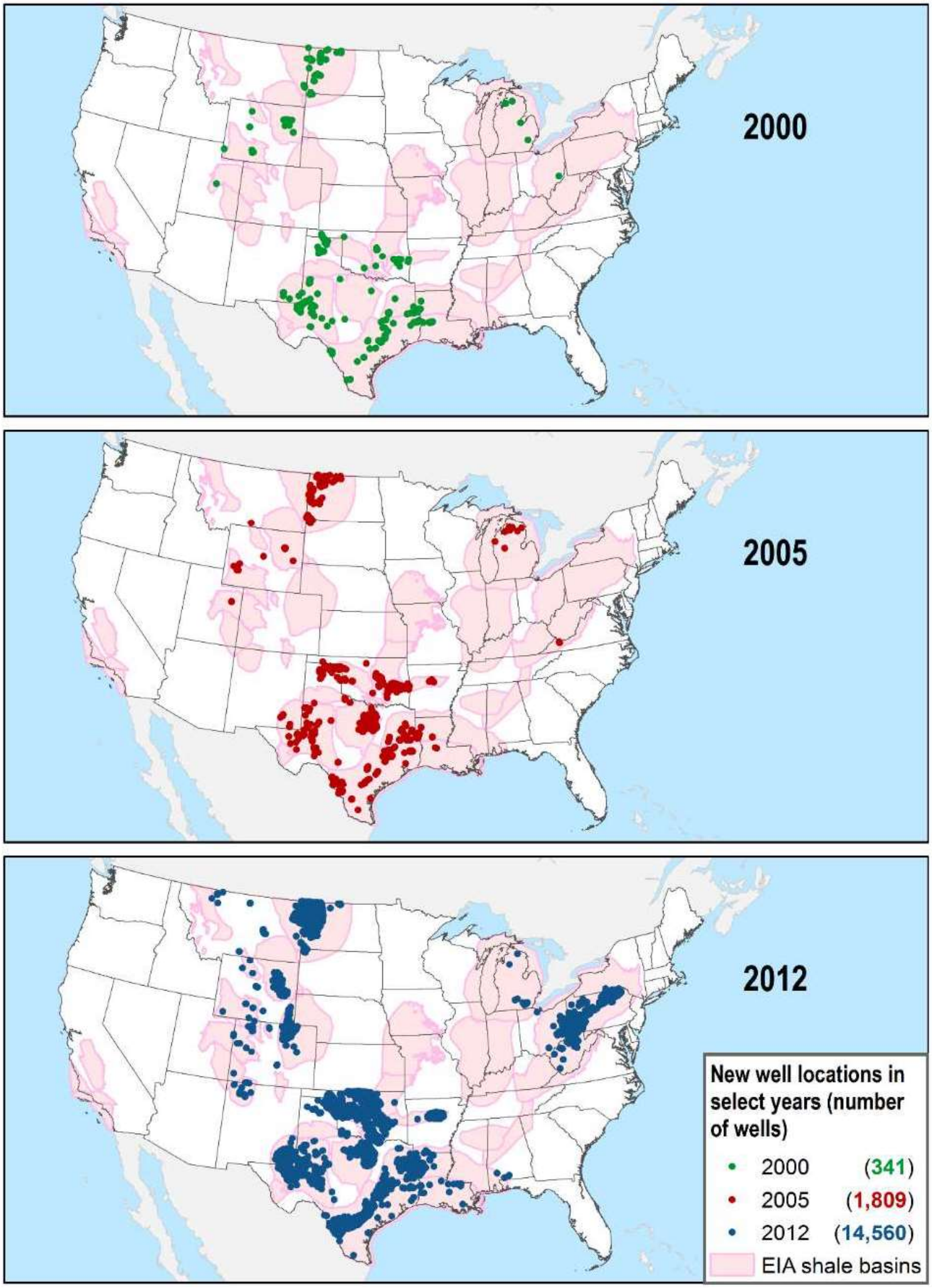


Figure 3-20. Location of horizontal wells that began producing oil or natural gas in 2000, 2005, and 2012.
Based on data from DrillingInfo (2014a).

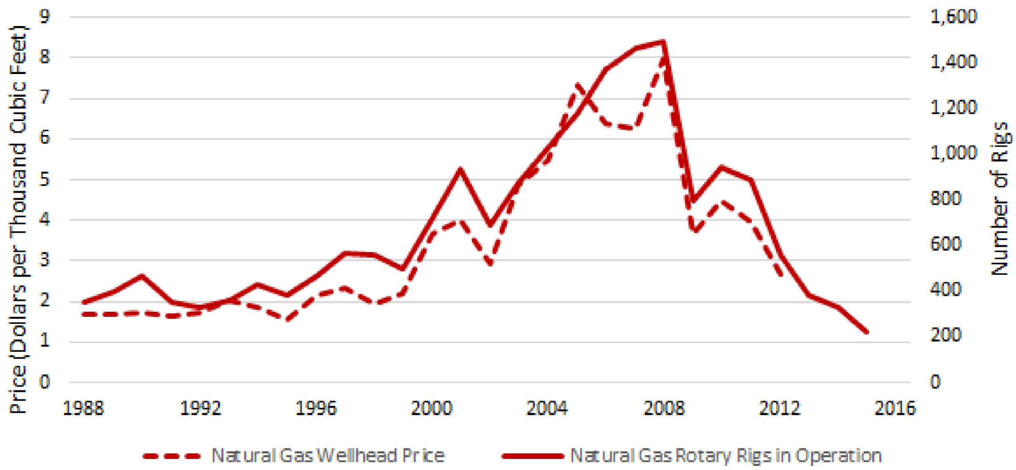


Figure 3-21. Natural gas prices and drilling activity, United States, 1988 to 2015.
Sources: EIA (2016b) and EIA (2016f).

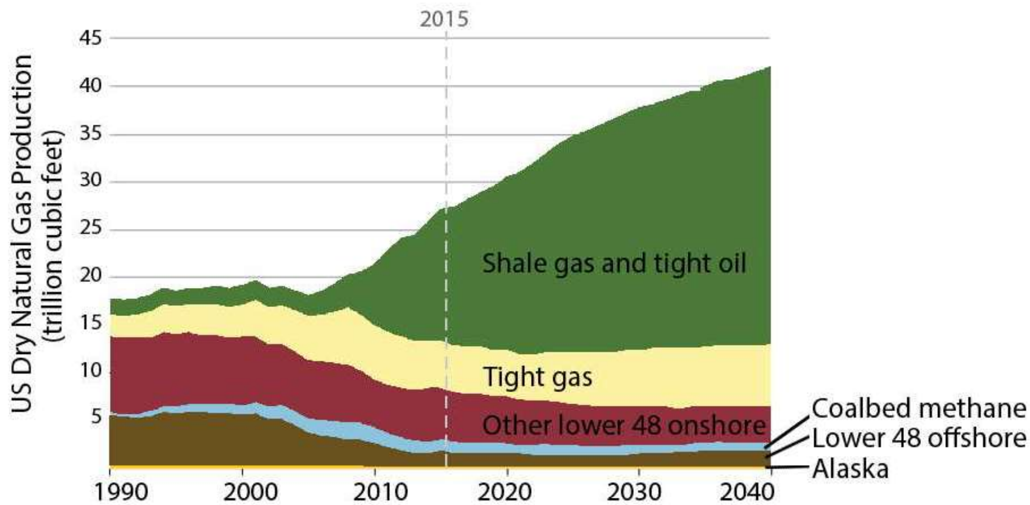


Figure 3-22. Historic and projected natural gas production by source (trillion cubic feet).
Source: EIA (2016a).

the details are subject to change. Nonetheless, a continuing increase in production is generally suggested and the locations of historical production identified in Figure 3-23 indicate areas of continued and future hydraulic fracturing activities for natural gas production.

The geographic concentration and trends in shale gas production by play (and identified by state) are shown in Figure 3-23. The Barnett Shale, where the modern hydraulic fracturing boom started, was the largest producer of shale gas until about 2010, producing 1.5 trillion cubic feet (tcf) (42.5 billion cubic meters [bcm]) that year and remains a significant producer. In 2009, the Marcellus and Haynesville plays produced 0.12 and 0.43 tcf (3.4 and 12.2 bcm), respectively, but by 2011, production from the Haynesville play surpassed that in the Barnett play, and by 2013 the Marcellus Shale surpassed both the Barnett and the Haynesville to become the play with the most production.

By 2014, the Marcellus play was producing 4.8 tcf (135.9 bcm) of gas annually, with the Eagle Ford, Haynesville, and Barnett each producing roughly 1.5 tcf (42.5 bcm). Estimates of technically recoverable resources, a general indicator of potential future production, are noted for the Marcellus (about 150 tcf [4.25 trillion cubic meters]), Haynesville (73 tcf [2.07 trillion cubic meters]), Eagle Ford in Texas (55 tcf [1.56 trillion cubic meters]), and Utica in Ohio, Pennsylvania, and West Virginia (55 tcf [1.56 trillion cubic meters]). This suggests that these four plays will be active contributors of shale gas production for the foreseeable future (EIA, 2013).[1] Other gas plays with significant resources include the Fayetteville in Arkansas, the Woodford in Oklahoma, and the Mancos in Colorado.

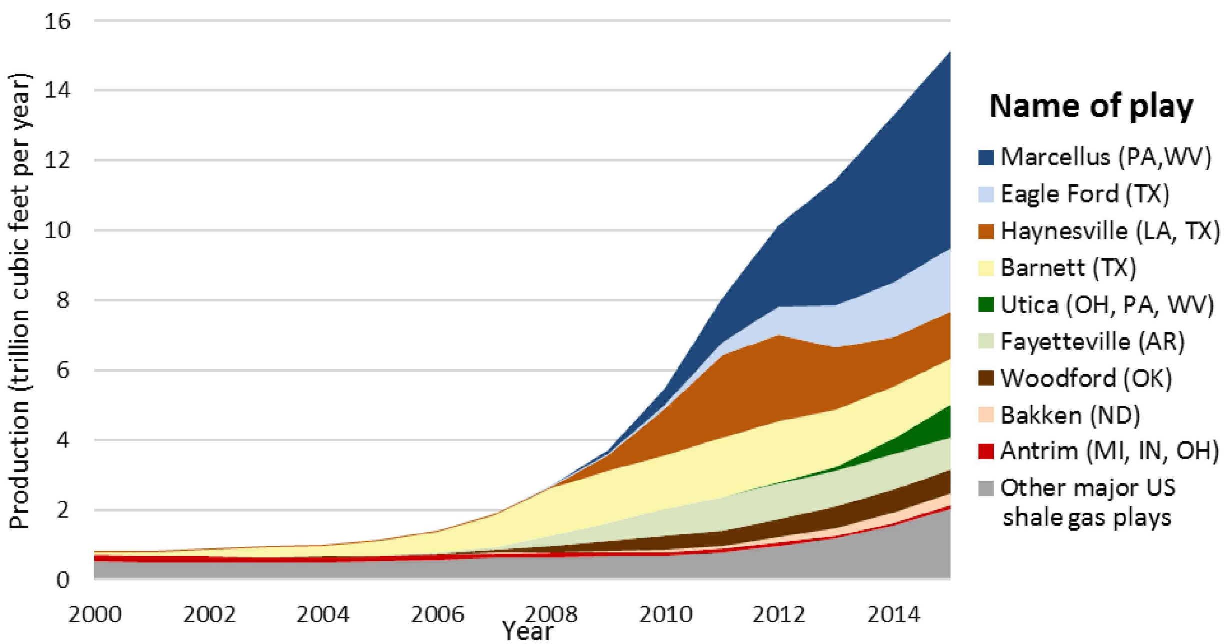


Figure 3-23. Production from U.S. shale gas plays, 2000-2014.
Source: EIA (2016g). The graph shows shale plays in the same vertical order as the legend.

3.5.2 Oil

While prices and drilling activity for natural gas were peaking between 2005 and 2008 and then falling (Figure 3-21), prices and drilling for oil were rising. These peaked between 2011 and 2014, and then rapidly declined as well (Figure 3-24). EIA projections to 2040 indicate a continued growth in total U.S. oil production, although the projected growth is not as fast or as large as that projected for natural gas. Tight oil production, presumably from hydraulically fractured wells, is expected to account for much of the projected growth (Figure 3-25); by 2040, tight oil is expected to account for nearly 65% of all U.S. crude oil production (EIA, 2016d). These production projections are dependent on estimated future prices of oil and other assumptions and, therefore, will likely be revised over time as energy markets and prices change. Currently, these projections

[1] Technically recoverable resources are the volumes of oil or natural gas that could be produced with current technology, regardless of oil and natural gas prices and production costs (EIA, 2013).

indicate a continuing, but lower rate of growth (as compared to the period from about 2005 to 2015). The locations of historical production identified in Figure 3-26 indicate areas of continued and future hydraulic fracturing activities for oil.

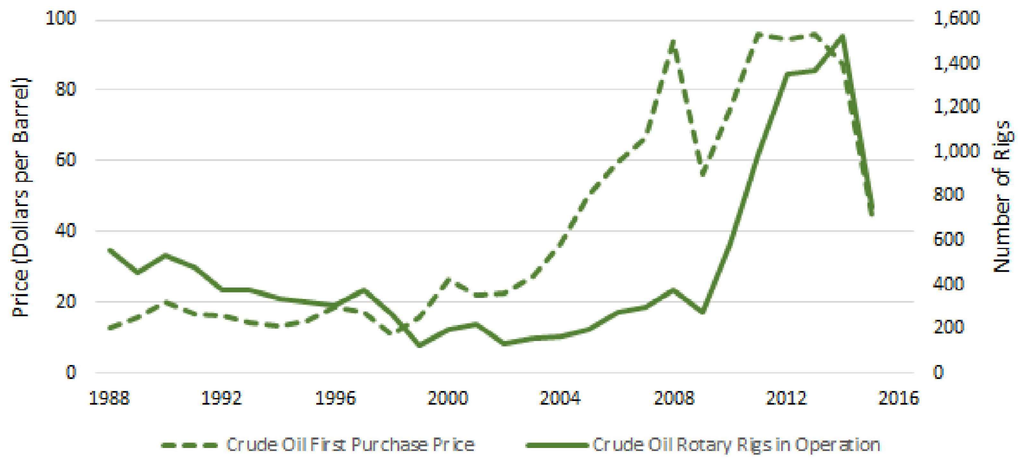


Figure 3-24. Crude oil prices and drilling activity, United States, 1988 to 2015.
Sources: EIA (2016b) and EIA (2016e).

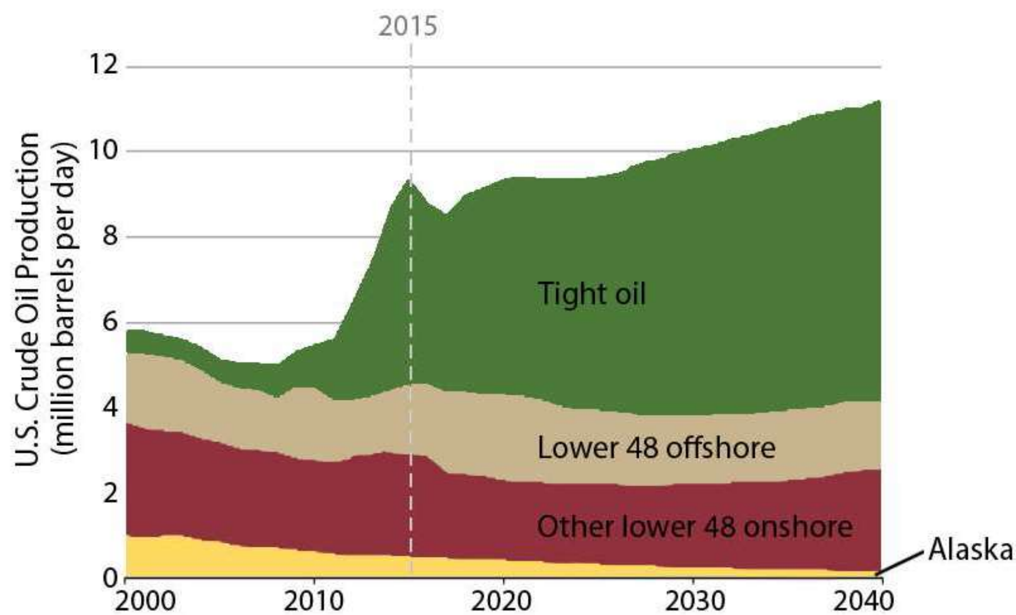


Figure 3-25. Historic and projected oil production by source (million barrels per day).
Source: EIA (2016a).

The geographic concentration and trends in tight oil production by play (and identified by state) are shown in Figure 3-26. Early tight oil production in the United States was centered in the Permian Basin in west Texas and New Mexico, at plays that included the Spraberry and the Bonespring. After 2009, the Bakken play (centered in western North Dakota) and the Eagle Ford play (in Texas) emerged as the largest-producing tight oil plays. Oil production in the Bakken increased from 99 million bbls (16,000 million L) in 2009 to 394 million bbls (63,600 million L) in 2014 (EIA, 2016g). Production from Eagle Ford increased from 12 million bbls (2,000 million L) in 2009 to 498 million bbls (79,100 million L) in 2014 (EIA, 2016g).

General estimates of potential resources suggest that future tight oil production in the United States will continue to be led by Texas and North Dakota. Technical recoverable resources are estimated at about 23 billion bbls (3,600 billion L) for the Bakken, about 21 billion bbls (3,300 billion L) for the Permian Basin, and about 10 billion bbls (1,600 billion L) for Eagle Ford (EIA, 2015). Other plays with significant estimated resources include the Niobrara-Codell in Colorado and Wyoming and the Granite Wash in Oklahoma and Texas (EIA, 2012).

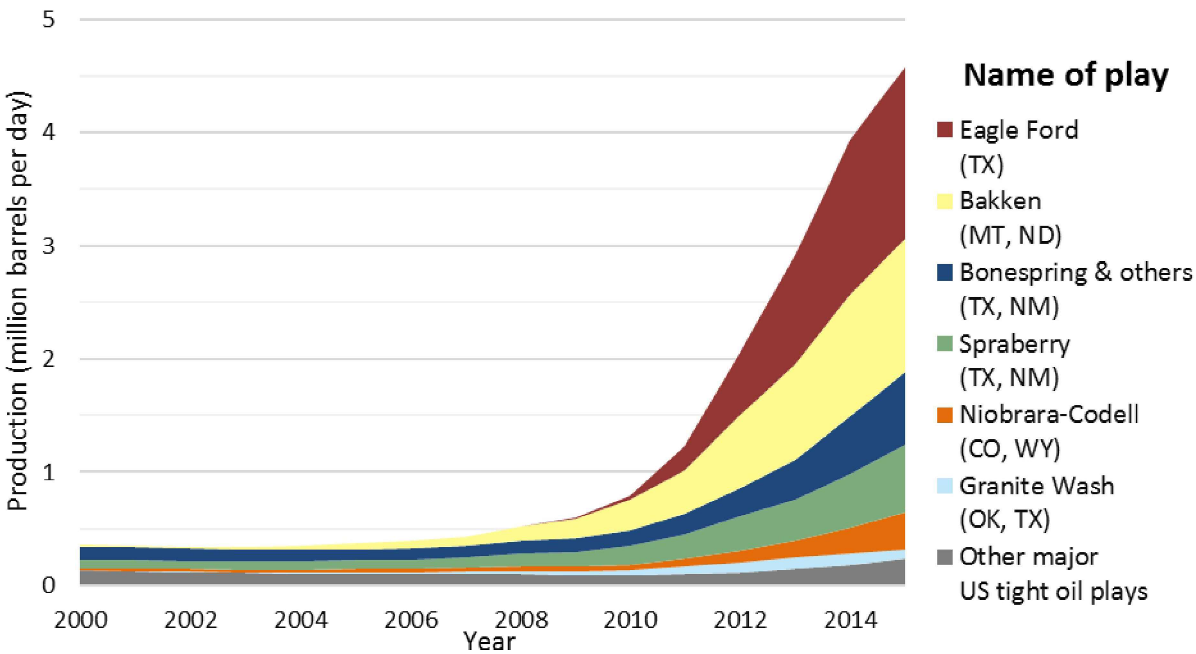


Figure 3-26. Production from U.S. tight oil plays, 2000-2014.
Source: EIA (2016g). The graph shows tight oil plays in the same vertical order as the legend.

3.6 Conclusions

Hydraulic fracturing is the injection of hydraulic fracturing fluids through the production well and into the subsurface oil or gas reservoir under pressures great enough to fracture the reservoir rock. The fractures allow for increased flow of oil and/or gas from the reservoir into the well. Water used in the hydraulic fracturing fluid is typically obtained from sources in the vicinity of the well. Water that naturally occurs in the oil and gas reservoir rocks often flows into the production well and through the well to the surface as a byproduct of the oil and gas production process. This byproduct water, commonly referred to as produced water, requires handling and management.

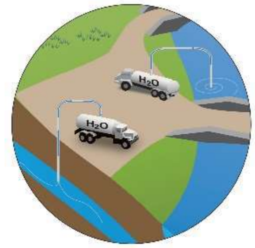
Many well site and operational activities are conducted to prepare a site and well for hydraulic fracturing and oil and/or gas production. The actual hydraulic fracturing event is of relatively short duration, usually several weeks or less, but it is also a phase of work with numerous complex operational activities to handle, mix, and inject the hydraulic fracturing fluid under pressure through the production well. The injected hydraulic fracturing fluid typically contains mostly water, includes a proppant (commonly sand) which ensures that the fractures remain propped open after injection, and contains less than two percent additives (chemicals) that improve the fluid properties for fracturing. These small percentages of additives, given the total volume of hydraulic fracturing fluids, mean that a typical hydraulic fracturing job can use tens of thousands of gallons of chemicals.

Since about 2005, the combination of hydraulic fracturing and directional drilling pioneered in the Barnett Shale in Texas has become widespread in the oil and gas industry. Hydraulic fracturing combined with directional drilling is now a standard industry practice. It has significantly contributed to the surge in United States oil and gas production, and accounted for slightly more than 50% of oil production and nearly 70% of gas production in 2015. Hydraulic fracturing has resulted in expanded production from unconventional shale and so-called tight oil or gas reservoirs that had previously been largely unused. This hydraulic fracturing-based production activity is geographically concentrated. About three-quarters of new hydraulic fracturing wells in 2011 and 2012 were located in five states (Texas, Colorado, Pennsylvania, North Dakota, and Oklahoma) with about half of all wells located in Texas.

There is no national database or complete national registry of wells that have been hydraulically fractured in the United States. Based on the data available from various commercial and public sources, we estimate that 25,000 to 30,000 new wells were drilled and hydraulically fractured in the United States annually between 2011 and 2014. In addition to these new wells, some existing wells not previously fractured were fractured, and some that had been fractured in the past were re-fractured. New drilling of hydraulic fracturing wells, influenced by oil and gas prices, peaked in the United States between 2005 and 2008 for gas and between 2011 and 2014 for oil. Following price declines, the number of new hydraulically fractured wells in 2015 was about 20,000. Future drilling and production will be influenced by future gas and oil prices. Despite recent declines in prices and new drilling, oil and gas production in the United States continues at historically high levels with projections of continued growth in the medium and long term led by hydraulic fracturing-based production from unconventional reservoirs.

This page is intentionally left blank.

Chapter 4. Water Acquisition



Abstract

In this chapter, the EPA examined the potential impacts of water withdrawals for hydraulic fracturing on drinking water resource quantity and quality, and identified common factors affecting the frequency and severity of impacts. Groundwater and surface water resources used for hydraulic fracturing also currently serve or in the future may serve as drinking water sources, and water withdrawals for hydraulic fracturing can affect the quantity or quality of the remaining drinking water resource.

Hydraulic fracturing used a median of 1.5 million gallons (5.7 million liters) of water per well from 2011 through early 2013. Surface water supplies almost all water used for hydraulic fracturing in the eastern United States, whereas surface water or groundwater is used in the West. Reuse of hydraulic fracturing wastewater as a percentage of injected volume is generally low, with a median of 5% according to an EPA literature survey. Greater reuse occurs where disposal options are limited (e.g., the Marcellus Shale in Pennsylvania) and not necessarily where water availability is lowest.

Hydraulic fracturing generally uses and consumes a relatively small percentage of water when compared to total water use, water consumption, and water availability at the national, state, and county scale. There are exceptions, however. For example, EPA's analysis shows that counties in southern and western Texas have relatively high hydraulic fracturing water withdrawals and low water availability. These findings indicate where impacts are more likely to occur or be severe, but local information (i.e., at the scale of the drinking water resource) is needed to determine whether potential impacts have been realized. In some example cases (e.g., the Eagle Ford Shale in Texas, the Haynesville Shale in Louisiana), local impacts to drinking water resource quantity have occurred in areas with increased hydraulic fracturing activity. In these instances, hydraulic fracturing water withdrawals contributed to local impacts along with other water users and the lack of precipitation.

Drought or seasonal times of low water availability can increase the frequency and severity of impacts, while water management practices such as the establishment of low-flow criteria (termed passby flows), shifting from fresh to brackish water sources, or increasing the reuse of wastewater for hydraulic fracturing can help protect drinking water resources.

Uncertainty about the extent of impacts on drinking water resources stems from the lack of available data at the local scale. The EPA could assess the potential for impacts at the county scale, but often could not determine whether impacts occurred at drinking water withdrawal locations.

Overall, hydraulic fracturing uses and consumes a relatively small percentage of water at the county scale, but not always, and impacts can still occur depending on the local balance between withdrawals and availability. Regional or local-scale factors, such as drought or water management, can modify this balance to increase or decrease the frequency or severity of impacts.

This page is intentionally left blank.

4. Water Acquisition

4.1 Introduction

Water is a crucial component of nearly all hydraulic fracturing operations, generally making up 90 – 97% of the total fluid volume injected into a well (Chapter 5).[1] Ground- and surface water resources that serve as sources of water for hydraulic fracturing also provide water for public water supplies or private drinking water wells. For this reason, water withdrawals for hydraulic fracturing can impact drinking water resources by changing the quantity or quality of the remaining resource.[2] In this chapter, we consider potential impacts of water acquisition for hydraulic fracturing on both drinking water resource quantity and quality, and, where possible, identify factors that affect the frequency or severity of impacts.[3]

We define impacts broadly in this assessment to include any change in the quantity or quality of drinking water resources; see Chapter 1 for more information. This definition applies reasonably well to the subsequent chapters (Chapters 5-8); however, by this definition, even the smallest water withdrawals would be considered impacts. Recognizing this, we focus on a smaller subset of potential impacts, specifically water withdrawals that have the potential to limit the availability of drinking water or alter its quality. Whether water withdrawals have this potential depends primarily on the balance between water use and availability at the local scale.[4,5] By "local" in this chapter, we refer to the scale at which impacts to drinking water resources are expected to occur. This usually means a given surface water (e.g., river or stream) or groundwater resource (i.e., aquifer), or a given watershed where we have detailed information about local water dynamics (e.g., case studies). We note the scale at which data are available and findings are reported.

---

[1] This range is based on multiple sources that either present hydraulic fracturing fluid composition as a function of volume (e.g., 95% of the total volume injected) or as a function of mass (e.g., 90% of the total mass injected). See Chapter 5 for additional information.

[2] Surface water withdrawals can affect water quality by altering in-stream flow and decreasing the dilution of pollutants or changing water chemistry (Section 4.5.3). Groundwater withdrawals may alter water quality by inducing vertical mixing of fresh groundwater with contaminated water from the land surface or underlying formations, or by promoting changes in reduction-oxidation conditions and mobilizing chemicals from geologic sources (Section 4.5.1).

[3] Water acquired for use in other oil and gas development steps besides hydraulic fracturing is beyond the scope of this chapter, including the water used in well drilling and well pad preparation and water removal for the production of coalbed methane. Furthermore, water released to the atmosphere via gas combustion is also outside the scope of this chapter.

[4] Throughout this chapter we use the terms "water use" and "water withdrawals" interchangeably to refer to the water that is acquired for hydraulic fracturing operations.

[5] There is no standard definition for water availability, and it has not been assessed recently at the national scale (U.S. GAO, 2014). Instead, a number of water availability indicators have been suggested (e.g., Roy et al., 2005). Here, availability is most often used to qualitatively refer to the amount of a location's water that could, currently or in the future, serve as a source of drinking water (U.S. GAO, 2014), which is a function of water inputs to a hydrologic system (e.g., rain, snowmelt, groundwater recharge) and water outputs from that system occurring either naturally or through competing demands of users. Where specific numbers are presented, we note the specific water availability indicator used.

A variety of factors can modify the balance between water use and availability. For example, multiple hydraulically fractured wells require more water than a single well, making it critical to assess the cumulative effects of multiple wells over a given area or time period. Furthermore, the combined effects of multiple water users pumping from the same aquifer can compound stress on already declining groundwater supplies. Alternatively, locally high rates of hydraulic fracturing wastewater reuse may help offset the need for fresh water withdrawals. These and other factors are discussed throughout the chapter.

This chapter proceeds roughly in two halves. In the first half, we address water use and consumption by hydraulic fracturing.[1] We provide an overview of the types of water used for hydraulic fracturing (Section 4.2); the amount of water used per well (Section 4.3); and then estimates of hydraulic fracturing water use and consumption at the national, state, and county scale, both in absolute terms and relative to total water use and consumption (Section 4.4). Although most available data and literature pertain to water use, we discuss water consumption because hydraulic fracturing consumes a substantial proportion of the water it uses, so that a proportion of the water is lost from the local hydrologic cycle. See Section 4.4 and Chapter 2 for more information.

In the second half of the chapter, we assess the potential for impacts by location in certain states (and major oil and gas regions within select states) where hydraulic fracturing currently occurs (Section 4.5; Appendix B.2). For each state and region, we discuss the water used and consumed by hydraulic fracturing, and then compare it to water availability. We do this using several lines of evidence: (1) literature information (both quantitative and qualitative) on state and regional hydraulic fracturing water use and availability; (2) comparisons between our county level estimates of hydraulic fracturing water use and an index of water availability; and (3) local case studies from the Eagle Ford play in Texas, the Upper Colorado River Basin in Colorado, and the Susquehanna River Basin in Pennsylvania.[2] The use of case studies provides insight into the local, sub-county scale, where impacts are most likely to be observed in both space and time.

Overall, this chapter provides a national assessment of where potential impacts to drinking water quantity and quality are most likely due to water acquisition for hydraulic fracturing. We utilize case studies where data are available to understand local dynamics and whether impacts are indeed realized. In the absence of case studies, we use county level data to assess where potential impacts are most likely. Finally, we identify the common factors affecting the frequency and severity of impacts. We provide a synthesis of our findings in Section 4.6.

---

[1] We refer specifically to "water consumption" when data are available or it is explicitly noted in the scientific literature. However, when specific information is not available, we use "water use" or "water withdrawals" as general terms to refer to both water use and consumption by hydraulic fracturing.

[2] The EPA's *Review of Well Operator Files for Hydraulically Fractured Oil and Gas Production Wells* (i.e., the "Well File Review;" see Text Box 6-1) was originally planned to inform the water acquisition stage of the hydraulic fracturing water cycle, but did not yield any useable information on this topic, and is therefore not cited as a source of information in this chapter. Although information in some well files was of good quality, the well files generally contained insufficient or inconsistent information on nearby surface water and groundwater quality, injected water volumes, and wastewater volumes and disposition; therefore, these data were not summarized (U.S. EPA, 2015n).

4.2 Types of Water Used

The three major sources of water for hydraulic fracturing are surface water (i.e., rivers, streams, lakes, and reservoirs), groundwater, and reused hydraulic fracturing wastewater.[1,2,3] These sources often vary in their initial water quality and in how they are provisioned to hydraulic fracturing operations. In this section, we provide an overview of the sources (Section 4.2.1), water quality (Section 4.2.2), and provisioning of water (Section 4.2.3) required for hydraulic fracturing. Detailed information on the types of water used by state and locality is presented in Section 4.5.

4.2.1 Source

Whether water used in hydraulic fracturing originates from surface water or groundwater resources is largely determined by the type of locally available water sources. Water transportation costs can be high, so the industry tends to acquire water from nearby sources if available (Nicot et al., 2014; Mitchell et al., 2013a; Kargbo et al., 2010). Surface water supplies most of the water for hydraulic fracturing in the eastern United States, whereas surface water or groundwater is used in the more semi-arid to arid western states. In western states that lack available surface water resources, groundwater generally supplies the majority of water needed for fracturing (Table 4-1). Brackish sources of groundwater can be important for reducing demand on fresh groundwater resources in certain regions (e.g., the Permian Basin and Eagle Ford Shale in Texas; see Section 4.5.1).[4] Local policies also may direct the industry to seek withdrawals from designated sources (U.S. EPA, 2013a); for instance, some states have encouraged the industry to seek water withdrawals from surface water rather than groundwater due to concerns over aquifer depletion. See Section 4.5.4 and Section 4.5.5 for more information.

[1] We use the term "hydraulic fracturing wastewater" to refer to produced water that is managed using practices that include, but are not limited to, reuse in subsequent hydraulic fracturing operations, treatment and discharge, and injection into disposal wells. The term 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 (see Chapter 8 and Appendix J, the Glossary, for more detail).

[2] Throughout this chapter we sometimes refer to "reused hydraulic fracturing wastewater" as simply "reused wastewater," because this is the dominant type of wastewater reused by the industry. When referring to other types of reused wastewater not associated with hydraulic fracturing (e.g., acid mine drainage, wastewater treatment plant effluent), we specify the source of the wastewater.

[3] We use the term "reuse" regardless of the extent to which the wastewater is treated (Nicot et al., 2014); we do not distinguish between reuse and recycling except when specifically reported in the literature.

[4] We use the term "fresh water" to qualitatively refer to water with relatively low TDS that is most readily and currently available for drinking water. We do not use the term to imply an exact TDS limit.

Table 4-1. Estimated proportions of hydraulic fracturing source water from surface water and groundwater.

Location	Surface water	Groundwater	Year or time period of estimate
Louisiana—Haynesville Shale	87%[a]	13%[a]	2009 - 2012
Oklahoma—Statewide	63%[b]	37%[b]	2011
Pennsylvania—Marcellus Shale, Susquehanna River Basin	92%[c]	8%[c]	2008 - 2013
Texas—Barnett Shale	50%[d]	50%[d]	2011 - 2013
Texas—Eagle Ford Shale	10%[e]	90%[e]	2011
Texas—TX-LA-MS Salt Basin[f]	30%[e]	70%[e]	2011
Texas—Permian Basin	0%[e]	100%[e]	2011
Texas—Anadarko Basin	20%[e]	80%[e]	2011
West Virginia—Statewide, Marcellus Shale	91%[g]	9%[g]	2012

[a] Percentages calculated from fracturing supply water usage data only. Rig supply water and other sources were excluded as they fall outside the scope of this assessment. Data from October 1, 2009, to February 23, 2012, for 1,959 Haynesville Shale natural gas wells (LA Ground Water Resources Commission, 2012).

[b] Proportion of surface water and groundwater permitted in 2011 by Oklahoma's 90-day provisional temporary permits for oil and gas mining. Temporary permits make up the majority of water use permits for Oklahoma oil and gas mining (Taylor, 2012).

[c] Calculated from SRBC (2016) data from July 2008 to December 2013.

[d] Nicot et al. (2014).

[e] Nicot et al. (2012).

[f] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

[g] Estimated proportions are for 2012, the most recent estimate for a full calendar year available from West Virginia DEP (2014). Data from the West Virginia DEP show the proportion of water purchased from commercial brokers as a separate category and do not specify whether purchased water originated from surface water or groundwater. Therefore, we excluded purchased water in calculating the relative proportions of surface water and groundwater shown in Table 4-1 (West Virginia DEP, 2014).

The reuse of wastewater from past hydraulic fracturing operations reduces the need for withdrawals of fresh surface water or groundwater.[1] In a survey of literature values from 10 states, basins, or plays, we found a median of 5% of the water used in hydraulic fracturing comes from reused hydraulic fracturing wastewater, with this percentage varying by location (Table 4-2).[2,3]

---

[1] Hydraulic fracturing wastewater may be stored on-site in open pits, which may also collect rainwater and runoff water. We do not distinguish between the different types of water that are collected on-site during oil and gas operations, and assume that most of the water collected on-site at well pads is hydraulic fracturing wastewater.

[2] Throughout this chapter, we preferentially report medians where possible because medians are less sensitive to outlier values than averages. Where medians are not available, averages are reported.

[3] This chapter examines reused wastewater as a percentage of injected volume because reused wastewater may offset total fresh water acquired for hydraulic fracturing. In contrast, Chapter 8 of this assessment discusses the total percentage of the generated wastewater that is reused rather than managed by different means (e.g., disposal in Class II wells). This distinction is sometimes overlooked, which can lead to a misrepresentation of the extent to which wastewater is reused to offset total fresh water used for hydraulic fracturing.

Table 4-2. Percentage of injected water volume that comes from reused hydraulic fracturing wastewater in various states, basins, and plays.

See Section 4.5 and Appendix B.2 for additional discussion of reuse practices by state and locality and variation over time where data are available.

State, basin, or play	Estimate of the percentage of injected water volume that comes from reused hydraulic fracturing wastewater[a]	Year or time period of estimate (NA = not available)
California—Monterey Shale	13%[b]	2014
Colorado—Wattenberg Field, Denver-Julesburg Basin	0%[c]	NA
Pennsylvania—Statewide	19%[d]	2014
Pennsylvania–Marcellus Shale, Susquehanna River Basin	16%[e]	2008 – 2013
Texas—Barnett Shale	5%[f]	2011
Texas—Eagle Ford Shale	0%[f]	2011
Texas—TX-LA-MS Salt Basin[g]	5%[f]	2011
Texas—Permian Basin (far west portion)	0%[f]	2011
Texas—Permian Basin (Midland portion)	2%[f]	2011
Texas—Anadarko Basin	20%[f]	2011
West Virginia—Statewide	15%[h]	2012
Overall Mean[i]	8%	
Overall Median[j]	5%	

[a] All estimates in this table refer to the percentage of injected water volume that comes from reused hydraulic fracturing wastewater. However, different literature sources used slightly different terminology when referring to this percentage. In the table footnotes below, we reference the terminology reported in the literature source cited.

[b] Produced water as a percentage of total water volume for 480 well stimulations according to completion reports between January 1, 2014, and December 10, 2014 (CCST, 2015a). All but two of these stimulations were conducted in Kern County, California (the remaining two were completed in Ventura County, California). Well stimulations mostly consisted of hydraulic fracturing operations, but also included smaller numbers of matrix acidizing and acid fracturing operations (CCST, 2015a).

[c] Reflects an assumption of reuse practices by Noble Energy in the Wattenberg Field of Colorado's Denver-Julesburg Basin, as reported by Goodwin et al. (2014).

[d] Percentage of recycled water used in hydraulic fracturing in 2014 based on data from the Pennsylvania Bureau of Topographic and Geologic Survey (Schmid and Yoxtheimer, 2015). This percentage was higher at 23% in 2013, but we present the most recent estimate available in the above table. The slight decline to 19% in 2014 may be explained by the fact that some completion reports had not yet been processed when these data were published, yet the data generally show an upward trend over time in reuse as a percentage of injected volume (Schmid and Yoxtheimer, 2015).

[e] Flowback water as a percentage of total water injected from July 2008 to December 2013 (SRBC, 2016). This percentage was 22% in 2013 alone (SRBC, 2016).

[f] Estimated percentage of recycling/reused water in 2011 (Nicot et al., 2012).

[g] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

[h] Reused fracturing water as a percentage of total water used for hydraulic fracturing in 2012, calculated from data provided by the West Virginia DEP (2014).

[i] Calculated based on the values presented in Table 4-2, excluding the value for Pennsylvania's Susquehanna River Basin to avoid double counting with the statewide value. The overall mean is not weighted by the number of wells in a given state, basin, or play.

[j] Calculated based on the values presented in Table 4-2, excluding the value for Pennsylvania's Susquehanna River Basin to avoid double counting with the statewide value. The overall median is not weighted by the number of wells in a given state, basin, or play.

Available data on reuse trends indicate increased reuse as a percentage of injected volume over time in both Pennsylvania and West Virginia, likely due to the lack of nearby disposal options in Class II injection wells regulated by the Underground Injection Control (UIC) Program (Section 4.5.3).

The reuse of wastewater for hydraulic fracturing is limited by the amount of water that returns to the surface during production (Nicot et al., 2012). In the first 10 days of well production, 5% to almost 50% of the hydraulic fracturing fluid volume can be collected, with values varying across geologic formations (Chapter 7, Table 7-1). Longer duration measurements are rare, but between 10% and 30% of the hydraulic fracturing fluid volume has been collected in the Marcellus Shale in Pennsylvania over nine years of production, while over 100% has been collected in the Barnett Shale in Texas over six years of production (Chapter 7, Table 7-2).[1] Assuming that 10% of hydraulic fracturing fluid volume is collected in the first 30 days and 100% of the wastewater is reused, it would take 10 wells to produce enough water to hydraulically fracture a new well. As more wells are hydraulically fractured in a given area, the potential for wastewater reuse increases.

The decision to reuse hydraulic fracturing wastewater appears to be driven by economics and the quality of the wastewater, and not concerns over local water availability (Section 4.2.2). Water transportation costs (i.e., trucking, piping), the availability of Class II wells, and local regulations can play a role in determining whether hydraulic fracturing wastewater is reused to offset the need for fresh water withdrawals (Schmid and Yoxtheimer, 2015). Besides hydraulic fracturing wastewater, other wastewaters may be reclaimed for use in hydraulic fracturing. These include acid mine drainage, wastewater treatment plant effluent, and other sources of industrial and municipal wastewater (Nicot et al., 2014; Ziemkiewicz et al., 2013). Limited information is available on the extent to which these other wastewaters are used.

4.2.2 Quality

Water quality is an important consideration when sourcing water for hydraulic fracturing. Fresh water is most often used to maximize hydraulic fracturing fluid performance and to ensure compatibility with the geologic formation being fractured. This finding is supported by the EPA's analysis of disclosures to the FracFocus Chemical Disclosure Registry (version 1.0; hereafter, the EPA FracFocus report) (U.S. EPA, 2015b), as well as by regional analyses from Texas (Nicot et al.,

[1] It is possible to collect over 100% of the hydraulic fracturing fluid volume because water from the formation returns to the surface along with the injected water.

2012) and the Marcellus Shale (Mitchell et al., 2013a).[1,2] Fresh water was the most commonly cited water source by companies included in an analysis of nine hydraulic fracturing service companies on their operations from 2005 to 2010 (U.S. EPA, 2013a). Three service companies noted that the majority of their water was fresh, because it required minimal testing and treatment (U.S. EPA, 2013a).[3] The majority of the nine service companies recommended testing for certain water quality parameters (pH and maximum concentrations of specific cations and anions) in order to ensure compatibility among the water, other fracturing fluid constituents, and the geologic formation (U.S. EPA, 2013a).

The reuse of hydraulic fracturing wastewater may be limited to an extent by water quality. Over the production life of a well, the quality of the wastewater produced begins to resemble the quality of the water naturally found in the geologic formation and may be characterized by high concentrations of total dissolved solids (TDS) (Goodwin et al., 2014). High concentrations of TDS and other individual dissolved constituents in wastewater, including specific cations (calcium, magnesium, iron, barium, strontium), anions (chloride, bicarbonate, phosphate, and sulfate), and microbial agents, can interfere with hydraulic fracturing fluid performance by producing scale in the borehole or by interfering with certain additives in the hydraulic fracturing fluid (e.g., high TDS may inhibit the effectiveness of friction reducers) (Gregory et al., 2011; North Dakota State Water Commission, 2010). Due to these limitations, wastewater can require treatment or blending with fresh water to meet the level of water quality desired in the hydraulic fracturing fluid formulation.[4]

Options for treating hydraulic fracturing wastewater to facilitate reuse are available and being used by the industry in some cases. For example, filter socks, centrifuge, dissolved air flotation, or settling technologies can remove suspended solids, and physical/chemical precipitation or electrocoagulation can remove dissolved metals (Schmid and Yoxtheimer, 2015). For more information on treatment of hydraulic fracturing wastewater, see Chapter 8.

[1] The FracFocus Chemical Disclosure Registry (often referred to as FracFocus; www.fracfocus.org) is a national hydraulic fracturing chemical disclosure registry managed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission. FracFocus was created to provide the public access to reported chemicals used for hydraulic fracturing within their area. It was originally established in 2011 (version 1.0) for voluntary reporting by participating oil and gas well operators. Six of the 20 states discussed in this assessment required disclosure to FracFocus at various points between January 1, 2011, and February 28, 2013, the time period analyzed by the EPA; another three of the 20 states offered the choice of reporting to FracFocus or the state during this same time period (see Appendix Table B-5 for states and disclosure start dates) (U.S. EPA, 2015b).

[2] Of all disclosures reviewed that indicated a source of water for the hydraulic fracturing base fluid, 68% listed "fresh" as the only source of water used. Note, 29% of all disclosures considered in the EPA's FracFocus report included information on the source of water used for the base fluid (U.S. EPA, 2015b).

[3] Service companies did not provide data on the percentage of fresh water versus non-fresh water used for hydraulic fracturing (U.S. EPA, 2013a).

[4] The EPA FracFocus report suggests that fresh water makes up the largest proportion of the base fluid when blended with water sources of lesser quality (U.S. EPA, 2015b).

4.2.3 Provisioning

Water for hydraulic fracturing is typically either self-supplied by the industry or purchased from public water systems.[1] Self-supplied water for fracturing generally refers to permitted direct withdrawals from surface water or groundwater or the reuse of wastewater. Nationally, self-supplied water is more common, although there is much regional variation (U.S. EPA, 2015b; CCST, 2014; Mitchell et al., 2013a; Nicot et al., 2012). Water purchased from municipal public water systems can be provided either before or after treatment (Nicot et al., 2014). Water for hydraulic fracturing is also sometimes purchased from smaller private entities, such as local land owners (Nicot et al., 2014).

4.3 Water Use Per Well

In this section, we provide an overview of the amount of water used per well during hydraulic fracturing. We discuss water use in the life cycle of oil and gas operations (Section 4.3.1) and national per well estimates and associated variability (Section 4.3.2). More detailed locality-specific information on water use per well is provided in Section 4.5.

4.3.1 Hydraulic Fracturing Water Use in the Life Cycle of Oil and Gas

Water is needed throughout the life cycle of oil and gas production and use, including both at the well for processes such as well pad preparation, drilling, and fracturing (i.e., the upstream portion), and later for end uses such as electricity generation, home heating, or transportation (i.e., the downstream portion) (Jiang et al., 2014; Laurenzi and Jersey, 2013). Most of the upstream water usage and consumption occurs during hydraulic fracturing (Jiang et al., 2014; Clark et al., 2013; Laurenzi and Jersey, 2013).[2] Water use per well estimates in this chapter focus on hydraulic fracturing in the upstream portion of the oil and gas life cycle, as the downstream portion of the lifecycle is outside the scope of this assessment.[3]

[1] According to Section 1401(4) of the Safe Drinking Water Act, a public water system is defined as system that provides water for human consumption from surface water or groundwater through pipes or other infrastructure to at least 15 service connections, or an average of at least 25 people, for at least 60 days per year. Public water systems may either be publicly or privately owned.

[2] Laurenzi and Jersey (2013) reported that hydraulic fracturing accounted for 91% of upstream water consumption, based on industry data for 29 wells in the Marcellus Shale. (91% was calculated from their paper by dividing hydraulic fracturing fresh water consumption (13.7 gal (51.9 L)/Megawatt-hour (MWh)) by total upstream fresh water consumption (15.0 gal (56.8 L)/MWh) and multiplying by 100). Similarly, Jiang et al. (2014) reported that 86% of water consumption occurred at the fracturing stage for the Marcellus Shale, based on Pennsylvania Department of Environmental Protection (PA DEP) data on 500 wells. The remaining water was used in several upstream processes (e.g., well pad preparation, well drilling, road transportation to and from the wellhead, and well closure once production ended). Clark et al. (2013) estimated lower percentages (30%–80%) of water use at the fracturing stage for multiple formations. Although their estimates for the fraction of water used at the fracturing stage may be low due to their higher estimates for transportation and processing, the estimates by Clark et al. (2013) similarly illustrate the importance of the hydraulic fracturing stage in water use, particularly in terms of the upstream portion of the life cycle.

[3] When the full life cycle of oil and gas production and use is considered (i.e., both upstream and downstream water use), most water is used and consumed downstream. For example, in a life cycle analysis of hydraulically fractured gas used for electricity generation, Laurenzi and Jersey (2013) reported that only 6.7% of water consumption occurred upstream (15.0 gal (56.8 L)/MWh), while 93.3% of fresh water consumption occurred downstream for power plant cooling via

4.3.2 National Estimates and Variability in Water Use Per Well for Hydraulic Fracturing

At its most basic level, the volume of water used per well for hydraulic fracturing equals the concentration of water in the hydraulic fracturing fluid multiplied by the total volume of the fluid injected. In turn, the total volume of fluid injected generally equals the volume of fluid in the fractures, plus the volume of the well itself, plus any fluid lost due to "leakoff" or other unintended losses.[1]

Nationally, most operators employ fracturing fluids with water as a base fluid, meaning the concentration of water in the fluid is high (U.S. EPA, 2015b; Yang et al., 2013; GWPC and ALL Consulting, 2009). The EPA inferred that more than 93% of reported disclosures to FracFocus used water as a base fluid (U.S. EPA, 2015b). The median reported concentration of water in the hydraulic fracturing fluid was 88% by mass, with 10th and 90th percentiles of 77% and 95%, respectively. Only roughly 2% of disclosures (761 wells) reported the use of non-aqueous substances as base fluids, typically either liquid-gas mixtures of nitrogen or carbon dioxide. Both of these formulations still contained substantial amounts of water, as water made up roughly 60% (median value) of the fluid in them (U.S. EPA, 2015b). Other formulations were rarely reported. Fluid formulations are discussed further in Chapter 5.

On average, hydraulic fracturing requires more than a million gallons (3.8 million liters) of water per well. Jackson et al. (2015) reported a national average of 2.4 million gal (9.1 million L) of water per well, calculated from FracFocus disclosures between 2010 and 2013. According to the EPA's project database of disclosures to FracFocus 1.0 (hereafter the EPA FracFocus 1.0 project database), the median volume of water used per well was 1.5 million gal (5.7 million L) between 2011 and early 2013, based on 37,796 disclosures nationally (U.S. EPA, 2015b, c).[2] Data on reported Information Handling Services well numbers and median volumes in Gallegos et al. (2015) show that overall per well volumes have increased in recent years from approximately 1.5 million gal (5.7 million L) in 2011 to 2.7 million gal (10.2 million L) in 2014.[3]

The recent increase in water use per well has been driven primarily by the proportional increase in horizontal wells (Gallegos et al., 2015) (Figure 4-1). Increases in horizontal well length affect total volumes injected primarily by allowing a larger fracture volume to be stimulated (Economides et al., 2013). As horizontal wells get longer, fracture, well, and total volumes all increase. Importantly, increases in the well length and water use per well do not necessarily mean an increase in water intensity (the amount of water used per unit energy extracted). Goodwin et al. (2014) found water

evaporation (209.0 gal (791.2 L)/MWh). Similar results were found for gas extraction in the Eagle Ford Shale (Scanlon et al., 2014b).

[1] Leakoff is the fraction of the hydraulic fracturing fluid that infiltrates into the formation (e.g., through an existing natural fissure) and is not recovered during production. This water lost to the formation can be a substantial fraction of the water injected (O'Malley et al., 2015). See Chapter 6 for more information about leakoff and some recent findings related to the relationship between hydraulic fracturing fluid volume and fracture volume.

[2] All water use data included in the EPA's FracFocus 1.0 project database were obtained from disclosures made to FracFocus. Although disclosures were made on a per well basis, a small proportion of the wells were associated with more than one disclosure (i.e., 876 out of 37,114, based on unique API numbers) (U.S. EPA, 2015c). For the purposes of this chapter, we discuss water use per disclosure in terms of water use per well.

[3] Derived from supporting information in Gallegos et al. (2015). Calculated by multiplying the median volume by the number of wells for each well type, then summing volumes across well types, and dividing by the total number of wells.

intensity did not increase in the Denver Basin despite increases in well length and water use per well.

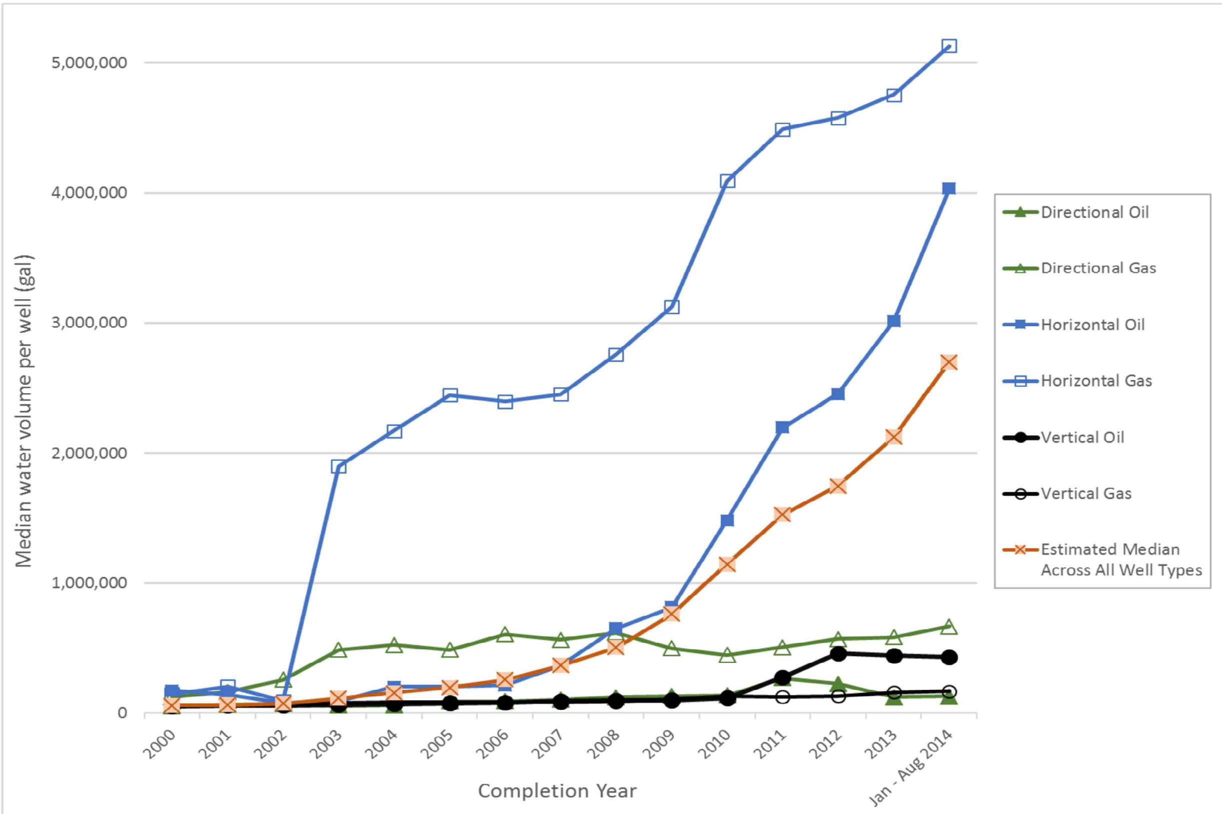


Figure 4-1. Median water volume per hydraulically fractured well nationally, expressed by well type and completion year.

Adapted using data from Gallegos et al. (2015). Note: shown in orange is the estimated median across all well types, derived from Gallegos et al. (2015) supporting information Tables S2 and S3. Calculated by multiplying the median volume by the number of wells for each well type, then summing volumes across well types, and dividing by the total number of wells for each year. This estimated median across all well types reflects the central tendency of the data, and was calculated because the individual data are proprietary and not published, preventing the calculation of an overall median.

There is substantial variation around these per well estimates. For instance, the 10th and 90th percentiles from the EPA FracFocus 1.0 project database are 74,000 gal and 6 million gal (280,000 L and 23 million L) per well, respectively.[1] Even in specific basins, plays, and within a single oil and gas field, water use per well varies widely. For example, Laurenzi and Jersey (2013) reported volumes ranging from 1 to 6 million gal (3.8 to 23 million L) per well (10th to 90th percentile) in the Wattenberg Field in Colorado.

Of the major unconventional formation types discussed in Chapter 2 (shales, tight formations-including tight sands or sandstones, and coalbeds), coalbeds generally require less water per well.

[1] Although the EPA FracFocus report shows 5th and 95th percentiles, we report 10th and 90th percentiles throughout this chapter to further reduce the influence of outliers.

Coalbed methane (CBM) comes from coal seams that often have a high initial water content and tend to occur at much shallower depths (U.S. EPA, 2015k). In part because of the shallower depths, shorter well lengths result in lower water use per well, often by an order of magnitude or more compared to operations in shales or tight formations (e.g., Murray, 2013).

4.4 Hydraulic Fracturing Water Use and Consumption at the National, State, and County Scale

In this section, we provide an overview of water use and consumption for hydraulic fracturing at the national, state, and county scale. We then compare these values to total water use and consumption at these scales. We do this to contextualize hydraulic fracturing water use and consumption with total water use and consumption, and to illustrate whether hydraulic fracturing is a relatively large or small user and consumer of water at these scales. Later, we compare hydraulic fracturing water use to water availability estimates at the county scale (Text Box 4-2).

Water use is water withdrawn for a specific purpose, part or all of which may be returned to the local hydrologic cycle. Water consumption is water that is removed from the local hydrologic cycle following its use (e.g., via evaporation, transpiration, incorporation into products or crops, consumption by humans or livestock), and is therefore unavailable to other water users (Maupin et al., 2014). Hydraulic fracturing water consumption can occur through evaporation from storage ponds, the retention of water in the subsurface through imbibition, or disposal in Class II wells, among other means.

Hydraulic fracturing water use is a function of the water use per well and the total number of wells fractured at a given spatial scale during the time period analyzed, calculated from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). Water consumption estimates are derived from United States Geological Survey (USGS) water use data, and therefore both use and consumption are presented with the published water use numbers being first.

4.4.1 National and State Scale

Hydraulic fracturing uses and consumes billions of gallons of water each year in the United States, but at the national and state scales, it is a relatively small user and consumer of water compared to total water use and consumption. According to the EPA's FracFocus 1.0 project database, hydraulic fracturing used 36 billion gal (136 billion L) of water in 2011 and 52 billion gal (197 billion L) in 2012, yielding an average annually of 44 billion gal (167 billion L) of water in 2011 and 2012 across all 20 states in the project database (U.S. EPA, 2015b, c). National water use for hydraulic fracturing can also be estimated by multiplying the water use per well by the number of wells hydraulically fractured. If the median water use per well (1.5 million gal) (5.7 million L) from the EPA's FracFocus 1.0 project database is multiplied by 25,000 to 30,000 wells fractured annually (Chapter 3), national water use for hydraulic fracturing is estimated to range from 38 to 45 billion gal (142 to 170 billion L) annually. Other calculated estimates have ranged higher than this, including estimates of approximately 80 billion gal (300 billion L) (Vengosh et al., 2014) and 50 to 72 billion gal (190-273 billion L) (U.S. EPA, 2015e). These estimates are higher due to differences in the estimated water use per well and the number of wells used as multipliers. For example, Vengosh et al. (2014) derived the estimate of approximately 80 billion gal (300 billion L) by multiplying an average of 4.0 million gal (15 million L) per well (estimated for shale gas wells) by 20,000 wells (the approximate total number of fractured wells in 2012).[1]

All of these estimates of water use for hydraulic fracturing are small relative to total water use and consumption at the national scale. The USGS compiles national water use estimates every five years in the National Water Census, with the most recent census conducted in 2010 (Maupin et al., 2014).[2] The USGS publishes water use, not consumption estimates, yet by applying consumption factors for each use category in the 2010 National Water Census, we derived estimates of total water consumption. We also used a consumption factor to estimate hydraulic fracturing water consumption from values in the EPA FracFocus 1.0 project database.[3] Comparing these estimates, average annual hydraulic fracturing water use in 2011 and 2012 was less than 1% of total 2010 annual water use for all of the 20 states combined where operators reported water use to FracFocus in 2011 and 2012. Hydraulic fracturing water consumption followed the same pattern when compared to total water consumption (Appendix Table B-1).[4]

At the state scale, hydraulic fracturing also generally uses billions of gallons of water, but accounts for a low percentage of total water use or consumption. Of all states in the EPA FracFocus 1.0 project database, operators in Texas used the most water (47% of water use reported in the EPA FracFocus 1.0 project database) (U.S. EPA, 2015c) (Appendix Table B-1). This was due to the large number of wells in that state, since hydraulic fracturing water use is proportional to the number of wells. Over 94% of reported water use occurred in just seven of the 20 states in the EPA FracFocus 1.0 project database (listed in order of highest statewide hydraulic fracturing water use): Texas, Pennsylvania, Arkansas, Colorado, Oklahoma, Louisiana, and North Dakota (U.S. EPA, 2015c) (Appendix Table B-1). Hydraulic fracturing is a small percentage when compared to total water use (<1%) and consumption (<3%) in each individual state (Appendix Table B-1). Other studies have shown similar results, with hydraulic fracturing water use and consumption ranging from less than

[1] This could result in an overestimation because the estimate of 20,000 wells was derived in part from FracFocus, and these wells are not necessarily specific to shale gas; they may include other types of wells that use less water (e.g., CBM). The estimate of 1.5 million gal (5.7 million L) per well based on the U.S. EPA (2015c) FracFocus 1.0 project database likely leads to a more robust estimate when used to calculate national water use for hydraulic fracturing because it includes wells from multiple formation types (i.e., shale, tight sand, and CBM), some of which use less water than shale gas wells on average.

[2] The National Water Census includes uses such as public supply, irrigation, livestock, aquaculture, thermoelectric power, industrial, and mining at the national, state, and county scale. The 2010 National Water Census included hydraulic fracturing water use in the mining category; there was no designated category for hydraulic fracturing alone.

[3] See footnotes for Appendix Table B-1 or for Table 4-3 for a description of the consumption estimate calculations.

[4] Water use percentages were calculated by averaging annual water use for hydraulic fracturing in 2011 and 2012 for a given state or county (U.S. EPA, 2015c), and then dividing by 2010 USGS total water use (Maupin et al., 2014) and multiplying by 100. Note, the annual hydraulic fracturing water use reported in FracFocus was not added to the 2010 total USGS water use value in the denominator, and is simply expressed as a percentage compared to 2010 total water use or consumption. This was done because of the difference in years between the two datasets, and because the USGS 2010 Water Census (Maupin et al., 2014) included hydraulic fracturing water use estimates in their mining category. This approach is consistent with that of other literature on this topic; see Nicot and Scanlon (2012). Consumption estimates were calculated in the same manner, except consumption, not use, values were employed. County level data from the USGS 2010 Water Census are available online at http://water.usgs.gov/watuse/data/2010/ (accessed November 11, 2014).

1% of total use in West Virginia (West Virginia DEP, 2013), Colorado (Colorado Division of Water Resources et al., 2014), and Texas (Nicot et al., 2014; Nicot and Scanlon, 2012), to approximately 4% in North Dakota (North Dakota State Water Commission, 2014).

4.4.2 County Scale

Water use and consumption for hydraulic fracturing is also relatively small in most, but not all, counties in the United States (Table 4-3; Figure 4-2; Figure 4-3a,b; and Appendix Table B-2). Based on the EPA FracFocus 1.0 project database, reported fracturing water use in 2011 and 2012 was less than 1% compared to 2010 USGS total water use in 299 of the 401 reporting counties (Figure 4-3a; Appendix Table B-2). However, hydraulic fracturing water use was 10% or more compared to total water use in 26 counties, 30% or more in nine counties, and 50% or more in four counties (Table 4-3; Figure 4-3a). McMullen County in Texas had the highest percentage at over 100% compared to 2010 total water use.[1] Total consumption estimates followed the same pattern, but with more counties in the higher percentage categories (hydraulic fracturing water consumption was 10% or more compared to total water consumption in 53 counties; 30% or more in 25 counties; 50% or more in 16 counties; and over 100% in four counties) (Table 4-3; Figure 4-3b).

Estimates based on the EPA's FracFocus 1.0 project database may form an incomplete picture of hydraulic fracturing water use in a given state or county, because the majority of states with data in the project database did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b). We conclude that this likely does not substantially alter the overall patterns observed in Figure 4-3a,b. See Text Box 4-1 for further details. These percentages also depend both upon the absolute water use and consumption for hydraulic fracturing and the relative magnitude of other water uses and consumption in that state or county. For instance, a rural county with a small population might have relatively low total water use prior to hydraulic fracturing.[2] Also, just because water is used in a certain county does not necessarily mean it originated in that county. The cost of trucking water can be substantial (Slutz et al., 2012), and the industry tends to acquire water from nearby sources when possible (Section 4.2.1); however, water can also be piped in from more distant, regional supplies. Despite these caveats, it is clear that hydraulic fracturing is generally a relatively small user (and consumer) of water at the county level, with the exception of a small number of counties where water use and consumption for fracturing can be high relative to other uses and consumption.

[1] Estimates of use or consumption exceeded 100% when hydraulic fracturing water use averaged for 2011 and 2012 exceeded total water use or consumption in that county in 2010.

[2] For example, McMullen County, Texas, mentioned above contains a small number of residents (707 people in 2010, according to the U.S. Census Bureau (2014).

Table 4-3. Average annual hydraulic fracturing water use and consumption in 2011 and 2012 compared to total annual water use and consumption in 2010, by county.

Only counties where hydraulic fracturing water was 10% or greater compared to 2010 total water use are shown (for full table, see Appendix Table B-2). Average annual hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus 1.0 project database (U.S. EPA, 2015c). Total annual water use data in 2010 from the USGS (Maupin et al., 2014). States listed by order of appearance in the chapter.

State	County	Total annual water use in 2010 (millions of gal)[a]	Average annual hydraulic fracturing water use in 2011 and 2012 (millions of gal)[b]	Hydraulic fracturing water use compared to total water use (%)[c]	Hydraulic fracturing water consumption compared to total water consumption (%)[c,d]
Texas	McMullen	657.0	745.9	113.5	350.4
	Karnes	1861.5	1055.2	56.7	120.1
	La Salle	2474.7	1288.7	52.1	93.7
	Dimmit	4073.4	1794.2	44.0	81.3
	Irion	1335.9	411.4	30.8	74.5
	Montague	3989.5	925.3	23.2	77.8
	De Witt	2394.4	546.6	22.8	48.6
	Loving	781.1	138.4	17.7	94.1
	San Augustine	1131.5	182.1	16.1	50.8
	Live Oak	1916.3	294.0	15.3	40.1
	Wheeler	6522.6	858.0	13.2	21.5
	Cooke	4533.3	454.3	10.0	29.9
Pennsylvania	Susquehanna	1617.0	751.3	46.5	123.4
	Sullivan	222.7	66.5	29.9	79.8
	Bradford	4354.5	1059.4	24.3	78.2
	Tioga	2909.1	566.3	19.5	47.3
	Lycoming	5854.6	704.6	12.0	33.8
West Virginia	Doddridge	405.2	78.5	19.4	69.4
Ohio	Carroll	1127.9	152.7	13.5	37.3
North Dakota	Mountrail	1248.3	449.4	36.0	98.3
	Dunn	1076.8	309.5	28.7	43.1
	Burke	394.2	63.6	16.1	40.8
	Divide	806.7	102.2	12.7	18.6

State	County	Total annual water use in 2010 (millions of gal)[a]	Average annual hydraulic fracturing water use in 2011 and 2012 (millions of gal)[b]	Hydraulic fracturing water use compared to total water use (%)[c]	Hydraulic fracturing water consumption compared to total water consumption (%)[c,d]
Arkansas	Van Buren	1587.8	899.6	56.7	168.8
Louisiana	Red River	1606.0	569.6	35.5	83.2
	Sabine	1522.1	395.2	26.0	76.6

[a] County level data accessed from the USGS website (http://water.usgs.gov/watuse/data/2010/) on November 11, 2014. Total water withdrawals per day were multiplied by 365 days to estimate total water use for the year (Maupin et al., 2014).

[b] Average of water used for hydraulic fracturing in 2011 and 2012 calculated from the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

[c] Percentages were calculated by averaging annual water use for hydraulic fracturing reported in FracFocus in 2011 and 2012 for a given state or county (U.S. EPA, 2015c), and then dividing by 2010 USGS total water use (Maupin et al., 2014) and multiplying by 100.

[d] Consumption values were calculated with use-specific consumption rates predominantly from the USGS, including 19.2% for public supply, 19.2% for domestic use, 60.7% for irrigation, 60.7% for livestock, 14.8% for industrial uses, 14.8% for mining (Solley et al., 1998), and 2.7% for thermoelectric power (Diehl and Harris, 2014). We used rates of 71.6% for aquaculture from Verdegem and Bosma (2009) ((evaporation per kg fish + infiltration per kg)/total water use per kg); and 82.5% for hydraulic fracturing (consumption value calculated by taking the median value for all reported produced water/injected water percentages in Tables 7-1 and 7-2 of this assessment and then subtracting from 100%). If a range of values was given, the midpoint was used. Note, this aspect of consumption is likely a low estimate since much of this produced water (injected water returning to the surface) is not subsequently treated and reused, but rather disposed of in Class II wells – see Chapter 8.

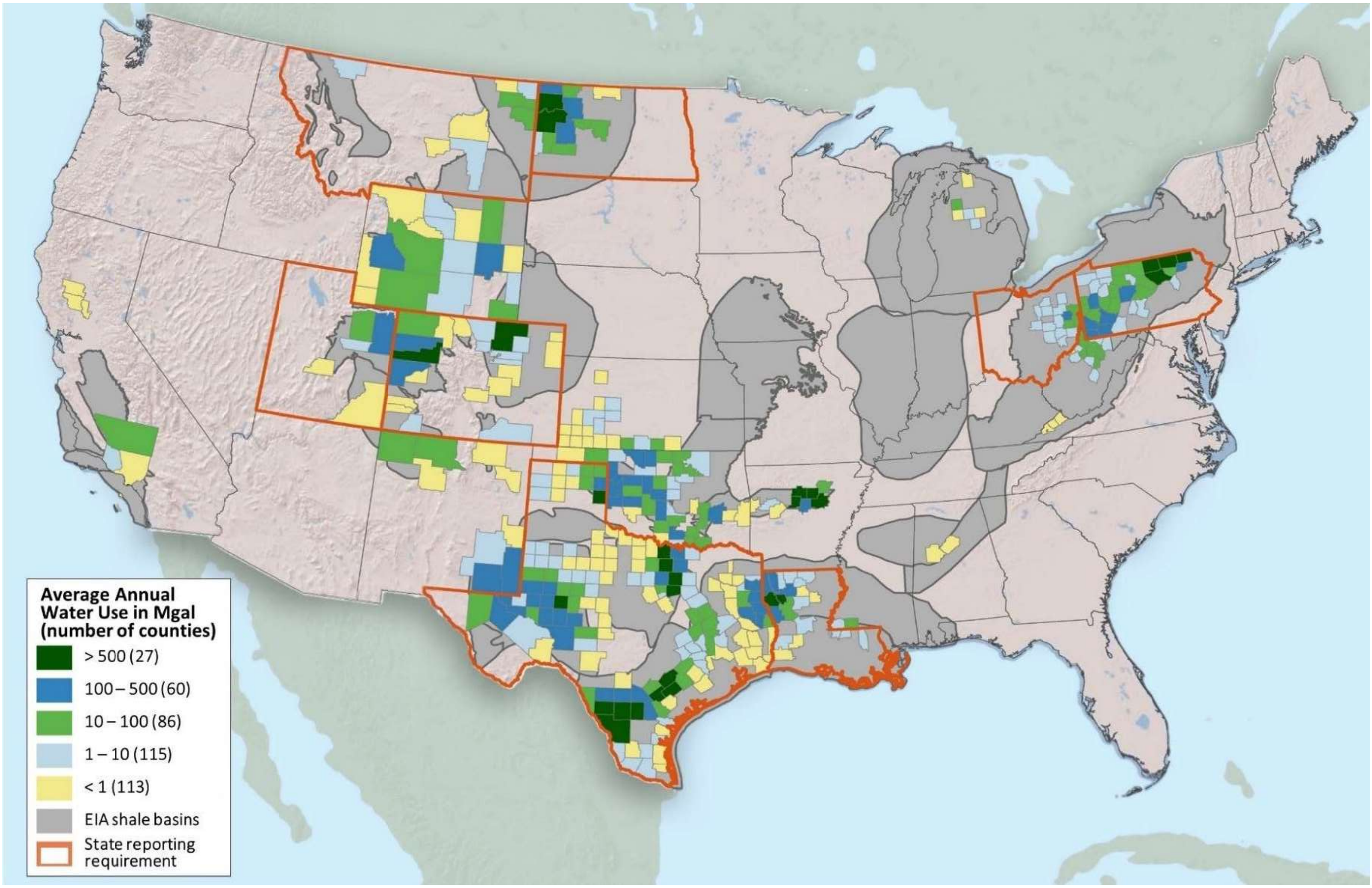
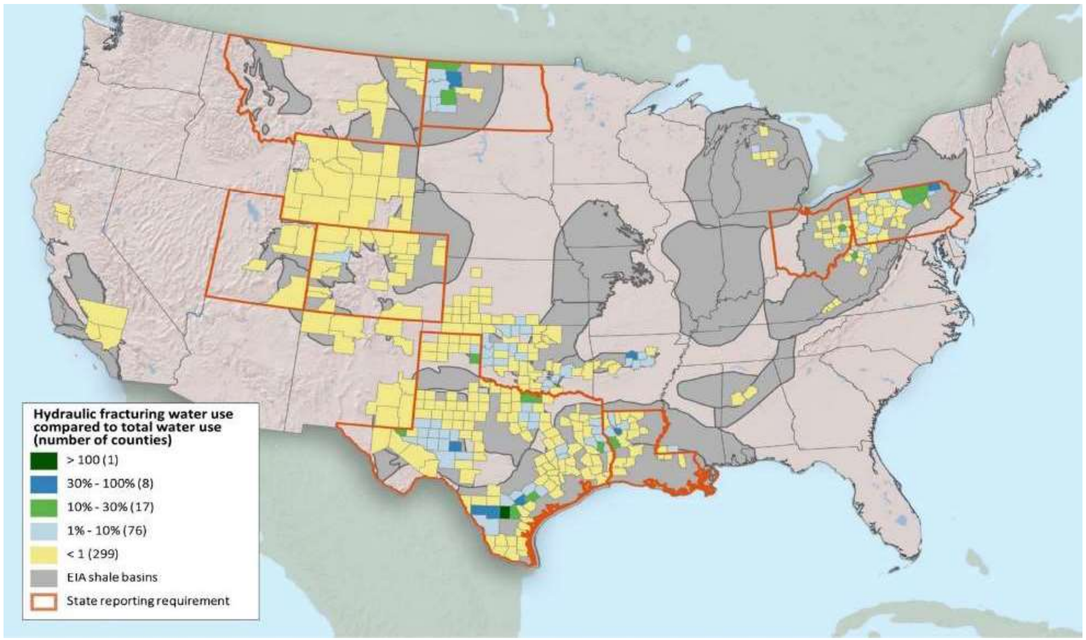
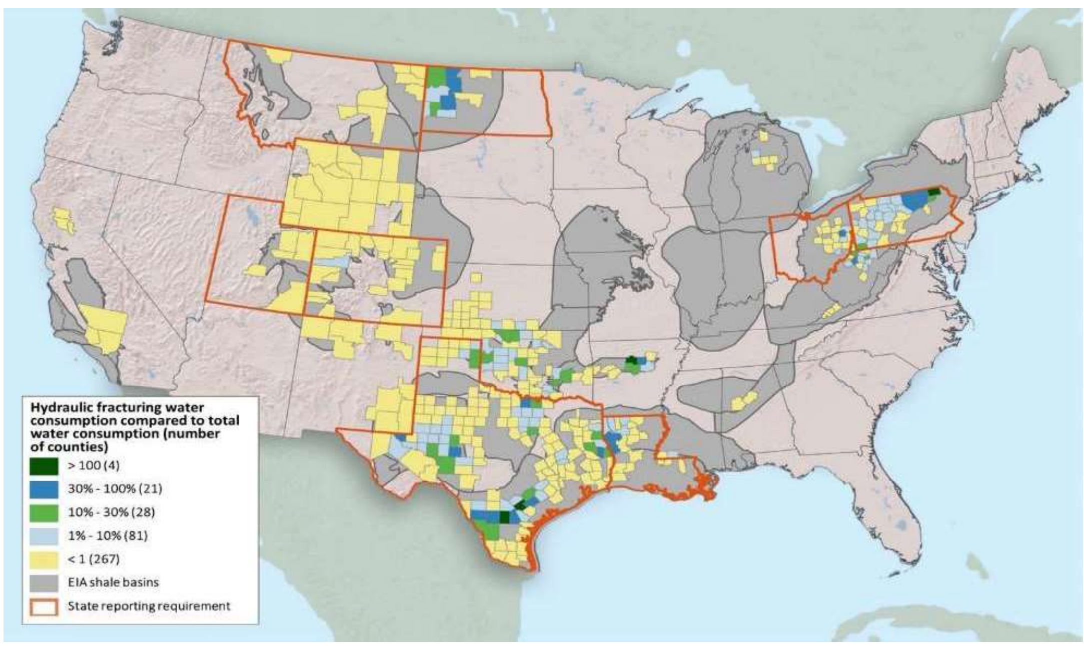


Figure 4-2. Average annual hydraulic fracturing water use in 2011 and 2012 by county.

Source: U.S. EPA (2015c). Water use in millions of gallons (Mgal). Counties shown with respect to major U.S. Energy Information Administration (EIA) shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012.



(a)



(b)

Figure 4-3. (a) Average annual hydraulic fracturing water use in 2011 and 2012 compared to total annual water use in 2010, by county, expressed as a percentage; (b) Average annual hydraulic fracturing water consumption in 2011 and 2012 compared to total annual water consumption in 2010, by county, expressed as a percentage.

Average annual hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus 1.0 project database (U.S. EPA, 2015c). Total annual water use data in 2010 from the USGS (Maupin et al., 2014). See Table 4-3 for descriptions of calculations for estimating consumption. Counties shown with respect to major U.S. EIA shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012. Note: Values over 100% denote counties where the average annual hydraulic fracturing water use or consumption in 2011 and 2012 exceeded the total annual water use or consumption in that county in 2010.

Text Box 4-1. Using the EPA's FracFocus 1.0 Project Database to Estimate Water Use for Hydraulic Fracturing.

The FracFocus Chemical Disclosure Registry (often referred to as FracFocus; www.fracfocus.org) is a national hydraulic fracturing chemical disclosure registry managed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission. FracFocus was created to provide the public access to reported chemicals used for hydraulic fracturing within their area. It was originally established in 2011 (version 1.0) for voluntary reporting by oil and gas well operators. The EPA used the data available from FracFocus between January 1, 2011 and February 28, 2013 to develop the EPA FracFocus 1.0 project database; the database and a related EPA report were both peer reviewed and published (U.S. EPA, 2015b, c). Six of the 20 states discussed in this assessment required disclosure to FracFocus at various points during this time; another three of the 20 states offered the choice of reporting to FracFocus or the state during this same time period (U.S. EPA, 2015b). Estimates based on the EPA's FracFocus 1.0 project database could form an incomplete picture of hydraulic fracturing water use, because most states with data in the project database (14 out of 20) did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b).

Water use for fracturing is a function of the water use per well and the total number of wells fractured over a given spatial area or a given period of time. For water use per well, we found seven literature values for comparison with values from the EPA's FracFocus 1.0 project database. On average, water use estimates per well in the project database were 77% of literature values (the median was 86%); Colorado's Denver Basin was the only location where the project database estimate as a percentage of the literature estimate was low (14%) (Appendix Table B-3). In general, water use per well estimates from the EPA's FracFocus 1.0 project database appear to align closely with the literature estimates for most areas for which we have data, with the exception of the Denver Basin of Colorado.

For the number of wells, we compared data in the EPA's FracFocus 1.0 project database to numbers available in state databases from North Dakota, Pennsylvania, and West Virginia (Appendix Table B-4). These were the state databases from which we could distinguish hydraulically fractured wells from other oil and gas wells. On average, we found that the EPA FracFocus 1.0 project database included 67% of the wells listed in state databases for 2011 and 2012 (Appendix Table B-4). Unlike North Dakota and Pennsylvania, West Virginia did not require operators to report fractured wells to FracFocus during this time period, possibly explaining its lower reporting rate. Multiplying the average EPA FracFocus 1.0 project database values of 77% for water use per well and 67% for well counts yields 52%. Thus, the EPA FracFocus 1.0 project database estimates for water use could be slightly over half of the estimates from these three state databases during this time period. These values are based on small sample sizes (seven literature values and three state databases) and should be interpreted with caution. Nevertheless, these numbers suggest that estimates based on the EPA's FracFocus 1.0 project database likely form an incomplete picture of hydraulic fracturing water use during this time period.

To assess how this might affect hydraulic fracturing water use estimates in this chapter, we doubled the water use value in the EPA's FracFocus 1.0 project database for each county, an adjustment much higher than any likely underestimation. Even with this adjustment, fracturing water use was still less than 1% compared to 2010 total water use in the majority of the 401 U.S. counties represented in the EPA FracFocus 1.0 project database (299 counties without adjustment versus 280 counties with adjustment). The number of counties where hydraulic fracturing water use was 30% or more of 2010 total county water use increased from nine to 21 with the adjustment.

These results indicate that most counties have relatively low hydraulic fracturing water use relative to total water use, even when accounting for likely underestimates. Since consumption estimates are derived from use, these will also follow the same pattern. Thus, potential underestimates based on the EPA's FracFocus 1.0 project database likely do not substantially alter the overall pattern shown in Figure 4-3. Rather, underestimates of hydraulic fracturing water use would mostly affect the percentages in the small number of counties where fracturing already constitutes a higher percentage of total water use and consumption.

4.5 Potential for Impacts by Location

The potential for hydraulic fracturing water acquisition to impact drinking water availability or alter its quality depends on the balance between water withdrawals and water availability at a given location. Where water availability is high compared to the volume of water withdrawn for hydraulic fracturing, this water use can be accomodated. However, where water availability is low and hydraulic fracturing water use is high, these withdrawals are more likely to impact drinking water resources. The balance between withdrawals and availability can vary greatly by geographic location. Moreover, a combination of regional or site-specific factors can alter this balance, making impacts more or less likely, or more or less severe. For these reasons, we discuss the various factors and potential for impacts by geographic location in the following section.

We organize this discussion by state, addressing 15 states accounting for almost all disclosures reported in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c): Texas (Section 4.5.1); Colorado and Wyoming (Section 4.5.2); Pennsylvania, West Virginia, and Ohio (Section 4.5.3); North Dakota and Montana (Section 4.5.4); Arkansas and Louisiana (Section 4.5.5), Oklahoma and Kansas (Appendix B.2.1); and Utah, New Mexico, and California (Appendix B.2.2). We highlight the states that best illustrate concepts relating to the potential for impacts, or factors that affect the frequency or severity of these impacts in Section 4.5; the remaining states are discussed in Appendix B.2. Within Section 4.5 and Appendix B, we address each state in order of most hydraulically fractured wells to least, and combine states with similar geographies or activity. For certain states, we address major oil and gas regions separately (e.g., the Permian Basin in Texas). Each section describes the number of fractured wells in that state or region, the type of water used, water use per well, and water use estimates at the county scale. We then discuss the potential for impacts by comparing water use and water availability and addressing factors (e.g., drought or the amount of water reused to offset fresh water use) that might alter the frequency or severity of impacts. As noted in the chapter introduction, we use several lines of evidence to evaluate the potential for impacts and factors for each location. We use the scientific literature, county level assessments, and local case studies where available.

4.5.1 Texas

Hydraulic fracturing in Texas accounts for the bulk of the activity reported nationwide, comprising 48% of the disclosures in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c) (Figure 4-4; Appendix Table B-5). There are five major basins in Texas: the Permian, Western Gulf (includes the Eagle Ford play), Fort Worth (includes the Barnett play), TX-LA-MS Salt (includes the Haynesville play), and the Anadarko (Figure 4-5); together, these five basins contain 99% of Texas' reported wells (Appendix Table B-5).

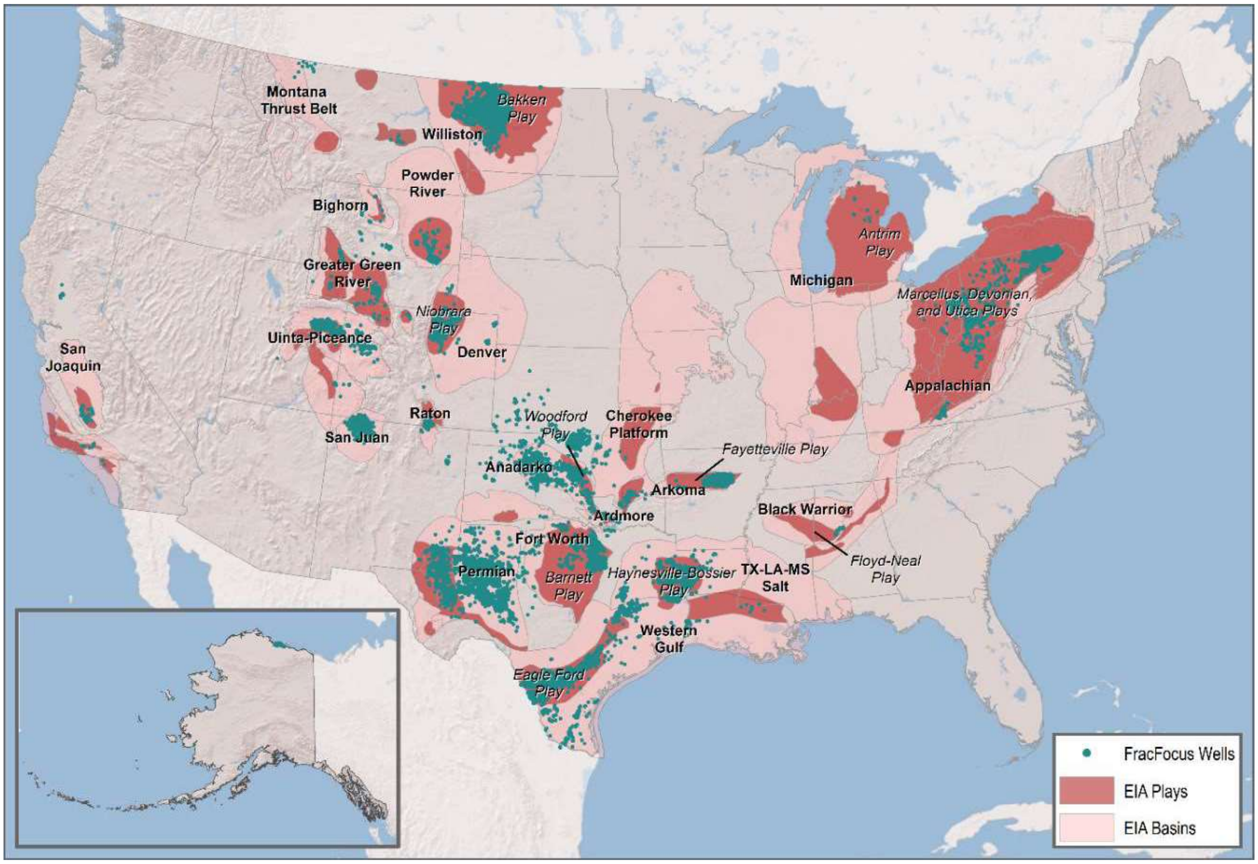


Figure 4-4. Locations of wells in the EPA FracFocus 1.0 project database, with respect to U.S. EIA shale plays and basins.

Note: Hydraulic fracturing can be conducted in geologic settings other than shale; therefore, some wells on this map are not associated with any EIA shale play or basin (EIA, 2015; U.S. EPA, 2015c).

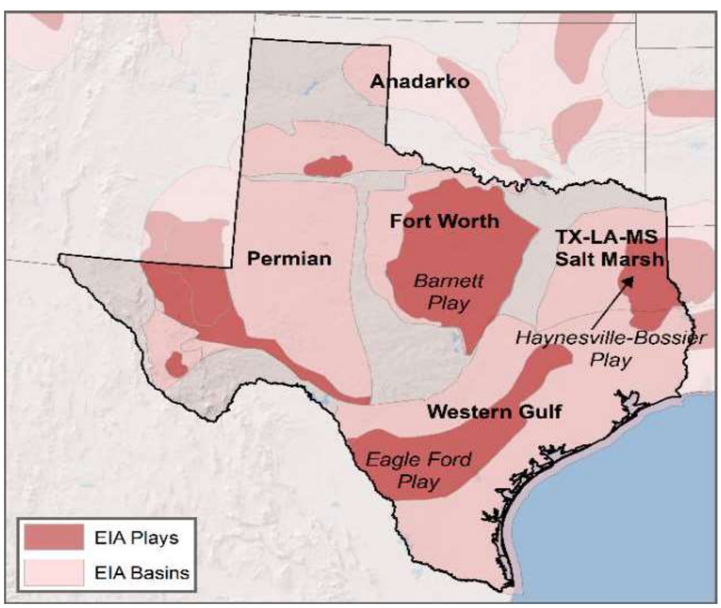


Figure 4-5. Major U.S. EIA shale plays and basins for Texas.

Source: EIA (2015).

Types of water used: What is known about water sources in Texas largely comes from direct surveys and interviews with industry operators and water suppliers (Nicot et al., 2014; Nicot et al., 2012). Overall, groundwater is the dominant source throughout most of the state (Nicot et al., 2014; Nicot et al., 2012) (Table 4-1). The exception is the Barnett Shale, where both surface water and groundwater are used in approximately equal proportions.

Hydraulic fracturing in Texas uses mostly fresh water (Nicot et al., 2012).[1] The exception is the far western portion of the Permian Basin, where brackish water makes up an estimated 80% of total hydraulic fracturing water use. Brackish water is used to a lesser extent in the Anadarko Basin, the Midland portion of the Permian Basin, and the Eagle Ford Shale (Table 4-4). Reuse of wastewater as a percentage of total water use is generally low (5% or less) in all major basins and plays in Texas, except for the Anadarko Basin in the Texas Panhandle, where it is 20% (Nicot et al., 2012) (Table 4-2).

Table 4-4. Estimated brackish water use as a percentage of total hydraulic fracturing water use in the main hydraulic fracturing areas of Texas, 2011.[a]
Adapted from Nicot et al. (2012).[b]

Play	Percentage
Barnett Shale	3%
Eagle Ford Shale	20%
Texas portion of the TX-LA-MS Salt Basin[c]	0%
Permian Basin—Far West	80%
Permian Basin—Midland	30%
Anadarko Basin	30%

[a] Nicot et al. (2012) define brackish water as any water with a total dissolved solids (TDS) content of >1,000 mg/L, but <35,000 mg/L, although they often limit that range to between 1,000 and 10,000 mg/L.

[b] Nicot et al. (2012) present the estimated percentages of brackish, recycled/reused, and fresh water relative to total hydraulic fracturing water use so that the percentages of the three categories sum to 100%.

[c] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.

The majority of water used in Texas for hydraulic fracturing is self-supplied via direct ground or surface water withdrawals (Nicot et al., 2014). Less often, water is purchased from local landowners, municipalities, larger water districts, or river authorities (Nicot et al., 2014).

Water use per well: Water use per well varies across Texas basins, with reported medians from 2011 to early 2013 of 3.9 million gal (14.8 million L) in the Fort Worth Basin, 3.8 million gal (14.4 million L) in the Western Gulf, 3.3 million gal (12.5 million L) in the Anadarko, 3.1 million gal (11.7 million L) in the TX-LA-MS Salt, and 840,000 gal (3.2 million L) in the Permian (Appendix

[1] The EPA FracFocus report shows that "fresh" was the only source of water listed in 91% of all disclosures reporting a source of water in Texas (U.S. EPA, 2015b). Nineteen percent of Texas disclosures included information related to water sources (U.S. EPA, 2015b).

Table B-5). Relatively low water use in the Permian Basin, which contains roughly half the reported wells in the state, is due to the abundance of vertical wells, mostly for oil extraction (Nicot et al., 2012).

Water use per well is increasing in most locations in Texas. In the Barnett Shale, water use per well increased from approximately 3 million gal (11 million L) in the mid-2000's to approximately 5 million gal (19 million L) in 2011 as the horizontal lengths of wells increased (Nicot et al., 2014). Similar increases in lateral length and water use per well were reported for the Texas-Haynesville, East Texas, and Anadarko basins, and most of the Permian Basin (Nicot et al., 2012; Nicot and Scanlon, 2012).[1]

Water use/consumption at the county scale: Water use and consumption for hydraulic fracturing can be significant in some Texas counties. Texas contains five of nine counties nationwide where operators used more than 1 billion gal (3.8 billion L) of water annually for hydraulic fracturing, and five of nine counties where fracturing water use in 2011 and 2012 was 30% or more compared to total water use in those counties in 2010 (Table 4-3, Figure 4-3a; Appendix Table B-2).[2]

According to detailed county level projections, water use for hydraulic fracturing is expected to increase with oil and gas production in the coming decades, peaking around the year 2030 (Nicot et al., 2012). These projections were made before the recent decline in oil and gas prices, and so are highly uncertain. If these projections hold, the majority of counties are expected to have relatively low water use for fracturing in the future, but hydraulic fracturing water use could equal or exceed 10%, 30%, and 50% compared to 2010 total county water use in 30, nine, and three counties, respectively, by 2030 (Appendix Table B-7).

Potential for impacts: Of all locations surveyed in this chapter, the potential for water quantity and quality impacts due to hydraulic fracturing water acquisition appears to be highest in southern and western Texas. This area includes the Anadarko, the Western Gulf (Eagle Ford play), and the Permian Basins. According to Ceres (2014), 28% and 87% of the wells fractured in the Eagle Ford play and Permian Basin, respectively, are in areas of high to extremely high water stress.[3] A comparison of hydraulic fracturing water use to water availability at the county scale also suggests the potential for impacts in this region (Text Box 4-2).

[1] It should be noted that energy production also increases with lateral lengths, and therefore, water use per unit energy produced—typically referred to as water intensity—may remain the same or decline despite increases in per-well water use (Nicot et al., 2014; Laurenzi and Jersey, 2013).

[2] Texas also contains 10 of the 25 counties nationwide where hydraulic fracturing water consumption was greater than or equal to 30% of 2010 total water consumption (Table 4-3). Nicot and Scanlon (2012) found similar variation among counties when they compared hydraulic fracturing water consumption to total county water consumption for the Barnett play. Their consumption estimates ranged from 581 million gal (2.20 billion L) in Parker County to 2.7 billion gal (10.2 billion L) in Johnson County, representing 10.5% and 29.7% compared to total water consumption in those counties, respectively. Fracturing in Tarrant County, part of the Dallas Fort-Worth area, consumed 1.6 billion gal (6.1 billion L) of water, 1.4% compared to total county water consumption (Nicot and Scanlon, 2012).

[3] Ceres (2014) compared well locations to areas categorized by a water stress index, characterized as follows: extremely high (defined as annual withdrawals accounting for greater than 80% of surface flows); high (40–80% of surface flows); or medium-to-high (20–40% of surface flows).

Text Box 4-2. Hydraulic Fracturing Water Use as a Percentage of Water Availability Estimates.

Researchers at Sandia National Laboratories assessed county level water availability across the continental United States (Tidwell et al., 2013). Assessments of water availability in the United States are generally lacking at the county scale, and this analysis—although undertaken for siting new thermoelectric power plants—can be used to assess potential impacts of hydraulic fracturing water withdrawals.

The authors generated annual water availability estimates for five categories of water: unappropriated surface water, unappropriated groundwater, appropriated water potentially available for purchase, brackish groundwater, and wastewater from municipal treatment plants (Tidwell et al., 2013). In the western United States, water is generally allocated by the principle of prior appropriation—that is, first in time of use is first in right. New development must use unappropriated water or purchase appropriated water from vested users. In their analysis, the authors assumed 5% of appropriated irrigated water could be purchased; they also excluded wastewater required to be returned to streams and the wastewater fraction already reused.

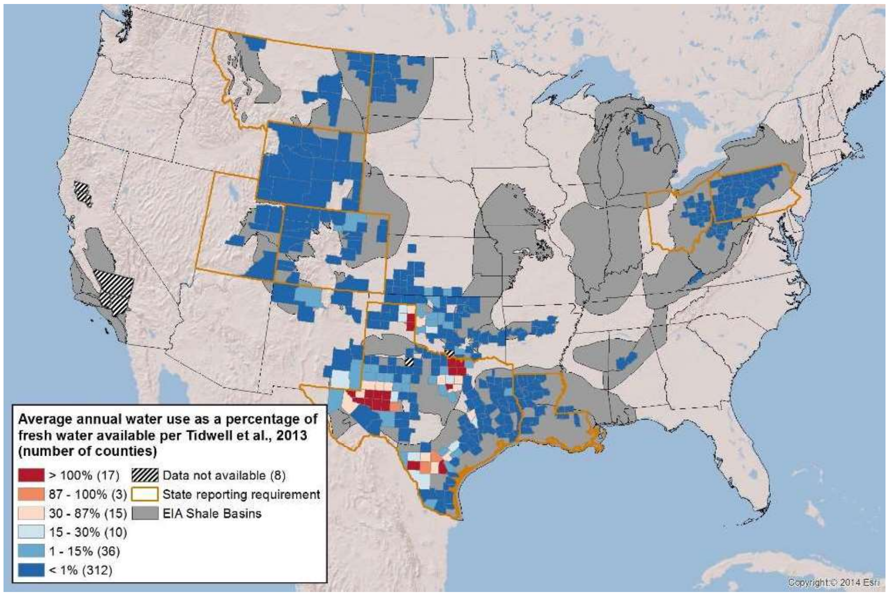
Given regulatory restrictions, they considered no fresh water to be available in California for new thermoelectric plants. Their definition of brackish water ranged from 3,000 to 10,000 ppm TDS, and from 50 to 2,500 ft (15-760 m) below the surface.

Combining their estimates of unappropriated surface water and groundwater and appropriated water potentially available for purchase, we derived a fresh water availability estimate for each county (except for those in California) and then compared this value to reported water use for hydraulic fracturing in 2011 and 2012 (U.S. EPA, 2015c). We also added the estimates of brackish groundwater and wastewater from municipal treatment plants to fresh water estimates to derive estimates of total water availability and did a similar comparison. Since the water availability estimates already take into account current water use for oil and gas operations, these results should be used only as indicator of areas where shortages might arise in the future. Here we focus on hydraulic fracturing water use compared to water availability. If we compared hydraulic fracturing water consumption to water availability, consumption would be lower relative to availability since by definition, water consumption is less than water use. Hence, water use versus availability acts as an upper-bound estimate, and includes consumption.

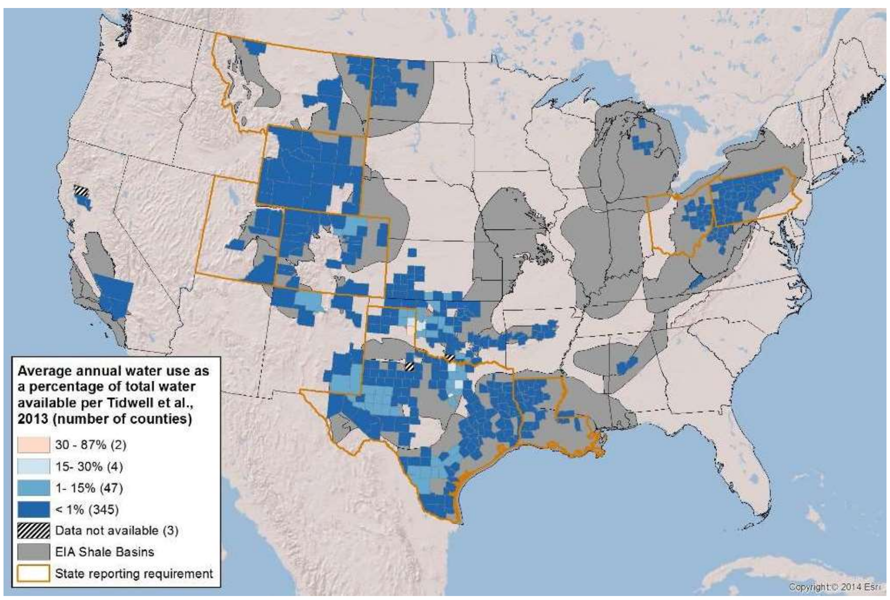
Overall, hydraulic fracturing water use represented less than 1% of fresh water availability in over 300 of the 395 counties analyzed (Figure 4-6a). This result suggests that there is ample water available at the county scale to accommodate hydraulic fracturing in most locations. However, there was a small number of counties where hydraulic fracturing water use was a relatively high percentage of fresh water availability. In 17 counties, fracturing water use actually exceeded the index of fresh water available; all of these counties were located in the state of Texas and were associated with the Anadarko, Barnett, Eagle Ford, and Permian basins/plays (Figure 4-5). In Texas counties with relatively high brackish water availability, hydraulic fracturing water use represented a much smaller percentage of total water availability (fresh + brackish + wastewater) (Figure 4-6b). This finding illustrates that potential impacts can be avoided or reduced in these counties through the use of brackish water or wastewater for hydraulic fracturing; a case study in the Eagle Ford play in southwestern Texas echoes this finding (Text Box 4-3).

(Text Box 4-2 is continued on the following page.)

Text Box 4-2 (continued). Hydraulic Fracturing Water Use as a Percentage of Water Availability Estimates.



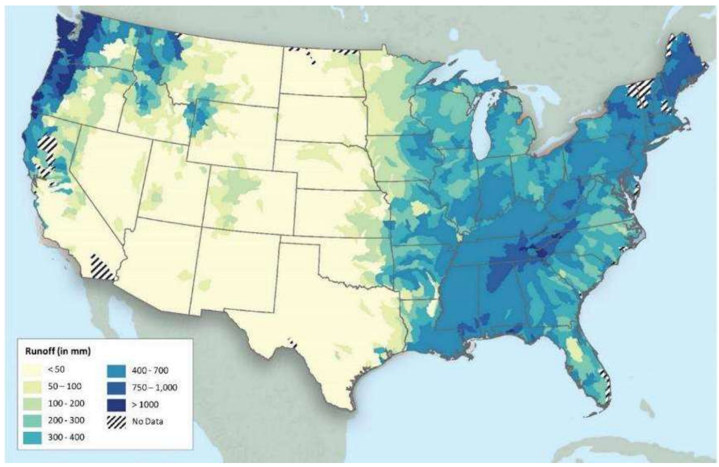
(a)



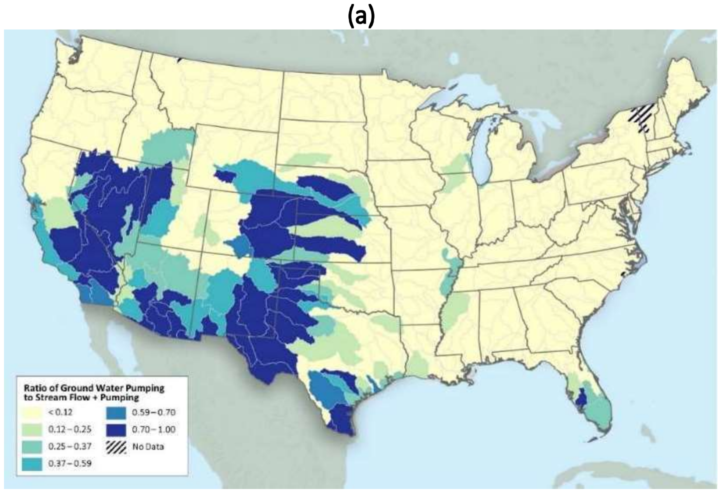
(b)

Figure 4-6. Average annual hydraulic fracturing water use in 2011 and 2012 compared to (a) fresh water available and (b) total water (fresh, brackish, and wastewater) available, by county, expressed as a percentage. Counties shown with respect to major U.S. EIA shale basins (EIA, 2015). Orange borders identify states that required some degree of reporting to FracFocus in 2011 and 2012. Data from U.S. EPA (2015c) and Tidwell et al. (2013); data from Tidwell et al. (2013) supplied from the U.S. Department of Energy (DOE) National Renewable Energy Laboratory on January 28, 2014 and available upon request from the U.S. DOE Sandia National Laboratories. The analysis by Tidwell et al. (2013) was done originally for thermoelectric power generation. As such, it was assumed that no fresh water could be used in California for this purpose due to regulatory restrictions, and therefore no fresh water availability data were given for California. The total water available for California is the sum of brackish water plus wastewater only.

Surface water availability is generally low in southern and western Texas (Figure 4-7a), and both fracturing operations and residents rely heavily on groundwater (Figure 4-7b). Similar to trends nationally, groundwater aquifers in Texas have experienced substantial declines caused by withdrawals (Konikow, 2013; TWDB, 2012; George et al., 2011). Groundwater in the Pecos Valley, Gulf Coast, and Ogallala aquifers in southern and western Texas is estimated to have declined by roughly 5, 11, and 44 mi^3 (21, 45.5, and 182 km^3), respectively, between 1900 and 2008 (Konikow, 2013).[1]



(a)



(b)

Figure 4-7. (a) Estimated annual surface water runoff from the USGS; (b) Reliance on groundwater as indicated by the ratio of groundwater pumping to stream flow and pumping.
Estimates for Figure 4-7a were calculated at the 8-digit hydrological unit code (HUC) scale by dividing annual average daily stream flow (from October 1, 2012, to September 30, 2013) by HUC area. Data accessed from the USGS (USGS, 2014c). Higher ratios (darker blues) in Figure 4-7b indicate greater reliance on groundwater. Figure adapted from Tidwell et al. (2012), using data provided by the U.S. Department of Energy's Sandia National Laboratories on December 12, 2014.

[1] The estimate of total net volumetric groundwater depletion for the Gulf Coast aquifer is the sum of the individual depletion estimates for the north (Houston area), central, and southern (Winter Garden area) parts of the Texas Gulf Coast aquifer. Groundwater depletion from the Carrizo-Wilcox aquifer is included in the estimate for the southern portion of the Gulf Coast aquifer (Konikow, 2013).

Groundwater quality degradation associated with aquifer pumping and the cumulative effects of all water users is well documented in the southern portion of the Ogallala aquifer. The quality of groundwater used by many private, public supply, and irrigation wells is poorest in the aquifer's southern portion, with elevated concentrations of TDS, chloride, nitrate, fluoride, manganese, arsenic, and uranium (Chaudhuri and Ale, 2014a; Gurdak et al., 2009; McMahon et al., 2007).[1] Extensive groundwater pumping can alter the quality of drinking water resources by inducing vertical mixing of high-quality groundwater with recharge water from the land surface that has been contaminated by nitrate or pesticides, or with lower-quality groundwater from underlying geologic formations (Gurdak et al., 2009; Konikow and Kendy, 2005). Pumping can also promote changes in reduction-oxidation (redox) conditions and thereby mobilize chemicals from geologic sources (e.g., uranium) (DeSimone et al., 2014). Similar patterns of groundwater quality degradation associated with prolonged aquifer depletion (i.e., salinization and contamination) have also been observed in other Texas aquifers, notably the northwest Edwards-Trinity (plateau), Pecos Valley, Carrizo-Wilcox, and southern Gulf Coast aquifers.[2]

The Texas Water Development Board (TWDB) estimates that overall demand for water (including water for hydraulic fracturing) out to the year 2060 will outstrip supply in southern and western Texas (TWDB, 2012). Furthermore, the TWDB expects groundwater supply in the major aquifers to decline by 30% between 2010 and 2060, mostly due to declines in the Ogallala aquifer (TWDB, 2012).[3,4] Irrigated agriculture is by far the dominant user of water from the Ogallala aquifer (Gurdak et al., 2009), but fracturing operations, along with other uses, now contribute to the aquifer's depletion.

The state has also experienced moderate to extreme drought conditions for much of the last decade, and the second-worst and longest drought in Texas history between March 2010 and November 2014 (TWDB, 2016; National Drought Mitigation Center, 2015) (Figure 4-8). Sustained drought conditions compound water availability concerns, and climate change is expected to place further stress on groundwater both now and in the future (Aghakouchak et al., 2014; Melillo et al., 2014) (Chapter 2). In their evaluation of the potential impact of climate change on groundwater recharge in the western United States, Meixner et al. (2016) show the largest declines in recharge are expected in specific aquifers in the southwestern United States, including the southern portion of the Ogallala aquifer, which is expected to receive 10% less recharge through the year 2050.

---

[1] Elevated levels of these constituents result from both natural processes and human activities, such as groundwater pumping (Chaudhuri and Ale, 2014a; Gurdak et al., 2009).

[2] Persistent salinity has been observed in west Texas, specifically in the southern Ogallala, northwest Edwards-Trinity (plateau), and Pecos Valley aquifers, largely due to prolonged irrigational groundwater pumping and ensuing alteration of hydraulic gradients leading to groundwater mixing (Chaudhuri and Ale, 2014b). High levels of groundwater salinization associated with prolonged aquifer depletion have also been documented in the Carrizo-Wilcox and southern Gulf Coast aquifers, underlying the Eagle Ford Shale in south Texas (Chaudhuri and Ale, 2014b; Konikow, 2013; Boghici, 2009). Further, elevated levels of constituents, including nitrate, lead, fluoride, chloride, sulfate, iron, manganese, and TDS, have been reported in the Carrizo-Wilcox aquifer (Boghici, 2009).

[3] TWDB (2012) defines groundwater supply as the amount of groundwater that can be produced given current permits and existing infrastructure. By contrast, TWDB (2012) defines groundwater availability as the amount of groundwater that is available regardless of legal or physical availability. Total groundwater availability in Texas is expected to decline by approximately 24% between 2010 and 2060 (TWDB, 2012).

[4] This message is echoed in the 2017 Texas State Water Plan (TWDB, 2016).

Groundwater moves slowly, and natural recharge rates are lower during times of drought (DeSimone et al., 2014). Consequently, as water withdrawals continue to outpace the rate of recharge, aquifer storage will decline further (USGS, 1999), potentially impacting both drinking water resource quantity and quality. For example, research from Steadman et al. (2015) in the Eagle Ford play shows that hydraulic fracturing groundwater consumption exceeds estimated recharge rates in the seven most active counties for drilling.

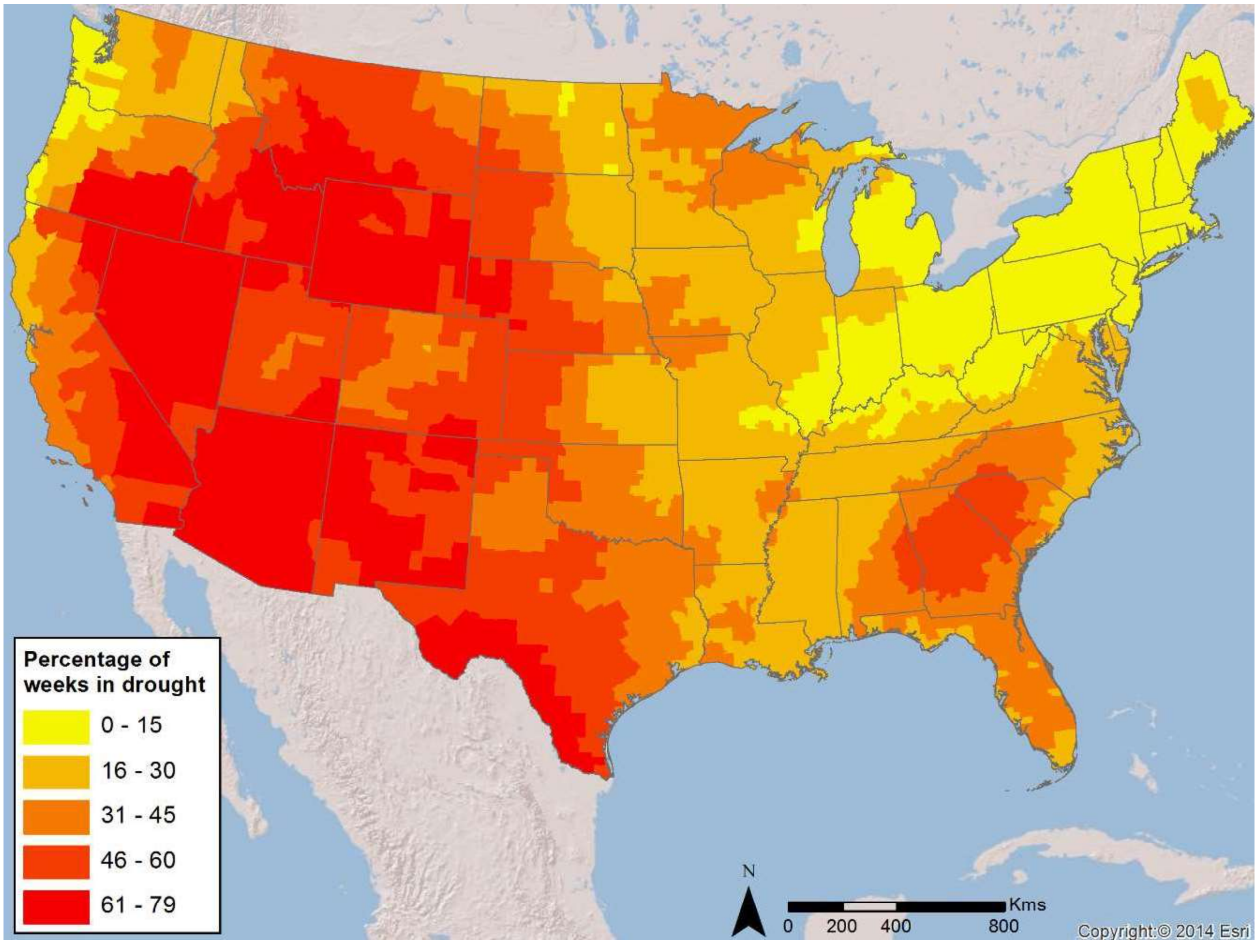


Figure 4-8. Percentage of weeks in drought between 2000 and 2013 by county.
Drought for a given week is defined as any portion of a given U.S. county having a weekly classification of moderate to exceptional drought (D1-D4 categorization) according to the National Drought Mitigation Center (http://droughtmonitor.unl.edu); number of weeks = 731.

A case study in the Eagle Ford play in southwestern Texas compared water demand for hydraulic fracturing with water supplies at the scale of the play, county, and 1 mi^2 (2.6 km^2) (Scanlon et al., 2014b). The authors observed generally adequate water supplies for hydraulic fracturing, except in specific locations, where they found excessive drawdown of groundwater locally in ~6% of the play area, with estimated declines of ~100-200 ft (31-61 m) after hydraulic fracturing activity increased in 2009 (Text Box 4-3).

Text Box 4-3. Case Study: Water Profile of the Eagle Ford Play, Texas.

Researchers from the University of Texas published a detailed case study of water supply and demand for hydraulic fracturing in the Eagle Ford play in southwestern Texas (Scanlon et al., 2014b). This effort assembled detailed information from state and local water authorities, and proprietary industry data on hydraulic fracturing, to develop a portrait of water resources in this 16-county area.

Scanlon et al. (2014b) compared water demand for hydraulic fracturing currently and over the projected play life (20 years) relative to water supply from groundwater recharge, groundwater storage (brackish and fresh), and stream flow. Using groundwater availability models developed by the Texas Water Development Board, they reported that water demand for hydraulic fracturing in 2013 was 30% of annual groundwater recharge in the play area, and over the 20-year play lifespan it was projected to be 26% of groundwater recharge, 5-8% of fresh groundwater storage, and 1% of brackish groundwater storage. The dominant water user in the play is irrigation (57 to 61% of water use, 62 to 65% of consumption), as compared with hydraulic fracturing (13% of water use and 16% of consumption). At the county level, projected water demand for hydraulic fracturing over the 20-year period was low relative to freshwater supply (ranging from 0.6-27% by county, with an average of 7.3%). Similarly, projected total water demand from all uses was low relative to supply, excluding two counties with high irrigation demands (Frio, Zavala), and one county with no known groundwater supplies (Maverick).

Although supply was found to be sufficient even in this semi-arid region, there were important exceptions, especially at sub-county scales. The researchers found no water level declines over much of the play area assessed (69% of the play area), yet in some areas they estimated groundwater drawdowns of 50 ft (15 m) or more (19% of the play area), 100 ft (31 m) or more (6% of the play area), and 200 ft (60 m) or more (approximately 2% of the play area). This was corroborated with well monitoring data that showed a sharp decline in water levels in several groundwater monitoring wells after hydraulic fracturing activity increased in 2009.

The researchers further concluded that shifting toward brackish groundwater is feasible, as evidenced by operators already doing so. This shift could further reduce impacts on fresh water resources and provide a large source of water for future hydraulic fracturing. In a 2011 estimate, approximately 20% of water used in the play came from brackish sources (Table 4-4), and anecdotal evidence suggests this practice has increased since then (Scanlon et al., 2014b). Projected hydraulic fracturing water use represents less than 1% of total brackish groundwater storage in the play area. By contrast, Scanlon et al. (2014b) concluded there is limited potential for reuse of wastewater in this play because of the small volumes that return to the surface during production (less than or equal to 5% of hydraulic fracturing water requirements).

In contrast to southern and western Texas, the potential for water quantity and quality effects appears to be lower in the north-central and eastern parts of the state, in areas including the Barnett and Haynesville plays. Residents obtain water for domestic use—which includes use of water for drinking—from a mixture of groundwater and surface water sources (Appendix Table B-6). Counties encompassing Dallas and Fort Worth rely mostly on publicly-supplied surface water (TWDB, 2012) (Appendix Table B-6). The Trinity aquifer in northeast Texas is projected to decline only slightly between 2010 and 2060 (TWDB, 2012). Nevertheless, Bene et al. (2007) estimate that hydraulic fracturing groundwater withdrawals will increase from 3% of total groundwater use in 2005 to 7%–13% in 2025, suggesting the potential for localized aquifer drawdown. Groundwater quality degradation associated with aquifer drawdown has been documented in the Trinity and Woodbine aquifers overlying much of the Barnett play, with both aquifers showing high levels of salinization (Chaudhuri and Ale, 2013).

Overall, the potential for impacts appears higher in western and southern Texas, compared to the northeast part of the state. Groundwater withdrawals for hydraulic fracturing, along with irrigation and other uses, may contribute to water quality degradation associated with intensive aquifer pumping in western and southern Texas. Areas with numerous high-capacity wells and large amounts of sustained groundwater pumping are most likely to experience groundwater quality degradation associated with withdrawals (Gurdak et al., 2009; McMahon et al., 2007). Further, given that Texas is prone to drought conditions and groundwater recharge is limited, the already declining aquifers in southern and western Texas are especially vulnerable to further groundwater depletion and resulting impacts to groundwater quantity and quality (Gurdak et al., 2009; Jackson et al., 2001). Impacts are likely to be localized drawdowns of groundwater, as shown by a detailed case study of the Eagle Ford play (Text Box 4-3). Scanlon et al. (2014b) suggested that a shift toward brackish water use could minimize potential future impacts to fresh water resources. This finding is consistent with our county level data (Text Box 4-2).

4.5.2 Colorado and Wyoming

Colorado had the second highest number of disclosures in the EPA FracFocus 1.0 project database, (13% of disclosures) (Figure 4-4 and Appendix Table B-5). We combine Colorado and Wyoming because of their shared geology of the Denver Basin (including the Niobrara play) and the Greater Green River Basin (Figure 4-9). There are three major basins reported for Colorado: the Denver Basin; the Uinta-Piceance Basin; and the Raton Basin. Together these basins contain 99% of reported wells in the state, although the bulk of the activity in Colorado is in the Denver Basin (Appendix Table B-5). Fewer wells (roughly 4% of disclosures in the EPA FracFocus 1.0 project database) are reported in Wyoming. There are two major basins reported for Wyoming (Greater Green River and Powder River) that together contain 86% of activity in the state (Appendix Table B-5).

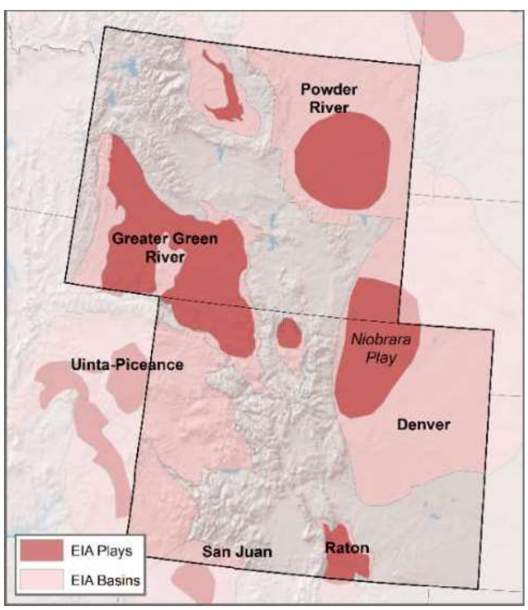


Figure 4-9. Major U.S. EIA shale plays and basins for Colorado and Wyoming.
Source: EIA (2015).

Types of water used: Water for hydraulic fracturing in Colorado and Wyoming comes from both groundwater and surface water, as well as reused wastewater (Colorado Division of Water Resources et al., 2014; BLM, 2013). Publicly available information on water sources for each state generally comes in the form of a list of potential sources, and detailed information on the types of water used for hydraulic fracturing is not readily accessible.[1] In northwestern Colorado's Garfield County (Uinta-Piceance Basin), the U.S. EPA (2015e) reports that any fresh water used for fracturing comes from surface water sources. In the Denver Basin (Niobrara play) of southeastern Wyoming, qualitative information suggests that groundwater supplies much of the water used for fracturing, although no data were available to characterize the ratio of groundwater to surface water withdrawals (AMEC Environment & Infrastructure, 2014; BLM, 2013; Tyrrell, 2012).

Non-fresh water sources, including industrial and municipal wastewater, brackish groundwater, and reused hydraulic fracturing wastewater, are sometimes listed as potential alternatives to fresh water for fracturing in both Colorado and Wyoming (Colorado Division of Water Resources et al., 2014; BLM, 2013); no data are available to show the extent to which these non-fresh water sources are used at the state or basin level. Based on discussions with industry, the U.S. EPA (2015e) reports that fresh water is used solely for drilling and reused wastewater supplies nearly all the water for hydraulic fracturing in Colorado's Garfield County. This estimate of reused wastewater as a percentage of injected volume is markedly higher than in other locations and likely results from the geologic characteristics of the Piceance tight sand formation, which has naturally high water content and produces large volumes of relatively high-quality wastewater (U.S. EPA, 2015e).

In contrast, a study by Goodwin et al. (2014) assumed no reuse of wastewater for hydraulic fracturing operations by Noble Energy in the Denver-Julesburg Basin of northeastern Colorado (Table 4-2). It is unclear whether this assumption is indicative of reuse practices of other companies in the Denver-Julesburg Basin. The difference in reused wastewater rates reported by the U.S. EPA (2015e) and Goodwin et al. (2014) may indicate an east-west divide in Colorado (i.e., low reuse in the east versus high reuse in the west), due at least in part to differences in wastewater volumes available for reuse. However, further information is needed to adequately characterize reuse patterns in Colorado.

Water use per well: Water use per well varies across Colorado, with median values of 1.8 million, 400,000, and 96,000 gal (6.8 million, 1.5 million, and 360,000 L) in the Uinta-Piceance, Denver, and Raton Basins, respectively, according to the EPA FracFocus 1.0 project database (Appendix Table B-5). Relatively low water volumes per well are reported in Wyoming (Appendix Table B-5). Low volumes reported for the Raton Basin of Colorado and the Powder River Basin of Wyoming are likely due to the prevalence of CBM extraction in these locations (U.S. EPA, 2015k; Sando et al., 2014).

More difficult to explain are the low volumes reported for the Denver Basin in the EPA FracFocus 1.0 project database. These values are lower than volumes reported in other non-CBM basins

[1] The Colorado Oil and Gas Conservation Commission collects information on the sources and quality of water used for hydraulic fracturing, including reused wastewater, with Form 5A, and has done so since June 2012; however, these data are in PDFs linked to individual wells and are not aggregated into a searchable database.

included in Appendix Table B-5. Goodwin et al. (2014) report much higher water use per well in the Denver Basin from 2010 to 2013, with a median of 2.8 million gal (10.6 million L) (although only usage for the Wattenberg Field was reported). Indeed, the 10th–90th percentiles (2.4-3.8 million gal) (9.1-14.4 million L) from Goodwin et al. (2014) are almost completely above those from the EPA FracFocus 1.0 project database for the Denver Basin (Appendix Table B-5).[1] However, it is difficult to draw clear conclusions because of differences in scale (i.e., field in Goodwin et al. (2014) versus basin in the EPA FracFocus 1.0 project database) and operators (i.e., Noble Energy in Goodwin et al. (2014) versus all in the EPA FracFocus 1.0 project database).

Trends in water use per well are generally lacking for Colorado, with the exception of those reported by Goodwin et al. (2014). They found that water use per well is increasing with well length in the Denver Basin; however, they also observed that water intensity (gallons of water per unit energy extracted) did not change, since energy recovery increased along with water use.

Water use/consumption at the county scale: Hydraulic fracturing operations in Colorado use billions of gallons of water, but this amount is a small percentage compared to total water used or consumed at the county scale. In both Garfield and Weld Counties, located in the Uinta-Piceance and Denver Basins, respectively, hydraulic fracturing used more than 1 billion gal (3.8 billion L) annually. Fracturing water use and consumption in these counties exceeded those in all other Colorado counties combined (Appendix Table B-2), but the water used for hydraulic fracturing in Garfield and Weld counties was less than 2% and 3% compared to 2010 total water use and consumption, respectively. In comparison, irrigated agriculture accounts for over 90% of the water used in both counties (Maupin et al., 2014). Overall, hydraulic fracturing accounts for less than 2% compared to 2010 total water use in all Colorado counties represented in the EPA FracFocus 1.0 project database (Appendix Table B-2). Water use estimates based on the EPA FracFocus 1.0 project database may be low relative to literature and state estimates (Text Box 4-1), but even if estimates from the project database were doubled, hydraulic fracturing water use and consumption would still be less than 4% and 6% compared to 2010 total water use and consumption, respectively, in each Colorado county.

In Wyoming, reported water use for hydraulic fracturing is small compared to Colorado (Appendix Table B-1). Fracturing water use and consumption did not exceed 1% of 2010 total water use and consumption, respectively, in any county (Appendix Table B-2). Unlike Colorado, Wyoming did not require disclosure to FracFocus during the time period analyzed by the EPA (U.S. EPA, 2015b) (Appendix Table B-5).

Colorado Division of Water Resources et al. (2014) projected that annual water use for hydraulic fracturing in the state would increase by approximately 16% between 2012 and 2015, but demand in later years is unclear. Even with an increase of 16% or more, hydraulic fracturing would still remain a relatively small user of water at the county scale in Colorado.

---

[1] Different spatial extents might explain these differences, since Goodwin et al. (2014) focus on 200 wells in the Wattenberg Field of the Denver Basin; however, Weld County is the center of activity in the Wattenberg Field, and the EPA FracFocus 1.0 project database contains 3,011 disclosures reported in Weld County, with a median water use per of 407,442 gal (1,542,340 L), similar to that for the basin as a whole.

Potential for impacts: The potential for water quantity and quality impacts due to hydraulic fracturing water withdrawals appears to be low at the county scale in Colorado and Wyoming because fracturing accounts for a low percentage of total water use and consumption (Figure 4-3a,b). This conclusion is also supported by the comparison of hydraulic fracturing water use to water availability at the county scale (Text Box 4-2; Figure 4-6a,b). However, counties in Colorado and Wyoming are large in their spatial extents, and any potential impacts will depend on site-specific factors affecting the balance between water use and availability at the local scale (i.e., at a given withdrawal point). In a multi-scale case study in the Upper Colorado River Basin, the U.S. EPA (2015e) did not identify any locations where fracturing currently contributed to locally high water use intensity due to the high rates of wastewater reuse reported. They did conclude, however, that future effects may be possible (Text Box 4-4).

Text Box 4-4. Case Study: Impact of Water Acquisition for Hydraulic Fracturing on Local Water Availability in the Upper Colorado River Basin.

The U.S. EPA (2015e) conducted a case study to explore the impact of hydraulic fracturing water demand on water availability at the river basin, county, and local scales in the semi-arid Upper Colorado River Basin (UCRB) of western Colorado. The study area overlies the Piceance geologic basin with natural gas in tight sands. Water withdrawal impacts were quantified using a water use intensity index (i.e., the ratio between the volume of water withdrawn at a site for hydraulic fracturing and the volume of available water). Researchers obtained detailed site-specific data on hydraulic fracturing water usage from state and regional authorities, and estimated available water supplies using observations at USGS gage stations and empirical and hydrologic modeling.

They found that water supplies accessed for oil and gas demand were concentrated in Garfield County, and most fresh water withdrawals were concentrated within the Parachute Creek watershed (198 mi^2). However, fresh water makes up a small proportion of the total water used for fracturing due to large quantities of high-quality wastewater produced from the Piceance tight sands. Based on discussions with industry, the U.S. EPA (2015e) reports that fresh water is used solely for drilling and reused wastewater supplies nearly all the water for hydraulic fracturing in Garfield County. Due to the high reuse rate, the U.S. EPA (2015e) did not identify any locations in the Piceance play where fracturing contributed to locally high water use intensity.

Scenario analyses demonstrated a pattern of increasing potential impact with decreasing watershed size in the UCRB. The U.S. EPA (2015e) examined hydraulic fracturing water use intensity under the current rates of both directional (S-shaped) and horizontal drilling. They showed that for the more water-intensive horizontal drilling, watersheds had to be larger to meet the same index of water use intensity (0.4) as that for directional drilling (100 mi^2 for horizontal drilling, as compared to 30 mi^2 for directional drilling). To date, most wells have been drilled directionally into the Piceance tight sands, although a trend toward horizontal drilling is expected to increase annual water use per well by about four times. Despite this increase, total hydraulic fracturing water use is expected to remain small relative to other users. Currently, irrigated agriculture is the largest water user in the UCRB.

Greater water demand could occur in the future if the water-intensive oil shale extraction industry becomes economically viable in the region. Projections for oil shale water demand indicate that the industry could increase water use for energy extraction in Garfield and Rio Blanco counties.

East of the Rocky Mountains in the Denver Basin, the potential for localized impacts exists given the combination of high hydraulic fracturing activity and low water availability (e.g., Weld County, Colorado), but lack of available data and literature at the local scale limits our ability to assess the potential for impacts in this location. Ceres (2014) concludes that all fractured wells in the Denver Basin are in high or extremely high water-stressed areas. Furthermore, the development of the Niobrara Shale in southeast Wyoming occurs in areas already impacted by high agricultural water use from the Ogallala aquifer, including the state's only three groundwater control areas, which were established as management districts in the southeast portion of the state in response to declining groundwater levels (AMEC Environment & Infrastructure, 2014; Wyoming State Engineer's Office, 2014; Tyrrell, 2012; Bartos and Hallberg, 2011). Groundwater withdrawals for hydraulic fracturing may have the potential to contribute to water quality degradation in these areas, depending on site-specific factors that may alter the balance between water use and availability.

Overall, the potential for impacts appears low at the county scale in Colorado and Wyoming, but local effects are certainly possible particularly east of the Rocky Mountains in the Denver Basin. Lack of available data and literature at the local scale limits our ability to assess the potential for impacts in this location.

4.5.3 Pennsylvania, West Virginia, and Ohio

Pennsylvania had the third most disclosures in the EPA FracFocus 1.0 project database (6.5% of disclosures) (Appendix Table B-5; Figure 4-4). We combine West Virginia and Ohio with Pennsylvania because they share similar geology overlying the Appalachian Basin (including the Marcellus, Devonian, and Utica stacked plays) (Figure 4-10); however, much less activity is reported in these two states (Appendix Table B-5).

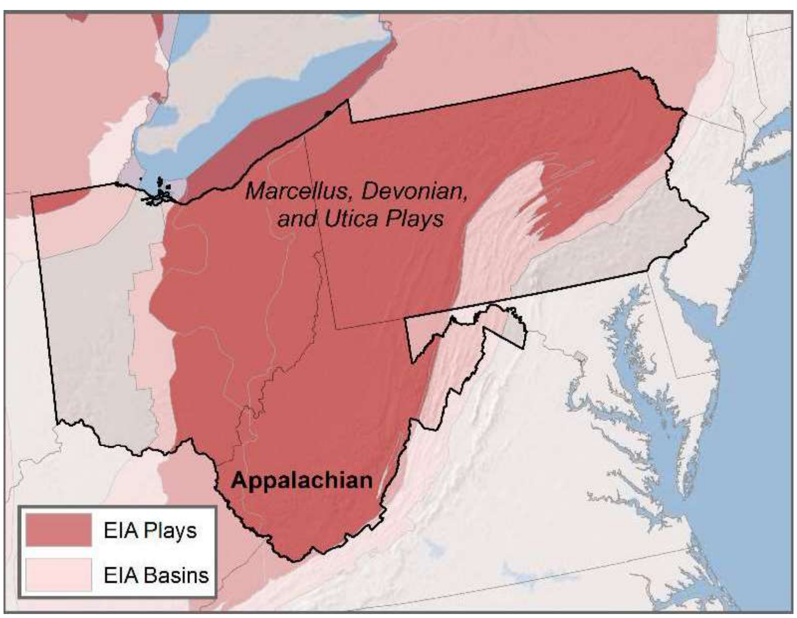


Figure 4-10. Major U.S. EIA shale plays and basins for Pennsylvania, West Virginia, and Ohio.
Source: EIA (2015).

Types of water used: Surface water is the primary water source for hydraulic fracturing in Pennsylvania, West Virginia, and Ohio (SRBC, 2016; Schmid and Yoxtheimer, 2015; West Virginia DEP, 2014; Mitchell et al., 2013a; West Virginia DEP, 2013; Ohio EPA, 2012b; STRONGER, 2011b) (Table 4-1). Further, the water used for hydraulic fracturing is most often fresh water in all three states. In both Pennsylvania's Susquehanna River Basin and throughout West Virginia, most water for hydraulic fracturing is self-supplied via direct withdrawals from surface water and groundwater (U.S. EPA, 2015e; West Virginia DEP, 2013). Operators also purchase water from public water systems, which may include a variety of commercial water brokers (West Virginia DEP, 2014; SRBC, 2013; West Virginia DEP, 2013). Municipal supplies are also used, particularly in urban areas of Ohio (STRONGER, 2011b).

Reused hydraulic fracturing wastewater as a percentage of total water used for fracturing was 19% in 2014 in Pennsylvania, and 15% in 2012 in West Virginia (Schmid and Yoxtheimer, 2015; West Virginia DEP, 2014) (Table 4-2). Available data indicate an increasing trend in reuse of wastewater over time in this region, likely due to the lack of nearby disposal options in Class II wells. Reused wastewater as a percentage of injected water volume ranged from approximately 2% to 19% in Pennsylvania (statewide) from 2009-2014 (Schmid and Yoxtheimer, 2015). This upward trend is also shown in Pennsylvania's SRB, where reuse as a percentage of total water injected reached 22% in 2013; the average reuse rate for 2008-2013 in the SRB was 16% (SRBC, 2016) (Table 4-2). In West Virginia, reuse as a percentage of injected volume ranged from 6% to 15% from 2010-2012 (West Virginia DEP, 2014). In Ohio's Marcellus and Utica Shales, reuse of wastewater is reportedly uncommon (STRONGER, 2011b), likely due to the prevalence of disposal wells in Ohio. See Chapter 8 for more information.

Aside from reused hydraulic fracturing wastewater, other types of wastewaters reused for hydraulic fracturing may include wastewater treatment plant effluent, treated acid mine drainage, and rainwater collected at various well pads (West Virginia DEP, 2014; SRBC, 2013; West Virginia DEP, 2013; Ziemkiewicz et al., 2013; Ohio EPA, 2012b). No data are available on the frequency of use of these other wastewaters.

Water use per well: Operators in these three states reported the third, fourth, and fifth highest median water use per well of the states we considered from the EPA FracFocus 1.0 project database, with 5.0, 4.2, and 3.9 million gal (18.9, 15.9, and 14.8 million L) in West Virginia, Pennsylvania, and Ohio, respectively (Appendix Table B-5). Hansen et al. (2013) report similar water use estimates for Pennsylvania and West Virginia for 2011 and 2012 (Appendix Table B-5). This correspondence is not surprising, as these estimates are also based on FracFocus data (via Skytruth). For 2011, the year overlapping with the time frame of the EPA FracFocus report (U.S. EPA, 2015b), Mitchell et al. (2013a) report an average of 2.3 million gal (8.7 million L) for vertical wells (54 wells) and 4.6 million gal (17.4 million L) for horizontal wells (612 wells) in the Pennsylvania portion of the Upper Ohio River Basin, based on records from PA DEP. The weighted average water use per well was 4.4 million gal (16.7 million L), similar to results based on the EPA FracFocus 1.0 project database listed above. In Pennsylvania's SRB, the long-term average water use per well from 2008-2013 was 4.3 million gal (16.3 million L). In 2013, the average water use per well increased to approximately 5.1 to 6.5 million gal (19.3 to 24.6 million L) due to increasing lengths of laterals in horizontal drilling (SRBC, 2016). Across the entire state of Pennsylvania, water use per well has increased over time, which may be explained by increasing horizontal well length, depth, and length of the completed interval (Schmid and Yoxtheimer, 2015).

Water use/consumption at the county scale: In this tri-state region, the highest water use for hydraulic fracturing is in northeastern Pennsylvania counties. On average, operators in Bradford County reported over 1 billion gal (3.8 billion L) used annually in 2011 and 2012 for fracturing; operators in three other counties (Susquehanna, Lycoming, and Tioga Counties) reported 500 million gal (1.9 billion L) or more used annually in each county (Table 4-3). On average, hydraulic fracturing water use is 3.2% compared to 2010 total water use for counties with disclosures in the EPA FracFocus 1.0 project database in these three states (Table 4-3; Appendix Table B-2). Susquehanna County in Pennsylvania has the highest percentages relative to 2010 total water use (47%) and consumption (123%).

Potential for impacts: Water availability is higher in Pennsylvania, West Virginia, and Ohio than in many western states, reducing the likelihood of impacts to drinking water resource quantity and quality. At the county scale, water supplies appear adequate to accommodate this use (Text Box 4-2; Figure 4-6a,b). However, impacts could still occur at the local scale (i.e., specific withdrawal points) as high water availability in a region does not preclude water stress, particularly if water withdrawals occur during seasonal low-flow periods (Entrekin et al., 2015). Without management of the rate and timing of withdrawals, surface water withdrawals for hydraulic fracturing have the potential to affect both drinking water quantity and quality (Mitchell et al., 2013a). For instance, withdrawals may alter natural stream flow regimes, potentially decreasing a stream's capacity to dilute contaminants (Gallegos et al., 2015; Mitchell et al., 2013a; Entrekin et al., 2011; NYSDEC, 2011; van Vliet and Zwolsman, 2008; IPCC, 2007; Environment Canada, 2004; Murdoch et al., 2000).

In a second, multi-scale case study, EPA showed that the potential for water acquisition impacts to drinking water resource quantity and quality increases at finer temporal and spatial resolutions (U.S. EPA, 2015e). They concluded that individual streams in Pennsylvania's SRB can be vulnerable to typical hydraulic fracturing water withdrawals depending on stream size, as defined by contributing basin area (U.S. EPA, 2015e) (Text Box 4-5). They observed infrequent (in less than 1% of withdrawals) high ratios of hydraulic fracturing water consumption to stream flow (high consumption-to-stream flow events). Further research from Barth-Naftilan et al. (2015) in Pennsylvania's Marcellus Shale (SRB and Ohio River Basin (ORB)) confirmed that stream flow alteration due to hydraulic fracturing surface water withdrawals increases at finer spatial scales (i.e., smaller watershed area). They showed that streams with drainage areas under 50 mi^2 (130 km^2) are the most vulnerable to stress induced by flow alteration (Barth-Naftilan et al., 2015).

Text Box 4-5. Case Study: Impact of Water Acquisition for Hydraulic Fracturing on Local Water Availability in the Susquehanna River Basin.

The U.S. EPA (2015e) conducted a second case study analogous to that in the UCRB (Text Box 4-4), to explore the impact of hydraulic fracturing water demand on water availability at the river basin, county, and local scales in the SRB in northeastern Pennsylvania. The study area overlies the Marcellus Shale gas reservoir. Water withdrawal impacts were quantified using a water use intensity index (Text Box 4-4). Researchers obtained detailed site-specific data on hydraulic fracturing water usage from state and regional authorities, and estimated available water supplies using observations at USGS gage stations and empirical and hydrologic modeling.

Most water for fracturing in the SRB is self-supplied by operators from rivers and streams with withdrawal points distributed throughout a wide geographic area. Public water systems provide a relatively small proportion of the water needed. Reuse of wastewater as a percentage of hydraulic fracturing fluid volume averaged 16% from 2008-2013, and has increased over time, reaching 22% in 2013 (SRBC, 2016) (Table 4-2). The Susquehanna River Basin Commission (SRBC) regulates water acquisition for hydraulic fracturing and issues permits that set limits on the volume, rate, and timing of withdrawals at individual withdrawal points; passby flow thresholds (hereafter, passby flows) halt water withdrawals during low flows.

The U.S. EPA (2015e) demonstrated that streams can be vulnerable from hydraulic fracturing water withdrawals depending on their size, as defined by contributing basin area. Small streams have the potential for impacts (i.e., high water use intensity) for all or most of the year. The U.S. EPA (2015e) showed an increased likelihood of impacts in small watersheds in the SRB (less than 10 mi^2 or 26 km^2). Furthermore, they showed that in the absence of passby flows, even larger watersheds (up to 600 mi^2 or 1,554 km^2) could be vulnerable during maximum withdrawal volumes and infrequent droughts. However, high water use intensity calculated from observed hydraulic fracturing withdrawals occurred at only a few withdrawal locations in small streams; local high water use intensity was not found at the majority of withdrawal points.

Detailed studies and state reports available throughout the Marcellus Shale region help provide an understanding of the potential impacts of hydraulic fracturing water withdrawals in both space and time at the local scale (SRBC, 2016; Barth-Naftilan et al., 2015; U.S. EPA, 2015e). In the SRB and ORB, water for hydraulic fracturing is taken from both large rivers and small headwater streams, with a considerable fraction of the water taken from small streams of small watersheds (Barth-Naftilan et al., 2015). The SRBC reports that most natural gas development in the SRB is focused in rural, headwater areas, where withdrawals have the potential to alter natural stream flow regimes (SRBC, 2016). In an analysis of the effects of water withdrawals on twelve streams in the SRB, Shank and Stauffer (2015) found that the largest withdrawals relative to stream size were from headwater streams, where daily withdrawals averaged 6.8% of average daily flows. However, they found water management in the form of low flow protections helped limit the potential for impacts.

Compared to conventional energy extraction, hydraulic fracturing consumes more water in a highly concentrated period of time (Patterson et al., 2016); thus, the cumulative impact of multiple wells withdrawing water from small streams, particularly during drought or seasonal low flows, has the potential to impact the quantity and quality of drinking water resources (Patterson et al., 2016). For instance, in modeling the potential future impact of hydraulic fracturing in the Delaware River Basin (DRB), Habicht et al. (2015) showed that under maximum well development, hydraulic fracturing water withdrawals from small streams could remove up to 70% of water during periods of low stream flow, and less than 3% during periods of normal stream flow.[1] Unlike groundwater withdrawals, any impacts to drinking water resource quantity and quality associated with surface water withdrawals are likely to persist for a shorter time period since the rate of replenishing water removed from the system is greater in surface water than groundwater (Alley et al., 1999) (Section 4.5.1).

The potential for water acquisition impacts to drinking water resource quality in this region is also greatest in small, unregulated streams, particularly under drought conditions or during seasonal low flows (U.S. EPA, 2015e; Vengosh et al., 2014; Mitchell et al., 2013a; Vidic et al., 2013; Rahm and Riha, 2012; Rolls et al., 2012; Kargbo et al., 2010; McKay and King, 2006). Surface water quality impacts may be of concern if a pollution discharge point (e.g., sewage treatment plant, agricultural runoff, or chemical spill) is immediately downstream of a hydraulic fracturing withdrawal point (U.S. EPA, 2015e; NYSDEC, 2011).[2] Potential water quality impacts associated with reduced water levels may also include possible interference with the efficiency of drinking water treatment plant operations, as increased contaminant concentrations in drinking water sources may necessitate additional treatment and ultimately impact drinking water quality (Water Research Foundation, 2014; Benotti et al., 2010).[3]

Water management policies in place in this region can help reduce the potential for impacts associated with hydraulic fracturing water withdrawals, including excessive lowering of water levels, unreliable water supplies, and degradation of water quality (SRBC, 2016; Barth-Naftilan et al., 2015; U.S. EPA, 2015e) (Text Box 4-5). For instance, the SRBC manages the quantity, location, and timing of withdrawals, using site-specific information to set instantaneous and daily withdrawal limits for all approved surface water and groundwater withdrawals. They also set low flow protections, known as passby flows, for most approved surface water withdrawals that require withdrawals to cease when stream flow drops below a prescribed threshold level (SRBC, 2016). Passby flows can reduce the frequency of high consumption-to-stream flow events, particularly in the smallest streams (Shank and Stauffer, 2015; U.S. EPA, 2015e).

Overall, there appears to be adequate surface water for hydraulic fracturing in Pennsylvania, West Virginia, and Ohio, but there is still the potential for impacts to both drinking water resource quantity and quality, particularly in small streams, if the rate and timing of withdrawals are not managed (U.S. EPA, 2015e). These potential impacts are expected to be localized in space (i.e.,

[1] Presently there is a moratorium on hydraulic fracturing in the DRB, which spans Pennsylvania, Delaware, New Jersey, and New York. Habicht et al. (2015) modeled the potential future environmental impact of hydraulic fracturing in the DRB should the moratorium be lifted, allowing hydraulic fracturing to expand into this region in the future.

[2] Aside from direct surface water withdrawals, unmanaged withdrawals from public water systems can cause cross-contamination if there is a loss of pressure, allowing the backflow of pollutants from tank trucks into the distribution system. The state of Ohio has issued a fact sheet relevant to this potential concern, intended specifically for public water systems providing water to oil and gas companies (Ohio EPA, 2012a). To prevent potential cross-contamination, Ohio requires a backflow prevention device at cross-connections. For example, bulk loading stations that provide public supply water directly to tank trucks are required to have an air-gap device at the cross-connection to prevent the backflow of contaminants into the public water system (Ohio EPA, 2012a).

[3] For instance, an increased proportion of organic matter entering a treatment plant may increase the formation of trihalomethanes, byproducts of the disinfection process formed as chlorine reacts with organic matter in the water being treated (Water Research Foundation, 2014).

occurring at specific withdrawal points), and time (e.g., low flow periods). Passby flows appear to be an effective water management tool for reducing the potential for impacts from surface water withdrawals.

4.5.4 North Dakota and Montana

North Dakota was fourth in the number of disclosures in the EPA FracFocus 1.0 project database (5.9% of disclosures) (Appendix Table B-5; Figure 4-4). We combine Montana with North Dakota, because both overlie the Williston Basin (which contains the Bakken play, shown in Figure 4-11), although many fewer wells are reported for Montana (Appendix Table B-5). The Williston Basin is the only basin with significant activity reported for either state, though other basins are also present in Montana (e.g., the Powder River Basin).

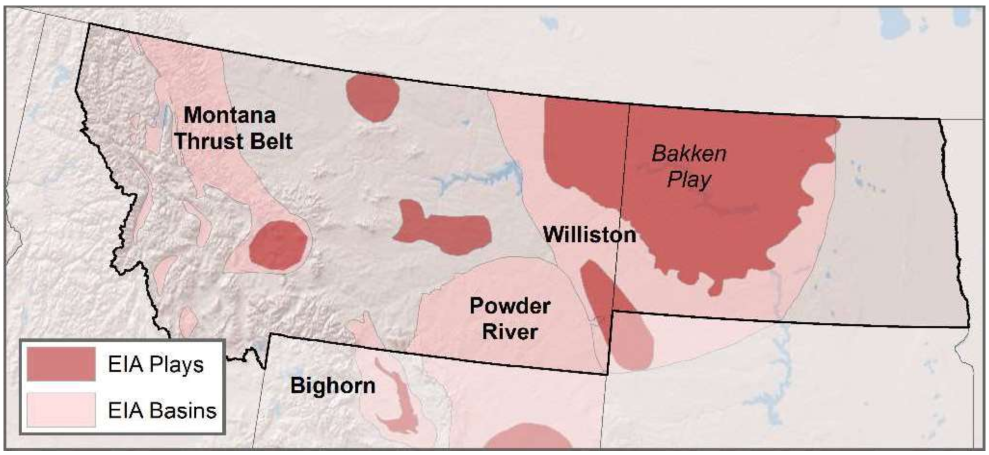


Figure 4-11. Major U.S. EIA shale plays and basins for North Dakota and Montana.
Source: EIA (2015).

Types of water used: Hydraulic fracturing in the Bakken play depends on both ground and surface water resources. Surface water from the Missouri River system provides the largest source of fresh water in the center of Bakken oil development (North Dakota State Water Commission, 2014; EERC, 2011, 2010; North Dakota State Water Commission, 2010). Apart from the Missouri River system, regional surface waters (e.g., smaller streams) do not provide a consistent supply of water for the oil industry due to seasonal stream flow variations. Sufficient stream flows generally occur only in the spring after snowmelt (EERC, 2011). Groundwater from glacial and bedrock aquifer systems has traditionally supplied much of the water needed for Bakken development, but concerns over limited groundwater supplies have led to limits on the number of new groundwater withdrawal permits issued (Ceres, 2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission, 2010).

The water used for Bakken development is mostly fresh. The EPA FracFocus report shows that "fresh" was the only source of water listed in almost all disclosures reporting a source of water in North Dakota (U.S. EPA, 2015b).[1] Reuse of Bakken wastewater is limited due to its high TDS, which

[1] Twenty-five percent of North Dakota disclosures included information related to water sources (U.S. EPA, 2015b).

presents challenges for treatment and reuse (Gadhamshetty et al., 2015). Industry is currently researching treatment technologies for reuse of this wastewater (Ceres, 2014; EERC, 2013, 2011).

Water for hydraulic fracturing is commonly purchased from municipalities or other public water systems in the region. The water is often delivered to trucks at water depots or transported directly to well pads via pipelines (EERC, 2011).

Water use per well: Water use per well is intermediate compared with other areas, with a median of 2.0 and 1.6 million gal (7.6 and 6.1 million L) per well in the Williston Basin in North Dakota and Montana, respectively, according to the EPA's FracFocus 1.0 project database (Appendix Table B-5). The North Dakota State Water Commission reports similar volumes (2.2 million gal (8.3 million L) per well on average for North Dakota) in a summary fact sheet (North Dakota State Water Commission, 2014).[1] Scanlon et al. (2016) show that average water use per well in the Bakken play has increased over time, from 580,000 gal (2.2 million L) in 2005 to 3.7 million gal (14.1 million L) in 2014, due in part to the increasing lengths of laterals in horizontal drilling.

In addition to water for hydraulic fracturing, Bakken wells may require "maintenance water" (Scanlon et al., 2016; Scanlon et al., 2014a). This extra water is reportedly needed because of the relatively high salt content of Bakken brine, potentially leading to salt buildup, pumping problems, and restriction of oil flow. Based on estimates from the North Dakota Department of Mineral Resources, Scanlon et al. (2016) report that approximately 400 – 600 gal (1,500 – 2,300 L) per day per each well may be required for well maintenance. Assuming a 15-year lifetime for wells, this could add up to 3.3 million gal (12.5 million L) per well of additional water (Scanlon et al., 2016).

Water use/consumption at the county scale: Water use for fracturing in this region is greatest in the northwestern corner of North Dakota (Gadhamshetty et al., 2015). Hydraulic fracturing water use in 2011 and 2012 averaged approximately 123 million gal (466 million L) per county in the two-state area, with use in McKenzie and Williams Counties in North Dakota exceeding 500 million gal (1.9 billion L) (Appendix Table B-2). There were four counties where 2011 and 2012 average hydraulic fracturing water use was 10% or more of 2010 total water use. Mountrail and Dunn Counties showed the highest percentages (36% and 29%, respectively). Outside of North Dakota's northwest corner, hydraulic fracturing used much less water in the rest of the state and Montana (Table 4-3; Appendix Table B-2).

Potential for impacts: In this region, there are concerns about over-pumping groundwater resources, but the potential for impacts appears to be low provided the Missouri River is determined to be a sustainable and usable source. This finding of a low potential for impacts is also supported by the comparison of hydraulic fracturing water use to water availability at the county scale (Text Box 4-2; Figure 4-6a,b). This area is primarily rural, interspersed with small towns. Residents rely on a mixture of surface water and groundwater for domestic use depending on the county, with most water supplied by local municipalities (Appendix Table B-6).

[1] The fact sheet is a stand-alone piece, and it is not accompanied by an underlying report.

The state of North Dakota and the U.S. Army Corps of Engineers concluded that groundwater resources in western North Dakota are not sufficient to meet the needs of the oil and gas industry (U.S. Army Corps of Engineers, 2011; North Dakota State Water Commission, 2010). All users combined currently withdraw approximately 6.2 billion gal (23.5 billion L) of water annually in an 11-county region in western North Dakota, already stressing groundwater supplies (U.S. Army Corps of Engineers, 2011). By comparison, the total needs of the oil and gas industry are projected to range from approximately 2.2 and 8.8 billion gal (8.3 and 33.3 billion L) annually by the year 2020 (U.S. Army Corps of Engineers, 2011).

Due to concerns for already stressed groundwater supplies, the state of North Dakota limits industrial groundwater withdrawals, particularly from the Fox Hills-Hell Creek aquifer (Ceres, 2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission, 2010). Currently, the oil industry is the largest industrial user of water from the Fox Hills-Hell Creek aquifer (North Dakota State Water Commission, 2010). Many farms, ranches, and some communities in western North Dakota rely on flowing wells from this artesian aquifer, particularly in remote areas that lack electricity for pumping; however, low recharge rates and withdrawals throughout the last century have resulted in steady declines in the formation's hydraulic pressure (North Dakota State Water Commission, 2010). Declines in hydraulic pressure do not appear to be associated with impacts to groundwater quality; rather, the state is concerned with maintaining flows for users (North Dakota State Water Commission, 2010).

To reduce demand for groundwater, the state is encouraging the industry to seek surface water withdrawals from the Missouri River system. The North Dakota State Water Commission concluded the Missouri River and its dammed reservoir, Lake Sakakawea, are the only plentiful and dependable water supplies for the oil industry in western North Dakota (North Dakota State Water Commission, 2010). In 2011, North Dakota authorized the Western Area Supply Project, by which Missouri River water (via the water treatment plant in Williston, North Dakota) will be supplied to help meet water demands, including for oil and gas development, of the state's northwest counties (WAWSA, 2011). In July 2012, the U.S. Army Corps of Engineers made available approximately 32.6 billion gal (123 billion L) of water per year from Lake Sakakawea for municipal and industrial water demands over the next ten years (U.S. Army Corps of Engineers, 2011). The Army Corps estimated that the oil and gas industry could use up to 8.8 billion gal (33.3 billion L) annually during this time period in the 11-county surrounding area, and included this as part of the 32.6 billion gal total (123 billion L) to be made available (U.S. Army Corps of Engineers, 2011). For context, annual water use for hydraulic fracturing in all North Dakota counties combined was approximately 2.2 billion gal (8.3 billion L) per year in 2011 and 2012 according to EPA's FracFocus 1.0 project database (Appendix Table B-2). As such, Lake Sakakawea appears to be an adequate resource to meet the water demands of hydraulic fracturing in the region at least in the near term.

4.5.5 Arkansas and Louisiana

Arkansas and Louisiana were ranked seventh and tenth in the number of disclosures in the EPA FracFocus 1.0 project database, respectively (Appendix Table B-5). Hydraulic fracturing activity in Louisiana occurs primarily in the TX-LA-MS Salt Basin, which contains the Haynesville play; activity in Arkansas is dominated by the Arkoma Basin, which contains the Fayetteville play (Figure 4-12).

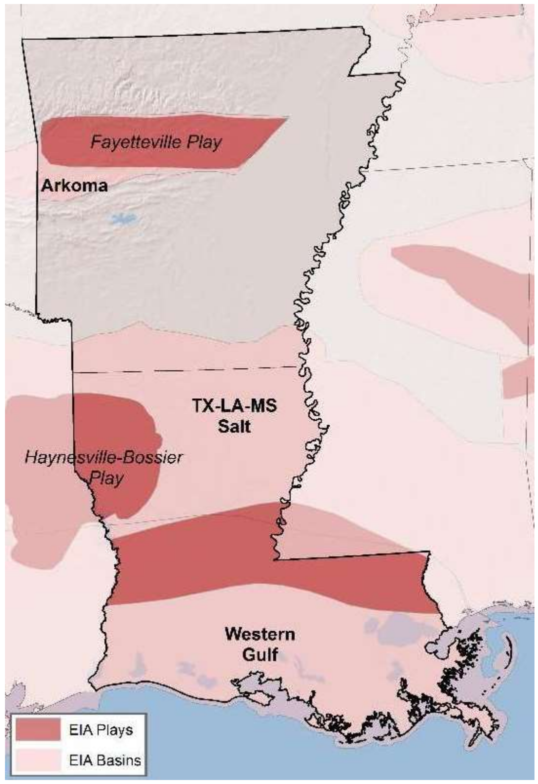


Figure 4-12. Major U.S. EIA shale plays and basins for Arkansas and Louisiana.
Source: EIA (2015).

Types of water used: Surface water is reported as the primary source of water for hydraulic fracturing operations in both Arkansas and Louisiana (ANRC, 2014; LA Ground Water Resources Commission, 2012; STRONGER, 2012). Quantitative information is lacking for Arkansas on the proportion of water sourced from surface versus groundwater. However, data are available for Louisiana, where an estimated 87% of water for hydraulic fracturing in the Haynesville Shale is from surface water (LA Ground Water Resources Commission, 2012) (Table 4-1). In 2008, during the early stages of development, hydraulic fracturing in Louisiana relied heavily on groundwater from the Carrizo-Wilcox aquifer, and concerns for the sustainability of groundwater resources prompted the state to encourage surface water withdrawals (LA Ground Water Resources Commission, 2012).

The EPA FracFocus report suggests that significant reuse of wastewater may occur in Arkansas to offset total fresh water used for hydraulic fracturing; 70% of all disclosures reporting a water source indicated a blend of "recycled/surface," whereas 3% of disclosures reporting a water source noted "fresh" as the exclusive water source (U.S. EPA, 2015b).[1] According to Veil (2011), Arkansas'

[1] Ninety-three percent of Arkansas disclosures included information related to water sources (U.S. EPA, 2015b).

Fayetteville Shale wastewater is of relatively good quality (i.e., low TDS), facilitating reuse.[1] Data are generally lacking on the extent to which hydraulic fracturing wastewater is reused in Louisiana.

Water use per well: Arkansas and Louisiana have the highest median water use per well of the states we considered from the EPA FracFocus 1.0 project database, at 5.3 million and 5.1 million gal (20.1 million and 19.3 million L), respectively (Appendix Table B-5).[2]

Water use/consumption at the county scale: On average, hydraulic fracturing uses 408 million gal (1.54 billion L) of water each year in Arkansas counties reporting activity, or 9.3% of 2010 total county water use (26.9% of total county consumption) (Appendix Table B-2). In 2011 and 2012, five counties dominated fracturing water use in Arkansas: Cleburne, Conway, Faulkner, Van Buren, and White Counties (Appendix Table B-2). Van Buren, which is sparsely populated and thus has relatively low total water use and consumption, is by far the Arkansas county highest in hydraulic fracturing water use and consumption relative to 2010 total water use and consumption (56% and 168%, respectively) (Table 4-3).

In Louisiana, hydraulic fracturing water use is concentrated in six parishes in the far northwestern corner of the state, associated with the Haynesville play.[3] On average in 2011 and 2012, hydraulic fracturing used 117 million gal (443 million L) of water annually per parish, representing approximately 3.6% and 10.8% of 2010 total water use and consumption, respectively (Appendix Table B-2). Operators in DeSoto Parish used the most water (over 1 billion gal (3.8 billion L) annually). Hydraulic fracturing water use and consumption was highest relative to 2010 total water use and consumption (35.5% and 83.2%, respectively) in Red River Parish (Table 4-3). These numbers may be low estimates, since Louisiana required disclosures to the state or FracFocus, and Arkansas required disclosures to the state but not FracFocus, during the time period analyzed (U.S. EPA, 2015b) (Appendix Table B-5).

Potential for impacts: Water availability is generally higher in Arkansas and Louisiana than in states farther west, reducing the potential for impacts to drinking water quantity and quality (Figure 4-6a, Figure 4-7a; Text Box 4-2). However, generally high water availability in this region does not preclude the potential for impacts at the local scale, particularly if surface water withdrawals occur during seasonal low flow periods. For instance, precipitation is highest in Arkansas in the late autumn and winter, with little rainfall occurring in the late spring and summer; thus, most small streams do not flow year round (Entrekin et al., 2015). Hydraulic fracturing surface water withdrawals from small streams during seasonal low flows have the potential to impact the quantity and quality of drinking water resources.

Additionally, in northwestern Louisiana, there are concerns about over-pumping of groundwater resources. Prior to 2008, most operators in the Louisiana portion of the Haynesville Shale used groundwater, withdrawing from the Carrizo-Wilcox, Upland Terrace, and Red River Alluvial aquifer

[1] Veil (2011) reports a range of 20,000-25,000 ppm TDS for Fayetteville Shale wastewater.

[2] According to STRONGER (2012) and STRONGER (2011a), both states require disclosure of information on water use per well, but this has not been synthesized into state level reports to date.

[3] Louisiana is divided into parishes, which are similar to counties in other states.

systems (LA Ground Water Resources Commission, 2012). To mitigate stress on groundwater, the state issued a water use advisory to the oil and gas industry that recommended Haynesville Shale operators seek alternative water sources to the Carrizo-Wilcox aquifer, which is predominantly used for public supply (LDEQ, 2008). Operators then transitioned to mostly surface water, with a smaller groundwater component (approximately 13% of all fracturing water used) (LA Ground Water Resources Commission, 2012). Of this groundwater component, the majority (approximately 74%) still came from the Carrizo-Wilcox aquifer (LA Ground Water Resources Commission, 2012).

Although the potential for hydraulic fracturing withdrawals to affect water supplies and water quality in the aquifer was reduced, it was not entirely eliminated. Despite Louisiana's water use advisory, a combination of drought conditions and higher than normal withdrawals (for all uses, not solely hydraulic fracturing) from the Carrizo-Wilcox and Upland Terrace aquifers caused several water wells to go dry in July 2011 (LA Ground Water Resources Commission, 2012). In August 2011, a groundwater emergency was declared for southern Caddo Parrish (LA Ground Water Resources Commission, 2012). Hydraulic fracturing withdrawals contributed to these conditions, alongside other users of water and the lack of precipitation.

4.6 Chapter Synthesis

In this chapter, we examined the potential for water acquisition for hydraulic fracturing to impact the quantity and quality of drinking water resources, and identified factors affecting the frequency or severity of impacts. Whether impacts occur from water acquisition for hydraulic fracturing depends on the local balance between water withdrawals and availability, and this balance can be modified by a combination of site or regional-specific factors. For this reason, information is needed at the local scale to determine whether impacts actually occur, yet this information is not available in many locations where hydraulic fracturing takes place; see Section 4.6.3 on Uncertainties below. Despite these limitations, our chapter used the scientific literature, county level assessments, and, where available, local case studies to point to areas with a higher potential for impacts; understand local dynamics, including example cases of impacts; and identify common factors that increase or decrease the frequency or severity of impacts. In this section, we summarize our major findings regarding hydraulic fracturing water acquisition activities, potential impacts, and these common factors (4.6.1 and 4.6.2). We then discuss uncertainities (4.6.3), and provide final conclusions (4.6.4).

4.6.1 Major Findings

The first half of this chapter focused on water acquisition activities, providing an overview of the types of water used (including sources, quality, and provisioning), water use per well, and water use and consumption at the national, state, and county scale. The three major types of water used for hydraulic fracturing are surface water, groundwater, and reused hydraulic fracturing wastewater. Because trucking can be a major expense, operators tend to use water sources as close to the well pad as possible. Operators usually self-supply surface water or groundwater directly, but may also obtain water from public water systems or other suppliers. Hydraulic fracturing operations in the eastern United States rely predominantly on surface water, whereas operations in more semi-arid to arid western states use either surface water or groundwater. There are areas of the country that rely entirely on groundwater supplies (e.g., western Texas).

Reuse of wastewater reduces the demand on fresh water sources, which currently supply the vast majority of water used for hydraulic fracturing. The proportion of the water used in hydraulic fracturing that comes from reused hydraulic fracturing wastewater is generally low; in a survey of literature values from 10 states, basins, or plays, we found a median value of 5%, with this percentage varying by location (Table 4-2).[1] Available data on reuse trends indicate increasing reuse of wastewater over time in both Pennsylvania and West Virginia, likely due to the lack of nearby disposal options in Class II wells. Reuse as a percentage of water injected is typically lower in other areas of the United States, likely in part because of the availability of disposal wells; see Chapter 8 for more information.

The median amount of water used nationally per hydraulically fractured well was approximately 1.5 million gal (5.7 million L) in 2011 through early 2013 based on the EPA analysis of FracFocus disclosures (U.S. EPA, 2015b, c). This increased to approximately 2.7 million gal (10.2 million L) in 2014, driven by a proportional increase in horizontal wells (estimated from data in Gallegos et al., 2015). These national estimates represent a variety of fractured well types, including types requiring much less water per well than horizontal shale gas wells. Thus, published estimates for horizontal shale gas wells are typically higher (e.g., approximately 4 million gal (15 million L) per well (Vengosh et al., 2014), and should not be applied to all fractured wells to derive national estimates. There was also wide variation within and among states and basins in the median per well water volumes reported in 2011 and 2012, from more than 5 million gal (19 million L) in Arkansas and Louisiana to less than 1 million gal (3.8 million L) in Colorado, Wyoming, Utah, New Mexico, and California (U.S. EPA, 2015c). This variation can result from several factors, including geologic formation, well length, and fracturing fluid formulation.

Hydraulic fracturing uses billions of gallons of water every year at the national and state scales, and even in some counties. When expressed relative to total water use or consumption at these scales, however, hydraulic fracturing generally accounts for only a small percentage, usually less than 1%. These percentages are higher though in specific counties. Annual hydraulic fracturing water use was 10% or more compared to 2010 total water use in 6.5% of counties with FracFocus disclosures in 2011 and 2012 in the EPA FracFocus 1.0 project database, 30% or more in 2.2% of counties, and 50% or more in 1.0% of counties (Appendix Table B-2). Consumption estimates follow the same pattern, with higher percentages in each category: hydraulic fracturing water consumption was 10%, 30%, and 50% or more of 2010 total water consumption in 13.5%, 6.2%, and 4.0% of counties with FracFocus disclosures in the EPA FracFocus 1.0 project database (Appendix Table B-2). Thus, hydraulic fracturing represents a relatively large user and consumer of water in these counties.

Whether water quantity or quality impacts occur from water acquisition for hydraulic fracturing depends on the local balance between water withdrawals and availability. From our survey of the literature and our county level assessments, southern and western Texas appear to have the

[1] Note that reused water as a percentage of total water injected differs from the percentage of wastewater that is reused. See Section 4.2 and Chapter 8 for more information.

highest potential for impacts of the areas assessed in this chapter, given the combination of high hydraulic fracturing water use, relatively low water availability, intense periods of drought, and reliance on declining groundwater resources; see Section 4.6.2 on Factors below. Importantly, our results do not preclude the possibility of local water impacts in areas with comparatively lower potential, nor do they necessarily mean impacts have occurred in the high potential areas. Our survey, however, provides an indicator of areas with higher *potential* for impacts, and could be used to target resources or future studies.

In two example cases, local impacts to drinking water resources occurred in areas with increased hydraulic fracturing activity. In a detailed case study, Scanlon et al. (2014b) observed generally adequate water supplies for hydraulic fracturing in the Eagle Ford play in southern Texas, except in specific locations. They found excessive drawdown of groundwater locally, with estimated declines of ~100-200 ft (30-60 m) in a small proportion of the play (~6% of the area) after hydraulic fracturing activity increased in 2009. In 2011, drinking water wells in an area overlapping with the Haynesville Shale ran out of water due to higher than normal groundwater withdrawals and drought (LA Ground Water Resources Commission, 2012). Hydraulic fracturing water withdrawals contributed to these conditions, along with other water users and the lack of precipitation. By contrast, two EPA case studies in the Upper Colorado and the Susquehanna River Basins found minimal impacts from hydraulic fracturing withdrawals currently (U.S. EPA, 2015e) (Sections 4.5.2, 4.5.3).

These site-specific findings emphasize the need to focus on regional and local dynamics when considering the impacts from hydraulic fracturing water withdrawals. The case studies and the scientific literature as a whole suggest some common factors that increase or decrease the frequency or severity of impacts. These are summarized in the section below.

4.6.2 Factors Affecting Frequency or Severity of Impacts

The potential for impacts depends on the combination of water withdrawals and water availability at a given withdrawal location. Where water withdrawals are relatively low compared to water availability, impacts are unlikely to occur. Where water withdrawals are relatively high compared to water availability, impacts are more likely.

Areas reliant on declining groundwater are particularly vulnerable to more frequent and severe impacts from cumulative water withdrawals, including withdrawals for hydraulic fracturing. Groundwater recharge rates can be extremely low, and groundwater pumping is exceeding recharge rates in many areas of the country (Konikow, 2013). When pumping exceeds recharge, the cumulative effects of withdrawals are manifested in declining water levels. For this reason, water levels in many aquifers in the United States have declined substantially over the last century (Konikow, 2013). Cumulative drawdowns can affect surface water bodies since groundwater can be the source of base flow in streams (Winter et al., 1998), and alter groundwater quality by mobilizing chemicals from geologic sources, among other means (DeSimone et al., 2014; Alley et al., 1999). Although in many of these areas (e.g., the Ogallala aquifer), irrigated agriculture is the dominant user of groundwater, hydraulic fracturing withdrawals now also contribute to declining groundwater levels. Hydraulic fracturing groundwater consumption, for example, exceeds estimated recharge rates in the seven most active hydraulic fracturing counties in the Eagle Ford Shale in southern Texas (Steadman et al., 2015). When necessary, state and local governments have encouraged or mandated industry to use surface water over groundwater, as evidenced in both Louisiana and North Dakota.

Among surface water sources, smaller streams, even in humid areas, are more vulnerable to frequent and severe impacts from withdrawals. A detailed EPA case study found that streams with the smallest contributing areas in northeastern Pennsylvania were particularly vulnerable to withdrawals (U.S. EPA, 2015e). Protecting smaller streams from excessive withdrawals is probably most important for aquatic life, but may also protect drinking water quantity and quality in certain instances.

Seasonal or long-term drought can also make impacts more frequent and severe for surface water and groundwater sources. Hot, dry weather depletes surface water bodies and reduces or prevents groundwater recharge, while water demand often increases simultaneously (e.g., for irrigation). The EPA case study in Pennsylvania found that even large streams could be vulnerable to withdrawals during times of low flows (U.S. EPA, 2015e). Much of the western United States has experienced prolonged periods of drought over the last decade (Figure 4-8). This dynamic will likely be magnified by future climate change in certain locations (Meixner et al., 2016).

By contrast to the above factors, consumption of water for hydraulic fracturing does not appear to substantially influence the frequency or severity of impacts. There are concerns that hydraulic fracturing permanently removes water from the hydrologic cycle, posing a threat to long-term water supplies. Since impacts occur locally and depend on the local water balance, impacts can occur regardless of whether the water is withdrawn and returned to the larger hydrologic cycle elsewhere or whether it is permanently sequestered underground. We acknowledge that whether the water is returned to the larger hydrologic cycle may make a difference for the water budget of a larger area, such as on the state, regional, or national scale. For example, water converted to steam during thermoelectric cooling in one location may condense and fall as precipitation in an adjacent state or region. At these larger scales, however, hydraulic fracturing water consumption is a very small fraction of total water availability.[1] Plus, at these scales, there are other larger factors that can affect regional water budgets, but which are out of scope for this assessment.[2] For these reasons, focusing on consumption distracts from the more salient issue that impacts depend upon the spatial and temporal balance between local water withdrawals and availability.

---

[1] For example, hydraulic fracturing used approximately 3.3 billion gal (12.5 billion L) of water on average annually in all Colorado counties with hydraulic fracturing activities combined according to FracFocus disclosures in 2011 and 2012 (Appendix B-1). Using the consumption rate of 82.5% yields a consumption estimate of approximately 2.7 billion gal (10.2 billion L). This would be approximately 0.1% of the fresh water and total water availability metrics used in Textbox 4-2 for all of those same counties combined (approximately 2.6 trillion gal (9.8 trillion L) of fresh water and total water available).

[2] The combustion of methane produced by hydraulic fracturing, for example, adds water molecules to the environment, and at large scales, this may affect regional water budgets. However, quantifying this is outside the scope of this assessment. Similarly, there are other larger factors (e.g., water used for cooling thermoelectric power plants) that can affect regional water budgets, but these are also outside the scope of this assessment.

There are also factors that can decrease the frequency and severity of any impacts from water withdrawals. The literature suggests that water management, particularly wastewater reuse, the use of brackish groundwater, the use of passby flows, and transitioning from limited groundwater sources to more abundant surface water sources can reduce impacts. Reuse is not a universal solution, since in many areas of the country wastewater volumes from one well are often a small percentage of the water needed to fracture the next well. In the Marcellus Shale, for instance, 100% reuse of the wastewater produced from one well means reducing fresh water demand by 10 or 30% for the next (Section 4.2.1; Chapter 7). Nevertheless, reuse can be an important local factor reducing fresh water demand.

Switching to brackish water is another means by which fresh water demand can be—and is in some locations—reduced. This is a source of alternative water in western and southern Texas, for example. In these areas, use of brackish water is currently reducing impacts to fresh water sources, and could with continued use reduce future impacts (Scanlon et al., 2014b; Nicot et al., 2012). Our county level estimates suggest that brackish water could readily meet the volume demanded by hydraulic fracturing in Texas.

Water management also includes passby flows, a low stream flow threshold below which withdrawals are not allowed. Evidence suggests passby flows can be effective in protecting streams from hydraulic fracturing water withdrawals (U.S. EPA, 2015e). Finally, as evidenced by examples in both North Dakota and Louisiana, water management may include transitioning from declining groundwater sources to surface water, if available.

4.6.3 Uncertainties

There are several uncertainties inherent in our assessment of the potential impacts of water acquisition for hydraulic fracturing. The largest uncertainties stem from the lack of literature and data on this subject at local scales. Because impacts occur at a given withdrawal point, our assessment could assess the potential for impacts, but often could not determine if potential impacts were realized in the absence of local data. The exceptions were local case studies from the Eagle Ford play in Texas, the Upper Colorado River Basin in Colorado, and the Susquehanna River Basin in Pennsylvania. Moreover, it is also not clear if local impacts, for example a drinking water well going dry, are likely to be documented in the scientific literature.

Other uncertainties arise from data limitations on the volume and types of water used or consumed for hydraulic fracturing, future water use projections, and water availability estimates. There are no nationally consistent data sources, and therefore, water use estimates must be based on multiple, individual pieces of information. For example, in their National Water Census, the USGS includes hydraulic fracturing in the broader category of “mining” water use, but hydraulic fracturing water use is not reported separately (Maupin et al., 2014). There are locations where average annual hydraulic fracturing water use in 2011 and 2012 in the EPA FracFocus 1.0 project database exceeded total mining water use in 2010, and one county where it exceeded all water use (U.S. EPA, 2015c; Maupin et al., 2014). This could be due to a rapid increase in hydraulic fracturing water use, differences in methodology between the two databases (i.e., the USGS 2010 National Water Census and the EPA FracFocus 1.0 project database), or both.

We used the EPA FracFocus 1.0 project database for water use estimates, which itself has limitations. Many states in the project database did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015b). We conclude that this likely does not change the overall hydraulic fracturing water use patterns observed across the United States (Text Box 4-1), but could affect particular county level estimates. Also, the database covered the time period of 2011 through early 2013. Thus, changes in the industry since then are not reflected in these data.

Hydraulic fracturing water use data that are often provided as water use associated with a particular well. While this is valuable information, the potential impacts of water acquisition for hydraulic fracturing could be better assessed if data were also available at the withdrawal point. If the total volume, date, location, and type (i.e., surface water or groundwater; and fresh, brackish, or reused wastewater) of each water withdrawal were documented, effects on availability could be better estimated. For example, surface withdrawal points could be aggregated by watershed or aquifer to estimate effects on downstream flow, groundwater levels, and water quality. Some of this information is available in disparate forms, but the lack of nationally consistent data on water withdrawal locations, timing, and amounts—data that are publicly available, easy to access and analyze—limits our assessment of potential impacts. The Susquehanna River Basin Commission collects this type of detailed data on hydraulic fracturing water withdrawals, but this type of information is not widely available across the nation.

Future hydraulic fracturing water use is also a source of uncertainty. Because water withdrawals and potential impacts are concentrated in certain localized areas, water use projections need to match this scale. Projections are available for Texas at the county scale, but more information at the county or sub-county scale is needed in other states with hydraulic fracturing activity and water availability concerns (e.g., northwest North Dakota, eastern Colorado). Due to a lack of data, we generally could not assess future water use and the potential for impacts in most areas of the country, nor could we examine these in combination with other relevant factors (e.g., climate change or population growth).

4.6.4 Conclusions

With notable exceptions, hydraulic fracturing uses and consumes a relatively small percentage of water when compared to total use, consumption, and availability at the national, state, and county scale. Despite this, impacts on drinking water resource quantity and quality from hydraulic fracturing water acquisition can occur at the local scale, because hydraulic fracturing water withdrawals are often concentrated in space and time, and impacts depend upon the local balance between withdrawals and availability. In two example cases, local impacts to drinking water resource quantity occurred in areas with increased hydraulic fracturing activity (e.g., in Texas's Eagle Ford play, and in Louisiana's Haynesville Shale). Declining groundwater resources, especially in the western United States, are particularly vulnerable to withdrawals, as are smaller streams, even in the more humid East. Finally, there are factors that increase or decrease the frequency and severity of impacts—included in this are times of low water availability, such as during drought, which can increase the frequency and severity of impacts, or conversely water management practices (e.g., shifting to brackish water, or passby flows), which can help protect drinking water resources.

Chapter 5. Chemical Mixing

Abstract

This chapter provides an analysis of the potential impacts on drinking water resources during the chemical mixing stage of the hydraulic fracturing water cycle and the factors governing the frequency and severity of these impacts. The chemical mixing stage includes the mixing of base fluid (90% to 97% by volume, typically water), proppant (2% to 10% by volume, typically sand), and additives (up to 2% by volume) on the well pad to make hydraulic fracturing fluid. This fluid is engineered to create and extend fractures in the targeted formation and to carry proppant into the fractures. Concentrated additives are delivered to the well pad and stored on site, often in multiple, closed containers, and moved around the well pad in hoses and tubing.

Changes in drinking water quality can occur if spilled fluids reach groundwater or surface water resources. In this assessment, a spill is considered to be any release of fluids. The EPA's analysis found that spills and releases of chemicals and fluids have occurred during the chemical mixing stage and have reached soil and surface water receptors. Spills of hydraulic fracturing fluids or additives included in the analysis had a median spill volume of 420 gal (1,590 L), with a range of 5 to 19,320 gal (9 to 72,130 L). Spills were caused most often by equipment failure or human error. The potential for spilled fluids to reach, and therefore impact, groundwater or surface water resources depends on the composition of the spilled fluid, spill characteristics, spill response activities, and the fate and transport of the spilled fluid.

The movement of spilled hydraulic fracturing fluids and chemicals through the environment is difficult to predict, because spills are site- and chemical-specific, and because hydraulic fracturing-related spills are typically complex mixtures of chemicals. Physicochemical properties, which depend on the molecular structure of a chemical, govern whether spilled chemicals volatilize, sorb, transform, and travel. Spill prevention practices and spill response activities can prevent spilled fluids from reaching ground or surface drinking water resources.

The severity of potential impacts on water quality from spills of additives or hydraulic fracturing fluids depends on the identity and amount of chemicals that reach ground or surface water resources, the hazards associated with the chemicals, and the characteristics of the receiving water body. The lack of monitoring following spills, along with the lack of publicly available information on the composition of additives and fracturing fluids, containment and mitigation measures in use, the proximity of chemical mixing to drinking water resources, and the fate and transport of spilled fluids limits the EPA's ability to fully assess potential impacts on drinking water resources and their frequency and severity. This chapter shows that spills of additives and hydraulic fracturing fluids during the chemical mixing stage of the hydraulic fracturing water cycle have occurred and have reached and impacted drinking water resources.

This page is intentionally left blank.

5. Chemical Mixing

5.1 Introduction

This chapter provides an analysis of the potential impacts on drinking water resources during the chemical mixing stage of the hydraulic fracturing water cycle and the factors governing the frequency and severity of these impacts. Chemical mixing is a complex process that requires the use of specialized equipment and a range of different additives to produce the fluid that is injected into a well to fracture the formation. This fluid, the hydraulic fracturing fluid, generally consists of a base fluid (typically water), a proppant (typically sand), and additives (chemicals), although there is no standard or single composition of hydraulic fracturing fluid used. The number, type, and amount of chemicals used to create the hydraulic fracturing fluid vary from well to well based on site- and operator-specific factors. Spills may occur at any point in the chemical mixing process.[1] The potential for spilled fluids to reach, and therefore impact, ground or surface water resources depends on the composition of the spilled fluid, spill characteristics, spill response activities, and the fate and transport of the spilled fluid. This chapter is structured around these concepts.

The chapter starts by discussing the characteristics of hydraulic fracturing fluids (Sections 5.2 to 5.4). This includes an introductory overview of the chemical mixing process (Section 5.2), a description of the different components of the hydraulic fracturing fluid (Section 5.3), the range of different chemicals used and their classes, the most frequently used chemicals nationwide, and volumes used (Section 5.4).[2] (Appendix H provides a list of chemicals that the EPA identified as being used in hydraulic fracturing fluids.)

The chapter continues with a discussion on how chemicals are managed on the well pad, the characteristics of spills when they occur, and spill response activities (Sections 5.5 to 5.7). This includes a description on how potential impacts of a spill on drinking water resources depends upon chemical management practices, such as storage, on-site transfer, and equipment maintenance (Section 5.5). A summary analysis of reported spills and their common causes at hydraulic fracturing sites is then presented (Section 5.6). Then, there is a discussion on the different efforts of spill prevention, containment, and mitigation (Section 5.7).

Next, the fate and transport of spilled chemicals is discussed (Section 5.8). This section includes how a chemical can move through the environment and transform, and what governs exposure concentrations of chemicals in the environment. Due to the complexities of the processes and the site-specific and chemical-specific nature of spills, it is difficult to develop a full assessment of their fate and transport. This section provides a general overview and discusses how the fate and transport of a chemical depends on site conditions, environmental conditions, physicochemical

[1] In this assessment, a spill is considered to be any release of fluids. Spills can result from accidents, fluid management practices, or illegal dumping.

[2] Chemical classes are groupings of different chemicals based on similar features, such as chemical structure, use, or physical properties. Examples of chemical classes include hydrocarbons, alcohols, acids, and bases.

properties of the released chemicals, fluid composition, volume of the release, the proximity to a drinking water resource, and the characteristics of the drinking water resource that is the receptor.

Next is an overview of on-going changes in chemical use in hydraulic fracturing, with an emphasis on industry efforts to reduce potential impacts from surface spills by using fewer and safer chemicals (Section 5.9). The chapter concludes by providing a synthesis, including a summary of findings, factors that affect frequency and severity of potential impacts, and a discussion of uncertainties and data gaps (Section 5.10).

Due to the limitations of available data and the scope of this assessment, it is not possible to provide a detailed analysis of all of the factors listed above. Data limitations preclude a quantitative analysis of the likelihood or severity of chemical spills or impacts. Spills that occur off-site, such as those during transportation of chemicals to the site or storage of chemicals in staging areas, are out of the scope of this assessment. This chapter qualitatively characterizes the potential for impacts on drinking water resources given the current understanding of overall operations and specific components of the chemical mixing process.

5.2 Chemical Mixing Process

Understanding the chemical mixing process is necessary to understand how, why, and when spills might occur. This section provides a general overview of the chemical mixing stage of the hydraulic fracturing water cycle (Carter et al., 2013; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008). Figure 5-1 shows a hydraulic fracturing site during the chemical mixing process. In our discussion, we focus on the types of additives used at each phase of the process. While similar processes are used to fracture horizontal and vertical wells, a horizontal well treatment is described here. Horizontal well treatments are likely to be more complex and therefore illustrative of the variety of practices that have become more prevalent over time with advances in technology (Chapter 3). A water-based system is described, because water is the most commonly used base fluid, appearing in more than 93% of FracFocus 1.0 disclosures between January 1, 2011 and February 28, 2013 (U.S. EPA, 2015a).[1] While the number and types of additives may vary widely, the basic chemical mixing process and the on-site layout of hydraulic fracturing equipment are similar across sites (BJ Services Company, 2009). Equipment used in the chemical mixing process typically consists of chemical storage trucks, water supply tanks, proppant supply, slurry blenders, a number of high-pressure pumps, a manifold, surface lines and hoses, and a central control unit. Detailed descriptions of specific additives and the equipment used in the process are provided in Sections 5.3 and 5.5, respectively.

[1] FracFocus (www.fracfocus.org) is a registry of information of water and chemical use in wells in which hydraulic fracturing is conducted. More details are provided in Text Box 5-1.

Figure 5-1. Representative hydraulic fracturing site showing equipment used on-site during the chemical mixing process.
The frac well head is located in the center bottom (green), the manifold runs down the middle, and high pressure pumps lead into the manifold from either side. Source: Schlumberger.

At a newly-drilled production well, the chemical mixing process begins after the drilling, casing, and cementing processes are finished and hydraulic fracturing equipment has been set up and connected to the well. The process can generally be broken down into one or more sequential stages with specific chemicals added at different phases during each stage phase to achieve a specific purpose (Knappe and Fireline, 2012; Fink, 2003). The process for water-based hydraulic fracturing is described in Figure 5-2 below.

The first phase is the cleaning and preparation of the well. The fluid used in this phase is often referred to as the pre-pad fluid, pre-pad volume, or spearhead. Acid is typically the first chemical introduced. Acid, with a concentration of 3% to 28% (by volume, typically hydrochloric acid, HCl), is used to clean any cement left inside the well from cementing the casing and dissolve any pieces of rock that may remain in the well that could block the perforations.[1] Acid is typically pumped directly from acid storage tanks or tanker trucks, without being mixed with other additives. The first, or pre-pad, phase may also involve mixing and injection of additional chemicals to facilitate the flow of fracturing fluid introduced in the next phase of the process. These additives may include biocides, corrosion inhibitors, friction reducers, and scale inhibitors (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

[1] Prior to the injection of the pad fluid, for wells that are cased in the production zone, the well casing is typically perforated to provide openings through which the pad fluid can enter the formation. A perforating gun is typically used to create small holes in the section of the well being fractured. The perforating gun is lowered into position in the horizontal portion of the well. An electrical current is used to set off small explosive charges in the gun, which creates holes through the well casing and out a short distance into the formation (Gupta and Valkó, 2007).

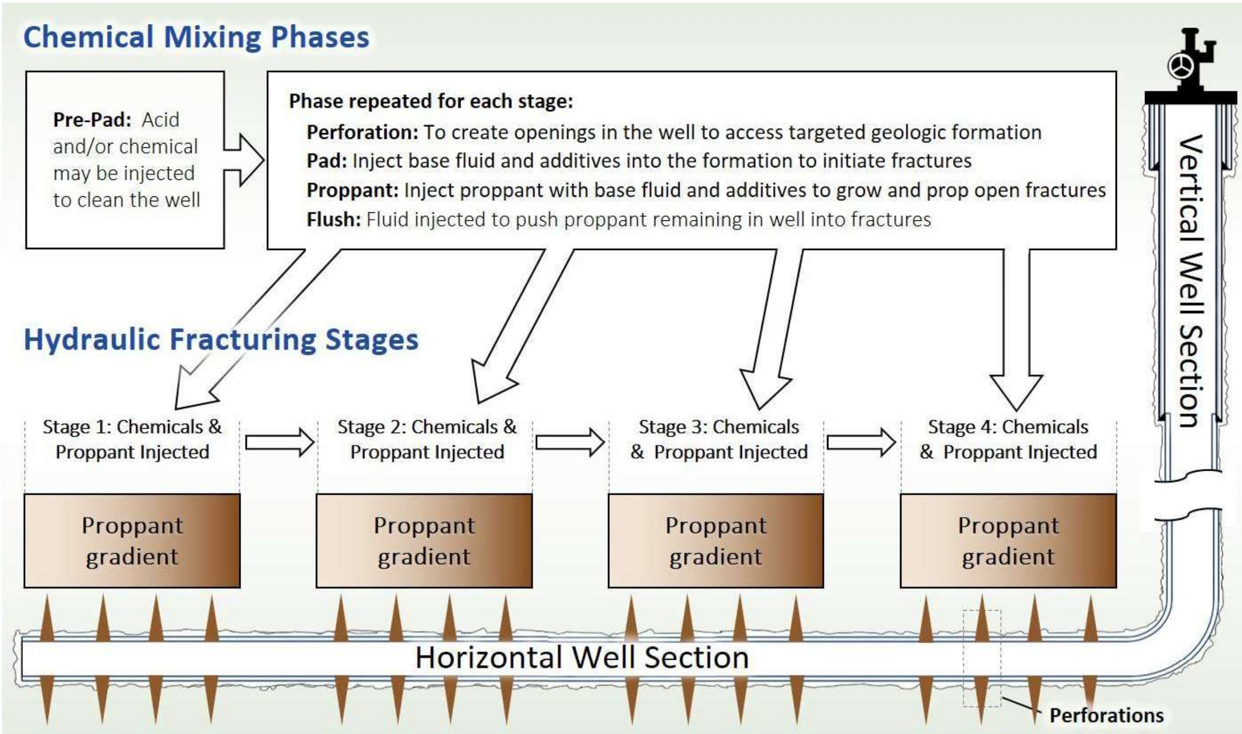


Figure 5-2. Overview of a chemical mixing process of the hydraulic fracturing water cycle.
This figure outlines the chemical mixing process for a generic water-based hydraulic fracture of a horizontal well. The chemical mixing phases outline the steps taken at the surface in the overall fracturing job, while the hydraulic fracturing stages outline how each section of the horizontal well would be fractured beginning with the toe of the well, shown on left-side. The proppant gradient represents how the proppant size may change within each stage of fracturing as the fractures are elongated. The chemical mixing process is repeated depending on the number of stages used for a particular well. The number of stages is determined in part by the length of the horizontal leg. In this figure, four stages are represented, but typically, a horizontal fracturing treatment would consist of 10 to 20 stages per well (Lowe et al., 2013). Fracturing has been reported to be done in as many as 59 stages (Pearson et al., 2013).

In the second phase, a hydraulic fracturing fluid, typically referred to as the pad or pad volume, is mixed, blended, and pumped down the well under high pressure to create fractures in the formation.[1] The pad is a mixture of base fluid, typically water, and additives and is designed to create, elongate, and enlarge fractures in the targeted geologic formation when injected under high pressure (Gupta and Valkó, 2007) (see Section 6.3 for additional information on fracture growth following injection). A typical pad consists of, at minimum, a mixture of water and friction reducer. A typical pad consists of, at minimum, a mixture of water and friction reducer. Other additives (see U.S. EPA (2015a) and Table 5-1) may be used to facilitate flow and kill bacteria (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008). The pad is pumped into the formation through perforations or sliding sleeves in the well casing.

[1] In terms of chemical mixing, "pad" is a term used to describe hydraulic fracturing fluid without solid at the start of the fracturing of the formation. In terms of the entire hydraulic fracturing process, the "well pad" or "pad" is the area of land where drilling occurs.

In the third phase, proppant, typically sand, is mixed into the hydraulic fracturing fluid. The proppant volume, as a proportion of the injected fluid, is increased gradually until the desired concentration in the fractures is achieved. Gelling agents, if used, are also mixed with the proppant and base fluid in this phase to increase the viscosity to help carry the proppant. Additional chemicals may be added to gelled fluids, initially to maintain viscosity and later to break down the gel and decrease viscosity, so the hydraulic fracturing fluid can more readily flow back out of the formation and through the well to facilitate production from the fractured formation (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

A final flush or clean-up phase may be conducted after the stage is fractured, with the primary purpose of maximizing well productivity. The flush is a mixture of water and additives that work to aid the placement of the proppant, clean out the chemicals injected in previous phases, and prevent microbial growth in the fractures (Knappe and Fireline, 2012; Fink, 2003).

The second, third, and fourth phases are repeated multiple times in a well with multi-stage hydraulic fracturing. For each stage, the well is typically perforated and fractured beginning at the end, or toe, of the well and proceeding backwards toward the bend or heel of the well, near the vertical section. In vertical wells, stages typically begin in deeper portions of the well and proceed shallower. Each fractured stage is isolated before the next stage is fractured. The number of stages sets how many times the chemical mixing process is repeated at the site surface (Figure 5-2). The number of stages increases with longer intervals of the well subjected to hydraulic fracturing (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

The number of stages per well can vary, with several sources suggesting between 10 and 20 stages is typical (GNB, 2015; Lowe et al., 2013).[1] The full range reported in the literature is much wider, with one source documenting between 1 and 59 stages per well (Pearson et al., 2013) and others reporting values within this range (NETL, 2013; STO, 2013; Allison et al., 2009). The number of stages per well seems to have increased over time. One study reports that the average number of stages per horizontal well rose from approximately 10 in 2008 to 30 in 2012 (Pearson et al., 2013). As more stages are used, the total volume of hydraulic fracturing fluid and chemicals increase. This increases the potential, frequency, and severity of surface spills associated with chemical mixing and thus potential impacts on drinking water resources.

In each of these phases, water is usually the primary component of the hydraulic fracturing fluid, though the exact composition of the fluid injected into the well changes over the duration of each stage. In water-based hydraulic fracturing, the composition, by volume, of a typical hydraulic fracturing fluid is 90% to 97% water, 2% to 10% proppant, and 2% or less additives (Carter et al., 2013; Knappe and Fireline, 2012; SWN, 2011).[2]

[1] The number of stages has been reported to be 6 to 9 in the Huron in 2009 (Allison et al., 2009), 13 to 32 in the Marcellus (NETL, 2013), and up to 40 by STO (2013).

[2] This range is based on a compilation of sources. Sources present compositions as by mass, by volume, or without specificity. Because of non-additive volumes, the composition by volume can be different before and after mixing. By mass: 90% water, 8-9% proppant, 0.5 to 1.5% additives (Knappe and Fireline, 2012); 88% water, 11% proppant, <1%

5.3 Overview of Hydraulic Fracturing Fluids

Hydraulic fracturing fluids are formulated to perform specific functions: create and extend the fracture and transport and place the proppant in the fractures (Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007).[1] The hydraulic fracturing fluid generally consists of three parts: (1) the base fluid, which is the largest constituent by volume, (2) the additives, and (3) the proppant. Additives, which can be a single chemical or a mixture of chemicals, are chosen to serve a specific purpose in the hydraulic fracturing fluid (e.g., friction reducer, gelling agent, crosslinker, biocide) (Spellman, 2012). Throughout this chapter, "chemical" is used to refer to an individual chemical substance (e.g., methanol, petroleum distillates).[2] Proppants are small particles, usually sand, mixed with fracturing fluid to hold fractures open so that the target hydrocarbons can flow from the formation through the fractures and up the wellbore. The combination of additives, and the mixing and injection process, varies based on a number of factors as discussed below. The additive combination determines the amount and type of equipment required for storage and, therefore, contributes to the determination of the potential for spills and impacts of those spills.

The particular composition of a hydraulic fracturing fluid is designed based on empirical experience, the geology and geochemistry of the production zone, economics, goals of the fracturing process, availability of the desired chemicals, and preference of the service company or operator (Montgomery, 2013; ALL Consulting, 2012; Klein et al., 2012; Ely, 1989).[3] No single set of specific chemicals is used at every site. Multiple types of fracturing fluids may be appropriate for a given site, and any given type of fluid may be appropriate at multiple sites. For the same type of fluid formulation, there can be differences in the additives, chemicals in those additives, and the concentrations selected. There are broad criteria for hydraulic fracturing fluid selection based on the targeted production zone temperature, pressure, water sensitivity, and permeability (Gupta and Valkó, 2007; Elbel and Britt, 2000). Figure 5-3 provides a general overview of the types of decisions to determine which fluid can be used for different situations. Similar fluids may be appropriate for different formations. For example, crosslinked fluids with 25% nitrogen foam (titanate or zirconate crosslink + 25% nitrogen) can be used in both gas and oil wells with high temperatures and

---

additive (as median maximum concentration) (U.S. EPA, 2015a), 94% water, 6% proppant, <1% additive (Sjolander et al., 2011), 88% water, 11% proppant, <1% additive (OSHA, 2014a, b). By volume: 95% water, 5% proppant, <1% additive (before mixing), 97% water, 2% proppant, <1% additive (after mixing) (Sjolander et al., 2011), 90% water, 10% proppant, <1% additive (before mixing), 95% water, 5% proppant, <1% additive (after mixing) (OSHA, 2014a, b), 98-99.5%, water and sand 0.5 to 2% additives (Spellman, 2012). Not specified: 99.9% water and sand, 0.1% chemicals (SWN, 2011), 98-99% water and proppant, 1-2 % additives (Carter et al., 2013).

[1] We use "hydraulic fracturing fluid" to refer to the fluid that is injected into the well and used to create and hold open fractures the formation.

[2] In this chapter, because of the way many chemicals are reported, we use the word "chemical" to refer to any individual chemical or chemical substance that has been assigned a CASRN (Chemical Abstracts Service Registry Number). A CASRN is a unique identifier for a chemical substance, which can be a single chemical (e.g., hydrochloric acid, CASRN 7647-01-0) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates (CASRN 64742-47-8), a complex mixtures of C9 to C16 hydrocarbons). For simplicity, we refer to both pure chemicals and chemical substances that are mixtures, which have a single CASRN, as "chemicals."

[3] Empirical experience tends to provide better result as operators gain experience at a new site or geology increases. When an operator moves to a new basin geology, there may be less than optimal results. With experience and understanding of the geology increases, the empirical evidence will inform what hydraulic fracturing fluid composition works better than others.

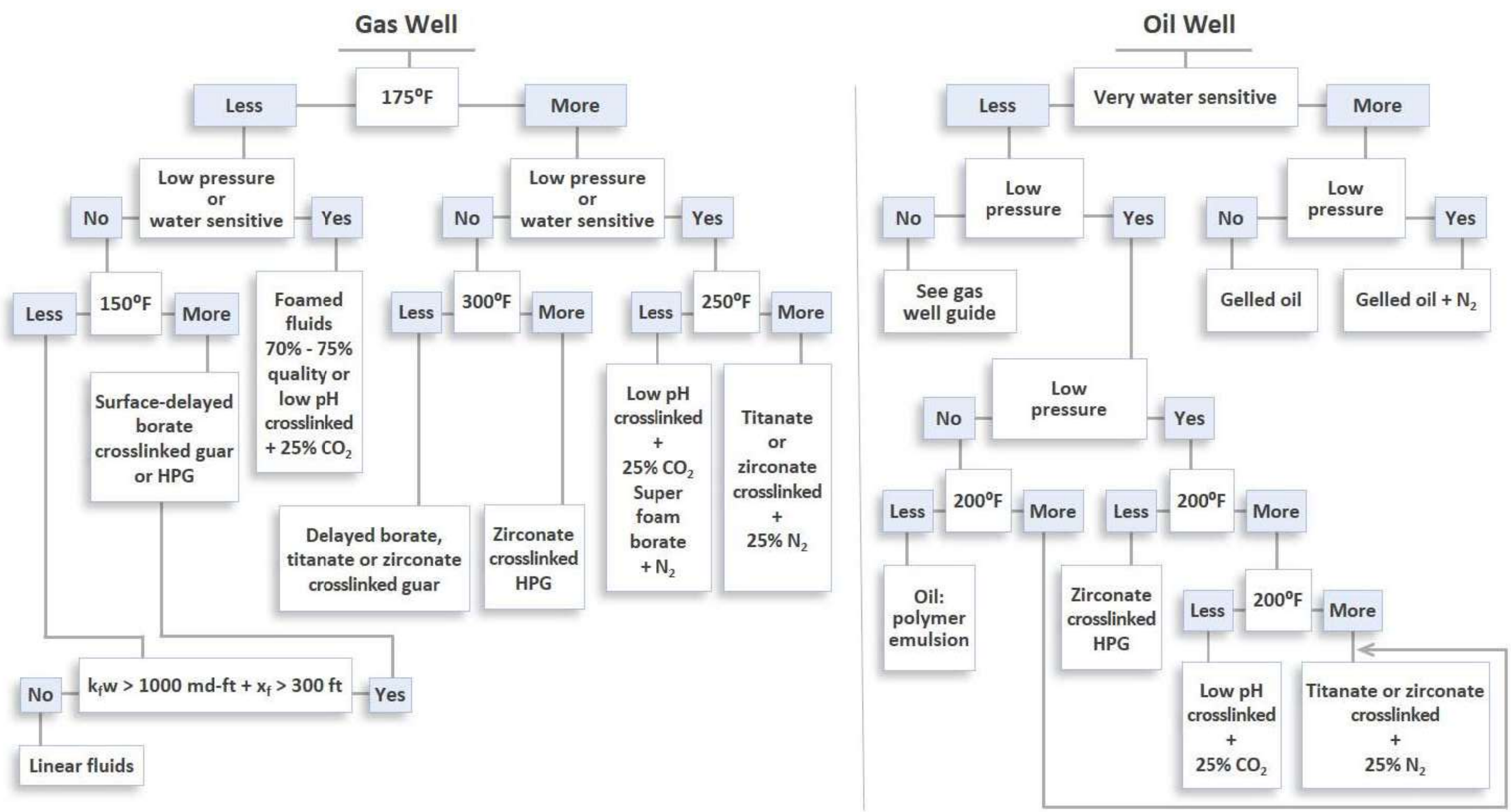


Figure 5-3. Example hydraulic fracturing fluid decision tree for gas and oil wells.
This decision tree figure serves as an example of the factors that determine the type of hydraulic fracturing fluid chosen to fracture a given formation, depending on whether the well will produce oil or gas. Factors include water sensitivity, formation temperature, and pressure. HPG is hydroxypropylguar, guar derivatized with propylene oxide. Parameters are: k_f, fracture permeability, w is the fracture width, and x_f is the fracture half-length. This figure was chosen to represent the differences between oil and gas wells and the types of decisions involved with choosing a fluid. This is adapted from Elbel and Britt (2000) and, as such, is dated to that time period. Since then, slickwater has become increasingly popular due to its simplicity and cheaper cost, and slickwater has often replaced linear and crosslinked gelled fluids, especially in shales. Other decision tree figures may exist.

variation in water sensitivity.[1,2] One of the most important properties in designing a hydraulic fracturing fluid is the viscosity (Montgomery, 2013).[3]

Table 5-1 provides a list of common types of additives, their functions, and the most frequently used chemicals for each purpose based on the EPA's analysis of disclosures to FracFocus 1.0 (U.S. EPA, 2015a, hereafter referred to as the EPA FracFocus 1.0 report), the EPA's project database of disclosures to FracFocus 1.0 (U.S. EPA, 2015c, hereafter referred to as the EPA FracFocus 1.0 project database), and other literature sources.[4] Additional information on more additives can be found in U.S. EPA (2015a).

A general description of typical hydraulic fracturing fluid formulations nationwide is difficult, because fracturing fluids vary from well to well. Based on the EPA FracFocus 1.0 report, the median number of chemicals reported for each disclosure was 14, with the 5th to 95th percentile ranging from four to 28 (see Appendix H for a list of hydraulic fracturing fluid chemicals). The median number of chemicals per disclosure was 16 for oil wells and 12 for gas wells (U.S. EPA, 2015a). Other sources have stated that between three and 12 additives and chemicals are used (Schlumberger, 2015; Carter et al., 2013; Spellman, 2012; GWPC and ALL Consulting, 2009).[5]

Water, the most commonly used base fluid for hydraulic fracturing, is inferred to be used as a base fluid in more than 93% of EPA FracFocus 1.0 disclosures (U.S. EPA, 2015c). Alternatives to water-based fluids, such as hydrocarbons and gases, including carbon dioxide and nitrogen-based foam, may also be used based on formation characteristics, cost, or preferences of the well operator or service company (ALL Consulting, 2012; GWPC and ALL Consulting, 2009). Non-aqueous base fluid ingredients were identified in 761 (2.2%) of EPA FracFocus 1.0 disclosures (U.S. EPA, 2015a). Gases and hydrocarbons may be used alone or blended with water; more than 96% of the disclosures identifying non-aqueous base fluids are blended (U.S. EPA, 2015a). There is no standard method to categorize the different fluid formulations (Patel et al., 2014; Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007). Therefore, we broadly categorize the fluids as water-based or alternative fluids.

[1] A crosslinked fluid is a fluid that has polymers that have been linked together through a chemical bond. A crosslink chemical is added to have the polymer chains linked together to form larger chemical structures with higher viscosity. The increased fracturing fluid viscosity allows the fluid to carry more proppant into the fractures. The fracturing fluid remains viscous until a breaking agent is introduced to break the cross-linked polymer.

[2] Water sensitivity refers to when a formation's physicochemical properties are affected in the presence of water. An example of a water sensitive formation would be one where the soil particles swell when water is added, reducing the permeability of the formation.

[3] Viscosity is a measure of the internal friction of fluid that provides resistance to shear within the fluid, informally referred to as how "thick" a fluid is. For example, custard is thick and has a high viscosity, while water is runny with a low viscosity. Sufficient viscosity is needed to create a fracture and transport proppant (Gupta and Valkó, 2007). In lower-viscosity fluids, proppant is transported by turbulent flow and requires more hydraulic fracturing fluid. Higher-viscosity fluids allows the fluid to carry more proppant, requiring less fluid but necessitating the reduction of viscosity after the proppant is placed (Rickman et al., 2008; Gupta and Valkó, 2007).

[4] A disclosure refers to all data submitted for a specific oil and gas production well for a specific fracture date.

[5] Sources may differ based on whether they are referring to additives or chemicals.

Table 5-1. Examples of common additives, their function, and the most frequently used chemicals reported to FracFocus for these additives.

The list of examples of common additives was developed from information provided in multiple sources (U.S. EPA, 2015a, c; Stringfellow et al., 2014; Montgomery, 2013; Vidic et al., 2013; Spellman, 2012; GWPC and ALL Consulting, 2009; Arthur et al., 2008; Gupta and Valkó, 2007; Gidley et al., 1989). The additive functions are based on information the EPA received from service companies (U.S. EPA, 2013a).

Additives	Function	Chemicals reported in 20% or more of disclosures in the EPA FracFocus 1.0 project database for given additive[a,b]
Acid	Dissolves cement, minerals, and clays to reduce clogging of the pore space	Hydrochloric acid
Biocide	Controls or eliminates bacterial growth, which can be present in the base fluid and may have detrimental effects on the long term well productivity	Glutaraldehyde; 2,2-dibromo-3-nitrilopropionamide
Breaker	Reduces the designed increase in viscosity of specialized treatment fluids such as gels and foams after the proppant has been placed and flowback commences to clean up the well	Peroxydisulfuric acid diammonium salt
Clay control	Prevents the swelling and migration of formation clays that otherwise react to water-based fluids	Choline chloride
Corrosion inhibitor	Protects the iron and steel components in the wellbore and treating equipment from corrosive fluids	Methanol; propargyl alcohol; isopropanol
Crosslinker	Increases the viscosity of base gel fluids by connecting polymer molecules	Ethylene glycol; potassium hydroxide; sodium hydroxide
Emulsifier	Facilitates the dispersion of one immiscible fluid into another by reducing the interfacial tension between the two liquids to achieve stability	2-Butoxyethanol; polyoxyethylene(10)nonylphenyl ether; methanol; nonyl phenol ethoxylate
Foaming agent	Generates and stabilizes foam fracturing fluids	2-Butoxyethanol; Nitrogen, liquid; isopropanol; methanol; ethanol
Friction reducer	Reduces the friction pressures experienced when pumping fluids through tools and tubulars in the wellbore	Hydrotreated light petroleum distillates
Gelling agent	Increases fracturing fluid viscosity allowing the fluid to carry more proppant into the fractures and to reduce fluid loss to the reservoir	Guar gum; hydrotreated light petroleum distillates
Iron control agent	Controls the precipitation of iron compounds (e.g., Fe_2O_3) from solution	Citric acid

Additives	Function	Chemicals reported in 20% or more of disclosures in the EPA FracFocus 1.0 project database for given additive[a,b]
Nonemulsifier	Separates problematic emulsions generated within the formation	Methanol; isopropanol; nonyl phenol ethoxylate
pH control	Affects the pH of a solution by either inducing a change (pH adjuster) or stabilizing and resisting change (buffer) to achieve desired qualities and optimize performance	Carbonic acid, dipotassium salt; potassium hydroxide; sodium hydroxide; acetic acid
Resin curing agents	Lowers the curable resin coated proppant activation temperature when bottom hole temperatures are too low to thermally activate bonding	Methanol; nonyl phenol ethoxylate; isopropanol; alcohols, C12-14-secondary, ethoxylated
Scale inhibitor	Controls or prevents scale deposition in the production conduit or completion system	Ethylene glycol; methanol
Solvent	Controls the wettability of contact surfaces or prevents or breaks emulsions[1]	Hydrochloric acid

[a] Chemicals (excluding water and quartz) listed in the EPA FracFocus 1.0 project database in more than 20% of disclosures for a given purpose when that purpose was listed as used on a disclosure (U.S. EPA, 2015c). These are not necessarily the active ingredients for the purpose, but rather are listed as being commonly present for the given purpose. Chemicals may be disclosed for more than a single purpose (e.g., 2-butoxyethanol is listed as being used as an emulsifier and a foaming agent).

[b] Analysis considered 32,885 disclosures and 615,436 ingredient records that met selected quality assurance criteria, including: completely parsed (parsing is the process of analyzing a string of symbols to identify and separate various components); unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; valid concentrations; and valid purpose. Disclosures that did not meet quality assurance criteria (5,645) or other, query-specific criteria were excluded from analysis.

5.3.1 Water-Based Fracturing Fluids

The advantages of water-based fracturing fluids are low cost, ease of mixing, and ability to recover and reuse the water. The disadvantages are that they have low viscosity, they create narrow fractures, and they may not provide optimal performance in water-sensitive formations (Montgomery, 2013; Gupta and Valkó, 2007) (Section 5.3.2). Water-based fluids can be as simple as water with a few additives to reduce friction, such as "slickwater," or as complex as water with crosslinked polymers, clay control agents, biocides, and scale inhibitors (Spellman, 2012). (See Figure 5-4 for a slickwater example.)

Gels may be added to water-based fluids to increase viscosity, which assists with proppant transport and results in wider fractures. Gelling agents include natural polymers, such as guar, starches, and cellulose derivatives, which require the addition of biocide to minimize bacterial growth (Spellman, 2012; Gupta and Valkó, 2007). Gels may be linear or crosslinked. Crosslinking

[1] Wettability is the ability of a liquid to maintain contact with a solid surface. When wettability is high, a liquid droplet will lie flat across a surface, maximizing the area of contact between the liquid and the solid. When wettability is low, a liquid droplet will approach a spherical shape, minimizing the area of contact between the liquid and solid.

increases viscosity without adding more gel. Gelled fluids require the addition of a breaker, which breaks down the gel after it carries in the proppant, to reduce fluid viscosity to facilitate fluid flowing back after treatment (Spellman, 2012; Gupta and Valkó, 2007). The presence of residual breakers may make it difficult to reuse recovered water (Montgomery, 2013).

5.3.2 Alternative Fracturing Fluids

Alternative hydraulic fracturing fluids can be used for water sensitive formations (i.e., formations where permeability is reduced when water is added) or as dictated by production goals (Halliburton, 1988). Examples of alternative fracturing fluids include acid-based fluids; non-aqueous-based fluids; energized fluids, foams, or emulsions; viscoelastic surfactant fluids; gels; methanol; and other unconventional fluids (Montgomery, 2013; Saba et al., 2012; Gupta and Hlidek, 2009; Gupta and Valkó, 2007; Halliburton, 1988).

Acid fracturing is generally used in carbonate formations without the use of a proppant. Fractures are initiated with a hydraulic fracturing fluid, and acid (gelled, foamed, or emulsified) is added to irregularly etch the wall of the fracture. The etching serves to prop open the formation, for a high-conductivity fracture (Spellman, 2012; Gupta and Valkó, 2007).

Non-aqueous fluids, like petroleum distillates and propane, are used in water-sensitive formations. Non-aqueous fluids may also contain additives, such as gelling agents, to improve performance (Gupta and Valkó, 2007). The use of non-aqueous fluids has decreased due to safety concerns, and because water-based and emulsion fluid technologies have improved (Montgomery, 2013; Gupta and Valkó, 2007). Methanol, for example, was previously used as a base fluid in water-sensitive reservoirs beginning in the early 1990s, but was discontinued in 2001 for safety concerns and cost (Saba et al., 2012; Gupta and Hlidek, 2009; Gupta and Valkó, 2007). Methanol is still widely used as an additive or in additive mixtures in hydraulic fracturing fluid formulations.

Energized fluids, foams, and emulsions minimize fluid leakoff in low pressure targeted geologic formations, have high proppant-carrying capacity, improve fluid recovery, and are sometimes used in water-sensitive formations (Barati and Liang, 2014; Gu and Mohanty, 2014; Spellman, 2012; Gupta and Valkó, 2007; Martin and Valko, 2007).[1] However, these treatments tend to be expensive, can require high pressure, and pose potential health and safety concerns (Montgomery, 2013; Spellman, 2012; Gupta and Valkó, 2007). Energized fluids (see Figure 5-4 for an example of an energized fluid composition) are mixtures of liquid and gas (Patel et al., 2014; Montgomery, 2013). Nitrogen (N_2) or carbon dioxide (CO_2), the gases used, make up less than 53% of the fracturing fluid volume, typically ranging from 20% to 30% by volume (Montgomery, 2013; Gupta and Valkó, 2007; Mitchell, 1970). Energized foams are liquid-gas mixtures, with nitrogen or carbon dioxide gas comprising more than 53% of the fracturing fluid volume, with a typical range of 65% to 80% by volume (Montgomery, 2013; Mitchell, 1970). Emulsions are liquid-liquid mixtures, typically a

[1] Leakoff is the fraction of the injected fluid that infiltrates into the formation (e.g., through an existing natural fissure) and is not recovered during production (Economides et al., 2007). See Chapter 6, Section 6.3 for more discussion on leakoff.

hydrocarbon (e.g., condensate or diesel) with water.[1] Both water-based fluids, including gels, and non-aqueous fluids can be energized fluids or foams.

Foams and emulsions break easily using gravity separation and are stabilized by using additives such as foaming agents (Gupta and Valkó, 2007). Emulsions may be used to stabilize active chemical ingredients or to delay chemical reactions, such as the use of carbon dioxide-miscible, non-aqueous fracturing fluids to reduce fluid leakoff in water-sensitive formations (Taylor et al., 2006).

Other types of fluids not addressed above include viscoelastic surfactant fluids, viscoelastic surfactant foams, crosslinked foams, liquid carbon dioxide-based fluid, and liquid carbon dioxide-based foam fluid, and hybrids of other fluids (King, 2010; Brannon et al., 2009; Curtice et al., 2009; Tudor et al., 2009; Gupta and Valkó, 2007; Coulter et al., 2006; Boyer et al., 2005; Fredd et al., 2004; MacDonald et al., 2003).

5.3.3 Tracers

Some chemicals are added to the fluid to act as tracers. Tracers are added to hydraulic fracturing fluid to assess the efficiency of fracturing and proppant placement. As an example, the efficiency of oil production from multistage fracturing was assessed by using 17 oil soluble tracers. Each tracer was used to assess production from a specific interval of the well (Catlett et al., 2013), although the specific compounds used were not identified (Table 5-2). Chemical classes of tracers and individual examples show a range of compounds employed including both inorganic and organic, and including radioactive elements, although only a few specific chemicals have been revealed. Of these, examples are proppant tracers and fluorocarbons. Although radioactive fluids have also been used for proppant tracing, a commonly-used approach has the short-half-life elements Antimony^{124}, Iridium^{192}, and Scandium^{46} bound to the proppants and gamma emissions are subsequently measured by a neutron-logging device (Sonnenfield et al., 2016; Odegard et al., 2015; Lowe et al., 2013; Osborn and McIntosh, 2011; McDaniel et al., 2010).[2,3] Of the organic tracers, 14 fluorinated organics have been identified through an analysis of FracFocus 2.0 disclosures (Konschnik and Dayalu, 2016). Three fluorinated tracers and Antimony^{124} were identified in produced water (Maguire-Boyle and Barron, 2014) (Appendix Table H-4).

Table 5-2. Classes and specifically identified examples of tracers used in hydraulic fracturing fluids.

Class	Specific Chemical[a]	Reference
Thiocyanates (SCN^-)	ND	Dugstad (2007)
Fluorobenzoic acids	ND	Dugstad (2007)

[1] Diesel is a mixture typically of C8 to C21 hydrocarbons. The shorthand “C8” is used to represent a hydrocarbon with 8 carbons. Thus “C21” represents a hydrocarbon with 21 carbons. Octane has 8 carbons and is thus a C8, and is a component of gasoline.

[2] Antimony^{124}: 60.2 days, Iridium^{192}: 74 days, Scandium^{46}: 83.8 days.

[3] Gadolinium^{155} and Gadolinium^{157} have been suggested as bound proppant tracers because of their high-gamma-capture cross-sections (Liu et al., 2015).

Class	Specific Chemical[a]	Reference
Radioactive tracers	Tritiated Water	Dugstad (2007)
	Tritiated Methanol	Dugstad (2007)
	Antimony^{124}	Silber et al. (2003)
	Iridium^{192}	Silber et al. (2003)
	Scandium^{46}	Silber et al. (2003)
Fluorocarbons	2,2,3,3,4,4,4-heptafluorobutyl undecylate	Maguire-Boyle and Barron (2014)
	2,3,4-Trifluorobenzoic acid	Konschnik and Dayalu (2016)
	2,4,5-Trifluorobenzoic acid	Konschnik and Dayalu (2016)
	2,4-Difluorobenzoic acid	Konschnik and Dayalu (2016)
	2,6-Difluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Chloro-4-fluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Fluorobenzoic acid	Konschnik and Dayalu (2016)
	2-Trifluoromethylbenzoic acid	Konschnik and Dayalu (2016)
	3-Trifluoromethylbenzoate	Konschnik and Dayalu (2016)
	4-(Trifluoromethyl)benzoic acid	Konschnik and Dayalu (2016)
	4-Chloro-2-fluorobenzoic acid	Konschnik and Dayalu (2016)
	4-fluoro-2-(trifluoromethyl)benzoic acid	Konschnik and Dayalu (2016)
	4-Fluoro-3-(trifluomethyl)benzoic acid	Konschnik and Dayalu (2016)
	Benzoic acid, 3,5-difluoro-	Konschnik and Dayalu (2016)
	cis-4-ethyl-5-octyl-2,2-*bis*(trifluoromethyl)-1,3 dioxolane	Maguire-Boyle and Barron (2014)
	p-Fluorobenzoic acid	Konschnik and Dayalu (2016)
	tri-fluoromethyl tetradeculate	Maguire-Boyle and Barron (2014)
Oil soluble alkyl esters	ND	Deans (2007)
Unstable emulsions	ND	Catlett et al. (2013)
Controlled-release polymers and solid tracers	ND	Salman et al. (2014)

[a] ND = none disclosed.

A different set of tracers have been proposed for identifying environmental impacts from hydraulic fracturing fluids (Kurose, 2014). These tracers are designed so that the fluids from individual wells are identifiable while having no environmental impact themselves. DNA and nanoparticles with magnetic properties made specifically for each well have been proposed for this purpose (Kurose, 2014).

5.3.4 Proppants

Proppants are small particles carried down the well and into fractures by hydraulic fracturing fluid. They hold the fractures open after the injection pressure has been released and the hydraulic fracturing fluid has been removed (Brannon and Pearson, 2007). The propped fractures provide a path for the hydrocarbon to flow from the reservoir. The EPA's analysis of FracFocus 1.0 data showed that 98% of disclosures reported sand as the proppant, making sand (i.e., quartz) the most commonly reported proppant (U.S. EPA, 2015a). Other proppants include man-made or specially engineered particles, such as high-strength ceramic materials or sintered bauxite (Schlumberger, 2014; Brannon and Pearson, 2007). Proppant types can be used individually or in combinations.

5.3.5 Example Hydraulic Fracturing Fluids

There is no standard composition of hydraulic fracturing fluid used across the United States, and the literature does not present any typical hydraulic fluid composition. In Figure 5-4, we present two examples of hydraulic fracturing fluid mixtures based on analyses conducted on the EPA FracFocus 1.0 project database (U.S. EPA, 2015c). These examples represent two different types of fluids used at two different wells. The first is a slickwater, and the second is an energized fluid.[1] Details of each fluid are presented in the figure along with pie charts of their composition, as given by maximum percent by mass of the total hydraulic fracturing fluid.

The first hydraulic fracturing fluid (Figure 5-4a), the slickwater, is composed of 87% water, 13% sand, and 0.05% chemicals, by mass. The fluid is 71% fresh water and 16% reused produced water, with a total water volume of 4,763,000 gal (18,030,000 L). The chemical composition consists of six different additive types (acid, friction reducer, biocide, scale inhibitor iron control, and corrosion inhibitor) and a total of 13 chemicals.

The second hydraulic fracturing fluid (Figure 5-4b), the energized fluid, is more complex and consists of 58% water, 28% nitrogen gas, 13% sand, and 1.5% additives, by mass, with a total water volume of 105,000 gal (397,000 L). The hydraulic fracturing fluid composition consists of 10 additives (acid, surfactant, foamer, corrosion inhibitor, biocide, friction reducer, breaker, scale inhibitor, iron control, and clay stabilizer) and a total of 28 chemicals.

[1] A slickwater is a hydraulic fracturing fluid designed to have a low viscosity to allow pumping at high rates. The critical additive in a slickwater is the friction reducer, which makes the fluid "slick."

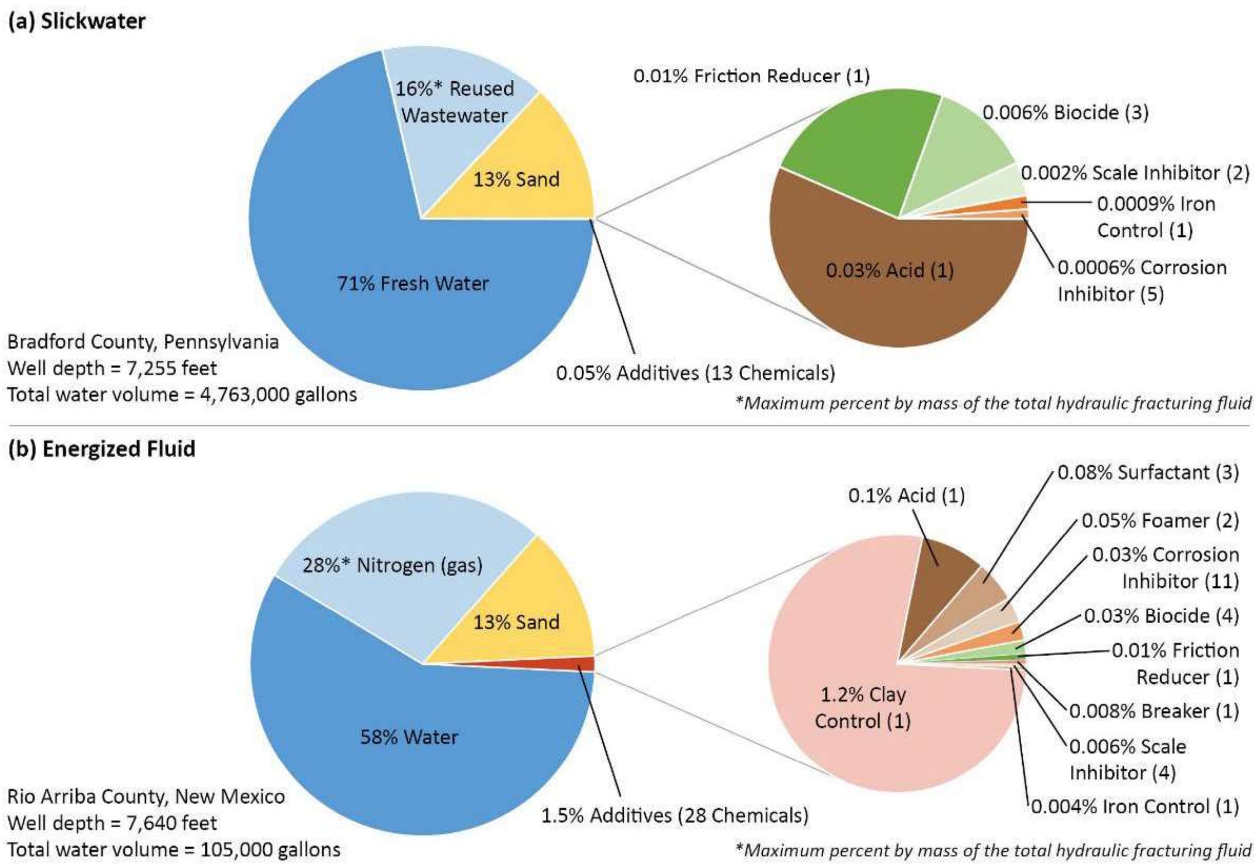


Figure 5-4. Example hydraulic fracturing fluids.
Example compositions of (a) slickwater and (b) energized fluid. The base fluid and proppants are on the left, and the additive breakdown is on the right. The number in parentheses represents the number of chemicals in that additive. See Table 5-1 for the function of different additives and the most common chemicals in those additives reported as based on the analysis of the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

These two examples give an idea of the difference in the compositions of two example hydraulic fracturing fluids. These compositions are the final mixture as if the entire fluid were mixed at once; they are generally not the actual composition at any given point in time. These compositions provide the potential composition of a spilled hydraulic fracturing fluid during the chemical mixing stage. Any of these ingredients (e.g., biocide) could be released by itself or mixed with the base fluid with other additives. The variability of hydraulic fracturing fluids from well to well and site to site makes it difficult to assess the potential of hydraulic fracturing additive or fluid release.

5.4 Frequency and Volume of Hydraulic Fracturing Chemical Use

This section highlights the different chemicals used in hydraulic fracturing fluids and discusses the frequency and volume of use. Using the EPA FracFocus 1.0 project database (Text Box 5-1), we focus our analysis on the individual chemicals that are used as ingredients in additive formulations, rather than on the complete mixture of chemicals that may be present in a hydraulic fracturing fluid. Operators can report information about well location, date of operations, and water and chemical use to the FracFocus registry. Chemicals are reported in FracFocus by using the chemical name and the Chemical Abstract Services Registration Number (CASRN), which is a unique number identifier for every chemical.[1] The information on specific chemicals, particularly those most commonly used, can be used to assess potential impacts on drinking water resources. The volume of chemicals stored on-site provides information on the potential volume of a chemical spill.

Text Box 5-1. The FracFocus Registry and EPA FracFocus Report.

The Ground Water Protection Council (GWPC) and the Interstate Oil and Gas Compact Commission (IOGCC) developed a national hydraulic fracturing chemical registry, FracFocus (www.fracfocus.org). Well operators can use the registry to disclose information about chemicals and water they use during hydraulic fracturing. As part of the EPA's Study of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources, the EPA published the report titled *Analysis of Hydraulic Fracturing Fluid Data from the FracFocus Chemical Disclosure Record Registry 1.0* (U.S. EPA, 2015a). For this report, the EPA accessed data from FracFocus 1.0 from January 1, 2011 to February 28, 2013, which included more than 39,000 disclosures (records of well data) in 20 states that had been submitted by operators prior to March 1, 2013. Accompanying the U.S. EPA (2015a) report is the published EPA FracFocus 1.0 project database, which. It supported analyses of FracFocus chemical and water use data (U.S. EPA, 2015c), and a report describing the details of data management for development of the project database (U.S. EPA, 2015b).

Submission to FracFocus was initially voluntary and varied from state to state. During the timeframe covered in the EPA FracFocus 1.0 report (January 2011 to February 2013), six of the 20 states with data submitted to FracFocus and included in the EPA FracFocus 1.0 project database began requiring operators to disclose chemicals used in hydraulic fracturing fluids to FracFocus (Colorado, North Dakota, Oklahoma, Pennsylvania, Texas, and Utah). Three other states started requiring disclosure to either FracFocus or the state (Louisiana, Montana, and Ohio), and five states required or began requiring disclosure to the state (Arkansas, Michigan, New Mexico, West Virginia, and Wyoming). Alabama, Alaska, California, Kansas, Mississippi, and Virginia did not have reporting requirements during the period of the EPA's study.

The EPA's analysis may or may not be nationally representative. Disclosures from the five states reporting the most disclosures to FracFocus (Texas, Colorado, Pennsylvania, North Dakota, and Oklahoma) comprise over 78% of the disclosures in the database; nearly half (47%) of the disclosures are from Texas. Thus, data from these states are most heavily represented in the EPA's analyses.

A disclosure reports the total water volume (in gallons) and the chemicals used in the fluid (as maximum ingredient concentration by mass both in the additive and in the hydraulic fracturing fluid). The actual mass of the chemicals used in the fluid are not reported. The fluid composition reported in the disclosure does not necessarily reflect the actual composition of the fluid at any time. Rather, the disclosure represents what the total composition of the fluid would be if all chemicals used were mixed together at their maximum reported concentration.

The EPA summarized information on the locations of the wells in the disclosures, water volumes used, and the frequency of use and maximum ingredient concentrations of the chemicals in the additives and the hydraulic fracturing fluid. Additional information can be found in the EPA FracFocus 1.0 report (U.S. EPA, 2015a) and in the EPA FracFocus 1.0 project database (U.S. EPA, 2015c).

[1] A CASRN and chemical name combination identify a chemical substance, which can be a single chemical (e.g., hydrochloric acid, CASRN 7647-01-0) or a mixture of chemicals (e.g., hydrotreated light petroleum distillates (CASRN 64742-47-8), a complex mixtures of C9 to C16 hydrocarbons). For simplicity, we refer to both pure chemicals and chemical substances that are mixtures, which have a single CASRN, as "chemicals."