hydrology

MDPI

Review

A Review of the Key Impacts of Deforestation and Wildfires on Water Resources with Regard to the Production of Drinking Water

Olivier Banton [1,2,*], Sylvie St-Pierre [1], Guillaume Banton [1], Nicolas Laures [3] and Anne Triganon [3]

[1] Hydriad Eau et Environnement, 443 Rte St-Genies, 30730 Saint-Bauzély, France
[2] Avignon Université, UMR 1114 EMMAH (AU/INRAE), 301 Rue Baruch de Spinoza, BP 21239, CEDEX 09, 84911 Avignon, France
[3] Agrivair Garrigue, Les Bouillens, RD139, 30130 Vergèze, France
* Correspondence: banton@hydriad.com

Abstract

Deforestation and wildfires drastically impact vegetation cover, consequently affecting water dynamics. These hazards alter the different components of the water cycle, including evapotranspiration, runoff, infiltration, and groundwater recharge. Overall, runoff increases while infiltration and groundwater recharge decrease. Furthermore, these hazards significantly alter the chemistry of both surface water and groundwater. The main changes to water quality relate to turbidity, bacterial load, mineralization and nutrients. Forest fires can also release contaminants such as heavy metals, polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs). Other contaminants can be introduced by products used in firefighting, such as retardants and perfluoroalkyl substances (PFAS). This paper reviews the impact of deforestation and wildfires on water resources, especially with a view to their use as raw water for drinking water production. The paper identifies the magnitude of the changes induced in water quantity and quality. Even if the results are climate- and site-specific, they provide an indication of the possible magnitude of these impacts. Finally, the various changes brought about by these hazards are ranked according to their potential impact on drinking water production.

Keywords: water resources; raw water; drinking water; forest; deforestation; wildfire

check for updates

Academic Editors: Bojan Đurin, Osman Fetoshi and Gabriel-Ionut Plavan

Received: 3 September 2025
Revised: 7 October 2025
Accepted: 11 October 2025
Published: 12 October 2025

Citation: Banton, O.; St-Pierre, S.; Banton, G.; Laures, N.; Triganon, A. A Review of the Key Impacts of Deforestation and Wildfires on Water Resources with Regard to the Production of Drinking Water. *Hydrology* **2025**, *12*, 271. https://doi.org/10.3390/hydrology12100271

1. Introduction

Both surface and groundwater resources have been discussed in various papers for their advantages and disadvantages in producing drinking water, e.g., Refs. [1–5]. Among the advantages of surface water as raw water for drinking water production are that it is easily accessible, pollution can be easily controlled, and treatment systems are well established (coagulation, filtration, and disinfection). However, its quality is highly variable due to the rapid changes in turbidity, organic matter, bacteriology, temperature, and chemistry. Surface water is easier to treat, but faces more variability. Surface water is also more vulnerable to contamination during extreme weather events [3].

Numerous works outline the advantages of using groundwater as a source of raw water, such as lower contamination risks. Using groundwater generally results in better quality and more consistent water, with low turbidity and a lower bacterial load. Groundwater has a better taste and stable temperature, and induces lower treatment costs (typically requiring only minimal disinfection or chemical adjustments). For its advantages, groundwater is often used as an emergency or crisis supply when surface sources fail. Groundwater is more stable but faces issues like over-extraction and saline intrusion [3]. Among the disadvantages, these resources can be limited and sometimes potentially overexploited, particularly in agricultural areas, arid regions and coastal zones. Furthermore, cleaning the resource is extremely difficult, if not impossible, once it is contaminated (e.g., by nitrate, pesticides, PFAS, microbes).

A robust drinking water treatment system is defined as one that provides excellent performance under normal conditions and experiences minimal deviation during periods of upset or challenge [6]. However, an increase in extreme weather events can cause changes to surface water quality parameters, such as turbidity, that are unprecedented and pose operational challenges to drinking water treatment plants [7].

Mirus et al. [8] reviewed hydrological disturbances with indirect or less pronounced human involvement, such as bark beetle infestation, wildfire and other natural hazards. Hydrological disturbance is defined as an abrupt event that changes the previously understood hydrological function of a system [9]. Modification of the territory—and, in particular, of its vegetation cover—has significant repercussions on water flow and quality [10,11]. For example, it was found that for every 1% increase in deforestation, access to clean water decreased by 1% [12].

After reviewing 137 cases, Andreassian [13] concluded that deforestation increases water yield, while reforestation decreases it. Filoso et al. [14] reviewed 308 study cases and summarized that reforestation decreases the total annual water yield in 80% of cases.

Deforestation increases soil erosion, resulting in higher levels of soil, sediment, and turbidity in the water, which increases the need for drinking water treatment. Numerous studies have shown a link between forest density and water quality [15–17]. Catchments that were predominantly forested (with over 70% forest cover) exhibited the lowest levels of concentration for most of the monitored quality parameters [15]. However, water quality deteriorates in densely populated areas despite high proportions of forest. Landscape configuration may explain 48% of the variation in water quality [16].

An inverse relationship was found between forest cover and turbidity [18], but a weak relationship between forest cover and total organic carbon (TOC). Converting 1% of a watershed from forest to developed land increases turbidity by 3.9%. An increase in TOC of 1% increases chemical costs by 0.46%, whereas an equivalent increase in turbidity increases these costs by only 0.19%.

Forest ecosystems enhance the absorption of nutrients, helping to maintain the quality of water resources [19,20]. An analysis of four land use change scenarios revealed that forest conversion and increased development resulted in higher average concentrations of total suspended solids (TSS) and total nitrogen (TN) at 13 out of 15 intake facilities [21]. Potential increases were found to be as high as 318% for TSS and 220% for TN. A German study of half-forested Water Protection Areas (WPAs) that were subject to deforestation (with 4.8% of the canopy cover lost within three years) found that nitrate concentrations more than doubled with severe forest dieback, while concentrations remained unchanged in undisturbed areas [22].

Forest cover has also been linked to better water quality [17] and reduced treatment costs [18,23,24]. Many studies have examined the relationships between the costs of treating drinking water, water quality, and changes in land use and land cover [18,25–34]. Liu et al. [32] suggest that the higher water yield and increased runoff associated with forest disturbance could impact water quality and consequently increase water treatment costs. In contrast, Mulatu et al. found that increasing forest cover in the catchment area substantially decreases the cost of water treatment chemicals. However, as the buffer distance increases, the contribution of forest cover to reducing treatment costs declines, although it still has a substantial effect on the cost of treating water.

Dearmont et al. [35] studied the effect of turbidity levels on chemical costs for surface water treatment plants in Texas, finding that a 1% reduction in turbidity reduced chemical costs by 0.27%. Meanwhile, Ernst et al. [25] demonstrated that 50–55% of variation in water treatment operating costs could be attributed to the percentage of forest cover in the catchment area. They also found that a 10% increase in forest cover results in a ~20% decrease in treatment costs.

Price et al. [36] showed that turbidity significantly impacts water treatment costs, with a 1% increase in nephelometric turbidity units (NTU) resulting in a 0.1% increase in costs (i.e., a deterioration in water quality). Nehra et al. [37] found that reducing turbidity and TOC by 1% would reduce treatment costs by 0.046–0.091% and 0.951–1.144%, respectively. They also found that a 1% loss of forest could increase treatment costs by 1.7%. Westling et al. [31] found that a 1% increase in forest cover results in a 0.801% decrease in turbidity of the raw water and a 0.228% decrease in chemical treatment costs.

Studies have shown that forests play a significant role in the various components of the water cycle and in water quality. Therefore, deforestation due to logging, decline or wildfires significantly impacts these components and their quality. A decline in the availability and quality of raw water sources poses a challenge to the production of drinking water and increases production costs.

The purpose of this paper is to review the impacts of deforestation and wildfire on the water resources which may subsequently affect the production of drinking water. It considers the impact on the various components of the water cycle, i.e., precipitation, evapotranspiration, runoff, infiltration, and groundwater recharge. It also examines the impact of deforestation and wildfire on water quality, in terms of turbidity, mineralization, bacteriology, heavy metals, volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), and anthropogenic chemical pollutants. This review does not examine the processes of erosion, the fate and role of organic matter in soil, or the process of revegetation following a hazard. It also does not consider factors other than deforestation and forest fires that may influence hydrological processes. It does not address the direct impact of deforestation and wildfires on the cost of water treatment, as this cost varies greatly depending on the country and its drinking water regulations.

2. Impacts of Deforestation

2.1. Impacts of Deforestation on the Water Cycle

The removal of forest cover reduces infiltration and increases surface runoff [38–45]. The removal of litter, resulting from deforestation and soil erosion, can increase peak runoff by around 1.4 to 1.5 and greatly reduce soil moisture [41]. A meta-analysis of the literature by [46] confirms that deforestation significantly affects rainfall fate and its distribution between runoff and infiltration.

Because forests transpire vast amounts of water, deforestation reduces evapotranspiration (ET) by between 2% and 50%, depending on the region. This alters local humidity. Studies using satellites and models confirm that reduced ET leads to less atmospheric moisture and loss of precipitation downwind [38,44–47]. With less evapotranspiration and infiltration into the soil, rainfall results in more runoff.

Van Luijk et al. [48] observed that infiltration rates varied from 26.1 to 28.7 mm/h under shrub vegetation, but from only 0.04 to 0.25 mm/h under degraded vegetation. This corresponds to a reduction factor of 115 to 650. The maximum humidity (under saturated soil conditions) was approximately 51% under shrub vegetation and around 37% under degraded conditions.

Woody plants improve water infiltration by creating macropores and protecting the structure of the soil. Although root-induced macropores promote infiltration, litter (i.e., decomposing organic debris that accumulates on the soil surface) and dense root systems can reduce the amount of water that reaches aquifer layers [49,50]. Massive deforestation can lead to a 50% decrease in recharge to karst aquifers, as well as reducing the recharge season from six months to three [51].

In a strongly seasonal subtropical dry climate (Malawi), it was found that an increase in deforestation of 1.0 percentage points has decreased access to clean drinking water by 0.93 percentage points [12]. In this study, clean drinking water is mainly associated with boreholes and protected wells, while unsafe water is associated with unprotected household wells, streams and rivers. During 2000–2010, the ratio of forest area in Malawi decreased by 14% or by 5.6 percentage points. This implies that the probability of accessing clean drinking water decreases by 5.2 (=5.6 × 0.93) percentage points. On the other hand, the study showed that a 1% decrease in rainfall decreases access to clean drinking water by 0.57 percentage points. Over this decade, deforestation has had a comparable impact (14%) on access to drinking water to the 9% decrease in rainfall caused by climate change.

The recolonization of deforested areas by woody vegetation (trees and shrubs) tends to have a favorable effect on infiltration rates. Indeed, reviving forests can improve infiltration, soil stability and baseflows, particularly in degraded areas, where there can be significant hydrological benefits [52]. However, this recolonization can further reduce groundwater recharge because trees and shrubs draw infiltration water, leading to a reduction in deep percolation and recharge [49–55]. Replacing agricultural surfaces with natural forests led to a 32% decrease in the total volume of water entering the aquifer and a significant drop in the water table level [56].

In semi-arid systems, woody encroachment often reduces groundwater recharge by consuming more water. Context (i.e., soil type and climate) is therefore key [57]. On degraded soils or agricultural land, reforestation typically increases infiltration. However, in certain situations, it can lead to a significant reduction in infiltration [58]. The impact on groundwater recharge also depends on geological formations and soil thickness [57]. In dry regions, recharge may increase via runoff transmission, whereas in more humid zones, evapotranspiration (ET) dominates and reduces flows [53,59].

The influence of woody plant encroachment on groundwater recharge is highly context-specific. It is not possible to establish a clear pattern of infiltration and groundwater recharge, since this influence is also controlled by soil and subsoil characteristics, topography, and climate.

Table 1 summarizes the main quantitative impacts of deforestation on water resources and potential consequences for drinking water production. Deforestation almost always leads to increased runoff. It also generally reduces infiltration and groundwater recharge. However, the extent of this impact can vary depending on the climatic and geological context. In quantitative terms, therefore, the potential impact on drinking water production will mainly concern the risk of a decrease in groundwater availability. This could result in a reduction in the volume of water available for production or a drop in the water table, causing wells or pumping systems to run dry. Since rivers rely on groundwater for their base flow, a decline in groundwater recharge can lead to reduced river and spring flow during periods of low water.

Table 1. Main quantitative impacts of deforestation on water resources and potential consequences for drinking water production.

Impact	References	Magnitude	Consequence for Drinking Water
Deforestation favors the runoff and erosion of surface litter and soil. Macro-porosity closes due to soil destructuration, compaction and macropore clogging by soil particles. This reduces infiltration and groundwater recharge.	[12,38–51]	Infiltration rates can be reduced by several orders of magnitude. A reduction in recharge of up to 50% has been reported.	Lower infiltration and groundwater recharge, and consequently lower amount of groundwater available as a source of raw water. This impact can last for many years. This can also result in reduced river and spring flow during periods of low water.
The recolonization of soils by woody plants does not immediately lead to the recovery of the soil's infiltration capacity. In fact, it can further reduce groundwater recharge due to the removal of soil water by roots.	[52–59]	A 32% decrease in recharge has been reported over several years following the re-establishment of woody vegetation.	

2.2. Impacts of Deforestation on the Water Quality

Deforestation had significant effects on the quality of surface water [60], including a notable decrease in electrical conductivity and changes in the ionic composition of the water, which suggests a dilution-related impact. Conversely, nitrate concentrations in streams increased significantly during the first two to three years after deforestation. Similar results were observed in other deforested catchments, and were attributed to increased leaching and reduced nutrient (N and P) uptake by biomass [61–65].

Watersheds with less than 35% forest cover experienced increased turbidity, suspended sediment, nutrients (nitrogen and phosphorus) and microbial load, whereas watersheds with 55% or more forest cover had much better water quality [66]. Above all, deforestation has led to a significant increase in turbidity and the bacteriological load in surface waters. However, there has been little or no increase in groundwater, as it is generally better protected from such parameters.

Nayaranan et al. [67] have shown that large-scale deforestations of the Amazon during the 2001–2020 period are associated with significant changes in sediment concentration in the eastern portion of the basin. In the heavily deforested eastern regions, the hydrogeomorphic response to deforestation occurs relatively rapidly (within a year), whereas the less disturbed western areas exhibit delays of 1 to 2 years before responses are observable.

Zhao et al. [68] studied the Upper Chao Phraya River Basin (UCPRB), which is a tropical monsoon basin located in Thailand. In the entire basin, a one percent reduction in the forest cover can increase the annual streamflow by 1.9%. In the upstream sub-basins, a one percent reduction in the forest cover can increase the annual streamflow and annual baseflow by 2.5–5.4% and 2.6–6.7%, respectively. A one percent reduction in the forest cover can increase the annual suspended sediment load by 8.7%.

Klein et al. [69] used continuous turbidity data during winter runoff seasons in 28 coastal watersheds. They showed that watersheds with more recent logging (higher percent harvested annually) have much higher turbidity (e.g., 10—percent exceedance turbidity levels) compared to pristine forested baselines.

Very few studies have addressed the impact of deforestation on groundwater quality. A significant increase in nitrate levels in groundwater during the first two to three years after deforestation [60]. Sheikhy-Narany et al. [70], meanwhile, examined nitrate concentrations measured in wells over a 25-year period (1989–2014), attempting to correlate them with changes in land use brought about by deforestation. According to their findings, nitrate concentrations increased by around 8.1% per year in agricultural wells and by around 3.9% in residential wells. By 2014, the proportion of areas where nitrate levels exceeded 10 mg-N/L (or 45 mg/L of NO_3, i.e., the nitrate standard for drinking water in many countries, e.g., Ref. [71]) had risen from 1% to 48%, indicating that nearly half of the wells were no longer in compliance with the drinking water standard. However, this evolution could be due to the combined effect of deforestation and changes in land use.

Deforestation has been linked to increased turbidity and worsening bacterial contamination in surface waters used as drinking water sources. Heavy rainfall post-deforestation increases the amount of suspended solids entering drinking water systems. This has led to higher turbidity and spikes in bacterial parameters [72].

Table 2 summarizes the main qualitative impacts of deforestation on water resources and potential consequences for drinking water production. Deforestation has a greater impact on surface water than on groundwater. The most significant impact is related to increases in total dissolved solids (turbidity), as well as in microbiological load. Increased turbidity mainly affects surface water and karst aquifers, and will have a significant impact on drinking water production, requiring increased flocculation, filtration and disinfection. It should also be noted that this turbidity may affect the effectiveness of water disinfection. In any case, continuous monitoring will be required so that treatments can be adjusted in real time. Deforestation will also alter the water's pH, which may affect the disinfection, coagulation and flocculation processes. Deforestation also promotes the release of nitrogen and phosphorus, which are produced when soil organic matter mineralizes and is no longer absorbed by vegetation. If the standards for drinking water are exceeded, the water will require appropriate treatment.

Table 2. Main qualitative impacts of deforestation on water resources and potential consequences for drinking water production.

Impact	References	Magnitude	Consequence for Drinking Water
Deforestation promotes the circulation of ions (both anions and cations), which contributes to changes in electrical conductivity and pH of waters.	[60,61,63]	No drastic changes have been reported.	Effect on disinfection efficiency, coagulation and flocculation
Sharp increase in turbidity and bacterial load in surface waters during rainy periods. Could also affect the groundwater in fractured or karstic aquifers.	[64,66–69,72]	Increase in load by several orders of magnitude.	Requires increased monitoring, filtration and disinfection, the latter of which is also losing its effectiveness.
Increased nitrogen and phosphorus concentrations due to a significant reduction in uptake by vegetation and mineralization of soil organic matter, affecting both surface water and groundwater.	[60–66,69,71]	A significant increase in nitrate levels in groundwater was reported, with 50% of wells showing concentrations that exceed the drinking water standard.	May cause drinking water standards to be exceeded, requiring appropriate treatment.

3. Impacts of Wildfires

3.1. Impacts of Wildfires on the Water Cycle

Many studies regard wildfires as a driver of deforestation or a special case thereof, given that fires facilitate forest degradation and often precede conversion to non-forest land use [73]. An analysis of the relationship between forest loss and fire [74], based on satellite-derived data from 2003 to 2018, found that, on average, 38 ± 9% of global forest loss was associated with fire. This proportion remained relatively stable throughout the study period. However, the impact on soil and water resources differs depending on whether deforestation or wildfires are involved.

Wildfires have a significant impact on the hydrological response of watersheds [75–79]. Atchley et al. [80] suggest that the increases in baseflow (i.e., watershed flow transiting groundwater) that are sometimes observed after a fire—see, e.g., Refs. [81,82]—may be related to increased water storage in the watershed. This is probably the result of increased soil moisture and groundwater recharge.

Johnk and Mays [83] found a correlation, which they consider to be causal, between wildfires and a temporary drop in the water table. They observed a significant decrease in recharge during the first year after the fire. Although some recovery occurs in subsequent years, infiltration capacity remains below previous levels [84–89]. Guzmán-Rojo et al. [87] report a reduction in groundwater recharge of around 40% in the first year, with recharge amounting to just 36 mm/year compared to 59 mm/year in unburned areas. The recharge deficit remained at around 10% after two years, suggesting that the soil's infiltration capacity had only partially recovered.

However, other studies have reported an increase in groundwater recharge and levels, which may be due to a decrease in evapotranspiration following vegetation mortality [80,90,91], or to the formation of macropores from burnt vegetation [92,93]. For example, Giambastiani et al. [94] found that the estimated recharge rates increased in the partially and completely burnt areas (219 and 511 mm/year, respectively) compared to the pristine pine forest area (73 mm/year). La Pasta Cordeiro et al. [95] showed that, in the first year after the wildfire, groundwater recharge in the burnt area increased from 20% to 40% of precipitation. Six years later, the groundwater level in the burnt area was still 10% higher than the historical average.

Atwood et al. [96] indicated that both surface runoff and infiltration likely increased in tandem. Their results suggest that the hydrological response to storms in post-fire environments is dynamic, involving more surface-subsurface exchange than was previously thought. This has important implications for vegetation regrowth and post-fire landslide hazards in the years following a wildfire.

A review of groundwater response reveals a range of outcomes, varying from substantial increases to notable decreases in recharge and baseflow, with some studies indicating negligible or short-lived effects [97]. This review concludes that in hydroclimatic settings where water input and evaporative demand cycles are out of sync, post-wildfire groundwater responses tend to be positive (i.e., increased flux or storage), whereas under low fire severity conditions or in vegetation types that quickly recover, groundwater responses tend to be negative (i.e., decreased flux or storage).

Fire burns vegetation and organic soil layers, often creating water-repellent or ash-sealed soils [98–104]. This reduces soil permeability by several orders of magnitude. Consequently, soil clogging with ash reduces permeability and infiltration capacity, thereby increasing surface runoff and limiting groundwater recharge [84–87,105,106]. Infiltration rates can decrease by over 90% [107]. Moody and Martin [108] have also shown a reduction in infiltration rates of between two and seven times post-fire. However, Bart and Tague [109] suggest that reduced evapotranspiration may influence groundwater recharge to a greater extent than increased soil hydrophobicity.

A wildfire typically causes an increase in runoff and erosion, directing precipitation towards watercourses rather than into the ground. However, the severity of the fire can influence recharge rates, either positively or negatively [80]. Therefore, groundwater recharge can increase or decrease depending on the severity, duration and physiographic setting of the fire. Úbeda and Sala [110] analyzed maximum runoff coefficients measured under different wildfire severities in northeastern Spain. During their study, they observed an increase in the coefficients from 6.3% to 10.0% for unburned forests, around 8.9% for forests subjected to low fire severity and from 33.6% to 37.4% for forests subjected to moderate or high fire severity. Therefore, the change in runoff coefficients was much greater for severe fire situations.

Moderate burns can open up the canopy and stimulate infiltration, whereas intense burns often have the opposite long-term effect [84,101]. Moody and Martin [108] investigated the effect of fire severity on hydraulic conductivity (i.e., soil permeability) and sorptivity (i.e., the soil's capacity to absorb and transmit water by capillary action), which are the two main parameters that control runoff, infiltration, and groundwater recharge. They found that the relationship between hydraulic conductivity and fire severity followed an exponentially decreasing function, while sorptivity followed a linearly decreasing function. Therefore, a severe fire reduces soil permeability more than sorptivity, whereas medium to less severe fires primarily influence the evolution of soil hydrological behavior through changes in sorptivity.

The time required for a watershed's hydrological functioning to return to its initial (pre-fire) state depends heavily on the rate at which vegetation cover is restored. González-Pelayo et al. [79] studied the relationship between fire frequency, soil water properties, and soil water dynamics. They found that water stress (a lack of soil humidity for plants) occurred 17 days earlier in areas affected by four fires and 10 days earlier on bare sites. Therefore, frequently burnt areas experience water stress more quickly than bare areas. Areas affected by a single fire exhibited superior vegetation resettlement compared to areas affected by four fires.

The increase in runoff generally persists for 2–6 years after the fire, with a peak in the first 1–2 years [111–114]. Mediterranean watersheds typically recover their initial hydrological functioning within 3 to 7 years, although the extent of the burnt area can extend this period [76,115]. Therefore, the effects of fires can be observed for one to two years after the fire, but may also be evident five to seven years later [107,116–118].

Wagenbrenner et al. [76] conducted a review of 28 studies (38 Mediterranean sites monitored for between three and 20 years) investigating how these sites recover their hydrological functioning following wildfires. Eighteen of the sites showed recovery within seven years, with delays ranging from zero years (i.e., no post-fire response) to seven years. However, no clear correlation was found between the time taken for hydrological recovery and either site or fire characteristics.

Table 3 summarizes the main quantitative impacts of wildfires on water resources and potential consequences for drinking water production. Wildfires leads to increased runoff. Depending on the climatic and geological context, infiltration and groundwater recharge may increase or decrease. In quantitative terms, therefore, the potential impact on drinking water production will mainly concern the risk of a decrease in groundwater availability. Since rivers rely on groundwater for their base flow, a decline in groundwater recharge can lead to reduced river and spring flow during periods of low water. This could result in a reduction in the volume of water available for production or a drop in the water table, causing wells or pumping systems to run dry.

Table 3. Main quantitative impacts of wildfires on water resources and potential consequences for drinking water production.

Impact	References	Magnitude	Consequence for Drinking Water
Wildfires significantly impact the distribution of rainfall between runoff and infiltration. Runoff is generally exacerbated. This impact can last for several years. It is most significant during the first two years, but it can last up to seven years. Depending on the climatic context, wildfires can lead to an increase or decrease in the rate of groundwater recharge and the levels of the water table.	[73–118]	Runoff may increase from 6.3% to 10.0% for unburned forests, around 8.9% for forests subjected to low fire severity and from 33.6% to 37.4% for forests subjected to moderate or high fire severity. Increases or decreases in groundwater are highly context-specific.	If there is a decline in groundwater recharge, there will be less groundwater available as a source of raw water. The resulting drop in the water table can cause wells to dry up or pumps to stop. This impact can last for many years. This can also result in reduced river and spring flow during periods of low water.
The severity of the fire determines the extent to which burning the soil and clogging its pores with ash reduces its permeability. This has a significant impact on infiltration and groundwater recharge. More frequent fires exacerbate impacts due to the difficulty of vegetation in reinstating itself and helping soils to recover their hydrological functionality.	[80,84,108,110,111]	Severe fire can exacerbate the changes and frequent fires can quadruple the impacts.	

3.2. Impacts of Wildfires on the Water Quality

Wildfires can significantly degrade the quality of river water in both the short and long term by increasing sediment loads and nutrient concentrations (nitrogen and phosphorus), as well as introducing toxic elements and modifying pH and temperature. A shift in groundwater hydrochemistry can also be observed due to ash and burnt organic matter, which modify the chemistry of infiltrated water by lowering pH and increasing ions, sometimes by a factor of 7 [86,89,119].

In the Mediterranean region, fires have been shown to greatly increase sediment flux and water turbidity, with a significant impact on water quality [120]. Lane et al. [117] report increases in suspended sediments of up to 200% after wildfires. Malmon et al. [121] report suspended sediment levels that were 10 to 100 times higher than previous, particularly in the first year. Smith et al. [122] report first-year exports ranging from 0.017 to 50 t per year per hectare over a wide range of catchment sizes (0.021–1655 km^2), representing an increase of between 1 and 1459 times the exports observed under unburned conditions. The maximum concentrations reported in the first year ranged from 11 to nearly 500,000 mg/L.

Following forest loss due to wildfires and salvage logging, streams that feed water treatment systems showed significantly higher turbidity levels (up to ~15 NTU versus ~5 NTU in unburned watersheds) [123]. This has an impact on the production of drinking water, due to higher levels of turbidity, organics and metals, which increase the cost and complexity of water treatment [122–124].

Wildfires generate high concentrations of nutrients (nitrogen—N and phosphorus—P), organic matter, and pollutants, such as heavy metals, in runoff water. Hampton et al. [125] conducted a meta-analysis of 121 sites, reporting median increases of 40–60% in nitrogen and phosphorus in rivers, as well as significant increases in dissolved organic carbon and suspended sediments. Wildfires also mobilize various metals, including Al, Fe, Mn, As, Pb, Cu and Ni, as well as PAHs.

In particular, ash releases nitrogen and phosphorus in soluble inorganic forms, promoting the eutrophication of waterways [122,126–130]. Smith et al. [122] report a wide range of exports to streams of total nitrogen (1.1 to 27 kg/ha/year) and total phosphorus (0.03 to 3.2 kg/ha/year) in the first year, representing 0.3 to 431 times the exports observed in unburnt forests. Nitrate exports ranged from 0.04 to 13.0 kg-N/ha/year (i.e., three to 250 times those observed in unburned forests).

Following a wildfire, recharge can therefore transfer significant nutrient fluxes to aquifers [119–133]. Ref. [119] reports a significant increase in nitrate concentrations in springs (by a factor of six after six to seven months), as well as a 1.3- to 2.2-fold increase in PAHs. Gunnarsdottir et al. [134] showed the presence of polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs) in groundwater, as well as an increase in heavy metals. PAHs were detected in the groundwater up to five months post-wildfire. The transfer of these constituents can peak in the months that follow [119,132] and continues to do so for at least nine years after the fire [133]. Several metals were also found at concentrations up to six times higher than the median. Solomon et al. [135] also identified VOC contamination in drinking water after the 2018 wildfires in California.

In addition to the direct risks they pose, wildfires can also lead to secondary contamination. This is particularly the case with retardants based on nitrogen, phosphorus and sulfur, and water additives such as aqueous film-forming foams, which are used to extinguish wildfires and can release per- and polyfluoroalkyl substances (PFAS).

Tobin et al. [136] found high nitrogen concentrations in California that subsequently declined over the following three to four years of monitoring. The spatial distribution of these concentrations correlated with that of nitrogen-based retardant products used in firefighting. However, Dimitriadou et al. [131] did not observe any increase in nitrogen or phosphorus attributable to wildfires, whether due to the combustion of organic matter or retardants.

PFAS (from aqueous film-forming foams) are highly mobile, toxic and persistent [137–139]. The fate of PFAS is influenced by organic carbon (OC), pH, cations and ionic strength [140–142], but the sorption of PFAS cannot be explained by OC, pH or clay content alone. Therefore, their fate in soil and groundwater is difficult to predict due to a lack of knowledge and hindsight, as well as uncertainties.

Table 4 summarizes the main qualitative impacts of wildfires on water resources and potential consequences for drinking water production. Wildfires affect both surface water and groundwater. A significant impact is related to increases in total dissolved solids (turbidity), as well as an increased microbiological load. This mainly concerns surface waters and karst aquifers. Increased turbidity will have a significant impact on drinking water production, requiring increased flocculation, filtration and disinfection. It should also be noted that this turbidity may affect the effectiveness of water disinfection. In any case, continuous monitoring will be required so that treatments can be adjusted in real time.

Wildfires will also alter the water's pH, which may affect the disinfection, coagulation and flocculation processes. They also cause an increase in metal concentrations and the release of nitrogen and phosphorus. These nutrients are produced when soil organic matter mineralizes and are no longer absorbed by vegetation. Polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs) can also be released by the combustion of organic matter. If the standards for drinking water are exceeded, the water will require appropriate treatment. Appropriate monitoring is required to identify such exceedances.

Table 4. Main qualitative impacts of wildfires on water resources and potential consequences for drinking water production.

Impact	References	Magnitude	Consequence for Drinking Water
Wildfires can alter the chemistry of water. This often results in a decrease in pH level.	[86,89,119]	Concentrations of ions can increase by up to sevenfold.	Effect on disinfection efficiency, coagulation and flocculation.
Wildfires can carry significant amounts of suspended matter into waterways due to the removal of the vegetation cover and humus layer that would normally protect the soil from erosion.	[120–124]	Suspended sediment rates up to 1000 times higher in streamflows.	Requires increased monitoring, filtration and disinfection, the latter of which is also losing its effectiveness
Wildfires can cause significant losses of nutrients (Nitrogen and Phosphorus).	[122,125–133]	Mean increases of 40–60% in nitrogen and phosphorus concentrations in rivers. Concentrations up to 400 times higher in the first year.	May cause drinking water standards to be exceeded, requiring appropriate treatment.
The combustion of plants releases polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs).	[119,132,135]	Found at much higher concentrations than normal for several years.	Increased monitoring and treatment if necessary.
Fire retardants may release nitrogen, phosphorus and sulfur. Water additives may also provide pollutants such as perfluoroalkyl substances (PFAS).	[136–142]	N, P and S concentrations may approach standards.	Increased monitoring and treatment if necessary.

Wildfires can lead to increased concentrations of nitrogen, phosphorus or sulfate, which are linked to the retardants used to fight them. PFAS (perfluoroalkyl substances) can be released from aqueous film-forming foams.

4. Discussion and Perspectives

Figure 1 summarizes the potential impacts and consequences of deforestation and wildfires on water resources and the production of drinking water. The main impacts and consequences are listed using a color code: blue = favorable; orange = unfavorable; red = very unfavorable. This enables the importance of each situation to be quickly grasped. For example, it shows that severe wildfire has the greatest impact on drinking water production in terms of both quantity and quality, while sustainable forestry (where only mature trees are cut) has the least impact. This table can be used to quickly identify the most significant quantitative or qualitative impacts associated with both types of risk.

Deforestation and wildfires can significantly impact the quantity and quality of water resources, with notable consequences for drinking water production. While most of the studies agree that these hazards modify runoff at the expense of infiltration and groundwater recharge, quantitative data on their impact on the different components of the water cycle is still scarce. The importance and duration of the impact are not fully documented, and are highly site-specific.

Although climate and vegetation (which are closely linked) play a significant role, they are not the only factors that determine the significance of the impacts. Without field measurements, it is impossible to accurately estimate this importance based solely on the most influencing factors. This is particularly true of the duration of the impacts, which varies significantly and appears to be closely linked to the dynamics of vegetation re-establishment.

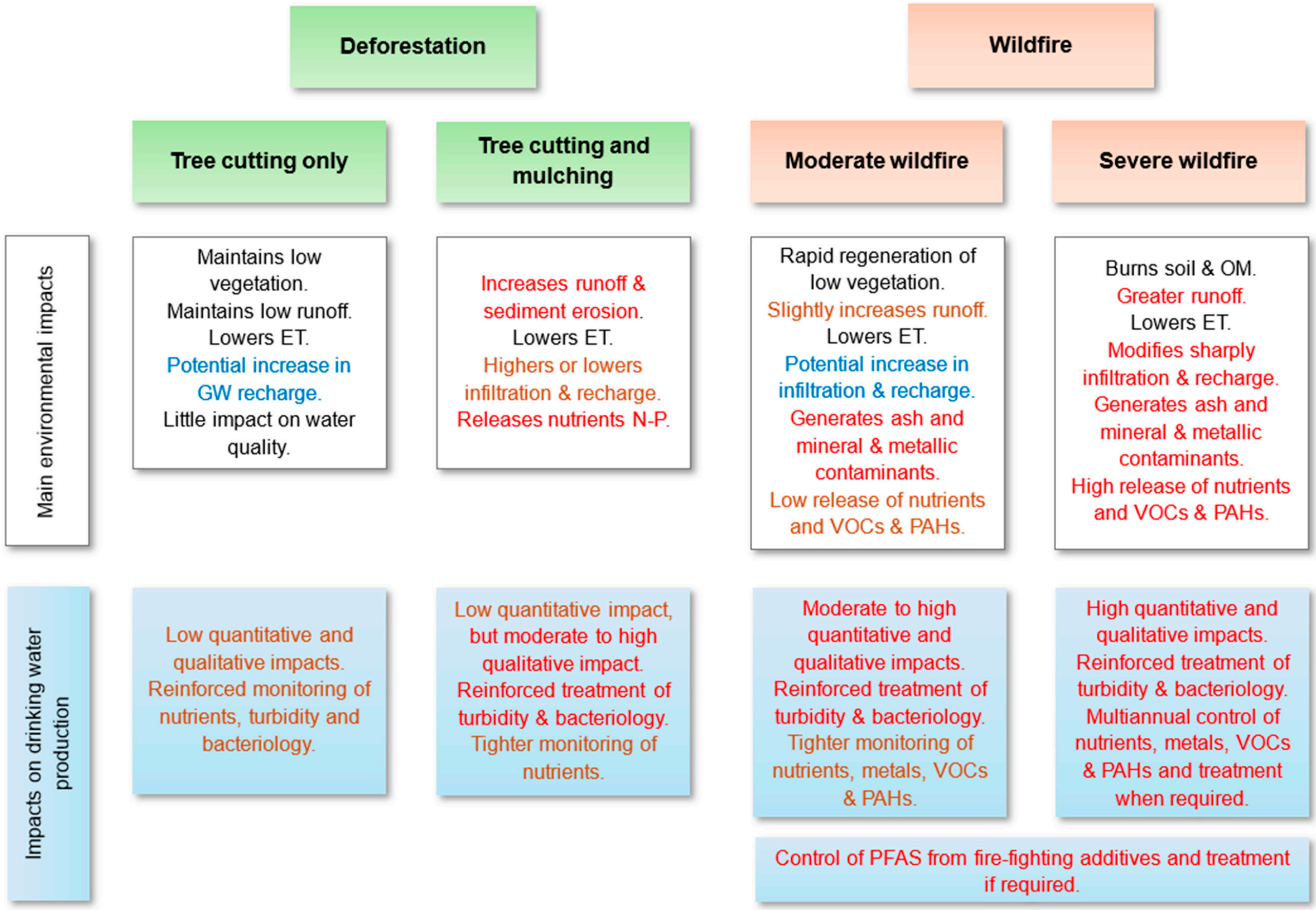

Figure 1. Summary of the impacts and consequences of deforestation and wildfires on water resources and drinking water production.

The lack of knowledge about quantitative impacts appears to stem from the limited number of gauged watersheds or the excessive size of those that are gauged. For this reason, it is not always possible to clearly identify the impact resulting from a change affecting only a small part of a watershed. This makes it difficult to carry out statistical analyses of a temporal or multivariate nature, which are needed to identify and quantify changes, and to determine the contribution of each factor to these changes.

In terms of drinking water production, the most damaging impact on the quantity of resources appears to be the changes in groundwater recharge, which consequently leads to low river flow rates. To quantify this impact, it is necessary to monitor groundwater levels and the discharge of springs and river base flow supported by groundwater. The ease with which the impacts can be identified depends on the size of the aquifer. The smaller the aquifer, the more noticeable the changes will be.

Deforestation and wildfires also impact water quality, leading to increased turbidity, bacterial load, metal concentrations, and nutrient levels. Wildfires can produce contaminants such as polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs), as well as contaminants related to firefighting products, such as perfluoroalkyl substances (PFAS). Therefore, quantitative monitoring must always be accompanied by qualitative monitoring in order to detect any undesirable substances or contaminants. The same recommendations apply to monitoring neighboring areas, as well as to the specific issue of unpredictable wildfires.

Water cycle sensitivity to forest change is then essential for understanding the magnitude of possible hydrological impacts caused by forest disturbance (e.g., de-forestation and wildfire) or forestation [143]. Many studies have examined the impact of deforestation and wildfires on surface water resources, but few have looked at groundwater, and very few from the point of view of drinking water production.

The impact of deforestation and wildfires on groundwater and base flow in watersheds is still a poorly understood and under-researched topic [144,145]. Zhang and Wei [146] estimated that future research and watershed management require a systematic approach that considers key contributing factors and the broad range of response variables related to hydrological services. All explanatory variables—including climate, forests, watershed properties, and the interactions and feedback between these factors—should be assessed.

To further reveal the mechanisms for variable hydrological response to forest change, it is necessary to study the water behavior through the whole critical zone extending from the top of the vegetation canopy through the soil and down to fresh bedrock and the bottom of the groundwater [147]. McDonnell et al. [148] concluded that the paired watershed approach is the most popular tool for quantifying the effects of forest watershed management on water sustainability. However, this approach does not often address the critical factor of water stored in the landscape. Future work needs to quantify storage in paired watershed studies to inform sustainable water management.

Mirus et al. [8] noted that the paucity of long-term recovery records (>5 years duration) also limits our ability to assess hydrologic resilience and distinguish between lasting versus ephemeral changes. Further development of conceptual models for disturbances and recovery trajectories are needed and will be possible with continued field monitoring, which underscores the utility of long-term experimental data [149]. Ebel et al. [150] found hydrologic impacts of disturbances may extend for longer than several years duration. Mirus et al. [8] concluded that future research will also contribute to the development of new conceptual frameworks and modeling approaches to meet land management objectives, hazard warning goals, and societal needs in an increasingly disturbed world.

In light of the impact of deforestation and wildfires on water resources, future research should consider the following aspects:

- Monitor small logged watersheds over several years. The size of the watershed should be proportionate to that of the deforested area. Measurements should begin a few years before deforestation starts and continue for several years after logging finishes, in order to measure the effect of vegetation re-establishing itself.
- To eliminate the effect of meteorological factors, this monitoring should also be carried out simultaneously in neighboring, unlogged watersheds.
- For wildfires, it is difficult to predict where and when they will occur. This makes it difficult to identify which watersheds require monitoring. This also makes it more challenging to accurately assess the quantitative impact of wildfires on ungauged watersheds. Nevertheless, it is crucial to monitor burnt watersheds immediately after wildfires in order to comprehend their effect on hydrology and the subsequent changes as vegetation re-establishes itself. Neighboring unburned watersheds should also be monitored.
- The impact on groundwater should be assessed in the same way. This should involve setting up piezometric monitoring and gauging springs or small draining watercourses. The same recommendations apply to the monitoring of neighboring aquifers (or parts of aquifers), as well as to the specific issue of unpredictable wildfires.

In any case, it is important to bear in mind that these monitoring activities must be carried out over many years in order to take into account the re-establishment of vegetation. It should also be remembered that the results are specific to the site and climate in question, and are not necessarily transferable to other contexts.

In the case of resources used for producing drinking water, regular quality monitoring carried out in accordance with the relevant regulations will be beneficial. It would therefore be important to include, in regulatory monitoring analyses, any undesirable substances and contaminants that may occur. This would enable verification of the qualitative impact of any hazards.

Author Contributions: Conceptualization, O.B., N.L. and A.T.; Funding acquisition, N.L. and A.T.; Literature review, O.B., S.S.-P. and G.B.; Visualization, O.B. and G.B.; Writing—original draft, O.B., S.S.-P. and G.B.; Writing—review and editing, N.L. and A.T. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Conflicts of Interest: The authors declare no conflicts of interest.

Abbreviations

The following abbreviations are used in this manuscript:

PAH	Polycyclic aromatic hydrocarbon
PFAS	Perfluoroalkyl and Polyfluoroalkyl Substances
VOC	Volatile organic compound

References

1. Katsanou, K.; Karapanagioti, H. Surface Water and Groundwater Sources for Drinking Water. In *Applications of Advanced Oxidation Processes (AOPs) in Drinking Water Treatment*; Gil, A., Galeano, L.A., Vicente, M.A., Eds.; Springer: Cham, Switzerland, 2017; Chapter 140. [CrossRef]
2. Chavez, M.; Gonzalez, F.R. Sustainable Management of Groundwater and Surface Water Resources for Drinking Water Supply. *Hydrol. Process.* **2018**, *32*, 1467–1482. [CrossRef]
3. Smith, B.D.; Thompson, K.W. Surface Water vs. Groundwater: Water Quality and Security Considerations. *Environ. Sci. Technol.* **2020**, *54*, 10256–10265. [CrossRef]
4. Hughes, D.; Pardo, A. Groundwater versus Surface Water as Sources of Drinking Water: A Case Study of Water Quality and Treatment. *Water Res.* **2021**, *176*, 115783. [CrossRef]
5. Juuti, P.S.; Juuti, R.P.; Katko, T.S.; Lipponen, A.M.; Luonsi, A.O. Groundwater Option in Raw Water Source Selection and Related Policy Changes in Finland. *Public Work. Manag. Policy* **2022**, *28*, 189–214. [CrossRef]
6. Huck, P.M.; Coffey, B.M. The importance of robustness in drinking-water systems. *J. Toxicol. Environ. Health Part A* **2004**, *67*, 1581–1590. [CrossRef]
7. Nemani, K.S.; Peldszus, S.; Huck, P.M. Practical Framework for Evaluation and Improvement of Drinking Water Treatment Robustness in Preparation for Extreme-Weather-Related Adverse Water Quality Events. *ACS EST Water* **2023**, *3*, 1305–1313. [CrossRef]
8. Mirus, B.B.; Ebel, B.A.; Mohr, C.H.; Zegre, N. Disturbance hydrology: Preparing for an increasingly disturbed future. *Water Resour. Res.* **2017**, *53*, 10007–10016. [CrossRef]
9. Ebel, B.A.; Mirus, B.B. Disturbance hydrology: Challenges and opportunities. *Hydrol. Process.* **2014**, *28*, 5140–5148. [CrossRef]
10. Cai, Z.; Wang, J.; Yang, Y.; Zhang, R. Influence of Vegetation Coverage on Hydraulic Characteristics of Overland Flow. *Water* **2021**, *13*, 1055. [CrossRef]
11. Nedbal, V.; Bernasová, T.; Kobesová, M.; Tesařová, B.; Vácha, A.; Brom, J. Impact of landscape management and vegetation on water and nutrient runoff from small catchments for over 20 years. *J. Environ. Manag.* **2025**, *373*, 123748. [CrossRef] [PubMed]
12. Mapulanga, A.M.; Naito, H. Effect of deforestation on access to clean drinking water. *Proc. Natl. Acad. Sci. USA* **2019**, *116*, 8249–8254. [CrossRef]
13. Andreassian, V. Waters and forests: From historical controversy to scientific debate. *J. Hydrol.* **2004**, *291*, 1–27. [CrossRef]
14. Filoso, S.; Bezerra, M.O.; Weiss, K.C.B.; Palmer, M.A. Impacts of forest restoration on water yield: A systematic review. *PLoS ONE* **2017**, *12*, e0183210. [CrossRef]

15. Kändler, M.; Blechinger, K.; Seidler, C.; Pavlů, V.; Šanda, M.; Dostál, T.; Krása, J.; Vitvar, T.; Štich, M. Impact of land use on water quality in the upper Nisa catchment in the Czech Republic and in Germany. *Sci. Total Environ.* **2017**, *586*, 1316–1325. [CrossRef] [PubMed]
16. Clément, F.; Ruiz, J.; Rodríguez, M.A.; Blais, D.; Campeau, S. Landscape diversity and forest edge density regulate stream water quality in agricultural catchments. *Ecol. Indic.* **2017**, *72*, 627–639. [CrossRef]
17. Caldwell, P.V.; Martin, K.L.; Vose, J.M.; Baker, J.S.; Warziniack, T.W.; Costanza, J.K.; Frey, G.E.; Nehra, A.; Mihiar, C.M. Forested watersheds provide the highest water quality among all land cover types, but the benefit of this ecosystem service depends on landscape context. *Sci. Total Environ.* **2023**, *882*, 163550. [CrossRef]
18. Warziniack, T.; Sham, C.H.; Morgan, R.; Feferholtz, Y. Effect of Forest Cover on Water Treatment Costs. *Water Econ. Policy* **2017**, *3*, 1750006. [CrossRef]
19. Mupepele, A.-C.; Dormann, C.F. Influence of forest harvest on nitrate concentration in temperate streams—A Meta-analysis. *Forests* **2017**, *8*, 5. [CrossRef]
20. Piffer, P.R.; Tambosi, L.R.; de Ferraz, S.F.B.; Metzger, J.P.; Uriarte, M. Native forest cover safeguards stream water quality under a changing climate. *Ecol. Appl.* **2021**, *31*, e02414. [CrossRef]
21. Gay, E.T.; Martin, K.L.; Caldwell, P.V. Projected land use changes will cause water quality degradation at drinking water intakes across a regional watershed. *PLoS Water* **2025**, *4*, e0000313. [CrossRef]
22. Winter, C.; Nguyen, T.V.; Musolff, A.; Lutz, S.R.; Rode, M.; Kumar, R.; Fleckenstein, J.H. Droughts can reduce the Nitrogen retention capacity of catchments. *Hydrol. Earth Syst. Sci.* **2023**, *27*, 303–318. [CrossRef]
23. Price, J.I.; Heberling, M.T. The Effects of Source Water Quality on Drinking Water Treatment Costs: A Review and Synthesis of Empirical Literature. *Ecol. Econ.* **2018**, *151*, 195–209. [CrossRef]
24. Fiquepron, J.; Garcia, S.; Stenger, A. Land use impact on water quality: Valuing forest services in terms of the water supply sector. *J. Environ. Manag.* **2013**, *126*, 113–121. [CrossRef]
25. Ernst, C.; Gullick, R.; Nixon, K. Protecting the source: Conserving forests to protect water. *Opflow* **2004**, *30*, 4–7. [CrossRef]
26. Sthiannopkao, S.; Takizawa, S.; Homewong, J.; Wirojanagud, W. Soil erosion and its impacts on water treatment in the northeastern provinces of Thailand. *Environ. Int.* **2007**, *33*, 706–711. [CrossRef] [PubMed]
27. Abildtrup, J.; Garcia, S.; Stenger, A. The effect of forest land use on the cost of drinking water supply: A spatial econometric analysis. *Ecol. Econ.* **2013**, *92*, 126–136. [CrossRef]
28. Honey-Rosés, J.; Acuna, V.; Bardina, M.; Brozovic, N.; Marcé, R.; Munné, A.; Sabater, S.; Termes, M.; Valero, F.; Vega, A.; et al. Examining the demand for ecosystem services: The value of stream restoration for drinking water treatment managers in the Llobregat River, Spain. *Ecol. Econ.* **2013**, *90*, 196–205. [CrossRef]
29. Vincent, J.R.; Ahmad, I.; Adnan, N.; Burwell, W.B.; Pattanayak, S.K.; Tan-Soo, J.-S.; Thomas, K. Valuing water purification by forests: An analysis of Malaysian panel data. *Environ. Resour. Econ.* **2016**, *64*, 59–80. [CrossRef]
30. Namugize, J.N.; Jewitt, G.; Graham, M. Effects of land use and land cover changes on water quality in the uMngeni river catchment, South Africa. *Phys. Chem. Earth* **2018**, *105*, 247. [CrossRef]
31. Westling, N.; Stromberg, P.M.; Swain, R.B. Can upstream ecosystems ensure safe drinking water—Insights from Sweden. *Ecol. Econ.* **2020**, *169*, 106552. [CrossRef]
32. Liu, N.; Caldwell, P.V.; Dobbs, G.R.; Miniat, C.F.; Bolstad, P.V.; Nelson, S.A.C.; Sun, G. Forested lands dominate drinking water supply in the conterminous United States. *Environ. Res. Lett.* **2021**, *16*, 084008. [CrossRef]
33. Mulatu, D.W.; Fentie, A.; Siikamäki, J. The Impact of forest and non-forest cover on drinking water treatment costs: Panel evidence from Ethiopia. *Water Environ. J.* **2021**, *35*, 772–790. [CrossRef]
34. Piaggio, M.; Siikamäki, J. The value of forest water purification ecosystem services in Costa Rica. *Sci. Total Environ.* **2021**, *789*, 147952. [CrossRef] [PubMed]
35. Dearmont, D.; McCarl, B.A.; Tolman, D.A. Costs of water treatment due to diminished water quality: A case study in Texas. *Water Resour. Res.* **1998**, *34*, 849–853. [CrossRef]
36. Price, J.I.; Renzetti, S.; Dupont, D.; Adamowicz, W.; Emelko, M.B. Production costs, inefficiency, and source water quality: A stochastic cost frontier analysis of Canadian Water utilities. *Land Econ.* **2017**, *93*, 1–11. [CrossRef]
37. Nehra, A.; Baker, J.S.; Caldwell, P.V.; Martin, K.L.; Warziniack, T.W.; Manner, R.H.; Mihiar, C.M.; Frey, G.E.; Costanza, J.K. The potential impact of forest loss on drinking water treatment costs in the southeastern U.S. *For. Policy Econ.* **2025**, *179*, 103603. [CrossRef]
38. François, M.; de Aguiar, T.R., Jr.; Mielke, M.S.; Rousseau, A.N.; Faria, D.; Mariano-Neto, E. Interactions Between Forest Cover and Watershed Hydrology: A Conceptual Meta-Analysis. *Water* **2024**, *16*, 3350. [CrossRef]
39. Bosch, J.M.; Hewlett, J.D. A review of catchment experiments to determine the effect of vegetation changes on water yield, and evapotranspiration. *J. Hydrol.* **1982**, *55*, 3–23. [CrossRef]
40. Zhang, L.; Dawes, W.R.; Walker, G.R. Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resour. Res.* **2001**, *37*, 701–708. [CrossRef]

41. Gomyo, M.; Kuraji, K. Effect of the litter layer on runoff and evapotranspiration using the paired watershed method. *J. Res.* **2016**, *21*, 306–313. [CrossRef]
42. Danáčová, M.; Földes, G.; Labat, M.M.; Kohnová, S.; Hlavčová, K. Estimating the Effect of Deforestation on Runoff in Small Mountainous Basins in Slovakia. *Water* **2020**, *12*, 3113. [CrossRef]
43. Wiekenkamp, I.; Huisman, J.A.; Bogena, H.R.; Vereecken, H. Effects of Deforestation on Water Flow in the Vadose Zone. *Water* **2020**, *12*, 35. [CrossRef]
44. Te Wierik, S.A.; Cammeraat, E.L.H.; Gupta, J.; Artzy-Randrup, Y.A. Reviewing the Impact of Land Use and Land-Use Change on Moisture Recycling and Precipitation Patterns. *Water Resour. Res.* **2021**, *57*, e2020WR029234. [CrossRef]
45. Shadmehri Toosi, A.S.; Batelaan, O.; Shanafield, M.; Guan, H. Land Use-Land Cover and Hydrological Modeling: A Review. *WIREs Water* **2025**, *12*, e70013. [CrossRef]
46. Woodward, C.; Shulmeister, J.; Larsen, J.; Jacobsen, G.E.; Zawadzki, A. The hydrological legacy of deforestation on global wetlands. *Science* **2014**, *346*, 844–847. [CrossRef]
47. Makarieva, A.; Nefiodov, A.V.; Nobre, A.D.; Bardi, U.; Shell, D.; Baudena, M.; Salexka, S.; Rammig, A. How transpiration by forests and other vegetation determines alternate moisture regimes. *arXiv* **2022**, arXiv:2204.07409v1. [CrossRef]
48. van Luijk, G.; Cowling, R.M.; Riksen, M.J.P.M.; Glenday, J. Hydrological implications of desertification: Degradation of South African semi-arid subtropical thicket. *J. Arid Environ.* **2013**, *91*, 14–21. [CrossRef]
49. Archer, S.R.; Andersen, E.M.; Predick, K.I.; Schwinning, S.; Steidl, R.J.; Woods, S.R. Woody Plant Encroachment: Causes and Consequences. In *Rangeland Systems*; Briske, D., Ed.; Springer Series on Environmental Management; Springer: Cham, Switzerland, 2017. [CrossRef]
50. Qiao, L.; Zou, C.B.; Stebler, E.; Will, R.E. Woody plant encroachment reduces annual runoff and shifts runoff mechanisms in the tallgrass prairie, USA. *Water Resour. Res.* **2017**, *53*, 4838–4849. [CrossRef]
51. Huntoon, P.W. Hydrogeologic characteristics and deforestation of the Stone Forest karst aquifers of South China. *Groundwater* **1992**, *30*, 167–176. [CrossRef]
52. van Meerveld, H.J.; Jones, J.P.G.; Ghimire, C.P.; Zwartendijk, B.W.; Lahitiana, J.; Ravelona, M.; Mulligan, M. Forest regeneration can positively contribute to local hydrological ecosystem services: Implications for forest landscape restoration. *J. Appl. Ecol.* **2021**, *58*, 755–765. [CrossRef]
53. Wilcox, B.P.; Basant, S.; Olariu, H.; Leite, P.A.M. Ecohydrological connectivity: A unifying framework for understanding how woody plant encroachment alters the water cycle in drylands. *Front. Environ. Sci.* **2022**, *10*, 934535. [CrossRef]
54. Wilcox, B.P.; Huang, Y. Woody plant encroachment paradox: Rivers rebound as degraded grasslands convert to woodlands. *Geophys. Res. Lett.* **2010**, *37*. [CrossRef]
55. Hrachowitz, M.; Stockinger, M.; Coenders-Gerrits, M.; van der Ent, R.; Bogena, H.; Lücke, A.; Stumpp, C. Reduction of vegetation-accessible water storage capacity after deforestation affects catchment travel time distributions and increases young water fractions in a headwater catchment. *Hydrol. Earth Syst. Sci.* **2021**, *25*, 4887–4915. [CrossRef]
56. Ruiz, M.C.; Valdés-Abellán, J.; Pla, C.; Fernández-Mejuto, M.; Benavente, D. Land Cover Changes and Their Influence on Recharge in a Mediterranean Karstic Aquifer (Alicante, Spain). *Land* **2023**, *12*, 128. [CrossRef]
57. Acharya, B.S.; Kharel, G.; Zou, C.B.; Wilcox, B.P.; Halihan, T. Woody Plant Encroachment Impacts on Groundwater Recharge: A Review. *Water* **2018**, *10*, 1466. [CrossRef]
58. Mongil-Manso, J.; Navarro-Hevia, J.; San Martín, R. Impact of Land Use Change and Afforestation on Soil Properties in a Mediterranean Mountain Area of Central Spain. *Land* **2022**, *11*, 1043. [CrossRef]
59. Qazi, N. Hydrological functioning of forested catchments, Central Himalayan Region, India. *For. Ecosyst* **2020**, *7*, 63. [CrossRef]
60. Płaczkowska, E.; Mostowik, K.; Bogena, H.R.; Leuchner, M. The Impact of Partial Deforestation on Solute Fluxes and Stream Water Ionic Composition in a Headwater Catchment. *Water* **2023**, *15*, 107. [CrossRef]
61. Rosén, K.; Aronson, J.A.; Eriksson, H.M. Effects of clear-cutting on streamwater quality in forest catchments in central Sweden. *For. Ecol. Manag.* **1996**, *83*, 237–244. [CrossRef]
62. Houlton, B.Z.; Driscol, C.T.; Fahey, T.J.; Likens, G.E.; Groffman, P.M.; Bernhardt, E.S.; Buso, D.C. Nitrogen Dynamics in Ice Storm Damaged Forest Ecosystems: Implications for Nitrogen Limitation Theory. *Ecosystems* **2003**, *6*, 431–443. [CrossRef]
63. Sajdak, M.; Siwek, J.P.; Wasak-Sęk, K.; Kosmowska, A.; Stańczyk, T.; Małek, S.; Żelazny, M.; Woźniak, G.; Jelonkiewicz, Ł.; Żelazny, M. Stream water chemistry changes in response to deforestation of variable origin (case study from the Carpathians, southern Poland). *Catena* **2021**, *202*, 105237. [CrossRef]
64. Noteboom, M.; Seidou, O.; Lapen, D.R. Predicting water quality trends resulting from forest cover change in an agriculturally dominated river basin in Eastern Ontario. *Water Qual. Res. J.* **2021**, *56*, 218–238. [CrossRef]
65. Robinson, K.-L.; Bogena, H.R.; Cammeraat, E.; Bol, R. Effects of Deforestation on Dissolved Organic Carbon and Nitrate in Catchment Stream Water revealed by Wavelet Analysis. *Front. Water* **2022**, *4*, 1003693. [CrossRef]
66. De Mello, K.; Valente, R.A.; Randhir, T.O.; Vettorazzi, C.A. Impacts of tropical forest cover on water quality in agricultural watersheds in southeastern Brazil. *Ecol. Indic.* **2018**, *93*, 1293–1301. [CrossRef]

67. Narayanan, A.; Cohen, S.; Gardner, J.R. Riverine sediment response to deforestation in the Amazon basin. *Earth Surf. Dynam.* **2024**, *12*, 581–599. [CrossRef]
68. Zhao, B.; Lei, H.; Yang, D.; Yang, S.; Santisirisomboon, J. Runoff and sediment response to deforestation in a large Southeast Asian monsoon watershed. *J. Hydrol.* **2022**, *606*, 127432. [CrossRef]
69. Klein, R.D.; Lewis, J.; Buffleben, M.S. Logging and turbidity in the coastal watersheds of northern California. *Geomorphology* **2012**, *139–140*, 136–144. [CrossRef]
70. Sheikhy Narany, S.T.; Aris, A.Z.; Sefie, A.; Keesstra, S. Detecting and predicting the impact of land use changes on groundwater quality, a case study in Northern Kelantan, Malaysia. *Sci. Total Environ.* **2017**, *599–600*, 844–853. [CrossRef]
71. Brindha, K.; Parimalarenganayaki, S.; Elango, L. Sources, toxicological effects and removal techniques of nitrates in groundwater: An Overview. *Indian J. Environ. Prot.* **2017**, *37*, 667–700.
72. Rasolofoson, R.A.; Ricketts, T.H.; Johnson, K.B.; Jacob, A.; Fisher, B. Forests moderate the effectiveness of water treatment at reducing childhood diarrhea. *Environ. Res. Lett.* **2021**, *16*, 064035. [CrossRef]
73. Hosonuma, N.; Herold, M.; De Sy, V.; De Fries, R.S.; Brockhaus, M.; Verchot, L.; Angelsen, A.; Romijn, E. An assessment of deforestation and forest degradation drivers in developing countries. *Environ. Res. Lett.* **2012**, *7*, 044009. [CrossRef]
74. van Wees, D.; van der Werf, G.R.; Randerson, J.T.; Andela, N.; Chen, Y.; Morton, D.C. The role of fire in global forest loss dynamics. *Glob. Chang. Biol.* **2021**, *27*, 2377–2391. [CrossRef]
75. Folton, N.; Andréassian, V.; Duperray, R. Hydrological impact of forest-fire from paired-catchment and rainfall–runoff modelling perspectives. *Hydrol. Sci. J.* **2015**, *60*, 1213–1224. [CrossRef]
76. Wagenbrenner, J.W.; Ebel, B.A.; Bladon, K.D.; Kinoshita, A.M. Post-wildfire hydrologic recovery in Mediterranean climates: A systematic review and case study to identify current knowledge and opportunities. *J. Hydrol.* **2021**, *602*, 126772. [CrossRef]
77. Soulis, K.X.; Generali, K.A.; Papadaki, C.; Theodoropoulos, C.; Psomiadis, E. Hydrological Response of Natural Mediterranean Watersheds to Forest Fires. *Hydrology* **2021**, *8*, 15. [CrossRef]
78. Moazeni, S.; Cerdà, A. The impacts of forest fires on watershed hydrological response. A review. *Trees For. People* **2024**, *18*, 100707. [CrossRef]
79. González-Pelayo, O.; Prats, S.A.; van den Elsen, E.; Malvar, M.C.; Ritsema, C.; Bautista, S.; Keizer, J.J. The effects of wildfire frequency on post-fire soil surface water dynamics. *Eur. J. For. Res.* **2024**, *143*, 493–508. [CrossRef]
80. Atchley, A.L.; Kinoshita, A.M.; Lopez, S.R.; Trader, L.; Middleton, R. Simulating Surface and Subsurface Water Balance Changes Due to Burn Severity. *Vadose Zone J.* **2018**, *17*, 180099. [CrossRef]
81. Kinoshita, A.M.; Hogue, T.S. Spatial and temporal controls on post-fire hydrologic recovery in Southern California watersheds. *Catena* **2011**, *87*, 240–252. [CrossRef]
82. Wine, M.; Cadol, D. Hydrologic effects of large southwestern USA wildfires significantly increase regional water supply: Fact or fiction? Environ. *Res. Lett.* **2016**, *11*, 085006. [CrossRef]
83. Johnk, B.T.; Mays, D.C. Wildfire Impacts on Groundwater Aquifers: A Case Study of the 1996 Honey Boy Fire in Beaver County, Utah, USA. *Water* **2021**, *13*, 2279. [CrossRef]
84. Guzmán-Rojo, M.; Fernandez, J.; d'Abzac, P.; Huysmans, M. Impacts of Wildfires on Groundwater Recharge: A Comprehensive Analysis of Processes, Methodological Challenges, and Research Opportunities. *Water* **2024**, *16*, 2562. [CrossRef]
85. Pradhan, N.R.; Floyd, I. Examining the effect of moisture thresholds on post-fire water-repellent soil: A large-scale modelling approach applied to the Upper Arroyo Seco watershed, California, USA. *Int. J. Wildland Fire* **2024**, *33*, WF22083. [CrossRef]
86. Rodríguez-Jiménez, E.; Cruz-Pérez, N.; Koritnik, J.; García-Gil, A.; Marazuela, M.Á.; Santamarta, J.C. Revealing the impact of wildfires on groundwater quality: Insights from Sierra de la Culebra (Spain). *Chemosphere* **2024**, *365*, 143375. [CrossRef] [PubMed]
87. Guzmán-Rojo, M.; Silva de Freitas, L.; Coritza Taquichiri, E.; Huysmans, M. Groundwater Vulnerability in the Aftermath of Wildfires at the El Sutó Spring Area: Model-Based Insights and the Proposal of a Post-Fire Vulnerability Index for Dry Tropical Forests. *Fire* **2025**, *8*, 86. [CrossRef]
88. Partington, D.; Thyer, M.; Shanafield, M.; McInerney, D.; Westra, S.; Maier, H.; Simmons, C.; Croke, B.; Jakeman, A.J.; Gupta, H.; et al. Predicting wildfire induced changes to runoff: A review and synthesis of modeling approaches. *WIREs Water* **2022**, *9*, e1599. [CrossRef]
89. Webb, R.W.; Litvak, M.E.; Brooks, P.D. The role of terrain-mediated hydroclimate in vegetation recovery after wildfire. *Environ. Res. Lett.* **2023**, *18*, 064036. [CrossRef]
90. Ebel, B.A. Simulated unsaturated flow processes after wildfire and interactions with slope aspect. *Water Resour. Res.* **2013**, *49*, 8090–8107. [CrossRef]
91. Maina, F.Z.; Siirila-Woodburn, E.R. Watersheds dynamics following wildfires: Nonlinear feedbacks and implications on hydrologic responses. *Hydrol. Process.* **2020**, *34*, 33–50. [CrossRef]
92. Nyman, P.; Sheridan, G.; Lane, P.N. Synergistic effects of water repellency and macropore flow on the hydraulic conductivity of a burned forest soil, south-east Australia. *Hydrol. Process.* **2010**, *24*, 2871–2887. [CrossRef]

93. Stoof, C.R.; Slingerland, E.C.; Mol, W.; van den Berg, J.; Vermeulen, P.J.; Ferreira, A.J.D.; Ritsema, C.J.; Parlange, J.-Y.; Steenhuis, T.S. Preferential flow as a potential mechanism for fire-induced increase in streamflow. *Water Resour. Res.* **2014**, *50*, 1840–1845. [CrossRef]
94. Giambastiani, B.M.; Greggio, N.; Nobili, G.; Dinelli, E.; Antonellini, M. Forest Fire Effects on Groundwater in a Coastal Aquifer (Ravenna, Italy). *Hydrol. Process.* **2018**, *32*, 2377–2389. [CrossRef]
95. La Pasta Cordeiro, M.; Nunes, J.P.; Condesso de Melo, M.T. Impact of wildfires on spatial and temporal evolution of groundwater recharge in an Atlantic pine forest: An integrated approach using field, remote sensing and modeling. *J. Hydrol. Reg. Stud.* **2025**, *59*, 102408. [CrossRef]
96. Atwood, A.; Hille, M.; Clark, M.K. Rengers, F.; Ntarlagiannis, D.; Townsend, K.; West, A.J. Importance of Subsurface Water for Hydrological Response During Storms in a Post-Wildfire Bedrock Landscape. *Nat. Commun.* **2023**, *14*, 3814. [CrossRef] [PubMed]
97. Walvoord, M.A.; Ebel, B.A.; Partridge, T.F.; Rey, D.M.; Rosenberry, D.O. Disparate Groundwater Responses to Wildfire. *Wiley Interdiscip. Rev. Water* **2025**, *12*, e70029. [CrossRef]
98. Ebel, B.A.; Moody, J.A.; Martin, D.A. Hydrologic conditions controlling runoff generation immediately after wildfire. *Water Resour. Res.* **2012**, *48*, 3529. [CrossRef]
99. Stein, E.D.; Brown, J.S.; Hogue, T.S.; Burke, M.P.; Kinoshita, A. Stormwater contaminant loading following southern California wildfires Get access Arrow. *Environ. Toxicol. Chem.* **2012**, *31*, 2625–2638. [CrossRef]
100. Kean, J.W.; McGuire, L.A.; Rengers, F.K.; Smith, J.B.; Staley, D.M. Amplification of post wildfire peak flow by debris. *Geophys. Res. Lett.* **2016**, *43*, 8545–8553. [CrossRef]
101. Havel, A.; Tasdighi, A.; Arabi, M. Assessing the hydrologic response to wildfires in mountainous regions. *Hydrol. Earth Syst. Sci.* **2018**, *22*, 2527–2550. [CrossRef]
102. Carvalho-Santos, C.; Marcos, B.; Nunes, J.P.; Regos, A.; Palazzi, E.; Terzago, S.; Monteiro, A.T.; Honrado, J.P. Hydrological Impacts of Large Fires and Future Climate: Modeling Approach Supported by Satellite Data. *Remote Sens.* **2019**, *11*, 2832. [CrossRef]
103. Yu, M.; Bishop, T.F.A.; Van Ogtrop, F.F. Assessment of the Decadal Impact of Wildfire on Water Quality in Forested Catchments. *Water* **2019**, *11*, 533. [CrossRef]
104. Williams, A.P.; Livneh, B.; McKinnon, K.A.; Lettenmaier, D.P. Growing impact of wildfire on western US water supply. *Proc. Natl. Acad. Sci. USA* **2022**, *119*, e2114069119. [CrossRef]
105. Imeson, A.C.; Verstraten, J.M.; van Mulligen, E.J.; Sevink, J. The effects of fire and water repellency on infiltration and runoff under Mediterranean type forest. *Catena* **1992**, *19*, 345–361. [CrossRef]
106. Carrà, B.G.; Bombino, G.; Denisi, P.; Plaza-Àlvarez, P.A.; Lucas-Borja, M.E.; Zema, D.A. Water Infiltration after Prescribed Fire and Soil Mulching with Fern in Mediterranean Forests. *Hydrology* **2021**, *8*, 95. [CrossRef]
107. Robichaud, P.R. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. *J. Hydrol.* **2000**, *231–232*, 220–229. [CrossRef]
108. Moody, J.A.; Martin, D.A. Post-fire, rainfall intensity-peak discharge relations for three mountainous watersheds in the Western USA. *Hydrol. Process.* **2001**, *15*, 2981–2993. [CrossRef]
109. Bart, R.R.; Tague, C.L. The impact of wildfire on baseflow recession rates in California. *Hydrol. Process.* **2017**, *31*, 1662–1673. [CrossRef]
110. Úbeda, X.; Sala, M. Variations in runoff and erosion in three areas with different fire intensities. *Geookodynamik* **1998**, *19*, 179–188.
111. Hubbert, K.R.; Wohlgemuth, P.M.; Beyers, J.L.; Narog, M.G.; Gerrard, R. Post-Fire Soil Water Repellency, Hydrologic Response, and Sediment Yield Compared between Grass-Converted and Chaparral Watersheds. *Fire Ecol.* **2012**, *8*, 143–162. [CrossRef]
112. Papadaki, C.; Theodoropoulos, C.; Soulis, K.X.; Generali, K.A.; Psomiadis, E.; Dimitriou, E. Effects of forest fires on headwater streamflow and the habitat suitability for benthic macroinvertebrates. *Hydrol. Sci. J.* **2022**, *67*, 1356–1371. [CrossRef]
113. Lucas-Borja, M.E.; Bombino, G.; Carrà, B.G.; D'Agostino, D.; Denisi, P.; Labate, A.; Plaza-Alvarez, P.A.; Zema, D.A. Modeling the Soil Response to Rainstorms after Wildfire and Prescribed Fire in Mediterranean Forests. *Climate* **2020**, *8*, 150. [CrossRef]
114. Thomas, M.A.; Lindsay, D.N.; Cavagnaro, D.B.; Kean, J.W.; McCoy, S.W.; Graber, A.P. The Rainfall Intensity-Duration Control of Debris Flows After Wildfire. *Geophys. Res. Lett.* **2023**, *50*, e2023GL103645. [CrossRef]
115. Secci, D.; Tanda, M.G.; Oria, M.D.; Todaro, V.; Fagandini, C. Impacts of climate change on groundwater droughts by means of standardized indices and regional climate models. *J. Hydrol.* **2021**, *603*, 127154. [CrossRef]
116. Mayor, A.G.; Bautista, S.; Llovet, J.; Bellot, J. Post-Fire Hydrological and Erosional Responses of a Mediterranean Landscape: Seven Years of Catchment-Scale Dynamics. *Catena* **2007**, *71*, 68–75. [CrossRef]
117. Lane, P.N.; Sheridan, G.J.; Noske, P.J.; Sherwin, C.B.; Costenaro, J.L.; Nyman, P.; Smith, H.G. Fire Effects on Forest Hydrology: Lessons from a Multi-Scale Catchment Experiment in SE Australia. *IAHS Publ.* **2012**, *353*, 137–143.
118. Zhou, Y.; Zhang, Y.; Vaze, J.; Lane, P.; Xu, S. Impact of Bushfire and Climate Variability on Streamflow from Forested Catchments in Southeast Australia. *Hydrol. Sci. J.* **2015**, *60*, 1340–1360. [CrossRef]

119. Mansilha, C.; Melo, A.; Martins, Z.E.; Ferreira, I.M.P.L.V.O.; Pereira, A.M.; Espinha Marques, J. Wildfire Effects on Groundwater Quality from Springs Connected to Small Public Supply Systems in a Peri-Urban Forest Area (Braga Region, NW Portugal). *Water* **2020**, *12*, 1146. [CrossRef]
120. Alexakis, D.D.; Hadjimitsis, D.G.; Agapiou, A. Integrated use of remote sensing, GIS and precipitation data for the assessment of soil erosion rate in the catchment area of "Yialias" in Cyprus. *Atmos. Res.* **2013**, *131*, 108–124. [CrossRef]
121. Malmon, D.V.; Reneau, S.L.; Katzman, D.; Lavine, A.; Lyman, J. Suspended sediment transport in an ephemeral stream following wildfire. *J. Geophys. Res. Earth Surf.* **2007**, *112*, F2. [CrossRef]
122. Smith, H.G.; Sheridan, G.J.; Lane, P.N.J.; Nyman, P. Wildfire effects on water quality in forest catchments: A review with implications for water supply. *J. Hydrol.* **2011**, *396*, 170–192. [CrossRef]
123. Emelko, M.B.; Silins, U.; Bladon, K.D.; Stone, M. Implications of land disturbance on drinking water treatability in a changing climate: Demonstrating the need for "source water supply and protection" strategies. *Water Res.* **2011**, *45*, 461–472. [CrossRef]
124. Porter, B.W.; Sowby, R.B.; Williams, G.P.; Limb, B.J.; Quinn, J.C.; Johnson, A. Mitigating Wildfire Impact on Water Quality through Climate-Based Financing: A Case Study of the Provo River Watershed. *ACS EST Water* **2025**, *5*, 649–658. [CrossRef]
125. Hampton, T.B.; Lin, S.; Basu, N.B. Forest fire effects on stream water quality at continental scales: A meta-analysis. *Environ. Res. Lett.* **2022**, *17*, 064003. [CrossRef]
126. Son, J.H.; Kim, S.; Carlson, K.H. Effects of Wildfire on River Water Quality and Riverbed Sediment Phosphorus. *Water Air Soil Pollut.* **2015**, *226*, 26. [CrossRef]
127. Rust, A.; Hogue, T.S.; Saxe, S.; Mccray, J.E. Post-fire water-quality response in the western United States. *Int. J. Wildland Fire* **2018**, *27*, 203–216. [CrossRef]
128. Klimas, K.; Hiesl, P.; Hagan, D.; Park, D. Prescribed fire effects on sediment and nutrient exports in forested environments: A review. *J. Environ. Qual.* **2020**, *49*, 793–811. [CrossRef] [PubMed]
129. De Palma-Dow, A.; McCullough, I.M.; Brentrup, J.A. Turning up the heat: Long-term water quality responses to wildfires and climate change in a hypereutrophic lake. *Ecosphere* **2022**, *13*, e4271. [CrossRef]
130. Kirn, L.; Mudarra, M.; Marín, A.; Andreo, B.; Hartmann, A. Improved Assessment of Groundwater Recharge in a Mediterranean Karst Region: Andalusia, Spain. In *EuroKarst 2016, Neuchâtel. Advances in Karst Science 2017*; Renard, P., Bertrand, C., Eds.; Springer: Cham, Switzerland, 2017. [CrossRef]
131. Dimitriadou, S.; Katsanou, K.; Charalabopoulos, S.; Lambrakis, N. Interpretation of the Factors Defining Groundwater Quality of the Site Subjected to the Wildfire of 2007 in Ilia Prefecture, South-Western Greece. *Geosciences* **2018**, *8*, 108. [CrossRef]
132. Orlova, J.; Olefeldt, D.; Yasinski, J.H.; Anderson, A.E. Effects of prescribed burn on nutrient and dissolved organic matter characteristics in peatland shallow groundwater. *Fire* **2020**, *3*, 53. [CrossRef]
133. Pennino, M.J.; Leibowitz, S.G.; Compton, J.E.; Beyene, M.T.; LeDuc, S.D. Wildfires can increase regulated nitrate, arsenic, and disinfection byproduct violations and concentrations in public drinking water supplies. *Sci. Total Environ.* **2022**, *804*, 149890. [CrossRef]
134. Gunnarsdottir, M.J.; Tómasdóttir, S.; Örlygsson, O.; Andradóttir, H.Ó.; Gardarsson, S.M. Impact of wildfires on the drinking water catchment for the capital area of Iceland—A case study. *Environ. Sci. Adv.* **2025**, *4*, 606–618. [CrossRef]
135. Solomon, G.M.; Hurley, S.; Carpenter, C.; Young, T.M.; English, P.; Reynolds, P. Fire and Water: Assessing Drinking Water Contamination After a Major Wildfire. *ACS EST Water* **2021**, *1*, 1878–1886. [CrossRef]
136. Tobin, B.W.; Schwartz, B.F.; Kelly, M.; Despain, J.D. Fire retardant and post-fire nutrient mobility in a mountain surface water—Karst groundwater system: The Hidden Fire, Sequoia National Park, California, USA. *Environ. Earth Sci.* **2014**, *73*, 951–960. [CrossRef]
137. Ghahremani, Z.; Iradukunda, P.; Farid, A. Post-wildfire soil and aquifer contamination: A review. *Jpn. Geotech. Soc. Spec. Publ.* **2021**, *9*, 175–179. [CrossRef]
138. Mussabek, D.; Söderman, A.; Imura, T.; Persson, K.M.; Nakagawa, K.; Ahrens, L.; Berndtsson, R. PFAS in the Drinking Water Source: Analysis of the Contamination Levels, Origin and Emission Rates. *Water* **2023**, *15*, 137. [CrossRef]
139. Benaafi, M.; Bafaqeer, A. Comprehensive Review of Global Perspectives on Per- and Polyfluoroalkyl Compounds: Occurrence, Fate, and Remediation in Groundwater Systems. *Water* **2024**, *16*, 1583. [CrossRef]
140. Sims, J.L.; Stroski, K.M.; Kim, S.; Killeen, G.; Ehalt, R.; Simcik, M.F.; Brooks, B.W. Global Occurrence and Probabilistic Environmental Health Hazard Assessment of Per- and Polyfluoroalkyl Substances (PFASs) in Groundwater and Surface Waters. *Sci. Total Environ.* **2022**, *816*, 151535. [CrossRef]
141. Johnson, G.R.; Brusseau, M.L.; Carroll, K.C.; Tick, G.R.; Duncan, C.M. Global distributions, source-type dependencies, and concentration ranges of per- and polyfluoroalkyl substances in groundwater. *Sci. Total Environ.* **2022**, *841*, 156602. [CrossRef]
142. Rasmusson, K.; Fagerlund, F. Per- and polyfluoroalkyl substances (PFAS) as contaminants in groundwater resource—A comprehensive review of subsurface transport processes. *Chemosphere* **2024**, *362*, 142663. [CrossRef]
143. Hou, Y.; Wei, X.; Zhang, M.; Creed, I.F.; McNulty, S.G.; Ferraz, S.F.B. A global synthesis of hydrological sensitivities to deforestation and forestation. *For. Ecol. Manag.* **2023**, *529*, 120718. [CrossRef]

144. Paul, M.J.; LeDuc, S.D.; Lassiter, M.G.; Moorhead, L.C.; Noyes, P.D.; Leibowitz, S.G. Wildfire induces changes in receiving waters: A review with considerations for water quality management. *Water Resour. Res.* **2022**, *58*, e2021WR030699. [CrossRef]
145. Elliott, S.M.; Hornberger, M.I.; Rosenberry, D.O.; Frus, R.J.; Webb, R.M. A Conceptual Framework to Assess Post-Wildfire Water Quality: State of the Science and Knowledge Gaps. *Water Resour. Res.* **2024**, *60*, e2023WR036260. [CrossRef]
146. Zhang, M.; Wei, X. Deforestation, forestation, and water supply. *Science* **2021**, *371*, 990–991. [CrossRef]
147. Grant, G.E.; Dietrich, W.E. The frontier beneath our feet. *Water Resour. Res.* **2017**, *53*, 2605–2609. [CrossRef]
148. McDonnell, J.J.; Evaristo, J.; Bladon, K.D.; Buttle, J.; Creed, I.F.; Dymond, S.F.; Grant, G.; Iroume, A.; Jackson, C.R.; Jones, J.; et al. Water sustainability and watershed storage. *Nat. Sustain.* **2018**, *1*, 378–379. [CrossRef]
149. Tetzlaff, D.; Carey, S.K.; McNamara, J.P.; Laudon, H.; Soulsby, C. The essential value of long-term experimental data for hydrology and water management. *Water Resour. Res.* **2017**, *53*, 2598–2604. [CrossRef]
150. Ebel, B.A.; Rengers, F.K.; Tucker, G.E. Observed and simulated hydrologic response for a first-order catchment during extreme rainfall three years after wildfire disturbance. *Water Resour. Res.* **2016**, *52*, 9367–9389. [CrossRef]