Median values of total radium in the Marcellus Shale ranged from about 1,000 pCi/L to less than 6,000 pCi/L, which are values far exceeding the industrial discharge limit of 60 pCi/L (Rowan et al., 2011) (Figure 7-6). In the Marcellus Shale, TENORM levels in produced water from unconventional reservoirs exceeded levels from conventional reservoirs levels by factors of 4 to 26 (PA DEP, 2015b) (Appendix Table E-8). The individual median concentrations in produced water from unconventional reservoirs of 11,300 pCi/L gross alpha, 3,445 pCi/L gross beta, and total radium of 7,180 pCi/L (Appendix Table E-8). TENORM has been identified in hydraulic fracturing fluid, presumably due to the reuse of produced water at levels from 2 to 4.5 times lower than produced water from unconventional reservoirs (PA DEP, 2015b) (Appendix Table E-8).

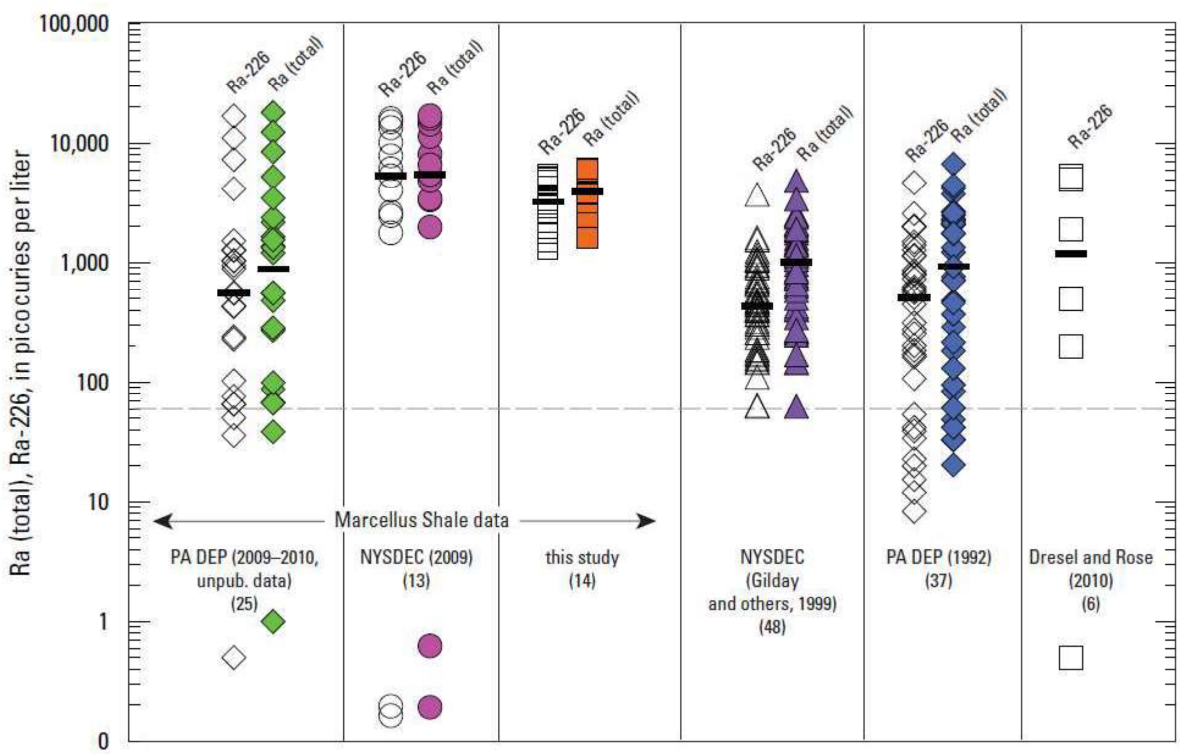

Figure 7-6. Data on radium 226 (open symbols) and total radium (filled symbols) for Marcellus Shale wells (leftmost three columns) and other formations (rightmost three columns).

Source: Rowan et al. (2011). The dashed line represents the industrial effluent discharge limit of 60 pCi/L set by the Nuclear Regulatory Commission. The black lines indicate the median concentrations, and the number of points in each dataset are shown in parentheses. Citations within the figure are provided in Rowan et al. (2011).

7.3.4.7 Organics

The organic content of produced water varies by well and lithology, but consists of naturally occurring and injected organic compounds (Lee and Neff, 2011). Of the organics detected by either routine or advanced analytical methods (Section 7.3.1), the majority are naturally occurring constituents of petroleum (Appendix Tables H-4 and H-5). These organics may be dissolved in water or, in the case of oil production, in the form of a separate or emulsified phase. Several classes of organic chemicals have been found in shale gas and CBM produced water, including aromatics, polyaromatic hydrocarbons, heterocyclic compounds, aromatic amines, phenols, phthalates, aliphatic alcohols, fatty acids, and nonaromatic compounds (list from Orem et al. (2014), see also: Hayes (2009), Benko and Drewes (2008), Orem et al. (2007), and Sirivedhin and Dallbauman (2004)). Compounds found in CBM waters included pyrene, phenanthrenone, alkyl phthalates, C_{12} through C_{18} fatty acids, and others. Similarly, compounds found in shale gas produced water included pyrene and perylene, ethylene glycol, diethylene glycol monododecyl ether, 2-(2-butoxyethoxy) ethanol, and others (Orem et al., 2014). Biomarkers—organic molecules characteristically produced by life forms, and unique to shale formations—have recently been suggested to fingerprint produced water (Hoelzer et al., 2016). More representative examples from five coal bed and two shale gas formations with reported concentrations are given in Appendix Tables E-9, E-11, and E-12, and the complete list of chemicals with CAS registry numbers identified by the EPA for this assessment appears in Appendix H. (See Appendix Table H-4 for chemicals with EPA-identified CAS numbers and Appendix Table H-5 for chemicals without.) Appendix Table E-13 lists concentrations of organic chemicals that were identified in three specific studies (Khan et al., 2016; Lester et al., 2015; Orem et al., 2007).

7.3.4.8 Hydraulic Fracturing Fluid Additives

Several chemicals used in hydraulic fracturing fluids have been identified in produced water. (Examples are shown in Table 7-6, Appendix Table E-10, and Appendix Tables H-4 and H-5.) Many of these chemicals were identified through advanced analytical procedures and equipment, and would not be expected to be found by routine analyses. Of note is that phthalates do not occur naturally. Their presence in produced water is due to either their use in hydraulic fracturing fluids; polyvinyl chloride (PVC) in well adhesives, valves, or fittings; or coatings on laboratory sample bottles (Orem et al., 2007).[1] Phthalates can also be used in drilling fluids, as breaker additives, or as plasticizers (Maguire-Boyle and Barron, 2014; Hayes and Severin, 2012a).[2] One of the produced water phthalates has been identified as a component of hydraulic fracturing fluid (di(2-ethylhexyl) phthalate) (Appendix Table H-2), while others have not, and those may originate from laboratory or field equipment.

Table 7-6. Examples of compounds identified in produced water that can be components of hydraulic fracturing fluid.

Appendix Tables H-4 and H-5 list chemicals identified in produced water and indicates those also identified as constituents of hydraulic fracturing fluid. Chemical or class designation in this table is taken directly from the text of the cited references except where noted, and may or may not reflect the chemical names from the Distributed Structure-Searchable Toxicity Database (DSSTox) show in Appendix Table H-4 or other chemicals listed in Appendix Table H-5.

Chemical or class	Use	Reference
2-Butanone	Solvent; microbial degradation product	Lester et al. (2015)

[1] Examples include di(2-ethylhexyl) phthalate, diisodecyl phthalate, and diisononyl phthalate (Orem et al., 2007).

[2] Specifically fatty acid phthalate esters (Maguire-Boyle and Barron, 2014).

Chemical or class	Use	Reference
2-Butoxyethanol	Acid dispersant, solvent, non-emulsifier	Thacker et al. (2015)
Acetone	Solvent; microbial degradation product	Lester et al. (2015)
Cocamidopropyl dimethylamine (C-7)	Foaming and lubrication enhancer	Lester et al. (2015)
Di(2-ethylhexyl) phthalate[a]	Derivative of polyvinyl chloride used in adhesives, valves, fittings or coatings of sample bottles	Orem et al. (2007)
Diethylene glycol monododecyl ether	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Dioctadecyl ester of phosphate phosphoric acid	Common lubricant	Maguire-Boyle and Barron (2014)
Ethylene glycol	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Fatty acid phthalate esters	(Related to) use in drilling fluids and breakers	Maguire-Boyle and Barron (2014)
Fluorocarbons	Tracers	Maguire-Boyle and Barron (2014)
Hexahydro-1,3,5-trimethyl-1,3,5-triazine-2-thione	Biocide	Orem et al. (2014)
Linear alkyl ethoxylates (C-4 to C-8, C-11 to C-14)	Enhancer of surfactant properties	Lester et al. (2015); Thurman et al. (2014)
Polyethylene glycol carboxylates (PEG-C-EO2 to PEG-C-EO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Polyethylene glycols (PEG-EO4 to PEG-EO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Polypropylene glycols (PPG-PO2 to PPG PO10)	Friction reducer, clay stabilizer, surfactants	Thurman et al. (2016)
Toluene	Solvent, scale inhibitor	Thacker et al. (2015)
Triethylene glycol monododecyl ether	Antifreeze, scale inhibitor, friction reducer	Orem et al. (2014)
Xylenes	Solvent, scale inhibitor	Thacker et al. (2015)

[a] Di(2-ethylhexyl) phthalate was named di-2-ethyl hexyl phthalate in Maguire-Boyle and Barron (2014).

7.3.4.9 Reactions within Formations

The introduction of hydraulic fracturing fluids into the target formation induces a number of changes to formation solids and fluids that influence the chemical evolution and composition of produced water. These changes can result from physical processes (e.g., rock fracturing and fluid mixing); geochemical processes (e.g., introducing oxygenated fluids of composition unlike that of the formation); and down hole conditions (elevated temperature, salinity, and pressure) that mobilize trace or major constituents into solution.

The creation of fractures exposes new formation surfaces to interactions involving hydraulic fracturing fluids and existing formation fluids. Formations in unconventional reservoirs targeted for development are composed of detrital, cement, and organic fractions. For example, elements potentially available for mobilization when exposed via fracturing include calcium, magnesium, manganese, and strontium in cement fractions, and silver, chromium, copper, molybdenum, niobium, vanadium, and zinc in organic fractions.

From organic compounds identified in five flowback samples and one produced water sample from the Fayetteville Shale, three possible types of reactions were identified by Hoelzer et al. (2016): hydrolysis of delayed acids, oxidant-caused halogenation reactions, and transformation of disclosed additives. First, delayed acids are used to “break” gel structures and would be intentionally introduced for their ability to cause in-formation reactions. Second, strong oxidants or other compounds introduced as breakers, along with elevated temperature and salinity, can trigger reactions between halogens (chloride, bromide, and iodide) and methane, acetone and pyrane resulting in halomethane compounds. A similar suggestion was made by Maguire-Boyle and Barron (2014). Low pH was found to promote oxidation of additives (Tasker et al., 2016). Third, known additives may react to form byproducts. Hoelzer et al. (2016) postulate examples from several types of compounds, two of these are the formation of benzyl alcohol from the hydraulic fracturing additive benzyl chloride, and abiotic and biotic reactions of phenols. In a study that used synthetic fracturing fluid, Tasker et al. (2016) reported that surfactants were recalcitrant to degradation under high pressure and temperature, which may explain the presence of the surfactant glycols in produced water as reported by Thurman et al. (2016) (Table 7-6), and the oxidation of other additives (gelling and some friction reducers (Table 5-1)) may explain their absence.

7.3.5 Spatial Trends in Produced Water Composition

As was reported for the volume of produced water (Section 7.2.2), the composition of produced water varies spatially on a regional to local scale according to the geographic and stratigraphic locations of each well within a hydraulically fractured production zone (Bibby et al., 2013; Lee and Neff, 2011). Spatial variability of produced water content occurs: (1) between plays of different rock sources (e.g., coal vs. sandstone); (2) between plays of the same rock type (e.g., Barnett Shale vs. Bakken Shale); and (3) within formations of the same source rock (e.g., northeastern vs. southwestern Marcellus Shale) (Barbot et al., 2013; Alley et al., 2011; Breit, 2002).

Geographic variability in produced water content has been established at a regional scale for conventional produced water. As an example, Benko and Drewes (2008) demonstrate TDS variability in conventional produced water among fourteen western geologic basins (e.g., Williston, San Juan, and Permian Basins). Median TDS in these basins range from as low as 4,900 mg/L in the Big Horn Basin to as high as 132,400 mg/L in the Williston Basin based on over 133,000 produced water samples from fourteen basins (Benko and Drewes, 2008).[1]

Average or median TDS of more than 100,000 mg/L has been reported for the Bakken (North Dakota, Montana) and Marcellus (Pennsylvania) formations; between 50,000 mg/L and 100,000 mg/L for the Barnett (Texas), and less than 50,000 mg/L for the Fayetteville (Arkansas) shale formations.[2] In tight formations, the average TDS was above 100,000 mg/L for the Devonian Sandstone (Pennsylvania) and Cotton Valley Group (Louisiana, Texas), between 50,000 mg/L and 100,000 mg/L for the Oswego (Oklahoma), and less than 50,000 mg/L for the Mesaverde Formation (Colorado, New Mexico, Utah, Wyoming). Maximum concentrations above 200,000 mg/L have been reported for the Marcellus, Bakken, Cotton Valley Group and Devonian Sandstone (Appendix Table E-2).

CBM produced waters had average TDS of less than 5,000 mg/L in the Powder River (Montana, Wyoming), Raton (Colorado, New Mexico), and San Juan (Arizona, Colorado, New Mexico, Utah) basins; while above 10,000 mg/L in the Black Warrior Basin (Alabama, Mississippi), which as noted above are due to the depositional history of these basins (Appendix Table E-3, Section 7.3.2).

Data further illustrating variability within both shale and tight gas reservoirs, as well as coalbed methane fields, at both the formation and local scales are presented and discussed in Appendix Section E.3.

7.4 Spill and Release Impacts on Drinking Water Resources

Surface spills of produced water from oil and gas production have occurred across the country and, in some cases, have caused impacts to drinking water resources. Released fluids can flow into nearby surface waters, if not contained on-site, or infiltrate into groundwater via soil. In this section, we first briefly describe the potential for spills from produced water handling equipment. Next, we address individually reported spill events. These have originated from pipeline leaks, well blowouts, well communication events, and leaking pits and impoundments. We then summarize several studies of aggregated spill data, which are based on state agency spill reports.

7.4.1 Produced Water Handling and Spill Potential

Throughout the production phase at oil and certain wet gas production facilities, produced water is stored in containers and pits that can contain free phase, dissolved phase, and emulsified crude oil. Since the crude oil is not efficiently separated out by the flow-through process vessels (such as

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[1] Data were drawn from the USGS National Produced Water Geochemical Database v2.0. Published updates made in October 2014 to the database (v2.1) are not reflected in this document.

[2] Because publications we are comparing may report either average or median values (but not uniformly both), we combine average and medians in this paragraph.

three-phase separators, heater treaters, or gun barrels), this crude oil can remain present in the produced water container or pit.

Produced water can be transferred to surface pits for long-term storage and evaporation. Surface pits are typically uncovered, earthen pits that may or may not be lined.[1] Unlined pits can lead to contamination of groundwater, especially shallow alluvial systems. Recovered fluids can overflow or leak from surface pits due to improper pit design and weather events.

Produced water that is to be treated or disposed of off-site is typically stored in storage tanks or pits until it can be loaded into transport trucks for removal (Gilmore et al., 2013). Tank storage systems are typically closed loop systems in which produced water is transported from the wellhead to aboveground storage tanks through interconnecting pipelines (GWPC and IOGCC, 2014). Failure of connections and lines during the transfer process or the failure of a storage tank can result in a surface release of fluids.

Depending on its characteristics, produced water can be recycled and reused on-site. It can be directly reused without treatment (after blending with freshwater), or it can be treated on-site prior to reuse (Boschee, 2014). As with other produced water management options, these systems also can spill during transfer of fluids.

7.4.2 Spills of Produced Water

7.4.2.1 Pipeline Leaks

Produced water is typically transported from the wellhead through a series of pipes or flowlines to on-site storage or treatment units (GWPC and IOGCC, 2014), or nearby injection wells. Faulty connections at either end of the transfer process or leaks or ruptures in the lines carrying the fluid can result in surface spills. A field report from PA DEP (2009b) described a leak from a 90-degree bend in an overland pipe carrying a mixture of produced water and freshwater between two pits. The impact included a “dull sheen” on the water and measured chloride concentration of 11,000 mg/L. The leak impacted a 0.4 mi (0.6 km) length of a stream, and fish and salamanders were killed. Beyond a confluence at 0.4 mi (0.6 km) with a creek, no additional dead fish were found. The release was estimated at 11,000 gal (42,000 L). In response to the incident, the pipeline was shut off, a dam was constructed for recovering the water, water was vacuumed from the stream, and the stream was flushed with fresh water (PA DEP, 2009b).

Another example of a pipeline release occurred in January 2015, when 70,000 bbls (2,940,000 gal or 11,130,000 L) of produced water containing petroleum hydrocarbons (North Dakota Department of Health, 2015) were released from a broken pipeline that crosses Blacktail Creek in Williams County, ND. The response included placing absorbent booms in the creek, excavating contaminated soil, removing oil-coated ice, and removing produced water from the creek. The electrical conductivity and chloride concentration in the water along the creek, the Little Muddy River, and Missouri River were found to be elevated above background levels, as were samples

[1] The use of the terms “impoundments” and “pits” varies and is described in Chapter 8. For the purposes of this section, the term “pits” will be generally used to cover all below-grade storage (but not above ground closed or open tanks).

taken from groundwater recovery trenches. Remediation work on this site continues as of the date of this writing (August, 2016).

7.4.2.2 Well Blowouts

Spills of produced water have occurred as a result of well blowouts. Fingerprinting of water from two monitoring wells in Killdeer, ND, was used to determine that brine contamination in the two wells resulted from a well blowout during a hydraulic fracturing operation. See the discussion in Section 6.2.2.1 for more information.

Another example of a well blowout associated with a hydraulic fracturing operation occurred in Clearfield County, PA. The well blew out, resulting in an uncontrolled flow of approximately 35,000 gal (132,000 L) of brine and fracturing fluid; some of the liquids reportedly reached a nearby stream (Barnes, 2010). The blowout occurred during drilling of plugs that were used to isolate fracture stages from each other. An independent investigation found that the primary cause of the incident was that the sole blowout preventer on the well had not been properly tested. In addition, the company did not have certified well control experts on hand or a written pressure control procedure (Vittitow, 2010).

In North Dakota, a blowout preventer failed, causing a release of between 50 and 70 bbls per day (2,100 gal/day or 7,900 L/day and 2,940 gal/day or 11,100 L/day) of produced water and oil (Reuters, 2014). Frozen droplets of oil and water sprayed on a nearby frozen creek. Liquid flowing from the well was collected and trucked offsite. A 3-ft (0.9-m) berm was placed around the well for containment. Multiple well communication events have also led to produced water spills ranging from around 700 to 35,000 gal (2,600 L to 130,000 L) (Vaidyanathan, 2013a). Well communication is described in Section 6.3.2.3.

The Chesapeake Energy ATGAS 2H well, located in Leroy Township, Bradford County, PA, experienced a wellhead flange failure on April 19, 2011, during hydraulic fracturing operations. Approximately 10,000 gal (38,000 L) of produced water spilled into an unnamed tributary of Towanda Creek, a state-designated trout stock fishery and a tributary of the Susquehanna River (USGS, 2013b; SAIC and GES, 2011). Chesapeake conducted post-spill surface water and groundwater monitoring (SAIC and GES, 2011).

Chesapeake concluded that there were short-term impacts to surface waters of a farm pond within the vicinity of the well pad, the unnamed tributary, and Towanda Creek following the event (SAIC and GES, 2011). The lower 500 ft (200 m) of the unnamed tributary exhibited elevated chloride, TDS, and specific conductance, which returned to background levels in less than a week. Towanda Creek experienced these same elevations in concentration, but only at its confluence with the unnamed tributary; elevated chloride, TDS, and specific conductance returned to background levels the day after the blowout (SAIC and GES, 2011).

7.4.2.3 Leaks from Pits and Impoundments

Leaks of produced water from on-site pits have caused releases as large as 57,000 gal (220,000 L) and have caused surface water and groundwater impacts (Vaidyanathan, 2013b; Levis, 2011; 2010c; PADEP, 2010). VOCs have been measured in groundwater near the Duncan Oil Field in New Mexico downgradient of an unlined pit storing produced water. More example releases from pits are described in Section 8.4.5.

Two of EPA's retrospective case studies evaluated potential impacts from produced water pits. The EPA retrospective case studies were designed to determine whether multiple lines of evidence might be found that could specifically link constituent(s) found in drinking water to hydraulic fracturing activities using the tiered assessment framework presented in Appendix Section E.6. A multiple-lines-of-evidence approach was used to evaluate potential cause-and-effect relationships between hydraulic fracturing activities and contaminant presence in groundwater. Such an approach is needed, because the presence of a constituent in groundwater that is also found in hydraulic fracturing fluids or produced water does not necessarily implicate hydraulic fracturing activities as the cause. This is because some constituents of hydraulic fracturing fluids or produced water are ubiquitous in society (i.e., BTEX), and some constituents of produced water can be present in groundwater as background constituents (i.e., methane, iron, and manganese).

Elements of the assessment framework include gathering background information, including pre-drilling sample results; developing a conceptual model of the site; and assessing multiple analytes to develop lines of evidence. Development of these requires adherence to sampling and quality assurance protocols to generate defensible data. Among many other quality assurance requirements, proper well purging and analyses of field and laboratory blanks are needed (Appendix Table E-17 and Figure E-15).

In the EPA's *Retrospective Case Study in Southwestern Pennsylvania: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2015j), elevated chloride concentrations and their timing relative to historical data suggested a recent groundwater impact on a private water well occurred near a pit. The water quality trends suggested that the chloride anomaly was related to the pit, but site-specific data were not available to provide a definitive assessment of the cause(s) and the longevity of the impact. Evaluation of other water quality parameters did not provide clear evidence of produced water impacts.

In the EPA's *Retrospective Case Study in Wise County, Texas: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2015l), impacts to two water wells were attributed to brine, but the data collected for the study were not sufficient to distinguish among multiple possible brine sources, including reserve pits, migration from underlying formations along wellbores, migration from underlying formation along natural fractures and a nearby brine injection well.

To aid in assessing impacts, a number of geochemical indicators and isotopic tracers for identifying oil and gas produced water have been identified. These include (Lauer et al., 2016; Warner et al., 2014a, b):

- Common ion ratios, including bromide/chloride and lithium/chloride;
- Isotope ratios, especially Strontium isotope ratios ($^{87}Sr/^{86}Sr$); and

- Enrichment of certain isotopes: $\delta^{18}O$, $\delta^{2}H$, $\delta^{7}Li$, $\delta^{13}C$-DIC, $\delta^{11}B$.[1]

For the case study, twelve geochemical indicators, including the bromine/chlorine (Br/Cl) and strontium isotope ratios, were considered for the well-water samples.[2] The results were used to assess whether the likelihood that the observed values originated with produced water (the aforementioned sources of brine), sea water, road salt, landfill leachate, sewage/septic tank leachate, and animal waste. In each sample evaluated, it was found that the water could have originated with one or more of the six sources. Thus these lines of evidence did not allow identification of neither a specific source nor a hydraulic fracturing source (Appendix Table E-18). A third well experienced similar impacts, and a landfill leachate source could not be ruled out in that case.

The case studies illustrate how multiple lines of evidence were needed to assess suspected impacts and that no single constituent or parameter could be used alone to assess potential impacts.

7.4.2.4 Other Sources

In the EPA's *Retrospective Case Study in Northeastern Pennsylvania: Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2014f), a pond was found to be impacted due to elevated chloride and TDS, along with strontium ratios ($^{87}Sr/^{86}Sr$) characteristic of Marcellus Shale produced water. Here, the suspected source of the impact was a well pad which had a hydrochloric acid spill, a possible produced water spill and been used for temporary storage of drill cuttings. The same mulidence fracturing impacts from constituents characteristic of produced water (TDS, chloride, sodium, barium, strontium and radium) found in three domestic wells located in an area with naturally occurring saline groundwater. Conversely, at a spring with organic chemical contamination but no associated chloride or TDS impacts, hydraulic fracturing activities were also ruled out.

An estimated 6,300 to 57,373 gal (24,000 to 217,280 L) of Marcellus Shale produced water was discharged through an open valve that drained a tank at XTO Energy Inc.'s Marquardt pad and flowed into a tributary of the Susquehanna River in November 2010 (U.S. EPA, 2016e; PA DEP, 2011c). Overland and subsurface flow of released fluids impacted surface water, a subsurface spring, and soil. Five hundred tons of contaminated soil were excavated, and an estimated 8,000 gal (30,000 L) of produced water was recovered (Science Applications International Corporation, 2010). Elevated levels of TDS, chloride, bromide, barium and strontium that indicated a release of produced water were present in the surface stream and a spring for roughly 65 days (U.S. EPA, 2016e). At that time the chloride concentration in the spring dropped below the state surface water standard of 250 mg/L. The impact extended a distance of approximately 1,400 ft (440 m) to the spring from the release point. Samples were taken in the tributary roughly 500 ft downstream from the spring, where chloride concentrations remained below the 250 mg/L standard throughout the sampling period, but were above the upstream concentrations (PA DEP, 2011c; Schmidley and Smith, 2011). Similarly, the total barium, total and dissolved iron, manganese and alkalinity concentrations remained below the Pennsylvania surface water quality standards at the downstream monitoring location throughout the monitoring period (Schmidley and Smith, 2011).

[1] DIC is dissolved inorganic carbon.

[2] The full list was: Br vs. B, Cl vs. Mg, Cl vs. Br, Cl vs. HCO_3,Cl vs. Ca, Cl vs. K, Cl vs. Na, Cl vs. SO_4, Cl/Br, Cl/I, K/Rb, $^{87}Sr/^{86}Sr$.

In Pennsylvania, discharges of brine were made into a storm drain that itself discharges to a tributary of the Mahoning River in Ohio. Analyses of the brine and drill cuttings that were discharged indicated the presence of contaminants, including benzene and toluene (U.S. Department of Justice, 2014). In California, an oil production company periodically discharged hydraulic fracturing wastewaters to an unlined sump for 12 days. It was concluded by the prosecution that the discharge posed a threat to groundwater quality (Bacher, 2013). These unauthorized discharges represent both documented and potential impacts on drinking water resources. However, data do not exist to evaluate whether such episodes are uncommon or whether they happen on a more frequent basis and remain largely undetected. Other cases of unpermitted discharges have been reported by various sources (Caniglia, 2014; Paterra, 2011).[1]

7.4.2.5 Data Compilation Studies

Three datasets were examined for produced water spill data. These included two published studies: a review of spills in Oklahoma that occurred prior to the onset of widespread high-volume hydraulic fracturing (Fisher and Sublette, 2005), and an EPA study of spills occurring between February 2006 and April 2012 on the well pads of hydraulically fractured wells (U.S. EPA, 2015m). The EPA spills study, *Review of state and industry spill data: characterization of hydraulic fracturing-related spills*, is described in Text Box 5-10. Because of data availability, EPA's study was dominated by data from Pennsylvania (21% of releases) and Colorado (48% of releases). Several difficulties are encountered in compiling and evaluating data on produced water spills and releases. Because states have differing minimum reporting levels, more spills are potentially reported in states with lower reporting limits.[2]

To include data from another state and to give results current to 2015, data from North Dakota were reviewed for this assessment.[3] Details on the procedures and results for non-produced water spills are given in Appendix Section E.5. The North Dakota Department of Health (NDDOH) collects data on environmental incidents and separately compiles oil field incidents; information is made available to the public at http://www.ndhealth.gov/EHS/Spills/. Of these incidents, most describe a release of oil, salt water, or other liquid. Of the remainder, a few describe releases of gas only.

For the period from November 2012 to November 2013, NDDOH reported 552 releases of produced water that were retained within the boundaries of the production or exploration facility and 104 that were not (North Dakota Department of Health, 2011). Thus, 16% of the releases were not contained within facility boundaries and had greater potential for impacting drinking water resources.

[1] Section 8.4 discusses permitted discharges of wastewater.

[2] For example, two agencies in the state of California manage different databases that both store information on spills associated with oil and gas production (CCST, 2015a). CCST (2015a) reported that the databases contain inconsistencies as to the number of spills and the details regarding those spills (e.g., quantity, chemical composition of the wastewater) resulting in uncertainty on the impacts spills have on the environment.

[3] Wirfs-Brock (2015) presented an analysis of North Dakota spill data through 2013.

7.4.2.6 Frequency of Spills and Releases

The EPA analyzed these data and found that, in recent years (2010-2015), there were between five and seven produced water spills per hundred active production wells (Figure 7-7). Spills declined between 2014 and 2015 (from 846 to 609), although the number of production wells increased. A study of 17 states indicated that there was an overall reduction of 8% in spills from 2014 to 2015, and an increase of 9% in Texas (King and Soraghan, 2016). More details on the data analysis are given in Appendix Section E.5, which includes results on North Dakota oil and spills of other types, including hydraulic fracturing fluids (as noted in Chapter 5).

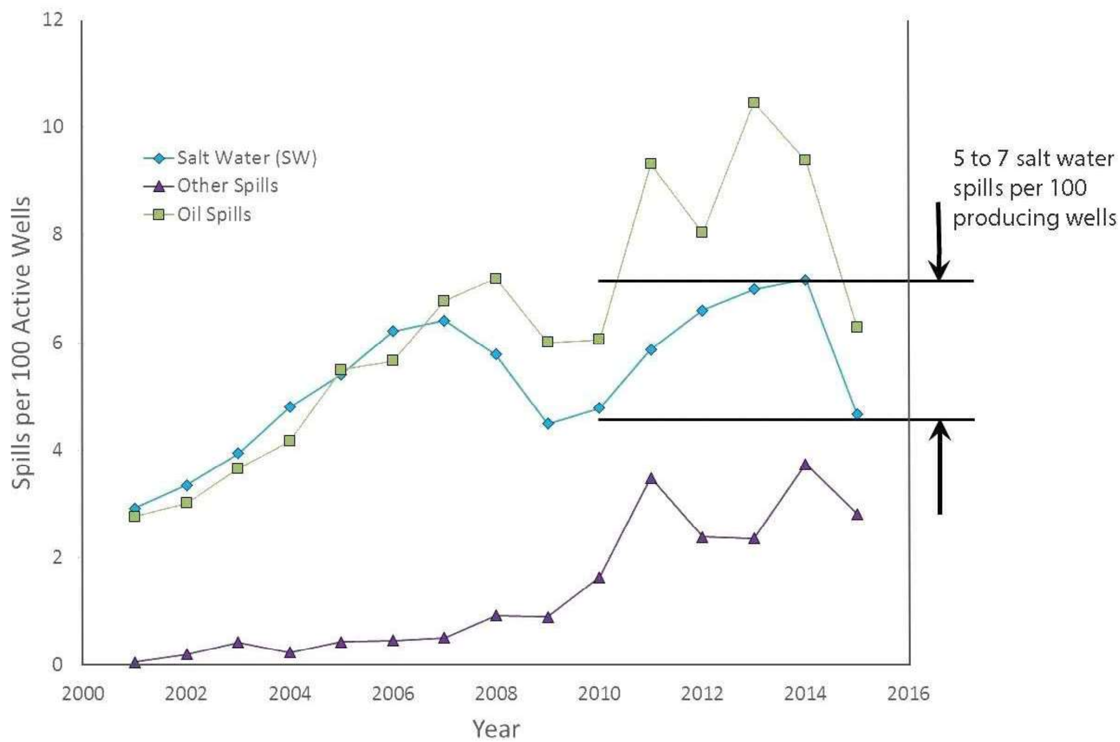

Figure 7-7. Produced water spill rates (spills per active wells) for North Dakota from 2001 to 2015 (Appendix Section E.5).

7.4.2.7 Produced Water Releases—Causes and Sources

The causes and sources identified for releases vary among the three datasets reviewed. North Dakota releases were dominated by leaks from various pieces of equipment, followed by "others," and various overflows (Figure 7-8). While the release rate declined from 2014 to 2015, the causes remained ranked relatively in the same order; notably fewer releases were attributed to "other" and more to equipment failure in 2015. The EPA's spills study found on- or near-well pad releases to be dominated by human error, unknown, and equipment failure (U.S. EPA, 2015m). The earlier

Oklahoma study was dominated by overflows, unpermitted discharges, and storms (Figure 7-9).[1] Storms can cause releases, as was noted after a major flood in northeastern Colorado that caused damage to produced water storage tanks releasing an estimated 43,000 gal (160,000 L) of produced water (COGCC, 2013).

The sources of releases are documented for the Oklahoma and EPA studies (Figure 7-10). The EPA cites storage, unknown, and hoses or lines as the major sources for its 225 well-pad releases. The earlier Oklahoma study cites unclassified, lines, and tanks as major sources of its 8,874 releases.

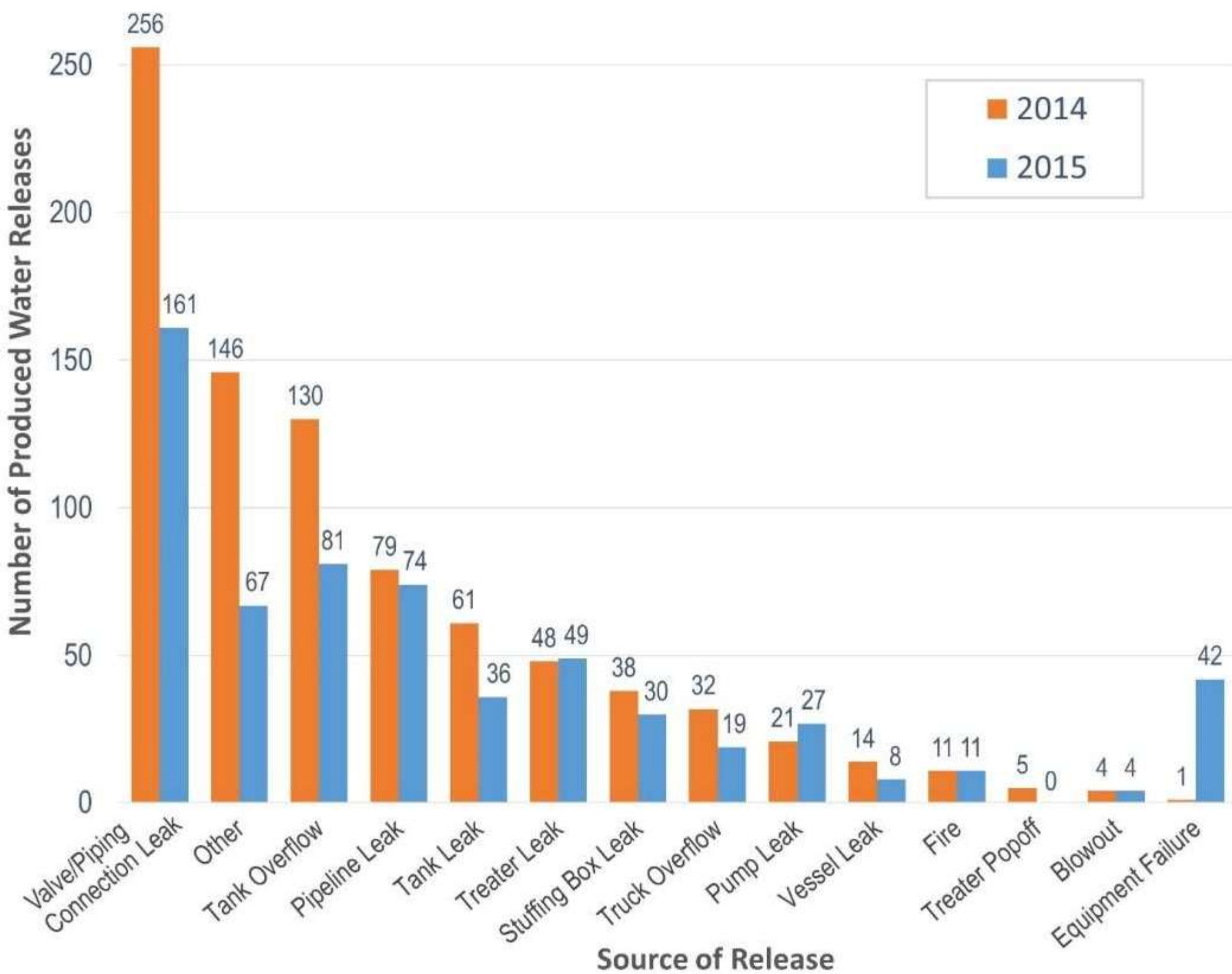

Figure 7-8. Number of produced water releases in North Dakota by cause for 2014 and 2015 (Appendix Section E.5).

[1] Some of the causes in the three studies may be more similar than they appear, because the categorization used in the different studies overlap. For example, the EPA categorized overflows as "human error;" blowouts, vandalism and weather as "other;" and corrosion as "equipment failure," while other studies listed these separately.

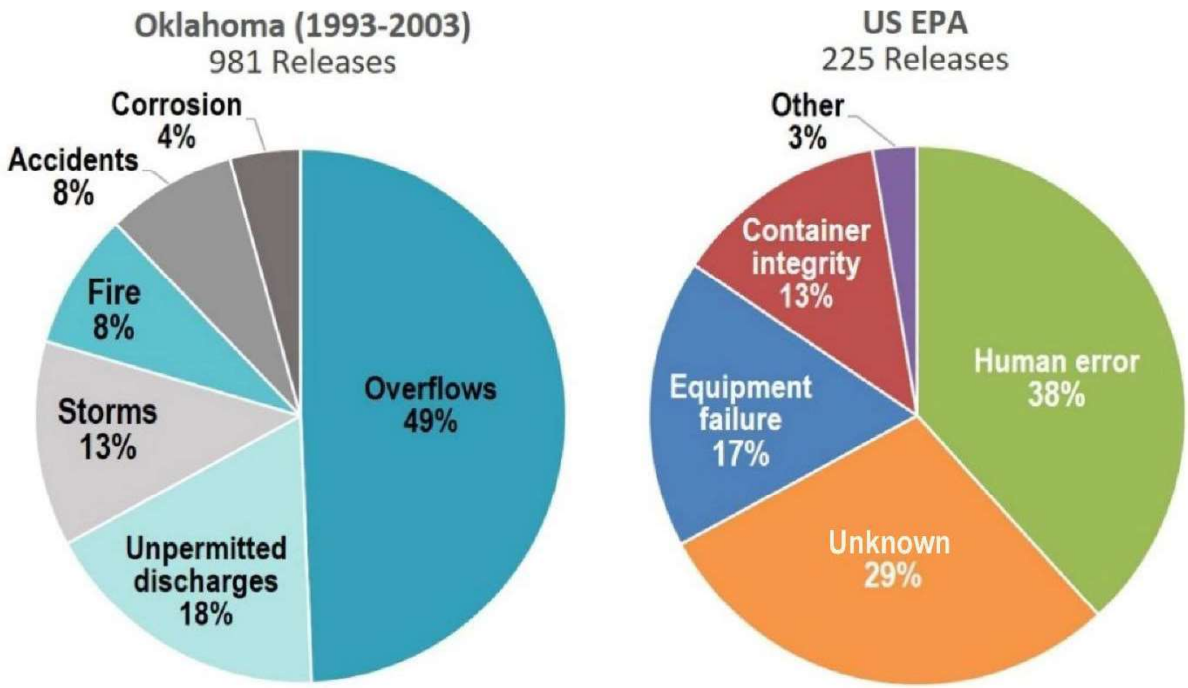

Figure 7-9. Distribution of spill causes in Oklahoma, pre-high volume hydraulic fracturing years of 1993-2003 (left) and in the EPA study of spills on production pads (right).
Data sources: left, Fisher and Sublette (2005); right, U.S. EPA (2015m).

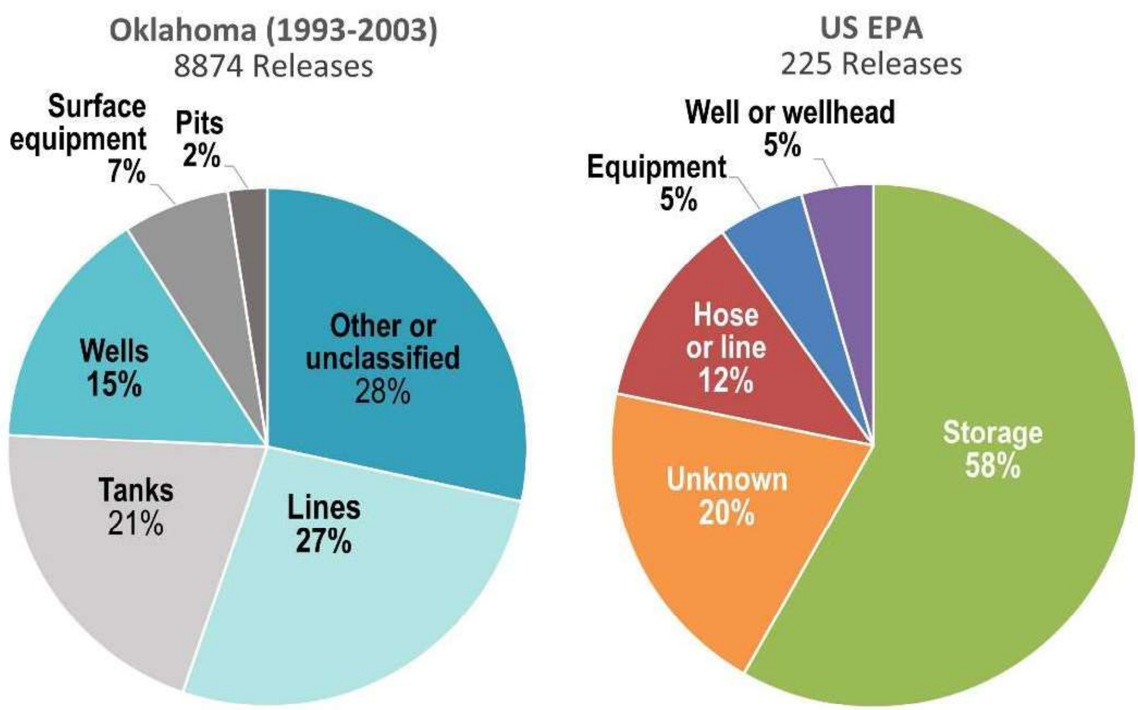

Figure 7-10. Distribution of spill sources in Oklahoma, pre-high volume hydraulic fracturing years of 1993-2003 (left) and in the EPA study of spills on production pads (right).
Data sources: left, Fisher and Sublette (2005); right, U.S. EPA (2015m).

7.4.2.8 The Volumes of Spilled Produced Water

The 2015 North Dakota spills were ranked from by the median volume, which is the level at which 50% of the spills are below this volume and 50% above (Figure 7-11).[1] Of the North Dakota spills in 2015, the highest median spill volume was caused by a blowout (2,400 gal, 91,000 L, left-most red box). The smallest median volume spill is approximately 10 times lower in volume (84 gal, 320 L). Spills larger than the median are of interest, because of their potential for impacting drinking water resources. The largest volume spill occurred from a pipeline break (2,900,000 gal, 11,000,000 L). The EPA spills study found the highest median volume spill was from equipment failure (1,700 gal, 6400 L), while the highest volume spill was due to container integrity (1,300,000 gal, 4,900,000 L) (Figure 7-12).

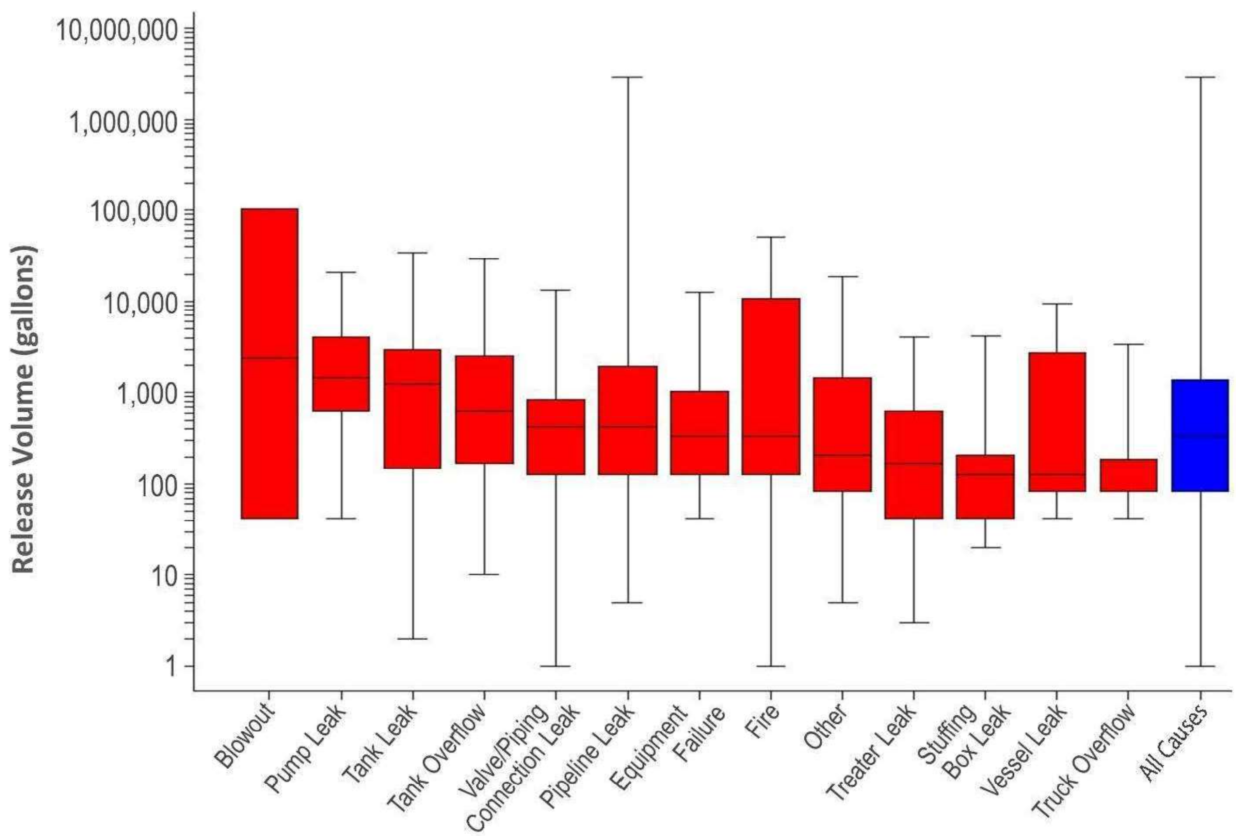

Figure 7-11. Volumes of 2015 North Dakota salt water releases by cause (leftmost 13 boxes in red), and all causes (last box in blue).

[1] These figures are called "box" plots or "box and whisker" plots. The rectangle in the middle represents the range of data from the 25th to 75th percentile. The line across the box represents the 50th percentile, also known as the median. Fifty percent of the data are below the median. The lines extending above and below the boxes represent the range of data from minimum to maximum. These concepts are illustrated in Appendix Figure E-6.

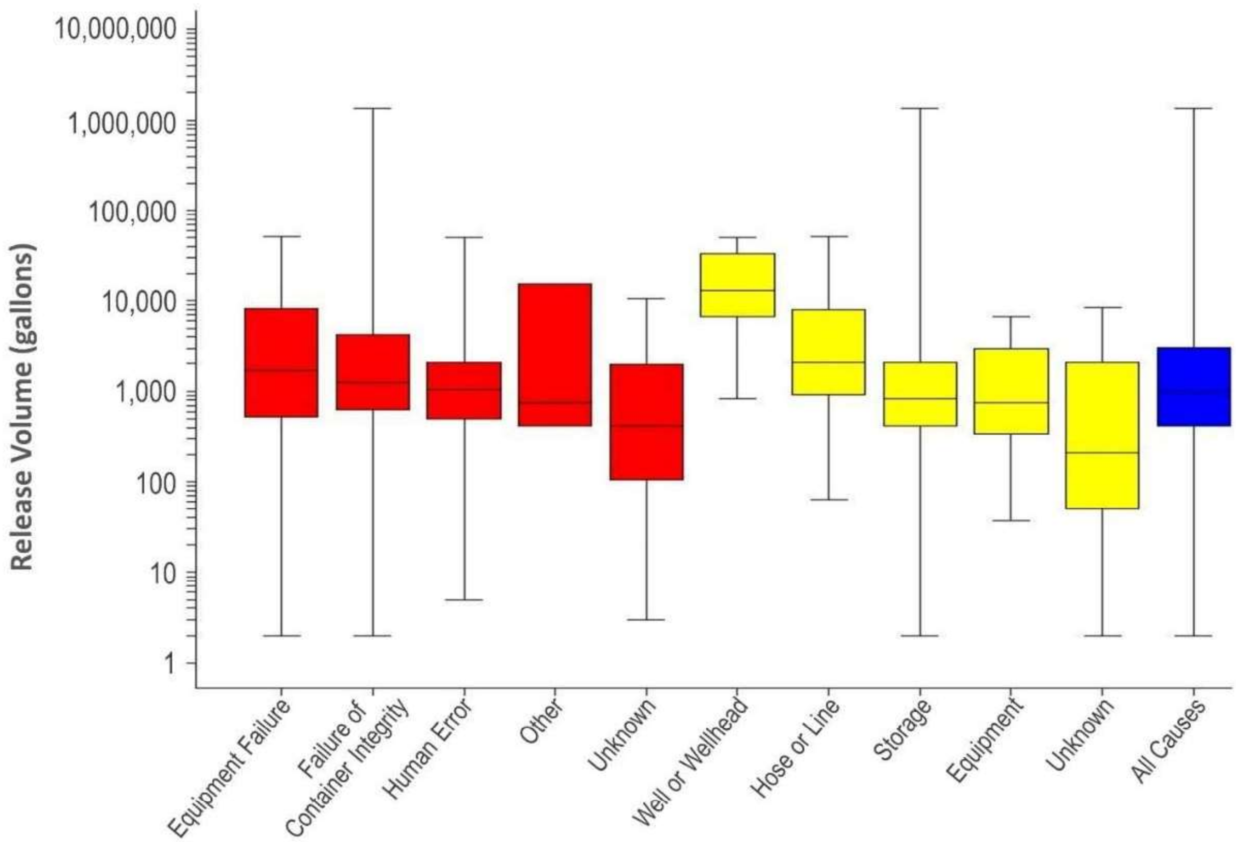

Figure 7-12. Volumes of produced water spills reported by the EPA for 2006 to 2012 by cause (the five left most boxes in red), source (the second five boxes in yellow), and all spills (blue). Calculated from Appendix B of U.S. EPA (2015m).

From the analyses, half of the spills are less than 1,000 gal (3,800 L) (EPA) and 340 gal (1,300 L) (North Dakota) (Figure 7-12, Figure 7-13, and medians in Table 7-7). The medians for the Oklahoma study were higher (overall 1,700 gal or 6,400 L; see Table 7-7 for yearly values) (Fisher and Sublette, 2005). These occurred in a different state and over an earlier time period, so a direct connection with the recent North Dakota and EPA results has not been made.

The skewed nature of the distributions are noted by the mean values being considerably higher than these medians (see Figure 7-13). In each case, this is caused by a small number of large spills. For 2015 in North Dakota, for example, there were 12 releases of 21,000 gal (79,000 L) or more; 5 of 42,000 gal (160,000 L) or more; and one of greater than 420,000 gal (1,600,000 L) (Appendix Table E-15). The largest spills from these data sets ranged from 1,000,000 gal (3,800,000 L) to 2,900,000 gal (11,000,000 L).

The EPA results give insight into recovery and reuse. Of the volume of spilled produced water, 16% was recovered for on-site use or disposal, 76% was reported as unrecovered, and the rest was unknown. The fewest spills occurred from wells and wellheads, but these spills had the greatest median volumes. Failure of container integrity was responsible for 74% of the volume spilled (U.S. EPA, 2015m).

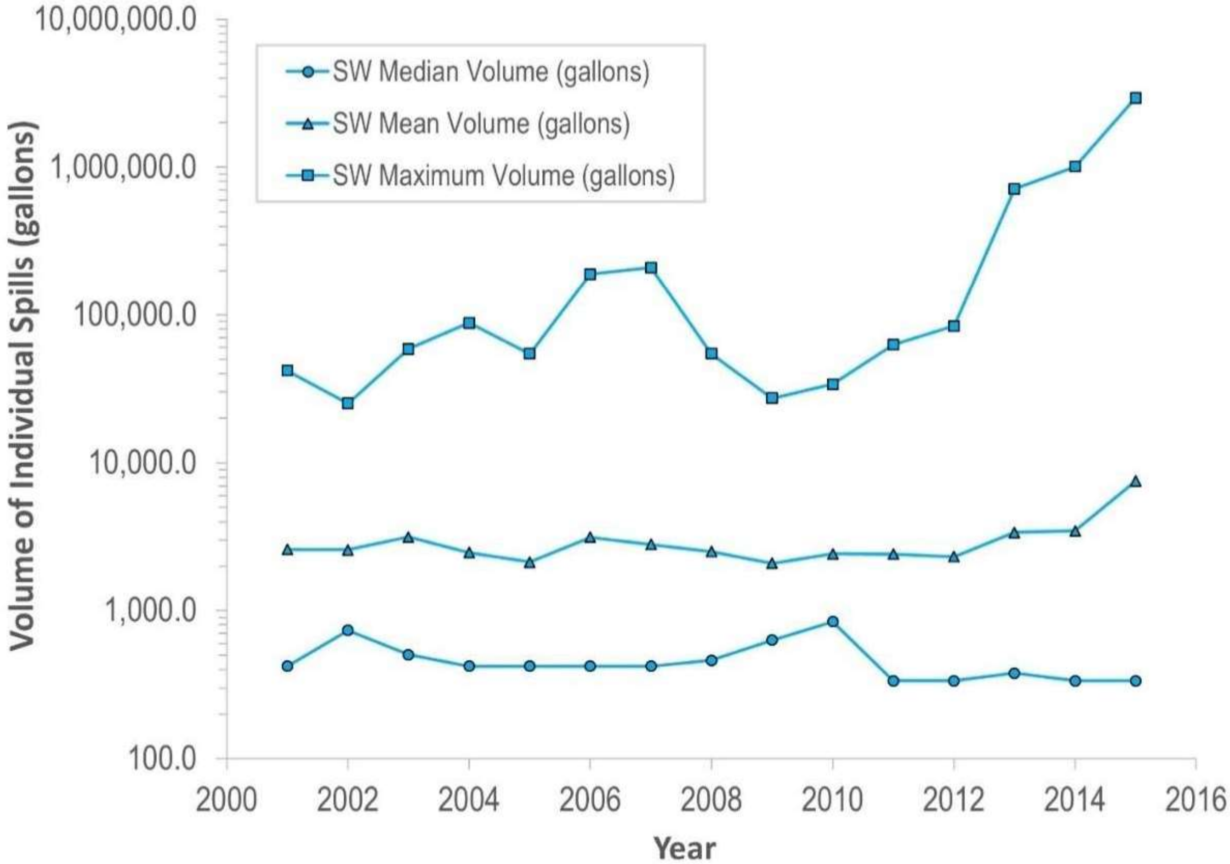

Figure 7-13. Median, mean, and maximum produced water spill volumes for North Dakota from 2001 to 2015.

Table 7-7. Summary of produced water release volumes.

Sources: U.S. EPA (2015m), Fisher and Sublette (2005), and Appendix Section E.5.

		Number		Minimum	25th percentile	Median	Mean	75th percentile	Maximum
Study	**Year(s)**	**Total**	**Quantified**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**
Oklahoma	1993-2002	7,916	2,365	0.0	630	1,700	7,000	4,200	3,400,000
	1993	373	161	0.4	420	1,500	3,900	4,200	46,000
	1994	844	333	0.4	420	1,600	5,400	4,200	84,000
	1995	913	333	0.0	420	1,500	3,700	4,200	63,000
	1996	880	333	4.2	630	2,100	6,500	4,200	420,000
	1997	806	270	0.4	630	1,900	6,000	4,200	120,000
	1998	825	236	2.1	798	4,900	2,100	4,200	105,000
	1999	886	218	10.5	840	2,100	6,600	4,200	120,000
	2000	853	155	4.2	840	2,100	5,600	5,040	210,000
	2001	826	144	21.0	840	2,100	31,000	6,510	3,400,000
	2002	710	182	0.8	630	1,700	5,500	3,276	130,000
U.S. EPA	2006-2012		225	2.1	420	1,008	10,920	2,982	1,344,000
North Dakota	2001		97	21.0	168	420	2,646	2,520	42,000
	2002		110	4.2	210	756	2,604	2,100	25,200
	2003		128	2.1	126	504	3,150	2,562	58,800
	2004		159	10.5	126	420	2,478	2,100	88,200
	2005		184	5.0	126	420	2,142	1,680	54,600
	2006		226	5.0	126	420	3,150	1,680	189,000
	2007		248	0.4	210	420	2,814	2,100	210,000

		Number		Minimum	25th percentile	Median	Mean	75th percentile	Maximum
Study	**Year(s)**	**Total**	**Quantified**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**	**(gal)**
North Dakota, cont.	2008		248	8.4	84	504	2,520	2,058	54,600
	2009		208	2.1	126	630	2,100	2,100	27,300
	2010		255	0.1	126	840	2,478	2,310	34,020
	2011		381	2.1	126	336	2,436	1,680	58,800
	2012		543	7.1	84	336	2,310	1,260	84,000
	2013		700	2.1	126	378	3,402	1,428	714,000
	2014		846	0.8	84	336	3,528	1,470	1,008,000
	2015		609	0.8	84	336	7,560	1,386	2,940,000

7.4.2.9 Environmental Receptors and Transport

Data from the EPA (U.S. EPA, 2015m) were used to show that some spills were known to impact environmental receptors: soil (141 spills, 340,000 gal, or 1.3 million L); surface water (17 spills, 170,000 gal, or 640,000 L); surface water and soil (13 spills); and groundwater (1 spill, 130 gal, or 490 L).[1] Although 1 spill was identified as reaching groundwater, the possible groundwater impact of 107 of the spills was unknown.

In summary, 18 produced water spills reached surface water or groundwater, accounting for 8% of the 225 cases and accounting for approximately 170,000 gal (640,000 L) of produced water. Spills with known volumes that reached a surface water body ranged from less than 170 gal (640 L) to almost 74,000 gal (280,000 L), with median of 5,900 gal (22,000 L). In 30 cases, it is unknown whether a spill of produced water reached any environmental receptor.

An assessment conducted by the California Council on Science and Technology (CCST, 2015a) states that between January 2009 and December 2014, 575 produced water spills were reported to the California Office of Emergency Services of which nearly 18 percent impacted waterways (CCST, 2015a). These spills occurred in areas where production from both unconventional and conventional reservoirs occurs. Additional studies of spill impacts are presented in Appendix Section E.5.3.

Studies of Environmental Transport of Released Produced Water

The processes that affected the fate and transport of spilled produced water (Figure 7-14) are the same as those processes that impact the fate and transport of spilled chemicals (Section 5.8). Produced water spills differ from the chemical spills as they are always primarily spills of water containing multiple chemicals. Additionally, produced water of high salinity is denser than water and may alter transport and transformation properties of the chemicals and soils.[2] If a spill occurs prior to treatment in an oil and water separator, the produced water can be spilled along with oil. In the environment, oil is transported as a separate phase liquid as it is immiscible with water. The oil phase may become trapped (similarly to how oil is trapped in oil reservoirs) and serve as a slowly dissolving source of hydrocarbons to the environment.

For example, Whittemore (2007) described a site with relatively little infiltration due to moderate to low permeability of silty clay soil and low permeability of underlying shale units. Thus, most, but not all, of the historically surface-disposed produced water at the site flowed into surface drainages. Observed historic levels of chloride in receiving waters resulted from the relative balance of produced water releases and precipitation runoff, with higher concentrations corresponding to low stream flows. Persistent surface water chloride contamination was attributed to slow flushing and discharge of contaminated groundwater.

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[1] Quoted volumes.

[2] Appendix Section E.7 describes the estimation of chemical properties for organic chemical constituents of produced water for baseline conditions of low TDS. Elevated salinity, as is common for produced water, would alter these values.

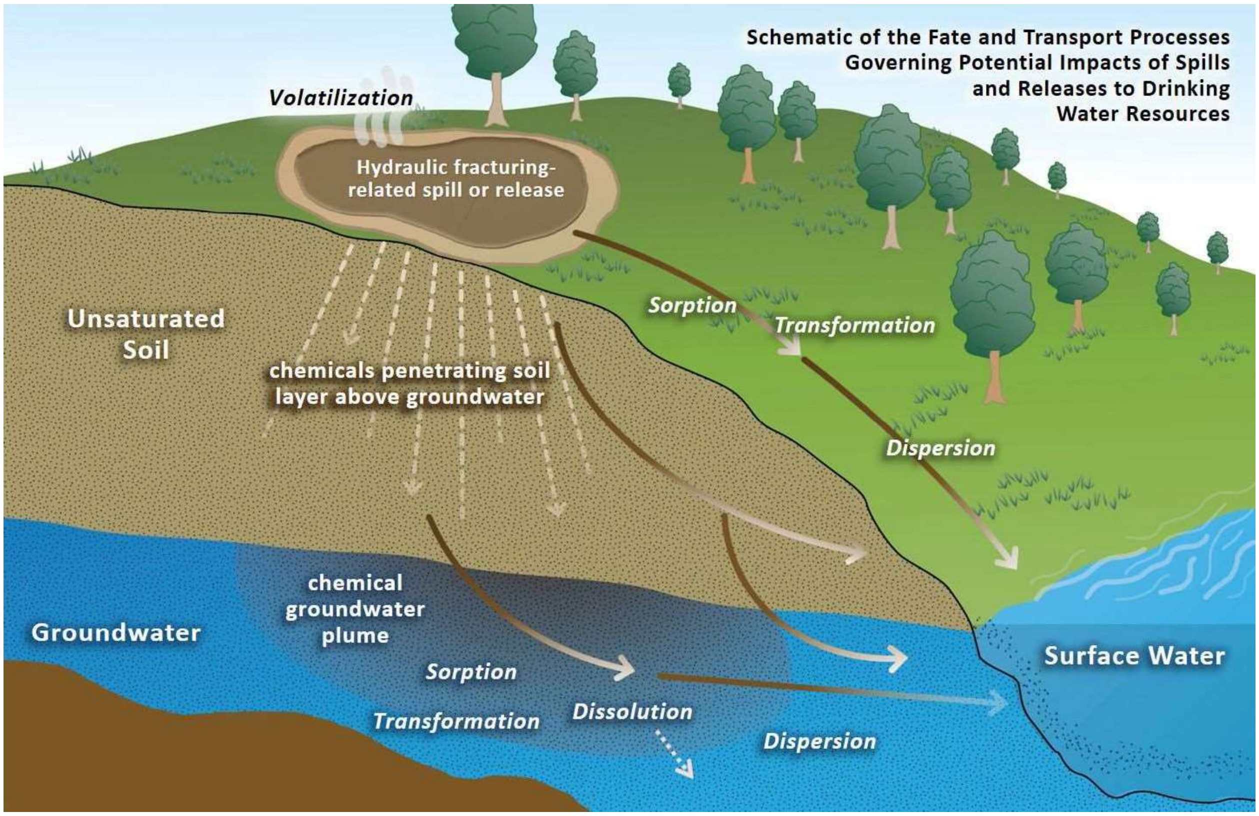

Figure 7-14. Schematic view of transport processes occurring during releases of produced water.

Because it is denser than freshwater, saline produced water can migrate downward through aquifers. Whittemore (2007) reported finding oilfield brine with a chloride concentration of 32,900 mg/L at the base of the High Plains aquifer. Where aquifers discharge to streams, saline stream water has been reported, although at reduced concentrations (Whittemore, 2007), likely due to diffusion within the aquifer and mixing with stream water. The stream flow rate, in part, determines mixing of substances in surface waters. High flows are related to lower chemical concentrations, and vice versa, as demonstrated for bromide in the Allegheny River (States et al., 2013).

7.5 Roadway Transport of Produced Water

Produced water is transported to treatment and disposal sites via pipeline, roadways, or railroad tankers. Accidents during transportation of hydraulic fracturing produced water are a possible mechanism leading to potential impacts to drinking water as truck-related releases have been reported. Nationwide data are not available, however, on the number of such accidents that result in impacts.

Crash rate estimates for Texas showed that commercial motor vehicle (CMV) crashes were correlated with oil and gas development activities over a recent period of increased oil and gas development (Quiroga and Tsapakis, 2015). As an example of the results, the number of new wells in the Permian Basin increased (by 61%) and so did rural CMV crashes (by 52%). For the Barnett Shale region, the number of new wells decreased (by 49%), and so did rural CMV crashes (by 34%). The correlations were strongest for the rural areas with oil and gas development (Permian and Eagle Ford).

Based on scenarios presented in Appendix Section E.8, the EPA estimated for this assessment the number of releases from truck crashes as having a chance of occurrence ranging between 1:110 and 1:13,000 over the lifetime of a producing well. The wide range of these estimates reflects both variable (distance and volume transported) and uncertain (crash rate) quantities. At 5,300 gal (20 m^3) per truckload, the volume from an individual spill would be low relative to the typical volume of water produced from a well. Several limitations are inherent in this analysis, including differing rural road and highway accident rates, differing transport distances, and differing amounts of produced water transported. Further, the estimates present an upper bound on impacts, because not all releases would reach or impact drinking water resources.

As for other types of impacts to drinking water resources, local effects can be significant despite the generally small numbers. For example, a brine-truck spill in Ohio resulted in concern for impacts to a drinking-water-source reservoir (Tucker, 2016).

7.6 Synthesis

Produced water is a by-product of oil and gas production and is that water that comes out of the well after hydraulic fracturing is completed and injection pressure is reduced. Produced water may contain hydraulic fracturing fluid, water from the surrounding formation, and naturally present hydrocarbons. Initially the chemistry of produced water reflects that of the hydraulic fracturing fluid. With time, the chemistry of the produced water becomes more similar to the water in the formation. Produced water is directly re-injected or stored at the surface for eventual reuse or disposal. Impacts to drinking water resources from produced water have been shown where spilled produced water entered surface water bodies or aquifers.

7.6.1 Summary of Findings

The volume and composition of produced water vary geographically, both within and among different production zones and with time and other site-specific factors. In most cases, there are high initial flow rates of produced water that last for a few weeks, followed by lower flow rates throughout the duration of gas production. The amount of fracturing fluid returned to the surface varies, and typically is less than 30%. In some formations (e.g., the Barnett Shale), the ultimate volume of produced water can exceed the volume of hydraulic fracturing fluid because of an inflow of water.

Knowledge of the composition of produced water comes from analysis of samples. Analysis of an individual sample is made much easier if the hydraulic fracturing and any equipment maintenance chemicals have been disclosed. Much of the chemical loading of produced water comes from naturally occurring material, both organic and inorganic, in the formation along with transformation products. As such, knowledge of produced water composition is uniquely dependent on sampling and analysis, which requires appropriate analytical methods. These are methods that can deal especially with high levels of TDS. Recently developed laboratory methods have greatly expanded the knowledge of organic chemicals in shale-gas and CBM produced waters, but these methods rely on advanced equipment and techniques. Routine methods of laboratory analysis do not detect many of the organic constituents of produced water.

The composition of produced water changes with time as the hydraulic fracturing fluid contacts the formation and mixes with the formation water. Typically it becomes more saline and more radioactive, if those constituents are present in the formation, while containing less DOC. The changing composition of produced water suggests that the potential concern for produced water spills also changes: initially the produced water may contain more hydraulic fracturing chemicals, later the concern may shift to the impact of high salinity water. Although varying within and between formations, shale and tight gas produced water typically contains high levels of TDS (salinity) and associated ionic constituents (bromide, calcium, chloride, iron, potassium, manganese, and sodium). Produced water can also contain toxic materials, including barium, cadmium, chromium, lead, mercury, nitrate, selenium, and BTEX. CBM produced water can have lower levels of salinity if its coal source was deposited under fresh water conditions, or if freshwater inflows to coal beds dilutes the formation water (Dahm et al., 2011). Many organic compounds have been identified in produced water. Most of these are naturally occurring constituents of petroleum. With the advent of advanced analytical techniques, more hydraulic fracturing fluid chemicals have been identified in produced water. These include some known tracer compounds, but others are known to exist whose identities have not yet been determined. Work has been done to identify environmentally benign tracers for assessing impacts, but these tracers have not been fully developed. Despite the presence in produced water of known hydraulic fracturing chemicals, the majority of organic and inorganic constituents of produced water come from the formation and cannot be minimized through actions of the operator. Throughout the formation-contact time, reactions occur between the constituents of the fracturing fluid and the formation.

Produced water spills have occurred across the country. From evaluation of data from across the United States and a focused study of North Dakota, the median produced water spill ranges from 336 to 1,000 gal (1,300 to 3,800 L). Although half of the spills are smaller than the median spill size, small numbers of much higher volume spills occur. In 2015, there were 12 spills in North Dakota greater than 21,000 gal (80,000 L), and one of 2,900,000 gal (11,000,000 L). From 2010 to 2015, there were approximately 5 to 7 produced water spills per hundred operating production wells. The major causes identified for these spills are container and equipment failures, human error, well communication, blowouts, pipeline leaks, and unpermitted discharges. Section 7.4.2 described impacts that were both of short and long term duration.

Highway transportation of produced water has resulted in crashes, but the impacts from these are unknown. Analysis of Texas crashes shows that as the oil and gas development activities increase, so do crashes, especially in rural areas. The EPA estimated the chance of a crash releasing produced water to range from 1:110 to 1:13,000.

7.6.2 Factors Affecting the Frequency or Severity of Impacts

The potential of spills of produced water to affect drinking water resources depends upon the release volume, duration, and composition, as well as watershed and water body characteristics. Larger spills of greater duration are more likely to reach a nearby drinking water resource than are smaller spills. Small releases, however, can impact resources where there are direct conduits from a source to receptor, such as fractures in rock. The composition of the spilled fluid also impacts the severity of a spill, as certain constituents are more likely to affect the quality of a drinking water resource.

Potential impacts to water resources from hydraulic fracturing related spills are expected to be affected by watershed and water body characteristics. For example, overland flow is affected by surface topography and surface cover. Infiltration of spilled produced water reduces the amount of water threatening surface water bodies. However, infiltration through soil can lead to groundwater impacts. Releases from pits can directly impact drinking water resources.

7.6.3 Uncertainties

The volume and some compositional aspects of produced water are known from published sources. The amount of hydraulic fracturing fluid returned to the surface is not well defined, because of the imprecise distinction between flowback and produced water. With regard to composition, TENORM and organics have the most limited data. Most of the available data on TENORM has come from the Marcellus Shale, where concentrations are typically high in comparison to the limited data available from other formations. Many organic constituents of produced water have been identified, and many of them are naturally occurring petroleum hydrocarbons. As methods improve and more data are collected, an increasing number of hydraulic fracturing fluid chemicals are being identified in produced water. Little is known concerning subsurface transformations and is reflected in only a few transformation products have been positively identified. Halogenation of organics has been noted, though.

Nationwide data on spills of produced water are limited in two primary ways: the completeness of reported data cannot be determined, and individual states' reporting requirements differ (U.S. EPA, 2015m). Therefore, the total number of spills occurring in the United States, their release volumes, and associated concentrations can only be estimated because of these underlying data limitations.

Spills vary in volume, duration, and composition, and most spill response focusses on immediate clean up, so several aspects of spills are not precisely characterized. The volume released is often a rough estimate, in part, because the spilled liquid spreads across the scene and is inherently difficult to measure. Simple measurements are often used to characterize the spill, rather than determining chemical concentrations (e.g., measuring electrical conductivity). As a consequence the suite of chemicals, and their concentrations, potentially impacting drinking water resources are usually unknown. Thus, the severity of impacts to drinking water resources is not usually well quantified.

Spills can originate from blowouts, well communication, aboveground or underground pipeline breaks, leaking pits, failed containers, human error (including unpermitted discharges, failure to detect spills, and failure to report spills) or unknown causes. The difference between these causes affects the location and size of the spill or release. For example, a container that fails may release a small amount of produced water, and be located on the well pad. A pipeline break may occur at a distance away from the well pad and release a larger amount of water from a bigger source (i.e., a pit). In addition, the factors governing transport of spilled fluid to a potential receptor vary by site: the presence and quality of secondary or emergency containment and spill response; the rate of overland flow and infiltration; the distance to a surface water body or drinking water well; and transport and fate processes. Impacts to drinking water resources from spills of produced water depend on environmental transport parameters, which can, in principle, be determined but are unlikely to be known or adequately specified in advance of a spill.

Because some constituents of produced water are constituents of natural waters (e.g., bromide in coastal surface waters) or can be released into the environment by other pollution events (e.g., benzene from gasoline releases, bromide from coal mine drainage), baseline sampling prior to impacts is one way to increase the certainty of an impact determination. Further sampling and investigation can be used to develop the linkage between a release and a documented drinking water impact. Appropriate sampling and analysis protocols, using quality assurance procedures, are essential for developing data that can withstand scrutiny. The EPA's northeastern Pennsylvania case study illustrates that the analytes that can be used to distinguish among types of water vary depending on the specifics of the situation. No single constituent or parameter could be used alone to assess impacts, and multiple lines of evidence were needed to assess the suspected impacts.

7.6.4 Conclusions

Produced water has the potential to affect the quality of drinking water resources if it enters into a surface water or groundwater body used as a drinking water resource. This can occur through spills at well pads or during transport of produced water. Specific impacts depend upon the spill itself, the environmental conditions surrounding the spill, water body and watershed characteristics, and the composition of the spilled fluid. The impacts from the majority of spills and releases is generally localized in extent as only the largest spills and releases impact large areas.

Chapter 8. Wastewater Disposal and Reuse

Abstract

This chapter addresses the practices and related impacts on drinking water resources that take place during the final stage of the hydraulic fracturing water cycle. This stage encompasses the management of wastewater, including disposal, reuse in hydraulic fracturing operations, or other uses. For this assessment, wastewater is defined as produced water from hydraulically fractured oil and gas wells that is managed by any of a number of strategies. The constituents of concern in hydraulic fracturing wastewaters that are most frequently noted include high total dissolved solids (TDS), chloride, bromide, and radionuclides (radium in particular). Other alkaline earth metals (e.g., barium), organics, and suspended solids, may be of concern as well.

Most hydraulic fracturing wastewater is managed by injection into Class II disposal wells. There are also "aboveground" management practices, which include reuse in subsequent hydraulic fracturing operations; treatment at a centralized waste treatment facility followed by reuse or discharge to surface water or a publicly owned treatment works; evaporation; irrigation; and direct discharge (under limited conditions). These practices can affect both surface water and groundwater.

Impacts on surface water arise from discharges of inadequately treated wastewater. In particular, bromide and iodide found in highly saline wastewaters can contribute to disinfection byproduct formation in downstream drinking water systems. If not removed during treatment, radium, metals, and organic compounds can also be discharged. Factors affecting the frequency and severity of impacts on surface waters include the wastewater's composition, its volume, and the processes used to treat it (common wastewater treatment processes do not significantly reduce the high TDS content in hydraulic fracturing wastewaters). In addition, site-specific factors such as local hydrology, size of the receiving water body, and other activities taking place in a watershed can affect the severity of the impact.

Pits and impoundments used for storage or disposal can impact surface water or groundwater through spills, leaks, and infiltration through soils. The frequency and severity of such impacts depend on pit construction and maintenance as well as proximity to drinking water resources. Unlined pits or those with compromised liners can cause long-lasting impacts on groundwater. Depth to the water table, soil properties, and the contaminants in the wastewater also affect the likelihood of impacts.

Characterizing the impacts from wastewater management associated with hydraulic fracturing is challenging given gaps in the data. Specifically, there are limited data on the wastewater volumes managed, on the influent and effluent concentrations and volumes from facilities that treat wastewater from hydraulic fracturing operations, and on wastewater residual characteristics and management of those residuals. Further, there is inadequate monitoring of drinking water resources for specific contaminants associated with hydraulic fracturing wastewater. However, the data that are available have shown that management of hydraulic fracturing wastewater through aboveground practices has affected the quality of water resources.

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8. Wastewater Disposal and Reuse

8.1 Introduction

The final stage of the hydraulic fracturing water cycle encompasses disposal and reuse of hydraulic fracturing wastewater. For the purposes of this assessment, "hydraulic fracturing wastewater" is defined as produced water from hydraulically fractured oil and gas wells that is being managed using practices that include, but are not limited to, reuse in subsequent hydraulic fracturing operations, treatment and discharge, and injection into disposal wells.[1,2,3] Although the term "wastewater" is generally used in this chapter, when more specific information about a wastewater is known (e.g., a source indicates the wastewater is flowback), that information is also noted.

Wells producing from oil and gas reservoirs generate produced water during the course of their productive lifespan. This produced water includes the often large volumes of flowback generated immediately after fracturing in deep wells with long horizontal sections. Flowback estimates vary by formation and are noted in Section 7.2.1 to range from about 300,000 to 10 million gal (1.14 to 37.8 million L) per well (Mantell, 2013; U.S. GAO, 2012). This large volume of initial flowback necessitates having a wastewater management strategy in place before hydraulic fracturing is initiated. Also, the longer-term generation of produced water requires ongoing wastewater management.

The majority of wastewater generated from all oil and gas operations in the United States is managed via Class II injection wells (Veil, 2015). Injection may be for either disposal or enhanced recovery. As hydraulic fracturing activity expands or diminishes, choices regarding disposal practices can change in a given region due to factors such as the quality and volume of the fluids; regulations; available infrastructure; and the feasibility and cost of treatment, reuse, and disposal options.

Several articles have noted potential effects of hydraulic fracturing wastewater on water resources (Vengosh et al., 2014; Olmstead et al., 2013; Rahm et al., 2013; States et al., 2013; Vidic et al., 2013; Rozell and Reaven, 2012; Entrekin et al., 2011). One study used probability modeling that indicated water pollution risk associated with gas extraction in the Marcellus Shale is highest for the wastewater disposal aspects of the operation (Rozell and Reaven, 2012). These concerns arise from

[1] The term "wastewater" is being used in this study as a general description of certain waters and is not intended to constitute a term of art for legal or regulatory purposes. This general description does not, and is not intended to, provide that the production, recovery, or recycling of oil, including the production, recovery, or recycling of flowback or produced water, constitutes "wastewater treatment" for the purposes of the Oil Pollution Prevention regulation (with the exception of dry gas operations), which includes the Spill Prevention, Control, and Countermeasure rule and the Facility Response Plan rule, 40 CFR 112 et seq.

[2] Disposal wells are Underground Injection Control (UIC) Class II wells, including those used for disposal (Class IID), enhanced oil recovery (Class IIR), and hydrocarbon storage (Class IIH).

[3] The term "reuse" is sometimes used to imply no treatment or basic treatment (e.g., media filtration) for the removal of constituents other than total dissolved solids (TDS), while "recycling" is sometimes used to convey more extensive treatment (e.g., reverse osmosis (RO)) to remove TDS (Slutz et al., 2012). In this document, the term "reuse" will be used to indicate use of wastewater for subsequent hydraulic fracturing, regardless of the level of treatment.

the elevated concentrations of chloride, bromide, radionuclides, and other constituents of concern found in many hydraulic fracturing wastewaters.

This chapter provides follow-on to Chapter 7, which discusses the per-well volumes of produced water (Section 7.2) and composition (Section 7.3), as well as the processes involved in its generation and impacts from a number of types of spills and releases. In this chapter, discussions are provided on management practices for hydraulic fracturing wastewater, available wastewater production information, and estimated aggregate volumes of wastewater generated for several states with active hydraulic fracturing (Section 8.2). As a complement to information on the composition of wastewaters in Chapter 7, Section 8.3 presents brief information on wastewater constituents and their relevance to wastewater management. Management methods used in recent years and their potential impacts on drinking water resources are described (Section 8.4). Based on background information provided in the earlier sections of the chapter, Section 8.5 discusses documented and potential impacts on drinking water resources from particular constituents, and a final synthesis discussion is provided (Section 8.6).[1]

8.2 Volumes of Hydraulic Fracturing Wastewater

This section provides a general overview of aggregate wastewater quantities generated in the course of hydraulic fracturing and subsequent oil and gas production, including estimates at regional and state levels. It also discusses methodologies used to produce these estimates and the associated challenges. (Chapter 7 provides a more in-depth discussion of the processes affecting produced water volumes and presents some typical per-well values and temporal patterns.) Wells also generate drilling fluid waste. Compared to produced water, however, drilling fluid wastewater can constitute a relatively small portion of the total wastewater produced (e.g., <10% in Pennsylvania during 2004-2013) (U.S. EPA, 2016d) and is not discussed further in this assessment.

Wastewater volume can be relevant to treatment costs, reuse options, and disposal capacities. IHS Global Insight suggests that as a general rule of thumb, the amount of flowback produced in the days or weeks after hydraulic fracturing is roughly comparable to the amount of produced water generated long-term over a span of years, which can vary considerably among wells (IHS, 2013). Thus, on a local level, operators can anticipate a relatively large volume of wastewater in the weeks following fracturing, with slower subsequent production of wastewater.

Wastewater volumes will most likely vary in the future as the amount and locations of hydraulic fracturing activities change and as existing wells age and move into the later phases of their production cycles. Substantial increases in wastewater production have occurred during times of increasing hydraulic fracturing activity. For instance, the average annual volume of wastewater

[1] This chapter makes use of background information collected by the EPA's Office of Water (OW) as part of the development of its recent pretreatment standards for wastewater from unconventional oil and gas formations (U.S. EPA, 2016d). The pretreatment standards apply to wastewater from crude oil and natural gas produced by a well drilled into shale and tight formations. Coalbed methane is beyond the scope of those standards. In this chapter, we consider wastewater generated by the hydraulic fracturing of those unconventional oil and gas formations included in the background research done by OW. But we also consider wastewater generated by hydraulic fracturing in coalbed methane and conventional formations.

generated by all gas production (both shale gas and conventional) in Pennsylvania quadrupled from the 2001-2006 period to the 2008-2011 period (Wilson and Vanbriesen, 2012).

However, although the total volume of wastewater might be expected to generally increase and decrease as oil and gas drilling and production changes, it is not necessarily a direct correlation. Data from the Pennsylvania Department of Environmental Protection (PA DEP) (PA DEP, 2016b) show trends in volumes of wastewater compared to gas produced from wells in the Marcellus Shale in Pennsylvania (Figure 8-1). Although the data show some variation, they demonstrate a general positive correlation between the volume of wastewater and the amount of produced gas until early 2015. At that time, Baker Hughes weekly rig counts also began to drop, declining from 85 in early January 2014 to 24 in early June 2016 (Baker Hughes, 2016). This suggests that a decline in overall drilling activity (generally a measure of new wells) can be associated with a decline in wastewater production, although the exact timing depends on whether there is a time delay between drilling and completion of a well and the start of production from that well.

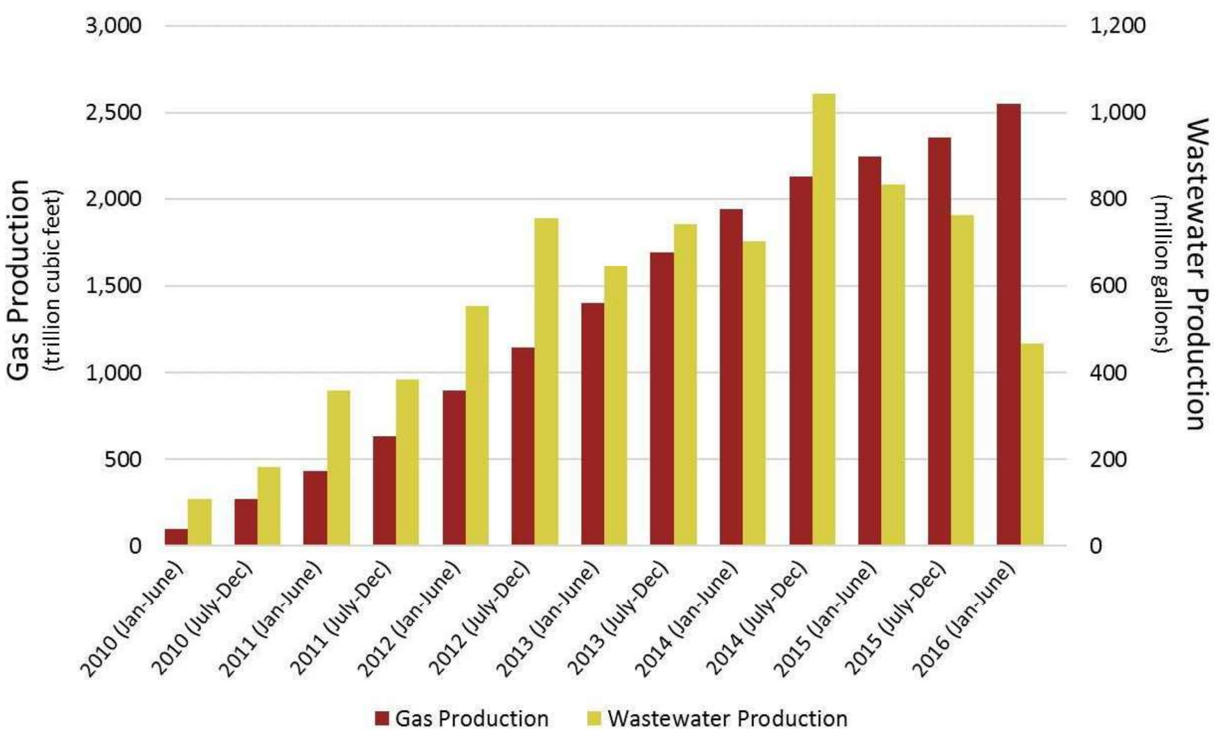

Figure 8-1. Wastewater (i.e., produced water and fracturing fluid waste) and produced gas volumes from unconventional (as defined by PA DEP) wells in Pennsylvania from January 2010 through June 2016.
Source: PA DEP (2016b).

Estimates of produced water compiled by Veil (2015) indicate that although oil and gas production in the United States increased by 29% and 22%, respectively, between 2007 and 2012, produced water volumes increased by less than 1%. There may be a number of factors contributing to this, including as noted by Veil (2015), a number of uncertainties associated with produced water estimates. First, wastewater generation varies from well to well, as do oil and gas production (see Chapter 7, Figure 7-1 for discussion of wastewater/produced water decline curves). The rates of decline in both wastewater volume and hydrocarbon production also vary among reservoirs. Additionally, some wells are drilled and completed but are not immediately put into production. Relationships between hydraulic fracturing activity, hydrocarbon production, and produced water volumes are likely reservoir- (and maybe production zone-) specific, and existing wells and production need to be considered to anticipate wastewater management needs.

8.2.1 National Level Estimate

Veil (2015) estimated that in 2012, U.S. onshore and offshore oil and gas production generated 889.59 billion gal (21.18 billion bbls) of produced water. This national-level estimate represents total oil and gas wastewater (from all oil and gas resources, and from wells hydraulically fractured and wells not hydraulically fractured). The estimate was compiled through a state-by-state analysis of survey data obtained from oil and gas agencies in the 31 states with active oil and gas production as well as the Department of Interior and U.S. EPA. However, Veil notes several issues with the data used for these estimates, including variability among states in data reporting, availability, and completeness. These issues may result in underestimation of the volumes of water produced (U.S. GAO, 2012). See Section 8.2.3 for more discussion on methods of estimating wastewater volumes.

8.2.2 Regional/State Level Estimates

A limited number of studies have described the geographic variability in oil and gas wastewater volumes. Veil (2015) reported that the top ten states nationwide for wastewater production in 2012 included Texas (35% of total), California (15% of total), Oklahoma (11% of total), and Wyoming (11% of total). A study by the Bureau of Land Management (BLM) (Guerra et al., 2011) states that in 2008, more than 80% of all oil and gas wastewater was generated in the western United States, with Texas producing the highest volume and Wyoming the second highest. The BLM report notes substantial wastewater from CBM wells, in particular those in the Powder River Basin (Wyoming). Figure 8-2 summarizes the wastewater volumes for these western states, demonstrating the wide variability from state to state (likely reflecting differences in the number of oil and gas production wells/production activity and reservoir geology). Although the authors do not identify all wastewater contributions from production involving hydraulic fracturing, the regions with established oil and gas production are likely to have methods and infrastructure available for management of hydraulic fracturing wastewater.

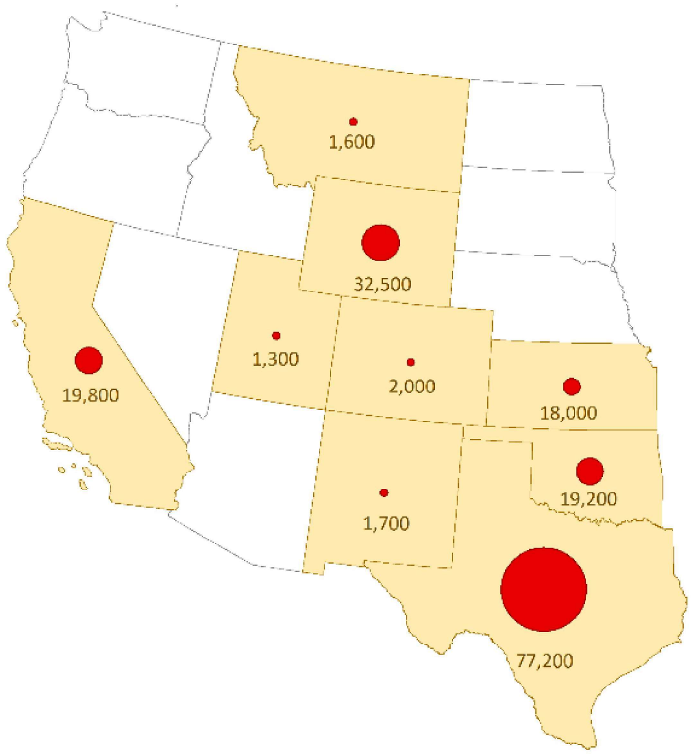

Figure 8-2. Wastewater quantities in the western United States (billions of gal per year).
Data from Guerra et al. (2011).

In the Marcellus region, waste data made public by the PA DEP have enabled analyses of wastewater volumes and trends since 2009. Estimates of produced water (including flowback or "fracing fluid waste" as well as "produced fluid") by Wunz (2015) and Shale Alliance for Energy Research Pennsylvania (SAFER PA, 2015) for 2014 are 1.73 and 1.64 billion gal (41.19 MMbl and 39.05 MMbl, respectively). The estimate compiled for this assessment is 0.65 billion gal (15.48 million bbls) for the first half of 2014 (Table 8-1). Variations among estimates reflect, among other factors, challenges in working with a dynamic database for which changes and corrections are ongoing.

Table 8-1 presents estimates of the volumes of hydraulic fracturing wastewater generated and the associated numbers of wells in North Dakota, Ohio, Pennsylvania, Texas (injected flowback only), and West Virginia. The data shown in this table were compiled for this assessment (except for West Virginia) and come primarily from records of produced water made publicly available on state websites.[1] These states are represented in Table 8-1 because the produced water volumes associated with hydraulic fracturing were readily identifiable. The data show that the increase in

[1] Data used for Table 8-1 were downloaded from state agency websites and compiled as needed (in either Microsoft Excel or Microsoft Access) for each state except West Virginia. Once compiled, data were filtered if needed and summed to produce estimates of wastewater production by year and a count of the numbers of wells generating the wastewater. Data were downloaded up through 2014. (Note that 2014 data for Pennsylvania and Texas are for partial years.) Differences in the years presented for the states are due to differences in data availability from the state agency databases. For West Virginia, data are from a report by Hansen et al. (2013) that compiled available flowback data from West Virginia.

the number of wells producing wastewater and the volumes of wastewater produced are generally consistent with the timing of the expansion of high-volume hydraulic fracturing and track with the increase in horizontal wells seen in Figure 3-20.

Several states with mature oil and gas industries (California, Colorado, New Mexico, Utah, and Wyoming) make produced water volumes publicly available by well as part of their oil and gas production data, but they do not directly indicate which wells have been hydraulically fractured. Some states (Colorado, Utah, Wyoming, and New Mexico) specify the producing formation or the basin along with volumes of hydrocarbons and produced water. Determining volumes of hydraulic fracturing wastewater for these states is challenging because there is a possibility of either inadvertently including wastewater from wells not hydraulically fractured or of missing volumes that should be included. This may be a particular problem where state terminology regarding what constitutes an unconventional resource or hydraulically fractured well is ambiguous or possibly different from other states. Appendix Table F-1 provides estimates of wastewater volumes in California, Colorado, New Mexico, Utah, and Wyoming in regions where hydraulic fracturing activity is taking place, along with notes on data limitations. The data in Table 8-1 and Appendix Table F-1 illustrate the challenges in both compiling a national estimate of hydraulic fracturing wastewater and comparing wastewater production among states due to dissimilar data types, presentation, and availability.

8.2.3 Estimation Methodologies and Challenges

Compiling and comparing data on wastewater production at the wide array of oil and gas locations in the United States presents challenges associated with data reporting and availability. Different approaches have been used to estimate state-specific and national wastewater volumes. Data from state agency websites and databases can be a ready source of information, whether publicly available and downloadable or provided directly by agencies upon request.

Veil (2015) notes that the reported volumes of produced water (e.g., reported by well in state production data) can be inaccurate or imprecise because produced water is not monitored continuously. Therefore, reported volumes may be estimates. Other issues such as data transcription errors or extrapolation of data can also affect reported volumes (Veil, 2015).

Using produced water volumes from state records to estimate the volume of wastewater regionally or nationally presents additional challenges due to a lack of consistency in data collection, availability, usability, completeness, and accuracy (Malone et al., 2015; Veil, 2015; U.S. GAO, 2012). Due to what are sometimes significant differences in the types of data collected and the mechanisms, formats, and definitions used, data cannot always be directly compared from state to state. This makes it difficult to aggregate volume data, and estimates may be better and more useful at a local or state level. Larger-scale estimates across regions or between states should be interpreted carefully.

Table 8-1. Estimated volumes (millions of gal) of wastewater based on state data for selected years and numbers of wells producing fluid.

State	Basin	Principal lithologies	Data type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
North Dakota	Williston	Shale	Produced water	2	3	130	790	1,900	4,500	8,500	9,700	From North Dakota Oil and Gas Commission website[a]. Data provided for six members of the Bakken Shale. Produced water includes flowback (reports are submitted within 30 days of well completion.)
			Wells	*161*	*152*	*844*	*2,083*	*3,303*	*5,036*	*6,913*	*8,039*	
Ohio	Appalachian	Shale	Primarily flowback	-	-	-	-	3	29	110	-	Data from Ohio DNR Division of Oil and Gas[b]. Utica data for 2011 and 2012. Utica and Marcellus data for 2013. Brine is noted by agency to be mostly flowback.
			Wells	-	-	-	-	*9*	*86*	*400*	-	
Pennsylvania	Appalachian	Shale	Flowback plus produced water (% flowback; % produced water)	-	-	-	180 (51%; 49%)	740 (46%; 54%)	1,100 (36%; 64%)	1,300 (27%; 73%)	650 (32%; 68%)	Waste data from PA DEP[c]. Second half of 2010 and first half of 2014. Data described as unconventional as defined by PADEP. Separate codes are provided by PA DEP for flowback and produced water.
			Wells	-	-	-	*1,232*	*2,434*	*4,039*	*5,015*	*5,150*	

State	Basin	Principal lithologies	Data type	2000	2004	2008	2010	2011	2012	2013	2014	Comments
Texas	Unspecified (entire state)	Shale, Sandstone	Flowback - injected volumes	-	-	-	-	490	2,200	3,100	2,000	Waste injection data from Texas Railroad Commission[d]. Monthly totals are provided for entire state. Oct - Dec for 2011, full years for 2012 and 2013, and Jan - Oct for 2014
West Virginia	Appalachian	Shale	Flowback - Estimated total disposed	-	-	-	120	110	59	-	-	Estimates from Hansen et al. (2013).[e]

[a] North Dakota Industrial Commission. Department of Mineral Resources. Bakken Horizontal Wells By Producing Zone: https://www.dmr.nd.gov/oilgas/bakkenwells.asp.

[b] Ohio Department of Natural Resources, Division of Oil and Gas Resources. Oil and Gas Well Production. http://oilandgas.ohiodnr.gov/production#ARCH1.

[c] PA DEP Oil and Gas Reporting website, https://www.paoilandgasreporting.state.pa.us/publicreports/Modules/Welcome/Agreement.aspx.

[d] Railroad Commission of Texas, Injection Volume Query, http://webapps.rrc.state.tx.us/H10/searchVolume.do;jsessionid=J3cgVHhK9nkwPrC7ZcWNMgyzF9LCYyR1NmvDy3F1QQ5wqXfcGNGN!1841197795?fromMain=yes&sessionId=143075601021612. Texas state data provide an aggregate total amount of flowback injected for the past few years. (Data on brine volumes injected do not differentiate hydraulically fractured wells and, therefore, well counts are not presented here.) These values are interpreted as estimates of generated flowback as based on reported quantities of "fracture water flow back" injected into Class IID wells.

[e] West Virginia flowback estimates from Hansen et al. (2013) are based on state data. Well counts that are explicitly associated with the flowback and total disposed volumes were not available.

To compile estimates of the production and management of hydraulic fracturing wastewater, there are additional challenges. Reporting of wastewater volumes may or may not include information that helps determine whether the producing well was hydraulically fractured (e.g., an indicator of resource type or formation). It also might not be clear whether volumes listed as 'produced water' include the flowback component. Some states (e.g., Colorado and Pennsylvania) include information on disposal and management methods along with production data, and others do not.

Given the limitations of comparing state databases, some studies have generated estimates of wastewater volume using water-to-gas and water-to-oil ratios along with the reports of hydrocarbon production (Murray, 2013). The reliability of any wastewater estimates made using this method would need to be evaluated in terms of the quality, timeframe, and spatial coverage of the available data, as well as the extent of the area to which the estimates will be applied. Water-to-hydrocarbon ratios are empirical estimates. Because these ratios show a wide variation among formations, reliable data are needed to formulate a ratio in a particular region.

Another approach to estimating wastewater volumes would entail multiplying per-well estimates of produced water production rates by the numbers of wells in a given area. Challenges associated with this approach include obtaining accurate estimates of the number of new and existing wells, along with accurate estimates of per-well water production both during the flowback period and during the production phase of the well's lifecycle. In particular, it can be challenging to correctly match per-well wastewater production estimates, which will vary by formation, with counts of wells, which may or may not be clearly associated with specific formations. Temporal variability in wastewater generation would also be difficult to capture and would add to uncertainty. Such an approach, however, may be attempted for order of magnitude estimates if the necessary data are available and reliable.

8.3 Wastewater Characteristics

Along with wastewater volumes, wastewater characteristics and the characteristics of residuals produced during treatment or storage are important for understanding the potential impacts of management and disposal of hydraulic fracturing wastewater on drinking water resources. This section provides brief highlights on several important constituents known to exist in hydraulic fracturing wastewaters and residuals. Chapter 7 provides more in-depth detail on the chemistry of produced water, and Chapter 9 discusses reference values and health effects associated with hydraulic fracturing wastewater constituents.

8.3.1 Wastewater

Wastewater composition is the result of naturally-occurring constituents originating in the formation solids and fluids as well as chemicals associated with the fracturing fluid. Discussion in this chapter focuses on constituents in hydraulic fracturing wastewater for which adequate information is available to assess documented and potential impacts on drinking water resources. There may also be unknown constituents in wastewaters for which analyses have not been performed. This is due, in part, to a lack of information on fracturing fluid ingredients identified as confidential business information (CBI). In addition, there are uncertainties about how fracturing fluid ingredients are degraded or removed in the subsurface. (See Chapter 5, Section 5.8 for a discussion of processes that can cause chemicals to degrade or transform in the subsurface.)

8.3.1.1 Total Dissolved Solids and Inorganics

Hydraulic fracturing wastewaters are generally high in total dissolved solids (TDS), especially those from shales and tight formations, with TDS values ranging from less than 1,000 mg/L to hundreds of thousands of mg/L (Section 7.3.4.4). The TDS in wastewaters from shale formations is typically dominated by sodium and chloride and may also include elevated concentrations of bromide, bicarbonate, sulfate, calcium, magnesium, barium, boron, strontium, radium, organics, and heavy metals (Chapman et al., 2012; Rowan et al., 2011; Blauch et al., 2009; Orem et al., 2007; Sirivedhin and Dallbauman, 2004).

Within each formation, the minimum and maximum values presented in Section 7.3.4.4 suggest spatial variation in TDS content that may need to be accommodated when considering management strategies such as reuse or treatment. In contrast to shales and sandstones, TDS values for wastewater from CBM formations are generally lower, with reported concentrations ranging from approximately 150 mg/L to 62,000 mg/L (DOE, 2014b; Dahm et al., 2011) (Appendix Table E-3). This results in fewer treatment challenges and a wider array of management options.

Constituents commonly found in TDS from hydraulic fracturing wastewaters may have potential health impacts or create treatment burdens on downstream drinking water systems if discharged at high concentrations to drinking water resources. Bromide, for example, can contribute to the increased formation of disinfection byproducts (DBPs) during drinking water treatment (Hammer and VanBriesen, 2012); see Section 8.5.1.

8.3.1.2 Organics

Less information is generally available about organic constituents in hydraulic fracturing wastewaters than about inorganic constituents, but there are now several studies reporting analyses of organic constituents (Chapter 7). The organic content in flowback waters can vary based on the chemical additives (e.g., biocides, antiscalants, gelling agents, breakers) used in hydraulic fracturing fluids and the chemistry of the formation, but the organics generally include polymers, oil and grease, volatile organic compounds (VOCs), and semi-volatile organic compounds (SVOCs) (Akob et al., 2016; Walsh, 2013; Hayes, 2009). Examples of other constituents detected include alcohols, naphthalene, acetone, and carbon disulfide, compounds that may be remnants of hydraulic fracturing fluid chemicals (Hayes and Severin, 2012b; Hayes, 2009) (Appendix E). Wastewater associated with CBM wells may have high concentrations of aromatic and halogenated organic contaminants potentially requiring treatment depending on how the wastewater will be managed (Pashin et al., 2014; Sirivedhin and Dallbauman, 2004). Concentrations of BTEX (benzene, toluene, ethylbenzene, and xylenes) in CBM produced waters are lower than in shale produced waters (Appendix Table E-9).

New research is focusing on transformation products generated in the subsurface; experimental work by Kahrilas et al. (2015) suggests that the biocide glutaraldehyde can be present in wastewaters along with its transformation products. Low molecular weight organic acids such as acetate, formate, and pyruvate have been detected in Marcellus wastewater, indicating microbial degradation of organic compounds in the fracturing fluid or formation (Akob et al., 2015).

8.3.1.3 Radionuclides

Radionuclides are constituents of concern in some hydraulic fracturing wastewaters, with most of the available data obtained for the Marcellus Shale in Pennsylvania (Appendix Table E-8). Results from a United States Geological Survey (USGS) report (Rowan et al., 2011) indicate that the predominant radionuclides in Marcellus Shale wastewater are radium-226 and radium-228. Radionuclides in produced fluids are considered 'technologically enhanced naturally-occurring radioactive material' (TENORM) because they have been exposed to the accessible environment.[1]

Although data regarding radionuclides in wastewater from formations other than the Marcellus Shale are limited, there is information on the naturally occurring radioactive material (NORM) in the formations themselves.[2] In particular uranium and thorium can be found in certain organic-rich black shales. High uranium content has been measured in the Marcellus, Barnett, Woodford, and other black shales (Swanson, 1955) (Section 7.3.4.6). Radium-226 and -228 are decay products of uranium and thorium and are soluble (Sturchio et al., 2001; Langmuir and Riese, 1985). Therefore wastewater from shales with high concentrations of uranium and thorium can contain radium, especially where TDS concentrations are also high (Rowan et al., 2011; Sturchio et al., 2001; Fisher, 1998). Section 7.3.3.2 provides further information on radionuclides in produced waters and in formations.

8.3.2 Constituents in Residuals

Depending on the wastewater and the treatment processes used, treatment residuals can consist of sludges, spent media (used filter materials), or brines. Residuals may require further treatment (e.g., dewatering sludges) prior to disposal (see Section 8.4.7 for further discussion on management of residuals). Residuals can contain constituents such as total suspended solids (TSS), TDS, metals, radionuclides, and organics. These constituents will be concentrated in the residuals, with the degree of concentration depending on the type of treatment employed. Processes such as electrodialysis and mechanical vapor recompression have been found to yield residuals with TDS concentrations in excess of 150,000 mg/L after treating waters with influent TDS concentrations of approximately 50,000 – 70,000 mg/L (Hayes et al., 2014; Peraki and Ghazanfari, 2014).

Also, TENORM in wastewaters can cause residual wastes to have gamma radiation emissions (Kappel et al., 2013). A laboratory study by Zhang et al. (2014b) estimated that the barium sulfate solids precipitated during treatment to remove barium and strontium from Marcellus Shale wastewater would also contain between 2,571 and 18,087 pCi/g of radium due to coprecipitation. Another similar study using mass balances calculated that sludge from a sulfate precipitation

[1] Technologically Enhanced Naturally Occurring Radioactive Material (TENORM) is defined by the EPA as naturally occurring radioactive materials (NORM) that have been concentrated or exposed to the accessible environment as a result of human activities such as manufacturing, mineral extraction, or water processing.

[2] Naturally Occurring Radioactive Materials (NORM) are radioactive materials found in nature that have not been moved or concentrated by human activities.

process would have an average radium concentration of 213 pCi/g (Silva et al., 2012). In sludge from lime softening processes, Silva et al. (2012) estimated a radium-226 concentration of 58 pCi/g, a level that would necessitate disposal as a low-level radioactive waste.

8.4 Wastewater Management Practices and Their Potential Impacts on Drinking Water Resources

Operators have several strategies for management of hydraulic fracturing wastewaters (Figure 8-3), with the most common choice being disposal via Class IID wells (Veil, 2015; Clark et al., 2013; Hammer and VanBriesen, 2012). Other practices include reuse in subsequent hydraulic fracturing operations (with varying levels of treatment), treatment at a centralized waste treatment facility (CWT) (often followed by reuse), evaporation (in arid regions), irrigation (with no discharge to waters of the United States), and direct discharge for livestock or agricultural use (allowed west of the 98th meridian). Up until 2011, treatment of unconventional oil and gas wastewaters (as defined by PA DEP) at publicly owned treatment works (POTWs) was a common practice for wastewater management in the Marcellus region (Lutz et al., 2013); this is discussed further in Text Box 8-1.

The methods shown in Figure 8-3 represent wastewater management strategies, not all of which would be used at the same facility. Descriptions of incidents of unpermitted disposal and resulting legal actions have also been publicly reported (Chapter 7). However, such events are not generally described in the scientific literature, and the prevalence of this type of activity is unclear. Additional sources of information about potential impacts exist, but some records are sealed (e.g., litigation records) and are not publicly accessible.

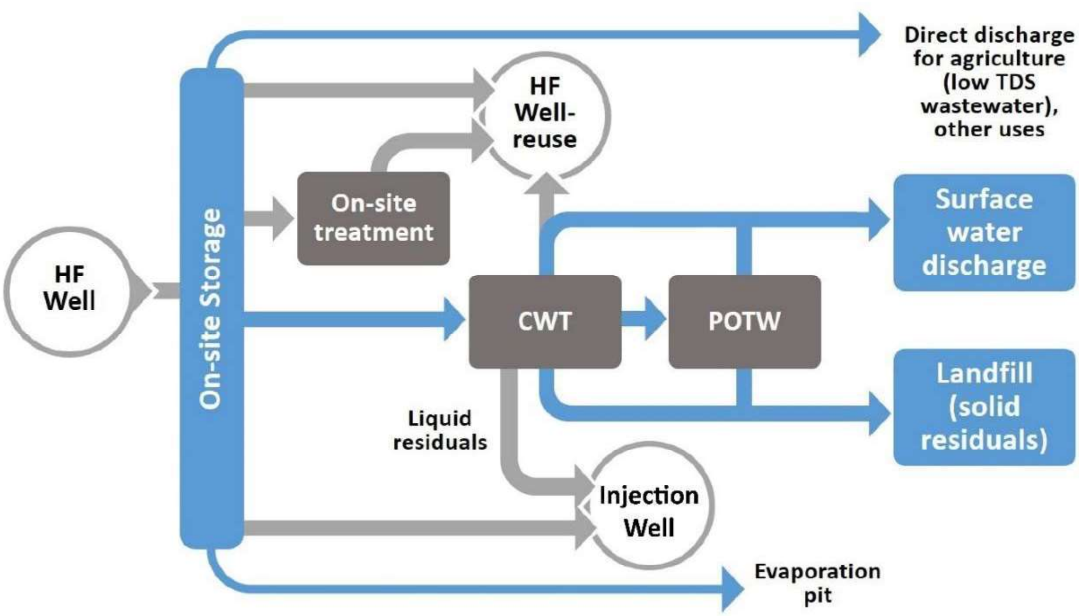

Figure 8-3. Schematic of wastewater management strategies.

Gray lines indicate management strategies that involve injection, either for reuse or disposal, and blue lines indicate management strategies that lead to other end points such as discharge, evaporation, landfills, or other uses.

Each of the wastewater management strategies can potentially lead to impacts on drinking water resources during some phase of their execution. Such impacts include, but are not limited to: accidental releases during transport (Chapter 7); discharges of treated wastewaters from CWTs or POTWs where treatment for certain constituents has been inadequate; migration of constituents to groundwater due to leakage from pits or land application of wastewaters; leakage from pits that reach surface waters (Chapter 7, Section 8.4.5); inappropriate management of liquid or solid residuals (e.g., leaching from landfills); or accumulation of constituents in sediments near outfalls of CWTs or POTWs that are treating or have treated hydraulic fracturing wastewater.[1]

A reliable census of oil and gas wastewater management practices nationwide is difficult to assemble due to a lack of consistent and comparable data among states. In addition, we do not know how often operators use more than one wastewater management strategy at a site (e.g., evaporation and injection), further complicating the tracking of wastewater management. As part of a data survey conducted by Veil (2015), some state agencies provided estimates of oil and gas wastewater volumes handled by several management practices (Table 8-2). These estimates illustrate how widespread injection for both enhanced recovery and for disposal is for managing oil and gas wastewater. The data also show regional differences in reuse and other practices. For hydraulic fracturing wastewaters, Table 8-3 illustrates nationwide variability in the primary wastewater management methods using qualitative and quantitative sources. Similar to Table 8-2, Table 8-3 shows disposal via underground injection predominates in most regions, and reuse is predominant in the Marcellus Region. (Table 8-3 does not include wastewater management in areas of CBM production.)

Management choices are affected by cost and a number of directly and indirectly related factors, including the chemical properties of the wastewater; the volume, duration, and flow rate of the wastewater generated; the feasibility of each option; the availability of necessary infrastructure; local, state, and federal regulations (Text Box 8-2); and operator discretion (U.S. GAO, 2012; NPC, 2011a). The economics (such as transport, storage, and disposal costs) and availability of treatment and disposal methods are of primary importance (U.S. GAO, 2012). For wastewater composition, there is limited information on the degradation or removal of fracturing fluid ingredients in the subsurface. Chemical disclosure requirements vary among states, and some fracturing fluid ingredients are claimed to be CBI. Therefore, the possible presence of unknown chemical constituents in wastewater contributes to uncertainty about the effectiveness and potential impacts of management strategies, particularly with regard to treatment efficacy.

[1] The term surface water as used in this assessment refers to surface waters that meet the definition of waters of the United States under the CWA (House Bill No. 1950, 2011).

Table 8-2. Estimated percentages of wastewater managed by practice and by state.
Source: Veil (2015). Estimates do not identify interstate transport (e.g., wastewater transported from PA to OH or WV for injection into disposal wells). Thus, there may have been some double counting of volumes in both the generating and receiving states.

Management practice	Percentage of produced water managed by practice and state											
	AR	CA	CO	NM	ND	OH	OK	PA	TX	UT	WV	WY
Injection for enhanced oil recovery	22	46	32	50 [d]	18	4.0	47	0	48	40	27	73
Injection for disposal	76	20	32	50 [d]	56	91	47	12	37	47	25	27
Surface discharge	0	2	10	no data	0	0	0	2.3	5.0 [f]	6	0	uncertain
Evaporation	0	21	9.0	no data	0	0		0	0	0	0	uncertain
Offsite commercial disposal	0.1 [a]	9	5.7 [c]	no data	26	Included in injection for disposal	6.0 [e]	0	10 [e]	7 [g]	28 [h]	uncertain
Beneficial reuse	1.1 [b]	no data	12 [b]	no data	0	5.0	0	85 (includes reuse for HF)	Est. 15-20 (flowback fluid)	0.5	uncertain	uncertain

[a] Land farm.

[b] Reuse for HF.

[c] Pits.

[d] Assumes even split with injection for enhanced oil recovery and injection for disposal.

[e] Injection.

[f] Fresh produced water.

[g] Evaporation ponds.

[h] Disposal wells.

Table 8-3. Management practices for wastewater from unconventional oil and gas resources.
Source: U.S. EPA (2016d).

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
Michigan	Antrim	Shale gas		XXX			Qualitative
Appalachian	Marcellus/Utica (PA)	Shale gas	XXX	XX	XX	Limited disposal wells in east	Quantitative
	Marcellus/Utica (WV)	Shale gas/oil	XXX	XX	X		Quantitative
	Marcellus/Utica (OH)	Shale gas/oil	XX	XXX	X		Mixed
Anadarko	Granite Wash	Tight gas	XX	XXX	X[a]		Mixed
	Mississippi Lime	Tight oil	X	XXX		Reuse/recycling limited but is being evaluated	Qualitative
	Woodford, Cana, Caney	Shale gas/oil	X	XXX	X[a]		Qualitative
Arkoma	Fayetteville	Shale gas	XX	XX	X[a]	Few existing disposal wells; new CWT facilities are under construction	Mixed
Fort Worth	Barnett	Shale gas	X	XXX	X[a]	Reuse/recycle not typically used due to high TDS early in flowback and abundance of disposal wells	Mixed
Permian	Avalon/Bone Springs, Wolfcamp, Spraberry	Shale/tight oil/gas	X	XXX	X[a]		Mixed
TX-LA-MS Salt	Haynesville	Tight gas	X	XXX		Reuse/recycle not typically used due to high TDS early in flowback and abundance of disposal wells	Mixed
West Gulf	Eagle Ford, Pearsall	Shale gas/oil	X	XXX	X		Mixed
Denver Julesburg	Niobrara	Shale gas/oil	X	XXX	X		Mixed
Piceance; Green River	Mesaverde/Lance	Tight gas	X	XX	X	Also managed through evaporation to atmosphere in ponds in this region	Qualitative

Basin	Formation	Resource type	Reuse	Injection for disposal	CWT facilities	Notes	Available data[b]
Williston	Bakken	Shale oil	X	XXX		Reuse/recycling limited but is being evaluated	Mixed

[a] CWT facilities identified in these formations are all operator-owned.

[b] This column indicates the type of data the EPA based the number of Xs on. In most cases, the EPA used a mixture of qualitative and quantitative data sources along with engineering judgment to determine the number of Xs.

XXX—The majority (≥50%) of wastewater is managed with this management practice; XX—A moderate portion (≥10% and <50%) of wastewater is managed with this management practice; X—This management practice has been documented in this location, but for a small (<10%) or unknown percent of wastewater. Blanks indicate the management practices have not been documented in the given location.

The availability and use of wastewater management strategies in a region can change over time as oil and gas production increases or decreases, regulations change, costs shift, and technologies evolve. Text Box 8-1 and Figure 8-4 illustrate shifting wastewater management practices in Pennsylvania as gas development in the Marcellus Shale increased and concerns over high-TDS discharges prompted a regulatory response. Reuse has increased substantially at well sites in Pennsylvania (labeled as "Reuse HF" in Figure 8-4) and wastewater management at CWTs has moved toward more facilities that provide wastewater for reuse and do not discharge (termed "zero-discharge facilities"). The estimated total reuse rate in Pennsylvania was 80% in 2012 and 90% in 2013 (PA DEP, 2015a). In contrast, wastewater disposal data in areas of Colorado where hydraulic fracturing takes place show a steady use of injection wells, an increase in surface water discharges, and a decrease in the use of on-site pits for evaporation since 2000 (Figure 8-5).

Another factor influencing reuse is the pace of hydraulic fracturing in the area. When hydraulic fracturing is active, demand for reuse is high. Some formations that are hydraulically fractured such as the Marcellus Shale and the Utica Shale are still in the early stages of development, with large potential resources not yet developed. For these plays, the need for wastewater treatment and/or reuse may remain high for decades to come, and the long-term wastewater management needs must be considered and addressed (SAFER PA, 2015).[1]

Researchers have developed optimization models to aid in the minimization of wastewater management costs as a part of comprehensive water management planning. For example, Yang et al. (2014) suggest an approach for reusing flowback in scheduled hydraulic fracturing events to minimize the operational costs of transportation, treatment, storage, and wastewater disposal. Another modeling study proposes an approach to minimize the total cost of water usage and wastewater treatment and disposal by optimizing capital costs (such as the costs of treatment units and storage pits) and operating costs for flowback management, treatment, storage, reuse, and wastewater disposal (Lira-Barragan et al., 2016).

Text Box 8-1. Temporal Trends in Wastewater Management – Experience of Pennsylvania.

Gross natural gas withdrawals from shale formations in the United States increased 518% between 2007 and 2012 (EIA, 2014b). This production increase has led to larger volumes of wastewater requiring appropriate management (Vidic et al., 2013; Gregory et al., 2011; Kargbo et al., 2010). The rapid increase in wastewater generated from hydraulically fractured oil and gas wells has led to many changes in wastewater disposal practices in the oil and gas industry. Changes have been most evident in Pennsylvania, which has experienced a more than 1,400% increase in natural gas production since 2000 (EIA, 2014b).

Lutz et al. (2013) estimated that total wastewater generation in the Marcellus region increased 570% between 2004 and 2013. The authors concluded that this increase has created stress on the existing wastewater disposal infrastructure. In 2010, concerns arose over elevated TDS in the Monongahela River

(*Text Box 8-1 is continued on the following page.*)

[1] As noted in Chapter 3, oil and gas prices influence new drilling activity. However, the links between oil and gas prices and the generation of wastewater (as a byproduct of production) appear to be less direct.

Text Box 8-1 (continued). Temporal Trends in Wastewater Management – Experience of Pennsylvania

basin, and studies linked high TDS (and, in particular, high bromide levels) to elevated DBP levels in drinking water systems (PA DEP, 2011a). In response, PA DEP amended Chapter 95 Wastewater Treatment Requirements under the Clean Streams Law for new discharges of TDS in wastewaters. This regulation is also informally known as the 2010 TDS regulation. The regulation disallowed any new direct discharges to streams as well as direct disposal at POTWs of hydraulic fracturing wastewater and set limits on treated discharges from new CWTs of 500 mg/L TDS, 250 mg/L chloride, 10 mg/L barium, and 10 mg/L strontium. Existing discharges were exempt.

In April 2011, PA DEP announced a request that by May 19, 2011, gas drilling operators voluntarily stop transporting wastewater from shale gas extraction (i.e., unconventional resources as defined by PA DEP) to the eight CWTs and seven POTWs that were exempt from the 2010 TDS regulation.[1] Follow-up letters from PA DEP to the owners of the wells specified that the role of bromides from Marcellus Shale wastewaters in the formation of total trihalomethanes (TTHM) was of concern due to the their potential public health impacts (PA DEP, 2011a).

In response to the request, the oil and gas industry in Pennsylvania accelerated the switch of wastewater deliveries from POTWs to CWTs for better removal of metals and suspended solids (Schmidt, 2013). Effluent sampling at two POTWs that had accepted Marcellus Shale wastewater showed that concentrations of bromide, chloride, barium, strontium, and sulfate dropped after the April 2011 request (Ferrar et al., 2013); data based on two sampling events, one before and one after May 2011).

Between early and late 2011, although reported wastewater production more than doubled, Marcellus Shale drilling companies in Pennsylvania reduced their use of CWTs that were exempt from the 2010 TDS regulation by 98%, and direct disposal of Marcellus Shale wastewater at POTWs was "virtually eliminated" (Hammer and VanBriesen, 2012).

Along with the decreased discharges from POTWs, there has been increased reuse of wastewater in the Marcellus Shale region. From 2008-2011, reuse of Marcellus wastewater for hydraulic fracturing increased, POTW treatment volumes decreased, tracking of wastewater improved, and wastewater transportation distances decreased (Rahm et al., 2013). Maloney and Yoxtheimer (2012) analyzed data from 2011 and found that reuse of flowback increased to 90% by volume. Eight percent of flowback was sent to CWTs. Brine water, which was defined as formation water, was reused (58%), disposed via injection well (27%), or sent to CWTs (14%). Of all the fluid wastes in the analysis, brine water was most likely to be transported to other states (28%). Maloney and Yoxtheimer (2012) also concluded that wastewater disposal to municipal sewage treatment plants declined nearly 100% from 47,221 bbls in the first half of 2011 to 408 bbls in the second half.

[1] An unconventional formation was defined in 2011 by the state of Pennsylvania as "A geological shale formation existing below the base of the Elk Sandstone or its geologic equivalent stratigraphic interval where natural gas generally cannot be produced at economic flow rates or in economic volumes except by vertical or horizontal wellbores stimulated by hydraulic fracture treatments or by using multilateral wellbores or other techniques to expose more of the formation to the wellbore." The EPA defines unconventional oil and gas as crude oil and natural gas produced by a well drilled into a shale and/or tight formation (including, but not limited to, shale gas, shale oil, tight gas, and tight oil). For the purpose of the rule, the definition of UOG does not include CBM (U.S. EPA, 2016d).

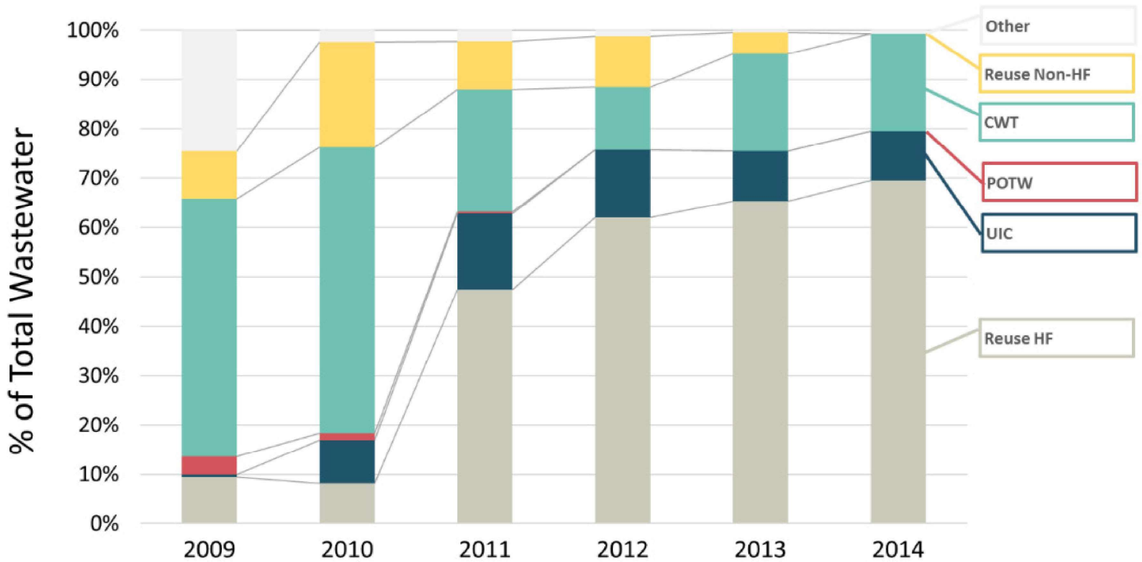

Figure 8-4. Percentages of total unconventional wastewater (as defined by PA DEP) managed via various practices for the second half of 2009 through the first half of 2014.
The volume sent to POTWs in 2013 was 0%. Note also that a majority of wastewater sent to CWTs is subsequently reused, so that when combined with "Reuse HF," the total reuse rate was approximately 90% in 2013. "Reuse HF" indicates on-site reuse. Source: Waste data from PA DEP (2015a).

Text Box 8-2. Regulations Affecting Wastewater Management.

Regulations affect wastewater management options and vary geographically as well as over time. At the Federal level, the EPA has promulgated national technology-based regulations, known as effluent limitations guidelines and standards (ELGs), for the oil and gas extraction industry, which can be found in 40 U.S. Code of Federal Regulations (CFR) Part 435. These ELGs do not apply to CBM discharges which are subject to technology based limits developed by permit writers on a case-by-case "best professional judgment" basis. The Onshore subcategory of the oil and gas, ELGs 40 CFR 125.3, Subpart C, prohibits the discharge of wastewater pollutants to waters of the United States from onshore oil and gas extraction facilities, with one exception in the arid west as discussed below. This "zero-discharge standard" means that, unless the exception applies, oil and gas wastewater pollutants cannot be discharged directly to waters of the United States. Operators have met this requirement through underground injection, reuse, or transfer of wastewater to POTWs and/or CWTs. The EPA finalized a rule in June 2016 that would prohibit operators from sending wastewater from unconventional oil and gas extraction to POTWs. Operators can continue to send wastewater to CWTs, which are subject to regulation under a separate set of ELGs in 40 CFR Part 437.

In addition, Subpart E of the oil and gas ELGs establishes an exception to the zero discharge standard west of the 98th meridian (the arid western portion of the continental United States), allowing discharges of produced water from onshore oil and gas extraction facilities to waters of the United States if the produced water has a use in agriculture or wildlife propagation when discharged into navigable waters. The term "use in agricultural or wildlife propagation" means that: (1) the produced water is of good enough quality to be used for wildlife or livestock watering or other agricultural uses; and (2) the produced water is actually put to

(*Text Box 8-2 is continued on the following page.)*

Text Box 8-2 (continued). Regulations Affecting Wastewater Management.

such use during periods of discharge (40 CFR 135.51(c)). Produced water discharged under this exception must not exceed an oil and grease concentration of 35 milligrams per liter (mg/L). Subpart E does not allow for discharge from sources other than produced water (i.e., drilling muds, drill cuttings, produced sands) to waters of the United States.

In addition to the technology-based limitations discussed above, the Clean Water Act (CWA) and the EPA's implementing regulations also require that permits include more stringent limits as necessary to meet applicable water quality standards. CWA Section 301(b)(1)(C); 40 CFR 122.44(d)(1).

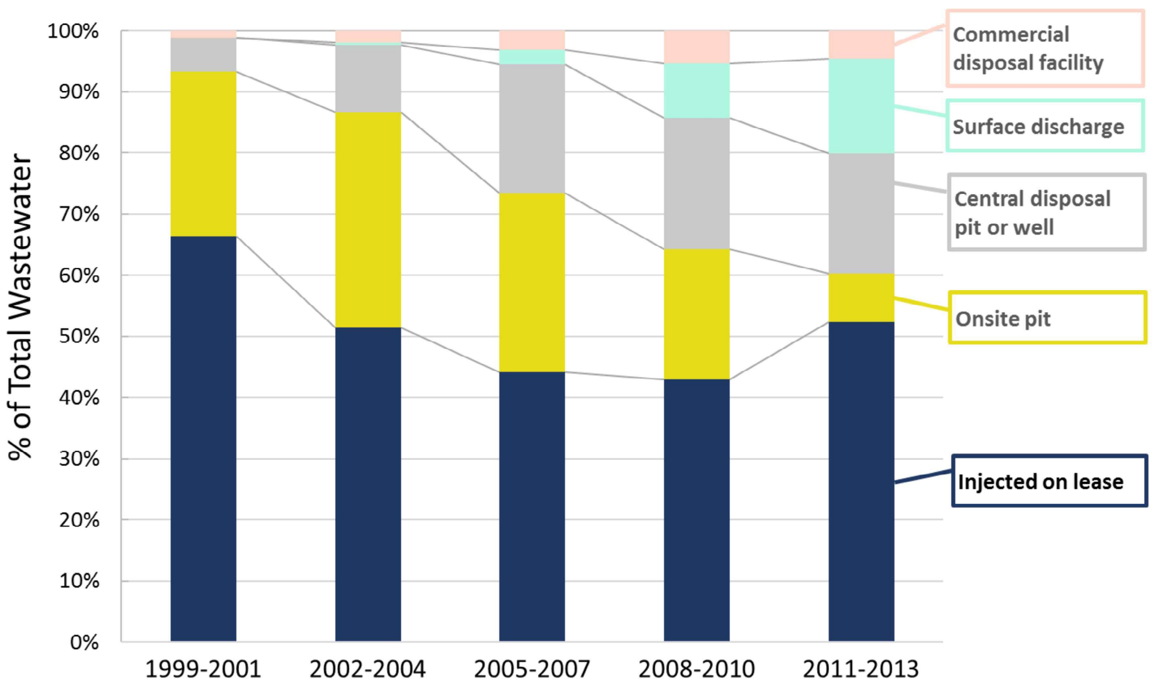

Figure 8-5. Management of wastewater in Colorado in regions where hydraulic fracturing is being performed.

See footnote for details on disposal codes.[1] Production data from Colorado Oil and Gas Conservation Commission (COGCC, 2015).

The following sections provide an overview of hydraulic fracturing wastewater management methods, with some discussion of the geographic and temporal variations in practices and their impacts on drinking water resources. In addition to currently used treatment and disposal methods, this section also briefly describes past treatment of hydraulic fracturing wastewater at

[1] Codes for wastewater disposal from COGCC are described by Veil (2015) as follows:

- Commercial disposal facility: water sent to commercial pits.
- On-site pit: most water evaporates, or excess water is hauled to disposal wells.
- Central disposal pit: Central facilities owned by a single producer to handle water from multiple wells (some recycled, much is injected).
- Injected on lease: Injected into wells, roughly half for enhanced recovery.
- Surface discharge: water is either fresh or treated to acceptable standards and discharged to surface water.

POTWs. More in-depth descriptions of treatment technologies applicable to hydraulic fracturing wastewater are available in Appendix F.

8.4.1 Underground Injection

Oil- and gas-related wastewater may be disposed of via Class II injection wells (disposal wells are referred to as Class IID whereas enhanced recovery wells are referred to as Class IIR) regulated by the UIC Program under the SDWA.[1] Nationwide, injection wells receive a large percentage of wastewater from the oil and gas industry, including wastewater associated with hydraulic fracturing. Veil (2015) estimates that in 2012, U.S. oil and gas production from onshore wells generated over 863 billion gal (20.56 billion bbls or 3.27 trillion L) of produced water, and of that volume, information on management was available for 97%. The study estimated that about 93% was injected into Class II wells, with about 47% injected into Class IID wells and 46% injected into Class IIR wells.[2]

The above national estimates are for the oil and gas industry as a whole. A good national estimate of the amount of hydraulic fracturing wastewater injected into Class II wells is difficult to develop due to lack of available information and data on injection of hydraulic fracturing wastewater. Management of hydraulic fracturing wastewater is not well tracked or made publicly available in many states (Pennsylvania being a notable exception). The local availability of Class IID wells along with generally low reuse rates, however, are consistent with Class IID wells being a primary means of wastewater management in many areas with hydraulic fracturing activity.

According to recently released data from 2012 and 2013, there are about 26,400 active Class IID wells in the United States, with more than 65% of these located in Texas, Oklahoma, and Kansas (Table 8-4). In Pennsylvania, on the other hand, there are currently nine operating disposal wells, and only three of these are commercially operated wells (at one facility) (SAFER PA, 2015). The location and number of Class IID wells is in part determined by geology (including depth and permeability of geologic formations appropriate for injection), permitting, and historical demand for disposal of oil and gas wastewater. The large Class IID well capacity in Texas, for example, is consistent with the availability of formations with suitable geology and the demand for wastewater disposal associated with a mature and active oil and gas industry. In contrast, injection plays a relatively small role in Marcellus Shale wastewater management in Pennsylvania—about 10% in 2013 and the first half of 2014 (PA DEP, 2015a).

[1] States may be given federal approval to run a UIC program under SDWA. UIC Class II wells include those used for disposal (Class IID), enhanced oil recovery (Class IIR), and hydrocarbon storage (Class IIH).

[2] Because some states surveyed by Veil (2015) do not distinguish between volumes injected for disposal versus enhanced recovery, assumptions and analyses were used in the study to estimate the two types of injection in some states.

Table 8-4. Distribution of active Class IID wells across the United States.
Data are primarily from 2012 and 2013. Source: U.S. EPA (2016d).

Geographic region (from the EIA)	State	Number of active disposal wells[a]	Average disposal rate per well (gpd/well)[b]	State disposal rate (MGD)
Alaska	Alaska	45	182,000	8.2
East	Illinois	1,054	—[c]	—[c]
	Michigan	772	16,200	13
	Florida	14	246,000	3.4
	Indiana	208	7,950	1.7
	Ohio	190	8,570	1.6
	West Virginia	64	6,970	0.45
	Kentucky	58	4,650	0.27
	Virginia	12	17,500	0.21
	Pennsylvania	9	6,380	0.057
	New York	10[d]	33.7	0.00034
Gulf Coast/Southwest	Texas	7,876	52,100	410
	Louisiana	2,448	40,300	99
	New Mexico	736	48,600	36
	Mississippi	499	24,200	12
	Alabama	85	53,300	4.5
Mid-Continent	Kansas	5,516	25,600	140
	Oklahoma	3,837	35,900	140
	Arkansas	640[e]	25,400	16
	Nebraska	113	19,100	2.2
	Missouri	11	2,270	0.025
	Iowa	3	—[c]	—[c]
Northern Great Plains	North Dakota	395	53,300	21
	Montana	199	32,700	6.5
	South Dakota	15	17,400	0.26

Geographic region (from the EIA)	State	Number of active disposal wells[a]	Average disposal rate per well (gpd/well)[b]	State disposal rate (MGD)
Rocky Mountains	Wyoming	335	107,000	36
	Colorado	292	48,800	14
	Utah	118	83,400	9.8
West Coast	California	826	86,800	72
	Nevada	10	54,600	0.55
	Oregon	9	—[c]	—[c]
	Washington	1	—[c]	—[c]
Total		**26,400**	**41,300**	**1,050**

Abbreviations: gpd—gal per day; MGD—million gal per day.

[a] Number of active disposal wells is based primarily on data from 2012 to 2013.

[b] Typical injection volumes per well are based on historical annual volumes for injection for disposal divided by the number of active disposal wells during the same year (primarily 2012 to 2013 data).

[c] Disposal rates and volumes are unknown.

[d] These wells are not currently permitted to accept extraction wastewater from production in unconventional reservoirs.

[e] Only 24 of the 640 active disposal wells in Arkansas are in the northern half of the state, close to the Fayetteville Shale.

The decision to inject hydraulic fracturing wastewater into Class IID wells depends in part on cost, including transportation costs. Therefore, the distance between the production well and a disposal well is an important consideration. For oil and gas producers, underground injection is a low cost management strategy unless significant trucking is needed to transport the wastewater to a disposal well (U.S. GAO, 2012).

Evaluation of documented or potential impacts on drinking water resources associated with disposal at Class IID injection wells is outside of the scope of this assessment. However, disposal wells play a significant role in the overall management of hydraulic fracturing water nationwide, and their availability and capacity are integral factors in determining which wastewater management strategies are used by operators in a given region. Should the feasibility of managing wastewater via underground injection become limited or less economically advantageous, operators will need to adjust their wastewater management programs. They may evaluate and implement other local practices such as sending wastewater to a CWT for treatment and discharge or reuse.

Recent events and studies, for example, have documented a link between wastewater injection and seismic activity in some locations in several states, including Oklahoma, Colorado, New Mexico, Arkansas, and Ohio (Weingarten et al., 2015; Wong et al., 2015). The Oklahoma Geological Survey (Andrews and Holland, 2015) "considers it very likely that the majority of recent earthquakes, particularly those in central and north-central Oklahoma, are triggered by the injection of produced water in disposal wells." Walsh and Zoback (2015) correlated wastewater injection from production wells (including hydraulically fractured wells) into Oklahoma's Arbuckle formation to the steep increase in seismic events observed in that state. Farther west, in the Raton Basin of southern Colorado and northern New Mexico, Rubinstein et al. (2014) presented several lines of evidence linking injection well disposal of CBM produced water to seismic events. Horton (2012) attributed a swarm of earthquakes in Northern Arkansas to hydraulic fracturing wastewater injection, and in a study evaluating multiple states in the mid-continent region, Weingarten et al. (2015) demonstrated a relationship between Class II wells (including both Class IID and Class IIR wells) and seismicity.

The local availability of Class IID wells and the capacity to accept large volumes of wastewater could be affected by these recent findings concerning seismic activity associated with injection (U.S. EPA, 2014c). Between 2011 and 2016, some state UIC programs modified their Class II wastewater injection regulations and permitting requirements. At least eight states (Arkansas, Colorado, Illinois, Kansas, Ohio, Oklahoma, Texas, and West Virginia) consider an assessment of seismicity in their Class II programs and have regulatory provisions for banning or shutting injection wells and/or modifying injection volumes and pressures if evidence indicates that a well is near a fault and/or is contributing to seismic activity.

As an example, Oklahoma has recently taken steps to reduce the risk of induced seismicity by implementing a regional strategy intended to reduce wastewater injection in certain regions (OCC OGCD, 2016). These actions affect over 10,000 square miles and 600 wastewater injection wells in western and central Oklahoma. The measures are intended to reduce wastewater injection in the area by 40% below 2014 totals, which will affect wastewater management and disposal practices across this large area.[1]

In terms of potential impacts on drinking water resources, Class IID facilities are subject to the same general considerations regarding wastewater storage and handling as other wastewater management sites and facilities (e.g., CWTs). Changes in surface water or groundwater quality due to general wastewater handling at these facilities may be another factor affecting wastewater management practices in some locations or regions. For example, Kell (2011) identified eight groundwater contamination incidents in Texas between 1993 and 2008 due to water releases from storage facilities associated with Class II well sites. A recent study by the United States Geological Survey documented impacts on surface water from hydraulic fracturing wastewater at a Class II disposal well site in central West Virginia (Akob et al., 2016). Water samples collected downstream from the facility were indicative of wastewater from hydraulic fracturing operations handled at the site. The authors documented elevated specific conductance and elevated TDS, sodium, chloride, barium, bromide, strontium, and lithium concentrations, and different strontium isotope ratios compared to those found in upstream, background waters. The study concluded that activities at the wastewater facility have affected water quality in a nearby stream. The pathways for the movement of wastewater into the local stream include several possibilities (e.g., leaks from storage ponds and tanks, transportation activities, previous site history).

[1] For additional information on strategies and initiatives regarding wastewater injection and inducted seismicity, see the following: KDHE (2014), States First Initiative (2014), and U.S. EPA (2014c).

8.4.2 Publicly Owned Treatment Works

POTWs are designed to treat local municipal wastewater and indirect discharges from industrial users. POTWs are also used to treat wastewater and other wastes from oil and gas operations in some eastern states. Although this is not a common method of treatment for oil and gas wastewaters in the United States, the scarcity of injection wells for waste disposal in Pennsylvania drove the need for disposal alternatives (Wilson and Vanbriesen, 2012). When development of the Marcellus Shale began, POTWs were used to treat wastewater originating from these oil and gas wells (Kappel et al., 2013; Soeder and Kappel, 2009). However, elevated concentrations of constituents in wastewater from the Marcellus region (halides, heavy metals, organic compounds, radionuclides, and salts) can pass through the treatment processes commonly used in POTWs and be discharged to receiving waters (Cusick, 2013; Kappel, 2013; Lutz et al., 2013; Schmidt, 2013). In addition, sudden, extreme salt fluctuations can disturb POTW biological treatment processes (Linarić et al., 2013; Lefebvre and Moletta, 2006).

The annual reported volume of oil and gas wastewater treated at POTWs in the Marcellus Shale region peaked in 2008 and has since declined significantly (Figure 8-6). As discussed in Text Box 8-1, this was in response to an April 2011 request from PA DEP asking operators to cease sending Marcellus Shale wastewater to 15 POTWs and CWTs that were exempt from the 2010 TDS regulation (Rahm et al., 2013). Although operators complied with the request in May 2011, non-Marcellus oil and gas produced water continued to be processed at these facilities (Ferrar et al., 2013; Lutz et al., 2013; Wilson and Vanbriesen, 2012).[1] In August 2016, the EPA finalized pretreatment standards prohibiting discharges of unconventional wastewater pollutants to POTWs (U.S. EPA, 2016d).

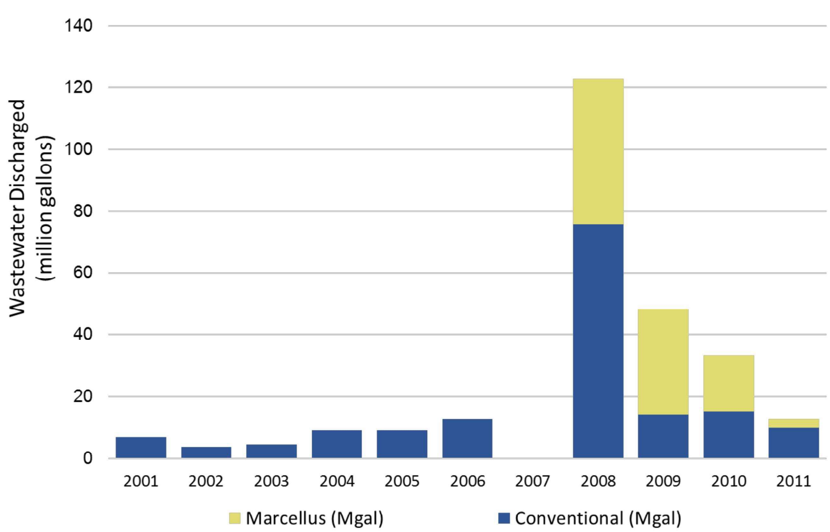

Figure 8-6. Oil and gas wastewater volumes discharged to POTWs from 2001-2011 in the Marcellus Shale. ("Conventional" is indicated by the authors as non-Marcellus wells and described as vertically drilled to shallower depths in more porous formations.)
Due to an unrecoverable data loss at the PA DEP, records for 2007 were not available. Source: Lutz et al. (2013).

[1] POTWs in Pennsylvania have likely been accepting waste considered conventional by Pennsylvania but unconventional by others based on the EPA's broader definition (Text Box 8-1).

8.4.3 Centralized Waste Treatment Facilities

A CWT facility is generally defined as one that accepts industrial materials (hazardous or non-hazardous, solid, or liquid) generated at another facility (off-site) for treatment or recovery (EPA, 2000). (Wastewater may also be treated at on-site mobile or semi-mobile facilities; see Appendix F for additional information.) The decision to treat hydraulic fracturing wastewater at a CWT and the level of treatment used depends upon several factors, such as a lack of proximity to Class II disposal wells; whether the wastewater might be reused for additional hydraulic fracturing jobs; the water quality needed if it will be reused; whether the treated wastewater can be discharged under the Subpart E agricultural and wildlife use exception in the arid west; and the water quality needed if it will be discharged to the waters of the United States. As a group, CWTs that accept oil and gas wastewater offer a wide variety of treatment capabilities and configurations (Text Box 8-3 and Appendix F).

Text Box 8-3. Wastewater Treatment Processes.

The constituents prevalent in hydraulic fracturing wastewater include TDS, TSS, radionuclides, organic compounds, and metals (Section 8.3 and Chapter 7). If the ultimate disposal or use of the wastewater necessitates treatment, a variety of technologies can be employed to remove or reduce these constituent concentrations.

The most basic treatment needed for oil and gas wastewaters, including those from hydraulic fracturing operations, is separation to remove TSS and oil and grease. This is accomplished through separation technologies including settling, skimming, hydrocyclones, dissolved air or induced gas flotation, media filtration, or biological aerated filters (Igunnu and Chen, 2014; Duraisamy et al., 2013; Barrett, 2010; Shammas, 2010).

Other treatment processes that may be used include media filtration after chemical precipitation for hardness and metals (Boschee, 2014); adsorption technologies for organics, heavy metals, and some anions (Igunnu and Chen, 2014); a variety of membrane processes (microfiltration, ultrafiltration, nanofiltration, reverse osmosis (RO)); and distillation technologies for metals and organics (Drewes et al., 2009).

Advanced processes, such as RO, or distillation methods, such as mechanical vapor recompression (MVR), are needed if the system requires significant reduction in TDS (Drewes et al., 2009; LEau LLC, 2008; Hamieh and Beckman, 2006). However, RO is typically only capable of treating TDS concentrations less than 35,000 mg/L (Shaffer et al., 2013), whereas distillation can effectively treat higher TDS waters (Hayes et al., 2014; Drewes et al., 2009). Extremely high TDS waters may require a series of advanced treatment processes, which can be very costly.

An emerging technology in hydraulic fracturing wastewater treatment is electrocoagulation, which has been used in mobile treatment systems to remove organics, TSS, and metals (Halliburton, 2014; Igunnu and Chen, 2014).

Appendix F provides more in-depth descriptions of technologies used to treat for hydraulic fracturing wastewaters and the constituents they remove. Also, Appendix Table F-4 provides an overview of influent and effluent results and removal percentages for constituents of concern at oil and gas treatment facilities reported in the literature (both conventional and unconventional) and the specific technology(ies) used to remove them. Section 8.4.7 discusses solid and liquid residuals, including treatment-related wastes.

The treated effluent from a CWT can be reused in hydraulic fracturing operations (also called zero-discharge), discharged directly to a receiving water under a National Pollutant Discharge Elimination System (NPDES) permit, discharged indirectly to a POTW, or a combination of these. Some CWTs may be configured so that they can either (1) partially treat the waste stream to suit the needs of operators who reuse it or (2) use more advanced treatment (i.e., TDS removal) if the treated wastewater will be discharged. Generally, the former option is less costly for the CWT, and some facilities that have permits to discharge do not do so continuously, opting to direct as much of the wastewater as possible for reuse. There are also CWTs permitted to discharge that do not have TDS removal capabilities. However, these facilities must still meet TDS discharge limits specified by their state. Appendix F contains additional information on treatment configurations, including examples of processes at several facilities treating oil and gas wastewater.

Facilities discharging treated wastewater to waters of the United States or POTWs are regulated under the Clean Water Act (CWA). For zero-discharge facilities, some states, including Pennsylvania and Texas, have adopted regulations to control permitting of these facilities or to encourage treatment and reuse. The PA DEP issues permits that allow zero-discharge CWTs to treat and release water back to oil and gas industries for reuse (see the Eureka Resources Facility in Williamsport, PA listed in Appendix Table F-6 as an example of a zero-discharge facility).[1]

In developing this assessment, we looked at NPDES permit information for several CWTs in the eastern United States treating wastewater from the Marcellus region and one near the Fayetteville Shale in Arkansas. The facilities include those with and without TDS removal capabilities, and some are undergoing upgrades to implement TDS removal. Some of the permits reviewed for this assessment are current, and others are expired and may be in the process of renewal. The permits require monitoring (with or without limits) for a range of constituents that may include chloride, TDS, TSS, total strontium, total barium, oil and grease, heavy metals, 5-day biological oxygen demand (BOD5), and a range of organic compounds (e.g., phenol, cresol, BTEX, phthalates), with the specific constituents varying by permit. Sample types for the facilities are generally 24-hour composites. The newer permits set limits for several important constituents such as chloride, TDS, TSS, total barium, total strontium, oil and grease, and a number of heavy metals. Bromide is generally either not included or is required to be reported but with no limit specified. However, limits on TDS will reduce bromide concentrations. Some permits require monitoring for total radium, uranium, and gross alpha, but no limits are specified. Note that these facilities do not necessarily discharge consistently because treated wastewater can be sent for reuse.

Although there are CWTs serving hydraulic fracturing operations throughout the country, the majority serve Marcellus Shale operations in Pennsylvania (Boschee, 2014). Of the 74 CWT facilities identified by the EPA (U.S. EPA, 2016d) as having accepted or having the ability to accept hydraulic fracturing wastewater (not counting facilities treating CBM wastewater), 40 are located in Pennsylvania (Table 8-5). Most are zero-discharge facilities, and many do not have treatment processes for TDS removal. Although several Pennsylvania facilities are permitted to discharge, Wunz (2015) found few that currently discharge (two CWTs in Pennsylvania, one in West Virginia,

[1] The facility is also permitted for indirect discharge to the Williamsport Sewer Authority.

Table 8-5. Number, by state, of CWT facilities that have accepted or plan to accept wastewater from unconventional oil and gas activities.

Source: U.S. EPA (2016d).

State	Unconventional formation(s) served	Zero discharge CWT facilities[a]		CWT facilities that discharge to a surface water or POTW[a]		CWT facilities with multiple discharge options[a]		Total known facilities
		Non-TDS removal	TDS removal	Non-TDS removal	TDS removal	Non-TDS removal	TDS removal	
AR	Fayetteville	2	0	0	0	0	1	**3**
CO	Niobrara, Piceance Basin	3 (1)	0	0	0	0	0	**3**
ND	Bakken	0	1 (1)	0	0	0	0	**1**
OH	Utica, Marcellus	10 (7)	0	1	0	0	0	**11**
OK	Woodford	2	0	0	0	0	0	**2**
PA	Utica, Marcellus	22	7(3)	8	0	0	3 (1)	**40**
TX	Eagle Ford, Barnett, Granite Wash	1	3	0	0	0	0	**4**
WV	Marcellus, Utica	4 (2)	0	0	0	1	1	**6**
WY	Mesaverde and Lance	0	2	0	0	0	2	**4**
Total		**44**	**13**	**9**	**0**	**1**	**7**	**74**

[a] Information is current as of 2014; it is possible that since 2014, some listed CWT facilities have closed and/or new CWT facilities not listed have begun operation. The number of facilities includes those that have not yet opened but are under construction, pending permit approval, or are in the planning stages. Facilities that are not accepting hydraulic fracturing wastewater but plan to accept it in the future are noted parenthetically and not included in the sum of total known facilities. Facilities handling CBM wastewater are not represented here.

and one in Ohio). According to EPA research (U.S. EPA, 2016d), the number of CWT facilities serving operators in the Marcellus and Utica Shales has increased since the mid-2000s, growing from roughly five in 2004 to over 40 in 2013. A similar trend has been noted for the Fayetteville Shale region in Arkansas, where there are fewer Class IID injection wells compared to the rest of the state (U.S. EPA, 2016d).

In other regions, a small number of newer facilities have emerged in the last several years, most often with TDS removal capabilities. In Texas, for example, two zero-discharge facilities with TDS removal capabilities are available to treat wastewater from the Eagle Ford Shale (beginning in 2011 and 2013), and one zero-discharge facility with TDS removal is located in the Barnett Shale region (operational since 2008). In Wyoming, there are four facilities in the region of the Mesaverde/Lance formations that started operating between 2006 and 2012. Two are zero-discharge facilities, and two have multiple discharge options; all are capable of TDS removal (U.S. EPA, 2016d).

Few states maintain a comprehensive list of CWT facilities, and the count provided by the EPA (U.S. EPA, 2016d) includes facilities that do not currently but plan to accept wastewater from unconventional formations. Therefore, the data in Table 8-5 do not precisely reflect the number of facilities currently handling hydraulic fracturing wastewaters. Other sources indicate either use of, or interest in, development of treatment facilities in other regions such as the Barnett Shale region (Hayes and Severin, 2012b), the Fayetteville (Veil, 2011), and other areas in Texas and Wyoming (Boschee, 2014, 2012). In addition, news releases and company announcements indicate that other wastewater treatment facilities are being planned (Greenhunter, 2014; Geiver, 2013; Purestream, 2013; Alanco, 2012; Sionix, 2011).

Use of specific types of CWTs has and will continue to shift due to drivers such as availability and cost of other disposal options (e.g., disposal wells), operator demand for reuse and the associated quality needed, developments in treatment, treatment costs, and regulatory changes. Practices in Pennsylvania over the last several years provide such an example. Between 2010 and 2013, the percentage of Marcellus wastewater treated at CWTs dropped from 52% to 20% (Figure 8-4), and the percentage of wastewater reused on-site rose to 65%, reflecting a shift in practice among operators. Among the percentage of the wastewater sent to CWTs, the portion sent to zero-discharge facilities for subsequent reuse rose from 10% to 65%. This is consistent with an increased emphasis on reuse in Pennsylvania. (See Section 8.4.4 for a discussion on reuse as a waste management practice.)

8.4.3.1 Relationship to Potable Surface Waters

Figure 8-7 shows the relationship between Pennsylvania potable water supplies and the CWTs that lie in their upstream watersheds. These surface waters, including streams, rivers, and waterbodies (e.g., lakes and reservoirs) have been evaluated by the PA DEP for attainment of a designated use of potable water supply as per the CWA Section 305(b) reporting and Section 303(d) listing. Ninety-four percent of the waterbodies and 98% of the streams and rivers were attaining their designated use in 2016. These stream segments may or may not currently have intakes for drinking water treatment plants. The map also shows the locations and types of CWTs that either currently accept unconventional oil and gas wastewater (as defined by PA DEP) or have accepted such wastewater

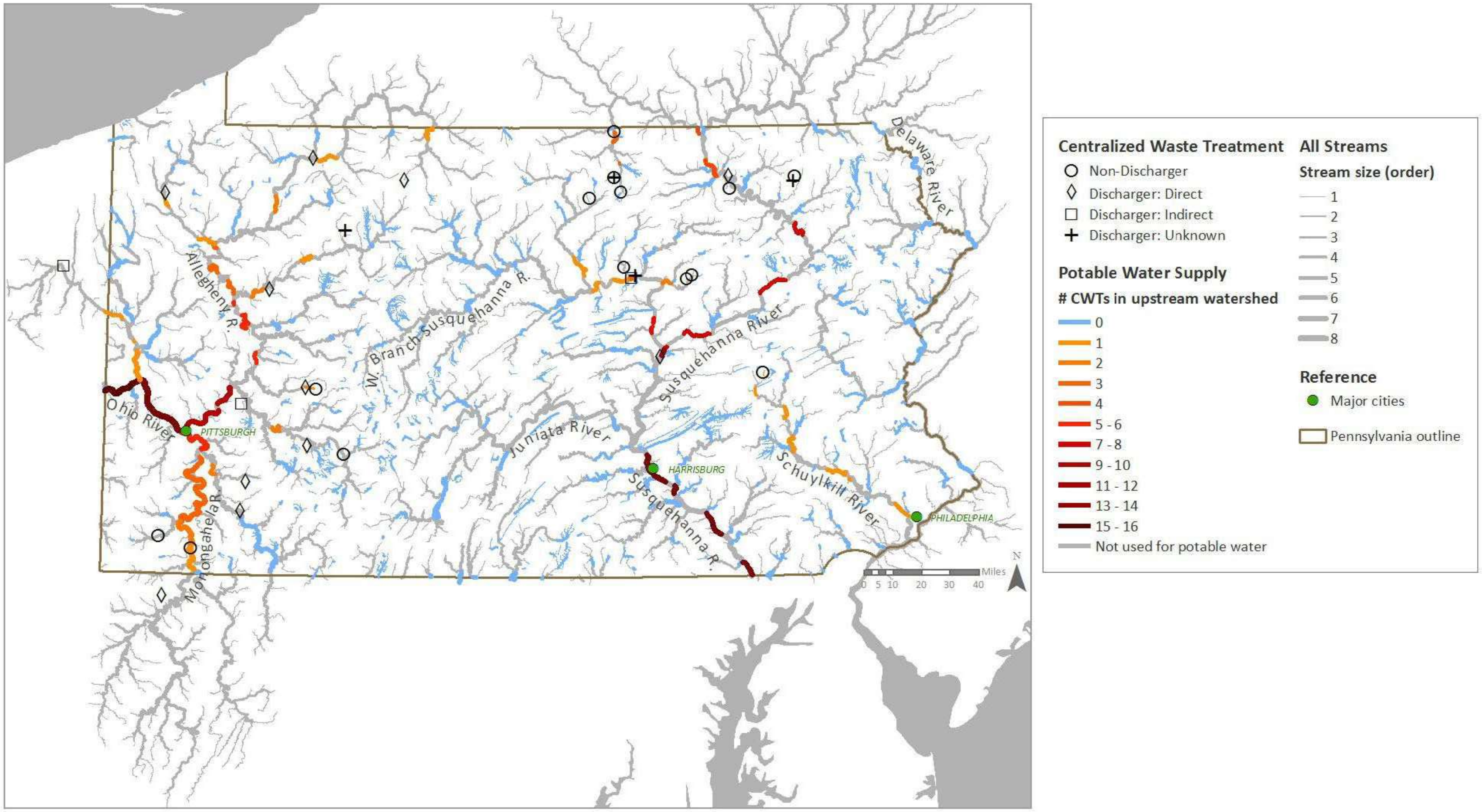

Figure 8-7. Map showing Pennsylvania surface water designated as potable water supplies and upstream CWTs.
Surface waters are colored orange to red to indicate the number of CWTs located in the upstream watershed. Blue surface waters have no upstream CWTs, and light gray lines show those not designated as potable water supplies. Symbols show the locations of CWTs that currently accept or have accepted unconventional oil and gas wastewater. Data sources: U.S. EPA (2016d), U.S. EPA (2016f), and PA DEP (2016b).

within approximately the last five years.[1] CWTs represented include both dischargers (direct and indirect) as well as zero-discharge facilities. For some facilities, we were not able to determine if the facility was zero-discharge or if it has a NPDES permit. The surface waters have been color-coded to indicate the number of CWTs that are located upstream. Darker red indicates more CWTs located in the upstream watershed, while blue indicates no upstream CWTs. Softer grey lines show portions of the stream network not designated for potable water supply. The thickness of the line indicates the size of the stream or river, categorized by the "stream order" designation.

The map provides a general illustration of how CWTs are situated within catchments in Pennsylvania, showing their spatial and general hydrologic relationships to streams that can serve as potable water supplies. The map shows that a given stream or waterbody may have a number of CWTs upstream, potentially contributing to combined impacts on surface water if there are spills or inadequately treated discharges. Note that the upstream catchment areas are large for the major rivers. Therefore, some rivers, such as the Ohio or Susquehanna, have as many as 15 or 16 upstream CWTs, although most are located far away. The map does not represent the effects of dilution on either discharges or spills; such an evaluation would necessitate currently unavailable data required to do a complete analysis of water quality. Note that many of the CWTs are zero-discharge facilities, and those that are permitted to discharge may do so intermittently. However, the storage and handling of wastewater at CWTs could impact nearby surface water through leaks and spills.

To more completely place these facilities in a watershed context, other types of discharges that could be occurring upstream should be taken into consideration. Impacts from hydraulic fracturing wastewater may be more problematic if there are additional pollutant sources within the watershed, increasing the cumulative effects of discharges and spills. For example, an EPA source apportionment study (U.S. EPA, 2015o) evaluated the relative contributions of bromide, chloride, nitrate, and sulfate from CWTs primarily treating hydraulic fracturing wastewater to the Allegheny River Basin and to two downstream public water system intakes. The study considered that the Allegheny River and its tributaries also receive runoff and discharges from an array of sources that include acid mine drainage and mining operations, coal-fired electric power stations, industrial wastewater treatment plants, and POTWs. It was concluded that CWTs treating oil and gas wastewater and coal-fired power plants with flue gas desulfurization were the primary contributors of bromide and chloride at the intakes (see Section 8.5.1 for further discussion), while nitrate and sulfate contributions were from POTWs and Acid Mine Drainage (U.S. EPA, 2015o).

8.4.3.2 Potential Impacts from CWTs

The potential impacts of managing hydraulic fracturing wastewater at CWTs depend on whether the CWT adequately treats for constituents of concern prior to discharge to surface water or a POTW, and whether treatment residuals are managed appropriately. Historically, CWTs have not

[1] The list of CWTs used to develop this map is based on best available data, including information in the technical development document supporting the new EPA unconventional oil and gas effluent limitation guidelines (U.S. EPA, 2016d) as well as data from PA DEP waste records. This information was supplemented with other publicly available descriptions of the facilities. The information may, however, not be complete, and the symbols may not definitively reflect the discharge status of a facility.

included processes to treat for constituents that are difficult to remove, such as the high concentrations of TDS found in wastewater from unconventional reservoirs. As a result, impacts on drinking water resources have included increased suspended solids and chloride concentrations downstream of discharging facilities that were accepting hydraulic fracturing wastewater (Olmstead et al., 2013) and elevated bromide concentrations and radium concentrations in CWT effluent (Warner et al., 2013a); see Sections 8.5.1 and 8.5.2. In addition, spills and leaks can occur in pits or impoundments associated with the storage of treated wastewater at CWTs (impacts related to spills and leaks from pits and impoundments are discussed in Section 8.4.5). Wastewater being transported by truck or pipeline to and from a CWT can also present a vulnerability for spills or leaks (Easton, 2014) (Chapter 7).

While selection of appropriate treatment processes is critical for CWTs that discharge to surface waters, there are also two important issues related to completeness of treatment that can have an impact. First, there may be unknown constituents in the wastewater. The effectiveness of treatment cannot be evaluated for constituents for which the wastewater has not been tested. This makes it challenging to know the degree to which effluent from a CWT is protective of public health. Second, even an efficient treatment process may not be able to reduce the concentrations of some constituents to levels that allow for discharge to a drinking water resource if influent concentrations are so high that they exceed the capabilities of the treatment technology(ies) to meet those discharge limits. For example, a facility described by Kennedy/Jenks Consultants (2002) removed a high percentage of boron (88%), but the effluent concentration of 1.9 mg/L (average influent concentration of 16.5 mg/L) was not low enough to meet California's action level of 1 mg/L. Thus, the influent concentration must be considered together with removal efficiency to determine whether the effluent quality will meet the requirements dictated by end use or by regulations.

Relatively few studies describe the ability of individual treatment processes to remove constituents from hydraulic fracturing wastewater. For this assessment, simple estimated effluent concentrations were calculated for several combinations of unit treatment processes, wastewater constituents, and influent concentrations (details are given in Appendix Table F-3). The purpose of the analysis was to illustrate the relative capabilities of a number of treatment processes and not to represent a complete treatment system. As an example, the estimates suggest that if wastewater contains radium with a concentration in the thousands of pCi/L, a 95% removal rate with chemical precipitation may result in an effluent that exceeds 100 pCi/L. Treatment of the same wastewater via distillation or reverse osmosis could result in effluent concentrations in the tens of pCi/L. This analysis suggests that attention should be paid to the capabilities of a planned treatment system for the full range of anticipated wastewater compositions.

To gain a better understanding of impacts, the USGS has conducted sampling for a wide array of water quality parameters in surface water and groundwater in the Monongahela River Basin in West Virginia to establish baseline water-quality conditions (Chambers et al., 2014). Future water quality sampling can be compared to this baseline to assess impacts from hydraulic fracturing activities. To address past impacts, Pennsylvania, having experienced water quality impacts on receiving streams due to discharges of high-TDS effluent modified their regulations to address these issues by setting water quality standards for CWT dischargers (Mauter and Palmer, 2014; Shaffer et al., 2013). (See Text Box 8-1.)

8.4.4 Wastewater Reuse for Hydraulic Fracturing

The reuse of hydraulic fracturing wastewater for subsequent hydraulic fracturing operations has increased in some regions of the country in recent years (Boschee, 2014, 2012; Gregory et al., 2011; Rassenfoss, 2011).[1] This practice is driven by factors that include cost (including treatment costs), the lack of availability of other management options (e.g., Class II disposal wells), and changes to state regulations (Boschee, 2014; Shaffer et al., 2013). Wastewater quality is a consideration; some constituents pose challenges for reuse and may necessitate treatment. For example, high concentrations of barium and sulfate can lead to scaling, and the presence of some constituents in wastewater can hinder crosslinking (Akob et al., 2016; Boschee, 2014). Hydraulic fracturing fluid formulations that can use high TDS waters (e.g., as high as 150,000 mg/L to over 300,000 mg/L) facilitate reuse with minimal treatment (Boschee, 2014; Mauter and Palmer, 2014). See Chapter 5 for more information regarding the chemical composition of hydraulic fracturing fluids and Appendix F for more discussion of considerations for reuse.

Reuse can be accomplished by blending either untreated or minimally treated hydraulic fracturing wastewater with fresh water to lower the TDS content (Boschee, 2014). Wastewater may be reused at a site with multiple wells, eliminating the need for transport to a CWT (Lester et al., 2015; Easton, 2014). Alternatively, wastewater can be treated at a CWT and then taken by operators for mixing with other water sources for reuse (Easton, 2014). Flowback may be preferable to later-stage produced water for reuse because of its lower TDS concentration. Also, it is typically generated in larger quantities from a single location as opposed to water produced later on, which is generated in smaller volumes over time from many different locations (Barbot et al., 2013; Maloney and Yoxtheimer, 2012). Reuse can reduce the costs associated with water acquisition and produced water management. Such economic and logistical benefits can be expected to inform ongoing wastewater management decisions.

Costs can be the most significant driver for reuse. For example, the costs of transporting wastewater from the generating well to the treatment facility and then to the new well can be weighed against the costs for transport to alternative locations (e.g., a disposal well). Trucking large quantities of water can be relatively expensive—from \$0.01 to \$0.19 per gallon (\$0.50 to \$8.00 per bbl)—rendering on-site treatment technologies and reuse economically competitive in some settings (Dahm and Chapman, 2014; Guerra et al., 2011). Reuse rates may also be driven by wastewater production rates compared to the demand for reuse, with both production and demand increasing in a region if more wells go into production or decreasing as plays mature (Lutz et al., 2013; Hayes and Severin, 2012b; Slutz et al., 2012). Other logistics to consider include proximity of the water sources for aggregation and sequencing of completion schedules (Mauter and Palmer,

[1] Reused hydraulic fracturing wastewater is discussed in Chapter 4 of this report (Water Acquisition) as well as in this chapter, though in a different context. The wastewater reuse rate described in this chapter is the amount or percentage of generated hydraulic fracturing wastewater that is managed through use in subsequent hydraulic fracturing operations. In contrast, Chapter 4 discusses reused wastewater as a source water and as one part of the base fluid for new fracturing fluid.

2014). A small survey by Mauter and Palmer (2014) indicates that the scheduling of well completions is complex, requiring optimization of labor, contractual issues, equipment usage, and water storage capacity among other factors. Boschee (2014) notes that in the Permian Basin, older conventional wells are linked by pipelines to a central disposal facility, facilitating movement of treated water to areas where it is needed for reuse. Companies drilling fewer wells or located in more remote areas may find reuse difficult because of challenges in consolidating wastewater from their wells or accessing wastewater from centralized facilities.

Regulations may also encourage reuse. For example, in 2013, the Texas Railroad Commission adopted rules eliminating the need for a permit when operators reuse on their own lease or transfer the fluids to another operator for reuse (Rushton and Castaneda, 2014). Any information on wastewater management practices in Texas that becomes available for the years after 2013 will allow evaluation of whether reuse has in fact increased.

A summary of reuse practices throughout the United States is hampered by the limited amount of data available for many regions of the country. However, current data indicate that extensive reuse takes place in the Marcellus region. Several studies using data from PA DEP data show that total reuse rates of oil and gas wastewater in Pennsylvania have risen over the last several years to between 85 and 90% (Table 8-6). This includes wastewater sent to CWTs to treat for reuse as well as reuse at the well sites without transfer to a CWT (labeled as "Reuse HF" in Figure 8-4). In particular, reuse of Marcellus wastewater at well sites in Pennsylvania has risen from about 8% in the second half of 2010 to nearly 70% in the first half of 2014 (PA DEP, 2015a). Schmid and Yoxtheimer (2015) report more recent data stating that in 2014, approximately 85% of Marcellus hydraulic fracturing wastewater was reused. Of that amount, 78% occurred on-site, and 22% was via CWTs.

Table 8-6. Estimated percentages of reuse of hydraulic fracturing wastewater.

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
East Coast[a]								
Marcellus, PA	Rahm et al. (2013)	9	8	25 – 48	67 – 80			
Marcellus, PA	Ma et al. (2014)		15 - 20				90	
Marcellus, PA	Shaffer et al. (2013)					90		
Marcellus, PA	Schmid and Yoxtheimer (2015)							85
Marcellus, PA	Hansen et al. (2013)	9	6	20	56			
Marcellus, PA	Maloney and Yoxtheimer (2012)				71.6			
Marcellus, PA	Tiemann et al. (2014)				72	87		

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
Marcellus, PA	Rassenfoss (2011)			~67 (general estimate) 96 (one company)				
Marcellus, PA	Wendel (2011)			75-85	90			
Marcellus, PA	Lutz et al. (2013)	13 (prior to 2011)			56			
Marcellus, PA (SW region)	Rahm et al. (2013)	~10	~15	~25-45	~70-80			
Marcellus, PA (NE region)	Rahm et al. (2013)	0	0	~55-70	~90-100			
Marcellus, WV	Hansen et al. (2013)			88	73	65 (partial year)		
Gulf Coast and Midcontinent								
Fayetteville	Veil (2011)			20 (single company target)				
Barnett	Rahm and Riha (2014), Nicot et al. (2012)				5 (general estimate – appears to cover recent years)			
Eagle Ford	Nicot and Scanlon (2012)				0	20 (estimate based on interviews)		
East Texas	Nicot and Scanlon (2012)				5			
Haynesville	Horner et al. (2014)						0	
Haynesville	Rahm and Riha (2014)				5 (general estimate – appears to cover recent years)			

Play or basin	Source and year	2008	2009	2010	2011	2012	2013	2014
West Coast and Upper Plains								
Denver-Julesburg (Weld County), CO	Sumi (2015)					54 (flow-back only)		
Bakken	Horner et al. (2014)						0	

[a] Studies focusing on the Marcellus Shale use waste data reports from PA DEP.

Reuse in the Marcellus region is higher in the northeastern part of Pennsylvania than in the southwestern portion where easier access to Class IID wells in Ohio makes disposal by injection more feasible (Rahm et al., 2013). Outside of the Marcellus region, reuse rates are lower. Ma et al. (2014) note that only a small amount of reuse is occurring in the Barnett Shale. Reuse has not yet been pursued aggressively in New Mexico or in the Bakken (North Dakota) (Horner et al., 2014; LeBas et al., 2013). Other sources, however, indicate growing interest in reuse, as evidenced in specialized conferences (e.g., "Produced Water Reuse Initiative 2014" on produced water reuse in Rocky Mountain oil and shale gas plays), and available state-developed information on reuse (e.g., fact sheet by the Colorado Oil and Gas Conservation Commission) (Colorado Division of Water Resources et al., 2014).

If hydraulic fracturing activity slows in an area that is currently reusing wastewater, demand for the wastewater may decrease and wastewater management practices may shift. Analysis by Wunz (2015) and data in Figure 8-1 suggest a decline in wastewater production in Pennsylvania. Wunz (2015) also notes that in the future, there could be a trend of more wastewater coming from late-stage produced water and less from flowback as more wells are in the production phase and fewer wells are being fractured. If the demand drops relative to production due to fewer wells being drilled and fractured, then the "excess" produced water will need to be managed by other means. Alternatives to reuse may include increased transport to disposal wells (e.g., those in Ohio), development of more disposal wells in Pennsylvania, or advanced treatment and discharge to surface water via CWTs that have TDS removal capabilities (SAFER PA, 2015; Wunz, 2015; Silva et al., 2014a).

8.4.4.1 Potential Impacts from Reuse

For companies employing reuse as a wastewater management strategy, surface spills and leaks can occur during wastewater transport to and from a treatment facility or from storage tanks/pits located at the treatment facility or at the well site. Releases may be due to failed infrastructure such as tank or pipe ruptures, from natural disasters such as floods or earthquakes, or incidents such as overfills, improper operations, or vandalism (CCST, 2015a; NYSDEC, 2011). If the spill or leak is not contained or otherwise mitigated, these releases could reach groundwater or surface water (CCST, 2015a; NYSDEC, 2011). See Chapter 7 for more discussion on types of spills associated with hydraulic fracturing activities, including storage and transport. See Section 8.4.5 for discussion of storage pits and associated impacts on drinking water resources.

With reuse there is the potential for accumulation of dissolved solids such as salts and TENORM in the wastewater over successive reuse cycles (see Section 7.3.4.6 and Section 8.5.2 for more information about TENORM). Because wastewater is often reused with minimal treatment, constituents resulting from time spent in the subsurface remain in the wastewater and can increase during additional hydraulic fracturing. This potentially concentrated wastewater can pose a bigger issue if a breach occurs in an on-site pit or tank storing this wastewater while awaiting reuse (Section 8.4.5; Chapter 7).

The issue of concentrating contaminants during reuse has not yet been quantitatively evaluated in the literature. Also, it is not known how much this problem would be mitigated due to the dilution of wastewater when reused as new fracturing fluid. Estimates of the percentages of reused wastewater in new fracturing fluids in Pennsylvania range from about 2% in 2009 to as much as 22% in 2013 (SRBC, 2016; Schmid and Yoxtheimer, 2015) (Chapter 4). However, data from Pennsylvania's TENORM study (PA DEP, 2015b) showed radium in some hydraulic fracturing fluids, presumably from a reused wastewater component. As reused wastewater continues to accumulate contaminants, the water will eventually need to be managed, either through treatment or injection.

8.4.5 Storage and Disposal Pits and Impoundments

The use of pits and impoundments as part of a wastewater management strategy is a historic as well as current practice in the oil and gas industry. These structures are either used for temporary storage (on-site at oil and gas production wells or off-site at CWTs or disposal wells) or they are intended for permanent disposal (evaporation or percolation). There are a variety of terms to describe these structures depending upon their use (Richardson et al., 2013); "pits," "impoundments," and "reserve pits" are some of the more common terms associated with wastewater management. The terms "impoundment" or "pond" are often used to refer to large area holding structures and are also used by some states for specific applications such as holding "freshwater" for fracturing fluid formulation (Quaranta et al., 2012). Definitions and terminology are not standardized and vary from state to state (Richardson et al., 2013). For the purposes of this section, the nomenclature will defer to the term used by the original author/regulating authority.

States govern the use and permitting of pits under their jurisdiction. Regulations vary from state to state regarding the circumstances in which pits can be used (e.g., chemical composition of the fluid), how they should be constructed, and whether they must be lined (e.g., proximity to drinking water resources and/or chemical composition of the fluid) (Richardson et al., 2013). Most states restrict the use of wastewater pits in environmentally sensitive areas. To avoid contamination events, some states are moving toward requiring closed loop systems (i.e., tanks) or injection wells rather than using pits for hydraulic fracturing wastewater storage. For example, Pennsylvania has modified their regulations (published October 8, 2016) to ban the use of pits for temporary storage of unconventional (as defined by PA DEP) wastewaters; many operators have already moved to closed-loop systems (PA DEP, 2016a). This development is particularly notable because of

Pennsylvania's heavy reliance on reuse for wastewater management, necessitating both on-site and off-site storage.

8.4.5.1 Locations and Numbers of Pits

The locations and number of existing pits (both for storage and for disposal) are not well documented in all states, and in the data found, pits associated with hydraulic fracturing operations were not specifically identified. With respect to larger pits for storage or disposal of wastewater, some states (e.g., Utah and Oklahoma) provide locational data on their websites. In 2016, the state of California began posting the number of active and inactive oil field produced water "ponds" (defined as unlined surface impoundments), both permitted and unpermitted, on their website. The July 2016 posting showed that 64% (682) of the 1,065 unlined ponds identified in the Central Valley and Central Coast of California were active. Of the active ponds, 21% (144) were not permitted (CA Water Board, 2016). Active ponds are primarily found in the southern San Joaquin Valley (CCST, 2015a). The EPA Region 8 conducted a survey of pits associated with oil and gas operations in Colorado, Montana, North Dakota, South Dakota, Utah, and Wyoming from 1996 through 2002. Results indicated there were approximately 28,000 pits at that time (U.S. EPA, 2003b).

In the absence of an inventory of pits in Pennsylvania, the organization SkyTruth led an effort using volunteers to produce a map of pits believed to be associated with drilling and hydraulic fracturing the Marcellus Shale (Manthos, 2014). The identification of pits was based on USDA aerial imagery taken in 2005, 2008, 2010, and 2013. SkyTruth acknowledges the uncertainties associated with identifying pits based on aerial images and volunteer labor. They have described their methodology as including multiple reviewers and QA/QC procedures. The study cannot differentiate ponds for drilling fluids and fracturing fluids from those for wastewater. Their preliminary findings indicate that the estimated number of ponds rose from 11 in 2005 to 529 in 2013, with the structures themselves increasing in size from a median size of 3,713 ft^2 (345 m^2) in 2005 to 66,844 ft^2 (6,210 m^2 in 2013. SkyTruth also notes that impoundments are not permanent and that of 581 ponds delineated in 2010, only 116 of them were found in the images from 2013.

Evaporation ponds, referred to as Commercial Oil Field Waste Disposal Facilities (COWDFs), are a waste management strategy most commonly used in the western states such as Utah, Wyoming, and Colorado (USFWS, 2014). According to a 2016 list of approved COWDFs posted by the Utah Division of Oil, Gas, and Mining (Utah Division of Oil, 2016), 20 facilities in Utah are approved to accept produced water. All are in the eastern part of the state where the Uinta and Paradox basins are found (unconventional shale formations). The Wyoming Department of Environmental Quality website, accessed in 2016, lists 35 active COWDFs (WDEQ, 2016b). The increase in hydraulic fracturing activity in Wyoming has resulted in significant increase in wastewater disposed of in COWDFs (USFWS, 2014). Data from the Colorado Oil & Gas Conservation Commission includes eight active evaporation pits, five of which are unlined (COGCC, 2016). Ninety-five other active pits are listed in Colorado, with descriptors such as "production," "multi-well pit," "skim," or "produced water." Seventy-one of these are unlined, and 22 have synthetic liners. Eleven pits are located in

Garfield County, where there is hydraulic fracturing activity. The Colorado data do not distinguish pits at centralized commercial facilities from on-site pits.

8.4.5.2 Unlined Storage Pits and Percolation Pits

Whether an unlined pit is designed and intended to percolate wastewater into the ground for disposal or if it is built for storage, it provides a pathway for wastewater to infiltrate into the subsurface and potentially reach groundwater. Such pits have been used historically for conventional oil and gas wastewater. More recently, they have received wastewater in areas where hydraulic fracturing takes place. States such as Montana and Wyoming allow unlined pits to be used for storage if the quality of the waste fluid meets specified limits and the pit is not in close proximity to environmentally sensitive areas such as drinking water resources, wetlands, and floodplains (Kuwayama et al., 2015b; Richardson et al., 2013).

In the past, several states have allowed unlined pits designed to dispose of wastewater via percolation into the subsurface. For example, until July 2015, percolation pits were permitted for wastewaters from hydraulically fractured wells in the Central Valley Region in California (Grinberg, 2016). The California Department of Conservation's Division of Oil, Gas, and Geothermal Resources (DOGGR) listed "evaporation-percolation" as the management method for almost 60% (190 million gal) of the wastewater generated via well stimulation in Kern County between 2011 and 2014 (CCST, 2015a). However, according to DOGGR's 2015 report addressing well stimulation activities in Kern County from January 1, 2014 through September 30, 2015, evaporation/percolation was not employed as a disposal option during that period (98% of the produced water was disposed of via operator-owned Class II injection wells, 1.75% was disposed of via commercial Class II injection wells, and 0.16% was reused).

While the practice of disposal via percolation pits has been discontinued in most states, as of July 2016, Wyoming's regulations still allow the use of percolation for disposing produced water specific to CBM operations in the Powder River Basin. To be permitted, the operator must demonstrate that the disposed fluid will comply with water quality standards of the Department of Environmental Quality (WYOGCC, 2015).

8.4.5.3 Evaporation Ponds

Evaporation is a simple water management strategy involving transporting wastewater to a pond or pit with a large surface area and allowing passive evaporation of the water from the surface (NETL, 2014; Clark and Veil, 2009). As discussed above, this disposal option, often referred to as a COWDF, is practical for drier climates of the western United States. Evaporation ponds have been used for oil and gas wastewater disposal in Montana, Colorado, Utah, New Mexico, and Wyoming (Veil et al., 2004). However, New Mexico no longer allows the use of pits for disposal (NM EMNRD OCD, 2013), and in Montana, evaporation ponds are no longer allowed because they do not put extracted water to a beneficial use (NRC, 2010). Figure 8-8 shows an example of a lined evaporation pit in Montana (DOE, 2006).

Figure 8-8. Lined evaporation pit in the Battle Creek Field (Montana).
Source: DOE (2006). Reproduced with permission from ALL Consulting.

As the water component of the wastewater is subject to evaporation, the fluid remaining in the pond becomes concentrated, and a sludge layer is formed. Remaining residual brines in the pond can be collected and disposed of via an underground injection well, and the solids can be taken to a landfill (see Section 8.4.7 for more details). In cold, dry climates, a freeze-thaw evaporation method has also been used to purify water from oil and gas wastewater (Boysen et al., 1999).

Nowak and Bradish (2010) describe the design, construction, and operation of two large commercial evaporation facilities in Southern Cross, Wyoming and Danish Flats, Utah. Each facility includes 14,000 gal (53,000 L) three-stage concrete receiving tanks, a sludge pond, and a series of five-acre (20,234 m^2) evaporation ponds connected by gravity or force-main underground piping. The Wyoming facility, which opened in 2008, consists of two ponds with a total capacity of approximately 84 million gal (2 million bbls or 318 million L). The Utah facility, open since 2009, consists of 13 ponds with a total capacity of approximately 218.4 million gal (5.2 million bbls or 826.6 million L). Each facility receives 0.42 to 1.47 million gal (10,000 to 35,000 bbls; 1.59 million to 5.56 million L) of wastewater per day from oil and gas production companies in the area.

Evaporation ponds or pits are subject to state regulatory agency approval and must meet state standards for water quality and quantity (Boysen et al., 2002).

8.4.5.4 Impacts and Potential Impacts from Pits and Impoundments

Pits containing hydraulic fracturing wastewater have the potential to impact drinking water resources if spills and overflows cause runoff to surface water or if wastewater percolates through the soil and reaches groundwater. In addition to contaminants in the wastewater itself, wastewater that reaches groundwater may mobilize constituents in pit bottoms or soils, and it may also reach hydrologically connected surface water. These impacts are amplified with increasing pit/impoundment size (Quaranta et al., 2012). Percolation may be accidental (through tears or improper installation of liner) or by design in unlined pits (Sumi, 2004).

Compromised pit liners can result in leaks, and extreme weather events, such as floods, can cause pits to overflow. An analysis of three state databases (New Mexico, Oklahoma, and Colorado) where pits and tanks have been used for storage of hydraulic fracturing wastewater found that for pits, the most common causes of spills were from overflows and liner malfunctions (Kuwayama et al., 2015b). For instance, of the 106 pit-related spills reported in New Mexico between 2000 and 2014, 33% were due to overflows and 26% were caused by liner malfunctions. Of the 62 tank spills reported, 44% were due to leaks, and 27% were related to overfilling (Kuwayama et al., 2015b). The types of constituents in pits that may be of concern from such events include VOCs, metals, TDS, oil, and TENORM (Kuwayama et al., 2015b).

Operational factors also influence potential impacts from pits and impoundments. These can include water level management (influent, seepage, spillage), the length of time water is stored in the pit/impoundment, the composition of the water, the local climate (rainfall and/or evaporation), and the transmission method (piped or delivered in an open channel) (NRC, 2010).

Construction and Capacity Issues

Construction requirements typically include specifications for features that can reduce the potential for impacts on groundwater or surface water. These can include liner specifications, depth to groundwater, secondary containment, setback requirements, freeboard, leak detection, and water quality monitoring (Kuwayama et al., 2015b).[1,2] For example, in a 2012 review of 19 states with shale gas development or potential for shale gas development, many states had setback requirements for pits in sensitive areas including surface water, wetlands, and floodplains. As of December 2015, however, 12 of the 19 states surveyed did not include setback requirements in their regulations. Many states did address the vertical separation of pits from the water table (e.g., 20 in (0.5 m) to seasonal high water table in PA; 10 ft (3 m) in WY; 50 ft (15 m) in NM) (Kuwayama et al., 2015b).

Despite construction standards, impacts on groundwater or surface water due to overflows, liner breaches, and other construction issues have been documented. In 2007 in Knox County, Kentucky, retention pits holding hydraulic fracturing flowback fluids overflowed into Acorn Fork Creek during the development of four natural gas wells (CCST, 2015a; Papoulias and Velasco, 2013). The incident caused the pH of the creek to drop from 7.5 to 5.6 and the conductivity to increase from 200 to 35,000 µS/cm. In addition, organics and metals including iron and aluminum formed precipitates in the stream. Fish and aquatic invertebrates were killed or distressed in the area of the stream affected by the release (Papoulias and Velasco, 2013).

[1] Setback is the distance between the pit and a stream, lake, building, or other feature or structure that needs protection.

[2] Freeboard is the vertical distance between the level of the water in an impoundment and the overflow elevation (an outfall or the lowest part of the berm).

Similarly, in 2009, Marcellus wastewater stored in an impoundment from a hydraulic fracturing operation in Washington County, Pennsylvania overflowed the bank of the impoundment and reached surface water (a tributary of Dunkle Run) (CCST, 2015a). NRC (2010) reported continuous overfilling of an impoundment in the Powder River Basin (Wyoming) with CBM produced water, resulting in significant erosion of a seasonal water channel. The CBM operator was required through litigation to manage flows to the impoundment to prevent overflows. The literature did not report specific impacts on groundwater or surface water from the Pennsylvania or Wyoming incidents.

In Pennsylvania in 2010, pit liner failure was reported to have impacted groundwater through leakage of Marcellus wastewater from six impoundments (Colaneri, 2014). Ziemkiewicz et al. (2014) note that a study of 15 pits and impoundments in West Virginia found that slope stability and liner deficiencies were common problems. Construction quality control and quality assurance were often inadequate; the authors found a lack of field compaction testing, use of improper soil types, excessive slope lengths, buried debris, and insufficient erosion control, although no breaches were reported. A statistical analysis of oil and gas violations in Pennsylvania found that structurally unsound impoundments or inadequate freeboard were the second most frequent type of violation, with 439 instances in the period from 2008 to 2010 (Olawoyin et al., 2013).

Unlined Pits

Impacts on groundwater from historic and current uses of unlined pits in the oil and gas industry have been documented. In a review of records spanning 25 years (1983 – 2007), 63 incidents of private water supply contamination from the infiltration of saline fluids from unlined or inadequately constructed reserve pits were identified in Ohio (Kell, 2011). The same study (Kell, 2011) identified 57 legacy (pre-1984) incidents in Texas involving groundwater contamination from unlined produced water disposal pits. Such pits were phased out in Texas by 1984, prompting a move towards disposal of oil and gas wastewater in disposal wells.

Kern County, California has experienced impacts on groundwater associated with unlined percolation pits. A 2014 study notes that there are hundreds of pits across Kern County and elsewhere in the state, stretching state resources for regulatory oversight (Grinberg, 2014). Past sampling of water in percolation pits has shown exceedances of California's Tulare Lake Basin Plan (Basin Plan), which specifies maximum levels permitted for discharges of oil field well wastewater to unlined ponds overlying groundwater (Grinberg, 2014).[1] For example, the McKittrick 1 and 1-3 pits are large percolation pits in Kern County near oil fields where most of the hydraulic fracturing in California takes place (Grinberg, 2014). The pits are situated close to a number of important resources. They are located within a few miles of the Kern River Flood Channel, the California State Water Project, farmland, and are in an area of high quality groundwater (Grinberg, 2014). Sampling of fluids in the pits dating back to 1997 showed consistent exceedances of Tulare Basin Plan standards for TDS, chlorides, and boron. Sampling also revealed the presence of BTEX, gasoline range organics (GRO), and diesel range organics (DRO) (MTA, 2014). Sampling of three monitoring

[1] The Basin Plan sets limits for salinity (1,000 µmhos/cm measured as electrical conductivity), chloride (175 mg/L), and boron (1 mg/L) (California Regional Water Quality Control Board Central Valley Region, 2015).

wells indicated that in 2004, a plume had migrated at least 4,000 ft (1,000 m) from the pits and was still detected in test wells in 2013. As of July 1, 2015, California's Code of Regulations includes a provision that no longer allows the use of pits, including percolation pits, for fluids produced from stimulated wells (Grinberg, 2016).

Unlined pits that were used from the 1960s until the mid-1990s for disposal of drilling muds and flowback and produced waters associated with hydraulic fracturing operations have been linked to groundwater contamination in Pavillion, Wyoming (Digiulio and Jackson, 2016; AME, 2015). A report by the Wyoming Oil and Gas Conservation Commission (WYOGCC) (WYOGCC, 2014a) summarizes site investigations and reclamation activities conducted by WOGCC, the Wyoming Department of Environmental Quality (WDEQ), and Encana Oil and Gas for pits in the Pavillion Well Field. The report includes information on samples collected between 2006 and 2013 from shallow groundwater in the vicinity of the pits. Some sites had detections for one or more of the following contaminants: GRO, DRO, BTEX, and/or naphthalene. Of the shallow groundwater sites with detections, some were associated with pits located within one-quarter mile of a domestic well. One of these sites exceeded clean-up levels established by the WDEQ Voluntary Remediation Program for DRO (13,000 µg/L) and benzene (110 µg/L).[1] The report noted that there was insufficient evidence to determine whether or not drinking water supply wells in the vicinity of the pits were contaminated by disposal of hydraulic fracturing wastewater in those pits (WYOGCC, 2014a).

Other examples in the literature include the detection of VOCs in groundwater downgradient of an unlined pit containing oil and gas wastewater near the Duncan Oil Field in New Mexico (Sumi, 2004) (Section 8.5). Groundwater impacts downgradient of an unlined pit in Oklahoma included high salinity (3500-25,600 mg/L) and the presence of VOCs (Kharaka et al., 2002). Neither New Mexico nor Oklahoma currently allows unlined pits for disposal or storage (OCC OGCD, 2015; NM EMNRD OCD, 2013).

Mobilization and Transport of Constituents

Groundwater impacts may result not just from constituents in the wastewater but also from mobilization of existing constituents in the soil or sediment. A CBM produced water impoundment in the Powder River Basin of Wyoming was studied for its impact on groundwater (Healy et al., 2011; Healy et al., 2008). Infiltration of water from the impoundment was found to create a perched water mound in the unsaturated zone above bedrock in a location with historically little recharge. Elevated concentrations of TDS, chloride, nitrate, and selenium were found at the site, with one lysimeter sample exceeding 100,000 mg/L of TDS (Healy et al., 2008). Most of the solutes found in the groundwater mound did not originate with the CBM produced water, but rather were the consequence of dissolution of previously existing salts and minerals (Healy et al., 2011).

Generally, the deeper that wastewater can move into an aquifer, as impacted by the volume and timing of the release, the longer the duration of contamination (Whittemore, 2007). Kharaka et al. (2007) reported on studies at a site in Oklahoma with one abandoned and two active unlined pits.

[1] WDEQ cleanup levels are derived from a combination of promulgated levels (MCL, state-assigned water quality standards) and risk-based cleanup level concentrations (WDEQ, 2016a).

Produced water from these pits penetrated 10 to 23 ft (3 to 7 m) thick shale and siltstone units, creating three plumes of high-salinity water (5,000 to 30,000 mg/L TDS). The impact of these plumes on the receiving water body (Skiatook Lake) was judged to be minimal, although the estimate was based on a number of notably uncertain transport quantities (Otton et al., 2007).

Vadose (unsaturated) zone transport was illustrated at a site in Oklahoma where two abandoned pits were major sources for releases of produced water and oil. Saline water from the pits flowed through thin soils and readily percolated into underlying permeable bedrock. Deeper, less-permeable bedrock was contaminated by salt water later in the history of the site, presumably due to fractures. The mechanisms proposed were vertical movement through permeable sand bodies, lateral movement along shale fractures, and possibly increased clay permeability due to the presence of highly saline water (Otton et al., 2007).

Summary

Collectively, the above examples show that regardless of the purpose of pits (storage or disposal), they present a potential pathway for wastewater constituents to impact groundwater or surface water. Good construction standards and practices, including liners, adequate freeboard, and setbacks, are important for minimizing potential impacts on both surface water and groundwater. Proper monitoring and maintenance (e.g., avoiding overfilling, maintaining the integrity of liners and berms) are also important for protecting surface water and groundwater. Unlined pits, in particular, can lead to groundwater contamination. This can be long-lasting, as evidenced by legacy impacts from older pits. Most states have phased out unlined disposal pits and unlined storage pits, but if such pits are still in use, they can provide ongoing potential sources of groundwater contamination (CCST, 2015a; Grinberg, 2014).

8.4.6 Other Management Practices and Issues

Additional strategies for wastewater management in some states include directly discharging to surface waters and land application. In particular, wastewater from CBM fracturing and production generally has lower TDS concentrations than wastewater from other types of unconventional formations and more readily lends itself to other uses.

8.4.6.1 Land Application and Road Spreading

Road spreading has been used as a disposal option for high-TDS wastewaters (brines) from conventional oil and gas production. Road spreading can be done for dust control and de-icing. Although recent data are not available, an American Petroleum Institute (API) survey estimated that approximately 75.6 million gal (1.8 million bbls or 286.2 million L) of wastewater was used for road spreading in 1995 (API, 2000). The API estimate does not specifically identify hydraulic fracturing wastewater. There is no current nationwide estimate of the extent of road spreading using hydraulic fracturing wastewater.

Road spreading with hydraulic fracturing wastewater is regulated primarily at the state level (Hammer and VanBriesen, 2012) and is prohibited in some states. For example, with annual approval of a plan to minimize the potential for pollution, PA DEP allows spreading of brines from conventional (as defined by PA DEP) wells for dust control and road stabilization. Hydraulic fracturing flowback, however, cannot be used for dust control and road stabilization (PA DEP, 2011b). In West Virginia, use of gas well brines for roadway de-icing is allowed per a 2011 memorandum of agreement between the West Virginia Division of Highways and the West Virginia Department of Environmental Protection, but the use of "hydraulic fracturing return fluids" is not permitted (Tiemann et al., 2014; West Virginia DEP, 2011).

Concerns about road application center on contaminants such as barium, strontium, and radium. A report from PA DEP analyzed several commercial rock salt samples and compared results with contaminants found in Marcellus Shale flowback samples. The results noted elevated barium, strontium, and radionuclide levels in Marcellus Shale brines compared with commercial rock salt (Titler and Curry, 2011). Another study found increases in metals (radium, strontium, calcium, and sodium) in soils ranging from 1.2 to 6.2 times the original concentrations (for radium and sodium, respectively), attributed to road spreading of wastewater from conventional oil and gas wells for de-icing (Skalak et al., 2014).

Potential impacts on drinking water resources from road spreading have been noted by Tiemann et al. (2014) and Hammer and VanBriesen (2012). These include potential effects of runoff on surface water and migration of brines to groundwater. Snowmelt can carry salts and other chemicals from the application site, and transport can increase if application rates are high or rain occurs soon after application (Hammer and VanBriesen, 2012). Research on the impacts of conventional road salt application has documented long-term salinization of both surface water and groundwater in the northern United States (Kelly, 2008; Kaushal et al., 2005). When conventional oil field brine was used in a controlled road spreading experiment, elevated chloride concentrations were detected in shallow groundwater (Bair and Digel, 1990). The amount of salt attributable to road application of hydraulic fracturing wastewaters has not been quantified.

To evaluate land application of solid wastes from oil and gas production, a laboratory study mimicking land spreading of conventional oilfield scales and sludges indicated that 20% of the radium in barite sulfate scales was released by microbial processes during incubation with soil (Matthews et al., 2006; Swann et al., 2004). Although the radium was then complexed with the soil, it would be more mobile and more bioavailable than when it was associated with the barite. Overall, potential effects on drinking water resources from land spreading are not well understood, including the amounts of hydraulic fracturing wastes that are managed by land spreading.

8.4.6.2 Management of Coalbed Methane Wastewater

Many, but not all, CBM wells are hydraulically fractured to enhance recovery, using fluids that range from water alone to more complex gel formulations with proppant (e.g., Engle et al., 2011; McCartney, 2011; NRC, 2010; Halliburton, 2008; U.S. EPA, 2004a). The literature indicates that hydraulic fracturing of CBM formations is being conducted in the San Juan, Raton, Piceance, and Uinta Basins, among others. Literature such as NRC (2010) notes that hydraulic fracturing may not be common in the Powder River Basin. Additionally, when CBM well stimulation does take place, it can be accomplished using very simple hydraulic fracturing fluid formulations (Chapter 3).

Wastewater from CBM wells can be managed like other hydraulic fracturing wastewater discussed above. However, the wastewater from CBM wells can also be of higher average quality (typically lower TDS content) than wastewater from other hydraulically fractured wells. The lower TDS content makes it more suitable for certain management practices and uses. A number of management strategies have been proposed or implemented, with varying degrees of treatment required depending on the quality of the wastewater and the intended use (Hulme, 2005; DOE, 2003, 2002). Although specific volumes managed through the practices discussed below are not well documented, qualitative information and considerations for feasibility are available and presented. The discussion below covers both dilute and higher-TDS wastewater from CBM formations.

The quality of CBM wastewater plays a large role in how the wastewater is managed. The TDS content can range from an average of nearly 1,000 mg/L in the Powder River Basin to an average of about 14,000 mg/L (and as high as approximately 62,000 mg/L) in the Black Warrior Basin (Appendix Table E-3). Data sources from about 2002 through 2008 indicate that operators in some basins such as the San Juan, Uinta, and Piceance, and Raton (in New Mexico), where TDS is typically higher compared to other basins (e.g., Powder River), manage most wastewater by injection into disposal wells (NRC, 2010; U.S. EPA, 2010a).

Discharge to rivers and streams, a management option governed by the CWA, may be permitted in cases where wastewater is of high quality.[1] To be discharged, the wastewater must meet technology-based effluent limitations established by the permitting authority on a case-by-case "best professional judgment" basis as well as any more stringent limitations necessary to meet applicable water quality standards. For example, as a means of protecting high-quality waters of the state, the Montana Supreme Court ruled in 2010 that treatment is required for all CBM produced water prior to discharge to surface water (NRC, 2010).

A 2008 EPA survey of CBM operators found that of the projects represented in the results, direct discharge to surface water was by far most prevalent in the Powder River Basin but was also reported as a management practice in the Green River, Raton, Black Warrior, Cahaba, Illinois, and Appalachian basins (U.S. EPA, 2013e, 2010a).[2] Discharges to surface water can provide habitat maintenance, restoration of wildlife-waterfowl fishery habitat, and flow augmentation to benefit downstream water users. However, hydrologic changes from such discharges could also have unanticipated effects on ecosystems previously adapted to intermittent streamflow.

Some CBM wastewater can be put to agricultural use, including livestock and wildlife watering, and crop irrigation. Livestock watering with CBM wastewater can be done using on-channel or off-channel impoundments, and irrigation is an area of active research (e.g., Engle et al., 2011; NRC, 2010). However, wastewater from some higher-salinity CBM basins (e.g., San Juan, Uinta, and Piceance) would need blending or treatment before such uses. Irrigation with treated CBM

[1] Although discharge to rivers and streams is generally prohibited under the EPA's oil and gas ELGs, the ELGs do not apply to CBM.
[2] These reports did not describe certain non-discharging wastewaters management strategies in basins with few operators in order to preserve CBI. The reports also do not provide information on hydraulic fracturing activities in the basins. Not also that results are presented by numbers of projects, which may vary in the number of wells they contain.

wastewater would be most suitable on coarse-textured soils for cultivation of salt-tolerant crops (DOE, 2003). NRC (2010) remarks that "use of CBM produced water for irrigation appears practical and sustainable," provided that appropriate measures are taken such as selective application, dilution or blending, appropriate timing, and rehabilitation of soils.

Although CBM wastewater is generally lower in TDS than wastewater associated with shale gas development, it can still have higher TDS concentrations than stream water. This poses concerns regarding the sodium adsorption ratio (SAR) for agricultural soils. A USGS study performed trend analysis of water quality at sampling sites in the Tongue and Powder River watersheds (Powder River Basin) (Sando et al., 2014). One of the study objectives was to determine possible effects of CBM produced water particularly in areas where the water was discharged to impoundments or upper reaches of in-stream channels for infiltration. Trend analysis showed potential effects of CBM production on downstream water quality (increases in sodium, alkalinity, and SAR) in the main-stem Powder River but found mixed results at the Tongue River sites (some appeared to be impacted by CBM activities while others did not) (Sando et al., 2014).

Sando et al. (2014) found that CBM pumping rates (i.e., discharge of produced water) were high relative to streamflow in the Powder River Basin. For the three main-stem Powder River sites, the CBM pumping rates were 26-34% of the 2001-2010 median streamflows. For one site in the Little Powder River watershed, the CBM pumping rate was 360% of 2001-2010 median streamflow. This underscores that in arid climates in the western United States, permitted discharges from CBM activities (whether hydraulically fractured or not) at a particular site may be large relative to the size of the receiving water and may sometimes dominate flows.

As noted above, a degree of treatment is needed (or required) for some uses. Plumlee et al. (2014) examined the feasibility, treatment requirements, and potential costs of several hypothetical uses for CBM wastewater. In several cases, costs for these uses were projected to be comparable to or less than estimated disposal costs. In one case study, use of CBM wastewater for streamflow augmentation or crop irrigation could potentially cost between $0.26 and $0.27 per bbl. For comparison, reported disposal costs in 2000-2001 ranged from $0.01 per bbl for a pipeline collection system with impoundment to $2.00 per bbl for hauling to disposal or treatment. The 2010 NRC report (NRC, 2010) noted that 15 to 18% of CBM produced water in the Powder River Basin was being treated to reduce SAR in order to satisfy NPDES permits for discharge.[1] If wastewater is treated to address SAR, reported costs are approximately $0.12 to $0.60/bbl (NRC, 2010).

The applicability of particular uses may be limited by ecological and regulatory considerations as well as the irregular nature of CBM wastewater production (voluminous at first, and then declining and halting after a period of years). Legal issues, including overlapping jurisdictions at the state level and senior water rights claims in over-appropriated basins (in western states) can also determine the use of CBM wastewater (Wolfe and Graham, 2002).

[1] SAR is the relative proportion of sodium to other cations in water. It is also an indication of risk to soil from alkalinity. The higher the SAR, the less suitable the water is for irrigation, and long-term use can damage soil structure.

8.4.6.3 Other Documented Uses of Hydraulic Fracturing Wastewater

Uses of wastewater from shales or other hydraulically fractured formations face many of the same possibilities and limitations as those associated with wastewater from CBM operations. The biggest difference is in the quality of the water. Wastewaters vary widely in water quality, with TDS values from shale and tight sand formations ranging from less than 1,000 mg/L TDS to hundreds of thousands of mg/L TDS (DOE, 2006) (Chapter 7). Wastewaters on the lower end of the TDS spectrum could be reused in many of the same ways as CBM wastewater, depending on the concentrations of potentially harmful constituents and applicable federal, state, and local regulations. High TDS wastewaters have more limited uses, and pre-treatment may be necessary (Shaffer et al., 2013; Guerra et al., 2011; DOE, 2006). Agricultural and wildlife uses are subject to the produced water daily effluent discharge limit of 35 mg/l for oil and grease.[1]

Potential uses for wastewater in the western United States include livestock watering, irrigation, streamflow supplementation, fire protection, road spreading, and industrial uses, with each having their own water quality requirements and applicability (Guerra et al., 2011). Guerra et al. (2011) summarized the least conservative TDS standards for five possible uses in the western United States that include 500 mg/L for drinking water (the drinking water secondary maximum contaminant level (SMCL)), 625 mg/L for groundwater recharge, 1,000 mg/L for surface water discharge, 1,920 mg/L for irrigation, and 10,000 mg/L for livestock watering. The authors estimated that wastewater from 88% of unconventional wells in the western United States could be used for livestock watering without TDS removal based on a maximum TDS concentration of 10,000 mg/L. However, wastewater from only 10% of unconventional wells could be used for surface discharge without treatment for TDS based on the least conservative standard among the western states of 1,000 mg/L TDS (Guerra et al., 2011). Guerra et al. (2011) indicate that in several basins in the western United States (e.g., Wind River, Green River, and Powder River), wastewater from 50% or more of oil and gas wells is suitable for agricultural use. In other basins (e.g., San Juan, Piceance, and Permian) over 50% of oil and gas wastewater is unsuitable for use without treatment. A 2006 Department of Energy (DOE) study pointed out that the quality necessary for use in agriculture depends on the plant or animal species involved and that in the Bighorn Basin in Wyoming, low-salinity wastewater is used for agriculture and livestock watering after minimal treatment to remove oil and grease (DOE, 2006).

Although TDS is a common criterion for water quality, there are also recommended limits or considerations for some metals, alkalinity, and nitrate in water for use in livestock watering, and for metals, SAR, electrical conductivity (ECw), and pH for water for irrigation (Guerra et al., 2011). Also, using TDS/salinity as the primary criterion may not be appropriate if wells contributing to the produced water have undergone hydraulic fracturing or if maintenance chemicals are being used on the well.

The water quality standards and monitoring requirements for direct discharge for use in irrigation or livestock watering include few specifications. In California, the California Council on Science and Technology (CCST, 2015a) notes that the testing and treatment required by the regional water

[1] 40 CFR 435.52(b).

quality control boards prior to use of produced water for irrigation do not include assessment for chemicals associated with hydraulic fracturing and that there are no policies prohibiting the use of hydraulic fracturing wastewaters for irrigation.

In the Wind River Basin in Wyoming, three NPDES permits were appealed by environmental groups due to concerns that the permits failed to address maintenance and hydraulic fracturing chemicals (Natural Resources Defense Council, 2015; PEER, 2015). The environmental groups argued that the EPA's regulations do not allow for the discharge of produced water containing chemicals from well treatment, and that, moreover, the EPA lacked sufficient information regarding the well treatment chemicals to determine whether the discharge would be "good enough quality" for wildlife and agricultural use, as required under the ELG regulations. As an example, the environmental groups pointed to MSDS information provided upon request for six maintenance products, which included toxic chemicals such as ethylene glycol, benzyl chloride, isopropanol, naphthalene, benzene, and xylene, among others. This raised concerns that produced water permitted for direct discharge may contain toxic chemicals or their degradation products. Ultimately, pursuant to a settlement agreement with the environmental groups and permittees, the EPA issued modified permits that included additional conditions for handling of and reporting about well stimulation and well maintenance chemicals.

8.4.7 Management of Solid and Liquid Residuals

Solid and liquid residuals associated with hydraulic fracturing wastewater are formed from treatment processes at CWTs, buildup of sludges in tanks and pits, and scale formation on pipes and equipment. These residuals must be managed and disposed of properly to avoid impacts on ground and surface water resources. (Note that drill cuttings and drilling muds are outside the scope of this chapter.)

8.4.7.1 Solid Residuals

The solid residuals produced at a CWT depend on the constituents in the untreated water and the treatment processes used and are likely to contain TSS, TDS, metals, radionuclides, and organics. Solid residuals can consist of sludges (from precipitation, filtration, settling units, and biological processes), spent media (filter media, adsorption media, or ion exchange media), and other material such as spent filter socks used to remove gross particulates. In addition, solids that accumulate in storage tanks and pits and scale that deposits on equipment are part of the residual load from a site. These residuals can constitute a considerable fraction of solid waste in an oil or gas production area.

Handling and disposal of residual sludges from treatment processes can present some of the biggest challenges associated with these technologies (Igunnu and Chen, 2014). Additional treatment may be applied to solid residuals including thickening, stabilization (e.g., anaerobic digestion), and dewatering processes prior to disposal. The solid residuals are then typically sent to a landfill, land spread on-site, or incinerated (Morillon et al., 2002). Land spreading is a waste management method in which wastes are spread over the soil surface and tilled into the soil to allow the hydrocarbons in the wastes to biodegrade (Smith et al., 1998); note that inorganic constituents