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ECOSYSTEMS

Watershed Responses to Climate Change-Driven Disturbances in Temperate Montane Ecosystems of the Western United States

Linnea A. Rock,[1]* Bryce Shoup,[2,3] Joshua A. Ajowele,[4,5] Félix Brédoire,[2] Isabella A. Oleksy,[6] Matthew Tetrick,[2] David G. Williams,[2] and Sarah M. Collins[1]

[1]Department of Zoology and Physiology, University of Wyoming, 1000 E. University Ave, Laramie, Wyoming 82071, USA; [2]Department of Botany, University of Wyoming, 1000 E. University Ave, Laramie, Wyoming 82071, USA; [3]Water Resources/Environmental Engineering Program, University of Wyoming, 1000 E. University Ave, Laramie, Wyoming 82071, USA; [4]Department of Biology, University of North Carolina-Greensboro, 321 McIver st, Greensboro, North Carolina 27412, USA; [5]Department of Ecosystem Science and Management, 1000 E. University Ave, Laramie, Wyoming 82071, USA; [6]Ecology and Evolutionary Biology Department, University of Colorado Boulder, Boulder, Colorado 80309, USA

ABSTRACT

Ecological disturbances driving state changes in ecosystems are likely to be exacerbated with climate shifts during this century. Temperate montane ecosystems of the Western United States (Mountain West) are especially vulnerable due to their low fertility, heterogeneous landscapes, and tight coupling between terrestrial and aquatic components. We review how catchment level pulse and press disturbances will intensify, and how they are reflected by coupled measurements of stream water chemistry and flow. Detecting effects on watershed processes can be complex and depend on the type and extent of disturbance. Within this context, we discuss the impacts of wildfire (pulse), bark beetle outbreaks (pulse), snowpack shifts (press), and progressive vegetation community shifts (press) on streamflow and chemistry dynamics (hydrochemographs). We used long-term data from three mid- to high-elevation watersheds as examples of how disturbances may influence stream hydrochemographs, including increased variability in winter nitrate export in more recent years with extremes in snowmelt runoff export; variable nitrate export when snow water equivalent was abnormally high or low; and high nitrate flux in the years immediately following sudden forest loss. These examples illustrate the need for long-term continuous monitoring to fill the gap in our understanding of the short- and long-term consequences of climate change-induced disturbances to watersheds. As disturbances increase in severity and frequency and induce ecological state changes, it is critical that we develop our understanding of impacts on downstream communities that depend sociologically and economically on water availability and quality.

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Linnea A. Rock and Bryce Shoup are considered joint first author.
David G. Williams and Sarah M. Collins are considered joint senior author.

**Corresponding author; e-mail:* lrock1@uwyo.edu

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HIGHLIGHTS

- Mountain West ecosystems are at risk as climate-driven disturbances intensify.
- Short- and long-term disturbances will alter water quality and quantity.
- Long-term stream monitoring can signal climate-driven shifts in watershed processes

INTRODUCTION

Shifts in climate during this century will drive changes in hydrology (Gergel and others 2017; Musselman and others 2017; Barnhart and others 2020) and vegetation dynamics (Billings and Bliss 1959; Parks and others 2019), and will increase the frequency of large-scale disturbances (Higuera and Abatzoglou 2021; Schapira and others 2021). These changes may be exacerbated by decreases in annual snowpack (Mote and others 2005, 2018; Pierce and others 2008; Gergel and others 2017), earlier snowmelt (Stewart and others 2004; Gergel and others 2017), earlier peak streamflow (Stewart and others 2005; Moore and others 2007), and increases in evapotranspiration (Milly and Dunne 2020; Hostetler and others 2021). Mountain West ecosystems are especially sensitive to climate-induced changes because of their low soil fertility, cold temperatures (Seastedt 2020), distinctive and sparse vegetation cover (Humphries 2020), and heterogeneous microclimates (Elias 2020). These climate shifts can change stream discharge and nutrient concentration which will alter ecosystem functioning, trickling downstream and altering ecosystem services. These changes, reflected in a stream's hydrograph (streamflow time series) and chemograph (water chemistry time series) (Davis and others 2013; Arora and others 2020), are due to a stream's ability to integrate heterogeneous processes in upstream source areas. Hence, streams can act as sentinels of how drainage basins are responding to climatic and biotic disturbances at multiple spatial and temporal scales (Davis and others 2013; Arora and others 2020; Yang and others 2022).

Climate-driven changes act as either a *pulse* disturbance—the result of an episodic event—or a *press* disturbance—the result of cumulative changes in environmental conditions. Pulse disturbances are single events that may occur over short timescales and induce rapid changes in ecosystem processes and properties. Common pulse disturbances in Mountain West ecosystems include events such as wildfires (Kipfmueller and Baker 2000; Larouche and others 2015; Cattau and others 2020) or insect outbreaks that damage vegetation (Kretchun and others 2016). Fire and insect outbreaks (Bentz and others 2009; Kretchun and others 2016) are often viewed as pulse events because the event itself occurs over a short time period, even though the resulting ecosystem changes may take years to be expressed in stream hydrochemograph signals. Alternatively, press disturbances alter ecosystem processes through slow gradual changes, such as from progressive multi-annual to decadal changes in climate that alter hydrology and vegetation (Harris and others 2018) or chronic nutrient deposition that alters balances of plant resource limitations (Smith and others 2009). Pulse and press disturbances in Mountain West ecosystems can unfold simultaneously in interrelated ways, exacerbated by climate change. For example, decreased snowpack may increase the chances for drought, leading to increased risk for insect outbreaks and wildfires. In this review, we explore the benefits of coupling the hydrograph and chemograph to understand more about watershed condition and reactions to disturbances.

THE HYDROCHEMOGRAPH AS AN INTEGRATED MEASURE OF WATERSHED PROCESSES AND CHANGE

Successful, long-term environmental monitoring programs provide process understanding at multiple scales and can detect impacts from pulse and press disturbances (Iwaniec and others 2021). These observations can generate new testable hypotheses and guide management improvements (Lindenmayer and Likens 2010). Taken at a single point in a stream, long-term, high-frequency measurements can detect shifts in biogeochemical processes operating within the stream and surrounding landscape (Laudon and Sponseller 2018; Lewis and others 2019). Additionally, hydrological and biogeochemical observations are likely to be leading indicators of climate-driven system changes (Laudon and Sponseller 2018). Hydrographs are commonly coupled with chemographs to assess impacts of terrestrial processes on water quality and to understand the dynamics resulting from multi-scale processes such as climate change, source water contribution, and watershed storage (Peters and Ratcliffe 1998; Gooseff and others 2002; Kirchner and Neal 2013; Oni and others 2013; Arora and others 2020). Concentration–discharge (cQ) analysis derived from analyzing the slopes of the flow and solute concentration covariance describes nutrient export.

The combination of the hydrograph and chemograph, hereafter the hydrochemograph, is a useful tool for understanding nutrient retention and release within a stream and the surrounding watershed. The coupled movement of water with solutes and nutrients is the ''fingerprint'' of the multi-scale ecological processes within the watershed (Knapp and others 2020), and any significant changes to those processes will likely be captured by the hydrochemograph. Examining nutrient hydrochemographs can provide insight into the influence on ecological processing, like seasonal shifts in nutrient export or in-stream productivity (Kincaid and others 2020). Long-term analysis of nutrient hydrochemographs can help highlight climate-driven pulse or press disturbances that can have repercussions on nutrient imbalances (Yang and others 2022). Local-scale changes such as nutrient retention and availability from the terrestrial to aquatic ecosystems (Rhoades and others 2017; Ren and others 2019b), changes in source water (MacNeille and others 2020), and surrounding vegetation shifts (Wondzell and others 2019) can be explained by the hydrochemograph. These observations may be especially useful in mountain streams, which are often oligotrophic and can be co-limited by nitrogen and phosphorus (Piper and others 2017). Monitoring water and nutrient flux in mountain streams could increase our awareness of how climate change-driven disturbances may impact water quantity and quality; for example, accelerated weathering (Crawford and others 2020) and drying vs. runoff events (Datry and others 2016; Sadro and others 2018; Davenport and others 2020) may mobilize sediment and nutrients from different parts of the watershed at varying magnitudes, and have implications for waters downstream (Kincaid and others 2020). These effects will have cascading and far-reaching effects on ecosystem services, from variable streamflow impacting reservoir storage and management (Willis and others 2011) to socioeconomic stresses in areas reliant on snowmelt as a substantial proportion of water source (Huning and AghaKouchak 2020). Evaluating the shifts in hydrochemographs may lead to deeper understanding of press and pulse disturbances which are projected to become more severe in future climate scenarios.

THE WATER CYCLE IN THE MOUNTAIN WEST

To begin understanding the impacts of climate-driven disturbances on the hydrochemograph, knowledge of the water cycle in the Mountain West is critical. Precipitation, runoff, water storage, evaporation, transpiration, and deep drainage processes impact streamflow and variations in streamflow can provide mechanistic understanding of the disturbance-driven changes. A thorough characterization of hydrologic pathways in mountain systems (left side of Figure 1) is fundamental for understanding disturbance-driven changes (right side of Figure 1). In the Mountain West, snowpack generally starts to accumulate in October–December and melts out in May–June. The late spring and early summer periods are dominated by runoff and shallow flow paths (surface water flows in Figure 1) (Somers and McKenzie 2020). Following the initial pulse of melt water, other water sources, particularly groundwater, become increasingly important to maintain baseflow (subsurface flows in Figure 1). The groundwater component is heavily dependent on local geologic and geomorphic features. Geomorphic features such as talus slopes can act in a twofold manner: first, as a fast-flowing reservoir in high-alpine settings they can have a large storage capacity (Somers and McKenzie 2020) and second, finer-grained deposits beneath the coarse blocks, if present, can act as slow-releasing aquifers capable of sustaining baseflow (Liu and others 2004). Other geomorphic features such as alluvium (Käser and Hunkeler 2016) and glacial moraines (Hood and Hayashi 2015) can also sustain baseflow during late summer months.

Another important flow path that historically was seen as contributing minimal water is groundwater flow through bedrock. Shallow and deep fractured bedrock (Frisbee and others 2011, 2017; Manning and others 2021) can contribute groundwater able to sustain streamflow during summer months (Somers and McKenzie 2020). While bedrock can contribute water to streamflow, the quantity is dictated by the geologic properties of the bedrock. For instance, fracture quantity and depth, permeability, porosity, and hydraulic conductivity can affect the amount of water stored and released. The hydrochemistry observed, in part, is dependent on the source of water (for example, talus fields, groundwaters) and the associated geo-

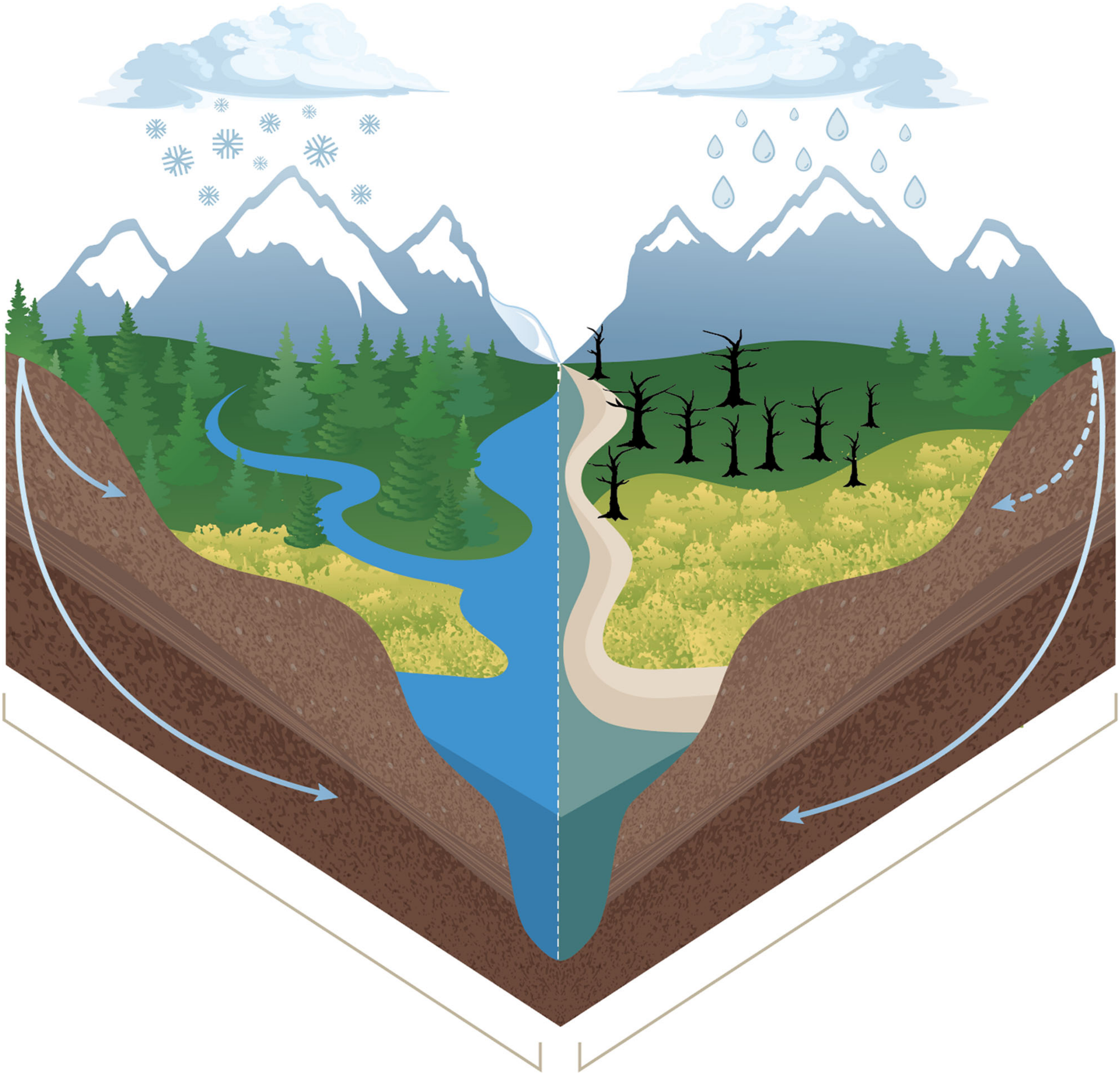

Figure 1. Historic (left) and future (right) conditions in Mountain West ecosystems. Warmer temperatures are altering hydrological processes by decreasing snowpack in high-elevation locations (Gergel and others 2017), inducing earlier melt of snowpack (Barnhart and others 2016; Musselman and others 2017), and shifting dominant precipitation inputs from snow to rain which increases flood-inducing rain-on-snow events (Musselman and others 2018). Shifts in vegetation are represented by forest migration to higher elevations, conversion of forest to non-forest vegetation at lower elevation, and conversion of high-alpine meadow to young forests creating patchy old growth stands mixed with younger new growth (Rust and Minckley 2020). Predicted increases in fire and insect outbreaks will lead to increased rates of forest conversion to non-forest vegetation (Rhoades and others 2017; Rust and Minckley 2020; Higuera and Abatzoglou 2021).

logic properties. Non-conservative ions, such as nitrate, have been shown to have elevated concentrations from talus field outflow compared to snowpack (Williams and others 1997; Campbell and others 2000), which is a function of microbial processes (Clark and others 2021). Nitrate export from talus fields will become increasingly important as groundwater contributions for sustaining baseflow during late summer increases due to changes in precipitation regimes. In groundwaters, conservative ions (for example, Cl^-, F^-, Ca^{2+}, Na^+) are usually enriched due to the sustained interaction with rock. This is further exemplified by groundwaters with increasing residence times containing increasing concentrations of solutes (Rademacher and others 2001). The relative contribution of groundwater discharge compared to other flow sources and associated geochemical signals likely dictate the sensitivity of shifts in the hydrochemograph to pulse and press disturbances that unfold at the surface. The use of hydrochemographs to detect disturbances may be most effective where groundwater discharge plays a minor role.

HYDROCHEMOGRAPH RESPONSE TO CLIMATE-DRIVEN DISTURBANCES

In this study, we used a combination of synthesizing existing literature and case study evidence to develop a conceptual framework that describes how hydrochemographs are useful tools for measuring ecosystem response to climate-driven disturbances. We focus on four disturbances that are likely to increase in frequency or intensity under future climate changes in high-elevation Mountain West watersheds: 1) the transition of precipitation from snow to rain; 2) chronic shifts in vegetation composition; 3) wildfires; and 4) insect outbreaks (Berghuijs and others 2014; Harris and others 2018; Coop and others 2020). Some are rapid, pulse disturbances (for example, fire and insect outbreaks). Others are gradual, press disturbances (for example, snow to rain, vegetation shifts due to slow processes like migration and regeneration). We present hypotheses for how watershed process changes triggered by those four disturbances are detected in the hydrochemograph (Figure 2). One way to assess the impact of these disturbances is having some form of long-term monitoring, providing the baseline information about the hydrochemistry and streamflow controlled by local environmental conditions and geology, but this can be limited by data availability. The conceptual framework presented here is intended to illustrate the utility of hydrochemographs as early warning signs of disturbance, for detecting historical disturbance, and for evaluating disturbance influence in Mountain West watersheds.

Hydrological Changes

As climate warms in the Mountain West, the dominant form of precipitation will shift from snow to rain that will move up in latitude and elevation (Klos and others 2014). This shift will increase the frequency and intensity of rain-on-snow (ROS) events that occur in the Mountain West (Trenberth 2011; Gergel and others 2017; Musselman and others 2018). As a result, lower elevations are likely to see a reduction, or even a suppression of snowpack, with only higher elevations receiving snow, reducing the total annual snowpack and expanding the intermittent snow zone (Gergel and others 2017). Warming is predicted to push the timing of snow melt to earlier in the spring leading to a prolonged snowmelt period (Barnhart and others 2016; Musselman and others 2017; Wu and others 2018). Prolonged snowmelt may reduce soil moisture, reducing the connection to streams (Barnhart and others 2016; Musselman and others 2017) altering current patterns of stream discharge (Berghuijs and others 2014; Fyfe and others 2017). Dust deposition is a factor that may accelerate the alteration of snowmelt and timing by decreasing snow albedo (Rhoades and others 2010). Increased dust deposition has been linked to drought (Prospero and Lamb 2003; Munson and others 2011) and livestock grazing (Schlesinger and others 1990; Neff and others 2005; Fernandez and others 2008). Additional risks include increased chances for large, damaging floods as the likelihood of ROS events increases (McCabe and others 2007; Musselman and others 2018). Ultimately, ROS events and associated flooding may decrease as snowpack decreases or disappears, especially at lower elevations (Musselman and others 2018).

Nutrient export will be impacted by the changes in hydrology. During peak snowmelt, the upper soils layers are flushed of dissolved organic carbon (DOC) (Winnick and others 2017) and soil nitrogen (Sickman and others 2003) that accumulated from the previous growing season. The changes in hydrology may reduce peak flow or extend the rising limb in early spring with similar responses reflected in stream DOC and nitrogen stored in the upper soils. Increasing the number of ROS events will increase the amount of nitrate entering streams earlier in the season. The increase in nitrate comes directly from rainfall as wet deposition and through melting snowpack containing nitrate from atmospheric deposition. The excess nitrate cannot accumulate in the soil profile due to overlying snowpack and an already saturated soil profile (Eimers and others 2007; Casson and others 2014), resulting in observable nitrate pulses in the chemograph. These pulses act as early flushes in the system leading to an overall decrease in nitrate during peak flow, but not necessarily a total reduction in nitrate entering the system on a yearly basis.

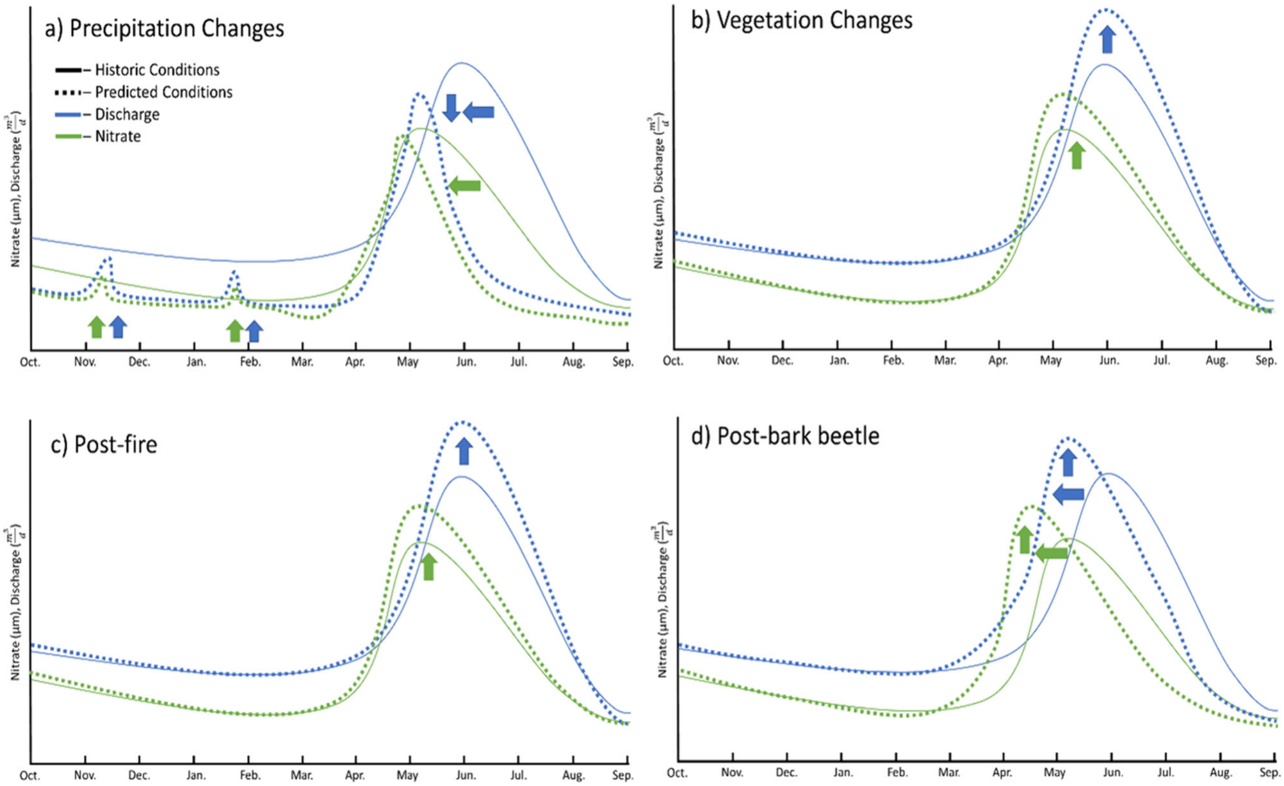

Figure 2. Hypothetical hydrochemographs representing **A** rain-on-snow events, **B** changing from a forested system to a meadow, **C** post-fire, **D** post-bark beetle. Arrows indicate direction of shift following the observed disturbance.

Figure 2a conceptualizes changes to the hydrochemograph resulting from changes in hydrology on an annual basis. Major changes in the hydrograph are represented by decreased and earlier peak runoff with several small peaks occurring through the winter months. A reduction in peak discharge results from decreased snowpack accumulation during winter months (Gergel and others 2017). The shift in timing is a product of warmer temperatures in the late spring and early summer months (Gergel and others 2017). Although the melt out period will be longer, it will not necessarily be reflected in the peak discharge curve due to reduced soil moisture leading to a reduced connection to the streams (Barnhart and others 2016; Musselman and others 2017). ROS events create minor peaks which are a combination of rainfall and snow melt (Pomeroy and others 2016). Nitrate follows a similar pattern as peak discharge with noticeable releases of nitrate during ROS event. As with the discharge peaks, minor nitrate peaks are a function of rainfall and snow melt (Eimers and others 2007; Casson and others 2014).

Vegetation Shifts

Mountain West forest vegetation has historically been resilient to disturbance events for over millenniums, with no major changes in species composition as long intervals between disturbances provided enough time for forest vegetation to recover (Romme and others 2016; Stegner and others 2019; Strand and others 2019). However, increased frequency and severity of stand-replacing disturbance events and unsuitable post-disturbance conditions for seedling establishment result in forest vegetation moving beyond a recovery threshold (Harvey and others 2016b; Hansen and others 2018; Davis and others 2019; Hoecker and others 2020; Parks and Abatzoglou 2020; Rodman and others 2020). Homogenization of the landscape may result in decline in forest resilience due to unavailable seed sources in the surrounding landscape (Harvey and others 2016a).

Interactions between disturbance types may have compounding effects on forest vegetation. For example, insect outbreak affecting mature trees could limit seed production prior to stand-replacing fire affecting forest vegetation recovery (Harvey and others 2013; Carlson and others 2017, 2020). In addition, climate-induced warmer and drier post-disturbance conditions reduces seedling densities and survival of established tree seedlings (Stevens-Rumann and others 2018; Hansen and Turner 2019) which may lead to forest conversion (Coop and others 2020) in the absence of forest management. Models of future climate scenarios, stand-replacing fire, and forest regeneration suggests forests will be replaced with non-forest vegetation in the Mountain West (Davis and others 2019; Parks and others 2019; Rammer and others 2021). With such an expected reduction in forest cover in the Mountain West, it is important to quantify the effect on ecosystem services, and to understand different trajectories that may occur under active forest management strategies (*for example*, post-wildfire salvage and planting vs. natural regeneration).

Forest vegetation provides important ecosystem services such as regulating water flow and quality by preventing flooding through evapotranspiration, precipitation interception, and absorption of nutrients (Bosch and Hewlett 1982; Hornbeck and others 1993; Rhoades and others 2011; Biederman and others 2014; Beier and others 2015). Litterfall can also contribute to these processes through the building and maintenance of soil organic matter, which is important for soil water retention and the maintenance of thriving microbial communities that immobilize nutrients and prevent leaching to the streams. Therefore, conversion of forested to non-forest vegetation has the tendency to disrupt livelihoods from increased flooding and nutrient input downstream, which could take up to 65 years to offset by recovering forest vegetation (Stednick 1996; Beier and others 2015; Caputo and others 2016). Changes in mountain vegetation resulting in increased bare ground promote nutrient export downstream, alleviating nitrogen and phosphorus limitations, which can reduce water quality through eutrophication (Visser and others 2016; Ren and others 2019b; Oleksy and others 2020) depending on substrate type, age, and soil mobile nutrient content. To understand the impacts of a switch from forest to non-forest vegetation, a conceptual hydrochemograph is presented in Figure 2b. During an annual period, both the hydrograph and chemograph experience increases in peak values, with the rest of the hydrochemograph behaving similarly to pre-disturbance conditions. The reduction of leaf area index associated with the loss of tree cover allows for higher snow accumulation and reduced transpiration, increasing peak discharge (VanShaar and others 2002). Concomitantly, nutrient export to streams increases, also exacerbated by the loss of vegetation uptake as observed for nitrate (Beier and others 2015; Caputo and others 2016). The changes presented in Figure 2b are from two time points; if each individual annual hydrochemograph were plotted, it would be a slow evolution with slight change between interannual variability.

Fire Disturbance

Fire regimes are rapidly changing in Mountain West ecosystems. More frequently, large fires are occurring as a result of climate-induced increases in fuel load combined with increased ignition frequency from lightning and anthropogenic activities (Kipfmueller and Baker 2000; Cattau and others 2020). Additionally, fire suppression policies have resulted in increased fire severity with a projected 65% loss of forest cover by 2098 (Hansen and others 2020). Increasing exposure to viable fuel (Beverly and others 2021) along with wildfire's positive correlation with fluctuating drought years (Dewar and others 2021) means the Mountain West is likely to continue to experience high-severity fires.

While fires can be considered a ''pulse'' disturbance, their impact on the hydrochemograph can often persist for years, especially following high-severity burns. In Mountain West ecosystems, streamflow responses after wildfires are highly heterogeneous and likely depend on several factors that interact to determine the magnitude of the increase in hydrologic flows: fire severity, post-fire precipitation, and stream network position. In general, across ecosystems, streamflow has been shown to increase for several years after a fire in forested watersheds, but many systems do recover to pre-fire discharge after 5–10 years (Saxe and others 2018; Williams and others 2022). The magnitude of the streamflow response is heavily influenced by the proportion of the watershed that burned at high severity (Hallema and others 2018) and the amount, intensity, and type of precipitation that falls in the years following wildfire disturbance (Wine and Cadol 2016). Specifically, watersheds with a larger proportion of high-severity burn area will be more vulnerable to extreme precipitation events due to limitations in vegetation recovery and reductions in soil infiltration rates (Saxe and others 2018; Wine and others 2018). Furthermore, streams lower in a river network are likely less sensitive to changes in hydrologic flows post-fire, and empirical work shows dampened responses to post-fire precipitation relative to lower-order streams (Reale and others 2015). While hydrologic responses can be variable, Figure 2c presents a generalized understanding of the impacts of wildfires on hydrology. Post-fire increases peak discharge with little to no change in baseflow conditions. This is a function of increased snowmelt, reduction of evapotranspiration, and decreased soil infiltration (Seibert and others 2010).

Although streamflow may increase after a fire, nutrient export does not always respond in tandem nor do all nutrients respond in synchrony. For instance, a recent meta-analysis showed that the largest increases in stream chemistry after wildfire were with nitrate, total phosphorus, and ammonium with relatively minor changes in phosphate, dissolved organic carbon, and total suspended solids (Hampton and others 2022). Often, investigators notice a several-year lag response in nutrient export after wildfire (Rhoades and others 2011). In some cases, particularly following the most extreme burns, nitrate concentrations may remain chronically in streams, fundamentally altering instream biogeochemical function, such as by increasing autotrophic productivity (Rhea and others 2021). The hypothetical chemograph presented in Figure 2c represents increases in observed nitrate during peak runoff. This hypothetical hydrochemograph behaves similarly to vegetation shifts in magnitude, but a key difference is the timescale the change occurs. Fire outbreak response will usually occur over a much shorter time span. Other stream constituents, like dissolved organic carbon or total suspended solids, show complex and context dependent responses to fire. For example, DOC levels post-fire may be re-volatilized, but in moderately burned catchment DOC may remain elevated for years following fire (Rhoades and others 2019).

Bark Beetle Outbreaks

Similar to fires, bark beetle outbreaks are a natural disturbance process in temperate mountain ecosystems, where insects such as the Mountain Pine Beetle (*Dendroctonus ponderosae*) cyclically kill about 2% of conifer trees in affected watersheds but have intensified in recent years due to climate change, management legacies, and insect adaptation (Raffa and others 2008; Bentz and others 2010; Williams and others 2010; Meddens and others 2012; Mitton and Ferrenberg 2012; Creeden and others 2014). As their geographical range is expanding with climate change, bark beetles are a key player in the shift of forest disturbance regimes (Bentz and others 2010; Weed and others 2013). Climate change is resulting in unprecedented increase in frequency and severity of disturbance events with the potential for more frequent interaction between different disturbance types (Higuera and Abatzoglou 2021; Schapira and others 2021). The evidence for whether insect outbreaks and fire are likely to interact is equivocal, with some studies noting that insect outbreaks prior to fire in the Western United States may have no effect on fire (Harvey and others 2013; Hart and others 2015; Andrus and others 2016). In contrast, previous fire disturbance in subalpine forest promotes resistance to insect disturbance as older trees that are more susceptible to insect attacks are replaced by more disconnected, younger trees that impeding the spread of subsequent insect outbreaks (Kulakowski and others 2012; Seidl and others 2016). The effect of a prior disturbance may be positive, negative or have no effect on subsequent disturbance depending on weather condition, intensity, and timing of disturbance (Harvey and others 2014a, 2014b; Andrus and others 2016; Meigs and others 2015), making it imperative to consider interaction among disturbance types in developing a framework for management in Mountain West ecosystems.

The effects of bark beetle outbreaks on the hydrochemograph are likely to be context dependent, but the alterations of hydrological and biogeochemical processes they induce are generally of lower magnitude than those of wildfires (Pugh and Gordon 2013). Bark beetle outbreaks typically unfold in three phases where changes in tree functioning and stand structure drive the sequence and intensity of the events recorded on the hydrograph (Wulder and others 2006; Mikkelson and others 2013a; Pugh and Gordon 2013). First, in the ‘‘green phase’’ during the weeks following the infection, the host trees die from the interruption of sap flow (Hubbard and others 2013) and reduced transpiration may increase soil moisture, but no effects on hydrology are anticipated during this phase (Pugh and Gordon 2013). In the second phase (‘‘red phase’’), dead trees may retain their needles for 1–3 years as they progressively turn red and fall (Wulder and others 2006). In this phase, canopy opening might not be important enough to allow for higher snow accumulation, but snow melt is accelerated by the needle litter that reduces albedo, eventually triggering earlier peak flow (Pugh and Small 2012; Pugh and Gordon 2013; Livneh and others 2015). Once all needles have fallen, dead standing trees enter the third phase (‘‘gray phase’’), which may last years to decades (Lewis and Hartley

2006; Mikkelson and others 2013a). The gray phase is likely to have increased streamflow due to higher snow accumulation and reduced transpiration, and earlier peak flow due to earlier and faster snowmelt (Love 1955; Bethlahmy 1974; Potts 1984; Pugh and Small 2012; Mikkelson and others 2013c; Livneh and others 2015). The hydrological response is expected to be proportional to the forest removed, with lesser response in dry regions or periods (Bosch and Hewlett 1982; Stednick 1996; Brown and others 2005; Pomeroy and others 2012; Biederman and others 2014).

In addition to their effects on hydrology, bark beetle disturbances are likely to influence nutrient cycling and stream chemographs. Following bark beetle outbreaks, elevated in-stream nutrients and organic matter have been observed (Clow and others 2011; Mikkelson and others 2013a, 2013b; Bearup and others 2014), partly due to increases in litter breakdown and vegetation productivity (Clow and others 2011). Old-growth stands, typically unmanaged and where mortality from bark beetle outbreaks is higher, are more prone to leak nitrate to streams than mixed-aged and managed stands (Rhoades and others 2017). Drier regions, with sparser understory vegetation and slower regeneration (Lotan and Critchfield 1990; Strong 2015), may be more prone to leach inorganic nitrogen (Rhoades and others 2017). With the prospect of earlier snow melt and/or patchier snowpack, soil microbes might not be able to efficiently immobilize inorganic nitrogen during winter, which could promote its export to streams at snowmelt (Brooks and Williams 1999; Mikkelson and others 2013a). The biogeochemical effects of bark beetle outbreaks can also extend beyond nutrients as organic matter and nitrate increase and soil pH decreases, mobilizing and leaching cations and metals from soils to stream water (Likens and others 1969; Bearup and others 2014; Mikkelson and others 2014). The intensity of the sorption, complexation, and precipitation mechanisms controlling metal mobility in soil and to the streams are likely to evolve through the different phases following a bark beetle outbreak, but observations are lacking (Bearup and others 2014).

While bark beetle outbreak effects occur over a multi-year period, Figure 2d conceptualizes the generalizable changes to the hydrochemograph. At intermediate timescales (1–3 years of the red phase), beetle outbreaks result in earlier snowmelt due to lower-snow albedo (increased needle litter) causing earlier peak discharge (Pugh and Small 2012; Pugh and Gordon 2013; Livneh and others 2015). At longer timescales (gray phase that may last decades), higher snow accumulation along with increased radiation results in earlier snowmelt and increased discharge (Love 1955; Bethlahmy 1974; Potts 1984; Pugh and Small 2012; Mikkelson and others 2013c; Livneh and others 2015). We would also expect increases in nitrate export in ecosystems with high nitrogen deposition (Zimmermann and others 2000; Huber and others 2004; Tokuchi and others 2004; Huber 2005; Jung and others 2021).

Assessing Change Using Hydrochemographs through Case Studies

Hydrochemographs are useful tools for examining water quantity and quality and can increase our understanding of watershed-scale changes in response to disturbances. Here, we use three examples of watersheds with high-frequency, long-term monitoring to demonstrate how we can detect and visualize responses to climate change-induced disturbances The three examples presented here emphasize the importance of long-term monitoring data for using the conceptual framework to detect, identify, and evaluate climate-driven disturbances via hydrochemographs. First, we evaluate stream response to multi-year drought and long-term warming air temperatures using data from Loch Vale, a high-elevation watershed in the central Rocky Mountains. We next examine variation between high- and low-snow years in another high-elevation watershed in the Rocky Mountains at Niwot Ridge. Finally, we evaluate the mid-elevation Andrews Experimental Forest and the consequences of mountainous forest clear-cut. The openly available data used in these case studies demonstrate the invaluable benefit of long-term data collection, allowing scientists to observe patterns and anomalies in streamflow and solute export.

In all three of our examples, we examine streamflow and nitrate export. Loch Vale data were gathered directly from their data maintenance site (USGS, US Geological Survey 2024), while Niwot Ridge (Caine 2021) and Andrews Experimental Forest data (Johnson and Fredriksen 2019) were gathered from Macrosheds using the Macrosheds R package (Rhea and others 2023; Vlah and others 2023). We chose these case studies because of the publicly available long-term, high-frequency data that highlights the importance of monitoring mountain streams to detect disturbances whose effects can propagate downstream at the basin scale. The three mid- to high-elevation case study locations are also already showing signs of climate change manifested as increasing summer air temperatures (Figure S1 Modeling and others 2015). We analyzed the data in the R programming language (R Core Team 2024). The tidyverse (Wickham and others 2019) package was used for data wrangling and visualization, and the dataRetrieval package's addWaterYear function was used to format the data into water year—October–October (De Cicco and others 2018); water years, rather than calendar years, were used for all analyses. Please see the supplement for more details and methods for each case study.

Loch Vale, Colorado

The Loch Vale outlet of a subalpine lake is at elevation 3105 m in the Rocky Mountains and represents a watershed minimally disturbed by direct human influences. We discretized the Loch Vale outlet data into three periods to distinguish typical climate periods from a drought period using Palmer Drought Severity Index, annual precipitation, and nitrate deposition data: 1990–2000, 2001–2007 (drought), and 2008–2019. The decades are further divided into three seasons (Figure S2a): winter (December to April), snowmelt runoff (May–June), and summer (July–November). The seasons coincide with strong snowmelt-dominated streamflow patterns with little to no flow over winter, increasing streamflow during snowmelt runoff, and decreasing streamflow during summer (Figure 3a).

As demonstrated by Mast and others (2014), nitrate concentrations peaked in the middle period (2001–2007) in response to drought (Figure 3a), then returned to pre-2000 levels during the 2008–2019 decade. The increased nitrate levels in the early 2000s were further distinguished from the other decades by the cumulative nitrate export curves with the highest sum during the drought period $>$ 500 kg greater than the typical sum (Figure 3b). Using cQ analysis, the long-term Loch Vale record revealed how the watershed can switch between periods of chemostasis (no change in concentration as streamflow changes), mobilization (increased concentration with streamflow), and dilution (decreased concentration with increased streamflow) within years and across time periods (Figure 3c–d). Most notably, winter cQ during the most recent time period 2008–2019 (blue triangles in Figure 3c–d) demonstrated stronger variability including periods of mobilization, resembling the snowmelt runoff slopes.

Three of the five lowest snow water equivalent (SWE) years in our record (1980–2019) were 2018 (379.890 kg m^{-2}), 2013 (430.0234 kg m^{-2}), and 2016 (505.160 kg m^{-2}), with the other two lowest occurring in 1981 and 1991 (NLDAS Project 2020). Low SWE (Figure S3a) along with increasing summertime air temperature (Figure S1a) may be indicative of earlier melt and greater precipitation as rain rather than snow, driving the winter cQ pattern toward mobilization. This trend is further evidenced by the streamflow duration curves that show winter streamflows at minimal values $<$ 83% of the time from 1990 to 2007, contrasted with minimal values met or exceeded as high as $>$ 99% in the most recent time period (blue lines in Figure 3e). Trends of earlier snowmelt and warming temperatures have been demonstrated across Colorado watersheds (Clow 2010). If winter mobilization trends continue, the hydrograph may shift toward earlier high flows, decreasing water and nutrient availability later in the season (Figure 2a).

Niwot Ridge, Colorado

Green Lake 4 is an alpine lake in the Green Lakes Valley at Niwot Ridge. Here we examine variation in the outlet hydrochemograph of Green Lake 4, located at elevation 3550 m in the Rocky Mountains. Like Loch Vale, this location is representative of a watershed minimally disturbed by direct human influences, with the exception of mining activity and water level manipulations at lower elevations in the watershed. We again found three seasons examining the average monthly SWE: winter (December–May), snowmelt runoff (June–July), and summer (August–November). The seasons coincide with strong snowmelt-dominated streamflow patterns with little to no flow over winter, increasing streamflow during snowmelt runoff, and decreasing streamflow during summer (Figure 4a).

We determined the highest (2011 at 2121.71 kg m^{-2}) and lowest (1991 at 464.972 kg m^{-2}) SWE years using cumulative annual SWE data from 1986–2019 (Figure S13b; NLDAS Project 2020), paired with when we had streamflow and nitrate data available. Comparing the high- and low-snow years (solid and dotted lines in Figure 4b, respectively) to the average hydrochemograph of the full data record (Figure 4a), nitrate concentrations were similar to average. High- and low-snow year nitrate demonstrated opposite summertime peaks with a spike in nitrate during the low-snow year and a drop during the high-snow year. High-snow

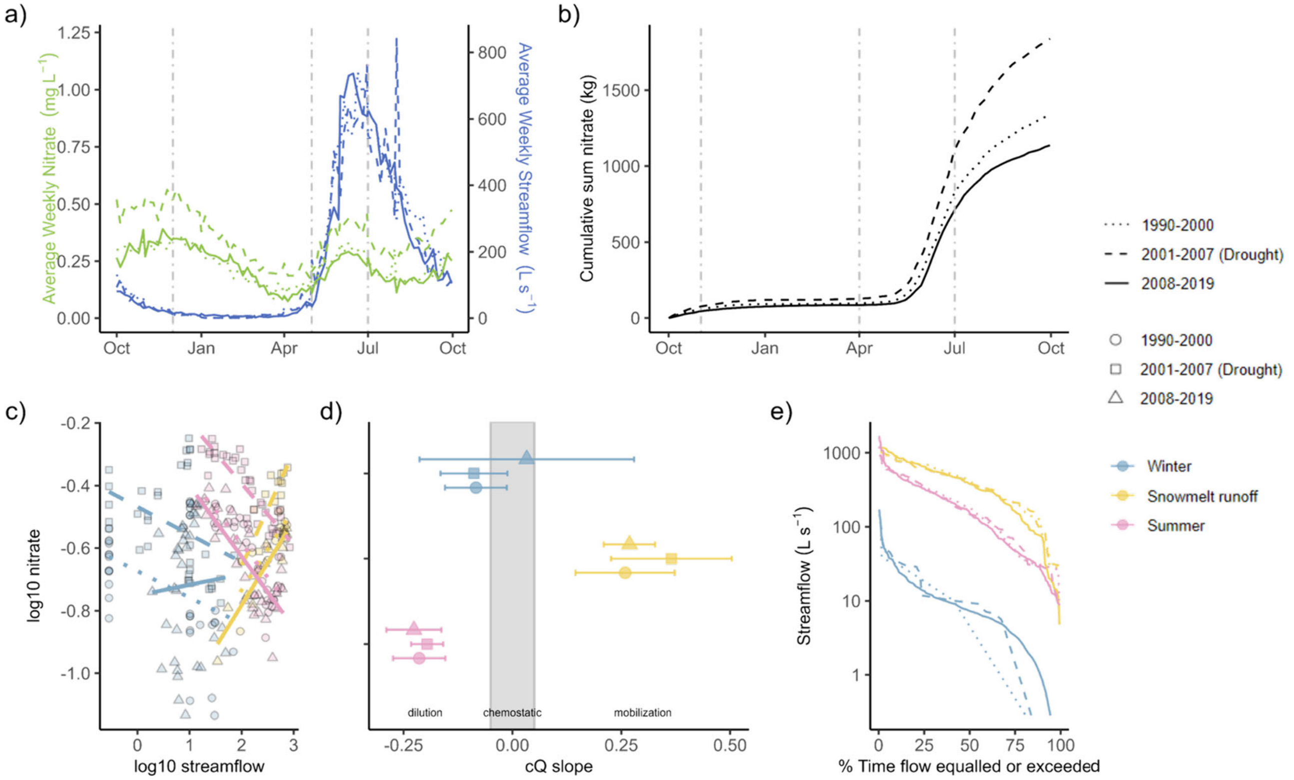

Figure 3. Hydrochemographs reveal seasonal and climate differences in the Loch Vale outlet in the Rocky Mountains, USA. Time periods are distinguished by line type and shape with 1990–2000 represented by dotted lines and circles, 2001–2007 (drought years) represented by dashed lines and squares, and 2008–2019 represented by solid lines and triangles. Seasons are divided into winter (December–April, blue), snowmelt runoff (May–June, yellow), and summer (July–November, pink). **a** Average weekly nitrate (green lines) and streamflow (blue lines) over the water year within each time period. **b** Average yearly cumulative sum of nitrate in kg over the water year, distinguished by time period. Vertical dashed lines in a) and b) display the seasonal breaks. **c** Seasonal nitrate concentration–discharge (cQ) on the log–log scale. d) Summary of seasonal cQ slopes (and 95% confidence intervals) with the gray bar representing the chemostatic range (slope <|0.05|). **e** Flow duration curves, distinguished by time period and season, displaying the percent of time streamflow rates were met or exceeded.

year streamflow was similar to average but peaked later in year. Streamflow was only collected in May during the low-snow year winter, making clear assessment of this season difficult. However, streamflow was substantially less than average during the low-snow year snowmelt runoff and summer seasons.

On average, nitrate–streamflow cQ slopes (circles in Figure 4c) demonstrate mobilization in winter (blue circle), high variability during snowmelt runoff (yellow circle), and dilution in summer (pink circle). Interestingly, both the high- and low-SWE years demonstrated dilution during the snowmelt runoff period (yellow square and triangle, respectively) but not during the summer period (pink square and triangle, respectively). During the high-SWE year (squares in Figure 4c), there was significant variation in winter (blue square) ranging from very dilute (slope of -4.97) to very mobile (slope of 3.32). Large swings in water availability are likely to increase in frequency with press disturbances like climate warming and precipitation shifts (Datry and others 2016; Davenport and others 2020) and could unpredictably mobilize nutrients or other solutes within the watershed (Kincaid and others 2020). As demonstrated in the Loch Vale case, high variability occurred in the most recent decade on record and may be indicative of warming temperatures and earlier onset of snowmelt. Changes like these observed in the hydrochemograph can serve as warnings of imbalanced nutrient content (Yang and others 2022), increased phytoplankton growth (Sadro and others 2018), and shifts in biotic communities (Datry and others 2016).

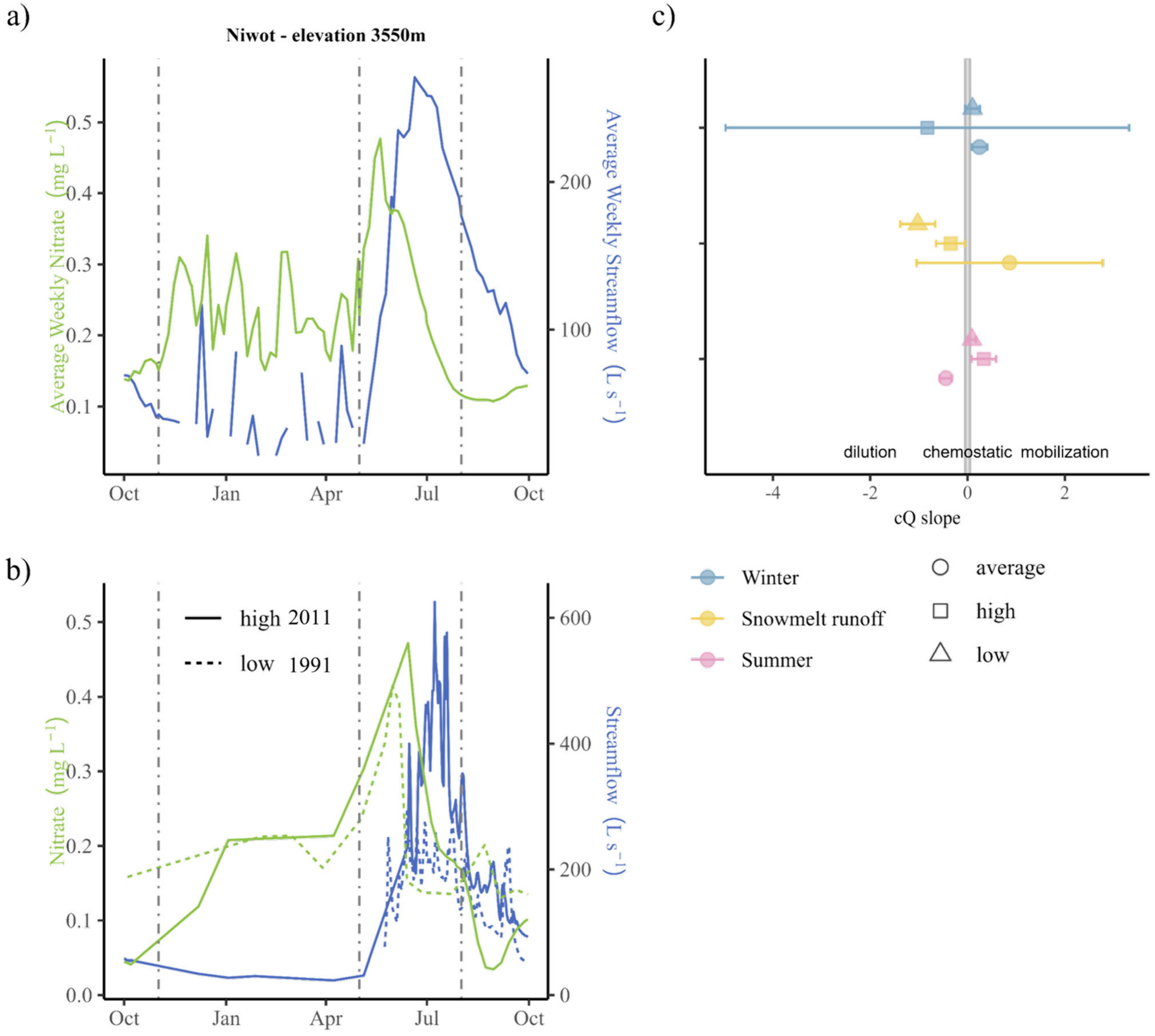

Figure 4. Hydrochemographs demonstrate variation in nutrient export between high- and low-snow years in the Green Lake 4 outlet in the Rocky Mountains, USA. **a** Average weekly nitrate concentrations (green lines) and streamflow (blue lines) from 1986–2019. b) Nitrate concentrations (green lines) and streamflow (blue lines) in the highest snow year (2011, solid lines) and lowest snow year (1991, dashed lines). Vertical dashed lines in a) and **b** display the seasonal breaks, with winter (December–May), snowmelt runoff (June–July), and summer (August–November). **c** Summary of nitrate concentration–discharge (cQ) slopes with 95% confidence intervals on the log–log scale. Gray bar represents the chemostatic range (slope $< |0.05|$).

HJ Andrews Experimental Watersheds, Oregon

The HJ Andrews Experimental Forest (HJA) site is a mid-elevation forested area in Oregon. In this example, we compared two watersheds with high-frequency, long-term monitoring. Watershed 6 has a maximum elevation of 1029 m, with the stream monitored at 878 m, and was clear-cut 100% in 1974. We compared Watershed 6 to a nearby control site, Watershed 8, which has a maximum elevation of 1182 m, with the stream monitored at 962 m. The two HJA watersheds are lower elevation and have different streamflow and watershed characteristics compared to our other two case studies but provided a broader range of examples from western montane watersheds with high-frequency, long-term monitoring.

In this example, we present a longer-term look at the Watersheds 6 and 8 hydrochemographs and observe the shift in nitrate concentrations, yet not in streamflow (Figure 5a). Furthermore, we observed a change in area-normalized nitrate flux (kg ha^{-1} day^{-1}) between the two watersheds after clear-cutting by examining the flux in Watershed 8 subtracted from the flux in Watershed 6 (Figure 5b). Area-normalized nitrate flux was calculated as:

$$\text{Flux} = \frac{(\text{nitrateconcentration}(\text{mg/L}) * \text{streamflow}(\text{L/s}) * \text{dailytimeinterval}(\text{s}))}{(\text{watershed area (ha)} * 1000000)}$$

Prior to the clear-cut logging in 1974, Watersheds 6 and 8 showed no substantial difference in their hydrochemographs. Logging occurred May–August 1974 in Watershed 6. The hydrochemographs demonstrate the loss of forested vegetation did not influence streamflow (in opposition to our prediction in Figure 2b), yet nitrate concentrations were higher in Watershed 6 than in the un-logged control Watershed 8 (Figure 5a, in agreement with our prediction in Figure 2b). The high nitrate values in Watershed 6 are further observed via the difference between area-normalized nitrate flux in Watersheds 6 and 8, in which the difference is higher between 1976 and 1980 compared to before clear-cutting and after (Figure 5b). The high nitrate concentrations continued until 1980, when Watershed 6 returned to baseline conditions similar to Watershed 8.

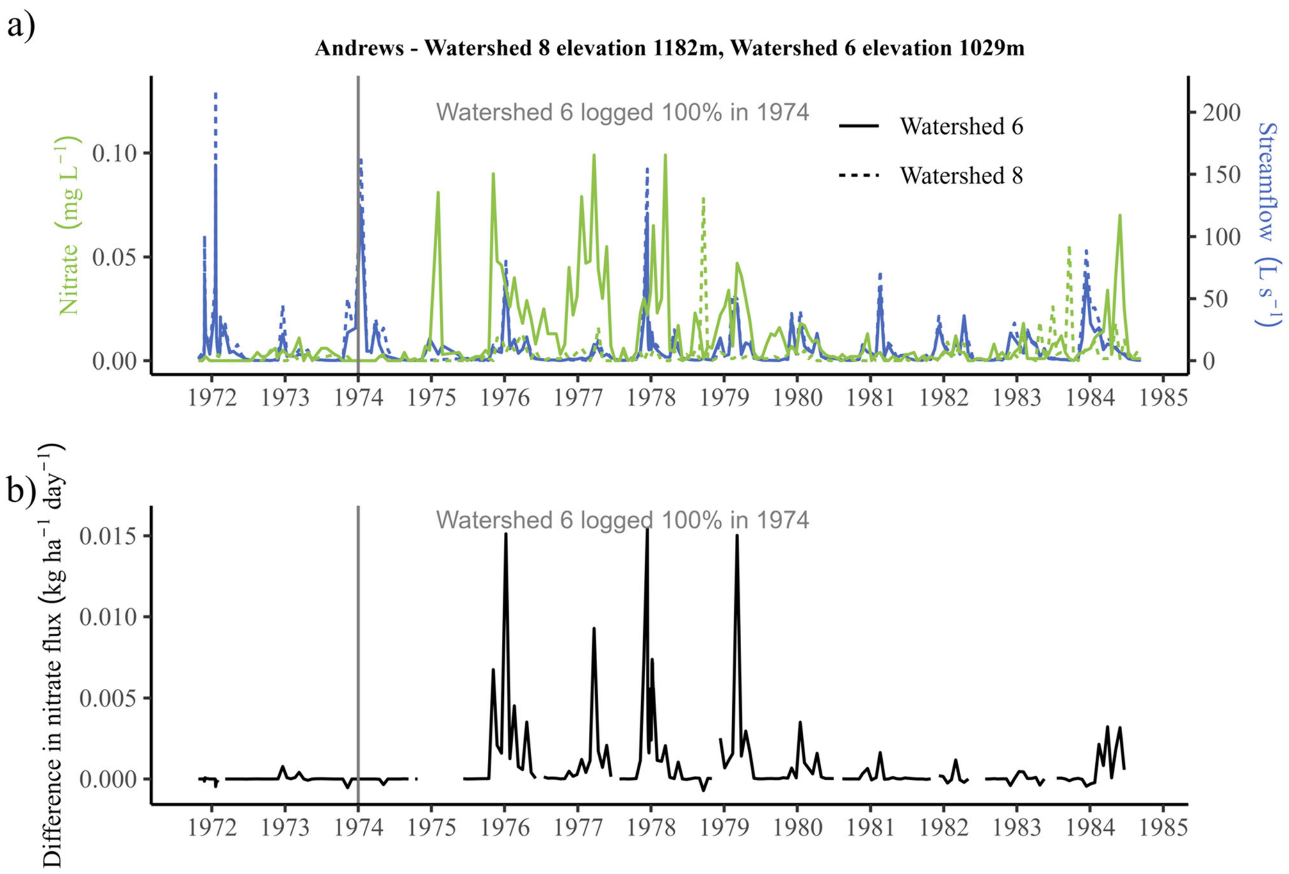

Figure 5. Hydrochemographs over a decade demonstrate high nutrient export after forest reduction in Watershed 6 compared to control Watershed 8 in montane Oregon, USA. **a** Nitrate concentrations are depicted in green and streamflow by blue. The watersheds are delineated by line type, with the control Watershed 8 shown by solid lines and the logged Watershed 6 shown by dashed lines. **b** Difference of area-normalized nitrate flux between Watershed 6 and Watershed 8. The vertical gray line at 1974 in both a and b displays the year Watershed 6 was clear-cut.

The clear-cutting disturbance in HJA Watershed 6 demonstrates the results of a pulse loss of forest vegetation. As a climate-induced disturbance, there may be differences between vegetation loss due to bark beetle (pulse disturbance) and a shift away from forested vegetation as community composition shifts (press disturbance). For both bark beetle and vegetation shifts, increased streamflow and nitrate are expected along with changes in peak timing with nuances depending on the disturbance. For example, long-lasting removal of large woody biomass due to press vegetation community composition shifts results in higher nitrate export to streams (Beier and others 2015; Caputo and others 2016). At near to intermediate timescales after a bark beetle outbreak, increased pine needle litter lowers snow albedo, resulting in earlier peak discharge (Pugh and Small 2012; Pugh and Gordon 2013; Livneh and others 2015). Forest cover in western montane ecosystems of the USA is expected to reduce and transition to non-forest vegetation as climate change progresses and disturbances persist (Stevens-Rumann and others 2018; Davis and others 2019; Coop and others 2020; Rammer and others 2021). As bare ground can increase nutrient export, we expect water quality degradation to accompany forest reduction (Visser and others 2016; Ren and others 2019a; Oleksy and others 2020).

IMPLICATIONS

High-elevation montane ecosystems may be especially susceptible to climate change. Mountain West aquatic environments, which provide important ecosystem services, will likely deteriorate as the consequences of climate change continue to be amplified. Increased fire, biotic stressors, precipitation occurring as rain rather than snow, and vegetation shifts are some of the main pulse and press disturbances changing Mountain West ecosystems. These can have cascading effects within alpine ecosystems that trickle to downstream ecosystem services. For example, as snow drought becomes more prevalent in the Western United States, there could be strong impacts on water quality and quantity in surrounding landscapes suffer, which could have socioeconomic consequences (Huning and AghaKouchak 2020). Disturbances occur naturally and often in the environment. The influence of and recovery from a disturbance varies with disturbance type (for example, pulse vs. press), the sensitivity of the ecosystem, and the severity of disturbance (Chapin and others 2011). While ecological memory may enhance resilience in some cases, changes in severity, frequency, magnitude, and timing of disturbances have the potential to restructure ecosystems (Johnstone and others 2016).

Biogeochemical cycling in high-elevation streams has shown dramatic responses to climate change-driven disturbances, like shifts in carbon and nutrient dynamics (Ren and others 2019b; Zhi and others 2020) and transformation of alpine streams from CO_2 sinks to sources (Ulseth and others 2018). Additionally, hydrologic conditions of alpine streams will be altered under a new climate with potential for both drying of streams with less snowpack (Datry and others 2016) and exponentially increased stormflows and flooding with rainfall (Davenport and others 2020). These extreme swings between drought and flood alter organismal communities, migration patterns, nutrient processing (Datry and others 2016), and overall threaten biodiversity (Pittock and others 2011). Variable flows can also be problematic for strategizing reservoir storage and management (Willis and others 2011). Understanding whether these changes are reversible and the magnitude of these changes on ecosystem health and services requires further research.

High-frequency, long-term monitoring of Mountain West streams' hydrology and nutrient fluxes can detect signals of pulse and press disturbances occurring over varying ecosystem scales. Often, ecological data are collected infrequently and over short periods of time, which have benefited our knowledge and understanding of ecological processes. However, longer datasets enable detection of early warning signs of climate-driven changes to critical processes and services (Williams and others 2007). The disturbances we discuss in this manuscript are already affecting the Mountain West. Increasing our monitoring efforts and developing more long-term, continuous monitoring sites are important steps to progress our grasp on these changes to the hydrochemograph, which could have profound consequences for downstream aquatic habitats and ecosystem services.

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DATA AVAILABILITY

The code and data used for analyses and to create the manuscript figures are located at https://zenodo.org/records/13770704. All raw stream chemistry and flow and climate data are openly available and cited.

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