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Cover photos: Top: *East Carolina University undergraduate engineering student, Deja Drummond, collects water quality data in the Tar-Pamlico river basin,* Credit: Stevon Creque. **Bottom:** *Undergraduate students (from right, Japhet, Elysee) and graduate student (left, Yvon) collecting plants' destructive samples from a field in Sutherland, Nebraska,* Credit: Dr. Arindam Malakar.
Back cover photo: *Inside Busch Stadium - St. Louis, MO*, Credit: Jackie Gillespie

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Letter from the Special Issue Editor

This journal, *Journal of Contemporary Water Research and Education*, has a vested interest in research, education and extension related to water resources science, engineering, management, and policy. Within this broader interest, the journal also puts value on publishing manuscripts from undergraduate research projects, class projects, and honors thesis with a focus on water research and education. The challenge with these types of manuscripts is the focus and depth should be narrow, time and resources are often limited (e.g., one semester or the equivalent of a 3 student credit hours), and peer-reviewed publication is often not the common standard nor goal for undergraduates (Fenn et al. 2010). The latter might be the most pressing challenge, unless the undergraduates have an interest in pursuing advanced degrees, particularly graduate school where the students will be doing research for theses and dissertations. Undergraduates need opportunities like this to frame their water research within broader scientific literature (Fox et al. 2017), especially if the next step in their career is graduate school.

The faculty advisors, mentors, and teachers also play an important role in pushing these types of undergraduate manuscripts forward, while maintaining the undergraduates as the lead author. Faculty can look at publishing undergraduate research with students as the lead authors as an innovative educational opportunity in water resources (Habib and Deshotel 2018), explaining how coauthors are selected and ordered, how to craft a readable story in science (e.g., see Mackay 1995), how you handle the peer-review process, and how to write acknowledgements. However, faculty including myself often fall into the trap of writing our stories in a complex nature, which limits the manuscript's ability to always tell a readable story.

The *Journal of Contemporary Water Research and Education* has provided and continues to provide undergraduates and faculty mentors with an opportunity to publish these manuscripts, while navigating the peer-review process. I personally find this a valuable contribution to scientific literature, as well as valuable educational opportunities in water resources. These undergraduates are part of the future workforce which will be tackling the pressing water problems and issues that we face locally, regionally and across our Nation. I hope you enjoy this Special Issue on undergraduate research.

Cheers,

Brian E. Haggard
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Effects of Rock Covering on Underlying Engineered Media in Bioretention Practices in Middle Tennessee, USA

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Abstract: Bioretention practices have become a common way to protect natural waterways in urban and suburban landscapes across the United States. However, optimal design, implementation, operation, and maintenance are still in need of study. A field survey of 52 bioretention practices was conducted in Davidson County, Tennessee, to address research questions related to operation and maintenance. A suite of site conditions were documented, such as size, signs of erosion, and dominant surface cover. Samples were collected from the surface of the engineered media layer and analyzed for organic matter content and bulk density. Vegetation was described in terms of dominant species and canopy cover. On average, the organic matter content of media under plant-based mulch cover was significantly greater than that under rock cover ($p = 0.002$). Bulk density of the surface media is strongly and inversely correlated to organic matter content; bulk density did not generally vary with bioretention area age and was highly variable within treatments. On average, the bulk density of the media under the plant-based mulch cover was significantly less than that under the rock cover. Media under the composite treatments had similar bulk density to both the plant-based mulch ($p = 0.233$) and the rock covers ($p = 0.132$). Plant canopy did not surpass 70% in practices with bulk density values above 1.55 g/cm^3. These results suggest that consideration should be made regarding the tradeoffs between utilizing rock coverings and potential for plant establishment impacts.

Keywords: *bioretention, urban water, runoff, green stormwater infrastructure, engineered media*

Urbanization plays a significant role in the loss and degradation of inland water systems in the United States (O'Driscoll et al. 2010) and across the globe (Millennium Ecosystem Assessment 2005). To combat threats posed to surface waterbodies, bioretention has been widely adopted as a form of green stormwater infrastructure (GSI) to manage the quantity and quality of urban stormwater runoff discharged to streams, creeks, rivers, and wetlands (Davis et al. 2009). Bioretention is a method of stormwater management using native plantings and soil conditioning (Coffman et al. 1994). Performance requirements for bioretention practices are commonly described in terms of capture volume, percolation and/or infiltration rates, and pollutant removal capacity. The design and operation of bioretention practices vary based on location-specific performance requirements. Functional processes at work in bioretention include hydraulic mixing, physical settling and straining, chemical adsorption and transformations, and biological uptake and conversion (Davis et al. 2009). Characteristics affecting these processes include, but are not limited to, size and contributing drainage area (Yang and Chui 2018), underlying soil characteristics (Davis et al. 2012), vegetation establishment (Muerdter et al. 2015; Dagenais et al. 2018), saturation and redox potential (Deitz and Clausen 2006), and local conditions like salting and climate (Soberg et al. 2017).

The integration of ecological, physical, chemical, and biological functions of soil, plants, and microorganisms has long been recognized

Research Implications

- Surface covering material selection in bioretention applications affects underlying media characteristics that are linked with performance.
- Organic matter content was greater under plant-based mulch covering than under rock coverings which may have implications for overall bioretention water quality function.
- Promoting vegetation health by not using rock surface coverings may result in better bioretention function.

as fundamental to bioretention function (Roy-Poirier et al. 2010). There is a growing body of knowledge shedding light on the interactions of engineered media, plants, and microbes in bioretention that impacts the physico-chemical properties of these systems (Skorobogatov et al. 2020). A study by Lucas and Greenway (2008) showed that the presence of vegetation improved nutrient removal as compared to no vegetation in bioretention mesocosms. Vijayaraghaven et al. (2021) used a bibliometric analysis to evaluate the specific role of bioretention components to outline desirable vegetation and media characteristics, and concluded that the performance of bioretention is yet to be fully optimized. There exists a need to better understand the interactions between design components and the potential impact of implementation decisions on bioretention function.

As the application of bioretention-based GSI matures, many design variations and adaptations have been deployed and evaluated at field scale in response to performance needs or operational concerns. Such adaptations include creating an internal water storage layer to enhance denitrification (Dougherty et al. 2007), nesting the practice within the footprint of a retention pond to address water quantity and quality issues and reduce overall infrastructure footprint (Chin 2017), using internal baffles to maximize mixing (Donaghue et al. 2022), using engineered media amendments like biochar and fungi to enhance pollutant removal (Mitchell et al. 2023), managing active storage with sensor-based controls (Persaud et al. 2019), and utilizing a reduced diversity or volunteer plant palette to help with vegetation maintenance while not hindering performance (Dagenais et al. 2018).

The use of a stone or river rock surface covering in place of conventional plant-based mulch is an example of a modification being implemented more commonly in Middle Tennessee and across the country. Metro Water Services Nashville-Davidson County (Metro) operates an Individual National Pollutant Discharge Elimination System (NPDES) permit to manage the separate storm sewer system (MS4) in Davidson County, Tennessee, USA, the county containing the fast-growing Nashville metropolitan area. Metro was an early adopter of green infrastructure technologies in Tennessee. Therefore, many other Tennessee MS4s look to Metro to provide a model based on the relatively long period of observation of practice performance. As the number of bioretention practices in Davidson County grew to over a thousand practices, rock surface covering was the most used surface cover in bioretention practices. The perceived advantages of rock covering over plant-based mulch include less washout during storms, ease of maintenance (less weed pressure), and preference in aesthetic appeal. However, the practice of using rock covering raises questions about the potential for impact to overall function of bioretention cells in terms of infiltrating water, filtering pollutants, and supporting the designed plant community.

In collaboration with municipal professionals at Metro, the research team conducted a field study with a goal to evaluate the impacts of rock surface covering on bioretention function. Bioretention function is the capturing, infiltrating, and filtering of pollutants from urban stormwater runoff, and porous soils and healthy vegetation are critical to these functions. Specific research questions for this study included: 1) Does surface cover affect media bulk density? 2) Does plant-based cover generate more organic matter than rock cover? 3) Does media bulk density affect plant canopy establishment? 4) What plant species are most observed? and 5) What conclusions can be drawn that may inform operation and maintenance activities to address common failures?

Methods

Study Site Selection

Davidson County, Tennessee, USA, lies in the Inner Basin ecoregion in the Cumberland River Basin in North-Central Tennessee. Fifty-two sites, out of the over one thousand practices, were selected in the operating area of Metro Water Services Nashville-Davidson County (Figure 1), capturing geographic variability throughout the service area with different surface covers (rock, organic, composite) and across a range of practice size (from 20 to 1,660 m^2) (Table 1). Practices were installed within the timeframe of 2009 to 2016, in either a commercial or residential land use setting. All practices were subject to the local requirement of using engineered media consisting of 70-85% sand, 10-30% silt plus clay, and 5-10% organic matter (by volume) (Metropolitan 2021).

Design documents were shared by Metro to the project team, describing each practice in terms of size, components, and placement in the larger development/landscape setting. These design documents were used to record pertinent design components, such as ponding depth, vegetation species (if specified), and presence of underdrain.

Field Methods

Each site was visited once during the summer of 2018 during dry weather conditions (not actively raining and no surface ponding). Measured site characteristics included size and dimension of depression, ponding depth, and thickness of mulch layer. Observations were recorded of the

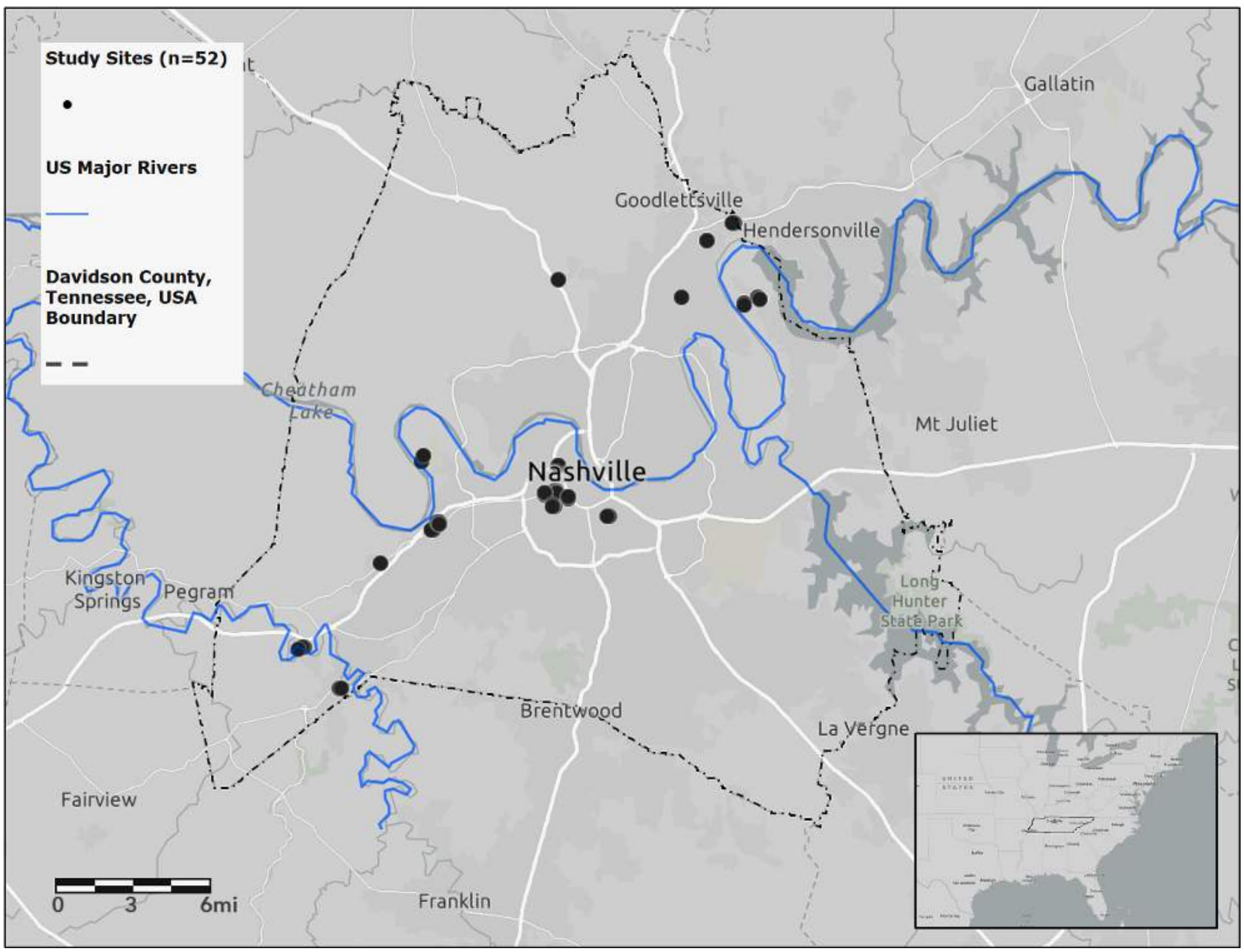

Figure 1. Map of study sites (n = 52) in Davidson County, Tennessee, USA. Note that some sites were co-located as separate bioretention cells at the same general location.

Table 1. Study site information including location, age, and pertinent characteristics.

Site ID	Latitude	Longitude	Sizes (m^2)	Year Built	Surface Cover
DG-1	36.11724	86.92367	367.1	2013	Composite
HC-2,3,4	36.13586	86.88746	89.7, 51.1, 166.9	2013	Composite
MN-1,6	36.15779	86.79991	84.4, 67.7	2010	Composite
SC-2,3	36.15662	86.80822	67.4, 65.6	2014	Composite
VD-1,3	36.14927	86.80086	259.4, 20.8	2014	Composite
AT-1,2,3	36.30008	86.69405	154.6, 62.1, 135.8	2014	Plant-based
BB-1,2	36.17526	86.89431	59.3, 119.4	2013	Plant-based
CB-1	36.17833	86.89227	141.2	2015	Plant-based
FW-1,2,3	36.26817	86.65768	1659.7, 933.4, 416.8	2014	Plant-based
HG-5	36.14013	86.88145	661.7	2009	Plant-based
MC-1,2	36.30956	86.67380	238.6, 665.8	2014	Plant-based
MC-2	36.31052	86.67556	665.8	2014	Plant-based
MN-2,3,4,5	36.15779	86.80014	92.6, 50.5, 46.1, 39.5	2010	Plant-based
MTA-1	36.27754	86.79753	159.4	2012	Plant-based
OH-1,2	36.26435	86.66808	136.8, 182.1	2012	Plant-based
RB-1,2,3	36.06929	86.97597	53.4, 53.1, 46.5	2013	Plant-based
SC-1,4,5	36.15675	86.80824	129.0, 254.7, 78.8	2014	Plant-based
SS-1,2	36.17236	86.79832	6.8, 6.8	2014	Plant-based
TA-1,2	36.14425	86.76282	62.7, 131.5	2016	Plant-based
VD-4	36.15015	86.80161	132.6	2014	Plant-based
AZ-1	36.26775	86.71164	122.9	2011	Rock
BM-1,2,3	36.15428	86.79139	162.0, 245.1, 255.5	2011	Rock
HC-1	36.13602	86.88670	50.4	2013	Rock
HG-1,2,3	36.14008	86.88361	109.4, 155.7, 45.8	2009	Rock
HG-4	36.13953	86.88208	64.1	2009	Rock
VD-2	36.14943	86.80173	201.8	2014	Rock
ZX-1,2	36.04577	86.95139	67.8, 31.6	2014	Rock

presence of fine sediment deposition, signs of erosion, shape of depression, vegetation health and abundance, and the presence of design components (e.g., forebay). Vegetation was documented with photographs of dominant plant species as well as a representative plant canopy cover photograph, using a mobile phone camera. Individual plant photographs were stored in a cloud location, shared with local plant experts (the Davidson County Master Gardeners), identified as accurately as possible, and compared to design documents (if available) for accuracy. Volunteer plants or weeds were not identified. Visual assessment of plant health was recorded. Canopy cover (%) was determined using the Canopeo (Oklahoma State University, Stillwater, OK) mobile application, which quantifies the proportion of an image with green pigment. It should be noted that this method did not allow for differentiation between plant species nor between designed plant community and volunteer vegetation.

Samples of the engineered media (n = 3) were collected for evaluating bulk density and organic matter content. Surface cover was removed to expose the top of the engineered soil layer. A bulk density hammer was used to push a 0.305 m long, 2.45 cm diameter acrylic core into the profile, extracted, and then capped with paraffin and foil. Samples were transported back to the laboratory in Knoxville, Tennessee, for analysis.

Laboratory Methods

Engineered media samples were dried for three days and mass measured to the nearest 0.00 g. Bulk density was determined as the mass of the dried media per volume of the sample (g/cm^3). Dried samples were then analyzed for organic matter content using the loss on ignition method. Samples were ignited at 400 degrees C for two hours, set to cool in a desiccator, and the mass determined. Organic matter content (%) was calculated by taking the difference in the masses of the dried sample and the ignited samples, dividing by the dry sample mass and multiplying by 100.

Statistical Methods

There were 52 independent sites used in the study. Sites were delineated into three categories based on surface cover: rock, plant-based, and composite. To be included in the rock category, at least 75% of the area of the practice needed to be covered in rock. Rock armoring in the inlet and outlet areas for energy dissipation was common, and not considered a factor for categorization. The plant-based category was assigned when mulch or plants covered the entire surface area (excluding energy dissipation areas). The composite category was assigned to the remainder of sites, where there was a mix of both plant-based and rock covering.

The Student t test (unequal variances) was used to evaluate the potential differences in media characteristics between the three surface cover categories (rock, plant-based, and composite). An alpha value of 0.05 was selected to show significance. Linear regression was used to determine if there was a relationship between media characteristics of bulk density and organic matter content, as well as between those characteristics and canopy cover.

The Shapiro-Wilk test was used to test the normality of the bulk density and organic matter content. For all coverage types, the bulk density and organic matter content are normally distributed at the 95% level of confidence. For the plant-based mulch cover and the composite cover, both bulk density and organic matter content are normally distributed at the 95% level of confidence; whereas, these properties are normally distributed at the 90% level of confidence for the rock cover. Therefore, all statistical tests and regressions were performed without data transformation.

Results

Media Characterization

Bulk density, organic matter, and canopy cover are reported in Table 2. On average, the organic matter content of media under plant-based mulch cover was significantly greater than that under rock cover (Table 2; $p = 0.002$). The media under the composite material has an organic matter content that was not different to that under the plant-based mulch ($p = 0.370$). The organic matter content of media under rock and composite materials was not different ($p = 0.099$). In general, the age of the bioretention areas did not significantly influence organic matter content within surface treatments, primarily due to the high variability in the

Table 2. Average bulk density (BD), organic matter (OM), vegetation cover (VC), and surface cover at each field site. NA – not applicable, no data collected for that site characteristic.

Site ID	BD (g/cm^3)	OM (%)	Surface Cover	VC (%)	Site ID	BD (g/cm^3)	OM (%)	Surface Cover	VC (%)
DG-1	0.89	14.4	Composite	50.9	MN-3	1.38	11.9	Plant-based	45.8
HC-2	1.54	3.9	Composite	71.1	MN-4	1.36	11.5	Plant-based	82.8
HC-3	1.54	4.8	Composite	52.6	MN-5	1.26	10.2	Plant-based	62.4
HC-4	1.58	5.5	Composite	59.1	MTA-1	1.15	12.9	Plant-based	69.4
MN-1	1.57	6.5	Composite	34.2	OH-1	1.69	3.1	Plant-based	55.2
MN-6	1.51	9.9	Composite	43.5	OH-2	1.74	2.1	Plant-based	35.1
SC-2	1.37	8.5	Composite	41.3	RB-1	1.01	11.6	Plant-based	97.0
SC-3	1.36	6.2	Composite	65.4	RB-2	0.86	11.8	Plant-based	99.3
VD-1	0.80	15.9	Composite	95.8	RB-3	0.89	15.9	Plant-based	98.7
VD-3	1.02	17.3	Composite	99.4	SC-1	1.36	7.7	Plant-based	31.1
AT-1	1.08	12.5	Plant-based	54.3	SC-4	1.41	6.9	Plant-based	90.3
AT-2	1.03	12.4	Plant-based	47.5	SC-5	1.34	6.2	Plant-based	44.1
AT-3	0.94	13.0	Plant-based	2.0	SS-1	0.97	NA	Plant-based	48.9
BB-1	1.44	5.4	Plant-based	83.1	SS-2	1.26	16.3	Plant-based	41.9
BB-2	1.46	5.1	Plant-based	91.9	TA-1	1.40	8.0	Plant-based	39.2
CB-1	1.10	13.8	Plant-based	69.5	TA-2	1.24	9.5	Plant-based	95.0
FW-1	0.84	12.5	Plant-based	94.0	VD-4	0.93	16.1	Plant-based	95.7
FW-2	0.93	15.9	Plant-based	71.0	AZ-1	1.61	1.7	Rock	36.6
FW-3	0.90	15.1	Plant-based	77.8	BM-1	1.59	4.8	Rock	27.2
HG-5	1.10	13.1	Plant-based	81.9	BM-2	1.69	4.5	Rock	31.1
MC-1	0.87	12.2	Plant-based	72.6	BM-3	1.69	5.7	Rock	35.2
MC-2	0.89	15.9	Plant-based	34.5	HC-1	1.40	4.4	Rock	53.2
MN-2	1.55	7.4	Plant-based	36.8	HG-1	1.74	4.9	Rock	60.6

measured values (Figure 2). The bulk density of the engineered media was strongly and inversely correlated to organic matter content (Figure 3). Similar to the organic matter content, bulk density did not generally vary with bioretention area age (Figure 4) and was highly variable within treatments. On average, the bulk density of the media under the plant-based mulch cover was significantly less than that under the rock cover ($p < 0.001$). The media under the composite treatments had similar bulk density to both the plant-based mulch ($p = 0.233$) and the rock covers ($p = 0.132$).

Vegetation

Canopy cover (%) varied widely between sites, from 16 to 99% among practices that contained living plants (Table 3). Two sites did not have any living plants. There was no significant relationship between canopy cover and any other variable.

Plants that were documented as present and healthy are listed in Table 4. The most observed herbaceous plants were common rush (*Juncus effusus*), black-eyed susan (*Rudbeckia fulgida*), and rose mallow (*Hibiscus lasiocarpos)*. The most observed shrubs were Virginia sweetspire (*Itea virginica*), summersweet (*Clethra alnifolia*), and inkberry holly (*Ilex glabra*). The most observed small tree was the sweetbay magnolia (*Magnolia virginiana*).

Discussion

This study evaluated the relationships between bioretention practice components of surface cover type, engineered media, and vegetated canopy cover. The strong inverse correlation between media bulk density and organic matter content supports conventional soil science knowledge about the same relationship in soil (Saini 1966). Since it has been shown that high organic matter in maturing bioretention cells has a positive relationship with trace metals measured in bioretention media (Costello et al. 2020), there are implications of surface cover selections on water quality treatment potential.

Significant differences in media bulk density between the rock covering and plant-based covering sites suggest that surface cover material influences

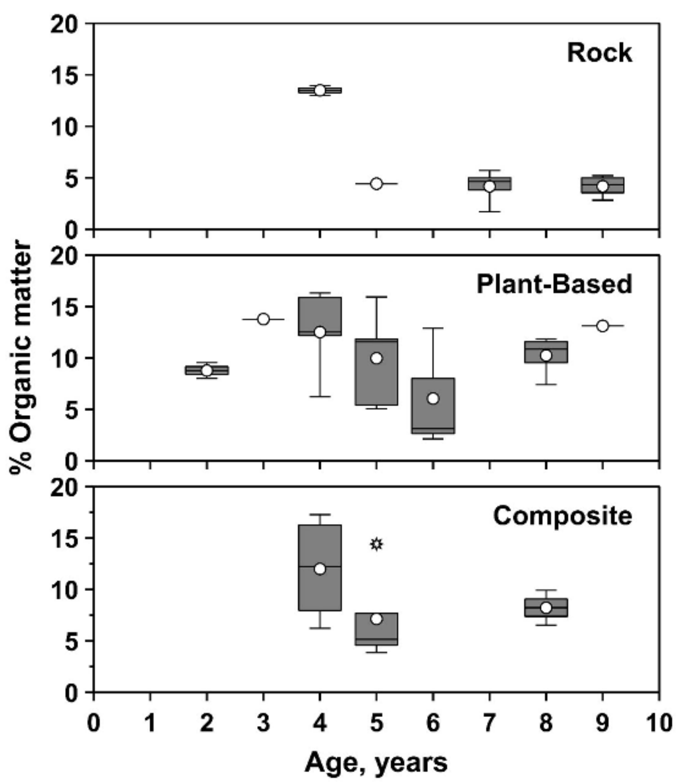

Figure 2. The organic matter content of soil media as a function of surface cover type and age.

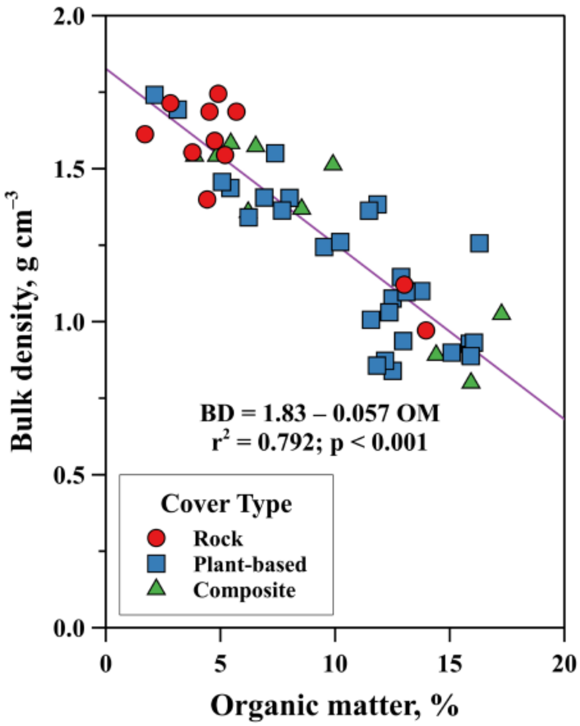

Figure 3. The relationship between bulk density and organic matter content of soil media as a function of surface cover type.

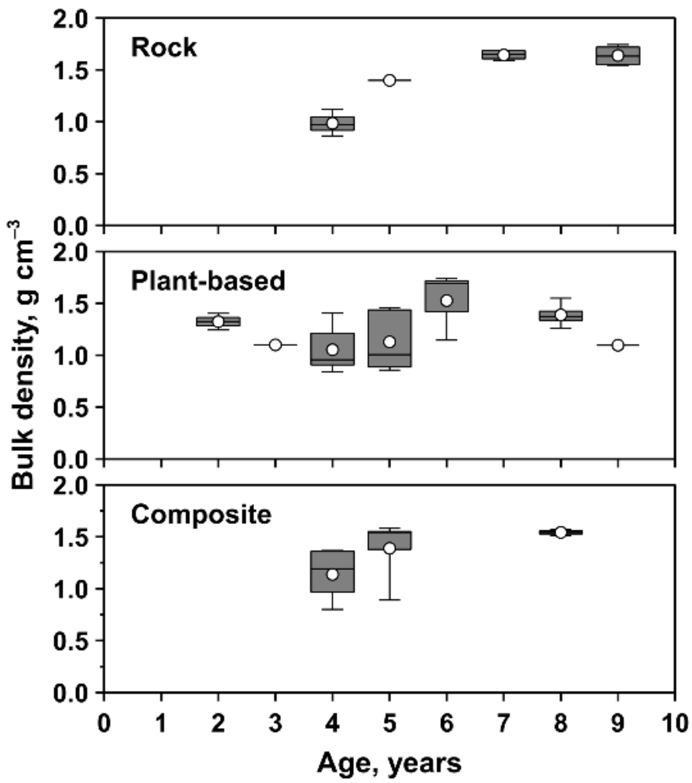

Figure 4. The bulk density of soil media as a function of surface cover type and age.

Table 3. The organic matter content and bulk density of surface media under various mulch cover types.†

Cover Type	Organic Matter Content %	Bulk Density g cm^{-3}
Plant-based (n = 20)	10.89 ± 4.02 a	1.19 ± 0.26 a
Rock (n = 12)	5.89 ± 3.93 b	1.51 ± 0.25 b
Composite (n = 10)	9.29 ± 4.90 ab	1.32 ± 0.30 ab

†Means ± standard deviations over all ages as a function of cover type. Mean followed by the same letter in the same column are not significantly different at the 95% confidence level.

Table 4. List of plants identified as present and healthy in bioretention study sites in Davidson County, TN, USA.

Common Name	Scientific Name
Ostrich Fern	*Matteuccia struthiopteris*
Switch Grass	*Panicum virgatum*
Common Rush	*Juncus effusus*
River Oats	*Chasmanthium latifolium*
Joe Pye Weed	*Eutrochium purpureum*
Butterflyweed	*Asclepias tuberosa*
Black-eyed Susan	*Rudbeckia fulgida*
American Alumroot	*Heuchera americana*
New England Aster	*Symphyotrichum novae-angliae*
Rose Mallow	*Hibiscus lasiocarpos*
Gray Dogwood	*Cornus racemosa*
Beautyberry	*Callicarpa americana*
Witch Hazel	*Hamamelis virginiana*
Smooth Hydrangea	*Hydrangea arborescens*
Inkberry Holly	*Ilex glabra*
Virginia Sweetspire	*Itea virginica*
Buttonbush	*Cephalanthus occidentalis*
Oakleaf Hydrangea	*Hydrangea quercifolia*
Summersweet	*Clethra alnifolia*
Sweetbay Magnolia	*Magnolia virginiana*

the underlying media, which has implications for the overall function of the practice. These findings suggest that rock surface covering used instead of plant-based mulch may adversely affect the function of bioretention systems in terms of storing and infiltrating stormwater runoff. There are also implications for mixing and associated treatment efficiencies of these practices. Studies have shown infiltration rates of bioretention cells to not diminish with age up to ten years (Spraakman and Drake 2021). However, the literature is more varied when examining bioretention function related to water quality treatment over time. Although the media sampled had various ages (from 2 to 9 years), the influence of age on the media characteristics, such as the accumulation of organic matter, was not evaluated. While it is evident that both organic matter and bulk density vary as a function of age under rock cover (and only under rock cover) (Figures 2 and 4), conclusions cannot be drawn about the influence of time (age). This would require the continuous sampling of the sites, beginning with installation. The measurements were all from different areas, and the initial conditions of each bioretention area were unknown.

The results also raise questions about the effect of surface covering selection and vegetation. Healthy vegetation aids in the physical, chemical, and biological processes needed for a fully functioning bioretention system (Muertder et al. 2018). Plant roots maintain soil structure and create macropores that enable fluid transport (Angers and Caron 1998), but the role of root-induced effects on media properties needs further investigation (Skorobogatov et al. 2020). Vegetation absorbs and dissipates energy, and the biomass aids in microbial processes. Plants directly uptake potential pollutants (e.g., nutrients and trace metals) (Mehmood et al. 2021). Vegetation also plays a significant role in the water budget and associated nutrient budgets in bioretention practices (Nocco et al. 2016). The plant community supports local wildlife (Kazemi et al. 2011), along with additional ecosystem services that provide co-benefits to humans. To this end, it is important to facilitate the establishment and maturation of a healthy plant community in bioretention practices to fully realize maximum functionality.

Compacted soil conditions may inhibit plant establishment. A soil bulk density of 1.6 g/cm^3 may adversely affect plant rooting capacity in sandy loam (Daddow and Warrington 1983). Media bulk density above this threshold was measured in more rock covered applications (6) than plant-based mulch covered applications (2). Though the canopy cover data varied widely, there are two implications based on the relationships between canopy cover and media characteristics. There was no canopy cover greater than 75% (performance criteria) (Metropolitan 2021) observed at sites where the media bulk density was greater than 1.55 g/cm^3, nor at locations with organic matter less than 5%. While rock is considered a permanent cover (as opposed to temporary cover like straw or some established seed), many performance requirements reference permanent vegetated cover to be greater than 80%. These findings show that more research is needed to evaluate the effect of rock coverings on meeting vegetation-focused performance requirements in bioretention applications.

There are other possible reasons for the differences in bulk density, organic matter, and vegetation characteristics observed in this study. The original hydrologic design may affect circumstances that influence the condition of media and vegetation. Installation practices, plant selection, and ongoing maintenance activities may also play a role in the observed conditions. Other interactions between the practice and adjacent topography, underlying soil, geology, and other site-specific conditions may also lead to differences in measured characteristics.

The plant species observed to be healthy and thriving in the studied bioretention practices (Table 4) may be favorable replacements where other selections have failed, or during bioretention renovations. This list is suitable for use in the Davidson County area but may also be useful for practitioners throughout the same ecoregion(s) depending on native status and site conditions. It is advised to check the native status of the species before specifying for a design or planting and give preference to those native to the region in which the application is to be installed.

Conclusion

This study found that organic matter content of bioretention media under plant-based mulch cover was significantly greater than that under rock cover. The bulk density of media was strongly and inversely correlated to organic matter content, and on average, was significantly less where plant-based cover was used rather than rock cover. These findings have implications for design and long-term maintenance. A functional goal of full plant canopy cover may help maintain soil structure, porosity, and infiltration capacity as well as support healthy vegetation. This functional goal will create a system that naturally replenishes media organic matter as part of the seasonal vegetation cycle, creating a more self-sustaining practice than one that utilizes dredged or quarried stone.

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Assessment of Recycled and Manufactured Adsorptive Materials for Phosphate Removal from Municipal Wastewater

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Abstract: Elevated concentrations of phosphorus (P) and other nutrients common in wastewater treatment plant (WWTP) effluent have been shown to contribute to the proliferation of harmful algal blooms, which may lead to fish kills related to aquatic hypoxia. Increased understanding of the negative effects associated with elevated P concentrations have prompted more strict regulation of WWTP effluent in recent years. The use of low-cost and potentially regenerative adsorptive phosphate filters has the potential to decrease P concentrations in WWTP effluent released to natural waters. This research focuses on assessing the capacities of recycled concrete aggregate (RCA), expanded slate, and expanded clay to remove phosphate from P-amended WWTP effluent. Results from a flow-through column study indicate that RCA consistently removed an average of 97% of phosphate over 20 weeks of continuous flow at an 8-hour hydraulic retention time (HRT). Expanded clay removed an average of 63% of introduced phosphate but decreased in removal capacity from 91 to 42% over the 20-week duration. Sorption data from batch studies were fitted to Langmuir models and RCA was shown to have the highest maximum sorption capacity (6.16 mg P/g), followed by expanded clay (3.65 mg P/g). RCA and expanded clay are promising options for use in passive filters for further reduction of phosphate from WWTP effluent.

Keywords: *Langmuir model passive filter, sorption, treatment*

Eutrophication is the process of accelerated plant and algae growth resulting from the introduction of excess nutrients into waters, potentially resulting in harmful algal blooms. Aerobic microorganisms then consume dissolved oxygen while breaking down the organic matter, resulting in lower levels of dissolved oxygen, known as water hypoxia (Paerl 2009). Increases in eutrophication levels can be caused by anthropogenic sources, such as domestic wastewater from wastewater treatment plants (WWTPs), or onsite wastewater systems, such as septic systems (Preisner et al. 2020). Inputs of phosphorus (P) from point and non-point source wastewater can negatively affect wildlife and human health. Limiting inputs of P to receiving waters can decrease the severity of eutrophication (Xie et al. 2013).

Hydrologic and climatic challenges, paired with aging infrastructure and increasingly stringent effluent regulatory limits, have incentivized WWTPs to seek alternative and supplemental treatment options. The use of low-cost and potentially regenerative adsorptive filters can remove P from wastewater and decrease concentrations released to natural bodies of water (White et al. 2021). These adsorptive materials typically contain compounds that can bind P, such as oxides of calcium (Ca), magnesium (Mg), iron (Fe), and aluminum (Al), which enable adsorption and precipitation processes to occur (Gubernat et al. 2020). Many natural, manufactured, and waste P adsorptive materials have been investigated; however, efficacy, cost, and availability of these materials vary widely (Cucarella and Renman 2009). Large volumes of concrete waste are

Research Implications

- Recycled concrete aggregate (RCA) and manufactured expanded clay may be used as passive adsorptive filters to further reduce phosphorus (P) loads from wastewater treatment plants (WWTPs).
- Potential negative environmental impacts from the use of adsorptive P filters, such as increased pH and leaching of ions, should be explored.
- Desorption characteristics of adsorptive P filters should be investigated to determine their potential as a substrate or soil amendment to grow crops, moving us toward a circular economy.

produced globally as older concrete buildings are demolished for new builds, and in many cases, this concrete is not reused (Deng and Wheatley 2018). Recycled concrete aggregate (RCA) has been investigated as a reactive material for P removal due to high lime content, with studies estimating up to 70 mg P/g maximum sorption capacity, according to the Langmuir model (Gubernat et al. 2020). Manufactured lightweight aggregates (e.g., slate and clay), which are natural minerals that have been crushed and heated to high temperatures, have also demonstrated high P sorption capacity – up to 12 mg P/kg (Zhu et al. 2003; Vohla et al. 2011; Baker et al. 2014; Gubernat et al. 2020).

While previous lab studies have proven many adsorbent materials to be effective at reducing P concentrations in various solutions, most are conducted at hydraulic retention times (HRTs) on the order of days (rather than hours) or exclusively as batch studies, and most do not assess actual effluent from a WWTP (Vohla et al. 2011; Wu et al. 2020). The objective of this study was to determine the phosphate sorption capacity of RCA, expanded slate, and expanded clay at the lab-scale under both batch and flow-through conditions using effluent from a local WWTP. Testing the materials at batch and flow-through conditions (i.e., at varying hydraulic conditions and scales) provides a more holistic understanding of how the materials would perform as adsorptive filters in a WWTP setting.

Materials and Methods

A locally produced RCA (Carolina Concrete Recycling, New Bern, NC, USA), a locally manufactured expanded slate (Stalite Lightweight Aggregate, Salisbury, NC, USA), a manufactured expanded clay (Filtralite®, Nordby, Norway), and a locally procured granite gravel (E.R. Lewis Construction Company, Greenville, NC, USA) as a control, were assessed to determine phosphate sorption efficacy. Two experiments were carried out: 1) a flow-through column study to characterize phosphate sorption efficacy under higher flow conditions than have generally been reported in the literature (Vohla et al. 2011), which could allow for treatment of higher volumes of water; and 2) a batch study to determine the phosphate sorption capacity of each material. All materials were sieved to the 1.00 to 3.35 mm particle size range using sieve pans. Effluent from a WWTP located in Greenville, NC, USA, was used in both studies and collected within one week prior to each experiment. Treated effluent was collected using a sump pump from the clear well, located after the UV disinfection process and just before release of treated effluent to a nearby river. Experimental conditions and sampling frequency are described below.

Flow-through Column Study

A column study was conducted to identify how much P was able to be removed in conditions similar to a WWTP. Physical properties of experimental materials are reported below (Table 1). Columns were filled with oven dried materials (tamping intermittently) and then saturated with water to fill the pore spaces. Porosity was calculated as the amount of added water divided by total column volume. Bulk density was calculated as the difference between empty and material-filled column weights divided by the column volume. As shown in Table 1, weight varied considerably amongst materials.

As shown in Figure 1, three replicate columns containing each material were made of clear polyvinyl chloride (PVC) and measured approximately 30 cm tall with a diameter of 10 cm (total volume approximately 2.36 L). Treated wastewater at the partnering WWTP typically

Table 1. Average porosity and bulk density values (n = 3) for materials used in batch and flow-through column study using recycled concrete aggregate (RCA), expanded clay and slate, and gravel.

Material	RCA	Expanded Clay	Expanded Slate	Gravel
Porosity (%)	48.8	31.6	33.0	83.7
Average Bulk Density (kg/m^3)	1184	1445	850	464

Figure 1. Experimental set-ups of flow-through column study (left) and sorption capacity batch study (right) in Coastal Ecological Engineering Lab at East Carolina University.

contains less than 1 mg PO_4-P/L; however, during periods of high rainfall or due to unanticipated influent characteristics, effluent concentrations can reach 5 mg PO_4-P/L or higher at times. During the course of this experiment, the WWTP effluent maintained concentrations below 1 mg PO_4-P/L when effluent was collected. To represent a "worst-case" scenario and ensure P removal was not limited by low P concentrations typically observed in the effluent, the WWTP effluent was amended with approximately 5 mg PO_4-P/L and continuously pumped into the bottom of each column up to the top where it was connected to an effluent tube as shown in Figure 1. A calculated (i.e., theoretical) 8-hour HRT was maintained throughout the duration of the study from March to July 2021. Actual HRT, which can be determined using tracers, was not investigated, though could shed light on potential preferential flow or dead spaces. The flowrate of influent water for each material type was calculated by dividing the product of column volume and porosity by the calculated HRT. In order to determine representative long-term P removal efficacy, samples were collected from the influent and effluent of each column one to two times each week on average and analyzed for phosphate using the Hach (Loveland, CO, USA) DR6000 spectrophotometer and test-n-tube kits using Method 10209/10210. Only samples for which the detected concentration of the accompanying standard was within +/-10% of the standard concentration value were considered for statistical analyses. The pH of samples was measured using a multiparameter benchtop meter

(Orion™ Versa Star Pro™, Thermo Fisher Scientific Inc., Waltham, MA, USA). The temperature, conductivity, and oxidation reduction potential (ORP) were measured using a YSI ProDSS.

Percent PO_4-P removal was calculated as follows:

$$\% \ Removal = \frac{(C_i - C_e)}{C_i}(100) \tag{1}$$

where C_i is the influent concentration and C_e is the column effluent concentration.

Average load removal was also calculated as follows for each material:

$$Load \ Removed = \frac{(\% \ Removal)}{100} * (C_{i\,avg}) * Q \tag{2}$$

where $C_{i\,avg}$ is the average influent concentration and Q is the flowrate.

Statistical comparisons were made between treatments (material type) using JMP Pro 17 (SAS Institute 2022). Shapiro-Wilk tests indicated that data were not normally distributed ($p < 0.0001$); therefore, non-parametric Kruskal-Wallis tests were used to determine if phosphate reduction, pH, specific conductivity, and ORP differed significantly ($\alpha = 0.05$) by material type. When significant differences were found, the Steel-Dwass post-hoc test was used to separate treatment medians to determine which specific material types differed. Though transformation of data followed by a parametric comparison test could have been utilized, a non-parametric approach was chosen to avoid potential over-compensation and distributional assumptions that may not adequately address non-normality and unequal variance issues (Mahachie John et al. 2013).

Sorption Capacity Batch Study

The second portion of the study was a batch study to determine the maximum P sorption capacity of each material. The materials were dried in an oven for 24 hours before use. For the first experimental trial, wastewater effluent was amended with monopotassium phosphate (KH_2PO_4) to achieve the following initial concentrations of P (C_i) in solution, similar to those selected by White et al. (2021): 0, 5, 10, 20, 30, 50, 100, 200, 500, 1000, 1500, and 2000 mg/L. As KH_2PO_4 was added to the wastewater effluent, pH decreased as shown in Table 2. Therefore, a second experimental run was carried out in which initial pH of KH_2PO_4-amended wastewater effluent was adjusted to the pH of the unamended wastewater (6.8) using potassium hydroxide to achieve the following initial concentrations of P (C_i) in solution: 0, 20, 50, 200, 500, and 1000 mg/L.

Each centrifuge tube contained 5 g of each material type along with 45 mL of amended wastewater effluent. There were three replicates of each material type for each different concentration. Once the weighed material and desired concentration of wastewater effluent were placed into the tube, the tubes were put into an orbital shaker for 24 hours at 150 rpm at room temperature (between 20 to 25°C), following methods by White et al. (2021). After 24 hours, samples were syringe-filtered (0.45 µm) and frozen until analysis (within 24 hours). All samples were analyzed for PO_4-P using a Smartchem 200 Discrete Analyzer (KPM analytics, Westborough, MA, USA) located in East Carolina University's Environmental Research Laboratory, using standard methods (APHA 2012).

Phosphorus sorption isotherms were created by plotting concentrations of sorbed PO_4-P (C_s, mg P/g substrate) for each replicate (n =3) by concentration of PO_4-P remaining in solution (C_{aq}, mg P/L). Sorbed P was calculated as follows:

$$C_s = \frac{\left((C_i - C_{aq})(V)\right)}{M_m} \tag{3}$$

Table 2. Unadjusted initial pH of wastewater effluent amended with KH_2PO_4 at the start of the first trial of the batch study.

KH_2PO_4-amended Initial Wastewater Effluent Concentration (mg P/L)	Initial pH
0	6.8
5	7
30	6.6
50	6.4
100	6.6
200	6.2
500	5.7
1000	5.3

where C_i is the initial concentration of PO_4-P in solution, V is the volume of solution (0.045 L), and M_m is the mass of material (5 g). Following methods by White et al. (2021), experimental data were fit to non-linear Langmuir models using a spreadsheet developed by Bolster (2010), which uses non-linear least squares regression to predict sorption capacity.

Results and Discussion

Flow-through Column Study

In the column study, 97% of P was removed by RCA and 63% of P was removed by expanded clay as shown in Figure 2, representing a significant difference ($p < 0.0001$). The P load removed by RCA was 10.1 mg/d as compared to 10.0 mg/d removed by expanded clay. Percent P reduction did not vary significantly between the expanded slate (average 0.6% reduction) and gravel (average 4.4% addition). Expanded clay decreased in removal capacity over the 20-week duration (from 91 to 42% removal), while RCA maintained consistent removal. Jensen et al. (2022) achieved similar (up to 100%) P removal in columns containing a variety of calcareous materials, with P inputs ranging from 3 to 22 mg P/L. Ádám et al. (2007) demonstrated above 90% P removal from a 10 mg P/L-amended secondary wastewater solution within columns containing expanded clay, with average HRTs of around four days. Potential contributing mechanisms of P removal include adsorption of phosphate ions due to high surface area and porosity (strong surface complex formation between Ca and Al compounds in RCA and with clay minerals in expanded clay), precipitation to form calcium phosphate minerals, ion exchange of hydroxide, chloride, or other ions, and surface complexation.

Columns were maintained at room temperature, with an average water temperature of approximately 21°C. The specific conductivity of effluent from columns containing RCA was higher than effluent from all other columns, as well as the influent (Table 3). This is likely due to leaching of ions, such as Ca, Mg, and bicarbonate, from the RCA, as has been documented in other studies (Engelsen et al. 2017). The ORP of effluent from columns containing RCA did not differ from columns containing expanded clay but was lower than the influent and effluent from columns containing expanded slate and gravel. The comparatively lower ORP values observed from effluent from columns containing RCA and expanded clay could be due to differences in surficial active sites which potentially allow for reduction of oxidizing agents in the water. Other studies have similarly observed increased P adsorption potential correlated with decreased ORP (Zhou et al. 2005; Andrés et al. 2018).

The pH did not vary between influent water and effluent coming from columns containing either gravel or expanded slate (Table 3; Figure 3), as

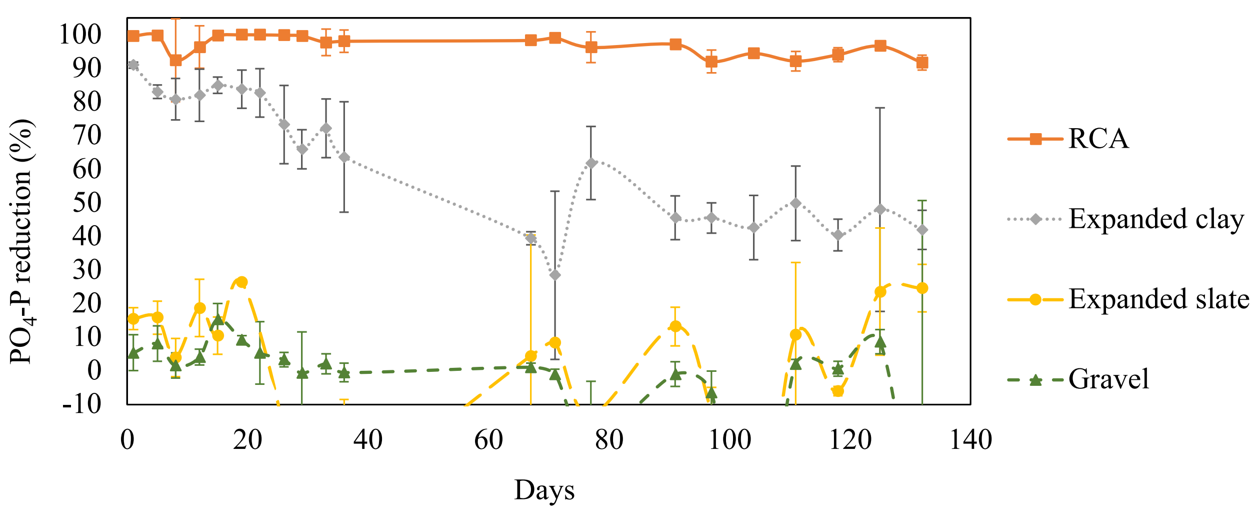

Figure 2. Average percent phosphate reduction for each material type over the duration of the flow-through study. Data points represented average percent reduction from three replicate columns as compared to the influent concentration with error bars representing standard deviation.

determined by statistical analyses. The effluent pH from columns containing RCA was greater than the effluent from columns containing expanded clay, which were both greater than the influent pH. The effluent pH from columns containing RCA decreased over the duration of the study (from 11.4 to 9.6), while effluent pH from columns containing expanded clay showed a decreasing trend over approximately the first month (from 9.4 to 7.8), followed by wider fluctuations of values over the remainder of the study. High pH values (up to 12.3) have been observed in other studies utilizing concrete, with pH generally increasing as particle size decreases (Gubernat et al. 2020). High pH values have also been observed from expanded clay within hybrid constructed wetlands, with effluent pH values between 8.1 to 8.8 observed in the first nine months of operation, then decreasing to 7.6 in the following three months (Põldvere et al. 2009).

Sorption Capacity Batch Study

The batch study revealed that the RCA and the expanded clay aggregate achieved the greatest modeled sorption capacities of the materials tested (Table 4; Figure 4). RCA achieved a 9.04 mg PO_4-P/g modeled maximum sorption capacity (R^2 =

Table 3. Average pH, specific conductivity, and oxidation reduction potential (ORP) values from influent and effluent water from flow-through column study, with standard deviation shown in parentheses. Treatments that share a letter are not significantly different (α = 0.05).

Material or Sample Type	pH		Specific Conductivity (uS/cm)		ORP (mV)	
Influent	7.2 (0.35)	C	495.0 (131.4)	B	220.8 (42.7)	AB
RCA	10.6 (0.61)	A	777.8 (310.7)	A	109.9 (82.4)	C
Expanded clay	8.7 (0.64)	B	572.6 (193.1)	B	174.1 (66.7)	BC
Expanded slate	7.3 (0.21)	C	491.7 (160.4)	B	229.4 (79.2)	AB
Gravel	7.2 (0.22)	C	467.2 (158.5)	B	232.0 (70.0)	A

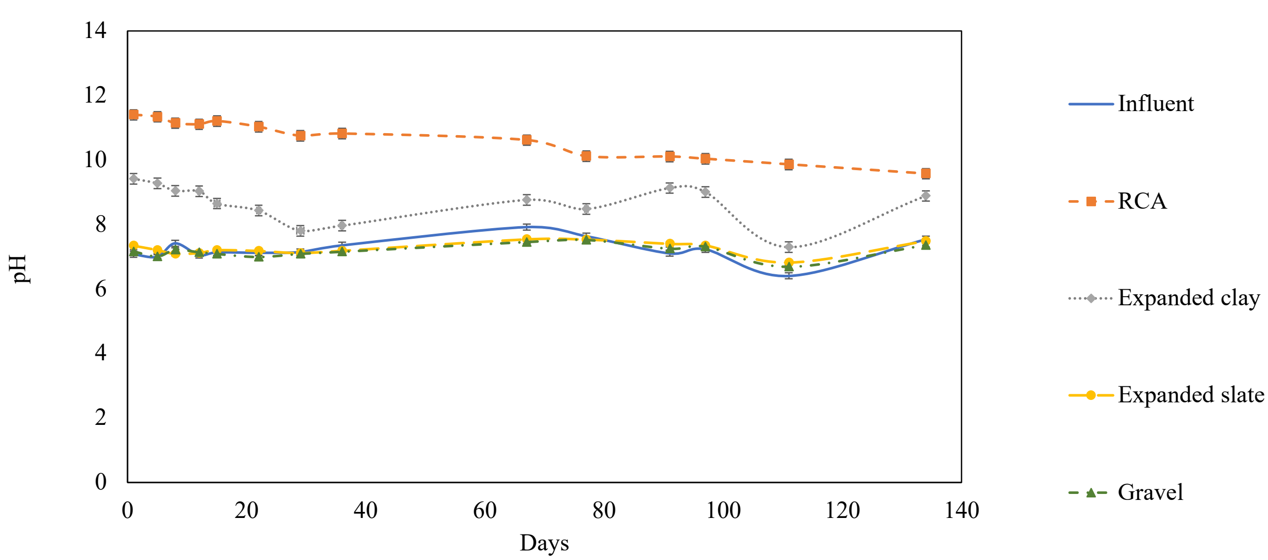

Figure 3. Average pH of influent and effluent (n = 3) water from columns over the duration of the flow-through study with error bars representing standard deviation.

0.99) and expanded clay achieved a 9.11 mg PO_4-P/g capacity (R^2 = 0.98) for the first experimental trial in which KH_2PO_4-amended wastewater pH was not adjusted. For the second experimental run of the batch study during which KH_2PO_4-amended wastewater pH was adjusted to the initial unamended wastewater pH, RCA achieved a 6.16 mg PO_4-P/g modeled maximum sorption capacity (R^2 = 0.97) and expanded clay achieved a 3.65 mg PO_4-P/g capacity (R^2 = 0.93). Both datasets for both experimental runs demonstrated good fit to the Langmuir model as indicated by the coefficients of determination. Coefficients of determination for expanded slate and gravel were negative, indicating that the Langmuir model is inappropriate for these data; therefore, modeled isotherms are not included in Figure 4. Differences in modeled maximum sorption capacities between the first and second experimental runs of the batch study highlight the impact that initial pH has on sorption capacity (as many studies have demonstrated), and the importance of adjusting pH for all tested P concentrations to ensure reliability of modeled sorption results (i.e., avoid confounding of experimental variables). Differences in modeled maximum sorption capacities between the first and second experimental runs of the batch study could also be attributed to differences in initial selected P concentrations in solution.

The modeled maximum sorption capacities of the materials tested in this study are comparable to those reported in a review by Gubernat et al. (2020; Table 4). Of the seven low-cost materials that Boyer et al. (2011) evaluated in a series of jar tests and mini-column experiments, recycled concrete was among the best performing materials for phosphate removal; however, the increase of pH by greater than 2 units was noted as an undesirable secondary change that could negatively impact ecosystem health of receiving waters. In some instances, increased alkalinity in effluent filter water may actually be desirable. For example, the WWTP from which effluent was used in this study is interested in potentially using adsorptive filters to reduce P concentrations in sludge digestor decant water, a sidestream process during which this nutrient-dense water is recycled back to the head of the plant (J. Manning, personal communication, 2021). An increase in pH (and alkalinity) of P filter effluent that is routed to the head of the plant could contribute to the pH buffering capacity within the WWTP itself. In situations where effluent from P filters would be directly discharged to receiving waters, mitigation steps would potentially need to be put in place to decrease the effluent pH to within the allowable range for that specific receiving waterbody. Mitigation options could include dilution of the P filter effluent with WWTP effluent that was not treated through the P filter, or implementation of another treatment step.

Table 4. Phosphorus sorption capacities (C_{smax}, mg P/g) for recycled concrete aggregate (RCA) and expanded clay, according to the Langmuir model (current study denoted by asterisk), as compared to values reported in Gubernat et al. (2020) for similar materials.

Material Type	C_{smax} (mg P/g)
RCA	6.16*
Expanded clay	3.65*
Autoclaved concrete	0.28 – 70.90
Biochar (raw)	2.39
Lightweight aggregate	2.50 – 12.00
Sand	0.06 – 0.13
Zeolite	0.46 – 2.19

Conclusion

Results from the flow-through column study indicate that the RCA and expanded clay materials effectively reduced P concentrations in P-amended WWTP effluent at a much lower HRT (higher flowrate) than has been reported in the literature (eight hours vs several days). The batch-scale study demonstrated that RCA and expanded clay also have the highest maximum sorption capacities of all materials tested in this study, according to the Langmuir model. RCA, the best performing material, was also associated with high effluent pH levels (10.6 on average). While such alkaline levels would not be suitable for release into the environment, this increased alkalinity may be useful in instances where P filter effluent could be recycled within a WWTP (i.e., to treat a sidestream process). These results highlight the potential of

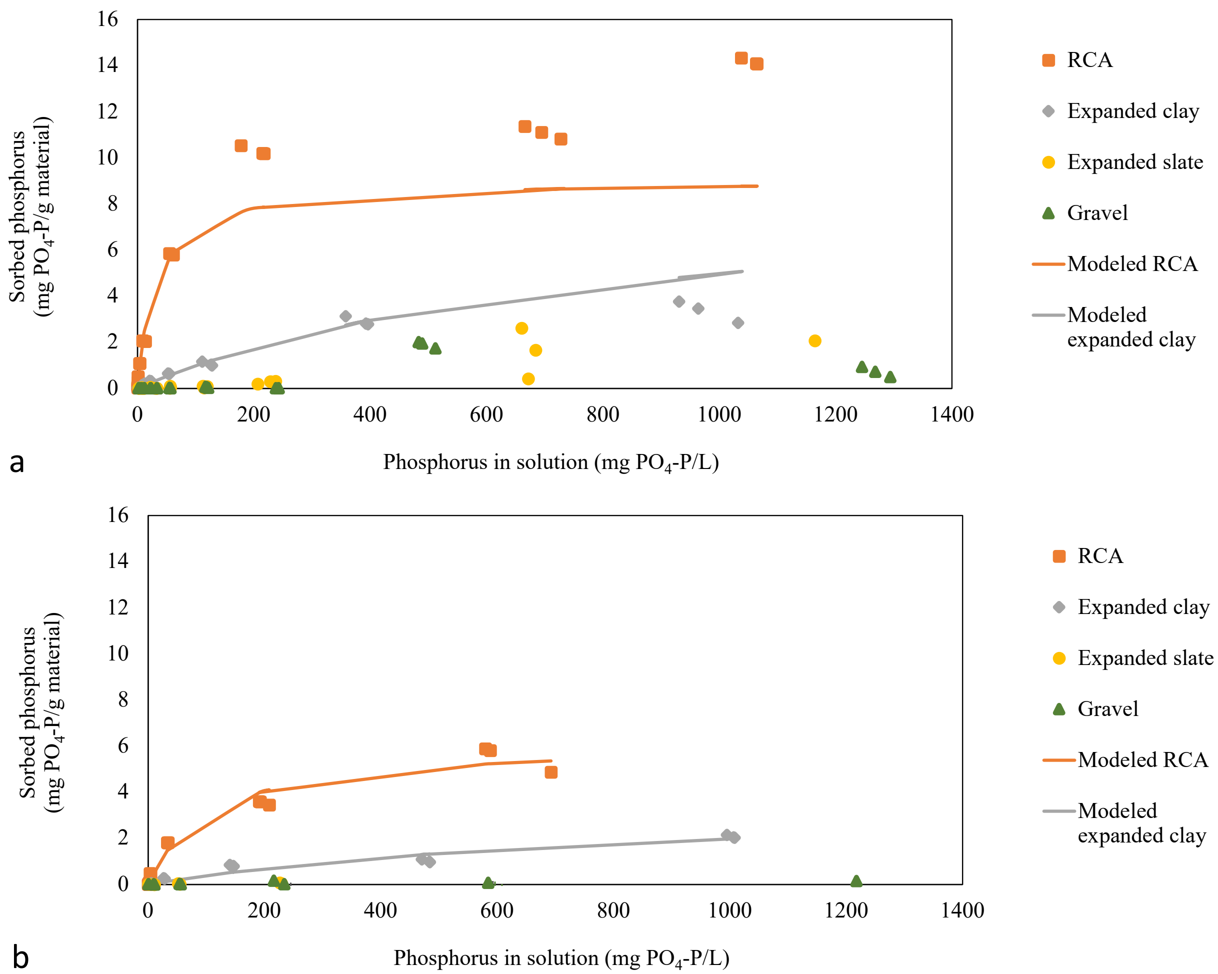

Figure 4. Phosphorus sorption isotherms of materials in phosphate-amended wastewater effluent for first experimental run of the batch study for which initial pH of KH_2PO_4-amended wastewater effluent was not adjusted (a) and second experimental batch study run for which initial pH of KH_2PO_4-amended wastewater was adjusted to unamended wastewater effluent pH using potassium hydroxide (b). Isotherm data were fit to a non-linear Langmuir model for RCA and expanded clay materials (solid lines).

recycled materials such as RCA to serve as a cost-effective add-on treatment technology to reduce P loads from WWTPs to receiving waters. These add-on technologies would not require substantial modification to existing infrastructure and would make beneficial reuse of a waste product.

The next steps for this study include investigation of the P desorption characteristics of adsorptive materials. Future work could also include exploration of P removal at various HRTs, temperatures, and starting P concentrations. Insight into mitigation measures to address increased pH of P filter effluent and other potential negative environmental impacts, as well as desorption behavior of materials, could enable investigation of spent P filter materials as a potential substrate or soil amendment for plant growth. Reuse of recycled materials for incorporation into passive filters and then for plant growth would serve to close the gap from production to disposal, bringing us closer to a circular bioeconomy.

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Changes in Streamflow Statistics and Catchment Land Uses Across Select USGS Gages in Northwest and West-central Arkansas

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Abstract: Since 1901, heavy rainfall events have increased in the United States in both intensity and frequency, and human population in the United States has increased, resulting in significant land use changes. Both trends contribute to an increase in observed flood magnitude and frequency. To determine if a relationship exists between land use/land cover and changing stream flows in northwest Arkansas, this study analyzed temporal changes in various flow statistics for 14 stream gages and compared the rates of change in flow statistics from gages on streams with watersheds that have varying land uses, i.e., urban, agricultural, and undeveloped. Mann-Kendall analysis was used to determine statistically significant changes in flow statistics, which were then compared to National Land Cover Dataset (NLCD) watershed land uses from 2001 and 2019. All analyzed gages had one or more flow statistics with at least a moderately significant increase, and all analyzed flow statistics showed at least moderately significant streamflow increases at two or more gages ($P < 0.100$). There were no decreases of any significance in any flow statistic at any gage. In general, urban land development did not happen on native prairies and forests but on previously agricultural land. Significant positive relationships were found between maximum yearly flow and 2019 urban land use, urban land use change from 2001 to 2019, and 2019 Human Development Index (HDI). A similar relationship was found to exist between yearly minimum flow and 2019 HDI. These results highlight the importance of considering the cost of potential stream bank erosion and flooding in future land use planning, permitting, and zoning.

Keywords: *streamflow statistics, days exceeding floods, land use, northwest Arkansas*

Flooding often causes extensive damage, so it is one of the major weather and climate disaster types tracked by the National Oceanic and Atmospheric Administration's (NOAA) National Centers for Environmental Information. In the United States, flooding takes 88 lives (NOAA 2021) and does $17 billion dollars of damage (FEMA 2020) annually. While deaths and cost of damage are the most common measures of flood damage, flood damage can cause utility outages, disrupt transportation and supply chains, and result in environmental problems like pollution.

Flood damage can be categorized into direct and indirect damage, then further differentiated by being tangible or intangible (Merz et al. 2010). Direct damage comes from physical contact with flood water, while indirect damage occurs outside of the flood location and/or time and is caused by direct damage. Tangible damage can be assessed in monetary value, while intangible damage cannot be assigned a value. Direct, tangible damage includes damage to buildings, property, and infrastructure. Direct, intangible damage includes loss of life and destruction of ecosystems. Indirect, tangible damage includes the disruption of transportation and other services outside of the flooded area because of direct damage to roads and infrastructure. Indirect, intangible damage includes

Research Implications

- The most prominent land use change across these watersheds appeared to be conversion of pasture to urban.
- Streamflow generally increased across all the selected United States Geological Survey (USGS) stream gages, and no statistics showed a significant decrease across the gages.
- Significant increases in streamflow were typically correlated with urban land use and or change in urban land use over time.
- The growing urban areas need to consider how increasing streamflow influence bank stability and potential flooding and frequency downstream.

psychological trauma and distrust in authorities. Often, commercial structures represent half of the monetary damages in flood prone zones (Shultz 2017). Regardless of how they are classified, the many types of flood damage have major economic, social, and environmental costs.

Flooding occurs when runoff exceeds the capacity of natural channels and manmade stormwater conveyance systems. Rainfall intensity, duration, and frequency influence runoff from the landscape, which occurs when rainfall exceeds interception, infiltration, evapotranspiration, and storage capacity. Due to climate change, temperatures are rising, and in turn, evaporation rates are also rising (Lin et al. 2017; UCAR 2021). In fact, atmospheric moisture in the United States is increasing at 5% per decade, which is expected to cause more precipitation and therefore more flooding (Trenberth 1998). The excess water vapor will likely increase precipitation outside of the subtropics (Dai et al. 2018) including temperate areas.

Although rainfall is a major factor that affects runoff rates across large spatial scales, runoff is also affected by local land use and factors such as land use change and/or development and resulting changes in vegetation cover, land slope, soil type and conditions, and impervious surfaces (USGS 2019). Removal of vegetation, compaction of soil, and increases in impervious surfaces increase runoff by lessening infiltration of rainfall into the soil. Grading a development site can either decrease runoff by decreasing land slopes, which increases time for infiltration to occur, or increase runoff by removing natural storage basins (NJDEP 2016).

Changes in land use, specifically involving urban development and conversion of forest to agricultural land, change the infiltration and storage capacity of a landscape. Urbanization increases impervious surfaces, which can cause flooding, channel degradation, and ecosystem disruption (Booth et al. 2002; Brown et al. 2005), "unless measures are taken to detain the runoff and control the rate of discharge off of newly developed sites" (City of Rogers 2018). Many states and municipalities require development sites to ensure post-developed runoff rates are less than pre-developed runoff rates for a few specific storm events (e.g., 1- and/ or 2-year, 24-hour storms; USEPA 2011). In theory, this should prevent increased flooding due to land development, but runoff calculation models are not perfect, and changing precipitation patterns are not necessarily considered.

Flooding frequency has increased by 2.5 times in northern mid-latitudes since the 2000's (Najibi and Devineni 2018), and flooding magnitude and frequency have also increased specifically in the United States (Berghuijs et al. 2017). This begs the question, which factors (precipitation or land use) that affect runoff, or both, is the major cause of the increased flooding? Since 1901, heavy rainfall events have increased in the United States in both intensity and frequency (Easterling et al. 2017), and population in the United States has increased, resulting in significant land use changes (Loveland et al. 2002). This study will evaluate discharge data from streams whose watersheds have experienced significant change in land use along with discharge data from streams whose watersheds have experienced little land use change. Specifically, changes in flow statistics were analyzed at each site in northwest Arkansas (NWA), including:

- number of days per year when mean daily flow surpassed given thresholds of moderate and severe flooding, and
- various annual flow statistics, including mean, selected percentiles, and peakflow.

This study analyzed changes in flow statistics over time for individual stream gages and compared rates of change in flow statistics for gages on streams with watersheds that have varying land uses, i.e., urban, agricultural, and undeveloped.

While this study focuses on changes in high flows, changes in low flows were also analyzed. Low flow is defined by the EPA as "flow of water in a stream during prolonged dry weather" (USEPA 2021). These low flows are not derived from direct runoff, but rather provided by groundwater discharge, subsurface return flows, surface discharge from lakes and marshes, or even melting glaciers in select regions (Smakhtin 2001). Low flows caused by groundwater recharge and subsurface return flows, which is the most prevalent low flow source in the study area, are affected by soil series distribution and infiltration, hydraulic characteristics of aquifers, evapotranspiration from the watershed, topography, and climate (Smakhtin 2001). Understanding changes in low and high flows are important for managing water supply, stormwater, waste-load allocation, reservoir storage, recreation, and wildlife conservation (Smakhtin 2001), as well as educational opportunities (Hutton and Allen 2021) and research needs (Bilotta and Peterson 2021).

Methods

Study Site Description

Data were obtained from 14 United States Geological Survey (USGS) stream gages across NWA and northeast Oklahoma using the National Water Information System (NWIS) where most of the drainage areas were in NWA (Table 1). The watersheds ranged in size from 18 km^2 (Jack Creek near Winfrey, AR USGS Site 07250974) to 1627 km^2 (Illinois River near Watts, OK USGS Site 07195500). The entirety of the period of record for each gage was used, with the longest continuous period of record being water years 1956 to 2021 (Illinois River near Watts, OK USGS Site 07195500). Some gages had gaps in their periods of record, such as Kings River near Berryville, AR (USGS Site 07050500) with a record of 1952 to 1975 and 1993 to 2021. The watersheds are primarily in Environmental Protection Agency (EPA) level 3 ecoregions Boston Mountains (38) and Ozark Highlands (39) (USEPA 2003).

Flow Statistics

Data were obtained for the 14 USGS stream gages using the NWIS (http://waterdata.usgs.gov). The average flow of each day (i.e., the mean daily discharge) from each gage was used to calculate each water year's maximum, minimum, mean, 10th percentile, 25th percentile, median, 75th percentile, and 90th percentile flow, and the number of days that had a mean flow meeting or exceeding the 1.01-year, 2-year, and 5-year flood event. These metrics will, hereafter, be referred to as the flow statistics.

The discharge for each return interval was calculated using a Log-Pearson III distribution. This distribution was chosen over a log-normal distribution because when both distribution types were plotted on log-normal and probability graph paper using the West Fork of the White River near Fayetteville data (USGS Site 07048550), the Log-Pearson III distribution fit the data better. Another reason this distribution was chosen is that it works for data with any skewness (Haan 1994). The Log-Pearson III distribution was used for each gage to maintain consistency, and the equation is:

$$\ln(X_t) = \ln(\overline{X}) * (1 + C_v K_t)$$

where X_t is the discharge of a flood with a t return period, $\overline{X}$ is the mean of the maximum yearly discharges, C_v is the coefficient of variation, and K_t is a frequency factor based on the return period, t, and coefficient of skewness, C_s.

$$C_v = \frac{\ln(\sigma)}{\ln(\overline{X})}$$

$$C_s = \frac{n\Sigma[\ln(X_i) - \ln(\overline{X})]^3}{(n - 1)(n - 2)\sigma^3}$$

where σ is the sample standard deviation of X, n is the number of water years, and X is the set of all observed maximum annual discharge. It should be noted that Log-Pearson III distribution equation used by Haan (1994) differs from USGS's Bulletin 17B Log-Pearson III distribution equation.

After the flows for each return interval were calculated, the Mann-Kendall test was used to determine if there was a trend with time in each of the flow statistics. The following steps were used to run each Mann-Kendall test:

1. List the specific flow statistics in chronological order, $x_1, x_2, \ldots, x_n$.
2. Determine if the difference $x_j - x_k$, called a pairwise comparison, is positive or negative, where $j > k$.

Table 1. Study site description including U.S. Geological Survey (USGS) gage name and number, latitude and longitude, watershed area, hydrologic unit code (HUC), ecoregion, and period of record used in stream flow analysis.

Gage Name	USGS Site Number	Latitude Longitude	Area (km^2)	HUC 8	Level 3 Ecoregion(s)	Period of Record
Flint Creek	07195800	36°15'22" 94°26'01"	38.5	11110103 Illinois	Ozark Highlands	1962-2021
Flint Creek	07195855	36°12'58" 94°36'19"	146.5	11110103 Illinois	Ozark Highlands	1980-2021
Frog Bayou	07250965	35°43'20" 94°06'49"	143.9	11110103 Frog-Mulberry	Boston Mountains	2001-2021
Illinois River	07194800	36°06'11" 94°20'40"	432.6	11110103 Illinois	Boston Mountains Ozark Highlands	2002-2021
Illinois River	07195500	36°07'48" 94°34'19"	1627.3	11110103 Illinois	Boston Mountains Ozark Highlands	1956-2021
Jack Creek	07250974	35°42'16" 94°05'30"	18.1	11110103 Frog-Mulberry	Boston Mountains	2002-2021
Jones Creek	07250935	35°44'09" 94°06'11"	53.1	11110103 Frog-Mulberry	Boston Mountains	2001-2021
Kings River	07050500	36°25'38" 93°37'15"	1366.2	11010001 Beaver Reservoir	Boston Mountains Ozark Highlands	1952-1975 1993-2021
Lee Creek	07249800	35°33'57" 94°31'55"	624.8	11110104 Kerr Reservoir	Boston Mountains	2000-2021
Mulberry River	07252000	35°34'37" 94°00'55"	966.0	11110201 Frog-Mulberry	Boston Mountains	1953-1995 1998-2021
Osage Creek	07195000	36°13'19" 94°17'18"	335.9	11110103 Illinois	Ozark Highlands	1953-1975 1996-2021
War Eagle Creek	07049000	36°12'00" 93°51'18"	684.1	11010001 Beaver Reservoir	Boston Mountains Ozark Highlands	1952-1970 1999-2021
West Fork	07048550	36°03'14" 94°04'59"	317.80	11010001 Beaver Reservoir	Boston Mountains Ozark Highlands	2002-2021
White River	07048600	36°04'23" 94°04'52"	1031.5	11010001 Beaver Reservoir	Boston Mountains Ozark Highlands	1963-1995 1999-2021

3. Compute S, where S equals total number of positive pairwise comparisons minus total number of negative pairwise comparisons.
4. Compute τ, where $\tau = S/[n(n-1)/2]$, n = number of data points.
5. Compute the standard deviation, σ_s, where $\sigma_s = \mathrm{sqrt}[(n/18)(n-1)(2n+5)]$.
6. Compute the Z score, $Z\tau$, where $Z\tau = (|S|-1)/\sigma_s$.
7. Determine the corresponding p value for $Z\tau$ based on a two-tailed standard normal distribution.

These steps were followed using Microsoft Excel for one site, and then automated using the programming language "R" with the tidyverse, rkt, and ggplot2 packages loaded from the R library.

Different α values were used to suggest different levels of significance. The α values were set to 0.01 for "highly significant" trends, 0.05 for "significant" trends, and 0.10 for "moderately significant" trends (Stogner 2000). The rate of change for each flow statistic was calculated using Theil-Sen Slope, which takes the median slope of the set of slopes between every combination of data points (Helsel et al. 2020). The Theil-Sen Slope was then converted into a percent change per year by dividing the Theil-Sen Slope by the mean value of the flow statistic.

Watersheds and Land Use Percentages

To obtain land use and land cover (LULC) data on each gage's watershed, the web toolkit Wikiwatershed and Model My Watershed (Stroud Water Research Center 2021) was used. The coordinates of each gage, as published by the USGS, were entered into Model My Watershed's search function. Often, this resulted in a location that was near, but not located exactly on, a bridge crossing over the stream. In such cases, it was assumed that the gage was on the bridge.

Once the exact location of the gage was determined, Model My Watershed was used to delineate the watershed of each gage. Model My Watershed reports LULC data from the National Land Cover Dataset (NLCD) for the delineated watershed. The oldest (2001) and newest (2019) NLCD data were used to calculate the land use percentages for each watershed and the land use change from 2001 to 2019 for each watershed.

The NLCD divides LULC into sixteen classifications. Those classifications were grouped into three basic LULC types to be analyzed. Open water, perennial ice/snow, deciduous forest, evergreen forest, mixed forest, shrub/scrub, woody wetlands, and emergent herbaceous wetlands were said to be "undeveloped." Barren land (rock/sand/clay), developed open space, low intensity, medium intensity, and high intensity were said to be "urban." Finally, pasture/hay (including grassland/herbaceous) and cultivated crops were said to be "agricultural" land use. A Human Development Index (HDI) was calculated by adding urban land use and agricultural land use percentages.

The percent change per year in each flow statistic with a moderate level of significance or greater ($\alpha < 0.10$) was paired with the land use percentages and the change in percentages in land use for each watershed, and linear regression was run using the Analysis ToolPak in Microsoft Excel. As with changes in the flow statistics, different α values were used to suggest different levels of significance, as previously defined.

Results and Discussion

Land Use and Changes

Based on the 2001 NLCD, the study watersheds had urban land use percentages ranging from 1.8% (Jack Creek near Winfrey, AR USGS Site 07250974) to 28.3% (Osage Creek near Elm Springs USGS site 07195000) with an arithmetic mean (hereafter referred to as average) of 7.9%. The agricultural land use in 2001 ranged from 4.6% (Mulberry River near Mulberry USGS site 07252000) to 60.6% (Flint Creek at Springtown, AR USGS site 07195800), with an average of 30.1%. When looking at combined human development, Osage Creek near Elm Springs had the highest HDI in 2001, in addition to the highest urban land use at 85.9%, while the Mulberry River near Mulberry (USGS site 07252000) had the lowest 2001 HDI at 7.5%. The average HDI was 38.0%, showing that in 2001 there was more undeveloped area on average across these watersheds than area manipulated by humans.

For the 2019 NLCD data, urban land use percentages ranged from 1.9% (Jack Creek near Winfrey, AR USGS site 07250974) to 42.3% (Osage Creek near Elm Springs USGS site

07195000) with an average of 9.9%. Agricultural land use in 2019 ranged from 5.5% (Mulberry River near Mulberry USGS site 07252000) to 62.3% (Flint Creek at Springtown, AR USGS site 07195800) with an average of 28.9%. HDI in 2001 ranged from 8.6% (Jack Creek near Winfrey, AR USGS site 07250974) to 87.7% (Osage Creek near Elm Springs USGS site 07195000) with an average of 38.7%. In 2019, as in 2001, the average watershed had less developed land (urban plus agriculture) at 38.7% than undeveloped land. The watershed with the maximum and minimum of each of the land use categories discussed was the same in 2019 as 2001, except for the minimum HDI occurring in the Jack Creek watershed instead of the Mulberry watershed.

Urban land use increased in all study watersheds from 2001 to 2019. Seven of the watersheds showed a small increase (< 1%) in urban land use, while three watersheds showed moderate increase (1.0 - 2.3%). The remaining two watersheds showed the largest increases in urban land use at 5.1% (Illinois River near Watts, OK USGS site 07195500) and 13.9% (Osage Creek near Elm Springs USGS site 07195000).

The agricultural land use from 2001 to 2019 generally decreased, with larger losses of 12.2% and 4.3% occurring in the watershed of Osage Creek near Elm Springs (USGS site 07195000) and Illinois River near Watts, OK (USGS site 07195500), respectively. The remaining watersheds had agricultural land use changes ranging from a decrease of 0.5% to an increase of 1.7%. The two watersheds with the largest increase in urban land use also had the largest decrease in agricultural land use, with the increase in urban land being very similar in magnitude to the decrease in agricultural land. These data suggest that urban development is primarily occurring in previously agricultural lands – not previously undeveloped lands. The same conclusion is drawn when examining the change in HDI.

The maximum change in HDI from 2001 to 2019 was 3.1% (Flint Creek at Springtown, AR USGS site 07195800), while all other watersheds had a change in HDI of 1.7% or less, including four watersheds with minor decreases in HDI (≤ 0.4%). The relatively low changes in HDI (and hence relatively low changes in undeveloped land) compared to the changes in urban and agricultural land suggest that urban development is occurring in land that was previously developed by humans (agricultural land) more than in existing undeveloped lands. In fact, the average increase in urban land use per watershed of 2.0% is likely due to an average 1.2% loss of agricultural land but only 0.8% loss of undeveloped land. However, watersheds with increased urban development have been estimated to have reduced ecosystem services and value, especially if HDI increases over time (Gashaw et al. 2018).

Flow Statistics

The changes overtime of 11 flow statistics at 14 sites were analyzed, showing 65 of the 154 possible changes to be at least moderately significant. All 65 of the at least moderately significant changes in the flow statistics were increases; no decreases were observed over the study period. Every gage that was analyzed had at least one flow statistic that increased with at least moderate significance.

Three sites had only one flow statistic that increased significantly over the period analyzed. Each of the three increasing flow statistics were related to high flows or flooding frequency. The 75th percentile in flows at Jones Creek at Winfrey, AR (USGS Site 070250935) increased by 5% per year from 2001 to 2021. The maximum flow at Flint Creek at Springtown, AR (USGS site 07195800) increased 0.8% per year from 1962 to 2021. The number of days where flows met or exceeded the 1.01-year flood at Lee Creek at Short, OK (USGS Site 07249800) increased 3.9% per year from 2000 to 2021.

Only two gages showed significant changes in the occurrence of the 2-year flood and 5-year flood. This is likely because the period of record that was analyzed was not long enough to show significant changes in such rare events. Because of this, the occurrences of the 2-year flood and the 5-year flood were not included in Table 3 or analyzed against watershed land use.

Three sites had at least moderately significant increases in every flow statistic. Osage Creek near Elm Springs (USGS Site 07195000) had highly significant changes in each of the flow statistics; however, its annual percent changes were moderate, ranging from 1.2% per year (75th

Table 2. Watershed land use percentages and change for each site from 2001 to 2019 (Undev is undeveloped land use, Urban is developed land uses, Agr is agricultural land use, and HDI = sum of Urban and Agr).

Name Site Number	NLCD 2001				NLCD 2019				2001-2019 Change			
	Undev	**Urban**	**Agr**	**HDI**	**Undev**	**Urban**	**Agr**	**HDI**	**Undev**	**Urban**	**Agr**	**HDI**
Flint Creek 07195800	32.9%	6.5%	60.6%	67.1%	29.8%	7.9%	62.3%	70.2%	-3.1%	1.4%	1.7%	3.1%
Flint Creek 07195855	32.3%	9.7%	58.0%	67.7%	30.7%	11.6%	57.8%	69.3%	-1.7%	1.8%	-0.2%	1.7%
Frog Bayou 07250965	89.0%	2.2%	8.8%	11.0%	89.2%	2.3%	8.6%	10.8%	0.2%	0.1%	-0.3%	-0.2%
Illinois River 07194800	40.4%	8.1%	51.5%	59.6%	40.0%	10.4%	49.7%	60.0%	-0.5%	2.3%	-1.8%	0.5%
Illinois River 07195500	33.2%	15.0%	51.8%	66.8%	32.4%	20.1%	47.5%	67.6%	-0.9%	5.1%	-4.3%	0.9%
Jack Creek 07250974	91.0%	1.8%	7.2%	9.0%	91.4%	1.9%	6.8%	8.6%	0.4%	0.0%	-0.4%	-0.4%
Jones Creek 07250935	88.4%	2.7%	8.9%	11.6%	88.6%	2.9%	8.6%	11.4%	0.2%	0.2%	-0.4%	-0.2%
Kings River 07050500	67.8%	4.9%	27.4%	32.2%	67.1%	5.1%	27.8%	32.9%	-0.6%	0.2%	0.4%	0.6%
Lee Creek 07249800	86.4%	2.8%	10.8%	13.6%	86.9%	2.8%	10.3%	13.1%	0.5%	0.1%	-0.6%	-0.5%
Mulberry River 07252000	92.5%	2.9%	4.6%	7.5%	91.2%	3.3%	5.5%	8.8%	-1.3%	0.4%	0.9%	1.3%
Osage Creek 07195000	14.1%	28.3%	57.6%	85.9%	12.3%	42.3%	45.4%	87.7%	-1.7%	13.9%	-12.2%	1.7%
War Eagle Creek 07049000	60.7%	5.1%	34.2%	39.3%	59.2%	5.5%	35.3%	40.8%	-1.5%	0.4%	1.1%	1.5%
West Fork 07048550	65.3%	13.5%	21.2%	34.7%	64.8%	14.5%	20.7%	35.2%	-0.5%	1.0%	-0.5%	0.5%
White River 07048600	74.7%	7.0%	18.3%	25.3%	74.3%	7.6%	18.1%	25.7%	-0.4%	0.5%	-0.2%	0.4%

and 90th percentiles) to 1.8% (minimum yearly flow). War Eagle Creek near Hindsville (USGS Site 07049000) and the Illinois River near Watts, OK (USGS Site 07195500) also had significant increases in each flow statistic. These two sites also had moderate annual percent changes ranging from 0.6% (Illinois River yearly maximum flow) to 1.4% (War Eagle Creek yearly minimum flow). Despite each of these watersheds having statistically significant increases across their flow regimes, the magnitude of increases were less than the significant increases of Frog Bayou at Winfrey (USGS Site 07250965), Illinois River at Savoy (USGS Site 07194800), Jones Creek at Winfrey, AR (USGS Site 07250935), and the West Fork of the White River East of Fayetteville (USGS Site 07048550). These sites had percent changes per year in various flow statistics ranging from 3.0 to 5.0% per year.

Of the sites that have several, but not all, significant increases in flow statistics, most significant changes were grouped in either high flows (75th percentile, 90th percentile, occurrence of the one-year flood, and yearly maximum flow) or low flows (yearly minimum flow, 10th percentile, and 25th percentile). Frog Bayou at Winfrey (USGS Site 07250965) and the West Fork of the White River East of Fayetteville (USGS Site 07048550) had significant increases in high flows. Flint Creek near West Siloam Springs (USGS Site 07195855), Illinois River at Savoy (USGS Site 07194800), and Jack Creek near Winfrey (USGS Site 07250974) had significant increases in low flows. The cutoff between high and low flows was the median flow, and median flow was the statistic that had the least number of significant changes (3 sites out of 14).

Relationship between Flow Statistics and Land Use

Annual percent changes in the flow statistics that were at least moderately significant were compared with several land use measures in their watersheds: percent urban in 2019, change in percent urban from 2001 to 2019, and HDI in 2019. The change in HDI from 2001 to 2019 was not included because the changes were relatively small compared to the other land use measures, as previously discussed.

The maximum yearly flow had a significant relationship with all three different land use statistics tested. The percent change per year in maximum yearly flow was significantly positively correlated to urban percent change from 2001 to 2019 ($p = 0.04$), as shown in Figure 1. The percent increase per year in maximum flows ranged between 0.2 and 1.5%, while the urban percent increase ranged from 0.2 to 13.9%. The slope of the relationship was 0.068, suggesting that increasing urban land use by 1% corresponds to a 0.068% increase per year in maximum flows. This relationship had one gage (Osage Creek near Elm Springs USGS Site 07195000) with a percent change per year in maximum flow and change in urban area of its watershed that were notably higher than those of every other gage in the comparison.

The percent change per year in maximum flow was also significantly positively correlated to the urban percentage of its watershed ($p = 0.04$). The range in percent changes per year in maximum flows was previously noted, while urban land use in 2019 ranged from 3.3 to 42.3%. The slope of the relationship was 0.024, suggesting that a 1% increase in urban land use from one watershed to another corresponds to a 0.024% increase per year in maximum flows. Again, this relationship had one gage with a percent change per year in maximum flow and percent urban area of its watershed in 2019 that were notably higher than those of every other gage in the comparison (Osage Creek near Elm Springs USGS Site 07195000).

Lastly, the percent change per year in maximum flow was significantly positively correlated to the 2019 HDI of its watershed ($p = 0.04$). The same percent changes per year in maximum flows were compared to 2019 HDI, which ranged from 8.8 to 87.7%. The slope of the relationship was 0.013, suggesting that a 1% increase in HDI from one watershed to another corresponds to a 0.013% yearly increase in maximum flows. This relationship had a well spread distribution of percent change in maximum flow and 2019 HDI.

The percent change per year in minimum flow was positively correlated to the HDI of its watershed in 2019 with moderate significance ($p = 0.06$). The percent change per year in minimum flow ranged from an increase of $< 0.1\%$ to an increase of 2.4%. The slope of the relationship was 0.021, suggesting that a 1% increase in HDI

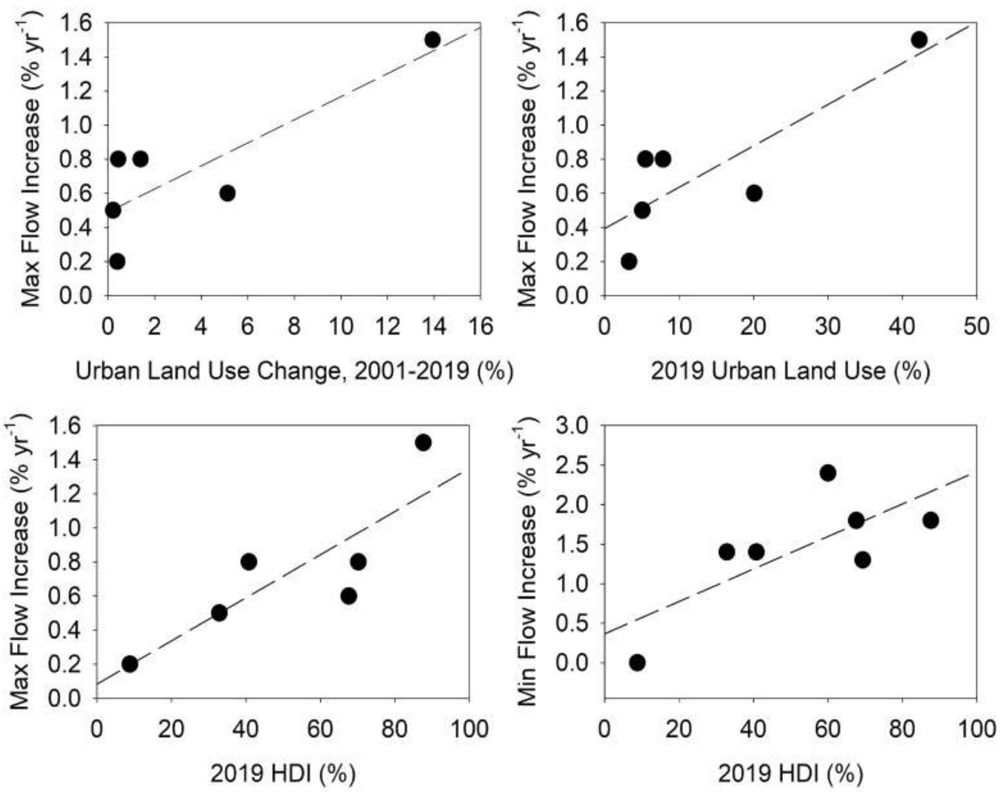

Figure 1. Significant relationships between percent change per year in minimum and maximum flow and watershed land use, including urban, urban plus agricultural (Human Development Index, HDI), and change in urban land use.

from one watershed to another corresponds to a 0.021% yearly increase in minimum flows. This relationship had one gage with a yearly percent change in minimum flow that was notably less than the rest of the gages (Mulberry River near Mulberry USGS site 07252000).

Low Flows

The initial objective of this study was to investigate high flows and flooding; however, we also found interesting trends in low flows. Various ideas exist about the effect of urbanization on baseflow in streams. One idea is that increased groundwater pumping (although not common in NWA) and decreased groundwater recharge caused by more impervious surfaces decrease baseflows (Brown et al. 2005); however, this was not observed in the study site region, as none of the analyzed gages had significant decreases in low flows (minimum, 10th percentile, and 25th percentile). It is assumed that minimum flow, 10th percentile flow, and sometimes 25th percentile flow represented baseflow conditions in these streams.

Another idea is that as populations in urban areas increase, wastewater treatment plant (WWTP) effluent can increase more than groundwater recharge decreases, therefore increasing baseflow in streams (Paul and Meyer 2001). Five out of the nine gages with significant increases in low flows have at least one if not multiple WWTPs in their watershed (Illinois River at Savoy USGS Site 07194800, Illinois River near Watts, OK USGS Site 07195500, Kings River near Berryville USGS Site 07050500, Osage Creek near Elm Springs USGS Site 07195000, and War Eagle Creek near Hindsville USGS Site 07049000).

The four gages with the largest percent increase per year in minimum flow (Illinois River at Savoy USGS Site 07194800, Illinois River near Watts, OK USGS Site 07195500, Osage Creek near Elm Springs USGS site 07195000, and Kings River near Berryville USGS site 07050500) all have at least one if not multiple WWTPs in their watersheds. As the population of the NWA

Table 3. Percent increase per year of annual flow statistics and days exceeding defined flood flows showing those that are moderately significant$^{\#}$ ($p < 0.1$), significant$^{\$}$ ($p < 0.05$), and highly significant* ($p < 0.01$), and the 1.01-year flood flow (as calculated).

Name Site Number	Min	Mean	Max	Percentiles 10th	 25th	 Median	 75th	 90th	1.01-Yr Flood Days	1.01-Yr Flow (cfs)
Flint Creek 07195800			0.8%$^{\#}$							53
Flint Creek 07195855	1.3%$^{\#}$			1.2%$^{\#}$	1.2%$^{\#}$					194
Frog Bayou 07250965							4.8%$^{\$}$	3.5%$^{\$}$	5.4%$^{\$}$	750
Illinois River 07194800	2.4%$^{\#}$			3.6%$^{\$}$	3.7%$^{\#}$					1718
Illinois River 07195500	1.8%*	0.7%$^{\$}$	0.6%$^{\#}$	1.2%*	1.0%*	0.6%$^{\$}$	0.6%$^{\#}$	0.6%$^{\#}$	0.8%$^{\#}$	3360
Jack Creek 07250974				<0.1%$^{\#}$	4.6%$^{\#}$					71
Jones Creek 07250935							5.0%$^{\$}$			107
Kings River 07050500	1.4%*		0.5%$^{\#}$	0.8%$^{\$}$	0.7%$^{\#}$					3906
Lee Creek 07249800									3.9%$^{\#}$	2750
Mulberry River 07252000	<0.1%*	0.5%*	0.2%$^{\$}$	0.5%*	0.5%$^{\#}$		0.5%$^{\$}$	0.6%$^{\$}$	0.5%$^{\$}$	3086
Osage Creek 07195000	1.8%*	1.4%*	1.5%*	1.6%*	1.6%*	1.3%*	1.2%*	1.2%*	1.6%*	408
War Eagle Creek 07049000	1.4%*	0.9%*	0.8%$^{\$}$	1.0%*	0.7%$^{\#}$	0.7%$^{\$}$	0.9%*	0.9%*	1.1%*	2069
West Fork 07048550		3.0%$^{\$}$					3.4%$^{\#}$	3.6%$^{\$}$	3.8%$^{\#}$	1566
White River 07048600					1.3%$^{\$}$		0.6%$^{\#}$			4611

metropolitan area increased from 347,045 in 2000 to 546,725 in 2020 (United States Census Bureau 2022), WWTP effluent discharges have increased to meet the needs of the growing population. For example, Northwest Arkansas Conservation Authority (NACA) WWTP obtained permits to increase effluent from 0.5 mgd (0.8 cfs) to 3.6 mgd (5.5 cfs) in 2009 and then to 7.2 mgd (11.2 cfs) in 2021 (ADEQ 2009; Smoot 2021). Even though there is a moderately significant, positive correlation between minimum flow change and 2019 HDI, the actual cause of the minimum flow increase is likely increased WWTP effluent, not watershed land use. The significant relationship between minimum flow change and 2019 HDI is likely attributed to the fact that HDI and WWTP effluent are both influenced by population growth. It should be noted that potable water for NWA comes from Beaver Lake, which is part of the White River Basin, but then is mostly discharged from WWTPs into the Illinois River watershed, which is essentially an inter-basin transfer of water.

It is interesting to note that urban watersheds that do not receive WWTP effluent but had increases in low flows, thus other factor(s) besides WWTP effluent must outweigh decreases in infiltration due to increased impervious surfaces. Another set of possible factors that would increase low flows is leakage from waterlines, sewers, and septic systems (USEPA 2022). These factors are most applicable in areas with increasing populations, such as the watershed of Flint Creek near West Siloam Springs (USGS Site 07195855) in which water use and wastewater increased, possibly increasing leakage. In Arkansas, 38% of households use septic tanks, although this percent is likely less in the NWA metropolitan area. But, with a failure rate of 10-20% (USEPA 2002), even a smaller percentage of households using septic tanks could increase groundwater and return flows to streams. Although septic tank failure is primarily a water quality issue, it also has an impact on the quantity of soil water and groundwater, and therefore baseflow.

For watersheds that do not have WWTP effluent discharge or a high population causing significant water/wastewater system leakages but do have increases in low flows, the most likely cause of increasing minimum flow is an increase in rainfall that leads to increased infiltration and groundwater recharge. This is consistent with a 2003 study in Iowa (Schilling and Libra 2003), which found increasing rainfall contributed more to streamflow as baseflow than it did as runoff. This could have occurred in the Mulberry River near Mulberry (USGS site 07252000) and Jack Creek near Winfrey (USGS site 07250974). In NWA, total yearly rainfall, based on water year as measured at Drake Field in Fayetteville (NWS 2022), has increased with moderate significance in the long term (1952-2021, $p = 0.097$) and the near term (2002-2021, $p = 0.081$). Increased rainfall could be a factor in low flow increases in all analyzed gages, not just the three gages listed above (McCabe and Wolock 2002; Rumsey et al. 2015).

High Flows

Of the gages analyzed, seven gages had at least moderately significant increases in the number of days with flow that met or exceeded the 1.01-year flood and three gages that had at least moderately significant increases in the number of days with flow that met or exceeded the 2-year flood. This has a large impact on channel morphology, as channel forming flow generally corresponds to the 1- to 3-year flood and most closely corresponds to the 1.5-year flood (NRCS 2001; Colorado Water Conservation Board 2006). However, frequencies of these smaller flood events (i.e., bank-full events) might be better analyzed using partial-duration flood series to better understand the occurrence of these events (Edwards et al. 2019).

These increases need to be monitored and controlled because uncontrolled channel morphology can have negative socioeconomic and ecological impacts (Hauer et al. 2011; Abubakar 2013). Such impacts include loss of agricultural land, destruction of utilities, and the alteration and/or destruction of aquatic habitats (Abubakar 2013). Increased erosion of stream banks increases phosphorous loadings because phosphorous is in the streamside soil and is often adsorbed to sediments (Son et al. 2011). Also, the destruction of riparian zones reduces the filtration of phosphorous before it reaches the streams (Tillery et al. 2003). Increased phosphorous loading leads to increased algae blooms and accelerated eutrophication (Tillery et al. 2003; Son et al. 2011). Additionally, as channels move from their natural floodplains, the effects of flooding are amplified due to decreased floodwater buffering and absorption (Pierce et al. 2012; Mondal and Patel 2018). Potential mitigation strategies include restoring riparian buffers, mechanical bank stabilization, and limiting human activity that increases high flows (Harmel et al. 1999; Abubakar 2013), though Mondal and Patel (2018) write that ecological approaches have grown in popularity over artificial mechanical stabilization methods.

The flood frequency analysis yielded a flow for each recurrence interval that is representative of the likelihood that the flow was met or exceeded in any one year based on the period of record. It should be noted, however, that the flood frequency analysis used to calculate the 1.01-, 2-, and 5-year flood flows were based on annual maximum flows, some of which had significant increases over time across selected streams. This means it is likely that the flows associated with these return intervals have increased over time across these sites. This is acceptable for the purposes of this study because the calculated 1.01-, 2-, and 5-year floods were used as thresholds to measure the number of days that met or exceeded those flows; they were not used to predict the likelihood of future flood events. The flow associated with a certain recurrence interval can increase over time due to increased large storm events, climate change, and urbanization (Raff et al. 2009).

Percent change per year in maximum flow was significantly positively correlated to urban land use and HDI in 2019, as well as change in urban land use from 2001 to 2019. Maximum flows show a greater response to increased rainfall in urban-dominated watersheds than rural watersheds (Changnon and Demissie 1996). Both urban-related relationships can be explained by increased runoff-related flow due to increased impervious surfaces, alteration and reduction of vegetation which decreases initial abstraction, and drainage systems that reduce the time it takes runoff to reach streams (USGS 2019). HEC-HMS models have been used to show that increases in streamflow are directly proportional to the rate of urbanization (Amini et al. 2011). It makes sense that urban development and other changes in a watershed produce changes in flow at the mouth of the watershed. A reason that could explain why the percent of urban land use of a watershed at a single point in time was a good predictor of change in maximum flows is that, as discussed previously, the rate of runoff due to increasing precipitation is amplified by urban land use (Changnon and Demissie 1996).

The creation of urban lands is not the only way humans develop landscapes. This study's HDI is comprised of urban and agricultural land use, bringing in the influence of pastures and agricultural land management on changes in stream flow statistics. The strong relationship between 2019 HDI and change in maximum flows is likely due to changes in soil quality and compaction and changes in vegetation in agricultural lands (O'Connell et al. 2007) in addition to the factors caused by urban changes. Undeveloped forests have greater infiltration rates than cultivated fields or grazed pastures (Bharati et al. 2002), meaning a greater amount of precipitation that falls on agricultural land becomes runoff and can contribute to maximum flows than precipitation that falls on undeveloped forest land.

The Osage Creek near Elm Springs (USGS site 07195000) watershed has more than double the 2019 urban land use and change in urban land use than those of the next highest analyzed watersheds. Also, its 2019 HDI is 18% higher than the watershed with the next highest 2019 HDI. For these reasons, it is no surprise that that gage at Osage Creek near Elm Springs showed the largest percent change per year in maximum flows (of those changes that were at least moderately significant) and had highly significant increases in all analyzed flow statistics, except for the number of days with flow exceeding the 5-year flood, which was significant, not highly significant. Additionally, the watershed collects effluent discharge from three major WWTPs: NACA, Rogers, and Springdale, which helps to explain the highly significant increases in low flows in this watershed.

Conclusion

While analyzing changes in flows across their flow regimes at various gages in NWA and northeast Oklahoma, along with the land use in their watersheds, the following conclusions were made:

- All analyzed gages had one or more flow statistics with at least a moderately significant increase, and all flow statistics increased at least moderately significantly at two or more gages.

- There were no decreases of any significance in any flow statistic at any gage.
- In general, the development of urban land did not happen on undeveloped land, but instead happened on land that was previously used for agriculture.
- Increases in yearly maximum flows were positively significantly correlated to 2019 urban land use, 2001 to 2019 change in urban land use, and 2019 HDI.
- Increases in yearly minimum flows were positively correlated to 2019 HDI with moderate significance.
- The growing urban areas need to consider how increasing streamflow influence bank stability and potential flooding and frequency downstream.

The increase in maximum flows and the occurrence of certain floods is concerning because of floods' damage to human life, property, and ecosystems. Knowing the relationships between flooding, flood frequency, land use, and changes over time could help city officials in NWA plan and regulate land development changes in ways that mitigate flooding.

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Case Study Article

On-field Agroecosystem Research Experience: An Undergraduate Perspective

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Abstract: Undergraduate hands-on research can foster innovation and critical thinking among young scholars to delve into real-world challenges. Specifically, exploring the critical nexus between water usage and agricultural yield, can foster academic growth and holds the key to addressing global food security in an era of increasing environmental constraints, where students can unlock insights crucial to enhancing crop yield and sustainability. Investigating the intricate relationship between water management and crop productivity through undergraduate research is exemplified in this article. Undergraduate students acquired hands-on research experience by collecting, processing, and analyzing destructive (crop biomass samples) and non-destructive (plant height, nodes, and leaf chlorophyll content) cropping system data on soybeans under irrigated and dryland production systems, where they worked closely with the farmer. Identifying the current research problem and study site selection, scientific decision-making during the field study, developing critical thinking while ensuring research communication skills, and quality assurance and quality control through technology during data collection and analysis were learning outcomes. The research highlights the observed distinct performance between irrigated and non-irrigated soybeans using non-destructive plant health and growth indicators with plant biomass, following appropriate quality control and assurance steps. Statistically, irrigated soybeans outperformed non-irrigated soybeans in terms of average plant height at maturity (irrigated: 97.0±1.7 cm vs. non-irrigated: 37.4±0.6 cm; $p<0.01$) and number of nodes on the mainstem (irrigated: 19.5±1.2 vs. non-irrigated: 12.6±0.8; $p<0.01$). Findings from this study can help ensure quality control and assurance in future cropping system projects. Through the agroecosystem study, we exhibit the importance and role of undergraduate research opportunities in developing the next generation of problem solvers.

Keywords: *undergraduate research, quality assurance, quality control, non-destructive, destructive, irrigated soybeans, non-irrigated soybeans*

Undergraduates interested in careers in scientific research should have real-world research experience and hands-on training (Thiry et al. 2012). Nevertheless, many students face the challenge of deciding what to do after graduation because they still need the technical skills required in the job market (Fortenberry 1993; Sabatini 1997). The demand for workforce and market skills prompted the development of a research-based course to encourage the involvement of young undergraduate research enthusiasts (Cavanagh et al. 2016). Various student research experiences have enhanced undergraduates' performance and interest in science, technology, engineering, and mathematics (STEM) research across the United States (Bruthers et al. 2021). Researchers and mentors believe that students would benefit from research experience, but they have yet to find the best ways to orient and guide them. It would be

Research Implications

- Undergraduate students benefit from research opportunities in on-field agroecosystem and water use efficiency.
- Soybean health and vegetative growth were significantly greater in irrigated areas of the field than in non-irrigated areas.
- Proper quality assurance and quality control, including photographs and audio recordings, can assist in eliminating errors in agricultural research.
- Learning outcomes from agroecosystem research can shape undergraduates' future research interests and enhance their problem-solving capabilities.

helpful to have an orientation that balances students' attitudes and expectations with the realities of the research experience. As they prepare to transition to graduate study or jobs, enthusiastic students bring an energy of curiosity and eagerness to learn through hands-on training in a real scenario (Adebisi 2022).

The learning activities include discussions with mentors, participation in group meetings, supervised opportunities to explore pertinent research material, and reflection on observations. Students learn scientific techniques such as research planning, modeling scientific breakthroughs, and data analysis through undergraduate research initiatives. Mentors should ideally assist students in assessing the credibility of scientific research and connecting their experiences to their expectations. Undergraduate research advances our understanding of critical issues like optimizing water use for agricultural yield. Hands-on training investigating the intricate relationship between water management and crop yield can become a foundation for understanding the necessity of sustainable agriculture.

Globally, soybean (*Glycine max L.* (Merr.)) is the first-grown legume and fifth most-grown crop (Boote et al. 1998; Kothari et al. 2022). However, climate change, rapid population growth, and high food demand have contributed to the complexity of the agroecosystem, the exploration of natural resources, and their adaptation to global environmental changes (Asseng et al. 2015; Jones et al. 2017). The complexity of agricultural systems related to the change in water use makes it quite tricky to define entire crop system operations in mathematical terms (Jones et al. 2017; Zhao et al. 2019). To fully visualize, represent, and predict the future yield of significant crop growth, crop simulation models have been developed for the past 40 years to assess the ability of farming systems to meet the world's food demand (Zhao et al. 2019). Previous studies highlighted the variation in future soybean yield prediction from different crop models and the limited understanding of soybean biological and mechanical processes in response to climate change (Belcher et al. 2004; Jägermeyr et al. 2021).

Through an undergraduate research opportunity, we collected ground truth destructive and non-destructive soybean data from irrigated and non-irrigated fields to quantify soybean growth and development in response to water usage, where we compared irrigated versus non-irrigated (rainfed) systems. Our undergraduate research-learning goal was to ensure the high quality of generated destructive and non-destructive data through scientific experimental design and developing critical thinking. We hypothesized that water usage can be very critical in crop production. We examined the nexus between water use and agricultural yield, a pivotal area for undergraduate research that can play a transformative role. With the undergraduate perspective linking water and yield through agroecosystem research experience, we had exposure to decision-making. We identify the need for high-quality destructive and non-destructive data with proper control and assurance and the critical role of communication skills for a successful undergraduate research project.

Materials and Methods

Study Area Description

This study collected destructive and non-destructive soybean data from a farmer's field in Sutherland, Nebraska (41.06, -101.15) in 2022 (Figure 1). The research area (56.8 hectares) was chosen to encompass a variety of soil types such that both irrigated and non-irrigated systems were covered (Figure 1A). The irrigated areas covered

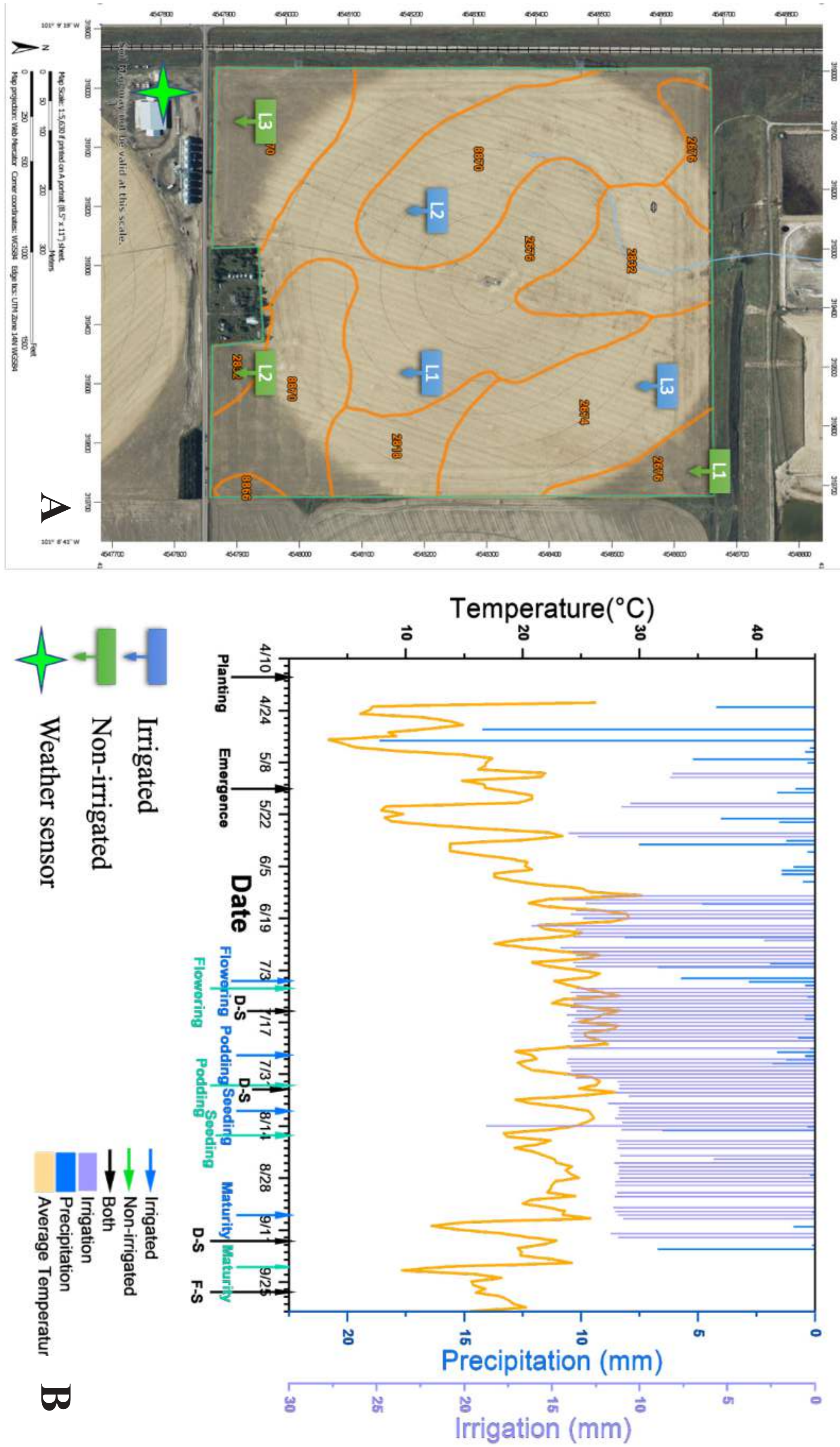

Figure 1. Study area (A) with different soil types (https://websoilsurvey.sc.egov.usda.gov) (left panel), weather and irrigation timeline (B) (right panel).

around 84% of the total field (47.3 hectares). In contrast, the non-irrigated area was the remaining land outside the center pivot radius, which covered approximately 16% of the total field (9.5 hectares). Localized weather stations were installed within 0.8 km from the field's center. Modern and cutting-edge technology, such as the availability of arable sensors (served as weather stations and provided both irrigation and precipitation water depths), soil type variability (Table 1), variable rate irrigation systems, and historical fertilizer and irrigation data, were available at the study sites. These datasets were critical in developing and verifying the cropping system model. However, our role was primarily to collect ground truth data in the farmer's field and link water use with observed yield while ensuring the proper quality of the collected data.

Non-destructive Plant Health and Growth Indicators

Irrigated and non-irrigated separate locations (Table 1) were selected based on soil types to represent experimental replicates. At each location, non-destructive (sampling without causing damage to the plants) data were collected from four randomly independent 1-m rows twice a week. At each representative location and 1-m row, coordinates were recorded, and locations (selected to represent major soil types) were named (location number and row number: e.g., L1-R1; L1-R2; … L1-R3; L3-R4).

For non-destructive sampling events, each calendar date of the visit, the planting date, the plant emergence date (typically takes around ten days, depending on soil temperature and moisture), the beginning pod date (when pods size were 5 mm long at one of the four uppermost nodes), the full pod date (when the pods' sizes were 2 cm at one of the four uppermost nodes), the flowering date (identified when plants have at least one flower on any node), and the full flowering date (identified when plants have an open flower at one of the two uppermost nodes) were noted.

Table 1. Study area map unit symbols with irrigation status, corresponding soil types, and area of interest (AOI) coverage (https://websoilsurvey.sc.egov.usda.gov).

Map Unit Symbol	Map Unit Name	Hectares in AOI	Percent of AOI
2674 (***Irrigated***)	Holdrege silt loam, 1 to 3 percent slopes, plains, and breaks	8.4	14.8%
2676 (***Non-irrigated***)	Holdrege silt loam, 3 to 7 percent slopes, eroded, plains, and breaks	19.3	34.0%
2818 (***Irrigated***)	Uly silt loam, 3 to 6 percent slopes, eroded	3.0	5.3%
2832 (***Irrigated***)	Uly-Coly silt loams, 6 to 11 percent slopes	5.5	9.6%
8866 (***Non-irrigated***)	Hord silt loam, 0 to 1 percent slopes, warm	0.3	0.5%
8870 (***Irrigated and Non-irrigated***)	Hord silt loam, 1 to 3 percent slopes	20.3	35.7%
	Totals for Area of Interest (AOI)	**56.8**	**100.0%**

The number of mainstem nodes in soybeans was frequently counted to evaluate how the plant grows vegetatively. The number of nodes on the mainstem was recorded on five to eight randomly selected plants in each randomly selected row in both irrigated and non-irrigated areas. Node 0 are the two cotyledon nodes, whereas node 1 is where the two unifoliolate leaflets are joined. All other nodes above the unifoliolate leaflets are numbered 2, 3, and so forth and hold trifoliolate leaflets (Kranz and Specht 2012).

Single-sided meter sticks (Eisco™, U.S.) were used to measure plant height from the cotyledonary node to the tip of the apex on five plants within a 1-m row. Portable SPAD 502 Plus Chlorophyll Meters (Spectrum Technologies Inc., U.S.) were used to measure plant leaves' health and nitrogen concentration in SPAD values (Ling et al. 2011), which were later converted into leaf chlorophyll concentration per unit area (Markwell et al. 1995). The calculated chlorophyll concentrations have been reported in Table 2.

Multiple photographs and audio recordings were captured at each stage for quality assurance and quality control (QA/QC) of the collected data. Upon capturing the images and recordings, we conducted a thorough review to identify any errors or typos, ensuring the accuracy and reliability of the data collection. Documenting this information in the field contributed to developing critical thinking skills in managing large datasets generated during agricultural research.

Destructive Plant Sampling

Monthly destructive samples were collected after full flowering from previously identified, marked, non-destructively sampled sites and whole plants were removed from three 1-m rows. At the time of harvest, three 2-m rows were sampled at each of the three locations. There were between 16 and 20 soybean plants in each 1-m row. All plants (including roots, stems, leaves, petioles, pods, and seeds) were entirely removed from the soil and placed in Ziploc® bags to be processed at the University of Nebraska-Lincoln laboratory. The date, location, and row number were meticulously labeled on each Ziploc® bag. Labeled Ziploc® bags containing samples were placed in insulated coolers containing ice packs to ensure sample freshness during travel to the laboratory. All leaves were separated from the mainstems for each plant and placed on aluminum foil. Three randomly selected leaves were taken from each pool of leaves from every plant and photographed alongside a Single Sided Meter Stick. Each leaf was individually folded in a pre-weighed aluminum foil with proper identification. Photographs were taken for QA/QC purposes, and foils were labeled with the date, location number, row number, plant number, and leaf number. The remaining leaves were bagged and folded in large aluminum foil, labeled with the date, location, row, and plant number, and identified as "pooled leaves."

All stems were separated, cut into 10 to 20 cm segments (from soil emergence or where stem color transitions from green to white), then labeled with the date, location number, row number, and plant number. Next, "pooled stems" were placed in a sizeable pre-weighed aluminum foil bag. Roots were discarded as they were not needed in this study. For seeds and pods per plant, the total number of pods and seeds per plant was counted and measured with a weighing balance (NTEP Precision BAL 620G 10MG, U.S.). The samples were bagged separately and identified as "pod samples" and "seed samples," respectively. Pooled leaves, stems, pod samples, and seed samples were weighed for fresh weights and placed in a drying oven (Fisher Scientific Isotemp General Purpose Heating and Drying Ovens, U.S.) at 65 °C. Dry weights were recorded until the constant dry weights (no change in dry weights) were achieved, and dry weights of each sample were recorded while keeping account of the labeling information.

Statistical Analysis

Statistical analyses were done in OriginPro Version 2023b (OrginLab Corporation, Northampton, MA, U.S.). The generated plant height and leaf chlorophyll concentration data were tested for normal distribution using the Shapiro-Wilk test followed by mean significance difference (two-sample paired t-test) between irrigated and non-irrigated. The mean difference in the number of nodes was calculated using the Wilcoxon rank sum test (elimination of normal distribution assumption).

Table 2. Average daily plant height (cm), number of nodes, and SPAD readings for both irrigated and non-irrigated sampling zones.

Sampling Days	Irrigated				Non-Irrigated			
DD/MM/YY	Height (cm)	Number of Nodes	SPAD	Chlorophyll (μmol m^{-2})	Height (cm)	Number of Nodes	SPAD	Chlorophyll (μmol m^{-2})
5/20/22	3.6±0.4	1.3±0.1	36.2±0.6	388.9±7.5	3.3±0.2	1.5±0.2	36.6±0.8	395.7±8.8
5/27/22	5.4±0.4	2.1±0.2	38.3±1.3	425.4±11.8	4.8±0.2	2.0±0.2	37.1±2.5	403.1±18.8
5/31/22	7.5±0.2	2.0±0.2	38.0±1.6	420.0±13.6	7.1±0.5	1.8±0.3	40.0±1.5	454.9±13
6/2/22	9.4±0.1	3.2±0.3	39.2±0.8	440.8±8.8	8.9±0.2	3.3±0.1	40.1±1.6	456.7±13.6
6/10/22	13.2±0.9	5.0±0.0	40.5±0.3	464.2±5.3	10.8±0.3	4.2±0.4	43.6±0.9	524.6±9.4
6/14/22	16.6±0.5	6.9±0.1	41.5±0.7	483.6±8.1	11.8±1.2	5.4±0.3	43.2±0.1	516.6±3.5
6/22/22	21.3±0.8	6.3±0.2	40.3±0.8	461.0±8.8	14.3±0.6	6.3±0.2	41.9±0.8	490.7±8.8
6/28/22	23.5±0.5	6.9±0.1	41.5±0.3	483.6±5.3	19.1±0.1	7.6±0.2	39.9±0.2	453.6±4.5
6/30/22	26.7±0.4	7.5±0.2	38.5±0.7	428.2±8.1	20.2±0.2	8.3±0.4	40.3±0.2	461.0±4.5
7/6/22	36.1±1.2	8.8±0.3	39.4±1.1	445.2±10.6	26.0±0.6	9.5±0.2	40.8±1.0	471.1±10.0
7/8/22	40.2±1.6	9.8±0.3	41.1±0.4	476.7±6.1	29.0±0.7	9.8±0.2	40.3±0.3	461.7±5.3
7/19/22	58.7±1.9	11.2±0.9	41.7±1.0	488.2±10.0	32.9±0.4	10.8±0.1	43.2±0.8	517.4±8.8
7/26/22	72.8±5.2	12.2±0.2	42.4±1.1	500.9±10.6	31.9±2.2	11.0±0.4	43.6±0.3	525.4±5.3
7/28/22	74.5±1.3	13.5±0.3	41.1±0.4	476.4±6.1	34.4±0.3	11.9±0.1	44.5±0.8	543.1±8.8
8/3/22	91.0±4.3	17.7±1.7	42.7±1.4	507.6±12.4	37.0±1.6	11.9±0.7	44.5±1.0	542.7±10
8/24/22	97.0±1.7	16.7±0.8	46.6±0.4	585.7±6.1	37.0±0.7	13.2±0.6	43.5±0.4	522.0±6.1
9/14/22	91.8±4.5	19.5±1.2	19.4±4.8	156.3±32.8	37.4±0.6	12.6±0.8	32.9±2.1	334.0±16.5

Results and Discussion

Increased soybean production has resulted from improved genetic potential (Vogel et al. 2021). However, in many places, production is diminished by water stress as their grain yield is linearly correlated to water usage (Sharda et al. 2019). Soybean water demand and usage have been linked to growth stage, soil type, and weather conditions. Generally, 381 to 635 mm of water is required to grow soybeans (Kranz and Specht 2012). In this study, irrigated soybeans received irrigation and precipitation water (968 and 112 mm, respectively), while non-irrigated soybeans only received precipitation water (112 mm).

Plant Growth and Plant Health Indicators

Seasonally, plant growth indicators (plant height and the number of nodes on the mainstems) were measured to monitor the difference in growth between irrigated and non-irrigated soybeans. Likewise, via SPAD, nitrogen status was measured to assess soybean's greenness as a plant health indicator. Plant height from emergence to harvest day (136 days) varied between irrigated and non-irrigated soybeans, but irrigated soybeans eventually surpassed non-irrigated soybeans, with the difference being significant ($p<0.01$) after day 24 from emergence. Irrigation and precipitation are essential factors in increasing yields and improving plant health. The growth trend for irrigated and non-irrigated soybeans was similar until July 8th, 2022, when the growth height for irrigated soybeans dramatically increased until harvest day (Figure 2A). The variance in plant height was induced by irrigation and precipitation volume received by irrigated soybeans (1,080 mm) and non-irrigated soybeans (112 mm). Overall, mean irrigated soybean plant heights (40.6±1.5 cm) were significantly greater ($p<0.01$) than non-irrigated plant heights (21.6±0.6 cm) for the entire season. Generally, the relationship between irrigation water and sufficiently bringing plants through several growth stages of development explained the variance in plant height.

Irrigated soybeans had more nodes than non-irrigated soybeans, suggesting more vegetative growth. Irrigated soybean nodes ranged from 15 to 20 nodes at maturity, while non-irrigated soybean nodes ranged from 10 to 15 (Figure 2B). The overall mean number of nodes was significantly greater ($p<0.01$) in irrigated soybeans (8.9±0.4) than in non-irrigated soybeans (7.8±0.3).

The SPAD meter, an instrument to measure plant nitrogen content, has been widely used in agricultural and research applications to assess plant nitrogen status (Ling et al. 2011). When comparing irrigated and non-irrigated areas, adequate water supplementation (irrigation and precipitation) benefit irrigated areas over non-irrigated areas in terms of vegetative growth and health (plant nitrogen concentration) (Wang et al. 2021). However, our findings show that soybean leaf chlorophyll concentration, as assessed by a portable SPAD meter, was not influenced by the amount of water applied to the field. Throughout the growing season, chlorophyll content was comparable in irrigated and non-irrigated soybeans, with averages ranging between 391 and 586 µmol m^{-2} before maturity (Figure 2C). Post-harvest, irrigated soybeans' SPAD measurements rapidly declined after maturity, which can be explained by chlorophyll breakdown during leaf senescence (Markwell et al. 1995; Hörtensteiner and Kräutler 2011). Overall, the maturity time difference and growth stages are linked to varying SPAD measurements (Ma et al. 1995). The harvest data concurred with the findings of plant health indices.

Crop Yield Difference between Irrigated and Non-irrigated at Harvest

The average final harvest weight of fresh seed, dry seed weight, and the number of seeds per 2-m row varied between irrigated and non-irrigated systems (Figure 3). Generally, soybean grain yield in the U.S. has increased as the total irrigated land has increased (Irwin et al. 2017). In our study, the total number of seeds harvested was 7.5 times greater in irrigated (5117±409 seeds) than in non-irrigated soybeans (680±180 seeds) ($p<0.01$). Fresh weight of seeds differed considerably ($p<0.01$), with seed weight (972±190 g) from irrigated soybeans being 11.7 times that of non-irrigated soybeans (83±24 g). Seed dry weight followed the same pattern ($p<0.01$), with irrigated soybean seed dry weight (738±72 g) being 12.6 times that of non-irrigated soybeans (58±19 g). The results from this study were expected based on the observed non-destructive plant health indices measured throughout the growing season. We did observe systemic differences in all destructive sampling post-flowering (data not shown here), which predicated the difference observed at harvest. Irrigated soybeans matured two weeks earlier than non-irrigated soybeans, which accounts for the significant difference between the two soybean systems' final harvests of destructive samples.

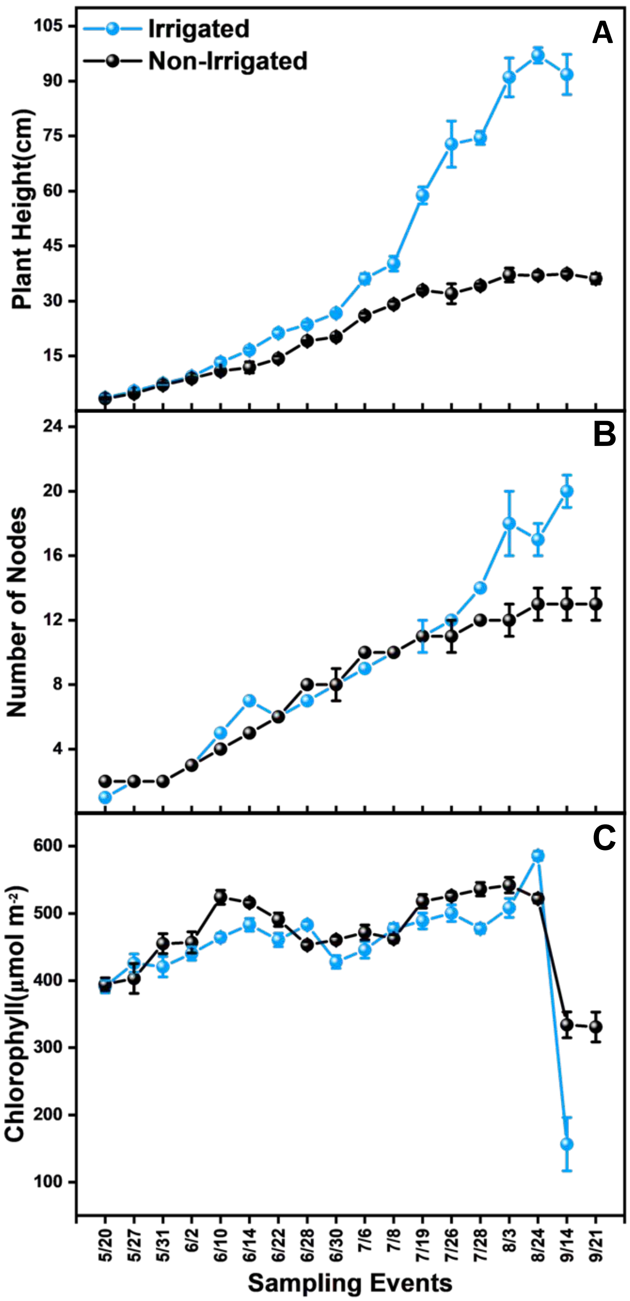

Figure 2. (A) Plant height within a 1-m row in both irrigated and non-irrigated areas, (B) the number of nodes within a 1-m row, and (C) leaf chlorophyll content in µmol m^{-2}.

Undergraduate Perspective in Agroecosystem Research

The job market needs more skilled workers with hands-on experience (Sabatini 1997). In the agricultural field, a lifelong learning experience should involve the active participation of undergraduates in research. Research experience and in-class activities are critical to a learning curve that integrates and provides valuable problem-solving capabilities to next-generation youth. Linking the critical role of water to yield through agroecosystem research, our hands get dirty while gaining the experience and knowledge needed to enter the job market. While few undergraduates see most of the classroom concepts' applications in real life, we gained crucial skills from participating in this agroecosystem research, including proper communication, extensive data handling, on-field data collection, post-processing data analysis, quality assurance, and quality control (Figure 4).

Communication was a critical learning objective among the skills learned from participating in agroecosystem research. Meeting farmers and knowing how they use scientific data, participating in decision-making meetings, and openly discussing project design and data collection objectives were exceptionally useful in solving agricultural issues.

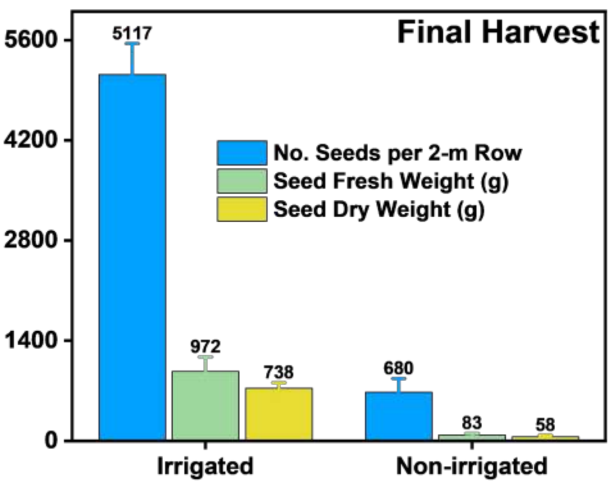

Figure 3. Irrigated and non-irrigated soybeans final harvest number of seeds per 2-m row, seed fresh weight, and seed dry weight (g).

Figure 4. Quality assurance (QA) and quality control (QC) non-destructive (A) number of plants per 1-m row, (B) comparing field pictures with Excel sheet for data transfer errors, (C) nodes and SPAD value pictures in the field, (D) plant height picture, and destructive QA/QC are exemplified in (E) picture of pods in field collected soybean samples, (F) leaf area measuring picture with scale, and (G) ensuring proper temperature of the drying oven.

Being part of the research team, undergraduates can strengthen their research thinking skills with the help of supervision (Craney et al. 2011). Before partaking in field and lab work, undergraduates know what they have learned in class. Working in the field motivates students and is essential to one's academic performance as an undergraduate. Participating in research as an undergraduate promotes critical thinking abilities and decision-making confidence. This study emphasized communication skills through appropriate field data collection.

Quality assurance and quality control are necessary data collection and processing procedures, which were crucial for achieving learning objectives (Sabatini 1997) and ensuring the validity of methods and data being used in research. Undergraduates would benefit significantly from increased opportunities to participate in research since doing research as an undergraduate prepares one for the future. In this study, when inputting data from the field, we would go back to the photographs to confirm the validity of the information entered in the Excel files. Figures 4A and 4B illustrate images captured in a 1-m row, and data entered in an Excel sheet, respectively. There are 18 plants in the image, and the labels L1 and R3 indicate that this sample belongs to location 1 and row 3. The numbers on the picture match the numbers in the Excel file; to ensure that clean data were well recorded, we refer to the picture and the recordings whenever we feel the data may contain certain inaccuracies. The project induced carefulness, eagerness, and preparedness for the job market. In addition, the outcomes of participating in undergraduate research were connecting with farmers and understanding the crucial role of water in agroecosystems. Overall, undergraduate education in agroecosystem research provides a foundation for understanding the complex relationships between agriculture and the environment and prepares students for research, consulting, and policy development careers.

Conclusions

Understanding the intricate dynamics of water management in agriculture is pivotal for addressing global food security challenges. Our research identifies the crucial role of water availability on crop production, reflected by more than a sevenfold difference in yield between irrigated and non-irrigated systems. This research's outcome reflects obtaining high-quality data while ensuring QA/QC by setting up an effective protocol during data collection. The precisely gathered data can support farmers in understanding management practices that boost yield, where effective data collection and analysis can foster decision-making and help verify new crop simulation models. For instance, knowing that irrigated fields have the potential to yield significantly more than non-irrigated fields with evidence and established plant health growth and health indicators would support farmers in making decisions when preparing and planning for their fields. On the other hand, this research provided an excellent opportunity for an undergraduate student to learn about the field data collection, data management, and analysis techniques.

There is a gap in undergraduate research; many students prefer internships, while others do not prefer hands-on studies. They are unaware of the opportunities, and some are not interested in them or think the compensation needs to improve (Tschepikow 2012; Stout 2018). From a larger perspective, undergraduate research dramatically benefits students. For a college student beginning to conduct research, this experience has been of utmost significance. The biggest lesson learned is the correct way to handle and process data. For undergraduates, most of what was learned in classes, in most cases, is not easily experienced in the actual world. Undergraduate research is a solution to help apply classroom lessons and identify contemporary problems and become future problem-solvers.

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Review Article

Residential Irrigation Restrictions and Water Conservation: A Review of Studies from 1978 to 2022

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Abstract: Urban water managers and policymakers have adopted demand management strategies to reduce water use and buffer against short-term water supply shortfalls. This article provides a systematic review of publications from 1978-2022 that examine the effectiveness of residential water use restrictions as the primary demand-side management tool. Our results indicate the significant overall effect of restrictions on reducing water consumption, with an average reduction of 12.3% from the 23 studies reviewed in this article. When evaluating effect strength by restriction type (mandatory versus voluntary), voluntary restrictions have a significantly lower effect than mandatory restrictions on water use. We also find an inverse correlation between the number of irrigation days allowed and the estimated effect strength.

Keywords: *residential irrigation restrictions, conservation policies, watering days*

Droughts worldwide are intensifying, with increased frequency, duration, and severity (Diffenbaugh et al. 2015; Keremane et al. 2017; Chiang et al. 2021). Climate-induced droughts, combined with population growth, have escalated pressures on urban water systems. Countries like Australia, South Africa, and the state of California have all had to develop various solutions to combat water scarcity resulting from these persistent droughts. With supply-side management options becoming increasingly limited due to the scarcity of untapped reservoirs, particularly in areas prone to recurrent droughts (Molle et al. 2010; Berbel and Esteban 2019), the focus has shifted to demand-side strategies for water management.

Demand-side management strategies, which include measures such as water pricing, financial incentives, and regulatory approaches like water quotas and usage restrictions, have taken precedence in urban water management (Olmstead et al. 2007; Olmstead and Stavins 2009; Mansur and Olmstead 2012; Baerenklau et al. 2014; Buck et al. 2021; Lee et al. 2021; Lee et al. 2022). A notable strategy is outdoor watering restrictions as an emergency response, which can be voluntary or mandatory. Such policies limit the number of days per week for watering (e.g., two days). During the 2020-2022 drought, for instance, California's urban water suppliers imposed restrictions on outdoor watering days (Nemati and Lee 2022). Additionally, in June 2022, the Metropolitan Water District of Southern California (MWD) introduced an Emergency Water Conservation Program, mandating one-day-per-week watering restrictions for millions in Los Angeles, Ventura, and San Bernardino Counties (Metropolitan Water District of Southern California (MWD) 2022).

These restrictions are not unique to the American Southwest; they are a global phenomenon. For example, in eastern Florida, 81 municipalities within the St. Johns River Water Management District have enforced watering restrictions, alternating between two days a week during dry seasons and one day during wet seasons (St. Johns River Water Management District 2022). Since 2011, Australia has enforced permanent emergency water restrictions in the Australian Capital Territory and Victoria (Australian Government Bureau of Meteorology 2022; Melbourne Water 2022;

Research Implications

- Analysis of 23 studies shows outdoor watering restrictions lead to a significant water demand reduction, with a reported average effect strength of 12.3%.
- Combining restrictions with other water conservation strategies like informational campaigns, rebates, and audits enhances their effectiveness.
- The success of mandatory restrictions depends on robust implementation, enforcement, and support from additional conservation policies.

Victorian State Government Environment 2022).

The effectiveness of restricting watering days can vary, being either mandatory, voluntary, or a combination of both, and is contingent upon the drought's severity, local climate, and geographic factors. A mild drought necessitates a less stringent response than a severe, prolonged one. For instance, the 2011-2016 California drought, the state's worst in over a millennium, called for a comprehensive policy approach (Griffin and Anchukaitis 2014; Browne et al. 2021). In such extreme cases, reducing irrigation days to once or twice weekly was a critical measure to close the significant gap between water supply and demand (Scauzillo 2017).

Although widely implemented by water agencies and policymakers, the effectiveness of outdoor watering restrictions has yielded inconsistent findings. Some studies report negligible impacts on water use (e.g., Robinson and Conley 2017; Hayden and Tsvetanov 2019; Dronyk-Trosper and Stitzel 2020), while others suggest reductions of 21 to 33% (e.g., Kenney et al. 2008; Mini et al. 2014; Browne et al. 2021). Analyzing various watering day strategies could clarify which are most effective at decreasing water consumption.

Our systematic review encompasses 23 studies from 1978 to 2022, investigating the effect of these restrictions on residential water use. Our objectives include a systematic review of the average effect of irrigation restrictions on residential water consumption, an examination of the variance in reported effects considering variables like location and season, and an assessment of the combined impact of irrigation restrictions with other conservation policies, such as audits, informational campaigns, and rebates.

Methods

We performed a systematic literature review using search terms ("watering days," "urban irrigation restrictions," and "water demand management") in various databases for publications studying the effectiveness of outdoor watering day restrictions. To reduce the risk of missing relevant studies, we applied the same search terms to various relevant journals, such as the *Journal of Utilities Policy*, *Environmental Economics and Management*, and *The American Water Works Association*.

We began the search on January 1, 2022, finishing the process on July 30, 2022. We searched without imposing restrictions on date or year, location, study design, study aim, or inclusion/exclusion criteria. Using the search procedure, we retrieved 112 articles published between 1978 and 2022. From this pool, we examined titles and abstracts, eliminated studies that did not focus on the effectiveness of irrigation restrictions, and estimated the amount of water saved. There were many articles on residential water conservation that instead focused on other policies or policy outcomes, such as price-based conservation strategies or welfare impacts of irrigation restrictions (e.g., Brennan et al. 2007).

The 23 articles identified as meeting the search criteria span 44 years of data in 12 distinct regions worldwide. The information from these articles was manually entered into a database, with each estimate of water savings as one observation. In this study, each reported "effect strength" in percentage terms is an observation for the study, defined as the percentage change in water use under irrigation restrictions. Note that each study could report more than one effect strength. A negative (positive) effect strength indicates a reduction (increase) in consumption due to the irrigation restrictions in place. Other factors entered for each observation include things such as the type and extent of the restrictions examined, concurrent water conservation policies, and study design.

One factor that is considered for water conservation policies is seasonality. Many irrigation restriction policies permit a different number of irrigation days for summer months and winter months. The differences in temperature, precipitation, and plant growth across seasons impact the irrigation demands. Water utilities respond by altering the number of watering days allowed by season. For this reason, the irrigation restriction effect strength was divided into the seasons from which the data were collected. The season variable was divided into three categories: "Summer," "Winter," and "Summer + Winter." Summer generally refers to April through September, and Winter refers to October through March. Summer + Winter refer to data collected across both time periods, most often over the entire year. The exact cutoff between summer and winter months is not uniform; the time frames given here broadly represent those used in the sample. Differences in seasonal effect strengths could be attributed to a strong association between seasonal changes in residential water demand and irrigation behavior (Kjelgren et al. 2000). Due to the higher temperatures and lower precipitation, people water their lawn more in the summer than the winter, meaning summer has a greater potential in reduction in the amount of water used in irrigation than winter.

Results

Summary of Peer-reviewed Articles Search Results

In Table 1, we provide a list of all 23 articles, study location, information on the irrigation restriction, and findings. This was the dataset used to examine the effectiveness of residential irrigation restrictions under varying circumstances, including time periods, locations, political situations, and conservation strategy bundles. The diverse circumstances in the dataset provide a unique look into which of these additional variables could lead to more successful implementation and effectiveness of residential irrigation restrictions.

The area with the greatest number of published studies was the Southwestern United States. California and Colorado were the subjects of six publications each, comprising more than half of the sample. Other regions with arid or semi-arid climates, such as Texas, Oklahoma, and New South Wales, were also represented in the dataset. Despite having climates and geographical features dissimilar to the other included regions, Florida, Massachusetts, Pennsylvania, and North Carolina were the subject of multiple publications, all within the last 16 years. Some publications chose to focus on mandatory restrictions without a limit on watering days (e.g., Grafton and Ward 2008). Some examined more stringent mandatory restrictions (e.g., Kenney et al. 2004; Browne et al. 2021). Others have examined both (e.g., Haque et al. 2013).

As indicated in Table 1, the overall estimated strengths range from the order of 1.6 to 34% reduction in water use (Maggioni 2015; Renwick and Green 2000). Some assessments of irrigation restrictions found them ineffective (e.g., Robinson and Conley 2017; Dronyk-Trosper and Stitzel 2020), while some found them to be significant tools for demand reduction (Anderson et al. 1980; Kenney et al. 2008).

Data from the 23 studies produced 251 total reported effects, summarized in Table 2. The average reported effect strength from the dataset was -0.123, meaning that, on average, irrigation restrictions lead to a -12.3% reduction in water consumption. When evaluating effect strength by restriction type (i.e., mandatory, voluntary, and mandatory plus voluntary), voluntary restrictions had a much lower effect than mandatory or mandatory plus voluntary restrictions. The high and low bounds for voluntary restrictions were estimated between no effect and roughly a 10% reduction.

Effect Strength by the Number of Irrigation Days Allowed

Figures 1 and 2 show the distribution of reported changes in water use across the number of irrigation days allowed. Figure 1 is on a per-study basis, taking the average estimated effect strength and number of irrigation days allowed in the study into a single point. This produced 23 points, one for each publication. The average estimated effect strength decreases as the number of permissible days increases, and vice versa. The maximum

Table 1. Summary of peer-reviewed literature on irrigation restrictions' effectiveness, with numbers in brackets indicating reported lowest and highest effect strength within each study.

Citation	State/Region	Watering Days Allowed	Additional Non-Price Strategies?	Overall Estimated Effect strength
Anderson et al. 1980	Colorado, U.S.	2	No	-0.304 [-0.197, -0.41]
Asci and Borisova 2014	Florida, U.S.	1-2	Yes	-0.173 [0.054, -0.556]
Browne et al. 2021	California, U.S.	1-2	Yes	-0.233 [-0.112, -0.338]
Dronyk-Trosper and Stitzel 2020	Oklahoma, U.S.	2-3	No	-0.018 [-0.007, -0.038]
Grafton and Ward 2008	New South Wales	7	No	-0.114 [-0.084, -0.144]
Halich and Stephenson 2006	Virginia, U.S.	-	Yes	-0.149 [-0.068, -0.154]
Haque et al. 2013	New South Wales	3-7	No	-0.158 [-0.0913, -0.201]
Haque et al. 2014	New South Wales	2-7	Yes	-0.113 [-0.039, -0.201]
Hayden and Tsvetanov 2019	California, U.S.	4	No	-0.00957 [-0.00635, -0.0256]
Kenney et al. 2004	Colorado, U.S.	1-2	Yes	-0.233 [0, -0.56]
Kenney et al. 2008	Colorado, U.S.	2	Yes	-0.334 [-0.031, -0.85]
Krohn 2019	Pennsylvania, U.S.	-	Yes	-0.0291 [-0.0037, -0.0498]
Maggioni 2015	Colorado, U.S.	-	Yes	-0.016 [-0.015, -0.017]
Miller 1978	Colorado, U.S.	2	Yes	-0.212 [-0.212, -0.212]
Mini et al. 2014	California, U.S.	2-7	Yes	-0.205 [-0.06, -0.35]
Renwick and Archibald 2018	California, U.S.	-	Yes	-0.155 [-0.151, -0.159]
Renwick and Green 2000	California, U.S.	-	Yes	-0.34 [-0.34, -0.34]
Robinson and Conley 2017	Massachusetts, U.S.	-	No	-0.018 [0.0263, -0.0385]
Shaw and Maidment 1987	Texas, U.S.	1-2	No	-0.0314 [0.0025, -0.0791]
Soliman 2022	California, U.S.	3	No	-0.195 [-0.153, -0.261]
Stone 2011	Colorado, U.S.	2	Yes	-0.063 [-0.0436, -0.0927]
Whitcomb 2008	Florida, U.S.	2	No	-0.0831 [0, -0.169]
Wichman et al. 2016	North Carolina, U.S.	2-3	No	-0.0897 [-0.029, -0.153]

Table 2. The summary statistics from the 23 publications included in this study.

	Mandatory	Voluntary	Mandatory & Voluntary	Overall
Total number of observations	179	51	19	251
Average reported effect strength*	-0.144	-0.049	-0.129	-0.123
Minimum reported effect strength	0.054	0.00	0.026	0.00
Maximum reported effect strength	-0.56	-0.097	-0.85	-0.85

*Negative effect strengths represent a reduction in water use, and positive effect strengths represent an increase in water use. For example, -0.123 means, on average, watering days restrictions lead to a 12.3% reduction in water use.

number of days allowed for irrigation is seven, which is equivalent to a voluntary restriction. Each successive decrease in the number of irrigation days allowed reduces the water used.

In Figure 2, each point is a reported effect strength (i.e., multiple reported effect strength per study), giving a single point to every reported effect strength in the database with a corresponding number of irrigation days allowed. Figure 2 displays similar trends to Figure 1. This is best seen by comparing the two extremes of the x-axis. Allowing irrigation seven days of the week yields little to no change in water use. In comparison, one to two watering days a week has been shown to provide a much more consistent and significant estimated reduction in demand.

An evident cluster of data points between one and three irrigation days is allowed in both figures. Irrigation restrictions are often implemented to reduce the number of allowed days to the minimum amount required to sustain grass. This is done to prevent users from overwatering their lawns by exceeding the recommended one to two days per week of watering in standard conditions.

Effect Strength by Season and Irrigation Restriction Type

Across all seasons and restriction types, the average estimated effect strength is -0.123, a 12.3% reduction in demand across the full sample. When grouped by restriction types, the difference between mandatory and voluntary irrigation restriction estimated reduction rate was clear. Mandatory restrictions, with an overall estimated effect strength equal to a 14.4% reduction, are nearly ten percentage points greater than voluntary restrictions at 4.96% (Table 3). While an estimated 5% reduction from voluntary restrictions is noteworthy, the upper limits of voluntary and mandatory restrictions illuminate the disparity between their ability to create significant demand reduction.

As illustrated in Table 3, when examining estimated effect strengths by season, overall, “Summer” had an estimated effect strength equal to a 15.3% reduction, compared to a 11.6% reduction for “Summer + Winter.” While “Winter” had a larger rate of reduction with -0.193, a sample size of three has limitations. When examining the average estimated effect strengths for mandatory restrictions, a similar relationship was apparent; mandatory restrictions have a reduction rate of 20.6% in the summer and 13.2% across both seasons. This trend does not hold when comparing seasonality under voluntary or mandatory and voluntary. However, a trend that continues was the greater average estimated effect strengths for mandatory restrictions compared to voluntary restrictions for both “Summer” and “Summer + Winter.” Further interpretation of seasonal effect strengths is difficult. Residential water use in the winter is primarily indoors, compared to summer, where a greater proportion of use is outdoors; this has led scholars to note the difficulty in drawing conclusions based on seasonal changes, coupled with potential changes in policy, conservation behaviors and attitudes, among other confounding factors (Browne et al. 2021).

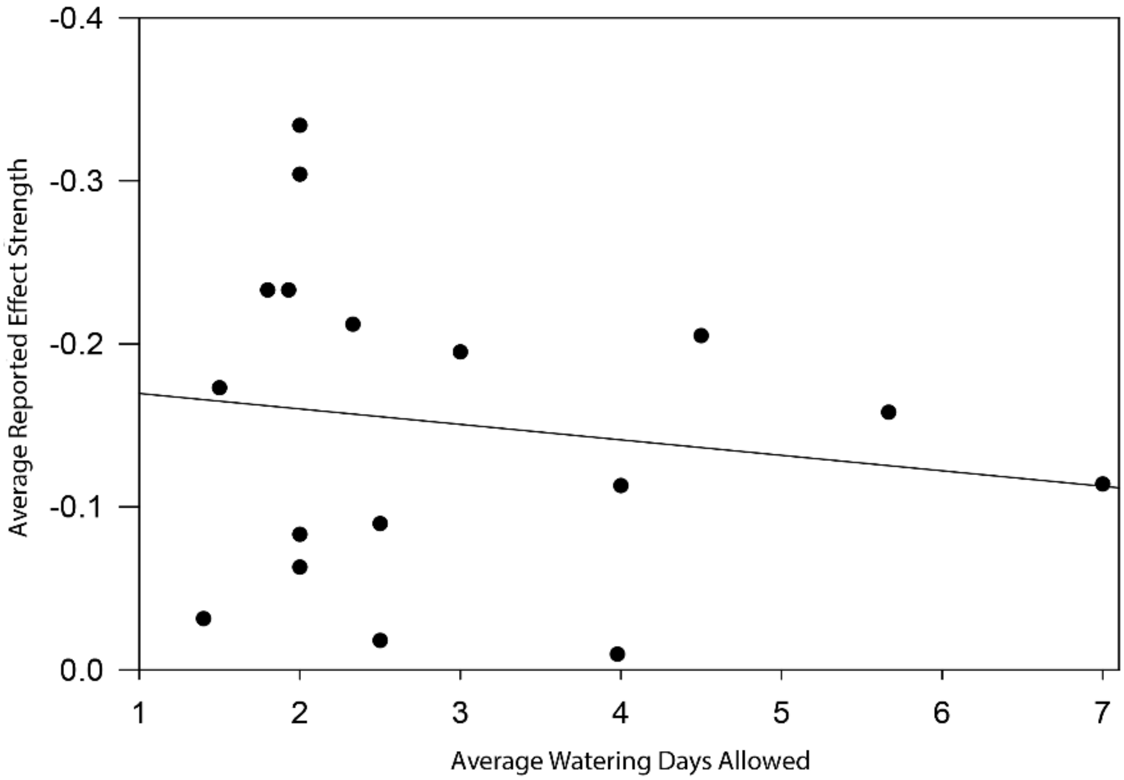

Figure 1. Reported effect strength (per study) by irrigation days allowed. The back solid line is the trend line.

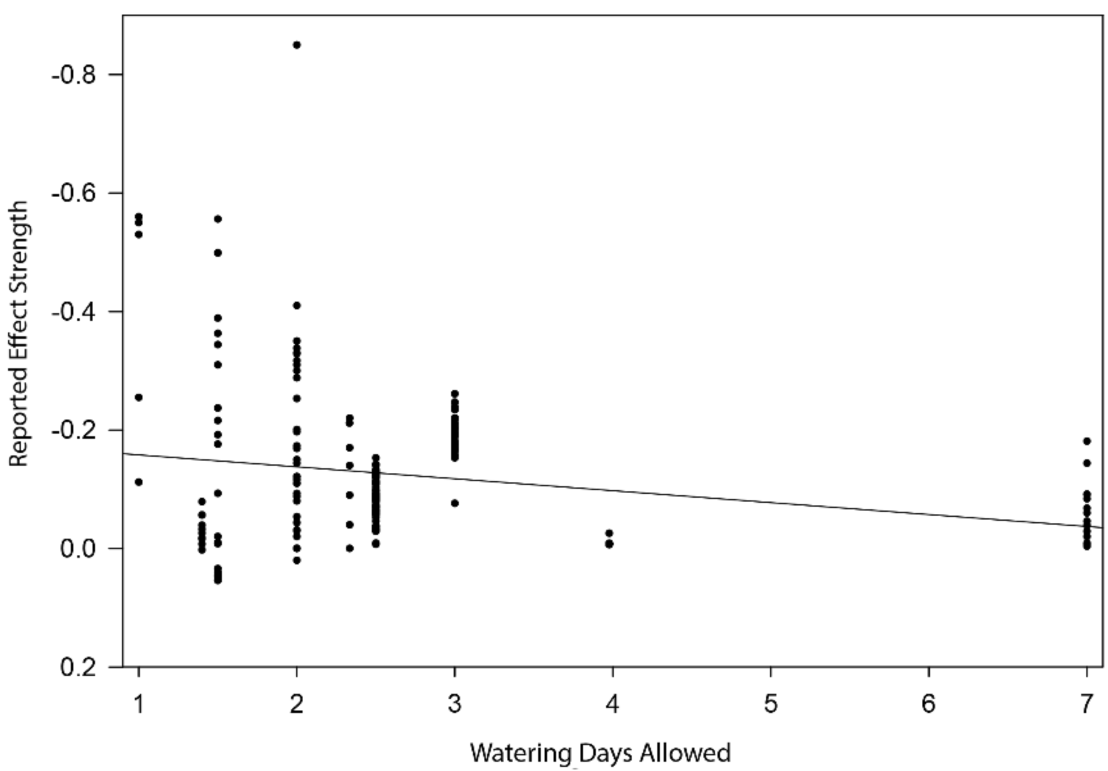

Figure 2. Reported effect strength by irrigation days allowed. This figure includes all the reported strengths in the study.

Table 3. The average reported effect strength (ARES) and number of observations (Obs.) by irrigation restriction type and season. Numbers in brackets report the reported effect strength range.

	Overall		Mandatory		Voluntary		Mandatory & Voluntary	
	ARES	*Obs.*	*ARES*	*Obs.*	*ARES*	*Obs.*	*ARES*	*Obs.*
Summer	-0.153 [0.026, - 0.56]	42	-0.206 [0.002, -0.56]	27	-0.0343 [0, -0.09]	10	-0.108 [-.038, -0.221]	5
Winter	-0.193 [-0.112, -0.256]	3	-0.193 [-0.12, -0.255]	3	-	-	-	-
Summer + Winter	-0.116 [0.054, -0.85]	206	-0.132 [0.054, -0.556]	150	-0.0522 [-0.004, -0.096]	41	-0.133 [-.007, -0.85]	15
Overall	-0.123	251	-0.144	180	-0.0496	51	-0.126	20

Effect Strength and Additional Conservation Policies

The impacts of the presence of additional conservation policies are presented in Table 4. The policies analyzed included three non-price policies: audit consultations, informational campaigns, and rebates. An audit consultation generally entails a government water consultant coming to a home to find water inefficiencies in the home and fix or suggest solutions to the issues found. Informational campaigns are wide-ranging education initiatives to teach better water use habits and new irrigation restriction regulations. Rebates are credits for discounts on water-efficient appliances such as low-flow toilets and shower heads. Price modifications account for a combination of two policies: price level changes and price structure changes. Audit consultations, informational campaigns, and rebates all correlate with a reduction in demand when used in conjunction with irrigation restrictions. The additional reduction effectiveness is to the order of 4.4, 6.3, and 5.6%, respectively. This contrasts with price modifications, where there is a negligible difference of 0.2%.

As noted in the methods and data section, these results could be misleading. For a strategy such as an informational campaign, they were not always mentioned by the studies and were thus marked as not being present when not mentioned. However, it is unlikely that an informational campaign was absent for more than a small selection of the sample, if at all. This applies in varying degrees to all the strategies recorded in the dataset. This creates non-representative figures with a disproportionately small number of observations.

Discussion

Based on the studies examined, mandatory residential irrigation restrictions are effective in reducing water demand. The degree of effectiveness varies between studies. Voluntary residential irrigation restrictions are ineffective; data on their effectiveness attribute little to no demand reduction across all studies examining it. Voluntary restrictions are unlikely to induce a meaningful reduction in usage or frequency without incentives to change outdoor irrigation habits. Mandatory restrictions often institute consequences for failed compliance, such as fines, rate increases, or even shutting off the water entirely. These enforcement standards likely induce the change that voluntary restrictions are not able to. This is shown in their average estimated effect strengths. Mandatory restrictions have an average 14.4% reduction in demand compared to an average 4.87% reduction for voluntary restrictions. Mandatory restrictions consistently outperform voluntary restrictions across seasons, locations, time periods, and concurrent policies.

Table 4. Average reported effect strength (ARES), number of observations (Obs.), and number of studies (# of Studies) by presence of additional conservation policies. Numbers in brackets report the reported effect strength range.

	With Conservation Policy			Without Conservation Policy		
Conservation Policies	*ARES*	*Obs.*	*# of Studies*	*ARES*	*Obs.*	*# of Studies*
Audit Consultation	-0.163 [0.054, -0.556]	23	3	-0.119 [0.0263, -0.85]	226	20
Informational Campaign	-0.173 [0, -0.85]	54	10	-0.109 [0.054, -0.56]	197	13
Rebate	-0.175 [-0.015, -0.85]	17	7	-0.119 [0.054, -0.56]	234	16
Price Modification	-0.122 [0.054, -0.85]	187	17	-0.124 [0.0263, -0.41]	64	7

Mandatory restrictions can be more likely to succeed through their implementation, enforcement, and additional conservation policies. It is not possible to force compliance without proper infrastructure and enforcement mechanisms. Similarly, without an effective strategy for implementation, the restrictions are unlikely to succeed. Examples of poor implementation include poor information dissemination, too few or too many irrigation days, or a lack of complimentary conservation policies. Avoiding these mistakes can produce better policy outcomes.

As the number of irrigation days allowed decreases, the amount of water conserved increases. The optimal number of irrigation days allowed is difficult to determine, however. It stands to reason that allowing six days of irrigation per week would not significantly change water demand. On the other hand, allowing a single day of irrigation would significantly reduce demand. According to the studies analyzed, the most common number of irrigation days allowed is around two. However, the effectiveness of two irrigation days per week is more mixed. The average effect strength of a two-day-per-week policy is a 16.13% reduction. The most optimistic study estimates a 33.4% demand reduction compared to the least optimistic estimate of a 1.8% demand reduction. The optimal number of irrigation days was not determined in this study. A higher order of demand reduction is induced by allowing fewer irrigation days. This trend, coupled with the ubiquity of two-day-per-week policies, suggests that they are likely optimal. A potential study on the optimal number of watering days could have a large impact on future policy decisions and can hopefully be completed in the future.

According to the studies analyzed for this review, mandatory restrictions reduce demand by 14.4% on average. Thirteen out of 23 of the studies analyzed include additional non-price policies, though this is almost certainly an underestimation. Given that the estimates of reduction from this review are based primarily on irrigation restrictions with policy bundles, the use of irrigation restrictions as the sole policy in many California agencies is unlikely to induce the change necessary to meet their conservation goals.

The research included in this review mirrors the sentiments of economists on price-based policies. While the consensus had been that price policies reduced demand in the short run, more recent analysis has argued that water is an inelastic commodity in the short run, making price-based policies ineffective in reducing consumption (Haque et al. 2013). Some have concluded that price-changing policy mainly falls on the poor while not creating significantly different policy outcomes between income groups (Wichman et al. 2016). The results indicated in Table 4, while not fully in line with the conclusions of economic and policy researchers, do point toward the ineffectiveness of price-based policies. Table 4 does, however, show the significant demand reduction created by non-price policies.

The increasing frequency and severity of droughts worldwide, and the subsequent need to reduce water consumption, will require more robust policies to further reduce demand. Water utility agencies should therefore seek to implement a diverse set of price and non-price strategies to optimally reduce demand. Not every agency has the means to employ all the conservation strategies discussed in this review. What works for one agency will not necessarily work for another. Irrigation restrictions are a valuable tool in reducing residential water use. Other price and non-price policies should be considered and implemented when instituting mandatory irrigation restrictions. Irrigation restrictions have a ceiling for reducing demand. When coupled with other compatible policies, further demand reduction is possible.

A common obstacle to policy implementation is political backlash. Water conservation policies require a change in lifestyle for the people living under them. This will inherently make them unpopular with a significant percentage of the municipality. While a more extreme policy package may be the most effective choice, the political repercussions may require decision-makers to implement a more conservative package.

Conclusion

Mandatory residential irrigation restrictions are growing in usage across the world in line with the increased frequency and severity of droughts. This review investigated the effectiveness of irrigation restrictions across various policies, locations, climates, and time periods through the analysis of 23 academic sources examining the effectiveness of irrigation restrictions. Using the data from these sources, we found evidence that such restrictions are likely to reduce residential water demand.

The effectiveness of mandatory irrigation restrictions was found to increase when the number of irrigation days allowed was decreased. Similarly, effectiveness was found to be higher with the implementation of additional non-price conservation measures such as rebates, informational campaigns, and audit consultations. The presence of price conservation measures is linked with negligible changes in demand. These results indicate that mandatory residential irrigation restrictions are effective in reducing demand and are more effective than voluntary restrictions. Additional policies are likely to increase the total reduction in water demand. A definitive best policy package is not provided, given the differing circumstances of each utility agency. However, introducing multiple conservation methods may produce better conservation outcomes.

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