Journal of Contemporary

Water Research & Education

Issue 175
April 2022

UCOWR
UNIVERSITIES COUNCIL ON WATER RESOURCES

A publication of the Universities Council on Water Resources
with support from Southern Illinois University Carbondale

JOURNAL OF CONTEMPORARY WATER RESEARCH & EDUCATION

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www.ucowr.org

ISSN 1936-7031

Cover photo: *Dry bed of an ephemeral stream draining caldera complex rocks of the Tucson Mountains.* Photo by Christopher Eastoe.
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Journal of Contemporary Water Research & Education

Issue No. 175 April 2022

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 175, Pages 1-14, April 2022

Understanding the Water Resources of a Mountain-block Aquifer: Tucson Mountains, Arizona

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Abstract: Water resources are limited in arid locations such as Tucson Basin. Residential development in the Tucson Mountains to the west of Tucson, Arizona, is limited by groundwater resources. Groundwater samples were collected from fractured bedrock and alluvial aquifers surrounding the Tucson Mountains to assess water quality and recharge history through measurement of stable O, H, and S isotopes; tritium; and ^{14}C. Most groundwater is a mixture of different ages but is commonly several thousand years old. A few sampling locations indicated a component of water recharged after the above-ground nuclear testing of the mid 1950s, and these sites may represent locations near where the aquifer receives present-day recharge. The Tucson Mountains also host sulfide deposits associated with fractures and replacement zones; these locally contribute to poor-quality groundwater. Projections of future climate predict intensifying drought in southwestern North America. In the study area, a combination of strategies such as rainwater harvesting, exploitation of renewable water, and low groundwater use could be used for sustainable use of the groundwater supply.

Keywords: *groundwater, fractured rock, isotopes, recharge, residence time, water supply, Arizona*

The Tucson Mountains form the western boundary of the northern part of Tucson Basin in southeastern Arizona, USA (Figure 1). The Tucson metropolitan area occupies much of Tucson Basin, which is the alluvial basin to the east of the mountains, and has spilled over into Avra Valley, the alluvial basin to the west. The mountains and their foothills constitute a biodiverse landscape of the Sonoran Desert. Most of the mountain range and part of the adjacent foothills are protected within the Saguaro National Park and Pima County Parks. As Tucson has grown, private land adjacent to the parks, including some of the larger valleys within the hard-rock range, has attracted low-density urban development. Several of these areas are at present beyond the reach of existing water and wastewater infrastructure and must rely on private wells, rainwater collection, or hauling water for domestic water supply, and individual disposal systems for wastewater.

The demand for water in the settled part of the mountain range continues to grow at a time when some private well owners report falling groundwater levels (Robert Webb, retired U.S. Geological Survey, oral communication 2017). Arizona has experienced drought since about 2000 (Arizona Department of Water Resources 2020a), manifested in Tucson by decrease in winter

Research Implications

- Groundwater in the Tucson Mountains occurs in poorly-connected rock fractures.
- Groundwater in caldera-complex volcanic rock is a mixture of late Pleistocene and pre-bomb, mainly summer recharge; little recharge occurs at present.
- Groundwater supply is limited, and of poor quality where affected by sulfide mineralization.
- Post-bomb recharge occurs in Oligocene volcanic rock and Cretaceous arkose, possibly providing a small, sustainable water supply.

rainfall (e.g., Eastoe and Dettman 2016). In the arid southwestern USA, future climate change is expected to result in higher temperatures (USGCRP 2017) and prolonged drought due to decreasing winter rain as the jet stream and storm tracks move poleward (Udall and Overpeck 2017). Future warming may decrease groundwater recharge as evapotranspiration increases. Management of water resources in the Tucson Mountains and similar mountain ranges can be informed by an improved understanding of the mountain-block aquifers.

Isotope studies of rainwater and groundwater in Tucson Basin and surrounding mountain ranges have contributed much to the understanding of the hydrology of the basin, and to the understanding of regional recharge mechanisms (Kalin 1994; Eastoe et al. 2004; Gu 2005; Eastoe and Dettman 2016; Eastoe and Gu 2016; Eastoe and Towne 2018). In the Tucson Basin, these studies have identified long-term mean isotope compositions in local precipitation, isotope lapse rates with altitude, domains of groundwater of different sources in basin alluvium, zones of basin alluvium in which recharge occurs rapidly, and evolution of groundwater sources beneath downtown Tucson. At regional scale, the studies have proposed multiple recharge mechanisms that appear to be zoned with respect to basin location.

Studying the hydrology of mountain blocks is commonly challenging because of paucity of field data and difficulty of access to sample locations (Wilson and Guan 2004). At the small scale of mountain headwater catchments, tracer studies have been combined with hydrologic flux observations and in some cases with modeling to constrain the relation between precipitation, soil storage, and streamflow (e.g., Katsuyama et al. 2005; Aishlin and McNamara 2011; Ajami et al. 2011; Gabrieli et al. 2012; Dwivedi et al. 2019). Isotope tracers have been applied at the scale of mountain blocks to track groundwater movement within mountain blocks (Winograd et al. 1998; Earman 2004) and to identify mountain-block recharge (MBR) to surrounding lowland aquifers (James et al. 2000; Manning and Solomon 2004; Thiros and Manning 2004; Wahi et al. 2008; Harris et al. 2010; Newton et al. 2012; Eastoe and Rodney 2014). Eastoe and Wright (2019) examined the distribution of isotope tracers in mountain-block groundwater of the southern Basin-and-Range Province, identifying several recharge mechanisms that appear to depend on altitude and lithology. Modeling of mountain-block topography and permeability predicts the partitioning of recharge between base flow in mountain streams and MBR, and the relationship of MBR to depth of fractures and topography (Welch and Allen 2012; 2014). Ren et al. (2019) used borehole observations in an experimental well field in granite to estimate hydrologic apertures of fractures and local fracture porosity; they noted that groundwater flow would also depend on fracture connectivity.

In the case of the Tucson Mountains, groundwater can be sampled from numerous private supply wells that occur in clusters in the northern part of the mountain block, over an area of about 75 km^2. Eastoe and Wright (2019) published a small isotope dataset (stable O and H isotopes, tritium, and ^{14}C) for wells in hard rock of the Tucson Mountains, and concluded that groundwater recharge in the range occurred by a mechanism (to be explained in detail below) that is unusual in other mountain blocks of southern Arizona. Beisner and Gray (2018) published a second dataset (stable O and H isotopes, sulfate isotopes, Sr, tritium, and ^{14}C) for eight groundwater samples adjacent to the range front in a small area near the Old Yuma Mine (Area Y, Figure 1), with the aim of identifying contamination emanating from the mine workings. To these datasets can be added stable O and H measurements with a few tritium and ^{14}C measurements for wells completed in alluvium near the outcrop boundaries of the mountain block. In this study, the isotope data are reviewed with the aim of providing information about the water resources of the Tucson Mountains, in particular the sources and ages of the groundwater in and near the mountain block, and the nature of the aquifer or aquifers.

Background

Study Area

The topography of the Tucson Mountains is highly varied. At one extreme is craggy landscape with cliffs and steep, V-shaped canyons. At the other extreme, rolling hills surround broad sandy or gravelly stream beds. The hard-rock outcrop of the range is surrounded by a broad, elliptical pediment consisting largely of alluvial-fan deposits. The elevation of the boundary between hard rock and the pediment ranges from 600 to 900 meters above sea level (masl), and the highest point is Wasson Peak at 1,428 masl. A semiarid to arid climate prevails. Precipitation occurs in two seasons: a season of frontal rain or snow events mainly between November and March, and a summer monsoonal season of convective rain systems between late June and September. In some years, tropical depressions bring additional rainfall in September or October. A long-term climate record is available for the Arizona-Sonora Desert Museum (ASDM) on the western flank of the range (Area D, Figure 1), where mean annual rainfall was 382 mm (69% in June-October) for 1971-2000 (Western Regional Climate Center 2020). Desert-scrub and desert-grassland vegetation types predominate (Rondeau et al. 2000).

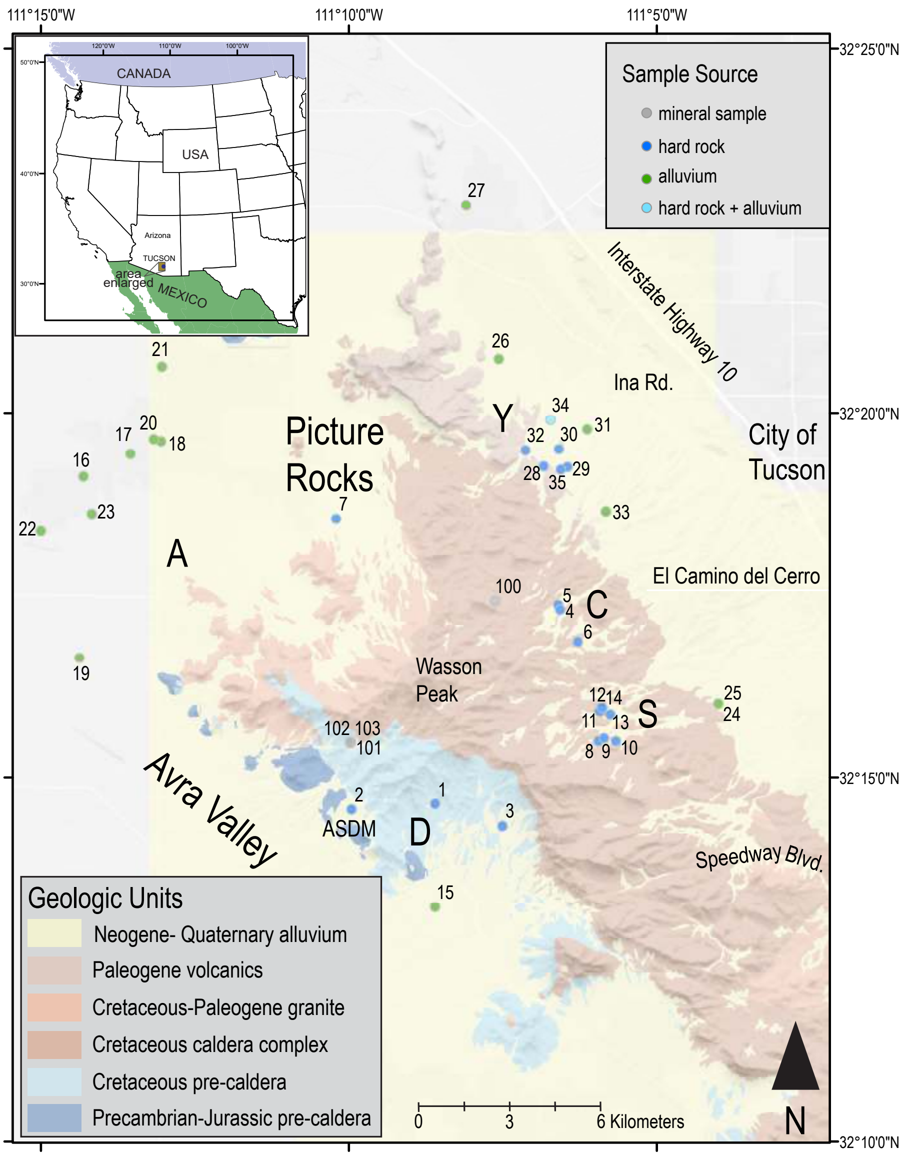

Figure 1. Map of the Tucson Mountains and surrounding area, showing geology (after Lipman 1993), and sample sites. A = Avra Valley Water Cooperative cluster; C = Camino del Cerro cluster; D = Desert Museum cluster; S = Sweetwater Drive cluster; Y = Old Yuma Mine cluster; ASDM = Arizona-Sonoran Desert Museum. Road map image is the intellectual property of Esri and is used herein under license. Copyright © 2014 Esri and its licensors. All rights reserved.

Geology

The Tucson Mountains are a fault-bounded block of crystalline rock within the Basin-and-Range Province (Fenneman 1931). The following description of the geology is from Lipman (1993) and Bezy (2005). The core of the range consists of a belt of felsic igneous rock of late Cretaceous to early Paleogene age including a supracrustal suite of rhyolitic tuff and megabreccia associated with the eruption of a large caldera, and coeval granitoids at the northwestern end of the belt. Pre-caldera units, comprising Paleozoic limestone and Mesozoic volcanic and terrestrial clastic sedimentary rocks including alluvial and minor lacustrine members, are overlain unconformably by the caldera-associated rocks along the western and southwestern flanks of the range. Post-caldera volcanic rocks of Oligocene age, mainly dacitic lava and pyroclastics, overlie the caldera rocks at the northern end of the range. Deformation of the crystalline rocks began with pre-Oligocene rotation (Hagstrum and Lipman 1991), followed by normal faulting associated with Neogene tectonic extension during the formation of the Basin-and-Range Province. Deformation has led to fracturing of the crystalline rocks.

Mineralization occurs as replacement bodies that are mainly controlled by northwest-trending fractures within late Cretaceous to early Paleogene sedimentary and volcanic rocks (Kinnison 1958). Sulfide mineralization occurs with skarn replacing thin limestone members of the Cretaceous Amole Arkose. Hypogene pyrite, galena, sphalerite, chalcopyrite, chalcocite, and molybdenite are recorded. Oxidation extends to a maximum of 13 m below the surface. In rhyolite tuff and megabreccia of the range near El Camino del Cerro, copper mineralization occurs with magnetite replacing volcanic rock. In the Old Yuma Mine, the most abundant sulfide is primary galena in a fracture zone cutting Cretaceous andesite and associated with a porphyritic dyke (Mindat.org 2020).

A broad set of alluvial fans flanks the Tucson Mountains to the east and west. The alluvial fans consist of gravel and sand transported from the mountains. The thickness of alluvium reaches 200 m within 1-2 km of the range front, according to well drillers' logs (Arizona Department of Water Resources 2020b).

Hydrogeology

Surface water is ephemeral throughout the Tucson Mountains. Springs are rare; only one spring (site 3, Figure 1) was sampled for this study. Limited information on groundwater occurrence is available in drillers' logs (Arizona Department of Water Resources 2020b). In settled areas of the mountains, groundwater is produced from domestic wells in hard rock. Available well logs provide insufficient detail to show whether water is produced from permeable strata or from fractures. In most cases, well depths are 100-200 m. Well owners in area C (Figure 1) reported declining water levels at the time of sampling. Groundwater is pumped from saturated alluvium adjacent to the mountain front, commonly from depths of 100-200 m.

Previous Work, Isotope Hydrology

Detailed studies of stable O, H, and S isotopes; tritium; and ^{14}C are available for the regional alluvial aquifer of Tucson Basin to the east of the Tucson Mountains (Eastoe et al. 2004; Gu 2005; Eastoe and Gu 2016). To the west of the range, Hess (1992) undertook a study including O and H stable isotopes in groundwater of the regional alluvial aquifer in Avra Valley. Long-term data on stable O and H isotopes in Tucson Basin precipitation were documented by Eastoe and Dettman (2016) and Wright (2001). Eastoe et al. (2011) reported a multi-year dataset for tritium in Tucson Basin precipitation. Data from these studies provided the basis for determination of isotope lapse rates in the mountain ranges surrounding Tucson Basin, and for two studies of regional recharge mechanisms. Eastoe and Towne (2018), in a study comparing recharge mechanisms of alluvial basins of the Basin-and-Range Province in Arizona, observed that basins in southern Arizona receive recharge of both summer and winter precipitation. Stable O and H isotope data are consistent with recharge occurring mainly during the wettest ~30% of months. Eastoe and Wright (2019) examined stable O and H isotope data in groundwater of mountain blocks in southern Arizona, including the Tucson Mountains. The pattern of isotope data in the Tucson Mountains is unusual in the region; the authors suggested that it represents mixing of younger and older recharge. The younger recharge resembles a mixture of summer and winter recharge for the wettest months, like that in Tucson Basin alluvium. The older recharge appears to be ancient precipitation of late Pleistocene to early Holocene age. Beisner and Gray (2018) presented a dataset for eight wells in a small area around the Old Yuma Mine, including stable O and H isotopes in water, stable S isotopes in sulfate, tritium, and stable C isotopes and ^{14}C in dissolved inorganic carbon. They interpreted the results in terms of groundwater age.

Methods

Groundwater samples were collected from domestic supply wells in continual use. Samples were analyzed at University of Arizona and U.S. Geological Survey (USGS) laboratories.

Isotope Analytical Methods – Area Y

Stable O and H isotopes (δ^{18}O and δ^{2}H) were measured at the USGS Reston Stable Isotope Laboratory in Reston, Virginia, using dual-inlet isotope ratio mass spectrometers (IRMS) on CO_2 and H_2 equilibrated at constant temperature with sample water following methods by Révész and Coplen (2008a; 2008b). The two-standard deviation (2σ) uncertainties are 0.2 per mil for δ^{18}O and 2 per mil for δ^{2}H. Results are reported relative to Vienna standard mean ocean water, VSMOW. The Reston Stable Isotope Laboratory measured δ^{34}S of sulfate extracted as $BaSO_4$ from water samples. Isotope measurements were made by continuous flow IRMS on SO_2 prepared using a Carlo Erba NC 2500 elemental analyzer (Révész et al. 2012). Measurements of ^{14}C and δ^{13}C ratios were made at the National Ocean Sciences Accelerator Mass Spectrometry facility at Woods Hole Oceanographic Institution, Massachusetts, by accelerator mass spectrometry (AMS) and IRMS, respectively, on CO_2 extracted by acid hydrolysis from water samples. AMS results are reported relative to international standard Oxalic Acid I. The University of Miami Tritium Laboratory, Miami, Florida, measured tritium by gas-proportional counting on H_2 gas prepared from water samples subjected to 60-fold electrolytic enrichment, with a reporting limit of 0.3 picocurie per liter (pCi/L), or 0.1 tritium unit (TU). Measurements are standardized relative to National Institute of Standards and Technology Standard Reference Material (NIST SRM) #4926.

Isotope Analytical Methods – Other Areas

Isotopic measurements were made at the Environmental Isotope Laboratory, University of Arizona, Tucson, Arizona. δ^{18}O and δ^{2}H were measured on an automated gas-source IRMS (Finnigan Delta S). For δ^{2}H measurement, water was reacted at 750°C with Cr metal in a Finnigan H/Device attached to the mass spectrometer. For δ^{18}O measurement, water was equilibrated with CO_2 at 15°C in an automated equilibration device coupled to the mass spectrometer. Standardization is based on international reference materials VSMOW and Standard Light Antarctic Precipitation (Coplen 1995). Analytical precision (1σ) is 0.9 ‰ or better for δ^{2}H and 0.08 ‰ or better for δ^{18}O (Eastoe and Dettman 2016). Measurements of δ^{34}S were made on $BaSO_4$ precipitated from solution at pH < 2, using a modified VG602C IRMS. Standardization is based on international standards OGS-1 and NBS123. Values of δ^{34}S are reported with an analytical precision of 0.13 ‰ (1σ). Tritium and ^{14}C were measured by liquid scintillation counting using Quantulus 1220 spectrophotometers. Tritium was measured on electrolytically enriched 0.18-L water samples, with a detection limit of 0.7 TU. Results are reported relative to NIST SRMs 4361 B and C. Dissolved inorganic carbon was extracted from 50-L water samples as $BaCO_3$, and the carbon was converted to benzene for measurement of ^{14}C. The detection limit was 0.4% modern carbon (pMC) for samples without dilution, and results are reported relative to Oxalic Acid I. Values of δ^{13}C were measured manually on CO_2 using a Finnigan Delta S mass spectrometer. The CO_2 was prepared from splits of the $BaCO_3$ extracted for ^{14}C measurement. Analytical precision was 0.1 ‰ (1σ), and measurements were calibrated using international standards NBS-19 and NBS-18.

Presentation of Data

Stable isotope measurements are expressed using δ-notation, e.g.: $\delta^{2}H = (R_{sample}/R_{standard} - 1) \times 1000$ ‰, where $R = {}^{2}H/{}^{1}H$ and the standard is VSMOW. The definitions of δ^{18}O, δ^{34}S, and δ^{13}C are analogous, with standards VSMOW for O, VCDT (Vienna-Canyon Diablo Troilite) for S, and VPDB (Vienna PeeDee Belemnite) for C.

Tritium data are expressed as TU, where 1 TU = 1 atom ^{3}H per 10^{18} atoms H. Measurements of ^{14}C are expressed as pMC without normalization, where 100 pMC corresponds to the ^{14}C content of atmospheric carbon in 1950, corrected for industrial emissions.

Results

Analysis includes previously published data from Beisner and Gray (2018) and Eastoe and Wright (2019). Additionally, previously unpublished data for groundwater from the alluvium flanking the Tucson Mountains are also included in Table 1.

Stable O and H Isotopes

Values of δ^{18}O and δ^2H for groundwater in hard rock form a linear array with a slope near 8, distinct from and to the right of the global meteoric water line (GMWL; Figure 2). The data array is also distinct from local meteoric water lines (LMWL) defined by seasonal means for all precipitation at 1,000 masl, or by seasonal means for the wettest ~30% of months at 1,000 masl (Figure 2B). Pairs of (δ^{18}O, δ^2H) range from (-7.2, -53 ‰) to (-9.9, -75 ‰); and much of the data range is present in each of four areas with multiple samples (Figure 2A). Groundwater from alluvium immediately east and west of the Tucson Mountains mainly plots on a modified LMWL (Figure 3) defined by precipitation in the wettest ~30% of months, for 1,000 masl (an approximate mean elevation for the mountain block), with slope 6.5, or for 740 masl (a typical elevation of the boundary between the mountain-block outcrop and surrounding alluvium) with slope 6.1. Pairs of (δ^{18}O, δ^2H) range mainly from (-7.7, -54 ‰) to (-8.6, -61 ‰).

Tritium and ^{14}C

Tritium measurements range from below detection to 6.8 TU. ^{14}C measurements range from 7.8 to 101.7 pMC. Relations between δ^{18}O, δ^2H, and pMC are shown in Figures 4A and 4B. Among samples with both tritium and ^{14}C data, tritium appears generally to increase with pMC, except for groundwater in area Y, where finite tritium is found only in samples with pMC near 100 (Figure 4C).

Stable C Isotopes

Values of δ^{13}C range from -7.9 to -14.7 ‰. In area Y, but not in other areas, δ^{13}C decreases as pMC increases (Table 1). In area Y, the δ^{13}C values probably represent mixing between soil-gas CO_2, with δ^{13}C values near or below -15 ‰, and rock-carbonate sources with δ^{13}C > -8 ‰. Across the study area, the latter may include Neogene pedogenic carbonate (mainly -1 to -2 ‰ in Tucson Basin alluvium, according to unpublished data of the University of Arizona Environmental Isotope Laboratory, oral communication, May 2021), Permian limestone (0 to +5 ‰; Veizer and Hoefs 1976), and Cretaceous lacustrine carbonate (δ^{13}C unknown). Soil-gas appears to predominate in groundwater with pMC near 100, drawn from Oligocene volcanic rock in area Y.

Stable S Isotopes

Values of δ^{34}S in groundwater from hard rock span a range of +0.9 to +6.9 ‰, with an outlier at +14.0 ‰ (Table 1). Two groundwater samples from alluvium, one east and one west of the range, have values of +5.3 and +5.4 ‰. These measurements are compared (Figure 5) with δ^{34}S ranges of sulfate in rainwater and dust in Tucson Basin and with Pliocene gypsum from the center of Tucson Basin (Gu 2005). Pliocene or older basin sediments may be present near the surface along the basin margins. In addition, three new measurements of δ^{34}S (+0.5, +0.6, and +1.4 ‰, on pyrite and sulfate crust from the base of a waste pile) were obtained from the Gould Mine (sites 101-103, Figure 1), where mineralization occurs as skarn replacing thin limestone lenses. A single value, +7.8 ‰, was obtained from jarositic limonite in the weathered zone at site 100.

Discussion

Recent Recharge

Values of δ^{18}O and δ^2H for all groundwater samples from hard rock conform to a single trend of slope near 8. Eastoe and Wright (2019) concluded that the data array represented groundwater of shorter and longer residence times, corresponding to its upper and lower ends, respectively. Addition of the data of Beisner and

Table 1. Groundwater sample data (Note: hr = hard rock; all = alluvium; m = meters; masl = meters above sea level; δ = delta, ‰, per mil; TU = tritium units; pMC = percent modern carbon).

Site number	Type	Aquifer	Date	Latitude (degrees)	Longitude (degrees)	Altitude (masl)	Depth (m)	$\delta^{18}O$ (‰)	$\delta^{2}H$ (‰)	Tritium (TU)	$\delta^{34}S$ (‰)	$\delta^{13}C$ (‰)	^{14}C (pMC)	Area (Fig. 1)
Groundwater from hard rock (Eastoe and Wright 2019)														
1	Well	hr	1999	32.244	-111.1435	871		-7.2	-53	5.4		-12.5		D
2	Well	hr	1998	32.2428	-111.1660	866	50.3	-7.2	-56	0.8		-7.9	38.9	D
3	Spring	hr	1999	32.2391	-111.1252	891	0.0	-7.9	-57	2.4		-10.7	70.0	D
4	Well	hr	7/27/2002	32.2896	-111.1102	805	150.9	-8.6	-64	<0.5	2	-8.6	33.1	C
5	Well	hr	7/27/2002	32.2885	-111.1097	828	152.4	-8.5	-62	<0.7	0.9	-10.5	28.0	C
6	Well	hr	2013	32.2812	-111.1049	825	274.4	-9.9	-75		3.4	-9.2	7.8	C
7	Well	hr	1998	32.3093	-111.1701	786	152.4	-8.1	-58	<0.7		-8.8	29.5	other
8	Well	hr	2009	32.2584	-111.0993	848	192.1	-9.4	-71			-8.6	28.4	S
9	Well	hr	2009	32.2591	-111.0978	836	122.0	-7.8	-57			-9.3	39.4	S
10	Well	hr	2009	32.2584	-111.0946	822		-8.7	-64					S
11	Well	hr	2009	32.2654	-111.0984	819	121.6	-8.6	-66					S
12	Well	hr	2009	32.2653	-111.0990	819		-7.6	-55					S
13	Well	hr	2009	32.2646	-111.0961	822	91.5	-8.7	-66					S
14	Well	hr	3/15/2003	32.2661	-111.0983	817		-8.6	-64	2.0	2.1	-10.4	53.5	S
Alluvium (Eastoe et al. 2004; previously unpublished)														
15	Well	all	5/12/1999	32.2205	-111.1435	802		-8.0	-55	1.0		-8.6	35.2	W flank
16	Well	all	11/10/1998	32.3191	-111.2385	659						-8.9	31.2	W flank
17	Well	all	11/10/1998	32.3242	-111.2258	668		-8.3	-59	<0.5	5.4			W flank
18	Well	all	11/13/1998	32.3270	-111.2175	675		-8.6	-61	<0.5				W flank
19	Well	all	10/3/1990	32.2775	-111.2396	674		-7.8	-55	<0.8		-9.1	38.0	W flank
20	Well	all	11/12/1998	32.3275	-111.2195	673		-8.5	-60	<0.6				W flank
21	Well	all	11/12/1998	32.3441	-111.2173	660		-8.5	-60	<0.6		-8.0	22.0	W flank
22	Well	all	11/12/1998	32.3065	-111.2500	656		-8.4	-59	<0.6				W flank
23	Well	all	11/12/1998	32.3104	-111.2362	673		-8.0	-57	<0.6				W flank
24	Well	all	7/21/1993	32.267	-111.0669	746	170.7	-7.6	-50	<0.7		-10.1	32.8	E flank
25	Well	all	7/13/1993	32.267	-111.0669	746	198.2	-7.7	-54	<0.7				E flank
26	Well	all	3/22/2003	32.3458	-111.1262	677	152.4	-8.4	-61	1.8	5.3	-11.0	65.0	E flank
27	Well	all	9/13/2000	32.381	-111.135	640		-8.2	-57	6.8				E flank
Old Yuma Mine area (Beisner and Gray 2018)														
28	Well	hr	01/11/16	32.32140	-111.11407	718.4	140.2	-7.6	-56	2.8	4.3	-13.9	101.7	Y
29	Well	hr	01/11/16	32.32127	-111.10762	725.7	213.4	-8.3	-63	<0.1	4.3	-11.6	49.0	Y
30	Well	hr	01/21/16	32.32528	-111.11002	710.0	118.9	-7.7	-57	<0.1	3.3	-11.9	71.9	Y
31	Well	all	01/29/16	32.32975	-111.10237	698.9	86.6	-8.2	-61	<0.1	3.4	-9.4	23.9	Y
32	Well	hr	02/08/16	32.32500	-111.11902	727.2		-7.3	-55	1.7	14.0	-14.7	99.7	Y
33	Well	all	02/09/16	32.31103	-111.09729	746.6	128.4	-8.9	-69	<0.1	6.9	-10.1	17.0	Y
34	Well	hr + all	02/29/16	32.33194	-111.11228	701.6	118.9	-7.1	-51	0.3	6.3	-9.5	40.9	Y
35	Well	hr	08/16/16	32.32068	-111.10949	723.4		-8.6	-65	<0.1	4.1	-12.1	42.2	Y

Mineral samples

Site number	Location	Latitude (degrees)	Longitude (degrees)	Mineral	$\delta^{34}S$ (‰)
100	Near Gila Monster Mine	32.2905	-111.1272	Jarosite	7.8
101	Gould Mine	32.2580	-111.1662	Sulfate crust	1.4
102	Gould Mine	32.2580	-111.1662	Pyrite	0.6
103	Gould Mine	32.2580	-111.1662	Pyrite	0.5

Gray (2018) for area Y confirms that conclusion (Figures 2A, 4A).

In both datasets, groundwater of short residence time occurs within the green ellipses of Figures 4A and 4B. Such water matches a modified LMWL defined by precipitation for the wettest ~30% of months at 1,000 masl, rather than the LMWL defined by amount-weighted seasonal mean precipitation for all months at 1,000 masl. The seasonal means are based on long-term data for 747 masl in Tucson Basin (Eastoe and Dettman 2016) and have been adjusted for altitude to 1,000 masl using isotope lapse rates from Eastoe et al. (2004). This behavior is typical of groundwater in neighboring alluvial basins, and corresponds to a regional mechanism in which recharge occurs from summer and winter precipitation during wettest months (Eastoe and Towne 2018). In the case of the Tucson Mountains, the contributions of summer precipitation are about 50 to 75%.

Groundwater with > 100 pMC or finite tritium > 1 TU or both (sites 1, 3, 14, 28, and 32; Figure 4) occurs at sites that have received recharge since 1953. Two sites (28 and 32) are wells completed in Oligocene volcanic rock, two (1 and 3) are in pre-caldera rock units in area D, and only one (14) is in the Cretaceous-early Paleogene caldera rocks in which most of the wells in areas Y, C, and S are completed. Note that the field of recent recharge (green ellipses in Figures 4A and 4B) also encompasses groundwater with pMC as low as 39, indicating that the recent recharge mechanism

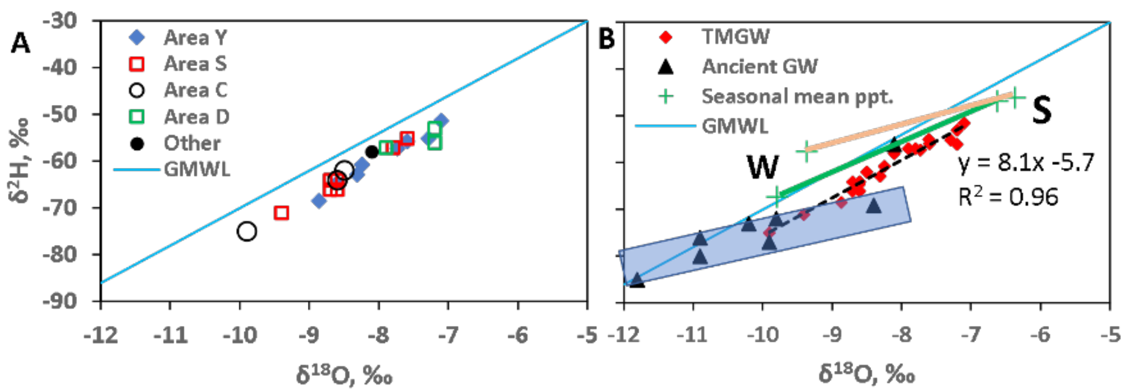

Figure 2. Plot of δ^2H vs. $\delta^{18}O$ for groundwater from the Tucson Mountains. A.) Classified by location (compare Figure 1 for cluster names). B.) In relation to mean isotope composition of seasonal precipitation at 1,000 masl, and isotope data for ancient groundwater (< 10% modern carbon) in the Tucson region (see text for data sources). TMGW = Tucson Mountains groundwater; GW = groundwater; GMWL = global meteoric water line (Craig 1961); S = summer; W = winter. Dashed line represents best fit regression line for TMGW data. Seasonal mean data with a brown tie-line are derived from data for all months in Tucson Basin; those with a green tie-line correspond to the wettest ~30% of months (Eastoe and Dettman 2016; Eastoe and Towne 2018).

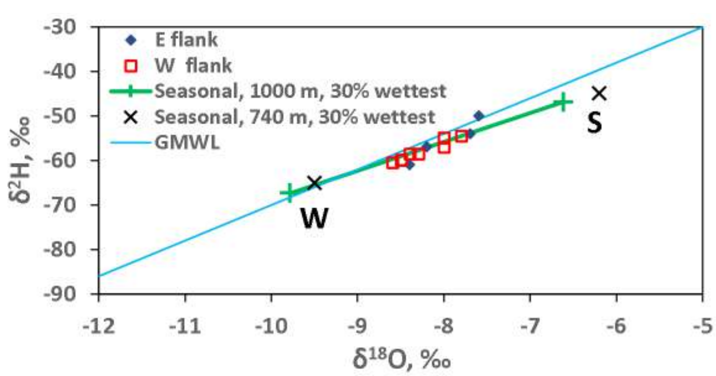

Figure 3. Plot of δ^2H vs. $\delta^{18}O$ for groundwater from wells completed in alluvium flanking the Tucson Mountains to the east (E) and west (W). Seasonal mean data for 740 and 1000 masl correspond to the wettest 30% of months (Eastoe and Dettman 2016; Eastoe and Towne 2018). GMWL = global meteoric water line (Craig 1961); S = summer; W = winter.

has operated for a considerable time, possibly thousands of years.

The number of examples is small, but these examples indicate that aquifer lithology influences the localization of recent recharge in the mountain block. Style of fracturing may play a role in enhancing recharge in certain lithologies; in addition, the type of soil profile developed on each rock type may play a role.

Residence Time of Low-δ^{18}O End Member

Beisner and Gray (2018) used criteria of Han and Plummer (2016) to establish which of their ^{14}C data could be corrected using a revised Fontes-Garnier method (Han and Plummer 2013). For instance, sample 33, containing 17 pMC, yielded corrected mean ages of 5,100 to 6700 years, the range reflecting assumptions about the pMC in dissolved rock carbonate. However, corrections of ^{14}C data using δ^{13}C as an indicator of dissolution of rock carbonate are problematic where mixing contributes to observed isotope compositions. First, mixing ratios are not accurately known. The bulk ^{14}C content, 17 pMC, might represent one of many possible mixing scenarios between older water with pMC < 17 and younger water with pMC > 17. Second, the correction equations are not constructed to account for mixing. An alternative approach to constraining the residence time of the low-δ^{18}O end member arises from its distinctive values of δ^{18}O and δ^2H. An increase in values of δ^{18}O (typically 2 to 3 ‰) and δ^2H is inferred in precipitation, commonly near the end of the Pleistocene, both in southwestern North America (e.g., Phillips et al. 1986) and globally (Jasechko et al. 2015). In southern Arizona, the shift occurred between 13,000 and 15,000 years ago, on the basis of a speleothem δ^{18}O record (Wagner et al. 2010)

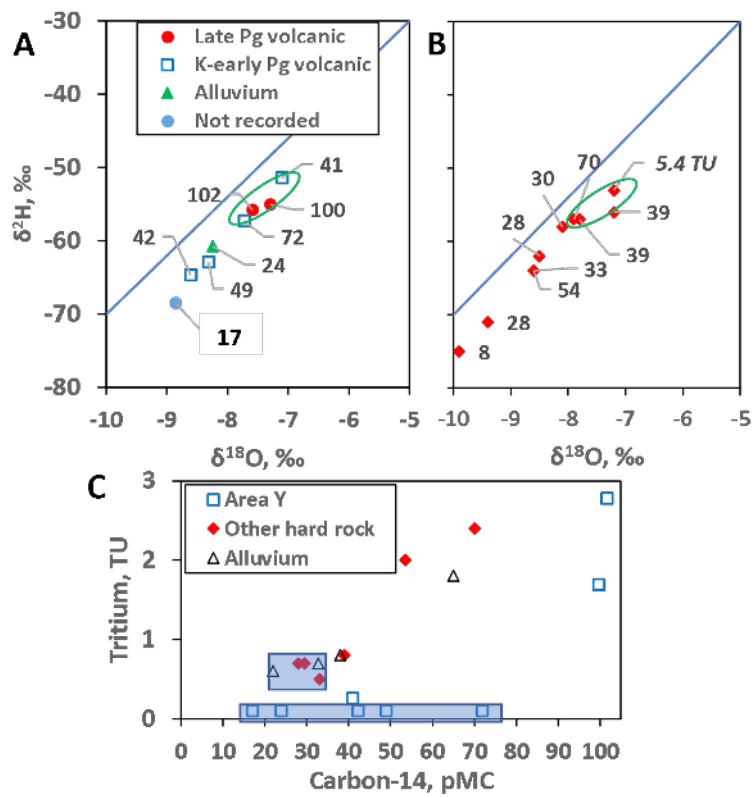

Figure 4. A.) Plot of δ^2H vs. δ^{18}O for groundwater from the Old Yuma Mine area (data of Beisner and Gray 2018). Data points are numbered corresponding to Figure 1 and Table 1, and classified according to the rock type in which each well was completed (records of Arizona Department of Water Resources 2020b). B.) Plot of δ^2H vs. δ^{18}O for groundwater from other groundwater hosted in hard rock (data of Eastoe and Wright 2019). In A and B, data points are labeled with ^{14}C content (% modern carbon, pMC, non-normalized) or tritium content (tritium units, TU), and the green ellipses indicate the field of post-bomb recharge. C.) Tritium vs. ^{14}C content in groundwater from the Tucson Mountains and flanking alluvium. Shaded blue rectangles enclose points for which tritium was below detection. For these points, the tritium value is plotted as the detection limit, 0.1 TU for area Y and 0.5-0.7 TU for other data.

from the Santa Rita Mountains, 70 km SSE of area S. The residence time of the low-$\delta^{18}O$ end member is therefore more than 13,000 years. Sample 6, with 8 pMC, falls on a broad evaporation trend (Figure 2B) defined by other ancient groundwater (< 10 pMC, uncorrected; data from Eastoe et al. 2004; Montgomery and Associates, Inc. 2009; Hopkins et al. 2014; Eastoe and Gu 2016; Tucci 2018; Schrag-Toso 2020) in the region around Tucson. Recharge of the low-$\delta^{18}O$ end member occurred from evaporated meteoric water. The seasonality of recharge in this case cannot be determined.

The presence of late Pleistocene recharge and the paucity of post-bomb recharge in most of the mountain block indicates that climate change has influenced the hydrology of the Tucson Mountains. Changes in recharge mechanism are probably related to the abundance of surface water, and may reflect climate change at the time-scale of the Holocene as indicated elsewhere in southwestern North America (Phillips et al. 1986; Wagner et al. 2010), or between the Little Ice Age and the present (discussed in a nearby study area by Eastoe 2020).

Groundwater Age, Flanking Alluvium

Most samples conform to a mixing line between mean winter and summer precipitation in the wettest months (Figure 3). A few samples contain finite tritium (sites 15, 26, and 27), indicating the presence of some post-1953 recharge. Several samples contain 22-35 pMC, indicating recharge that may be thousands of years old. The low-$\delta^{18}O$ end member discussed in the previous section is absent in the alluvium. Therefore, there is no evidence for recharge older than 13,000 years in the alluvium.

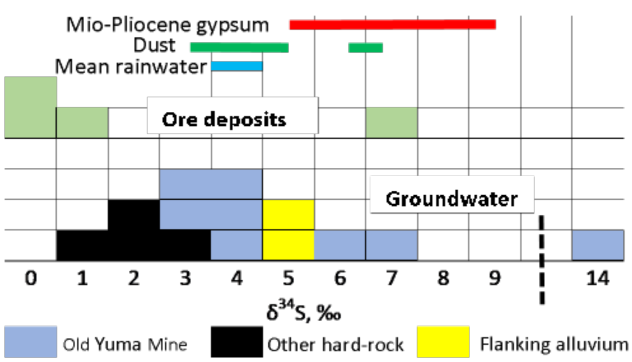

Figure 5. Frequency histogram of $\delta^{34}S$ data in groundwater and ore-related mineral samples, in relation to amount-weighted mean precipitation, dust and gypsum evaporite from central Tucson Basin (Gu 2005).

Nature of the Hard-rock Aquifer

Groundwater in the hard rock of the Tucson Mountains may reside in one or more porous strata, or in fractures with or without hydrologic connection. Groundwater with distinctive isotope compositions is closely juxtaposed in areas Y, C, and S (Figure 1). This is clearest in area S, where sites 8 ($\delta^{18}O$ = -9.4 ‰; 28.4 pMC, little dissolved Fe^{2+}) and 9 ($\delta^{18}O$ = -7.8 ‰; 39.4 pMC, containing dissolved Fe^{2+}) are about 100 m apart. Other wells, sites 10-13, within a few hundred meters of sites 8 and 9, produce water with $\delta^{18}O$ between -7.6 and -9.4 ‰. These observations are consistent with an aquifer or aquifers consisting of a poorly-connected system of fractures.

Mountain-block Recharge

At site 31 and possibly site 33, groundwater is pumped from basin-fill alluvium. At both sites, $\delta^{18}O$ and $\delta^{2}H$ data conform to the general pattern for the hard-rock aquifer (Figure 2A) and values of pMC, 33 and 17 respectively, are the lowest in area Y. Mountain-block (i.e., subsurface) recharge into alluvium is indicated near these sites. Other samples from flanking alluvium near the Tucson Mountains have a different pattern of $\delta^{18}O$ and $\delta^{2}H$ data (Figure 3), indicating that mountain-front recharge (i.e., from the surface where mountain drainages intersect the range front) predominates.

Water Quality

Groundwater from hard rock with $\delta^{34}S$ values lower than +3.5 ‰ probably contains a mixture of rain and dust sulfate with sulfate from oxidation of ore sulfide (Figure 5). Where sulfide oxidation has occurred in the hard-rock aquifer, the groundwater is also likely to contain dissolved base metals. At site 9, the well owner reported dissolved iron in the groundwater. The single sample with $\delta^{34}S$ = +14 ‰ occurs with the highest sulfate concentration, 134 ppm, in area Y (Beisner and Gray 2018). Groundwater in this well smelled of H_2S, consistent with bacterial sulfate reduction as the reason for the high $\delta^{34}S$ value.

Implications for Water Supply

Areas Y, C, and S, with low-density urban development, rely on groundwater pumped from a system of fractured-rock aquifers. The isotope evidence is consistent with little connection between fractures. Available volumes of water are therefore limited, and likely to vary from fracture to fracture. Tritium and ^{14}C data provide little evidence of replacement of groundwater by post-1953 recharge in these areas. Even if post-1953 recharge was initially present and has been removed by pumping of shallower groundwater, such water does not appear to have been replaced in recent decades. In area C (sites 4 and 5), static water levels were falling at the time of sampling based on information provided by well owners. Recharge to the mountain block under present conditions appears to be slow to non-existent. Water supply therefore appears limited, and at many sites is dependent on recharge that occurred thousands of years ago. In the absence of municipal water supply, collection of rainwater from roofs or hauling of water from elsewhere may be necessary to supplement waning groundwater supply. Capture of rainwater would have insignificant effect on recharge, given that little or no post-bomb recharge appears to be occurring in most of the mountain block.

Sustainable water supply may be possible where post-bomb replenishment of groundwater is occurring, in areas D, S, and Y. Targeted exploration, for example in the Oligocene volcanic rocks at the north end of the Tucson Mountains, may locate a renewable, but not necessarily large, water supply.

Conclusions

In the Tucson Mountains, stable O and H isotope data proved to be useful in identifying groundwater mixing and constraining groundwater residence times. Measurements of ^{14}C and tritium were useful in identifying post-bomb recharge. S isotope data helped to explain water quality issues.

Groundwater in fractured-rock aquifers in the Tucson Mountains is a mixture of recharge of different ages. Younger water, recharged since about 13,000 before the present, is a mixture of summer and winter recharge occurring during wettest months; in general, summer recharge has predominated. A similar recharge mechanism operates in alluvium flanking the range. Older groundwater has low ^{14}C content and a $\delta^{18}O$ signature consistent with recharge before 13 Ka. The seasonality of the older recharge is not known. Mountain-block recharge from fractured rock to basin alluvium occurred locally near the Old Yuma Mine. Post-bomb recharge occurs in Oligocene volcanic rock and Cretaceous sedimentary rock, but is uncommon in the Cretaceous-early Paleogene caldera complex that makes up most of the mountain block. These units might provide a renewable groundwater resource. The water-bearing fractures in the rest of the range appear to be poorly connected and receive little recharge at present. Water supply in the mountain block is therefore limited in volume, and is of variable quality where sulfide mineralization is present.

Acknowledgements

CJE acknowledges private well owners for permission to obtain samples; Jay Cole, John Schiff, and Robert Webb are thanked for help in contacting well owners; and the Environmental Isotope Laboratory at the University of Arizona is thanked for funding the analyses other than those in area Y. Analyses in area Y were funded by the U.S. Geological Survey. The authors thank two anonymous reviewers for their constructive comments. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 175, Pages 15-32, April 2022

"I Believe I Can and Should": Self-efficacy, Normative Beliefs and Conservation Behavior

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Abstract: This study examines the social-psychological drivers of conservation action among landowners in Minnesota. In particular, we apply an integrated norm activation theory to understand landowner conservation behavior. Data were collected through a self-administered mail survey of 3,000 landowners in La Crescent and Reno Watersheds in Southeastern Minnesota and analyzed using structural equation modeling. Study findings show that landowners' conservation action is driven by their feelings of personal obligation, and beliefs about whether one is capable of taking actions to influence outcomes (i.e., self-efficacy). Landowners who feel a sense of personal obligation and believe that their actions can make a difference are more likely to take conservation actions. Further, landowners who believe it is their personal responsibility to protect water and perceive social expectations are more likely to develop feelings of personal obligation. Importantly, this study highlights the role of self-efficacy as an activator of personal norm, as well as a driver of conservation behavior. Our study suggests that strategies that appeal to landowners' sense of personal responsibility and self-expectations, promote conservation action as a social norm, and build landowners' self-efficacy or confidence in their ability to make a difference, are likely to be successful.

Keywords: *water conservation, landowner behavior, norms*

Conservation professionals and environmental managers throughout the state of Minnesota invest considerable time and money on outreach, education, and technical assistance programs to promote conservation practice adoption and protect invaluable water resources. Despite these efforts, non-point source (NPS) pollution continues to be of significant concern across the state. Every county in the state has an impaired water body. Altogether, more than 5,000 water bodies are listed as impaired for one or multiple uses. This includes more than 1,800 impaired water bodies in the Upper Mississippi River Basin (MPCA 2016). The Upper Mississippi River Basin, which includes large portions of the states of Illinois, Iowa, Minnesota, Missouri, and Wisconsin, provides life-sustaining ecosystem services for wildlife habitat, cultural preservation, public water supply, navigation, commerce, and recreation. Water impairments in the Upper Mississippi River Basin have significant impacts on ecosystem functioning and community well-being within Minnesota, as well as in downstream communities and the Gulf of Mexico (HTF 2018).

Research Implications

- Feelings of personal obligation and beliefs about one's ability to make a difference are key drivers of landowner conservation behavior.
- Study findings show that conservation outreach and programming that appeal to landowners' sense of responsibility and personal norms are likely to motivate landowners to take conservation action.
- Programs that build landowners' self-efficacy, or confidence in their ability to make a difference, also are essential to supporting and sustaining conservation behaviors.

Current approaches to managing NPS pollution in Minnesota and across the Midwestern U.S. rely predominantly on voluntary action of landowners, agricultural producers, residents, and other resource users. How to best engage and inspire conservation action among key actors is a critical question for environmental management agencies and organizations (Nelson et al. 2017). Protecting and restoring water is particularly challenging in a state like Minnesota where 75% of its land is in private ownership. In Minnesota, Soil and Water Conservation Districts (SWCDs) and watershed districts (WDs) play a prominent role in private land conservation. SWCDs and WDs develop comprehensive plans, implement capital improvements, provide technical and financial assistance to landowners, and develop educational and outreach programs that promote natural resource conservation (MNBWSR 2019a; 2019b). WDs and SWCDs that work directly with landowners to install conservation practices largely rely on landowners to initiate the process. Thus, for these agencies and organizations, understanding landowners—what motivates and constrains their conservation decisions and actions—is essential to their work and to making programs and practices appealing.

Landowner conservation decision-making is complex, and there are no universal predictors or models for conservation action (Prokopy et al. 2008; 2019). Researchers have investigated the drivers of private-sphere (e.g., adoption of conservation practices) and public-sphere (e.g., civic engagement) conservation behavior. Past research has associated multiple types of variables, including land and landowner characteristics (e.g., land size, tenure, education, age, gender), and economic factors (e.g., income, land value) (Manzo and Weinstein 1987; Smith 1994; Koehler and Koontz 2008; Larson and Lach 2010; Prokopy et al. 2019) with behavior. Studies also have examined the social-psychological determinants of conservation action. Constructs such as environmental attitudes and awareness, perceived practice characteristics (e.g., Reimer et al. 2012; Arbuckle and Roesch-McNally 2015), attachment to land (e.g., Ryan et al. 2003), self-efficacy (e.g., Perry and Davenport 2020), values, and norms (e.g., Pradhananga and Davenport 2019) have been linked with conservation practice adoption. A review of qualitative studies examining motivations and barriers to conservation practice use identified several motivators including environmental awareness, concern, trust in information sources, and farmers' stewardship identity, as well as barriers such as negative perceptions about conservation practices and perceived risks of practice adoption (Ranjan et al. 2019). Similarly, increased levels of civic engagement and participation in conservation initiatives have been associated with feelings of personal responsibility (Story and Forsyth 2008), pro-ecological worldview and trust (Larson and Lach 2010), self-efficacy (Martinez and McMullin 2004), community attachment and environmental concern (Brehm et al. 2004; 2006; Pradhananga and Davenport 2017), and personal norm (Raymond et al. 2011; Pradhananga et al. 2015; 2017; Vaske et al. 2020). We build on this line of research by investigating the social-psychological drivers of conservation action among landowners in Minnesota. In particular, we examine conservation as an "other" interest or pro-social (as opposed to self-interest) behavior, and apply an integrated norm activation theory to understand landowner conservation behavior.

Normative Approach to Conservation Behavior

Theories such as the norm activation theory (NAT) posit that individual actions that have consequences for others are moral choice situations. In moral choice situations, feelings of personal obligation, or personal norm, strongly influence one's behavior. Individuals take actions that are consistent with their internal self-evaluations (Schwartz 1977).

According to moral approach theories such as the NAT and Value-Belief-Norm (VBN) theory (Stern 2000), values and beliefs activate personal norm, which influences behavior. This conceptualization of cognitions from values, beliefs, and norms to behavior is consistent with the cognitive hierarchy theory, which postulates that human cognitions are organized in a hierarchy from values to behaviors (Fulton et al. 1996). The specific beliefs that activate personal norm are awareness of consequences of one's actions or an environmental condition (i.e., awareness of consequences), and beliefs about responsibility for those consequences (i.e., ascription of responsibility) (Schwartz 1977; Stern 2000). There is ample empirical support for the relationships posited in the NAT and VBN theory (Bamberg and Möser 2007). Studies have demonstrated the positive effect of personal norm in a wide range of behavioral contexts including water conservation (Harland et al. 2007; Landon et al. 2017), recycling (Nigbur et al. 2010), energy conservation (Ibtissem 2010), and willingness to accept climate change strategies (Nilsson et al. 2004). The NAT, VBN theory, and related concepts have also been applied in the context of landowner and farmer conservation behavior. Past work in this area has provided evidence to support links between personal norms and conservation practice adoption (Pradhananga and Davenport 2019), participation in conservation programs (Johansson et al. 2013), conservation of native vegetation (Raymond et al. 2011), and civic engagement in water management (e.g., Pradhananga et al. 2015; 2017). For example, a study of farmer conservation practice adoption in Minnesota reported that farmer personal norm was a direct predictor of practice adoption (Pradhananga and Davenport 2019). In a study of Swedish landowners, Nilsson et al. (2004) found that landowners who participated in forest preservation or wetland restoration programs reported higher levels of awareness of consequences, personal responsibility, and personal norm than those that did not participate. Johansson et al. (2013) reported that landowners who had participated in conservation programs were more aware of the consequences of threats to biodiversity, ascribed greater responsibility to themselves, and felt a personal obligation to participate in biodiversity conservation than landowners who did not participate. More recently, Vaske et al. (2020) found that normative beliefs influenced farmers' decisions to participate in conservation programs without compensation. Related constructs such as the "good farmer identity" (McGuire et al. 2015), particularly the conservationist identity, characterized by stewardship ethic and long-term environmental concern, have also been shown to be related to conservation behaviors (e.g., McGuire et al. 2015; Dixon et al. 2021). For example, a study of farmers in Iowa found that wildlife conservationist identity was significantly related to likelihood of wildlife management practice use (e.g., using weedy fencerows, avoiding mowing) (Dixon et al. 2021).

Research in this area has also explored the norm activation process, in particular the relationships among awareness of consequences, ascription of responsibility, and personal norm. The NAT also defines ascription of responsibility and personal norm as distinct constructs. While responsibility is a measure of one's "sense of connection or relatedness" to a situation or individual in need, personal norms are "directed toward the performance of specific acts" (Schwartz 1977, p. 246). Further, denial of responsibility can also act as a defense mechanism, even in situations where feelings of obligation are activated. Past work in this area has provided empirical evidence to suggest that ascription of responsibility and personal norms are distinct psychological constructs (e.g., Stern 2000; Harland et al. 2007). Applications of the VBN theory suggest a chain of relationships where awareness of consequences influences ascription of responsibility, which in turn affects personal norm (e.g., Stern 2000; De Groot and Steg 2009; Pradhananga et al. 2017). Thus, ascription of responsibility appears to be a more proximal determinant of personal norm than awareness of consequences.

Subjective norm, or social pressure to take action (Ajzen 1991) has been shown to influence conservation behaviors and behavioral intentions (e.g., Corbett 2002; Pradhananga et al. 2015; Ranjan et al. 2019; Knapp et al. 2020). People are more likely to take action if they believe that others important to them approve of that behavior (Ajzen 1991). Studies have provided empirical support for the relationship between subjective norms and landowner conservation behavior. For example, in a study of private landowners in Texas, Sorice and Conner (2010) reported a significant influence of subjective norm on landowners' intentions to enroll in an incentive program to protect endangered species. A study of cattle ranchers also found that subjective norm was a significant predictor of intentions to engage in wildlife management (Willcox et al. 2012). Vaske et al. (2020) reported a significant influence of subjective norm on Illinois farmers' intention to participate in conservation programs. While a meta-analysis of studies applying the Theory of Planned Behavior (TPB) to environmental behavior found generally weak relationship between subjective norm and behavioral intention (Armitage and Conner 2001), other studies have provided support for the influence of subjective norms on personal norms and behavior (Bamberg and Möser 2007; Klöckner 2013). While not explicitly included in the NAT, Schwartz (1977) suggests that subjective norms may be internalized as personal norms, which in turn influence behavior. Literature in this area suggests that the extent of social pressure one feels to take actions such as using conservation practices can have an influence on feelings of personal obligation and intentions to take action.

Self-efficacy and Behavior

In the social cognitive theory, Bandura (1977; 2001) argues that human agency is characterized by beliefs about one's capability to achieve goals or outcomes, also defined as self-efficacy. Self-efficacy represents human capacity of self-reflectiveness to evaluate their own motivations and values. Beliefs about one's efficacy influences "how people feel, think, and act" (Bandura 1990, p. 128). Beliefs about whether or not one is capable of taking actions affect what actions people take and how much effort they put into performing a behavior (Bandura 2001). In the context of landowner conservation behavior, confidence in one's ability to use conservation practices (i.e., self-efficacy) can be expected to affect an individual's intentions to take conservation action.

Research has consistently linked self-efficacy with behaviors related to public health (e.g., health promotion, disease prevention, physical activity) (Bandura 1998; Plotnikoff et al. 2008). While not extensively applied to environmental behaviors, a subset of studies have linked self-efficacy with environmental behaviors such as recycling (e.g., Tabernero and Hernandez 2011), transportation choice (e.g., Jugert et al. 2016), invasive species management (e.g., Clarke et al. 2021a; 2021b), and landowner conservation behavior (e.g., Wu and Mweemba 2010; Perry and Davenport 2020). For example, a study of residents in Spain (Tabernero and Hernández 2011) reported that residents who perceived a greater capacity to recycle (i.e., higher levels of self-efficacy) engaged in more recycling behaviors. Self-efficacy has also been found to be positively associated with intentions to conserve energy (Lee and Tanusia 2016), and support for biodiversity (Clayton et al. 2017). A qualitative assessment of farmer decision-making identified low levels of perceived self-efficacy as a significant barrier to conservation agriculture (Perry and Davenport 2020). A study of farmers in Iran reported a significant effect of self-efficacy on farmers' water conservation behavior (Yazdanpanah et al. 2015). Studies of family forest owners have also reported a significant influence of self-efficacy on their intentions to engage in invasive plant management (Clarke et al. 2021a; 2021b).

While self-efficacy has not been applied extensively to landowner conservation behavior, two related constructs, perceived ability and perceived behavioral control, have received much attention. Perceived ability, or perceptions about the availability of resources to take action (Schwartz 1977), and perceived behavioral control (i.e., perceptions about the level of ease or difficulty of performing a behavior) (Ajzen 1991) have been shown to affect conservation action (Harland et al. 2007; Chan and Bishop 2013; Pradhananga et al. 2017; Scalco et al. 2017; Wilson et al. 2018; Pradhananga and Davenport 2019) as well as personal norm (Pradhananga et al. 2015; Pradhananga and Davenport 2019). In the NAT, ability to take action is postulated as a necessary precondition for the activation of personal norms. Feelings of personal obligation to take action, or personal norms, are more likely to be activated if one believes that they have the ability to take such action. Further, denial of ability may neutralize personal norms even when they have been formed (Schwartz 1977).

While perceptions about the ease or difficulty of taking an action may affect confidence in one's ability to attain outcomes (i.e., self-efficacy), self-efficacy and perceived behavioral control are distinct constructs. Self-efficacy is a broader concept that incapsulates the idea of perceived ability to act (e.g., use conservation practices) to attain certain outcomes (e.g., improve water quality). Constructs such as self-efficacy, perceived ability, and perceived behavioral control are useful in understanding factors that may constrain conservation norms and behaviors. In the current paper, we integrate self-efficacy in the NAT to examine the relationships among self-efficacy, personal norm, and behavioral intention, in the context of water resource management. Specifically, in this study's conceptual model (Figure 1), we hypothesize that personal norm will have a positive influence on intended conservation behavior. Self-efficacy, ascription of responsibility, and subjective norm are hypothesized as positive predictors of personal norm.

Methods

We administered a mail survey with 3,000 landowners, including agricultural landowners in La Crescent and Reno Watersheds in southeastern Minnesota. The sampling frame was generated using a list of property owners obtained from Winona and Houston Counties' publicly available landowner parcel data. The sample consisted of Winona and Houston County landowners who live within the two study watersheds. A random sample of 1,500 landowners from each watershed were selected for survey mailing. An adapted Dillman et al. (2014) Tailored Design Method was used to increase response rate, and included three waves of mailing. Each mailing included a questionnaire, cover letter, map of the watershed, and a postage-paid envelope. The surveys were administered from March to July 2018.

The questionnaire inquired about landowners' beliefs about water pollution, perspectives on water management, engagement in conservation behaviors, and sociodemographic information. The survey questionnaire was designed based on past research, particularly around conservation behavior in the Midwest (Prokopy et al. 2008; Pradhananga et al. 2015; Pradhananga and Davenport 2019).

Study Site

The Mississippi River-La Crescent Watershed stretches across Winona and Houston Counties. Pine Creek is the largest stream in the watershed (MPCA 2018a). The major land cover in the watershed is forest (47%), with 27% of the watershed in cropland (MNDNR 2015a). Major resource concerns in the watershed include soil erosion, total suspended solids, low dissolved oxygen, nitrate, and degradation of stream habitat (USDA NRCS 2007; MPCA 2018b). Stretches of the Pine Creek and Mississippi River are listed as impaired due to *Escherichia coli* and polychlorinated biphenyl (PCB) (MPCA 2016). The Mississippi River-Reno Watershed is located entirely in Houston County. Crooked Creek and Winnebago Creek are the largest streams in the watershed (MPCA 2018a). The major land cover in the watershed is cropland (42%), followed by forest (37%) (MNDNR 2015b). Soil loss and

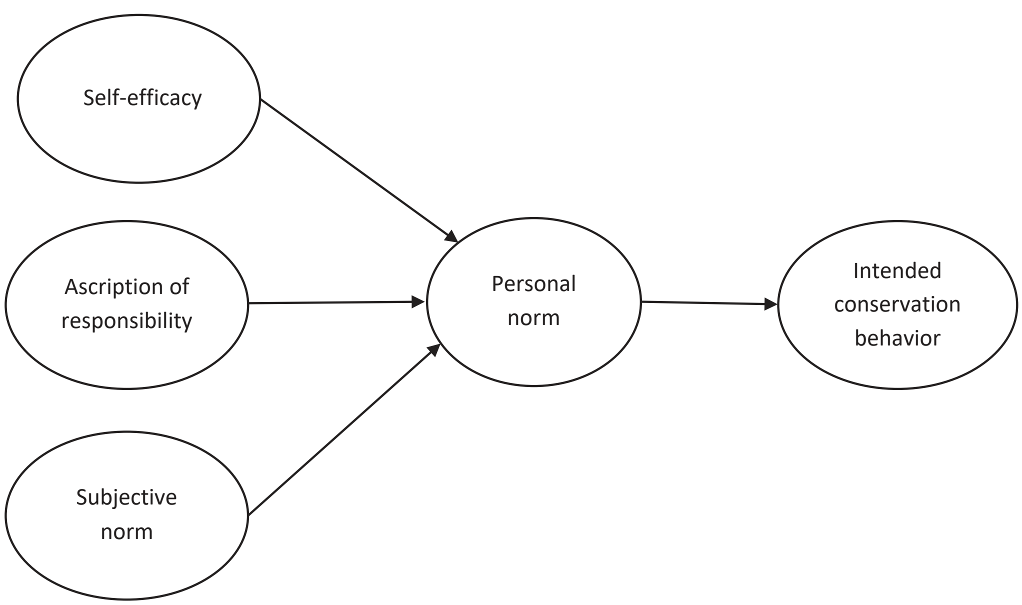

Figure 1. Study conceptual model.

oxygen depletion are major resource concerns in the watershed (USDA NRCS 2008). Stretches of Crooked Creek and Winnebago Creek are listed as impaired for *E.coli* and aquatic macroinvertebrate bioassessments (MPCA 2016). Residents in the Houston and Winona Counties are predominantly White (97% and 94%, respectively) and non-Hispanic (U.S. Census Bureau 2019, American Community Survey 5-year estimates). Median age varies; Winona County residents (Med = 35) overall are younger than Houston County residents (Med = 45), and gender identity reported is evenly split between male and female. About 25% of residents in Houston County and 30% of residents in Winona County have a Bachelor's degree or higher, and median income is about $60,000 in each county (Table 1).

Measures

Ascription of responsibility was measured using two items adapted from Pradhananga et al. (2019). An example item measured is "It is my personal responsibility to help protect water." Respondents were asked to rate two statements on a five-point Likert type scale from strongly disagree (-2) to strongly agree (+2). ***Subjective norm*** was measured using two items based on suggestions from Ajzen (1991) and adapted from past empirical work (e.g., Karppinen 2005; Bernath and Roschewitz 2008; Pradhananga et al. 2015). Items included "People who are important to me expect me to use conservation practices on my land" and "People who are important to me expect me to maintain my land/farm in a way that does not contribute to water resource problems." Respondents rated each statement on a five-point Likert type scale from strongly disagree (-2) to strongly agree (+2). ***Self-efficacy*** was measured using three items rated on a four-point scale from not at all capable (0) to very capable (3). Following recommendations from Bandura (2006) and adapted from an application in a study about recycling (Tabernero and Hernández 2011), the response scale was developed as a unipolar scale. The question stem was framed as "To what extent do you believe you are capable of the following?" The items included "Using a new conservation practice on the land/farm," and "Changing land use practices to reduce impacts on water resources." ***Personal norm*** was measured using three items adapted from past applications of normative theories to conservation behavior (e.g., Harland et al. 2007; Pradhananga et al. 2015; 2019). Respondents rated each statement on a five-point Likert type scale from strongly disagree (-2) to strongly agree (+2). An example item is "I feel a personal obligation to use conservation practices on my land/property." ***Intended conservation***

Table 1. Study area demographic characteristics.

		Houston	Winona
Gender:	Male	50.2%	49.5%
	Female	49.8%	50.5%
Origin:	Hispanic or Latino	1.1%	3.0%
Race:	White alone	96.9%	93.6%
	Other races	2.2%	4.7%
	Two or more races	0.9%	1.7%
Age:	Median (of all resident population)	45.3	35.2
	65 years and over (of 18 and over population)	26.5%	20.0%
Education:	Bachelor's degree or higher	24.8%	30.1%
Income:	Median income	$60,382	$59,329

Source: U.S. Census Bureau, 2019 American Community Survey 5-year estimates.

behavior was measured using two items on a five-point scale from most certainly not (-2) to most certainly will (+2). Intentions to engage in two behaviors were measured: "Use a new conservation practice on my land," and "Contact conservation assistance professionals about water resource initiatives."

Analysis

Convergent and discriminant validity were assessed using composite reliability and average variance extracted (AVE) (Fornell and Larcker 1981). AVE scores greater than 0.5, and composite reliability greater than 0.7 indicate adequate convergent validity (Raykov 1997; Hair et al. 2010). Discriminant validity is achieved if the correlations between latent constructs do not exceed the square root of AVE for either construct in the pair being compared (Fornell and Larcker 1981).

We used structural equation modeling to test the hypothesized relationships in the conceptual model (Figure 1). Model fit was examined by assessing several model fit indices. We considered the model to have adequate fit to the data if it had a relative chi-square (χ^2/df) of five or less (Schumacker and Lomax 2004), a root mean square error of approximation (RMSEA) less than 0.07 (Steiger 2007), standardized root mean square residual (SRMR) less than 0.08 (Hu and Bentler 1999), and incremental fit index (IFI) greater than 0.95 (Kline 2016). The model was estimated using the full information maximum likelihood method in LISREL 8.80.

We conducted mediation analysis to assess the direct and indirect effects of the exogenous variables (i.e., ascription of responsibility, subjective norm, and self-efficacy) on intended conservation behavior (Hayes 2013). The indirect effect of each exogenous variable on intentions for conservation behavior was calculated as the product of the predictor's (i.e., ascription of responsibility, subjective norm, self-efficacy) effect on the mediator (i.e., personal norm), and the mediator's effect on the criterion variable (i.e., intended conservation behavior). The Sobel test (Sobel 1986) was used to determine if the indirect effects were significant.

Results

Response Rate and Respondent Profile

Overall, 597 landowners completed the survey for a response rate of 23%. Response rates were 23% in La Crescent and 21% in Reno Watersheds.

Most respondents in both La Crescent (77%) and Reno (80%) Watersheds identified as male. A vast majority of respondents characterized their race and ethnicity as White (La Crescent: 98%, Reno: 92%). Median age among La Crescent (48% 65 years of age and over) and Reno (45% 65 years of age and over) Watershed respondents was 65 and 64, respectively. Almost half of the respondents in La Crescent Watershed (42%), and about one-third of respondents in Reno Watershed (35%) had attained at least a college bachelor's degree. A majority of respondents (59%) reported an annual household income of $75,000 or more in La Crescent, and 48% of Reno Watershed respondents reported an annual household income of $75,000 or more. Most respondents in La Crescent (82%) and Reno (66%) Watersheds did not use their land for agricultural production.

Comparisons with census statistics reveal that the survey respondent sample includes higher proportions of older adult (65 years of age and over) and of male-identifying residents than those residing in the two counties (Table 1). Demographic statistics also suggest that survey respondents overall have higher formal education attainment and income than area residents. Though these differences are consistent with other studies using similar sampling frames in rural areas (i.e., county property owner identification lists), it is important to acknowledge that the voices of residents who are younger, identify as female, or have lower household incomes are underrepresented in this study.

A majority of respondents believed that they are moderately to very capable of using a new conservation practice (57%) and maintaining conservation practices (70%) on their land/farm. A vast majority of respondents somewhat to strongly agreed that it is their personal responsibility to help protect water (88%), and to make sure that what they do on their land does not contribute to water resource problems (89%). Most respondents somewhat to strongly agreed that people who are important to them expect them to use conservation practices on their land (60%), and maintain their land in a way that does not contribute to water resource problems (72%). A majority of respondents also agreed that they feel a personal obligation to do whatever they can to prevent water pollution (84%), maintain their land/farm in a way that does not contribute to water resource problems (85%), and use conservation practices on their land/property (76%). Intentions of conservation behavior, however, were generally low. Only about a quarter of respondents reported that they probably to most certainly will use a new conservation practice on their land (26%), and fewer reported that they probably or most certainly will contact conservation assistance professionals about water resource initiatives (16%). While intentions to engage in conservation behaviors are low in the study watersheds, it must be noted that only 28% of La Crescent Watershed and 34% of Reno Watershed respondents reported that they used their land for agricultural production. Further, most of the outreach from conservation assistance professionals such as SWCDs in the study area focuses on farmers, rather than non-farm landowners. These factors may explain the low levels of intentions among survey respondents. The survey also inquired about landowners' current use of conservation practices. Most respondents reported using practices such as "using fertilizers/pesticides on lawns and gardens at recommended rates" (80%) and "planting trees as windbreak on land/property" (72%). Smaller proportions of respondents reported using practices such as "rain barrel or cistern to store water" (25%), and "rain garden" (15%). However, not all practices (e.g., cover crops and conservation tillage) are applicable to all landowners surveyed.

Structural Equation Modeling

Composite reliability exceeded the threshold of 0.7 for all latent constructs. Factor loadings of observed measures on latent constructs ranged between 0.71 and 0.91 (Table 2). The AVE of latent constructs ranged between 0.55 and 0.81. AVE square root scores of all latent constructs were greater than factor correlations between pairs of latent constructs (Table 3). These results demonstrate acceptable convergent and discriminant validity. These findings demonstrate that the latent constructs in the conceptual model, including ascription of responsibility and personal norms are distinct constructs.

The structural model with ascription of responsibility, subjective norm, and self-efficacy as exogenous variables, and personal norm and intended conservation behavior as endogenous variables demonstrated adequate model fit (Figure 2). Relative chi-square of the model was less than 5 ($\chi^2/df = 2.45$). RMSEA value was below the threshold of 0.07 (RMSEA = 0.049, 90% confidence interval: 0.038-0.061). IFI was 0.98, above the 0.95 threshold. The paths from self-efficacy ($\beta = 0.17$, $t = 3.83$), ascription of responsibility ($\beta = 0.38$, $t = 6.70$), and subjective norm ($\beta = 0.24$, $t = 4.58$) to personal norm were statistically significant. Personal norm was a statistically significant positive predictor of intended conservation behavior ($\beta = 0.22$, $t = 3.20$). Self-efficacy also had a direct and positive effect on intended conservation behavior ($\beta = 0.20$, $t = 3.52$). The model explained 18% of the variance in intended conservation behavior, and 40% of the variance in personal norm. The statistically significant indirect effects of the exogenous variables, ascription of responsibility, subjective norm, and self-efficacy on intended conservation behavior suggest that the relationship between the exogenous variable and intended conservation behavior is mediated by personal norm (Table 4). However, we also found that the direct effect of self-efficacy on intended conservation behavior was significant. This suggests that the effect of self-efficacy on intended conservation behavior is not completely mediated by personal norm.

Discussion

This paper contributes to the body of knowledge supporting a normative basis for pro-environmental behavior and arguing that self-interest alone does not fully capture what compels landowners to take conservation action. In this study we used an integrated norm activation model to examine landowner personal norms and conservation behaviors. Specifically, we investigated the influence of self-efficacy, personal responsibility, and subjective norms on respondents' personal norms and ultimately, their intentions to take conservation action. Findings indicate that landowners who feel a personal moral obligation to protect water have

Table 2. Descriptive statistics, reliability analysis, and factor loadings of items measuring constructs in the structural model.

Latent Variable	Survey Item	Mean*	SD	Factor Loadings (λ)	Composite Reliability (ρ)
Ascription of responsibility[a]	It is my personal responsibility to help protect water	1.36	0.82	0.78	0.81
	It is my personal responsibility to make sure that what I do on the land doesn't contribute to water resource problems	1.45	0.78	0.87	
Subjective norm[a]	People who are important to me expect me to use conservation practices on my land	0.75	0.88	0.89	0.90
	People who are important to me expect me to maintain my land/farm in a way that does not contribute to water resource problems	0.95	0.86	0.91	
Self-efficacy[b]	Using a new conservation practice on the land/farm	2.64	1.04	0.92	0.91
	Maintaining conservation practices on the land/farm	2.92	1.03	0.85	
	Changing land use practices to reduce impacts on water resources	2.65	1.07	0.85	
Personal norm[a]	I feel a personal obligation to do whatever I can to prevent water pollution	1.28	0.85	0.85	0.88
	I feel a personal obligation to maintain my land/farm in a way that does not contribute to water resource problems	1.34	0.88	0.90	
	I feel a personal obligation to use conservation practices on my land/property	1.09	0.94	0.78	
Intended conservation behavior[c]	Use a new conservation practice on my land	-0.13	1.05	0.78	0.71
	Contact conservation assistance professionals (e.g., my soil and water conservation district or the Natural Resources Conservation Service) about water resource initiatives	-0.39	0.98	0.70	

[a]Variables measured on a 5-point scale from *strongly disagree* (-2) to *strongly agree* (2). [b]Variables measured on a 4-point scale from *not at all capable* (0) to *very capable* (3). [c]Variables measured on a 5-point scale from *most certainly not* (-2) to *most certainly will* (2). SD = Standard Deviation.

stronger intentions to take actions, or in our case, to use a new conservation practice on their land and to contact conservation professionals about water resource initiatives. These findings confirm past research demonstrating a link between personal norms and pro-environmental behaviors in a wide range of behavioral contexts (Bamberg and Möser 2007), including farmer conservation behavior (Pradhananga and Davenport 2019), household water use (Harland et al. 2007), and household sanitation (Poortvliet et al. 2018). Our study results are important to conservation professionals because they offer evidence that protecting water is viewed by many landowners as a self-expectation, a moral obligation. For these individuals, protecting water is consistent with the concept of "self" and evokes positive self-evaluations (Schwartz 1977). Research examining farmers' identity has also shown that farmers with a conservationist or stewardship identity are more likely to engage in conservation behaviors (e.g., McGuire et al. 2015; Dixon et al. 2021).

Personal norms are also significant because the study shows they fully mediate the effect of responsibility and perceived social expectations (i.e., subjective norms) on conservation behaviors. In other words, the two antecedent beliefs, responsibility and perceived social expectations, do not directly influence intentions to act. They must first be internalized or activated as self-expectations to protect water and ultimately as intentions to act. Programs that appeal to landowners’ sense of personal responsibility and promote conservation as a social norm are likely to activate feelings of personal obligation, which in turn affects conservation behavior.

Of the three antecedent beliefs in the model, ascription of personal responsibility had the

Table 3. Discriminant validity matrix.

Constructs[a]	AR	SE	SN	PN	ICB
AR	**0.83** (0.69)				
SE	0.39	**0.87** (0.76)			
SN	0.57	0.34	**0.90** (0.81)		
PN	0.58	0.39	0.51	**0.84** (0.71)	
ICB	0.32	0.34	0.33	0.38	**0.74** (0.55)

[a]AR = Ascription of responsibility; SE = Self-efficacy; SN = Subjective norm; PN = Personal norm; CB = Conservation behavior. Note: Off-diagonal elements are correlations between constructs. Diagonal elements (bold) are the square root of average variance extracted (AVE) between the constructs and their indicators (AVE scores in parentheses). To meet the criteria for discriminant validity, off-diagonal elements should be less than 0.85 and AVE square root scores should be larger than correlations in the same row and column.

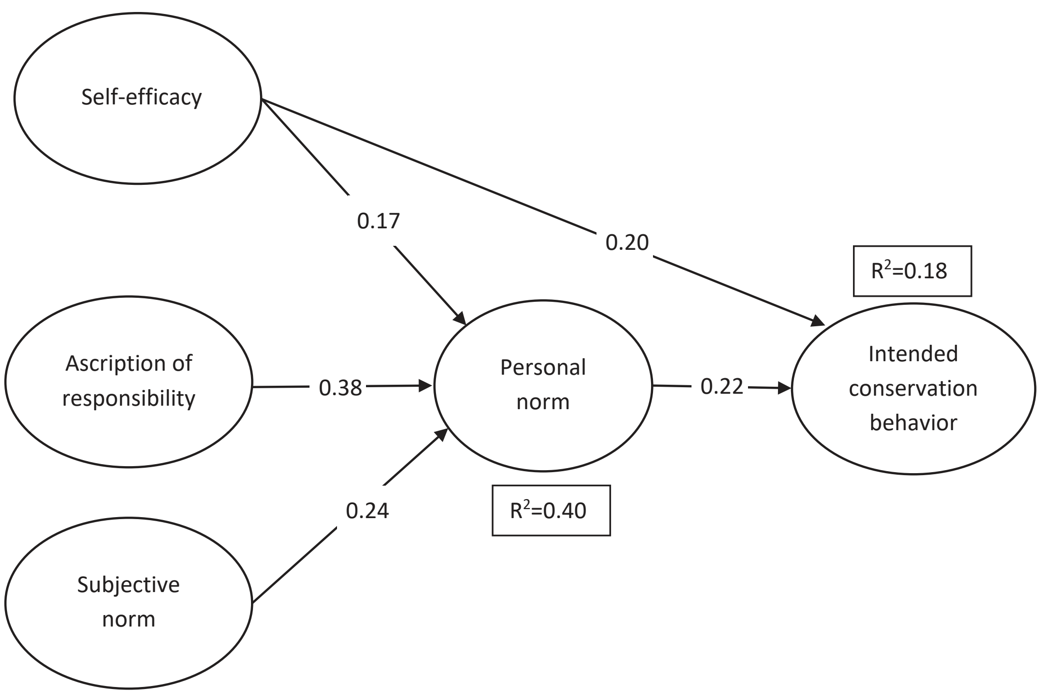

Figure 2. Standardized solution for structural model of beliefs, personal norms, and intended conservation behavior. Only statistically significant ($p \leq 0.05$) paths shown in figure. Chi-square (χ^2, $df = 44$) = 107.90; $\chi^2/df = 2.45$; Root Mean Square Error of Approximation (RMSEA) = 0.049 (90% CI: 0.038-0.061); Incremental Fit Index (IFI) = 0.98.

Table 4. Indirect effects of self-efficacy, ascription of responsibility, and subjective norm on landowners' intended conservation behavior.

Indirect Effect	Product of Unstandardized Coefficients	Z-statistic	p-value
SE[a]→PN[b]→ICB[c]	0.04	2.527	0.011*
AR[d]→PN→ICB	0.08	2.815	0.005*
SN[e]→PN→ICB	0.05	2.629	0.008*

[a]Self-efficacy; [b]Personal norm; [c]Intended conservation behavior; [d]Ascription of responsibility; [e]Subjective norm; *Statistically significant ($p \leq 0.05$).

biggest effect on personal norms. This finding is consistent with past applications of moral theories (e.g., NAT and VBN) which purport that feeling personally responsible for addressing a problem activates personal norms to take action (Stern et al. 1999; Harland et al. 2007; Pradhananga and Davenport 2019). In our study, landowners who believe it is their personal responsibility to protect water have higher self-expectations to take action.

Subjective norms, or perceived social expectations, also influence personal norms. Schwartz (1977) argues that behavioral norms shared by members of a group become self-expectations for individual group members as they are "learned" through social interactions (p. 231). This study's findings lend further credence to the subjective norm internalization process posited by Schwartz (1977) and are consistent with more recent research reporting a positive effect of social norms on personal norms in multiple behavioral contexts including landowner conservation behavior (Bamberg and Möser 2007; Klöckner 2013; Pradhananga et al. 2015). Importantly, our study conceptualized social norms as the expectations of people *important* to the respondent, assuming that important people, rather than society at large, have a bigger effect on personal norm development. For example, the broader farming community may not have established social norms for conservation. Yet, normative influences of *important* others such as family and like-minded conservationist farmers may influence norm activation among farmers.

A unique contribution of our work is the inclusion of the concept self-efficacy as an antecedent belief in the model. This approach is consistent with Bandura's (2001) social cognitive theory and Schwartz's (1977) NAT. Findings here demonstrate the role believing in one's ability to make a difference has in developing a personal norm and taking action to protect water. We found that self-efficacy has a direct effect on behavioral intention, as well as an indirect effect through personal norms. The effect of self-efficacy on personal norms is consistent with the norm activation process outlined in the NAT which suggests that perceived ability activates personal norms (Schwartz 1977). Lack of ability may also have a "neutralizing" effect on personal norms (Schwartz 1977, p. 246). Even when personal norms are activated, without the ability to take action, individuals may not be able to follow through on their feelings of obligation, which can result in negative self-evaluations. Further, the NAT also suggests that personal norms are activated when individuals believe that there are actions that can address a problem. This study shows that self-efficacy, or perceptions of one's ability to take action (e.g., use conservation practices) to meet certain outcomes (e.g., improve water quality) (Bandura 2001) also activates personal norms for water protection. Landowners are more likely to feel a sense of personal obligation to take action if they believe that they are capable of taking actions that are effective at addressing water resource problems.

Past studies have reported links between high levels of self-efficacy and pro-environmental behaviors including recycling (Tabernero and Hernández 2011), energy conservation (Lee and Tanusia 2016), support for biodiversity (Clayton et al. 2017), farmer decision-making (Perry and Davenport 2020), and landowner engagement in invasive species management (e.g., Clarke et al. 2021a; 2021b). Perceptions about whether one is capable of influencing outcomes not only affect behavioral choices, but also the amount and persistence of effort one is willing to put into the behavior (Bandura 2001). When people perceive that they do not have control over outcomes, they are less likely to take action and exert high or persistent effort into taking action (Bandura 2001). Adopting a new conservation practice is considered a “high-cost” pro-environmental behavior (Esfandiar et al. 2020), suggesting that it requires considerable time, money, and effort to undertake. Thus, the sentiments, “I believe I can” and “My actions will make a difference” become crucibles of conservation action. Two distinct dimensions of self-efficacy are essential to behavior: perceived ability to perform a conservation practice (e.g., Pradhananga et al. 2015) and perceived efficacy of the practice itself to ameliorate the problem. Besides attenuating action, low self-efficacy beliefs can lead to feelings of frustration, guilt, and hopelessness (Perry and Davenport 2020). Higher levels of self-efficacy, on the other hand, can lead to landowner engagement in actions to protect natural resources (e.g., Clarke et al. 2021a; 2021b).

One limitation of this study is that we focus on behavioral intentions and not behaviors. While intentions to act are positively correlated with actual behaviors, studies have also noted inconsistencies between intentions and behaviors (e.g., Sheeran and Webb 2016). Further, we did not account for the influence of past or current use of conservation practices on landowners’ intentions to use conservation practices in the future. Opportunities for future research exist in examining the influence of past use of conservation practices on landowners’ beliefs and intentions to use conservation practices in the future.

From a practical perspective, this study makes several key assertions that inform conservation programming. First, personal ethics are major drivers of conservation behavior. Conservation programs that focus solely on self-interest appeals, conventional science communication, and incentives like technical and financial assistance, may not have the return on investment intended, especially for those whose decisions hinge on feeling personally responsible for water protection, believing neighbors or local officials expect them to take action, or knowing they can make a difference in water outcomes. Second, low self-efficacy is doubly important as both a constraint to feeling morally obligated to act and as a barrier to the behavior itself, even when feelings of obligation exist. Not knowing how to take action, not believing one has the ability to take action, and not feeling that the action will make a meaningful difference are potentially high hurdles for conservation programming to overcome.

Our study suggests that strategies that appeal to landowners’ sense of personal responsibility and self-expectations, promote conservation action as a social norm, and build landowners’ self-efficacy, or confidence in their ability to make a difference are essential to supporting and sustaining conservation behaviors. Since beliefs about personal responsibility and social expectations do not have direct effect on intentions to engage in conservation, strategies that emphasize the activation of norm are more likely to be successful. Studies examining a range of norm-based intervention strategies such as benchmarking and commitment have shown to be effective in inspiring behavior change (Abrahamse et al. 2005; 2007; de Snoo et al. 2010). Research shows that benchmarking, or providing feedback about one’s behaviors and the actions others are taking, leads to normative pressure to keep up with others (Abrahamse et al. 2005; de Snoo et al. 2010; 2013; Lokhorst et al. 2010). For example, applications of benchmarking to farmer behavior have shown that farmers who received feedback comparing their conservation actions with others spent more time on conservation (e.g., de Snoo et al. 2010). In La Crescent and Reno Watersheds, providing feedback to landowners about their use of conservation practices compared to their neighbors may be useful in promoting social norms of conservation and increasing landowner engagement in conservation.

Commitment-making, or asking people to commit to taking action can activate personal norms in a decision-making situation (McKenzie-Mohr 2000; Lokhorst et al. 2010). Research in this area has found that benchmarking, along with commitment can influence farmers’ engagement in conservation (e.g., de Snoo et al. 2010; Lokhorst et al. 2010). Further, asking landowners to make small commitments such as contacting conservation professionals can lead to participation in more substantial activities such as conservation programs (Kennedy 2010). A critical practical question for conservation educators, extension agents, and field staff is "Who are the important people with influence on landowners' self-expectations and conservation decision-making?" This survey effort (Pradhananga et al. 2019) and other similar studies (e.g., Pradhananga et al. 2014; 2018) of Minnesota landowners reveal that family members, neighbors, and local conservation agencies are among the most influential groups when conservation decisions are made. Knowing those influential referent groups and engaging them in community-centered conservation program development and implementation is likely to make a difference.

Finally, this study shows that programs to build landowners' self-efficacy are needed to promote conservation behaviors. Bandura (2012) outlines four main sources of self-efficacy: 1) enactive experiences (e.g., mastery, resiliency), 2) vicarious experience (e.g., social models of success), 3) social persuasion (e.g., reinforcement of positive self-image and reduction of self-doubt), and 4) emotional and physical states. More recently, Perry and Davenport (2020) identified sources of farmers' self-efficacy to engage in conservation agriculture. The authors identified personal achievement in soil conservation and precision agriculture, observing others' success, and peer feedback as primary sources of self-efficacy. Feedback plays a critical role in building self-efficacy. Feedback that highlights social models of success can be a useful tool to enhance landowners' self-efficacy. For example, strategies such as sharing success stories of water protection can help establish conservation as a community norm and build landowners' self-efficacy. Programs and communication campaigns that provide social and ecological feedback about the outcomes of conservation practices (e.g., erosion control, water quality improvements) are strategies to build self-efficacy. Providing honest and localized social and biophysical feedback about conservation practice impacts, including benchmarking to demonstrate what others are doing and their successes and challenges creates transparency and enables community-driven dialogue.

Conclusion

Study findings show that landowners' conservation action is driven by their feelings of personal obligation, and beliefs about whether one is capable of taking actions to influence outcomes (i.e., self-efficacy). Landowners who feel a sense of personal obligation and believe that they can take actions that can make a difference are more likely to take conservation actions. Further, landowners who believe it is their personal responsibility to protect water and perceive social expectations are more likely to develop feelings of personal obligation. Importantly, this study highlights the significance of self-efficacy as an activator of personal norm, as well as a driver of conservation behavior.

Acknowledgements

The authors would like to thank Winona County, Minnesota, and the Minnesota Pollution Control Agency for funding this research.

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General questions about the conference can be directed to Karl Williard (ucowr@siu.edu), Executive Director of UCOWR.

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