestry can also improve infiltration and preferential flow (Bargués Tobella et al. 2014). In Australia, eucalyptus-pasture agroforestry and alley farming were adopted to lower groundwater table and prevent stream and land salinization (which are distinct from source water protection objectives) (Bari & Schofield 1991; Noorduijn et al. 2010). For the last example, it also reveals that we need to consider the context and purpose of implementing agroforestry systems when evaluating water quantity impacts, as they may not necessarily be "beneficial" from a source water protection standpoint. As with other agricultural BMPs, empirical evidence that directly addresses agroforestry and downstream or larger-scale water quantity impacts via changes in runoff or infiltration is again missing.

It is noteworthy that the impacts of agroforestry on hydrology may resemble those in forests and forested catchments, such as (1) the trade-offs between positive impacts (e.g. increases in baseflow or reduction in runoff peak) due to enhanced soil water recharge and infiltration on the one hand, and negative impacts (e.g. reduced baseflow or decreases in soil water reserves) on the other hand due to the higher water use of trees and evapotranspiration, or (2) the hydrologic responses to rainfall at the catchment level (Rodríguez-Blanco et al. 2012; Palleiro et al. 2014) (discussed in the earlier section of this document).

In theory, grass and forest buffer strips in agricultural landscapes can affect water quantity via reduced runoff and/or increased infiltration and soil water storage. As noted earlier, the studies on buffer strips reviewed focus primarily on water quality outcomes (e.g. erosion control and sediment transport) (Udawatta et al. 2002; McKergow et al. 2004; Anderson et al. 2009; Asam et al. 2012; Hernandez-Santana et al. 2013). As a result, changes in runoff or infiltration generally have not been linked to impacts on flow regime or other water quantity provisions of interest further downstream. One modeling study from the UK demonstrated the potential flood reduction benefits of riparian buffers (Gao et al. 2016).

Our review found a small number of modeling studies related to groundwater recharge. Two studies in India demonstrated that agricultural BMPs, including *in situ* rainwater conservation, irrigation management and modifying rice-wheat areas, could stabilize the water table and sustain groundwater recharge (Ambast et al. 2006; Dourte et al. 2014). But these studies also questioned the potential effectiveness of conservation agriculture in rice-wheat systems for reducing groundwater depletion (Balwinder-Singh et al. 2015). Another study modeled groundwater recharge under various agricultural

---

theoretical; and Ghazavi et al. (2008) evaluated hedgerow buffer strips and their influence on soil-water dynamics and groundwater recharge at the local scale in France.

BMP implementation scenarios in the U.S. Midwest and estimated an increase in groundwater recharge between 23 and 36% (Legge et al. 2013).

For flood reduction, while a number of studies pointed out such potential due to decreased surface or overland runoff volumes, it is also clear that changes at the plot- or field-scale will not necessarily lead to a reduction in flooding downstream. It is critical to evaluate the effect of agricultural BMPs on flood peaks, in terms of their size, duration and timing (O'Connell et al. 2007; Deasy et al. 2014).

Ranching BMPs

Ranching BMPs may affect water quantity at the watershed scale by altering vegetation and soil properties. Similar to agricultural BMPs, empirical evidence on the impacts of ranching BMPs is heavily skewed towards soil hydrologic properties (changes in infiltration and runoff) or water quality (Gilley et al. 1996; Webber et al. 2010; Teague et al. 2011; Pilon et al. 2017), and studies evaluating watershed-scale hydrologic impacts are lacking. We found one modeling study that simulated the long-term wet and dry season hydrologic impacts of different grazing management practices in a watershed in north central Texas and identified water conservation and flood risk reduction benefits resulting from multi-paddock grazing (Park et al. 2017). Another case study, while not directly focusing on the impact of ranching BMPs, evaluated long-term trends in streamflow from large rangeland watersheds in central Texas, and suggested that improved range conditions may reduce stormflows via improving soil infiltration (Wilcox et al. 2008). However, the authors also concluded that, for many semiarid rangelands, large-scale shrub clearing (not necessarily a BMP) in combination with sound range management will not lead to significant, if any, increases in streamflow.

Fire risk management.

Impacts of prescribed forest thinning—one type fire risk management—on runoff, baseflow or water supply can be inferred from existing knowledge on forest hydrology and thinning as a forestry practice (Bent 2001; Dung et al. 2011, 2012; Sun et al. 2015; Gokbulak et al. 2016). No empirical studies on thinning for the purpose of reducing wildfire risk and restoring forest health were identified through this review. One study noted that "any impact of forest thinning on water supply in Central Arizona is considered by the USDA to be an incidental benefit of management. Nevertheless, it is potentially important" (Simonit et al. 2015). Two modeling efforts simulated the impact of the Four Forest Restoration Initiative in Arizona (with the goal of mitigating fire risk through forest thinning) on the long-term water balance, and showed both risks and benefits in terms increasing annual water yield and altering reservoir sedimentation rates (Simonit et al. 2015; Moreno et al. 2016).

Road management

The literature (empirical studies, modeling, and review) on the impacts of roads on water quantity—including surface runoff and peak flows—is relatively limited, focused primarily on forested lands in North America (e.g. Thomas & Megahan 1998, 2001; Jones et al. 2000; Jones & Grant 2001; Foltz et al. 2009; Dymond et al. 2014; Wesemann et al. 2017). Studies on linkages between road management practices and hydrologic benefits are even more scarce and geographically limited (Luce 1997; Switalski et al. 2004; Kolka & Smidt 2004; Lloyd et al. 2013). Most evidence reviewed focused on soil hydraulic properties or changes in infiltration and runoff, but did not link these improvements to flow changes at watershed scale. One review paper considered different road removal techniques and identified critical knowledge gaps including a lack of evidence on the impact of road removal on natural stream and floodplain function restoration (Switalski et al. 2004). The importance of topographic and geologic contexts was also highlighted (Switalski et al. 2004; Al-Chokhachy et al. 2016).

Overall, within the context of forest hydrology and discussions on the role of soil, there seems to be a reasonably clear theoretical pathway between road management and the potential benefits to flood reduction (that is, in forested lands, activities that enhance soil water recharge and infiltration may lead to potential reduction in runoff peak, and flood mitigation further downstream). However, as with other source water protection activities, empirical evidence on the connections between changes in infiltration or runoff and downstream water quantity is largely unavailable.

Literature reviewed

Acreman M, Arthington AH, Colloff MJ, Couch C, Crossman ND, Dyer F, Overton I, Pollino CA, Stewardson MJ, Young W. 2014. Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. Frontiers in Ecology and the Environment **12**:466–473.

Acreman M, Holden J. 2013. How Wetlands Affect Floods. Wetlands **33**:773–786.

Acreman MC, Fisher J, Stratford CJ, Mould DJ, Mountford JO. 2007. Hydrologic science and wetland restoration: some case studies from Europe. Hydrology and Earth System Sciences Discussions **11**:158–169.

Afinowicz JD, Munster CL, Wilcox BP. 2005. Modeling Effects of Brush Management on the Rangeland Water Budget: Edwards Plateau, Texas. Journal of the American Water Resources Association **41**:181–193.

Al-Chokhachy R, Black TA, Thomas C, Luce CH, Rieman B, Cissel R, Carlson A, Hendrickson S, Archer EK, Kershner JL. 2016. Linkages between unpaved forest roads and streambed sediment: why context matters in directing road restoration: Unpaved roads and streambed sediment linkages. Restoration Ecology **24**:589–598.

Ambast SK, Tyagi NK, Raul SK. 2006. Management of declining groundwater in the Trans Indo-Gangetic Plain (India): Some options. Agricultural Water Management **82**:279–296.

Anderson SH, Udawatta RP, Seobi T, Garrett HE. 2009. Soil water content and infiltration in agroforestry buffer strips. Agroforestry Systems **75**:5–16.

Andréassian V. 2004. Waters and forests: from historical controversy to scientific debate. Journal of Hydrology **291**:1–27.

Angier J, McCarty G, Prestegaard K. 2005. Hydrology of a first-order riparian zone and stream, mid-Atlantic coastal plain, Maryland. JOURNAL OF HYDROLOGY **309**:149–166.

Araya T, Cornelis WM, Nyssen J, Govaerts B, Getnet F, Bauer H, Amare K, Raes D, Haile M, Deckers J. 2012. Medium-term effects of conservation agriculture based cropping systems for sustainable soil and water management and crop productivity in the Ethiopian highlands. FIELD CROPS RESEARCH **132**:53–62.

Araya T, Nyssen J, Govaerts B, Deckers J, Cornelis WM. 2015. Impacts of conservation agriculture-based farming systems on optimizing seasonal rainfall partitioning and productivity on vertisols in the Ethiopian drylands. SOIL & TILLAGE RESEARCH **148**:1–13.

Asam Z-Z, Kaila A, Nieminen M, Sarkkola S, O'Driscoll C, O'Connor M, Sana A, Rodgers M, Xiao L. 2012. Assessment of phosphorus retention efficiency of blanket peat buffer areas using a laboratory flume approach. ECOLOGICAL ENGINEERING **49**:160–169.

Ayars JE, Evans RG. 2015. Subsurface Drainage—What's Next?. Irrigation and Drainage **64**:378-92.

Balwinder-Singh, Humphreys E, Gaydon DS, Sudhir-Yadav. 2015. Options for increasing the productivity of the rice-wheat system of north west India while reducing groundwater depletion. Part 2. Is conservation agriculture the answer? FIELD CROPS RESEARCH **173**:81–94.

Bargués Tobella A, Reese H, Almaw A, Bayala J, Malmer A, Laudon H, Ilstedt U. 2014. The effect of trees on preferential flow and soil infiltrability in an agroforestry parkland in semiarid Burkina Faso. WATER RESOURCES RESEARCH **50**:3342–3354.

Bari M, Schofield N. 1991. Effects of Agroforestry-Pasture Associations on Groundwater Level and Salinity. AGROFORESTRY SYSTEMS **16**:13–31.

Baumhardt RL, Schwartz RC, Jones (retired) OR, Scanlon BR, Reedy RC, Marek G. 2017. Long-Term Conventional and No-Tillage Effects on Field Hydrology and Yields of a Dryland Crop Rotation. Soil Science Society of America Journal **81**:200.

Beck HE, Bruijnzeel LA, van Dijk AIJM, McVicar TR, Scatena FN, Schellekens J. 2013. The impact of forest regeneration on streamflow in 12 mesoscale humid tropical catchments. Hydrology and Earth System Sciences **17**:2613–2635.

Bedford B. 1999. Cumulative effects on wetland landscapes: Links to wetland restoration in the United States and southern Canada. WETLANDS **19**:775–788.

Beechie T, Richardson JS, Gurnell AM, Negishi J. 2013. Watershed Processes, Human Impacts, and Process-based Restoration. Page Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats. John Wiley & Sons, Chichester, West Sussex ; Hoboken, NJ.

Benegas L, Ilstedt U, Roupsard O, Jones J, Malmer A. 2014. Effects of trees on infiltrability and preferential flow in two contrasting agroecosystems in Central America. AGRICULTURE ECOSYSTEMS & ENVIRONMENT **183**:185–196.

Bent GC. 2001. Effects of forest-management activities on runoff components and ground-water recharge to Quabbin Reservoir, central Massachusetts. Forest Ecology and Management **143**:115–129.

Berg MD, Marcantonio F, Allison MA, McAlister J, Wilcox BP, Fox WE. 2016. Contrasting watershed-scale trends in runoff and sediment yield complicate rangeland water resources planning. Hydrology and Earth System Sciences **20**:2295–2307.

Bjorneberg DL, Kanwar R s., Melvin SW. 1996. Tillage and Crop Rotation Effects on Subsurface Drainage Response to Rainfall. Transactions of the ASAE **39**:2147–2154.

Boswell JS, Olyphant GA. 2007. Modeling the hydrologic response of groundwater dominated wetlands to transient boundary conditions: Implications for wetland restoration. Journal of Hydrology **332**:467–476.

Bradshaw CJA, Sodhi NS, Peh KS-H, Brook BW. 2007. Global evidence that deforestation amplifies flood risk and severity in the developing world. Global Change Biology **13**:2379–2395.

Brown AE, Western AW, McMahon TA, Zhang L. 2013. Impact of forest cover changes on annual streamflow and flow duration curves. JOURNAL OF HYDROLOGY **483**:39–50.

Brown AE, Zhang L, McMahon TA, Western AW, Vertessy RA. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology **310**:28–61.

Bruijnzeel LA. 2004. Hydrologic functions of tropical forests: not seeing the soil for the trees? Agriculture, Ecosystems & Environment **104**:185–228.

Bruijnzeel LA. 2016, June 9. Hydrologic impacts of tropical land degradation, natural regeneration and reforestation. Cuenca, Ecuador.

Bruijnzeel LA, Kappelle M, Mulligan M, Scatena FN. 2010. Tropical montane cloud forests: state of knowledge and sustainability perspectives in a changing world. Pages 691–740 Tropical Montane Cloud Forests: Science for Conservation and Management. Cambridge University Press, Cambridge. Available from http://ebooks.cambridge.org/ref/id/CBO9780511778384 (accessed June 29, 2017).

Bruijnzeel LA, Mulligan M, Scatena FN. 2011. Hydrometeorology of tropical montane cloud forests: emerging patterns. Hydrologic Processes **25**:465–498.

Bullock A, Acreman M. 2003. The role of wetlands in the hydrologic cycle. Hydrology and Earth System Sciences **7**:358–389.

Buytaert W, Célleri R, De Bièvre B, Cisneros F, Wyseure G, Deckers J, Hofstede R. 2006. Human impact on the hydrology of the Andean páramos. Earth-Science Reviews **79**:53–72.

Buytaert W, Iñiguez V, De Bièvre B. 2007. The effects of afforestation and cultivation on water yield in the Andean páramo. Forest Ecology and Management **251**:22–30.

Capon SJ et al. 2013. Riparian Ecosystems in the 21st Century: Hotspots for Climate Change Adaptation? Ecosystems **16**:359–381.

Celleri R, Feyen J. 2009. The Hydrology of Tropical Andean Ecosystems: Importance, Knowledge Status, and Perspectives. Mountain Research and Development **29**:350–355.

Chow TL, Rees HW, Daigle JL. 1999. Effectiveness of terraces/grassed waterway systems for soil and water conservation: A field evaluation. Journal of Soil and Water Conservation **54**:577–583.

Crespo PJ, Feyen J, Buytaert W, Bücker A, Breuer L, Frede H-G, Ramírez M. 2011. Identifying controls of the rainfall-runoff response of small catchments in the tropical Andes (Ecuador). Journal of Hydrology **407**:164–174.

Dadson SJ et al. 2017. A restatement of the natural science evidence concerning catchment-based "natural" flood management in the UK. Proceedings of the Royal Society A: Mathematical, Physical and Engineering Science **473**:20160706.

Daniel J. 1999. Influence of wheat tillage practices on shallow groundwater recharge. JOURNAL OF SOIL AND WATER CONSERVATION **54**:560–564.

Deasy C, Titman A, Quinton JN. 2014. Measurement of flood peak effects as a result of soil and land management, with focus on experimental issues and scale. Journal of Environmental Management **132**:304–312.

Dixon SJ, Sear DA, Odoni NA, Sykes T, Lane SN. 2016. The effects of river restoration on catchment scale flood risk and flood hydrology: The Effects of River Restoration on Catchment Scale Flood Risk. Earth Surface Processes and Landforms **41**:997–1008.

Dourte DR, Shukla S, Haman DZ, Reddy MD, Devi MU, Mani A. 2014. Agricultural management impacts on groundwater: simulations of existing and alternative management options in Peninsular India. Hydrologic Processes **28**:5021–5033.

Dudley N, Stolton S, Owusu RA. 2003. Running pure: the importance of forest protected areas to drinking water : a research report for the World Bank/WWF Alliance for Forest Conservation and Sustainable Use. WWF International ; World Bank, Gland, Switzerland; Washington, D.C. Available from http://search.ebscohost.com/direct.asp?db=eih&jid=%228NVA%22&scope=site (accessed June 22, 2017).

Dung BX, Gomi T, Miyata S, Sidle RC, Kosugi K, Onda Y. 2012. Runoff responses to forest thinning at plot and catchment scales in a headwater catchment draining Japanese cypress forest. JOURNAL OF HYDROLOGY **444**:51–62.

Dung BX, Miyata S, Gomi T. 2011. Effect of forest thinning on overland flow generation on hillslopes covered by Japanese cypress. Ecohydrology **4**:367–378.

Dymond SF, Aust WM, Prisley SP, Eisenbies MH, Vose JM. 2014. Application of a Distributed Process-Based Hydrologic Model to Estimate the Effects of Forest Road Density on Stormflows in the Southern Appalachians. Forest Science **60**:1213–1223.

Ellison D et al. 2017. Trees, forests and water: Cool insights for a hot world. Global Environmental Change **43**:51–61.

Ellison D, Futter MN, Bishop K. 2012. On the forest cover-water yield debate: from demand- to supply-side thinking. GLOBAL CHANGE BIOLOGY **18**:806–820.

Farley KA, Jobbagy EG, Jackson RB. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology **11**:1565–1576.

Filoso S, Bezerra MO, Weiss KCB, Palmer MA. 2017. Impacts of forest restoration on water yield: A systematic review. PLoS ONE **12**:e0183210. https://doi.org/10.1371/journal.pone.0183210Flerchinger GN, Seyfried MS, Hardegree SP. 2016. Hydrologic response and recovery to prescribed fire and vegetation removal in a small rangeland catchment: Hydrologic Recovery to Rangeland Vegetation Disturbance. Ecohydrology **9**:1604–1619.

Foltz RB, Copeland NS, Elliot WJ. 2009. Reopening abandoned forest roads in northern Idaho, USA: Quantification of runoff, sediment concentration, infiltration, and interrill erosion parameters. Journal of Environmental Management **90**:2542–2550.

Gao J, Holden J, Kirkby M. 2016. The impact of land-cover change on flood peaks in peatland basins. WATER RESOURCES RESEARCH **52**:3477–3492.

Garcia-Chevesich PA, Neary DG, Scott DF, Benyon RG, Reyna T. 2017. Forest management and the impact on water resources: A review of 13 countries. Page 203. Technical Document 37. United Nations Educational, Scientific, and Cultural Organization (UNESCO), International Hydrologic Program, Paris, France. Available from http://unesdoc.unesco.org/images/0024/002479/247902e.pdf.

Ghazavi G, Thomas Z, Hamon Y, Marie JC, Corson M, Merot P. 2008. Hedgerow impacts on soil-water transfer due to rainfall interception and root-water uptake. Hydrologic Processes **22**:4723–4735.

Gilley JE, Patton BD, Nyren PE, Simanton JR. 1996. Grazing and Haying Effects On Runoff and Erosion from a Former Conservation Reserve Program Site. Applied Engineering in Agriculture **12**:681–684.

Gokbulak F, Sengonul K, Serengil Y, Ozhan S, Yurtseven I, Uygur B, Ozcelik MS. 2016. Effect of Forest Thinning on Water Yield in a Sub-Humid Mediterranean Oak-Beech Mixed Forested Watershed. WATER RESOURCES MANAGEMENT **30**:5039–5049.

Gomez-Delgado F et al. 2011. Modelling the hydrologic behaviour of a coffee agroforestry basin in Costa Rica. HYDROLOGY AND EARTH SYSTEM SCIENCES **15**:369–392.

Grayson R, Holden J, Rose R. 2010. Long-term change in storm hydrographs in response to peatland vegetation change. JOURNAL OF HYDROLOGY **389**:336–343.

Gregory SV, Swanson FJ, McKee WA, Cummins KW. 1991. An Ecosystem Perspective of Riparian Zones: Focus on links between land and water. BIOSCIENCE **41**:540–551.

Guzmán P, Batelaan O, Huysmans M, Wyseure G. 2015. Comparative analysis of baseflow characteristics of two Andean catchments, Ecuador. Hydrologic Processes **29**:3051–3064.

Havstad KM, Peters DPC, Skaggs R, Brown J, Bestelmeyer B, Fredrickson E, Herrick J, Wright J. 2007. Ecological services to and from rangelands of the United States. Ecological Economics **64**:261–268.

Hernandez-Santana V, Zhou X, Helmers MJ, Asbjornsen H, Kolka R, Tomer M. 2013. Native prairie filter strips reduce runoff from hillslopes under annual row-crop systems in Iowa, USA. JOURNAL OF HYDROLOGY **477**:94–103.

Ilstedt U et al. 2016. Intermediate tree cover can maximize groundwater recharge in the seasonally dry tropics. Scientific Reports **6**. Available from http://www.nature.com/articles/srep21930 (accessed July 6, 2017).

Ilstedt U, Malmer A, Verbeeten E, Murdiyarso D. 2007. The effect of afforestation on water infiltration in the tropics: A systematic review and meta-analysis. Forest Ecology and Management **251**:45–51.

Inserillo EA, Green MB, Shanley JB, Boyer JN. 2017. Comparing catchment hydrologic response to a regional storm using specific conductivity sensors. HYDROLOGIC PROCESSES **31**:1074–1085.

Javaheri A, Babbar-Sebens M. 2014. On comparison of peak flow reductions, flood inundation maps, and velocity maps in evaluating effects of restored wetlands on channel flooding. Ecological Engineering **73**:132–145.

Jobbágy EG, Jackson RB. 2004. Groundwater use and salinization with grassland afforestation. Global Change Biology **10**:1299-1312.

Jones JA, Grant GE. 2001. Comment on "Peak flow responses to clear-cutting and roads in small and large basins, Western Cascades, Oregon: A second opinion" by R. B. Thomas and W. F. Megahan. Water Resources Research **37**:175–178.

Jones JA, Swanson FJ, Wemple BC, Snyder KU. 2000. Effects of Roads on Hydrology, Geomorphology, and Disturbance Patches in Stream Networks. Conservation Biology **14**:76–85.

Kolka RK, Palik BJ, Tersteeg DP, Bell JC. 2011. Effects of Riparian Buffers on Hydrology of Northern Seasonal Ponds. TRANSACTIONS OF THE ASABE **54**:2111–2116.

Kolka RK, Smidt MF. 2004. Effects of forest road amelioration techniques on soil bulk density, surface runoff, sediment transport, soil moisture and seedling growth. Forest Ecology and Management **202**:313–323.

Kramer L., Burkart MR, Meek DW, Jaquis RJ, James DE. 1999. Field-scale watershed evaluations on deep-loess soils: II. Hydrologic responses to different agricultural land management systems. Journal of Soil and Water Conservation **54**:705–710.

Legge JT, Doran PJ, Herbert ME, Asher J, O'Neil G, Mysorekar S, Sowa S, Hall KR. 2013. From model outputs to conservation action: Prioritizing locations for implementing agricultural best management practices in a Midwestern watershed. JOURNAL OF SOIL AND WATER CONSERVATION **68**:22–33.

Leys A, Govers G, Gillijns K, Poesen J. 2007. Conservation tillage on loamy soils: explaining the variability in interrill runoff and erosion reduction. European Journal of Soil Science **58**:1425–1436.

Li J et al. 2017. Grassland restoration reduces water yield in the headstream region of Yangtze River. Scientific Reports **7**. Available from http://www.nature.com/articles/s41598-017-02413-9 (accessed June 13, 2017).

Li J, Shi W. 2015. Effects of alpine swamp wetland change on rainfall season runoff and flood characteristics in the headwater area of the Yangtze River. CATENA **127**:116–123.

Liu Y, Engel BA, Flanagan DC, Gitau MW, McMillan SK, Chaubey I. 2017. A review on effectiveness of best management practices in improving hydrology and water quality: Needs and opportunities. Science of The Total Environment **601**:580-93.Lloyd RA, Lohse KA, Ferre T. 2013. Influence of road reclamation techniques on forest ecosystem recovery. Frontiers in Ecology and the Environment **11**:75–81.

Luce CH. 1997. Effectiveness of Road Ripping in Restoring Infiltration Capacity of Forest Roads. Restoration Ecology **5**:265–270.

Mallik AU, Newaz S, Mackereth RW, Shahi C. 2011. Geomorphic changes of headwater systems 3-23 years after forest harvesting by clearcutting. ECOSPHERE **2**.

Martinet MC, Vivoni ER, Cleverly JR, Thibault JR, Schuetz JF, Dahm CN. 2009. On groundwater fluctuations, evapotranspiration, and understory removal in riparian corridors. Water Resources Research **45**:W05425.

Martinez-Martinez E, Nejadhashemi AP, Woznicki SA, Love BJ. 2014. Modeling the hydrologic significance of wetland restoration scenarios. Journal of Environmental Management **133**:121–134.

McGarry D, Bridge BJ, Radford BJ. 2000. Contrasting soil physical properties after zero and traditional tillage of an alluvial soil in the semi-arid subtropics. Soil and Tillage Research **53**:105–115.

McGlynn BL, McDonnell JJ. 2003. Quantifying the relative contributions of riparian and hillslope zones to catchment runoff. Water Resources Research **39**. Available from http://doi.wiley.com/10.1029/2003WR002091 (accessed June 20, 2017).

McKergow L, Prosser I, Grayson R, Heiner D. 2004. Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 1. Riparian hydrology. AUSTRALIAN JOURNAL OF SOIL RESEARCH **42**:473–484.

McLaughlin DL, Cohen MJ. 2013. Realizing ecosystem services: wetland hydrologic function along a gradient of ecosystem condition. Ecological Applications **23**:1619–1631.

Moore GW, Owens MK. 2012. Transpirational Water Loss in Invaded and Restored Semiarid Riparian Forests. RESTORATION ECOLOGY **20**:346–351.

Moreno HA, Gupta HV, White DD, Sampson DA. 2016. Modeling the distributed effects of forest thinning on the long-term water balance and streamflow extremes for a semi-arid basin in the southwestern US. Hydrology and Earth System Sciences **20**:1241–1267.

Morley TR, Reeve AS, Calhoun AJK. 2011. The Role of Headwater Wetlands in Altering Streamflow and Chemistry in a Maine, USA Catchment. JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION **47**:337–349.

Mosquera GM, Lazo PX, Célleri R, Wilcox BP, Crespo P. 2015. Runoff from tropical alpine grasslands increases with areal extent of wetlands. CATENA **125**:120–128.

Muñoz-Villers LE, Holwerda F, Susana Alvarado-Barrientos M, Geissert D, Marin-Castro B, Gomez-Tagle A, McDonnell J, Asbjornsen H, Dawson T, Bruijnzeel LA. 2015. Hydrologic effects of cloud forest conversion in central Veracruz, Mexico. BOSQUE **36**:395–407.

Noorduijn SL, Ghadouani A, Vogwill R, Smettem KRJ, Legendre P. 2010. Water table response to an experimental alley farming trial: dissecting the spatial and temporal structure of the data. Ecological Applications **20**:1704–1720.

Nosetto MD, Jobbágy EG, Paruelo JM. 2005. Land-use change and water losses: the case of grassland afforestation across a soil textural gradient in central Argentina. Global Change Biology. **11**:1101-1117.

Nosetto MD, Jobbágy EG, Tóth T, Jackson RB. 2008. Regional patterns and controls of ecosystem salinization with grassland afforestation along a rainfall gradient. Global Biogeochemical Cycles **22.**

Ochoa-Tocachi BF et al. 2016. Impacts of land use on the hydrologic response of tropical Andean catchments. HYDROLOGIC PROCESSES **30**:4074–4089.

O'Connell PE, Ewen J, O'Donnell G, Quinn P. 2007. Is there a link between agricultural land-use management and flooding? Hydrology and Earth System Sciences **11**:96–107.

Palleiro L, Rodriguez-Blanco ML, Taboada-Castro MM, Taboada-Castro MT. 2014. Hydrologic response of a humid agroforestry catchment at different time scales. HYDROLOGIC PROCESSES **28**:1677–1688.

Palmer M, Allan JD, Meyer J, Bernhardt ES. 2007. River restoration in the twenty-first century: Data and experiential future efforts. RESTORATION ECOLOGY **15**:472–481.

Palmer MA et al. 2005. Standards for ecologically successful river restoration: Ecological success in river restoration. Journal of Applied Ecology **42**:208–217.

Palmer MA, Reidy Liermann CA, Nilsson C, Fl?rke M, Alcamo J, Lake PS, Bond N. 2008. Climate change and the world's river basins: anticipating management options. Frontiers in Ecology and the Environment **6**:81–89.

Park JY, Ale S, Teague WR, Dowhower SL. 2017. Simulating hydrologic responses to alternate grazing management practices at the ranch and watershed scales. Journal of Soil and Water Conservation **72**:102–121.

Perry LG, Reynolds LV, Beechie TJ, Collins MJ, Shafroth PB. 2015. Incorporating climate change projections into riparian restoration planning and design. ECOHYDROLOGY **8**:863–879.

Pilon C, Moore PA Jr, Pote DH, Pennington JH, Martin JW, Brauer DK, Raper RL, Dabney SM, Lee J. 2017. Long-term Effects of Grazing Management and Buffer Strips on Soil Erosion from Pastures. JOURNAL OF ENVIRONMENTAL QUALITY **46**:364–372.

Potter KW. 1991. Hydrologic impacts of changing land management practices in a moderate-sized agricultural catchment. Water Resources Research **27**:845–855.

Rassam DW. 2005. Impacts Of Hillslope And Floodplain Characteristics On Groundwater Dynamics: Implications For Riparian Denitrification. Pages 2735–2741 in Zerger, A and Argent, RM, editor. MODSIM 2005: INTERNATIONAL CONGRESS ON MODELLING AND SIMULATION: ADVANCES AND APPLICATIONS FOR MANAGEMENT AND DECISION MAKING: ADVANCES AND APPLICATIONS FOR MANAGEMENT AND DECISION MAKING.

Rittenburg RA, Squires AL, Boll J, Brooks ES, Easton ZM, Steenhuis TS. 2015. Agricultural BMP Effectiveness and Dominant Hydrologic Flow Paths: Concepts and a Review. JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION **51**:305–329.

Rodríguez-Blanco ML, Taboada-Castro MM, Taboada-Castro MT. 2012. Rainfall–runoff response and event-based runoff coefficients in a humid area (northwest Spain). Hydrologic Sciences Journal **57**:445–459.

Salemi LF, Groppo JD, Trevisan R, Marcos de Moraes J, de Paula Lima W, Martinelli LA. 2012. Riparian vegetation and water yield: A synthesis. Journal of Hydrology **454**–**455**:195–202.

Scott DF, Bruijnzeel LA, Mackensen J. 2005. The hydrologic and soil impacts of forestation in the tropics. Pages 622–651 in M. Bonell and L. A. Bruijnzeel, editors. Forests, Water and People in the Humid Tropics. Cambridge University Press, Cambridge.

Seavy NE, Gardali T, Golet GH, Griggs FT, Howell CA, Kelsey R, Small SL, Viers JH, Weigand JF. 2009. Why Climate Change Makes Riparian Restoration More Important than Ever: Recommendations for Practice and Research. Ecological Restoration **27**:330–338.

Shafroth P, Cleverly J, Dudley T, Taylor J, Van Riper C, Weeks E, Stuart J. 2005. Control of Tamarix in the Western United States: Implications for water salvage, wildlife use, and riparian restoration. ENVIRONMENTAL MANAGEMENT **35**:231–246.

Shafroth PB, Beauchamp VB, Briggs MK, Lair K, Scott ML, Sher AA. 2008. Planning riparian restoration in the context of Tamarix control in western North America. RESTORATION ECOLOGY **16**:97–112.

Shipitalo M, Dick W, Edwards W. 2000. Conservation tillage and macropore factors that affect water movement and the fate of chemicals. SOIL & TILLAGE RESEARCH **53**:167–183.

Simonit S, Connors JP, Yoo J, Kinzig A, Perrings C. 2015. The Impact of Forest Thinning on the Reliability of Water Supply in Central Arizona. PLOS ONE **10**:e0121596.

Skaggs RW, Fausey NR, Evans RO. 2012. Drainage water management. Journal of soil and water conservation **67**:167A-172A.

Sloan BP, Mantilla R, Basu NB, Fonley M. 2017. Hydrologic Impacts of Subsurface Drainage from the Field to Watershed Scale. Hydrologic Processes **31:** 3017-3028..

Smakhtin V. 2001. Low flow hydrology: a review. Journal of Hydrology **240**:147–186.

Staes J, Rubarenzya MH, Meire P, Willems P. 2009. Modelling hydrologic effects of wetland restoration: a differentiated view. Water Science & Technology **59**:433.

Sun G, Caldwell PV, McNulty SG. 2015. Modelling the potential role of forest thinning in maintaining water supplies under a changing climate across the conterminous United States. HYDROLOGIC PROCESSES **29**:5016–5030.

Switalski T, Bissonette J, DeLuca T, Luce C, Madej M. 2004. Benefits and impacts of road removal. Frontiers in Ecology and the Environment **2**:21–28.

Teague WR, Dowhower SL, Baker SA, Haile N, DeLaune PB, Conover DM. 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrologic properties in tall grass prairie. Agriculture, Ecosystems & Environment **141**:310–322.

Thomas RB, Megahan WF. 1998. Peak flow responses to clear-cutting and roads in small and large basins, Western Cascades, Oregon: A second opinion. Water Resources Research **34**:3393–3403.

Thomas RB, Megahan WF. 2001. Reply [to “Comment on ?Peak flow responses to clear-cutting and roads in small and large basins, Western Cascades, Oregon: A Second Opinion? by R. B. Thomas and W. F. Megahan”]. Water Resources Research **37**:181–183.

Udawatta R, Krstansky J, Henderson G, Garrett H. 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. JOURNAL OF ENVIRONMENTAL QUALITY **31**:1214–1225.

Valentin C et al. 2008. Runoff and sediment losses from 27 upland catchments in Southeast Asia: Impact of rapid land use changes and conservation practices. AGRICULTURE ECOSYSTEMS & ENVIRONMENT **128**:225–238.

van Dijk AIJM, van Noordwijk M, Calder IR, Bruijnzeel SLA, Schellekens J, Chappell NA. 2009. Forest-flood relation still tenuous - comment on “Global evidence that deforestation amplifies flood risk and severity in the developing world” by C. J. A. Bradshaw, N.S. Sodi, K. S.-H. Peh and B.W. Brook. Global Change Biology **15**:110–115.

van Noordwijk M, Tanika L, Lusiana B. 2017a. Flood risk reduction and flow buffering as ecosystem services - Part 1: Theory on flow persistence, flashiness and baseflow. Hydrology and Earth System Sciences **21**:2321–2340.

van Noordwijk M, Tanika L, Lusiana B. 2017b. Flood risk reduction and flow buffering as ecosystem services - Part 2: Land use and rainfall intensity effects in Southeast Asia. Hydrology and Earth System Sciences **21**:2341–2360.

Verhoeven JTA, Soons MB, Janssen R, Omtzigt N. 2008. An Operational Landscape Unit approach for identifying key landscape connections in wetland restoration. Journal of Applied Ecology **45**:1496–1503.

von Freyberg J, Radny D, Gall HE, Schirmer M. 2014. Implications of hydrologic connectivity between hillslopes and riparian zones on streamflow composition. JOURNAL OF CONTAMINANT HYDROLOGY **169**:62–74.

Wahren A, Schwaerzel K, Feger K-H. 2012. Potentials and limitations of natural flood retention by forested land in headwater catchments: evidence from experimental and model studies. JOURNAL OF FLOOD RISK MANAGEMENT **5**:321–335.

Wang X, Shang S, Qu Z, Liu T, Melesse AM, Yang W. 2010. Simulated wetland conservation-restoration effects on water quantity and quality at watershed scale. Journal of Environmental Management **91**:1511–1525.

Webber DF, Mickelson SK, Ahmed SI, Russell JR, Powers WJ, Schultz RC, Kovar JL. 2010. Livestock grazing and vegetative filter strip buffer effects on runoff sediment, nitrate, and phosphorus losses. Journal of Soil and Water Conservation **65**:34–41.

Wesemann J, Herrnegger M, Schulz K. 2017. Beeinflussung von Quelleinzugsgebieten durch Forststraßen und Waldwirtschaft mit besonderem Schwerpunkt auf Karstgebiete – eine Literaturstudie für die Praxis (Impacts of Forest Roads and Forestry on Spring Catchments with Focus on Karst Areas – A review for Practitioners). Hydrologie und Wasserbewirtschaftung **61**:6–26.

Wilcox BP. 2002. Shrub Control and Streamflow on Rangelands: A Process Based Viewpoint. Journal of Range Management **55**:318.

Wilcox BP. 2007. Does rangeland degradation have implications for global streamflow? Hydrologic Processes **21**:2961–2964.

Wilcox BP, Huang Y. 2010. Woody plant encroachment paradox: Rivers rebound as degraded grasslands convert to woodlands: RIVERS REBOUND AS WOODLANDS EXPAND. Geophysical Research Letters **37**:n/a-n/a.

Wilcox BP, Huang Y, Walker JW. 2008. Long-term trends in streamflow from semiarid rangelands: uncovering drivers of change. Global Change Biology **14**:1676–1689.

Wilcox BP, Owens MK, Knight RW, Lyons RK. 2005. Do woody plants affect streamflow on semiarid karst rangelands? Ecological Applications **15**:127–136.

Wilcox BP, Thurow TL. 2006. Emerging Issues in Rangeland Ecohydrology: Vegetation Change and the Water Cycle. Rangeland Ecology & Management **59**:220–224.

Wine ML, Zou CB. 2012. Long-term streamflow relations with riparian gallery forest expansion into tallgrass prairie in the Southern Great Plains, USA. FOREST ECOLOGY AND MANAGEMENT **266**:170–179.

Yang W, Wang X, Liu Y, Gabor S, Boychuk L, Badiou P. 2010. Simulated environmental effects of wetland restoration scenarios in a typical Canadian prairie watershed. Wetlands Ecology and Management **18**:269–279.

Zedler JB. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Environment **1**:65–72.

Zedler JB, Kercher S. 2005. Wetland Resources: Status, Trends, Ecosystem Services, and Restorability. Annual Review of Environment and Resources **30**:39–74.

Zhang L, Mitsch WJ. 2005. Modelling hydrologic processes in created freshwater wetlands: an integrated system approach. Environmental Modelling & Software **20**:935–946.

Agricultural practices and crop yield

To what extent can specific agricultural best management practices result in crop yield gains or cost reductions?

Conservation agriculture

Conservation Agriculture (CA) has three principles, namely minimal soil disturbance/conservation tillage/no-till, permanent soil cover/residue retention (with crops, cover crops, or a mulch of crop residues) and crop rotations—encapsulating a wide variety of agriculture practices (as defined by FAO). While we attempted to disambiguate findings on individual practice as opposed to practices in combination, often it was challenging to do so given the inter-dependency of activities like mulching and tillage.

Besides empirical studies at the experiment field level, a number of recent reviews and meta-analyses summarized empirical studies across regions and continents. For example, we found reviews on CA in Zimbabwe (Mashingaidze et al. 2012; Mafongoya et al. 2016), Africa (Giller et al. 2009), Central Asia (Kienzler et al. 2012), Australia (Kirkegaard et al. 2014), Sub-Saharan Africa and South Asia (Brouder & Gomez-Macpherson 2014; Palm et al. 2014; Stevenson et al. 2014), and at the global-scale (Rusinamhodzi et al. 2011; Scopel et al. 2013; Pittelkow et al. 2015a). In the scientific literature, there is agreement that evidence concerning crop yield outcomes using CA is highly context-dependent, with mixed results—yields are generally lower, but variations are significant depending on conditions and factors.

Examples of key literature:

- Brouder & Gomez-Macpherson (2014) assessed the quality of reviews studying the impact of CA on smallholder crop yield in Sub-Saharan Africa and South Asia from 1993 to 2012, collected empirical evidence on different crop types, and identified knowledge gaps (insufficient data, lack of rigor, etc.).
- Pittelkow et al. (2015a) conducted a comprehensive global-scale meta-analysis of 610 studies (across 48 crops and 63 countries), showing the general direction of impact (reduced yield reductions) but also highlighting important nuances. For instance, when combined with residue retention and crop rotation, the negative impacts of no-till can be minimized.
- Giller et al. (2015) reviewed the evidence base for CA and argued that many of the claims made for CA were not supported by science, even in large-scale agriculture (both crop yield and economic costs and benefits were included in the review). Their analysis suggests an increasing trend in pragmatic adoption on larger mechanized farms, while limited uptake of CA by smallholder farmers in developing countries due to the lack of resources.
- An OECD report also provided a comprehensive review of soil and water conservation practices, focusing on CA in particular (OECD 2016). Several economic studies are included, suggesting a small cost advantage over conventional soil farm practices (5-10%) on average, although results vary widely across sites, depending a soil and crop types, and/or in developed versus developing

countries. This review also suggested that yields generally decreased under soil and water conservation practices, but increased under rain-fed agro-systems in dry climates.

For no-till/zero-till/reduced tillage/conservation tillage, either as a stand-alone practice or as an integral part of CA, our review found a large body of literature that specifically looked at its impact on crop yield. Examples include studies from Brazil (Bolliger et al. 2006); North America (Toliver et al. 2012; Ogle et al. 2012); Europe (Van den Putte et al. 2010); North America and Europe (Cooper et al. 2016); China (He et al. 2010; Huang et al. 2015; Zhao et al. 2017); and globally (Pittelkow et al. 2015b). Pittelkow et al. (2015b) extended the dataset used in the earlier review of CA (Pittelkow et al. 2015a) to 678 studies representing 50 crops and 63 countries, employing a more expansive definition of no-till compared to CA. These meta-analyses also identified factors influencing the overall yield response to no-till.

For cover crops and related practices (residue retention, mulching, etc.), fewer studies were found beyond the context of CA (e.g. China, crop residue retention on rice yield: Huang et al. 2013; North America, winter cover crops: Marcillo & Miguez 2017; Northern Europe: Valkama et al. 2015; global, but focusing on legume companion plants: Verret et al. 2017).

Global scale meta-analyses, often showing an average yield decrease under CA practices, have constraints on predictive power at a regional scale. Nonetheless, these studies provide helpful information on factors (including climate, soil, crop type, crop management practices, etc.) contributing to yield gaps. The higher methodological bar imposed for including studies in meta-analyses means that poorly documented studies are excluded, and this in turn can produce geographic and other biases.

Agroforestry

Agroforestry is an increasingly prominent example of a working landscape practice that can provide multiple economic, cultural and ecological benefits. Compared CA, studies and reviews on the yield or economic benefits are more diverse, and include documented changes in biodiversity and/or ecosystem services (ES) provision (such as soil erosion control, soil fertility, or biomass production) in addition to changes in yields. For instance, several studies[10] looked broadly at tradeoffs among biodiversity, ES and crop yield or income (e.g. Steffan-Dewenter et al. 2007; Clough et al. 2011; De Beenhouwer et al. 2013; Torralba et al. 2016); while other studies measured how changes in pollination services or pest control services affect crop yield or household income in agroecosystems.

A small number of empirical studies documented direct effects of agroforestry on crop productivity but, due to the limited number of observations and their geographic diversity, it is impossible to infer a general direction of impact. Similar to CA, effects are context-dependent and influenced by the type of agroforestry system. A few examples are given below:

[10] Study findings include: bats and birds may increase crop yield in Indonesian cacao agroforestry (Maas et al. 2013); increases in species richness may indirectly increase household income (Cardozo et al. 2015); and pest-suppressive crop diversification in agroecosystems (intercropping, inclusion of flowering plants, etc.) may interfere with crop production and reduce yield (Letourneau et al. 2011).

- Fertilizer tree systems were developed in Africa to build soil organic matter and fertility. Cases from southern Africa (Malawi, Tanzania, Mozambique, Zambia, and Zimbabwe) showed promising results, leading to increases in maize yield (Akinnifesi et al. 2008; Ajayi et al. 2011).
- Highly diverse, dense cacao agroforestry may generate greater yields, cash incomes and household benefits through product diversification in Central America (Cerda et al. 2014); timber yield from smallholder agroforestry systems (silvopastoral, coffee, cacao, and living fences) may also be profitable if management practices were improved in Honduras and Nicaragua (de Sousa et al. 2016).
- In central Ethiopian highlands, while Eucalyptus boundary plantations reduce crop (tef and wheat) yields, timber products may compensate for yield reductions (Kidanu et al. 2005); on the south coast of Western Australia, Eucalyptus agroforestry to reduce salinization risk revealed mixed results in terms of crop and pasture production as well as economic returns (Sudmeyer & Simons 2008); in the eastern Himalayas, nitrogen-fixing alder in cardamom cultivation increased production and yield through accelerated nutrient cycling (Sharma et al. 2008).

Other BMPs

A few other studies looked at collections of practices that include but are not limited to agroforestry, CA, or integrated nutrient management (e.g. Pretty et al. 2006; Sileshi et al. 2008; Bayala et al. 2012), showing examples of both crop yield increases and decreases.

An OECD report (2016) also reviewed a broader collection of farm management practices including organic farming, integrated pest management, biotechnology, and precision agriculture. The review also observed a variety of empirical and modeling studies on nutrient management, irrigation management, and other practices. Given the expansive number of agricultural practices that may be categorized as "best management practices", a more targeted literature review would likely better identify gaps for specific BMPs.

Beyond yield

Yield increases are not the only motivator for farmer adoption of agricultural BMPs. For CA, the potential for reduced costs—such as fuel savings—may help drive adoption. However, these benefits may be limited for smallholders (Giller et al. 2015). On the contrary, labor costs may be a limiting factor for the adoption of agricultural BMPs (Giller et al. 2009; Rusinamhodzi 2015). Reviews of on-farm economics suggest high heterogeneity (Mercer 2004; Montambault & Alavalapati 2005; Andersson & D'Souza 2014, 2014; Corbeels et al. 2014; Pannell et al. 2014). Our review also found a growing body of literature in regard to uncertainties around crop yield gains and labor requirements concerning CA in smallholder agriculture, especially in Africa (Giller et al. 2009; Andersson & D'Souza 2014; Cheesman et al. 2017).

In addition to the annual impacts of agricultural BMPs, it is also necessary to consider longer-term outcomes, especially for farms with relatively poor soils where the comparative costs of erosion can be greater (particularly in locations with less topsoil where yield decreases from soil loss can occur sooner). Hypothetically, despite possible near-term crop yield reductions from adopting CA (compared to conventional farming), reduced soil erosion over a longer time horizon (e.g. 20 years) could actually result in higher net yields over time. However, such long-term, indirect benefits may be relatively hard to appreciate from a farmer livelihood point of view.

Literature reviewed

Ajayi OC, Place F, Akinnifesi FK, Sileshi GW. 2011. Agricultural success from Africa: the case of fertilizer tree systems in southern Africa (Malawi, Tanzania, Mozambique, Zambia and Zimbabwe). International Journal of Agricultural Sustainability **9**:129–136.

Akinnifesi FK, Chirwa PW, Ajayi OC, Sileshi G, Matakala P, Kwesiga FR, Harawa H, Makumba W. 2008. Contributions of Agroforestry Research to Livelihood of Smallholder Farmers in Southern Africa: 1. Taking Stock of the Adaptation, Adoption and Impact of Fertilizer Tree Options. Agricultural Journal **3**:58–75.

Andersson JA, D'Souza S. 2014. From adoption claims to understanding farmers and contexts: A literature review of Conservation Agriculture (CA) adoption among smallholder farmers in southern Africa. Agriculture, Ecosystems & Environment **187**:116–132.

Bayala J, Sileshi GW, Coe R, Kalinganire A, Tchoundjeu Z, Sinclair F, Garrity D. 2012. Cereal yield response to conservation agriculture practices in drylands of West Africa: A quantitative synthesis. Journal of Arid Environments **78**:13–25.

Bolliger A, Magid J, Amado JCT, Skóra Neto F, Ribeiro M de F dos S, Calegari A, Ralisch R, de Neergaard A. 2006. Taking Stock of the Brazilian "Zero-Till Revolution": A Review of Landmark Research and Farmers' Practice. Pages 47–110 Advances in Agronomy. Elsevier. Available from http://linkinghub.elsevier.com/retrieve/pii/S0065211306910025 (accessed May 30, 2017).

Brouder SM, Gomez-Macpherson H. 2014. The impact of conservation agriculture on smallholder agricultural yields: A scoping review of the evidence. Agriculture, Ecosystems & Environment **187**:11–32.

Cardozo EG, Muchavisoy HM, Silva HR, Correa Zelarayan ML, Alves Leite MF, Rousseau GX, Gehring C. 2015. Species richness increases income in agroforestry systems of eastern Amazonia. AGROFORESTRY SYSTEMS **89**:901–916.

Cerda R, Deheuvels O, Calvache D, Niehaus L, Saenz Y, Kent J, Vilchez S, Villota A, Martinez C, Somarriba E. 2014. Contribution of cocoa agroforestry systems to family income and domestic consumption: looking toward intensification. AGROFORESTRY SYSTEMS **88**:957–981.

Cheesman S, Andersson JA, Frossard E. 2017. Does closing knowledge gaps close yield gaps? On-farm conservation agriculture trials and adoption dynamics in three smallholder farming areas in Zimbabwe. The Journal of Agricultural Science **155**:81–100.

Clough Y et al. 2011. Combining high biodiversity with high yields in tropical agroforests. Proceedings of the National Academy of Sciences **108**:8311–8316.

Cooper J et al. 2016. Shallow non-inversion tillage in organic farming maintains crop yields and increases soil C stocks: a meta-analysis. Agronomy for Sustainable Development **36**. Available from http://link.springer.com/10.1007/s13593-016-0354-1 (accessed May 27, 2017).

Corbeels M et al. 2014. Understanding the impact and adoption of conservation agriculture in Africa: A multi-scale analysis. Agriculture, Ecosystems & Environment **187**:155–170.

De Beenhouwer M, Aerts R, Honnay O. 2013. A global meta-analysis of the biodiversity and ecosystem service benefits of coffee and cacao agroforestry. Agriculture, Ecosystems & Environment **175**:1–7.

de Sousa KFD, Detlefsen G, Virginio Filho E de M, Tobar D, Casanoves F. 2016. Timber yield from smallholder agroforestry systems in Nicaragua and Honduras. AGROFORESTRY SYSTEMS **90**:207–218.

Giller KE, Andersson JA, Corbeels M, Kirkegaard J, Mortensen D, Erenstein O, Vanlauwe B. 2015. Beyond conservation agriculture. Frontiers in Plant Science **6**. Available from http://journal.frontiersin.org/Article/10.3389/fpls.2015.00870/abstract (accessed May 23, 2017).

Giller KE, Witter E, Corbeels M, Tittonell P. 2009. Conservation agriculture and smallholder farming in Africa: The heretics' view. Field Crops Research **114**:23–34.

He J, Li H-W, Wang Q-J, Gao H-W, Li W-Y, Zhang X-M, McGiffen M. 2010. The adoption of conservation tillage in China: Conservation tillage in China. Annals of the New York Academy of Sciences **1195**:E96–E106.

Huang M, Zhou X, Cao F, Xia B, Zou Y. 2015. No-tillage effect on rice yield in China: A meta-analysis. Field Crops Research **183**:126–137.

Huang S, Zeng Y, Wu J, Shi Q, Pan X. 2013. Effect of crop residue retention on rice yield in China: A meta-analysis. Field Crops Research **154**:188–194.

Kidanu S, Mamo T, Stroosnijder L. 2005. Biomass production of Eucalyptus boundary plantations and their effect on crop productivity on Ethiopian highland vertisols. AGROFORESTRY SYSTEMS **63**:281–290.

Kienzler KM, Lamers JPA, McDonald A, Mirzabaev A, Ibragimov N, Egamberdiev O, Ruzibaev E, Akramkhanov A. 2012. Conservation agriculture in Central Asia-What do we know and where do we go from here? FIELD CROPS RESEARCH **132**:95–105.

Kirkegaard JA, Conyers MK, Hunt JR, Kirkby CA, Watt M, Rebetzke GJ. 2014. Sense and nonsense in conservation agriculture: Principles, pragmatism and productivity in Australian mixed farming systems. Agriculture, Ecosystems & Environment **187**:133–145.

Letourneau DK et al. 2011. Does plant diversity benefit agroecosystems? A synthetic review. Ecological Applications **21**:9–21.

Liu Y, Engel BA, Flanagan DC, Gitau MW, McMillan SK, Chaubey I. 2017. A review on effectiveness of best management practices in improving hydrology and water quality: Needs and opportunities. Science of The Total Environment **601**:580-93.

Maas B, Clough Y, Tscharntke T. 2013. Bats and birds increase crop yield in tropical agroforestry landscapes. ECOLOGY LETTERS **16**:1480–1487.

Mafongoya P, Rusinamhodzi L, Siziba S, Thierfelder C, Mvumi BM, Nhau B, Hove L, Chivenge P. 2016. Maize productivity and profitability in Conservation Agriculture systems across agro-ecological regions in Zimbabwe: A review of knowledge and practice. Agriculture, Ecosystems & Environment **220**:211–225.

Marcillo GS, Miguez FE. 2017. Corn yield response to winter cover crops: An updated meta-analysis. Journal of Soil and Water Conservation **72**:226–239.

Mashingaidze N, Madakadze C, Twomlow S, Nyamangara J, Hove L. 2012. Crop yield and weed growth under conservation agriculture in semi-arid Zimbabwe. SOIL & TILLAGE RESEARCH **124**:102–110.

Mercer DE. 2004. Adoption of agroforestry innovations in the tropics: A review. Agroforestry Systems **61**–**62**:311–328.

Montambault JR, Alavalapati JRR. 2005. Socioeconomic research in agroforestry: a decade in review. Agroforestry Systems **65**:151–161.

OECD. 2016. Farm Management Practices to Foster Green Growth. OECD Publishing. Available from http://www.oecd-ilibrary.org/agriculture-and-food/farm-management-practices-to-foster-green-growth_9789264238657-en (accessed May 30, 2017).

Ogle SM, Swan A, Paustian K. 2012. No-till management impacts on crop productivity, carbon input and soil carbon sequestration. Agriculture, Ecosystems & Environment **149**:37–49.

Palm C, Blanco-Canqui H, DeClerck F, Gatere L, Grace P. 2014. Conservation agriculture and ecosystem services: An overview. Agriculture, Ecosystems & Environment **187**:87–105.

Pannell DJ, Llewellyn RS, Corbeels M. 2014. The farm-level economics of conservation agriculture for resource-poor farmers. Agriculture, Ecosystems & Environment **187**:52–64.

Pittelkow CM, Liang X, Linquist BA, van Groenigen KJ, Lee J, Lundy ME, van Gestel N, Six J, Venterea RT, van Kessel C. 2015a. Productivity limits and potentials of the principles of conservation agriculture. Nature **517**:365–368.

Pittelkow CM, Linquist BA, Lundy ME, Liang X, van Groenigen KJ, Lee J, van Gestel N, Six J, Venterea RT, van Kessel C. 2015b. When does no-till yield more? A global meta-analysis. Field Crops Research **183**:156–168.

Pretty JN, Noble AD, Bossio D, Dixon J, Hine RE, Penning de Vries FWT, Morison JIL. 2006. Resource-Conserving Agriculture Increases Yields in Developing Countries. Environmental Science & Technology **40**:1114–1119.

Rusinamhodzi L. 2015. Tinkering on the periphery: Labour burden not crop productivity increased under no-till planting basins on smallholder farms in Murehwa district, Zimbabwe. FIELD CROPS RESEARCH **170**:66–75.

Rusinamhodzi L, Corbeels M, van Wijk MT, Rufino MC, Nyamangara J, Giller KE. 2011. A meta-analysis of long-term effects of conservation agriculture on maize grain yield under rain-fed conditions. Agronomy for Sustainable Development **31**:657–673.

Scopel E et al. 2013. Conservation agriculture cropping systems in temperate and tropical conditions, performances and impacts. A review. Agronomy for Sustainable Development **33**:113–130.

Sharma E, Sharma R, Sharma G, Rai SC, Sharma P, Chettri N. 2008. Values and Services of Nitrogen-Fixing Alder Based Cardamom Agroforestry Systems in the Eastern Himalayas. Page 393+ in Snelder, DJ and Lasco, RD, editor. SMALLHOLDER TREE GROWING FOR RURAL DEVELOPMENT AND ENVIRONMENTAL SERVICES: LESSONS FROM ASIA. SPRINGER, PO BOX 17, 3300 AA DORDRECHT, NETHERLANDS.

Sileshi G, Akinnifesi FK, Ajayi OC, Place F. 2008. Meta-analysis of maize yield response to woody and herbaceous legumes in sub-Saharan Africa. Plant and Soil **307**:1–19.

Steffan-Dewenter I et al. 2007. Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. Proceedings of the National Academy of Sciences **104**:4973–4978.

Stevenson JR, Serraj R, Cassman KG. 2014. Evaluating conservation agriculture for small-scale farmers in Sub-Saharan Africa and South Asia. Agriculture, Ecosystems & Environment **187**:1–10.

Sudmeyer RA, Simons JA. 2008. Eucalyptus globulus agroforestry on deep sands on the southeast coast of Western Australia: The promise and the reality. AGRICULTURE ECOSYSTEMS & ENVIRONMENT **127**:73–84.

Toliver DK, Larson JA, Roberts RK, English BC, De La Torre Ugarte DG, West TO. 2012. Effects of No-Till on Yields as Influenced by Crop and Environmental Factors. Agronomy Journal **104**:530.

Torralba M, Fagerholm N, Burgess PJ, Moreno G, Plieninger T. 2016. Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. Agriculture, Ecosystems & Environment **230**:150–161.

Tscharntke T et al. 2011. Multifunctional shade-tree management in tropical agroforestry landscapes - a review: Multifunctional shade-tree management. Journal of Applied Ecology **48**:619–629.

Valkama E, Lemola R, Känkänen H, Turtola E. 2015. Meta-analysis of the effects of undersown catch crops on nitrogen leaching loss and grain yields in the Nordic countries. Agriculture, Ecosystems & Environment **203**:93–101.

Van den Putte A, Govers G, Diels J, Gillijns K, Demuzere M. 2010. Assessing the effect of soil tillage on crop growth: A meta-regression analysis on European crop yields under conservation agriculture. European Journal of Agronomy **33**:231–241.

Verret V, Gardarin A, Pelzer E, Médiène S, Makowski D, Valantin-Morison M. 2017. Can legume companion plants control weeds without decreasing crop yield? A meta-analysis. Field Crops Research **204**:158–168.

Zhao X, Liu S-L, Pu C, Zhang X-Q, Xue J-F, Ren Y-X, Zhao X-L, Chen F, Lal R, Zhang H-L. 2017. Crop yields under no-till farming in China: A meta-analysis. European Journal of Agronomy **84**:67–75.

Impacts of anthropogenic change: climate and land use

What are the expected relative magnitudes of contributions to hydrologic outcomes from anthropogenic climate change as compared to land use/land cover change?

Study characteristics

There is a growing body of literature comparing hydrologic impacts attributable to climate change and land use change—nearly two-thirds of identified studies (20 of 33) were published in the last 5 years. The existing literature generally comprises two primary types of studies: retrospective and prospective, each accounting for roughly half of identified publications. Most (25) studies assessed streamflow (annual or sub-annual discharge) as a primary outcome of interest. Sediment (6) and groundwater recharge (2) were the next most commonly investigated outcomes. Other outcomes (6) included nutrient loads, temperature, BOD, and instream habitat condition.

Retrospective studies attempt to model historical outcomes, and then use these validated models to identify the relative contributions of climate versus land cover variables through a variety of approaches. Prospective studies similarly develop validated models to predict outcomes, but additionally incorporate one or more future scenarios of climate and/or land use change.

When assessing these studies in aggregate, there are important considerations related to scale: both temporal and spatial. Both retrospective and prospective studies varied in the time periods investigated. Retrospective studies generally considered mid-20th century to recent modern times, but at least two studies considered time periods extending to the end of the 19th century (Owens & Walling, 2002; Yang et al., 2015). Similarly, prospective studies demonstrated considerable variability in defining time periods for future conditions. Some studies considered conditions (that are presently) just a few years away (2020) (Combalicer & Im, 2012; Kharel et al., 2016; Trisurat, Eawpanich, & Kalliola, 2016), while others evaluated potential end of 21st century conditions (Chung, Park, & Lee, 2011; Ma et al., 2014; Montenegro & Ragab, 2012; van Roosmalen, Sonnenborg, & Jensen, 2009; Woltemade & Hawkins, 2016). For prospective studies, additional variability concerns the selection of future scenarios (climate and land use) for evaluation.

Even greater variability was observed in terms of the study area of interest. Watersheds ranged in size from just 35 km^2 to 2.8 million km^2 (median ~2,000 km^2). Importantly, the influence of different drivers of hydrologic change can vary across different spatial scales (DeBano et al., 2016; Yang et al., 2015). In terms of geographic representation, watersheds in China accounted for the largest number of identified studies (14). The United States (7) and Europe (5) held the next highest representation within surveyed literature. While Latin America only accounted for two studies (Brazil and Argentina), it is important to note that our search was restricted to English language results. Only a single study with direct relevance was identified within Africa (Legesse, Vallet-Coulomb, & Gasse, 2003).

Recognizing this variability in study design (in terms outcomes, time periods, and spatial extents, among others) is critical for assessing the interpretability of study findings in aggregate.

Literature reviewed

Bari, M. A., Silberstein, R. P., & Aryal, S. K. (2011). The impact of climate change and plantation development on streamflow in the Denmark River catchment, Western Australia. In Chan, F and Marinova, D and Anderssen, RS (Ed.), 19TH INTERNATIONAL CONGRESS ON MODELLING AND SIMULATION (MODSIM2011) (pp. 3539–3545).

Bronstert, A., Niehoff, D., & Bürger, G. (2002). Effects of climate and land-use change on storm runoff generation: present knowledge and modelling capabilities. Hydrologic Processes, 16(2), 509–529. https://doi.org/10.1002/hyp.326

Buendia, C., Batalla, R. J., Sabater, S., Palau, A., & Marce, R. (2016). Runoff Trends Driven by Climate and Afforestation in a Pyrenean Basin. LAND DEGRADATION & DEVELOPMENT, 27(3), 823–838. https://doi.org/10.1002/ldr.2384

Chung, E.-S., Park, K., & Lee, K. S. (2011). The relative impacts of climate change and urbanization on the hydrologic response of a Korean urban watershed. Hydrologic Processes, 25(4), 544–560. https://doi.org/10.1002/hyp.7781

Combalicer, E. A., & Im, S. (2012). Change Anomalies of Hydrologic Responses to Climate Variability and Land-use Changes in the Mt. Makiling Forest Reserve. JOURNAL OF ENVIRONMENTAL SCIENCE AND MANAGEMENT, (1), 1–13.

DeBano, S. J., Wooster, D. E., Walker, J. R., McMullen, L. E., & Horneck, D. A. (2016). Interactive influences of climate change and agriculture on aquatic habitat in a Pacific Northwestern watershed. ECOSPHERE, 7(6). https://doi.org/10.1002/ecs2.1357

Diaz, R. A., & Querner, E. P. (2005). Estimating climate change effect upon flood risk reduction by afforestation. In Wagener, T and Franks, S and Gupta, HV and Bogh, E and Bastidas, L and Nobre, C and Galvao, CDO (Ed.), Regional Hydrologic Impacts of Climatic Change - Impact Assessment and Decision Making (Vol. 295, pp. 249–253). Retrieved from https://www.researchgate.net/profile/Erik_Querner/publication/40796847_Estimating_climate_change_effect_upon_flood_risk_reduction_by_afforestation/links/00463533857c749063000000/Estimating-climate-change-effect-upon-flood-risk-reduction-by-afforestation.pdf

Guo, J., Su, X., Singh, V. P., & Jin, J. (2016). Impacts of Climate and Land Use/Cover Change on Streamflow Using SWAT and a Separation Method for the Xiying River Basin in Northwestern China. Water, 8(5), 192. https://doi.org/10.3390/w8050192

He, Y., Wang, F., Mu, X., Guo, L., Gao, P., & Zhao, G. (2017). Human activity and climate variability impacts on sediment discharge and runoff in the Yellow River of China. Theoretical and Applied Climatology, 129(1–2), 645–654. https://doi.org/10.1007/s00704-016-1796-8

Hoyer, R., & Chang, H. (2014). Assessment of freshwater ecosystem services in the Tualatin and Yamhill basins under climate change and urbanization. APPLIED GEOGRAPHY, 53, 402–416. https://doi.org/10.1016/j.apgeog.2014.06.023

Johnson, T., Butcher, J., Deb, D., Faizullabhoy, M., Hummel, P., Kittle, J., … Witt, J. (2015). Modeling Streamflow and Water Quality Sensitivity to Climate Change and Urban Development in 20 U.S. Watersheds. JAWRA Journal of the American Water Resources Association, 51(5), 1321–1341. https://doi.org/10.1111/1752-1688.12308

Kharel, G., Zheng, H., & Kirilenko, A. (2016). Can land-use change mitigate long-term flood risks in the Prairie Pothole Region? The case of Devils Lake, North Dakota, USA. Regional Environmental Change, 16(8, SI), 2443–2456. https://doi.org/10.1007/s10113-016-0970-y

Legesse, D., Vallet-Coulomb, C., & Gasse, F. (2003). Hydrologic response of a catchment to climate and land use changes in Tropical Africa: case study South Central Ethiopia. Journal of Hydrology, 275(1), 67–85. https://doi.org/10.1016/S0022-1694(03)00019-2

Liu, Wenbin, Cai, T., Fu, G., Zhang, A., Liu, C., & Yu, H. (2013). The streamflow trend in Tangwang River basin in northeast China and its difference response to climate and land use change in sub-basins. ENVIRONMENTAL EARTH SCIENCES, 69(1), 51–62. https://doi.org/10.1007/s12665-012-1933-3

Liu, Wenfei, Wei, X., Liu, S., Liu, Y., Fan, H., Zhang, M., … Zhan, M. (2015). How do climate and forest changes affect long-term streamflow dynamics? A case study in the upper reach of Poyang River basin. ECOHYDROLOGY, 8(1), 46–57. https://doi.org/10.1002/eco.1486

Liu, Y., Zhang, X., Xia, D., You, J., Rong, Y., & Bakir, M. (2013). Impacts of Land-Use and Climate Changes on Hydrologic Processes in the Qingyi River Watershed, China. JOURNAL OF HYDROLOGIC ENGINEERING, 18(11, SI), 1495–1512. https://doi.org/10.1061/(ASCE)HE.1943-5584.0000485

Liu, Z., Yao, Z., Huang, H., Wu, S., & Liu, G. (2014). LAND USE AND CLIMATE CHANGES AND THEIR IMPACTS ON RUNOFF IN THE YARLUNG ZANGBO RIVER BASIN, CHINA. LAND DEGRADATION & DEVELOPMENT, 25(3), 203–215. https://doi.org/10.1002/ldr.1159

Lopez-Moreno, J. I., Zabalza, J., Vicente-Serrano, S. M., Revuelto, J., Gilaberte, M., Azorin-Molina, C., … Tague, C. (2014). Impact of climate and land use change on water availability and reservoir management: Scenarios in the Upper Aragon River, Spanish Pyrenees. SCIENCE OF THE TOTAL ENVIRONMENT, 493, 1222–1231. https://doi.org/10.1016/j.scitotenv.2013.09.031

Ma, X., Lu, X. X., van Noordwijk, M., Li, J. T., & Xu, J. C. (2014). Attribution of climate change, vegetation restoration, and engineering measures to the reduction of suspended sediment in the Kejie catchment, southwest China. HYDROLOGY AND EARTH SYSTEM SCIENCES, 18(5), 1979–1994. https://doi.org/10.5194/hess-18-1979-2014

Ma, Xing, Xu, J., Luo, Y., Aggarwal, S. P., & Li, J. (2009). Response of hydrologic processes to land-cover and climate changes in Kejie watershed, south-west China. HYDROLOGIC PROCESSES, 23(8), 1179–1191. https://doi.org/10.1002/hyp.7233

Ma, Xing, Xu, J., & van Noordwijk, M. (2010). Sensitivity of streamflow from a Himalayan catchment to plausible changes in land cover and climate. HYDROLOGIC PROCESSES, 24(11), 1379–1390. https://doi.org/10.1002/hyp.7602

Mishra, V., Cherkauer, K. A., Niyogi, D., Lei, M., Pijanowski, B. C., Ray, D. K., … Yang, G. (2010). A regional scale assessment of land use/land cover and climatic changes on water and energy cycle in the upper Midwest United States. INTERNATIONAL JOURNAL OF CLIMATOLOGY, 30(13), 2025–2044. https://doi.org/10.1002/joc.2095

Montenegro, S., & Ragab, R. (2012). Impact of possible climate and land use changes in the semi arid regions: A case study from North Eastern Brazil. JOURNAL OF HYDROLOGY, 434, 55–68. https://doi.org/10.1016/j.jhydrol.2012.02.036

Owens, P., & Walling, D. (2002). Changes in sediment sources and floodplain deposition rates in the catchment of the River Tweed, Scotland, over the last 100 years: The impact of climate and land use change. EARTH SURFACE PROCESSES AND LANDFORMS, 27(4), 403–423. https://doi.org/10.1002/esp.327

Qin, G., Jin, Y., & Geng, S. (2012). Impact of Climate and Land-Use Changes on Water Security for Agriculture in Northern China. JOURNAL OF INTEGRATIVE AGRICULTURE, 11(1), 144–150.

Records, R. M., Arabi, M., Fassnacht, S. R., Duffy, W. G., Ahmadi, M., & Hegewisch, K. C. (2014). Climate change and wetland loss impacts on a western river's water quality. HYDROLOGY AND EARTH SYSTEM SCIENCES, 18(11), 4509–4527. https://doi.org/10.5194/hess-18-4509-2014

Trisurat, Y., Eawpanich, P., & Kalliola, R. (2016). Integrating land use and climate change scenarios and models into assessment of forested watershed services in Southern Thailand. ENVIRONMENTAL RESEARCH, 147, 611–620. https://doi.org/10.1016/j.envres.2016.02.019

US EPA National Center for Environmental Assessment, O. of the D. D., & Johnson, T. (2013). Watershed Modeling to Assess the Sensitivity of Streamflow, Nutrient, and Sediment Loads to Potential Climate Change and Urban Development in 20 U.S. Watersheds (Final Report) [Reports & Assessments]. Retrieved May 31, 2017, from https://cfpub.epa.gov/ncea/global/recordisplay.cfm?deid=256912

van Roosmalen, L., Sonnenborg, T. O., & Jensen, K. H. (2009). Impact of climate and land use change on the hydrology of a large-scale agricultural catchment. Water Resources Research, 45(7), W00A15. https://doi.org/10.1029/2007WR006760

Woltemade, C. J., & Hawkins, T. W. (2016). Stream Temperature Impacts Because of Changes in Air Temperature, Land Cover and Stream Discharge: Navarro River Watershed, California, USA. RIVER RESEARCH AND APPLICATIONS, 32(10), 2020–2031. https://doi.org/10.1002/rra.3043

Xie, X., Liang, S., Yao, Y., Jia, K., Meng, S., & Li, J. (2015). Detection and attribution of changes in hydrologic cycle over the Three-North region of China: Climate change versus afforestation effect. AGRICULTURAL AND FOREST METEOROLOGY, 203, 74–87. https://doi.org/10.1016/j.agrformet.2015.01.003

Xu, X., Yang, D., Yang, H., & Lei, H. (2014). Attribution analysis based on the Budyko hypothesis for detecting the dominant cause of runoff decline in Haihe basin. JOURNAL OF HYDROLOGY, 510, 530–540. https://doi.org/10.1016/j.jhydrol.2013.12.052

Yang, Q., Tian, H., Friedrichs, M. A. M., Liu, M., Li, X., & Yang, J. (2015). HYDROLOGIC RESPONSES TO CLIMATE AND LAND-USE CHANGES ALONG THE NORTH AMERICAN EAST COAST: A 110-YEAR HISTORICAL RECONSTRUCTION. JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION, 51(1), 47–67. https://doi.org/10.1111/jawr.12232

Zhao, Y., & Yu, X. (2013). Effects of climatic variability and human activity on runoff in the Loess Plateau of China. FORESTRY CHRONICLE, 89(2), 153–161. https://doi.org/10.5558/tfc2013-032