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# Impact of Eastern Redcedar Encroachment on Water Resources in the Nebraska Sandhills

Yaser Kishawi
*University of Nebraska-Lincoln*, ykishawi2@unl.edu

Aaron R. Mittelstet
*University of Nebraska-Lincoln*, amittelstet2@unl.edu

Troy E. Gilmore
*University of Nebraska-Lincoln*, gilmore@unl.edu

Dirac Twidwell
*University of Nebraska-Lincoln*, dirac.twidwell@unl.edu

Tirthankar Roy
*University of Nebraska-Lincoln*, roy@unl.edu

## Authors

Yaser Kishawi, Aaron R. Mittelstet, Troy E. Gilmore, Dirac Twidwell, Tirthankar Roy, and Nawaraj Shrestha

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Yaser Kishawi,[1] Aaron R. Mittelstet,[1] Troy E. Gilmore,[1,2]
Dirac Twidwell,[3] Tirthankar Roy,[4]
and Nawaraj Shrestha[2]

1 Department of Biological Engineering, University of Nebraska–Lincoln, 223 L. W. Chase Hall, Lincoln, NE 68583-0726, United States
2 School of Natural Resources, University of Nebraska–Lincoln, 101 Hardin Hall, Lincoln, NE 68583-0961, United States
3 Department of Agronomy and Horticulture, University of Nebraska–Lincoln, Lincoln, NE 68503-0984, United States
4 Department of Civil and Environmental Engineering, University of Nebraska–Lincoln, 1110 S. 67th St., Omaha, NE 68182-0178, United States

*Corresponding author* — A.R. Mittelstet, amittelstet2@unl.edu

**Abstract**
Worldwide, tree or shrub dominated woodlands have encroached into herbaceous dominated grasslands. While very few studies have evaluated the impact of Eastern Redcedar (redcedar) encroachment on the water budget, none have analyzed the impact on water quality. In this study, we evaluated the impact of redcedar encroachment on the water budget in the Nebraska Sand Hills and how the decreased streamflow would increase nitrate and atrazine concentrations in the Platte River. We calibrated a Soil and Water Assessment Tool (SWAT model) for streamflow, recharge, and evapotranspiration. Using a moving window with a dilate morphological filter, encroachment scenarios of 11.9%, 16.1%, 28.0%, 40.6%, 57.5%, 72.5% and 100% were

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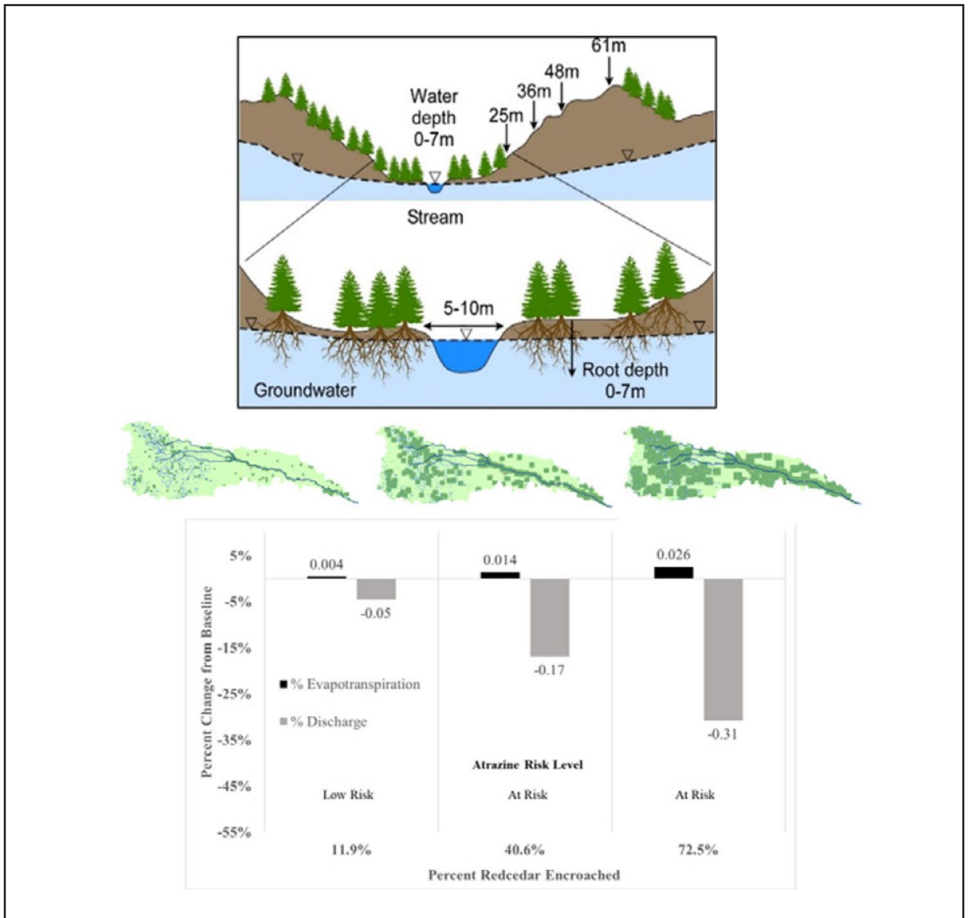
developed and simulated by the calibrated model. At 11.9% and 100% encroachment, streamflow was reduced by 4.6% and 45.5%, respectively in the Upper Middle Loup River, a tributary to the Platte River. Percolation and deep aquifer recharge increased by 27% and 26% at 100% encroachment. Streamflow in the Platte River, a major water source for Omaha and Lincoln, would decrease by 2.6%, 5.5% and 10.5% for 28%, 57.5%, and 100% encroachment of the Loup River watershed, respectively. This reduction in streamflow could increase nitrate and atrazine concentrations in the Platte River by 4 to 15% and 4 to 30%, respectively. While the density of redcedar is minimal, it is important to manage their encroachment to prevent reductions in streamflow and potential increases in pollutant concentrations.

**Keywords:** Soil and Water Assessment Tool, Loup River, Platte River, Atrazine, Baseflow-dominated watershed

**Highlights**

- Streamflow for 100% redcedar encroachment reduced by 46% compared to historical flows .
- Encroachment increases atrazine by 4 to 30% compared to historical concentrations.
- Nitrate increases from 0.89–0.94 mg/L to 0.98–1.02 mg/L for 100% encroachment.



## 1. Introduction

Globally, grasslands are shifting rapidly to woody-plant dominance (Anadón et al., 2014; Eldridge et al., 2011; Zou et al., 2014). This phenomenon is known as woody-plant encroachment and is driving severe changes to numerous ecosystem services in grasslands worldwide. Woody-plant encroachment into grasslands has reduced biodiversity and food production (Fuhlendorf et al., 2008; Ratajczak et al., 2012), altered hydrological function and decreased freshwater supply (Schreiner-McGraw et al., 2020; Zou et al., 2018), and increased risk of natural disasters associated with large wildfire occurrence (Donovan et al., 2020; Fuhlendorf et al., 2008; Ratajczak et al., 2012; Zou et al., 2018). The underlying cause for woody-plant expansion into grasslands is owed to interacting drivers that span local-to-global scales (O'Connor et al., 2020), including global increases in $CO_2$, shifts in temperature and precipitation patterns, the displacement of diverse and free-ranging native herbivore species with domesticated livestock, increasing seed dispersal through human transportation and tree plantings, and global departures in anthropogenically-driven fire regimes (Archer et al., 2017; Wilcox et al., 2022).

One of the most notorious species encroaching into grasslands in North America is Eastern Redcedar (redcedar) (Engle et al., 2008; Fogarty et al., 2020; Twidwell et al., 2013). Redcedar (*Juniperus virginiana*) is native to North America but historically was limited in abundance due to fire (Axmann and Knapp, 1993). Meneguzzo and Liknes (2015) found that redcedar has increased range from the eastern coasts of the U.S. to the Midwest states reaching as far as western Nebraska. They concluded that the redcedar geographic distribution in the central U.S. is considered widespread, with the most significant increases occurring in Nebraska and Missouri during the early 2000s.

Several additional studies have quantified the redcedar encroachment in Nebraska, including the Nebraska Sand Hills (NSH), the principle recharge area for the High Plains Aquifer (Adane et al., 2017; Adane and Gates, 2014; Awada et al., 2013). According to Walker and Hoback (2007), the rate of redcedar encroachment was 2% annually, and in the past 30 years, the coverage has increased to 30% in the Loess Canyons in southeastern Lincoln County in Nebraska. In the last 20 years, redcedar has noticeably increased from 10,000 to 300,000 trees (30 times) in the NSH (Nebraska Forest Service, 2016).

The origins of redcedar encroachment and the underlying mechanisms driving woody-plant expansion are well understood in the NSH. Government-backed incentives programs have advocated for the planting of redcedar into open grasslands in North America for >100-years (Donovan et al., 2018; Gardner, 2009). More than 1.8 million redcedar trees were distributed for planting in the Great Plains in 2001, with more trees planted in Nebraska than any other state (Ganguli et al., 2008). Redcedar is planted as a windbreak or shelterbelt in the NSH to provide shelter for livestock, homes, and other structures, but redcedar windbreaks serve as a seed-source for encroachment into Sand Hills grassland in the absence of frequent disturbances (Fogarty et al., 2020). Prior to human transportation and dispersal through tree planting programs, redcedar was a rare feature in the NSH. Redcedar is one of the most fire-sensitive tree species in the Great Plains and was restricted to sites where it could escape fire damage (Twidwell et al., 2021); however, fire is now mostly absent from the Sand Hills and alternative disturbances (e.g., mechanical treatments) have not been applied in a way that prevents woody-plant expansion (Fogarty et al., 2020; Roberts et al., 2018; Scholtz et al., 2021).

The impacts of redcedar encroachment in the Great Plains have been evaluated by scientists since the beginning of the 20th century. Based on a comprehensive review by Bielski et al. (2017), the impact of redcedar on soil was documented in the 1940s, the impact on livestock production in the 1970s, and recently the impact on society. For example, it was found that high calcium content (>2%) of redcedar foliage changed soils from acidic to alkaline, thus raising the pH of the soils and impacting the native vegetation. The impacts can be categorized into social costs, changes in biodiversity and productivity and biophysical changes. Redcedar control costs for public schools (or land in related trusts), a social cost, have increased by $250,000 since 2006, and the displacement of herbaceous production by woody plants has decreased livestock production by 75% in multiple long-term experimental investigations. Wildfire risk has shifted from frequent grass-driven fires to infrequent juniper driven crown fires with longer flames lengths (Bielski et al., 2017). Biodiversity and productivity have been impacted for several species; for example, grassland birds and prairie chickens have been replaced by woodland/shrubland birds in areas with >10% encroachment. The richness of grassland species declined by 88% in redcedar encroached areas.

Finally, the biophysical impacts include carbon storage shifts from beneath grasslands (96%) to above ground in redcedar (52%). This shift to aboveground storage has increased the potential for rapid losses of carbon due to disturbance factors such as drought, wildfire, disease, or insect outbreaks in the encroached areas (Bielski et al., 2017).

Multiple studies have evaluated the impacts of redcedar on water resources, yet none have evaluated their impact in the NSH. Zou et al. (2018) reviewed the impacts of redcedar proliferation on water resources in the U.S. Great Plains. The study concluded that watersheds with redcedar encroachment have increased evapotranspiration and precipitation loss to canopy interception. This leads to soil moisture depletion and reductions in surface runoff and deep recharge. A study by Wine and Hendrickx (2013) investigated the bio-hydrologic effects of redcedar encroachment into Oklahoma grassland. The study found that the average evapotranspiration from grassland and redcedar was 787 mm (95% of precipitation) and 798 mm (97% of precipitation), respectively. The impacts of redcedar encroachment on water resources in Nebraska, specifically in the NSH, are significant for multiple reasons. First, the NSH is one of the last remaining intact prairie regions in the world (Scholtz and Twidwell, 2022) and has only recently begun to experience woody encroachment (Fogarty et al., 2020). Second, landscape transformation (from rangeland to redcedar) is believed to reduce groundwater recharge and discharge to the stream system. Lastly, the Loup River, which drains the NSH, is a major tributary to the Platte River, a vital waterway for Nebraska. Understanding the hydrological impacts of redcedar encroachment will play an important role in the sustainability of the High Plains Aquifer and the Nebraska rivers and ecosystems. In this study, we use the Soil and Water Assessment Tool (SWAT) to model the impacts of redcedar encroachment on streamflow, evapotranspiration and recharge at the watershed scale within the NSH. Few studies have applied the SWAT model to evaluate the impacts of redcedar on water resources (Qiao et al., 2015; Starks and Moriasi, 2017). Starks and Moriasi (2017) conducted a modeling study in the central reach of the North Canadian River basin in central Oklahoma. They found that if rangeland was replaced by redcedar completely (100% encroachment), a reduction in stream discharge could reach 112% of the current municipal water demand and 89% of the projected 2060 demand. This was supported by Zou et al. (2018), where they assessed the impacts of redcedar proliferation on water resources in the Great Plains, U.S. using the SWAT model. They found that a complete conversion from rangeland to redcedar would result in a reduction in streamflow throughout the year between 20 and 40% depending on the aridity of the climate. None of these studies evaluated the impact of redcedar encroachment on water quality, a recommended topic for future research identified by Zou et al. (2018). The objectives of this study were to evaluate the (1) impacts of the current and future redcedar encroachments on the streamflow, evapotranspiration and recharge in the NSH and (2) implications of redcedar encroachment on the water quantity and quality in the Platte River, a vital river in Nebraska. We hypothesize that as redcedar encroachment increases, streamflow and recharge will decrease, evapotranspiration will increase and nitrate and atrazine concentrations in the Platte River will increase.

## 2. Materials and methods

### *2.1. Study area*

The NSH is an area of vegetated sand dunes located in central to western Nebraska with a total area of approximately 50,000 km$^2$ (Ahlbrandt and Fryberger, 1980; Smith, 1965; Sweeney and Loope, 2001). It consists mainly of interdunal basins, connected with an unconfined aquifer, and hosts around 4700 lakes and over 2000 km$^2$ of wetlands (Dappen et al., 2007). The climate in the NSH is semiarid, with annual precipitation ranging from 406 mm (west) to 610 mm (east) and an average temperature of 8.9 °C (Ahlbrandt and Fryberger, 1980). Currently the watershed consists of 93.6% pasture, 4.3% wetland and only 0.2% forest (predominantly redcedar) (Dewitz, 2019). With a maximum saturated thickness of about 300 m in western Nebraska (Miller and Appel, 1997), the NSH overlies the majority of unconfined groundwater storage within the High Plains Aquifer. Historically there has been little evidence of reductions in groundwater storage beneath the NSH (Haacker et al., 2016; McGuire, 2017; Peterson et al., 2016), but the shallow water table in much of the NSH increases vulnerability to redcedar encroachment and climate change (Adane et al., 2019; Burbach and Joeckel, 2006; Zou et al., 2018).

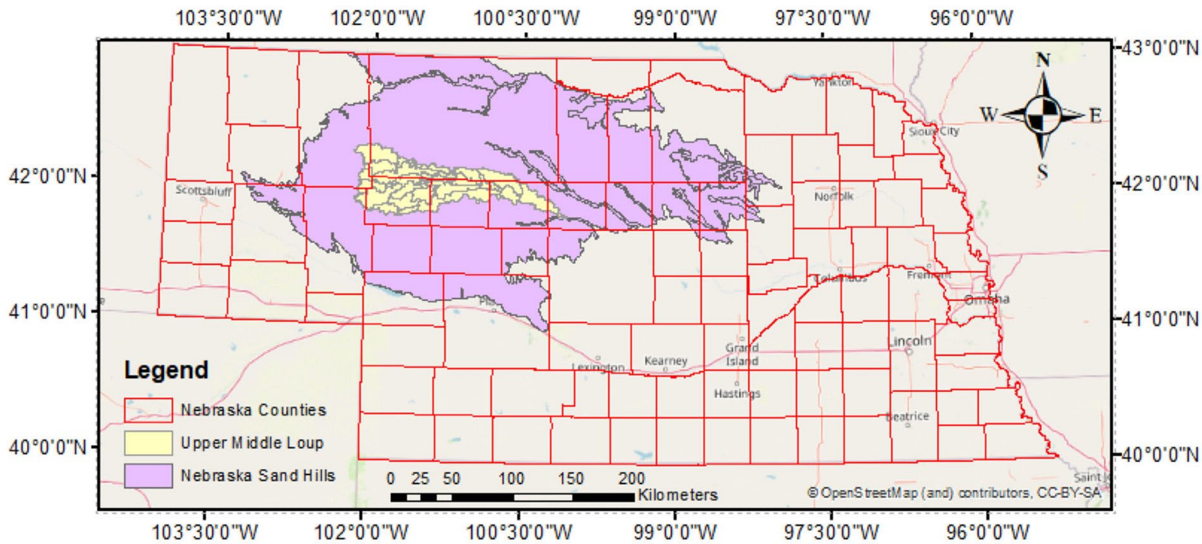



**Fig. 1.** Location of the Upper Middle Loup watershed compared with Nebraska Sandhills and Nebraska state map.

The Upper Middle Loup River (UMLR) watershed is in the center of the NSH with an area of 4950 km$^2$ (**Fig. 1**). The UMLR watershed is baseflow dominated (95% baseflow) (Szilagyi et al., 2011a,b). **Fig. 2**a illustrates the water depth in the NSH, estimated as the difference between the 30-m digital elevation model (DEM) (USGS, 1999) and spring 1995

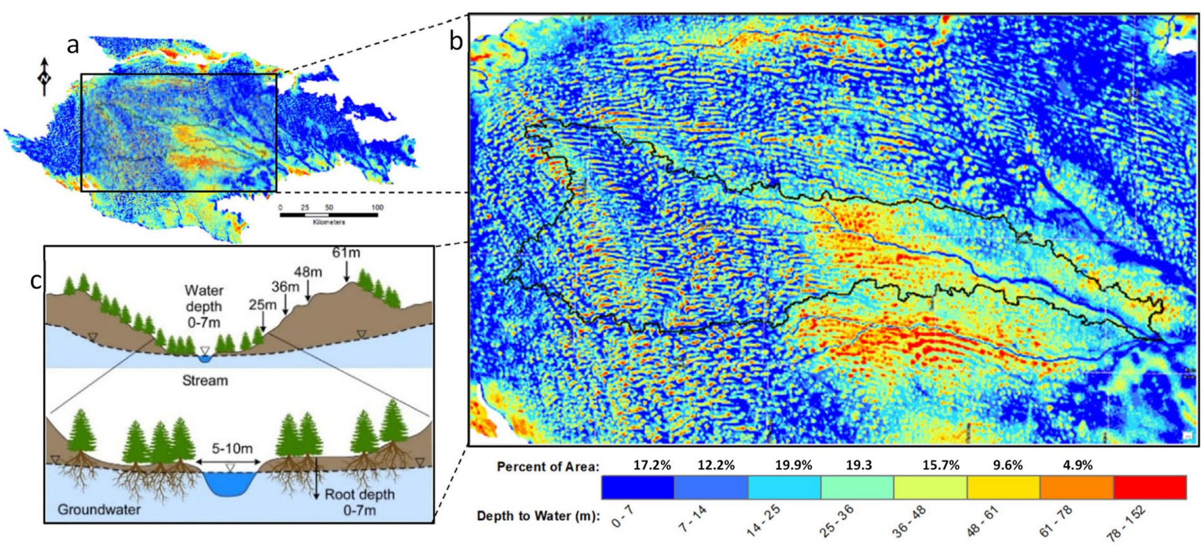



**Fig. 2.** (a) The depth of water (m) in the Nebraska Sand Hills (source: Rossman et al., 2014) and (b) the Upper Middle Loup watershed, (c) schematic drawing of Eastern Redcedar root penetration into the vadose zone and water table.

water table data. As shown in Fig. 2b, the water depths near the streams, where redcedar usually persist, is 0–4 m. This is within the range of redcedar roots and would probably be impacted with redcedar encroachments, especially where water depths are shallow. According to Anderson (2003), redcedar roots can penetrate 7.5 m which increases the access to the water table to >17%. Fig. 2c shows conceptually how redcedar can have access to an unconfined aquifer and can limit seepage to the lake and streams.

### *2.2. Eastern Redcedar encroachment*

While currently <1% of our study area is redcedar, 256,653 ha of grassland was converted to woody vegetation, predominantly redcedar, from 2007 to 2017 in Nebraska (Fogarty et al., 2020). Nearly 21,000 ha of Sandhills grassland was converted to woody vegetation, mostly redcedar. The encroachment rate has increased significantly east, north and south of our study area (Fogarty et al., 2020). Since redcedar is fire-intolerant, their population has been historically controlled by wildfires. Reduction in wildfires since European settlement and the increased redcedar plantings for windbreaks has resulted in increased redcedar encroachment. **Fig. 3** illustrates a common image in the NSH where trees are emerging in locations where they had not been before.

Starks and Moriasi (2017) evaluated redcedar encroachment impacts on stream discharge based on three scenarios (one to simulate the baseline encroachment, a second to simulate encroachment removal redcedar-to-grassland, and a third scenario grassland-to-redcedar by 10% increments up to 100%). However, the spatial distribution and expansion of redcedar were not considered in conjunction with the 10% incremental changes. In this study, the impact of redcedar on water resources in the UMLR watershed was evaluated using encroachment scenarios that incorporated different spatial distributions and expansion. Encroachment scenarios were created based on existing redcedar cover and spatial variation of encroachment representative of the present and potential future environmental conditions. The baseline scenario (present condition, 4a) had <1% encroachment in the watershed area. The additional encroachment scenarios, developed from baseline scenario using neighborhood approach, represent redcedar coverage of 11.9%, 16.1%, 28.0%, 40.6%, 57.5%, 72.5%, and 100% of the watershed

**Fig. 3.** Eastern Redcedar (redcedar) near a stream in the study area in the Nebraska Sand Hills. At the top of the image, one can see single trees emerging on the hillside. Red arrows pointing to redcedar trees. Picture taken by authors.

area. In these scenarios, only rangeland was encroached while other land uses (e.g., lakes, wetlands, urban) remained constant. For example, 100% encroachment means that 100% of the pasture (93.6% of total watershed area) is converted to redcedar. Each of the scenarios are simulated as though the conversion is immediate, while in reality it will take decades or centuries to reach some of these encroachment scenarios. Though 100% rangeland to redcedar coverage is unlikely and would take 50 years at a 2% encroachment rate, we have no way of knowing if the encroachment rate will increase (due to increased planting) or decrease (due to more controlled fires). These scenarios will demonstrate the potential impact if these encroachment scenarios are realized.

The baseline scenario was created by combining the evergreen and mixed forest land cover classes from National Land Cover Database (NLCD) 2016. Redcedar is the dominant tree species comprising at least 90% of the conifer basal area in the Great Plains (Filippelli et al., 2020). With redcedar encroachment at the early stages and occurrence of redcedar as understory species, we assumed major proportion of mixed forest is comprised of redcedar in mixed forest areas. The NLCD data derived from Landsat images at 30 m resolution do not detect such occurrence of redcedar and therefore we assumed that including the mixed forest could also compensate for undetected redcedar that are significant for future encroachment scenarios. The baseline map was classified as a binary image, the presence of redcedar is represented with a value of

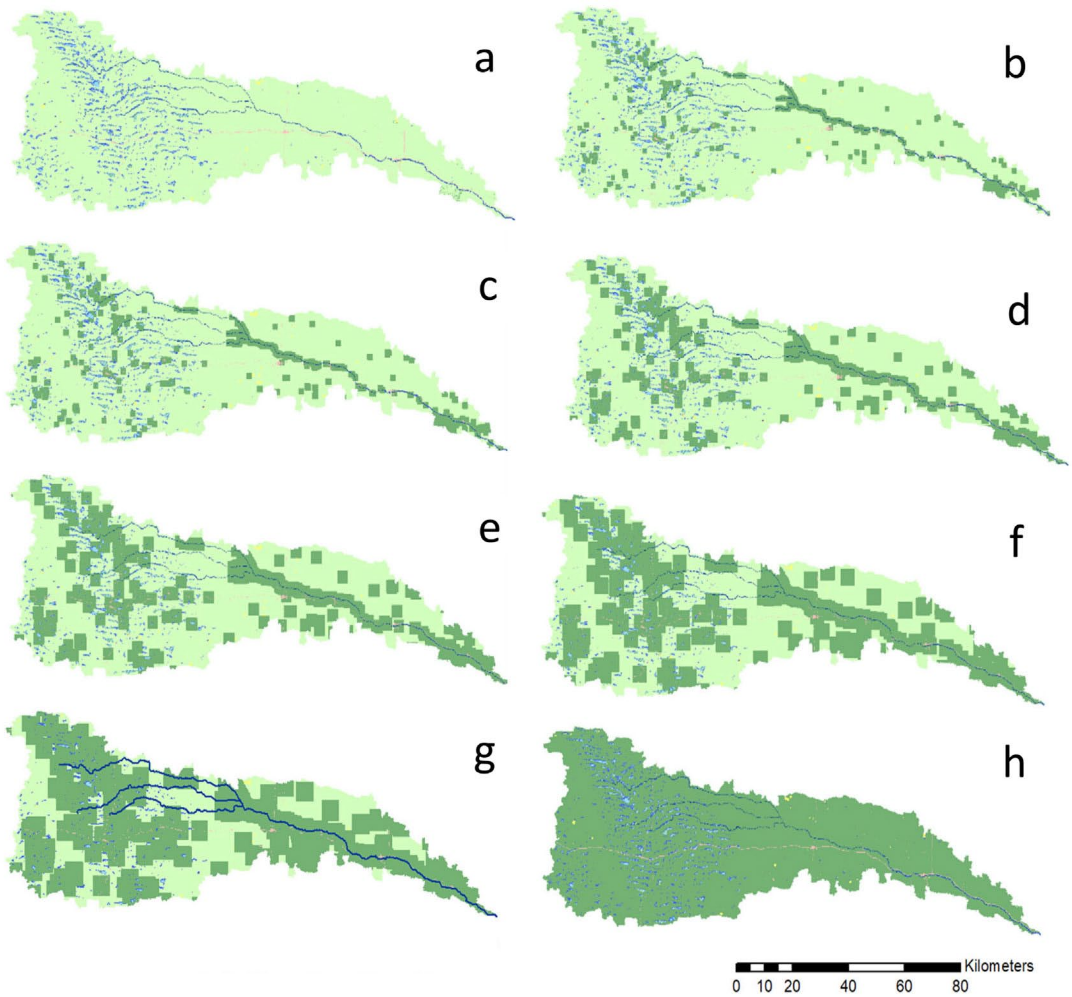



**Fig. 4.** Eastern Redcedar (ERC), represented by dark green color, encroachment scenarios (a) baseline scenario <1%, (b) 11.9% (c) 16.1%(d) 28.0%(e) 40.6% (f) 57.5% and (g) 72.5% and (h) 100%.

1 while absence with 0. The binary image was passed through a moving window (3 × 3 m to 7 × 7 m) with a dilate morphological filter (Haralick et al., 1987). When a binary image passes through the process of dilation, the area with value of 0 (non-redcedar/no encroachment) is replaced by 1 (redcedar) representing an encroachment occurring. The process was iterated to create redcedar encroachment percentages of 11.9%, 16.1%, 28.0%, 40.6%, 57.5%, 72.5% and 100% as shown in **Fig. 4.**

**Table 1** Percentage of each land use, soil type and slope for the Upper Middle Loup River watershed used in the SWAT model.

| *LULC* | *Area%* | *Soil type* | *Area%* | *Slope* | *Area%* |
|---|---|---|---|---|---|
| Water | 1.07% | NEW† | 0.44% | 0–2% | 19.44% |
| Wetlands | 4.27% | NE081 | 1.94% | 2–4% | 16.65% |
| Urban land | 0.68% | NE133 | 40.61% | 4–6% | 14.86% |
| Forest | 0.19% | NE134 | 17.08% | 6–10% | 24.74% |
| Pasture | 93.56% | NE135 | 6.00% | >10% | 24.31% |
| Corn | 0.23% | NE137 | 33.16% | – | – |
| – | – | NE146 | 0.77% | – | – |

† SWAT defined NEW soil type where soil data is missing. The model creates a new soil profile and the user should populate the parameters. This is usually created where waterbodies cover the soil map and no information is available about the bed soil.

### *2.3. SWAT model setup*

ArcSWAT version 2012.10._5.21 was used to set up the SWAT model using a 30 m DEM, NLCD land cover, and STATSGO soil layer. **Table 1** shows the different land cover, soil types, and slope and their percentage within the UMLR watershed. Different combinations of land use, soil type, and slope produce unique hydrological response units (HRUs). A total of 1439 HRUs and 37 subbasins were generated (**Fig. 5**).

Daily rainfall and temperature data from 1981 to 2019 were obtained from the PRISM model (Parameter-elevation Relationships on Independent Slopes Model) (Manatsa et al., 2008) and downloaded through the PRISM Explorer (PRISM Climate Group, 2004). Although the water

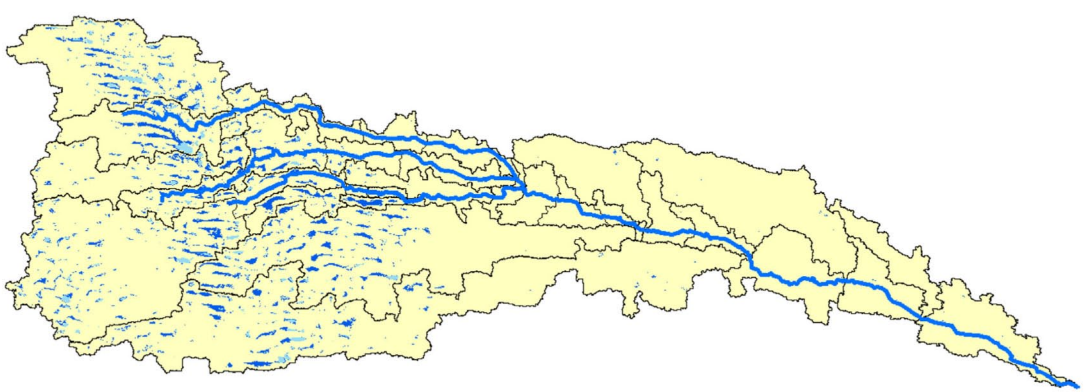


**Fig. 5.** Upper Middle Loup River watershed with the 37 SWAT model subbasins and the ponds, wetlands and streams.

**Table 2** Pond and wetland SWAT model parameters for Upper Middle Loup River watershed.

| *Pond parameters* | *Parameters definition* | *Value* |
|---|---|---|
| PND_FR | Fraction of subbasin area that drains into ponds | Varies |
| PND_PSA | Ponds surface area when filled to principal spillway (ha) | Varies |
| PND_PVOL | Water Vol. stored when filled to principal spillway ($m^3$) | Varies |
| PND_ESA | Ponds surface area when filled to emergency spillway (ha) | Varies |
| PND_EVOL | Water Vol. stored when filled to emergency spillway ($m^3$) | Varies |
| PND_VOL | Initial water Vol. in ponds ($m^3$) | Varies |
| PND_K | Hydraulic conductivity of ponds bottom (mm/h) | 0.5 |
| PNDEVCOEFF | Ponds evaporation coefficient | 0.1 |
| *Wetland parameters* | *Parameter definition* | *Value* |
| WET_FR | Fraction of subbasin area that drains into wetlands | Varies |
| WET_NSA | Wetlands surface area when filled to normal water level (ha) | Varies |
| WET_NVOL | Water Vol. stored when filled to normal water level ($m^3$) | Varies |
| WET_MXSA | Wetlands surface area when filled to maximum water level (ha) | Varies |
| WET_MXVOL | Water Vol. stored when filled to maximum water level ($m^3$) | Varies |
| WET_VOL | Initial water Vol. in wetlands ($m^3$) | Varies |
| WET_K | Hydraulic conductivity of wetlands bottom (mm/h) | 0.5 |
| WETEVCOEFF | Wetlands evaporation coefficient | 0.1 |

bodies within the model (wetlands and ponds) represent only 5% of the total area, they are concentrated within the upstream subbasins and are discharge regions (Rossman et al., 2019). Therefore, special focus was given to populate the wetland and pond parameters within the SWAT model, as shown in **Table 2**. The values of the parameters were mainly related to the area of each pond/wetland, which was calculated from the NLCD land use map. The estimation of the depth and volume of the ponds/wetlands parameters were based on the method mentioned by Evenson et al. (2015, 2016, 2018) and Muhammad et al. (2019). Additionally, the actual stream was defined based on aerial images (U.S. Geological Survey, 2020) to ensure better representation.

The aquifer characteristics were evaluated, including the soil subsurface compositions, hydraulic conductivity, specific yield, lag time, and groundwater depths (Pettijohn and Chen, 1962; Rossman et al., 2014). To simulate the travel of water throughout the watershed, the lag time was applied to the SWAT model based on the values extracted from Rossman's groundwater model (Rossman et al., 2019). Lag times ranged from 39 to 4500 days. Additionally, the soil hydraulic conductivity and aquifer properties were modified based on USGS sub-soil map (Pettijohn and

**Table 3** Pasture (PAST) and Eastern Redcedar (FRST) parameters to simulate encroachment in the Upper Middle Loup River watershed.

| *Parameters* | *Parameters definition* | *PAST value* | *FRST value* |
|---|---|---|---|
| CANMX | Maximum canopy storage (mm $H_2O$) | 10 | 28 |
| Sol_K | Saturated hydraulic conductivity (mm/h) | 70 | 116 |
| Sol_ZMX | Maximum rooting depth of soil profile (mm) | 525 | 8024 |
| CN2 | Initial SCS runoff curve number for moisture condition II | 50 | 37 |
| BLAI | Maximum potential leaf area index | 4 | 5 |
| CHTMX | Maximum canopy height (m) | 0.5 | 10 |
| RDMX | Maximum root depth (m) | 2 | 3.5 |

Chen, 1962). For the conversion of pasture to redcedar (to simulate encroachment) different parameters were modified as shown in **Table 3** according to the literature review (Afinowicz et al., 2005; Ahl et al., 2008; Caterina, 2012).

### *2.4. SWAT model calibration*

Once the SWAT model was created, it was then calibrated for streamflow, evapotranspiration, and recharge. Most modeling studies only calibrate streamflow (Bailey et al., 2016; Dhami et al., 2018; Starks and Moriasi, 2017), and very few calibrate more than one hydrologic component (Jin and Jin, 2020). Thus, calibrating each component of the hydrologic cycle reduces the uncertainty in the results. Streamflow was calibrated and validated from 2000–2009 and 2010–2019, respectively. The daily streamflow data from the only USGS stream gauge (06776500), located at the watershed outlet, was downloaded for the period from January 2000 to December 2019. With the high baseflow index, the variability in streamflow was very low. The Q95/Q5 is only 1.41 with a Q95 of 16.3 $m^3s^{-1}$ and Q5 of 11.5 $m^3s^{-1}$ (Hobza and Schepers, 2018). The average streamflow was 13.4 $m^3s^{-1}$ with a daily maximum and minimum of 19.7 and 10.3 $m^3s^{-1}$, respectively.

Another reason for calibrating evapotranspiration and recharge is due to the high uncertainty in the measured streamflow measurements. The bed is sandy and highly mobile, thus developing a reliable rating curve is a challenge. From 2000 to 2021 there were 200 streamflow measurements taken by the USGS. Though the average streamflow from the rating curve was 15.3 $m^3s^{-1}$ compared to measured discharge of 14.4 $m^3s^{-1}$, the $R^2$ and NSE were 0.05 and −5.56, respectively. The average error was 15.3%. The gauge height ranged from 0.92 to 1.67m. The error for streamflow measurements from 0.92m to 1.25m was 8.8% compared to an average of 228% for streamflow with a gauge height exceeding 1.25 m. In summary, the daily stream flow measurements derived from the rating curve have high uncertainty during high flow events. This is supported by Harmel et al. (2006) where they calculate uncertainty in streamflow measurements. The error is a function of individual streamflow measurements, stage-discharge relationship, continuous stage measurement and effect of streambed condition. Our study site has a shifty channel and mobile, unstable bed, which have +/−20% and +/−10% uncertainty, respectively. For measurements taken in ideal conditions, uncertainty is +/−6% vs +/−20% for poor conditions. Using Eq. (1) from Harmel et al. (2006), the probable error for ideal and poor conditions was calculated as 23% and 30%, respectively.

$$PER = \sqrt{\sum_{i=1}^{n} E\frac{2}{1}} + E\frac{2}{2} \dots \pm E\frac{2}{n}$$

where PER=probable error range (+/−), n = number of potential error sources, $E_1$, $E_2$, … $E_n$ = uncertainty associated with each potential error source (+/−). $E_1$ is the individual streamflow measurement (+/−6% and +/20% uncertainty for ideal and poor conditions, respectively), $E_2$ is the stage-discharge relationship (+/−20% uncertainty for shifty channel) and $E_3$ is the effect of streambed condition (+/−10% uncertainty for mobile, unstable bed). We assumed the uncertainty associated with the continuous stage measurement was minimal.

Monthly evapotranspiration and annual recharge estimated by Szilagyi et al. (2011a,b) were used in model calibration. Szilagyi et al. (2011a,b) applied a Calibration-Free Evapotranspiration Mapping Technique (CREMAP) covering the entire state of Nebraska. It provided a monthly evapotranspiration estimation from January 2000 to December 2009. This was used to calibrate evapotranspiration, not only for the entire watershed but also for each land use, specifically pasture and redcedar. The average evapotranspiration from the CREMAP model for the study area was 482 mm. Average pasture and redcedar evapotranspiration was 479 mm and 511 mm, respectively. In the NSH, there is very little runoff, so that it is often assumed that precipitation minus evapotranspiration is equal to recharge. Szilagyi et al. (2011a,b) calculated the mean annual groundwater recharge across Nebraska using MODIS where the study utilizes 1-km 8-day composited MODIS surface temperature and basic atmospheric data. The annual average recharge for the UMLR watershed is approximately 47 mm $yr^{-1}$ compared to long-term mean recharge >140 mm per year estimated in eastern Nebraska (Szilagyi et al., 2011a,b). See Supplemental information for more details on evapotranspiration and recharge in the study area.

A combination of manual and autocalibration using SWATCUP was used. Modeling results were evaluated using Nash-Sutcliffe Efficiency (NSE), $R^2$, and percent bias (PBIAS). NSE (Nash and Sutcliffe, 1970), which ranges from $-\infty$ to 1, is often used to examine hydrological models' predictive power. The NSE equation is

$$NSE = 1 - \left[ \frac{\sum_{i=1}^{n} (Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n} (Y_i^{obs} - \overline{Y^{obs}})^2} \right]$$

where: $Y_i^{obs}$ is the $i$th observation for the constituent being evaluated; $Y_i^{sim}$ is the $i$th simulated for the constituent being evaluated; $\overline{Y^{obs}}$ is the mean of observed data for the constituent being evaluated, and $n$ is the total number of observations.

The coefficient of determination is a key output of regression analysis (Thomas and Tiemann, 2015). The $R^2$ equation is

$$r^2 = \frac{ESS}{TSS} = \frac{\textit{sum of squares explained by regression}}{\textit{total sum of squares}}$$

$$TSS = ESS + USS = \sum(\hat{y} - \bar{y})^2 + \sum(y - \hat{y})^2$$

where: $\bar{y}$ is the average value of the dependent variable; $y$ represents the observed values of the dependent variable, and $\hat{y}$ denotes the estimated value of $y$ for the given $x$ value.

Percent Bias (PBIAS) measures the average tendency of the simulated values whether they are larger or smaller than their observed counterparts. A PBIAS of 0.0 is optimal, where low-magnitude values indicate more accurate model simulations (Yapo et al., 1996). In hydrological models, PBIAS can help to examine the model tendency towards underestimation (positive values) or tendency towards overestimation (negative values) (Van Liew et al., 2005).

$$\text{PBIAS} = \frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right)}{\sum_{i=1}^{n}\left(Y_i^{obs}\right)} \times 100$$

where $Y_i^{obs}$ is the $i$th observation for the constituent being evaluated and $Y_i^{sim}$ is the $i$th simulated value of the constituent being evaluated.

### *2.5. Eastern Redcedar encroachment simulations*

The streamflow, evapotranspiration, and recharge simulated by the calibrated SWAT model constituted the baseline scenario. Each redcedar encroachment scenario (11.9%, 16.1%, 28%, 40.6%, 57.5% and 72.5%) was compared to the baseline. The encroachment percentages indicate the percentage of pasture that was converted to redcedar in the calibrated SWAT model. For details on how pasture hydrologic response units (HRUs) were converted to redcedar, see Supplemental information. Thus, the 100% encroachment scenario means that all HRUs that are classified as pasture are converted to redcedar, and the relevant parameters in the SWAT model (i.e., CN, Max depth root, Canopy Maximum Interception Capacity CANMX) were changed to reflect the redcedar parameters.

#### *2.5.1. Water quality analysis*

We assumed that streamflow in the Loup River near Genoa (USGS gauge 06792500) (**Fig. 6**) would mimic that of the UMLR watershed due to redcedar encroachment. Since it is unknown if the concentration would increase or remain the same with a reduction in streamflow, we calculated the modified concentration in the Loup River in two ways: 1) nitrate and atrazine concentration remains the same and 2)mass of nitrate and atrazine remains the same. This method was applied to each encroachment scenario. The streamflow (15-minute data) was selected when the water-quality sample was taken. The mass of nitrate and atrazine was then calculated near Duncan (USGS gauge 06774000) on the Platte River. Measured concentrations of nitrate and atrazine were available at gauge stations 06792500 and 06774000 for the years 2010 to 2014. Based on the baseline and encroachment scenarios, the concentration for nitrate and atrazine was estimated downstream of the confluence of the Loup and Platte Rivers to assess the impact of encroachment on water quality

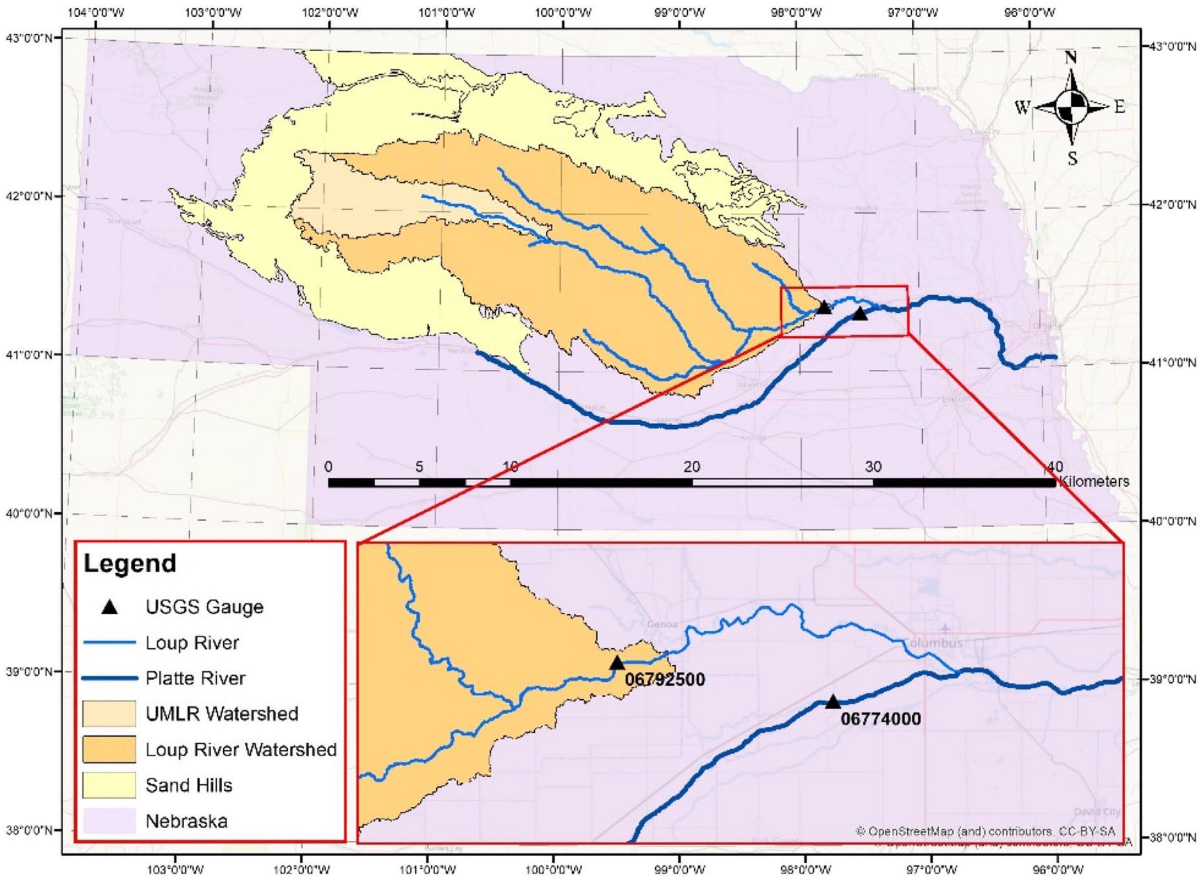



**Fig. 6.** Map illustrating the Loup River watershed, Upper Middle Loup River (UMLR) watershed, Sand Hills and the location of the USGS gauge stations on the Loup River (06792500) and Platte River (06774000). Gauge station 06792500 is located 52.9 km upstream of the confluence with the Platte River and the gauge station 06774000 is located 15.8 km upstream of the confluence with the Loup River.

in the Platte River. Due to the uncertainty in streamflow measurements, we used the upper and lower bounds of streamflow to calculate the atrazine and nitrate concentrations.

Additionally, a risk factor was calculated for nitrate and atrazine exposure for the Platte River for each encroachment scenario based on the method adopted by Hansen et al. (2019). This study identified the risk of nitrate and atrazine exposure individually (single risk factor, SRF) and together (dual risk factor, DRF) for nitrate and atrazine in several locations within Nebraska. The SRF values range from <0.8 safe; 0.8 to 1.0 Low Risk; 1.0 to 2.0 At Risk and >2.0 is considered High Risk. A study by

Rhoades et al. (2013) showed that exposure to nitrate (2 mg/L) and atrazine (3 μg/L) together increased the occurrences of Non-Hodgkin Lymphoma in Nebraska (DRF). The DRF ranged from: 0 = Very Low Risk; 1 = Low Risk; 2 = Medium-Low Risk; 3 = Medium Risk; 4 = Medium- High Risk; 5 = High Risk; and 6 = Very High Risk. For more details on the risk factor analysis, see Supplemental information.

## 3. Results and discussion

When presenting the results, it is important to do so with a clear understanding of the assumptions. The following assumptions were made in this analysis: 1) redcedar encroachment scenarios simulated are hypothetical and only represent what may potentially occur; 2) Trees classified as mixed forest in the NLCD were assumed to be ERC; 3) Impact of ERC encroachment on streamflow in the Loup River mimics the impact from the UMLR watershed; 4) Streamflow and atrazine and nitrate concentrations at the USGS gauge stations on the Loup River and Platte River are the same when they converge at the confluence; 5) Nitrate and atrazine concentrations from 2010 to 2014 are representative of their current and future concentrations.

### *3.1. SWAT model calibration and validation*

The calibrated model was compared to monthly-observed streamflow and evapotranspiration and average annual recharge. **Table 4** lists the parameters used in the SWAT calibration, their default range, and calibrated values. The calibrated values apply to every HRU while the calibrated values vary for lateral travel time (LAT_TTIME.hru) curve number (CN.mgt), and saturated hydraulic conductivity (Sol_K.sol). Lateral travel time ranged from 39 to 4500 days, curve number from 31 to 92 and saturated hydraulic conductivity from 2.9 to 450 mm $hr^{-1}$ (**Fig. 7**). It was found that streamflow was most sensitive to the groundwater- (i.e., GW_Delay) and HRU-related parameters (i.e., LAT_TTIME and CANMX). For ET, GW_REVAP and soil-related parameters (i.e., Sol_AWC, Sol_K) were the most influential. Autocalibration was initially conducted before applying different manual calibration simulations using the mentioned parameters.

**Table 4** SWAT model calibration parameters, default ranges and the calibrated values for the Upper Middle Loup River watershed.

| *SWAT parameters* | *Default range* | *Calibrated value* | *Parameter definition* |
|---|---|---|---|
| GW_REVAP.gw | 0.02–10 | 8 | Groundwater "revap" coefficient. |
| GW_DELAY.gw | 0–500 days | 31 | Groundwater delay time (days). |
| ESCO.hru | 0–1 | 0 | Soil evaporation compensation factor. |
| EPCO.hru | 0–1 | 0.1 | Plant uptake compensation factor. |
| CN.mgt | Varies | Varies | Initial SCS runoff curve number for moisture condition II |
| LAT_TTIME.hru | 40–4500 | Varies | Lateral flow travel time (days) |
| CANMX.hru | 0–28 | 0, 10, 28 | Maximum canopy storage (mm $H_2O$) |
| Sol_AWC.sol | 0–1 | 0.22 | Available water capacity of the soil layer (mm $H_2O$/mm soil) |
| Sol_K.sol | 2–450 | Varies | Saturated hydraulic conductivity (mm/h) |
| PND_FR.pnd | 0–1 | 0.1 | Fraction of subbasin area that drains into ponds. |
| WET_FR.pnd | 0–1 | 0.1 | Fraction of subbasin area that drain into wetlands. |
| PLAPS.sub | −500–500 | 3.2 | Precipitation lapse rate (mm $H_2O$/km). |
| TLAPS.sub | −50–50 | 0 | Temperature lapse rate (°C/km). |

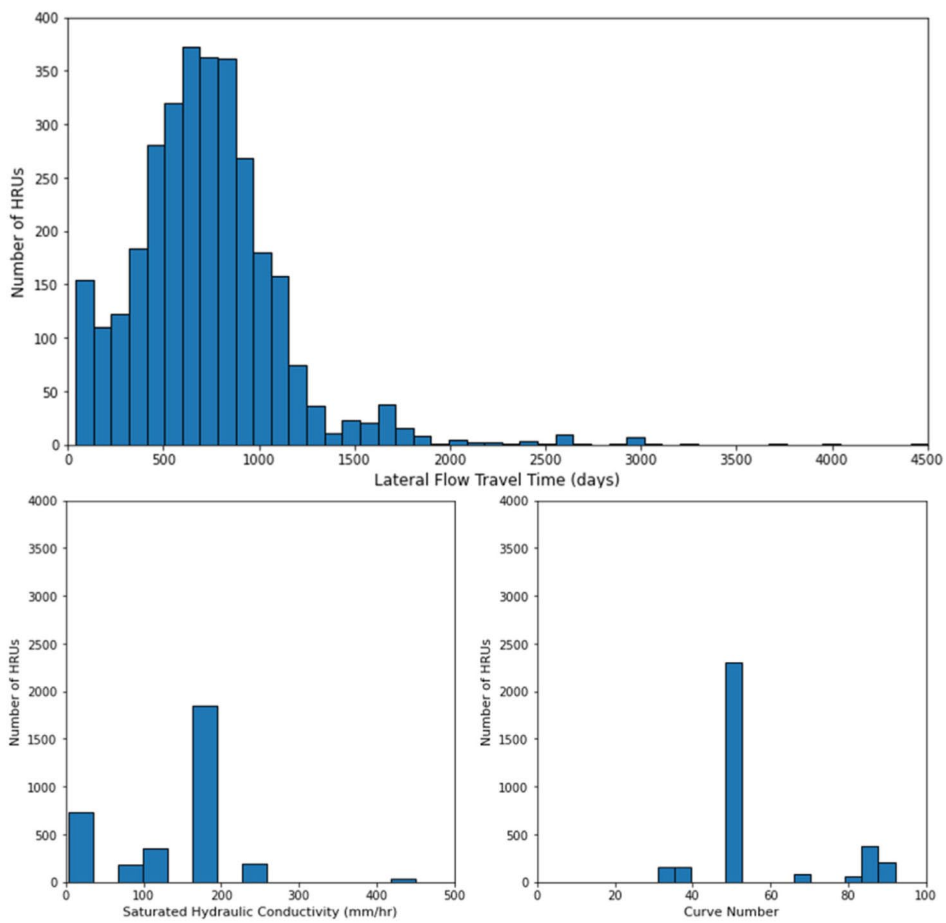



**Fig. 7.** Histograms illustrating the number of HRUs for the various values for lateral travel time, saturated hydraulic conductivity and curve number.

The simulated streamflow was close to the measured streamflow pattern over the calibration (2000–2009) and validation periods (2010–2019) (S1). For the calibration period, the $R^2$, NSE, and PBIAS were 0.01, 0.05, and 2.6%, respectively. For the validation period the $R^2$, NSE, and PBIAS were 0.03, −1.09, and 4.8%, respectively. Though the NSE had an unsatisfactory rating, the PBIAS was rated as Very Good based on Moriasi et al. (2007). The simulated streamflow for both calibration and validation periods were close to the observed average. For the calibration period, the average simulated streamflow was 12.1 $m^3s^{-1}$ compared to the observed 13.2 $m^3s^{-1}$. For the validation period, the simulated streamflow was 16.3 $m^3s^{-1}$ compared to 14.4 $m^3s^{-1}$ for the observed streamflow. Additionally, the model had a 0.95 baseflow percentage which approaches the recorded baseflow near the UMLR outlet based on USGS Science Data Catalog (SDC). Hobza and Schepers (2018) stated that the streamflow out of the NSH, groundwater discharges (defined as baseflow) ranges from 0.8 to 0.95. In the only other study where SWAT was used in the NSH, modeling statistics were also poor, with NSE of −1.13, and PBIAS of 21.1 (Strauch and Linard, 2009). We hypothesize that this is due to the high level of uncertainty of the rating curves derived in the sandy streams of the NSH, especially at high flow events. The NSE from our model was actually higher than the NSE when comparing the streamflow based on the rating curve (observed) to the measured streamflow.

The simulated monthly evapotranspiration for the entire watershed (2000–2009) was 483 mm compared to the CREMAP model average of 482 mm (~99%). The NSE and PBIAS metrics were 0.63 (Satisfactory) and −0.51 (Very Good), respectively. Additionally, evapotranspiration model results for pasture and redcedar were similar to the observed values with NSE, $R^2$, and PBIAS of 0.64, 0.65, and 0.82% for pasture and 0.45, 0.46, and 0.01% for redcedar, respectively. Szilagyi et al. (2011a,b) and Rossman et al. (2014) estimated an average recharge across the UMLR watershed of 47 and 56 mm $yr^{-1}$ from 2000 to 2009, respectively. The calibrated SWAT model simulated an average of 54.2 mm $yr^{-1}$ thus showing good agreement. With our model simulations yielding streamflow, evapotranspiration, and recharge values comparable to measured values, our encroachment scenario results will thus have less uncertainty. Overall, the average streamflow, evapotranspiration and recharge simulated values match quite well with the observed values. Even though the modeled streamflow had a low NSE, we believe the model simulates the overall hydrology satisfactorily.

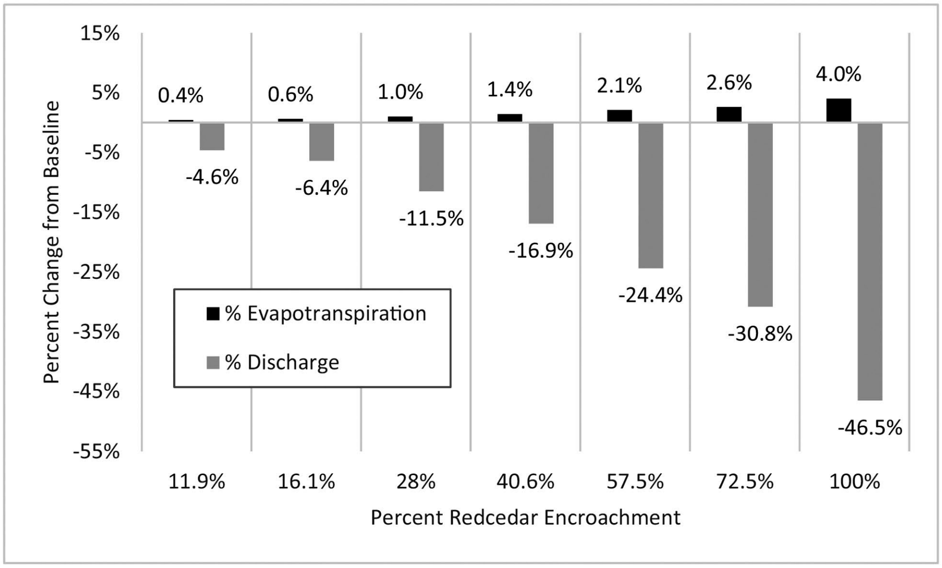



**Fig. 8.** Changes in streamflow and evapotranspiration for various levels of redcedar encroachment in the Upper Middle Loup River watershed. Each encroachment scenario was compared to the baseline scenario with no encroachment (<1% redcedar coverage).

### 3.2. Encroachment scenarios

Different encroachment scenarios were simulated using the calibrated SWAT model. The scenarios were conducted by changing the pasture classes in the modified land use into redcedar. The general trend indicates that as the encroachment percentage increases, average streamflow decreases (**Fig. 8**). An increase of redcedar to 11.9% yielded a reduction in streamflow from 12.1 to 11.5 $m^3s^{-1}$. In addition, the fact that the watershed is baseflow-dominated (i.e., >90% of the lateral flow is baseflow) makes the impacts of interception from redcedar roots significant. Thus, any increase of redcedar reduces lateral flow (feeding the streams). As encroachment continues to 41% and 100%, discharge to the UMLR decreases by 24.4% and 46.5%, respectively.

As shown in **Table 5**, different hydrological components were evaluated for each of the simulated encroachment scenarios. The evapotranspiration increases as the encroachment percentage increases. 11.9% encroachment resulted in a 0.43% increase in evapotranspiration compared to the baseline, while a 100% encroachment resulted in a 4% increase. On the contrary, the streamflow decreased as the encroachment

**Table 5** Water balance components from each encroachment scenario compared to the baseline scenario. Units are all in mm. Example of storage would be water stored in ponds. Adding up the sum of each parameter yields the amount of precipitation.

| | *Baseline* | *11.9%* | *16.1%* | *28%* | *40.6%* | *57.5%* | *72.5%* | *100%* |
|---|---|---|---|---|---|---|---|---|
| Precipitation | 659.3 | 659.3 | 659.3 | 659.3 | 659.3 | 659.3 | 659.3 | 659.3 |
| Runoff | 3.7 | 3.4 | 3.3 | 3.1 | 2.9 | 2.7 | 2.5 | 2.0 |
| Percolation | 75.6 | 77.4 | 78.2 | 80.5 | 83.1 | 86.5 | 89.5 | 95.7 |
| Evapotranspiration | 483.2 | 485.3 | 486.0 | 488.0 | 490.0 | 493.1 | 495.6 | 502.7 |
| Deep recharge | 3.8 | 3.9 | 3.9 | 4.0 | 4.2 | 4.3 | 4.5 | 4.8 |
| Lateral flow | 67.6 | 64.7 | 63.5 | 60.1 | 56.4 | 51.3 | 47.0 | 36.4 |
| Storage | 25.4 | 24.6 | 24.4 | 23.6 | 22.7 | 21.4 | 20.2 | 17.7 |

increased with a −4.59% reduction for the first 11.9% of encroachment and reducing to −46.5% with 100% encroachment. Zou et al. (2018) stated that a complete conversion of rangeland to redcedar (i.e., 100%) encroachment would lead to streamflow(i.e., discharge) reductions of 20–40%. Similarly, while the evapotranspiration increased (with the increase of encroachment), the percolation increased from 75.6 mm to 95.7 mm at full encroachment (Table 5). Though several studies have shown that recharge decreases from redcedar encroachment, others have shown increases. This can be seen from the results in our study where the ranges of Sol_K are from 2.91 to 450 mm $hr^{-1}$ with the majority of HRUs (2202 out of total 3342 HRUs) having a range of Sol_K from 116 to 180 mm $hr^{-1}$, where an decrease in lateral flow was accompanied by an increase in both percolation and deep recharge.

### *3.3. Impact of redcedar encroachment on the Platte River*

Though <1% of the Loup River watershed is currently redcedar, the rate of encroachment has increased recently (Lower Loup NRD, 2017). The land use in the Loup River watershed is similar to the UMLR watershed, though there is less pasture in the Loup River watershed (80% vs 94%) and more cropland (15% vs <1%). Assuming the impact on streamflow of redcedar encroachment would be similar in the Loup River watershed as the UMLR watershed, we evaluated the potential impact of redcedar encroachment on the entire Loup River watershed and its impact on the Platte River streamflow and water quality based on the results obtained from the UMLR watershed encroachment scenarios.

The average streamflow in the Middle Loup and Loup Rivers from 2000 to 2019 was 13.8 and 47.2 $m^3s^{-1}$ at Dunning and Genoa, respectively. The streamflow on the Platte River was 49.8 $m^3s^{-1}$ at Duncan (upstream of confluence) for the same period. Therefore, nearly 50% of the water in the Platte River downstream of the confluence is from the Loup River. In 2004 and 2006 (dry years), 79% of the water at the confluence was from the Loup River. During the drier months, August to November, 61% of the water at the confluence originated from the Loup River from 2000 to 2019. Any impact on the streamflow from the redcedar encroachment will not only influence the streamflow in the Loup River, but also in the Platte River. For example, the discharge at Genoa would decrease from 47.2 to 41.8 $m^3s^{-1}$, 35.7 $m^3s^{-1}$, and 25.3 $m^3s^{-1}$ for 28%, 57.5%, and 100% encroachment, respectively.

This reduction in streamflow to the Platte River will also impact water quality, a concept noted by Zou et al. (2015). The assessment of streamflow at USGS gauges, Genoa and Duncan, is important to estimate the potential changes in water quality on the Platte River. The average nitrate concentrations were 0.55 mg/L and 1.40 mg/L while the atrazine concentrations were 0.60 μg/L and 1.42 μg/L at Genoa (Loup River) and Duncan (Platte River) stations, respectively. The concentration of atrazine and nitrate on the Platte River at the confluence of the Loup River ranges from 1.20–1.25 μg/L and 0.89–0.94 mg/L when accounting for the uncertainty in streamflow. The increase in atrazine and nitrate concentrations is minimal for 11.9% encroachment, but significant for 100% encroachment. Accounting for the reduced streamflow to the Platte River, the concentration of atrazine would increase by 4–30%. This range accounts for uncertainty in streamflow and whether the concentration of atrazine or its mass remains the same for the Loup River. The concentrations would increase to 1.30 to 1.56 μg/L. The nitrate in the Platte River would increase by 4–15% with concentrations ranging from 0.98 to 1.02 mg/L.

For the baseline scenario (no encroachment) the single risk factor (SRF) for nitrate is considered Safe for each year from 2010 to 2014 on a scale from 0 (Considered Safe) to 3 (High Risk). Atrazine is considered Safe for each year except for 2013 where it is considered High Risk. At 100% encroachment, the SRF for nitrate remains Safe for each year, but for atrazine the SRF increases to 1 (Low Risk) for 2009 and 2010, 2 (Risk) for 2012 and remains 3 (High Risk) for 2013. For the DRF, each year is considered Very Low for the baseline except for 2013 where it is considered Medium. At 100% encroachment, the risk factor increases from Very Low to Low for both 2010 and 2012. Though the risk factor doesn't change much, there is a clear increase in nitrate and atrazine concentrations from redcedar encroachment. These predictions support the hypotheses that an increase in redcedar encroachment in the Loup River watershed will impact both water quantity and quality in the Platte River. While our modeled results have substantial uncertainties, they highlight the general concept that the amount of water discharging from NSH streams is relevant to cities of Lincoln and Omaha in terms of water security and water resources management plans.

Future work includes the coupling of SWAT with MODFLOW. The SWAT model is limited in its ability to simulate groundwater processes. Using SWAT-MODFLOW in this baseflow-dominated system will improve modeling results. This coupling of the surface and the groundwater model can help improve the calibration of the model, thus improve the estimation of streamflow. Coupling with MODFLOW will also provide the change in the water table as redcedar encroachment increases. The reduction in the water table will impact the thousands of lakes and wetlands, as well as the ecosystem, in the NSH. Future work should also include climate change. This would include not only temperature and precipitation, but also $CO_2$ concentrations. The encroachment rate will certainly be affected by each climate variable and whether the variables increase or decrease and by how much.

Another aspect of future improvement would be more accurate redcedar encroachment predictions by applying machine learning or some ecological model to better predict the spatial and temporal spread of ERC. Incorporating climate change models will not only impact the water resources directly but will influence the growth rate of ERC based on changes in temperature, precipitation and $CO_2$.

## 4. Conclusions

Grasslands worldwide are shifting to woody-plant dominance. This woody plant encroachment is causing major changes to multiple ecosystem services. The cause of the woody-plant expansion into the world's grasslands is from global increases in $CO_2$, shifts in temperature and precipitation patterns, displacement of diverse and free-ranging native herbivore species with domesticated livestock, increasing seed dispersal through human transportation and tree plantings, and global departures in anthropogenically-driven fire regimes. In North America, the Eastern Redcedar has encroached into grasslands due to the reduction in wildfires and their planting for windbreaks. This study focuses on redcedar encroachment in the Nebraska Sandhills.

In this study, the SWAT model was utilized to examine the impacts of redcedar encroachment on the water resources in the Nebraska Sand Hills, a major recharge zone for the High Plains Aquifer, and the Platte River, a major water source for the cities of Omaha and Lincoln. Based on the results, we accept our hypothesis that as redcedar encroachment increases, streamflow will decrease, evapotranspiration will increase and nitrate and atrazine concentrations in the Platte River will increase. However, we reject our hypothesis that recharge will also decrease. The scenarios ranged from 11.9% to 100% encroachment. The results showed a reduction in streamflow as the encroachment increased. With full encroachment, the flow in the Upper Middle Loup River was reduced by nearly half (53% of the original flow), and the evapotranspiration increased 4.04% from baseline. Assuming the same reduction in streamflow for the Loup River watershed, a major tributary to the Platte River, the streamflow in the Platte River will decrease from by 5.6% and 22.6% for 28% and 100% encroachment, respectively.

In addition, we evaluated the impacts of the Redcedar encroachment on water quality in the Loup and Platte Rivers. The range of concentrations accounts for the uncertainty in streamflow and whether the nitrate/atrazine concentration or mass remains in the same in the Loup River. The reduction in streamflow could increase nitrate concentrations in the Platte River from 4 to 15% and atrazine concentrations would increase from 4 to 30% at 100% encroachment.

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**CRediT authorship contribution statement** — Yaser Kishawi: Methodology, Validation, Formal analysis, Investigation, Writing – original draft. Aaron R. Mittelstet: Conceptualization, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition. Troy E. Gilmore: Conceptualization, Writing – review & editing. Dirac Twidwell: Conceptualization, Writing – review & editing, Funding acquisition. Tirthankar Roy: Writing – review & editing. Nawaraj Shrestha: Methodology, Formal analysis, Investigation.

**Data availability** Data will be made available on request.

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**Supplementary data** Supplementary data follows the **References**.

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**Hydrological Response Unit (HRU) Land Use Change**

Due to the conceptual setup of the SWAT model, HRUs are a non-spatial feature. This means that any HRU with its unique features (i.e., slope, land use, soil type) can be anywhere within the subbasin and cannot be assigned to a unique location. Thus, it is impossible to reflect the generated encroachment maps from satellite images to the current model. This limits the opportunity to target a specific HRU without changing every HRU that holds the same features. To overcome the limitation, a modification to the HRU files with pasture was performed. By segmenting the original pasture (i.e., HRUs with pasture) into several classes and changing only the name of the land use, without changing other parameters, creates an additional number of HRUs. The segmentation was conducted by overlaying encroachment scenarios, generated using satellite image analysis, and slicing the original shapefile that holds the pasture into additional shapefiles, and naming the new classes PA10, PA15, PA25, PA40, PA55 and PA70. The new names represent the same encroachment scenarios 11.9%, 16.1%, 28%, 40.6%, 57.5% and 72.5% respectively as shown in Figure 4. This segmentation of pasture resulted in an increased number of associated HRU files from 1493 to 3342. The encroachment scenarios were then simulated by selecting the related pasture shapefile (and corresponding HRU files) based on the desired percentage of encroachment then changing the relevant parameters in the selected HRUs files.

**Water Quality Analysis**

Additionally, a risk factor was calculated for nitrate and atrazine exposure for the Platte River for each encroachment scenario based on the method adopted by Hansen et al. (2019). This study identified the risk of nitrate and atrazine exposure individually (single risk factor, SRF) and together (dual risk factor, DRF) for nitrate and atrazine in several locations within Nebraska. The SRF was determined based on sets of measured data and maximum contamination level (MCL) at 10 mg $L^{-1}$ for nitrate and 3 µg $L^{-1}$ for atrazine (Rhoades et al. 2013). The SRF analysis is:

$$RF_{MCL}^{95th\%} = \frac{95perc(MEC_i)}{MCL}$$

were $RF_{MCL}^{95th\%}$ is the risk factor for the contaminant based on the 95$^{th}$ percentile of the MEC and MCL, $MEC_i$ is the measured environmental concentration at time i (based on yearly step according to the collected data), MCL is the maximum contaminant level according to Environmental Protection Agency acceptable concentrations for drinking water. were $RF_{MCL}^{95th\%}$values range from < 0.8 safe; 0.8 to 1.0 Low Risk; 1.0 to 2.0 At Risk and > 2.0 is considered High Risk. A study by Rhoades et al.(2013) showed that exposure to nitrate (2 mg $L^{-1}$) and atrazine (3 µg $L^{-1}$) together increased the occurrences of Non-Hodgkin Lymphoma in Nebraska (DRF). The DRF ranged from*: 0 = Very Low Risk; 1 = Low Risk; 2 = Medium-Low Risk; 3 = Medium Risk; 4 = Medium-High Risk; 5 = High Risk; and 6 = Very High Risk.*

**Evapotranspiration and Recharge**

Billesbach and Arkebauer (2012) performed a long-term direct measurement of evapotranspiration and surface water balance in NSH from 2003-2009. This study found that the three ecosystems in the NSH (sub-irrigated meadow, dry valley, and uplands) behaved in a different way. The annual ET for sub-irrigated meadow, dry valley, and uplands were 735 mm, 462 mm, and 280 mm, respectively. However, these measurements could only be considered point measurements compared to the NSH scale. The monthly evapotranspiration estimates resulted in an $R^2$ of 0.8 – 0.9, while the annual estimates had $R^2$ between 0.7-0.8. The mean annual evapotranspiration estimated remained within 10% of the measured values.

Another study by Rossman et al. (2014) evaluated how vadose zone lag time effects groundwater recharge in the NSH. They found that the recharge ranged from -204 mm (discharge in some areas where the lakes are clustered towards the western side of UML) to 143 mm (mainly towards the eastern and southern parts of the UMLR watershed). Gilmore et al. (2019) used regional water table patterns to estimate the recharge rates in shallow aquifers. In the NSH, the mean annual recharge rates vary depending on soil type, land cover, climatic situation, and slope. The eastern part of the NSH has higher recharge rates (100-276 mm $yr^{-1}$) compared to the western parts, where recharge rates (0-60 mm $yr^{-1}$) are lower, or even discharge (0 to -386 mm $yr^{-1}$) can occur. For the UMLR watershed, the main discharge area is in the western part of the watershed, where lakes and wetlands are concentrated.

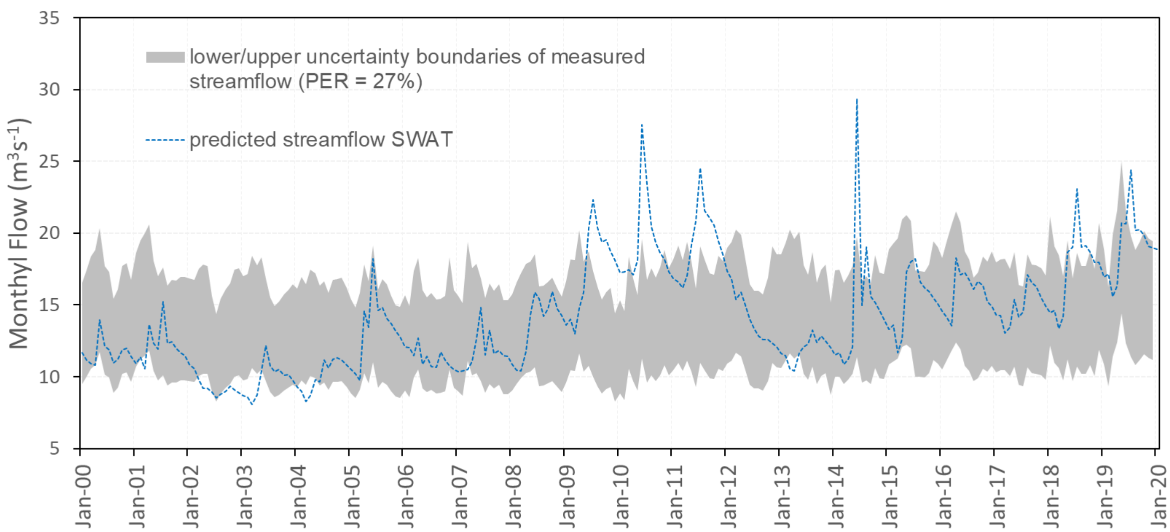


Figure S1: SWAT simulated vs observed discharge for the Upper Middle Loup watershed for the calibration (2000-2009) and validation periods (2010-2020).

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