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**Key Points:**

- Runoff ratios, nitrate, and dissolved organic carbon exhibited unique thresholds in area burned, which differed with burn severity
- Post-fire hydrologic changes enhanced dissolved organic carbon transport to streams, while nitrate transport remained relatively unaffected
- Dissolved organic carbon responses were sensitive to area burned and burn severity, while nitrate was sensitive to area burned and aridity

**Supporting Information:**

Supporting Information may be found in the online version of this article.

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# When Do Riverine Systems "Feel the Burn"? Simulating How Burn Extent and Severity Modulate Hydrologic Controls on Biogeochemical Export

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**Abstract** Wildfires impact terrestrial landscapes and downstream river corridors through shifts in vegetation and soil properties leading to downstream hydrologic and water quality impacts. The magnitude of these impacts depend on a complex and interconnected set of wildfire, landscape, and aquatic processes. Here, we isolate the impact of post-fire hydrologic changes on streamflow, nitrate, and dissolved organic carbon using the Soil and Water Assessment Tool (SWAT) model. We explore how responses differ across burn severity and area burned in two test basins: a humid forested basin and a semi-arid mixed land use basin. We ran 1830 wildfire simulations testing impacts of area burned, burn severity, and post-fire precipitation on streamflow, nitrate, and dissolved organic carbon. Our work suggests that area burned thresholds differ with burn severity and analyte. Additionally, post-fire transport of dissolved organic carbon was sensitive to both area burned and severity, while nitrate was primarily sensitive to area burned. Despite a muted ($-9.5$ to 5.7 mm yr$^{-1}$ change) hydrologic response in the semi-arid basin, the model predicted large (7%–288% increase) shifts in dissolved organic carbon, suggesting that post-fire shifts in flow pathways and soil properties are key in its response. The limited shifts in nitrate responses in the simulations highlight that terrestrial post-fire transformations, rather than hydrologic changes, may control the increases in stream nitrate often observed post-fire. As wildfire regimes are shifting, improving understanding of post-fire nutrient export responses is critical to protect freshwater resources and aquatic ecosystems.

**Plain Language Summary** Wildfires can significantly alter landscapes, with downstream impacts to streams. We explored how the size and severity of wildfires can impact streamflow and water quality. Additionally, we investigated whether there were thresholds, a value above which wildfire caused observable impacts, in the area burned required to observe a response. To answer these questions we used a process-based model to run 1830 wildfire scenarios in a humid basin and a semi-arid basin to simulate the wildfire response on streamflow, and two common water quality parameters (nitrate, and dissolved organic carbon). Overall, we determined that thresholds were unique to the water quality metric and were influenced by both burn severity and how dry the system's climate is (aridity). Additionally, our work underscored the importance of understanding nutrient cycling in streams after a fire, by highlighting differences between our modeled responses and observational studies. Continued work is needed to accurately represent post-fire processes in models, improving predictions and allowing us to better manage post-fire impacts.

## 1. Introduction

Wildfire regimes are shifting, with an increased incidence of large, severe wildfires and longer wildfire seasons (Brown et al., 2021; Ellis et al., 2022; Jain et al., 2017). This shift is important as wildfires can substantially alter the terrestrial landscape of watersheds and affect the river corridors that drain them. Loss of vegetation and altered soil physicochemical properties after a wildfire can lead to hydrologic changes in a watershed, including altered flow paths (Atwood et al., 2023; Rey et al., 2023; Van der Sant et al., 2018), increases in peak flows and annual water yields (Beyene et al., 2021; Hallema et al., 2017; Saxe et al., 2018), and increased flashiness (Neary et al., 2003). Wildfire can also impact key constituents of stream biogeochemical cycling, such as nitrate and

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dissolved organic carbon (DOC) fluxes (Paul et al., 2022), through changes in their transport (e.g., hydrologic changes), and their reactivity (e.g., changes in speciation).

The current understanding of the influence of wildfires on nitrate and DOC transport in streams is highly variable (Rhoades et al., 2019; Rust et al., 2019; Santos et al., 2019; Wampler et al., 2024). For example, nitrate, a soluble and leachable form of inorganic nitrogen, is often elevated in streams after wildfire, but typically recovers to pre-fire concentrations within three to five years after wildfires (Bladon et al., 2008; Mast et al., 2016; Rhoades et al., 2011). However, stream nitrate can remain elevated for decades, especially in catchments burned at high severity with delayed vegetation recovery (Rhoades et al., 2019). Moreover, stream nitrate may also have high spatial variability along a stream network depending on topographic, vegetation, and wildfire characteristics (Rhea et al., 2022). Similarly, DOC release from soils and export through streams is also highly variable after wildfire (Paul et al., 2022). Wildfires can deposit significant amounts of carbon-rich ash and increase transport to the river through shallow organic layers, leading to increases in DOC observed post-fire (Burton et al., 2016; Emelko et al., 2011; Harris et al., 2015; Hohner et al., 2016). However, it can also consume significant amounts of both the organic soil layer and vegetation, removing significant sources of organic matter in the ecosystem which can lead to decreases in riverine DOC (Betts & Jones, 2009; Rodríguez-Cardona et al., 2020; Santos et al., 2019; Wampler et al., 2024). Both of these constituents play an important role in aquatic ecosystems, including regulating productivity (Bernhardt & Likens, 2002; Dodds & Smith, 2016; Wetzel, 1995).

The non-uniform DOC and nitrate responses suggest that there are possible tipping points or non-linearities for water quality post-fire, with downstream implications for drinking water treatment (Hohner et al., 2019; Smith et al., 2011). These tipping points, or thresholds, have been observed for wildfire disturbance impacts on streamflow. Impacts to streamflow are generally not observable until at least 20% of the watershed has been disturbed (Beyene et al., 2021; Caldwell et al., 2020; Hallema et al., 2018; Williams et al., 2022). While it has been suggested that this threshold can also be applied for water quality responses (Murphy et al., 2023), thresholds for water quality metrics remain inconclusive for many parameters, largely due to data limitations (Raoelison et al., 2023). Quantification of thresholds for water quality responses is particularly important for post-fire management decisions (Murphy et al., 2023). The influence of wildfires on river corridors has increased over the last several decades, cumulatively impacting ∼11% of total western US river lengths since 1984 (Ball et al., 2021). Thus, mechanistic linkages between wildfires and riverine biogeochemistry responses are increasingly important to understanding source water quality vulnerability and post-fire management decisions.

While post-fire hydrologic and water quality responses are often considered to be a result of differences in wildfire characteristics, more recent work suggests that responses can also be controlled by differences in landscape and climate characteristics (Ebel, 2013; Murphy et al., 2015; Rust et al., 2019; Saxe et al., 2018). In particular, aridity (P/PET) has been observed to be a strong control on post-fire infiltration (Sheridan et al., 2015), vegetation recovery (Puig-Gironès et al., 2017), soil organic carbon (Pellegrini et al., 2023) and streamflow (Biederman et al., 2022; Saxe et al., 2018; Wampler et al., 2023; Wine et al., 2018). However, there has been limited work exploring the impact of aridity on post-fire DOC and nitrate responses in streams (Hampton et al., 2022; Rust et al., 2019; Wampler et al., 2024). Current research suggests that DOC increases in more arid streams post-fire, though data on these relationships are limited (Wampler et al., 2024). Stream nitrate responses in separate meta-analyses were found to both not correlate (Hampton et al., 2022) and correlate with aridity (Rust et al., 2019). Given the strong relationship between hydrology and aridity post-fire, more work is needed to explore the relationship between aridity and water quality responses post-fire.

Post-fire ecosystem carbon and nitrogen responses are complex and interconnected (Gustine et al., 2022; Hanan, D'Antonio, et al., 2016; Hudiburg et al., 2023; Santín et al., 2015; Strain et al., 2024), which can make it challenging to disentangle the mechanisms driving post-fire changes in DOC and nitrate in streams. Process based models—which are built on our conceptual understanding of physical processes—are well suited to investigate these mechanisms, as processes can be isolated and tested under controlled conditions that would be impossible in the field (Ebel et al., 2023; Partington et al., 2022). Models also provide an opportunity to perform controlled testing of different wildfire scenarios (burn severity and area burned), providing the data required to determine potential area burned thresholds (Partington et al., 2022). Recent reviews have noted that many different models have been used to model post-fire hydrologic and water quality responses, with different strengths and weaknesses to each model (Basso et al., 2022; Ebel et al., 2023; Partington et al., 2022; Shephard et al., 2025). In particular, the Soil and Water Assessment Tool (SWAT) model is well suited to investigate biogeochemical impacts, due to its relatively robust representation of carbon and nitrogen cycling (Neitsch et al., 2011; Shephard et al., 2025). SWAT has previously been used to simulate both post-fire organic carbon (Loiselle et al., 2020) and nitrate (Basso et al., 2020) concentrations. One of the limitations of these modeling efforts is that Basso et al. (2020) and Loiselle et al. (2020) did not simulate post-fire shifts in biogeochemical cycling. This is likely because these processes are complex, highly variable over time and space, and thus difficult to incorporate into many current process-based models (Basso et al., 2022). However, the exclusion of post-fire biogeochemical changes provides the opportunity to isolate the role of hydrologic cycling and hydrologic shifts, in particular, in controlling post-fire in-stream DOC and nitrate responses. Past work has suggested that both shifts in hydrology and biogeochemical cycling could be leading to the observed post-fire changes in nitrate and DOC, but were unable to disentangle the effects of the two different processes (Richardson et al., 2024; Santos et al., 2019). Thus, simulating only hydrologic shifts post-fire allows us to start to disentangle these mechanisms leading to changes in stream nitrate and DOC dynamics.

We performed a series of process-based model simulations to mechanistically explore the impact of area burned and burn severity on streamflow and the subsequent transport of nitrate and DOC across two basins with contrasting land use and aridity. The models enabled us to isolate the role of wildfire on transport processes and to compare responses across different aridities, removing confounding variables present in field-based inter-regional comparisons. Our objectives were to better understand how burn severity, area burned, and aridity influence post-fire streamflow and riverine nitrate and DOC concentrations and export. We hypothesized that: (H1) Runoff ratios and nitrate and DOC concentrations would increase with increasing percent area burned and burn severity, based on previous observations across watersheds (Rust et al., 2019; Williams et al., 2022), and (H2) thresholds in runoff ratios and water quality responses related to catchment area burned would exist, with thresholds varying with burn severity and watershed catchment aridity.

Results from this mechanistic modeling exercise may help to contextualize variable water quality responses observed empirically post-fire and provide a foundation for predicting the influences of area burned and severities on downstream water quality with shifting wildfire regimes.

## 2. Materials and Methods

### 2.1. Model Setup and Calibration

The Soil Water Assessment Tool (SWAT) model was used to simulate wildfire impacts across two test basins. While it was originally developed for agriculture, SWAT has recently been adapted for wide ranges of watershed modeling applications, including wildfire (Basso et al., 2020; Loiselle et al., 2020; Wampler et al., 2023) and DOC (Du et al., 2020; X. Zhang et al., 2013). QSWAT in QGIS was used to build the models (QGIS: 3.34.9-Prizren; QSWAT: 1.7.2).

In SWAT, the basin is delineated by the stream network into subbasins, then further split into hydrologic response units (HRUs) which are areas with similar slope, soil type, and land use. Inputs into the model include a 30 m DEM (NASA Shuttle Radar Topography Mission (SRTM), 2013) which was used for elevation and slope, soil layer (SSURGO, Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture, n.d.), land use layer (30 m, Dewitz, 2020), and climate data: precipitation and temperature (Daymet, Thornton et al., 2022). These data sets were downloaded using the elevatr, soilDB, FedData, and daymetR packages in R (Beaudette et al., 2025; Bocinsky, 2024; Hollister et al., 2023; Hufkens, 2023; R Core Team, 2024). Solar radiation, relative humidity, and wind speed were generated using the weather generator within SWAT (WGEN_US_COOP_1960_2010). We used the read-in method in SWAT for potential evapotranspiration (PET), using MODIS potential ET data (Running et al., 2021), which was interpolated to a daily resolution using the "StructTS" function in R using a local linear trend model (R Core Team, 2024; Running et al., 2021).

We created two test basins inspired by real watersheds: a humid, forested basin and a semi-arid, mixed land use basin to run our series of in-silico scenarios for hypothesis testing (Figure 1). To ensure the in-silico scenarios in our test basins were accurately representing physical processes, the models were calibrated for streamflow and evapotranspiration (ET) from 2005 to 2017 based on real watersheds. The humid, forested basin used watershed characteristics and hydrologic calibration parameters from the American River basin in Washington, USA (USGS gauge 12488500) and the semi-arid, mixed land use basin used watershed characteristics and hydrologic calibration parameters from the Tule River basin in California, USA (USGS gauge 11204100). The two basins are

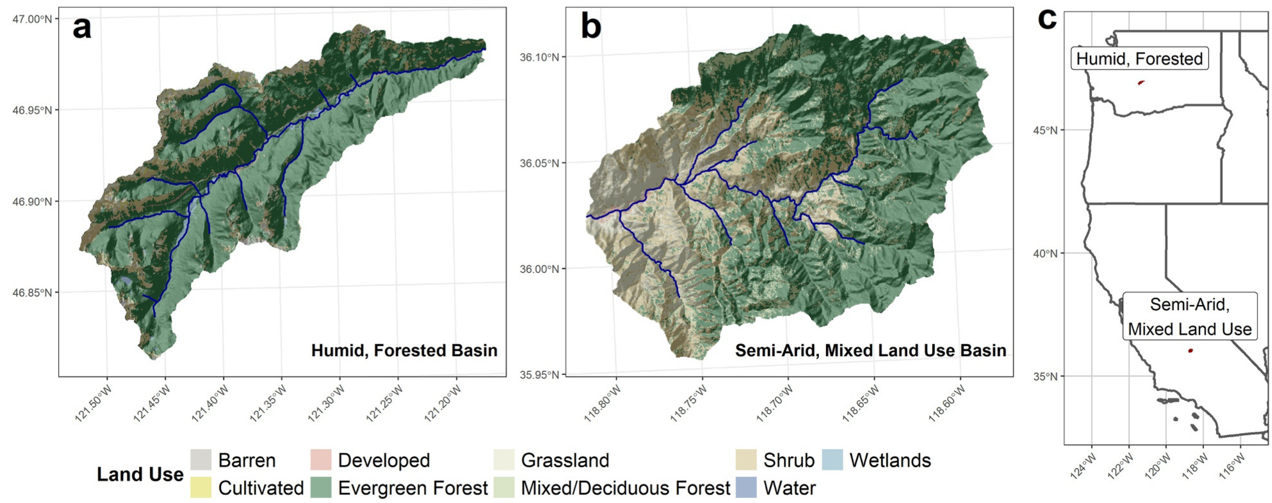



**Figure 1.** Maps showing the land use (Dewitz, 2020) and streams (represented by blue lines, U.S. Geological Survey, 2020) for the test basins: (a) humid, forested basin and (b) semi-arid, mixed land use basin. (c) The location of the basins in the Western US.

similar in size, elevation, and slope but vary in aridity which is a high order control on climate, hydrology, and natural land cover, making them ideal archetypes to test the impact of aridity on post-fire hydrology and water quality (Table 1). We calibrated the models for streamflow at the basin outlets using daily and monthly data from the U.S. Geological Survey streamflow gauges (U.S. Geological Survey, n.d.). We calibrated ET at the HRU level for all major land cover types using monthly aggregated MODIS ET data (Running et al., 2021), choosing a representative HRU for each land cover. While we lacked available water quality to calibrate the nitrate and DOC, observations from each region were used to ensure the magnitudes were within reasonable ranges (Wampler et al., 2025). Additionally, visual checks were performed to ensure model outputs were realistic based on expert opinion (i.e., relationships between nitrate and DOC and streamflow, biomass by land cover, and soil water over time). For more detailed information on calibration and parameters used, see the supplemental material (Text S1 and Tables S1, S2 in Supporting Information S1).

### 2.2. Description of Model Processes

The SWAT model incorporates all major hydrologic processes including precipitation, evapotranspiration, infiltration, runoff, and storage using the water balance equation. The portion of precipitation partitioned into ET is a function of the potential ET, leaf area index (LAI), soil surface cover and available water (Neitsch et al., 2011). The remaining water on the landscape can be routed to streams through surface runoff or infiltrate into the soils and eventually be routed to streams as lateral subsurface flow, or groundwater (shallow or deep). The time it takes for precipitation to travel through the different flow paths ranges from a few days for lateral flow to months to year(s) for groundwater flows where the deeper groundwater is slower to respond to recharge (Arnold et al., 2013, Table S2 in Supporting Information S1).

The SWAT model incorporates the major nitrogen cycling processes (Figure 2), simulating nitrogen fixing, denitrification, nitrification, and mineralization within the soils to transform nitrogen between organic and mineral species. Specifically, nitrogen is added via plant residue, nitrogen fixation by bacteria, and rain and removed via plant uptake, leaching, volatilization, denitrification, and erosion (Neitsch et al., 2011). The amount of nitrate leached and transported to streams is a function of the amount of nitrate in the soil layers and the water flow through those layers (Neitsch et al., 2011). Once in the stream, nitrate concentrations within the model can be impacted by algae growth and nitrification (Neitsch et al., 2011).

Recent work has aimed to improve organic carbon processes within SWAT (Du et al., 2020; Zhang et al., 2013). Within the soil layers, organic matter is cycled between plant litter, microbial biomass, and recalcitrant and labile pools. Movement between the different pools is controlled by water availability, temperature, tillage, and clay and sand content (Zhang et al., 2013). Like nitrate, DOC movement is controlled as a function of the availability in the

**Table 1**
*Descriptions of the Climate, Landscape, Hydrologic, and Water Quality Characteristics of the Modeled Test Basins*

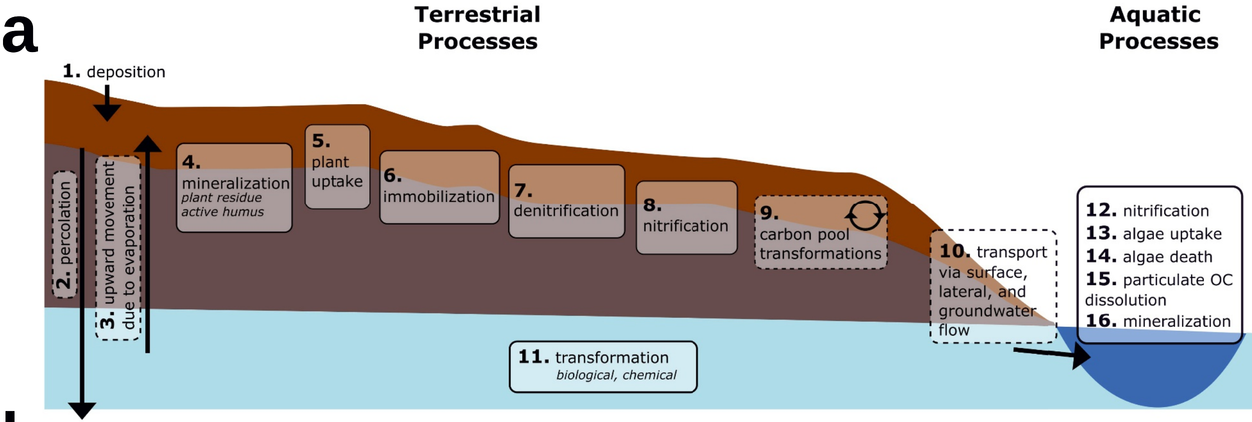
| | Humid, forested basin | Semi-arid, mixed land use basin | Data source |
|---|---|---|---|
| Area ($km^2$) | 205.63 | 249.98 | Calibrated Model |
| Average Elevation (m) | 1477 | 1359 | NASA Shuttle Radar Topography Mission (SRTM) (2013) |
| Average Slope (degrees) | 42.4 | 38.7 | NASA Shuttle Radar Topography Mission (SRTM) (2013) |
| Aridity Index (P/PET) | 1.95 | 0.31 | Running et al. (2021), Thornton et al. (2022) |
| Average Annual Precipitation (mm) | 1891 | 638 | Thornton et al. (2022) |
| Land Use (%)[a] | | | |
| Evergreen Forest | 90.99 | 64.58 | Dewitz (2020) |
| Deciduous Forest | – | 2.47 | Dewitz (2020) |
| Wetland Forest | 0.22 | – | Dewitz (2020) |
| Rangeland Shrubland | 7.49 | 22.05 | Dewitz (2020) |
| Rangeland Grasses | 1.30 | 10.90 | Dewitz (2020) |
| Calibrated Model Outputs | | | |
| Annual Runoff Ratio | 0.72 | 0.38 | Calibrated model outputs |
| Average Annual Flow (mm) | 1365 | 244 | Calibrated model outputs |
| Average Annual ET (mm) | 477 | 356 | Calibrated model outputs |
| Flow Partitioning (%) | | | Calibrated model outputs |
| Surface Flow | 2.24 | 1.50 | Calibrated model outputs |
| Lateral Flow | 68.9 | 40.6 | Calibrated model outputs |
| Shallow Groundwater Flow | 15.3 | 36.6 | Calibrated model outputs |
| Deep Groundwater Flow | 13.5 | 21.2 | Calibrated model outputs |
| Average Streamflow ($m^3/s$)[b] | 8.84 | 1.91 | Calibrated model outputs |
| Average DOC (mg/L)[b] | 0.55 | 0.21 | Calibrated model outputs |
| Average Nitrate (mg/L)[b] | 0.08 | 0.10 | Calibrated model outputs |

[a]Land uses making up less than 10%–12% of a subbasin were dissolved into other land uses in the model (Text S1 in Supporting Information S1). [b]Average determined from calibrated model across calibration period between 2005 and 2018.

soil and the water movement through the soil. Once in the stream, organic carbon can cycle between particulate and dissolved fractions, algae, and inorganic carbon (Du et al., 2020).

### 2.2.1. Wildfire Scenarios

Wildfire was represented in the model using a modified version of the wildfire module (SWAT-F) developed by Wampler et al. 2023. The module adjusts parameters (saturated hydraulic conductivity, soil available water capacity, soil bulk density, soil and plant uptake evaporation compensation factors, curve number, manning's "n" for overland flow, and maximum leaf area index, Table S3 in Supporting Information S1) within SWAT to represent the effects of a low, moderate, or high severity wildfire on the landscape at an HRU scale, where an increase in burn severity is represented by larger changes in vegetation, soil, and runoff properties (see Wampler et al., 2023 and Table S3 in Supporting Information S1 for details). To better represent flow and transport processes in the model, we modified the original fire module by removing the land use changes, and replacing it with a decrease in the maximum LAI for burned HRUs, which impacts plant transpiration and biomass generation within the SWAT model. A drop in LAI is often observed post-fire, reflective of a loss of vegetation post-fire (McMichael et al., 2004). The decrease in LAI was determined for each basin by calculating the actual average decrease in MODIS LAI (Myneni et al., 2015) for actual wildfires in each test basin at each severity level based on methodology outlined in Shi et al. (2024). It should be noted that the module is focused on post-fire hydrologic changes, so it does not include any other changes to the sources or processing of nitrate and DOC a




b

| Process | Controlling Factors |
|---|---|
| 1. Deposition (wet and dry) | - |
| 2. Percolation | water flow, saturated water content, [$NO_3^-$]* |
| 3. Upward movement due to evaporation | evaporation, saturated water content, [$NO_3^-$]* |
| 4. Mineralization (soils) | saturated water content, soil temperature†, [$NO_3^-$]*, [P]* |
| 5. Plant uptake | plant demand†, [$NO_3^-$]* |
| 6. Immobilization | [$NH_4$]*, plant C:N ratio* |
| 7. Denitrification | saturated water content, soil temperature†, soil carbon*, [$NO_3^-$]* |
| 8. Nitrification | saturated water content, [$NH_4$]* |
| 10. Transport via surface, lateral flow, groundwater flow | water flow, [$NO_3^-$]* |
| 11. Transformation | - |
| 12. Nitrification (stream) | travel time†, water temperature*, stream oxygen content*, [$NH_4$]* |
| 13. Algae uptake | algae biomass and growth* |

c

| Process | Controlling Factors |
|---|---|
| 2. Percolation | water flow, saturated water content, soil bulk density, [DOC]*, microbial biomass* |
| 4. Mineralization (soils) | soil texture* |
| 9. Carbon pool transformations | saturated water content, soil temperature†, soil texture* |
| 10a. Transport via surface and lateral flow | water flow, saturated water content, soil bulk density, [DOC]*, microbial biomass* |
| 10b. Transport via groundwater flow | water flow, [DOC]* |
| 14. Algae death | travel time†, water temperature*, algae biomass and growth* |
| 15. Particulate OC dissolution | travel time†, water temperature*, [particulate OC]* |
| 16. Mineralization (stream) | travel time†, water temperature*, [DOC]* |

**Figure 2.** Conceptual diagram (a) illustrating the major SWAT processes controlling movement of nitrate and DOC through the landscape to streams. Factors controlling these processes are noted in tables (b) for nitrate, and (c) for DOC. Factors with no superscript are directly affected by SWAT-F, factors indirectly affected by SWAT-F are denoted with a dagger (†), and factors not affected by SWAT-F are denoted by a star (*).

post-fire. Thus, the fire scenarios are representing how post-fire hydrologic shifts impact the transport of these nutrients.

To investigate the impact of area burned we developed 21 fire scenarios with increasing area burned between 0% and 100% burned in intervals of 5%, where the 0% scenario was considered the baseline model with no wildfire. To investigate the impact of burn severity, each of the area burned scenarios were run at low, moderate, and high severities. To determine what part of the simulated basin would be burned in each scenario, we started with actual wildfires that burned in the American and Tule river basins in 2017. Prior to 2017, neither basin had experienced significant wildfire in the last 20 years. Burn severity maps from the Monitoring Trends in Burn Severity database (MTBS, MTBS Project, 2021) were used to extract the mean burn severity as the difference in normalized burn ratio (dNBR) in each HRU. These dNBR values were used to determine which HRUs burned in each area burn scenario where the more severely burned HRUs were selected to burn in the lower area burned scenarios. Since we selected burned areas at the HRU level, we were not able to reach exact percentages, but all scenarios were within 2.2% of the target percentage burned (Figure S1 in Supporting Information S1). We ran the model for one year (11 August 2017- 10 August 2018) after the start of the hypothetical wildfire in each basin (set as 11 August 2017). On the fire date, all HRUs selected to burn have their parameters modified to reflect the burned conditions described previously. We limited the model runs to one year post-fire, as SWAT-F does not account for post-fire recovery.

Each fire scenario was run 30 times, using different precipitation model inputs for each scenario, allowing us to estimate an average impact across different post-fire precipitations. The precipitation inputs were generated by replacing the precipitation for the post-fire year with precipitation from previous years (1988–2017) one year at a time, matching the same day of year similar to the method used in Wampler et al. (2023). We ran these different precipitation scenarios as past work has shown that post-fire water quality impacts can vary based on post-fire precipitation (Murphy et al., 2015; Wampler et al., 2024). Overall, there were a total of 1830 scenarios per basin, with 3660 scenarios total.

### 2.2.2. Streamflow and Water Quality Outputs

Daily streamflow, nitrate, and DOC concentrations and loads at the basin outlets were extracted from model outputs. For each scenario, we calculated the annual average (concentration) or annual sum (streamflow and loads) for each "fire year," where each fire year was August 11th to August 10th of the following year, to match the year following the wildfire scenario. Raw load outputs from the model were converted to concentrations using the daily modeled streamflow values. Since annual water yields are highly variable, we wanted to normalize streamflow based on the input precipitation to better isolate the impact of wildfire on streamflow. To do this we took the average of the daily precipitation inputs into the model and calculated the annual precipitation which was divided by the annual streamflow across the basin to get the annual runoff ratio for each year.

### 2.3. Threshold Determination

To examine wildfire impacts, we compared the predicted post-fire streamflow and water quality to 30 years of unburned simulation data using the same precipitation inputs as the wildfire scenarios. A fire impact was defined as when the runoff ratios, concentrations, and loads post-fire exceeded the 30 years interannual variability of the unburned scenarios. Threshold values for each metric were determined in two steps.

First, for each analyte and burn severity category, a best-fit line was fit to the simulation data; this allowed us to more accurately resolve the threshold values since we did not run an infinite number of area burned scenarios. We tested seven potential best-fit models with the best model determined by the lowest Akaike information criterion (AIC) value (see Supporting Information S1).

After best fit lines were determined, we utilized methods similar to those used by Williams et al. (2022) to determine when the best fit line crossed the unburned 30-year interannual variability. Briefly, bootstrapping was used to determine the distribution of the unburned sample mean, sampling the unburned scenarios 1000 times with replacement to generate the distribution of the mean unburned value. The distribution was used to determine the 95% quantile of the unburned mean. Threshold values were defined as where the best-fit line crossed the 95% quantile of the unburned mean to the nearest tenth of a percent, which was considered to be significantly different from the unburned scenarios, as these likely represented different populations ($\alpha < 0.05$), and thus a significant fire impact was predicted.

## 3. Results

### 3.1. Simulated Hydrologic Responses of Area Burned and Burn Severity Scenarios

We simulated changes in annual water yields, nitrate, and DOC following wildfire scenarios across area burned simulations using three different burn severities in the two test basins: a humid, forested basin and semi-arid, mixed land use basin. For annual water yields, we noted that both area burned and burn severity interacted to control the post-fire response (Figure 3a). However, the flow response varied substantially between the two test basins. In the humid, forested basin annual water yields averaged across all wildfire percent area burned simulations were predicted to increase by $17.6 \pm 11.2$ mm yr$^{-1}$ for low severity, $39.3 \pm 11.3$ mm yr$^{-1}$ for moderate severity, and $52.1 \pm 11.5$ mm yr$^{-1}$ for high severity. Comparatively, in the semi-arid, mixed land use basin, water yields were predicted to decrease by $9.5 \pm 5.2$ mm yr$^{-1}$ for low severity, $3.5 \pm 5.2$ mm yr$^{-1}$ for moderate severity, and increase by $3.7 \pm 5.3$ mm yr$^{-1}$ for high severity.

The runoff ratio responses also differed between basins (Figure 3b). In the humid, forested basin, runoff ratios were quite consistent across the 30 unburned scenarios, leading to a narrow 95% confidence interval for the mean unburned runoff ratio compared to the semi-arid, mixed land use basin where runoff ratios were much more variable across the different precipitation scenarios, leading to a larger confidence interval. This difference in unburned variability, paired with the differences in annual water yield changes, resulted in very different threshold responses. For the semi-arid, mixed land use basin, all the mean burn scenario responses were within the 95% quantile of the mean runoff ratio (Table 2), suggesting that even at 100% burned at high severity, annual

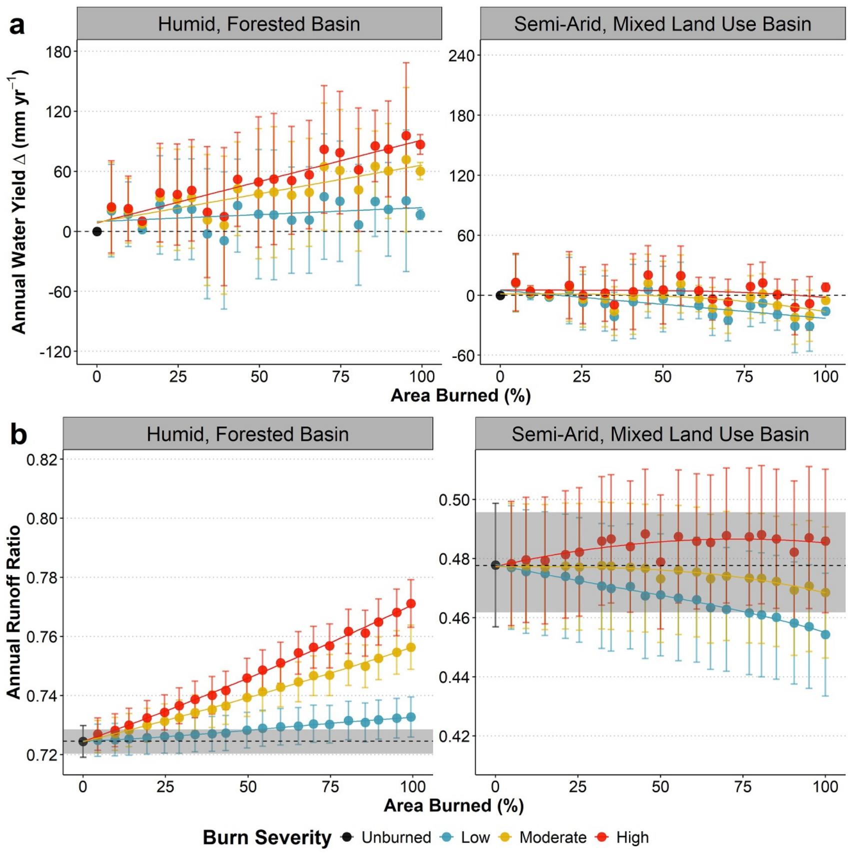



**Figure 3.** (a) Change in annual water yields and (b) annual runoff ratios at the outlet of the test basins in the year following the wildfire simulations. The bars are the 95% confidence interval for each scenario. The baseline unburned simulation is shown in black as 0% burned. The gray box displays the 5% and 95% quantiles of the bootstrapped mean annual runoff ratios in the basin without wildfire with the dashed line representing the median. Best-fit lines are displayed for each severity group (Table S4 in Supporting Information S1).

water yields may not significantly increase outside of normal inter-annual variability and may even decrease. Conversely, for the humid, forested basin, burn scenarios exceeded threshold values at 53.1% for low severity, 14.7% for moderate severity, and 10.1% for high severity (Table 2).

For both basins, wildfire caused increases in surface runoff with an increased 0.2%–37.0% of annual streamflow coming from surface runoff (Figure 4). The increase in surface flow was primarily due to decreases in lateral (defined as saturated subsurface within SWAT) flow; groundwater flows remained relatively stable, with minimal (−6.1% to 0.85%) changes in the fire scenarios. For example, lateral flow decrease the most markedly with

**Table 2**
*Thresholds in Area Burned (%) Required to Exceed Unburned Variability for Runoff Ratios and Nitrate and Dissolved Organic Carbon Concentrations and Loads*

| | | Thresholds in area burned (%) | | | | |
|---|---|---|---|---|---|---|
| Basin | Burn severity | Runoff ratio | Average nitrate concentration | Annual nitrate load | Average DOC concentration | Annual DOC load |
| Humid, Forested Basin | Low | 53.1 | – | 84.8 | 19.3 | 37.5 |
| | Moderate | 14.7 | – | 25.7 | 12.5 | 27.6 |
| | High | 10.1 | 70.3 | 8.8 | 8.4 | 23.5 |
| Semi-Arid, Mixed Land Use | Low | – | 66.2 | 93.5 | 14.0 | 24.7 |
| | Moderate | – | 52.3 | 91.6 | 5.2 | 11.0 |
| | High | – | 43.7 | – | 3.1 | 7.6 |

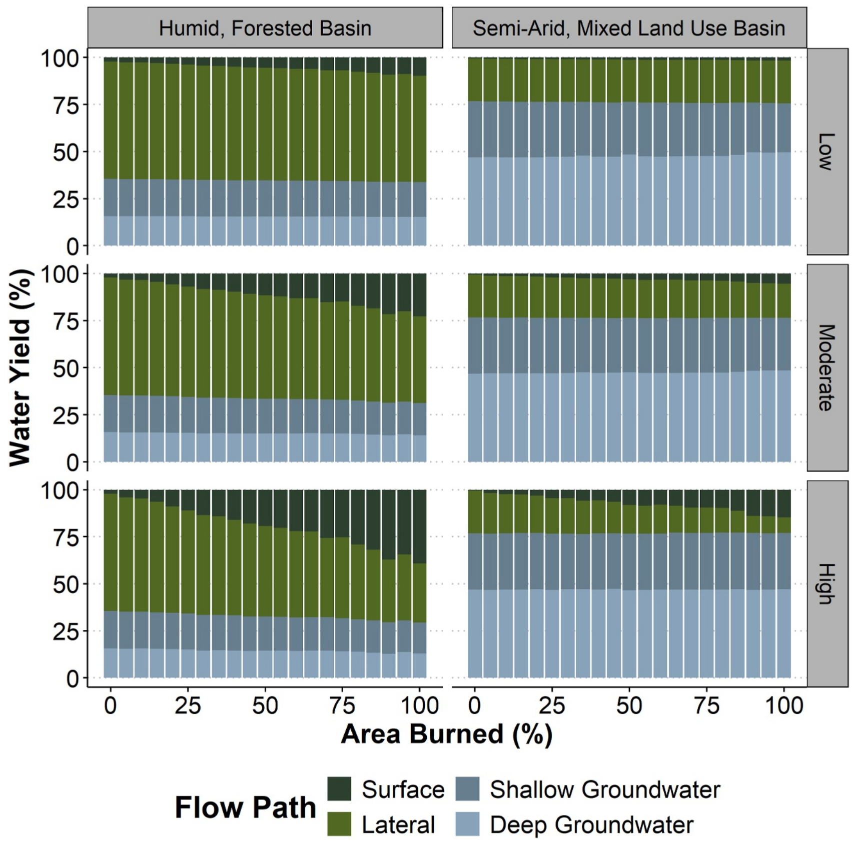



**Figure 4.** Changes in the relative percentage of each flow path for the year following the simulated wildfire across the area burned scenarios at low, moderate and high severity for the two test basins.

increased area burned in the humid forested basin under high severity simulations (Figure 4). Surface flows increased more in the humid basin than the semi-arid basin (Figure 4), with the semi-arid basin water yields dominated by groundwater flow across all scenarios.

### 3.2. Simulated Water Quality Responses to Altered Hydrology Across Area Burned and Burn Severity Scenarios

#### 3.2.1. Nitrate

In the humid, forested basin at low and moderate severity, average nitrate concentrations were predicted to change between −0.004 to 0.0006 mg $L^{-1}$ (−5 to 1%) from unburned scenarios. For high severity scenarios, concentration changes were more variable, predicted to change between −0.001 and 0.009 mg $L^{-1}$ (−1 to 11%). Due to the minimal changes in nitrate in the basin, only the high severity scenarios exceeded the unburned threshold at 70.3% burned (Table 2). Conversely, in the semi-arid, mixed land use basin, we predicted increases in nitrate concentrations with increased area burned for all three severity levels (Figure 5a). Increased nitrate in this basin was relatively insensitive to burn severity, with similar increases in average concentration across all burn severity levels, ranging from −0.00008 to 0.014 mg $L^{-1}$ (−1 to 30%, Figure 5a). Despite similar increases across severity levels, thresholds did vary somewhat by burn severity, decreasing from 66.2% at low severity, 52.3% at moderate severity, and 43.7% at high severity (Table 2).

The relationship between area burned and nitrate loads post-fire was non-linear, with the most noticeable increased rates at higher (>75%) area burned in both test basins (Figure 5a). However, there were very different responses across fire scenarios between the two basins. The humid, forested basin was sensitive to changes in nitrate loads with changes in burn severity (Table 2), with the threshold values decreasing substantially with increased burn severity (low: 84.8%, moderate: 25.7%, high: 8.8%). Comparatively, the semi-arid, mixed land use basin was quite insensitive to changes in nitrate load with increasing burn severity, only exceeding the unburned

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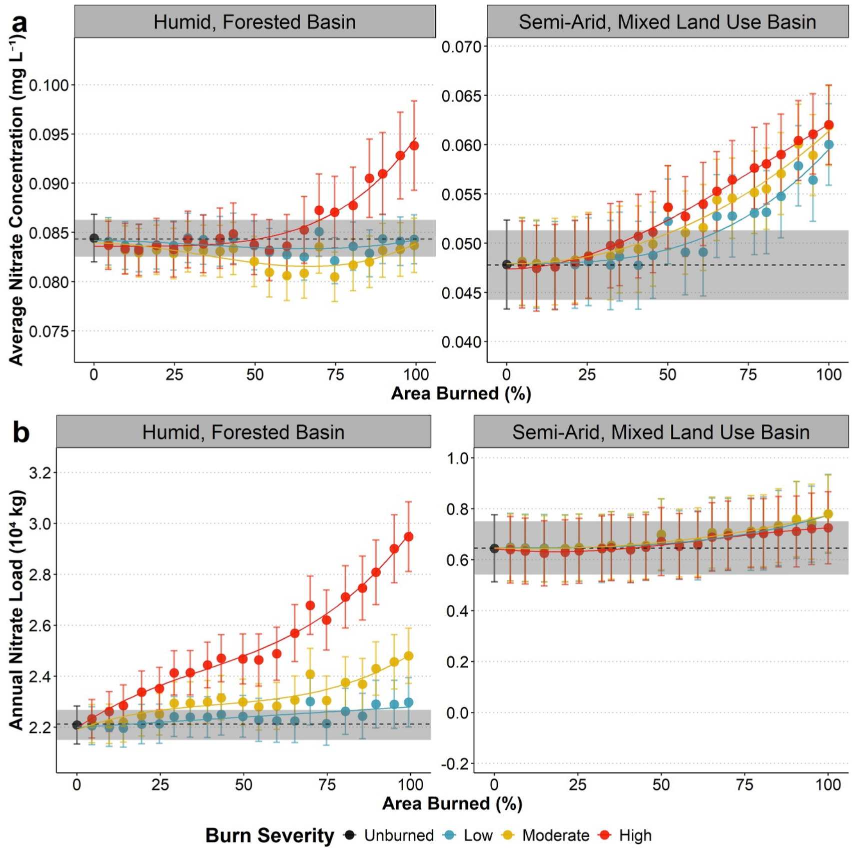


**Figure 5.** (a) Average annual concentration of nitrate given in mg $L^{-1}$ for nitrate concentrations and (b) annual loads of nitrate given in $10^4$ kg $yr^{-1}$ at the outlet of the test basins in the year following the wildfire simulations. The bars are the 95% confidence interval for each scenario. The baseline unburned simulation is shown in black as 0% burned. The gray box displays the 5% and 95% quantiles of the bootstrapped mean annual runoff ratios in the basin without wildfire with the dashed line representing the median. Best-fit lines are displayed for each severity group (Table S4 in Supporting Information S1).

threshold when a large portion of the basin had burned (>90%). Burn severity was also less influential in the semi-arid, mixed land use basin. In fact, the high severity scenarios predicted slightly lower nitrate loads than the low and moderate severity scenarios (Figure 5b).

### 3.2.2. Dissolved Organic Carbon

Compared to nitrate, the response in DOC concentrations were more similar between the two test basins; both exhibited nearly linear responses with increased area burn (Figure 6a). For the humid, forested basin, increases in DOC concentrations were much more sensitive to area burned than burn severity. Concentrations ranged from 0.01 to 0.37 mg $L^{-1}$ (1%–67%) for low severity, 0.01–0.51 mg $L^{-1}$ (2%–95%) for moderate severity, and 0.02–0.59 mg $L^{-1}$ (3%–108%) for high severity scenarios. Greater increases for the semi-arid, mixed land use basin were predicted with increases in DOC concentrations with a greater sensitivity to burn severity. Concentrations ranged from 0.01 to 0.15 mg $L^{-1}$ (7%–75%) for low severity, 0.03–0.38 mg $L^{-1}$ (16%–191%) for moderate severity, and 0.05–0.58 mg $L^{-1}$ (24%–288%) for high severity scenarios. Due to the large increases in DOC concentration observed, thresholds were relatively low. For the humid, forested basin, thresholds were between 8.4%–19.3% and 3.1%–14.0% for the semi-arid mixed land use basin (Table 2).

Average annual DOC loads exhibited similar patterns as DOC concentrations (Figure 6b) The semi-arid, mixed land use basin had lower thresholds of significant changes in DOC loads: 24.7% for low severity, 11.0% for moderate severity, and 7.6% for high severity (Table 2). The thresholds for the humid, forested basin were less variable across severity levels, with thresholds of 37.5% for low severity, 27.6% for moderate severity and 23.5% for high severity (Table 2).

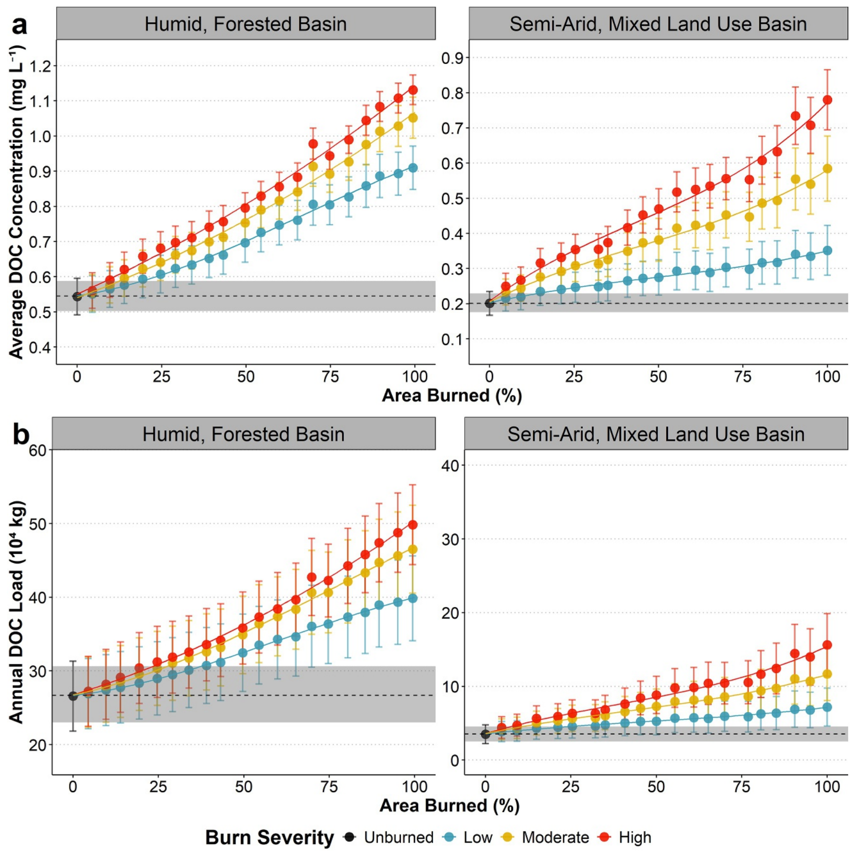


**Figure 6.** (a) Average annual concentration of DOC given in mg $L^{-1}$ for DOC concentrations and (b) annual loads of dissolved organic carbon (DOC) given in $10^4$ kg $yr^{-1}$ at the outlet of the test basins in the year following the wildfire simulations. The bars are the 95% confidence interval for each scenario. The baseline unburned simulation is shown in black as 0% burned. The gray box displays the 5% and 95% quantiles of the bootstrapped mean annual runoff ratios in the basin without wildfire with the dashed line representing the median. Best-fit lines are displayed for each severity group (Table S4 in Supporting Information S1).

## 4. Discussion

### 4.1. Using Model-Observation Comparisons to Infer Post-Fire Mechanisms

#### 4.1.1. Hydrology

The range of increases in annual water yields in the wildfire simulations (0.85–86.9 mm $yr^{-1}$, Figure 3a) were consistent with previous modeling (Wampler et al., 2023) and observational (Blount et al., 2020; Guo et al., 2021; Hallema et al., 2019; Niemeyer et al., 2020) studies which report increases in annual water yields due to wildfire (excluding climate effects) between 1.2 and 478.9 mm $yr^{-1}$, across a range of burn severities, area burned percentages, and climates. Furthermore, observations across 26 humid burned basins suggest that the median increase in flow due to wildfire is 16 mm $yr^{-1}$, which is well within the range we predicted (Hallema et al., 2019).

However, in the semi-arid, mixed land use basin, we predicted decreased annual water yields in low and moderate severity simulations, and small increases in water yields in high severity simulations, with no runoff ratio scenarios being significantly higher than the unburned scenarios (Figure 3). Recent work has highlighted the importance of aridity in the post-fire flow responses, suggesting that more arid basins are more likely to result in decreased runoff ratios post-fire (Biederman et al., 2022; Saxe et al., 2018). Indeed, minimal (<10 mm $yr^{-1}$) or statistically insignificant post-fire shifts in annual streamflow in semi-arid and arid basins have been observed in Colorado, California, and New Mexico, USA (Hampton & Basu, 2022; Havel et al., 2018; Wine & Cadol, 2016). While we would typically expect wildfire to increase water yields due to decreases in evapotranspiration (ET, Kang et al., 2024), this is not always the case. ET may increase post-fire due to increased transpiration from remaining understory vegetation, increased sublimation of snow, or increased soil evaporation (Goeking & Tarboton, 2022). Indeed, in the simulations, increased basin-wide ET was predicted for low severity burns regardless of basin aridity, however moderate severity burns resulted in a small decrease in ET (Figure S2 in Supporting Information S1). Previous SWAT wildfire modeling has also predicted increases in ET, attributing the increase to shifts in land use, increased soil evaporation, and increased sublimation (Loiselle et al., 2020). Given that we did not modify land uses or snow parameters, the increase in ET is likely due to an increase in soil evaporation in our simulations. Decreases in infiltration represented in the model would result in more available water in the surficial soils, while the reduced LAI would reduce soil cover allowing more evaporation.

It is also possible that we predicted minimal ($-9.5$ to 5.7 mm $yr^{-1}$ change) shifts in annual water yields in the semi-arid, mixed land use basin because it is groundwater dominated (Table 1). Past work has suggested that basins with a large proportion of groundwater are likely to exhibit muted stream hydrologic responses to wildfire (Bart & Tague, 2017; Beyene et al., 2021; Buma & Livneh, 2017; Jung et al., 2009; Rey et al., 2023). The muted response is thought to be due to the long residence times of groundwater, buffering post-fire changes and delaying any response observed in the stream (Jung et al., 2009; Rey et al., 2023). Specifically, our modeled basins had calibrated lag times on the order of a year, roughly the length of the post-fire simulation so changes to groundwater may not be observed in this study (Table S2 in Supporting Information S1).

Surface runoff (0.2%–37.0%) increased in the simulations for both basins, primarily due to decreases in lateral flow (Figure 4). This increase in surface runoff is expected post-fire due to increased net precipitation (C. H. S. Williams et al., 2019) and decreased infiltration due to alterations to soil properties post-fire (Moody et al., 2016; Robichaud, 2000). Indeed, past research has observed the relative proportion of streamflow from surface runoff to increase between 0% and 41% in semi-arid basins (Havel et al., 2018; Jung et al., 2009) and 16%–25% in humid basins (Venkatesh et al., 2020). Similar to our findings, it has often been observed that soil water and lateral flow decrease post-fire with an increase in surface runoff (Atchley et al., 2018; Jung et al., 2009; Maina & Siirila-Woodburn, 2020; Venkatesh et al., 2020). However, groundwater flows may also increase, due to decreased vegetation allowing for more groundwater recharge (Jung et al., 2009; Rey et al., 2023). We did not predict an increase in groundwater flows across the simulations (Figure 2), this is likely due to the long lag times and 1-year simulation period. However, decreases in infiltration and reductions in vegetation, which were represented in SWAT-F may allow for greater surface evaporation (Atchley et al., 2018; Ebel, 2013a) which may also contribute to the limited ($-3.7$% to 2.7%) changes in groundwater flow.

Contrary to our findings of large (0.5%–37%) increases in surface runoff in the humid, forested basin (Figure 4), surface runoff is relatively uncommon in Pacific Northwest environments where the test basin is based (Wondzell & King, 2003). It has been suggested that low surface runoff post-fire is partly attributable to rapid post-fire understory growth, which is often observed in humid regions due to ample water and nutrient availability (Buma & Livneh, 2017; Wondzell & King, 2003), which is not simulated in our model. Additionally, while SWAT-F simulated significant decreases in saturated hydraulic conductivity based on work in Colorado, USA (Moody et al., 2016), work in more humid regions suggest that this decrease may not always be observed (Sheridan et al., 2007; Wondzell & King, 2003). However, we lack the data to include variable saturated hydraulic conductivity in the wildfire simulations. Thus, this disconnect suggests that more work is needed to mechanistically understand the processes generating surface runoff across different landscape types in order to more accurately represent post-fire flow partitioning and produce simulations more consistent with field observations across a range of climates.

#### 4.1.2. Nitrate

Minimal changes in stream nitrate were predicted ($-4.7$% to 29.7%, Figure 5a), contrary to field-based observations, where increases in nitrogen are observed in about 77% of observational studies (Paul et al., 2022). While concentrations tended to increase with increased area burn, the magnitude of the increases were much less than expected based on the literature—16% to 1,600% in humid basins (Bladon et al., 2008; Bush et al., 2024; Caldwell et al., 2020; Coble et al., 2023) and ∼15% to 1,000% in semi-arid basins (Hauer & Spencer, 1998; Mast et al., 2016; Rhoades et al., 2011; Santos et al., 2019). Nitrate cycling post-fire involves a complex set of interconnected processes across terrestrial and aquatic ecosystems. Long-term post-fire nitrate responses depend on microbial biomass, soil moisture, temperature, pH, ammonium availability, vegetation uptake from remaining vegetation or regrowing vegetation, and erosion of sediment and ash (Hanan et al., 2017; Hanan, Schimel, et al., 2016; Robichaud et al., 2016; Strain et al., 2024), all of which impact the concentrations and forms of nitrogen available for transport to aquatic ecosystems. Given that our model predicted much smaller changes in nitrate than field observations, this suggests that these missing biogeochemical processes may play a key role in controlling the magnitude of post-fire nitrate responses in the environment.

Specifically, our results that isolate hydrologic effects of wildfire on stream nitrate suggest that the large increases in nitrate observed post-fire are due to shifts in nitrogen cycling and pools, rather than shifts in transport. Nitrate is highly mobile, being easily transported along hydrologic flow paths (Burt et al., 1993). Wildfire can exacerbate this through vegetation loss, causing soils to increase the "leakiness" of nitrate (Hanan, D'Antonio, et al., 2016). While nitrogen sources can be directly altered during the wildfire due to volatilization, many of the other processes affecting nitrate occur over months/years post-fire, like inputs of ammonium during the fire which can be converted to nitrate (Gustine et al., 2022; Smithwick et al., 2005). This often leads to a lagged response in soil nitrate, with peak soil nitrate observed 7–12 months post-fire (Wan et al., 2001). Indeed, many have found that in the initial post-fire period nitrate concentrations are minimally altered with the highest nitrate concentrations often occurring months or years after wildfire (Crandall et al., 2021; Oliver et al., 2012; Rhoades et al., 2011; Sánchez et al., 2023; Santos et al., 2019; Uzun et al., 2020).

#### 4.1.3. Dissolved Organic Carbon

Wildfire increased the ability to transport DOC, as illustrated in the model outputs, but due to post-fire shifts in solubility or soil carbon stocks following real wildfires, decreases or no changes in DOC may be observed in the environment. Compared to nitrate, we predicted much larger increases in DOC concentrations with our simulations (1.4%–288%, Figure 6). Increases in organic carbon are observed in about 50% of observational studies, with no change and decreases observed in 25% of studies (Paul et al., 2022). Thus, while the magnitude of increases are consistent with observational studies in humid basins (−18% to 100%, Bush et al., 2024; Caldwell et al., 2020; Coble et al., 2023; Emelko et al., 2011; Wampler et al., 2024) and semi-arid basins (−98% to 260%, Betts & Jones, 2009; Chow et al., 2019; Harris et al., 2015; Hohner et al., 2016; Mast & Clow, 2008; Murphy et al., 2015; Oliver et al., 2012; Revchuk & Suffet, 2014; Rhoades et al., 2019; Richardson et al., 2024; Santos et al., 2019; Uzun et al., 2020; Wagner et al., 2015), we notably did not predict any decreases in DOC concentrations post-fire due to wildfire-induced changes in hydrologic transport alone.

Decreases in stream DOC post-fire have been linked to increased groundwater flows, which are typically low in DOC (Santos et al., 2019; Wampler et al., 2024), decreases in soil organic matter (Rodríguez-Cardona et al., 2020; Wampler et al., 2024), and decreased solubility of organic matter (Wagner et al., 2018). In the model, the amount of DOC leached to streams is a function of available soil carbon, while available soil carbon is a function of fresh plant residue and soil water, temperature, texture, and tillage (X. Zhang et al., 2013). Plant residue, soil water, and soil temperature are impacted by changes made in the SWAT-F module (Figure 2). However, we did not predict an increase in groundwater flow contributions (Figure 4) and we did not simulate immediate post-fire changes in soil organic matter due to combustion or shifts in the chemical composition of organic matter. While we cannot disentangle which mechanisms could be driving decreases in DOC in observational studies, we can attribute the lack of modeled DOC decreases to the lack of changes in soil organic matter content and composition in our simulations (Table S5 in Supporting Information S1).

The increase we predicted in DOC indicates that the ability to transport DOC increases with wildfire (Figure 6). This has previously been noted by researchers who measured increases in the concentration-flow relationship post-fire (Akie et al., 2024; Richardson et al., 2024). However, if sources and solubility of DOC are decreased post-fire, we may not see the same increases we predicted as the system becomes source limited. Work in California, Colorado, and South Carolina USA noted a similar trend, with noticeable increases in DOC concentrations during the first flushing period post-fire, but with concentrations returning to unburned conditions relatively quickly following fire (Olivares et al., 2019; Uzun et al., 2020; Wagner et al., 2015). Similarly, studies two and three years post-fire noted a decrease in DOC concentrations relative to unburned sites, specifically during flushing periods (Betts & Jones, 2009; Wampler et al., 2024), suggesting that once initial ash inputs have been flushed (Wagner et al., 2015) or stabilized (Bodí et al., 2014), decreased stocks and solubility of DOC (Johnson et al., 2007; Miesel et al., 2015) on the landscape dominate the post-fire response.

Given the limited hydrologic response post-fire (Figure 3), it was surprising that the semi-arid, mixed land use basin exhibited large shifts in DOC concentrations across the burn scenarios (Figure 6). Increases in annual loads

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were smaller in the semi-arid, mixed land use basin than the humid, forested basin, likely reflecting the muted hydrologic response. However, if increased flow was the only thing driving increased DOC in our model, a similar minimal shift in DOC concentrations in the semi-arid, mixed land use basin would be expected, which was not the case. The strong response in the semi-arid basin suggests that shifts in flow pathways (Figure 4) and soil properties can still have a substantial impact on post-fire DOC concentrations, even when no net changes in flow are detected. This response could be due to the shift toward more surface runoff, since there is more organic matter available to transport on the landscape surface (Figure 2).

### 4.2. Post-Fire Hydrologic and Biogeochemical Thresholds

#### 4.2.1. Hydrology

The area burned thresholds we predicted in the humid, forested basin for runoff ratios—53.1% for low severity, 14.7% for moderate severity, and 10.1% for high severity (Figure 3b)—are, on average, consistent with existing literature which has generally observed a ~20% area burned threshold for streamflow changes (Caldwell et al., 2020; Hallema et al., 2018; Hampton & Basu, 2022; Khaledi et al., 2022; A. P. Williams et al., 2022). Large variability exists in thresholds reported, with some finding this threshold to be lower (7%, Buma & Livneh, 2017) or higher (70%, Guo et al., 2021) than 20%. In an analysis of 32 basins across the USA, the effect of low severity burns was too small to be detected, even when large areas of the watershed had burned (Hallema et al., 2018). Thus, the fact that the threshold areas span both lower and higher than the ~20% determined from observational studies could be explained by mixed burn severity present in observational studies but missing from the model. While difficult to parse out in observational studies, due to burn severities occurring in a mosaic across a burned watershed, we suggest that differences in severity is a potential driver of the ranges in threshold responses observed across field studies.

We did not predict any thresholds for runoff ratios in the semi-arid, mixed land use basin. This is consistent with observations from coastal California, USA where runoff ratios and flow changes never exceeded pre-fire variability (Newcomer et al., 2023). The lack of response was attributed to lower severity, heterogeneous burns, and dry conditions post-fire. However, the simulations herein found no thresholds across a large range of burn severities and post-fire precipitation. We believe the lack of thresholds in the semi-arid, mixed land use basin could be due to large variability in runoff ratios across the 30 years of precipitation scenarios (Figure 3b). Newcomer et al. (2023) also noted that climate and post-fire storms exert a stronger control on runoff ratios than wildfire, suggesting that in drier basins post-fire changes may be masked by post-fire precipitation trends.

#### 4.2.2. Nitrate

Despite only modeling transport changes, the predicted thresholds for nitrate concentrations at moderate to high severity (52.3%–43.7%) were remarkably consistent with observed thresholds of 45% for basins burned at moderate to high severity in a semi-arid region (Rhoades et al., 2011). While not specifically investigating thresholds, additional work in semi-arid and sub-humid regions noted that a wildfire response was not or minimally observed in basins burned less than 30%–37% (Mast & Clow, 2008; Richardson et al., 2024), lower thresholds than those predicted in this study. Overall, much of the literature on post-fire nitrate impacts focuses either on a small number of sites (Bladon et al., 2008; Mast & Clow, 2008; Uzun et al., 2020), or measures nitrate at a single site over time (Betts & Jones, 2009; Mast et al., 2016), making it challenging to determine thresholds in observed nitrate responses. This work, while focused on nitrate impacts from a transport lens, provides a starting point in designing studies to determine thresholds in the field.

#### 4.2.3. Dissolved Organic Carbon

Thresholds for DOC concentrations were closely linked to burn severity, with lower area burned thresholds (3.1%–8.4%) for high severity burns relative to moderate (5.2%–12.5%) and low severity burns (14%–19.3%, Table 2) across both test basins. While relatively linear thresholds were predicted in this study, field observations suggest that observed DOC may be complicated by organic matter availability and composition. The low thresholds we predicted were consistent with observations from Colorada, USA, where a wildfire response was observed in a basin with only 8% of its area burned (Chow et al., 2019). However, they also observed that basins with a high burn extent (>75% area burned with 50%–60% at high severity) had statistically similar concentrations to unburned basins. The nonlinearity was attributed to a lack of organic matter sources and vegetation recovery in the highly burned basins. Post-fire organic matter transformations may also be a contributing factor to the non-linearity. Recent work using burned leachates determined that for many vegetation types the highest concentrations of DOC is from low and moderate burns (Roebuck et al., 2025). Thus, observational thresholds for DOC are likely difficult to derive, as DOC dynamics are a net outcome of a myriad of processes beyond hydrologic transport pathways.

### 4.3. Controls on Post-Fire Biogeochemical Responses

#### 4.3.1. Hydrology

Our model predictions highlight the importance of considering the hydrologic context of post-fire nutrient dynamics. While most post-fire studies tend to only report concentration, as it is easily measurable, biogeochemical loads may provide a different picture (Raymond & Saiers, 2010). For example, in our humid basin, we predicted no change in nitrate concentration at a 100% moderate severity burn (Figure 5), however, due to increased streamflows post-fire (Figure 3), nitrate loads were greater by 2,714 kg a year, a 11% increase in annual export (Figure 5). In forested basins in the Western US, which are often nitrogen limited (Cen et al., 2025; Elser et al., 2007), this increase could be enough to cause harmful algae blooms and impact ecosystem nitrogen cycling (Paerl & Scott, 2010), even with no change in stream nitrate concentrations observed.

#### 4.3.2. Area Burned

The simulations showed relationships between area burned and nitrate concentrations (semi-arid, mixed land use basin, Figure 5) and nitrate loads (humid, forested basin, Figure 5). This is consistent with past work that has observed a relationship between nitrate and area burned (Caldwell et al., 2020; Rhoades et al., 2011; Rust et al., 2019). Interestingly, two studies which found area burned was less important than basin area, elevation, climate classification, and pre-fire concentrations (Cavaiani et al., 2024; Hampton et al., 2022) were larger scale meta analyses, suggesting that climate and landscape characteristics can result in variable post-fire responses, as we predicted across the two basins (Figure 5).

Contrary to our findings (Figure 6), past observational studies have not observed a relationship between DOC and area burned, with elevation, basin area, and discharge exerting stronger controls on post-fire DOC dynamics (Barton et al., 2023; Cavaiani et al., 2024). The models focused on test basins with similar basin areas and elevations (Table 1), allowing us to explore the impact of area burned without these drivers. Given that we did predict differences across burn severities, this suggests that while elevation, basin area, and discharge may be more important controls on post-fire DOC concentrations across basins, area burned can still influence post-fire DOC transport.

#### 4.3.3. Burn Severity

Burn severity had minimal impacts on post-fire nitrate concentrations, particularly in semi-arid basins (Figure 5). Past work provides mixed messages about the relationship between nitrate and burn severity. While no relationship between burn severity and nitrate concentrations has been found in semi-arid regions (Sánchez et al., 2023; Santos et al., 2019), relationships between nitrate and burn severity have been observed in humid regions (Caldwell et al., 2020; Coble et al., 2023). Past work suggests that post-fire vegetation is a strong control on nitrate concentrations (Rhea et al., 2022), thus the differential response to vegetation recovery in humid basins versus semi-arid basins (Puig-Gironès et al., 2017) could be leading to the predicted difference. Since we did not model vegetation recovery processes, it makes sense neither basin exhibited a strong relationship with burn severity.

DOC concentrations were closely linked to burn severity across both test basins (Figure 6). This relationship has been observed in summer baseflow (Santos et al., 2019) and fall flushes (Roebuck et al., 2022; Wampler et al., 2024) in Mediterranean climates. However, these studies found that higher burn severity was related to decreased DOC concentrations, not the increase we predicted. The observational studies linked the decreases observed to increased groundwater contributions of low DOC water or decreases in landscape organic matter. Since we did not predict increases in groundwater contributions (Figure 4) and did not simulate any fire-induced changes to organic matter pools, this is likely why we did not predict the decreasing relationship found in field studies.

#### 4.3.4. Aridity

Post-fire water quality responses varied between the two test basins. The results suggest that aridity could be a driver of differential post-fire nitrate responses, through changes in nitrate transport (Figure 5). Past work in the Western US has noted that nitrate increases post-fire were larger in more humid basins (Rust et al., 2019). While this corroborates the idea that aridity is an important control, we found the opposite trend, that concentrations increased more due to wildfire in the semi-arid, mixed land used basin (Figure 5). Rust et al. (2019) suggested the increase in humid regions was due to higher precipitation allowing more transport of nitrate to streams. We may have predicted the opposite trend because we were unable to account for changes in nitrate stocks and sources post-fire. The humid, forested basin showed increased ability to transport nitrate through an increased nitrate load (Figure 5b), but perhaps concentrations were not increased due limited nitrate availability—past work has suggested that undisturbed humid forests have tight nutrient cycling with minimal losses due to efficient incorporation in the soil-microbe-plant system (Perakis & Hedin, 2001).

Compared to nitrate, DOC responses were relatively similar between the two basins (Figure 6), suggesting that post-fire transport of DOC may be similar regardless of aridity. However, aridity has been found to be an important driver of DOC concentrations within a burned basin, with higher DOC concentrations in more arid portions of the watershed during drying and dry seasons post-fire (Wampler et al., 2024). This pattern of increased DOC concentrations in more arid basins has also been observed across unburned basins (Kerins & Li, 2023), suggesting that the relationship observed by Wampler et al. (2024) was not strictly due to wildfire, but rather natural climate and landscape processes. Past work has also observed that changes in soil organic carbon post-fire is a function of aridity, with larger losses of organic matter in more arid basins (Pellegrini et al., 2023). However, since we did not model wildfire-caused changes in soil carbon, this is likely why aridity was not particularly important for DOC in our simulations.

## 5. Conclusions

In field studies, it can be difficult to disentangle the impacts of climate and landscape from wildfire, for example, past work has found that groundwater inputs can mask post-fire DOC responses (Wampler et al., 2024). Our study allowed us to isolate the impacts of area burned and burn severity from other confounding factors, allowing us to determine that both area burned and burn severity do both separately impact the ability to transport DOC and nitrate post-fire. These responses in reality are, however, likely muted by the complex interplay of area burn, burn severity, climate, and landscape characteristics driving stream nutrient dynamics post-fire. Alterationed in biogeochemical processing of these non-conservative parameters during transport from the landscape to streams may further alter their behaviors post-fire (Table S5 in Supporting Information S1). Additionally, post-fire recovery further complicates our understanding of post-fire biogeochemistry, as processes impacting nutrient cycling recover on different timescales (Dove et al., 2020; Muñoz-Rojas et al., 2016; Nave et al., 2011; Prendergast-Miller et al., 2017).

While much research is needed to better understand the myriad of processes affecting post-fire stream biogeochemistry—and how such processes interact with each other—our results suggest a few key processes to start with as a focus of future work. First, while post-fire hydrology is relatively well characterized, our findings on flow path changes suggest that more work is needed to better characterize post-fire soil infiltration properties across a range of landscapes (Ebel & Martin, 2017; Ebel & Moody, 2020; Ebel et al., 2023). Secondly, future work should work to incorporate shifts in the stocks and pools of nutrients post-fire (Table S5 in Supporting Information S1). This seemed to be a key driver of the differences between our modeled results and observational studies. While some work has been done in this area (i.e., Lopez et al., 2024; J. Zhang et al., 2023), we need to reach a more mechanistic understanding to allow us to represent these processes in process-based models. Lastly, we need to work to better understand the recovery trajectories of different post-fire processes (Table S5 in Supporting Information S1). Our findings are limited to the immediate post-fire period as we do not currently have process-based understanding of how key hydrologic and landscape processes recover through time post-fire. Overall, our study highlights the importance of area burned and burn severity on post-fire stream biogeochemical responses moderated by hydrologic shifts and underscores the importance of continued work to mechanistically understand post-fire reactive transport processes. Addressing these limitations will enable us to build more accurate models, improving predictions of post-fire impacts and the ability to respond and mitigate wildfire impacts.

10.1029/2025WR040678



## Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

## Data Availability Statement

All data and code are publicly available on the Environmental System Science Data Infrastructure for a Virtual Ecosystem (ESS-DIVE) repository at https://doi.org/10.15485/2529445 (Wampler et al., 2025).

The following programs were used for creating, running, and calibrating the SWAT model:

- SWAT Model (https://swat.tamu.edu/)
- 2W2E GmbH (2019)

The following publicly available data sets were used in model calibration and model simulations:

- Dewitz (2020)
- Monitoring Trends in Burn Severity (https://mtbs.gov/direct-download)
- Myneni et al. (2015)
- NASA Shuttle Radar Topography Mission (SRTM) (2013)
- Running et al. (2021)
- Soil Survey Geographic (SSURGO) Database (https://sdmdataaccess.sc.egov.usda.gov)
- Thornton et al. (2022)
- U.S. Geological Survey Water Data for the Nation (https://waterdata.usgs.gov/nwis/rt).

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