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United States
Environmental Protection
Agency

# Assessment of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking Water Resources

**Office of Research and Development**
**Washington, D.C.**

United States
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NOTICE

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# List of Acronyms/Abbreviations

| Acronym | Definition | Acronym | Definition |
|---|---|---|---|
| **2BE** | 2-butoxyethanol | **CM** | chemical mixing |
| **ACToR** | Aggregated Computational Toxicology Resource database | **COGCC** | Colorado Oil and Gas Conservation Commission |
| **AMEC** | AMEC Environment & Infrastructure, Inc. | **CWCB** | Colorado Water Conservation Board |
| **ANRC** | Arkansas Natural Resources Commission | **CWT** | centralized waste treatment |
| **AO** | administrative order | **CWTF** | centralized water treatment facility |
| **AOGC** | Arkansas Oil and Gas Commission | **DBNM** | dibromochloronitromethane |
| **API** | American Petroleum Institute | **DBP** | disinfection by-products |
| | | **DecaBDE** | decabromodipheyl ether |
| **ATSDR** | Agency for Toxic Substance and Disease Registry | **DfE** | Design for the Environment |
| | | **DI** | Drilling Info, Inc. |
| **AWWA** | American Water Works Association | **DMR** | Discharge Monitoring Report |
| | | **DNR** | Department of Natural Resources |
| **BLM** | Bureau of Land Management | **DO** | dissolved oxygen |
| **BTEX** | benzene, toluene, ethylbenzene, and xylenes | **DOC** | dissolved organic carbon |
| | | **DOE** | U.S. Department of Energy |
| **CARES** | Casella Altela Regional Environmental Services | **DOGGR** | California Department of Conservation's Division of Oil, Gas & Geothermal Resources |
| **CASRN** | chemical abstract services registration number | | |
| **CBI** | confidential business information | **DOJ** | U.S. Department of Justice |
| **CBM** | coalbed methane | **DOT** | U.S. Department of Transportation |
| **CCST** | California Council on Science and Technology | **DRO** | diesel range organics |
| **CDWR** | Colorado Division of Water Resources | **EERC** | Energy and Environmental Research Center, University of North Dakota |
| **CFR** | Code of Federal Regulations | **EIA** | U.S. Energy Information Administration |
| **CICAD** | Concise International Chemical Assessment Document | | |
| | | **EPA** | U.S. Environmental Protection Agency |

| Acronym | Definition | Acronym | Definition |
|---------|-----------|---------|-----------|
| **EPA OW** | U.S. Environmental Protection Agency's Office of Water | **IRIS** | Integrated Risk Information System |
| **EPI** | estimation programs interface | **IUPAC** | International Union of Pure and Applied Chemistry |
| **ERCB** | Energy Resource Conservation Board | **KWO** | Kansas Water Office |
| **ERG** | Eastern Research Group | **LOAEL** | lowest observed adverse effect level |
| **ESN** | Environmental Services Network | **MCDA** | multicriteria decision analysis |
| **FRS** | fluids recovery services | **MCL** | maximum contaminant level |
| **GES** | Groundwater & Environmental Services, Inc. | **MCLG** | maximum containment level goal |
| **GNB** | Government of New Brunswick | **MCOR** | Marcellus Center for Outreach and Research |
| **GRO** | gasoline range organics | **MIT** | mechanical integrity test |
| **GTI** | Gas Technology Institute | **MRL** | minimum risk level |
| **GWPC** | Ground Water Protection Council | **MSC** | Marcellus shale coalition |
| **HBCD** | hexabromocyclododecane | **MT GWIC** | Montana Ground Water Information Center |
| **HDPE** | high-density polyethylene | **MTBE** | methyl tert-butyl ether |
| **HF** | hydraulic fracturing | **MVR** | mechanical vapor recompression |
| **HHBP** | Human Health Benchmarks for Pesticides | **NAS** | National Academy of Sciences |
| **HISA** | Highly Influential Scientific Assessment | **NDDMR** | North Dakota Department of Mineral Resources |
| **HUC** | hydrological unit code | **NDDOH** | North Dakota Department of Health |
| **IAEA** | International Atomic Energy Agency | **NDSWC** | North Dakota State Water Commission |
| **IARC** | International Agency for Research on Cancer | **NETL** | National Energy Technology Laboratory |
| **IOGCC** | Interstate Oil and Gas Compact Commission | **NGO** | non-governmental organization |
| **IPCC** | Intergovernmental Panel on Climate Change | **NM OCD** | New Mexico Oil Conservation Division |
| **IPCS** | International Programme on Chemical Safety | | |

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| Acronym | Definition | Acronym | Definition |
|---------|-----------|---------|-----------|
| **NM OSE** | New Mexico Office of the State Engineer | **PAH** | polycyclic aromatic hydrocarbon |
| **NOAEL** | no observed adverse effect level | **PFBC** | Pennsylvania Fish and Boat Commission |
| **NORM** | naturally occurring radioactive material | **PDL** | positive determination letter |
| | | **PMF** | Positive Matrix Factorization |
| **NPC** | National Petroleum Council | **POD** | point-of-departure |
| **NPDES** | National Pollution Discharge Elimination System | **POTW** | publicly owned treatment work |
| **NRC** | National Resource Council | **PPRTV** | provisional peer-reviewed toxicity value |
| **NTP** | U.S. National Toxicology Program | | |
| **NYSDEC** | New York State Department of Environmental Conservation | **QA** | quality assurance |
| | | **QAPP** | quality assurance project plan |
| **O&G** | oil and gas | **QC** | quality control |
| **ODNR** | Ohio Department of Natural Resources | **QSAR** | Quantitative Structure Activity Relationship |
| **ODNR, DMRM** | Ohio Department of Natural Resources, Division of Mineral Resources Management | **RfD** | reference dose |
| | | **RfV** | reference value |
| | | **RO** | reverse osmosis |
| **OEPA** | Ohio Environmental Protection Agency | **SAB** | Science Advisory Board |
| **ORD** | Office of Research and Development | **SAIC** | Science Applications International Corporation |
| **OSF** | oral slope factor | **SDWA** | Safe Drinking Water Act |
| **OSHA** | Occupational Safety & Health Administration | **SDWIS** | safe drinking water information system |
| **OSWER** | Office of Solid Water and Emergency Response | **SEECO** | Southern Electrical Equipment Company |
| **OWRB** | Oklahoma Water Resources Board | **SGEIS** | supplemented generic environmental impact statement |
| **PA DCNR** | Pennsylvania Department of Conservation and Natural Resources | **SHS MSC** | statewide health standards for medium-specific concentrations |
| **PA DEP** | Pennsylvania Department of Environmental Protection | **SMCL** | secondary maximum contaminant level |

| Acronym | Definition | Acronym | Definition |
|---------|-----------|---------|-----------|
| **SPE** | Society of Petroleum Engineers | **USGAO** | U.S. Government Accountability Office |
| **SRB** | Susquehanna River basin | **USGS** | U.S. Geological Survey |
| **SRBC** | Susquehanna River Basin Commission | **UWS** | Universal Well Services |
| **STO** | Statoil | **VOC** | volatile organic compounds |
| **STRONGER** | State review of oil and natural gas environmental regulations | **WAWSA** | Western Area Water Supply Authority |
| **SVOC** | semi-volatile organic compounds | **WFR** | Well File Review |
| **SWE** | Southwestern Energy | **WHO** | World Health Organization |
| **TARM** | TerrAqua Resource Management | **WRF** | Water Research Foundation |
| **TBA** | tert-butyl alcohol | **WVDEP** | West Virginia Department of Environmental Protection |
| **TDS** | total dissolved solids | **WWTP** | wastewater treatment plant |
| **TENORM** | technologically enhanced naturally occurring radioactive material | **WYOGCC** | Wyoming Oil and Gas Conservation Commission |
| **THM** | trihalomethane | | |
| **TIPRO** | Texas Independent Producers and Royalty Owners Association | | |
| **TMDL** | total maximum daily load | | |
| **TOC** | total organic carbon | | |
| **TPH** | total petroleum hydrocarbons | | |
| **TSS** | total suspended solids | | |
| **TTHM** | total trihalomethane | | |
| **TWDB** | Texas Water Development Board | | |
| **TXRRC** | Texas Railroad Commission | | |
| **UCRB** | Upper Colorado River basin | | |
| **UIC** | underground injection control | | |
| **UOG** | unconventional oil and gas | | |

# Preface

1   The U.S. Environmental Protection Agency (EPA) is conducting a study of the potential impacts of
2   hydraulic fracturing for oil and gas on drinking water resources. This study was initiated in Fiscal
3   Year 2010 when Congress urged the EPA to examine the relationship between hydraulic fracturing
4   and drinking water resources in the United States. In response, EPA developed a research plan
5   (*Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources*) that was
6   reviewed by the Agency's Science Advisory Board (SAB) and issued in 2011. A progress report on
7   the study (*Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources:*
8   *Progress Report)*, detailing the EPA's research approaches and next steps, was released in late 2012
9   and was followed by a consultation with individual experts convened under the auspices of the SAB.

10  The EPA's study includes the development of several research projects, extensive review of the
11  literature and technical input from state, industry, and non-governmental organizations as well as
12  the public and other stakeholders. A series of technical roundtables and in-depth technical
13  workshops were held to help address specific research questions and to inform the work of the
14  study. The study is designed to address research questions posed for each stage of the hydraulic
15  fracturing water cycle:

16   • Water Acquisition: What are the possible impacts of large volume water withdrawals
17      from  ground and surface waters on drinking water resources?

18   • Chemical Mixing: What are the possible impacts of surface spills of hydraulic fracturing
19      fluid on or near well pads on drinking water resources?

20   • Well Injection: What are the possible impacts of the injection and fracturing process on
21      drinking  water resources?

22   • Flowback and Produced Water: What are the possible impacts of surface spills of flowback
23      and produced water on or near well pads on drinking water resources?

24   • Wastewater Treatment and Waste Disposal: What are the possible impacts of
25      inadequate  treatment of hydraulic fracturing wastewaters on drinking water resources?

26  This report, *Assessment of the Potential Impacts of Hydraulic Fracturing for Oil and Gas on Drinking*
27  *Water Resources*, includes both the literature review and results from the research projects
28  conducted as part of the EPA's study. It will undergo independent, external peer review in
29  accordance with Agency policy and all of the peer review comments received will be considered in
30  the development of the final report.

31  The EPA's study will contribute to the understanding of the potential impacts of hydraulic
32  fracturing activities for oil and gas on drinking water resources and the factors that may influence
33  those impacts. The study will help facilitate and inform dialogue among interested stakeholders,
34  including Congress, other Federal agencies, states, tribal government, the international community,
35  industry, non-governmental organizations, academia, and the general public.

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## Acknowledgements

# Executive Summary

# Executive Summary

1  Since the early 2000s, oil and natural gas production in the United States has been transformed
2  through technological innovation. Hydraulic fracturing, combined with advanced directional
3  drilling techniques, made it possible to economically extract oil and gas resources previously
4  inaccessible. The resulting surge in production increased domestic energy supplies and brought
5  economic benefits to many areas of the United States.

6  The growth in domestic oil and gas production also raised concerns about potential impacts to
7  human health and the environment, including potential effects on the quality and quantity of
8  drinking water resources. Some residents living close to oil and gas production wells have reported
9  changes in the quality of drinking water and assert that hydraulic fracturing is responsible for these
10 changes. Other concerns include competition for water between hydraulic fracturing activities and
11 other water users, especially in areas of the country experiencing drought, and the disposal of
12 wastewater generated from hydraulic fracturing.

13 The U.S. Congress urged the U.S. Environmental Protection Agency (EPA) to study the relationship
14 between hydraulic fracturing and drinking water. This report synthesizes available scientific
15 literature and data to assess the potential for hydraulic fracturing for oil and gas to change the
16 quality or quantity of drinking water resources, and identifies factors affecting the frequency or
17 severity of any potential changes. This report can be used by federal, tribal, state, and local officials;
18 industry; and the public to better understand and address any vulnerabilities of drinking water
19 resources to hydraulic fracturing activities.

## What is Hydraulic Fracturing?

20 Hydraulic fracturing is a stimulation technique used to increase oil and gas production from
21 underground rock formations. Hydraulic fracturing involves the injection of fluids under pressures
22 great enough to fracture the oil- and gas-producing formations. The fluid generally consists of
23 water, chemicals, and proppant (commonly sand). The proppant holds open the newly created
24 fractures after the injection pressure is released. Oil and gas flow through the fractures and up the
25 production well to the surface.

26 Hydraulic fracturing has been used since the late 1940s and, for the first 50 years, was mostly used
27 in vertical wells in conventional formations.[1] Hydraulic fracturing is still used in these settings, but
28 the process has evolved; technological developments (including horizontal and directional drilling)
29 have led to the use of hydraulic fracturing in unconventional hydrocarbon formations that could not
30 otherwise be profitably produced (see Figure ES-1). These formations include:

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[1] Conventional formations often allow oil and natural gas to flow to the wellbore without hydraulic fracturing and typically contain trapped oil and natural gas that migrated from other subsurface locations. Hydraulic fracturing can be used to enhance oil and gas production from these formations. In unconventional formations, hydraulic fracturing is needed to extract economical quantities of oil and gas.

1  • **Shales.** Organic-rich, black shales are the source rocks in which oil and gas form on
2     geological timescales. Oil and gas are contained in the pore space of the shale. Some shales
3     contain predominantly gas or oil; many shale formations contain both.

4  • **Tight formations**. "Tight" formations are relatively low permeability, non-shale,
5     sedimentary formations that can contain oil and gas. Like in shales, oil and gas are
6     contained in the pore space of the formation. Tight formations can include sandstones,
7     siltstone, and carbonates, among others.

8  • **Coalbeds**. In coalbeds, methane (the primary component of natural gas) is generally
9     adsorbed to the coal rather than contained in the pore space or structurally trapped in the
10    formation. Pumping the injected and native water out of the coalbeds after fracturing
11    serves to depressurize the coal, thereby allowing the methane to desorb and flow into the
12    well and to the surface.
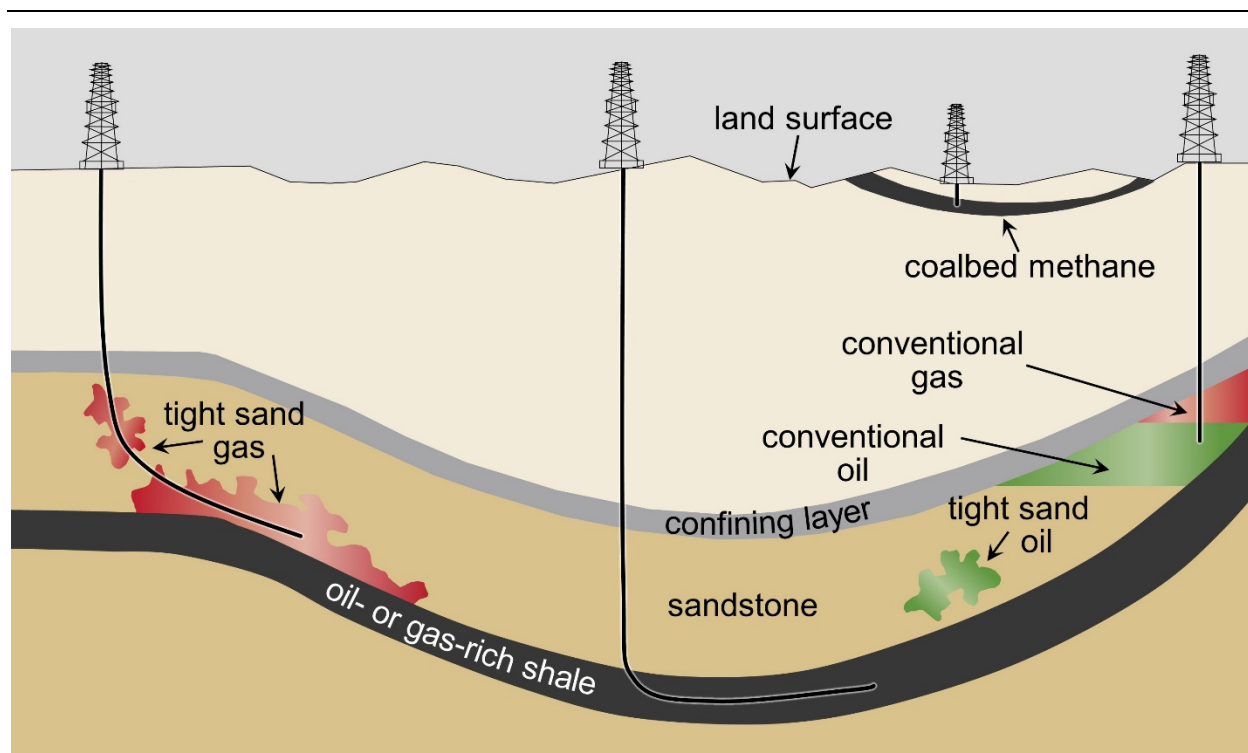


**Figure ES-1. Schematic cross-section of general types of oil and gas resources and the orientations of production wells used in hydraulic fracturing.**

Shown are conceptual illustrations of types of oil and gas wells. A vertical well is producing from a conventional oil and gas deposit (right). In this case, a gray confining layer serves to "trap" oil (green) or gas (red). Also shown are wells producing from unconventional formations: a vertical coalbed methane well (second from right); a horizontal well producing from a shale formation (center); and a well producing from a tight sand formation (left). Note: Figure not to scale. Modified from USGS (2002) and Newell (2011).

1   The combined use of hydraulic fracturing with horizontal (or more generically, directional) drilling
2   has led to an increase in oil and gas activities in areas of the country with historical oil and gas
3   production, and an expansion of oil and gas activities to new regions of the country.

## Scope of the Assessment

4   We defined the scope of this assessment by the following activities involving water that support
5   hydraulic fracturing (i.e., the hydraulic fracturing water cycle; see Figure ES-2):[1]

6   • **Water acquisition:** the withdrawal of ground or surface water needed for hydraulic
7       fracturing fluids;

8   • **Chemical mixing:** the mixing of water, chemicals, and proppant on the well pad to create
9       the hydraulic fracturing fluid;

10  • **Well injection:** the injection of hydraulic fracturing fluids into the well to fracture the
11      geologic formation;

12  • **Flowback and produced water:** the return of injected fluid and water produced from the
13      formation (collectively referred to as produced water in this report) to the surface, and
14      subsequent transport for reuse, treatment, or disposal; and

15  • **Wastewater treatment and waste disposal:** the reuse, treatment and release, or
16      disposal of wastewater generated at the well pad, including produced water.

17  This assessment reviews, analyzes, and synthesizes information relevant to the potential impacts of
18  hydraulic fracturing on drinking water resources at each stage of the hydraulic fracturing water
19  cycle. Impacts are defined as any change in the quality or quantity of drinking water resources.
20  Where possible, we identify the mechanisms responsible or potentially responsible for any impacts.
21  For example, a spill of hydraulic fracturing fluid is a mechanism by which drinking water resources
22  could be impacted.

23  Drinking water resources are defined within this report as any body of ground water or surface
24  water that now serves, or in the future could serve, as a source of drinking water for public or
25  private use. This is broader than most federal and state regulatory definitions of drinking water and
26  encompasses both fresh and non-fresh bodies of water. Trends indicate that both types of water
27  bodies are currently being used, and will continue to be used in the future, as sources of drinking
28  water.

29  This assessment focuses on the potential impacts from activities in the hydraulic fracturing water
30  cycle on drinking water resources. We do this so federal, tribal, state, and local officials; industry;
31  and the public can better understand and address any vulnerabilities of drinking water resources to
32  hydraulic fracturing activities. We do not address other concerns raised about hydraulic fracturing
33  specifically or about oil and gas exploration and production activities more generally. Activities that

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[1] In this assessment, we refer to the "EPA" when referencing other EPA studies. If a conclusion or analysis was done
specifically by the authors of this assessment, we refer to it and its findings in the first person.

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1   are not considered include: acquisition and transport of constituents of hydraulic fracturing fluids
2   besides water (e.g., sand mining and chemical production) outside of the stated water cycle; site
3   selection and well pad development; other infrastructure development (e.g., roads, pipelines,
4   compressor stations); site reclamation; and well closure. A summary and evaluation of current or
5   proposed regulations and policies is beyond the scope of this report. Additionally, this report does
6   not discuss the potential impacts of hydraulic fracturing on other water users (e.g., agriculture or
7   industry), other aspects of the environment (e.g., seismicity, air quality, or ecosystems), worker
8   health or safety, or communities. Furthermore, this report is not a human health risk assessment. It
9   does not identify populations that are exposed to chemicals, estimate the extent of exposure, or
10  estimate the incidence of human health impacts.
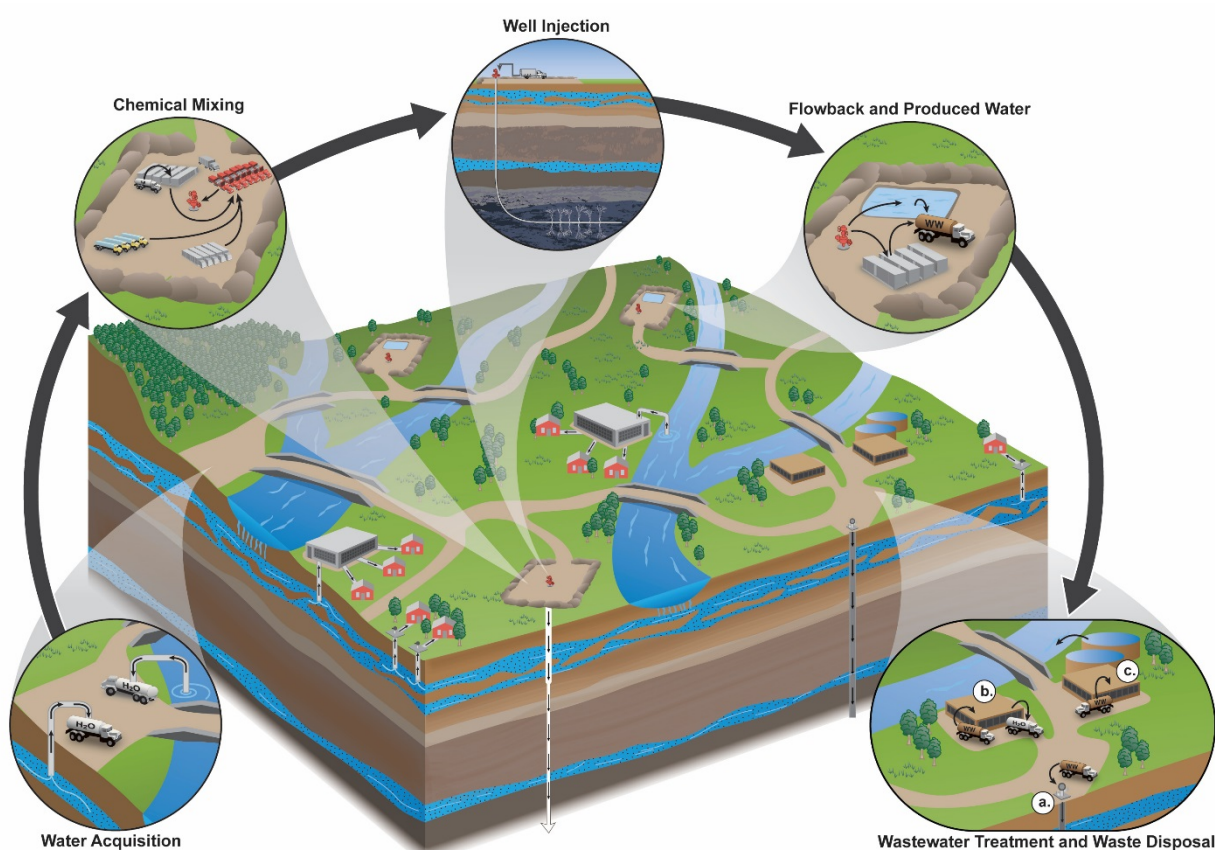


**Figure ES-2. The stages of the hydraulic fracturing water cycle.**

Shown here is a generalized landscape depicting the activities of the hydraulic fracturing water cycle and their relationship to each other, as well as their relationship to drinking water resources. Arrows depict the movement of water and chemicals. Specific activities in the "Wastewater Treatment and Waste Disposal" inset are (a)underground injection control (UIC) well disposal, (b) wastewater treatment and reuse, and (c) wastewater treatment and discharge at a centralized waste treatment (CWT) facility. Note: Figure not to scale.

## Approach

1   This assessment relies on relevant scientific literature and data. Literature evaluated included
2   articles published in science and engineering journals, federal and state government reports, non-
3   governmental organization (NGO) reports, and industry publications. Data sources examined
4   included federal- and state-collected data sets, databases maintained by federal and state
5   government agencies, other publicly-available data and information, and data, including
6   confidential and non-confidential business information, submitted by industry to the EPA.[1] The
7   relevant literature and data complement research conducted by the EPA under its *Plan to Study the*
8   *Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (hereafter referred to as the
9   "Study Plan") and published by scientific journals or as peer-reviewed EPA reports; those articles
10  and reports are cited throughout this assessment. The research topic areas and projects described
11  in the Study Plan were designed to meet the data and information needs of this assessment and
12  were developed with substantial expert and public input.

## Proximity of Current Activity and Drinking Water Resources

13  Thousands of wells are drilled and fractured every year in the United States, with activities
14  concentrated in specific locations. We estimate 25,000-30,000 new wells were drilled and
15  hydraulically fractured annually in the United States between 2011 and 2014. Additional, pre-
16  existing wells (wells more than one year old that may or may not have been hydraulically fractured
17  in the past) were also likely fractured. Hydraulic fracturing took place in at least 25 states between
18  1990 and 2013. The EPA's analysis of disclosures made to FracFocus 1.0 (hereafter "FracFocus")
19  contained wells from 20 of these states.[2] Almost half of these wells were in Texas. Colorado was a
20  distant second, while Pennsylvania and North Dakota were third and fourth, respectively. Hydraulic
21  fracturing activities were further localized within the 20 states. Of the approximately 1,500
22  counties or county equivalents in these 20 states, slightly over 400 contained all of the wells
23  disclosed to FracFocus during this time period. In Colorado, over 85% of the hydraulically fractured
24  wells disclosed were located in two counties. The price of gas and oil may cause short term
25  volatility in the number of wells drilled and fractured per year, yet hydraulic fracturing is expected
26  to continue to expand and drive an increase in domestic oil and gas production in coming decades.

27  Hydraulically fractured wells can be located near residences and drinking water resources.
28  Between 2000 and 2013, approximately 9.4 million people lived within one mile of a hydraulically

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[1] Some information provided to the EPA in response to two separate information requests to service companies and well operators was claimed as confidential business information.
[2] FracFocus is a publicly accessible website (www.fracfocus.org) managed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission where oil and gas production well operators may disclose information voluntarily or pursuant to state requirements about the ingredients used in hydraulic fracturing fluids at individual wells. The EPA analyzed disclosures from FracFocus 1.0 for over 38,000 oil and gas production wells hydraulically fractured between January 1, 2011 and February 28, 2013. A disclosure refers to data submitted for a specific oil and gas production well for a specific fracture date. Most wells had only one disclosure, but a small number of wells (876 wells) had multiple disclosures. For the purposes of this Executive Summary, we equate disclosures with wells when discussing this study.

1   fractured well. Approximately 6,800 sources of drinking water for public water systems were
2   located within one mile of at least one hydraulically fractured well during the same period. These
3   drinking water sources served more than 8.6 million people year-round in 2013.

4   Although proximity of hydraulic fracturing activities to a drinking water resource is not in of itself
5   sufficient for an impact to occur,, it does increase the potential for impacts. Residents and drinking
6   water resources in areas experiencing hydraulic fracturing activities are most likely to be affected
7   by any potential impacts, should they occur. However, hydraulic fracturing can also affect drinking
8   water resources outside the immediate vicinity of a hydraulically fractured well; a truck carrying
9   wastewater could spill or a release of inadequately treated wastewater could have downstream
10  effects.

## Major Findings

11  From our assessment, we conclude there are above and below ground mechanisms by which
12  hydraulic fracturing activities have the potential to impact drinking water resources. These
13  mechanisms include water withdrawals in times of, or in areas with, low water availability; spills of
14  hydraulic fracturing fluids and produced water; fracturing directly into underground drinking
15  water resources; below ground migration of liquids and gases; and inadequate treatment and
16  discharge of wastewater.

17  We did not find evidence that these mechanisms have led to widespread, systemic impacts on
18  drinking water resources in the United States. Of the potential mechanisms identified in this report,
19  we found specific instances where one or more mechanisms led to impacts on drinking water
20  resources, including contamination of drinking water wells. The number of identified cases,
21  however, was small compared to the number of hydraulically fractured wells.

22  This finding could reflect a rarity of effects on drinking water resources, but may also be due to
23  other limiting factors. These factors include: insufficient pre- and post-fracturing data on the quality
24  of drinking water resources; the paucity of long-term systematic studies; the presence of other
25  sources of contamination precluding a definitive link between hydraulic fracturing activities and an
26  impact; and the inaccessibility of some information on hydraulic fracturing activities and potential
27  impacts.

28  Below, we provide a synopsis of the assessment's key findings, organized by each stage of the
29  hydraulic fracturing water cycle. We provide answers to the research questions presented in the
30  Study Plan and Chapter 1. While come citations are provided here, individual chapters should be
31  consulted for additional detail and citations.

### *Water Acquisition*

32  Water is a major component of nearly all hydraulic fracturing operations. It typically makes up
33  almost 90% or more of the fluid volume injected into a well, and each hydraulically fractured well
34  requires thousands to millions of gallons of water. Cumulatively, hydraulic fracturing activities in
35  the United States used on average 44 billion gal of water a year in 2011 and 2012, according to the

1     EPA's analysis of FracFocus disclosures. Although this represents less than 1% of total annual water
2     use and consumption at this scale, water withdrawals could potentially impact the quantity and
3     quality of drinking water resources at more local scales.[1]


### Research Questions: Water Acquisition

- ### What are the types of water used for hydraulic fracturing?

4     Water for hydraulic fracturing typically comes from surface water, ground water, or reused
5     hydraulic fracturing wastewater. Hydraulic fracturing operations in the eastern United States
6     generally rely on surface water, while operations in the more semi-arid to arid western states
7     generally use mixed supplies of surface and ground water. In the Marcellus Shale in Pennsylvania,
8     for example, most water used for hydraulic fracturing originates from surface water, whereas
9     surface and ground water are used in approximately equal proportions in the Barnett Shale in
10    Texas (see Figure ES-3a,b). In areas that lack available surface water (e.g., western Texas), ground
11    water supplies most of the water needed for hydraulic fracturing.

12    Across the United States, the vast majority of water used in hydraulic fracturing is fresh, although
13    operators also make use of lower-quality water, including reused hydraulic fracturing wastewater.[2]
14    Based on available data, the median reuse of wastewater as a percentage of injected volumes is 5%
15    nationally, with the percentage varying by location.[3] Available data on reuse trends indicate
16    increased reuse of wastewater over time in both Pennsylvania and West Virginia. Reuse as a
17    percentage of injected volumes is lower in other areas, including regions with more water stress,
18    likely because of the availability of disposal wells. For example, reused wastewater is
19    approximately 18% of injected volumes in the Marcellus Shale in Pennsylvania's Susquehanna
20    River Basin, whereas it is approximately 5% in the Barnett Shale in Texas (see Figure ES-3a,b).

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[1] Water use is water withdrawn from ground- or surface water for a specific purpose, part or all of which may be returned to the local hydrologic cycle. If no water is returned, water use equals water consumption. 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). In the case of hydraulic fracturing, water can be consumed by the loss of injected water to subsurface zones or via underground disposal of wastewaters, among other means.

[2] In this assessment, hydraulic fracturing "wastewater" refers to both produced water and any other water generated as a hydraulic fracturing site.  As used in this assessment, the term "wastewater" is not intended to constitute a term of art for legal or regulatory purposes.

[3] Reused wastewater as a percentage of injected water differs from the percentage of wastewater that is managed through reuse, as opposed to other wastewater management options. For example, in the Marcellus in Pennsylvania, approximately 18% of injected water is reused produced water, while approximately 70% of wastewater or more is managed through reuse (Figure ES-3a).

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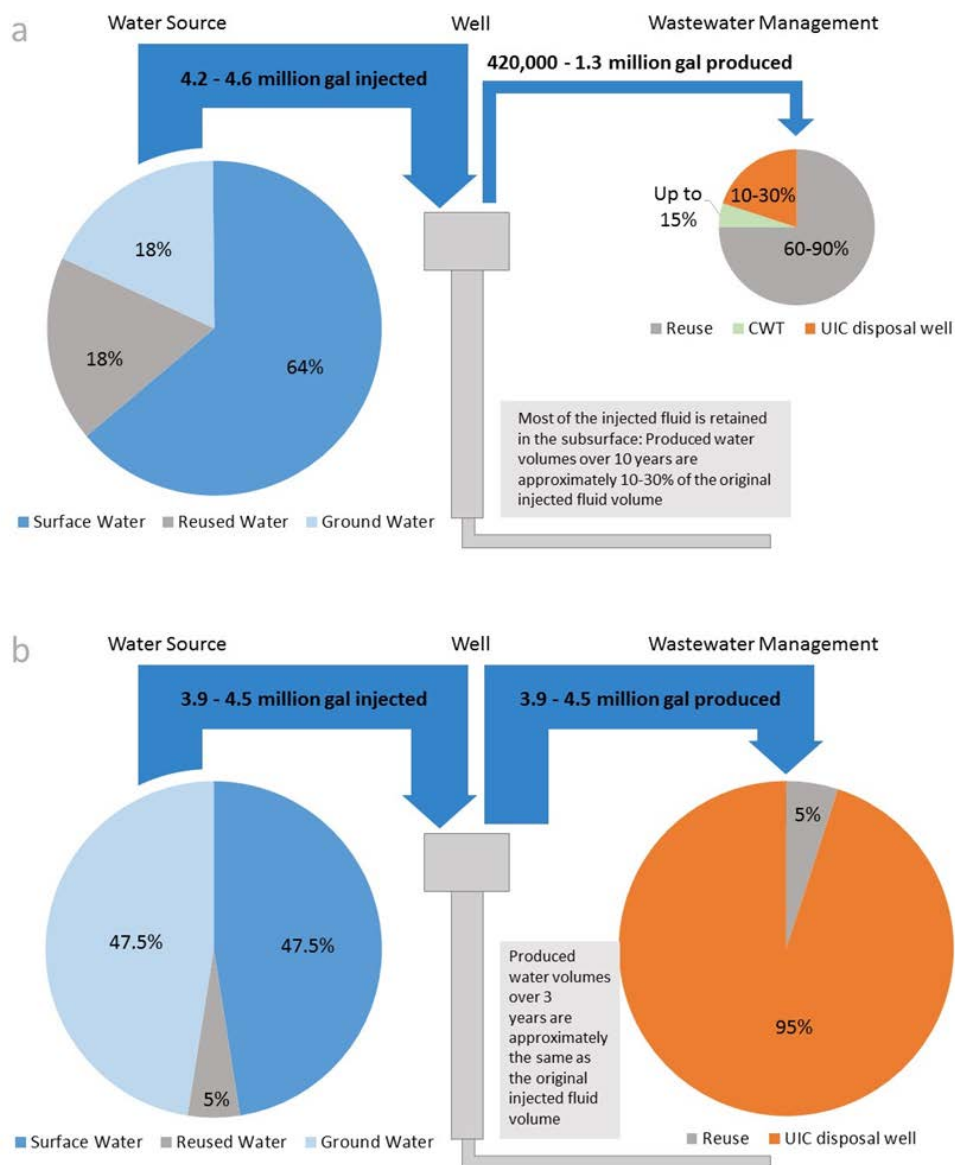
**Figure ES-3. Water budgets representative of practices in the Marcellus Shale in the Susquehanna River Basin in Pennsylvania (a) and the Barnett Shale in Texas (b).**

Pie size and arrow thickness represent the relative volume of water as it flows through the hydraulic fracturing water cycle. Wastewater going to a centralized waste treatment (CWT) facility may be either discharged to surface water or reused. Wastewater going to an underground injection control (UIC) well is disposed of below ground. These examples represent typical water management practices as depicted for the most recent time period reviewed by this assessment. They do not represent any specific well. Note: Values for Marcellus Shale are specific to the Susquenhanna River Basin, except for the produced water volumes. The longest-term measurement available was from the West Virginia portion of the the Marcellus Shale.

- ***How much water is used per well?***

1   The national median volume of water used per hydraulically fractured well is approximately 1.5
2   million gal (5.7 million L), according to the EPA's analysis of FracFocus disclosures. This estimate
3   likely represents a wide variety of fractured well types, including vertical wells that generally use
4   much less water per well than horizontal wells. Thus, published estimates for horizontal shale gas
5   wells are typically higher (e.g., approximately 4 million gallons (Vengosh et al., 2014)). There is also
6   wide variation within and among states and basins in the median water volumes used per well,
7   from more than 5 million gal (19 million L) in Arkansas, Louisiana and West Virginia to less than 1
8   million gal (3.8 million L) in California, New Mexico, and Utah, among others. This variation results
9   from several factors, including well length, formation geology, and fracturing fluid formulation.

- ***How might cumulative water withdrawals for hydraulic fracturing affect drinking water quantity?***

10  Cumulatively, hydraulic fracturing uses billions of gallons of water each year at the national and
11  state scales, and even in some counties. As noted above, hydraulic fracturing water use and
12  consumption are generally less than 1% of total annual water use and consumption at these scales.
13  However, there are a few counties in the United States where these percentages are higher. For
14  2011 and 2012, annual hydraulic fracturing water use was 10% or more compared to 2010 total
15  annual water use in 6.5% of counties with FracFocus disclosures analyzed by the EPA, 30% or more
16  in 2.2% of counties, and 50% or more in 1.0% of counties. Consumption estimates followed the
17  same general pattern. In these counties, hydraulic fracturing is a relatively large user and consumer
18  of water.

19  High fracturing water use or consumption alone does not necessarily result in impacts to drinking
20  water resources. Rather, impacts result from the combination of water use or consumption and
21  water availability at local scales. In our survey of published literature, we did not find a case where
22  hydraulic fracturing water use or consumption alone caused a drinking water well or stream to run
23  dry. This could indicate an absence of effects or a lack of documentation in the literature we
24  reviewed. Additionally, water availability is rarely impacted by just one use or factor alone. In
25  Louisiana, for example, the state requested hydraulic fracturing operations switch from ground to
26  surface water, due to concerns that ground water withdrawals for fracturing could, in combination
27  with other uses, adversely affect drinking water supplies.

28  The potential for impacts to drinking water resources from hydraulic fracturing water withdrawals
29  is highest in areas with relatively high fracturing water use and low water availability. Southern
30  and western Texas are two locations where hydraulic fracturing water use, low water availability,
31  drought, and reliance on declining ground water has the potential to affect the quantity of drinking
32  water resources. Any impacts are likely to be realized locally within these areas. In a detailed case
33  study of southern Texas, Scanlon et al. (2014) observed generally adequate water supplies for
34  hydraulic fracturing, except in specific locations. They found excessive drawdown of local ground
35  water in a small proportion (approximately 6% of the area) of the Eagle Ford Shale. They suggested
36  water management, particularly a shift towards brackish water use, could minimize potential future
37  impacts to fresh water resources.

1    The potential for impacts to drinking water quantity due to hydraulic fracturing water use appears
2    to be lower—but not eliminated—in other areas of the United States. Future problems could arise if
3    hydraulic fracturing increases substantially in areas with low water availability, or in times of water
4    shortages. In detailed case studies in western Colorado and northeastern Pennsylvania, the EPA did
5    not find current impacts, but did conclude that streams could be vulnerable to water withdrawals
6    from hydraulic fracturing. In northeast Pennsylvania, water management, such as minimum stream
7    flow requirements, limits the potential for impacts, especially in small streams. In western North
8    Dakota, ground water is limited, but the industry may have sufficient supplies of surface water from
9    the Missouri River system. These location-specific examples emphasize the need to focus on
10   regional and local dynamics when considering potential impacts of hydraulic fracturing water
11   acquisition on drinking water resources.

- ***What are the possible impacts of water withdrawals for hydraulic fracturing on water quality?***

12   Water withdrawals for hydraulic fracturing, similar to all water withdrawals, have the potential to
13   alter the quality of drinking water resources. Ground water withdrawals exceeding natural
14   recharge rates decrease water storage in aquifers, potentially mobilizing contaminants or allowing
15   the infiltration of lower quality water from the land surface or adjacent formations. Withdrawals
16   could also decrease ground water discharge to streams, potentially affecting surface water quality.
17   Areas with large amounts of sustained ground water pumping are most likely to experience
18   impacts, particularly drought-prone regions with limited ground water recharge.

19   Surface water withdrawals also have the potential to affect water quality. Withdrawals may lower
20   water levels and alter stream flow, potentially decreasing a stream's capacity to dilute
21   contaminants. Case studies by the EPA show that streams can be vulnerable to changes in water
22   quality due to water withdrawals, particularly smaller streams and during periods of low flow.
23   Management of the rate and timing of surface water withdrawals has been shown to help mitigate
24   potential impacts of hydraulic fracturing withdrawals on water quality.

## Chemical Mixing

25   Hydraulic fracturing fluids are developed to perform specific functions, including: create and
26   extend fractures, transport proppant, and place proppant in the fractures. The fluid generally
27   consists of three parts: (1) the base fluid, which is the largest constituent by volume and is typically
28   water; (2) the additives, which can be a single chemical or a mixture of chemicals; and (3) the
29   proppant. Additives are chosen to serve a specific purpose (e.g., adjust pH, increase viscosity, limit
30   bacterial growth). Chemicals generally comprise a small percentage (typically 2% or less) of the
31   overall injected fluid volume. Because over one million gallons of fluids are typically injected per
32   well, thousands of gallons of chemicals can be potentially stored on-site and used during hydraulic
33   fracturing activities.

34   On-site storage, mixing, and pumping of chemicals and hydraulic fracturing fluids have the potential
35   to result in accidental releases, such as spills or leaks. Potential impacts to drinking water resources
36   from spills of hydraulic fracturing fluids and chemicals depend on the characteristics of the spills,
37   and the fate, transport, and the toxicity of chemicals spilled.

### Research Questions: Chemical Mixing

- **What is currently known about the frequency, severity, and causes of spills of hydraulic fracturing fluids and additives?**

1  The frequency of on-site spills from hydraulic fracturing could be estimated for two states, but not
2  for operations nationally or for other areas. Frequency estimates from data and literature ranged
3  from one spill for every 100 wells in Colorado to between approximately 0.4 and 12.2 spills for
4  every 100 wells in Pennsylvania.[1] These estimates include spills of hydraulic fracturing fluids and
5  chemicals, and produced water reported in state databases. Available data generally precluded
6  estimates of hydraulic fracturing fluid and/or chemical spill rates separately from estimates of an
7  overall spill frequency. It is unknown whether these spill estimates are representative of national
8  occurrences. If the estimates are representative, the number of spills nationally could range from
9  approximately 100 to 3,700 spills annually, assuming 25,000 to 30,000 new wells are fractured per
10  year.

11  The EPA characterized volumes and causes of hydraulic fracturing-related spills identified from
12  selected state and industry data sources. The spills occurred between January 2006 and April 2012
13  in 11 states and included 151 cases in which fracturing fluids or chemicals spilled on or near a well
14  pad. Due to the methods used for the EPA's characterization of spills, these cases were likely a
15  subset of all fracturing fluid and chemical spills during the study's time period. The reported
16  volume of fracturing fluids or chemicals spilled ranged from 5 gal to more than 19,000 gal (19 to
17  72,000 L), with a median volume of 420 gal (1,600 L) per spill. Spill causes included equipment
18  failure, human error, failure of container integrity, and other causes (e.g., weather and vandalism).
19  The most common cause was equipment failure, specifically blowout preventer failure, corrosion,
20  and failed valves. More than 30% of the 151 fracturing fluid or chemical spills were from fluid
21  storage units (e.g., tanks, totes, and trailers).

- **What are the identities and volumes of chemicals used in hydraulic fracturing fluids, and how might this composition vary at a given site and across the country?**

22  In this assessment, we identified a list of 1,076 chemicals used in hydraulic fracturing fluids. This is
23  a cumulative list over multiple wells and years. These chemicals include acids, alcohols, aromatic
24  hydrocarbons, bases, hydrocarbon mixtures, polysaccharides, and surfactants. According to the
25  EPA's analysis of disclosures to FracFocus, the number of unique chemicals per well ranged from 4
26  to 28, with a median of 14 unique chemicals per well.

27  Our analysis indicates that chemical use varies and that no single chemical is used at all well sites
28  across the country, although several chemicals are widely used. Methanol, hydrotreated light
29  petroleum distillates, and hydrochloric acid were reported as used in 65% or more of wells,
30  according to FracFocus disclosures analyzed by the EPA. Only 32 chemicals, excluding water,
31  quartz, and sodium chloride, were used in more than 10% of wells according to the EPA's analysis

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[1] Spill frequency estimates are for a given number of wells over a given period of time. These are not annual estimates nor are they for the lifetime of a well.

1    of FracFocus disclosures. The composition of hydraulic fracturing fluids varies by state, by well, and
2    within the same service company and geologic formation. This variability likely results from several
3    factors, including the geology of the formation, the availability and cost of different chemicals, and
4    operator preference.

5    Estimates from the EPA's database developed from FracFocus suggest median volumes of
6    individual chemicals injected per well ranged from a few gallons to thousands of gallons, with an
7    overall median of 650 gal (2,500 L) per chemical per well. Based on this overall median and
8    assuming 14 unique chemicals are used per well, an estimated 9,100 gal (34,000 L) of chemicals
9    may be injected per well. Given that the number of chemicals per well ranges from 4 to 28, the
10   estimated volume of chemicals injected per well may range from approximately 2,600 to 18,000 gal
11   (9,800 to 69,000 L).

- ***What are the chemical, physical, and toxicological properties of hydraulic fracturing chemical additives?***

12   Measured or estimated physicochemical properties were obtained for 453 chemicals of the total
13   1,076 chemicals reported in hydraulic fracturing fluids. We could not estimate physicochemical
14   properties for the inorganic chemicals or mixtures. The 453 chemicals have a wide range of
15   physicochemical properties.

16   Properties affecting the likelihood of a spilled chemical reaching and impacting a drinking water
17   resource include mobility, solubility, and volatility. Of the 453 chemicals for which physicochemical
18   properties were available, 18 of the top 20 most mobile ones were reported in the EPA's FracFocus
19   database for 2% or less of wells. Choline chloride and tetrakis (hydroxymethyl) phosphonium were
20   exceptions and were reported in 14% and 11% of wells, respectively. These two chemicals appear
21   to be relatively more common, and, if spilled, would move quickly through the environment with
22   the flow of water. The majority of the 453 chemicals associate strongly with soils and organic
23   materials, suggesting the potential for these chemicals to persist in the environment as long-term
24   contaminants. Many of the 453 chemicals fully dissolve in water, but their aqueous solubility varies
25   greatly. Few of the chemicals volatilize, and thus a large proportion of most hydraulic fracturing
26   chemicals tend to remain in water.

27   Oral reference values and oral slope factors meeting the criteria used in this assessment were not
28   available for the majority of chemicals used in hydraulic fracturing fluids, representing a significant
29   data gap for hazard identification.[1,2] Reference values and oral slope factors are important for
30   understanding the potential human health effects resulting from exposure to a chemical. Chronic
31   oral reference values and/or oral slope factors from selected federal, state, and international
32   sources were available for 90 (8%) of the 1,076 chemicals used in hydraulic fracturing fluids. From

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[1] A reference value is an estimate of an exposure to the human population (including susceptible subgroups) for a given duration that is likely to be without an appreciable risk of adverse health effects over a lifetime. Reference value is a generic term not specific to a given route of exposure.
[2] An oral slope factor is an upper-bound, approximating 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent.

1    U.S. federal sources alone, chronic oral reference values were available for 73 chemicals (7%) of the
2    1,076 chemicals, and oral slope factors were available for 15 chemicals (1%). Of the 32 chemicals
3    reported as used in at least 10% of wells in the EPA's FracFocus database (excluding water, quartz,
4    and sodium chloride), seven (21%) have a federal chronic oral reference value. Oral reference
5    values and oral slope factors are a key component of the risk assessment process, although
6    comprehensive risk assessments that characterize the health risk associated with exposure to these
7    chemicals are not available.

8    Of the chemicals that had values available, the health endpoints associated with those values
9    include the potential for carcinogenesis, immune system effects, changes in body weight, changes in
10    blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and
11    developmental toxicity. However, it is important to note that evaluating any potential risk to human
12    populations would require knowledge of the specific chemicals that are present at a particular site,
13    whether or not humans are exposed to those chemicals and, if so, at what levels and for what
14    duration, and the toxicity of the chemicals. Since most chemicals are used infrequently on a
15    nationwide basis, potential exposure is likely to be a local or regional issue, rather than a national
16    issue. Accordingly, consideration of hazards and risks associated with these chemical additives
17    would be most useful on a site-specific basis and is beyond the scope of this assessment.

- ***If spills occur, how might hydraulic fracturing chemical additives contaminate drinking water resources?***

18    There are several mechanisms by which a spill can potentially contaminate drinking water
19    resources. These include overland flow to nearby surface water, soil contamination and eventual
20    transport to surface water, and infiltration and contamination of underlying ground water. Of the
21    151 spills characterized by the EPA, fluids reached surface water in 13 (9% of 151) cases and soil in
22    97 (64%) cases. None of the spills of hydraulic fracturing fluid were reported to have reached
23    ground water. This could be due to an absence of impact; however, it can take several years for
24    spilled fluids to infiltrate soil and leach into ground water. Thus, it may not be immediately
25    apparent whether a spill has reached ground water or not.

26    Based on the relative importance of each of these mechanisms, impacts have the potential to occur
27    quickly, be delayed short or long periods, or have a continual effect over time. In Kentucky, for
28    example, a spill impacted a surface water body relatively quickly when hydraulic fracturing fluid
29    entered a creek, significantly reducing the water's pH and increasing its conductivity (Papoulias
30    and Velasco, 2013).

## *Well Injection*

31    Hydraulic fracturing fluids are injected into oil or gas wells under high pressures. The fluids flow
32    through the well (commonly thousands of feet below the surface) into the production zone (i.e., the
33    geologic formation being fractured) where the fluid injection pressures are sufficient to create
34    fractures in the rock.

35    There are two major subsurface mechanisms by which the injection of fluid and the creation and
36    propagation of fractures can lead to contamination of drinking water resources: (1) the unintended

1   movement of liquids or gases out of the production well or along the outside of the production well
2   into a drinking water resource via deficiencies in the well's casing or cement, and (2) the
3   unintended movement of liquids or gases from the production zone through subsurface geologic
4   formations into a drinking water resource. Combinations of these two mechanisms are also
5   possible.

### Research Questions: Well Injection

- **How effective are current well construction practices at containing fluids—both liquids and gases—before, during, and after fracturing?**

6   Production wells are constructed to access and convey hydrocarbons from the formations in which
7   they are found to the surface, and to isolate fluid-bearing zones (containing oil, gas, or water) from
8   each other. Typically, multiple casings are emplaced and cemented along the wellbore to protect
9   and isolate the oil and/or natural gas from the formations it must travel through to reach the
10  surface.

11  Below ground drinking water resources are often separated from the production well using casing
12  and cement. Cemented surface casing, in particular, is an important well construction feature for
13  isolating drinking water resources from liquids and gases that may move through the subsurface. A
14  limited risk modeling study of selected injection wells in the Williston Basin in North Dakota
15  suggests that the risk of aquifer contamination from leaks inside the well to the drinking water
16  resource decreases by a factor of approximately one thousand when surface casing extends below
17  the bottom of the drinking water resource ([Michie and Koch, 1991](#)). Most wells used in hydraulic
18  fracturing operations have casing and a layer of cement to protect drinking water resources, but
19  there are exceptions: a survey conducted by the EPA of oil and gas production wells hydraulically
20  fractured by nine oil and gas service companies in 2009 and 2010 estimated that at least 3% of the
21  wells (600 out of 23,000 wells) did not have cement across a portion of the casing installed through
22  the protected ground water resource identified by well operators. The absence of cement does not
23  in and of itself lead to an impact.  However, it does reduce the overall number of casing and cement
24  barriers fluids must travel through to reach ground water resources.

25  Impacts to drinking water resources from subsurface liquid and gas movement may occur if casing
26  or cement are inadequately designed or constructed, or fail. There are several examples of these
27  occurrences in hydraulically fractured wells that have or may have resulted in impacts to drinking
28  water resources. In one example, an inner string of casing burst during hydraulic fracturing, which
29  resulted in a release of fluids on the land surface and possibly into the aquifer near Killdeer, North
30  Dakota. The EPA found that, based on the data analysis performed for the study, the only potential
31  source consistent with conditions observed in two impacted monitoring wells was the blowout that
32  occurred during hydraulic fracturing ([U.S. EPA, 2015j](#)). In other examples, inadequately cemented
33  casing has contributed to impacts to drinking water resources. In Bainbridge, Ohio, inadequately
34  cemented casing in a hydraulically fractured well contributed to the buildup of natural gas and high
35  pressures along the outside of a production well. This ultimately resulted in movement of natural
36  gas into local drinking water aquifers ([Bair et al., 2010](#); [ODNR, 2008](#)). In the Mamm Creek gas field

1    in Colorado, inadequate cement placement in a production well allowed methane and benzene to
2    migrate along the production well and through natural faults and fractures to drinking water
3    resources ([Science Based Solutions LLC, 2014](#); [Crescent, 2011](#); [COGCC, 2004](#)). These cases illustrate
4    how construction issues, sustained casing pressure, and the presence of natural faults and fractures
5    can work together to create pathways for fluids to migrate toward drinking water resources.

6    Fracturing older wells may also increase the potential for impacts to drinking water resources via
7    movement of gases and liquids from the inside of the production well or along the outside of the
8    production well to ground water resources. The EPA estimated that 6% of 23,000 oil and gas
9    production wells were drilled more than 10 years before being hydraulically fractured in 2009 or
10   2010. Although new wells can be designed to withstand the stresses associated with hydraulic
11   fracturing operations, older wells may not have been built or tested to the same specifications and
12   their reuse for this purpose could be of concern. Moreover, aging and use of the well can contribute
13   to casing degradation, which can be accelerated by exposure to corrosive chemicals, such as
14   hydrogen sulfide, carbonic acid, and brines.

- ***Can subsurface migration of fluids—both liquids and gases—to drinking water resources occur, and what local geologic or artificial features might allow this?***

15   Physical separation between the production zone and drinking water resources can help protect
16   drinking water. Many hydraulic fracturing operations target deep formations such as the Marcellus
17   Shale or the Haynesville Shale (Louisiana/Texas), where the vertical distance between the base of
18   drinking water resources and the top of the shale formation may be a mile or greater. Numerical
19   modeling and microseismic studies based on a Marcellus Shale-like environment suggest that
20   fractures created during hydraulic fracturing are unlikely to extend upward from these deep
21   formations into shallow drinking water aquifers.

22   Not all hydraulic fracturing is performed in zones that are deep below drinking water resources.
23   For example, operations in the Antrim Shale (Michigan) and the New Albany Shale
24   (Illinois/Indiana/Kentucky) take place at shallower depths (100 to 1,900 ft or 30 to 579 m), with
25   less vertical separation between the formation and drinking water resources. The EPA's survey of
26   oil and gas production wells hydraulically fractured by nine service companies in 2009 and 2010
27   estimated that 20% of 23,000 wells had less than 2,000 ft (610 m) of measured distance between
28   the point of shallowest hydraulic fracturing and the base of the protected ground water resources
29   reported by well operators.

30   There are also places in the subsurface where oil and gas resources and drinking water resources
31   co-exist in the same formation. Evidence indicates that hydraulic fracturing occurs within these
32   formations. This results in the introduction of fracturing fluids into formations that may currently
33   serve, or in the future could serve, as a source of drinking water for public or private use. According
34   to the data examined, the overall frequency of occurrence of this practice appears to be low, with
35   the activity generally concentrated in some areas in the western United States. The practice of
36   injecting fracturing fluids into a formation that also contains a drinking water resource directly
37   affects the quality of that water, since some of the fluid likely remains in the formation following
38   hydraulic fracturing. Hydraulic fracturing in a drinking water resource is a concern in the short-

1    term (should there be people currently using these zones as a drinking water supply) and the long-
2    term (if drought or other conditions necessitate the future use of these zones for drinking water).

3    Liquid and gas movement from the production zone to underground drinking water resources may
4    also occur via other production wells or injection wells near hydraulic fracturing operations.
5    Fractures created during hydraulic fracturing can intersect nearby wells or their fracture networks,
6    resulting in the flow of fluids into those wells. These well communications, or "frac hits," are more
7    likely to occur if wells are close to each other or on the same well pad. In the Woodford Shale in
8    Oklahoma, the likelihood of well communication was less than 10% between wells more than 4,000
9    ft (1,219 m) apart, but rose to nearly 50% between wells less than 1,000 ft (305 m) apart (Ajani and
10   Kelkar, 2012). If an offset well is not able to withstand the stresses applied during the hydraulic
11   fracturing of a neighboring well, well components may fail, which could result in a release of fluids
12   at the surface from the offset well. The EPA identified incidents in which surface spills of hydraulic
13   fracturing-related fluids were attributed to well communication events.

14   Older or inactive wells—including oil and gas wells, injection wells, or drinking water wells—near a
15   hydraulic fracturing operation may pose an even greater potential for impacts. A study in Oklahoma
16   found that older wells were more likely to be negatively affected by the stresses applied by
17   hydraulic fracturing in neighboring wells (Ajani and Kelkar, 2012). In some cases, inactive wells in
18   the vicinity of hydraulic fracturing activities may not have been plugged properly—many wells
19   plugged before the 1950s were done so with little or no cement. The Interstate Oil and Gas Compact
20   Commission estimates that over one million wells may have been drilled in the United States prior
21   to a formal regulatory system being in place, and the status and location of many of these wells are
22   unknown (IOGCC, 2008). State programs exist to plug identified inactive wells, and work is on-
23   going to identify and address such wells.

### Flowback and Produced Water

24   Water, of variable quality, is a byproduct of oil and gas production. After hydraulic fracturing, the
25   injection pressure is released and water flows back from the well. Initially this water is similar to
26   the hydraulic fracturing fluid, but as time goes on the composition is affected by the characteristics
27   of the formation and possible reactions between the formation and the fracturing fluid. Water
28   initially produced from the well after hydraulic fracturing is sometimes called flowback in the
29   literature, and the term appears in this assessment. However, hydraulic fracturing fluids and any
30   formation water returning to the surface are often referred to collectively as produced water. This
31   definition of produced water is used in this assessment.

32   The amount of produced water varies, but typically averages 10% to 25% of injected volumes,
33   depending upon the amount of time since fracturing and the particular well (see Figure ES-3a).
34   However, there are exceptions to this, such as in the Barnett Shale in Texas where the total volume
35   of produced water can equal or exceed the injected volume of hydraulic fracturing fluid (see Figure
36   ES-3b). Flow rates are generally high initially, and then decrease over time throughout oil or gas
37   production.

1   Impacts on drinking water resources have the potential to occur if produced water is spilled and
2   enters surface water or ground water. Environmental transport of chemical constituents in
3   produced water depends on the characteristics of the spill (e.g., volume and duration), the
4   composition of spilled fluids, and the characteristics of the surrounding environment.

### *Research Questions: Flowback and Produced Water*

- ***What is currently known about the frequency, severity, and causes of spills of flowback and produced water?***

5   Surface spills of produced water from hydraulically fractured wells have occurred. As noted in the
6   Chemical Mixing section above, the frequency of on-site spills from hydraulic fracturing activities
7   could be estimated for two states, but not nationally. Estimates of spill frequencies at hydraulic
8   fracturing sites in Colorado and Pennsylvania, including spills of produced water, ranged from
9   approximately 0.4 to 12.2 spills per 100 wells. Available data generally precluded estimates of
10  produced water spill rates separately from estimates of overall spill frequency. Away from the well,
11  produced water spills from pipelines and truck transport also have the potential to impact drinking
12  water resources.

13  The EPA characterized spill volumes and causes for 225 cases in which produced water spilled on
14  or near a well pad. These spills occurred between January 2006 and April 2012 in 11 states. The
15  median reported volume per produced water spill was 990 gallons (3,750 L), more than double that
16  for spills of hydraulic fracturing fluids and chemicals. The causes of produced water spills were
17  reported as human error, equipment failure, container integrity failure, miscellaneous causes (e.g.,
18  well communication), and unknown causes. Most of the total volume spilled (74%) for all 225 cases
19  combined was caused by a failure of container integrity.

- ***What is the composition of hydraulic fracturing flowback and produced water, and what factors might influence this composition?***

20  A combination of factors influence the composition of produced water, including: the composition
21  of injected hydraulic fracturing fluids, the type of formation fractured, subsurface processes, and
22  residence time. The initial chemical composition of produced water primarily reflects the chemistry
23  of the injected fluids. At later times, the chemical composition of produced water reflects the
24  geochemistry of the fractured formation.

25  Produced water varies in quality from fresh to highly saline, and can contain high levels of major
26  anions and cations, metals, organics, and naturally occurring radionuclides. Produced water from
27  shale and tight gas formations typically contains high levels of total dissolved solids (TDS) and ionic
28  constituents (e.g., bromide, calcium, chloride, iron, potassium, manganese, magnesium, and
29  sodium). Produced water also may contain metals (e.g., barium, cadmium, chromium, lead, and
30  mercury), and organic compounds such as benzene. Produced water from coalbed methane
31  typically has much lower TDS levels compared to other produced water types, particularly if the
32  coalbed was deposited under fresh water conditions..

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1    We identified 134 chemicals that have been detected in hydraulic fracturing produced water. These
2    include chemicals added during the chemical mixing stage, as well as naturally occurring organic
3    chemicals and radionuclides, metals, and other constituents of subsurface rock formations
4    mobilized by the hydraulic fracturing process. Data on measured chemical concentrations in
5    produced water were available for 75 of these 134 chemicals.

6    Most of the available data on produced water content are for shale and coalbed methane
7    formations, while less data are available for tight formations, such as sandstones. The composition
8    of produced water must be determined through sampling and analysis, both of which have
9    limitations—the former due to challenges in accessing production equipment, and the latter due to
10   difficulties identifying target analytes before analysis and the lack of appropriate analytical
11   methods. Most current data are for inorganic chemicals, while less data exist for organic chemicals.
12   Many more organic chemicals were reported as used in hydraulic fracturing fluid than have been
13   identified in produced water. The difference may be due to analytical limitations, limited study
14   scopes, and undocumented subsurface reactions.

- ***What are the chemical, physical, and toxicological properties of hydraulic fracturing flowback and produced water constituents?***

15   The identified constituents of produced water include inorganic chemicals (cations and anions, i.e.,
16   metals, metalloids, non-metals, and radioactive materials), organic chemicals and compounds, and
17   unidentified materials measured as total organic carbon and dissolved organic carbon. Some
18   constituents are readily transported with water (i.e., chloride and bromide), while others depend
19   strongly on the geochemical conditions in the receiving water body (i.e., radium and barium), and
20   assessment of their transport is based on site-specific factors. We were able to obtain actual or
21   estimated physicochemical properties for 86 (64%) of the 134 chemicals identified in produced
22   water.

23   As in the case of chemicals in hydraulic fracturing fluid, chemical properties that affect the
24   likelihood of an organic chemical in produced water reaching and impacting drinking water
25   resources include: mobility, solubility, and volatility. In general, physicochemical properties suggest
26   that organic chemicals in produced water tend to be less mobile in the environment. Consequently,
27   if spilled, these chemicals may remain in soils or sediments near spill sites. Low mobility may result
28   in smaller dissolved contaminant plumes in ground water, although these chemicals can be
29   transported with sediments in surface water or small particles in ground water. Organic chemical
30   properties vary with salinity, and effects depend on the nature of the chemical.

31   Oral reference values and/or oral slope factors from selected federal, state, and international
32   sources were available for 83 (62%) of the 134 chemicals detected in produced water. From U.S.
33   federal sources alone, chronic oral reference values were available for 70 (52%) of the 134
34   chemicals, and oral slope factors were available for 20 chemicals (15%). Of the chemicals that had
35   values available, noted health effects include the potential for carcinogenesis, immune system
36   effects, changes in body weight, changes in blood chemistry, pulmonary toxicity, neurotoxicity, liver
37   and kidney toxicity, and reproductive and developmental toxicity. As noted above, evaluating any
38   potential risk to human populations would require knowledge of the specific chemicals that are

1  present at a particular site, whether or not humans are exposed to those chemicals and, if so, at
2  what levels and for what duration, and the toxicity of the chemicals. The chemicals present in
3  produced water can vary based on the formation and specific well, due to differences in fracturing
4  fluid formulation and formation geology. Accordingly, consideration of hazards and risks associated
5  with these chemicals would be most useful on a  site-specific basis and is beyond the scope of this
6  assessment.

- ***If spills occur, how might hydraulic fracturing flowback and produced water contaminate drinking water resources?***

7  Impacts to drinking water resources from spills or releases of produced water depend on the
8  volume, timing, and composition of the produced water. Impacts are more likely the greater the
9  volume of the spill, the longer the duration of the release, and the higher the concentration of
10  produced water constituents (i.e., salts, naturally occurring radioactive material, and metals).

11  The EPA characterization of hydraulic fracturing-related spills found that 8% of the 225 produced
12  water spills included in the study reached surface water or ground water. These spills tended to be
13  of greater volume than spills that did not reach a water body. A well blowout in Bradford County,
14  Pennsylvania spilled an estimated 10,000 gal (38,000 L) of produced water into a tributary of
15  Towanda Creek, a state-designated trout fishery. The largest volume spill identified in this
16  assessment occurred in North Dakota, where approximately 2.9 million gal (11 million L) of
17  produced water spilled from a broken pipeline and impacted surface and ground water.

18  Chronic releases can and do occur from produced water disposed in unlined pits or impoundments,
19  and can have long-term impacts. Ground water impacts may persist longer than surface water
20  impacts because of lower flow rates and decreased mixing. Plumes from unlined pits used for
21  produced water have been shown to persist for long periods and extend to nearby surface water
22  bodies.

## *Wastewater Management and Waste Disposal*

23  Hydraulic fracturing generates large volumes of produced water that require management. In this
24  section we refer to produced water and any other waters generated onsite by the single term
25  "wastewater." Clark and Veil (2009) estimated that, in 2007, approximately one million active oil
26  and gas wells in the United States generated 2.4 billion gal per day (9.1 billion L per day) of
27  wastewater. There is currently no reliable way to estimate what fraction of this total volume can be
28  attributed to hydraulically fractured wells. Wastewater volumes in a region can increase sharply as
29  hydraulic fracturing activity increases.

30  Wastewater management and disposal could affect drinking water resources through multiple
31  mechanisms, including: inadequate treatment of wastewater prior to discharge to a receiving
32  water, accidental releases during transport or leakage from wastewater storage pits, unpermitted
33  discharges, migration of constituents in wastewaters following land application, inappropriate
34  management of residual materials from treatment, or accumulation of wastewater constituents in
35  sediments near outfalls of centralized waste treatment facilities (CWTs) or publicly owned
36  treatment works (POTWs) that have treated hydraulic fracturing wastewater. The scope of this

1   assessment excludes potential impacts to drinking water from the disposal of hydraulic fracturing
2   wastewater in underground injection control (UIC) wells.

### Research Questions: Wastewater Management and Waste Disposal

- ***What are the common treatment and disposal methods for hydraulic fracturing wastewater, and where are these methods practiced?***

3   Hydraulic fracturing wastewater is managed using several options, including: disposal in UIC wells
4   (also called disposal wells); through evaporation ponds; treatment at CWTs, followed by reuse or
5   by discharge to either surface waters or POTWs; reuse with minimal or no treatment; and land
6   application or road spreading. Treatment of hydraulic fracturing wastewater by POTWs was used in
7   the past in Pennsylvania. This decreased sharply following new state-level requirements and a
8   request by the Pennsylvania Department of Environmental Protection (PA DEP) for well operators
9   to stop sending Marcellus Shale wastewater to POTWs (and 15 CWTs) discharging to surface
10  waters.

11  Wastewater management decisions are generally based on the availability and associated costs
12  (including transportation) of disposal or treatment facilities. A survey of state agencies found that,
13  in 2007, more than 98% of produced water from the oil and gas industry was managed via
14  underground injection ([Clark and Veil, 2009](#)). Available information suggests that disposal wells are
15  also the primary management practice for hydraulic fracturing wastewater in most regions in the
16  United States (e.g., the Barnett Shale; see Figure ES-3b). The Marcellus Shale region is a notable
17  exception, where most wastewater is reused because of the small number of disposal wells in
18  Pennsylvania (see Figure ES-3a). Although this assessment does not address potential effects on
19  drinking water resources from the use of disposal wells, any changes in cost of disposal or
20  availability of disposal wells would likely influence wastewater management decisions.

21  Wastewater from some hydraulic fracturing operations is sent to CWTs, which may discharge
22  treated wastewater to surface waters, POTWs, or back to well operators for reuse in other
23  hydraulic fracturing operations. Available data indicate that the use of CWTs for treating hydraulic
24  fracturing wastewater is greater in the Marcellus Shale region than other parts of the country. Most
25  of the CWTs accepting hydraulic fracturing wastewater in Pennsylvania cannot significantly reduce
26  TDS, and many of these facilities provide treated wastewater to well operators for reuse and do not
27  currently discharge treated wastewater to surface water.

28  Reuse of wastewater for subsequent hydraulic fracturing operations may require no treatment,
29  minimal treatment, or more extensive treatment. Operators reuse a substantial amount (ca. 70-
30  90%) of Marcellus Shale wastewater in Pennsylvania (see Figure ES-3a). Lesser amounts of reuse
31  occur in other areas (e.g., the Barnett Shale; see Figure ES-3b). In certain formations, such as the
32  Bakken Shale in North Dakota, there is currently no indication of appreciable reuse.

33  In some cases, wastewater is used for land applications such as irrigation or road spreading for
34  deicing or dust suppression. Land application has the potential to introduce wastewater
35  constituents into surface water and ground water due to runoff and migration of brines. Studies of

1  road spreading of conventional oil and gas brines have found elevated levels of metals in soils and
2  chloride in ground water.

- ***How effective are conventional POTWs and commercial treatment systems in removing
  organic and inorganic contaminants of concern in hydraulic fracturing wastewater?***

3  Publicly owned treatment works using basic treatment processes are not designed to effectively
4  reduce TDS concentrations in highly saline hydraulic fracturing wastewater—although specific
5  constituents or constituents groups can be removed (e.g., metals, oil, and grease by chemical
6  precipitation or other processes). In some cases, wastewater treated at CWTs may be sent to a
7  POTW for additional treatment and discharge. It is blended with POTW influent to prevent
8  detrimental effects on biological processes in the POTW that aid in the treatment of wastewater.

9  Centralized waste treatment facilities with advanced wastewater treatment options such as reverse
10 osmosis, thermal distillation, or mechanical vapor recompression, reduce TDS concentrations and
11 can treat contaminants currently known to be in hydraulic fracturing wastewater. However, there
12 are limited data on the composition of hydraulic fracturing wastewater, particularly for organic
13 constituents. It is unknown whether advanced treatment systems are effective at removing
14 constituents that are generally not tested for.

- ***What are the potential impacts from surface water disposal of treated hydraulic fracturing
  wastewater on drinking water treatment facilities?***

15 Potential impacts to drinking water resources may occur if hydraulic fracturing wastewater is
16 inadequately treated and discharged to surface water. Inadequately treated hydraulic fracturing
17 wastewater may increase concentrations of TDS, bromide, chloride, and iodide in receiving waters.
18 In particular, bromide and iodide are precursors of disinfection byproducts (DBPs) that can form in
19 the presence of organic carbon in drinking water treatment plants or wastewater treatment plants.
20 Drinking water treatment plants are required to monitor for certain types of DBPs, because some
21 are toxic and can cause cancer.

22 Radionuclides can also be found in inadequately treated hydraulic fracturing wastewater from
23 certain shales, such as the Marcellus. A recent study by the PA DEP (2015b) found elevated radium
24 concentrations in the tens to thousands of picocuries per liter and gross alpha and gross beta in the
25 hundreds to thousands of picocuries per liter in effluent samples from some CWTs receiving oil and
26 gas wastewater. Radium, gross alpha, and gross beta were also detected in effluents from POTWs
27 receiving oil and gas wastewater (mainly as effluent from CWTs), though at lower concentrations
28 than from the CWTs. Research in Pennsylvania also indicates the accumulation of radium in
29 sediments and soils affected by the outfalls of some treatment plants that have handled oil and gas
30 wastewater, including Marcellus Shale wastewater, and other wastewaters (PA DEP, 2015b;
31 Warner et al., 2013a). Mobilization of radium from sediments and potential impacts on downstream
32 water quality depend upon how strongly the radium has sorbed to sediments. Impacts may also
33 occur if sediment is resuspended (e.g., following storm events). There is no evidence of radionuclide
34 contamination in drinking water intakes due to inadequately treated hydraulic fracturing
35 wastewater.

1   Hydraulic fracturing wastewaters contain other constituents such as barium, boron, and heavy
2   metals. Barium in particular has been documented in some shale gas produced waters. Little data
3   exist on metal and organic compound concentrations in untreated and treated wastewaters in
4   order to evaluate whether treatment is effective, and whether there are potential downstream
5   effects on drinking water resources when wastewater is treated and discharged.

## Key Data Limitations and Uncertainties

6   This assessment used available data and literature to examine the potential impacts of hydraulic
7   fracturing from oil and gas on drinking water resources nationally. As part of this effort, we
8   identified data limitations and uncertainties associated with current information on hydraulic
9   fracturing and its potential to affect drinking water resources. In particular, data limitations
10  preclude a determination of the frequency of impacts with any certainty. These limitations and
11  uncertainties are discussed in brief below.

### *Limitations in Monitoring Data and Chemical Information*

12  While many activities conducted as part of the hydraulic fracturing water cycle take place above
13  ground, hydraulic fracturing itself occurs below ground and is not directly observable. Additionally,
14  potential mechanisms identified in this assessment may result in impacts to drinking water
15  resources that are below ground (e.g., spilled fluids leaching into ground water). Data that could be
16  used to characterize the presence, migration, or transformation of chemicals in the subsurface
17  before, during, and after hydraulic fracturing were found to be scarce relative to the number of
18  hydraulically fractured oil and gas production wells. Specifically, local water quality data needed to
19  compare pre- and post-hydraulic fracturing conditions are not consistently collected or readily
20  available. The limited amount of data collected before and during hydraulic fracturing activities
21  reduces the ability to determine whether hydraulic fracturing affected drinking water resources in
22  cases of alleged contamination.

23  Information (identity, frequency of use, physicochemical and toxicological properties, etc.) on the
24  chemicals associated with the hydraulic fracturing water cycle is not complete and limits
25  understanding of potential impacts on drinking water resources. Well operators claimed at least
26  one chemical as confidential at more than 70% of wells reported to FracFocus and analyzed by the
27  EPA. The identity of these chemicals, and other chemicals in produced water, are needed to
28  understand their properties and would also help inform what chemicals to test for to establish
29  baseline conditions and to test for in the event of a suspected drinking water impact. Of the 1,173
30  total chemicals identified by the EPA in hydraulic fracturing fluid and flowback and produced
31  water, 147 have chronic oral reference values and/or oral slope factors from the sources that met
32  the selection criteria for inclusion in this assessment. Because the majority of chemicals identified
33  in this report do not have chronic oral reference values and/or oral slope factors, risk assessors at
34  the local and regional level may need to use alternative sources of toxicity information that could
35  introduce greater uncertainties.

### *Other Contributing Limitations*

1  We found other limitations that hamper the ability to fully assess the potential impacts of hydraulic
2  fracturing on drinking water resources nationally. These include the number and location of
3  hydraulically fractured wells, the location of drinking water resources, and information on changes
4  in industry practices. The lack of a definitive well count particularly contributes to uncertainties
5  regarding total water use or total wastewater volume estimates, and would limit any kind of
6  cumulative impact assessment. Lack of specific information about private drinking water well
7  locations and the depths of drinking water resources in relation to hydraulically fractured rock
8  formations and well construction features (e.g., casing and cement) limits the ability to assess
9  whether subsurface drinking water resources are isolated from hydraulically fractured oil and gas
10 production wells. Finally, this assessment is a snapshot in time, and the industry is rapidly changing
11 (e.g., the number of wells fractured, the location of activities, and the chemicals used). It is unclear
12 how changes in industry practices could affect potential drinking water impacts in the future.
13 Consideration of future development scenarios was not a part of this assessment, but such an
14 evaluation could help establish potential short- and long-term impacts to drinking water resources
15 and how to assess them.

## Conclusions

16 Through this national-level assessment, we have identified potential mechanisms by which
17 hydraulic fracturing could affect drinking water resources. Above ground mechanisms can affect
18 surface and ground water resources and include water withdrawals at times or in locations of low
19 water availability, spills of hydraulic fracturing fluid and chemicals or produced water, and
20 inadequate treatment and discharge of hydraulic fracturing wastewater. Below ground mechanisms
21 include movement of liquids and gases via the production well into underground drinking water
22 resources and movement of liquids and gases from the fracture zone to these resources via
23 pathways in subsurface rock formations.

24 We did not find evidence that these mechanisms have led to widespread, systemic impacts on
25 drinking water resources in the United States. Of the potential mechanisms identified in this report,
26 we found specific instances where one or more of these mechanisms led to impacts on drinking
27 water resources, including contamination of drinking water wells. The cases occurred during both
28 routine activities and accidents and have resulted in impacts to surface or ground water. Spills of
29 hydraulic fracturing fluid and produced water in certain cases have reached drinking water
30 resources, both surface and ground water. Discharge of treated hydraulic fracturing wastewater has
31 increased contaminant concentrations in receiving surface waters. Below ground movement of
32 fluids, including gas, most likely via the production well, have contaminated drinking water
33 resources. In some cases, hydraulic fracturing fluids have also been directly injected into drinking
34 water resources, as defined in this assessment, to produce oil or gas that co-exists in those
35 formations.

36 The number of identified cases where drinking water resources were impacted are small relative to
37 the number of hydraulically fractured wells. This could reflect a rarity of effects on drinking water

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1   resources, or may be an underestimate as a result of several factors. There is insufficient pre- and
2   post-hydraulic fracturing data on the quality of drinking water resources. This inhibits a
3   determination of the frequency of impacts. Other limiting factors include the presence of other
4   causes of contamination, the short duration of existing studies, and inaccessible information related
5   to hydraulic fracturing activities.

6   This state-of-the-science assessment contributes to the understanding of the potential impacts of
7   hydraulic fracturing on drinking water resources and the factors that may influence those impacts.
8   The findings in this assessment can be used by federal, state, tribal, and local officials; industry; and
9   the public to better understand and address any vulnerabilities of drinking water resources to
10  hydraulic fracturing activities. This assessment can also be used to help facilitate and inform
11  dialogue among interested stakeholders, and support future efforts, including: providing context to
12  site-specific exposure or risk assessments, local and regional public health assessments, and
13  assessments of cumulative impacts of hydraulic fracturing on drinking water resources over time
14  or over defined geographic areas of interest. Finally, and most importantly, this assessment
15  advances the scientific basis for decisions by federal, state, tribal, and local officials, industry, and
16  the public, on how best to protect drinking water resources now and in the future.

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# Chapter 1

# Introduction

# 1. Introduction

## 1.1. Background

1   Since the early 2000s, oil and natural gas production in the United States has been transformed
2   through the technological innovations of hydraulic fracturing and directional drilling. Hydraulic
3   fracturing is a stimulation technique used to increase production of oil and gas. It involves the
4   injection of fluids under pressures great enough to fracture the oil- and gas-production formations.
5   Hydraulic fracturing in combination with advanced directional drilling techniques has made it
6   possible to economically extract hydrocarbons from unconventional resources, such as shale, tight
7   formations, and coalbeds.[1] It can also enhance production from conventional resources. The surge
8   in use of hydraulic fracturing and associated technologies has significantly increased domestic
9   energy supplies (see Chapter 2) and brought economic benefits to many areas of the United States.

10  The growth in domestic oil and gas exploration and production– the direct result of the expanded
11  use of hydraulic fracturing– has also raised concerns about its potential for impacts to human
12  health and the environment. Specific concerns have been raised by the public about the effects of
13  hydraulic fracturing on the quality and quantity of drinking water resources. Some residents living
14  close to oil and gas production well sites report changes in the quality of ground water resources
15  used for drinking water and assert that hydraulic fracturing is responsible for these changes. Other
16  concerns include competition for water between hydraulic fracturing operations and other water
17  users, especially in areas of the country experiencing drought, and the disposal of wastewater
18  generated from hydraulic fracturing. In response to public concerns, the U.S. Congress urged the
19  U.S. Environmental Protection Agency (EPA) to study the relationship between hydraulic fracturing
20  and drinking water (H.R. Rep. 111-316, 2009). In 2011, the EPA published its *Plan to Study the*
21  *Potential Impacts of Hydraulic Fracturing on Drinking Water Resources* (U.S. EPA, 2011c; hereafter
22  Study Plan). The research described in the Study Plan began the same year. In 2012, the EPA issued
23  *Potential Impacts of Hydraulic Fracturing on Drinking Water Resources: Progress Report* (U.S. EPA,
24  2012f; hereafter Progress Report) in order to update the public on the status of the research being
25  conducted under the Study Plan. In this report, we review and synthesize scientific literature,
26  including the publications resulting from the EPA's research and information provided by
27  stakeholders, to assess the potential for hydraulic fracturing for oil and gas to change the quality or
28  quantity of drinking water resources. This report also identifies factors affecting the frequency or
29  severity of any potential impacts.

## 1.2. Scope

30  This assessment focuses on hydraulic fracturing in onshore oil and gas wells in the contiguous
31  United States; limited available information on hydraulic fracturing in Alaska is included. To the

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[1] Unconventional resources is an umbrella term for oil and natural gas that is produced by means that do not meet the criteria for conventional production. What has qualified as unconventional at any particular time is a complex function of resource characteristics, the available exploration and production technologies, the economic environment, and the scale, frequency, and duration of production from the resource (see Text Box 2-2).

1    extent possible, this assessment addresses hydraulic fracturing in all types of oil- and gas-bearing
2    formations in which it is conducted, including shale, so-called 'tight' formations (e.g., certain
3    sandstones, siltstones, and carbonates), coalbeds, and conventional reservoirs. It tends to focus on
4    hydraulic fracturing in shale, which reflects the relatively large amount of literature and available
5    data on hydraulic fracturing in this type of geologic formation.

6    The scope of activities examined in this assessment is defined by the hydraulic fracturing water
7    cycle. This cycle encompasses activities involving water that support hydraulic fracturing and
8    consists of five stages: (1) acquisition of water needed to create hydraulic fracturing fluids; (2)
9    mixing of water and chemicals on the well pad to create hydraulic fracturing fluids; (3) injection of
10   hydraulic fracturing fluids into the well to fracture the geologic formation; (4) management of
11   flowback and produced water on the well pad and in transit for reuse, treatment, or disposal; and
12   (5) reuse, treatment and discharge, or disposal of hydraulic fracturing wastewater (see Figure
13   1-1).[1,2,3,4]

14   Activities within the hydraulic fracturing water cycle can take place on or near the well pad or some
15   distance away. On-site activities include mixing and injecting hydraulic fracturing fluids and
16   capturing flowback and produced water. Water withdrawals and wastewater treatment and
17   disposal may occur in the same watershed, adjacent watersheds, or watersheds many miles away
18   from the production site.

19   This assessment focuses on impacts on drinking water resource quantity and quality. Consistent
20   with the Study Plan (U.S. EPA, 2011c), drinking water resources are defined broadly within this
21   report as any body of ground water or surface water that now serves, or in the future could serve,
22   as a source of drinking water for public or private use. This is broader than most regulatory
23   definitions of "drinking water" and encompasses both fresh and non-fresh bodies of water, since
24   trends indicate both types of water bodies are now and in the future will be used as sources of
25   drinking water (see Chapter 3). We note that drinking water resources provide not only water that
26   individuals actually drink but also water used for many additional purposes such as cooking and
27   bathing.

28   We assess potential effects on drinking water resources from business-as-usual operations as well
29   as from accidents and unintended releases that may occur during the hydraulic fracturing water
30   cycle (see Table 1-1).

---

[1] Hydraulic fracturing fluids are engineered fluids, typically consisting of a base fluid, additives, and proppants, that are pumped under high pressure into the well to create and hold open fractures in the formation.
[2] Flowback is defined multiple ways in the literature. In general, it is either fluids predominantly containing hydraulic fracturing fluid that return from a well to the surface or a process used to prepare the well for production (see Chapter 7).
[3] Produced water is water that flows from oil and gas wells.
[4] Hydraulic fracturing wastewater is flowback and 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 (see Chapter 8).
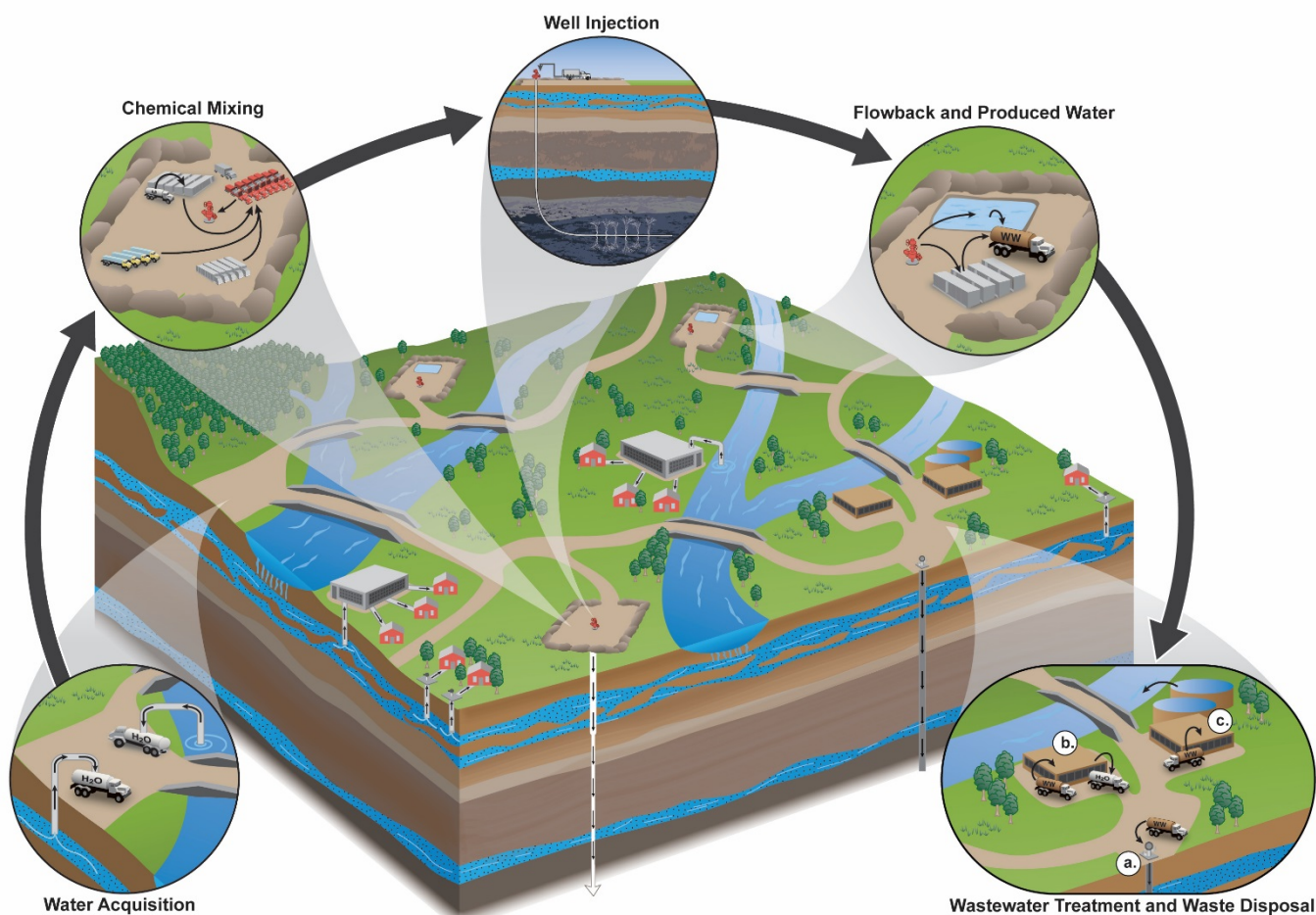
**Figure 1-1. Conceptualized view of the stages of the hydraulic fracturing water cycle.**

Shown here is a generalized landscape depicting the activities of the hydraulic fracturing water cycle and their relationship to each other, as well as their relationship to drinking water resources. Activities may take place in the same watershed or different watersheds and close to or far from drinking water resources. Drinking water resources are any body of ground water or surface water that now serves, or in the future could serve, as a source of drinking water for public or private use. Arrows depict the movement of water and chemicals. Specific activities in the "Wastewater Treatment and Waste Disposal" inset are (a) underground injection control (UIC) well disposal, (b) wastewater treatment and reuse, and (c) wastewater treatment and discharge at a centralized waste treatment (CWT) facility. Note: Figure not to scale.

*This document is a draft for review purposes only and does not constitute Agency policy.*

**Table 1-1. Stages of the hydraulic fracturing water cycle have various potential effects on drinking water resources.**

The potential effects addressed in this assessment, and how they are related to the activities within each stage, are summarized here.

| Water cycle stage | Activities or processes potentially affecting drinking water resources | Potential drinking water effects addressed in this assessment | | | |
| --- | --- | --- | --- | --- | --- |
| | | Quality | | Quantity | |
| | | Ground water | Surface water | Ground water | Surface water |
| **Water acquisition** | Water withdrawals | X | X | X | X |
| **Chemical mixing** | Spills of hydraulic fracturing fluids | X | X | | |
| **Well injection** | Subsurface migration of hydraulic fracturing fluids or formation fluids | X | X | | |
| **Flowback and produced water** | Spills of flowback or produced water | X | X | | |
| **Wastewater treatment and waste disposal** | Discharge of untreated or inadequately treated wastewater and inappropriate disposal of waste solids | X | X | | |

1 As part of the assessment, we evaluated immediate, near-term, and long-term effects on drinking
2 water resources. For example, we considered how surface spills of hydraulic fracturing fluids may
3 potentially have immediate or near-term impacts on neighboring surface water and shallow ground
4 water quality (see Chapters 5 and 7). We also considered how the potential release of hydraulic
5 fracturing fluids in the subsurface may take years to impact ground water resources, because
6 liquids and gas often move slowly in the subsurface (see Chapter 6). Additionally, effects may be
7 detected near the activity or at some distance away. For instance, we considered that, depending on
8 the constituents of treated hydraulic fracturing wastewater discharged to a stream and the flow in
9 that stream, drinking water resource quality could be affected a significant distance downstream
10 (see Chapter 8).

11 This assessment focuses predominantly on activities supporting a single well or multiple wells on a
12 single well pad, accompanied by a more limited discussion of cumulative activities and the effects
13 that could result from having many wells on a landscape. Studies of cumulative effects are generally
14 lacking, but we use the scientific literature to address this topic where possible.[1]

15 We address *mechanisms* for impacts as well as *impacts* of hydraulic fracturing for oil and gas on
16 drinking water resources. In general, a mechanism is the means or series of events that links an
17 activity to an impact, while an impact is the end result of a mechanism and represents a change in
18 the entity of interest. Specific definitions used in this assessment are provided below.

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[1] Cumulative effects refer to combined changes in the environment that can take place as a result of multiple activities over time and/or space.

- A **mechanism** is a means or series of events by which an activity within the hydraulic fracturing water cycle has been observed to change the quality or quantity of drinking water resources.

- A **suspected mechanism** is a means or series of events by which hydraulic fracturing activities could logically have resulted in an observed change in the quality or quantity of drinking water resources. Available evidence may or may not be sufficient to determine if it is the *only* mechanism that caused the observed change.

- A **potential mechanism** is a means or series of events by which hydraulic fracturing activities could logically or theoretically (for instance, based on modeling) change the quality or quantity of drinking water resources but one that has not yet been observed.

- An **impact** is any observed change in the quality or quantity of drinking water resources, regardless of severity, that results from a mechanism.

- A **potential impact** is any change in the quality or quantity of drinking water resources that could logically occur as the result of a mechanism or potential mechanism but has not yet been observed.

Potential mechanisms and impacts, as well as suspected mechanisms, are addressed because data required to document mechanisms and impacts may be inaccessible, incomplete, or nonexistent. In addition, evidence may be insufficient to isolate the contribution of hydraulic fracturing to changes in the quality or quantity of drinking water resources from other human activities occurring nearby. We anticipate that our understanding of mechanisms and impacts will be advanced as the scientific community continues to evaluate potential health and environmental effects of hydraulic fracturing.

In this assessment, we also identify and discuss factors affecting the frequency or severity of changes to avoid a simple inventory of all specific situations in which hydraulic fracturing might alter drinking water quality or quantity. This allows knowledge about the conditions under which effects are likely or unlikely to occur to be applied to new circumstances (e.g., a new area of oil or gas development where hydraulic fracturing is expected to be used) and could inform the development of strategies to prevent impacts. Although no attempt has been made in this assessment to identify or evaluate comprehensive best practices for states, tribes, or the industry, we describe ways to avoid or reduce the impacts of hydraulic fracturing activities as they have been reported in the scientific literature. A summary and evaluation of current or proposed regulations and policies is beyond the scope of this report.

For this assessment, we did not conduct site-specific predictive modeling to quantitatively estimate environmental concentrations of contaminants in drinking water resources, although modeling studies conducted by others are described. Further, this report is not a human health risk assessment. It does not identify populations that are exposed to chemicals or other stressors in the environment, estimate the extent of exposure, or estimate the incidence of human health impacts (see Chapter 9).

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1   This assessment focuses on the potential impacts from activities in the hydraulic fracturing water
2   cycle on drinking water resources. It does not address all concerns that have been raised about
3   hydraulic fracturing nor about oil and gas exploration and production more generally. Activities
4   that are not considered include acquisition and transport of constituents of hydraulic fracturing
5   fluids besides water (e.g., sand mining and chemical production); site selection and well pad
6   development; other infrastructure development (e.g., roads, pipelines, compressor stations); site
7   reclamation; and well closure. We consider these activities to be outside the scope of the hydraulic
8   fracturing water cycle and, therefore, their impacts are not addressed in this assessment.
9   Additionally, this report does not discuss the potential impacts of hydraulic fracturing on other
10  water uses (e.g., agriculture or industry), other aspects of the environment (e.g., air quality or
11  ecosystems), worker health or safety, or communities.

## 1.3. Approach

12  This assessment relies on scientific literature and data that address topics within the scope of the
13  hydraulic fracturing water cycle. Scientific journal articles and peer-reviewed EPA reports that have
14  been published containing results from the EPA's hydraulic fracturing study comprise one set of
15  applicable literature. Other literature evaluated includes articles published in science and
16  engineering journals, federal and state government reports, non-governmental organization (NGO)
17  reports, and oil and gas industry publications. Data sources examined include federal- and state-
18  collected data sets, databases curated by federal and state government agencies, other publicly
19  available data and information, and data including confidential and non-confidential business
20  information submitted by industry to the EPA.[1]

### 1.3.1. EPA Hydraulic Fracturing Study Publications

21  The research topic areas and projects described in the Study Plan were developed with substantial
22  expert and public input, and they were designed to meet the data and information needs of this
23  assessment. As such, published, peer-reviewed results of the research conducted under the Study
24  Plan are incorporated and cited frequently throughout this assessment. As is customary in
25  assessments that synthesize a large body of literature and data, the results of EPA research are
26  contextualized and interpreted in combination with the other literature and data described in
27  Section 1.3.2. The articles and EPA reports themselves that give complete and detailed project
28  results can be found on the EPA's hydraulic fracturing website (www.epa.gov/hfstudy). For ease of
29  reference, a description of the individual projects, the type of research activity they represent (i.e.,
30  analysis of existing data, scenario evaluation, laboratory study, or case study), and the
31  corresponding citations of published articles and EPA reports that are referenced in this
32  assessment can be found in Appendix H.

### 1.3.2. Literature and Data Search Strategy

33  The EPA used a broad search strategy to identify approximately 3,700 sources of scientific
34  information that could be applicable to this assessment. This search strategy included both

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[1] Information was provided to the EPA by nine hydraulic fracturing service companies in response to a September 2010 information request and by nine oil and gas well operators in response to an August 2011 information request.

1    requesting input from scientists, stakeholders, and the public about relevant data and information,
2    and thorough searching of published information and applicable data.[1]

3    Over 1,400 articles, reports, data, and other sources of information were obtained through outreach
4    to the public, stakeholders, and scientific experts. The EPA requested material through many
5    venues, as follows. We received recommended literature from the Science Advisory Board (SAB),
6    the EPA's independent federal scientific advisory committee, from its review of the EPA's draft
7    Study Plan; its consultation on the EPA's Progress Report (U.S. EPA, 2012f); and during an SAB
8    briefing on new and emerging information related to hydraulic fracturing in fall 2013. Subject
9    matter experts and stakeholders also recommended literature through a series of technical
10   workshops and roundtables organized by the EPA between 2011 and 2013. In addition, the public
11   submitted material to the SAB during the SAB review of the draft Study Plan, Progress Report, and
12   briefing on emerging information, as well as in response to a formal request for data and
13   information posted in the *Federal Register* (EPA-HQ-ORD-2010-0674) in November 2012. The
14   submission deadline was extended from April to November 2013 to provide the public with
15   additional opportunity to provide input to the EPA.

16   Approximately 2,300 additional sources were identified by conducting searches for material that
17   could be applicable to the assessment via online scientific databases and federal, state, and
18   stakeholder websites. We searched these databases and websites in particular for (1) materials
19   addressing topics not covered by the documents submitted by experts, stakeholders, and the public
20   as noted above, and (2) newly emerging scientific studies. Multiple targeted and iterative searches
21   on topics determined to be within the scope of the assessment were conducted until fall 2014. After
22   that time, we largely included newer literature as it was recommended to us during our internal
23   technical reviews or as it came to our attention and was determined to be important for filling a gap
24   in information. In many cases, our searches uncovered the same material submitted by the public,
25   but approximately 2,300 new sources were also identified.

### 1.3.3.  Literature and Data Evaluation Strategy

26   We evaluated the literature and data identified in the search strategy above using the five
27   assessment factors outlined by the EPA Science Policy Council in *A Summary of General Assessment*
28   *Factors for Evaluating the Quality of Scientific and Technical Information* (U.S. EPA, 2003). The
29   factors are (1) applicability and utility, (2) evaluation and review, (3) soundness, (4) clarity and
30   completeness, and (5) uncertainty and variability. Table 1-2 lists these factors along with the
31   specific criteria for each that were developed for this assessment. We first evaluated all materials
32   for applicability. If "applicable" under the criteria, the reference was evaluated on the basis of the
33   other four factors.

34   Our objective was to consider and then cite literature in the assessment that fully conforms to all
35   criteria defining each assessment factor. However, the preponderance of literature on some topics
36   did not fully conform to some aspects of the outlined criteria. For instance, there were many white

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[1] This study did not review information contained in state and federal enforcement actions concerning alleged contamination of drinking water resources.

1  papers and reports in technical areas in which independent peer review is not standard practice or
2  is not well documented. Therefore, we included references in the assessment that were not peer-
3  reviewed but that addressed topics not found in the peer-reviewed literature, that provided useful
4  background information, or that corroborated conclusions in the peer-reviewed literature.

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**Table 1-2. Criteria developed for the five factors used to evaluate literature and data cited in this assessment.**

Criteria are consistent with those outlined by the EPA's Science Policy Council (U.S. EPA, 2003). Criteria are incorporated into the Quality Assurance Project Plans for this assessment (U.S. EPA, 2014g, 2013d).

| Factor | Criteria |
|---|---|
| **Applicability** | Document provides information useful for assessing the potential pathways for hydraulic fracturing activities to change the quality or quantity of drinking water resources, identifies factors that affect the frequency and severity of impacts, or suggests ways that potential impacts may be avoided or reduced. |
| **Review** | Document has been peer-reviewed. |
| **Soundness** | Document relies on sound scientific theory and approaches, and conclusions are consistent with data presented. |
| **Clarity/completeness** | Document provides underlying data, assumptions, procedures, and model parameters, as applicable, as well as information about sponsorship and author affiliations. |
| **Uncertainty/variability** | Document identifies uncertainties, variability, sources of error, and/or bias and properly reflects them in any conclusions drawn. |

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### 1.3.4.  Quality Assurance and Peer Review

5  The use of quality assurance (QA) and peer review helps ensure that the EPA conducts high-quality
6  science that can be used to inform policymakers, industry, and the public. QA activities performed
7  by the EPA ensure that the agency's environmental data are of sufficient quantity and quality to
8  support the data's intended use. The EPA prepared a programmatic Quality Management Plan (U.S.
9  EPA, 2014h) for all of the research conducted under the EPA's Study Plan, including the review and
10 synthesis of the scientific literature in this assessment. The hydraulic fracturing Quality
11 Management Plan describes the QA program's organizational structure; defines and assigns QA and
12 quality control (QC) responsibilities; and describes the processes and procedures used to plan,
13 implement, and assess the effectiveness of the quality system. The broad plan is then supported by
14 more detailed QA Project Plans (QAPPs). For instance, the QAPPs developed for this assessment
15 provide the technical approach and associated QA/QC procedures for our data and literature search
16 and evaluation strategies introduced in Section 1.3.2 and 1.3.3 (U.S. EPA, 2014g, 2013d). A QA audit
17 was conducted by the QA Manager during the preparation of this assessment in order to verify that
18 the appropriate QA procedures, criteria, reviews, and data verification were adequately performed
19 and documented. Identifying uncertainties is another aspect of QA; uncertainty, including data gaps
20 and data limitations, is discussed throughout this assessment.

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1    This report is classified as a Highly Influential Scientific Assessment (HISA), defined by the Office of
2    Management and Budget (OMB) as a scientific assessment that (1) could have a potential impact of
3    more than $500 million in any year or (2) is novel, controversial, or precedent-setting or has
4    significant interagency interest (OMB, 2004). The OMB describes specific peer review requirements
5    for HISAs. The EPA often engages the SAB as an external federal advisory committee to conduct
6    peer reviews of high-profile scientific matters relevant to the agency. Members of an ad hoc panel,
7    the same panel that was convened under the auspices of the SAB to provide comment on the
8    Progress Report, will also provide comment on this assessment.[1] Panel members were nominated
9    by the public and chosen to create a balanced review panel based on factors such as technical
10   expertise, knowledge, experience, and absence of any real or perceived conflicts of interest.

## 1.4.  Organization

11   This assessment begins with a general description of hydraulic fracturing activities and the role of
12   hydraulic fracturing in the oil and gas industry in the United States (see Chapter 2). It follows with a
13   characterization of drinking water resources in the continental United States, with a focus on areas
14   in which we estimate hydraulic fracturing has taken place over the time period of 2000–2013 (see
15   Chapter 3).

16   Chapters 4 through 8 are organized around the stages of the hydraulic fracturing water cycle (see
17   Figure 1-1) and address the potential for activities conducted during those stages to change the
18   quality or quantity of drinking water resources. Each of the stages is covered by a separate chapter.
19   There is also a chapter devoted to an examination of the properties of chemicals and constituents
20   that have been or may be used in hydraulic fracturing fluids or present in flowback and produced
21   water (see Chapter 9).

22   Each chapter addresses research questions developed under the Study Plan, as data and
23   information allow (see Table 1-3). Concise answers appear in text boxes at the end of each chapter.
24   The final chapter provides major conclusions and a synthesis of information presented across the
25   assessment. It also highlights significant gaps in information that contribute to uncertainties about
26   those conclusions (see Chapter 10).

**Table 1-3. Research questions addressed by this assessment.**

> Each chapter addresses research questions developed under the Study Plan. Chapters 2 and 3 develop
> background on hydraulic fracturing and drinking water resources, respectively.

| Chapter and water cycle stage | Research questions |
|---|---|
|  |  |

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[1] Information about this process is available online at http://yosemite.epa.gov/sab/sabproduct.nsf/
02ad90b136fc21ef85256eba00436459/b436304ba804e3f885257a5b00521b3b!OpenDocument.

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| Chapter and water cycle stage | Research questions |
|---|---|
| **Chapter 4 - Water Acquisition** | • What are the types of water used for hydraulic fracturing?<br><br>• How much water is used per well?<br><br>• How might cumulative water withdrawals for hydraulic fracturing affect drinking water quantity?<br><br>• What are the possible impacts of water withdrawals for hydraulic fracturing on water quality? |
| **Chapter 5 - Chemical Mixing** | • What is currently known about the frequency, severity, and causes of spills of hydraulic fracturing fluids and chemical additives?<br><br>• What are the identities and volumes of chemicals used in hydraulic fracturing fluids, and how might this composition vary at a given site and across the country?<br><br>• What are the chemical and physical properties of hydraulic fracturing chemical additives?<br><br>• If spills occur, how might hydraulic fracturing chemical additives contaminate drinking water resources? |
| **Chapter 6 - Well Injection** | • How effective are current well construction practices at containing fluids- both liquids and gases- before, during, and after fracturing?<br><br>• Can subsurface migration of fluids- both liquids and gases- to drinking water resources occur, and what local geologic or artificial features might allow this? |
| **Chapter 7 - Flowback and Produced Water** | • What is currently known about the frequency, severity, and causes of spills of flowback and produced water?<br><br>• What is the composition of hydraulic fracturing flowback and produced water, and what factors might influence this composition?<br><br>• What are the chemical and physical properties of hydraulic fracturing flowback and produced water constituents?<br><br>• If spills occur, how might hydraulic fracturing flowback and produced water contaminate drinking water resources? |

| Chapter and water cycle stage | Research questions |
|---|---|
| **Chapter 8 - Wastewater Treatment and Waste Disposal** | • What are the common treatment and disposal methods for hydraulic fracturing wastewater, and where are these methods practiced? <br><br> • How effective are conventional publicly owned treatment works and commercial treatment systems in removing organic and inorganic contaminants of concern in hydraulic fracturing wastewater? <br><br> • What are the potential impacts from surface water disposal of treated hydraulic fracturing wastewater on drinking water treatment facilities? |
| **Chapter 9 - Hazard Evaluation of Chemicals Across the Water Cycle Stages** | • What are the toxicological properties of hydraulic fracturing chemical additives? <br><br> • What are the toxicological properties of hydraulic fracturing flowback and produced water constituents? |

## 1.5. Intended Use

1   We expect that this report, as a synthesis of the science, will contribute to the understanding of the
2   potential impacts of hydraulic fracturing on drinking water resources and the factors that may
3   influence those impacts. The data and findings in this report can be used by federal, tribal, state,
4   and local officials; industry; and the public to better understand and address any vulnerabilities of
5   drinking water resources to hydraulic fracturing activities.

6   We expect this report will be used to help facilitate and inform dialogue among interested
7   stakeholders, including Congress, other federal agencies, states, tribal governments, the
8   international community, industry, NGOs, academia, and the general public. Additionally, the
9   identification of knowledge gaps will promote greater attention to these areas by researchers.

10  We also expect this report may support future assessment efforts. For instance, we anticipate that it
11  could contribute context to site-specific exposure or risk assessments of hydraulic fracturing, to
12  regional public health assessments, or to assessments of cumulative impacts of hydraulic fracturing
13  on drinking water resources over time or over defined geographic areas of interest.

14  Finally, and most importantly, this assessment advances the scientific basis for decisions by federal,
15  state, tribal, and local officials; industry; and the public on how best to protect drinking water
16  resources now and in the future.

## 1.6. References for Chapter 1

H.R. Rep. 111-316. Department of the Interior, Environment, and Related Agencies Appropriation Act, 2010: Conference report (to accompany H.R. 2996), (2009). http://www.gpo.gov/fdsys/pkg/CRPT-111hrpt316/pdf/CRPT-111hrpt316.pdf

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# Chapter 2

# Hydraulic Fracturing, Oil and Gas Production, and the U.S. Energy Sector

# 2. Hydraulic Fracturing, Oil and Gas Production, and the U.S. Energy Sector

1   This chapter provides general background information useful for understanding the in-depth
2   technical chapters that follow. We describe the process and purpose of hydraulic fracturing and the
3   situations and settings in which it is used (Section 2.1). Then, to place hydraulic fracturing in the
4   context of well site operations, we describe activities from site assessment and selection through
5   production to site closure. This helps illustrate the intensive nature of activities during the
6   relatively short hydraulic fracturing phase during the life of a production well (Section 2.2). Finally,
7   we characterize the prevalence of hydraulic fracturing in the United States, its importance in the oil
8   and gas industry today and into the future, and its role in the U.S. energy sector (Sections 2.3 and
9   2.4).

## 2.1.  What is Hydraulic Fracturing?

10  Hydraulic fracturing is a stimulation technique used to increase production of oil and gas. Hydraulic
11  fracturing involves the injection of fluids under pressures great enough to fracture the oil- and gas-
12  production formations. Hydraulic fracturing fluid transfers the pressure generated by equipment at
13  the surface into the subsurface to create fractures, and it carries and places the proppant into the
14  fractures so that they remain "propped" open after the injection pumping pressure is terminated
15  ([Gupta and Valkó, 2007](#)). Oil and gas can then flow through the fractures into the well and through
16  the well to the surface. Hydraulic fracturing has been used since the late 1940s and for the first
17  almost 50 years was used in vertical wells in conventional hydrocarbon reservoirs.[1] Hydraulic
18  fracturing is still used in these settings, but the process has evolved; technological developments
19  have led to the use of hydraulic fracturing in low-permeability (unconventional) hydrocarbon
20  reservoirs that could not otherwise be profitably produced (see Text Box 2-1). Wells stimulated by
21  hydraulic fracturing may be vertical, deviated, or horizontal in orientation (see Figure 2-1), and
22  they may be newly drilled or older at the time the fracturing is done.

23

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[1] A conventional reservoir is a reservoir in which buoyant forces keep hydrocarbons in place below a sealing caprock. Reservoir and fluid characteristics of conventional reservoirs typically permit oil or natural gas to flow readily into wellbores. The term is used to make a distinction from shale and other unconventional reservoirs, in which gas might be distributed throughout the reservoir at the basin scale, and in which buoyant forces or the influence of a water column on the location of hydrocarbons within the reservoir are not significant.

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*This document is a draft for review purposes only and does not constitute Agency policy.*

**Text Box 2-1. Is Hydraulic Fracturing "New"?**

Hydraulic fracturing in one form or another has been in use since the late 1940s, when a fracturing technique was patented by the Stanolind Oil and Gas Company and licensed to the Halliburton Oil Well Cementing Company. There are precedents that go back even further: reports from the early days of the oil and gas industry in the mid-19th century show producers trying to increase production by pumping fluids or dropping explosives into wells (Montgomery and Smith, 2010). Throughout its history, hydraulic fracturing has been used as a production technique to increase, or "stimulate," production from a well (some hydraulic fracturing methods are used to stimulate production in water wells, which is outside the scope of this report).

The groundwork for the transformation to modern hydraulic fracturing was laid in the 1970s and early 1980s, when a coalition of private companies, government agencies, and industry groups began sponsoring research into shale gas development technologies. During that period, Congress began to offer tax incentives to induce producers to apply the developing technologies in the field (Wang and Krupnick, 2013; EIA, 2011a; Yergin, 2011). The first horizontal wells were drilled in the mid-1980s in the Austin Chalk oil-bearing formation in Texas (Pearson, 2011; Haymond, 1991). Directional drilling and other emerging technologies matured in the late 1990s. In 2001, the Mitchell Energy company found a way to economically fracture the Barnett Shale in Texas. The company was bought by Devon Energy, a company with advanced experience in horizontal drilling. In 2002, seven wells were drilled and developed in the Barnett Shale using both horizontal drilling and hydraulic fracturing. Fifty-five more wells were completed in 2003 (Yergin, 2011). The techniques were rapidly adopted and further developed by others. By 2003/2004, modern hydraulic fracturing in the Barnett Shale was producing more gas than all other shale gas wells in the rest of the country (mostly shallow shale gas production in the Appalachian and Michigan Basins, see Section 2.4.1) (DOE, 2011b; Montgomery and Smith, 2010). By 2005, the new techniques were being used in low-permeability hydrocarbon plays outside of Texas, and modern hydraulic fracturing soon became the industry standard, driving the surge in U.S. production of natural gas.

Despite the long history of hydraulic fracturing, the culmination of technical innovations in the early 2000s represent an appreciable change. These innovations have made hydraulic fracturing economical enough to become standard practice in the oil and gas industry. Modern hydraulic fracturing (sometimes referred to as high-volume hydraulic fracturing) is characterized by the use of long horizontal wells and higher volumes of more complex mixtures of water, proppants, and chemical additives for injection as compared to earlier fracturing practices. Wells are often deep and long: shale gas production wells are commonly 5,000 to 13,500 ft (1,524 to 4,115 m) deep with long horizontal sections of 2,000 to 5,000 ft (610 to 1,524 m) or more in length. Other important advances occurred in oil and gas geophysical survey techniques (such as downhole telemetry and 3D seismic imaging) (Wang and Krupnick, 2013; EIA, 2011a). Hydraulic fracturing continues to be conducted in vertical production wells as well as conventional reservoirs using some of these newer techniques. Modern hydraulic fracturing has made it possible to extract resources in previously untapped hydrocarbon-bearing geologic settings, altering and expanding the geographic range of oil and gas production activities.
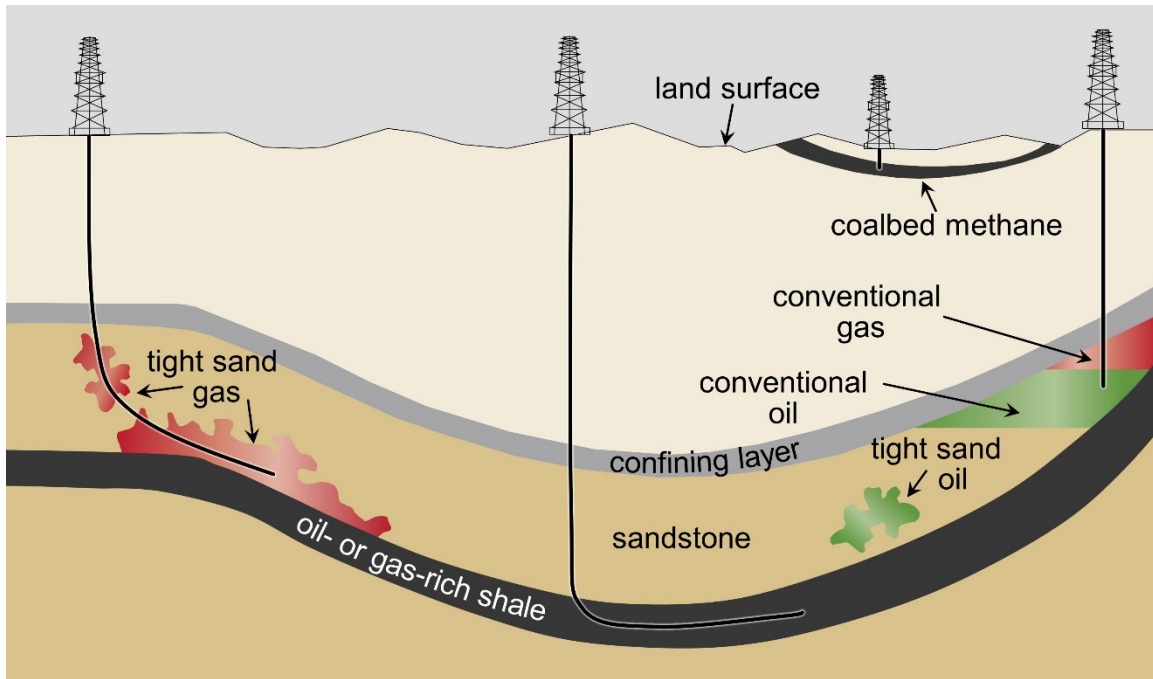
**Figure 2-1. Schematic cross-section of general types of oil and gas resources and the orientations of production wells used in hydraulic fracturing.**

Shown are conceptual illustrations of types of oil and gas wells. A vertical well is producing from a conventional oil and gas deposit (right). In this case, a thin, gray confining layer serves to "trap" oil (green) or gas (red). Also shown are wells producing from unconventional formations: a vertical coalbed methane well (second from right); a horizontal well producing from a shale formation (center); and a deviated well producing from a tight sand formation (left). Note: Figure not to scale. Modified from USGS (2002) and Newell (2011).

1  Historically, oil and gas have been extracted from conventional reservoirs that develop when
2  hydrocarbons formed in deeper geologic source formations migrate until they accumulate
3  underneath an impermeable layer (see Figure 2-1). Extraction practices vary. In settings where a
4  reservoir is permeable enough and under enough pressure to yield a relatively high rate of
5  hydrocarbon flow into a well, the economic extraction of oil and/or gas may be as simple as using a
6  drilled well to enable hydrocarbons to flow to the surface under the natural pressure of the
7  reservoir. In other cases, producers may inject water and/or carbon dioxide under pressure into
8  the reservoir via one or more nearby wells to help move and enhance production of the oil and gas.
9  But essentially, producers are drawing on hydrocarbons that have already accumulated in a
10 relatively accessible form.

11 Hydraulic fracturing is one of several methods used to enhance production from oil and gas
12 reservoirs. It is distinct from other methods of hydrocarbon extraction (known generally as
13 enhanced recovery techniques) that involve injecting fluids to influence either reservoir pressure,
14 fluid viscosity, or both. The primary purpose of hydraulic fracturing is to increase the surface area
15 of the reservoir rock by creating fractures that are propped open, allowing the hydrocarbon to flow
16 from the rock through the fractures to the well and through the well up to the surface.

1   Hydraulic fracturing, in conjunction with horizontal and directional drilling, has made it possible to
2   economically extract oil and gas from "unconventional" geologic formations (see Text Box 2-2),
3   such as the relatively low permeability shales in which oil and gas form (see Figure 2-1). With
4   modern horizontal drilling techniques, producers can, for example, drill a single well that follows
5   the contours of a relatively thin, horizontal shale formation. Such drilling allows fracturing to be
6   conducted in a long horizontal section of the well that accesses an extensive portion of the oil- or
7   gas-bearing formation. Unconventional formations include:

8   • **Shales.** Organic-rich black shales are the source rocks in which oil and gas form on geologic
9      timescales. Shales have very low permeability, and the hydrocarbons are contained in the
10     pore space in the shales. Some shales produce predominantly gas and others predominantly
11     oil; often there will be some coproduction of gas from oil wells and coproduction of liquid
12     hydrocarbons from gas wells (USGS, 2013a; EIA, 2011a).

13  • **Tight formations.** "Tight" sands (sandstones), siltstone, carbonates, etc., are relatively low
14     permeability, non-shale, sedimentary formations that can contain hydrocarbons. The
15     hydrocarbons are contained in the pore space of the formations. There is a continuum in
16     permeability between "tight" formations which require hydraulic fracturing to be produced
17     economically and sandstone (and other) formations that do not. In the literature, "tight gas"
18     is generally distinguished from "shale gas," while oil resources from shale and tight
19     formations are frequently lumped together under the label "shale oil" or "tight oil"
20     (Schlumberger, 2014; USGS, 2014a).

21  • **Coalbeds.** Hydraulic fracturing can be used to extract methane (the primary component of
22     natural gas) from coal seams. In coalbeds, the methane is adsorbed to the coal surface
23     rather than contained in pore space or structurally trapped in the formation. Pumping the
24     injected and formation water out of the coalbeds after fracturing serves to depressurize the
25     coal, thereby allowing the methane to desorb and flow into the well and to the surface
26     (USGS, 2000).

---

**Text Box 2-2. "Conventional" Versus "Unconventional."**

27  The terms "conventional" and "unconventional" are widely used in the literature to distinguish types of oil
28  and gas reservoirs, plays, wells, production techniques, and more. In this report, the terms are used to
29  distinguish different types of hydrocarbon resources: "conventional" resources are those that can
30  economically be extracted using long-established technologies, and "unconventional" resources are those
31  whose extraction has become economical only with the advances that have occurred in modern hydraulic
32  fracturing (often coupled with directional drilling) in recent years.

33  Note that as modern hydraulic fracturing has become industry standard, the word "unconventional" is less
34  apt than it once was to describe these resources. In a sense, "the unconventional has become the new
35  conventional" (NETL, 2013).

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36  Although the goal of stimulation by hydraulic fracturing is the same wherever it is employed, the
37  way it is accomplished varies due to a number of factors. General location and geologic conditions,

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1  whether the well is existing or newly drilled, the proximity of the well to infrastructure and raw
2  materials, operator preferences, and other factors can affect how a hydraulic fracturing operation is
3  designed and carried out. Technological advances have made it possible to drill deeper and longer
4  horizontal wells, to conduct fracturing through longer portions of the well, and to place multiple
5  wells on a single well pad (NETL, 2013; Montgomery and Smith, 2010). Many facets of hydraulic
6  fracturing-related technology have changed since they were first pioneered (see Text Box 2-1). How
7  hydraulic fracturing is practiced now (especially in the long horizontal wells) is different from how
8  it was conducted during the first decades of its use. As operators gain experience with both
9  evolving and new technologies, practices will continue to change.

10  The following three maps show the locations of major shale oil and gas resources, tight gas
11  resources, and coalbed methane resources, respectively, in the continental United States (see
12  Figure 2-2, Figure 2-3, and Figure 2-4). These maps represent resources that are being exploited
13  now or could be exploited in the future. Hydraulic fracturing continues to be used to enhance
14  production in conventional reservoirs (not shown), although it is uncertain how often this occurs.

15  The formations hydraulically fractured for gas or oil vary in their depth below the surface. For
16  example, the Marcellus Shale (found primarily in Pennsylvania, New York, and West Virginia) is
17  found at depths of 4,000 to 8,500 ft (1,200 to 2,600 m), the Barnett Shale (Texas) is found at depths
18  of 6,500 to 8,500 ft (2,000 to 2,600 m), and the Haynesville-Bossier Shale (Louisiana and Texas) is
19  found at depths of 10,500 to 13,500 ft (3,200 to 4,100 m) (NETL, 2013). These represent some of
20  the largest gas-producing shale formations or shale plays. However, some other plays are
21  shallower. Parts of the Antrim (Michigan), Fayetteville (Arkansas), and New Albany (Indiana and
22  Kentucky) shale plays, for example, are less than 2,000 ft (600 m) deep (NETL, 2013; GWPC and
23  ALL Consulting, 2009). Exploitation of thin coal seams often takes place close to the surface as well.
24  In the San Juan Basin (New Mexico), coal seams are 550 to 4,000 ft (170 to 1,200 m) deep; in the
25  Powder River Basin (Wyoming and Montana) they are 450 to greater than 6,500 ft (140 to 2,000 m)
26  deep, and in the Black Warrior Basin (Alabama and Mississippi) depths can range from the ground
27  surface to 3,500 ft (1,100 m) (ALL Consulting, 2004). See Chapter 6 for more information on the
28  depths of these formations and plays.
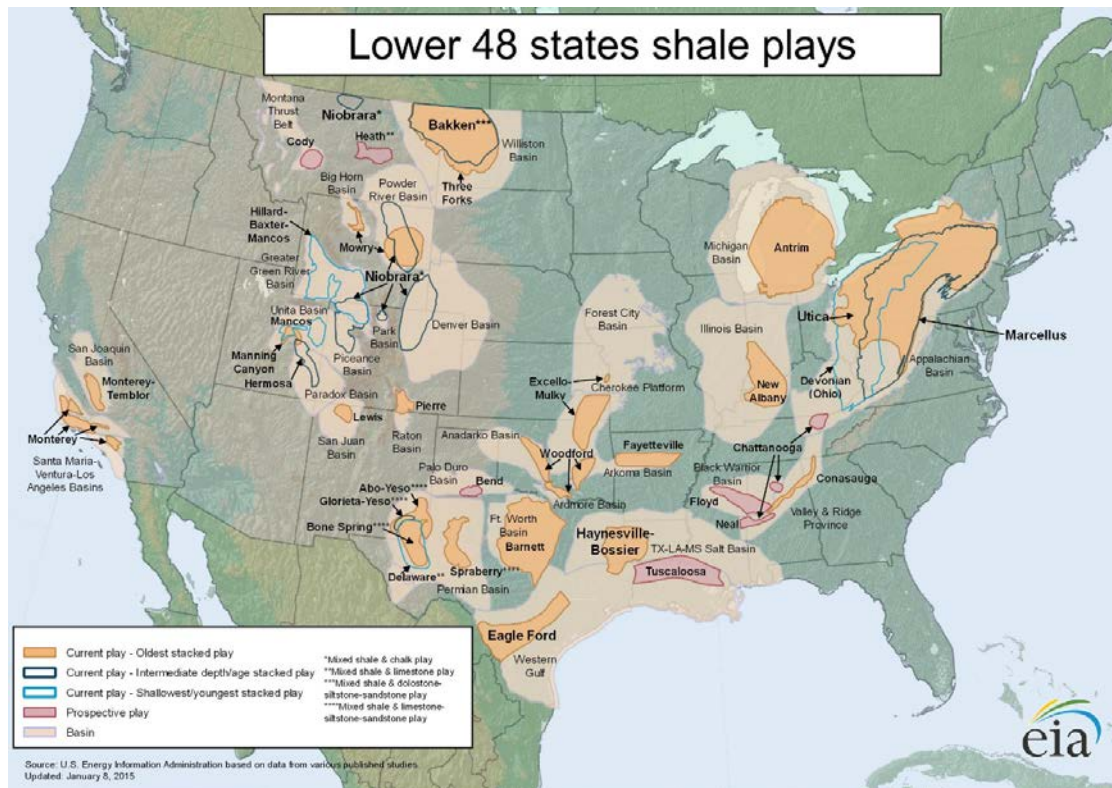
**Figure 2-2. Shale gas and oil plays in the lower 48 United States.**

Source: EIA (2015b).
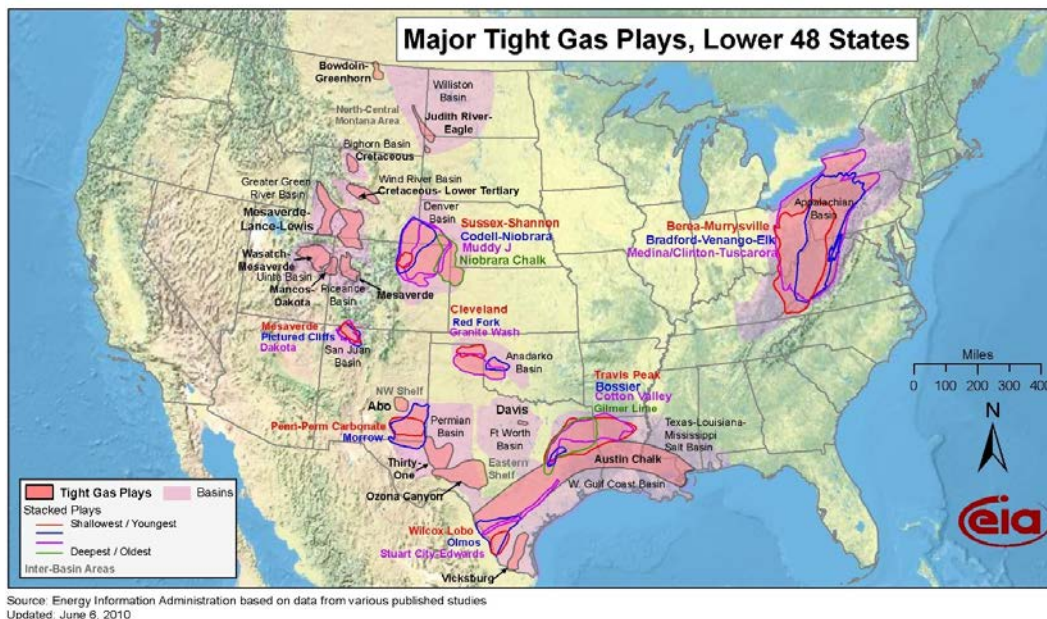
**Figure 2-3. Tight gas plays in the lower 48 United States.**
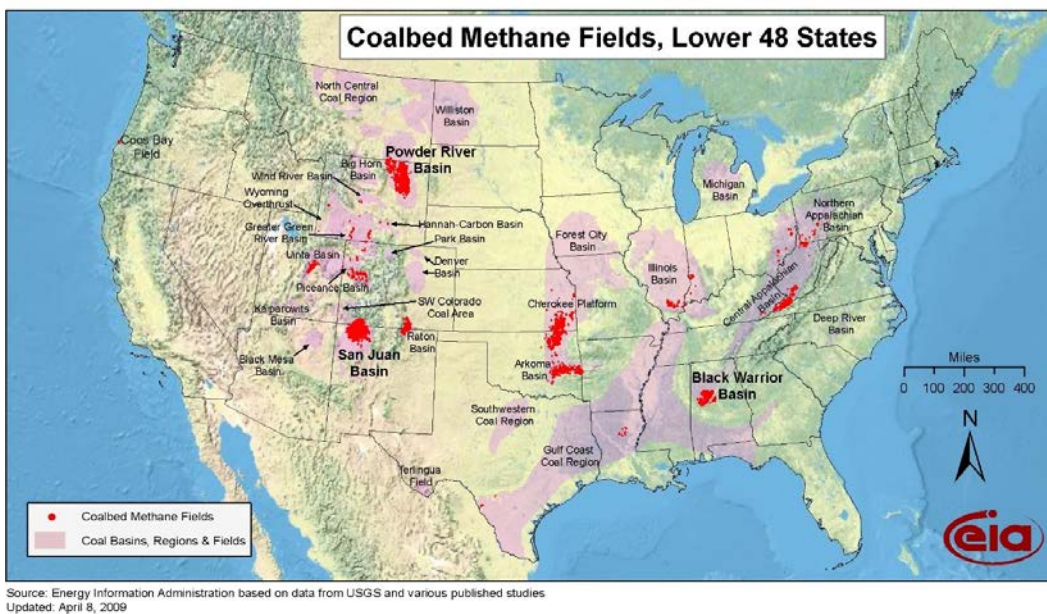
     Source: EIA (2011b).



**Figure 2-4. Coalbed methane fields in the lower 48 United States.**

     Source: EIA (2011b).

## 2.1.   Hydraulic Fracturing and the Life of a Well

1   Hydraulic fracturing itself is a relatively short-term process, with the timeframe for a typical
2   fracturing treatment being two to 10 days during which fluids are injected into the well to fracture
3   the oil- and gas-bearing geologic formations ([Halliburton, 2013](); [NYSDEC, 2011]()). However, it is a
4   period of intense activity— the most activity that takes place at a well site during its existence.

5   In this section, we briefly describe some of the supporting and ancillary activities that take place at
6   the well site, from initial site development through production and ultimately to closure (see Figure
7   2-5). This time period likely ranges from years to decades, depending on factors such as rate of
8   depletion of the oil or gas, cost of production, and the price of oil and gas. The rate of oil and gas
9   depletion in the reservoir is somewhat uncertain in unconventional formations because there is
10   relatively little history on which to base predictions.

11   The overview of well operations presented in this section is broad and is provided to illustrate
12   common activities and describe some specific operational details. The details of well preparation,
13   operations, and closure vary from company to company, from play to play, from jurisdiction to
14   jurisdiction, and from well to well. The various activities involved in well development and
15   operations can be conducted by the well owner and/or operator, owner/operator representatives,
16   service companies, or other third parties contractors working for the well owner.
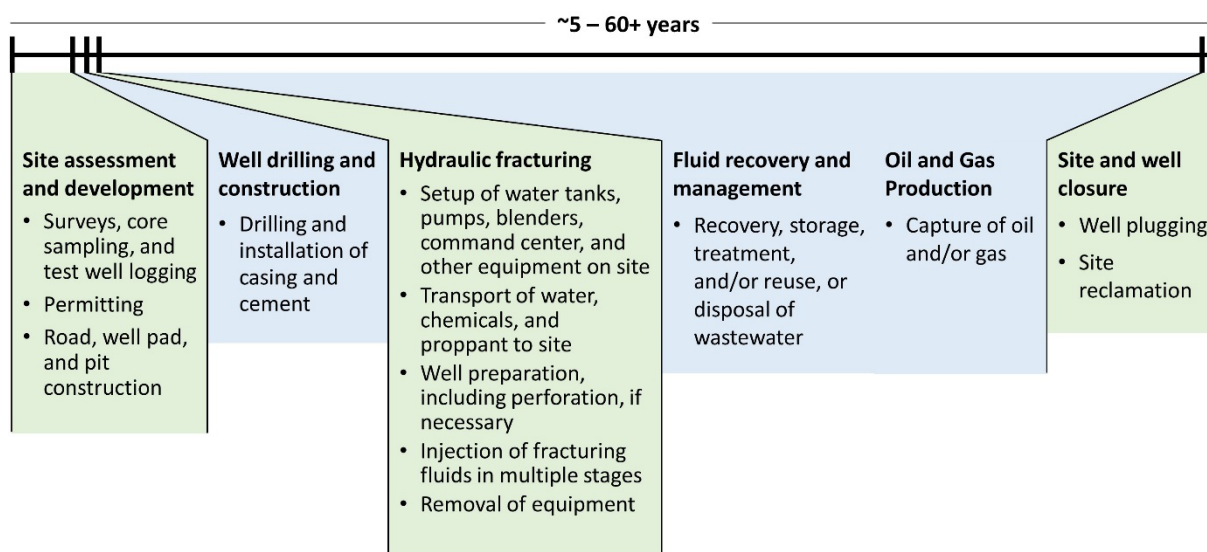


**Figure 2-5. Generalized timeline and summary of activities that take place during the operational phases of an oil or gas well site operation in which hydraulic fracturing is used.**

Relative duration of phases is approximate.

### 2.1.1. Site and Well Development

1   Numerous activities occur to assess and develop the site and to drill and construct the production
2   well before hydraulic fracturing and production can occur.

#### *2.1.1.1. Site Assessment and Development*

3   Identifying a geologically suitable well site requires integrating data from geophysical surveys
4   (including seismic surveys) that help to delineate subsurface features with other geologic
5   information from rock core samples. Cores may be obtained while drilling exploratory wells or test
6   holes. Core samples provide firsthand information on the characteristics of the oil- or gas-bearing
7   formation, such as porosity, permeability, and details about the quantities and qualities of the
8   hydrocarbon resource. Drilling rates and drill cuttings help identify the strata being drilled through
9   and can help confirm and correlate stratigraphy and formation depths, including the depths of
10  water-bearing formations.[1] Well logging (also known as wireline logging) is especially useful
11  combined with core analysis for understanding the properties of formations (Kundert and Mullen,
12  2009).[2]

13  Logistical factors involved in the selection of the well drilling site include topography; proximity to
14  facilities such as roads, pipelines, and water sources; well spacing considerations; well setback
15  requirements; potential for site erosion; location relative to environmentally sensitive areas; and
16  proximity to populated areas (Drohan and Brittingham, 2012; Arthur et al., 2009a). Before
17  developing the site and initiating well drilling, the oil and gas company (or their representative)
18  obtains a mineral rights lease, negotiates with landowners, and applies for a drilling permit from
19  the appropriate state and local authorities. During the project, leases and permissions are also
20  needed for other activities including performing seismic surveys and drilling exploratory holes
21  (Hyne, 2012). This initial site assessment phase of the process may take several months (King,
22  2012).

23  Site preparation is necessary to enable equipment and supplies to reach the well area. Typically, the
24  site is surveyed first, and then an access road may need to be built to accommodate truck traffic
25  (Hyne, 2012). The operator then levels and grades the site to manage drainage and to allow
26  equipment to be hauled to and placed on site. Next, the operator may excavate and grade several
27  impoundments or storage pits near the well pad. In some cases, steel tanks may be used to hold
28  fluids instead of, or in addition to, pits. The pits may hold water intended for drilling fluids,
29  materials generated during drilling such as used drilling mud and drill cuttings, or the flowback and
30  produced waters after fracturing (Hyne, 2012). Pit construction is generally governed by local
31  regulations; federal regulations may also apply on federal and Indian Country. In some areas,
32  regulations may require pits to be lined to prevent fluid seepage into the shallow subsurface or may

---

[1] Drill cuttings are ground rock produced by the drilling process.

[2] Well logging consists of a continuous measurement of physical properties in or around the well with electrically powered instruments to infer formation properties. Measurements may include electrical properties (resistivity and conductivity), sonic properties, active and passive nuclear measurements, measurements of the wellbore, pressure measurement, formation fluid sampling, sidewall coring tools and others. Measurements may be taken via a wireline, which is a wire or cable that is used to deploy tools and instruments downhole and that transmits data to the surface.

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1     prohibit pits altogether. Some sites have piping along the surface of the well pad or in the shallow
2     subsurface that delivers water used for hydraulic fracturing, removes flowback and produced
3     water, or transports the oil and gas once production begins ([Arthur et al., 2009a](#)).

4     After site and well pad preparation, drill rigs and associated equipment (e.g., the drill rig platform,
5     drilling mud system components, generators, chemical storage tanks, blowout preventer, fuel
6     storage tanks, cement pumps, drill pipe, and casing) are moved on and off the pad at the different
7     stages of well drilling and completion. During drilling and completion, well pads can range in size
8     from less than an acre to several acres depending on the scope of the operations ([King, 2012](#);
9     [NYSDEC, 2011](#)).

10    *Well Drilling and Construction*

11    Construction of the production well involves the drilling of the hole (or wellbore), along with the
12    installation and cementing of a series of casing strings to support the wellbore and isolate and
13    protect both the hydrocarbons being produced and any water-bearing zones through which the
14    well passes.[1] In certain settings, some portions of the well can be completed as open holes.[2] Details
15    on these and other well construction activities are presented in Chapter 6 and Appendix D.

16    The operator begins drilling by lowering and rotating the drill string, which consists of the drill bit,
17    drill pipe (see Figure 2-6), and drill collars (heavy pieces of pipe that add weight to the bit). The
18    drill pipe attaches to the drill bit, rotating and advancing the bit; as drilling advances, new sections
19    of pipe are added at the surface, enabling the drilling to proceed deeper ([Hyne, 2012](#)). A drilling
20    fluid is circulated during drilling.[3] The drilling fluid, which may be water-based or oil-based, is
21    pumped down to the drill bit, where it cools and lubricates the drill bit, counterbalances downhole
22    pressures, and lifts the drill cuttings to the surface ([King, 2012](#)).

23    Although all wells are initially drilled vertically, finished well orientations include vertical, deviated,
24    and horizontal. The operator selects the well orientation that will provide access to the targeted
25    zone(s) within a formation and that will align the well with existing fractures and other geologic
26    structures to optimize production. Deviated wells may be "S" shaped or continuously slanted.
27    Horizontal wells have lateral sections oriented approximately 90 degrees from the vertical portion
28    of the well. In wells completed horizontally, the lengths of these laterals can range from 2,000 to
29    5,000 ft (610 to 1,524 m) or more ([Hyne, 2012](#); [Miskimins, 2008](#); [Bosworth et al., 1998](#)).[4]
30    Horizontal wells are instrumental in accessing productive areas of thin and laterally extensive oil-
31    and gas-bearing shales. Although the portion of hydraulically fractured wells that are horizontal is
32    growing, in some areas, such as California, hydraulic fracturing is still primarily conducted in
33    vertical wells ([CCST, 2015](#)).

---

[1] Casing is steel pipe that is lowered into a wellbore. Casing extends from the bottom of the hole to the surface.
[2] An open hole completion is a well completion that has no casing or liner set across the reservoir formation, allowing the produced fluids to flow directly into the wellbore.
[3] Drilling fluid is any of a number of liquid and gaseous fluids and mixtures of fluids and solids (as solid suspensions, mixtures, and emulsions of liquids, gases, and solids) used when drilling boreholes ([Schlumberger, 2014](#)).
[4] A lateral is a horizontal section of a well.

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**Figure 2-6. Pulling drill pipe onto the drilling platform.**

Source: Joshua Doubek, Wikicommons, CC-BY-SA-3.0.

1  The drilling and well construction proceeds with repeated steps (the drill string is lowered, rotated,
2  drilled to a certain depth, pulled out, and then the casing is lowered into the hole, set, and
3  cemented). Successively smaller diameters of casing are used as the hole is drilled deeper (see
4  Figure 2-7). Selection and installation of the casing strings is important for several purposes,
5  including isolating hydrocarbon reservoirs from nearby aquifers, isolating over-pressured zones,
6  and transporting hydrocarbons to the surface (Hyne, 2012). Newly installed casing strings are
7  cemented in place before drilling continues (or before the well is completed in the instance of the
8  production casing). The cement protects the casing from corrosion by formation fluids, stabilizes
9  the casing and the wellbore, and prevents fluid movement along the well between the outside of the
10  casing and wellbore (Renpu, 2011). The well can be cemented continuously from the surface down
11  to the production zone of the well. Partially cemented wells are also possible with, for example,
12  cement from the surface to some distance below the deepest fresh water-bearing formation and
13  perhaps cement across other deeper formations. Chapter 6 and Appendix D contain more details on
14  casing and cement.

**Figure 2-7. Sections of surface casing lined up and being prepared for installation at a well site in Colorado.**

Photo credit: Gregory Oberley (U.S. EPA).

1  When drilling, casing, and cementing are finished, the well can be completed in the production zone
2  in several ways. The production casing may be cemented all the way through the production zone
3  and perforated prior to hydraulic fracturing in the desired locations. Alternatively, operators may
4  use an open hole completion, in which the casing is set just into the production zone and cemented.
5  The remainder of the wellbore within the production zone is left open with no cement (Hyne,
6  2012). Once all aspects of well construction are completed, the operator can remove the drilling rig,
7  install the wellhead, and prepare the well for stimulation by hydraulic fracturing and subsequent
8  production.

### 2.1.1.  Hydraulic Fracturing

9  Hydraulic fracturing is typically a short, intense, repetitive process requiring specialized equipment
10  and (for high volume horizontal wells) large amounts of water, chemicals, and proppant. Machinery
11  and equipment are often brought to the site mounted on trucks and remain that way during use.
12  Tanks, totes, and other storage containers of various sizes holding water and chemicals are also
13  transported and installed on site. Figure 2-8 shows a well pad prepared for hydraulic fracturing
14  with the necessary equipment and structures.

**Figure 2-8. Hydraulic fracturing operation in Troy, PA.**

Site with all equipment on site in preparation for injection. Source: NYSDEC (2011).

### 2.1.1.2.  Injection Process

1  Prior to injection, hydraulic fracturing fluids are mixed using specialized feeding and mixing
2  equipment. The mixing is generally performed mechanically on a truck-mounted blender and is
3  electronically monitored and controlled by the operator in a separate van (see Chapter 5).
4  Numerous hoses and pipes are used to transfer hydraulic fracturing fluid components from storage
5  units to the mixing equipment and ultimately to the wellhead.

6  A wellhead assembly is temporarily installed on the wellhead during the fracture treatment to
7  allow high pressures and volumes of proppant-laden fluid to be injected into the well. Pressures
8  required for fracturing can vary widely depending on depth, formation pressure, and rock type.
9  Fracturing pressures have been reported ranging from 4,000 psi to 12,000 psi (Ciezobka and Salehi,
10  2013; Abou-Sayed et al., 2011; Thompson, 2010). The pressure during fracturing is measured using
11  pressure gauges, which can be installed at the surface and/or downhole (Ross and King, 2007).
12  Figure 2-9 shows two wellheads side-by-side being prepared for fracturing.
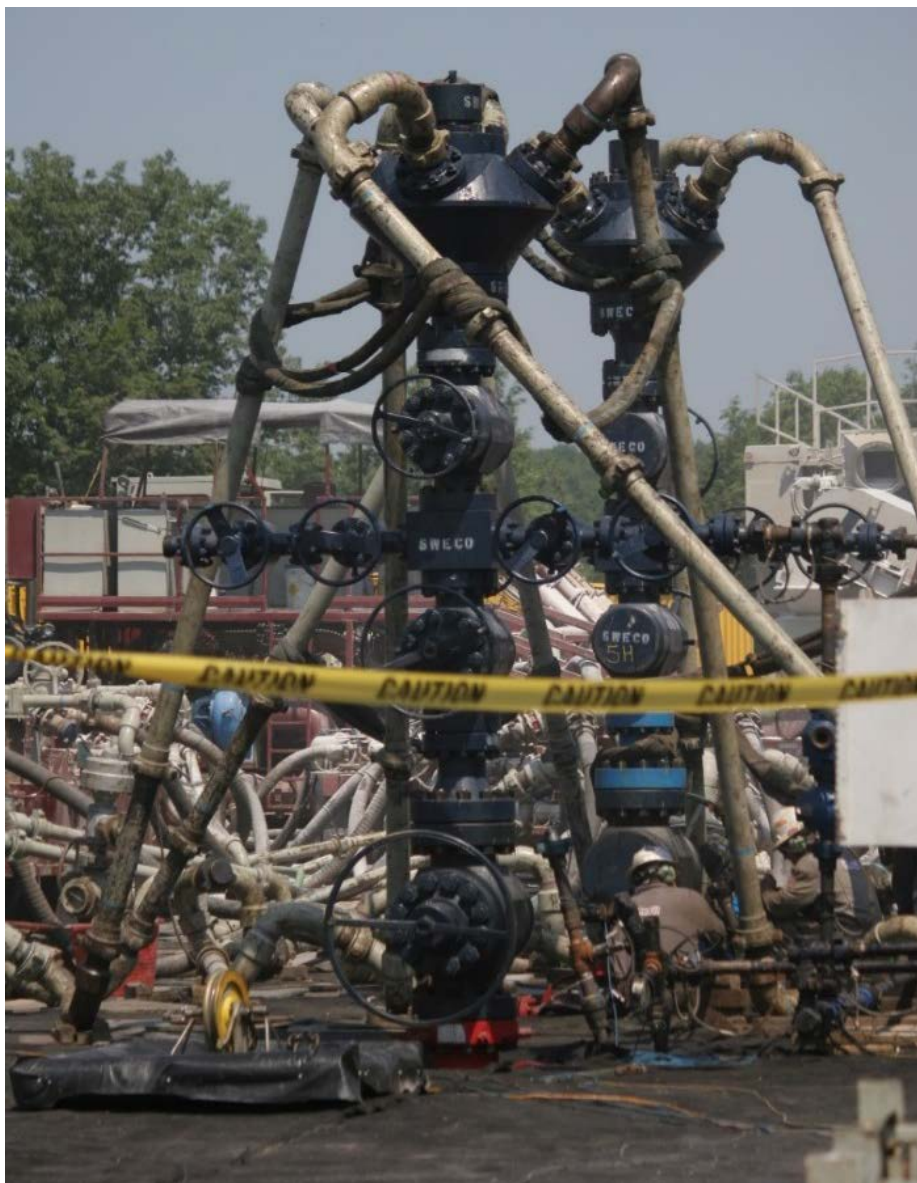
**Figure 2-9. Two wellheads side-by-side being prepared for hydraulic fracturing at a well site in Pennsylvania.**

Photo credit: Mark Seltzer (U.S. EPA).

1  The entire length of the well in the production zone is not fractured all at once; instead, shorter
2  lengths or segments of the well in the production zone are isolated and fractured in "stages" (Lee et
3  al., 2011). Each stage of a fracturing job can consist of phased injection of different fluids consisting
4  of varying components (i.e., chemicals and additives). These different fluids (1) remove excess
5  drilling fluid or cement from the formation (often using acid) (GWPC and ALL Consulting, 2009),
6  (2) initiate fractures ("pad fluid" without proppant), (3) carry the proppant (Hyne, 2012), and
7  (4) flush the wellbore to ensure that all proppant-laden fluids reach the fractures. Each phase

1    requires moving up to millions of gallons of fluids around the site through various hoses and lines,
2    blending the fluids, and injecting them at high pressures down the well.

3    The total number of stages depends on the formation properties and the orientation and length of
4    the well. As technology has improved, the lengths of laterals in horizontal wells and the numbers of
5    stages per well have tended to increase (NETL, 2013; Pearson et al., 2013). The number of stages
6    per well can vary, with several sources suggesting that between 10 and 20 is typical (GNB, 2015;
7    Lowe et al., 2013). The full range reported in the literature is much wider, with one source
8    documenting between 1 and 59 stages per well (Pearson et al., 2013) and others reporting values
9    within this range (NETL, 2013; STO, 2013; Allison et al., 2009). For more details on hydraulic
10   fracturing stages, see Chapter 5, Section 5.2.

11   The induced fractures are designed to achieve the optimum drainage of hydrocarbons from the
12   reservoir formations. Engineers can design fracture systems using modeling software that requires
13   a significant amount of data on formation permeability, porosity, in situ stress, mineralogy, and
14   geologic barrier locations, among other factors (Holditch, 2007). Microseismic monitoring during
15   fracturing can be used to characterize the horizontal and vertical extent of the fractures created and
16   assist with the design of future fracturing jobs (Cipolla et al., 2011). Post-fracture monitoring of
17   pressure or tracers can also help characterize the results of a fracturing job. More details of
18   injection, fracturing, and related monitoring are provided in Chapter 6 and Appendix D.

### 2.1.1.3.  Fracturing Fluids

19   The fracturing fluids injected into the well serve a variety of purposes and require chemical
20   additives to perform properly (see Chapter 5, Section 5.3). Depending on the geologic setting,
21   reservoir geochemistry, production type, proppant size, and other factors, operators typically
22   choose to use one of several common types of fracturing fluid systems (Arthur et al., 2014;
23   Spellman, 2012; Gupta and Valkó, 2007). Water-based fracturing fluids are the most common, but
24   other fluid types can be used such as: foams or emulsions made with nitrogen, carbon dioxide, or
25   hydrocarbons; acid-based fluids; and others (Montgomery, 2013; Saba et al., 2012; Gupta and
26   Hlidek, 2009; Gupta and Valkó, 2007; Halliburton, 1988). The most common water-based fluid
27   systems are slickwater formulations, which are typically used in very low permeability reservoirs,
28   and gelled fracturing fluids, which can be used in reservoirs with higher permeability (Barati and
29   Liang, 2014).[1,2] More details of hydraulic fracturing fluid systems are discussed in Section 5.3.
30   Importantly, chemical usage in the industry is continually changing as processes are tested and
31   refined by companies. Shifts in fluid formulations are driven by economics, technological
32   developments, and concerns about environmental and health impacts.

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[1] Slickwater is a type of fracturing fluid that consists mainly of water with a very low portion of additives like polymers that serve as friction reducers to reduce friction loss when pumping the fracturing fluid downhole (Barati and Liang, 2014).

[2] Gelled fluids are fracturing fluids that are usually water-based with added gels to increase the fluid viscosity to aid in the transport of proppants (Spellman, 2012; Gupta and Valkó, 2007).

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1   The largest constituent of a typical hydraulic fracturing fluid is water (see Figure 2-10). The water
2   sources used for hydraulic fracturing base fluid include ground water, surface water, treated
3   wastewater, and reused flowback or produced water from other wells (URS Corporation, 2011;
4   Blauch, 2010; Kargbo et al., 2010).[1] The water may be brought to the production well site via trucks
5   or piping, or it may be locally sourced (for example, pumped from a local river or obtained from a
6   water well tapping local ground water). Selection of water sources depends upon availability, cost,
7   quality of the water, and the logistics of delivering it to the site. Chapter 4 provides additional
8   details on water acquisition and the amount of water used for hydraulic fracturing.



**Figure 2-10. Water tanks (blue, foreground) lined up for hydraulic fracturing at a well site in central Arkansas.**

Photo credit: Martha Roberts (U.S. EPA).

9    Proppants are, by volume, second to the base fluid in the hydraulic fracturing fluid system. Silicate
10   minerals, most notably quartz sand, are the most commonly used proppants. Increasingly, silicate
11   proppants are being coated with resins that help prevent development and flowback of particles or
12   fragments of particles. Ceramic materials, such as those based on calcined (heated) bauxite or
13   calcined kaolin (mullite) are also used as proppants due to their high strength and resistance to
14   crushing and deformation (Beckwith, 2011).

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[1] Base fluid is the fluid into which additives and proppants are mixed to formulate a hydraulic fracturing fluid.

1    Additives comprise relatively small percentages of hydraulic fracturing fluid systems, generally
2    constituting ≤2.0% of the fluid (GWPC and ALL Consulting, 2009). The EPA analyzed additive data
3    in the EPA FracFocus project database 1.0 and estimated that hydraulic fracturing additives in 2011
4    and 2012 totaled 0.43% of the total amount of fluid injected for hydraulic fracturing (U.S. EPA,
5    2015a). Note that this small percentage can total tens of thousands of gallons of chemical additives
6    for a typical high-volume hydraulic fracturing job (see Chapter 5, Section 5.4 for details on additive
7    volumes). A given additive may consist of a single chemical ingredient, or it may have multiple
8    ingredients. The mix of chemicals used in any particular fracturing job is influenced by the
9    properties of the target formation, the amount and type of proppant that needs to be carried,
10   operator preference, and to some degree, by local or regional availability of chemicals and potential
11   interactions between chemicals (King, 2012). Chapter 5 includes details on the number, types, and
12   estimated quantities of chemicals that can be used in hydraulic fracturing.

## 2.1.2.   Fluid Recovery, Management, and Disposal

13   When the injection pressure is reduced at the end of the fracturing process, the direction of fluid
14   flow reverses, with some of the injected hydraulic fracturing fluid flowing into the well and to the
15   surface along with some naturally-occurring fluids from the production zone (NYSDEC, 2011). The
16   fluid is initially a portion of the injected fluid, which decreases over the first few weeks or months
17   until produced water originating from the fractured oil- or gas-bearing rock formation
18   predominates. This recovery of produced water continues over the life of the well (Barbot et al.,
19   2013). Chapter 7 presents descriptions and discussions of the composition and quantities of fluids
20   recovered at the well, referred to as flowback and produced water.

21   The hydraulic fracturing flowback and produced water (sometimes referred to as hydraulic
22   fracturing wastewater), as well as any other liquid waste from the well pad itself (e.g., rainwater
23   runoff), is typically stored on-site in impoundments (see Figure 2-11) or tanks. This wastewater can
24   be moved offsite via truck or pipelines. The majority of these hydraulic fracturing wastewaters
25   nationally are managed through disposal into deep Class II injection wells regulated under the
26   Underground Injection Control (UIC) program under the Safe Drinking Water Act (see Chapter 8).
27   Other management strategies include treatment followed by discharge to surface water bodies, or
28   reuse for subsequent fracturing operations either with or without treatment (U.S. EPA, 2012f; U.S.
29   GAO, 2012). Decisions regarding wastewater management are driven by factors such as cost
30   (including costs of storage and transportation), availability of facilities for treatment, reuse, or
31   disposal, and regulations (Rassenfoss, 2011). Wastewater management is yet another aspect of
32   fracturing-related oil and gas production that is changing significantly. Chapter 8 contains details of
33   the treatment, reuse and recycling, and disposal of wastewater.

**Figure 2-11. Impoundment on the site of a hydraulic fracturing operation in central Arkansas.**

Photo credit: Caroline E. Ridley (U.S. EPA).

### 2.1.3. Oil and Gas Production

1   After hydraulic fracturing, equipment is removed and partial site reclamation may take place if
2   drilling of additional wells or laterals is not planned (NYSDEC, 2011). Operators may dewater, fill
3   in, and regrade pits that are no longer needed. Parts of the pad may be reseeded, and the well pad
4   may be reduced in size (e.g., from 3 to 5 acres (1 to 2 hectares) during the drilling and fracturing
5   process to 1 to 3 acres (0.4 to 1 hectares) during production) (NYSDEC, 2011).

6   Wells may be shut-in immediately after completion if there is no infrastructure to receive the
7   product or if prices are unfavorable. Prior to bringing a well into production, the operator typically
8   runs a production test to determine the maximum flow rate the well can sustain and to optimize
9   equipment settings (Hyne, 2012; Schlumberger, 2006). Such tests may be repeated throughout the
10  life of the well. During production, monitoring (e.g., mechanical integrity testing, corrosion
11  monitoring), including any compliance with state monitoring requirements, may be conducted to
12  enable operators to be sure that the well is operating as intended.

13  In the case of gas wells, the produced gas typically flows through a flowline to a separator that
14  separates the gas from water or any liquid hydrocarbons (NYSDEC, 2011). The finished gas is sent
15  to a compressor station where it is compressed to pipeline pressure and sent to a pipeline for sale.
16  Production at oil wells proceeds similarly, although oil/water or oil/water/gas separation occurs

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1   most typically on the well pad, no compressor is needed, and the oil can be hauled (by truck or
2   train) or piped from the well pad.

3   During the life of the well it may be necessary to perform workovers to maintain or repair portions
4   or components of the well and replace old equipment. Such workovers involve ceasing production
5   and removing the wellhead, and may include cleaning out sand or deposits from the well, repairing
6   casing, replacing worn well components such as tubing or packers, or installing or replacing lift
7   equipment to pump hydrocarbons to the surface (Hyne, 2012). In some cases, wells may be
8   recompleted after the initial construction, with re-fracturing if production has decreased (Vincent,
9   2011). Recompletion also may include additional perforations in the well at a different interval to
10  produce from a different formation than originally done, lengthening the wellbore, or drilling new
11  laterals from an existing wellbore.

12  As of 2012, Shires and Lev-On (2012) suggested that the rate of re-fracturing in natural gas wells
13  was about 1.6%. Analysis for the EPA's 2012 Oil and Gas Sector New Source Performance Standards
14  indicated a re-fracture rate of 1% for gas wells (U.S. EPA, 2012d). In the EPA's Inventory of U.S.
15  Greenhouse Gas Emissions and Sinks (U.S. EPA, 2015g), the number of gas wells that were re-
16  fractured in a given year as a percent of the total existing population of hydraulically fractured
17  producing gas wells in a given year ranges from 0.3% to 1% across the 1990-2013 period.

### 2.1.1.4. Production Rates and Duration

18  The production life of a well depends on a number of factors, such as the amount of hydrocarbons
19  in place, the reservoir pressure, production rate, and the economics of well operations. It may be as
20  short as three or four years in deep-water, high-permeability formations and as long as 40 to 60
21  years in onshore tight gas reservoirs (Ross and King, 2007). In hydraulically fractured wells in
22  unconventional reservoirs, production is often characterized by a rapid drop followed by a slower
23  decline compared to conventional hydrocarbon production wells (Patzek et al., 2013). However,
24  most modern, high-volume fractured wells are less than a decade old. Consequently, there is a
25  limited historical basis to determine the full extent of the production decline (Patzek et al., 2013)
26  and to ultimately determine how much they will produce.

### 2.1.4. Site and Well Closure

27  Once a well reaches the end of its useful life, it is plugged, and the well site is closed. If a wellbore is
28  not properly plugged, fluids from higher pressure zones may eventually migrate through the
29  wellbore to the surface or to other zones such as fresh water aquifers (NPC, 2011b). Plugging is
30  usually performed according to state regulations governing the locations and materials for plugs
31  (Calvert and Smith, 1994). Operators typically use cement plugs placed across fresh water
32  formations and oil or gas formations (NPC, 2011b). Some surface structures can be left in place, and
33  the local topography and land cover are restored to predevelopment conditions to the extent
34  possible, per state regulations. The wellhead and any surface equipment are removed.
35  Impoundments are dewatered, filled in, and graded. The well casing is typically cut off below the
36  surface and a steel plate or cap is emplaced to seal the top of the casing and wellbore (API, 2010a),
37  although there may also be an aboveground marker used in some locations. Some states require
38  notification of the landowner or a government agency of the location of the well.

## 2.2.    How Widespread is Hydraulic Fracturing?

1    Hydraulic fracturing activity in the United States and worldwide is substantial. One industry
2    cumulative estimate stated that by the time of writing in 2010, close to 2.5 million fracture
3    treatments had been performed globally ([Montgomery and Smith, 2010](#)). In 2002, the Interstate Oil
4    and Gas Compact Commission (IOGCC) stated that close to 1 million wells had been hydraulically
5    fractured in the United States since the 1940s ([IOGCC, 2002](#)). A recent U.S. Geological Survey
6    (USGS) publication analyzed 1 million hydraulically fractured wells and 1.8 million hydraulic
7    fracturing treatment records from the United States from 1947 to 2010 ([USGS, 2015](#)). Although
8    some form of hydraulic fracturing has been used for more than 60 years, the technological
9    advancements that combined hydraulic fracturing and directional drilling in the early 2000s
10    resulted in the new era of modern hydraulic fracturing, which uses higher volumes of fracturing
11    fluids than were typically used in prior decades. Modern hydraulic fracturing is typically associated
12    with horizontal wells producing from unconventional shale reservoirs, but hydraulic fracturing
13    continues to be done in vertical wells in conventional reservoirs also. This ongoing mix of
14    traditional and modern hydraulic fracturing activities makes estimates of the total number of
15    hydraulic fracturing wells challenging.

16    The following series of images illustrates hydraulic fracturing activities and the scale of those
17    activities in the United States. Figure 2-12 (taken in Springville Township, in northeastern
18    Pennsylvania) and Figure 2-13 (taken near Williston, in northwestern North Dakota) show
19    individual well pads in the context of the local landscape. Landsat images in Figure 2-14 and Figure
20    2-15 provide satellite views of areas in northwest Louisiana and southeast Wyoming, respectively,
21    where hydraulic fracturing activities currently occur as identified by the well pads in the images.
22    These images serve to illustrate activity at a wider scale, though they are not representative of all
23    hydraulic fracturing activities in the eastern or western United States. The light red circles around
24    some of the well pads identify them as hydraulic fracturing wells that were reported by well
25    operators to the FracFocus registry (as summarized in the EPA FracFocus project database 1.0)
26    ([U.S. EPA, 2015b](#)). (The FracFocus well locations reflect information in the EPA FracFocus project
27    database for well operations reporting hydraulic fracturing activities between January 2011 and
28    February 2013. The Landsat images are from a later period, July and August of 2014, so additional
29    well pads in the images now may be represented in the FracFocus registry.)

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**Figure 2-12. Aerial photograph of a well pad and service road in Springville Township, Pennsylvania.**

Image © J Henry Fair / Flights provided by LightHawk.



**Figure 2-13. Aerial photograph of hydraulic fracturing activities near Williston, North Dakota.**

Image © J Henry Fair / Flights provided by LightHawk.
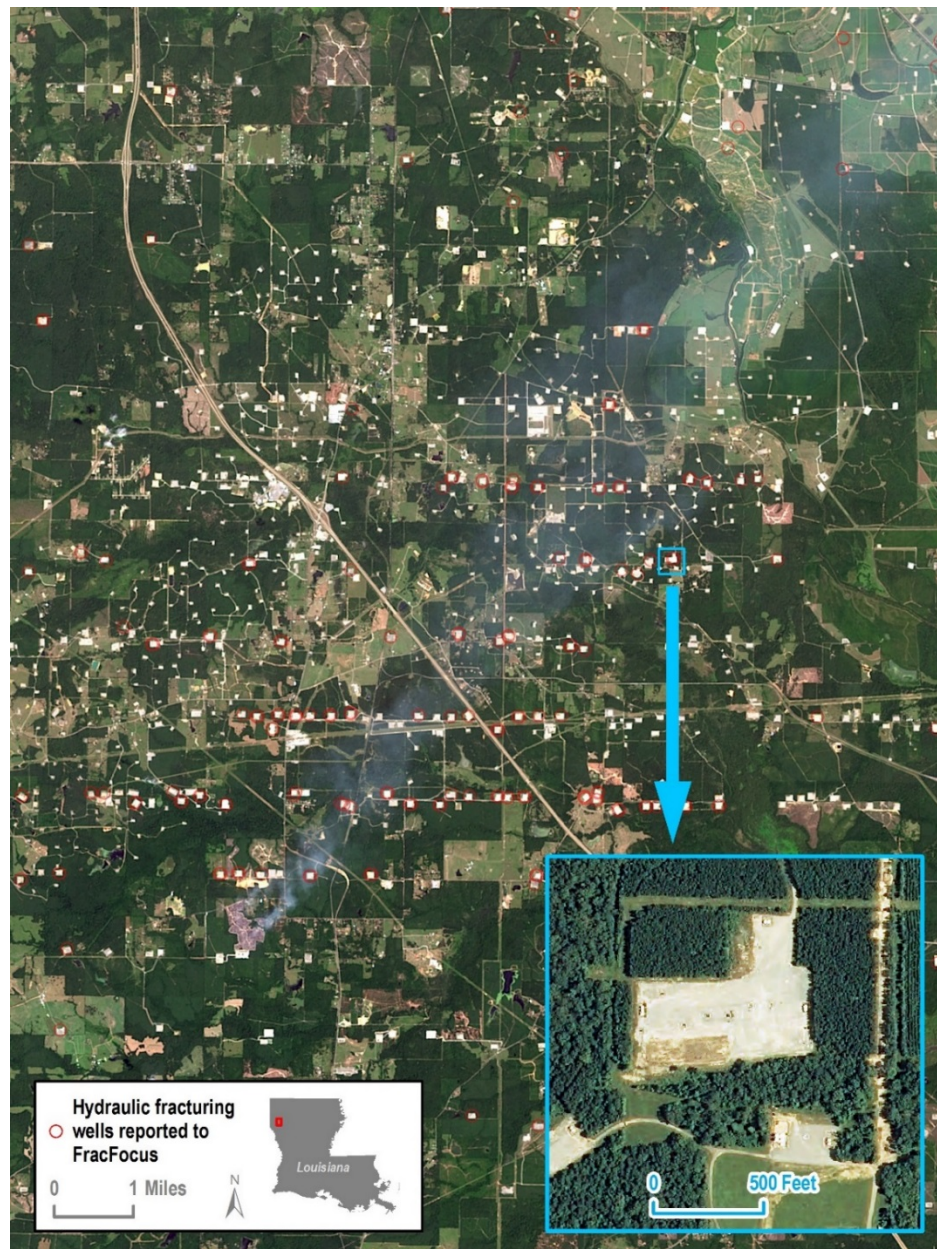
**Figure 2-14. Landsat photo showing hydraulic fracturing well sites near Frierson, Louisiana.**

Source: Imagery from USGS Earth Resources Observation and Science, Landsat 8 Operational Land Imager (scene LC80250382014232LGN00) captured August 20, 2014 and accessed on May 1, 2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/).

Inset imagery from USDA National Agriculture Imagery Program (entity M 3209351_NE 15_1_20130703_20131107) captured July 3, 2013 and accessed May 1, 2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/).

FracFocus well locations are from the EPA FracFocus project database 1.0 (U.S. EPA, 2015b).
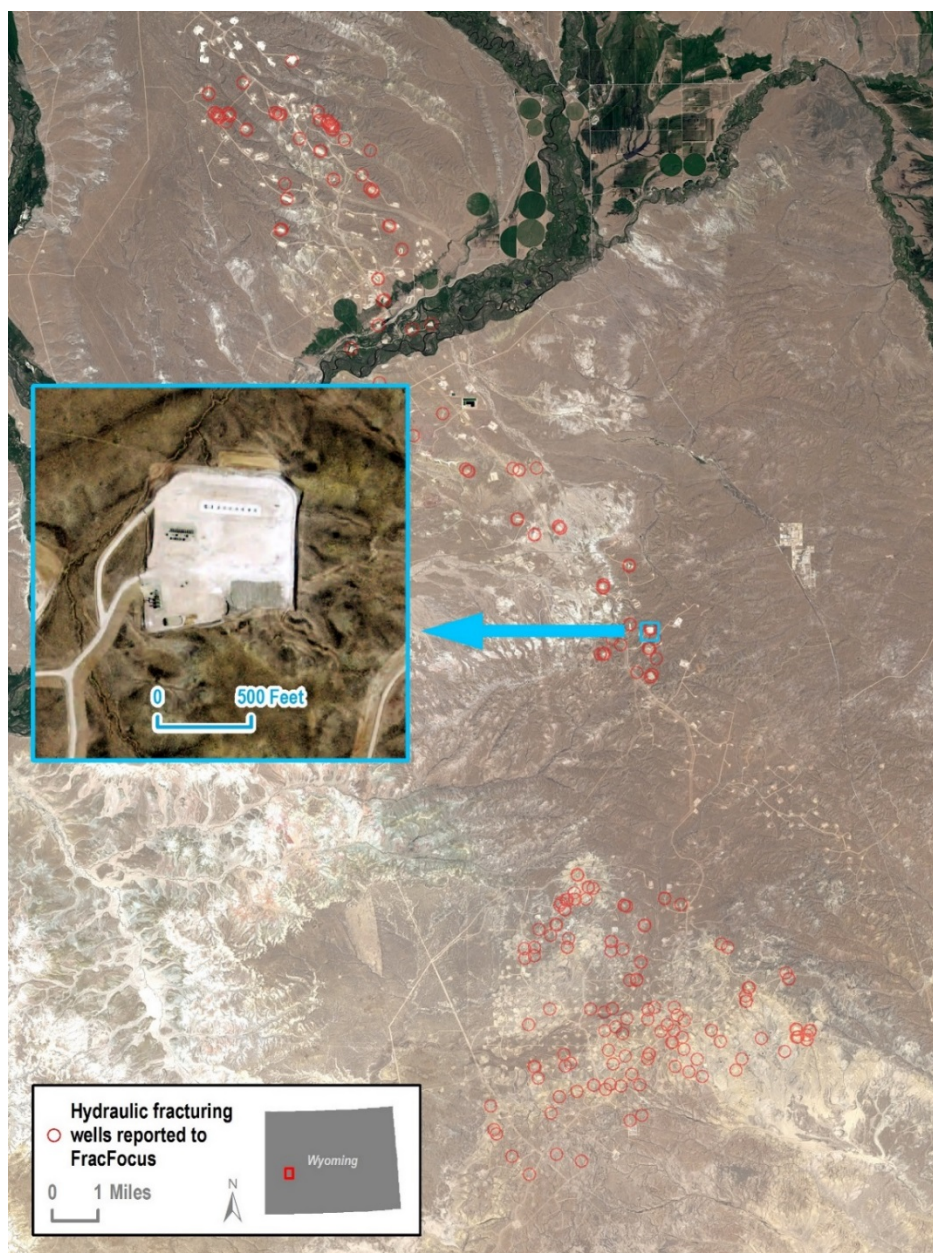
**Figure 2-15. Landsat photo showing hydraulic fracturing well sites near Pinedale, Wyoming.**

Source: Imagery from USGS Earth Resources Observation and Science, Landsat 8 Operational Land Imager (scene LC80370302014188LGN00) captured July 7, 2014 and accessed May 1, 2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/).

Inset imagery from USDA National Agriculture Imagery Program (entity M 4210927_NW 12_1_20120623_20121004) captured June 23, 2012 and accessed May 1, 2015 from USGS's EarthExplorer (http://earthexplorer.usgs.gov/).

FracFocus well locations are from the EPA FracFocus project database 1.0 (U.S. EPA, 2015b).

1    The maps in Figure 2-16 show recent changes nationally in the geography of oil and gas production
2    through the increased use of horizontal drilling, which occurs together with hydraulic fracturing.

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1    Some traditional oil- and gas-producing parts of the country, such as Texas, have seen an expansion
2    of historically strong production activity as a result of the deployment of horizontal drilling and
3    modern hydraulic fracturing. Pennsylvania, a century ago one of the leading oil- and gas-producing
4    states, has seen a resurgence in oil and gas activity. Other states currently experiencing a steep
5    increase in production activity, such as North Dakota, Arkansas, and Montana, have historically
6    produced less oil and gas and are therefore undergoing new development.
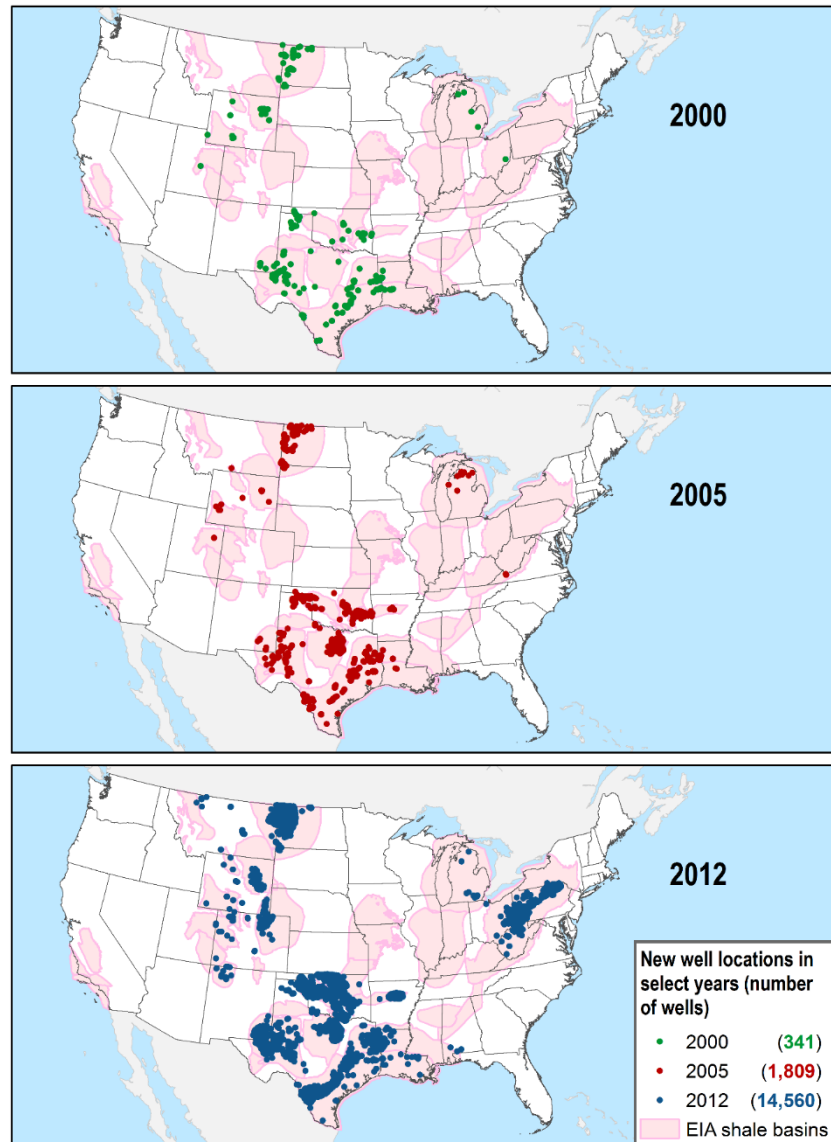
**Figure 2-16. Location of horizontal wells that began producing oil or natural gas in 2000, 2005, and 2012, based on data from DrillingInfo (2014a).**

### 2.2.1. Number of Wells Fractured per Year

1  We estimate that from roughly 2011 to 2014, approximately 25,000 to 30,000 new oil and gas wells
2  were hydraulically fractured each year. Additional, pre-existing wells (wells more than one year old
3  that may or may not have been hydraulically fractured in the past) were also likely fractured each
4  year. Since the early 2000s, the percentage of all hydraulically fractured wells that are either
5  horizontal or deviated has steadily grown. Our estimates are based on data detailed below from
6  several public and private sector organizations that track drilling and various aspects of hydraulic
7  fracturing activity. There is no complete database or registry of wells that are hydraulically
8  fractured in the United States. Another source of uncertainty is the rate at which relatively new
9  hydraulic fracturing wells are re-fractured or the rate at which operators use older, existing wells
10  for hydraulic fracturing. Future trends in the number of wells hydraulically fractured per year will
11  be affected by the cost of well operation and the price of oil and gas. Scenarios of increasing, flat,
12  and decreasing hydraulic fracturing activity all appear to be possible (Weijermars, 2014).

13  The number of wells reported to the FracFocus registry provides a low estimate of the number of
14  hydraulically fractured wells.[1] As of early April 2015, the FracFocus registry reported receiving
15  information on a cumulative total of approximately 95,000 fracturing jobs, or roughly 22,400 per
16  year over the 51-month period from January 2011 through March 2015 (GWPC, 2015). In a more
17  detailed review of FracFocus data from 2011 and 2012, the EPA found there were approximately
18  14,000 and 22,500 fracturing jobs reported to the FracFocus website in those years, respectively,
19  across 20 states (U.S. EPA, 2015a). These 2011 and 2012 numbers are likely underestimates of
20  wells hydraulically fractured annually, in part because FracFocus reporting was voluntary for most
21  states for at least a portion of 2011 to 2012 (though the increase from 2011 to 2012 in part reflects
22  more states requiring reporting to the registry). Hydraulic fracturing practices may alternately (or
23  in addition to FracFocus) be tracked by states. Compared to state records of hydraulic fracturing
24  from North Dakota, Pennsylvania, and West Virginia in 2011 and 2012, we found that the count of
25  wells based on records submitted to FracFocus was an underestimate of the number of fracturing
26  jobs in those states by an average of approximately 30% (see Text Box 4-1).

27  An additional estimate of the number of hydraulically fractured wells can be obtained from
28  DrillingInfo, a commercial database compiling data from individual state oil and gas agencies
29  (DrillingInfo, 2014a). The data indicate an increase in the number of new hydraulically fractured
30  wells drilled each year, from approximately 12,800 in 2000 to slightly more than 21,600 in 2005, to
31  nearly 23,000 in 2012. The number of new horizontal wells (which are likely all hydraulically
32  fractured) show a significant increase, from 344 (about 1% of all new production wells) in 2000, to
33  1,810 in 2005, to 14,560 (nearly 41% of all new production wells) in 2012 (see Figure 2-16).

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[1] The FracFocus registry was developed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission. Oil and gas well operators can use the FracFocus registry to disclose information about hydraulic fracturing well locations, and water and chemical use during hydraulic fracturing operations. Submission of information to FracFocus was initially voluntary (starting in January 2011), but now about half of the 20 states represented in FracFocus have enacted reporting requirements for well operators that either mandate reporting to FracFocus or allow it as one reporting option. FracFocus data are discussed in more detail in Chapter 4 (regarding water volumes) and Chapter 5 (regarding chemical use). For more information see www.fracfocus.org and U.S. EPA (2015a).

1   Because DrillingInfo data do not directly report whether a well has been hydraulically fractured, we
2   relied on properties of the well and the oil or gas producing formation to infer which wells were
3   hydraulically fractured and when. First, we assumed that *all* horizontal wells were hydraulically
4   fractured in the year they started producing. Second, we assumed that all wells within a shale,
5   coalbed, or low-permeability formation, regardless of well orientation, were hydraulically fractured
6   in the year they started producing.[1]

7   We used well-specific data provided by oil and gas well operators to the EPA to supplement our
8   estimates of hydraulic fracturing using DrillingInfo data (U.S. EPA, 2015o). Matching wells in each
9   dataset using API well numbers, we found that 80% of 171 newly drilled wells known to be
10  fractured in 2009 and 2010 according to their well files were correctly identified as fractured using
11  well and formation properties in DrillingInfo.[2] We did not correctly identify all of the vertical or
12  deviated wells that were known to be fractured. (We were unable to identify wells for which
13  hydraulic fracturing was inferred using the properties in DrillingInfo but were not fractured.) This
14  comparison suggests that the estimates of hydraulically fractured wells from DrillingInfo are likely
15  underestimates.

16  Another source of estimates is from a U.S. Geological Survey publication that reviewed data from
17  the commercial IHS database of U.S. oil and gas production and well data (USGS, 2015). The study
18  period was from 1947 through 2010. The authors estimated a total of approximately 277,000
19  hydraulically fractured wells between 2000 and 2010 (compared to close to 212,000 during the
20  same time period estimated based on DrillingInfo data). This is roughly 25,000 wells per year over
21  that time period. Approximately three-quarters of these wells were vertical. Reflecting advances in
22  directional drilling technology over the decade ending in 2010, the percentage of total wells
23  fractured that were horizontal or deviated wells grew from less than 10% to over 60%.

24  Well counts tracked by Baker Hughes provide another estimate of new wells fractured annually.
25  Since 2012, this oilfield service company has published a quarterly count of new wells spudded; it
26  includes only new inland U.S. wells "identified to be significant consumers of oilfield services and
27  supplies."[3] A reported total of 36,824 oil and gas wells were spudded in the United States in 2012,
28  with new wells per quarter fluctuating between about 8,500 and 9,500 (Baker Hughes, 2014b).
29  While 100% of new wells are probably not hydraulically fractured (see below for estimates of
30  hydraulic fracturing rates in new wells), a count of new wells also does not include hydraulic
31  fracturing taking place in older, existing wells.

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[1] The assignment of formation type (shale, coalbed, low-permeability, or conventional) for each well was based on a crosswalk of information on basin/play provided in DrillingInfo (2014a) with expert knowledge of those basins/plays at EIA (2012a). If formation type could not be determined, it was considered conventional by default. This is similar methodology to that used by the EPA for its greenhouse gas inventory (U.S. EPA, 2013c).

[2] An API well number is a unique identifying number given to each oil and gas well drilled in the United States. The system was developed by the American Petroleum Institute.

[3] To spud a well is to start the well drilling process by removing rock, dirt, and other sedimentary material with the drill bit.

1  Data collected under the EPA's Greenhouse Gas Reporting Program (GHGRP) provide information
2  on completions and workovers with hydraulic fracturing (i.e., re-fracturing) of gas wells. Data
3  reported to GHGRP for years 2011 to 2013 suggest that 9-14% of the gas wells reported to be
4  hydraulically fractured in each year were pre-existing wells undergoing re-fracturing ([U.S. EPA,
5  2014e](#)).[1] The GHGRP requirements do not include reporting of re-fracturing in oil wells, and other
6  data sources for information specifically on re-fracturing of existing oil wells compared to initial
7  fracturing of oil wells were not identified. For comparison, an EPA survey of an estimated 23,200 oil
8  and gas production wells that were hydraulically fractured by nine oil and gas service companies in
9  2009 and 2010 suggests that 42% of the wells were pre-existing (i.e., more than one year old) when
10  they were hydraulically fractured ([U.S. EPA, 2015o](#)). Differences in data (including data from
11  different years and data from gas wells only (GHGRP) versus oil and gas wells, for instance),
12  definitions, and assumptions used to estimate the percentage of pre-existing wells hydraulically
13  fractured in a year could account for the different results.

14  In summary, determination of the national scope of hydraulic fracturing activities in the United
15  States is complicated by a lack of a centralized source of information and the fact that well and
16  drilling databases each track different information. There is also uncertainty about whether
17  information sources are representative of the nation, whether they include data for all production
18  types, whether they represent only modern (high volume) hydraulic fracturing, and whether they
19  include activities in both conventional and unconventional reservoirs. Taking these limitations into
20  account, however, it is reasonable to assume that between approximately 25,000 and 30,000 new
21  wells (and, likely, additional pre-existing wells) were hydraulically fractured each year in the
22  United States from about 2011 to 2014.

### 2.2.2. Hydraulic Fracturing Rates

23  Estimates of hydraulic fracturing rates, or the proportion of all oil and gas production wells that are
24  associated with hydraulic fracturing, also indicate widespread use of the practice. Based on an
25  assessment described above of data from [DrillingInfo (2014a)](#), hydraulic fracturing rates have
26  increased over time. From 2005 to 2012, rates of hydraulic fracturing increased from 57% to 64%
27  of all new production wells (including oil wells, gas wells, and wells producing both oil and gas).

28  In 2009, industry consultants stated that hydraulic fracturing was used on nearly 79% of all wells
29  and more than 95% of "unconventional" wells ([IHS, 2009](#)). A 2010 article in an industry publication
30  noted "some believe that approximately 60% of all wells drilled today are fractured" ([Montgomery
31  and Smith, 2010](#)). Of 11 important oil and gas producing states that responded to an IOGCC survey
32  (Arkansas, Colorado, Louisiana, New Mexico, North Dakota, Ohio, Oklahoma, Pennsylvania, Texas,
33  Utah, and West Virginia), ten estimated that 78% to 99% of new oil and gas wells in their states
34  were hydraulically fractured in 2012; Louisiana was the one exception, reporting a fracturing rate
35  of 3.9% in 2012 ([IOGCC, 2015](#)). Although estimates of fracturing rates are variable, largely ranging
36  from near 60% to over 90% (as described above), they are often higher for gas wells than they are
37  for oil wells. A 2010 to 2011 industry survey of 20 companies involved in natural gas production

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[1] The GHGRP reporting category that covers re-fracturing is "workovers with hydraulic fracturing."

1  found that 94% of the wells that they operated were fractured; among those, roughly half were
2  vertical and half were horizontal (Shires and Lev-On, 2012).

## 2.3.    Trends and Outlook for the Future

3  Fossil fuels are the largest source of all energy generated in the United States. They currently
4  comprise approximately 80% of the energy produced (EIA, 2014f). However, the mix of fossil fuels
5  has shifted in recent years. Coal, the leading fossil fuel produced by the U.S. since the 1980s, has
6  experienced a significant decrease in production. In 2007, coal accounted for approximately 33% of
7  U.S. energy production, and by 2013 it decreased to approximately 24% (EIA, 2014f). On the other
8  hand, natural gas production has risen to unprecedented levels, and oil production has resurged to
9  levels not seen since the 1980s (see Figure 2-17). Oil went from accounting for 15% of U.S. energy
10  production to 19% between 2007 and 2013, and natural gas (both dry and liquid) went from 31%
11  to 35% (EIA, 2014f).

12  Below, we discuss recent and projected shifts in oil and natural gas production that can primarily
13  be attributed to hydraulic fracturing and directional drilling technologies.

### 2.3.1.   Natural Gas (Including Coalbed Methane)

14  Natural gas production in the United States peaked in the early 1970s, reached those levels again in
15  the mid-1990s, and between the mid- to late-2000s has increased to even higher levels (see Figure
16  2-17). The recent increase in total gas production has been driven almost entirely by shale gas (see
17  Figure 2-18).

18  As natural gas prices fell between 2008 and 2012 (EIA, 2014e), drilling of new natural gas wells
19  declined markedly (EIA, 2014g) (see Figure 2-19). Nevertheless, natural gas production is expected
20  to increase over the coming decades (see Figure 2-18). EIA (2013b) predicts that shale gas
21  production will more than double between 2011 and 2040 and that the portion of total natural gas
22  production represented by shale gas will increase from one-third to one-half. The EIA projects
23  steady growth in the development of tight gas as well (about a 25% increase in production over the
24  30-year period) and delayed growth in the development of coalbed methane resources, for which
25  production is not expected to increase again until sufficiently high natural gas prices are realized
26  around 2035. Overall, the EIA projects that the share of U.S. natural gas production from shales,
27  tight formations, and coalbeds will increase from 65% in 2011 to nearly 80% in 2040.
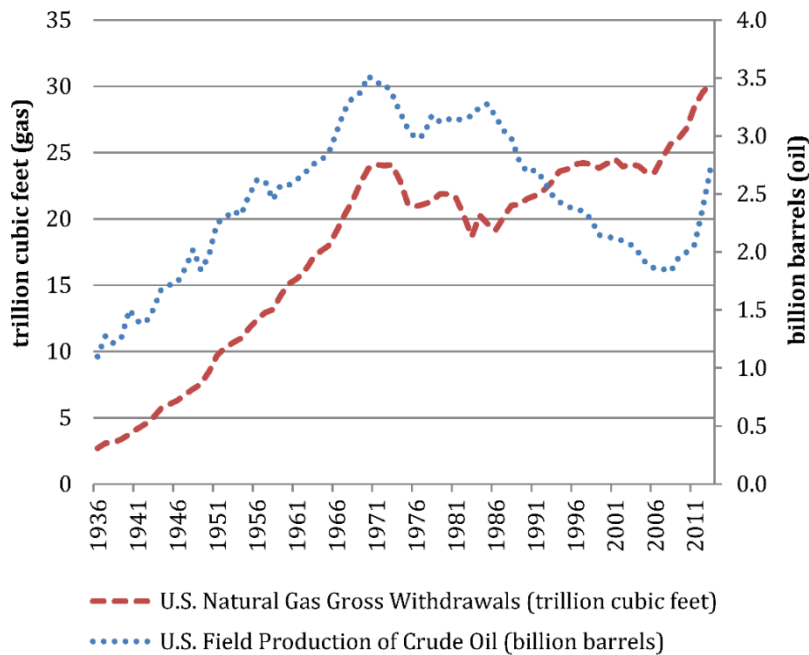
**Figure 2-17. Trends in U.S. oil and gas production.**

Source: EIA (2013d) and EIA (2014d).
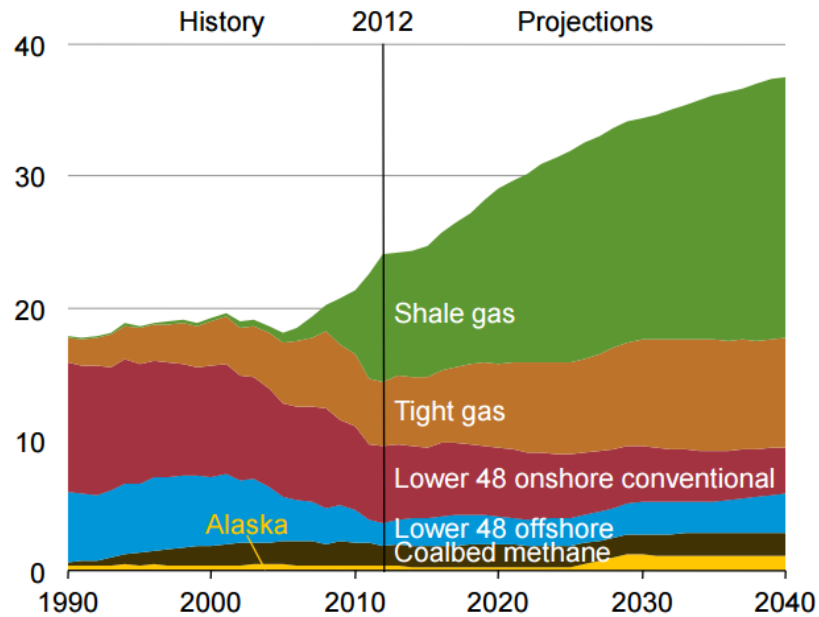


**Figure 2-18. Historic and projected natural gas production by source (trillion cubic feet).**

Source: EIA (2014a).
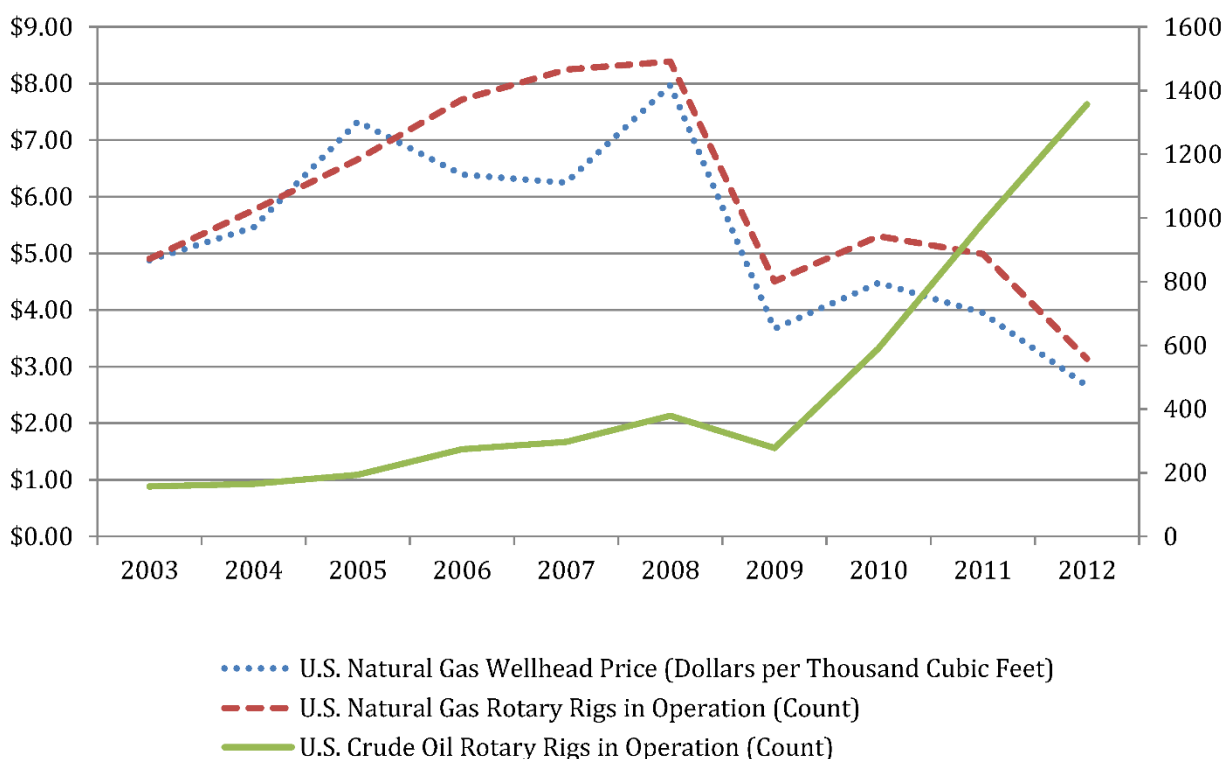
**Figure 2-19. Natural gas prices and oil and gas drilling activity, 2008–2012.**

Source: EIA (2014e), EIA (2014g), and EIA (2013b).

1    Shale gas production varies by play (see Figure 2-20a). Until 2010, the Texas Barnett Shale was the
2    play with the most production. Although production from the Barnett Shale is still significant,
3    production has increased sharply in other plays. By 2012, production from the Haynesville play (on
4    the Louisiana/Texas border) surpassed that in the Barnett play, and by 2013 the Marcellus Shale
5    (in the Appalachian Basin underlying Pennsylvania, West Virginia, and other states) was the play
6    with the most production. Because technically recoverable resources are an order of magnitude
7    higher in the Marcellus than in any other U.S. shale gas play, it is likely that the Marcellus Shale will
8    be very active in shale gas production for the foreseeable future (EIA, 2011a).[1]

9    In the 1970s, most tight gas production in the United States was in the San Juan Basin centered in
10   New Mexico. As modern hydraulic fracturing came into common usage in the mid-2000s, the lead in
11   tight gas production shifted to Texas (especially East Texas) and the Rocky Mountain states (Vidas
12   and Hugman, 2008).

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[1] Technically recoverable resources represent the volumes of oil and natural gas that could be produced with current
technology, regardless of oil and natural gas prices and production costs (EIA, 2013c).
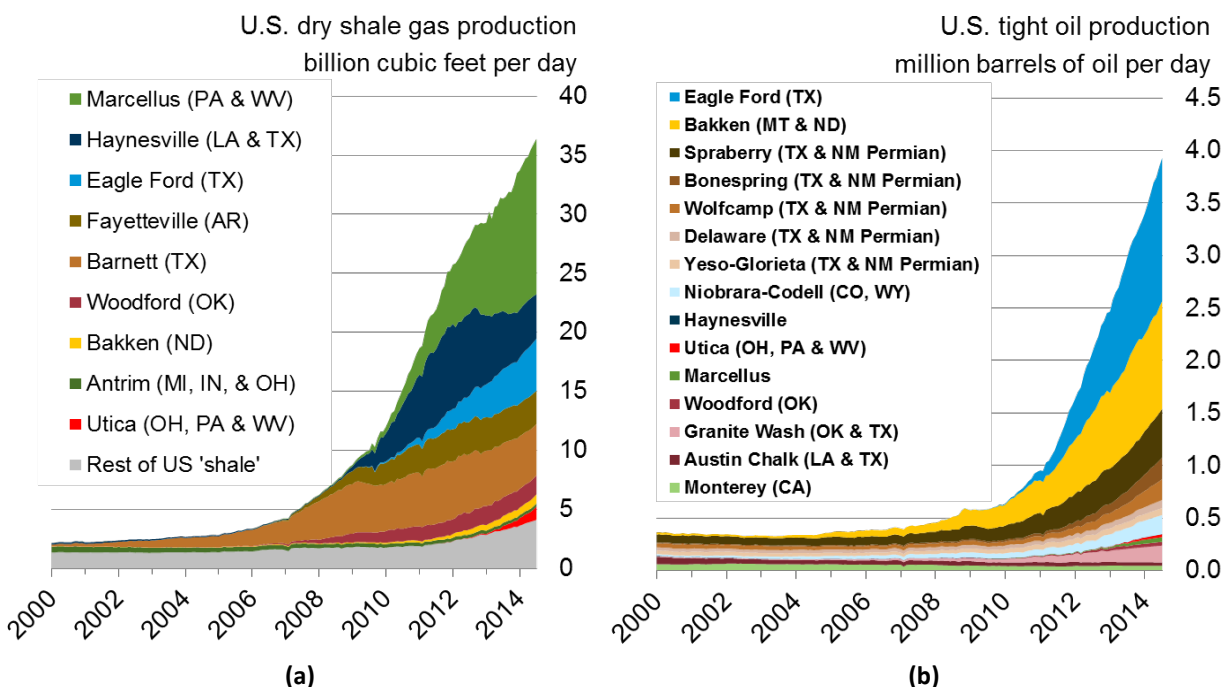
**Figure 2-20. (a) Production from U.S. shale gas plays, 2000–2014, in billion cubic feet per day; (b) Production from U.S. tight oil plays, 2000-2014.**

Tight oil includes oil from shale and other tight formations, plus lease condensate from natural gas production. Source: EIA (2012c).

1  Modern coalbed methane production techniques were pioneered in the Black Warrior Basin in
2  Alabama and in the San Juan Basin (Vidas and Hugman, 2008). With the use of hydraulic fracturing,
3  most coalbed methane production in the United States now comes from the San Juan Basin and
4  from Rocky Mountain Basins (e.g., the Uinta-Piceance Basin in Colorado and Utah and the Powder
5  River Basin centered in Wyoming) (Vidas and Hugman, 2008).

### 2.3.2.  Oil

6  The EIA data indicate that as drilling activity for natural gas declined between 2008 and 2012,
7  drilling for oil increased by a similar order of magnitude (see Figure 2-19). Figure 2-21 shows past
8  and projected future trends in U.S. oil production and importation (EIA, 2013a). Note that this
9  graph shows production and importation in millions of barrels (bbl) *per day*. The current surge in
10 tight oil production is expected to continue until the latter part of the current decade and then
11 taper, while conventional oil production is projected to remain fairly level. However, downward
12 trends in the price of oil since mid-2014 are not reflected in these projections.

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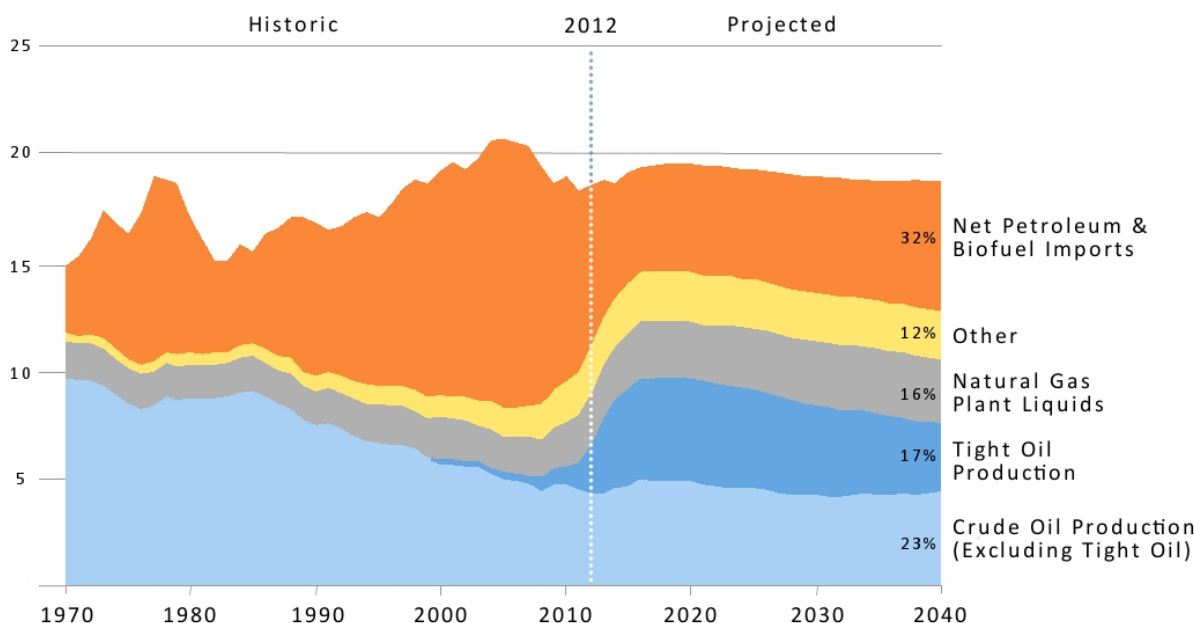
**Figure 2-21. U.S. petroleum and other liquid fuels supply by source, past and projected future trends (million barrels per day).**

Source: EIA (2013a).

1  Like shale gas production, tight oil production varies by play (Figure 2-20b). The Bakken Shale play,
2  centered in western North Dakota, is important for shale oil production with production increasing
3  from 123 million bbl (20 billion L) in 2011 to 213 million bbl (34 billion L) in 2012. Proved reserves
4  in the Bakken have increased from almost 2 billion to over 3 billion bbl (316 billion L to 503 billion
5  L). The Eagle Ford play in Texas is another major area of shale oil activity, with production
6  increasing from 71 million bbl (11 billion L) in 2011 to 210 million bbl (33 billion L) in 2012, and
7  proved reserves increasing from 1.25 billion to 3.4 billion bbl (199 billion to 536 billion L) (EIA,
8  2014b). Oil production from the Eagle Ford surpassed that from the Bakken in 2013 (EIA, 2014h).
9  Among other shale oil plays that might become important in future domestic U.S. oil production, the
10 Niobrara (centered in Colorado) and Austin Chalk (in Texas, Louisiana, and Mississippi) are
11 believed to have quantities of recoverable resources on the same order of magnitude as the Bakken
12 and Eagle Ford plays (EIA, 2012b).

## 2.4.  Conclusion

13 Since about 2005, the combination of hydraulic fracturing and horizontal drilling pioneered in the
14 Barnett Shale have become widespread in the oil and gas industry. Hydraulic fracturing is now a
15 standard industry practice and has significantly contributed to a surge in U.S. production of both oil
16 and gas. Modern hydraulic fracturing has resulted in additional types of geological formations being
17 tapped, and sometimes these formations are located in regions of the country new to intensive oil

1  and gas exploration and production. In other areas, the improved techniques have made possible a
2  resurgence of production.

3  An estimated 25,000 to 30,000 new wells drilled in the United States were hydraulically fractured
4  as a production-enhancing technique in each year from 2011 to 2014. Additional pre-existing wells
5  were also fractured. Since the early 2000s, the percentage of all hydraulically fractured wells that
6  are either horizontal or deviated has steadily grown. Reserves of oil and gas that are now accessible
7  with modern hydraulic fracturing are considerable, and if technical improvements outpace
8  depletion of oil and gas resources, the quantity of resources that are deemed economically and
9  technically recoverable may continue to grow. Given current trends, it appears likely that hydraulic
10  fracturing will continue to play an important role in the oil and gas industry, and the United States'
11  energy portfolio, in the decades ahead.

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# Chapter 3

# Drinking Water Resources in the United States

# 3. Drinking Water Resources in the United States

1   Consideration of how and where hydraulic fracturing activities potentially impact drinking water
2   resources requires an understanding of both the activities and the potentially impacted resources.
3   In Chapter 2, we provided background on hydraulic fracturing and in this chapter, we provide an
4   overview of drinking water resources in the United States. We describe the use of these resources,
5   including patterns in current use and trends for future use (Section 3.1). We then characterize the
6   spatial distribution of hydraulically fractured wells and current surface and ground water supplies
7   throughout the United States (Section 3.2) to evaluate where potential impacts of hydraulic
8   fracturing on drinking water resources may occur.

## 3.1. Current and Future Drinking Water Resources

9   In this assessment, drinking water resources are defined broadly as any body of ground water or
10  surface water that now serves, or in the future could serve, as a source of drinking water for public
11  or private use. Drinking water resources provide not only water that individuals actually drink but
12  also water used for many additional purposes such as cooking and bathing. Our definition of
13  drinking water resources includes both fresh and non-fresh bodies of water.

14  The average American uses about 90 gal (341 L) of drinking water per day for indoor and outdoor
15  purposes (e.g., drinking, food preparation, washing clothes and dishes, flushing toilets, and
16  watering lawns or gardens (Maupin et al., 2014; AWWA, 1999). Drinking water is supplied to
17  households by either public water systems (PWSs) or private water systems (private ground water
18  wells and surface water intakes).[1] In 2011, approximately 270 million people (86% of the
19  population) in the United States relied on water supplied to their homes by one of the more than
20  51,000 community water systems (Maupin et al., 2014; U.S. EPA, 2013b).[2] These systems provided
21  households with nearly 24 billion gal (91 billion L) of water per day (Maupin et al., 2014).[3] In areas
22  without service by PWSs, approximately 43 million people (14% of the population) relied on
23  private sources for drinking water, and private water systems account for about 3.6 billion gal (14
24  billion L) of daily water withdrawals (Maupin et al., 2014).

25  Drinking water resources can be surface waters such as rivers, streams, lakes, or reservoirs, as well
26  as ground water aquifers. In 2011, approximately 70% of the population receiving drinking water
27  from PWSs relied on surface water, and 30% relied on ground water (U.S. EPA, 2013b). However,

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[1] Public water systems (PWSs) provide water for human consumption from surface or ground water through pipes or other infrastructure to at least 15 service connections or serve an average of at least 25 people for at least 60 days a year (U.S. EPA, 2012e). Private (non-public) water systems serve fewer than 15 connections and fewer than 25 individuals (U.S. EPA, 1991).

[2] The EPA categorizes public water systems as either community water systems, which supply water to the same population year-round, or non-community water systems, which supply water to at least 25 of the same people at least six months per year, but not year-round. Approximately 101,000 non-community water systems provide water to non-residential facilities (e.g., schools, small businesses, churches, and campgrounds (U.S. EPA, 2013b).

[3] The U.S. Geological Survey (USGS) compiles data in cooperation with local, state, and federal environmental agencies to produce water-use information aggregated at the county, state, and national levels. Every five years, data at the county level are compiled into a national water use census and state-level data are published. The most recent USGS water use report was released in 2014, and contains water use estimates from 2010 (Maupin et al., 2014; USGS, 2014b).

1  the relative importance of surface and ground water sources for supplying drinking water varies
2  geographically (see Figure 3-1). Most larger PWSs rely on surface water and are located in urban
3  areas (U.S. EPA, 2011b), whereas most smaller PWSs rely on ground water and are located in rural
4  areas (U.S. EPA, 2014j, 2013b). In fact, more than 95% of households in rural areas obtain their
5  drinking water from ground water aquifers (U.S. EPA, 2011b).
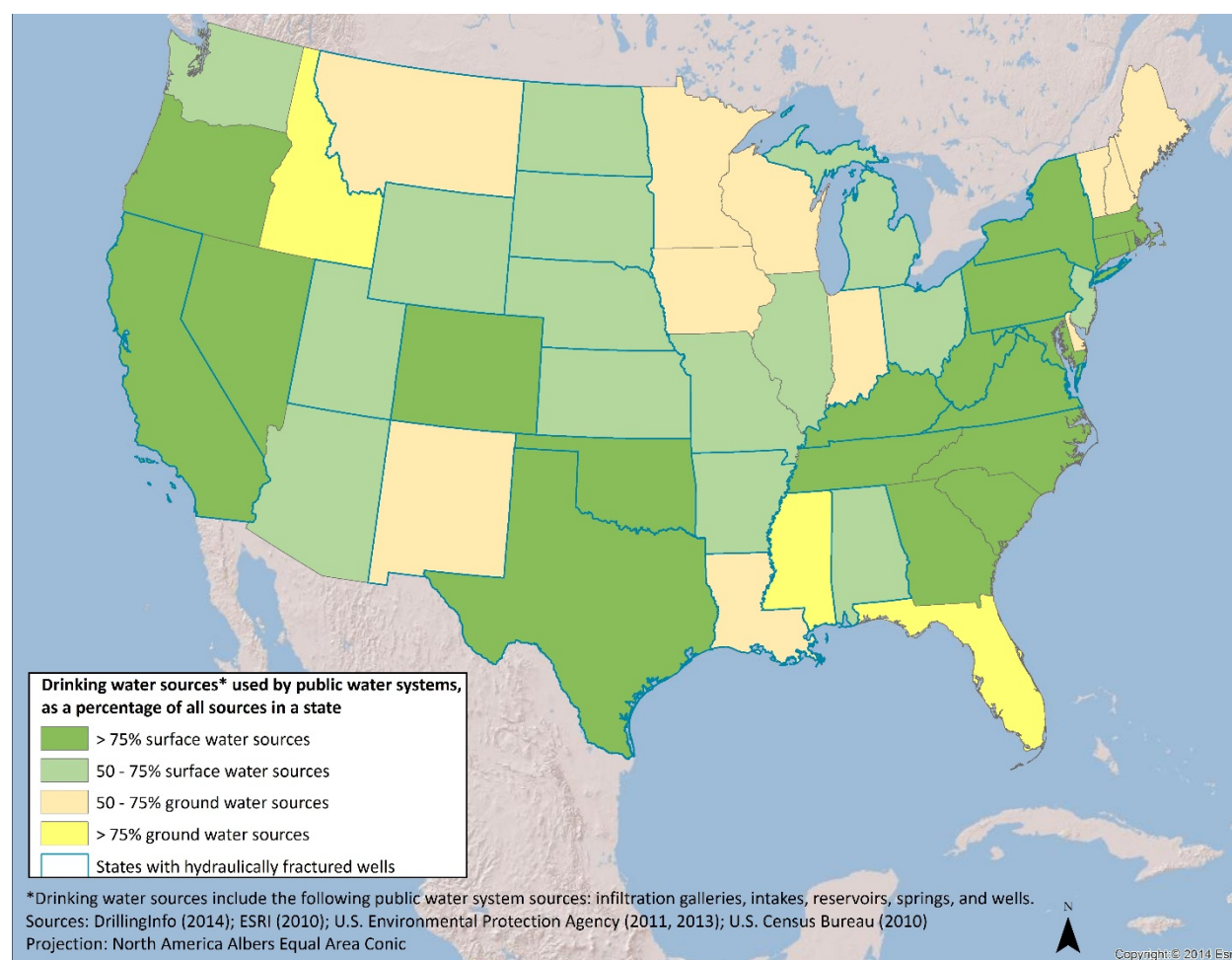


**Figure 3-1. Geographic variability in drinking water sources for public water systems.**

> The relative importance of surface and ground water as drinking water sources varies by state. The
> public water system sources used in this analysis include infiltration galleries, intakes, reservoirs,
> springs, and wells. States with hydraulically fractured wells were identified from DrillingInfo data.

6  The future availability of drinking water resources that are considered fresh in the United States
7  will be affected by changes in climate and water use (Georgakakos et al., 2014; U.S. Global Change

1    [Research Program, 2009](https://...)).[1] Since 2000, about 30% of the total area of the contiguous United States
2    has experienced moderate drought conditions and about 20% has experienced severe drought
3    conditions ([National Drought Mitigation Center, 2015](https://...); [U.S. EPA, 2015r](https://...)). Declines in surface water
4    resources have already led to increased withdrawals and cumulative net depletions of ground
5    water in some areas ([Castle et al., 2014](https://...); [Georgakakos et al., 2014](https://...); [Konikow, 2013a](https://...); [Famiglietti et al.,](https://...)
6    [2011](https://...)). Other sources of water that might not be considered fresh, such as wastewater from sewage
7    treatment plants, brackish (containing 3,000–10,000 mg/L TDS) and saline (containing more than
8    10,000 mg/L TDS) surface and ground water, as well as seawater (containing about 35,000 mg/L
9    TDS) are also increasingly being used to meet water demand. Through treatment or desalination,
10   these water sources can reduce the use of high-quality, potable fresh water for industrial processes,
11   irrigation, recreation, and toilet flushing (i.e., non-potable uses). In addition, in 2010, approximately
12   355 million gal per day (1.3 billion L per day) of treated wastewater was reclaimed through potable
13   reuse projects ([NRC, 2012](https://...)). Such projects use reclaimed wastewater to augment surface drinking
14   water resources or to recharge aquifers that supply drinking water to PWSs ([NRC, 2012](https://...); [Sheng,](https://...)
15   [2005](https://...)).

16   An increasing number of states are developing new water supplies to augment existing water
17   through reuse of reclaimed water, recycling of storm water, and desalination ([U.S. GAO, 2014](https://...)). Most
18   desalination programs currently use brackish water, although plans are underway to expand the
19   use of seawater for desalination in some states. States with the highest installed capacity for
20   desalination include Florida, California, Arizona, and Texas ([Cooley et al., 2006](https://...)). It is likely that
21   various water treatment technologies will continue to expand drinking water resources beyond
22   those currently being considered for use as drinking water. Therefore, these potential future
23   sources are also considered drinking water resources in this assessment.

## 3.2. The Proximity of Drinking Water Resources to Hydraulic Fracturing Activity

24   The colocation of hydraulic fracturing activities with surface and ground water increases the
25   potential for impacts to current and future drinking water resources ([Vengosh et al., 2014](https://...); [Entrekin](https://...)
26   [et al., 2011](https://...)). In this section, we analyze the aboveground proximity of hydraulically fractured well
27   sites, drinking water resources (including the location of surface water bodies and ground water
28   wells that supply public water systems), and populated areas.[2]

29   To determine the spatial relationship between hydraulically fractured wells and populated areas,
30   we analyzed the locations of the approximately 273,000 oil and gas wells that were hydraulically
31   fractured in 25 states between 2000 and 2013 (see Chapter 2) with respect to where people live
32   (i.e., census blocks).[3] Nationwide, approximately 9.4 million people lived within one mile of a

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[1] Fresh water qualitatively refers to water with relatively low TDS that is most readily available for drinking water currently.

[2] The vertical proximity of ground water resources to geologic formations and hydraulic fracturing operations is addressed in Chapter 6.

[3] In the analyses in this chapter, we only include the oil and gas production wells that we identified were hydraulically fractured using criteria outlined in Chapter 2 and that began producing between 2000 and 2013. The well data found in DrillingInfo may not represent the full year for 2013 since the frequency with which DrillingInfo updates the database varies by state. The final update performed by DrillingInfo for 2013 ranges by state from June 2013 to December 2013.

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1 hydraulically fractured well for some period of time between 2000 and 2013 (DrillingInfo, 2014a;
2 U.S. Census Bureau, 2010); more than 5.7 million people lived within half a mile of a hydraulically
3 fractured well.

4 We then analyzed trends in the proximity of hydraulically fractured wells to highly populated areas.
5 For this analysis, we considered metropolitan areas (areas with more than 50,000 people) and
6 micropolitan areas (areas with 10,000 to 49,999 people) (U.S. Census Bureau, 2013c).[1]
7 Approximately 81,300 (30%) of new wells hydraulically fractured between 2000 and 2013 were
8 located within a metropolitan or micropolitan area (see Figure 3-2) (DrillingInfo, 2014a; U.S.
9 Census Bureau, 2013c; U.S. EPA, 2013b; ESRI, 2010). From 2000 to 2008, the number of new wells
10 hydraulically fractured per year within metropolitan and micropolitan areas increased 300%; the
11 proportion of wells hydraulically fractured per year in metropolitan and micropolitan areas almost
12 doubled over the same eight-year period (see Figure 3-3).[2] From 2008 to 2012, however, the
13 number of wells hydraulically fractured per year in metropolitan and micropolitan areas decreased
14 by about half in comparison to the peak of approximately 10,000 wells in 2008 (see Figure 3-3),
15 whereas hydraulic fracturing in areas outside of metropolitan and micropolitan areas increased or
16 remained relatively constant (DrillingInfo, 2014a; U.S. Census Bureau, 2013b).

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[1] Metropolitan and micropolitan combined statistical areas are geographic entities delineated by the Office of
Management and Budget. Specifically, a metropolitan combined statistical area is a core urban area of 50,000 or more
people while a micropolitan combined statistical area is an urban core of at least 10,000, but less than 50,000, people (U.S.
Census Bureau, 2013c). These terms are referred to as metropolitan and micropolitan areas in this assessment.
[2] For comparison, the DrillingInfo data indicate an increase in the number of wells estimated to be hydraulically fractured
each year, regardless of location, from approximately 12,800 in 2000 to slightly more than 21,600 in 2005 to nearly
23,000 in 2012, the last year for which complete data are available.

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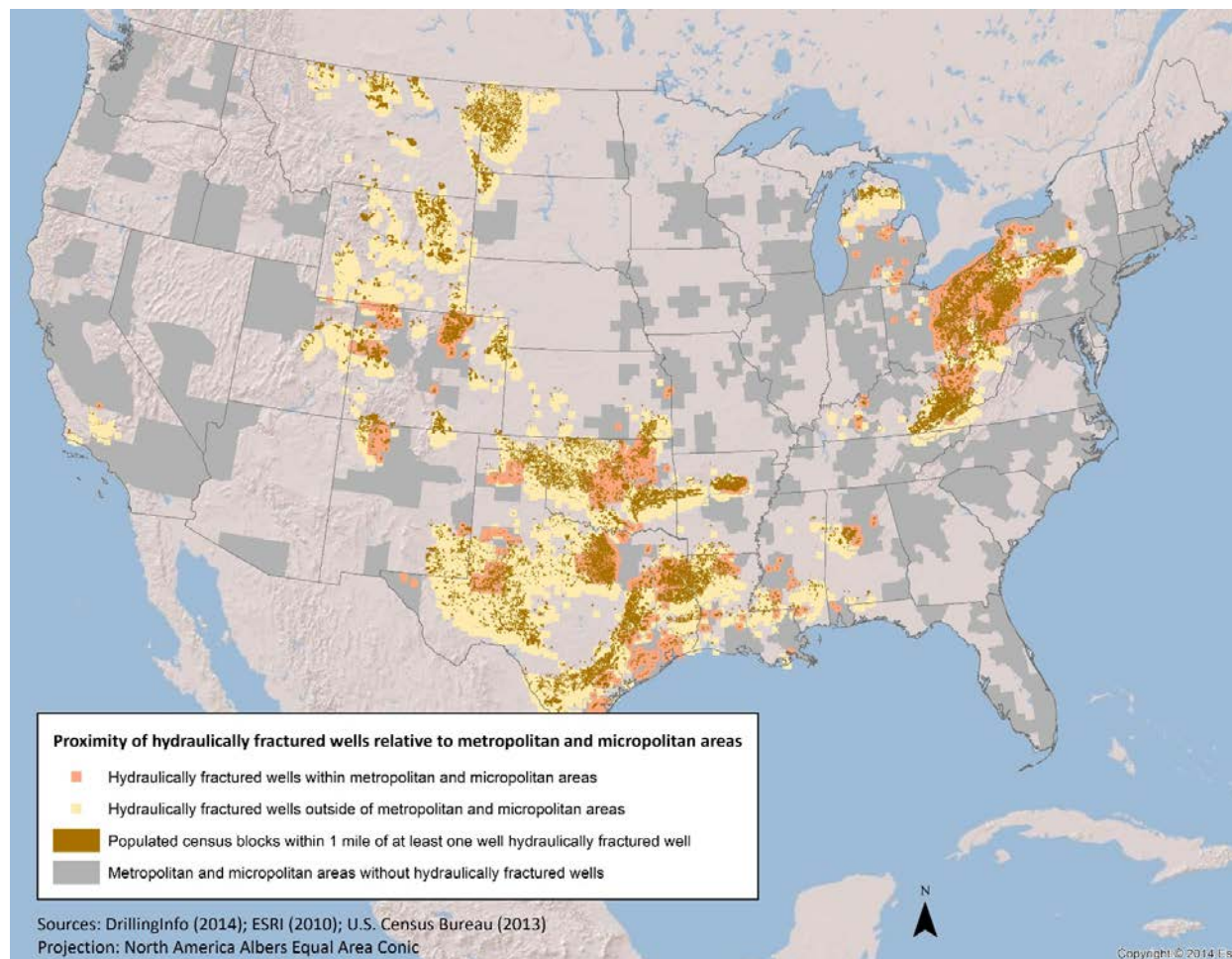
**Figure 3-2. Proximity of hydraulically fractured wells relative to populated areas.**

The estimates of hydraulically fractured wells from 2000 to 2013 developed from the DrillingInfo data were based on several assumptions described in Chapter 2.
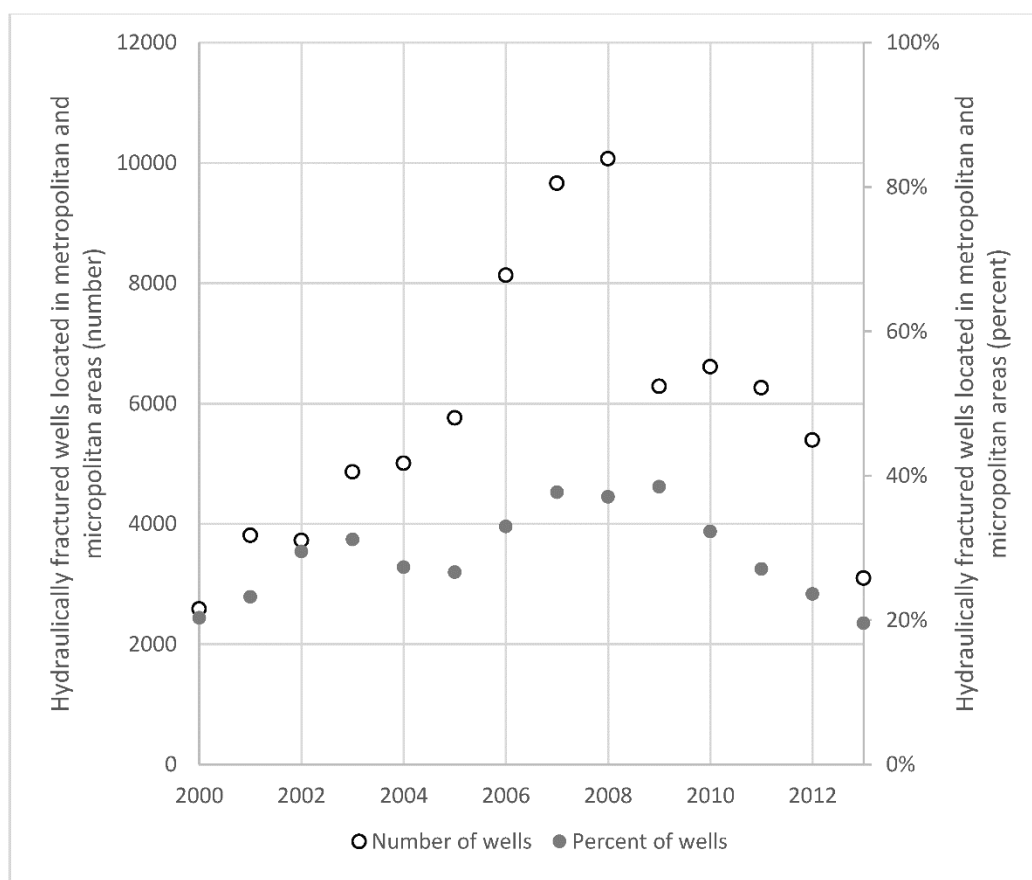
**Figure 3-3. Temporal trends (2000–2013) in the number and percent of hydraulically fractured wells located within populated areas.**

> The estimates of hydraulically fractured wells from 2000 to 2013 developed from the DrillingInfo data were based on several assumptions described in Chapter 2. The graph shows the number of hydraulically fractured wells by the year they started producing. Well data may not be complete for 2013 since final updates to the database for 2013 ranged from June 2013 to December 2013, depending on the state. Original data from DrillingInfo (2014a) and U.S. Census Bureau (2013c).

1  We next considered the proximity of hydraulically fractured wells to water sources for PWSs. We
2  present proximity from both the vantage point of hydraulically fractured wells (e.g., on average,
3  how far away is the nearest PWS source?) and from the vantage point of PWSs (e.g., if there is at
4  least one fractured well within 1 mile of a PWS, are there usually more?).

5  Based on the 2000–2013 DrillingInfo data, the distance from hydraulically fractured wells to the
6  nearest source supplying a PWS ranged from 0.01 to 41 miles, with an average distance of 6.2 miles
7  (9.9 km) and a median distance of 4.8 miles (7.6 km) (DrillingInfo, 2014a; U.S. EPA, 2014j). These
8  PWS sources included both surface water sources (e.g., infiltration galleries, intakes, reservoirs, and
9  springs) and ground water wells. An estimated 21,900 of hydraulically fractured oil and gas wells
10  (8%) were within 1 mile of at least one PWS source (see Figure 3-4). The maximum number was 40
11  PWS sources within 1 mile of a single hydraulically fractured well.

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1   Between 2000 and 2013, approximately 6,800 PWS sources had a hydraulically fractured well
2   within a 1 mile radius. Most of these PWS sources were located in Colorado, Louisiana, Michigan,
3   North Dakota, Ohio, Oklahoma, Pennsylvania, Texas, and Wyoming (see Figure 3-5). These PWS
4   sources had an average of seven fractured wells and a maximum of 144 fractured wells within that
5   one mile proximity. They also supplied water to 3,924 PWSs—1,609 of which are community water
6   systems—that served more than 8.6 million people year-round in 2013 (U.S. EPA, 2014j; U.S.
7   Census Bureau, 2013a; U.S. EPA, 2013b).[1]

8   We also analyzed the location of hydraulically fractured wells relative to populations where a high
9   proportion (≥30%, or twice the national average) obtain drinking water from private systems
10  (private ground water wells and surface water intakes).[2] Between 2000 and 2013, approximately
11  3.6 million people obtained drinking water from private systems in counties with at least one
12  hydraulically fractured well (DrillingInfo, 2014a; USGS, 2014b), and approximately 740,000 people
13  obtained drinking water from private supplies in counties with more than 400 fractured wells
14  (DrillingInfo, 2014a; USGS, 2014b) (see Figure 3-6).[3] These counties were located in Colorado,
15  Kentucky, Michigan, Montana, New Mexico, New York, Oklahoma, Pennsylvania, Texas, and
16  Wyoming (see Figure 3-6).

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[1] All PWS types were included in the locational analyses performed. However, only community water systems were used
to calculate the number of customers obtaining water from a PWS with at least one source within 1 mile of a hydraulically
fractured well. If non-community water systems are included, the estimated number of customers increases by 533,000
people (U.S. EPA, 2012e).
[2] There is no national data set of private water systems. The USGS estimates the proportion of the population reliant on
private water systems, referred to as the "self-supplied population," by county, based on estimates of the population
without connections to a public water system (Maupin et al., 2014). We used the USGS estimates for this analysis.
[3] Approximately 14% of the U.S. population is supplied by private water systems (Maupin et al., 2014). In this analysis, we
only considered counties in which more than double the national average—that is, at least 30% of the county's
population—was supplied by private water systems.
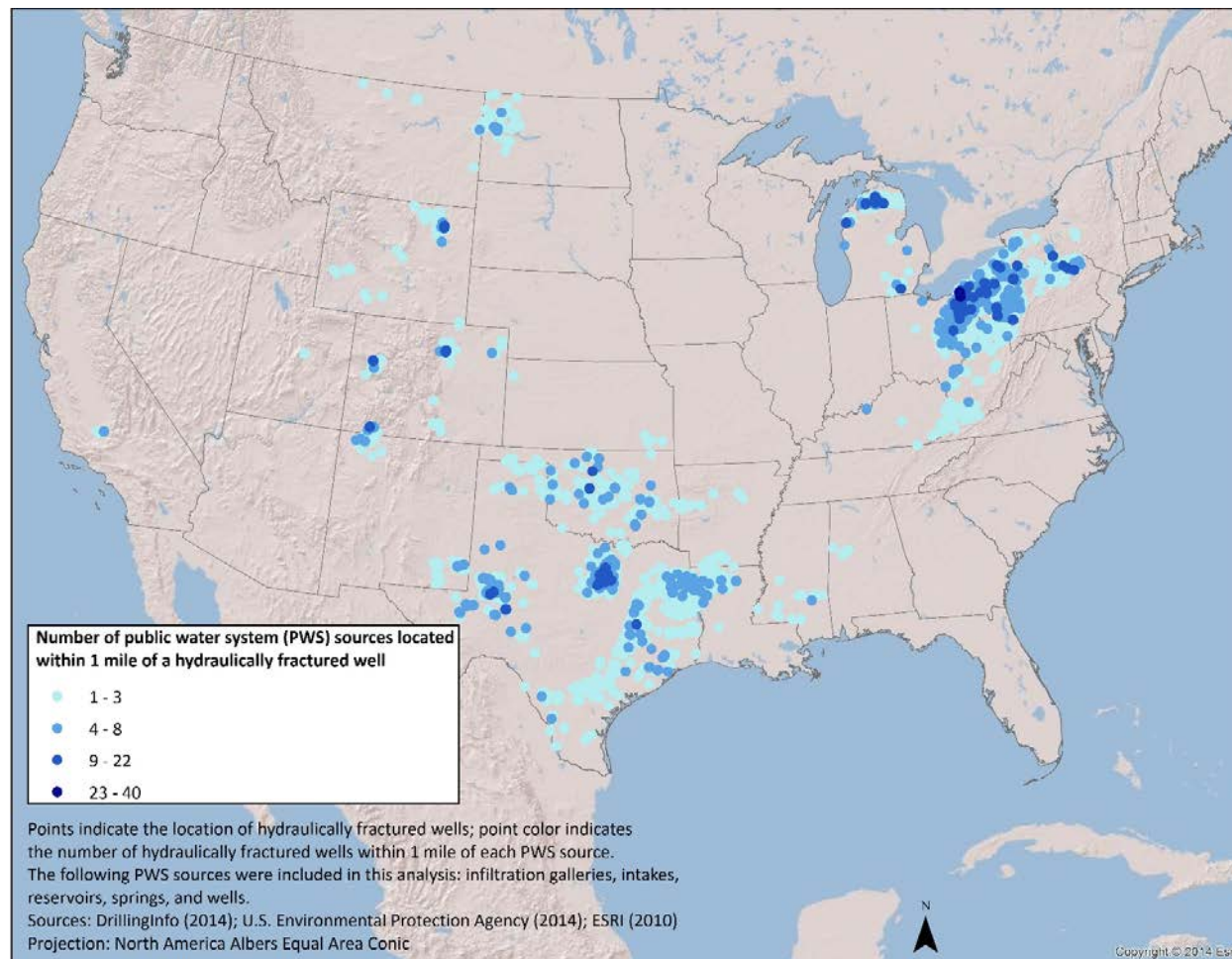
**Figure 3-4. Location and number of public water system (PWS) sources located within 1 mile of a hydraulically fractured well.**

Points indicate the location of hydraulically fractured wells; point color indicates the number of hydraulically fractured wells within 1 mile of each PWS source. The following PWS sources were included in this analysis: infiltration galleries, intakes, reservoirs, springs, and wells. The estimates of wells hydraulically fractured from 2000 to 2013 developed from the DrillingInfo data were based on several assumptions described in Chapter 2.

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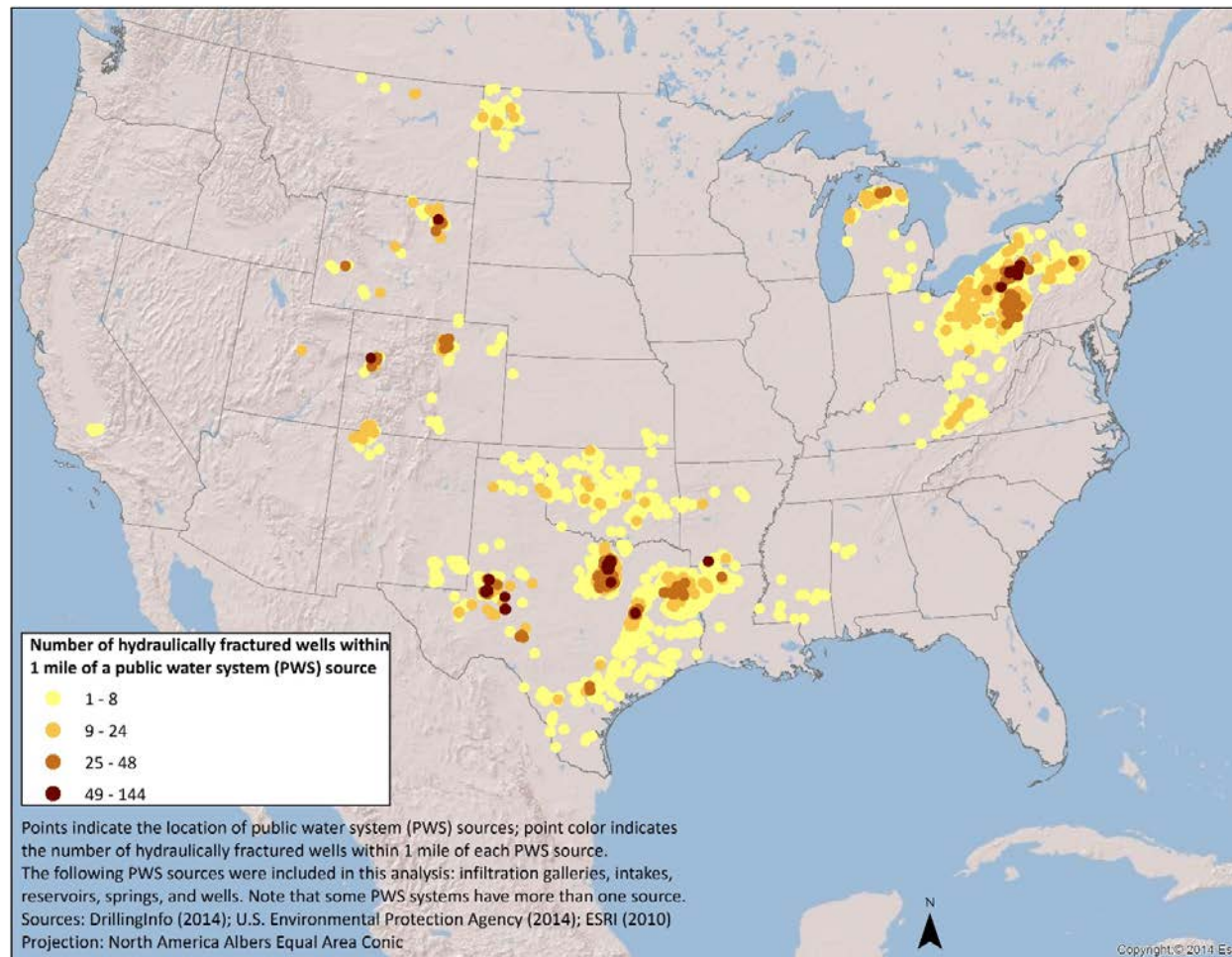
**Figure 3-5. The location of public water system sources within 1 mile of hydraulically fractured wells.**

Points indicate the location of public water system (PWS) sources; point color indicates the number of hydraulically fractured wells within 1 mile of each PWS source. The estimates of wells hydraulically fractured from 2000 to 2013 developed from the DrillingInfo data were based on several assumptions described in Chapter 2.

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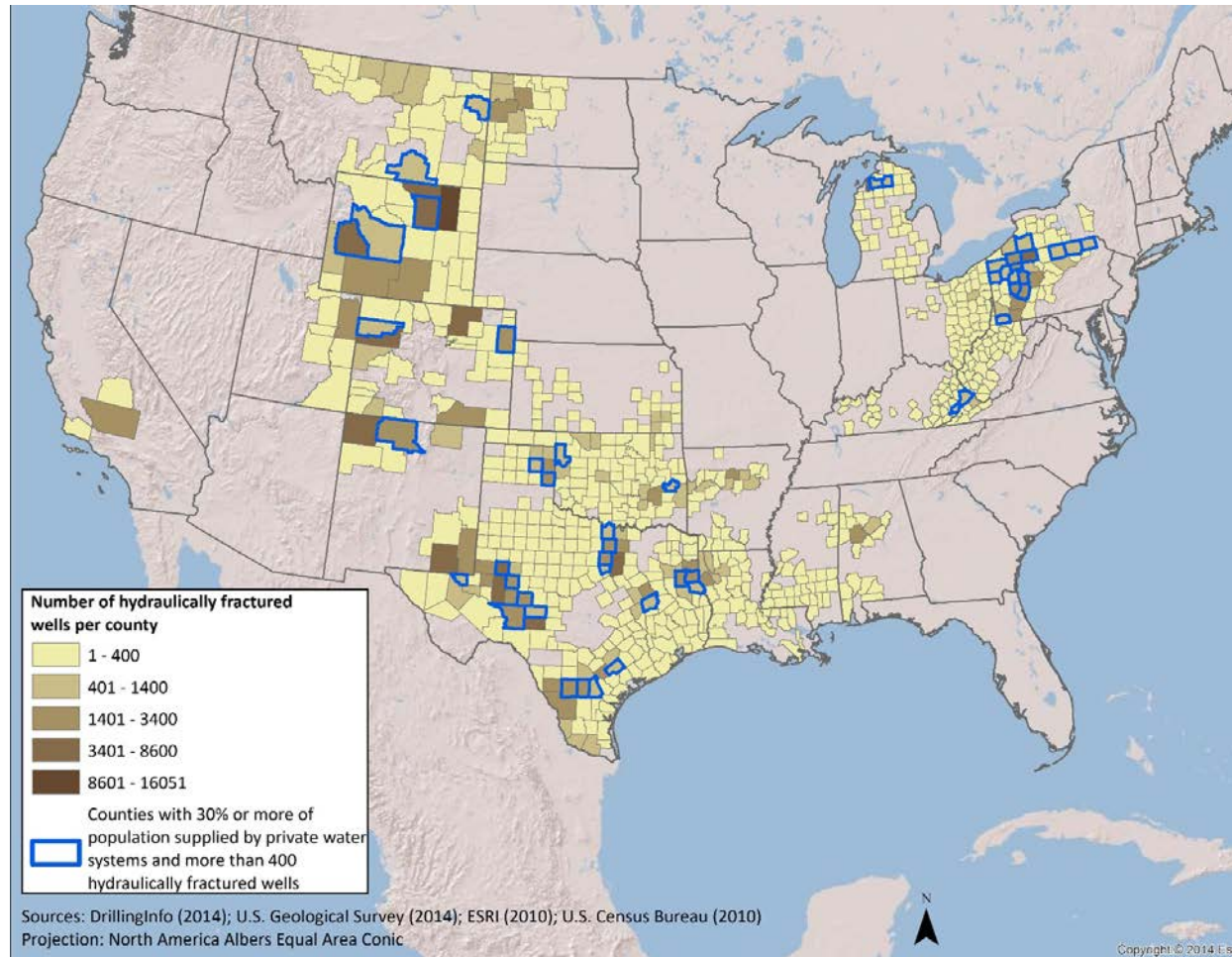
**Figure 3-6. Co-occurrence of hydraulic fracturing activity and populations supplied by private water systems.**

Color indicates the number of hydraulically fractured wells per county. The estimates of wells hydraulically fractured from 2000 to 2013 developed from the DrillingInfo data were based on several assumptions described in Chapter 2. Counties with more than 400 hydraulically fractured wells and in which at least 30% of the population is supplied by private water systems are outlined in blue.

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## 3.3. Conclusion

1   The evaluation of potential hydraulic fracturing impacts on drinking water resources in the United
2   States depends on an understanding of how the country's current and future drinking water needs
3   are and will be met. The U.S. population requires sufficient drinking water resources—that is,
4   bodies of fresh or non-fresh surface or ground water that now serve, or in the future could serve, as
5   a source of water for drinking water for public or private use—to meet everyday needs. Currently,
6   most people in the United States rely on water supplied to their homes via public water systems,
7   and most of this water comes from fresh surface water bodies. Shortages in fresh water availability
8   in the United States, especially in the western United States, have already led some states to
9   augment their water supplies with other water sources (e.g., brackish and saline surface and
10  ground water, seawater, and reclaimed wastewater), suggesting that additional water bodies may
11  provide drinking water as the quantity and quality of existing sources change.

12  The colocation of hydraulic fracturing activities with drinking water resources increases the
13  potential for these activities to affect the quality and quantity of current and future drinking water
14  resources. While close proximity of hydraulically fractured wells to drinking water resources does
15  not necessarily indicate that an impact has or will occur, information about the relative location of
16  wells and water supplies is an initial step in understanding where potential impacts might occur.

17  Millions of people live in areas where their drinking water resources are located near hydraulically
18  fractured wells. While most hydraulic fracturing activity from 2000 to 2013 did not occur in close
19  proximity to public water supplies, a sizeable number of hydraulically fractured wells (21,900)
20  were located within 1 mile of at least one PWS source (e.g., infiltration galleries, intakes, reservoirs,
21  springs and ground water wells). Approximately 6,800 sources of drinking water for public water
22  systems, serving more than 8.6 million people year-round, were located within 1 mile of at least one
23  hydraulically fractured well. An additional 3.6 million people obtain drinking water from private
24  systems in counties with at least one hydraulically fractured well and in which at least 30% of the
25  population is reliant on private water systems.

**Text Box 3-1. Major Findings**

*Current and future drinking water resources*

- Most of the U.S. population (270 million in 2011, or 86%) relies on water supplied to their homes through a public water system, 70% of which comes from surface water and 30% of which comes from ground water.

- An estimated 14% of the U.S. population relies on private water systems for drinking water.

- An increasing number of states are developing new drinking water supplies via reuse of reclaimed water, recycling of storm water, and desalination. These new supplies can augment existing water sources.

- Most of the U.S. population (270 million in 2011, or 86%) relies on water supplied to their homes through a public water system, 70% of which comes from surface water and 30% of which comes from ground water.

- An estimated 14% of the U.S. population relies on private water systems for drinking water.

- An increasing number of states are developing new drinking water supplies via reuse of reclaimed water, recycling of storm water, and desalination. These new supplies can augment existing water sources.

*Proximity of drinking water resources to hydraulic fracturing activity*

- Nationwide, while most hydraulic fracturing activity from 2000 to 2013 did not occur in close proximity to public water supplies, a sizeable number of hydraulically fractured wells (21,900) were located within 1 mile of at least one PWS source.

- The distance between wells that were hydraulically fractured between 2000 and 2013 and the nearest source supplying a PWS ranged from 0.01 to 41 miles, with an average distance of 6.2 miles (9.9 km).

- An estimated 6,800 public water system sources were located within 1 mile of a hydraulically fractured oil and gas well between 2000 and 2013. These PWS sources supplied water to 3,924 public water systems and served more than 8.6 million people year-round in 2013.

- Approximately 9.4 million people lived within 1 mile of at least one hydraulically fractured oil and gas well between 2000 and 2013.

- Approximately 3.6 million people obtain drinking water from private systems in counties with at least one hydraulically fractured well and in which at least 30% of the population (i.e., double the national average) is reliant on private water systems.

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# Chapter 4

# Water Acquisition

# 4. Water Acquisition

## 4.1. Introduction

1  Water is a crucial component of nearly all hydraulic fracturing operations, making up
2  approximately 90% or more of fluid injected into a well (U.S. EPA, 2015a; GWPC and ALL
3  Consulting, 2009). Given that at least 25,000 to 30,000 wells may be fractured each year
4  (Chapter 2), and that each well requires thousands to millions of gallons of water (Section 4.3), the
5  potential exists for effects on the quantity of drinking water resources. Large volume water
6  withdrawals also could alter the quality of drinking water resources by decreasing dilution of
7  pollutants by surface waters, or in the case of ground water, allowing the infiltration of lower-
8  quality water from the land surface or adjacent formations.

9  In this chapter, we consider potential effects of water acquisition for hydraulic fracturing on both
10  drinking water resource quantity and quality, and where possible, identify factors that affect the
11  frequency or severity of impacts. We define drinking water resources broadly, to include not just
12  currently designated drinking waters, but waters that could in the future be used as drinking water
13  sources (see Chapter 1). Although most available data and literature pertain to water use, we
14  discuss water consumption where possible.[1]

15  We provide an overview of the types of hydraulic fracturing water used (Section 4.2); the amount of
16  water used per well (Section 4.3); and cumulative water use and consumption estimates
17  (Section 4.4).[2] We then discuss these three factors for 15 states where hydraulic fracturing
18  presently occurs and consider the potential for hydraulic fracturing water withdrawals to affect
19  water quantity and quality in localities within those states (Section 4.5). We primarily discuss
20  results at the state and county level because data are most available at these scales. Moreover,
21  states and localities often differ in industry activity, formation type, and water availability, all of
22  which affect potential impacts.[3] Lastly, we provide a synthesis that summarizes major findings,
23  factors affecting the frequency or severity of impacts, uncertainties, and conclusions (Section 4.6).

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[1] 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 Underground Injection Control (UIC) Class II injection wells.
[2] In this chapter, cumulative annual water use or water consumption refers to the amount of water used or consumed by all hydraulic fracturing wells in a given area per year.
[3] 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.

## 4.2. Types of Water Used

1   Water used for hydraulic fracturing generally comes from surface water (i.e., rivers, streams, lakes,
2   and reservoirs), ground water aquifers, or reused hydraulic fracturing wastewater.[1,2,3] These
3   sources can vary in their initial water quality and in how they are provisioned to hydraulic
4   fracturing operations. In this section, we provide an overview of the sources (Section 4.2.1), water
5   quality (Section 4.2.2), and provisioning of water (Section 4.2.3) required for hydraulic fracturing.
6   Detailed information on the types of water used by state and locality is presented in Section 4.5.

### 4.2.1. Source

7   Whether water used in hydraulic fracturing originates from surface or ground water resources is
8   largely determined by the amount of water needed and the type of locally available water sources.
9   Water transportation costs can be high, so the industry tends to acquire water from nearby sources
10  if available ([Nicot et al., 2014](#); [Mitchell et al., 2013a](#); [Kargbo et al., 2010](#)). Surface water is typically
11  available to supply most of the water needed in the eastern United States, whereas mixed supplies
12  of surface and ground water are used in the more semi-arid to arid western states. In western
13  states that lack available surface water resources, ground water supplies the majority of water
14  needed for fracturing unless alternative sources, such as reused wastewater, are available and
15  utilized. Local policies also may direct the industry to seek withdrawals from designated sources
16  ([U.S. EPA, 2013a](#)): for instance, some states have encouraged the industry to seek water
17  withdrawals from surface water rather than ground water due to concerns over aquifer depletion
18  (Section 4.5).

19  The reuse of wastewater from past hydraulic fracturing operations can reduce the need for fresh
20  surface or ground water and offset total new water withdrawals for hydraulic fracturing.[4,5] Based
21  on available data, the median reuse of wastewater as a percentage of injected volume is 5%
22  nationally, but this percentage varies by location (Table 4-1).[6,1]

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[1] 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.

[2] 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.

[3] We use "wastewater" as a general term to include both flowback and produced water that may be reused in hydraulic fracturing; we do not distinguish between flowback and produced water except when specifically reported in the literature.

[4] 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.

[5] 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.

[6] 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.

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1    The reuse of wastewater for hydraulic fracturing is limited by the amount of water that returns to
2    the surface during production (Nicot et al., 2012). In the first 10 days of well production, 5% to
3    almost 50% of injected fluid volume may be collected, with values varying across geologic
4    formations (see Chapter 7, Table 7-1). Longer duration measurements are rare, but between 10%
5    and 30% of injected fluid volume has been collected in the Marcellus Shale in Pennsylvania over 9
6    years of production, while over 100% has been collected in the Barnett Shale in north-central Texas
7    over six years of production (see Chapter 7, Table 7-2). Assuming that 10% of injected fluid volume
8    is collected in the first 30 days and the reuse rate is 100%, it would take 10 wells to produce
9    enough water to hydraulically fracture a new well. As more wells are hydraulically fractured in a
10   given area, the potential for wastewater reuse increases.

11   Besides hydraulic fracturing wastewater, other wastewaters may be reclaimed for use in hydraulic
12   fracturing. These may include acid mine drainage, wastewater treatment plant effluent, and other
13   sources of industrial and municipal wastewater (Nicot et al., 2014; Ziemkiewicz et al., 2013).
14   Limited information is available on the extent to which these other wastewaters are used.

**Table 4-1. Percentage of injected water volume that comes from reused hydraulic fracturing wastewater in various states, basins, and plays.**

States listed by order of appearance in the chapter. See Section 4.5 for additional discussion of reuse practices by state and locality and variation over time where data are available.

| State, basin, or play | Available estimate | Year of estimate (NA = not available) |
|---|---|---|

[1] 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 injection wells). This distinction is sometimes overlooked, which sometimes leads to a misrepresentation of the extent to which wastewater is reused to offset total fresh water used for hydraulic fracturing.

| State, basin, or play | Available estimate | Year of estimate (NA = not available) |
|---|---|---|
| Texas—Barnett Shale | 5%[a] | 2011 |
| Texas—Eagle Ford Shale | 0%[a] | 2011 |
| Texas—TX-LA-MS Salt Basin[b] | 5%[a] | 2011 |
| Texas—Permian Basin (far west portion) | 0%[a] | 2011 |
| Texas—Permian Basin (Midland portion) | 2%[a] | 2011 |
| Texas—Anadarko Basin | 20%[a] | 2011 |
| Colorado—Garfield County, Uinta-Piceance Basin | 100%[c] | NA |
| Colorado—Wattenberg Field, Denver-Julesburg Basin | 0%[d] | NA |
| Pennsylvania—Marcellus Shale, Susquehanna River Basin | 18%[e] | 2012 |
| West Virginia—Marcellus Shale, Statewide | 15%[f] | 2012 |
| California—Monterey Shale, Statewide | 4%[g] | 2014 |
| **Overall Mean**[h] | 15% | |
| **Overall Median**[i] | 5% | |

[a] Estimated percentage of recycling/reused water in 2011 (Nicot et al., 2012).
[b] Nicot et al. (2012) refer to this region of Texas as the East Texas Basin.
[c] Based on industry practices reported in U.S. EPA (2015c).
[d] 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).
[e] Volume of flowback injected as a percentage of total water injected, 2012 (Hansen et al., 2013). This is the most recent estimate available. For 2008 to 2011, reuse as a percentage of injected volume averaged 13%, with a median of 8%, according to U.S. EPA (2015c).
[f] Reused fracturing water as a percentage of total water used for hydraulic fracturing, 2012, calculated from data provided by the West Virginia DEP (2014).
[g] Reported data on planned hydraulic fracturing operations as described in 249 well stimulation notices submitted during the first half of January 2014 to CCST (2014). Of these notices, 4% indicated planned use of produced water (sometimes blended with fresh water) for fracturing, while 96% indicated planned use of only fresh water.
[h] The overall mean is not weighted by the number of wells in a given state, basin, or play.
[i] The overall median is not weighted by the number of wells in a given state, basin, or play.

### 4.2.2. Quality

1    Water quality is an important consideration when sourcing water for hydraulic fracturing. Fresh
2    water is often preferred to maximize hydraulic fracturing fluid performance and to ensure
3    compatibility with the geologic formation being fractured. This finding is supported by the EPA's
4    analysis of disclosures to FracFocus 1.0 (hereafter the EPA FracFocus report) (U.S. EPA, 2015a), as
5    well as by regional analyses from Texas (Nicot et al., 2012) and the Marcellus (Mitchell et al.,

1    2013a).[1,2] Fresh water was the most commonly cited water source by companies included in an

2    analysis of nine hydraulic fracturing service companies on their operations from 2005 to 2010 (U.S.

3    EPA, 2013a). Three service companies noted that the majority of their water was fresh because it

4    required minimal testing and treatment (U.S. EPA, 2013a).[3] The majority of the nine service

5    companies recommended testing for certain water quality parameters (pH and maximum

6    concentrations of specific cations and anions) in order to ensure compatibility among the water,

7    other fracturing fluid constituents, and the geologic formation (U.S. EPA, 2013a).

8    The reuse of hydraulic fracturing wastewater may be limited by water quality. As a hydraulically

9    fractured well ages, the wastewater quality begins to resemble the water quality of the geologic

10   formation and may be characterized by high TDS (Goodwin et al., 2014). High concentrations of

11   TDS and other individual dissolved constituents in wastewater, including specific cations (calcium,

12   magnesium, iron, barium, strontium), anions (chloride, bicarbonate, phosphate, and sulfate), and

13   microbial agents, can interfere with hydraulic fracturing fluid performance by producing scale in

14   the wellbore or by interfering with certain chemical additives in the hydraulic fracturing fluid (e.g.,

15   high TDS may inhibit the effectiveness of friction reducers) (Gregory et al., 2011; North Dakota

16   State Water Commission, 2010). Due to these limitations, wastewater may require treatment to

17   meet the level of water quality desired in the hydraulic fracturing fluid formulation. Minimal

18   treatment or blending of wastewater and fresh water is sometimes done to dilute high TDS or other

19   constituents. Fresh water typically makes up the largest proportion of the base fluid when blended

20   with water sources of lesser quality (U.S. EPA, 2015a).[4] However, direct reuse of wastewater with

21   minimal or no treatment is sometimes possible with higher-quality wastewater (U.S. EPA, 2015c)

22   (Section 4.5.2). No data are currently available to characterize the relative frequency of reuse done

23   with treatment, minimal treatment, or no treatment.

### 4.2.3. Provisioning

24   Water for hydraulic fracturing is typically either self-supplied by the industry or purchased from

25   public water systems.[5] Self-supplied water for fracturing generally refers to permitted direct

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[1] FracFocus is a national hydraulic fracturing registry for oil and gas well operators to disclose information about hydraulic fracturing well locations, and water and chemical use during hydraulic fracturing operations developed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission (U.S. EPA, 2015a). The registry was originally established in 2011 for voluntary reporting. However, 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, 2015a).

[2] Of all disclosures to FracFocus 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, 2015a).

[3] Service companies did not provide data on the percentage of fresh water versus non-fresh water used for hydraulic fracturing (U.S. EPA, 2015a).

[4] In FracFocus disclosures indicating that fresh water was used in any combination with "recycled," "produced," or "brine," the median concentration of fresh water across all states ranged from 69% to 93% (U.S. EPA, 2015a).

[5] 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 or ground water 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.

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1    withdrawals from surface or ground water or the reuse of wastewater. Nationally, self-supplied
2    water is more common, although there is much regional variation (U.S. EPA, 2015a; CCST, 2014;
3    Mitchell et al., 2013a; Nicot et al., 2012). Public water systems encompass a variety of water
4    suppliers (U.S. EPA, 2015c). Water purchased from municipal public water systems can be provided
5    either before or after treatment (Nicot et al., 2014). Water for hydraulic fracturing is also
6    sometimes purchased from smaller private entities, such as local land owners (Nicot et al., 2014).

## 4.3.  Water Use Per Well

7    In this section, we provide an overview of the amount of water used per well during hydraulic
8    fracturing. We discuss water use in the life cycle of oil and gas operations (Section 4.3.1), national
9    patterns and associated variability (Section 4.3.2), as well as the factors affecting water use per well
10   including well length, geology, and fracturing fluid formulation (Section 4.3.3). More detailed state-
11   and 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

12   Water is needed throughout the life cycle of oil and gas production and use, including both at the
13   well for processes such as well pad preparation, drilling, and fracturing (i.e., the upstream portion),
14   and later for end uses such as electricity generation, home heating, or transportation (i.e., the
15   downstream portion) (Jiang et al., 2014; Laurenzi and Jersey, 2013). Most of the water used and
16   consumed in the upstream portion of the life cycle occurs during hydraulic fracturing (Jiang et al.,
17   2014; Clark et al., 2013; Laurenzi and Jersey, 2013).[1] Water use per well estimates in this chapter
18   focus on hydraulic fracturing in the upstream portion of the oil and gas life cycle.[2]

### 4.3.2.  National Patterns of Water Use Per Well for Fracturing

19   Hydraulic fracturing for oil and gas requires a large volume of water to create sufficient pressures.
20   According to the EPA's project database of disclosures to FracFocus 1.0 (hereafter the EPA
21   FracFocus project database), the median volume of water used per well, based on

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[1] 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, 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.

[2] 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 evaporation (209.0 gal (791.2 L)/MWh).

1 37,796 disclosures nationally, was 1.5 million gal (5.7 million L) (U.S. EPA, 2015b).[1] There was
2 substantial variability around this median, however, with 10th and 90th percentiles of 74,000 and 6
3 million gal (280,000 and 23 million L) per well, respectively.[2] Even in specific basins and plays,
4 water use per well varied widely. Water injected also can vary within a single field; Laurenzi and
5 Jersey (2013) reported volumes for the Wattenberg Field of the Niobrara play ranging from 1 to 6
6 million gal (3.8 to 23 million L) per well (10th to 90th percentile).

### 4.3.3   Factors Affecting Water Use Per Well

7 Water use varies depending on many factors, including well length, geology, and the composition of
8 the fracturing fluid.

9 *Well length:* Well length is a principal driver of the amount of water used per well. Increases in well
10 length affect total water volumes injected primarily by allowing a larger fracture volume to be
11 stimulated (Economides et al., 2013). Fracture volume is the volume of the fractures in the geologic
12 formation that fill with hydraulic fracturing fluid. The total volume of injected fluid equals fracture
13 volume plus the volume of the wellbore itself, plus any fluid lost due to "leakoff" or other
14 unintended losses.[3] Thus, as wells get longer, fracture, well, and total volumes all increase. This is
15 particularly evident in longer horizontal wells versus vertical wells. For example, median water use
16 in horizontal gas wells was over 35 times higher than in vertical gas wells (2.9 million gal vs. 82,000
17 gal (11 million L vs. 310,000 L), respectively) between the years 2000 and 2010 (USGS, 2015).

18 *Geology:* Geologic characteristics also influence the amount of water used per well. There are three
19 major formation types: shales, tight sands, and coalbeds (see Chapter 2). Reported differences in
20 water use for shales versus tight sands are rare. However, Nicot et al. (2012) reported that total
21 water use in tight sand formations is less than half of that of shale in Texas, although results were
22 not reported per well.

23 In contrast to hydrocarbons from shales and tight sands, coalbed methane (CBM) comes from coal
24 seams that often have a high initial water content and tend to occur at much shallower depths (U.S.
25 EPA, 2015l). Thus, dewatering is often necessary to stimulate production of CBM. In addition,
26 geologic pressures are lower (leading to higher permeability) and well lengths are shorter, all of
27 which result in lower water use per well. Water use per well in CBM operations can be lower by an
28 order of magnitude or more compared to operations in shales or tight sands. For example, Murray
29 (2013) reported water use across formations in Oklahoma, and found that water use in the
30 CBM-dominated Hartshorn Formation was much lower than in the shale gas-dominated Woodford
31 Formation.

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[1] Water use data from the EPA's FracFocus project database were obtained from disclosures made to FracFocus 1.0. 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, 2015b). For the purposes of this chapter, we discuss water use per disclosure in terms of water use per well.
[2] 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.
[3] 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. See Chapter 6 for more information about leakoff.

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1   *Fracturing Fluid Type:* The majority of wells use fracturing fluids that consist mostly of water (U.S.
2   EPA, 2015a; Yang et al., 2013; GWPC and ALL Consulting, 2009). The EPA inferred that more than
3   93% of reported disclosures to FracFocus used water as a base fluid (U.S. EPA, 2015a). The median
4   reported concentration of water in the hydraulic fracturing fluid was 88% by mass, with 10th and
5   90th percentiles of 77% and 95%, respectively. Only roughly 2% of disclosures (761 wells) reported
6   the use of non-aqueous substances as base fluids, typically either liquid-gas mixtures of nitrogen
7   (643 disclosures, 84% of non-aqueous formulations) or carbon dioxide (83 disclosures, 11% of
8   non-aqueous formulations). Both of these formulations still contained substantial amounts of
9   water, as water made up roughly 60% (median value) of fluid in them (U.S. EPA, 2015a). Other
10  formulations were rarely reported. Non-aqueous formulations are discussed further in Chapter 5.

## 4.4. Cumulative Water Use and Consumption

11  In this section we provide an overview of cumulative water use and consumption for hydraulic
12  fracturing at the national, state, and county scales. We then compare these values to total water use
13  and consumption. We discuss both use and consumption because hydraulic fracturing is both a user
14  and consumer of water. Water use refers to water withdrawn for a specific purpose, part or all of
15  which may be returned to the local hydrologic cycle. Water consumption refers to water that is
16  removed from the local hydrologic cycle following its use, and is therefore unavailable to other
17  users (Maupin et al., 2014). Hydraulic fracturing water consumption can occur through such means
18  as evaporation from storage ponds, the retention of water in the subsurface through imbibition, or
19  disposal in UIC Class II injection wells. In the latter two cases, the water consumed is generally
20  completely removed from the hydrologic cycle. In this section, water consumption estimates are
21  derived from USGS water use data, and therefore both use and consumption are presented with the
22  published water use numbers being first.

### 4.4.1. National and State Scale

23  Cumulatively, hydraulic fracturing uses and consumes billions of gallons of water each year in the
24  United States, but at the national or state scale, it is a relatively small user (and consumer) of water
25  compared to total water use and consumption. According to the EPA's FracFocus project database,
26  hydraulic fracturing used 36 billion gal (136 billion L) of water in 2011, and 52 billion gal (197
27  billion L) in 2012; therefore, hydraulic fracturing used an annual average of 44 billion gal (167
28  billion L) of water in 2011 and 2012 across all 20 states in the project database (U.S. EPA, 2015a, b).
29  Cumulative national water use for hydraulic fracturing can also be estimated by multiplying the
30  water use per well by the number of wells hydraulically fractured. If the median water use per well
31  (1.5 million gal) (5.7 million L) from the EPA's FracFocus project database is multiplied by 25,000
32  to 30,000 wells fractured annually (see Chapter 2), cumulative national water use for hydraulic
33  fracturing is estimated to range from 37.5 to 45.0 billion gal (142 to 170 billion L) annually. Other
34  calculated estimates have ranged higher than this, including estimates of approximately 80 billion
35  gal (300 billion L) (Vengosh et al., 2014) and 50-72 billion gal (190-273 billion L) (U.S. EPA, 2015c).
36  These estimates are higher due to differences in the estimated water use per well and the number
37  of wells used as multipliers. For example, (Vengosh et al., 2014) derived the estimate of
38  approximately 80 billion gal (300 billion L) by multiplying an average of 4.0 million gal (15 million

1    L) per well (estimated for shale gas wells) by 20,000 wells (the approximate total number of
2    fractured wells in 2012).[1]

3    All of these estimates of cumulative water use for hydraulic fracturing are small relative to total
4    water use and consumption at the national scale. For example, in the combined 20 states where
5    operators reported water use to FracFocus in 2011 and 2012 (U.S. EPA, 2015b), annual hydraulic
6    fracturing water use and consumption averaged over those two years was less than 1% of total
7    annual water use and consumption in 2010 (see Appendix Table B-1).[2,3]

8    At the state scale, hydraulic fracturing also generally uses billions of gallons of water cumulatively,
9    but accounts for a low percentage of total water use or consumption. Of all states, operators in
10   Texas used the most water cumulatively (47% of cumulative water use reported in the EPA
11   FracFocus project database) (U.S. EPA, 2015b) (see Appendix Table B-1). This was due to the large
12   number of wells in that state. Over 94% of reported cumulative water use occurred in just seven of
13   the 20 states in the EPA FracFocus project database: Texas, Pennsylvania, Arkansas, Colorado,
14   Oklahoma, Louisiana, and North Dakota (U.S. EPA, 2015b). Hydraulic fracturing is a small
15   percentage when compared to total water use (<1%) and consumption (<3%) in each individual
16   state (see Appendix Table B-1). Other studies have shown similar results, with hydraulic fracturing
17   water use and consumption ranging from less than 1% of total use in West Virginia (West Virginia
18   DEP, 2013), Colorado (Colorado Division of Water Resources; Colorado Water Conservation Board;
19   Colorado Oil and Gas Conservation Commission, 2014), and Texas (Nicot et al., 2014; Nicot and
20   Scanlon, 2012), to approximately 4% in North Dakota (North Dakota State Water Commission,
21   2014).

### 4.4.2.  County Scale

22   Cumulative water use and consumption for hydraulic fracturing is also relatively small in most, but
23   not all, counties in the United States (see Table 4-2, Figure 4-1, and Figure 4-2a,b). Reported

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[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 EPA FracFocus project database likely leads to a more robust estimate when used to calculate national cumulative 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 (U.S. EPA, 2015b).

[2] The USGS compiles water use estimates approximately every five years in the National Water Census including the 1995 Census in Solley et al. (1998); 2005 Census in Kenny et al. (2009); and 2010 Census in Maupin et al. (2014). The 2010 version is the most updated version available. The Census includes uses such as public supply, irrigation, livestock, aquaculture, thermoelectric power, industrial, and mining at the national, state, and county scale. The 2010 Census included hydraulic fracturing water use in the mining category; there was no designated category for hydraulic fracturing alone.

[3] Percentages were calculated by averaging annual water use for hydraulic fracturing in 2011 and 2012 for a given state or county (U.S. EPA, 2015b), 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 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). See footnotes for Appendix Table B-1 and Table 4-2 for description of the consumption estimate calculations.

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1    fracturing water use in FracFocus in 2011 and 2012 was less than 1% compared to 2010 USGS total
2    water use in 299 of the 401 reporting counties (U.S. EPA, 2015b) (see Figure 4-2a and Appendix
3    Table B-2). However, hydraulic fracturing water use was 10% or more compared to total water use
4    in 26 counties, 30% or more in nine counties, and 50% or more in four counties (see Table 4-2 and
5    Figure 4-2a). McMullen County in Texas had the highest percentage at over 100% compared to
6    2010 total water use.[1] Total consumption estimates followed the same pattern, but with more
7    counties in the higher percentage categories (hydraulic fracturing water consumption was 10% or
8    more compared to total water consumption in 53 counties; 30% or more in 25 counties; 50% or
9    more in 16 counties; and over 100% in four counties) (see Table 4-2 and Figure 4-2b). Note,
10   estimates based on the EPA's FracFocus project database may form an incomplete picture of
11   hydraulic fracturing water use in a given state or county because the majority of states with data in
12   the project database did not require disclosure to FracFocus during the time period analyzed (U.S.
13   EPA, 2015a). We conclude that this likely does not substantially alter the overall patterns observed
14   in Figure 4-2a,b (see Text Box 4-1 for further details).

15   These percentages depend both upon the absolute water use and consumption for hydraulic
16   fracturing and the relative magnitude of other water uses and consumption in that state or county.
17   For instance, a rural county, with a small population, might have relatively low total water use prior
18   to hydraulic fracturing.[2] Also, just because water is used in certain county does not necessarily
19   mean it originated in that county. While the cost of trucking water can be substantial (Slutz et al.,
20   2012), and the industry tends to acquire water from nearby sources when possible (see Section
21   4.2.1), water can also be piped in from more distant, regional supplies. Despite these caveats, it is
22   clear that hydraulic fracturing is generally a relatively small user (or consumer) of water at the
23   county level, with the exception of a small number of counties where water use and consumption
24   for fracturing can be high relative to other uses and consumption.

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[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)).

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**Table 4-2. Annual average 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). Annual average hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus project database (U.S. EPA, 2015b). 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] | Annual average 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 |

| State | County | Total annual water use in 2010 (millions of gal)[a] | Annual average 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] |
|---|---|---|---|---|---|
| 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 |
| 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 as reported to FracFocus (U.S. EPA, 2015b).
[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, 2015b), 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 (USGS, 2014h). 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 UIC Class II injection wells—see Chapter 8).
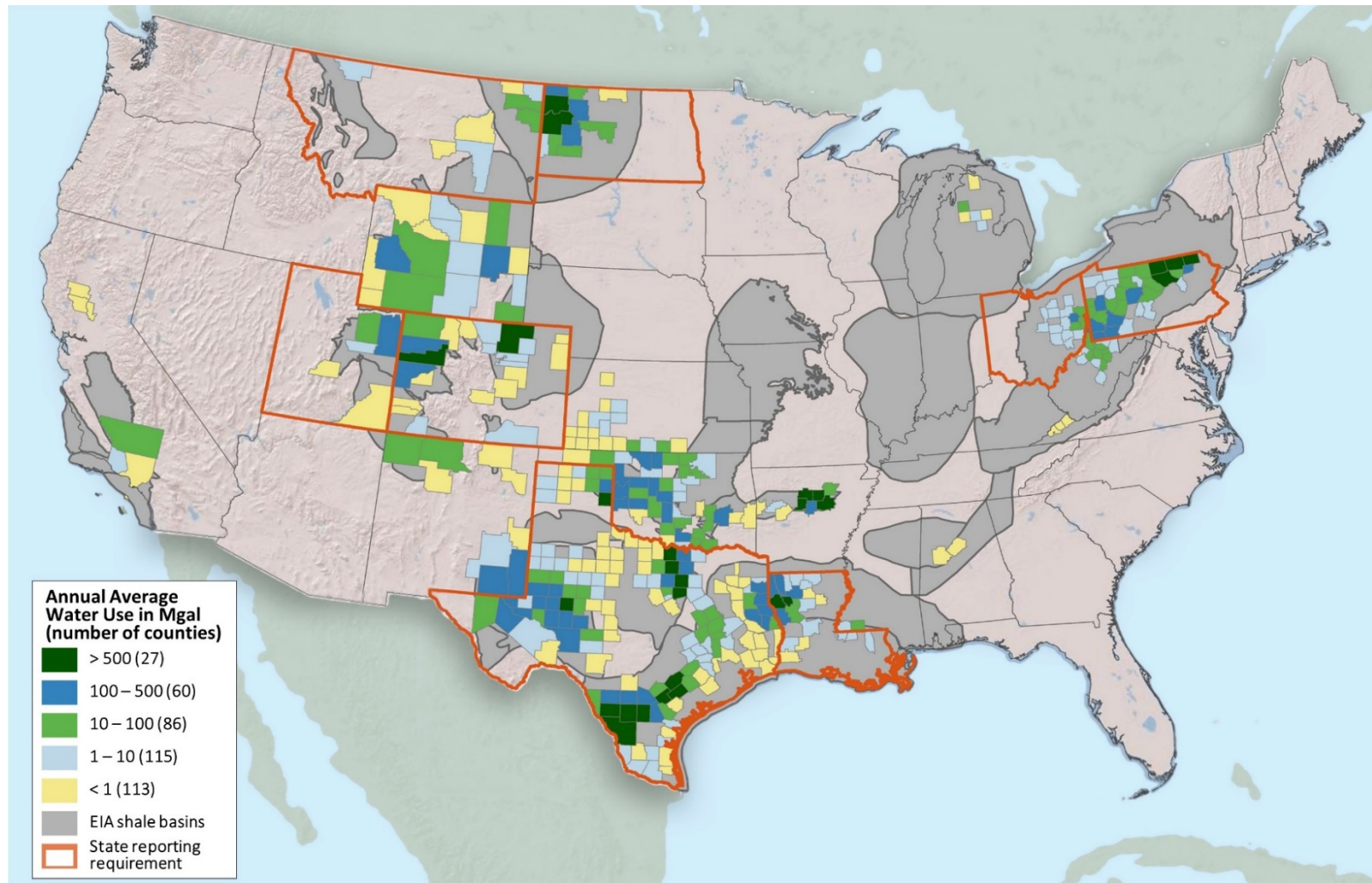
**Figure 4-1. Annual average hydraulic fracturing water use in 2011 and 2012 by county (U.S. EPA, 2015b).**

Source: (U.S. EPA, 2015b). Water use in millions of gallons (Mgal). Counties shown with respect to major U.S. Energy Information Administration (EIA) shale basins (EIA, 2015b). Orange borders identify states that required some degree of reporting to FracFocus 1.0 in 2011 and 2012.

*This document is a draft for review purposes only and does not constitute Agency policy.*
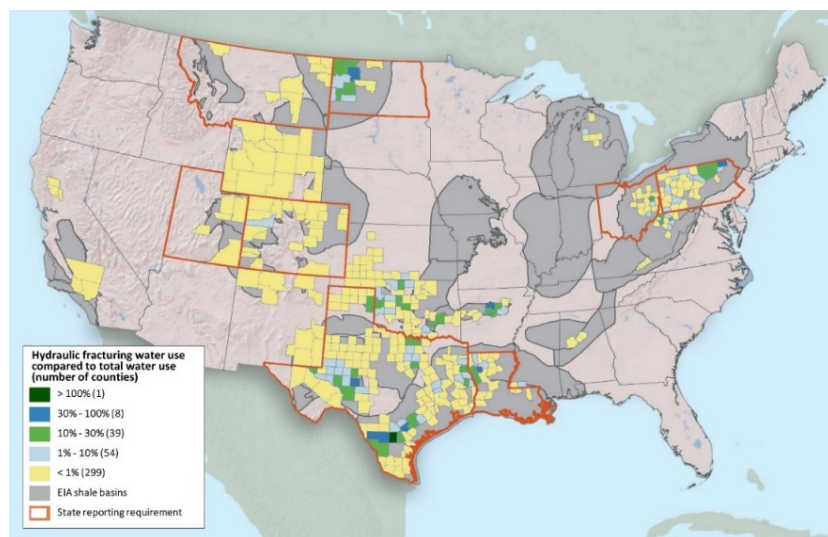
**(a)**
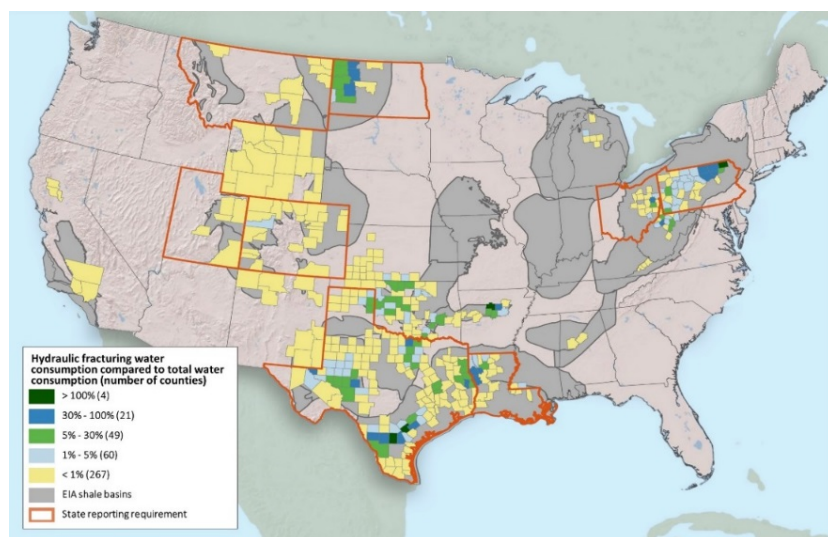


**(b)**

**Figure 4-2. (a) Annual average hydraulic fracturing water use in 2011 and 2012 compared to total annual water use in 2010, by county, expressed as a percentage; (b) Annual average hydraulic fracturing water consumption in 2011 and 2012 compared to total annual water consumption in 2010, by county, expressed as a percentage.**

Annual average hydraulic fracturing water use data in 2011 and 2012 from the EPA's FracFocus project database (U.S. EPA, 2015b). Total annual water use data in 2010 from the USGS (Maupin et al., 2014). See Table 4-2 for descriptions of calculations for estimating consumption. Counties shown with respect to major U.S. EIA shale basins (EIA, 2015b). Orange borders identify states that required some degree of reporting to FracFocus 1.0 in 2011 and 2012. Note: Values over 100% denote counties where the annual average hydraulic fracturing water use or consumption in 2011 and 2012 exceeded the total annual water use or consumption in that county in 2010.

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

1  FracFocus is a national hydraulic fracturing registry managed by the Ground Water Protection Council and the Interstate
2  Oil and Gas Compact Commission (GWPC, 2015). The registry was established in 2011 for voluntary reporting. However,
3  six of the 20 states discussed in this assessment required disclosure to FracFocus at various points between January 1,
4  2011 and February 28, 2013, the time period analyzed by the EPA; another three of the 20 states offered the choice of
5  reporting to FracFocus or the state during this same time period (U.S. EPA, 2015a). Estimates based on the EPA's
6  FracFocus project database likely form an incomplete picture of hydraulic fracturing water use because most states with
7  data in the project database (14 out of 20) did not require disclosure to FracFocus during the time period analyzed (U.S.
8  EPA, 2015a).

9  Cumulative water use for fracturing is a function of the water use per well and the total number of wells fractured. For
10 water use per well, we found seven literature values for comparison with values from the EPA's FracFocus project
11 database. On average, water use estimates per well in the project database were 77% of literature values (the median was
12 86%); Colorado's Denver Basin was the only location where the project database estimate as a percentage of the
13 literature estimate was low (14%) (see Appendix Table B-3). In general, water use per well estimates from the EPA's
14 FracFocus project database appear to provide a reasonable approximation for most areas for which we have data, with
15 the exception of the Denver Basin of Colorado.

16 For number of wells, we compared data in the EPA's FracFocus project database to numbers available in state databases
17 from North Dakota, Pennsylvania, and West Virginia (see Appendix Table B-4). These were the state databases from
18 which we could distinguish hydraulically fractured wells from total oil and gas wells. On average, we found that the EPA
19 FracFocus project database included 67% of the wells listed in state databases for 2011 and 2012 (see Appendix Table B-
20 4). Unlike North Dakota and Pennsylvania, West Virginia did not require operators to report fractured wells to FracFocus
21 during this time period, possibly explaining its lower reporting rate. Multiplying the average EPA FracFocus project
22 database values of 77% for water use per well and 67% for well counts yields 52%. Thus, the EPA FracFocus project
23 database estimates for water use could be slightly over half of the estimates from these three state databases during this
24 time period. These values are based on a small sample sizes (7 literature values and 3 state databases) and should be
25 interpreted with caution. Nevertheless, these numbers at the very least suggest that estimates based on the EPA's
26 FracFocus project database may form an incomplete picture of hydraulic fracturing water use during this time period.

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

33 These results indicate that most counties have relatively low hydraulic fracturing water use, relative to total water use,
34 even when accounting for likely underestimates. Since consumption estimates are derived from use, these will also follow
35 the same pattern. Thus, potential underestimates based on the EPA's FracFocus project database likely do not
36 substantially alter the overall pattern shown in Figure 4-2. Rather, underestimates of hydraulic fracturing water use
37 would mostly affect the percentages in the small number of counties where fracturing already constitutes a higher
38 percentage of total water use and consumption.

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## 4.5. Potential for Water Use Impacts by State

39 High fracturing water use or consumption alone does not necessarily result in impacts to drinking
40 water resources. Rather, impacts most often result from the combination of water use and water
41 availability at a given withdrawal point. Where water availability is high compared to water

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1   withdrawn for hydraulic fracturing, this water use can be accomodated. However, where water
2   availability is low compared to use, hydraulic fracturing withdrawals are more likely to impact
3   drinking water resources. Water management, such as the type of water used or the timing or
4   location of withdrawals, can modify this relationship. All of these factors can vary considerably by
5   location.

6   Besides potential water quantity effects, water withdrawals for hydraulic fracturing have the
7   potential to alter the quality of drinking water resources. This possibility is not unique to the oil and
8   gas industry, as any large-volume water withdrawal has the potential to affect water quality.
9   Although there is little research that specifically connects water withdrawals for hydraulic
10  fracturing to potential water quality impacts, multiple studies have described the impact of drought
11  or industrial withdrawals on water quality (Georgakakos et al., 2014; Whitehead et al., 2009;
12  Murdoch et al., 2000; Schindler, 1997). For instance, in the absence of controls, surface water
13  withdrawals can lower water levels and alter stream flows, potentially decreasing a stream's
14  capacity to dilute contaminants (Mitchell et al., 2013a; Entrekin et al., 2011; NYSDEC, 2011; van
15  Vliet and Zwolsman, 2008; IPCC, 2007; Environment Canada, 2004; Murdoch et al., 2000).
16  Furthermore, ground water withdrawals exceeding natural recharge rates may lower the water
17  level in aquifers, potentially mobilizing contaminants or allowing the infiltration of lower-quality
18  water from the land surface or adjacent formations (USGS, 2003; Jackson et al., 2001).

19  In the following section, we assess the potential for water quantity and quality impacts by location,
20  organized by state. We focus our discussion on the 15 states that account for almost all disclosures
21  reported in the EPA FracFocus project database (U.S. EPA, 2015b): Texas (Section 4.5.1); Colorado
22  and Wyoming (Section 4.5.2); Pennsylvania, West Virginia, and Ohio (Section 4.5.3); North Dakota
23  and Montana (Section 4.5.4); Oklahoma and Kansas (Section 4.5.5); Arkansas and Louisiana
24  (Section 4.5.6); and Utah, New Mexico, and California (Section 4.5.7).[1] Each section describes the
25  extent of hydraulic fracturing activity in that state or group of states; the type of water used in
26  terms of source, quality, and provisioning; and the water use per well. We then discuss cumulative
27  estimates and the potential for impacts to drinking water resources in the context of water
28  availability.

29  We have ordered the states by the number of hydraulically fractured wells reported, and combined
30  states with similar geographies or activity. Most of the available data did not allow us to assess the
31  potential for impacts at a finer resolution than the county scale. Any potential adverse impacts are
32  most likely to be observed locally at a particular withdrawal point. Therefore, our analysis most
33  often suggests where the potential for impacts exists, but does not indicate where impacts will
34  occur at the local scale. Where possible, we utilize local-scale case studies in southern Texas,
35  western Colorado, and eastern Pennsylvania to provide details at a much finer resolution, and offer
36  insight into whether any impacts from water acquisition for hydraulic fracturing were realized in
37  these areas.

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[1] We do not highlight the remaining five states included in the EPA FracFocus project database because of low reported
activity: Virginia (90 disclosures), Alabama (55), Alaska (37), Michigan (15), and Mississippi (4).

### 4.5.1. Texas

1  Hydraulic fracturing in Texas accounts for the bulk of the activity reported nationwide, comprising
2  48% of the disclosures in the EPA FracFocus project database (U.S. EPA, 2015b) (see Figure 4-3 and
3  Appendix Table B-5). There are five major basins in Texas: the Permian, Western Gulf (includes the
4  Eagle Ford play), Fort Worth (includes the Barnett play), TX-LA-MS Salt (includes the Haynesville
5  play), and the Anadarko (see Figure 4-4); together, these five basins contain 99% of Texas' reported
6  wells (see Appendix Table B-5).
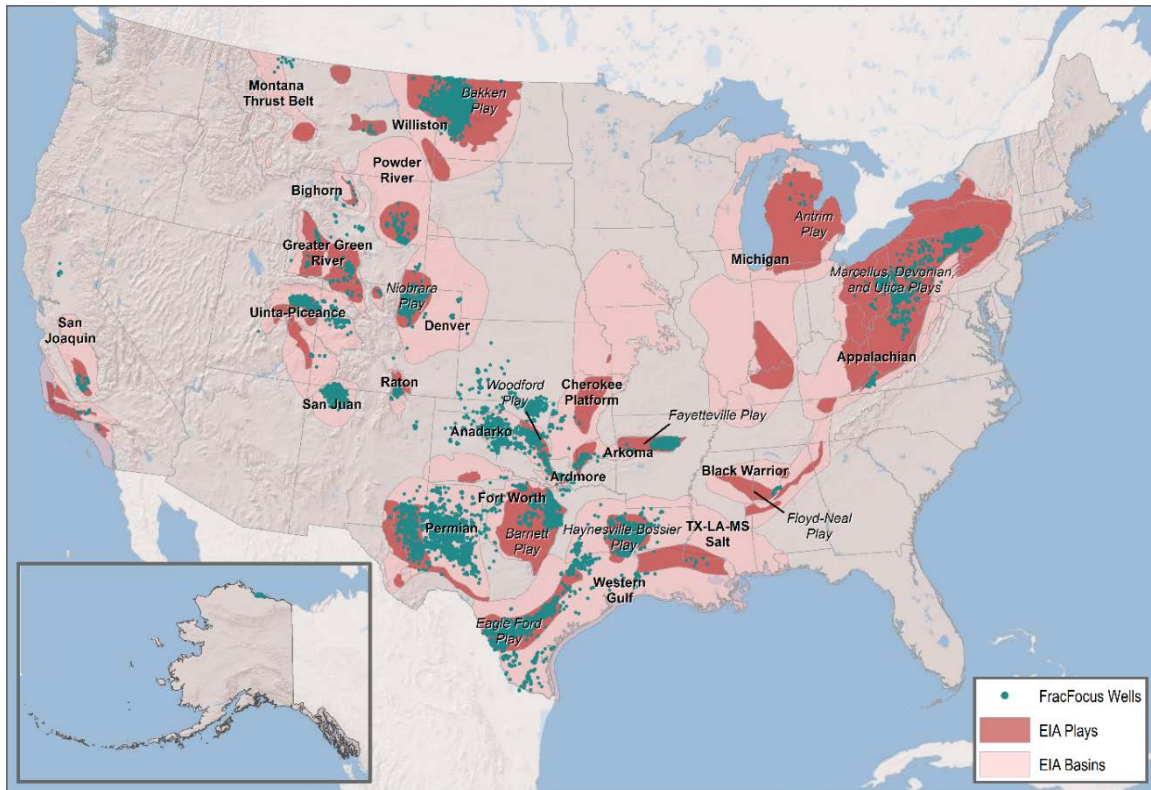


**Figure 4-3. Locations of wells in the EPA FracFocus project database, with respect to U.S. EIA shale plays and basins (EIA, 2015; US. EPA, 2015b).**

Note: Hydraulic fracturing is conducted in geologic settings other than shale; therefore, some wells on this map are not associated with any EIA shale play or basin. (EIA, 2015b; U.S. EPA, 2015b).
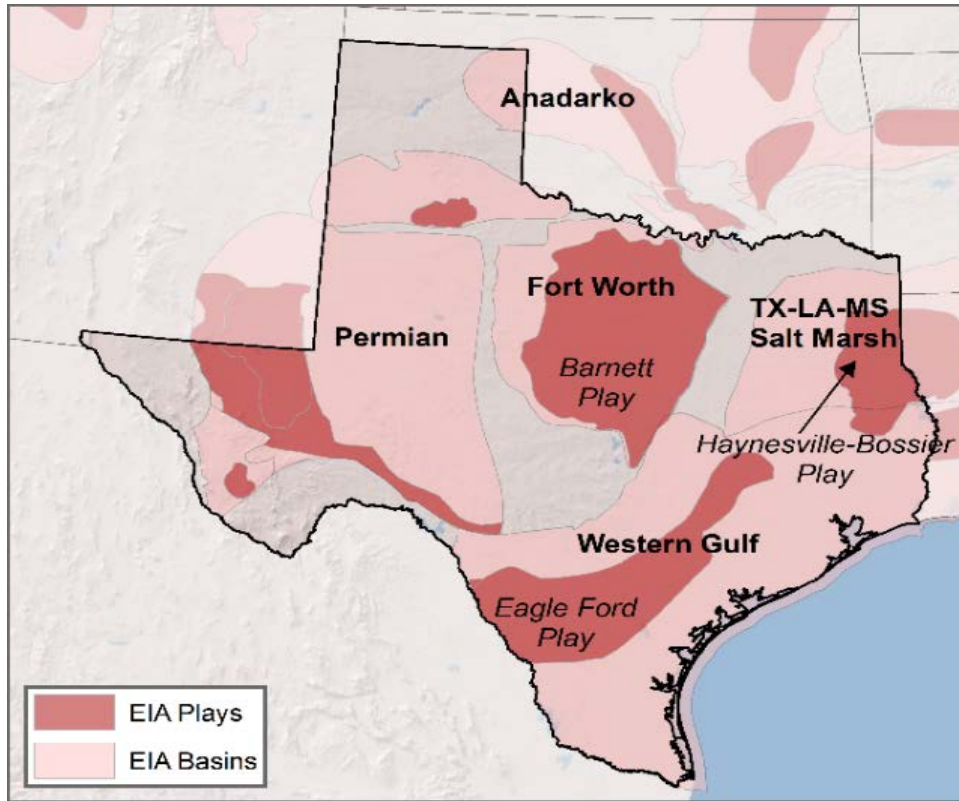
**Figure 4-4. Major U.S. EIA shale plays and basins for Texas (EIA, 2015).**

Source: ([EIA, 2015b](#))

1    *Types of water used:* What is known about water sources in Texas largely comes from direct surveys
2    and interviews with industry operators and water suppliers ([Nicot et al., 2014](#); [Nicot et al., 2012](#)).
3    Overall, ground water is the dominant source throughout most of the state ([Nicot et al., 2014](#); [Nicot et](#)
4    [al., 2012](#)) (see Table 4-3). The exception is the Barnett Shale, where both surface and ground water
5    are used in approximately equal proportions.

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

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[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, 2015a](#)). However, 19% of Texas disclosures included information related to water sources ([U.S. EPA, 2015a](#)).

**Table 4-3. Estimated proportions of hydraulic fracturing source water from surface and ground water.**

States listed by order of appearance in the chapter.

| Location | Surface water | Ground water |
|---|---|---|
| Texas—Barnett Shale | 50%[a] | 50%[a] |
| Texas—Eagle Ford Shale | 10%[b] | 90%[b] |
| Texas—TX-LA-MS Salt Basin[c] | 30%[b] | 70%[b] |
| Texas—Permian Basin | 0%[b] | 100%[b] |
| Texas—Anadarko Basin | 20%[b] | 80%[b] |
| Pennsylvania—Marcellus Shale, Susquehanna River Basin | 78%[d] | 22%[d] |
| West Virginia—Statewide, Marcellus Shale | 91%[e] | 9%[e] |
| Oklahoma—Statewide | 63%[f] | 37%[f] |
| Louisiana—Haynesville Shale | 87%[g] | 13%[g] |

[a] Nicot et al. (2014).

[b] Nicot et al. (2012).

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

[d] Estimated proportions are for 2011 (U.S. EPA, 2015c).

[e] 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 or ground water. Therefore, we excluded purchased water in calculating the relative proportions of surface and ground water shown in Table 4-3 (West Virginia DEP, 2014).

[f] Proportion of surface and ground water 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).

[g] Data from October 1, 2009, to February 23, 2012, for 1,959 Haynesville Shale natural gas wells (LA Ground Water Resources Commission, 2012).

**Table 4-4. Brackish water use as a percentage of total hydraulic fracturing water use in Texas' main hydraulic fracturing areas, 2011.**

Adapted from Nicot et al. (2012).[a]

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

[a] 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%.

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

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

4  *Water use per well*: Water use per well varies across Texas' basins, with reported medians of
5  3.9 million gal (14.8 million L) in the Fort Worth Basin, 3.8 million gal (14.4 million L) in the
6  Western Gulf, 3.3 million gal (12.5 million L) in the Anadarko, 3.1 million gal (11.7 million L) in the
7  TX-LA-MS Salt, and 840,000 gal (3.2 million L) in the Permian (see Appendix Table B-5). Relatively
8  low water use in the Permian Basin, which contains roughly half the reported wells in the state, is
9  due to the abundance of vertical wells, mostly for oil extraction (Nicot et al., 2012).

10  Water use per well is increasing in most locations in Texas. In the Barnett Shale, water use per
11  horizontal well increased from a median of 1.25 million gal (4.73 million L) in 2001 to 4.7 million
12  gal (17.8 million L) in 2012, as the number of wells and horizontal lengths increased (Nicot et al.,
13  2014). Similar increases in lateral length and water use per well were reported for the
14  Texas-Haynesville, East Texas, Anadarko, and most of the Permian Basin (Nicot et al., 2012; Nicot
15  and Scanlon, 2012).[1]

---

[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).

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

7  According to detailed county-level projections, water use for hydraulic fracturing is expected to
8  increase with oil and gas production in the coming decades, peaking around the year 2030 (Nicot et
9  al., 2012). The majority of counties are expected to have relatively low cumulative water use for
10 fracturing in the future, but cumulative hydraulic fracturing water use could equal or exceed 10%,
11 30%, and 50% compared to 2010 total county water use in 30, nine, and three counties,
12 respectively, by 2030 (see Appendix Table B-7). Thus, potential hydraulic fracturing water
13 acquisition impacts in Texas may be most likely to occur over the next 15–25 years as water
14 demand for fracturing is highest.

15 *Potential for impacts:* Of all locations surveyed in this chapter, the potential for water quantity and
16 quality impacts due to hydraulic fracturing water use appears to be highest in western and
17 southern Texas. This area includes the Anadarko, the Western Gulf (Eagle Ford play), and the
18 Permian Basins. According to Ceres (2014), 28% and 87% of the wells fractured in the Eagle Ford
19 play and Permian Basin, respectively, are in areas of high to extremely high water stress.[3] A
20 comparison of hydraulic fracturing water use to water availability at the county scale also suggests
21 the potential for impacts (see Text Box 4-2 and Figure 4-5). The Texas Water Development Board
22 estimates that overall demand for water (including water for hydraulic fracturing) out to the year
23 2060 will outstrip supply in southern and western Texas (TWDB, 2012). Moreover, the state has
24 experienced moderate to extreme drought conditions for much of the last decade (National Drought
25 Mitigation Center, 2015). The 2012 Texas State Water Plan emphasizes that "in serious drought
26 conditions, Texas does not and will not have enough water to meet the needs of its people, its
27 businesses, and its agricultural enterprises" (TWDB, 2012).

28

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[1] 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 (see Table 4-2).

[2] 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 cumulative 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 19.3% 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).

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*This document is a draft for review purposes only and does not constitute Agency policy.*

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

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

5    The authors generated annual availability estimates for five categories of water: unappropriated surface
6    water, unappropriated ground water, appropriated water potentially available for purchase, brackish
7    groundwater, and wastewater from municipal treatment plants (Tidwell et al., 2013). In the western United
8    States, water is generally allocated by the principle of prior appropriation—that is, first in time of use is first
9    in right. New development must use unappropriated water or purchase appropriated water from vested
10   users. In their analysis, the authors assumed 5% of appropriated irrigated water could be purchased; they
11   also excluded wastewater required to be returned to streams and the wastewater fraction already reused.
12   Given regulatory restrictions, they considered no fresh water to be available in California for new
13   thermoelectric plants.

14   Combining their estimates of unappropriated surface and ground water and appropriated water potentially
15   available for purchase, we derived a fresh water availability estimate for each county (except for those in
16   California) and then compared this value to reported water use for hydraulic fracturing (U.S. EPA, 2015b). We
17   also added the estimates of brackish and wastewater to fresh water estimates to derive estimates of total
18   water availability and did a similar comparison. Since the water availability estimates already take into
19   account current water use for oil and gas operations, these results should be used only as indicator of areas
20   where shortages might arise in the future.

21   Overall, hydraulic fracturing water use represented less than 1% of fresh water availability in over 300 of the
22   395 counties analyzed (see Figure 4-5a). This result suggests that there is ample water available at the county
23   scale to accommodate hydraulic fracturing in most locations. However, there was a small number of counties
24   where hydraulic fracturing water use was a relatively high percentage of fresh water availability. In 17
25   counties, fracturing water use actually exceeded the index of fresh water available; all of these counties were
26   located in the state of Texas and were associated with the Anadarko, Barnett, Eagle Ford, and Permian
27   basins/plays (see Figure 4-4). In Texas counties with relatively high brackish water availability, hydraulic
28   fracturing water use represented a much smaller percentage of total water availability (fresh + brackish +
29   wastewater) (see Figure 4-5b). This finding illustrates that potential impacts can be avoided or reduced in
30   these counties through the use of brackish water or wastewater for hydraulic fracturing; a case study in the
31   Eagle Ford play in southwestern Texas confirms this (see Text Box 4-3).

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**Text Box 4.2 (continued): Hydraulic Fracturing Water Use as a Percentage of Water Availability Estimates.**
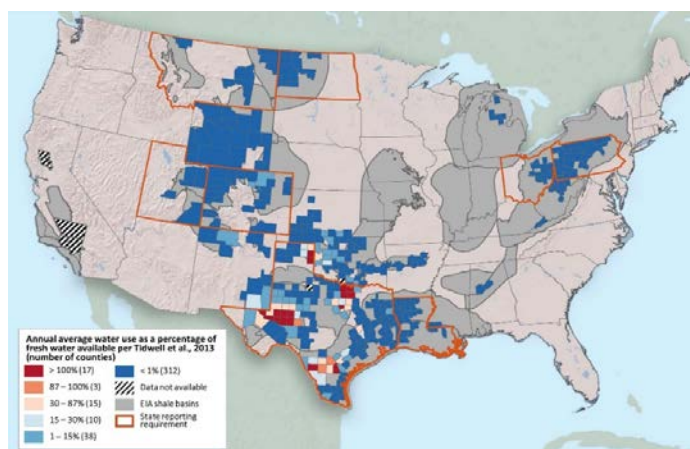


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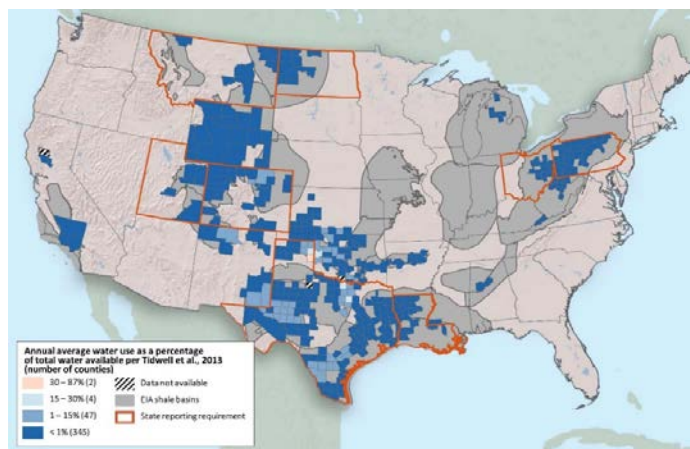


b

**Figure 4-5. Annual average 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, 2015b). Orange borders identify states that required some degree of reporting to FracFocus 1.0 in 2011 and 2012. Data from U.S. EPA (2015b) 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 (a). The total water available for California is the sum of brackish water plus wastewater only (b).

1    Surface water availability is generally low in western and southern Texas (Figure 4-6a), and both
2    fracturing operations and residents rely heavily on ground water (Figure 4-6b). Similar to trends
3    nationally, ground water aquifers in Texas have experienced substantial declines caused by
4    withdrawals (Konikow, 2013b; TWDB, 2012; George et al., 2011). Ground water in the Pecos Valley,
5    Gulf Coast, and Ogallala aquifers in southern and western Texas is estimated to have declined by
6    roughly 5, 11, and 43 cubic miles (21, 45.5, and 182 cubic kilometers), respectively, between 1900
7    and 2008 (Konikow, 2013b).[1] The Texas Water Development Board expects ground water supply in
8    the major aquifers to decline by 30% between 2010 and 2060, mostly due to declines in the
9    Ogallala aquifer (TWDB, 2012).[2] Irrigated agriculture is by far the dominant user of water from the
10   Ogallala aquifer (USGS, 2009), but fracturing operations, along with other uses, now contribute to
11   the aquifer's depletion.

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[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. Ground water depletion from the Carrizo-Wilcox aquifer is included in the estimate for the southern portion of the Gulf Coast aquifer (Konikow, 2013b).

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

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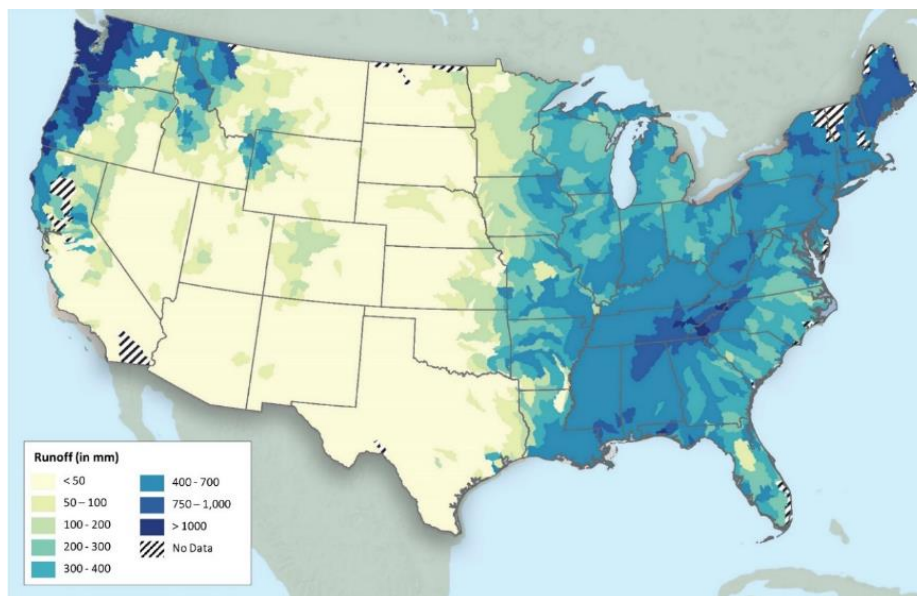
**(a)**
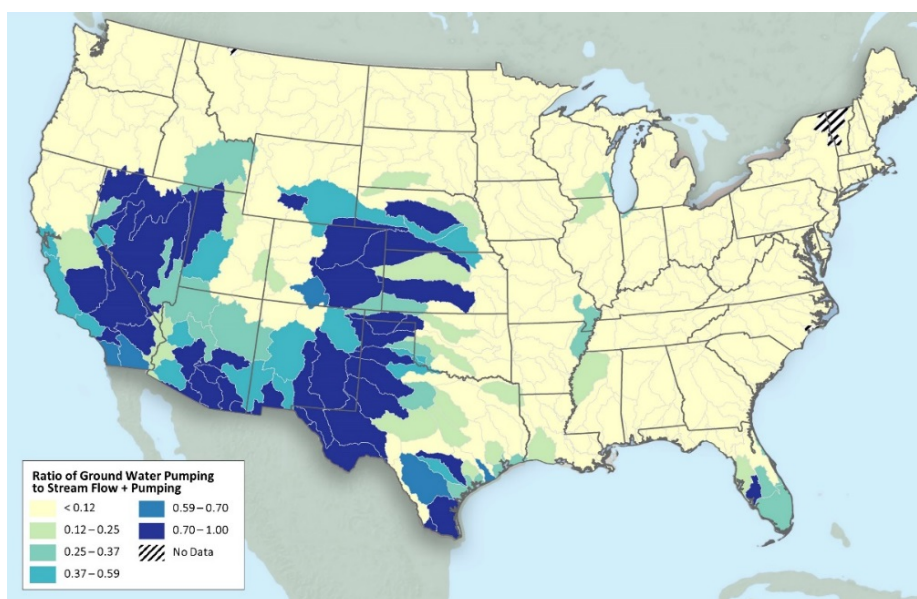


**(b)**

**Figure 4-6. (a) Estimated annual surface water runoff from the USGS; (b) Reliance on ground water as indicated by the ratio of ground water pumping to stream flow and pumping.**

Estimates for Figure 4-6a 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, 2014g). Higher ratios (darker blues) in Figure 4-6b indicate greater reliance on ground water. Figure redrawn from Tidwell et al. (2012), using data provided by the U.S. Department of Energy's Sandia National Laboratories on December 12, 2014.

1   Extensive ground water pumping can induce vertical mixing of high-quality ground water with
2   recharge water from the land surface that has been contaminated by nitrate or pesticides, or with
3   lower-quality ground water from underlying geologic formations (USGS, 2009; Konikow and Kendy,
4   2005). Ground water quality degradation associated with aquifer pumping is well documented in
5   the southern portion of the Ogallala aquifer in the Texas panhandle. The quality of ground water
6   used by many private, public supply, and irrigation wells is poorest in the aquifer's southern
7   portion, with elevated concentrations of TDS, chloride, nitrate, fluoride, manganese, arsenic, and
8   uranium (Chaudhuri and Ale, 2014a; USGS, 2009, 2007). Elevated levels of these constituents result
9   from both natural processes and human activities such as ground water pumping (Chaudhuri and
10  Ale, 2014a; USGS, 2009). Similar patterns of ground water quality degradation (i.e., salinization and
11  contamination) have also been observed in other Texas aquifers.[1]

12  Ground water withdrawals for hydraulic fracturing, along with irrigation and other uses, may
13  contribute to water quality degradation associated with intensive aquifer pumping in western and
14  southern Texas. Areas with numerous high-capacity wells and large amounts of sustained ground
15  water pumping are most likely to experience ground water quality degradation associated with
16  withdrawals (USGS, 2009, 2007). Given that Texas is prone to drought conditions, ground water
17  recharge is limited, making the already declining aquifers in southern and western Texas especially
18  vulnerable to further ground water depletion and resulting potential impacts to ground water
19  quality (USGS, 2009; Jackson et al., 2001).

20  This survey of the available literature and data points to the potential for impacts in southern and
21  western Texas, but generally does not indicate whether impacts will occur at the local scale around
22  specific withdrawal points. An exception is a case study in the Eagle Ford play of southwestern
23  Texas that compared water demand for hydraulic fracturing with water supplies at the scale of the
24  play, county, and one square mile (Scanlon et al., 2014). The authors observed generally adequate
25  water supplies for hydraulic fracturing, except in specific locations, where they found excessive
26  drawdown of local ground water in a small proportion (~6% of the area) of the Eagle Ford play
27  (see Text Box 4-3).

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[1] Persistent salinity has also been observed in west Texas, specifically in the southern Ogallala, northwest Edwards-
Trinity (plateau), and Pecos Valley aquifers, largely due to prolonged irrigational ground water pumping and ensuing
alteration of hydraulic gradients leading to ground water mixing (Chaudhuri and Ale, 2014b). High levels of ground water
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, 2013b; 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).

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

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

5  Scanlon et al. (2014) compared water demand for hydraulic fracturing currently and over the projected play
6  life (20 years) relative to water supply from ground water recharge, ground water storage (brackish and
7  fresh), and stream flow. Using detailed ground water availability models developed by the Texas Water
8  Development Board, they reported that water demand for hydraulic fracturing in 2013 was 30% of annual
9  ground water recharge in the play area, and over the 20-year play lifespan it was projected to be 26% of
10  groundwater recharge, 5-8% of fresh groundwater storage, and 1% of brackish ground water storage. The
11  dominant water user in the play is irrigation (62 to 65% of water use, 53 to 55% of consumption), as
12  compared with hydraulic fracturing (13% of water use and 16% of consumption). At the county level,
13  projected water demand for hydraulic fracturing over the 20-year period was low relative to freshwater
14  supply (ranging from 0.6-27% by county, with an average of 7.3%). Similarly, projected total water demand
15  from all uses was low relative to supply, excluding two counties with high irrigation demands (Frio, Zavala),
16  and one county with no known ground water supplies (Maverick).

17  Although supply was found to be sufficient even in this semi-arid region, there were important caveats
18  especially at sub-county scales. The researchers found no water level declines over much of the play area
19  assessed (69% of the play area), yet in some areas they estimated ground water drawdowns of up to 50 feet
20  (12% of the play area), and in others of 100 feet or more (6% of the play area). This was corroborated with
21  well monitoring data that showed a sharp decline in water levels in several ground water monitoring wells
22  after hydraulic fracturing activity increased in 2009. The researchers concluded that any impacts in these
23  locations could be minimized if brackish ground water were used. Projected hydraulic fracturing water use
24  represents less than 1% of total brackish ground water storage in the play area. By contrast, they concluded
25  there is limited potential for reuse of wastewater in this play because of small volumes available (less than or
26  equal to 5% of hydraulic fracturing water requirements).

27  The potential for water quantity and quality effects appears to be lower in north-central and
28  eastern Texas, in areas including the Barnett and Haynesville plays. Residents obtain water for
29  domestic use—which includes use of water for drinking—from a mixture of ground water and
30  surface water sources (see Appendix Table B-6). Counties encompassing Dallas and Fort Worth rely
31  mostly on publically-supplied surface water (TWDB, 2012) (see Appendix Table B-6).

32  Although the Trinity, the major aquifer in northeast Texas, is projected to decline only slightly
33  between 2010 and 2060 (TWDB, 2012), Bene et al. (2007) estimate that hydraulic fracturing
34  ground water withdrawals will increase from 3% of total ground water use in 2005 to 7%–13% in
35  2025, suggesting the potential for localized aquifer drawdown and potential impacts to water
36  quality. Additionally, ground water quality degradation associated with aquifer drawdown has been
37  documented in the Trinity and Woodbine aquifers underlying much of the Barnett play, with both
38  aquifers showing high levels of salinization (Chaudhuri and Ale, 2013).

1 Overall, the potential for impacts appears higher in western and southern Texas, compared to the
2 northeast part of the state. Impacts are likely to be localized drawdowns of ground water, as shown
3 by a detailed case study of the Eagle Ford play (see Text Box 4-3). Scanlon et al. (2014) suggested
4 that a shift towards brackish water use could minimize potential future impacts to fresh water
5 resources. This finding is consistent with our county-level data (see Text Box 4-2).

### 4.5.2. Colorado and Wyoming

6 Colorado had the second highest number of disclosures in the EPA FracFocus project database,
7 (13% of disclosures) (see Figure 4-3 and Appendix Table B-5). We combine Colorado and Wyoming
8 because of their shared geology of the Denver Basin (including the Niobrara play) and the Greater
9 Green River Basin (see Figure 4-7). There are three major basins reported for Colorado: the Denver
10 Basin; the Uinta-Piceance Basin; and the Raton Basin. Together these basins contain 99% of
11 reported wells in the state, although the bulk of the activity in Colorado is in the Denver Basin (see
12 Appendix Table B-5). Fewer wells (roughly 4% of disclosures) are present in Wyoming. There are
13 two major basins reported for Wyoming (Greater Green River and Powder River) that together
14 contain 86% of activity in the state (see Appendix Table B-5).
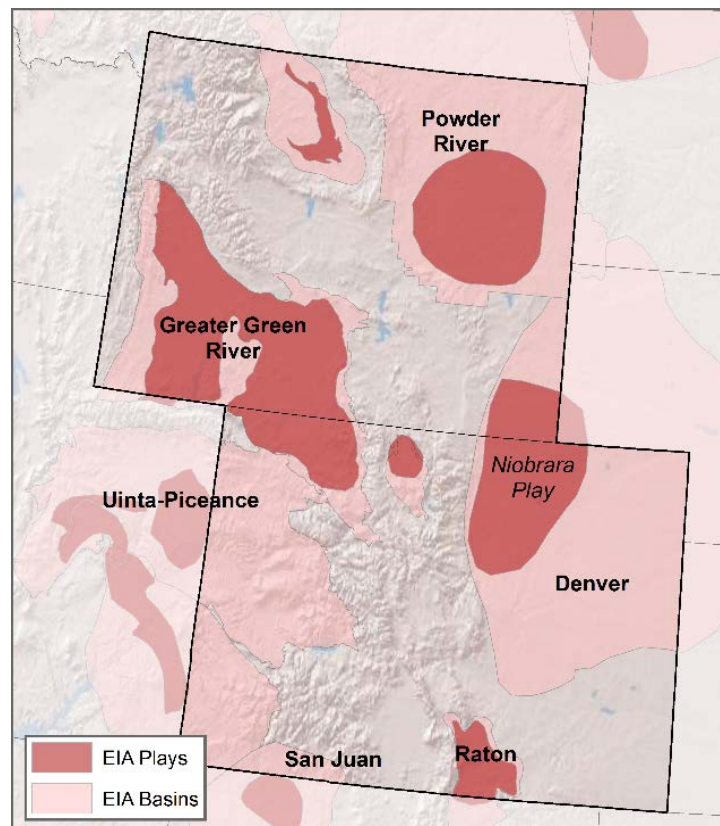


**Figure 4-7. Major U.S. EIA shale plays and basins for Colorado and Wyoming (EIA, 2015).**

Source: (EIA, 2015b)

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1   *Types of water used:* Water for hydraulic fracturing in Colorado and Wyoming comes from both
2   ground water and surface water, as well as reused wastewater (Colorado Division of Water
3   Resources; Colorado Water Conservation Board; Colorado Oil and Gas Conservation Commission,
4   2014; BLM, 2013b). The only publicly available information on water sources for each state is a list
5   of potential sources; it does not appear that either state provides more specific information on
6   water sources for hydraulic fracturing. In the Uinta-Piceance Basin of northwestern Colorado, the
7   EPA (2015c) reports that most of the fresh water used for fracturing comes from surface water,
8   although fresh water sources make up a small proportion of the total water used. In the Denver
9   Basin (Niobrara play) of southeastern Wyoming, qualitative information suggests that ground
10  water supplies much of the water used for fracturing, although no data were available to
11  characterize the ratio of ground water to surface water withdrawals (AMEC, 2014; BLM, 2013b;
12  Tyrrell, 2012).

13  Non-fresh water sources (e.g., industrial and municipal wastewater, brackish ground water, and
14  reused hydraulic fracturing wastewater) are sometimes listed as potential alternatives to fresh
15  water for fracturing in both Colorado and Wyoming (Colorado Division of Water Resources;
16  Colorado Water Conservation Board; Colorado Oil and Gas Conservation Commission, 2014; BLM,
17  2013b); no data are available to show the extent to which these non-fresh water sources are used at
18  the state or basin level. In northwest Colorado's Garfield County (Uinta-Piceance Basin), the EPA
19  (2015c) reports that fresh water is used solely for drilling and that reused wastewater supplies
20  nearly all the water for hydraulic fracturing (see Table 4-1). This estimate of reused wastewater as
21  a percentage of injected volume is markedly higher than in other locations and results from the
22  geologic characteristics of the Piceance tight sand formation, which has naturally high water
23  content and produces large volumes of relatively high-quality wastewater (U.S. EPA, 2015c).

24  In contrast, a study by Goodwin et al. (2014) assumed no reuse of wastewater for hydraulic
25  fracturing operations by Noble Energy in the Denver-Julesburg Basin of northeastern Colorado (see
26  Table 4-1). It is unclear whether this assumption is indicative of reuse practices of other companies
27  in the Denver-Julesburg Basin. The difference in reused wastewater rates reported by the EPA
28  (2015c) and Goodwin et al. (2014) may indicate an east-west divide in Colorado (i.e., low reuse in
29  the east versus high reuse in the west), due at least in part to differences in wastewater volumes
30  available for reuse. However, further information is needed to adequately characterize reuse
31  patterns in Colorado.

32  *Water Use per Well:* Water use per well varies across Colorado, with median values of 1.8 million,
33  400,000, and 96,000 gal (6.8 million, 1.5 million, and 363,000 L) in the Uinta-Piceance, Denver, and
34  Raton Basins, respectively according to the EPA FracFocus project database (see Appendix Table B-
35  5). Low water volumes per well are reported in Wyoming (see Appendix Table B-5). Low volumes
36  reported for the Raton Basin of Colorado and the Powder River Basin of Wyoming are due to the
37  prevalence of CBM extraction in these locations (U.S. EPA, 2015l; USGS, 2014d).

38  More difficult to explain are the low volumes reported for the Denver Basin in the EPA FracFocus
39  project database. These values are lower than any other non-CBM basin reported in Appendix Table
40  B-5. Goodwin et al. (2014) report much higher water use per well in the Denver Basin, with a

1   median of 2.8 million gal (10.6 million L) (although only usage for the Wattenberg Field was
2   reported). Indeed, the 10th–90th percentiles (2.4–3.8 million gal) (9.1 to 14.4 million L) from
3   Goodwin et al. (2014) are almost completely above those from the EPA FracFocus project database
4   for the Denver Basin (see Appendix Table B-5).[1] It is difficult to draw clear conclusions because of
5   differences in scale (i.e., field in Goodwin versus basin in the project database) and operators (i.e.,
6   Noble Energy in Goodwin versus all in the project database). However, it seems plausible that the
7   EPA FracFocus project database may be incomplete for estimating the amount of water used per
8   well in the Denver Basin.

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

13  *Cumulative water use/consumption:* Hydraulic fracturing operations in Colorado cumulatively use
14  billions of gallons of water, but this amount is a small percentage compared to total water used or
15  consumed at the county scale. Operators in both Garfield and Weld Counties, located in the Uinta-
16  Piceance and Denver Basins, respectively, use more than 1 billion gal (3.8 billion L) annually.
17  Fracturing water use and consumption in these counties exceed those in all other Colorado counties
18  combined (see Appendix Table B-2), but the water used for hydraulic fracturing in Garfield and
19  Weld counties is less than 2% and 3% compared to 2010 total water use and consumption,
20  respectively. In comparison, irrigated agriculture accounts for over 90% of the water used in both
21  counties (Maupin et al., 2014; Kenny et al., 2009). Overall, hydraulic fracturing accounts for less
22  than 2% compared to 2010 total water use in all Colorado counties represented in the EPA
23  FracFocus project database (see Appendix Table B-2). Water use estimates based on the EPA
24  FracFocus project database may be low relative to literature and state estimates (Text Box 4-1), but
25  even if estimates from the project database were doubled, hydraulic fracturing water use and
26  consumption would still be less than 4% and 5.5% compared to 2010 total water use and
27  consumption, respectively, in each Colorado county.

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

33  The Colorado Division of Water Resources et al. (2014) project that annual water use for hydraulic
34  fracturing in the state will increase by approximately 16% between 2012 and 2015, but demand in

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[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 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.

1    later years is unclear. Even with an increase of 16% or more, hydraulic fracturing would still
2    remain a relatively small user of water at the county scale in Colorado.

3    *Potential for impacts:* The potential for water quantity and quality impacts appears to be low at the
4    county scale in Colorado and Wyoming, because fracturing accounts for a low percentage of total
5    water use and consumption (see Figure 4-2a,b). This conclusion is also supported by the
6    comparison of hydraulic fracturing water use to water availability at the county scale (see Text Box
7    4-2 and Figure 4-5a,b). However, counties in Colorado and Wyoming may be too large to detect the
8    potential for impacts, and local scale studies help provide details at a finer resolution. In a multi-
9    scale case study in western Colorado, the EPA (2015c) also did not observe any impacts in the
10   Upper Colorado River Basin. Due to the high reuse rate of wastewater, they did not identify any
11   locations where fracturing currently contributed to locally high water use intensity. They did
12   conclude, however, that future water use effects were possible (see Text Box 4-4).

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

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

20   They found that water supplies accessed for oil and gas demand were concentrated in Garfield County, and most fresh
21   water withdrawals were concentrated within the Parachute Creek watershed (198 mi$^2$). However, fresh water makes up a
22   small proportion of the total water used for fracturing due to large quantities of high-quality wastewater produced from
23   the Piceance tight sands. Fresh water is used only for drilling, and the water used for fracturing is reported to be 100%
24   reused wastewater (see Table 4-1). Due to the high reuse rate, The EPA (2015c) did not identify any locations in the
25   Piceance play where fracturing contributed to locally high water use intensity.

26   Scenario analyses demonstrated a pattern of increasing potential impact with decreasing watershed size in the UCRB. The
27   EPA (2015c) examined hydraulic fracturing water use intensity under the current rates of both directional (S-shaped) and
28   horizontal drilling. They showed that for the more water-intensive horizontal drilling, watersheds had to be larger to
29   meet the same index of water use intensity (0.4) as that for directional drilling (100 mi$^2$ for horizontal drilling, as
30   compared to 30 mi$^2$ for directional drilling). To date, most wells have been drilled directionally into the Piceance tight
31   sands, although a trend toward horizontal drilling is expected to increase annual water use per well by about 4 times.
32   Despite this increase, total hydraulic fracturing water use is expected to remain small relative to other users. Currently,
33   irrigated agriculture is the largest water user in the UCRB.

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

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1  East of the Rocky Mountains in the Denver Basin, sub-county effects may be possible given the
2  combination of high hydraulic fracturing activity and low water availability, but lack of available
3  data and literature at this scale limits our ability to assess the potential for impacts in this location.
4  Ceres (2014) concludes that all fractured wells in the Denver Basin are in high or extremely high
5  water-stressed areas. Furthermore, the development of the Niobrara Shale in southeast Wyoming
6  occurs in areas already impacted by high agricultural water use from the Ogallala aquifer, including
7  the state's only three ground water control areas, which were established as management districts
8  in the southeast portion of the state in response to declining ground water levels (AMEC, 2014;
9  Wyoming State Engineer's Office, 2014; Tyrrell, 2012; Bartos and Hallberg, 2011). Ground water
10 withdrawals for hydraulic fracturing may have the potential to contribute to water quality
11 degradation particularly in these areas.

12 Overall, the potential for impacts appears low at the county scale in Colorado and Wyoming, but
13 sub-county effects may be possible particularly east of the Rocky Mountains in the Denver Basin.
14 Lack of available data and literature at the local scale limits our ability to assess the potential for
15 impacts in this location.

### 4.5.3. Pennsylvania, West Virginia, and Ohio

16 Pennsylvania had the third most disclosures in the EPA FracFocus project database (6.5% of
17 disclosures) (see Appendix Table B-5 and Figure 4-3). We combine West Virginia and Ohio with
18 Pennsylvania because they share similar geology overlying the Appalachian Basin (including the
19 Marcellus, Devonian, and Utica stacked plays) (see Figure 4-8); however, much less activity is
20 reported in these two states (see Appendix Table B-5).
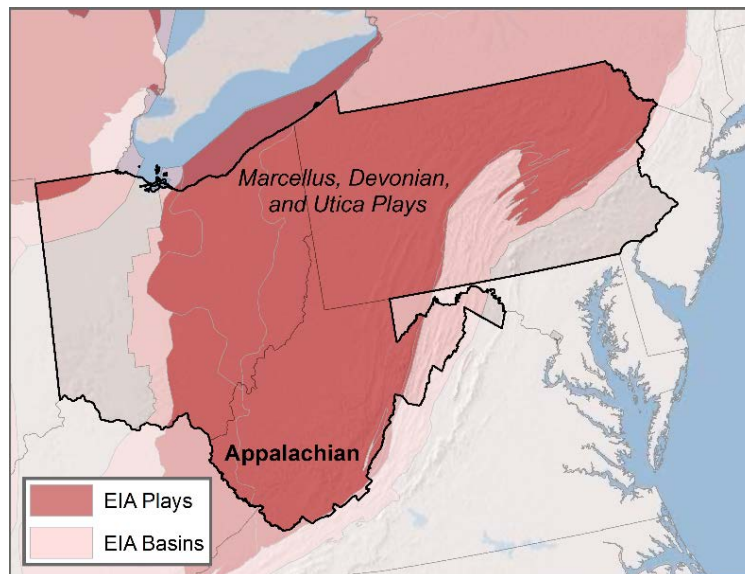


**Figure 4-8. Major U.S. EIA shale plays and basins for Pennsylvania, West Virginia, and Ohio (EIA, 2015).**

Source: (EIA, 2015b).

1　*Types of water used*: Surface water is the primary water source for hydraulic fracturing in
2　Pennsylvania, West Virginia, and Ohio (Mitchell et al., 2013a; SRBC, 2013; West Virginia DEP, 2013;
3　Ohio EPA, 2012b). Available data for Pennsylvania are specific to the Susquehanna River Basin
4　(SRB), where hydraulic fracturing water is sourced mostly from surface water (SRBC, 2013) (see
5　Table 4-3). The industry also uses mostly surface water in West Virginia (West Virginia DEP, 2014,
6　2013) (see Table 4-3). Although specific data are not available, state reports indicate that most
7　water for hydraulic fracturing in Ohio's Marcellus or Utica Shale formations is sourced from nearby
8　surface water bodies (Ohio EPA, 2012b; STRONGER, 2011b).

9　Given that surface water is the primary water source, the water used for hydraulic fracturing is
10　most often fresh water in all three states. In both Pennsylvania's SRB and throughout West Virginia,
11　most water for hydraulic fracturing is self-supplied via direct withdrawals from surface water and
12　ground water (U.S. EPA, 2015a; West Virginia DEP, 2013). Operators also purchase water from
13　public water systems, which may include a variety of commercial water brokers (West Virginia
14　DEP, 2014; SRBC, 2013; West Virginia DEP, 2013). Municipal supplies may be used as well,
15　particularly in urban areas of Ohio (STRONGER, 2011b).

16　Reused hydraulic fracturing wastewater accounted for an estimated 18% and 15% of total water
17　used for fracturing in 2012 in Pennsylvania's SRB and West Virginia, respectively (West Virginia
18　DEP, 2014; Hansen et al., 2013; SRBC, 2013) (see Table 4-1). Available data indicate increased reuse
19　of wastewater over time in this region likely due to the lack of nearby disposal options; from 2010-
20　2012 reused wastewater as a percentage of injected water volume ranged from 10% to 18% and
21　6% to 15% in Pennsylvania's SRB and West Virginia, respectively (West Virginia DEP, 2014; Hansen
22　et al., 2013). In Ohio's Marcellus and Utica Shales, reuse of wastewater is reportedly uncommon
23　(STRONGER, 2011b), potentially due to the prevalence of disposal wells in Ohio (see Chapter 8).

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

29　*Water Use per Well*: Operators in these three states reported the third, fourth, and fifth highest
30　median water use nationally in the EPA FracFocus project database, with 5.0, 4.2, and
31　3.9 million gal (18.9, 15.9, and 14.8 million L) per well in West Virginia, Pennsylvania, and Ohio,
32　respectively (U.S. EPA, 2015b) (see Appendix Table B-5). Hansen et al. (2013) report similar water
33　use estimates for Pennsylvania and West Virginia (see Appendix Table B-5). This correspondence is
34　not surprising, as these estimates are also based on FracFocus data (via Skytruth). For 2011, the
35　year overlapping with the time frame of the EPA FracFocus report (U.S. EPA, 2015a), Mitchell et al.
36　(2013a) report an average of 2.3 million gal (8.7 million L) for vertical wells (62 wells) and
37　4.6 million gal (17.4 million L) for horizontal wells (612 wells) in the Pennsylvania portion of the
38　Ohio River Basin, based on records from PA DEP. The weighted average water use per well was
39　4.4 million gal (16.7 million L), similar to results based on the EPA FracFocus project database
40　listed above.

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1   *Cumulative water use/consumption:* In this tri-state region, highest cumulative water use for
2   hydraulic fracturing is in northeastern Pennsylvania counties. On average, operators in Bradford
3   County reported over 1 billion gal (3.8 billion L) used annually in 2011 and 2012 for fracturing;
4   operators in three other counties (Susquehanna, Lycoming, and Tioga Counties) cumulatively
5   reported 500 million gal (1.9 billion L) or more used annually (see Table 4-2). On average,
6   hydraulic fracturing water use is 3.2% compared to 2010 total county water use for counties with
7   disclosures in the EPA FracFocus project database in these three states (see Table 4-2 and
8   Appendix Table B-2). Susquehanna County in Pennsylvania has the highest percentages relative to
9   2010 total water use (47%) and consumption (123%).

10  *Potential for impacts*: Water availability is higher in Pennsylvania, West Virginia, and Ohio than in
11  many western states, reducing the likelihood of impacts to drinking water quantity and quality. At
12  the county scale, water supplies appear adequate to accommodate this use (Tidwell et al., 2013)
13  (see Text Box 4-2 and Figure 4-5a,b).

14  However, impacts could still occur at specific withdrawal points. In a second, multi-scale case study,
15  EPA researchers concluded that individual streams in this region can be vulnerable to typical
16  hydraulic fracturing water withdrawals depending on stream size, as defined by contributing basin
17  area (U.S. EPA, 2015c) (see Text Box 4-5). They observed infrequent (in less than 1% of
18  withdrawals) high ratios of hydraulic fracturing water consumption to stream flow (high
19  consumption-to-stream flow events). Passby flows can reduce the frequency of high consumption-
20  to-stream flow events, particularly in the smallest streams (U.S. EPA, 2015c).[1]

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[1] A passby flow is a prescribed, low stream flow threshold below which withdrawals are not allowed. The SRBC uses passby flows to protect streams in the Susquehanna River Basin, an area including much of eastern Pennsylvania (U.S. EPA, 2015c).

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

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

8   Most water for fracturing in the SRB is self-supplied from rivers and streams with withdrawal points
9   distributed throughout a wide geographic area. Public water systems provide a relatively small proportion of
10  the water needed. Reuse of wastewater makes up approximately 13% to 18% of injected fluid volume on
11  average, as reported by the EPA ([2015c](#)) for 2008 to 2011 and [Hansen et al. (2013)](#) for 2012, respectively
12  (see Table 4-1). The Susquehanna River Basin Commission (SRBC) regulates water acquisition for hydraulic
13  fracturing and issues permits that set limits on the volume, rate, and timing of withdrawals at individual
14  withdrawal points; passby flow thresholds halt water withdrawals during low flows.

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

23  Without management of the rate and timing of withdrawals, surface water withdrawals for
24  hydraulic fracturing have the potential to affect both water quantity and quality ([Mitchell et al.,](#)
25  [2013a](#)). Potential effects are generally applicable, but are especially relevant in this region because
26  surface water is the primary water source for hydraulic fracturing in Pennsylvania, West Virginia,
27  and Ohio. Of greatest concern are small, unregulated streams, particularly under drought
28  conditions or during seasonal low flows ([U.S. EPA, 2015c](#); [Vengosh et al., 2014](#); [Mitchell et al.,](#)
29  [2013a](#); [Vidic et al., 2013](#); [Rahm and Riha, 2012](#); [Rolls et al., 2012](#); [Kargbo et al., 2010](#); [McKay and](#)
30  [King, 2006](#)). Surface water quality impacts may be of concern if a pollution discharge point (e.g.,
31  sewage treatment plant, agricultural runoff, or chemical spill) is immediately downstream of a
32  hydraulic fracturing withdrawal ([U.S. EPA, 2015c](#); [NYSDEC, 2011](#)).[1] Water quality impacts

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[1] 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](#)).

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1 associated with reduced water levels may also include possible interference with the efficiency of
2 drinking water treatment plant operations, as increased contaminant concentrations in drinking
3 water sources may necessitate additional treatment and ultimately impact drinking water quality
4 (Water Research Foundation, 2014; Benotti et al., 2010).[1]

5 Overall, there appears to be adequate surface water for hydraulic fracturing, but there is the
6 potential for impacts to both drinking water quantity and quality, particularly in small streams, if
7 withdrawals are not managed (U.S. EPA, 2015c).

### 4.5.4.  North Dakota and Montana

8 North Dakota was fourth in the number of disclosures in the EPA FracFocus project database (5.9%
9 of disclosures) (see Appendix Table B-5 and Figure 4-3). We combine Montana with North Dakota
10 because both overlie the Williston Basin (which contains the Bakken play, shown in Figure 4-9),
11 although many fewer wells are reported for Montana (see Appendix Table B-5). The Williston Basin
12 is the only basin with significant activity reported for either state, though other basins are also
13 present in Montana (e.g., the Powder River Basin).
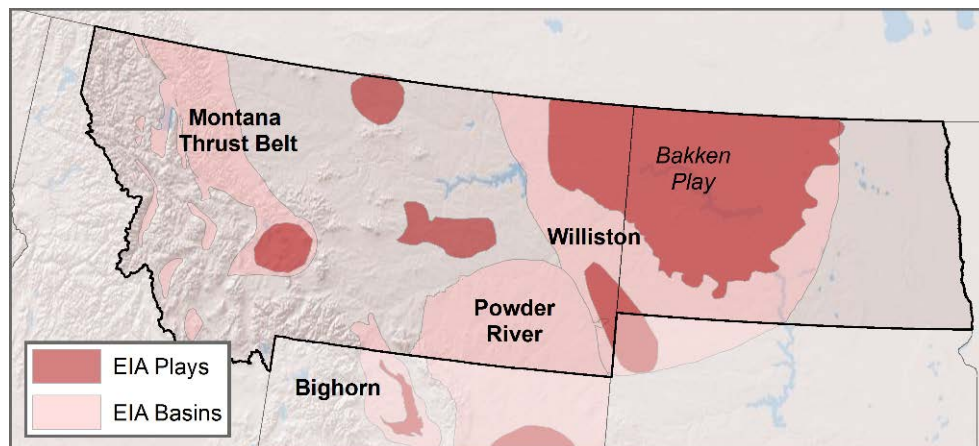


**Figure 4-9. Major U.S. EIA shale plays and basins for North Dakota and Montana (EIA, 2015b).**

Source: (EIA, 2015b).

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[1] 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 react with organic matter in the water being treated (Water Research Foundation, 2014).

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1    *Types of water used:* Hydraulic fracturing of the Bakken play underlying much of western North
2    Dakota and northeastern Montana depends on both ground and surface water resources. Surface
3    water from the Missouri River system provides the largest source of fresh water in the center of
4    Bakken oil development (North Dakota State Water Commission, 2014; EERC, 2011, 2010; North
5    Dakota State Water Commission, 2010). Apart from the Missouri River system, regional surface
6    waters (i.e., small streams) do not provide a consistent supply of water for the oil industry due to
7    seasonal stream flow variations. Sufficient stream flows generally occur only in the spring after
8    snowmelt (EERC, 2011). Ground water from glacial and bedrock aquifer systems has traditionally
9    supplied much of the water needed for Bakken development, but concerns over limited ground
10   water supplies have led to limits on the number of new ground water withdrawal permits issued
11   (Ceres, 2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission,
12   2010).

13   The water used for Bakken development is described as mostly fresh. The EPA FracFocus report
14   shows that "fresh" was the only source of water listed in almost all disclosures reporting a source of
15   water in North Dakota (U.S. EPA, 2015a).[1] Reuse of Bakken wastewater is limited due to its quality
16   of high TDS, which presents challenges for treatment and reuse. However, the industry is
17   researching treatment technologies for reuse of this wastewater (Ceres, 2014; EERC, 2013, 2011).

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

21   *Water Use per Well*: Water use per well is intermediate compared with other areas, with a median
22   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
23   Montana, respectively according to the EPA's FracFocus project database (see Appendix Table B-5).
24   The North Dakota State Water Commission reports similar volumes (2.2 million gal (8.3 million L)
25   per well on average for North Dakota) in a summary fact sheet (North Dakota State Water
26   Commission, 2014).[2]

27   A presentation by the North Dakota Department of Mineral Resources (NDDMR) suggests that
28   Bakken wells require an average of 600 gal (2,300 L) per day of "maintenance water" in addition to
29   the initial water for hydraulic fracturing (North Dakota Department of Mineral Resources, 2013).[3]
30   This extra water is reportedly needed because of the relatively high salt content of Bakken brine,
31   potentially leading to salt buildup, pumping problems, and restriction of oil flow. According to the
32   NDDMR, maintenance water can contribute to large additional volumes over a typical well life span
33   (6.6–8.8 million gal (25-33 million L) over 30–40 years). It is unclear whether this phenomenon is
34   restricted to the Bakken play.

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[1] However, 25% of North Dakota disclosures included information related to water sources (U.S. EPA, 2015a).
[2] The fact sheet is a stand-alone piece, and it is not accompanied by an underlying report.
[3] The NDDMR's presentation that mentions the issue of maintenance water was later picked up and reported on by *National Geographic* (http://news.nationalgeographic.com/news/energy/2013/11/131111-north-dakota-wells-maintenance-water/) and by Ceres (2014). Peer-reviewed studies on the Bakken also report on maintenance water (e.g., Scanlon et al., 2014), but they refer to the same original sources.

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1  *Cumulative water use/consumption:* Cumulative water use for fracturing in this region is greatest in
2  the northwestern corner of North Dakota. In counties with 2011 and 2012 disclosures to FracFocus,
3  fracturing water use averaged approximately 123 million gal (466 million L) per county annually in
4  the two-state area, with use in McKenzie and Williams Counties in North Dakota exceeding
5  500 million gal (1.9 billion L) per year (see Appendix Table B-2). There are four counties where
6  2011 and 2012 average hydraulic fracturing water use was 10% or more of 2010 total water use.
7  Mountrail and Dunn Counties showed the highest percentages. Outside of North Dakota's northwest
8  corner, the rest of the state and Montana showed little cumulative water use from hydraulic
9  fracturing (see Table 4-2 and Appendix Table B-2).

10  *Potential for impacts:* In this region, there are concerns about over-pumping ground water
11  resources, but the potential for impacts appears to be low provided the Missouri River is
12  determined to be a sustainable and usable source. This finding of a low potential for impacts is also
13  supported by the comparison of hydraulic fracturing water use to water availability at the county
14  scale (see Text Box 4-2 and Figure 4-5a,b.) This area is primarily rural, interspersed with small
15  towns. Residents use a mixture of surface water and ground water for domestic use depending on
16  the county, with most water supplied by local municipalities (see Appendix Table B-6).

17  The state of North Dakota and the U.S. Army Corps of Engineers concluded that ground water
18  resources in western North Dakota are not sufficient to meet the needs of the oil and gas industry
19  (U.S. Army Corps of Engineers, 2011; North Dakota State Water Commission, 2010). All users
20  combined currently withdraw approximately 6.2 billion gal (23.5 billion L) of water annually in an
21  11-county region in western North Dakota, already stressing ground water supplies (U.S. Army
22  Corps of Engineers, 2011). By contrast, the total needs of the oil and gas industry are projected to
23  range from approximately 2.2 and 8.8 billion gal (8.3 and 33.3 billion L) annually by the year 2020
24  (U.S. Army Corps of Engineers, 2011).

25  Due to concerns for already stressed ground water supplies, the state of North Dakota limits
26  industrial ground water withdrawals, particularly from the Fox Hills-Hell Creek aquifer (Ceres,
27  2014; Plummer et al., 2013; EERC, 2011, 2010; North Dakota State Water Commission, 2010).
28  Currently, the oil industry is the largest industrial user of water from the Fox Hills-Hell Creek
29  aquifer in western North Dakota (North Dakota State Water Commission, 2010). Many farms,
30  ranches, and some communities in western North Dakota rely on flowing wells from this artesian
31  aquifer, particularly in remote areas that lack electricity for pumping; however, low recharge rates
32  and prolonged withdrawals throughout the last century have resulted in steady declines in the
33  formation's hydraulic pressure (North Dakota State Water Commission, 2010). Declines in
34  hydraulic pressure do not appear to be associated with impacts to ground water quality; rather, the
35  state is concerned with maintaining flows for users through conservation (North Dakota State
36  Water Commission, 2010).

37  To reduce pressure on ground water, the state is encouraging the industry to seek surface water
38  withdrawals from the Missouri River system, which if used, may be an adequate resource. The
39  North Dakota State Water Commission concluded the Missouri River and its dammed reservoir,
40  Lake Sakakawea, are the only plentiful and dependable water supplies for the oil industry in

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1 western North Dakota (North Dakota State Water Commission, 2010). In 2011, North Dakota
2 authorized the Western Area Supply Project, by which Missouri River water (via the water
3 treatment plant in Williston, North Dakota) will be supplied to help meet water demands, including
4 for oil and gas development, of the state's northwest counties (WAWSA, 2011). Industrial surface
5 water withdrawals are presently allowed in Lake Sakakawea on a temporary and controlled basis
6 while the U.S. Army Corps of Engineers conducts a multi-year study to determine whether surplus
7 water is available to meet the demands of regional municipal and industrial users (U.S. Army Corps
8 of Engineers, 2011).

### 4.5.5. Oklahoma and Kansas

9 Oklahoma had the fifth most disclosures in the EPA FracFocus project database (5.0% of
10 disclosures) (see Appendix Table B-5, and Figure 4-3). Three major basins— the Anadarko, which
11 includes the Woodford play; the Arkoma, which includes the Fayetteville play; and the Ardmore,
12 which includes the Woodford play—contain 67% of the disclosures in Oklahoma (see Figure 4-9
13 and Appendix Table B-5). Few wells were reported for Kansas (Kansas disclosures comprise 0.4%
14 of the EPA FracFocus project database), but because of the shared geology of the Cherokee Platform
15 across the two states, we group Kansas with Oklahoma. Oklahoma and Kansas were two of the
16 three states where a large fraction of wells were not associated with a basin defined by the U.S. EIA
17 (U.S. EPA, 2015b) (see Appendix Table B-5).[1]

18 *Types of water used:* Water for hydraulic fracturing in Oklahoma and Kansas comes from both
19 surface and ground water (Kansas Water Office, 2014; Taylor, 2012). Data on temporary water use
20 permits in Oklahoma (which make up the majority of water use permits for Oklahoma oil and gas
21 mining) show that, in 2011, approximately 63% and 37% of water for hydraulic fracturing came
22 from surface and ground water, respectively (Taylor, 2012) (see Table 4-3). General water use in
23 Oklahoma follows an east-west divide, with the eastern half dependent on surface sources and the
24 western half relying heavily on ground water (OWRB, 2014). Water obtained for fracturing is
25 assumed to fit this pattern as well. No data are available on the proportion of hydraulic fracturing
26 water that is sourced from surface versus ground water resources in Kansas.

27 For both Oklahoma and Kansas, no data are available to describe the extent to which reused
28 wastewater is used as a percentage of total injected volume. However, the quality of Oklahoma's
29 Woodford Shale wastewater has been described as low in TDS, and thus reuse could reduce the
30 demand for fresh water (Kuthnert et al., 2012).

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[1] Alaska was the other state in the EPA FracFocus project database where the U.S. EIA shale basins did not adequately describe well locations, with all 37 wells in Alaska not associated with a U.S. EIA basin. For all other states, U.S. EIA shale basins captured 86%–100% of the wells in the EPA FracFocus project database (U.S. EPA, 2015b).

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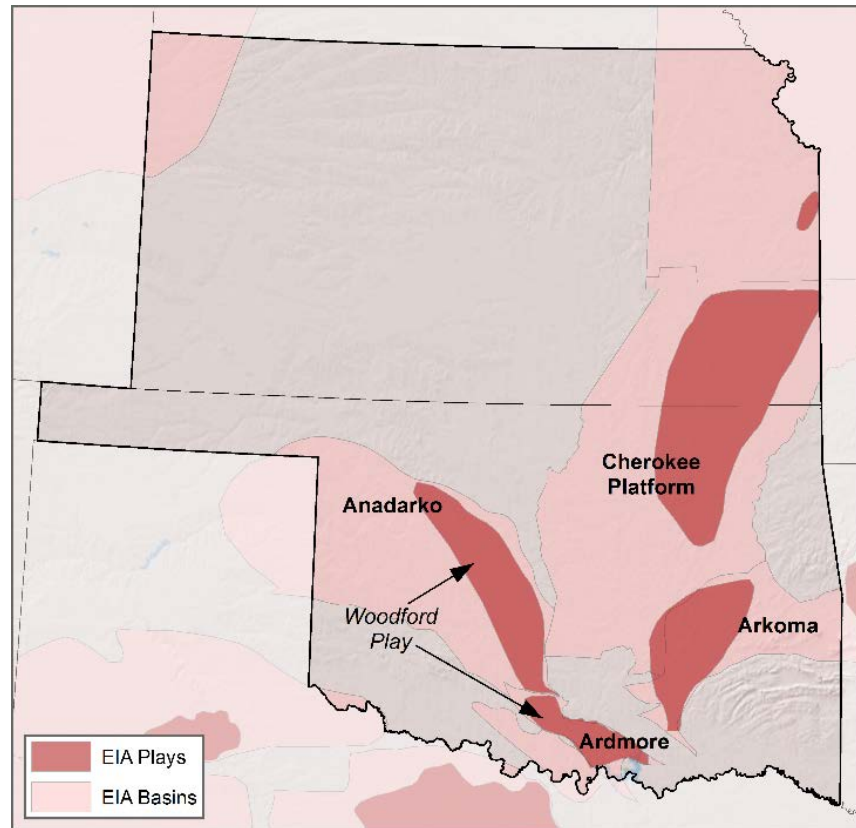
**Figure 4-10. Major U.S. EIA shale plays and basins for Oklahoma and Kansas (EIA, 2015).**

Source: (EIA, 2015b)

1   *Water Use per Well:* State-level estimates of median water use per well in Oklahoma include 2.6
2   million gal (9.8 million L) and 3 million gal (11 million L) [U.S. EPA (2015b) and, Murray (2013),
3   respectively]. Water use for hydraulic fracturing increased from 2000 to 2011, driven by volumes
4   required for fracturing horizontal wells across the state (Murray, 2013). Within the state there are
5   wide ranges in water use for different formations. According to the EPA FracFocus project database,
6   the Ardmore and Arkoma Basins of Oklahoma, had the highest median water use in the country,
7   with medians of 8.0 and 6.7 million gal (30.3 and 25.4 million L) per well, respectively; whereas the
8   Anadarko Basin had lower median water use per well and higher disclosure counts (3.3 million gal
9   (12.5 million L), 935 disclosures) (see Appendix Table B-5). Wells not associated with a U.S. EIA
10  basin had a median of 1.9 million gal (7.2 million L) per well (592 disclosures) (see Appendix Table
11  B-5). It is not clear why lower water volumes were reportedly used in unassociated wells, but
12  Oklahoma has several CBM deposits in the eastern part of the state where very low water use has
13  been reported (Murray, 2013). Median water use per well in Kansas was 1.5 million gal (5.7 million
14  L), focused mostly in a five-county area in the south-central and southwest portions of the state
15  (see Appendix Table B-5).

16  *Cumulative water use/consumption:* Cumulatively, operators reported using an average of
17  71.9 million gal (272.2 million L) of water annually in Oklahoma counties with disclosures; in

1  Kansas, this value is only 3.5 million gal (13.2 million L) (see Appendix Table B-2). Average
2  hydraulic fracturing water use in 2011 and 2012 did not exceed 10% of 2010 total water use in any
3  county in Oklahoma or Kansas (see Appendix Table B-2). However, there were six counties in
4  Oklahoma (Alfalfa, Canadian, Coal, Pittsburg, Rogers Mills, and Woods) where fracturing water
5  consumption exceeded 10% of 2010 total county water consumption.

6  *Potential for impacts:* The potential for effects on drinking water resources appears to be low in
7  Oklahoma and Kansas, since hydraulic fracturing water use and consumption are generally low as a
8  percentage of total water use and consumption. This finding is generally supported by the
9  comparison of cumulative fracturing water use to water availability at the county scale (see Text
10 Box 4-2 and Figure 4-5a,b). If impacts to water quantity or quality do occur, however, they are more
11 likely to happen in western Oklahoma than in the eastern half of the state or Kansas. Of the six
12 Oklahoma counties where fracturing consumption exceeded 10% of 2010 water consumption,
13 three (Alfalfa, Canadian, and Roger Mills) are in the western half of the state where surface water
14 availability is lowest (Figure 4-6a). Surface water is fully allocated in the Panhandle and West
15 Central regions, encompassing much of the state's northwestern quadrant (OWRB, 2014). As a
16 result, residents generally rely on ground water in western Oklahoma (see Appendix Table B-6),
17 and it is likely that fracturing does as well.

18 Projecting out to 2060, Oklahoma's Water Plan concludes that aquifer storage depletions are likely
19 in the Panhandle and West Central regions due to over-pumping, particularly for irrigation (OWRB,
20 2014). Ground water depletions are anticipated to be small relative to storage, but will be the
21 largest in summer months and may lead to higher pumping costs, the need for deeper wells, lower
22 water yields, and detrimental effects on water quality (OWRB, 2014). Drought conditions are likely
23 to exacerbate this problem, and Oklahoma's Water Plan specifically mentions the potential for
24 climate change to affect future water supplies in the state (OWRB, 2014). In the adjacent Texas
25 Panhandle, future irrigation needs may go unmet (TWDB, 2012), and this may be the case in
26 western Oklahoma as well.

27 Aquifer depletions in western Oklahoma may be associated with ground water quality degradation,
28 particularly under drought conditions. The central portion of the Ogallala aquifer underlying the
29 Oklahoma Panhandle and western Oklahoma contains elevated levels of some constituents (e.g.,
30 nitrate) due to over-pumping, although generally it is of better quality than the southern portion of
31 the aquifer (USGS, 2009). Additional ground water withdrawals for hydraulic fracturing in western
32 Oklahoma may add to these water quality issues, particularly in combination with other substantial
33 water uses (e.g., irrigation) (USGS, 2009).

### 4.5.6.  Arkansas and Louisiana

34 Arkansas and Louisiana were ranked seventh and tenth in the number of disclosures in the EPA
35 FracFocus project database, respectively (see Appendix Table B-5). Hydraulic fracturing activity in
36 Louisiana occurs primarily in the TX-LA-MS Salt Basin, which contains the Haynesville play; activity
37 in Arkansas is dominated by the Arkoma Basin, which contains the Fayetteville play (Figure 4-11).

1   *Types of water used:* Surface water is reported as the primary source of water for hydraulic
2   fracturing operations in both Arkansas and Louisiana (ANRC, 2014; LA Ground Water Resources
3   Commission, 2012; STRONGER, 2012). Quantitative information is lacking for Arkansas on the
4   proportion of water sourced from surface versus ground water. However, data are available for
5   Louisiana, where an estimated 87% of water for hydraulic fracturing in the Haynesville Shale is
6   sourced from surface water (LA Ground Water Resources Commission, 2012) (see Table 4-3). In
7   2008, during the early stages of development, hydraulic fracturing in Louisiana relied heavily on
8   ground water from the Carrizo-Wilcox aquifer, although concerns for the sustainability of ground
9   water resources have more recently prompted the state to encourage surface water withdrawals
10  (LA Ground Water Resources Commission, 2012).

11  The EPA FracFocus report suggests that significant reuse of wastewater may occur in Arkansas to
12  offset total fresh water used for hydraulic fracturing; 70% of all disclosures reporting a water
13  source indicated a blend of "recycled/surface," whereas only 3% of disclosures reporting a water
14  source noted "fresh" as the exclusive water source (U.S. EPA, 2015a).[1] According to Veil (2011),
15  Arkansas' Fayetteville Shale wastewater is of relatively good quality (i.e., low TDS), potentially
16  facilitating reuse. Data are generally lacking on the extent to which hydraulic fracturing wastewater
17  is reused to offset total fresh water use in Louisiana.

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[1] 93% of Arkansas disclosures included information related to water sources (U.S. EPA, 2015a).
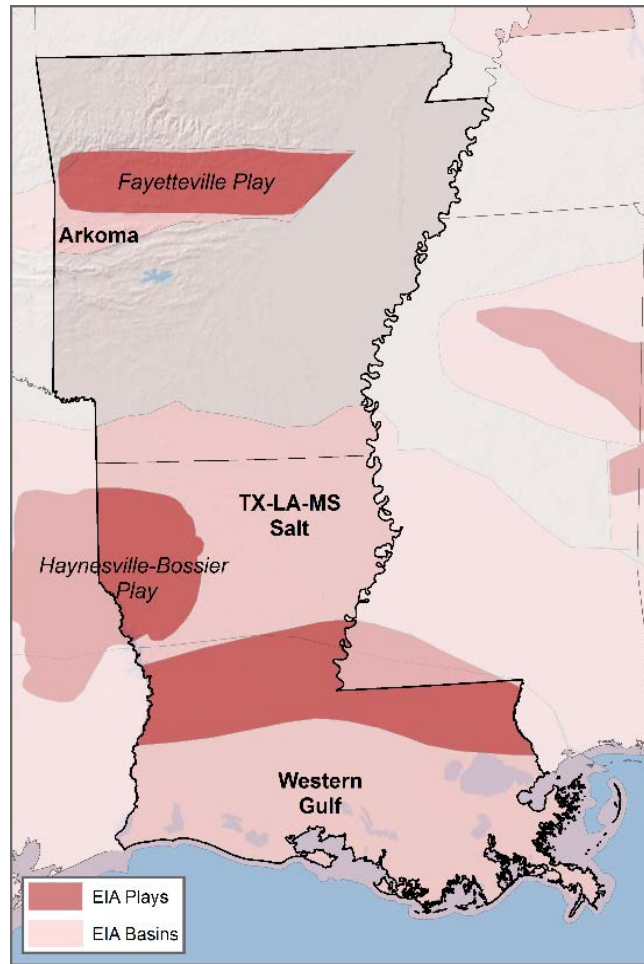
**Figure 4-11. Major U.S. EIA shale plays and basins for Arkansas and Louisiana (EIA, 2015b).**

Source: (EIA, 2015b).

1  *Water Use per Well:* Arkansas and Louisiana have the highest median water use per well in the
2  nation, at 5.3 million and 5.1 million gal (20.1 million and 19.3 million L), respectively based on the
3  EPA FracFocus project database (see Appendix Table B-5).[1]

4  *Cumulative water use/consumption:* On average, hydraulic fracturing operations cumulatively use
5  408 million gal (1.54 billion L) of water each year in Arkansas counties reporting activity, or 9.3%
6  of 2010 total county water use (26.9% of total county consumption) (see Appendix Table B-2). In
7  2011 and 2012, five counties dominated fracturing water use in Arkansas: Cleburne, Conway,
8  Faulkner, Van Buren, and White Counties (see Appendix Table B-2). Van Buren, which is sparsely
9  populated and thus has relatively low total water use and consumption, is by far the county highest

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[1] 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.

1    in hydraulic fracturing water use and consumption relative to 2010 total water use and
2    consumption (56% and 168%, respectively) (see Table 4-2).

3    In Louisiana, fracturing water use is concentrated in six parishes in the far northwestern corner of
4    the state, associated with the Haynesville play.[1] On average in 2011 and 2012, hydraulic fracturing
5    used 117 million gal (443 million L) of water annually per parish, representing approximately 3.6%
6    and 10.8% of 2010 total water use and consumption, respectively (see Appendix Table B-2).
7    Operators in De Soto Parish used the most water (over 1 billion gal (3.8 billion L) annually).
8    Fracturing water use and consumption was highest relative to 2010 total water use and
9    consumption (35.5% and 83.2%, respectively) in Red River Parish (see Table 4-2). These numbers
10   may be low estimates since Louisiana required disclosures to the state or FracFocus and Arkansas
11   required disclosures to the state, but not FracFocus, during the time period analyzed (U.S. EPA,
12   2015a) (see Appendix Table B-5).

13   *Potential for impacts:* Water availability is generally higher in Arkansas and Louisiana than in states
14   farther west, reducing the potential for impacts to drinking water quantity and quality (Figure 4-6a,
15   Text Box 4-2, and Figure 4-5). There are, however, concerns about over-pumping of ground water
16   resources in northwestern Louisiana. Prior to 2008, most operators in the Louisiana portion of the
17   Haynesville Shale used ground water, withdrawing from the Carrizo-Wilcox, Upland Terrace, and
18   Red River Alluvial aquifer systems (LA Ground Water Resources Commission, 2012). To mitigate
19   stress on ground water, the state issued a water use advisory to the oil and gas industry that
20   recommended Haynesville Shale operators seek alternative water sources to the Carrizo-Wilcox
21   aquifer, which is predominantly used for public supply (LDEQ, 2008). Operators then transitioned
22   to mostly surface water, with a smaller ground water component (approximately 12% of all
23   fracturing water used) (LA Ground Water Resources Commission, 2012). Of this ground water
24   component, the majority (approximately 74%) still came from the Carrizo-Wilcox aquifer (LA
25   Ground Water Resources Commission, 2012).

26   Although the potential for hydraulic fracturing withdrawals to affect water supplies and water
27   quality in the aquifer appears greatly reduced, it is not entirely eliminated. Despite Louisiana's
28   water use advisory, a combination of drought conditions and higher than normal withdrawals (for
29   all uses, not solely hydraulic fracturing) from the Carrizo-Wilcox and Upland Terrace aquifers
30   caused several water wells to go dry in July 2011. In August 2011, a ground water emergency was
31   declared for southern Caddo Parish (LA Ground Water Resources Commission, 2012). There are
32   hydraulic fracturing wells in southern Caddo Parish (U.S. EPA, 2015b), and so it is possible that
33   fracturing withdrawals contributed to the problem of declines in ground water in this instance.

### 4.5.7. Utah, New Mexico, and California

34   Together, Utah, New Mexico, and California accounted for approximately 9% of disclosures in the
35   EPA FracFocus project database (3.8%, 3.1% and 1.9% of disclosures, respectively) (see Appendix
36   Table B-5 and Figure 4-3). Almost all reported hydraulic fracturing in Utah and California were in

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

1  the Uinta-Piceance Basin (99%) and San Joaquin Basin (95%), respectively. Activity in New Mexico
2  mostly occurs in the Permian and San Juan Basins, which together comprised 96% of reported
3  disclosures in that state (see Figure 4-12).
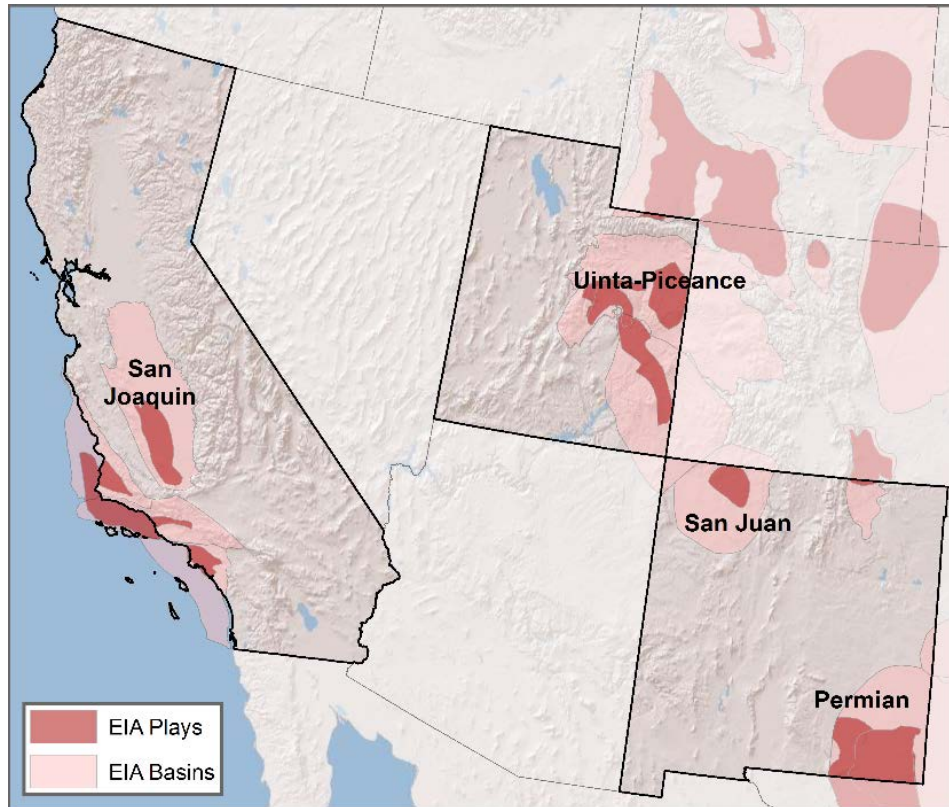


**Figure 4-12. Major U.S. EIA shale plays and basins for Utah, New Mexico, and California (EIA, 2015).**

Source: (EIA, 2015b).

4  *Types of water used*: Of these three states, California has the most information available on the
5  sources of water used for hydraulic fracturing. Most current and proposed fracturing activity is
6  focused in Kern County in the San Joaquin Basin, where well stimulation notices indicate that
7  operators depend mainly on surface water purchased from nearby irrigation districts (CCST, 2014).
8  California irrigation districts receive water allocated by the State Water Project, and deliveries may
9  be restricted or eliminated during drought years (CCST, 2014).[1] In addition to publicly-supplied
10 surface water, operators also may self-supply a smaller proportion of water from on-site ground
11 water wells (CCST, 2014). Operators use primarily fresh water for hydraulic fracturing (96% of well

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[1] The California State Water Project is water storage and distribution system maintained by the California Department of Water Resources, which provides water for urban and agricultural water suppliers in Northern California, the San Francisco Bay Area, the San Joaquin Valley, the Central Coast, and Southern California (California Department of Water Resources, 2015).

1   stimulation notices reported); reused wastewater (sometimes blended with fresh water) is used in
2   small amounts relative to total water use (4% of well stimulation notices reported) (CCST, 2014) (see
3   Table 4-1).

4   The source, quality, and provisioning of water used for hydraulic fracturing in Utah and New
5   Mexico are not well characterized. The 2010 New Mexico water use report summarizes
6   withdrawals for a variety of water use categories. In 2010, mining water use (which includes water
7   used for oil and gas production) consisted of 26% and 74% of surface and ground water
8   withdrawals, respectively (NM OSE, 2013). Assuming that hydraulic fracturing follows the same
9   pattern as other mining water uses (e.g., for metals, coal, geothermal), water for hydraulic
10  fracturing in New Mexico would be supplied primarily by ground water withdrawals. To our
11  knowledge, no data are available to characterize the source of water for hydraulic fracturing
12  operations in Utah. In addition, no data are available to describe the extent to which reused
13  wastewater is used as a proportion of total water injected for either Utah or New Mexico.

14  *Water use per well:* Median water use per well in Utah, New Mexico, and California is lower than in
15  other states in the EPA FracFocus project database: Utah ranks 13th (approximately 302,000 gal
16  (1.14 million L)), New Mexico ranks 14th (approximately 175,000 gal (662,000 L)), and California
17  ranks 15th (approximately 77,000 gal (291,000 L)) out of the 15 states (see Appendix Table B-5). A
18  likely explanation for the low water use per well in Utah and New Mexico is the prevalence of CBM
19  in the Uinta (Utah) and San Juan (New Mexico) Basins. Low water use per well in California is
20  attributed to the prevalence of vertical wells and the use of crosslinked gels. Vertical wells
21  dominate because the complex geology precludes long horizontal drilling and fracturing (CCST,
22  2014).

23  For California, the California Council on Science and Technology (CCST) reports average water use
24  per well of 130,000 gal (490,000 L), which agrees with the state average of approximately 131,700
25  gal (498,500 L) according to the EPA FracFocus project database (CCST, 2014) (see Appendix Table
26  B-5); this is expected because estimates from CCST are also based on data submitted to FracFocus.

27  *Cumulative water use/consumption:* Operators in Utah, New Mexico, and California report using low
28  cumulative amounts of water compared to most other states (see Appendix Table B-1). Only four
29  counties (Duchesne and Uintah Counties in Utah, and Eddy and Lea Counties in New Mexico)
30  required more than 50 million gal (189 million L) annually (see Appendix Table B-2). Fracturing
31  water use and consumption did not exceed 1% of 2010 total water use and consumption in any
32  county.

33  *Potential for impacts:* The potential for water quantity and quality impacts from hydraulic
34  fracturing water withdrawals in Utah, New Mexico, and California appears to be low at present (see
35  Text Box 4-2 and Figure 4-5a,b). Hydraulic fracturing does not use or consume much water
36  compared to other users or consumers in these states. As in other states, this does not preclude
37  sub-county effects, and this finding of low potential for impacts could change if fracturing activities
38  increase beyond present levels. This is particularly the case because these states generally have low
39  surface water availability (see Figure 4-6a) and high ground water dependence (see Figure 4-6b),

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1   and have experienced frequent periods of drought over the last decade (National Drought
2   Mitigation Center, 2015).

## 4.6.  Chapter Synthesis

3   In this chapter we examine the potential for water acquisition for hydraulic fracturing to affect
4   drinking water quantity and quality. The potential for impacts largely depends on water use,
5   consumption, and availability. Water management—in terms of the type of water used, the timing
6   or location of water withdrawals, or other factors—also can play a role. Because all of these factors
7   vary considerably from place-to-place, any impacts that occur will be location-specific and occur at
8   the spatial scale of the specific drinking water resource (i.e., the particular stream, watershed, or
9   local ground water aquifer). Therefore, it is important to consider the potential for hydraulic
10  fracturing impacts by location.

11  We examine the potential for impacts by considering (1) the types of water used for hydraulic
12  fracturing; (2) the amounts of water used per well; (3) cumulative estimates of water used and
13  consumed for hydraulic fracturing; and (4) a state-by-state assessment of the potential for impacts
14  based on water use, consumption, and availability. We often could not assess the potential for
15  impacts at a finer resolution than the county scale due to lack of available local-scale data for most
16  areas. Thus, our assessment suggests areas that are more likely than others to experience impacts,
17  but does not necessarily indicate that these impacts will occur. Three case studies (southern Texas,
18  western Colorado, and eastern Pennsylvania), provide an in-depth examination at finer scales, and
19  we rely on those where possible (see Text Boxes 4-3, 4-4, and 4-5).

### 4.6.1.  Major Findings

20  Water for hydraulic fracturing typically comes from surface water, ground water, or reused
21  wastewater. Because trucking can be a major expense, operators often use water sources as close to
22  well pads as possible. Operators usually self-supply surface or ground water directly, but also may
23  obtain water secondarily through public water systems or other suppliers. Hydraulic fracturing
24  operations in the eastern United States generally rely on surface water, whereas operations in more
25  semi-arid to arid western states use mixed surface and ground water supplies. In areas that lack
26  available surface water (e.g., western Texas), ground water supplies most of the water needed for
27  fracturing unless alternative sources, such as reused wastewater, are available and utilized.

28  The vast majority of water used for hydraulic fracturing nationally comes from fresh water sources,
29  although some operators also use lower-quality water (e.g., hydraulic fracturing wastewater,
30  brackish ground water, or small proportions of acid mine drainage and wastewater treatment plant
31  effluent). The use of non-fresh sources can reduce competition for current drinking water
32  resources. Nationally, the proportion of reused wastewater is generally low as a percentage of
33  injected volume; based on available data, the median reuse of wastewater as a percentage of
34  injected volume is 5% nationally, but this percentage varies by location (see Table 4-1).[1] Available

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[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).

1    data on reuse trends indicate increasing reuse of wastewater over time in both Pennsylvania and
2    West Virginia, likely due to the lack of nearby disposal options. Reuse as a percentage of water
3    injected appears to be low in other areas, likely in part because of the relatively high availability of
4    disposal wells (see Chapter 8).

5    The median amount of water used per hydraulically fractured well, based on national disclosures to
6    FracFocus, is approximately 1.5 million gal (5.7 million L) of water (U.S. EPA, 2015a, b). This
7    estimate represents a variety of fractured well types, including types that use much less water per
8    well than horizontal shale gas wells. Thus, published estimates for horizontal shale gas wells are
9    typically higher (e.g., approximately 4 million gal (15 million L) per well (Vengosh et al., 2014)).
10   There is also wide variation within and among states and basins in the median water volumes
11   reported per disclosure, from more than 5 million gal (19 million L) in Arkansas and Louisiana to
12   less than 1 million gal (3.8 million L) in Colorado, Wyoming, Utah, New Mexico, and California (U.S.
13   EPA, 2015b). This variation results from several factors, including well length, formation geology,
14   and fracturing fluid formulation (see Section 4.3.3).

15   Cumulatively, hydraulic fracturing uses billions of gallons of water every year at the national and
16   state scales, and even in some counties. When expressed as a percentage compared to total water
17   use or consumption at these scales, however, hydraulic fracturing water use and consumption is
18   most often a small percentage, generally less than 1%. This percentage may be higher in specific
19   areas. Annual hydraulic fracturing water use was 10% or more compared to 2010 total water use in
20   6.5% of counties with FracFocus disclosures in 2011 and 2012, 30% or more in 2.2% of counties,
21   and 50% or more in 1.0% of counties (U.S. EPA, 2015a). Consumption estimates follow the same
22   general pattern, but with slightly higher percentages in each category. In these counties, hydraulic
23   fracturing represents a relatively large user and consumer of water.

24   High hydraulic fracturing water use or consumption alone does not necessarily result in impacts to
25   drinking water resources. Rather, the potential for impacts depends on both water use or
26   consumption and water availability at a given withdrawal point. Our state-by-state assessment
27   examines the intersection between water use or consumption and availability at the county scale.
28   This approach suggests where the potential for impacts exists, but does not indicate where impacts
29   will occur at the local scale. Where possible, we use local-scale case studies in Texas, Pennsylvania,
30   and Colorado to provide details at finer spatial scales.

31   In our survey of the published literature, we did not find a case where hydraulic fracturing water
32   use by itself caused a drinking water well or stream to run dry. This could indicate an absence of
33   hydraulic fracturing effects on water availability; alternatively, it could reflect that these events are
34   not typically documented in the types of literature we reviewed. Water availability is rarely
35   impacted by just one use or factor alone. For example, drinking water wells in an area overlapping
36   with the Haynesville Shale in northwest Louisiana ran out of water in 2011, due to higher than
37   normal withdrawals and drought (LA Ground Water Resources Commission, 2012). Hydraulic
38   fracturing water use in the area may have contributed to these conditions, along with other water
39   uses and the lack of precipitation. Other impacts to drinking water quantity or quality (e.g.,

1    declining aquifer levels, decreased stream flow, increased pollutant concentrations) also may occur
2    before wells and streams actually go dry.

3    The potential for impacts due to hydraulic fracturing water withdrawals is highest in areas with
4    relatively high fracturing water use and low water availability. Southern and western Texas are two
5    locations where hydraulic fracturing water use combined with low water availability, drought, and
6    reliance on declining ground water sources has the potential to affect the quantity and quality of
7    drinking water resources. Fracturing withdrawals combined with other intensive uses, particularly
8    irrigation, could contribute to ground water quality degradation. Any impacts are likely to be
9    realized locally within these areas. In a detailed case study of southern Texas, Scanlon et al. (2014)
10   observed generally adequate water supplies for hydraulic fracturing, except in specific locations.
11   They found excessive drawdown of local ground water in a small proportion (∼6% of the area) of
12   the Eagle Ford play. They suggested water management, particularly a shift towards brackish water
13   use, could minimize potential future impacts to fresh water resources (see Text Box 4-3). County-
14   level data confirm that high brackish water availability in Texas may help offset hydraulic
15   fracturing water demand (see Text Box 4-2).

16   Comparatively, the potential for hydraulic fracturing water acquisition impacts to drinking water
17   quantity and quality appears to be lower—but not entirely eliminated—in other areas of the United
18   States. Detailed case studies in western Colorado and northeastern Pennsylvania did not show
19   impacts, despite indicating that streams could be vulnerable to water withdrawals from hydraulic
20   fracturing (U.S. EPA, 2015c). High wastewater reuse rates in western Colorado eliminated the need
21   for more fresh water withdrawals. In northeast Pennsylvania, water withdrawals for hydraulic
22   fracturing could result in high water consumption-to-stream flow events, but water management
23   (e.g., passby flows) limited the potential for impacts, especially on small streams (U.S. EPA, 2015c).
24   In western North Dakota, ground water is limited, but the industry may have sufficient supplies of
25   surface water from the Missouri River system. These location-specific examples emphasize the
26   need to focus on regional and local dynamics when considering the potential impacts of hydraulic
27   fracturing water acquisition on drinking water resources.

## 4.6.2.  Factors Affecting Frequency or Severity of Impacts

28   The potential for hydraulic fracturing water use to affect drinking water resource quantity or
29   quality depends primarily on the amount of water used or consumed versus water availability at a
30   given withdrawal point. Potential impacts to drinking water resources reflect all uses, including
31   hydraulic fracturing demands, compared to available water. Areas with high water use, low water
32   availability, slowly replenishing sources, and/or episodic water shortages (e.g., seasonal or longer-
33   term droughts) are more vulnerable to potential impacts. Areas with high water availability relative
34   to existing uses, high rainfall distributed throughout the year, or high storage capacity, are less
35   likely to be affected.

36   Water management can alter this dynamic between water use and availability. The type of water
37   used (e.g., fresh, brackish, reused hydraulic fracturing wastewater, other wastewaters) is a major
38   factor that can either increase or decrease the potential for impacts. Replacing a fresh water source
39   with another type of water can reduce the demand for fresh water and decrease potential

1  competition for drinking water. Brackish ground water use may reduce the demand for fresh water
2  and decrease competition for drinking water currently, but this may change if desalinization for
3  drinking water becomes more prevalent in the future (see Chapter 3).

4  The timing and location of water withdrawals can also affect the potential for impacts, particularly
5  for surface water withdrawals. Withdrawing water from small streams is more likely to result in a
6  high-consumption-to-stream flow event than removing water from larger streams (U.S. EPA,
7  2015c). Withdrawals during periods of low stream flow are also more likely to result in impacts
8  than withdrawals during high flow periods. Hydraulic fracturing operations may have the ability to
9  withdraw water during periods of high stream flow, and store it for future use during drier periods.

### 4.6.3. Uncertainties

10  There are several uncertainties inherent in our assessment of hydraulic fracturing water use and
11  potential effects on drinking water quantity and quality. The largest uncertainties stem from the
12  lack of literature and data on this subject at local scales, and the question of whether any impacts
13  would be documented in the types of literature we reviewed.

14  We used a state-by-state approach to identify areas where potential impacts are likely, based on
15  relatively high fracturing water use and low water availability. Typically, only data at the county-
16  scale were available. Because impacts occur at smaller spatial scales (i.e., at water withdrawal
17  sites), our assessment suggests the potential for impacts, but does not indicate whether impacts
18  will occur. In only a few places could we use local case studies to determine if potential impacts
19  were realized; these case studies show that local factors can greatly affect whether drinking water
20  resources are impacted.

21  In our survey of the published literature, we did not find a case where hydraulic fracturing water
22  use alone caused a drinking water well or stream to run dry. This could indicate an absence of
23  hydraulic fracturing effects on water availability, or it could reflect that these events are not
24  typically documented in the types of literature we reviewed. Water availability is rarely impacted
25  by just one use or factor alone. These issues may have limited our findings.

26  Other uncertainties arise from data limitations regarding the volume and types of water used or
27  consumed for hydraulic fracturing, future water use projections, and water availability estimates.
28  There are no nationally consistent data sources, and therefore water use estimates must be based
29  on multiple, individual pieces of information. For example, in their National Water Census, the USGS
30  includes hydraulic fracturing in the broader category of "mining" water use, but hydraulic
31  fracturing water use is not reported separately (Maupin et al., 2014). There are locations where
32  annual average hydraulic fracturing water use in 2011 and 2012 exceeded total mining water use in
33  2010, and one county where it exceeded all water use (U.S. EPA, 2015b; Maupin et al., 2014). This
34  could be due to a rapid increase in hydraulic fracturing water use, differences in methodology
35  between the two databases (i.e., the USGS 2010 National Water Census and the EPA FracFocus
36  project database), or both.

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1   The EPA FracFocus project database represents the most extensive database currently available to
2   estimate hydraulic fracturing water use. However, estimates based on the project database form an
3   incomplete picture of hydraulic fracturing water use because most states with data in the project
4   database did not require disclosure to FracFocus during the time period analyzed (U.S. EPA, 2015a)
5   (see Text Box 4-1). We conclude that this likely does not change the overall hydraulic fracturing
6   water use patterns observed across the United States, but could affect our assessment of the
7   potential impacts in specific locations.

8   Hydraulic fracturing water use data are often provided in terms of water use per well. While this is
9   valuable information, the potential impacts of water acquisition for hydraulic fracturing could be
10  better assessed if data were also available at the withdrawal point. If the total volume, date, and
11  location of each water withdrawal were documented, the quality of the water used and potential
12  effects on availability could be better estimated. For example, surface withdrawal points could be
13  aggregated by watershed to estimate effects on downstream flow. Alternatively, if the location and
14  depth of ground water pumping were documented, these could be aggregated to assess effects on a
15  given aquifer. Some of this information is available in disparate forms, but the lack of nationally
16  consistent data on water withdrawal locations, timing, and amounts—data that are publicly
17  available, easy to access, and easy to analyze—limits our assessment of hydraulic fracturing water
18  use.

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

### 4.6.4.  Conclusions

27  Water acquisition for hydraulic fracturing has the potential to impact drinking water resources by
28  affecting drinking water quantity and quality (see Text Box 4-6). In our survey of the published
29  literature, we did not find a case where hydraulic fracturing water use by itself caused a drinking
30  water well or stream to run dry. However, the potential for impacts to drinking water quantity and
31  quality exists and is highest in areas with relatively high fracturing water use and low water
32  availability. Southern and western Texas are two locations where the potential appears highest due
33  to the combined effects of high hydraulic fracturing activity, low water availability, drought, and
34  reliance on declining ground water sources. Even in locations where water is generally plentiful,
35  localized impacts can still occur in certain instances. Excessive ground water pumping can cause
36  localized drawdowns; surface water withdrawals can affect stream flow, particularly in smaller
37  streams or during low flow periods. These findings emphasize the need to focus on regional and
38  local dynamics when examining potential impacts of hydraulic fracturing water acquisition on
39  drinking water quantity and quality.

**Text Box 4-6. Research Questions Revisited.**

*What are the types of water used for hydraulic fracturing?*

- Water for hydraulic fracturing typically comes from surface, ground water, or reused wastewater. Operators often use water sources as close to well pads as possible as trucking is a major expense. Operators usually self-supply surface or ground water directly, but also may obtain water secondarily through public water systems or other suppliers. Hydraulic fracturing operations in the eastern United States generally rely on surface water, whereas operations in more semi-arid to arid western states use mixed surface and ground water supplies. In areas that lack available surface water (e.g., western Texas), ground water supplies most of the water needed for fracturing unless alternative sources, such as reused wastewater, are available and utilized.

- The vast majority of water used nationally comes from fresh water sources, although some operators also use lower-quality water (e.g., hydraulic fracturing wastewater, brackish ground water, or small proportions of acid mine drainage and wastewater treatment plant effluent). The use of non-fresh sources can reduce competition for current drinking water resources. Nationally, the proportion of reused wastewater is generally low as a percentage of injected volume; based on available data, median reuse of wastewater across all basins and plays is 5% of injected volume (see Table 4-1). 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. Reuse as a percentage of water injected appears to be low in other areas, likely in part because of the relatively high availability of disposal wells (see Chapter 8).

*How much water is used per well?*

- The median amount of water used per hydraulically fractured well, based on national disclosures to FracFocus, is approximately 1.5 million gal (5.7 million L) of water (U.S. EPA, 2015a, b). This estimate represents a variety of fractured well types. There is also wide variation within and among states and basins in the median water volumes reported per disclosure, 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, 2015b). This variation results from several factors, including well length, formation geology, and fracturing fluid formulation (see Section 4.3.3).

- Trends indicate that water use per well is increasing in certain locations as horizontal well lengths increase. This may not, however, increase water use per unit energy extracted.

1     ***How might cumulative water withdrawals for hydraulic fracturing affect drinking water quantity?***

2      •      Cumulatively, hydraulic fracturing uses billions of gallons of water every year at the national and state
3      scales, and even in some counties. When expressed as a percentage compared to total water use or
4      consumption at these scales, however, hydraulic fracturing water use and consumption is most often a
5      small percentage, generally less than 1%. This percentage may be higher in specific areas. Annual
6      hydraulic fracturing water use was 10% or more compared to 2010 total water use in 6.5% of counties
7      with FracFocus disclosures in 2011 and 2012, 30% or more in 2.2% of counties, and 50% or more in
8      1.0% of counties ([U.S. EPA, 2015a](#)). Consumption estimates follow the same general pattern, but with
9      slightly higher percentages in each category. In these counties, hydraulic fracturing represents a
10      relatively large user and consumer of water.

11      •      High hydraulic fracturing water use or consumption alone does not necessarily result in impacts to
12      drinking water resources. Rather, the potential for impacts depends on both water use or consumption
13      and water availability at a given withdrawal point. Our state-by-state assessment examines the
14      intersection between water use or consumption and availability at the county scale. This approach
15      suggests where the potential for impacts exists, but does not indicate where impacts will occur at the
16      local scale. Local-scale case studies help provide details at finer spatial scales.

17      •      In our survey of the published literature, we did not find a case where hydraulic fracturing water use by
18      itself caused a drinking water well or stream to run dry. This could indicate an absence of hydraulic
19      fracturing effects on water availability, or it could reflect that these events are not typically documented
20      in the types of literature we reviewed. Water availability is rarely impacted by just one use or factor
21      alone. For example, drinking water wells in an area overlapping with the Haynesville Shale in northwest
22      Louisiana ran out of water in 2011, due to higher than normal withdrawals and drought ([LA Ground
23      Water Resources Commission, 2012](#)). Hydraulic fracturing water use in the area may have contributed to
24      these conditions, along with other water uses and the lack of precipitation. Other impacts to drinking
25      water quantity or quality (e.g., declining aquifer levels, decreased stream flow, increased pollutant
26      concentrations) also may occur before wells and streams actually go dry.

27      •      The potential for impacts due to hydraulic fracturing water withdrawals is highest in areas with
28      relatively high fracturing water use and low water availability. Southern and western Texas are two
29      locations where hydraulic fracturing water use combined with low water availability, drought, and
30      reliance on declining ground water sources has the potential to affect the quantity of drinking water
31      resources. Any impacts are likely to be realized locally within these areas. In a detailed case study of
32      southern Texas, [Scanlon et al. (2014)](#) observed generally adequate water supplies for hydraulic
33      fracturing, except in specific locations. They found excessive drawdown of local ground water in a small
34      proportion (~6% of the area) of the Eagle Ford play. They suggested water management, particularly a
35      shift towards brackish water use, could minimize potential future impacts to fresh water resources (see
36      Text Box 4-3). County-level data confirm that high brackish water availability in Texas may help offset
37      hydraulic fracturing water demand (see Text Box 4-2).

38      •      The potential for hydraulic fracturing water acquisition impacts to drinking water quantity and quality
39      appears to be lower—but not entirely eliminated—in other areas of the United States. Detailed case
40      studies in western Colorado and northeastern Pennsylvania did not show impacts, despite indicating that
41      streams could be vulnerable to water withdrawals from hydraulic fracturing ([U.S. EPA, 2015c](#)). High
42      wastewater reuse rates in western Colorado eliminated the need for more fresh water withdrawals. In
43      northeast Pennsylvania, water withdrawals for hydraulic fracturing could result in high water
44      consumption-to-stream flow events, but water management (e.g., passby flows) limited the potential for
45      impacts, especially on small streams ([U.S. EPA, 2015c](#)). In western North Dakota, ground water is limited,
46      but the industry may have sufficient supplies of surface water from the Missouri River system. These

1   location-specific examples emphasize the need to focus on regional and local dynamics when considering
2   the potential impacts of hydraulic fracturing water acquisition on drinking water resources.

3   ***What are the possible impacts of water withdrawals for hydraulic fracturing on water quality?***

4   • Water withdrawals for hydraulic fracturing, similar to all water withdrawals, have the potential to alter
5   the quality of drinking water resources. Ground water withdrawals exceeding natural recharge rates
6   decrease water storage in aquifers, potentially mobilizing contaminants or allowing the infiltration of
7   lower-quality water from the land surface or adjacent formations. Withdrawals could also decrease
8   ground water discharge to streams, potentially affecting surface water quality. Areas with numerous
9   high-capacity wells and large amounts of sustained ground water pumping are most likely to experience
10   impacts, particularly in drought-prone regions with limited ground water recharge.

11   • Surface water withdrawals also have the potential to affect water quality. Withdrawals may lower water
12   levels and alter stream flow, potentially decreasing a stream's capacity to dilute contaminants. Case
13   studies by the EPA show that streams can be vulnerable to changes in water quality due to water
14   withdrawals, particularly smaller streams and during periods of low flow (U.S. EPA, 2015c). Management
15   of the rate and timing of surface water withdrawals can help mitigate potential impacts of fracturing
16   withdrawals on water quality.

17   • Like water quantity effects, any effects of water withdrawals on water quality will likely occur nearest the
18   withdrawal point, again emphasizing the need for location specific assessments.

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# Chapter 5

# Chemical Mixing

# 5. Chemical Mixing

## 5.1. Introduction

1   This chapter addresses the potential for on-site spills of chemicals used in the chemical mixing
2   process to affect the quality of drinking water resources. Chemical mixing is a complex process that
3   requires the use of specialized equipment and a range of different additives to produce the
4   hydraulic fracturing fluid that is injected into the well. The number, type, and volume of chemicals
5   used vary from well to well based on site- and company-specific factors. Spills may occur at any
6   point in the hydraulic fracturing process. Chemicals may spill from on-site storage and containment
7   units; from interconnected hoses and pipes used to transfer chemicals to and from mixing and
8   pumping units, and tanker trucks; and from the equipment used to mix and pressurize chemical
9   mixtures that are pumped down the well. The potential for a spill to affect the quality of a drinking
10  water resource is governed by three overarching factors: (1) fluid characteristics (e.g., chemical
11  composition and volume), (2) chemical management and spill characteristics, and (3) chemical fate
12  and transport (see Figure 5-1). This chapter is organized around the three factors.
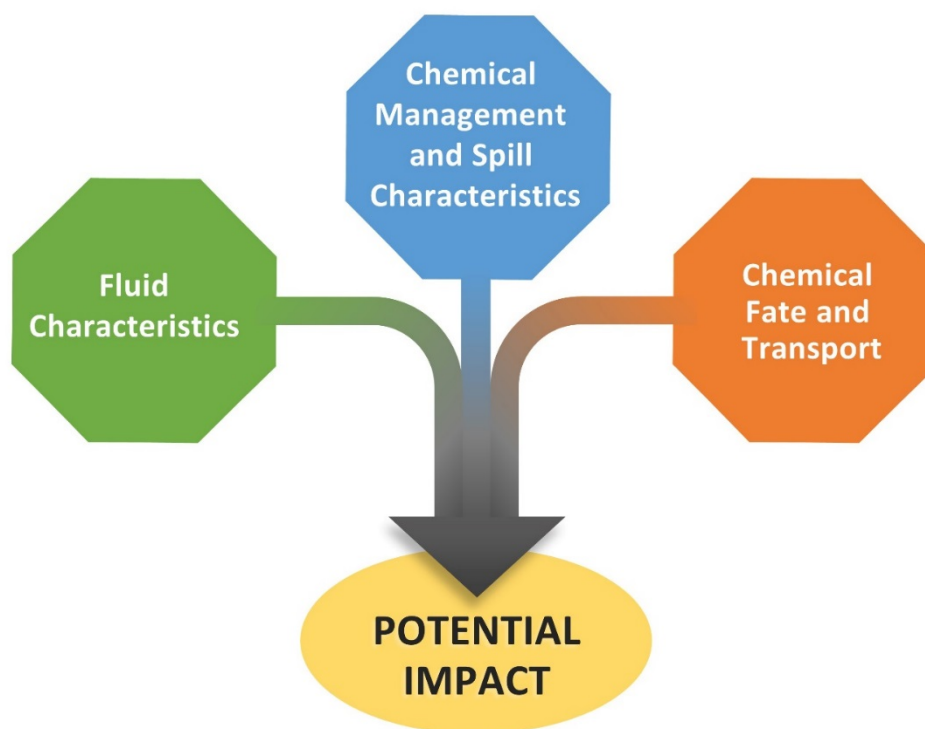


**Figure 5-1. Factors governing potential impact to drinking water resources.**

Factors include (1) fluid characteristics (e.g., chemical composition and volume), (2) chemical
management and spill characteristics, and (3) chemical fate and transport.

1   Section 5.2 provides an introductory overview of the chemical mixing process. The number and
2   volume of chemicals used and stored on-site are affected by such variables as the type, size, and
3   goals of the operation; formation characteristics; depth of the well; the length of the horizontal leg;
4   and the number of fracturing phases and stages.

5   Section 5.3 describes the different components of the hydraulic fracturing fluid, generally
6   comprised of the base fluid, proppant, and additives, which may be either individual chemicals or
7   mixtures. The composition of the hydraulic fracturing fluid is engineered to meet specific criteria.
8   The total amount and types of additives vary according to the characteristics of the well, site
9   geology, economics, availability, and the production goals (e.g., Maule et al., 2013). Section 5.4
10  presents the wide range of different chemicals used and their classes, the most frequently used
11  chemicals nationwide and from state-to-state, and volumes used.[1] Appendix A provides a list of
12  chemicals that the EPA identified as being used in hydraulic fracturing fluids based on eight
13  sources.

14  Sections 5.5 to 5.7 discuss how chemicals are managed on-site, how spills may occur, and the
15  different approaches for addressing spills. Section 5.5 describes how the potential impact of a spill
16  on drinking water resources depends upon chemical management practices, such as storage, on-
17  site transfer, and equipment maintenance. Section 5.6 discusses spill prevention, containment, and
18  mitigation. A summary analysis of reported spills and their common causes at hydraulic fracturing
19  sites is presented in Section 5.7.

20  Section 5.8 discusses the fate and transport of spilled chemicals. Spilled chemicals may react and
21  transform into other chemicals, travel from the site of release to a nearby surface water, or leach
22  into the soils and reach ground water. Chemical fate and transport after a release depend on site
23  conditions, environmental conditions, physicochemical properties of the released chemicals, and
24  the volume of the release.

25  Section 5.9 provides an overview of on-going changes in chemical use in hydraulic fracturing, with
26  an emphasis on efforts by industry to reduce potential impacts from surface spills by using fewer
27  and safer chemicals. A synthesis and a discussion of limitations are presented in Section 5.10.

28  Factors affecting the frequency and severity of impacts to drinking water resources from surface
29  spills include size and type of operation, employee training and experience, standard operating
30  procedures, quality and maintenance of equipment, type and volume of chemical spilled,
31  environmental conditions, proximity to drinking water resources, spill prevention practices, and
32  spill mitigation measures. Due to the limitations of available data and the scope of this assessment,
33  it is not possible to provide a detailed analysis of all of the factors listed above. Data limitations also
34  preclude a quantitative analysis of the likelihood or magnitude of chemical spills or impacts. Spills
35  that occur off-site, such as those during transportation of chemicals or storage of chemicals in
36  staging areas, are out of scope. This chapter qualitatively characterizes the potential for impacts to

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[1] 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, pesticides, acids, and bases.

1    drinking water resources given the current understanding of overall operations and specific
2    components of the chemical mixing process.

## 5.2.  Chemical Mixing Process

3    An understanding of the chemical mixing process is necessary to understand how, why, and when
4    spills that may affect drinking water resources might occur. This description provides a general
5    overview of chemical mixing in the context of the overall hydraulic fracturing process (Carter et al.,
6    2013; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

7    Figure 5-2 shows a hydraulic fracturing site during the chemical mixing process. The discussion
8    focuses on the types of additives used at each phase of the process. While similar processes are
9    used to fracture horizontal and vertical wells, a horizontal well treatment is described here because
10   it is likely to be more complex and because horizontal hydraulic fracturing has become more
11   prevalent over time with advances in hydraulic fracturing technology. A water-based system is
12   described because water is the most commonly used base fluid, appearing in more than 93% of
13   FracFocus disclosures between January 1, 2011 and February 28, 2012 (U.S. EPA, 2015a).



**Figure 5-2. Hydraulic fracturing site showing equipment used on-site during the chemical mixing process.**

Source: Industry source.

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1　While the number and types of additives may widely vary, the basic chemical mixing process is
2　similar across sites. The on-site layout of hydraulic fracturing equipment is also similar from site to
3　site (BJ Services Company, 2009). Equipment used in the chemical mixing process typically consists
4　of chemical storage trucks, water supply tanks, proppant supply, slurry blenders, a number of high-
5　pressure pumps, a manifold, surface lines and hoses, and a central control unit. Detailed
6　descriptions of specific additives and the equipment used in the process are provided in Sections
7　5.3 and 5.5, respectively.

8　The chemical mixing process begins after the drilling, casing, and cementing processes are finished
9　and hydraulic fracturing equipment has been set up and connected to the well. The process can
10　generally be broken down into sequential phases with specific chemicals added at each phase to
11　achieve a specific purpose (Knappe and Fireline, 2012; Fink, 2003). Phases may overlap. The
12　process for water-based hydraulic fracturing is outlined in Figure 5-3 below.
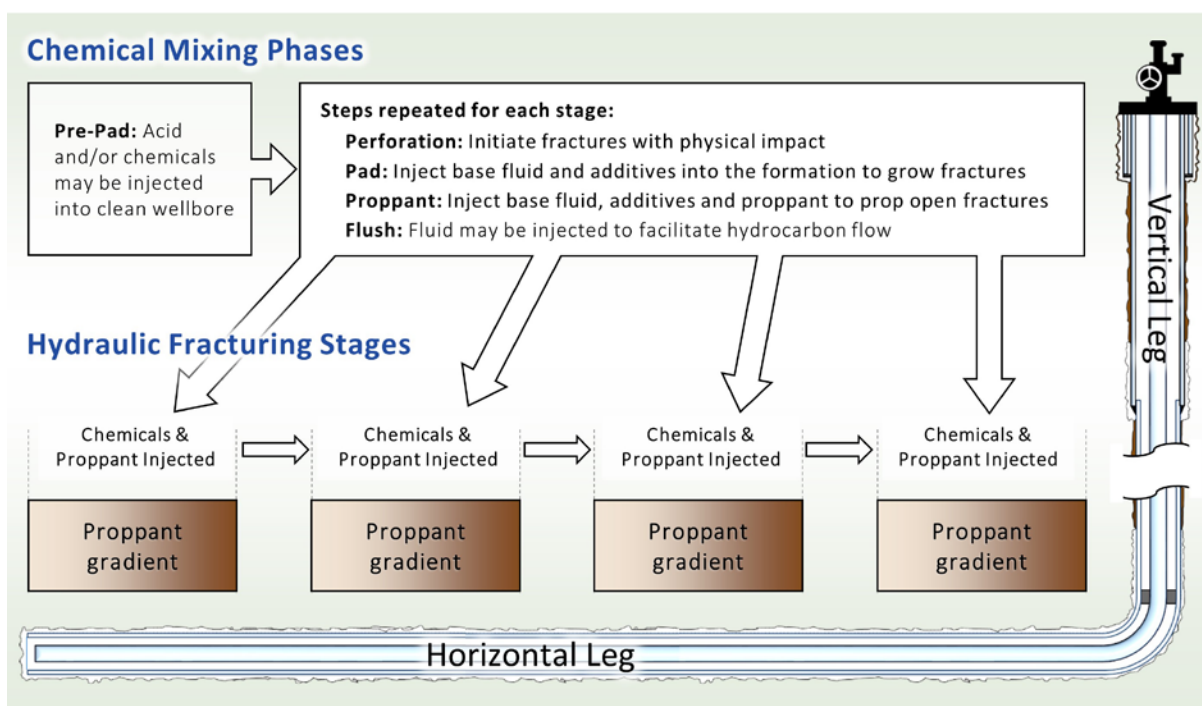


**Figure 5-3. Overview of a chemical mixing process of the hydraulic fracturing water cycle.**

> 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 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).

1    The first phase of the process consists of the cleaning and preparation of the well. The fluid used in
2    this phase is often referred to as the pre-pad fluid or pre-pad volume. Acid is typically the first
3    chemical introduced. Acid, with a concentration of 3%–28% (typically hydrochloric acid, HCl), is
4    used to adjust pH, clean any cement left inside the well from cementing the casing, and dissolve any
5    pieces of rock that may remain in the well and could block the perforations. Acid is typically
6    pumped directly from acid storage tanks or tanker trucks, without being mixed with other
7    additives. The first, or pre-pad, phase may also involve mixing and injection of additional chemicals
8    to facilitate the flow of fracturing fluid introduced in the next phase of the process. These additives
9    may include biocides, corrosion inhibitors, friction reducers, and scale inhibitors (Carter et al.,
10   2013; King, 2012; Knappe and Fireline, 2012; Spellman, 2012; Arthur et al., 2008).

11   In the second phase, a hydraulic fracturing fluid, typically referred to as the pad or pad volume, is
12   mixed, blended, and pumped down the wellbore to create fractures in the formation. The pad is a
13   mixture of base fluid, typically water, and additives. The pad is designed to create, elongate, and
14   enlarge fractures along the natural channels of the formation when injected under high pressure
15   (Gupta and Valkó, 2007). A typical pad consists of, at minimum, a mixture of water and friction
16   reducer. The operator may also add other additives (see U.S. EPA (2015a) and Table 5-1) used to
17   facilitate flow and kill bacteria (Carter et al., 2013; King, 2012; Knappe and Fireline, 2012;
18   Spellman, 2012; Arthur et al., 2008). The pad is pumped into the formation through perforations in
19   the well casing (see Text Box 5-1).

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**Text Box 5-1. Perforation.**

20   Prior to the injection of the pad, the well casing is typically perforated to provide openings through which the
21   pad fluid can enter the formation. A perforating gun is typically used to create small holes in the section of the
22   wellbore being fractured. The perforating gun is lowered into position in the horizontal portion of the well.
23   An electrical current is used to set off small explosive charges in the gun, which creates holes through the well
24   casing and out a short, controlled distance into the formation (Gupta and Valkó, 2007).

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25   In the third phase, proppant, typically sand, is mixed into the hydraulic fracturing fluid. The
26   proppant volume, as a proportion of the injected fluid, is increased gradually until the desired
27   concentration in the fractures is achieved. Gelling agents, if used, are also mixed in with the
28   proppant and base fluid in this phase to increase the viscosity and carry the proppant. Additional
29   chemicals may be added to gelled fluids, initially to maintain viscosity and later to break the gel
30   down into a more readily removable fluid. (Carter et al., 2013; King, 2012; Knappe and Fireline,
31   2012; Spellman, 2012; Arthur et al., 2008).

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

36   The second, third, and fourth phases are repeated multiple times in a horizontal well, as the
37   horizontal section, or leg, of the wellbore is typically fractured in multiple segments referred to as

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1    stages. For each stage, the well is typically perforated and fractured beginning at the end, or toe, of
2    the wellbore and proceeding backwards toward the vertical section. Each fractured stage is isolated
3    before the next stage is fractured. The number of stages corresponds directly to the number of
4    times the chemical mixing process is repeated at the site surface (see Figure 5-3). The number of
5    stages depends upon the length of the leg (Carter et al., 2013; King, 2012; Knappe and Fireline,
6    2012; Spellman, 2012; Arthur et al., 2008).

7    The number of stages per well can vary, with several sources suggesting between 10 and 20 is
8    typical (GNB, 2015; Lowe et al., 2013).[1] The full range reported in the literature is much wider, with
9    one source documenting between 1 and 59 stages per well (Pearson et al., 2013) and others
10   reporting values within this range (NETL, 2013; STO, 2013; Allison et al., 2009). It also appears that
11   the number of stages per well has increased over time. For instance, in the Williston Basin the
12   average number of stages per horizontal well rose from approximately 10 in 2008 to 30 in 2012
13   (Pearson et al., 2013).

14   In each of these phases, water is the primary component of the hydraulic fracturing fluid, though
15   the exact composition of the fluid injected into the well changes over the duration of each stage. In
16   water-based hydraulic fracturing, water typically comprises between 90% and 94% of the
17   hydraulic fracturing fluid, proppant comprises 5% to 9%, and additives comprise the remainder,
18   typically 2% or less (Carter et al., 2013; Knappe and Fireline, 2012; SWN, 2011). The exception to
19   this typical fluid composition may be when a concentrated acid is used in the initial cleaning phase
20   of the fracturing process.

## 5.3.   Overview of Hydraulic Fracturing Fluids

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

34   The particular composition of hydraulic fracturing fluids is selected by a design engineer based on
35   empirical experience, the formation, economics, goals of the fracturing process, availability of the

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[1] The number of stages has been reported to be 6 to 9 in the Huron in 2009 (Allison et al., 2009), 25 and up in the Marcellus (NETL, 2013), and up to 40 by STO (2013).

1    desired chemicals, and preference of the service company or operator (Montgomery, 2013; ALL
2    Consulting, 2012; Klein et al., 2012; Ely, 1989). No single set of specific chemicals is used at every
3    site. Multiple types of fracturing fluids may be appropriate for a given site and any given type of
4    fluid may be appropriate at multiple sites. For the same type of fluid formulation, there can be
5    differences in the additives, chemicals, and concentrations selected. There are broad criteria for
6    hydraulic fracturing fluid selection based on the fracturing temperatures, formation permeability,
7    fracturing pressures, and formation water sensitivity, as shown in Figure 5-4 (Gupta and Valkó,
8    2007; Elbel and Britt, 2000). One of the most important properties in designing a hydraulic
9    fracturing fluid is the viscosity (Montgomery, 2013).[1]

10    Figure 5-4 provides a general overview of which fluids can be used in different situations. As an
11    example, crosslinked fluids with 25% nitrogen foam (titanate or zirconate crosslink + 25% $N_2$) can
12    be used in both gas and oil wells with high temperatures with variation in water sensitivity.

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[1] 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).
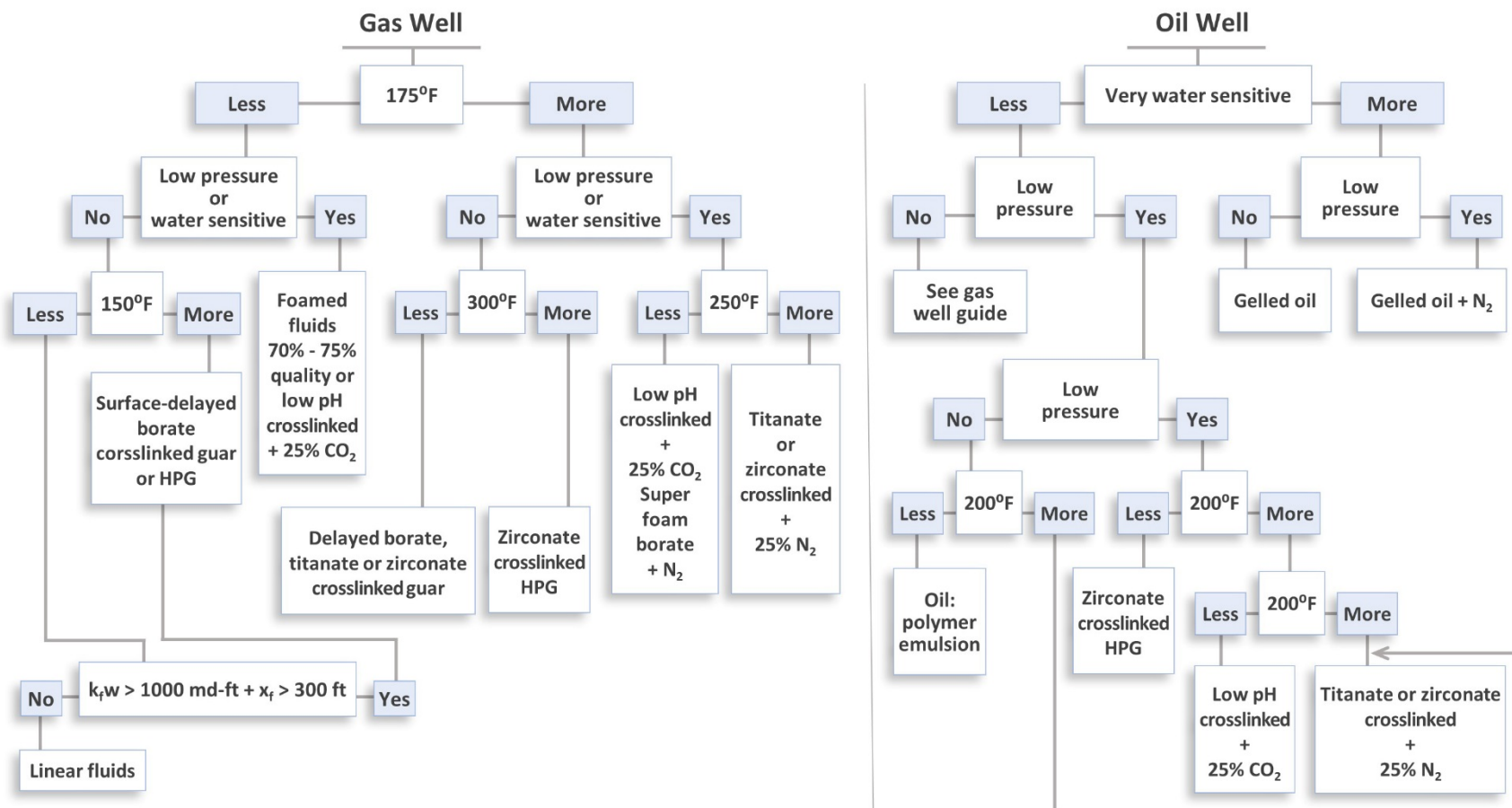
**Figure 5-4. Example fracturing fluid decision tree for gas and oil wells.**

Adapted from Elbel and Britt (2000).

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

**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, b; 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% of FracFocus disclosures for additive[a,b] |
|---|---|---|
| **Acid** | Dissolves cement, minerals, and clays to reduce clogging of the pore space | Hydrochloric acid |
| **Biocide** | Controls or eliminates bacteria, which can be present in the base fluid and may have detrimental effects on the fracturing process | Glutaraldehyde; 2,2-dibromo-3-nitrilopropionamide |
| **Breaker** | Reduces the viscosity of specialized treatment fluids such as gels and foams | Peroxydisulfuric acid diammonium salt |
| **Clay control** | Prevents the swelling and migration of formation clays in reaction 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 |

| Additives | Function | Chemicals reported in ≥20% of FracFocus disclosures for additive[a,b] |
|---|---|---|
| **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 from solution | Citric acid |
| **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 | Hydrochloric acid |

[a] Chemicals (excluding water and quartz) listed as reported to FracFocus in more than 20% of disclosures for a given purpose when that purpose was listed as used on a disclosure. 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; 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.

1 A general description of typical hydraulic fracturing fluid formulations nationwide is difficult
2 because fracturing fluids vary from well to well. Based on the FracFocus report, the median number
3 of chemicals reported for each disclosure was 14, with the 5th to 95th percentile ranging from four to
4 28. The median number of chemicals per disclosure was 16 for oil wells and 12 for gas wells (U.S.
5 EPA, 2015b). Other sources have stated that between three and 12 additives and chemicals are
6 used (Schlumberger, 2015; Carter et al., 2013; Spellman, 2012; GWPC and ALL Consulting, 2009).[1]

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

### 5.3.1. Water-Based Fracturing Fluids

17 The advantages of water-based fracturing fluids are low cost, ease of mixing, and ability to recover
18 and recycle the water. The disadvantages are low viscosity, the narrowness of the fractures created,
19 and they may not provide optimal performance in water-sensitive formations (see Section 5.3.2)
20 (Montgomery, 2013; Gupta and Valkó, 2007). Water-based fluids can be as simple as water with a
21 few additives to reduce friction, such as "slickwater," or as complex as water with crosslinked
22 polymers, clay control agents, biocides, and scale inhibitors (Spellman, 2012).

23 Gels may be added to water-based fluids to increase viscosity, which assists with proppant
24 transport and results in wider fractures. Gelling agents include natural polymers, such as guar,
25 starches, and cellulose derivatives, which requires the addition of biocide to minimize bacterial
26 growth (Spellman, 2012; Gupta and Valkó, 2007). Gels may be linear or crosslinked. Crosslinking
27 increases viscosity without adding more gel. Gelled fluids require the addition of a breaker, which
28 breaks down the gel after it carries in the proppant, to reduce fluid viscosity to facilitate fluid
29 flowing back after treatment. (Spellman, 2012; Gupta and Valkó, 2007). The presence of residual
30 breakers may make it difficult to reuse recovered water (Montgomery, 2013).

### 5.3.2. Alternative Fracturing Fluids

31 Alternative hydraulic fracturing fluids can be used for water-sensitive formations (i.e., formations
32 where permeability is reduced when water is added) or as dictated by production goals
33 (Halliburton, 1988). Examples of alternative fracturing fluids include acid-based fluids; non-
34 aqueous-based fluids; energized fluids, foams or emulsions; viscoelastic surfactant fluids; gels;

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

1  methanol; and other unconventional fluids ([Montgomery, 2013](#); [Saba et al., 2012](#); [Gupta and Hlidek,](#)
2  [2009](#); [Gupta and Valkó, 2007](#); [Halliburton, 1988](#)).

3  **Acid fracturing** removes the need for a proppant and is generally used in carbonate formations.
4  Fractures are initiated with a viscous fracturing fluid, and the acid (gelled, foamed, or emulsified) is
5  added to irregularly etch the wall of the fracture and prop open the formation for a higher
6  conductivity fracture ([Spellman, 2012](#); [Gupta and Valkó, 2007](#)).

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

15 **Energized fluids, foams, and emulsions** minimize fluid leakoff, have high proppant-carrying
16 capacity, improve fluid recovery, and are sometimes used in water-sensitive formations ([Barati and](#)
17 [Liang, 2014](#); [Gu and Mohanty, 2014](#); [Spellman, 2012](#); [Gupta and Valkó, 2007](#); [Martin and Valko,](#)
18 [2007](#)).[1] However, these treatments tend to be expensive, require high pressure, and pose potential
19 health and safety concerns ([Montgomery, 2013](#); [Spellman, 2012](#); [Gupta and Valkó, 2007](#)).
20 **Energized fluids** are mixtures of liquid and gas ([Patel et al., 2014](#); [Montgomery, 2013](#)). Nitrogen
21 ($N_2$) or carbon dioxide ($CO_2$), the gases used, make up less than 53% of the fracturing fluid volume,
22 typically ranging from 25% to 30% by volume ([Montgomery, 2013](#); [Gupta and Valkó, 2007](#);
23 [Mitchell, 1970](#)). **Energized foams** are liquid-gas mixtures, with $N_2$ or $CO_2$ gas comprising more than
24 53% of the fracturing fluid volume, with a typical range of 70% to 80% by volume ([Mitchell, 1970](#)).
25 **Emulsions** are liquid-liquid mixtures, typically a hydrocarbon (e.g., condensate or diesel) with
26 water, with the hydrocarbon typically 70% to 80% by volume.[2] Both water-based fluids, including
27 gels, and non-aqueous fluids can be energized fluids or foams.

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

32 **Other types of fluids** not addressed above include viscoelastic surfactant fluids, viscoelastic
33 surfactant foams, crosslinked foams, liquid carbon dioxide-based fluid, and liquid carbon dioxide-
34 based foam fluid, and hybrids of other fluids ([King, 2010](#); [Brannon et al., 2009](#); [Curtice et al., 2009](#);

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[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.
[2] Diesel is a mixture typically of C8 to C21 hydrocarbons.

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1    Tudor et al., 2009; Gupta and Valkó, 2007; Coulter et al., 2006; Boyer et al., 2005; Fredd et al., 2004;
2    MacDonald et al., 2003).

3    Alternative fluids have been developed to work in tight formations, shales, and coalbeds, where
4    production is based on desorption of the natural gas, or in formations where the fracturing fluid
5    must displace a fluid that is already in place.

### 5.3.3. Proppants

6    Proppants are small particles carried down the well and into fractures by fracturing fluid. They hold
7    the fractures open after hydraulic fracturing fluid has been removed (Brannon and Pearson, 2007).
8    The propped fractures provide a path for the hydrocarbon to flow from the reservoir. Sand is most
9    commonly used, but other proppants include man-made or specially engineered particles, such as
10   resin-coated sand, high-strength ceramic materials, or sintered bauxite (Schlumberger, 2014;
11   Brannon and Pearson, 2007). Proppant types can be used individually or in combinations.

## 5.4. Frequency and Volume of Hydraulic Fracturing Chemical Use

12   This section highlights the different chemicals used in hydraulic fracturing and discusses the
13   frequency and volume of use. Based on the U.S. EPA analysis of the FracFocus 1.0 database (see Text
14   Box 5-2), we focus our analysis on individual chemicals rather than mixtures of chemicals used as
15   additives. Chemicals are reported to FracFocus by using the chemical name and the Chemical
16   Abstract Services Registration Number (CASRN), which is a unique number identifier for every
17   chemical substance.[1] The information on specific chemicals, particularly those most commonly
18   used, can be used to assess potential impacts to drinking water resources. The volume of chemicals
19   stored on-site provides information on the potential volume of a chemical spill.

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[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."

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**Text Box 5-2. 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](www.fracfocus.org)). Well operators can use the registry to disclose information about chemicals they use during hydraulic fracturing. The EPA accessed data from FracFocus 1.0 from January 1, 2011 to February 28, 2013, which included more than 39,000 disclosures from 20 states that had been submitted by operators prior to March 1, 2013.

Submission to FracFocus was initially voluntary and varied from state to state. During the timeframe of the EPA's study, six of the 20 states with data in the 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.

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. The EPA's analysis may or may not be nationally representative.

The EPA summarized information on the locations of the wells in the disclosures, water volumes used, and the frequency of use and concentrations (% by mass, reported as maximum ingredient concentration) of the chemicals in the additives and the hydraulic fracturing fluid. Additional information can be found in the EPA FracFocus report ([U.S. EPA, 2015a](U.S. EPA, 2015a)).

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The EPA compiled a list of 1,076 chemicals known to be have been used in the hydraulic fracturing process (see a full list, methodology, and the source citations in Appendix A). The chemicals used in hydraulic fracturing fall into different chemical classes and include both organic and inorganic chemicals. The chemical classes of commonly used hydraulic fracturing chemicals include but are not limited to:

- Acids (e.g., hydrochloric acid, peroxydisulfuric acid, acetic acid, citric acid).
- Alcohols (e.g., methanol, isopropanol, ethylene glycol, propargyl alcohol, ethanol).
- Aromatic hydrocarbons (e.g., benzene, naphthalene, heavy aromatic petroleum solvent naphtha).
- Bases (e.g., sodium hydroxide, potassium hydroxide).
- Hydrocarbon mixtures (e.g., petroleum distillates).
- Polysaccharides (e.g., guar gum).
- Surfactants (e.g., poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy, 2-butoxyethanol).
- Salts (e.g., sodium chlorite, dipotassium carbonate).

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1 | **Text Box 5-3. Confidential Business Information (CBI)**

2 | This assessment relies in large part upon information provided to the EPA or to other organizations. The
3 | submitters of that information (e.g., businesses that operate wells or perform services to hydraulically
4 | fracture the well) may view some of the information as confidential business information (CBI), and
5 | accordingly asserted CBI claims to protect such information. Information deemed to be CBI may include
6 | information such as trade secrets or other proprietary business information, entitled to confidential
7 | treatment under Exemption 4 of the Freedom of Information Act (FOIA) and other applicable laws. FOIA and
8 | the EPA's CBI regulations may allow for information claimed as CBI provided to the EPA to be withheld from
9 | the public, including in this document.

10 | The EPA evaluated data from FracFocus 1.0, a national hydraulic fracturing chemical registry used and relied
11 | upon by some states, industry groups and non-governmental organizations. A company submitting a
12 | disclosure to FracFocus may choose to not report the identity of a chemical it considers CBI. As part of the
13 | EPA's analysis, more than 39,000 FracFocus 1.0 disclosures over the period January 1, 2013 to March 1, 2013
14 | were analyzed and more than 70% of disclosures contained at least one chemical designated as CBI. Of the
15 | disclosures containing CBI chemicals, there was an average of five CBI chemicals per disclosure (U.S. EPA,
16 | 2015a). The prevalence of CBI claims in FracFocus 1.0 limits completeness of the data set.

17 | Consistent with the hydraulic fracturing study plan, data were submitted by nine service companies to the
18 | EPA regarding chemicals used in hydraulic fracturing from 2005 to 2009. Because this submission was to the
19 | EPA, the EPA was given the actual names and CASRNs of any chemicals the company considered CBI. This
20 | included a total of 381 CBI chemicals, with a mean of 42 CBI chemicals per company and a range of 7 to 213
21 | (U.S. EPA, 2013a).

### 5.4.1.  National Frequency of Use of Hydraulic Fracturing Chemicals

22 | The EPA reported that 692 chemicals were reported to FracFocus 1.0 for use in hydraulic fracturing
23 | from January 1, 2011, to February 28, 2013, with a total of 35,957 disclosures (U.S. EPA, 2015a).[1]

24 | Table 5-2 presents the 35 chemicals (5% of all chemicals identified in the EPA's study) that were
25 | reported in at least 10% of the FracFocus 1.0 disclosures for all states reporting to FracFocus
26 | during this time. This table also includes the top four additives that were reported to include the
27 | given chemical in FracFocus disclosures from January 1, 2011 to February 28, 2013.

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[1] The EPA reported that 692 chemicals were reported to FracFocus 1.0 for use in hydraulic fracturing from January 1, 2011, to February 28, 2013, with a total of 35,957 disclosures. Chemicals may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates), and they each have a single CASRN. Of these 692 chemicals, 598 had valid fluid and additive concentrations (34,675 disclosures). Sixteen chemicals were removed because they were minerals listed as being used as proppants. This left a total of 582 chemicals (34,344 disclosures).

**Table 5-2. Chemicals reported to FracFocus 1.0 from January 1, 2011 to February 28, 2013 in 10% or more disclosures, with the percent of disclosures for which each chemical is reported and the top four reported additives for the chemical.**

For chemicals with fewer than four reported additives, the table presents all additives (U.S. EPA, 2015b).

| No. | Chemical name[a] | CASRN | Percent of disclosures[b] | Chemical used in these additives (four most common, FracFocus database)[c] |
|---|---|---|---|---|
| 1 | Methanol | 67-56-1 | 72% | corrosion inhibitors, surfactants, non-emulsifiers, scale control |
| 2 | Hydrotreated light petroleum distillates[d] | 64742-47-8 | 65% | friction reducers, gelling agents and gel stabilizers, crosslinkers and related additives, viscosifiers |
| 3 | Hydrochloric acid | 7647-01-0 | 65% | acids, solvents, scale control, clean perforations |
| 4 | Water | 7732-18-5 | 48% | acids, biocides, clay control, scale control |
| 5 | Isopropanol | 67-63-0 | 47% | corrosion inhibitors, non-emulsifiers, surfactants, biocides |
| 6 | Ethylene glycol | 107-21-1 | 46% | crosslinkers and related additives, scale control, corrosion inhibitors, friction reducers |
| 7 | Peroxydisulfuric acid, diammonium salt | 7727-54-0 | 44% | breakers and breaker catalysts, oxidizer, stabilizers, clean perforations |
| 8 | Sodium hydroxide | 1310-73-2 | 39% | crosslinkers and related additives, biocides, pH control, scale control |
| 9 | Guar gum | 9000-30-0 | 37% | gelling agents and gel stabilizers, viscosifiers, clean perforations, breakers and breaker catalysts |
| 10 | Quartz[e] | 14808-60-7 | 36% | breakers and breaker catalysts, gelling agents and gel stabilizers, scale control, crosslinkers and related additives |
| 11 | Glutaraldehyde | 111-30-8 | 34% | biocides, surfactants, crosslinkers and related additives, sealers |
| 12 | Propargyl alcohol | 107-19-7 | 33% | corrosion inhibitors, inhibitors, acid inhibitors, base fluid |
| 13 | Potassium hydroxide | 1310-58-3 | 29% | crosslinkers and related additives, pH control, friction reducers, gelling agents and gel stabilizers |
| 14 | Ethanol | 64-17-5 | 29% | surfactants, biocides, corrosion inhibitors, fluid foaming agents and energizers |

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| No. | Chemical name[a] | CASRN | Percent of disclosures[b] | Chemical used in these additives (four most common, FracFocus database)[c] |
|---|---|---|---|---|
| 15 | Acetic acid | 64-19-7 | 24% | pH control, iron control agents, acids, gelling agents and stabilizers |
| 16 | Citric acid | 77-92-9 | 24% | iron control agents, scale control, gelling agents and gel stabilizers, pH control |
| 17 | 2-Butoxyethanol | 111-76-2 | 21% | surfactants, corrosion inhibitors, non-emulsifiers, fluid foaming agents and energizers |
| 18 | Sodium chloride | 7647-14-5 | 21% | breakers/breaker catalysts, friction reducers, scale control, clay control |
| 19 | Solvent naphtha, petroleum, heavy arom.[f] | 64742-94-5 | 21% | surfactants, non-emulsifiers, inhibitors, corrosion inhibitors |
| 20 | Naphthalene | 91-20-3 | 19% | surfactants, non-emulsifiers, corrosion inhibitors, inhibitors |
| 21 | 2,2-Dibromo-3-nitrilopropionamide | 10222-01-2 | 16% | biocides, clean perforations, breakers and breaker catalysts, non-emulsifiers |
| 22 | Phenolic resin | 9003-35-4 | 14% | proppants, biocides, clean perforations, base fluid |
| 23 | Choline chloride | 67-48-1 | 14% | clay control, clean perforations, base fluid, biocides |
| 24 | Methenamine | 100-97-0 | 14% | proppants, crosslinkers and related additives, biocides, base fluid |
| 25 | Carbonic acid, dipotassium salt | 584-08-7 | 13% | pH control, proppants, acids, surfactants |
| 26 | 1,2,4-Trimethylbenzene | 95-63-6 | 13% | surfactants, non-emulsifiers, corrosion inhibitors, inhibitors |
| 27 | Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides[g] | 68424-85-1 | 12% | biocides, non-emulsifiers, corrosion inhibitors, scale control |
| 28 | Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture)[h] | 127087-87-0 | 12% | surfactants, friction reducers, non-emulsifiers, inhibitors |
| 29 | Formic acid | 64-18-6 | 12% | corrosion inhibitors, acids, inhibitors, pH control |
| 30 | Sodium chlorite | 7758-19-2 | 11% | breakers/breaker catalysts, biocides, oxidizer, proppants |
| 31 | Nonyl phenol ethoxylate | 9016-45-9 | 11% | non-emulsifiers, resin curing agents, activators, friction reducers |

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| No. | Chemical name[a] | CASRN | Percent of disclosures[b] | Chemical used in these additives (four most common, FracFocus database)[c] |
|---|---|---|---|---|
| **32** | Tetrakis(hydroxymethyl)phosphonium sulfate | 55566-30-8 | 11% | biocides, scale control, clay control |
| **33** | Polyethylene glycol | 25322-68-3 | 11% | biocides, non-emulsifiers, surfactants, clay control |
| **34** | Ammonium chloride | 12125-02-9 | 10% | friction reducers, crosslinkers and related additives, scale control, clay control |
| **35** | Sodium persulfate | 7775-27-1 | 10% | breakers and breaker catalysts, oxidizer, pH control |

[a] Chemical refers to chemical substances with a single CASRN, these may be pure chemicals (e.g., methanol) or chemical mixtures (e.g., hydrotreated light petroleum distillates).

[b] Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

[c] Analysis considered 32,885 disclosures and 615,436 ingredient records that met selected quality assurance criteria, including: completely parsed; 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.

[d] Hydrotreated light petroleum distillates (CASRN 64742-47-8) is a mixture of hydrocarbons, in the C9 to C16 range.

[e] Quartz (CASRN 14808-60-7) the proppant most commonly reported, was also reported as an ingredient in other additives U.S. EPA (2015a).

[f] Heavy aromatic solvent naphtha (petroleum) (CASRN 64742-94-5) is mixture of aromatic hydrocarbons, in the C9 to C16 range.

[g] Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides (CASRN 68424-85-1) is a mixture of benzalkonium chloride with carbon chains between 12 and 16.

[h] Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture) (CASRN 127087-87-0) is mixture with varying length ethoxy links.

1  There is no single chemical used at all wells across the nation. Methanol is the most commonly used
2  chemical, reported at 72.1% of wells in FracFocus 1.0, and is associated with 33 types of additives,
3  including corrosion inhibitors, surfactants, non-emulsifiers, and scale control (U.S. EPA, 2015b).
4  Table 5-2 also shows the variability in different chemicals reported to FracFocus 1.0. The
5  percentage of disclosures reporting a given chemical suggests the likelihood of that chemical's use
6  at a site. Only three chemicals (methanol, hydrotreated light petroleum distillates, and hydrochloric
7  acid) were used at more than half of the sites nationwide, and only 12 were used at more than one-
8  third.

9  In addition to providing information on frequency of use, FracFocus 1.0 data provides the
10  maximum concentration by mass of a given chemical in an additive. For example, for the most
11  frequently used chemical, methanol, the median maximum additive concentration reported in
12  FracFocus disclosures is 30%, by mass, with a range of 0.44% to 100% (5th to 95th percentile). This
13  suggests that methanol is generally used as part of a mixture of chemicals in the hydraulic
14  fracturing fluid, and may be stored in a mixture of chemicals or as pure methanol. This wide range

1 of possible concentrations of methanol further complicates assessing the potential impact of spills,
2 as the properties of the fluid will depend on the different chemicals present and on their
3 concentrations. For all chemicals, spills of a highly concentrated chemical can have different
4 potential impacts than spills of dilute mixtures.

### 5.4.2.  Nationwide Oil versus Gas

5 FracFocus 1.0 data also can elucidate the differences between the chemicals used for oil production
6 and those used for gas production, providing a better understanding of potential spill impacts from
7 each. Table C-1 and C-2 in Appendix C present the chemicals reported in at least 10% of all gas (34
8 chemicals) and oil (39 chemicals) disclosures nationwide.

9 Many of the same chemicals are used for oil and gas, but some chemicals are used more frequently
10 in oil production and others more frequently in gas.[1] For example, hydrochloric acid is the most
11 commonly reported chemical for gas wells (73% of disclosures); it is the fifth most frequently
12 reported chemical for oil wells (58% of disclosures). However, both oil and gas operators each
13 reports using methanol in 72% of disclosures. Methanol is the most common chemical used in
14 hydraulic fracturing fluids at oil wells and the second most common chemical in hydraulic
15 fracturing fluids at gas wells.

### 5.4.3.  State-by-State Frequency of Use of Hydraulic Fracturing Chemicals

16 We conducted a state-by-state analysis of chemical use based on FracFocus 1.0 disclosures (U.S.
17 EPA, 2015b). Some states reported more disclosures than other states, because they have relatively
18 more hydraulic fracturing activity and/or greater numbers of disclosures to FracFocus 1.0.
19 Reporting can bias national numbers towards those states with a disproportionate number of
20 disclosures. For example, the EPA (2015a) reported that Texas had 16,405 of the 34,675
21 disclosures with parsed ingredients and valid CASRNs and concentrations, making up almost half
22 (47%) of all disclosures for the 20 states reporting to FracFocus 1.0. We attempt to account for the
23 possible effect of having a large number of disclosures in Texas by looking at a compilation of the
24 top 20 chemicals reported to FracFocus for all states.

25 Table 5-3 presents and ranks chemicals reported most frequently to FracFocus 1.0 for each state
26 (U.S. EPA, 2015b). There are 94 unique chemicals comprising the top 20 chemicals for each state,
27 indicating similarity in chemical usage among states.

28 Methanol is reported in 19 of the 20 (95%) states. Alaska is the only state in which methanol is not
29 reported (based on the state's 20 disclosures to FracFocus). The percentage of disclosures
30 reporting use of methanol ranges from 38% (Wyoming) to 100% (Alabama, Arkansas).

31 Ten chemicals (excluding water) are among the 20 most frequently reported in 14 of the 20 states.
32 These chemicals are: methanol; hydrotreated light petroleum distillates; ethylene glycol;

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[1] This separation was done solely based on whether it was an oil or gas disclosure. The analysis did not separate out reservoir factors, such as temperature, pressure, or permeability, which may be important factors for which chemicals are used.

1   isopropanol; quartz; sodium hydroxide; ethanol; guar gum; hydrochloric acid; and peroxydisulfuric
2   acid, diammonium salt.[1] These 10 chemicals are also the most frequently reported chemicals
3   nationwide.

4   By performing this analysis by state, we observed that methanol is used across the continental U.S.
5   (not Alaska), and there are 9 other chemicals that are frequently used across the U.S. Beyond those,
6   however, there are a number of different chemicals that are used in one state more commonly than
7   others and many may not be used at all in other states. This suggests that there is regional
8   variability in some chemicals and a common set of the same chemicals that are frequently used.

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[1] Quartz was the most commonly reported proppant and also reported as an ingredient in other additives (U.S. EPA, 2015a).

**Table 5-3. The percentage of disclosures of the 20 most commonly reported chemical by state, where a chemical is reported in at least three states.**

The 20 most frequently reported chemicals were identified for all 20 states that reported to FracFocus 1.0 (U.S. EPA, 2015b). The chemicals were ranked by counting the number of states where that chemical was in the top 20. Chemicals were then ranked so that chemicals used most widely among the most states come first. Methanol is reported in 19 of 20 states, so methanol is ranked first. A chemical was only presented on the list if it were reported in at least three states, resulting in 33 chemicals. The full table of top 20 chemicals (91 chemicals) is presented in Appendix C.

| Chemical name | CASRN | Percentage of disclosures per state[a] | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | AL | AK | AR | CA | CO | KS | LA | MI | MS | MT | NM | ND | OH | OK | PA | TX | UT | VA | WV | WY |
| Methanol | 67-56-1 | 100% | | 100% | 39% | 63% | 79% | 59% | 93% | 75% | 63% | 91% | 53% | 52% | 70% | 69% | 78% | 79% | 61% | 64% | 38% |
| Distillates, petroleum, hydrotreated light | 64742-47-8 | | 45% | 56% | 55% | 74% | 90% | 84% | 100% | 100% | 60% | 63% | 47% | 84% | 70% | 60% | 66% | 75% | | 82% | 51% |
| Ethylene glycol | 107-21-1 | 100% | 100% | 22% | 60% | | 63% | 34% | 71% | 75% | 49% | 45% | 36% | 57% | 47% | 34% | 59% | 85% | 28% | 59% | |
| Isopropanol | 67-63-0 | 100% | 65% | 44% | | 57% | 25% | 51% | 79% | | 64% | 62% | 37% | 49% | 42% | 31% | 48% | 53% | 54% | 31% | 43% |
| Quartz | 14808-60-7 | | 100% | | 89% | 23% | 23% | 37% | | 50% | 64% | 68% | 46% | 45% | 27% | | 43% | 40% | | 22% | 30% |
| Sodium hydroxide | 1310-73-2 | | 100% | 21% | 69% | 22% | 28% | 53% | | 50% | 54% | 30% | 52% | | 27% | 17% | 46% | 37% | | | 57% |
| Ethanol | 64-17-5 | | | 45% | | 50% | 80% | 42% | | 100% | | 47% | 27% | 60% | 46% | 16% | 21% | | 63% | 54% | 25% |
| Guar gum | 9000-30-0 | | 50% | | 93% | | | 49% | | 50% | 43% | 63% | 55% | 51% | 25% | 23% | 43% | 43% | | 23% | 69% |
| Hydrochloric acid | 7647-01-0 | 100% | | 99% | | 53% | 85% | 57% | | | 23% | 79% | | 99% | 76% | 96% | 71% | 85% | 86% | 96% | |
| Peroxydisulfuric acid, diammonium salt | 7727-54-0 | | 50% | | 83% | | 22% | 27% | 57% | | 62% | 75% | 55% | 64% | 39% | | 54% | 39% | | 54% | 64% |
| Propargyl alcohol | 107-19-7 | | | 61% | | | 71% | 30% | 36% | | | 68% | | 49% | 41% | 58% | 39% | 36% | 28% | 58% | |
| Glutaraldehyde | 111-30-8 | | | 55% | | | 75% | 36% | | 50% | | 57% | | 72% | 55% | 34% | 40% | | | 71% | 22% |
| Naphthalene | 91-20-3 | 100% | | | | 30% | 42% | 29% | 86% | | 49% | | 43% | | 25% | | | 38% | 9% | | |
| 2-Butoxyethanol | 111-76-2 | 100% | 100% | | | | | | 79% | | 37% | | | | | 21% | 24% | 53% | 89% | 26% | |

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| Chemical name | CASRN | Percentage of disclosures per state[a] | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | AL | AK | AR | CA | CO | KS | LA | MI | MS | MT | NM | ND | OH | OK | PA | TX | UT | VA | WV | WY |
| Citric acid | 77-92-9 | | | | | | 46% | | | | | 40% | | 66% | 36% | 29% | 24% | 79% | 80% | 41% | |
| Sodium chloride | 7647-14-5 | | | | | 35% | | 41% | | 50% | | | 25% | | | 21% | | 9% | | 22% | 23% |
| Solvent naphtha, petroleum, heavy aromatics | 64742-94-5 | | | | | 33% | 43% | | | | 70% | | 49% | | 31% | 17% | | | 9% | | 35% |
| Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides | 68424-85-1 | | | 28% | | | 54% | | | 50% | | | | 37% | 33% | 16% | | | | 22% | |
| 2,2-Dibromo-3-nitrilopropionamide | 10222-01-2 | 100% | | | | 49% | | | 71% | 36% | | | | | | 34% | | | 28% | | |
| Potassium hydroxide | 1310-58-3 | | | | | | | 34% | | 100% | 60% | | 59% | 73% | | 39% | | | | | |
| Acetic acid | 64-19-7 | | | | | 21% | 28% | | | | | | | | 24% | 31% | | | | | 31% |
| Choline chloride | 67-48-1 | | | | | 27% | | | | | | 34% | | | 38% | | | 52% | | 57% | |
| Polyethylene glycol | 25322-68-3 | 100% | | | | | | | 50% | | 36% | | 29% | | | 29% | | | | | |
| 1,2,4-Trimethylbenzene | 95-63-6 | | | | | 27% | 40% | | | | | | 25% | | | | | | 9% | | |
| Ammonium chloride | 12125-02-9 | | | 21% | | 28% | | | | | | | | | | 31% | | | | 21% | |
| Diatomaceous earth, calcined | 91053-39-3 | | 100% | | 71% | | | | | | | 38% | | | | | | 35% | | | |
| Didecyl dimethyl ammonium chloride | 7173-51-5 | | | 24% | | | | | | 50% | | | | | 32% | | | | | 21% | |
| Sodium chlorite | 7758-19-2 | | | | | | | 35% | | 100% | | | 24% | | | | | | | | 23% |
| Sodium erythorbate | 6381-77-7 | | | 33% | | 30% | | | | | | | | | | | | 13% | | 32% | |
| N,N-Dimethylformamide | 68-12-2 | | | | | | | | | | | | | 47% | 20% | | | 33% | | | |

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| Chemical name | CASRN | Percentage of disclosures per state[a] | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | AL | AK | AR | CA | CO | KS | LA | MI | MS | MT | NM | ND | OH | OK | PA | TX | UT | VA | WV | WY |
| **Nonyl phenol ethoxylate** | 9016-45-9 | | | | | | | | | | | 30% | | | | | | 36% | 32% | | |
| **Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture)** | 127087-87-0 | | | | | 25% | 40% | | | | | | | | | | | | 9% | | |
| **Sodium persulfate** | 7775-27-1 | | | | | | | | | 100% | | | | | | 16% | | | | | 26% |
| **Tetramethylammonium chloride** | 75-57-0 | | | | | | | | | | 44% | | 29% | | | | | | | | 26% |

[a] Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

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### 5.4.4. Volumes of Chemicals Used

1    Understanding the volume of chemicals used at each hydraulic fracturing site is important for
2    understanding potential impacts of chemicals to drinking water resources, because the chemical
3    volume governs how much will be stored on-site, the types of containers required, and the total
4    amount that could spill. While the on-site operator has precise knowledge of the composition and
5    volume of chemicals stored on-site, this information is generally not publicly available. We
6    conducted a comprehensive review of publicly available sources and found two sources (OSHA,
7    2014a, b; Sjolander et al., 2011) that identify specific chemicals used at a hydraulic fracturing site
8    and provide information on volumes. These are presented in Table 5-4. The volume of chemicals
9    totaled 7,500 gal (28,000 L) and 14,700 gal (56,000 L) for the two sources, with a mean volume for
10   an individual chemical of 1,900 gal (7,000 L) and 1,225 gal (4,600 L), respectively. The range of
11   volumes for each chemical used is 30 to 3,690 gal (114 to 14,000 L).

**Table 5-4. Example list of chemicals and volumes used in hydraulic fracturing.**

Volumes are for wells with and unknown number of stages and at least one perforation zone. Every well and fluid formation is unique. Volumes may be larger for longer horizontal laterals and with a greater number of stages.

| Ingredient | Examples | Sjolander et al. (2011)[a] | | Occupational Safety and Health Administration (OSHA, 2014a, b)[b] | |
| --- | --- | --- | --- | --- | --- |
| | | Volume (gal) or mass (lbs) | Percent overall[c] | Volume (gal) | Percent by volume |
| Water | | 4,000,000 gal | 94.62 | 2,700,000 | 90 |
| Proppant | Sand | ~ 1,500,000 lbs[d] | 5.17 | 285,300 | 9.51 |
| Acid | Hydrochloric acid or muriatic acid | 1,338 gal | 0.03 | 3,690 | 0.123 |
| Friction reducer | Polyacrylamide, mineral oil | 2,040 gal | 0.05 | 2,640 | 0.088 |
| Surfactant | Isopropanol | | | 2,550 | 0.085 |
| Potassium chloride | | | | 1,800 | 0.06 |
| Gelling agent | Guar gum or hydroxymethyl cellulose | -[e] | -[e] | 1,680 | 0.056 |
| Scale inhibitor | Ethylene glycol, alcohol, and sodium hydroxide | | | 1,290 | 0.043 |

| Ingredient | Examples | Sjolander et al. (2011)[a] | | Occupational Safety and Health Administration (OSHA, 2014a, b)[b] | |
|---|---|---|---|---|---|
| | | Volume (gal) or mass (lbs) | Percent overall[c] | Volume (gal) | Percent by volume |
| pH buffer | Carbonate | | | 330 | 0.011 |
| Preservative | Ammonium persulfate | | | 300 | 0.01 |
| Crosslinker | Borate salts | -[e] | -[e] | 210 | 0.007 |
| Iron control | Citric acid | -[e] | -[e] | 120 | 0.004 |
| Corrosion inhibitor | n,n-Dimethyl formamide | -[e] | -[e] | 60 | 0.002 |
| Biocide / antimicrobial agent | Glutaraldehyde, ethanol, methanol | 2,040 gal | 0.05 | 30 | 0.001 |
| Gel-breaker | Ammonium persulfate | -[e] | -[e] | | |
| All chemicals | | 7,458 gal | 0.21 | 14,700 | 0.49 |
| Chemical Volume: Mean (full range) | | 1,864.5 gal (1,338 – 2,040 gal) | | 1,225 (30 – 3,690) | |

[a] Adapted from Penn State "Water Facts" publication entitled "Introduction to Hydrofracturing" (Sjolander et al., 2011). Composite from two companies: Range Resources, LLC, and Chesapeake Energy, which released in July 2010 the chemistry and volume of materials typically used in their well completions and stimulations.

[b] Adapted from a table generated by the OSHA for use in a training module (OSHA, 2014a, b).

[c] As presented in Sjolander et al. (2011); does not explicitly state percent by mass or by volume.

[d] The Penn State publication presented proppant in pounds instead of gallons.

[e] Listed as an ingredient, but no information on volume or percentage.

1    Because of the limited information on chemical volumes publicly available, we estimated chemical
2    volumes used across the nation based on the information provided in the FracFocus database.
3    Figure 5-5 plots median estimated chemical volumes, ranked from high to low, with the range of 5th
4    to 95th percentiles.[1] Volumes used are presented for the 74 chemicals that were reported to
5    FracFocus in at least 100 disclosures and for which density data were available.[2] The estimated
6    median volumes vary widely among the different chemicals, covering a range of near zero to 27,000
7    gal (98,000 L). The mean of the estimated median volumes was 650 gal (2,500 L).[3]

8    With the median chemical volume, we can estimate total chemical volume for all chemicals used.
9    Based on the above mean of median chemical volumes of 650 gal (2,500 L) per chemical, and given
10   that the median number of chemicals used at a site is 14 (U.S. EPA, 2015a), an estimated 9,100 gal

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[1] Volumes were estimated using FracFocus disclosures. The total hydraulic fracturing fluid volume reported was used to calculate the total fluid mass by assuming the fluid has a density of 1 g/mL. This is a simplifying assumption based on the fact that more than 93% of disclosures are inferred to use water as a base fluid. Water had a median concentration of 88% by mass in the fracturing fluid. Some disclosures reported using brine, which has a density between 1.0 and 1.1 g/mL. This would introduce at most an error of 10% for the fluid calculation (the difference of a chemical being present at 10 versus 9 gal, 1,000 versus 900 gal). We also assume that the mass of chemicals present in calculating the total fluid mass is negligible. Given that ≤2% of the fluid volume is non-water chemicals, and assuming the density of which is 3 mg/L, the error introduced is approximately 6%. For reference, for the chemicals we are calculating volumes, chlorine dioxide is the densest at 2.757 mg/L. Chemical with densities less than 1 mg/L introduce approximately <1% error.

Next, the mass of each chemical per disclosure was calculated. Each chemical is reported to FracFocus 1.0 as a maximum concentration by mass in the hydraulic fracturing fluid. This introduces error, as we only know that it is equal to or less than this mass fraction. In the U.S. EPA (2015a) EPA's analysis of the FracFocus 1.0 database, an additive is comprised of three chemicals with maximum ingredient concentration of 60% in the additive and a maximum concentration of 0.22% in the hydraulic fracturing fluid. Each of the three chemicals cannot be present at 60%. We have no way to know the actual proportions of each chemical in the additive and thus must calculate chemical mass based on the given information. Therefore, our calculations likely overestimate actual volumes. However, in some cases, the concentration in the additive that is given is less than 100% and only one chemical is listed in the additive. In these cases, it appears that the disclosure is reporting the concentration of that chemical in water. Hydrogen chloride is listed as the sole ingredient in the acid additive, and the maximum concentration is 40% by mass. In this case, the HCl is diluted down to 40%, so the total volume would be underestimated.

After all the chemical masses are calculated, the volume is calculated by dividing chemical mass by density.

Given the limited information available, due to the limits of the FracFocus database and general lack of publicly available data, and despite the errors associated with these calculations, these calculations provide context for the general magnitude of volumes for each of the chemicals used on-site. These calculations are used to calculate median volumes for each chemical. These volume calculations are for the chemicals themselves, not the additives.

Analysis considered 34,495 disclosures and 672,358 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; criteria for water volumes; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (4,035) or other, query-specific criteria were excluded from analysis.

[2] Density data were gathered from Reaxys® and other sources as noted. Reaxys® (http://www.elsevier.com/online-tools/reaxys) is an online database of chemistry literature and data. Direct density source, as provided by Reaxys®, is provided in Table C-7 in Appendix C.

[3] Reporting records to FracFocus 1.0 was required in six of the 20 states between January 1, 2011 and February 28, 2013. An additional three states required disclosure to either FracFocus or the state, and five states required reporting to the state. Reporting to FracFocus 1.0 was optional in other states. Some states changed their reporting requirements during the course of the study. The FracFocus 1.0 database therefore does not encompass all data on chemicals used in hydraulic fracturing. As stated in Text Box 4-2, this mix of voluntary versus mandatory disclosure requirements limits the completeness of data extracted from FracFocus 1.0 for estimating hydraulic fracturing fluid. According to a comparison between FracFocus reported fluid volumes and literature values, water use per well was reported to be about 86% of the actual used (median of estimated values. See Chapter 4, Text Box 4-1). If the fluid volume is underreported, then estimated chemical volumes based on fluid volume would be similarly underestimated. Using the underreporting of 86%, then the estimated median chemical volume would be 760 gal.

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1  (34,000 L) of chemicals may be used per well. Given that the number of chemicals per well ranges
2  from 4 to 28 (U.S. EPA, 2015a), the total volume of chemicals per well may range from 2,600 to
3  18,000 gal (9,800 to 69,000 L).

4  Another way to estimate total volume of chemicals per well is to use the estimated median volume
5  of 1.5 million gal (5.7 million L) of fluid used to fracture a well (see Chapter 4) (U.S. EPA, 2015a)
6  and assume that up to 2% of that volume are chemicals added to base fluid (Carter et al., 2013;
7  Knappe and Fireline, 2012), resulting in up to 30,000 gal (114,000 L) of chemicals used per well.
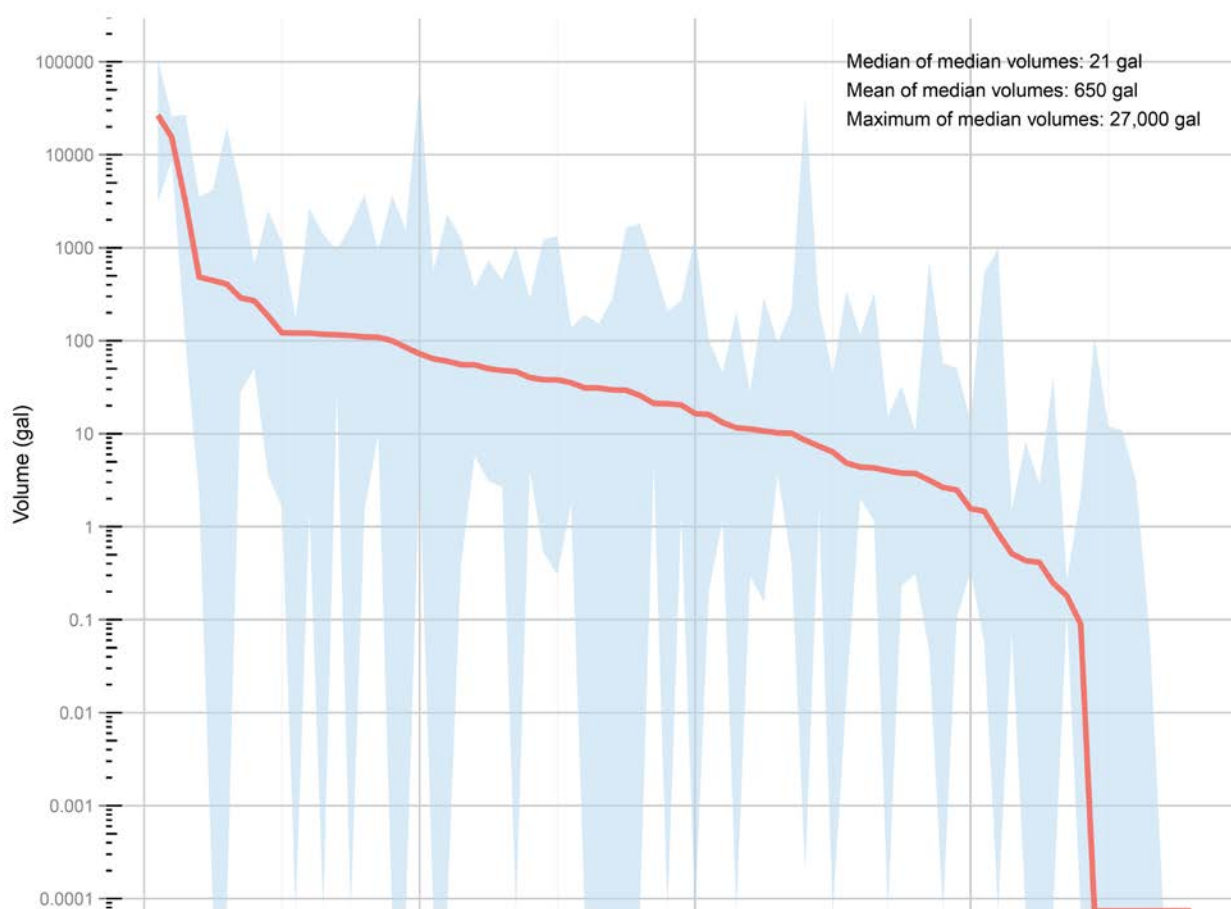


**Figure 5-5. Estimated median volumes for chemicals reported in at least 100 FracFocus**
**disclosures by February 28, 2013 for use in hydraulic fracturing from**
**January 1, 2011 to February 28, 2013.**

Shaded area represents the zone of 5% and 95% confidence limits. Derived from (U.S. EPA, 2015b).

8   Using the mean of the median chemical volumes from disclosures in FracFocus 1.0, we can also
9   estimate volume per additive and extrapolate to estimate on-site chemical storage. If we assume
10  three to five chemicals per additive, then total volume per additive stored on-site would
11  approximate 1,900 to 3,200 gal (7,400 to 12,000 L). On-site containers generally store 20% to

1  100% more additive volume than needed ([Houston et al., 2009](#); [Malone and Ely, 2007](#)). This would
2  suggest that 2,300 to 6,500 gal (8,800 to 25,000 L) per additive are stored on-site.

3  The volume that may be released during a spill depends on where in the chemical mixing process
4  the spill occurs. Spills from chemical or additive containers may result in higher volume spills than
5  the estimated volumes in Figure 5-5 suggest. However, if the spill is of the hydraulic fracturing fluid
6  downstream of the blender, then the total volume of chemical spilled may be less than the
7  estimated total volumes held on site.

## 5.5. Chemical Management and Spill Potential

8  This section provides a description of the primary equipment used in the chemical mixing and well
9  injection processes, along with a discussion of the spill vulnerabilities specific to each piece of
10 equipment. Equipment breakdown or failure can trigger a spill itself, and it can also lead to a
11 suspension of activity and the disconnection and reconnection of various pipes, hoses, and
12 containers. Each manipulation of equipment poses additional potential for a spill. The EPA found
13 that approximately one-third of chemical spills on or near the well pad related to hydraulic
14 fracturing resulted from equipment failure ([U.S. EPA, 2015n](#)). When possible, we describe
15 documented spills associated with or attributed to specific pieces of equipment, in text boxes in the
16 relevant subsections.

17 Hydraulic fracturing operations are mobile and must be assembled at each well site, and each
18 assembly and disassembly presents a potential for spills. Equipment used in the chemical mixing
19 and well injection processes typically consists of chemical storage trucks, oil storage tanks/tanker
20 trucks; a slurry blender; one or more high-pressure, high-volume fracturing pumps; the main
21 manifold; surface lines and hoses; and a central control unit. There are many potential sources for
22 leaks and spills in this interconnected system.

23 Equipment varies in age and technological advancement depending upon service company
24 standards and costs associated with purchase and maintenance. Older equipment may have
25 experienced wear and tear, which may be a factor in spills caused by equipment failure. New
26 equipment may be more automated, reducing opportunities for human error. Information detailing
27 the extent of technological and age differences in fracturing equipment across sites and operators is
28 limited. Table 5-5 provides a description of typical hydraulic fracturing equipment.

**Table 5-5. Examples of typical hydraulic fracturing equipment and their functions.**

| Equipment | Function |
|---|---|
| Acid transport truck | Transports acids to job sites, the truck has separate compartments for multiple acids or additives. |
| Chemical storage truck | Transport chemicals to the site in separate containment units or totes. Chemicals are typically stored on and pumped from the chemical storage truck. |
| Base fluid tanks | Store the required volume of base fluid to be used in the hydraulic fracturing process. |
| Proppant storage units | Hold proppant and feed it to the blender via a large conveyor belt. |
| Blender | Takes fluid (e.g., water) from the fracturing tanks and proppant (e.g., sand) from the proppant storage unit and combines them with additives before transferring the mixture to the fracturing pumps |
| High-pressure fracturing pumps | Pressurize mixed fluids received from the blender and injected into the well. |
| Manifold trailer with hoses and pipes | A transfer station for all fluids. Includes a trailer with a system of hoses and pipes connecting the blender, the high-pressure pumps, and the fracturing wellhead. |
| Fracturing wellhead or frac head | A wellhead connection that allows fracture equipment to be attached to the well. |

1  While the primary equipment and layout is generally the same across well sites, the type, size, and
2  number of pieces of equipment may vary depending on a number of factors (Malone and Ely, 2007):

3  • The size and type of the fracture treatment;

4  • The number of wells drilled per well pad;

5  • The location, depth, and length of the fractures;

6  • The volumes and types of additives, proppants, and fluids used; and

7  • The operating procedures of the well operator and service company (e.g., some companies
8     require backup systems in case of mechanical failure, while others do not).

9  Figure 5-6 provides a schematic diagram of a typical layout of hydraulic fracturing equipment.
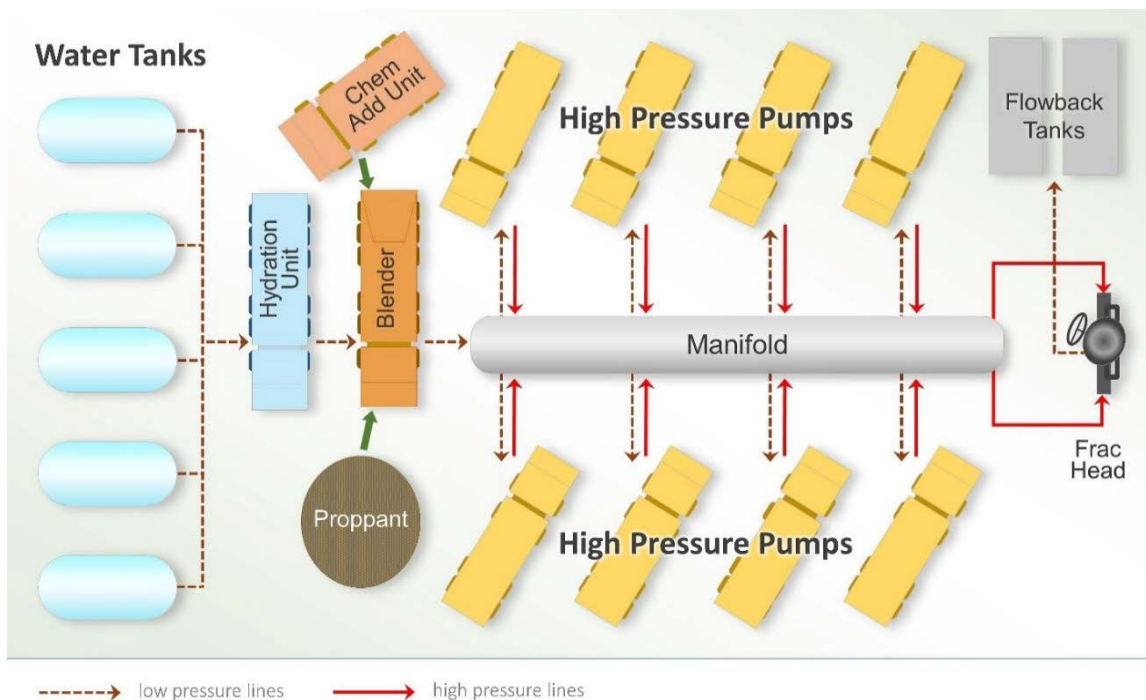
**Figure 5-6. Typical hydraulic fracturing equipment layout.**

> This illustration shows how the various components of a typical hydraulic fracturing site fit together. Numbers of pumps and tanks vary from site to site. Some sites do not use a hydration unit as the gel is batch mixed prior to the treatment (Olson, 2011; BJ Services Company, 2009).

### 5.5.1. Storage

1    This section provides an overview of publicly available information on storage and containment of
2    chemicals used in the hydraulic fracturing process. Most public sources provide general
3    information on the types and sizes of containment units. While operators maintain a precise
4    inventory of volumes of chemicals stored and used for each site, this information is typically not
5    made public.

6    The volumes of each chemical used are based on the size and site-specific characteristics of each
7    fracture treatment. Sites often store an excess of the design volume of chemicals for contingency
8    purposes. Malone and Ely (2007) indicate that companies typically store an excess of 20% of the
9    required chemical inventories on location. Houston et al. (2009) recommends maintaining an on-
10   site chemical reserve of 100% extra beyond what is necessary for the fracturing treatment
11   (Houston et al., 2009). See Text Box 5-4 for documented spills from storage units.

---

**Text Box 5-4. Spills from Storage Units.**

1 Of the 151 spills of chemicals, additives, or fracturing fluid discussed and evaluated in (U.S. EPA, 2015n) (see
2 Text Box 5-13 for more information), 54 spills were from storage units. Storage units include smaller totes or
3 tanks used for storing individual chemicals or additives, and larger tanks containing fracturing fluid. These
4 spills resulted from equipment failure, failure of storage integrity, or human error. Sixteen of these spills were
5 due to failure of container integrity, which includes holes and cracks in containers, demonstrating the need
6 for properly constructed and maintained storage units. The remaining spills from storage containers resulted
7 from human error or equipment malfunctions, or had an unknown cause.

### 5.5.1.1.  Hydraulic Fracturing Base Fluid Storage

8 Base fluids used in hydraulic fracturing are typically stored on-site in large volume tanks. Non-
9 water-based fluids may be stored in specialized containment units designed to prevent or minimize
10 releases. For example, nitrogen and carbon dioxide must be stored in compressed gas or cryogenic
11 liquid cylinders, as required by U.S. Department of Transportation (DOT) and OSHA regulations.
12 Due to the large volume of base fluid storage tanks (about 21,000 gal (80,000 L) (Halliburton,
13 1988), uncontrolled spills could damage other storage units and equipment, which could result in
14 additional spills. Fresh water used as a base fluid is generally not a source of concern for spills.
15 Reused wastewater, brine, and non-aqueous base fluids have the potential to adversely impact
16 drinking water resources in the event of a spill. An example of a documented spill of hydraulic
17 fracturing fluid is presented in Text Box 5-5.

---

**Text Box 5-5. Spills of Fracturing Fluid Documented to Impact Surface Water.**

18 In September 2009, two spills of hydraulic fracturing fluid occurred at the same site in Pennsylvania. A total
19 of approximately 7,350 gal (28,000 L) of fluid comprised of a mixture of water, gel, and friction reducer
20 leaked and migrated to Stevens Creek. While the causes of the spills are not clear, it appears that a
21 pressurized line may have broken and a seal may have failed (U.S. EPA, 2015n, Appendix B Line 307;
22 Lustgarten, 2009).

### 5.5.1.2.  Chemical Additive Storage

23 Additives are typically stored on-site in the containers in which they were transported and
24 delivered. The chemical additive trailer typically consists of a flatbed truck or van enclosure that
25 holds a number of chemical totes, described below, and is equipped with metering pumps that feed
26 chemicals to the blender. Depending on the size and type of the fracturing operation, there may be
27 one or more chemical additive trailers per site (ALL Consulting, 2012; NYSDEC, 2011). Additives
28 constitute a relatively small portion of fluids used in a hydraulic fracturing fluid, although they can
29 still be used in volumes ranging from the tens to tens of thousands of gallons.

30 The storage totes generally remain on the transportation trailers, but they also may be unloaded
31 from the trailers and transferred to alternative storage areas before use. Our investigation did not
32 find much information on how often, when, or why these transfers occur. Additional transfers and

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1  movement can increase the likelihood of a spill. See Text Box 5-6 for documented spills from an
2  additive storage unit.

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**Text Box 5-6. Spill from Additive (Crosslinker) Storage Tote.**

3  During a tote transfer in Pennsylvania, a tote of crosslinker fell off a forklift, spilling approximately 15–20 gal
4  (60–80 L) onto the well pad. The area was scraped clean with a backhoe and placed in a lined containment
5  area (U.S. EPA, 2015n, Line 309).

---

6  The most commonly used chemical totes are 200–400 gal (760–1,500 L) capacity polyethylene
7  containers that may be reinforced with steel or aluminum mesh (NYSDEC, 2011). Metal containers
8  of the same capacity may also be used (ALL Consulting, 2012; UWS, 2008). The totes are typically
9  equipped with bottom release ports, which enable the direct feed of the additives to the blending
10 equipment (NYSDEC, 2011). Spills may occur if lines are improperly connected to these ports or if
11 the connection equipment is faulty.



**Figure 5-7. Metal and high-density polyethylene (HDPE) chemical additive units.**

The image on the left depicts metal totes (industry source). The image on the right depicts plastic
totes (NYSDEC, 2011).

12 Certain additives require specialized containment units with added spill prevention measures. For
13 example, additives containing methanol may be subject to federal safety standards, and industry
14 has developed guidance on methanol's safe storage and handling (Methanol Institute, 2013).

15 Dry additives are typically transported and stored on flatbed trucks in 50 or 55 lb (23 or 25 kg)
16 bags, which are set on pallets containing 40 bags each (NYSDEC, 2011; UWS, 2008; Halliburton,
17 1988). Proppants are stored on-site in large tanks or bins with typical capacities of 220 to 440 lb
18 (100 to 200 kg) (ALL Consulting, 2012; BJ Services Company, 2009; Halliburton, 1988).

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### 5.5.1.3. Acid Storage

1   Acids are generally stored on-site in the containment units in which they are transported and
2   delivered. A typical acid transport truck holds 3,000 to 5,000 gal (11,400 to 19,000 L) of acid and
3   can have multiple compartments to hold different kinds of acid (Arthur et al., 2009b). Acids such as
4   hydrochloric acid and formic acid are corrosive and can be extremely hazardous in concentrated
5   form. Therefore, acid transport trailers and fracture tanks must be lined with chemical-resistant
6   coating designed to prevent leakage and must meet applicable DOT regulatory standards (pursuant
7   to 40 CFR 173) designed to prevent or minimize spills.

8   Acid fracture treatments may use thousands of gallons (thousands of liters) of acid per treatment
9   (Spellman, 2012). Given the large volumes used, failure of containment vessels during storage or
10  failure of connections and hoses during pumping could result in high-volume acid spills. Eight spills
11  (out of 105 spills from state data sources) of acid or fracturing fluid containing acid were reported
12  to state data sources examined by the EPA (2015n). The median volume of these acid spills was
13  approximately 1,600 gal (6,000 L) (Lines 240, 241, 248, 258, 264, 272, 278, and 281 in Appendix B
14  of U.S. EPA (2015n)).

15  Details of a documented acid spill are presented in Text Box 5-7.

---

**Text Box 5-7. Spill of Acid from Storage Container.**

16  In July 2014, over 20,000 gal (76,000 L) of hydrochloric acid spilled from a storage container when a flange
17  malfunctioned. The acid spilled into a nearby alfalfa field, where it was contained with an emergency berm
18  (Phillips, 2014; Wertz, 2014).There is no information on how much leached into soils or if the spill reached
19  drinking water resources.

---

### 5.5.1.4. Gel Storage

20  Gels can be added to hydraulic fracturing fluid using either batch or continuous (also called "on-the-
21  fly") mixing systems. Gelling agents and gel slurries are stored differently on-site and may pose
22  different potential spill scenarios depending on whether the site is using batch or continuous
23  mixing processes (BJ Services Company, 2009).

### 5.5.1.5. Batch Mixing

24  In a typical batch mixing process, powdered gelling agents and related additives (e.g., buffers,
25  surfactants, biocides) are mixed on-site with base fluid water in large tanks (typically 20,000 gal or
26  76,000 L) (BJ Services Company, 2009; Halliburton, 1988).

27  The number of gel slurry tanks used varies based on site-specific conditions and the size of the
28  fracture job. These tanks may be subject to leaks or overflow during the batch mixing process and
29  during storage prior to injection. One of the disadvantages of batch mixing is the need for multiple
30  suction hoses to draw pre-gelled fluids from storage tanks into the blender, which may increase the
31  potential for spills. Yeager and Bailey (2013) state that a drawback of batch mixing is the "fluid
32  spillage and location mess encountered when pre-mixing tanks," suggesting that small spills are not

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1  uncommon during batch mixing. Details of a documented gel slurry spill are presented in Text Box
2  5-8.

---

**Text Box 5-8. Spill of Gel during Mixing.**

3  On April 9, 2010, a company was mixing a gel slurry for an upcoming fracture job. The tank had developed a
4  crack, which allowed approximately 10,000 gal (38,000 L) of water mixed with 60 gal (230 L) of gel to leak
5  out. The mixture did not reach a water receptor, and absorbents were used to clean up the gel (U.S. EPA,
6  2015n, Appendix B Line 220).

---

### 5.5.1.6. Continuous Mixing (On-the-Fly)

7  In continuous mixing operations, powdered gels are typically replaced with liquid gel concentrates
8  (Allen, 2013; BJ Services Company, 2009). Operators prepare dilute gelling agents as needed using
9  specialized hydration units (BJ Services Company, 2009). Liquid gel concentrates may be stored on-
10  site in single-purpose tanker trucks (Harms and Yeager, 1987), but are more often stored in
11  specialized mixing and hydration units (Ayala et al., 2006). Continuous mixing requires less
12  preparation than batch mixing but typically requires more equipment (BJ Services Company, 2009;
13  Browne and BD, 1999), which may increase the possibility for spills resulting from equipment
14  malfunctions or human error.

### 5.5.2. Hoses and Lines

15  High- and low-pressure hoses and lines are used to transfer hydraulic fracturing fluids from storage
16  units to specialized mixing and pumping equipment and ultimately to the wellhead. A discussion of
17  the different types of hoses and lines and possible points of failure is provided below. The following
18  photograph shows an example of hoses and lines at a hydraulic fracturing site.

---

**Figure 5-8. A worker adjusts hoses at a hydraulic fracturing site near Mead, Colorado.**

Source: AP Photo/Brennan Linsley.

1   Suction pumps and hoses move large volumes of base fluid to the blender. Incomplete or damaged
2   seals in inlet or outlet connections can cause fluid leaks at the connection points. Improperly fitted
3   seals also severely limit or eliminate suction lift, which may impair the suction pump and increase
4   spill potential. Suction hoses themselves are susceptible to leaks due to wear and tear. Equipment
5   providers recommend hoses be closely inspected to ensure they are in good operating condition
6   prior to use (Upstream Pumping, 2015; BJ Services Company, 2009; Malone and Ely, 2007).

7   Discharge hoses transfer additives from containment vessels or totes to the blender. Given the
8   potential for concentrated chemicals to spill during transfer from storage totes to the blender, it is
9   particularly important that these hoses are in good condition and that connector seals or washers
10   fit properly and are undamaged. Discharge hoses are also used to carry fracturing fluid pumped
11   from the blender via the low-pressure manifold to the high-pressure pumps. Proppant-heavy fluids
12   are pumped through discharge hoses at relatively low rates. If a sufficient flow rate is not
13   maintained, proppant may settle out, damaging pumps and creating potential for spills or leaks
14   (Upstream Pumping, 2015; BJ Services Company, 2009; Malone and Ely, 2007).

15   High-pressure flow lines convey pressurized fluids from the high-pressure pumps into the high-
16   pressure manifold, and from the manifold into the wellbore. High-pressure flow lines are subject to
17   erosion caused by the high-velocity movement of abrasive, proppant-laden fluid. Curved sections of
18   flow lines (e.g., swivel joints) where abrasive fluids are forced to turn corners are particularly
19   subject to erosion and are more likely to develop stress cracks or other defects that may result in a
20   leak or spill. Safety restraints are typically used to prevent movement of flow lines in the event of

1  failure and to help control spills. High-pressure flow lines are pressure-tested to detect fatigue or
2  stress cracks prior to the fracturing treatment (OSHA, 2015; BJ Services Company, 2009; Arthur et
3  al., 2008; Malone and Ely, 2007; Halliburton, 1988).

4  Nineteen spills of chemicals or fracturing fluids associated with leaks from hoses or lines had a total
5  spill volume of 12,756 gal (48,300 L), with a median volume of 420 gal (1,600 L) (U.S. EPA, 2015n).

### 5.5.3. Blender

6  The blender is the central piece of equipment used to create the fracturing fluid for injection. It
7  moves, meters, and mixes precise amounts of the base fluid, additives, and proppant and pumps the
8  mixed slurry to high-pressure pumping equipment (BJ Services Company, 2009; Malone and Ely,
9  2007; Halliburton, 1988). A typical blender consists of a centrifugal suction pump for pulling base
10  fluid, one or more chemical metering pumps to apportion the additives, one or more proportioners
11  to measure and feed proppant, and a central agitator tank where fluid components are mixed
12  together.

13  The blending process is monitored to ensure that a uniform mixture is maintained regardless of
14  injection rates and volumes. Excessive or reduced rates of flow during treatment can cause the
15  blender to malfunction or to shut down, which may result in spills. (Malone and Ely, 2007;
16  Halliburton, 1988). For aqueous hydraulic fracturing fluid blends, spills that occur downstream of
17  the blender will be a dilute mixture comprised primarily of water with a low concentration (less
18  than or equal to 2%) of chemicals. Details of a spill from a blender are presented in Text Box 5-9.

---

**Text Box 5-9. Spill of Hydraulic Fracturing Fluid from Blender.**

19  In May 2006, a blender malfunctioned during a fracture job in Oklahoma. Approximately 294 gal (1,100 L) of
20  fluid spilled into a nearby wheat field. The fluid consisted of hydrochloric acid, clay stabilizer, diesel, and
21  friction reducer. Contaminated soil was removed by the operator (U.S. EPA, 2015n, Appendix B Line 249).

---

### 5.5.4. Manifold

22  A trailer-mounted manifold and pump system functions as a central transfer station for all fluids
23  used to fracture the well. The manifold is a collection of low- and high-pressure pipes equipped
24  with multiple fittings for connector hoses. Fluids are pumped from the blender through the low-
25  pressure manifold hoses, which distribute it to high-pressure pump trucks. Pressurized slurry is
26  sent from the pump trucks through high-pressure manifold lines and into additional high pressure
27  lines that lead to the wellhead (Malone and Ely, 2007).

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**Figure 5-9. Manifold (pointed to by the white arrow).**

Source: Halliburton.

1  Manifold and pump system components require varying amounts of manual assembly and undergo
2  varying amounts of pre-testing (Malone and Ely, 2007). Improperly tested parts may be more likely
3  to break or lose functionality, leading to a spill. In manifolds requiring more manual assembly, there
4  may be more opportunities for human error. The EPA (U.S. EPA, 2015n) identified seven spills
5  sourced from manifolds. Three of these spills, out of the 105 chemical or hydraulic fracturing fluid
6  spills reported to state data sources, were fracturing fluid that resulted from either human error of
7  equipment failure. These three spills were responsible for approximately 5,000 gal (19,000 L) of
8  spilled fluids (U.S. EPA, 2015n, Appendix B Lines 35, 141, 160).

### 5.5.5. High-Pressure Fracturing Pumps

9  High-pressure fracturing pumps take the fracturing fluid mixture from the blender, pressurize it,
10 and propel it down the well. Typically, multiple high-pressure, high-volume fracturing pumps are
11 needed for hydraulic fracturing (Upstream Pumping, 2015). Such pumps come in a variety of sizes.
12 Bigger pumps move greater volumes of fluid at higher pressures; therefore, spills from these pumps
13 may be larger. Smaller pumps may require more operators and more maintenance (BJ Services
14 Company, 2009), and therefore may result in more frequent spills.

**Figure 5-10. High-pressure pumps on either side of the manifold.**

> Source: http://drillingahead.com/roger-payne/gallery/14826/20000-psi-frac-near-caldwelltexas-2005#gallery_img.

1  The "fluid ends" of hydraulic fracturing pumps are the pump components through which fluids are
2  moved and pressurized. Pump fluid ends must withstand enormous pressure and move a large
3  volume of abrasive fluid high in solids content. They have multiple parts (e.g., seals, valves, seats
4  and springs, plungers, stay rods, studs) that can wear out under the stress of high-pressure
5  pumping (Upstream Pumping, 2015). Given the sustained pressures involved, careful maintenance
6  of fluid ends is necessary to prevent equipment failure (Upstream Pumping, 2015; API, 2011).
7  Details of a documented spill from a fracture pump are presented in Text Box 5-10.

---

**Text Box 5-10. Spill of Fluid from Fracture Pump.**

8  In December 2011, a fluid end on a fracture pump developed a leak, spilling approximately 840 gal (3,200 L)
9  of fracturing fluid. A vacuum truck was used to recover the spilled fluid, and all affected soils were scheduled
10 to be neutralized and taken to a landfill at the end of the job, after removal of the equipment (U.S. EPA, 2015n,
11 Appendix B Line 14).

---

### 5.5.6. Surface Wellhead for Fracture Stimulation

1   A wellhead assembly, often referred to as a frac head or frac stack, is temporarily installed on the
2   wellhead during the fracture treatment. The frac head assembly allows high volumes of high-
3   pressure proppant-laden fluid to be injected into the formation (OSHA, 2015; Halliburton, 2014;
4   Stinger Wellhead Protection, 2010). The temporary frac head is equipped with specialized isolation
5   tools so that the wellhead is protected from the effects of pressure and abrasion.



**Figure 5-11. Multiple fracture heads.**

Source: DOE/NETL.

6   As with all components of hydraulic fracturing operations, repeated and prolonged stress from
7   highly pressurized, abrasive fluids may lead to equipment damage. The presence of minute holes or
8   cracks in the frac head may result in leaks when pressurized fluids are pumped. In addition, surface
9   blowouts or uncontrolled fluid releases may occur at the frac head because of valve failure or
10  failure of other components of the assembly. Details of a documented frac head failure are
11  presented in Text Box 5-11. Details on the Killdeer, ND, blowout and associated spill are presented
12  in Text Box 5-12.

---

**Text Box 5-11. Spill from Frac Head Failure.**

1  On March 2, 2011, a frac head failed during fracturing operations in Colorado. Approximately 8,400 gal
2  (32,000 L) of slickwater fracturing fluid leaked. The majority of the spill was contained on-site, though a small
3  amount ran off into a nearby cornrow. Some of the fluid was recovered, and saturated soils were scraped and
4  stockpiled on the well pad (U.S. EPA, 2015n, Appendix B Line 75).

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**Text Box 5-12. The Killdeer Case Study.**

5  In September 2010, a blowout occurred in the Franchuk 44-20 SWH well, in Dunn County near Killdeer, ND.
6  Hydraulic fracturing fluids, oil, and flowback water spilled onto the land and possibly entered the Killdeer
7  aquifer, which is a source of drinking water. The EPA investigated a reported blowout event at the Killdeer
8  site as part of a retrospective case study. The study area is comprised of historical oil and gas production and
9  current hydraulic fracturing. The discussion below was taken from the EPA Killdeer case study (U.S. EPA,
10  2015j).

11  **Methods**

12  Water quality samples were collected from three domestic wells, nine monitoring wells (installed by
13  Terracon), two supply wells, one municipal well, and one state well during three rounds in July 2011, October
14  2011, and October 2012. The geochemistry of water samples was investigated by analyzing major ions, trace
15  metals, methane/ethane gas concentrations, volatile organic compounds, semivolatile organic compounds,
16  glycol ethers, diesel- and gasoline-range organics, low-molecular-weight acids, and selected stable isotopes.
17  Data collected from this study were statistically compared with historical water quality data retrieved from
18  the literature and national water quality databases. To help determine whether hydraulic fracturing
19  processes were a cause of alleged impacts on water quality, detailed environmental record searches were
20  conducted to help identify other potential contaminant sources.

21  **Results**

22  Three study wells, NDGW09, NDGW08, and NDGW07, were excluded from the comparisons with historical
23  data. NDGW09 was excluded since it was screened in the Sentinel Butte aquifer so a comparison with
24  historical Killdeer aquifer water quality data was not appropriate. NDGW08 and NDGW07 had significant
25  differences in water quality compared to the remaining study wells. These two wells showed differences in
26  ion concentrations (e.g., chloride, calcium, magnesium, sodium, strontium) as well as tert-butyl alcohol (TBA).
27  The remaining study wells were then compared with historical data to determine if these wells represented
28  background water quality of the Killdeer aquifer. This comparison between the remaining study wells and
29  Killdeer aquifer historical water quality data indicated that these remaining study wells were in general
30  consistent with the historical background data and then used for the data analysis as background wells. There
31  were limited detections of other organic compounds in the study wells. In most cases, with the exception

32  of TBA, the detected organic compounds could not be directly linked to the blowout or hydraulic fracturing,
33  as these chemicals could have originated from other sources including vehicular traffic, generators used to
34  power well pumps, flaring of methane from the pad production wells, and cement used to repair a well the
35  day prior to sampling.

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1 | Comparisons of TBA between the study data and historical data could not be made since no historical data for
2 | TBA were found for the Killdeer aquifer. TBA data were compared with the background study wells and were
3 | found to be different. Based on the analysis of other potential sources of contamination, the EPA determined
4 | that the only other potential sources of TBA were gasoline spills, leaky underground storage tanks, and
5 | hydraulic fracturing fluids. The data from this study suggest the TBA resulted from the degradation of tert-
6 | butyl hydroperoxide used during the hydraulic fracturing of the Franchuk well since MTBE and other
7 | signature compounds associated with gasoline or fuels were not present in NDGW08 and NDGW07.

8 | **Conclusion**

9 | The analysis of data from NDGW008 and NDGW07 indicated that the main impact on water quality was from
10 | briny water and TBA mixing with Killdeer aquifer water in these wells. In all cases, the fingerprinting
11 | techniques used indicated that the impacts on NDGW07 and NDGW08 were consistent with deep formation
12 | brines underlying the Killdeer study location. Based on the data analysis performed for the Killdeer case
13 | study, the observed impacts on NDGW07 and NDGW08 were likely caused by the blowout that occurred at
14 | the Franchuk 44-20. This evidence, along with the absence of another plausible candidate cause, strongly
15 | suggests impact on a drinking water resource from the blowout during the hydraulic fracturing of the
16 | Franchuk 44-20 SWH well.

## 5.6. Spill Prevention, Containment, and Mitigation

17 | Several factors influence spill prevention, containment, and mitigation, including Federal, State, and
18 | local regulations and company practices. State regulations governing spill prevention, containment,
19 | and mitigation at hydraulic fracturing facilities vary in scope and stringency (Powell, 2013; GWPC,
20 | 2009). Employee training and equipment maintenance are also factors in effective spill prevention,
21 | containment, and mitigation. Analysis of these factors was outside the scope of this assessment.

22 | Hydraulic fracturing operating companies themselves may develop and implement spill prevention
23 | and containment procedures. The American Petroleum Institute has a guidance document *Practices*
24 | *for Mitigating Surface Impacts Associated with Hydraulic Fracturing* (API, 2011). The document
25 | describes practices currently used in the oil and natural gas industry to minimize potential surface
26 | environmental impacts. As another example, the province of New Brunswick, Canada, released
27 | rules for industry on responsible environmental management of oil and natural gas activities (GNB,
28 | 2013). It was beyond the scope of this assessment to evaluate the efficacy of the practices in these
29 | documents or the extent to which they are implemented.

30 | Spill containment systems include primary, secondary, and emergency containment systems.
31 | Primary containment systems are the storage units, such as tanks or totes, in which fluids are
32 | intentionally kept. Secondary containment systems, such as liners and berms installed during site
33 | set-up, are intended to contain spilled fluids until they can be cleaned up. Emergency containment
34 | systems, such as berms, dikes, and booms, can be implemented temporarily in response to a spill.

35 | The EPA investigated spill containment and mitigation measures in an analysis of spills related to
36 | hydraulic fracturing activities (U.S. EPA, 2015n). Of the approximately 25% of reports that included
37 | information on containment, the most common types of containment systems referenced in the
38 | hydraulic fracturing-related spill records included berms, booms, dikes, liners, and pits, though

1  many of the spill reports did not indicate specific containment measures. Some spills were reported
2  to breach the secondary containment systems. Breaches of berms and dikes were most commonly
3  reported.

4  In cases where secondary containment systems were not present or were inadequate, operators
5  sometimes built emergency containment systems. The most common were berms, dikes, and
6  booms, but there were also instances where ditches, pits, or absorbent materials were used to
7  contain the spilled fluid. Absorbent materials were generally used when small volumes (10–200 gal
8  or 40–800 L) of additives or chemicals were spilled (U.S. EPA, 2015n). There was not enough
9  information to detail the regularity of emergency containment systems or their effectiveness.

10  Remediation is the action taken to clean up a spill and its affected environmental media. The most
11  commonly reported remediation activity, mentioned in approximately half of the hydraulic
12  fracturing-related spill records evaluated by the EPA, was removal of spilled fluid and/or affected
13  media, typically soil (U.S. EPA, 2015n). Other remediation methods reported by the EPA (U.S. EPA,
14  2015n) included the use of absorbent material, vacuum trucks, flushing the affected area with
15  water, and neutralizing the spilled material.

## 5.7. Overview of Chemical Spills Data

16  Spills of hydraulic fracturing fluids have occurred across the country and have affected the quality
17  of drinking water resources (U.S. EPA, 2015n; Brantley et al., 2014; COGCC, 2014; Gradient, 2013).
18  Spills may infiltrate drinking water resources by reaching surface water, or by leaching into the
19  ground water. Potential impacts depend upon a variety of factors including the chemical spilled,
20  environmental conditions, and actions taken in response to the spill. However, due to a lack of
21  available data, little is known about the prevalence and severity of actual drinking water impacts.

### 5.7.1. EPA Analysis of Spills Associated with Hydraulic Fracturing

22  The EPA (2015n) (see Text Box 5-13 for additional information) evaluated 457 spills related to
23  hydraulic fracturing activities on or near the well pad. Of these spills, 151 spills were of chemicals,
24  additives, or fracturing fluids. Information in the spill reports included: spill causes (e.g., human
25  error, equipment failure), sources (e.g., storage tank, hose or line), volumes, and environmental
26  receptors.

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**Text Box 5-13. EPA Review of State and Industry Spill Data: Characterization of Hydraulic Fracturing-Related Spills.**

27  The EPA (2015n) used data gathered from select state and industry sources to characterize hydraulic
28  fracturing-related spills with respect to volumes spilled, materials spilled, sources, causes, environmental
29  receptors, containment, and responses. For the purposes of the study, hydraulic fracturing-related spills were
30  defined as those occurring on or near the well pad before or during the injection of hydraulic fracturing fluids
31  or during the post-injection recovery of fluids. Because the main focus of this study is to identify hydraulic
32  fracturing-related spills on the well pad that may reach surface or ground water resources, the following
33  topics were not included in the scope of this project: transportation-related spills, drilling mud spills, and
34  spills associated with disposal through underground injection control wells.

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1   Data on spills that occurred between January 2006 and April 2012 were obtained from nine states with
2   online spill databases or other data sources, nine hydraulic fracturing service companies, and nine oil and gas
3   production well operators. The data sources used in this study contained over 36,000 spills. The EPA
4   searched each spill report for keywords related to hydraulic fracturing (e.g., frac, glycol, flowback). Spill
5   records from approximately 12,000 spills (33 percent of the total number of spills reviewed) contained
6   insufficient information to determine whether the event was related to hydraulic fracturing. Of the spills with
7   sufficient information, the EPA identified approximately 24,000 spills (66%) as not related to hydraulic
8   fracturing on or near the well pad. The remaining 457 spills (approximately 1%) occurred on or near the well
9   pad and were definitively related to hydraulic fracturing. These 457 spills occurred in 11 different states over
10  six years (January 2006 and April 2012).

11  The EPA categorized spills according to the following causes: equipment failure, human error,
12  failure of container integrity, other (e.g., well communication, weather, vandalism), and unknown.
13  Figure 5-12 presents the percent distribution of chemical or fracturing fluid spills associated with
14  each cause. Over half of the spills were collectively caused by equipment failure (34%) and human
15  error (25%). Approximately one-quarter of the spill causes were unknown or not reported. A
16  report analyzing spills in Colorado is generally consistent with the EPA's findings (COGCC, 2014).[1]
17  Colorado found that equipment failure was the dominant spill cause, accounting for over 60% of
18  spills between 2010 and 2013, followed by human error accounting for 20%–25% of spills.

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[1] The COGCC report included all materials related to oil and gas production and were therefore not specific to chemical and fracturing fluid spills.
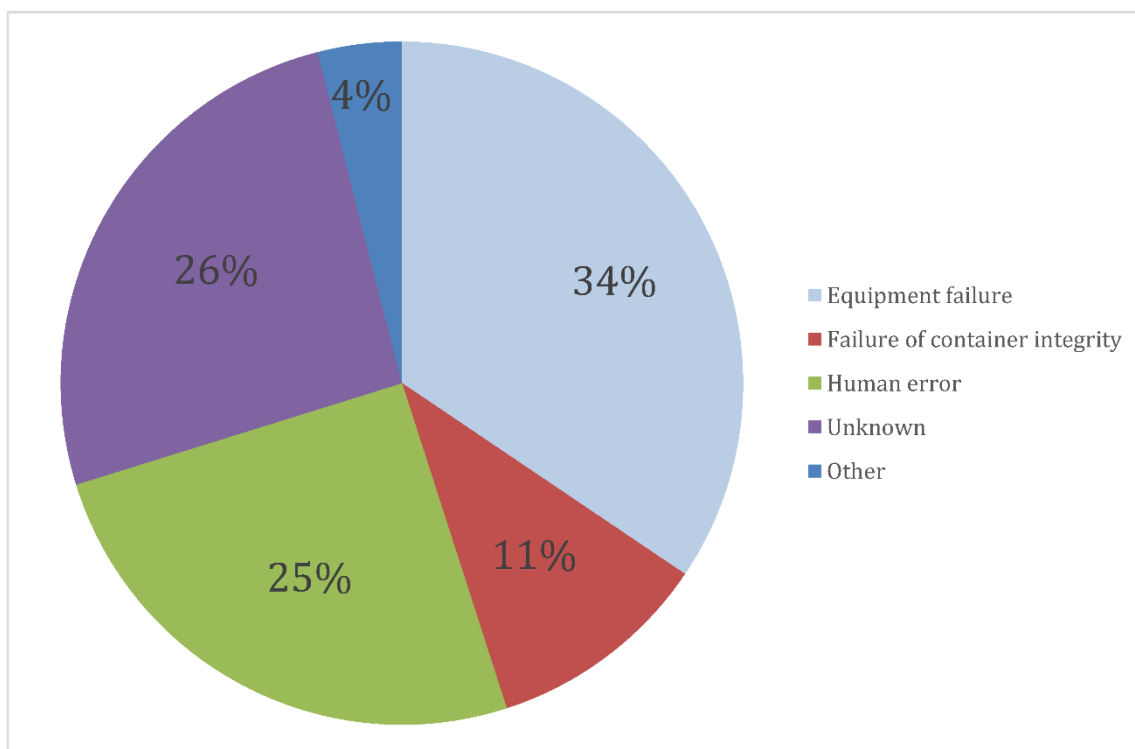
**Figure 5-12. Distribution of the causes of 151 hydraulic fracturing-related spills of chemicals and fracturing fluid.**

Data from U.S. EPA (2015n).

1   Spills in the EPA Spills Report were also categorized by the following sources: storage, equipment,
2   well or wellhead, hose or line, and unknown. Figure 5-13 presents the percent distribution for the
3   chemical or fracturing fluid spills associated with each source. Storage units (e.g., chemical totes,
4   fracturing fluid tanks) were the predominant source of spills, accounting for 36% (54 spills) of spill
5   sources. Spills from storage units were predominantly caused by human error (39%), followed by
6   failure of container integrity (30%). Spills from equipment were the next most common known
7   source (18%), followed by spills from hoses or lines (13%). Twenty-eight percent of spills had an
8   unknown source.
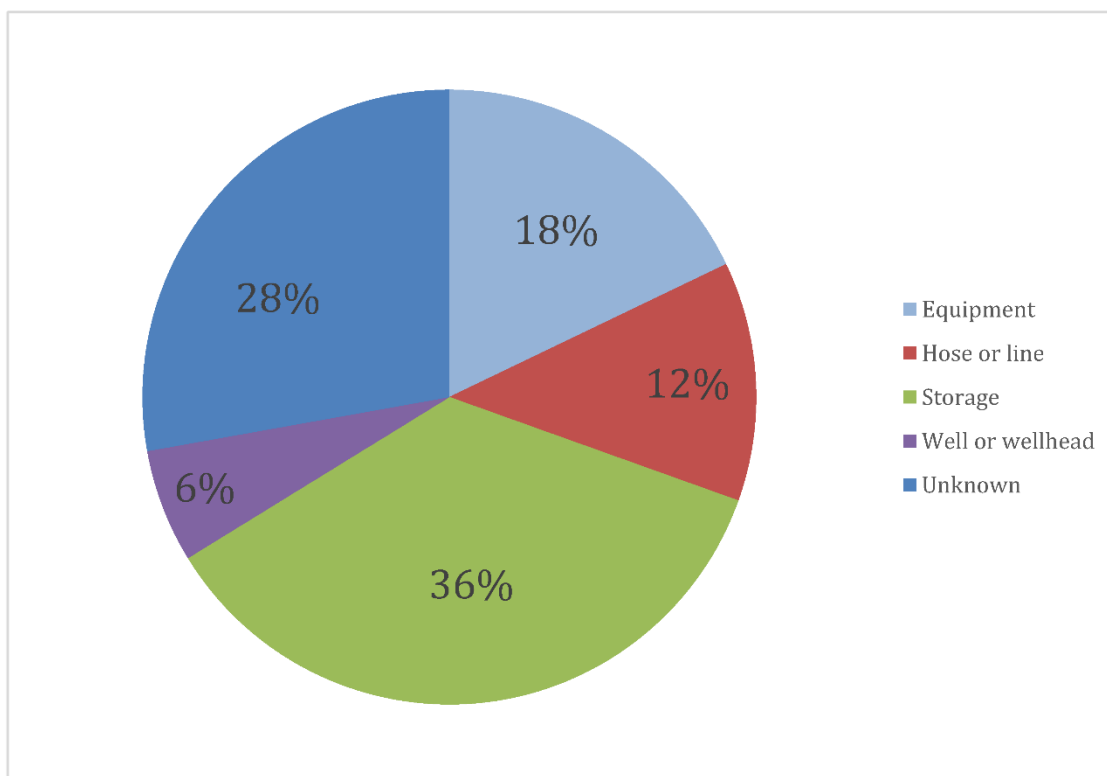
**Figure 5-13. Percent distribution of sources of 151 hydraulic fracturing-related spills of chemicals or fracturing fluid.**

Data from U.S. EPA (2015n).

1    The reported total volume of 125 of 151 chemical or hydraulic fracturing fluid spills was
2    approximately 184,000 gal (697,000 L). The volume was unknown for 26 of these spills. The spills
3    ranged in volume from 5 to more than 19,000 gal (19 to 72,000 L), with a median volume of 420 gal
4    (1,600 L). The largest source of spills was storage containers, which released approximately 83,000
5    gal (314,000 L) of spilled fluid. Spills from wells or wellheads are often associated with high spill
6    volumes. Nine instances of spills at the well or wellhead were associated with approximately
7    46,000 gal (174,000 L) of spilled fluid (see Figure 5-14). The high pressure associated with frac
8    head blowouts has led to large, high-volume spills (see Text Box 5-11).
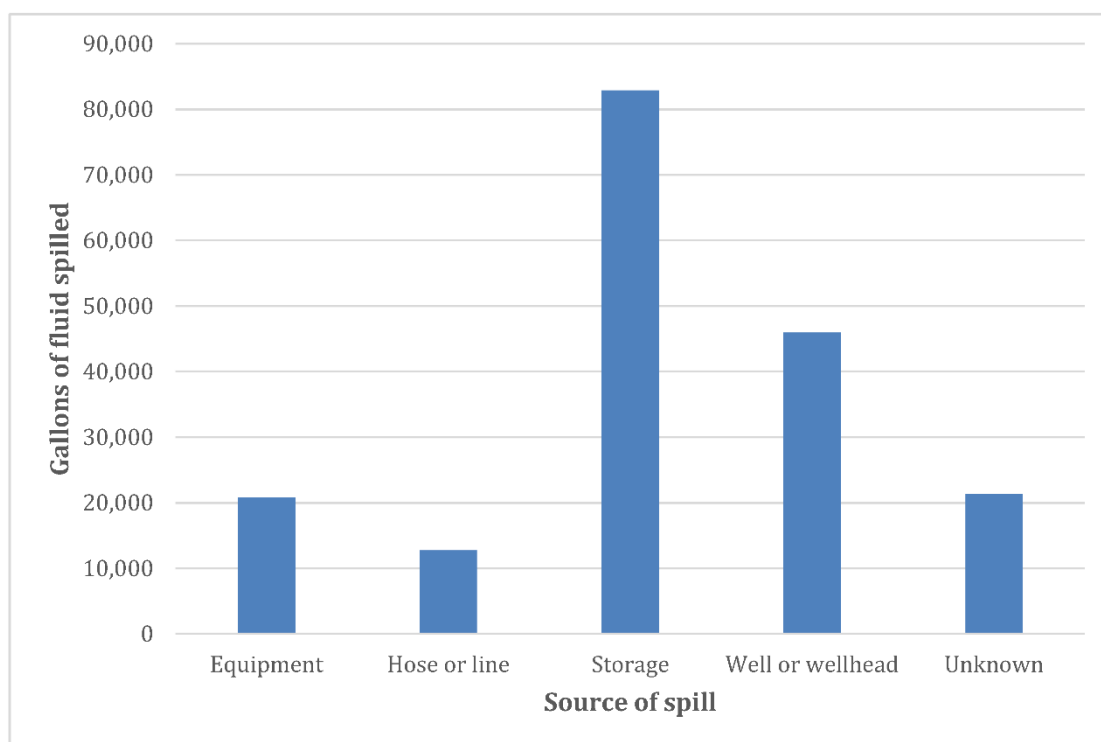
**Figure 5-14. Total volume of fluids spilled for 151 hydraulic fracturing-related spills of chemicals and fracturing fluid, by spill source.**

Data from U.S. EPA (2015n).

1  Figure 5-15 presents the number of chemical or fracturing fluid spills that reached environmental
2  receptors, by receptor type. Environmental receptors (i.e., surface water, ground water, soil) were
3  identified for 101 of the 151 spills, or 67% of the spills in the EPA's analysis (U.S. EPA, 2015n). Soil
4  was by far the dominant environmental receptor, with 97 spills reaching soil. Thirteen spill reports
5  indicated that the spilled fluid had reached surface water. Nine spill reports identified both soil and
6  surface water as a receptor. No spill report identified ground water as a receptor. The data contain
7  few post-spill analyses, so ground water contamination may have occurred but have not been
8  identified. Additionally, several years may be required for a spilled fluid to leach into the ground
9  water and therefore impact on a ground water receptor may not be immediately apparent. Storage
10  units were the predominant sources of spills reaching an environmental receptor.
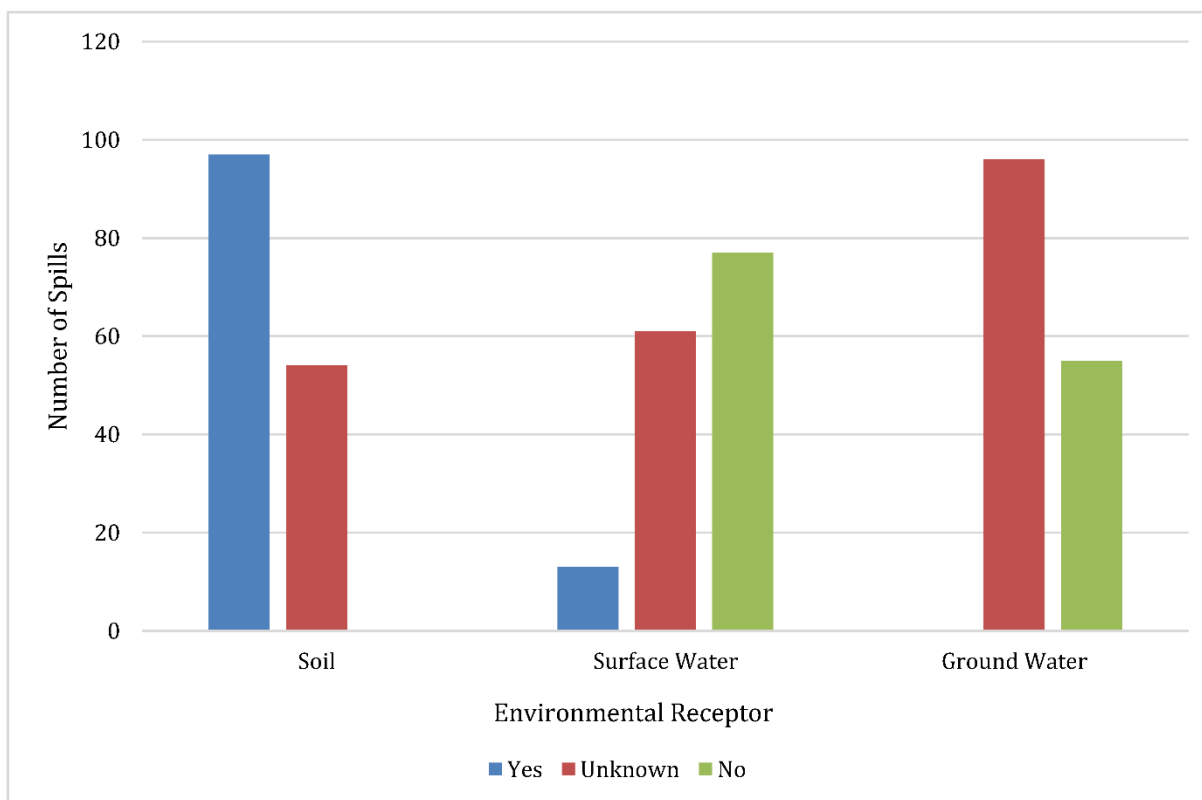
**Figure 5-15. Number of hydraulic fracturing-related spills of chemicals or fracturing fluid that reported whether an environmental receptor was reached.**

"Unknown" refers to hydraulic fracturing related spill events for which environmental receptors were specified as unknown or not identified (positively or negatively).
Data from U.S. EPA (2015n).

1  Six spills from storage containers reached a surface water receptor. Thirty-eight of the spills from
2  storage units reached a soil receptor. If a spill was confined to a lined well pad, for example, it might
3  not have reached the soil, but most incident reports did not include whether the well pad was lined
4  or unlined. Regarding spills of hydraulic fluids and chemicals from storage containers, 16 spills
5  were due to failure of container integrity, which includes holes and cracks in containers, and
6  overflowing containers as a result of human error or equipment malfunctions.

7  The EPA analysis demonstrates that spills of chemicals, additives, and fracturing fluids do occur at
8  well sites and reach both soil and surface water receptors.

### 5.7.2.  Other Spill Reports

9   Surface spills related to hydraulic fracturing activities are not well documented in the scientific
10  literature, though some evidence of spills and impacts to environmental media exists (e.g., U.S. EPA,

1   2015j; Brantley et al., 2014; Gross et al., 2013; Papoulias and Velasco, 2013). For example,

2   Papoulias and Velasco (2013) demonstrated that fracturing fluid spilled into surface water likely

3   contributed to the distress and deaths of the threatened blackside dace fish in Kentucky. A variety

4   of chemicals entered the creek and significantly reduced the stream's pH and increased stream

5   conductivity. Using data from post-spill sampling reports in Colorado, Gross et al. (2013) identified

6   concentrations of benzene, toluene, ethylbenzene, and xylene in ground water samples, which the

7   authors attributed to numerous hydraulic fracturing-related spills. The COGCC (2014) published a

8   report analyzing all spills reported to the state of Colorado between 2010 and 2013, and found that

9   approximately 8% of them were related to hydraulic fracturing. Based on the EPA's analyses (U.S.

10  EPA, 2015n) and available scientific data, we estimate spill rates of chemicals and hydraulic

11  fracturing fluid range from 0.4 and 12.2 spills for every 100 wells. (See Text Box 5-14 for additional

12  information.)

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**Text Box 5-14. Spill Rates.**

13  Several studies have estimated the frequency of hydraulic fracturing-related spills. Three studies (Rahm et al.,

14  2015; Brantley et al., 2014; Gradient, 2013) calculated a spill rate for the Marcellus Shale in Pennsylvania

15  using reports from the Pennsylvania Department of Environmental Protection (PA DEP) Oil and Gas

16  Compliance Report Database, and here we estimate an on-site spill rate for Colorado. The PA DEP database

17  provides a searchable format based on Notices of Violations from routine inspections or investigations of spill

18  reports or complaints. Each study had different criteria for inclusion, presented in Table 5-6, resulting in a

19  range of rates even when using the same data source. Spills of hydraulic fracturing fluids, flowback/produced

20  water are estimated to occur at a rate of 0.4 per 100 wells fractured. Spills related to hydraulic fracturing

21  activity are estimated to occur at a rate between 3.3 to 12.2 spills per 100 wells installed (PA DEP data) (see

22  Table 5-6).

23  In its study of spills related to hydraulic fracturing, the EPA determined that spill reports from the Colorado

24  Oil and Gas Conservation Commission (COGCC) Information System were the most detailed spill reports from

25  among the nine state data sources investigated and generally provided more of the information needed to

26  determine whether a spill was related to hydraulic fracturing (U.S. EPA, 2015n). Here, we estimate the spill

27  rate in Colorado by dividing the number of hydraulic fracturing-related spills identified by the EPA (U.S. EPA,

28  2015n, Appendix B)(Appendix B in U.S. EPA, 2015b) by the number of wells hydraulically fractured in

29  Colorado for specific time periods between January 2006 and April 2012. We used three data sources to

30  estimate the number of wells: (1) there were 172 reported spills in Colorado for the 15,000 wells fractured

31  from January 2006 to April 2012 (DrillingInfo, 2012), (2) there were 50 reported spills in Colorado for the

32  3,559 wells fractured from January 2011 to April 2012 (U.S. EPA, 2015b), and (3) there were 41 reported

33  spills in Colorado for the 3,000 wells fractured from September 2009 to October 2010 (U.S. EPA, 2013a).

34  From these data we estimate an average of 1.3 reported spills on or near the well pad for every 100

35  hydraulically fractured wells.

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36

**Table 5-6. Estimations of spill rates.**

Spill rates from four different sources. Each source used different criteria to identify and include spills in their analysis.

| Spill rate[a] | Data source | Time period | Inclusion criteria | Information source |
|---|---|---|---|---|
| 0.4 | PA DEP[b] Media | 2008–2013 | Volume spilled > 400 gal; Spill reported to reach water body. | Brantley et al. (2014)[d] |
| 3.3 | PA DEP[b] | 2009-2012 | "Unconventional" well; Spills with unknown volumes not included, Includes any spill during HF activities | Gradient (2013)[e] |
| 12.2 | PA DEP[b] | 2007-July 2013 | "Unconventional" well based on environmental violation rates. | Rahm et al. (2015)[e] |
| 1.3 | COGCC[c] | Jan 2006 – May 2012 | Specifically related to hydraulic fracturing on or near well pad | U.S. EPA (2013a)[d] |

[a]Spill rate is the number of reported spills per 100 wells.

[b]PA DEP (http://www.depreportingservices.state.pa.us/ReportServer/Pages/ReportViewer.aspx?/Oil_Gas/OG_Compliance)

[c]COGCC (https://cogcc.state.co.us/cogis/IncidentSearch.asp)

[d]Spill rate is calculated as the number of spills per 100 wells fractured.

[e]Spill rate is calculated as the number of spills per 100 wells installed.

**Text Box 5-14 (Spill Rates), continued:**

1　The spill rates presented in Table 5-6 are based on spill reports found in two state data sources and are
2　limited by both the spills reported in the state data sources and the inclusion criteria defined by each of the
3　studies. Spills identified from state data sources are likely a subset of the total number of spills that occurred
4　within a state for a specified time period. Some spills may not be recorded in state data sources because they
5　do not meet the spill reporting requirements in place at the time of the spill. Additionally, the PA DEP Notices
6　of Violation may include spills not specifically related to hydraulic fracturing, such as spills of drilling fluids.

7　The inclusion criteria used by each of the studies affects which spills are used to calculate a spill rate. More
8　restrictive criteria, such as only counting spills that were greater than 400 gallons, results in a lower number
9　of spills being used for estimating spill rates, while less restrictive criteria, such as all spills from wells
10　marked unconventional in the PA DEP database, results in a greater number of spills being used for
11　estimating spill rates. Rahm et al. applied the least restrictive criteria of the four studies (i.e., spills from
12　unconventional wells) when identifying spills, while Brantley et al. applied more restrictive criteria (i.e., spills
13　of >400 gallons in which spilled fluids reached a surface water body). This may account for the different spill
14　rates calculated by these two studies.

15　Based on previous studies and the analysis here, hydraulic fracturing-related spills rates in Pennsylvania and
16　Colorado range from 0.4 and 12.2 per 100 wells. These numbers may not be representative of national spill
17　rates or rates in other regions. If this range is applied nationally however, assuming between 25,000 and
18　30,000 wells are fractured each year, we would expect between approximately 100 and 3,700 spills annually
19　from hydraulic fracturing.

## 5.8. Fate and Transport of Chemicals

20　This section provides an overview of fate and transport of the range of chemicals used in hydraulic
21　fracturing fluid, including the physicochemical properties of these chemicals, and an overview of
22　the potential for a spilled chemical to affect drinking water resources. A general overview of the
23　processes governing the fate and transport of a chemical spill is shown in Figure 5-16. A chemical
24　spill has the potential to migrate to and have an impact on drinking water resources. Once spilled,
25　there are different paths that chemicals can travel and different processes they may undergo.
26　Chemicals may react and transform into other chemicals, volatilize, travel to surface water, leach
27　into and partition to soils, and/or reach ground water. The potential path and the severity of the
28　impact of a spill depend on different factors, including the site conditions; environmental
29　conditions; climate; weather; and chemical properties, concentration, and volume of the release.
30　The point in the chemical mixing process where the spill occurs affects potential impact. If the spill
31　occurs before chemicals are mixed into the base fluid, the chemicals will spill in their most
32　concentrated form. If the hydraulic fracturing fluid spills, then the chemicals will be more diluted,
33　and there may be effects on persistence and mobility due to interactions among chemicals present.
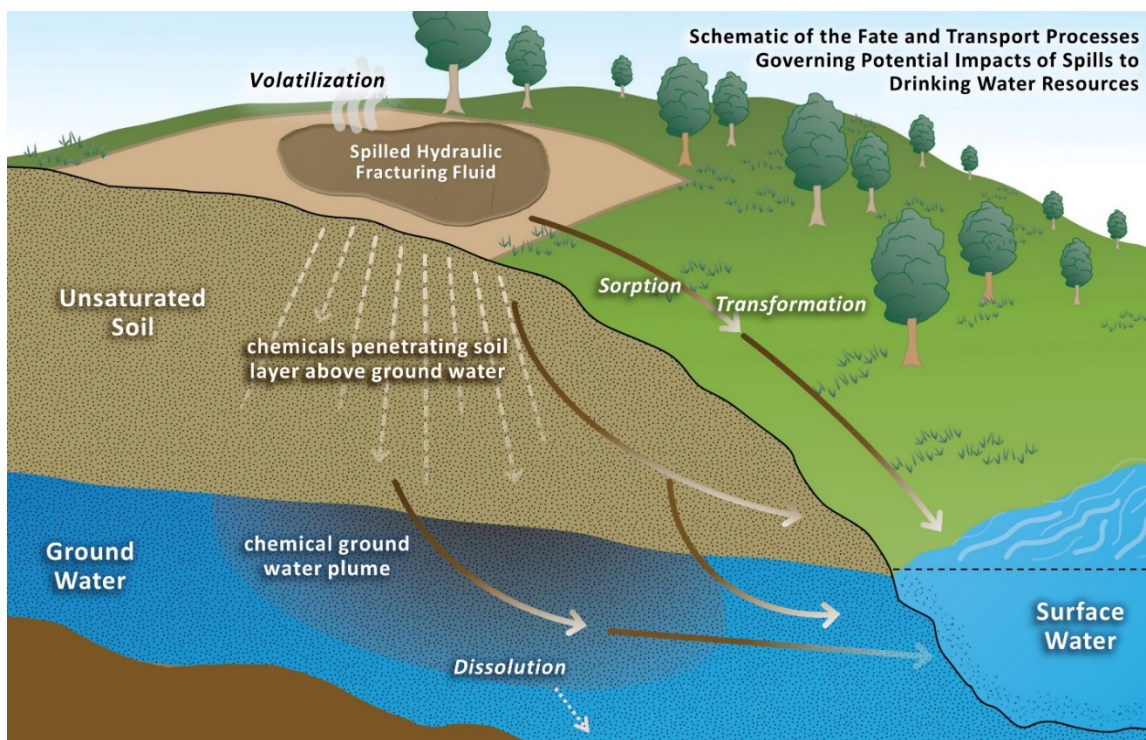
**Figure 5-16. Fate and transport schematic for a spilled hydraulic fracturing fluid.**

Schematic shows the potential paths and governing processes that spilled chemicals, which may lead to potential impacts to drinking water resources.

1  For inorganic chemicals, the properties and processes governing fate and transport depend on pH,
2  oxidation state, presence of iron oxides, soil organic matter, cation exchange capacity, and major ion
3  chemistry (U.S. EPA, 1996).[1] Transport of these chemicals into ground water depends on the nature
4  of ground water flow and flow through the unsaturated zone above the water table. Potential
5  transformations of inorganic chemicals differ from those of organic chemicals.[2] Some inorganic
6  anions (i.e., nitrate, chloride, and bromide) move with their carrier liquid and are affected mostly by
7  physical transport mechanisms. For many inorganic chemicals, transport is driven by the physical
8  flow processes (advection and dispersion), sorption, and precipitation. The relative role of each of
9  these depends on both chemical and environmental characteristics.[3,4]

10  Determining the fate and transport of organic chemicals and mixtures is a complex problem,
11  because of the many processes and different environmental media (air, soil, water) that can have an

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[1] Cation exchange capacity is the total amount of cations (positively charged ions) that a soil can hold. For example, when metal ions like $Ca^{2+}$ and $Na^+$ pass through the soil, they adhere and remain attached to the soil.
[2] The unsaturated zone is also referred to as the vadose zone. Meaning "dry," the vadose zone is the soil zone above the water table that is only partially filled by water, hence "unsaturated zone."
[3] Advection is a mechanism for moving chemicals in flowing water, where a chemical moves along with the flow of the water itself.
[4] Sorption is the general term used to describe the partitioning of a chemical between soil and water and depends on the nature of the solids and the properties of the chemical.

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1    impact. Unlike inorganic chemicals, organic chemicals degrade, which may affect their movement
2    and potential impact. Schwarzenbach et al. (2002) formalized a general framework for organic
3    chemical transport, where transport and transformation depend on both the nature of the chemical
4    and the properties of the environment. The fate and transport of organic compounds in soils has
5    been presented in the literature (e.g., Bouchard et al., 2011; Rivett et al., 2011; Abriola and Pinder,
6    1985a, b) and in textbooks (e.g., Domenico and Schwartz, 1997; Schnoor, 1996; Freeze and Cherry,
7    1979).

### 5.8.1.  Potential Paths

8    Chemicals, additives, and hydraulic fracturing fluids that are released into the environment travel
9    along different potential paths, as detailed in Figure 5-16. Liquids may flow overland to nearby
10   surface water or infiltrate the subsurface, where they may eventually reach the underlying ground
11   water or travel laterally to reach surface water. Movement can occur quickly or be delayed and
12   have a later or longer-term impact. Surface water and ground water gain or lose flow to each other,
13   and may transport chemicals in the process.

14   A dry chemical (e.g., gelling agents, biocides, friction reducers) released to the environment will
15   generally remain where it is spilled. Any spill that is not removed could act as a long-term source of
16   contamination. Wind could cause the chemical to disperse, or rain could dissolve a soluble chemical.
17   Dissolved chemicals may infiltrate into soil or flow overland. Insoluble chemicals and those sorbed
18   to soil particles could be mobilized by rain events via runoff and erosion.

### 5.8.1.1.  *Movement across the Land Surface*

19   In low permeability soils, there may be little infiltration and greater overland flow. Higher
20   permeability soils will allow fluid to penetrate into the soil layer. In either case, some of the
21   chemicals in the fluid may sorb to the soil particles and the vegetation, and then these chemicals
22   may be mobilized during rainfall, runoff, or erosion. As rainwater percolates through the soil, it may
23   dissolve stored chemicals, which can then migrate toward ground water.

24   The type of release is also important. If the spill is a slow leak, then the liquid may pond and the
25   affected area will expand slowly. If a more rapid release occurs, like a blowout or tank failure, then
26   momentum may result in greater overland movement and less soil infiltration during the event.

### 5.8.1.2.  *Movement through the Subsurface*

27   The unsaturated and saturated zones are the two zones of soils below the ground surface.
28   Movement through the unsaturated zone is driven by the depth of ponding of the spilled fluid,
29   gravity, and capillary properties of the subsurface.[1] In fractured rock or highly permeable soils,
30   fluids may move quickly through the subsurface. In low permeability soil, the movement of the fluid
31   is slower. As chemicals pass through the subsurface, some may sorb to soil or remain in the open
32   spaces between soil particles, effectively slowing their movement. Chemicals may again be

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[1] Capillarity occurs because of the forces of attraction of water molecules to themselves (cohesion) and to other solid substances such as soils (adhesion).

1    mobilized during future precipitation events, resulting in infiltration towards ground water or
2    movement through the unsaturated zone towards surface water.

3    Fluids that move through the subsurface into the saturated zone will move in the direction of the
4    flowing ground water. Generally, fluids travel further in systems with high ground water flow rates
5    and high recharge (e.g., sandy aquifers in humid climates) than in systems with low flow and low
6    recharge. Chemicals may sorb to suspended soil particles or complex with naturally occurring
7    chemicals (e.g., dissolved organic carbon) and be transported with the flowing water.[1] These
8    mechanisms can mobilize sparingly soluble chemicals that would otherwise be immobile.

### 5.8.2.  Physicochemical Properties

9    Three physicochemical properties are useful to describe the movement of chemicals in the
10   environment: (1) $K_{ow}$, the octanol-water partition coefficient, (2) the aqueous solubility, and (3) the
11   Henry's law constant.[2] These properties describe whether a chemical will sorb to soil and organic
12   matter or stay in water ($K_{ow}$), how much of a chemical may dissolve in water (aqueous solubility),
13   and whether a chemical will tend to remain in the water or volatilize (Henry's law constant).[3]

14   The $K_{ow}$ measures the relative hydrophobicity (chemical prefers to be in oil, log $K_{ow}$ >0) and
15   hydrophilicity (chemical prefers to be in water, log $K_{ow}$ <0) of a chemical. Aqueous solubility is the
16   maximum amount of a chemical that will dissolve in water in the presence of a pure chemical;
17   solubility generally serves as an upper bound on possible concentrations. The Henry's law constant
18   is the ratio of the concentration of a chemical in air (or vapor pressure) to the concentration of that
19   chemical in water.

20   Estimates and measured values for physicochemical properties were obtained by using the
21   Estimation Program Interface Suite 4.1 (see Appendix C).[4] Of the 1,076 chemicals the EPA listed as
22   used in hydraulic fracturing (see Appendix A), EPI Suite™ has estimated properties for 453 (42%).
23   EPI Suite™ does not have data available for the remaining 58% of the chemicals. The 453 chemicals
24   with physicochemical property data were chemicals with structures that are considered suitably
25   representative of the substance to compute properties within the constraints of EPI Suite™
26   software. Only unique defined organic desalted structures were submitted for property calculation.
27   Figure 5-17 presents histograms of all 453 of the chemicals, sorted by four physicochemical
28   parameters: measured log $K_{ow}$ ($n$ = 247, 23%), estimated log $K_{ow}$ (n=453, 42%) estimated log of the

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[1] Complexation is a reaction between two chemicals that form a new complex, either through covalent bonding or ionic
forces. This often results in one chemical solubilizing the other.
[2] The octanol-water partition coefficient ($K_{ow}$) represents the ratio of the solubility of a compound in octanol (a nonpolar
solvent) to its solubility in water (a polar solvent) in a mixture of the two. The higher the $K_{ow}$, the more nonpolar the
compound.
[3] We present the physicochemical parameter values using $log_{10}$ because of the wide range of values that these parameters
cover.
[4] EPI Suite™, version 4.1, http://www.epa.gov/opptintr/exposure/pubs/episuite.htm (U.S. EPA, 2012b). The EPI
(Estimation Programs Interface) Suite™ is a Windows®-based suite of physicochemical property and environmental fate
estimation programs developed by the EPA Office of Pollution Prevention and Toxics and Syracuse Research Corporation.
EPI Suite™ provides estimates of physicochemical properties for organic chemicals and has a database of measured values
for physicochemical properties when available. EPI Suite™ cannot estimate parameters for inorganic chemicals.

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1   aqueous solubility ($n$ = 453, 42%), and estimated log of Henry's law constant (at 25°C, $n$ = 453,
2   42%).
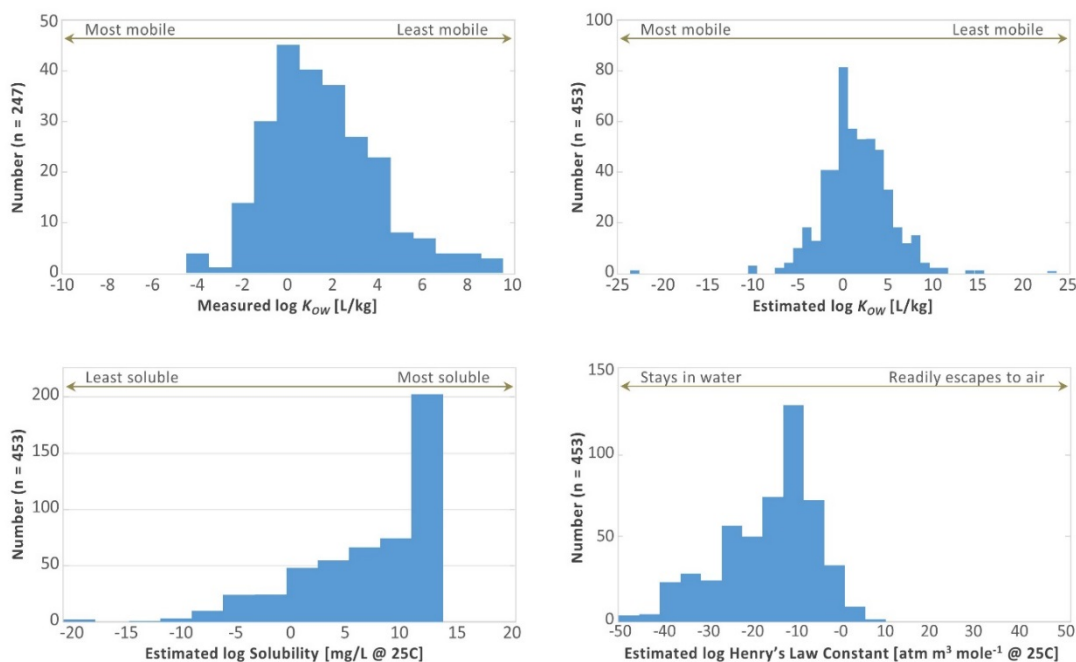


**Figure 5-17. Histograms of physicochemical properties of chemicals used in the hydraulic fracturing process.**

> Measured values of log $K_{ow}$ (upper left). Estimated physicochemical properties for log $K_{ow}$ (upper right), log Solubility (lower left), and log Henry's law constant (lower right) for all chemicals. Physicochemical properties (log $K_{ow}$, solubility, and Henry's Law constant) estimated by EPI Suite™.

3    The EPA also used EPI Suite™ to determine the physicochemical properties for 19 chemicals
4    provided to the EPA as confidential business information (CBI) (See Text Box 5-3 for discussion on
5    CBI).[1] The CBI chemical physicochemical properties are plotted as histograms in Figure 5-18. The
6    values of the physicochemical properties of known and CBI chemicals are similar, covering similar
7    ranges centered on similar values, suggesting that even though these chemicals are not publicly
8    known, their physicochemical properties are not appreciably different from the known chemicals.
9    This means that their fate and transport would not be appreciably different than the chemicals that
10   are publicly known.

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[1] Well operators may specify certain ingredients as confidential business information (CBI) and not disclose the chemicals used to FracFocus. The CASRNs of a range of CBI chemicals were provided to USEPA by 9 service companies.
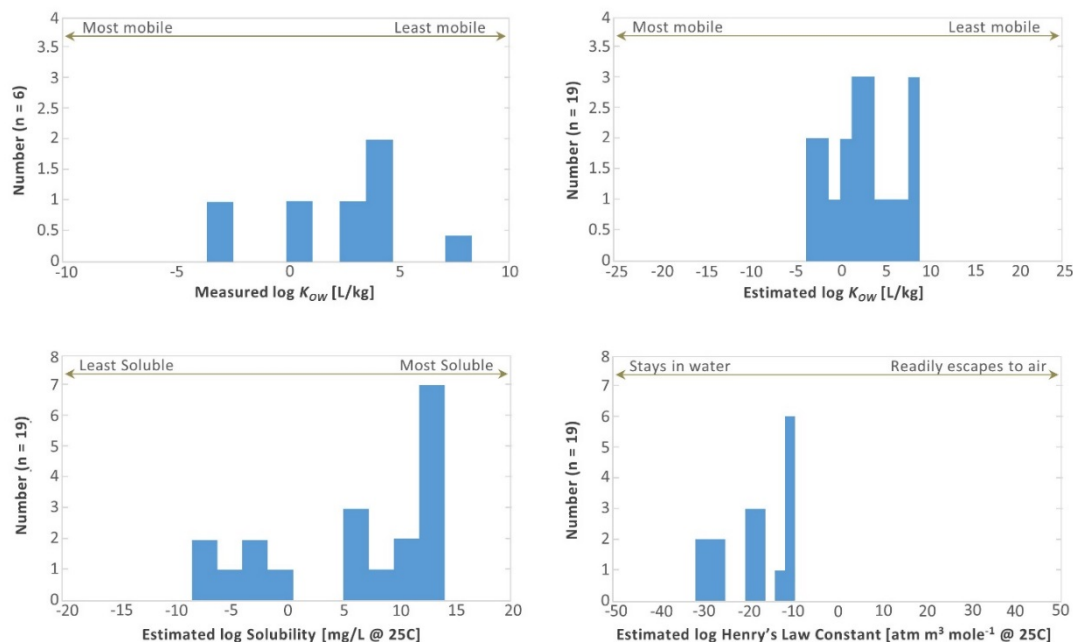
**Figure 5-18. Histograms of physicochemical properties of confidential chemicals used in the hydraulic fracturing process.**

Source: ([U.S. EPA, 2013a](#))
Measured values of log $K_{ow}$ (upper left). Estimated physicochemical properties for log $K_{ow}$ (upper right), log solubility (lower left), and log Henry's law constant (lower right) for all chemicals. Physicochemical properties (log $K_{ow}$, solubility, and Henry's Law constant) estimated by EPI Suite™.

### 5.8.3. Mobility of Chemicals

1   Figure 5-17 shows the distribution of the three properties. The log $K_{ow}$ distribution demonstrates
2   that the chemicals cover a wide range from the more mobile to the less mobile. The more
3   hydrophilic chemicals are more mobile (i.e., they move with water). The more hydrophobic
4   chemicals tend to associate with organic matter and soil particles and to be less mobile in the
5   environment, and they may serve as long-term sources of contamination. A large number of the
6   chemicals fall near log $K_{ow}$ = 0, which indicates that these chemicals are likely to associate roughly
7   equally with organic or aqueous phases. However, overall the log $K_{ow}$ values are skewed positively,
8   indicating less mobile chemicals, which may result in their being later-term or long-term sources of
9   impact to drinking water. The log S values span a wide range from fully miscible to sparingly
10  soluble. Many of the chemicals have high aqueous solubilities, with a large number being fully
11  miscible. Most of the chemicals have log Henry's law constants less than 0, indicating that most are
12  not volatile. Once these chemicals dissolve into water they will tend to stay there rather than
13  volatilize. Therefore, volatilization does not generally serve as a loss process for most hydraulic
14  fracturing chemicals.

15  Table 5-7 and Table 5-8 present the 20 most mobile and least mobile chemicals, known to be used
16  in hydraulic fracturing fluids, respectively, as ranked by log $K_{ow}$. These were taken from the list of

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1   515 chemicals with estimated values for physicochemical properties. These tables also include
2   values for aqueous solubility and Henry's law constant, as well as frequency of use, based on
3   FracFocus disclosures (U.S. EPA, 2015a).

4   Table 5-7 shows the chemicals that have the *lowest* log $K_{ow}$ and are, thus, the *most* mobile. These
5   chemicals are fully miscible (i.e., they will mix completely with water), which means they may move
6   through the environment at high concentrations, leading to greater severity of impact. These
7   chemicals generally have low volatility, based on their negative log Henry's law constants (i.e., will
8   remain in water and will not be lost to the air). These chemicals will dissolve in water and move
9   rapidly through the environment (e.g., via infiltration into the subsurface or via overland flow to
10  surface waters). Chemicals exhibiting this combination of properties have greater potential to cause
11  immediate impacts to drinking water resources. Most of the chemicals in the table were
12  infrequently reported (≤2% of wells) in FracFocus (U.S. EPA, 2015a). However, choline chloride
13  (14% of wells), used for clay control, and tetrakis(hydroxymethyl)phosphonium sulfate (11% of
14  wells), a biocide, were more commonly reported.

**Table 5-7. Ranking of the 20 most mobile organic chemicals, as determined by the largest log $K_{ow}$, with CASRN, percent of wells where the chemical is reported from January 1, 2011 to February 28, 2013 (U.S. EPA, 2015b), and physicochemical properties (log $K_{ow}$, solubility, and Henry's Law constant) as estimated by EPI Suite™.**

(U.S. EPA, 2015b)
For organic salts, parameters are estimated using the desalted form.

| Rank | Chemical Name | CASRN | Percent of wells (U.S. EPA, 2015b)[a] | Estimated Log $K_{ow}$ (unitless)[b] | Estimated Water Solubility (mg/L @ 25°C)[c] | Estimated Henry's Law Constant (atm m$^3$/mole @ 25°C)[d] |
|---|---|---|---|---|---|---|
| 1 | 1,2-Ethanediaminium, N,N'-bis[2-[bis(2-hydroxyethyl)methylammonio]ethyl]-N,N'-bis(2-hydroxyethyl)-N,N'-dimethyl-, tetrachloride | 138879-94-4 | 2% | -23.19 | $1.00 \times 10^6$ | $2.33 \times 10^{-35}$ |
| 2 | Phosphonic acid, [[(phosphonomethyl)imino]bis[2,1-ethanediylnitrilobis(methylene)]]tetrakis- | 15827-60-8 | 0.2% | -9.72 | $1.00 \times 10^6$ | NA |
| 3 | Phosphonic acid, [[(phosphonomethyl)imino]bis[2,1-ethanediylnitrilobis(methylene)]]tetrakis-, sodium salt | 22042-96-2 | 0.07% | -9.72 | $1.00 \times 10^6$ | NA |

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| Rank | Chemical Name | CASRN | Percent of wells (U.S. EPA, 2015b)[a] | Estimated Log $K_{ow}$ (unitless)[b] | Estimated Water Solubility (mg/L @ 25°C)[c] | Estimated Henry's Law Constant (atm m³/mole @ 25°C)[d] |
|---|---|---|---|---|---|---|
| 4 | Phosphonic acid, [[(phosphonomethyl)imino]bis[2,1-ethanediylnitrilobis(methylene)]]tetrakis-, ammonium salt (1:x) | 70714-66-8 | NA | -9.72 | $1.00 \times 10^6$ | NA |
| 5 | Phosphonic acid, (((2-[(2-hydroxyethyl)(phosphonomethyl)amino)ethyl)imino]bis(methylene))bis-, compd. with 2-aminoethanol | 129828-36-0 | NA | -6.73 | $1.00 \times 10^6$ | $5.29 \times 10^{-42}$ |
| 6 | 2-Hydroxy-N,N-bis(2-hydroxyethyl)-N-methylethanaminium chloride | 7006-59-9 | NA | -6.7 | $1.00 \times 10^6$ | $4.78 \times 10^{-19}$ |
| 7 | N-(3-Chloroallyl)hexaminium chloride | 4080-31-3 | 0.02% | -5.92 | $1.00 \times 10^6$ | $1.76 \times 10^{-8}$ |
| 8 | 3,5,7-Triazatricyclo(3.3.1.1(superscript 3,7))decane, 1-(3-chloro-2-propenyl)-, chloride, (Z)- | 51229-78-8 | NA | -5.92 | $1.00 \times 10^6$ | $1.76 \times 10^{-8}$ |
| 9 | (2,3-dihydroxypropyl)trimethylammonium chloride | 34004-36-9 | NA | -5.8 | $1.00 \times 10^6$ | $9.84 \times 10^{-18}$ |
| 10 | Phosphonic acid, [[(phosphonomethyl)imino]bis[6,1-hexanediylnitrilobis(methylene)]]tetrakis- | 34690-00-1 | 0.006% | -5.79 | $1.00 \times 10^6$ | NA |
| 11 | [Nitrilotris(methylene)]tris-phosphonic acid pentasodium salt | 2235-43-0 | 0.5% | -5.45 | $1.00 \times 10^6$ | $1.65 \times 10^{-34}$ |
| 12 | Aminotrimethylene phosphonic acid | 6419-19-8 | 2% | -5.45 | $1.00 \times 10^6$ | $1.65 \times 10^{-34}$ |
| 13 | Choline chloride | 67-48-1 | 14% | -5.16 | $1.00 \times 10^6$ | $2.03 \times 10^{-16}$ |
| 14 | Choline bicarbonate | 78-73-9 | 0.2% | -5.16 | $1.00 \times 10^6$ | $2.03 \times 10^{-16}$ |
| 15 | alpha-Lactose monohydrate | 5989-81-1 | NA | -5.12 | $1.00 \times 10^6$ | $4.47 \times 10^{-22}$ |
| 16 | Lactose | 63-42-3 | NA | -5.12 | $1.00 \times 10^6$ | $4.47 \times 10^{-22}$ |

| Rank | Chemical Name | CASRN | Percent of wells ([U.S. EPA, 2015b](#))[a] | Estimated Log $K_{ow}$ (unitless)[b] | Estimated Water Solubility (mg/L @ 25°C)[c] | Estimated Henry's Law Constant (atm m³/mole @ 25°C)[d] |
|---|---|---|---|---|---|---|
| 17 | Tetrakis(hydroxymethyl)phosphonium sulfate | 55566-30-8 | 11% | -5.03 | $1.00 \times 10^6$ | $9.17 \times 10^{-13}$ |
| 18 | Disodium ethylenediaminediacetate | 38011-25-5 | 0.6% | -4.79 | $1.00 \times 10^6$ | $1.10 \times 10^{-16}$ |
| 19 | Nitrilotriacetamide | 4862-18-4 | NA | -4.75 | $1.00 \times 10^6$ | $1.61 \times 10^{-18}$ |
| 20 | 1,3,5-Triazine-1,3,5(2H,4H,6H)-triethanol | 4719-04-4 | 0.2% | -4.67 | $1.00 \times 10^6$ | $1.08 \times 10^{-11}$ |

[a] Some of the chemicals in these tables have NA (not available) listed as the number of wells, which means that these chemicals have been used in hydraulic fracturing, but they were not reported to FracFocus program for the time period of the study (January 1, 2011, to February 28, 2013) ([U.S. EPA, 2015b](#)). Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.
[b] Log $K_{ow}$ is estimated using the KOWWIN™ model, which uses an atom/fragment contribution method.
[c] Water solubility is estimated using the WSKOWWIN™ model, which estimates a chemical's solubility from $K_{ow}$ and any applicable correction factors.
[d] Henry's Law constant is estimated using the HENRYWIN™ model using the bond contribution method.

1  Table 5-8 shows the chemicals that have the *highest* log $K_{ow}$ and are, thus, the *least* mobile. The
2  estimated aqueous solubilities for some of these chemicals are extremely low, with highest
3  solubilities of <10 μg/L. Therefore, the concentration of these chemicals dissolved in water will be
4  low. The estimated Henry's law constants are more variable for these low-mobility chemicals.
5  Chemicals with high log $K_{ow}$ values (>0) and high Henry's law constants will sorb strongly to organic
6  phases and solids and may volatilize. However, their strong preference for the organic or solid
7  phase may slow or reduce volatilization. The chemicals with low Henry's law constants will readily
8  sorb to organic phases and solids. Less mobile chemicals will move slowly through the soil and have
9  potentially delayed and longer-term impacts to drinking water resources. Seven of the chemicals in
10 Table 5-8 were reported to FracFocus ([U.S. EPA, 2015b](#)). Five were reported infrequently (<1% of
11 wells). Tri-n-butyltetradecylphosphonium chloride (6% of wells), used as a biocide, and
12 C>10-alpha-alkenes (8% of wells), a mixture of alpha-olefins with carbon numbers greater than 10
13 used as a corrosion inhibitor, were more commonly reported. The least mobile organic chemical is

1  sorbitan, tri-(9Z)-9-octadecenoate, a mineral oil co-emulsifier (0.05% of wells), with an estimated
2  log $K_{ow}$ of 22.56.[1]

**Table 5-8. Ranking of the 20 least mobile organic chemicals, as determined by the largest log $K_{ow}$, with CASRN, percent of wells where the chemical is reported from January 1, 2011 to February 28, 2013 (U.S. EPA, 2015b), and physicochemical properties (log $K_{ow}$, solubility, and Henry's Law constant) as estimated by EPI Suite™.**

Source: (U.S. EPA, 2015b)
For organic salts, parameters are estimated using the desalted form.

| Rank | Chemical Name | CASRN | Percent of wells (U.S. EPA, 2015b)[a] | Estimated Log $K_{ow}$ (unitless)[b] | Estimated Water Solubility (mg/L @ 25°C)[c] | Estimated Henry's Law Constant (atm m³/ mole @ 25°C)[d] |
|---|---|---|---|---|---|---|
| 1 | Sorbitan, tri-(9Z)-9 octadecenoate | 26266-58-0 | 0.05% | 22.56 | $1.12 \times 10^{-19}$ | $4.02 \times 10^{-11}$ |
| 2 | Fatty acids, C18-unsatd., dimers | 61788-89-4 | NA | 14.6 | $2.31 \times 10^{-10}$ | $4.12 \times 10^{-08}$ |
| 3 | Sorbitan sesquioleate | 8007-43-0 | 0.02% | 14.32 | $2.31 \times 10^{-11}$ | $7.55 \times 10^{-12}$ |
| 4 | Tri-n-butyltetradecyl-phosphonium chloride | 81741-28-8 | 6% | 11.22 | $7.90 \times 10^{-7}$ | $2.61 \times 10^{-1}$ |
| 5 | Sodium bis(tridecyl) sulfobutanedioate | 2673-22-5 | NA | 11.15 | $7.46 \times 10^{-9}$ | $8.51 \times 10^{-11}$ |
| 6 | 1-Eicosene | 3452-07-1 | NA | 10.03 | $1.26 \times 10^{-5}$ | $1.89 \times 10^{1}$ |
| 7 | D&C Red 28 | 18472-87-2 | NA | 9.62 | $1.64 \times 10^{-8}$ | $6.37 \times 10^{-21}$ |
| 8 | C.I. Solvent Red 26 | 4477-79-6 | NA | 9.27 | $5.68 \times 10^{-5}$ | $5.48 \times 10^{-13}$ |
| 9 | 1-Octadecene | 112-88-9 | NA | 9.04 | $1.256 \times 10^{-4}$ | $1.07 \times 10^{1}$ |
| 10 | Alkenes, C>10 alpha- | 64743-02-8 | 8% | 8.55 | $3.941 \times 10^{-4}$ | $8.09 \times 10^{0}$ |
| 11 | Dioctyl phthalate | 117-84-0 | NA | 8.54 | $4.236 \times 10^{-4}$ | $1.18 \times 10^{-5}$ |
| 12 | Benzene, C10-16-alkyl derivs. | 68648-87-3 | 0.5% | 8.43 | $2.099 \times 10^{-4}$ | $1.78 \times 10^{-1}$ |

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[1] Sorbitan, tri-(9Z)-9-octadecenoate, CASRN 26266-58-0, is soluble in hydrocarbons and insoluble in water, listed as an effective coupling agent and co-emulsifier for mineral oil (Santa Cruz Biotechnology, 2015; ChemicalBook, 2010).

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| Rank | Chemical Name | CASRN | Percent of wells ([U.S. EPA, 2015b])[a] | Estimated Log $K_{ow}$ (unitless)[b] | Estimated Water Solubility (mg/L @ 25°C)[c] | Estimated Henry's Law Constant (atm m³/ mole @ 25°C)[d] |
|---|---|---|---|---|---|---|
| 13 | Di(2-ethylhexyl) phthalate | 117-81-7 | NA | 8.39 | $1.132 \times 10^{-3}$ | $1.18 \times 10^{-5}$ |
| 14 | 1-Octadecanamine, N,N-dimethyl- | 124-28-7 | NA | 8.39 | $8.882 \times 10^{-3}$ | $4.51 \times 10^{-3}$ |
| 15 | N,N-dimethyloctadecylamine hydrochloride | 1613-17-8 | NA | 8.39 | $8.882 \times 10^{-3}$ | $4.51 \times 10^{-3}$ |
| 16 | Butyryl trihexyl citrate | 82469-79-2 | 0.03% | 8.21 | $5.56 \times 10^{-5}$ | $3.65 \times 10^{-9}$ |
| 17 | 1-Hexadecene | 629-73-2 | NA | 8.06 | $1.232 \times 10^{-3}$ | $6.10 \times 10^{0}$ |
| 18 | Benzo(g,h,i)perylene | 191-24-2 | NA | 7.98 | $7.321 \times 10^{-4}$ | $1.26 \times 10^{-2}$ |
| 19 | Dodecylbenzene | 123-01-3 | NA | 7.94 | $1.015 \times 10^{-3}$ | $1.34 \times 10^{-1}$ |
| 20 | Isopropanolamine dodecylbenzene | 42504-46-1 | 0.02% | 7.94 | $1.015 \times 10^{-3}$ | $1.34 \times 10^{-1}$ |

[a] Some of the chemicals in these tables have NA (not available) listed as the number of wells, which means that these chemicals have been used in hydraulic fracturing, but they were not reported to FracFocus program for the time period of the study (January 1, 2011, to February 28, 2013) ([U.S. EPA, 2015b]). Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.
[b] Log $K_{ow}$ is estimated using the KOWWIN™ model, which uses an atom/fragment contribution method.
[c] Water solubility is estimated using the WSKOWWIN™ model, which estimates a chemical's solubility from Kow and any applicable correction factors.
[d] Henry's Law constant is estimated using the HENRYWIN™ model using the bond contribution method.

1  Table 5-9 shows the EPI Suite™ estimated physicochemical property values of the 20 chemicals
2  most frequently reported to FracFocus nationwide, with estimated mean and median volumes
3  based on FracFocus data. Most have log $K_{ow}$ < 1, meaning that they are generally hydrophilic and
4  will associate with water. These chemicals also have very high solubilities, so they will be mobile in
5  the environment and go where the water goes. These chemicals have the potential for immediate
6  impacts to drinking water resources. Naphthalene has a measured log $K_{ow}$ = 3.3 with an estimated
7  solubility of 142.1 mg/L, which means it will be less mobile in the environment. Naphthalene will
8  sorb to particles and move slowly through the environment, and have the potential to act as long-

1     term sources of contamination.[1] All of these chemicals have low Henry's law constants, so they tend

2     not to volatilize.

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[1] Chemicals may have the potential to be long-term sources of contamination because they move slowly through the environment. In this discussion, we are not accounting for biodegradation or other transformation processes, which may reduce the persistence of certain chemicals in the environment. Under the right conditions, for example, naphthalene is biodegradable, which may reduce or remove it from the environment, and thus may not be a long-term source of contamination.

**Table 5-9. The 20 chemicals reported most frequently nationwide for hydraulic fracturing based on reported FracFocus 1.0 disclosures (U.S. EPA, 2015b), with EPI Suite™ physicochemical parameters where available, and estimated mean and median volumes of those chemicals, where density was available.**

Source: (U.S. EPA, 2015b)

Excludes water, sodium chloride, and quartz. NA means that the physicochemical parameter is not provided by EPI Suite™. For organic salts, parameters are estimated using the desalted form.

| Rank | Chemical Name | CASRN | Number Of Wells Using Chemical (% of wells) | Log $K_{ow}$ (unitless) | | Water Solubility Estimate From Log $K_{ow}$ (mg/L @ 25$^o$C) | Henry's Law Constant (atm m$^3$/mole @ 25$^o$C) | | | Estimated Volume, per disclosure (gal) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Estimated | Measured | | Estimated, Bond Method | Estimated, Group Method 25 | Measured | Mean | Median |
| 1 | Methanol | 67-56-1 | 24,753 (72%) | -0.63 | -0.77 | $1.00 \times 10^6$ | $4.27 \times 10^{-6}$ | $3.62 \times 10^{-6}$ | $4.55 \times 10^{-6}$ | 1,218 | 110 |
| 2 | Distillates, petroleum, hydrotreated light | 64742-47-8 | 22,463 (65%) | NA | NA | NA | NA | NA | NA | NA | NA |
| 3 | Hydrochloric acid | 7647-01-0 | 22,380 (65%) | NA | NA | NA | NA | NA | NA | 28,320 | 3,110 |
| 4 | Isopropanol | 67-63-0 | 16,039 (47%) | 0.28 | 0.05 | $4.024 \times 10^5$ | $7.52 \times 10^{-6}$ | $1.14 \times 10^{-5}$ | $8.10 \times 10^{-6}$ | 2,095 | 55 |
| 5 | Ethylene glycol | 107-21-1 | 15,800 (46%) | -1.2 | -1.36 | $1.00 \times 10^6$ | $1.31 \times 10^{-7}$ | $5.60 \times 10^{-11}$ | $6.00 \times 10^{-8}$ | 614 | 184 |
| 6 | Peroxydisulfuric acid, diammonium salt | 7727-54-0 | 14,968 (44%) | NA | NA | NA | NA | NA | NA | NA | NA |
| 7 | Sodium hydroxide | 1310-73-2 | 13,265 (39%) | NA | NA | NA | NA | NA | NA | 551 | 38 |
| 8 | Guar gum | 9000-30-0 | 12,696 (37%) | NA | NA | NA | NA | NA | NA | NA | NA |
| 9 | Glutaraldehyde | 111-30-8 | 11,562 (34%) | -0.18 | NA | $1.672 \times 10^5$ | $1.10 \times 10^{-7}$ | $2.39 \times 10^{-8}$ | NA | 1,313 | 122 |
| 10 | Propargyl alcohol | 107-19-7 | 11,410 (33%) | -0.42 | -0.38 | $9.355 \times 10^5$ | $5.88 \times 10^{-7}$ | NA | $1.15 \times 10^{-6}$ | 183 | 2 |
| 11 | Potassium hydroxide | 1310-58-3 | 10,049 (29%) | NA | NA | NA | NA | NA | NA | NA | NA |

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| Rank | Chemical Name | CASRN | Number Of Wells Using Chemical (% of wells) | Log $K_{ow}$ (unitless) | | Water Solubility Estimate From Log $K_{ow}$ (mg/L @ 25°C) | Henry's Law Constant (atm m³/mole @ 25°C) | | | Estimated Volume, per disclosure (gal) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Estimated | Measured | | Estimated, Bond Method | Estimated, Group Method 25 | Measured | Mean | Median |
| 12 | Ethanol | 64-17-5 | 9,861 (29%) | -0.14 | -0.31 | $7.921 \times 10^5$ | $5.67 \times 10^{-6}$ | $4.88 \times 10^{-6}$ | 5.00E-06 | 831 | 121 |
| 13 | Acetic acid | 64-19-7 | 8,186 (24%) | 0.09 | -0.17 | $4.759 \times 10^5$ | $5.48 \times 10^{-7}$ | $2.94 \times 10^{-7}$ | $1.00 \times 10^{-7}$ | 646 | 47 |
| 14 | Citric acid | 77-92-9 | 8,142 (24%) | -1.67 | -1.64 | $1.00 \times 10^6$ | $8.33 \times 10^{-18}$ | NA | $4.33 \times 10^{-14}$ | 163 | 20 |
| 15 | 2-Butoxyethanol | 111-76-2 | 7,347 (21%) | 0.57 | 0.83 | $6.447 \times 10^4$ | $9.79 \times 10^{-8}$ | $2.08 \times 10^{-8}$ | $1.60 \times 10^{-6}$ | 385 | 26 |
| 16 | Solvent naphtha, petroleum, heavy arom. | 64742-94-5 | 7,108 (21%) | NA | NA | NA | NA | NA | NA | NA | NA |
| 17 | Naphthalene | 91-20-3 | 6,354 (19%) | 3.17 | 3.3 | $1.421 \times 10^2$ | $5.26 \times 10^{-4}$ | $3.7 \times 10^{-4}$ | $4.4 \times 10^{-4}$ | 72 | 12 |
| 18 | 2,2-Dibromo-3-nitrilopropionamide | 10222-01-2 | 5,656 (16%) | 1.01 | 0.82 | $2.841 \times 10^3$ | $6.16 \times 10^{-14}$ | NA | $1.91 \times 10^{-8}$ | 183 | 5 |
| 19 | Phenolic resin | 9003-35-4 | 4,961 (14%) | NA | NA | NA | NA | NA | NA | NA | NA |
| 20 | Choline chloride | 67-48-1 | 4,741 (14%) | -5.16 | NA | $1.00 \times 10^6$ | $2.03 \times 10^{-16}$ | NA | NA | 2,131 | 290 |

Note: Analysis considered 34,675 disclosures and 676,376 ingredient records that met selected quality assurance criteria, including: completely parsed; unique combination of fracture date and API well number; fracture date between January 1, 2011, and February 28, 2013; valid CASRN; and valid concentrations. Disclosures that did not meet quality assurance criteria (3,855) or other, query-specific criteria were excluded from analysis.

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### 5.8.4. Transformation Processes

1    It is important to understand the processes governing transformation of chemicals in the
2    environment. The transformation of a chemical reduces its concentration over time. Chemicals may
3    completely degrade before reaching a drinking water resource. Transformation processes may be
4    biotic or abiotic. The transformation process may transform a chemical into a less or more harmful
5    chemical.

6    Biodegradation is a biotic process where microorganisms transform a chemical from its original
7    form into another chemical. For example, the biodegradation pathway of methanol is
8    $CH_3OH \rightarrow CH_2O \rightarrow CHOOH \rightarrow CO_2$, or methanol $\rightarrow$ formaldehyde $\rightarrow$ formic acid $\rightarrow$ carbon dioxide.
9    This pathway shows how the original chemical transforms through a series of steps until it becomes
10   the final product, carbon dioxide. Some chemicals are readily biodegraded, while others break
11   down slowly over time. Biodegradation is a highly site-specific process, requiring nutrients, a
12   carbon source, water, and an energy source. A highly biodegradable chemical could be persistent if
13   the conditions for biodegradability are not met. Conversely, a highly biodegradable chemical could
14   biodegrade quickly under the right conditions, before it can impact a water resource. The
15   relationship between mobility and biodegradability is complex, and therefore a variety of factors
16   can influence a particular compound's movement through the environment.

17   Abiotic processes, such as oxidation, reduction, photochemical reaction, and hydrolysis, can
18   transform or break apart chemicals. In hydrolysis, for example, a water molecule substitutes for a
19   group of atoms. The typical results are products that are more polar than the original, and thus
20   have different physicochemical properties than the original compounds ([Schwarzenbach et al.,](#)
21   [2002](#)).[1]

### 5.8.5. Fate and Transport of Chemical Mixtures

22   Chemicals at hydraulic fracturing sites are often present as mixtures, which may act differently in
23   the environment than individual chemicals do. Individual chemicals can affect the fate and
24   transport of other chemicals in a mixture primarily by changing their solubility and biodegradation
25   rates.

26   Mixtures of chemicals may be more mobile than individual chemicals due to cosolvency, which
27   increases solubility in the aqueous phase. Methanol and ethanol are examples of cosolvent alcohols
28   used frequently in hydraulic fracturing fluids ([U.S. EPA, 2015a](#)). The presence of either greatly
29   increases BTEX solubility ([Rasa et al., 2013](#); [Corseuil et al., 2011](#); [Heermann and Powers, 1998](#)).[2] By
30   increasing solubility, ethanol can affect the fate and transport of other compounds. For example,

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[1] A polar molecule is a molecule with a slightly positive charge at one part of the molecule and a slightly negative charge on another. The water molecule, $H_2O$, is an example of a polar molecule, where the molecule is slightly positive around the hydrogen atoms and negative around the oxygen atom.
[2] BTEX is an acronym for benzene, toluene, ethylbenzene, and xylenes. These chemicals are a group of single ringed aromatic hydrocarbons based on the benzene structure. These compounds are found in petroleum and are of specific importance because of their potential health effects.

1    BTEX has been observed to travel farther in the subsurface in the presence of ethanol (Rasa et al.,
2    2013; Corseuil et al., 2011; Corseuil et al., 2004; Powers et al., 2001; Heermann and Powers, 1998).

3    The presence of surfactants lowers fluid surface tension and increases solubility of organic
4    compounds, and can mobilize less soluble/less mobile organic compounds. Two common
5    surfactants reported to FracFocus 1.0 from January 1, 2011 to February 2013 were 2-
6    butoxyethanol (CASRN 111-76-2, 21% of disclosures) and poly(oxy-1,2-ethanediyl)-nonylphenyl-
7    hydroxy (mixture) (CASRN 127087-87-0, 20% of disclosures). Additionally, surfactants can
8    mobilize bacteria in the subsurface, which can increase the impact of pathogens on drinking water
9    resources (Brown and Jaffé, 2001).

10   When chemicals are present as mixtures, one chemical may decrease or enhance the
11   biodegradability of another through inhibition or co-metabolism. The process of inhibition can
12   occur when multiple chemicals compete for the same enzyme, so only one chemical is degraded at a
13   time, which can ultimately slow biodegradation of each of the chemicals present. For example, the
14   biodegradation of ethanol and methanol may inhibit the biodegradation of BTEX or other organic
15   compounds present (Rasa et al., 2013; Powers et al., 2001). Co-metabolism may increase the
16   biodegradation rate of other compounds. For example, when methane or propane is present with
17   tetrachloroethylene, the enzyme produced by bacteria to degrade methane also degrades
18   tetrachloroethylene (e.g., Alvarez-Cohen and Speitel, 2001 and references therein). For the
19   purposes of chemicals used in hydraulic fracturing, the presence of other chemicals in additives and
20   hydraulic fracturing fluids could result in increased or decreased biodegradation if the chemicals
21   are spilled. A chemical that may have otherwise been biodegradable may be inhibited and act as a
22   long-term source.

### 5.8.6.   Site and Environmental Conditions

23   Environmental conditions at and around the spill site affect the movement and transformation of
24   the chemical. We discuss the following: site conditions (e.g., proximity, land cover, and slope), soil
25   conditions (e.g., permeability and porosity), and weather and climate.

26   The proximity of a spill to a drinking water resource, either laterally in the case of a surface water
27   body or downward for an aquifer, affects the potential for impact. Land cover will affect how
28   readily a fluid moves over land. For example, more rugged land cover such as forest will impede
29   flow, and an asphalt road will facilitate flow. A spill that occurs on or near a sloped site may move
30   overland faster, making it more likely to reach a nearby surface water body. Flatter surfaces would
31   result in a greater chance for infiltration to the subsurface, which could increase the potential for a
32   ground water impact.

33   Soil characteristics that affect the transport and transformation of spill chemicals include soil
34   texture (e.g., clay, silt, sand), permeability, porosity, and organic content.[1,1] Fluids will move more

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[1] Permeability of a soil describes how easily a fluid can move through the soil. Under a constant pressure, a fluid will move
faster in a high permeability soil than the same fluid in a low permeability soil.

1   quickly through permeable soil (e.g., sand) than through less permeable soil (e.g., clay). A soil with a
2   high porosity provides more volume to hold water and spilled chemicals. Another important factor
3   for a site is the organic content, of which there are two competing types: soil organic carbon and
4   dissolved organic carbon. Each type of carbon acts as a strong substance for chemicals to adhere to.
5   Soil organic carbon present in a solid phase, such as dead and decaying leaves and roots, is not
6   mobile and slows the movement of chemicals through the soil. Dissolved organic carbon (DOC)
7   moves with the water and can act as a shuttling mechanism to move insoluble chemicals across the
8   surface and through the subsurface. Chemicals may also associate with particulates and colloids,
9   which may act as an important transport mechanism.

10   Weather and climate conditions also affect the fate and transport of a spilled chemical. After a
11   spilled chemical stops moving, rainfall may remobilize the chemical. The amount, frequency, and
12   intensity of precipitation will impact volume, distance, and speed of chemical movement.
13   Precipitation may carry chemicals downward or overland, and it can cause erosion, which may
14   move sorbed chemicals overland.

### 5.8.7.   Peer-Reviewed Literature on the Fate and Transport of Hydraulic Fracturing Fluid Spills

15   There has been limited peer-reviewed research investigating the fate and transport of chemicals
16   spilled at hydraulic fracturing sites. Aminto and Olson (2012) modeled a hypothetical spill of
17   1,000 gal (3,800 L) of hydraulic fracturing fluid using equilibrium partitioning. The authors
18   evaluated how 12 chemicals typically used for hydraulic fracturing in the Marcellus Shale would
19   partition amongst different phases: air, water, soil, and biota.[2] They presented a ranking of
20   concentrations for each phase. In water, they showed that sodium hydroxide (a pH buffer), 4,4-
21   dimethyl oxazolidine (a biocide), hydrochloric acid (a perforation clean-up additive), and 3,4,4-
22   trimethyl oxazolidine (a biocide) had the highest simulated water concentrations; however, these
23   concentrations depended on the chemicals included in the simulated mixture and the
24   concentrations of each. Their analysis also suggested that after a spill, a large fraction would enter
25   the air and leave the soil; however, some constituents would be left behind in the water, soil, and
26   biota compartments, which could effectively act as long-term contamination sources. Aminto and
27   Olson (2012) only studied this one scenario. Other scenarios could be constructed with different
28   chemicals in different concentrations. These scenarios may result in different outcomes with
29   greater impacts.

### 5.8.8.   Potential and Documented Fate and Transport of Documented Spills

30   There is limited information on the fate and transport of hydraulic fracturing fluids and chemicals.
31   In this section, we highlight the potential and documented impacts for three documented spills (U.S.
32   EPA, 2015n). In each case, we provide the documented and potential paths (surface, subsurface, or
33   combination) and the associated fate and transport governing processes by which a spill has been

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[1] Porosity of a soil describes the amount of empty space for a given volume of soil. The porosity describes how much air, water, or hydraulic fluid a given volume of soil can hold.
[2] The chemicals they investigated included: sodium hydroxide, ethylene glycol, 4,4-dimethyl oxazolidine, 3,4,4-trimethyl oxazolodine, 2-amino-2-methyl-1-propanol, formamide, glutaraldehyde, benzalkonium chloride, ethanol, hydrochloric acid, methanol, and propargyl alcohol.

1   documented to or has the potential to have an impact on drinking water resources. The three cases
2   involve a tank overflow with a surface water impact, a human error blender spill with a soil impact,
3   and an equipment failure that had no impact. These three spills were chosen to highlight cases
4   where there was a documented impact, a potential impact, and no impact.

5   In the first documented spill, shown in Figure 5-19, a tank overflowed twice, releasing a total of
6   7,350 gal (980 ft$^3$ or 28 m$^3$).[1] The spilled fluid was documented as containing a friction reducer and
7   gel. The spill traveled across the land surface, crossed a road, and then continued to a nearby
8   stream. The spill affected wetlands and a stream, where fish were killed. The fish kill indicates that
9   the chemicals present were in high enough concentrations to have an adverse impact.
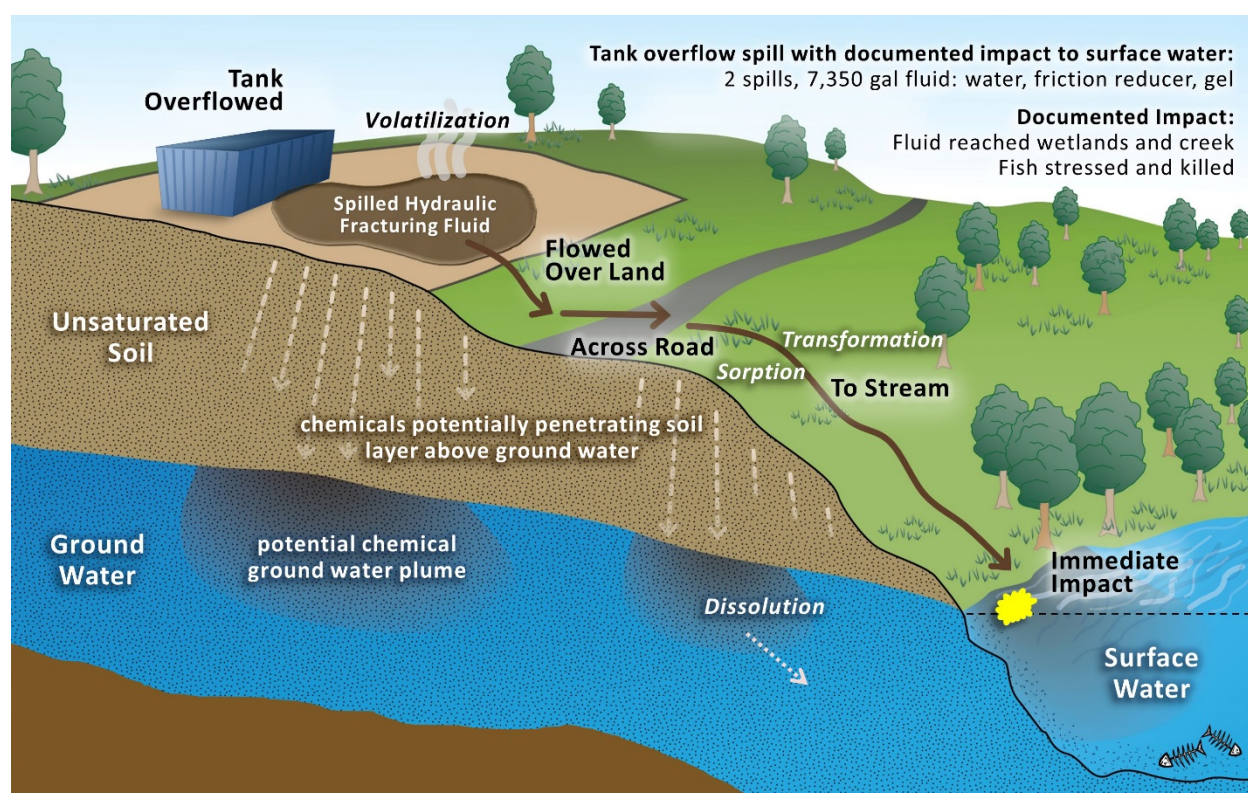


**Figure 5-19. Fate and Transport Spill Example: Case 1.**

Spills information from U.S. EPA (2015n).

10   For this first spill, the documented path was overland flow from the tank to the stream with a
11   documented, immediate impact. In addition to this documented path, there are potential paths for
12   potential impacts to drinking water resources. The spilled chemicals may have penetrated into the
13   soils or sorbed to soils and vegetation as the fluid moved across the ground towards the stream.

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[1] We provide the total volume of the spill in gallons as well as cubic length (cubic feet and cubic meters), because it may be a little harder to visualize how far a volume of 7,300 gal might travel.

1   Chemicals could be mobilized during later rainfall, runoff, or erosion events. Chemicals that
2   infiltrated the subsurface could serve as long-term sources, as well as travel laterally across the
3   unsaturated zone, or continue downwards to the ground water. Additionally, some chemicals could
4   be lost to transformation processes. The lack of reported soil or ground water sampling data
5   prevents the ability to know if these potential paths occurred or not.

6   The second documented spill (U.S. EPA, 2015n, line 144), shown in Figure 5-20, occurred when a
7   cap was left off the blender, and 504 gal (70 ft$^3$ or 2 m$^3$) of biocide and hydraulic fracturing fluid
8   were released; 294 gal (39 ft$^3$ or 1.1 m$^3$) were retained by a dike with a lined secondary
9   containment measure, demonstrating the partial effectiveness of this containment mechanism.
10  However, 210 gal (28 ft$^3$ or 0.8 m$^3$) did run off-site and were vacuumed up. There was no
11  documented impact to surface or ground water.
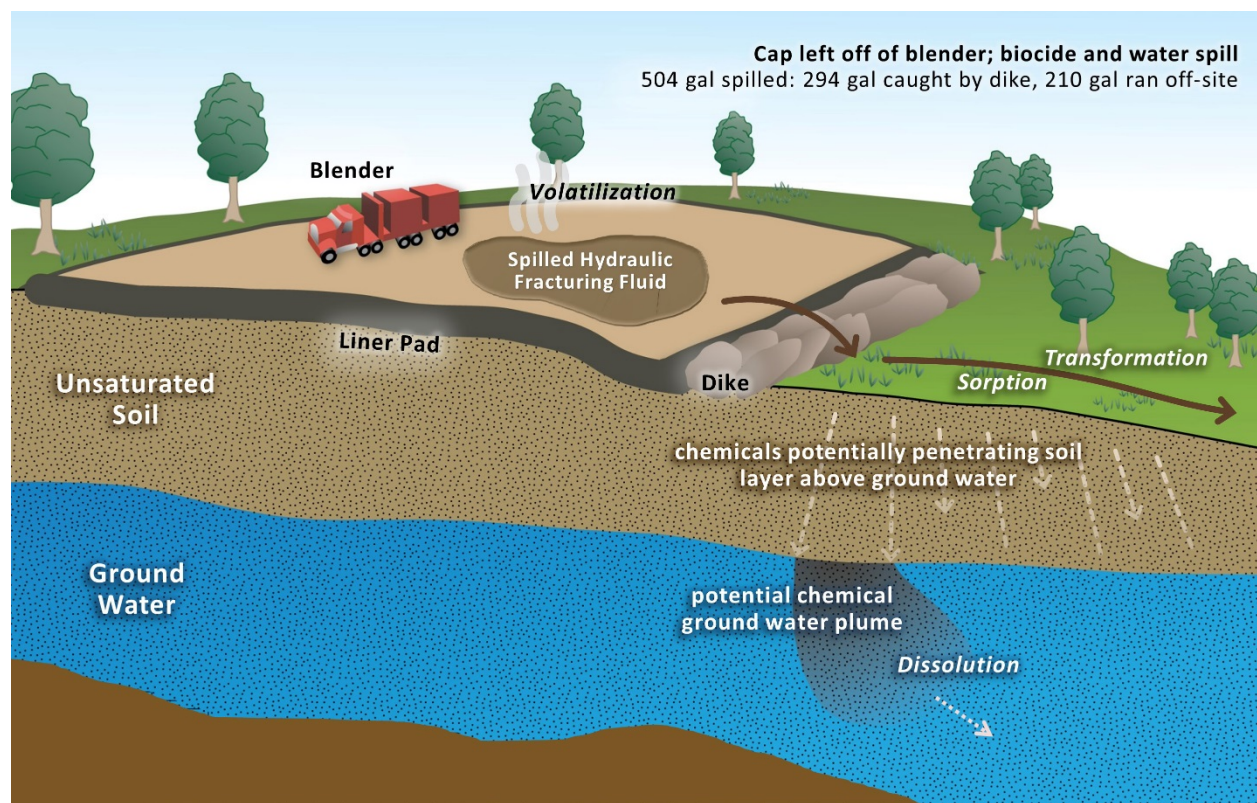


**Figure 5-20. Fate and Transport Spill Example: Case 2.**

Spills information from U.S. EPA (2015n).

12  In this second case, the uncontained 210 gal could have infiltrated the subsurface, creating a
13  potential path to ground water. There is no documented information on the composition of the
14  spilled fluid. Highly mobile chemicals would have penetrated the soil more quickly than less mobile
15  chemicals, which would have sorbed to soil particles. As the chemicals penetrated into the soil,
16  some could have moved laterally in the unsaturated zone, or traveled downward to the water table

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1   and moved with the ground water. These chemicals could have served as a long-term source. These
2   chemicals could have transformed into other chemicals with different physicochemical properties,
3   and any volatile chemicals could have moved to the air as a loss process. As in the first case, there
4   was no reported sampling of soil or groundwater, so there is no way to demonstrate whether
5   chemicals did or did not follow this path.

6   In the third documented spill (U.S. EPA, 2015n, line 188), shown in Figure 5-21, 630 gal (84 ft$^3$ or
7   2.4 m$^3$) of crosslinker spilled onto the well pad when a hose wore off at the cuff. The spill was
8   contained in the berm and an on-site vacuum truck was used to clean up the spill. No impact to soil
9   or water was reported.
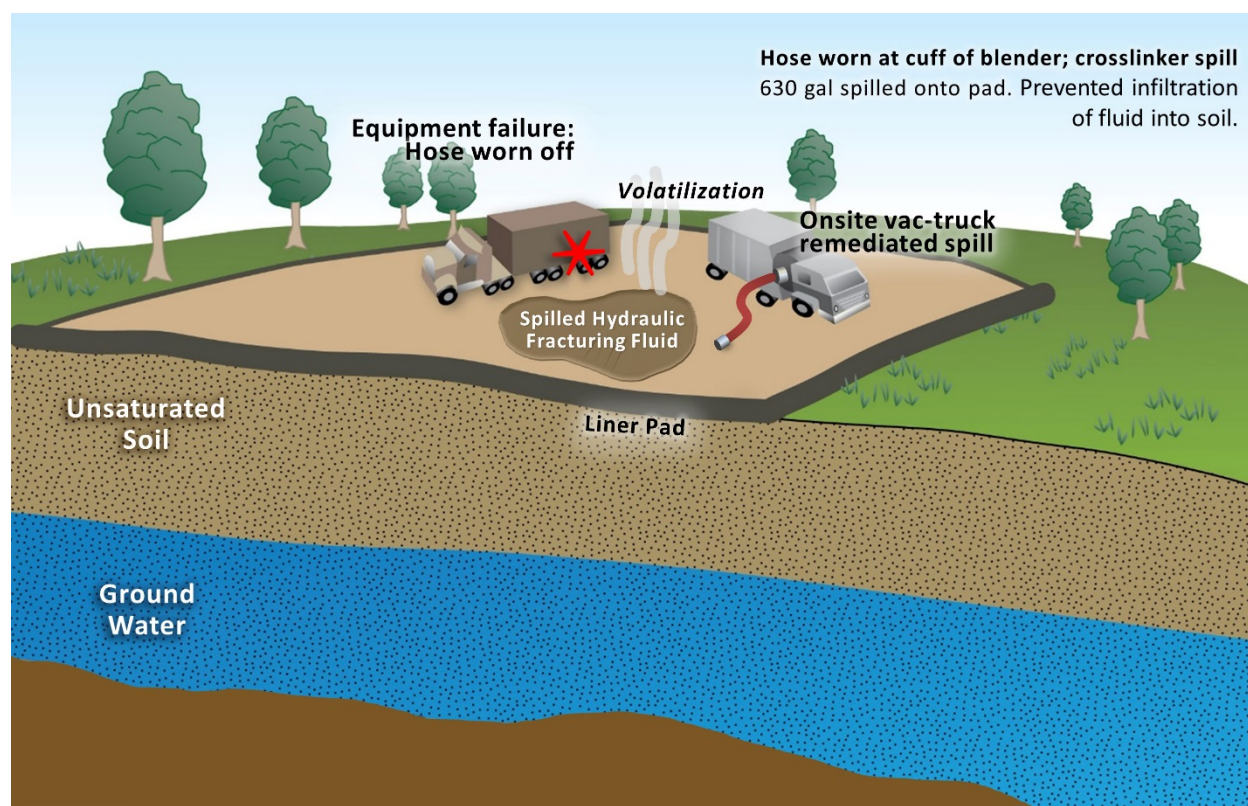


**Figure 5-21. Fate and Transport Spill Example: Case 3.**

Spills information from U.S. EPA (2015n).

10  For this third case, we do not have any information on whether the well pad was lined or not. If the
11  site had a liner, the spill could have been fully contained, not infiltrated the subsurface, and been
12  fully cleaned up. Without a liner or if the liner was not completely successful (e.g., had a tear), the
13  potential paths would have been similar to those above in the second case, where the chemicals
14  may sorb to the soils and penetrate into the subsurface. There was no reported sampling of soil or
15  ground water to determine whether or not chemicals migrated into the soil.

## 5.9. Trends in Chemicals Use in Hydraulic Fracturing

1    This section provides an overview of ongoing changes in chemical use in hydraulic fracturing, with
2    an emphasis on efforts to reduce potential impacts from surface spills by using fewer and safer
3    chemicals.

4    Representatives from oil and gas companies, chemical companies, and non-profits are working on
5    strategies to reduce the number and volume of chemicals used and to identify safer chemicals
6    (Waldron, 2014). Southwestern Energy Company, for example, is developing an internal chemical
7    ranking tool (SWN, 2014), and Baker Hughes is working on a hazard ranking system designed for
8    wide-scale external use (Baker Hughes, 2014a; Brannon et al., 2012; Daulton et al., 2012; Brannon
9    et al., 2011). Environmental groups, such as the Environmental Defense Fund, are also developing
10   hazard rating systems (Penttila et al., 2013). Typical criteria used to rank chemicals include
11   mobility, persistence, biodegradation, bioaccumulation, toxicity, and hazard characteristics. In this
12   report, toxicity and a methodology to rank chemical hazards of hydraulic fracturing chemicals is
13   discussed in Chapter 9.

14   The EPA has not conducted a comprehensive review of efforts to develop safer hydraulic fracturing
15   chemicals. However, the following are some specific examples of efforts that companies cite as part
16   of their efforts toward safer chemical use:

17   • A renewable citrus-based replacement for conventional surfactants (Fisher, 2012);

18   • A crosslinked gel system comprised of chemicals designated as safe food additives by the
19      U.S. Food and Drug Administration (Holtsclaw et al., 2011);

20   • A polymer-free gel additive (Al-Ghazal et al., 2013);

21   • A dry, hydrocarbon-free powder to replace liquid gel concentrate (Weinstein et al., 2009);

22   • Biodegradable polymers (Irwin, 2013);

23   • The use of ultraviolet light to control bacteria (Rodvelt et al., 2013);

24   • New chelating agents that reduce the use of strong acids (LePage et al., 2013), and

25   • The recovery and reuse of flowback and produced water as hydraulic fracturing fluids,
26      which may reduce need to add additional chemicals (Horn et al., 2013).

27   In addition to efforts to address environmental concerns, the oil and gas industry continues to
28   research and develop less expensive and more effective fracturing fluid additives. A review of the
29   EPA's new chemicals program found that from 2009 to April 2015, the Agency received pre-
30   manufacturing notices (PMN) for about 110 chemicals that have the potential for use as hydraulic
31   fracturing fluid additives. Examples include chemicals intended for use as clay control agents,
32   corrosion inhibitors, gel crosslinkers, emulsifiers, foaming agents, hydrate inhibitors, scale
33   inhibitors, and surfactants. At the time of PMN submission, these chemicals were not in commercial
34   use in the United States. As of April 2015, the EPA had received 30 notices of commencement,
35   indicating that some of those chemicals are now used commercially.

1    The FracFocus 1.0 data extracted by the EPA cannot be used to identify temporal trends in additive
2    usage. A data set with a much longer duration of data collection would be needed to distinguish
3    actual temporal trends from the normal diversity of chemicals in use as a result of geologic and
4    geographic variability. However, the current FracFocus 1.0 database provides a point of comparison
5    for use in the future.

## 5.10. Synthesis

6    Chemical mixing is the process by which a base fluid, chemicals, and proppant are mixed prior to
7    injection into the well. This chapter addressed the potential for on-site spills of chemicals used in
8    the hydraulic fracturing process to affect the quality of drinking water resources, which is governed
9    by three overarching factors: (1) fluid characteristics, (2) chemical management and spill
10   characteristics, and (3) chemical fate and transport.

### 5.10.1. Summary of Findings

11   Documented on-site chemical spills have occurred during the chemical mixing process and reached
12   soil and surface water receptors, with potential impacts to drinking water resources. The EPA
13   analysis of 497 spills reports found no documented impacts to ground water from those particular
14   chemical spills, though there was little information on post-spill testing and sampling (U.S. EPA,
15   2015n). The EPA's case study in Killdeer, ND strongly suggests that there was impact to ground
16   water, but it is unclear if the path was via the surface spill caused by the blowout (U.S. EPA, 2015j).
17   The EPA found 151 spills of chemicals or fracturing fluid on or near the well pad in a six-year time
18   period. The chemical spills were primarily caused by equipment failure (34%), closely followed by
19   human error (25%). The remaining spills were caused by a failure of container integrity, weather,
20   vandalism, well communication, or unknown causes. Reported spills cover a large range of volumes,
21   from five to 19,000 gal (19 to 72,000 L), with a median of 420 gal (1,700 L).

22   If a spilled fluid reaches a drinking water resource, the potential to affect the water quality is largely
23   governed by the fluid characteristics. A typical water-based fracturing fluid is composed of 90%–
24   94% water, 5%–9% proppant, and less than 2% chemical additives (Carter et al., 2013; Knappe and
25   Fireline, 2012). According to the EPA's analysis of disclosures to FracFocus 1.0, approximately 93%
26   of hydraulic fracturing fluids are inferred to use water as a base fluid (U.S. EPA, 2015a). Non-
27   aqueous constituents, such as nitrogen, carbon dioxide, and hydrocarbons, are also used as base
28   fluids or used in combination with water as base fluids.

29   The EPA has identified 1,076 unique chemicals used in hydraulic fracturing fluids. The chemicals
30   include acids, aromatic hydrocarbons, bases, hydrocarbon mixtures, polysaccharides, and
31   surfactants. Of the 1,076 chemicals, 453 have physicochemical properties in the EPI Suite™
32   database. These chemicals range from fully miscible to insoluble, and from highly hydrophobic to
33   highly hydrophilic. The majority of the chemicals are not volatile.

34   According to the EPA's analysis of FracFocus, a median of 14 chemicals are used per well, with a
35   range of four to 28 (5th and 95th percentiles). The volumes used range from tens to tens of
36   thousands of gallons (tens to tens of thousands of liters) per well; therefore, operators typically

1     store chemicals on-site in large volumes (typically 200 to 400 gal (760–1,500 L) totes), often in
2     multiple containers. The ten most common chemicals (excluding quartz) are methanol,
3     hydrotreated light petroleum distillates, hydrochloric acid, isopropanol, ethylene glycol,
4     peroxydisulfuric acid diammonium salt, sodium hydroxide, guar gum, glutaraldehyde, and
5     propargyl alcohol. These chemicals are present in multiple additives. Methanol was reported in
6     72% of the FracFocus disclosures, and hydrotreated light petroleum distillates and hydrochloric
7     acid were both reported in over half the disclosures (U.S. EPA, 2015b).

### 5.10.2. Factors Affecting the Frequency or Severity of Impacts

8     The potential for spills from the chemical mixing process to affect drinking water resources
9     depends on three factors: fluid characteristics, chemical management and spill characteristics, and
10    chemical fate and transport. Specific factors affecting the frequency and severity of impacts include
11    size and type of spill, volume of chemicals spilled, type of chemicals and their properties,
12    combinations of chemicals spilled, environmental conditions, proximity to drinking water
13    resources, employee training and experience, quality and maintenance of equipment, and spill
14    containment and mitigation.

15    The size and type of a fracturing operation, including the number of wellheads, the depth of the
16    well, the length of the horizontal leg, and the number of stages and phases, affect the likelihood and
17    potential impacts of spills. Larger operations may require larger volumes of chemicals, more
18    storage containers, more equipment, and additional transfers between different pieces of
19    equipment. Larger storage containers increase the maximum volume of a spill or leak from a
20    storage container, and additional transfers between equipment increase the possibility of human
21    error.

22    The type of chemical spilled governs how it will move and transform in the environment. More
23    mobile chemicals move faster through the environment, causing a quicker impact. More mobile
24    chemicals are also generally more soluble and may reach the drinking water resource at higher
25    concentrations. Less mobile chemicals will move more slowly, and may have delayed and longer-
26    term impacts, at lower concentrations. The severity of impact is also governed by how the chemical
27    adversely impacts water quality. Water quality impacts may range from aesthetic effects (e.g., taste,
28    smell) to adverse health effects.

29    The environmental conditions at and around the spill site affect the fate and transport of a given
30    chemical. These conditions include soil properties, climate, weather, and terrain. Permeable soils
31    may allow for rapid transport of the spilled fluid through the soil and into a nearby drinking water
32    resource. Precipitation can re-mobilize trapped chemicals and move them over land or through the
33    subsurface.

34    The proximity of a spill to drinking water resources affects the frequency and severity of impact.
35    The closer a spill is to a drinking water a resource, the higher potential to reach it. Also, as a fluid
36    moves toward a drinking water resource, it may decrease in concentration, which will affect the
37    severity of an impact. More concentrated chemicals have the potential to have a bigger impact on

1 water quality. The characteristics of the drinking water resource will also influence the magnitude
2 of the impact of a spill.

3 The most successful way to prevent impacts to drinking water resources is to prevent spills from
4 occurring in the first place and to quickly and effectively contain spills. Effective spill containment
5 and mitigation measures can prevent or reduce the frequency and severity of impacts. Spill
6 containment measures include well pad containment liners, diversion ditches, berms, dikes,
7 overflow prevention devices, drip pans, and secondary containers. These may prevent a spill from
8 reaching soil and water receptors. Spill mitigation, including removing contaminated soils,
9 vacuuming up spilled fluids, and using sorbent materials may limit the severity of a spill.
10 Implementation of these measures varies from site-to-site and may not always be effective.

### 5.10.3. Uncertainties

11 The lack of information regarding the composition of chemical additives and fracturing fluids,
12 containment and mitigation measures in use, and the fate and transport of spilled fluids greatly
13 limits our ability to assess potential impacts to drinking water resources.

14 There is no standard design for hydraulic fracturing fluids. Detailed information on the chemicals
15 used is limited, and volumes of chemicals stored on-site are generally not publicly available. These
16 limitations in data preclude the ability to know what volumes of chemicals may be spilled.
17 FracFocus, which currently holds the most comprehensive information on water and chemicals
18 used in hydraulic fracturing fluids, identifies well-specific chemicals and the concentration of those
19 chemicals as a maximum percentage of the mass of fracturing fluid. Accuracy and completeness of
20 original FracFocus disclosure information was not verified. In applying the EPA-standardized
21 chemical list to the ingredient records in the FracFocus database, standardized chemical names
22 were assigned to only 65% of the ingredient records from the more than 36,000 unique, fully
23 parsed disclosures. The remaining ingredient records could not be assigned a standardized
24 chemical name and were excluded from analyses (U.S. EPA, 2015a).

25 Operators may specify certain ingredients as confidential business information (CBI) and not
26 disclose the chemical used. More than 70% of disclosures to FracFocus contained at least one CBI
27 chemical. Of disclosures with at least one CBI chemical, the average number of CBI chemicals was
28 five. Approximately 11% of all ingredients were reported to FracFocus as CBI (U.S. EPA, 2015a). No
29 data are available in FracFocus for any chemical listed as CBI. Therefore, FracFocus CBI chemicals
30 are not included in analyses of volume, physical properties, or any other analysis in this
31 assessment, although we were able to do limited physicochemical analysis of 19 CBI chemicals.

32 Of the 1,076 hydraulic fracturing fluid chemicals identified by the EPA, 623 did not have estimated
33 physicochemical properties reported in the EPI Suite™ database. Knowing the chemical properties
34 of a spilled fluid is essential to predicting how and where it will travel in the environment. Although
35 we can make some generalizations about the physicochemical properties of these chemicals and
36 how spilled chemicals may move in the environment, the distribution of properties could change if
37 we obtained data for all known fracturing fluid chemicals (as well as for those listed as CBI).

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1  In order to determine the potential impact of a spill, the physicochemical properties, the site-
2  specific environmental conditions, and proximity to drinking water resources must be known. This
3  information is generally lacking.

4  There is a lack of baseline surface water and ground water quality data. This lack of data limits our
5  ability to assess the relative change to water quality from a spill or attribute the presence of a
6  contaminant to a specific source.

7  In addition to limited information on chemical usage, we cannot complete a thorough assessment of
8  the potential impact of chemical spills due to limited information on actual spills. Data sources used
9  in the EPA's spills analysis do not cover all states with hydraulic fracturing activity. The available
10 data provide limited information on the types and volumes of chemicals spilled, spill causes,
11 containment and mitigation measures, and sources of spills. In addition, there is little available data
12 on impacts of spills, due to a lack of baseline data and incomplete documentation of follow-up
13 actions and testing.

14 In general, then, we are limited in our ability to fully assess potential impacts to drinking water
15 resources from chemical spills, based on available current information. To improve our
16 understanding we need: more information on the chemical composition of additives and fracturing
17 fluid; the physicochemical properties of chemicals used; baseline monitoring and field studies of
18 spilled chemicals; drinking water resources quality conditions before and after hydraulic fracturing
19 is performed; detailed site-specific environmental conditions; more information on the
20 containment and mitigation measures and their effectiveness; and the types and volumes of spills.

### 5.10.4. Conclusions

21 The chemical mixing stage of the hydraulic fracturing process has the potential to cause impacts to
22 drinking water resources by way of surface spills of chemicals and fracturing fluids. There are
23 documented chemical spills at fracturing sites, but a lack of available data limits our ability to
24 determine impacts. Potential impacts to drinking water resources are governed by the fluid
25 characteristics, chemical management and spill characteristics, and the fate and transport of spilled
26 chemicals through the environment.

**Text Box 5-16. Research Questions Revisited.**

*What is currently known about the frequency, severity, and causes of spills of hydraulic fracturing fluids and additives?*

- The frequency of on-site spills from hydraulic fracturing operations could be obtained for two states. Frequency estimates from data and literature ranged from 0.4 to 1.3 spills for every 100 wells hydraulically fractured in Pennsylvania and Colorado, respectively, and between 3.3 and 12.2 spills for every 100 wells installed in Pennsylvania (Rahm et al., 2015; U.S. EPA, 2015n; Brantley et al., 2014; Gradient, 2013).[1] These estimates include spills of hydraulic fracturing chemicals and fluids, and flowback and produced water reported in state databases. It is unknown whether these spill estimates are representative of national occurrences. Estimates of the frequency of on-site spills from hydraulic fracturing operations were unavailable for other areas. If the estimates are representative, the number of spills nationally could range from 100 to 3,700 spills annually, assuming 25,000 to 30,000 new wells are fractured per year.

- In an analysis of spills, EPA characterized volumes and causes of hydraulic fracturing-related spills identified from selected state and industry data sources. The spills occurred between January 2006 and April 2012 in 11 states and included 151 cases in which fracturing fluid or additives spilled on or near a well pad (U.S. EPA, 2015n). These cases were likely a subset of all chemical and fracturing fluid spills during the study's time period. The reported volume of chemicals or fracturing fluid spilled ranged from 5 gal to more than 19,000 gal (19 to 72,000 L), with a median volume of 420 gal (1,600 L) per spill. Spill causes included equipment failure, human error, failure of container integrity, and other causes (e.g., weather and vandalism). The most common cause was equipment failure. Specific causes of equipment failure included blowout preventer failure, corrosion, and failed valves. More than 30% of the chemical or fracturing fluid spills characterized by the EPA came from fluid storage units (e.g., tanks, totes, and trailers) (U.S. EPA, 2015n).

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[1] Spill frequency estimates are for a given number of wells over a given period of time. These are not annual estimates nor are they for over a lifetime of the wells.

***What are the identities and volumes of chemicals used in hydraulic fracturing fluids, and how might this composition vary at a given site and across the country?***

- In this assessment, we compiled a list of 1,076 chemicals used to formulate hydraulic fracturing fluids. These chemicals include acids, alcohols, aromatic hydrocarbons, bases, hydrocarbon mixtures, polysaccharides, and surfactants. This is a cumulative list over multiple wells and years. Operators used an median of 14 unique chemicals per well according to the EPA's analysis of disclosures to FracFocus (U.S. EPA, 2015a).

- Our analysis showed that chemical use varies and that no single chemical is used at all well sites across the country, although several chemicals are widely used. Methanol, hydrotreated light petroleum distillates, and hydrochloric acid were reported in 65% or more of FracFocus disclosures analyzed by the EPA (U.S. EPA, 2015a). The composition of hydraulic fracturing fluids varies by state, by well, and within the same service company and geologic formation. This variability likely results from several factors, including the geology of the formation, the availability and cost of different chemicals, and operator preference (U.S. EPA, 2015a).

- Estimates from the EPA's database developed from disclosures made to FracFocus suggest median volumes of individual chemicals injected per well ranged from a few gallons to thousands of gallons, with a median of 650 gal (2,500 L) per chemical per well (U.S. EPA, 2015b). If 14 unique chemicals are used per well, then an estimated 9,100 gal (34,000 L) of chemicals may be injected per well (U.S. EPA, 2015a).

***What are the chemical and physical properties of hydraulic fracturing chemical additives?***

- Measured or estimated physicochemical properties were obtained for 453 of the 1,076 chemicals reported in hydraulic fracturing fluids. The wide variety of chemicals results in a wide range of physicochemical properties.

- Many hydraulic fracturing chemicals fully dissolve in water, but the aqueous solubilities range from fully miscible to sparingly soluble.

- The octanol-water partition coefficient ranges from the highly hydrophilic to the highly hydrophobic. Many chemicals used in hydraulic fracturing fluid fall in the middle of this range, suggesting that they will divide equally between water and solid phase, so that they may move slower through the environment than those that associate more with water. More chemicals will associate strongly with soils and organic materials, suggesting the potential of these chemicals to be long-term contaminants if they are spilled.

- There are few hydraulic fracturing chemicals that are volatile. Most hydraulic fracturing chemicals will tend to remain in water as opposed to volatilizing to the air.

1 • The chemicals for which we know physicochemical properties are not necessarily the chemicals most
2 frequently reported as used in hydraulic fracturing activities. Of the 453 chemicals for which
3 physicochemical properties were available, 18 of the top 20 most mobile chemicals were included in 2%
4 or less of disclosures (U.S. EPA, 2015b). However, two more common, but highly mobile chemicals,
5 choline chloride and tetrakis (hydroxymethyl) phosphonium sulfate were reported in 14% and 11% of
6 disclosures, respectively. These two chemicals are thus relatively more common, and, if spilled, their
7 properties of high mobility means they would move quickly through the environment with the flow of
8 water.

9 ***If spills occur, how might hydraulic fracturing chemical additives contaminate drinking water***
10 ***resources?***

11 • When chemicals are spilled, there are several paths by which a chemical could contaminate drinking
12 water resources. The chemical could flow overland to nearby surface water, penetrate into the soil that
13 could travel laterally and impact surface waters, or infiltrate and contaminate the underlying ground
14 water.

15 • Of the 151 spills characterized by the EPA, fluids reached surface water in 13 (9%) and soil in 97 (64%)
16 of those cases. None of the spills reportedly reached ground water (U.S. EPA, 2015n), but it could take
17 several years for spilled fluids to infiltrate soil and leach into ground water. Thus, it may not be
18 immediately known whether a spill reaches ground water or not

19 • The timing of a potential impact varies, but it could occur quickly, be delayed, have a continual impact
20 over time, or occur much later. Which path the spill takes depends on different conditions, such as
21 distance to a water receptor, spill volume, soil characteristics, and the physicochemical properties of the
22 chemical.

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# Chapter 6

# Well Injection

# 6. Well Injection

## 6.1. Introduction

1    To conduct hydraulic fracturing, fluids (primarily water, mixed with the types of chemicals and
2    proppant described in Chapter 5) are injected into a well under high pressure.[1] These fluids flow
3    under pressure through the well (sometimes thousands of feet below the surface), then exit the
4    well and move into the formation, where they create fractures in the rock. This process is also
5    known as a fracture treatment or a type of stimulation.[2] The fractures, which typically extend
6    hundreds of feet laterally from the well, are designed to remain within the production zone to
7    access as much oil or gas as possible, while using no more water or chemicals than necessary to
8    complete the operation.[3]

9    Production wells are sited and designed primarily to optimize production of oil or gas, which
10   requires isolating water-bearing formations and those containing the hydrocarbons to be exploited
11   from each other. This isolation can also protect drinking water resources. Appropriately sited,
12   designed, constructed, and operated wells and hydraulic fracturing treatments can reduce the
13   potential for impacts to drinking water resources. However, problems with the well's components
14   or improperly sited, designed, or executed hydraulic fracturing operations (or combinations of
15   these) could lead to adverse effects on drinking water resources.

16   The well and the geologic environment in which it is located are a closely linked system, often
17   designed with multiple barriers (i.e., isolation afforded by the well's casing and cement and the
18   presence of multiple layers of subsurface rock) to prevent fluid movement between oil/gas zones
19   and drinking water resources. Therefore, in this chapter we discuss (1) the well (including its
20   construction and operation) and (2) features in the subsurface geologic formations that could
21   provide or have provided pathways for migration of fluids to drinking water resources. If present
22   and in combination with the existence of a fluid and a physical force that moves the fluid, these
23   pathways can lead to impacts on drinking water resources throughout the life of the well, including
24   during and after hydraulic fracturing.[4]

25   Fluids can move **via pathways adjacent to or through the production well** that are created in
26   response to the stresses exerted during hydraulic fracturing operations (see Section 6.2). While
27   wells are designed and constructed to isolate fluids and maximize the production of oil and gas,

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[1] A fluid is a substance that flows when exposed to an external pressure; fluids include both liquids and gases.
[2] In the oil and gas industry, "stimulation" has two meanings—it refers to (1) injecting fluids to clear the well or pore spaces near the well of drilling mud or other materials that create blockage and inhibit optimal production (i.e., matrix treatment) and (2) injecting fluid to fracture the rock to optimize the production of oil or gas. This chapter focuses on the latter.
[3] The "production zone" (sometimes referred to as the target zone) refers to the portion of a subsurface rock zone that contains oil or gas to be extracted (sometimes using hydraulic fracturing). "Producing formation" refers to the larger geologic unit in which the production zone occurs.
[4] The primary physical force that moves fluids within the subsurface is a difference in pressure. Fluids move from areas of higher pressure to areas of lower pressure when a pathway exists. Density-driven buoyancy may also serve as a driving force. See Section 6.3.

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1   inadequate construction or degradation of the casing or cement can allow fluid movement that can
2   change the quality of drinking water resources. Potential issues associated with wells may be
3   related to the following:

4   • Casing (e.g., faulty, inadequate, or degraded casing or other well components, as
5       influenced by the numbers of casings; the depths to which these casings are set;
6       compatibility with the geochemistry of intersected formations; the age of the well;
7       whether re-fracturing is performed; and other operational factors) and

8   • Cement (e.g., poor, inadequate, or degraded cement, as influenced by a lack of cement in
9       key intervals; poor-quality cement; improper or inadequate placement of cement; or
10      degradation of cement over time).

11  Fluid movement can also occur **via induced fractures and/or other features within subsurface**
12  **formations** (see Section 6.3). While the hydraulic fracturing operation may be designed so that the
13  fractures will remain within the production zone, it is possible that, in the execution of the
14  hydraulic fracturing treatment, fractures can extend beyond their designed extent. Four scenarios
15  associated with induced fractures may contribute to fluid migration or communication between
16  zones:

17  • Flow of injected and/or displaced fluids through pore spaces in the rock formations out of
18      the production zone due to pressure differences and buoyancy effects.

19  • Fractures extending out of oil/gas formations into drinking water resources or zones that
20      are in communication with drinking water resources or fracturing into zones containing
21      drinking water resources.

22  • Fractures intersecting artificial structures, including abandoned or active (producing)
23      offset wells near the well that is being stimulated (i.e., well communication) or abandoned
24      or active mines.[1]

25  • Fractures intersecting geologic features that can act as conduits, such as existing
26      permeable faults and fractures.

27  In this chapter, we describe the conditions that can contribute to or cause the development of the
28  pathways listed above, the evidence for the existence of these pathways, and potential impacts or
29  impacts on drinking water resources associated with these pathways.

30  The interplay between the well and the subsurface features is complex, and sometimes it is not
31  possible to identify what specific element is contributing to or is the primary cause of an impact to
32  drinking water resources. For example, concerns have been raised regarding stray gas detected in
33  ground water in natural gas production areas (for additional information about stray gas, see
34  Sections 6.2.2 and 6.3.2.4).[2] Stray gas migration is a technically complex phenomenon, because

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[1] An abandoned well refers to a well that is no longer being used or cannot be used because of its poor condition.
[2] Stray gas refers to the phenomenon of natural gas (primarily methane) migrating into shallow drinking water resources or to the surface.

1  there are many potential naturally occurring or artificially created routes for migration of gas into
2  aquifers (including along production wells and via naturally existing or induced fractures), and it is
3  challenging to determine the source of the natural gas and whether the mobilization is related to oil
4  or gas production activities.

5  Furthermore, identifying cases where contamination of drinking water resources occurs due to oil
6  and gas production activities—including hydraulic fracturing operations—requires extensive
7  amounts of site and operational data, collected before and after hydraulic fracturing operations.
8  Where such data does exist and provides evidence of contamination, we present it in the following
9  sections. We do not attempt to predict which of these pathways is most likely to occur or to lead to
10  a drinking water impact, or the magnitude of an impact that might occur as a result of migration via
11  any single pathway, unless the information is available and documented based on collected data.

## 6.2. Fluid Migration Pathways Within and Along the Production Well

12  In this section, we discuss pathways for fluid movement along or through the production well used
13  in the hydraulic fracturing operation. While these pathways can form at any time within any well,
14  the repeated high pressure stresses exerted during hydraulic fracturing operations may make
15  maintaining integrity of the well more difficult (Council of Canadian Academies, 2014). In Section
16  6.2.1, we present the purpose of the various well components and typical well construction
17  configurations. Section 6.2.2 describes the pathways for fluid movement that can potentially
18  develop within the production well and wellbore and the conditions that lead to pathway
19  development, either as a result of the original design of the well, degradation over time or use, or
20  hydraulic fracturing operations.

21  While we discuss casing and cement separately, it is important to note that these are related—
22  inadequacies in one of these components can lead to stresses on the other. For example, flaws in
23  cement may expose the casing to corrosive fluids. Furthermore, casing and cement work together in
24  the subsurface to form a barrier to fluid movement, and it may not be possible to distinguish
25  whether integrity problems are related to the casing, the cement, or both. For additional
26  information on well design and construction, see Appendix D.

### 6.2.1. Overview of Well Construction

27  Production wells are constructed to convey hydrocarbon resources from the reservoirs in which
28  they are found to the surface and also to isolate fluid-bearing zones (containing oil, gas, or fresh
29  water) from each other. Multiple barriers are often present, and they act together to prevent both
30  horizontal movement (in or out of the well) and vertical movement (along the wellbore from deep
31  formations to drinking water resources). Proper design and construction of the casing, cement, and
32  other well components in the context of the location of drinking water resources and maintaining
33  mechanical integrity throughout the life of a well are necessary to prevent migration of fracturing
34  fluids, formation fluids, and hydrocarbons into drinking water resources.

1    A well is a multiple-component system that typically includes casing, cement, and a completion
2    assembly, and it may be drilled vertically, horizontally, or in a deviated orientation.[1] These
3    components work together to prevent unintended fluid movement into, out of, or along the well.
4    Due to the presence of multiple barriers within the well and the geologic system in which it is
5    placed, the existence of one pathway for fluid movement does not necessarily mean that an impact
6    to a drinking water resource has occurred or will occur.

7    Casing primarily acts as a barrier to lateral movement of fluids, and cement primarily acts as a
8    barrier to unintended vertical movement of fluids. Together, casing and cement are important in
9    preventing fluid movement into drinking water resources, and are the focus of this section.  Figure
10   6-1 illustrates the configurations of casing and cement that may occur in oil and gas production
11   wells, including the types of casing strings that may be present, the potential locations of cement,
12   and other features. The figure depicts an idealized representation of the components of a
13   production well; it is important to note that there is a wide variety in the design of hydraulically
14   fractured oil and gas wells in the United States (U.S. EPA, 2015o), and the descriptions in the figure
15   or in this chapter do not represent every possible well design.

### 6.2.1.1.  Casing

16   Casing is steel pipe that is placed into the drilled wellbore to maintain the stability of the wellbore,
17   to transport the hydrocarbons from the subsurface to the surface, and to prevent intrusion of other
18   fluids into the well and wellbore (Hyne, 2012; Renpu, 2011). A long continuous section of casing is
19   referred to as a casing string, which is composed of individual lengths of casing (known as casing
20   joints) that are threaded together using casing collars. In different sections of the well, multiple
21   concentric casing strings (of different diameters) can be used, depending on the construction of the
22   well.

23   The presence of multiple layers of casing strings can isolate and protect geologic zones containing
24   drinking water. In addition to conductor casing, which prevents the hole from collapsing during
25   drilling, one to three other types of casing may be also present in a well. The types of casing include
26   (from largest to smallest diameter) surface casing, intermediate casing, and production casing
27   (GWPC, 2014; Hyne, 2012; Renpu, 2011). One or more of any of these types of casing may be
28   present in a well. Surface casing often extends from the wellhead down to the base (bottom) of the
29   drinking water resource to be protected. Wells also may be constructed with liners, which are
30   anchored or suspended from inside the bottom of the previous casing string, rather than extending
31   all the way to the surface, and production tubing, which is used to transport the hydrocarbons to
32   the surface.

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[1] For the purposes of this assessment, a well's "orientation" refers to the direction in which the well is drilled, and "deviation" is used to indicate an orientation that is neither strictly vertical nor strictly horizontal. However, in industry usage, "deviation" is also used as a generic term to indicate well orientation.

Wells used in hydraulic fracturing operations are constructed with multiple layers of casing and cement to isolate fluid-bearing zones from each other. While there is no "typical well," some of the more common elements are presented in this figure. For additional information on the design and construction of wells, see Appendix D.

Wells are constructed with multiple layers of casing that maintain wellbore stability, transport hydrocarbons to the surface, and prevent intrusion of fluids into the well and **wellbore (1)**.

**Surface casing (2)** extends from the **wellhead (3)** down to or below the base (bottom) of the **ground water resource (4)** to be protected. The surface casing is cemented from its base to the surface to isolate the ground water resource and prevent fluid movement. (Wells may also have a **conductor casing** (not shown) to prevent unconsolidated material from collapsing into the wellbore.) The **cement shoe (5)** controls the placement of cement and prevents it from flowing back into the casing after it has been placed.

When used, **intermediate casing (6)** can reduce pressure on weak formations or allow better control of over-pressured formations, and it extends from the surface through the formation(s) of concern.

The **production casing (7)** extends to the end of the wellbore in the **production zone (8)**—which, in many cases, is thousands of feet below the surface—and is cemented in place. In some wells, a production liner is used in place of production casing. The production liner is hung from the next largest casing string by a **hanger (9)** that is attached to the casing and typically is cemented in place from the surface. **Tubing (10)**, when used, conveys hydrocarbons to the surface; it is installed after hydraulic fracturing operations and is not cemented in place. If the well has an **open hole completion** (not shown), the production casing extends just into the production zone and the entire length of the wellbore through the production zone is uncased.

At the bottom of the production casing, a **packer (11)**, a mechanical sealing device, may be set at the lower end of the tubing to create and seal off the **annulus (12)**, the space between the tubing and casing (or between two casing strings), and to keep fluid from migrating along the wellbore. **Perforations (13)** may be made through the casing and cement, using small explosive charges; fracturing is conducted through the perforations. Fracturing fluids are pumped into the well through the tubing, if present, or through the production casing (if the well has an open hole completion).

**Cement (14)** isolates fluid-containing formations from each other, protects the casing from exposure to formation fluids, and adds strength to the casing.

Note: Figure not to scale; other configurations are possible.

**Figure 6-1. Overview of well construction.**

1   Among the wells represented by the Well File Review (see Text Box 6-1), between one and three
2   casing strings were present (the Well File Review did not evaluate conductor casings). A
3   combination of surface and production casings was most often reported, followed by a combination
4   of surface, intermediate, and production strings. All of the production wells used in hydraulic
5   fracturing operations in the Well File Review had surface casing, while approximately 39% of the
6   wells (an estimated 9,100 wells) had intermediate casing, and 94% (an estimated 21,900 wells) had
7   production casing (U.S. EPA, 2015o).

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**Text Box 6-1. The Well File Review.**

8   The EPA conducted a survey of onshore oil and gas production wells that were hydraulically fractured by nine
9   oil and gas service companies in the continental United States between approximately September 2009 and
10  September 2010. The *Review of Well Operator Files for Hydraulically Fractured Oil and Gas Production Wells:*
11  *Well Design and Construction* (U.S. EPA, 2015o), referred to as the Well File Review, presents the results of the
12  survey and describes, for these wells: well design and construction characteristics, the relationship of well
13  design and construction characteristics to drinking water resources, and the number and relative location of
14  well construction barriers (i.e., casing and cement) that can block pathways for potential subsurface fluid
15  movement.

16  The results of the survey are based on information provided by well operators for a statistically
17  representative sample of 323 hydraulically fractured oil and gas production wells. The EPA did not attempt to
18  independently and systematically verify data supplied by well operators. Consequently, results from analyses
19  based on these data are of the same quality as the supplied data.

20  Results of the survey are presented as rounded estimates of the frequency of occurrence of hydraulically
21  fractured production well design or construction characteristics with 95% confidence intervals. The results
22  are statistically representative of an estimated 23,200 (95% confidence interval: 21,400-25,000) onshore oil
23  and gas production wells hydraulically fractured in 2009 and 2010 by the nine service companies.

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24  Hydraulic fracturing operations impose a variety of stresses on the well components. The casing
25  should be designed with sufficient strength to withstand the stresses it will encounter during the
26  installation, cementing, fracturing, production, and postproduction phases of the life of the well.
27  These stresses, illustrated in Figure 6-2, include burst pressure (the interior pipe pressure that will
28  cause the casing to burst), collapse pressure (the pressure applied to the outside of the casing that
29  will cause it to collapse), tensile stress (the stress related to stretching exerted by the weight of the
30  casing or tubing being raised or lowered in the hole), compression and bending (the stresses that
31  result from pushing along the axis of the casing or bending the casing), and cyclical stress (the
32  stress caused by frequent or rapid changes in temperature or pressure). Casing strength can be
33  increased by using high-strength alloys or by increasing the thickness of the casing. In addition, the
34  casing must be resistant to corrosion from contact with the formations and any fluids that might be
35  transported through the casing, including hydraulic fracturing fluids, brines, and oil or gas.

Note: Figure not to scale

Stresses Exerted on Well Casings
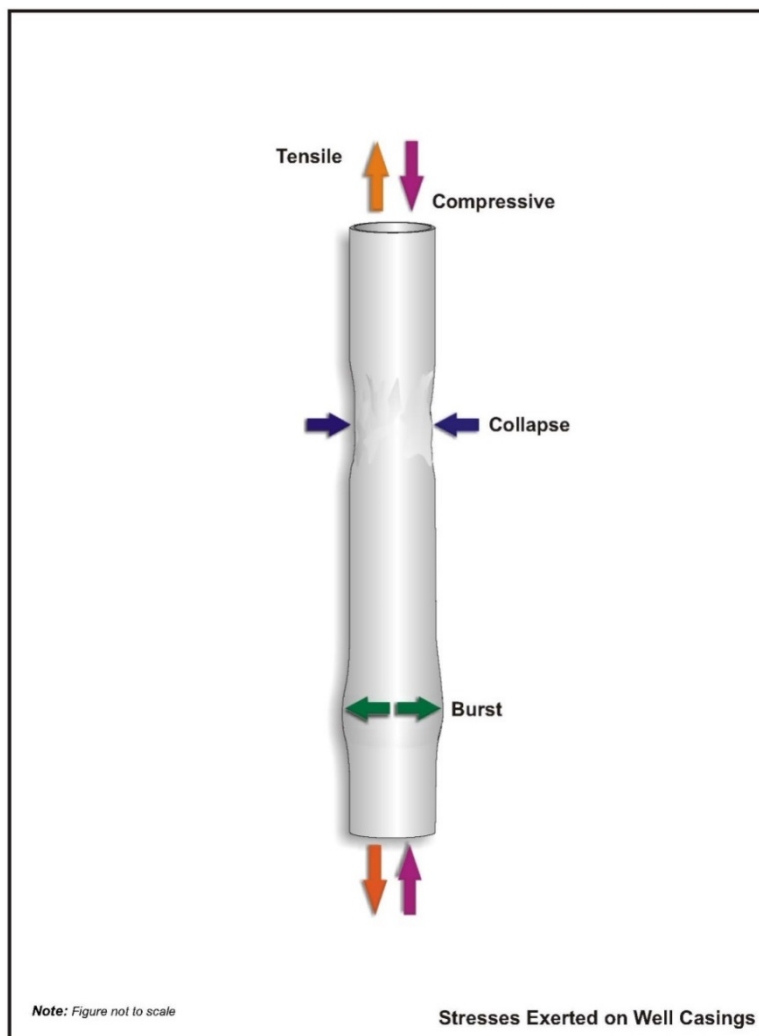
**Figure 6-2. The various stresses to which the casing will be exposed.**

> In addition to the stresses illustrated, the casing will be subjected to bending and cyclical stresses.
> Source: U.S. EPA (2012c).

### 6.2.1.2. Cement

1  Cement is one of the most important components of a well for providing zonal isolation and
2  reducing impacts on drinking water. Cement isolates fluid-containing formations from each other,
3  protects the casing from exposure to formation fluids, and provides additional strength to the
4  casing. The strength of the cement and its compatibility with the formations and fluids encountered
5  are important for maintaining well integrity through the life of the well.

6  The cement does not always need to be continuous along the entire length of the well in order to
7  protect drinking water resources; rather, protection of drinking water resources depends on a good
8  cement seal across the appropriate subsurface zones, including all fresh water- and hydrocarbon-
9  bearing zones. One study in the Gulf of Mexico found that there was no breakdown in isolation

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1  between geologic zones with pressure differentials as high as 14,000 psi as long as there was at
2  least 50 ft (15 m) of high-quality cement between the zones ([King and King, 2013](#)).

3  Most wells have cement behind the surface casing, which is a key barrier to contamination of
4  drinking water resources. The surface casings in nearly all of the wells used in hydraulic fracturing
5  operations represented in the Well File Review (93% of the wells, or an estimated 21,500 wells)
6  were fully cemented.[1] None of the wells studied in the Well File Review had completely uncemented
7  surface casings.

8  The length and location of cement behind intermediate and production casings can vary based on
9  the presence and locations of over-pressured formations, formations containing fluids, or
10  geologically weak formations (i.e., those that are prone to structural failure when exposed to
11  changes in subsurface stresses). State regulations and economics also play a role; 25 out of 27 oil
12  and gas producing states surveyed by the Ground Water Protection Council require some minimum
13  amount of cementing on the production casing above the producing zone ([GWPC, 2014](#)). In general,
14  the intermediate casings of the wells studied in the Well File Review were fully cemented; while
15  among production casings, about half were partially cemented, about a third were fully cemented,
16  and the remainder were either uncemented or their cementing status was undetermined. Among
17  the approximately 9,100 wells represented in the Well File Review that are estimated to have
18  intermediate casing, the intermediate casing was fully cemented in an estimated approximately
19  7,300 wells (80%) and partially cemented in an estimated 1,700 wells (19%). Production casings
20  were partially cemented in 47% of the wells, or approximately 10,900 wells ([U.S. EPA, 2015o](#)).

21  The Well File Review also estimated the number of wells with a continuous cement sheath along the
22  outside of the well. An estimated 6,800 of the wells represented in the study (29%) had cement
23  from the bottom of the well to the ground surface, and approximately 15,300 wells (66%) had one
24  or more uncemented intervals between the bottom of the well and the ground surface. In the
25  remaining wells, the location of the top of the cement was uncertain, so no determination could be
26  made regarding whether the well had a continuous cement sheath along the outside of the well
27  ([U.S. EPA, 2015o](#)).

28  A variety of methods are available for placing the cement, evaluating the adequacy of the cementing
29  process and the resulting cement job, and repairing any identified deficiencies. Cement is most
30  commonly emplaced by pumping the cement down the inside of the casing to the bottom of the
31  wellbore and then up the space between the outside of the casing and the formation (or the next
32  largest casing string). This method is referred to as the primary cement job and can be performed
33  as a continuous event in a single stage (i.e., "continuous cementing") or in multiple stages (i.e.,
34  "staged cementing"). Staged cementing may be used when, for example, the estimated weight and

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[1] The Well File Review defined fully cemented casings as casings that had a continuous cement sheath from the bottom of the casing to at least the next larger and overlying casing (or the ground surface, if surface casing). Partially cemented casings were defined as casings that had some portion of the casing that was cemented from the bottom of the casing to at least the next larger and overlying casing (or ground surface), but were not fully cemented. Casings with no cement anywhere along the casing, from the bottom of the casing to at least the next larger and overlying casing (or ground surface), were defined as uncemented.

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1    pressure associated with standard cement placement could damage weak zones in the formation
2    (Crook, 2008).

3    Deficiencies in the cementing process can result from poor centering or lost cement.[1] Poor
4    centering of the casing within the wellbore can cause uneven cement placement around the
5    wellbore, leading to the formation of thin weak spots that are prone to creation of uncemented
6    channels around the casing and loss of integrity (Kirksey, 2013). If any deficiencies or defects in the
7    primary cement job are identified, remedial cementing may be performed.

8    In over 90% of wells studied in the Well File Review, the casing strings were cemented using
9    primary cement methods. Secondary or remedial cementing was used on an estimated
10   4,500 casings (8%) most often on surface and production casings and less often on intermediate
11   casings. The remedial cementing techniques employed in these wells included cement squeezes,
12   cement baskets, and pumping cement down the annulus (U.S. EPA, 2015o). See Appendix D for
13   more information on remedial cementing techniques.

14   A variety of logs are available to evaluate the quality of cement behind the well casing. Among wells
15   in the Well File Review, the most common type of cement evaluation log run was a standard
16   acoustic cement bond log (U.S. EPA, 2015o). Standard acoustic cement bond logs are used to
17   evaluate both the extent of the cement placed along the casing and the cement bond between the
18   cement, casing, and wellbore.[2] Cement bond indices calculated from standard acoustic cement bond
19   logs on the wells in the Well File Review showed a median bond index of 0.7 just above the
20   hydraulic fracturing zone; this value decreased to 0.4 over a measured distance of 5,000 ft (1,524
21   m) above the hydraulic fracturing zone (U.S. EPA, 2015o). While standard acoustic cement bond
22   logs can give an average estimate of bonding, they cannot alone indicate zonal isolation, because
23   they may not be properly run or calibrated (Boyd et al., 2006; Smolen, 2006). One study of 28 wells
24   found that cement bond logs failed to predict communication between formations 11% of the time
25   (Boyd et al., 2006). In addition, they cannot discriminate between full circumferential cement
26   coverage by weaker cement and lack of circumferential coverage by stronger cement (King and
27   King, 2013; Smolen, 2006). A few studies have compared cement bond indices to zonal isolation,
28   with varying results. For example, Brown et al. (1970) showed that among 16 South American wells
29   with varying casing size and cement bond indices, a cemented 5.5 in (14 cm) diameter casing with a
30   bond index of 0.8 along as little as five feet can act as an effective seal. The authors also suggest that
31   an effective seal in wells having calculated bond indices differing from 0.8 are expected to have an
32   inverse relationship between bond index and requisite length of cemented interval, with longer
33   lengths needed along casing having a lower bond index. Another study recommends that wells
34   undergoing hydraulic fracturing should have a given cement bond over an interval that is three
35   times the length that would otherwise be considered adequate for zonal isolation (Fitzgerald et al.,

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[1] Lost cement refers to a failure of the cement or the spacer fluid used to wash the drilling fluid out of the wellbore to be circulated back to the surface, indicating that the cement has escaped into the formation.
[2] Cement bond logs are used to calculate a bond index, which varies between 0 and 1, with 1 representing the strongest bond and 0 representing the weakest bond.

1  1985). Conversely, King and King (2013) concluded that field tests from wells studied by Flournoy
2  and Feaster (1963) had effective isolation when the cement bond index ranged from 0.31 to 0.75.

### 6.2.1.3. Well Orientation

3  A well can be drilled and constructed with any of several different orientations: vertical, horizontal,
4  and deviated. The well's orientation can be important, because it affects the difficulty of drilling,
5  constructing, and cementing the well. In particular, as described below, constructing and cementing
6  horizontal wells present unique challenges (Sabins, 1990). In a vertical well, the wellbore is vertical
7  throughout its entire length, from the wellhead at the surface to the production zone. Deviated
8  wells are drilled vertically but are designed to deviate from the vertical direction at some point
9  such that the bottom of the well is at a significant lateral distance away from the point in the
10  subsurface directly under the wellhead. In a horizontal well, the well is drilled vertically to a point
11  known as the kickoff point, where the well turns toward the horizontal, extending into and parallel
12  with the approximately horizontal targeted producing formation (see  Figure 6-1).

13  The use of horizontal wells, particularly in unconventional reservoirs, is increasing (DrillingInfo,
14  2014b; Valko, 2009). Among wells evaluated in the Well File Review (i.e., over the period of
15  September 2009 to September 2010), about 66% were vertical, 11% were horizontal, and about
16  23% were deviated wells (U.S. EPA, 2015o).[1] This is generally consistent with information available
17  in industry databases—of the approximately 16,000 oil and gas wells used in hydraulic fracturing
18  operations in 2009 (one of the years for which the data for the Well File Review were collected),
19  39% were vertical, 33% were horizontal, and 28% were either deviated or the orientation was
20  unknown (DrillingInfo, 2014b). Note that among natural gas wells used in hydraulic fracturing
21  operations, 49.5% of wells in the DrillingInfo database were horizontal. The use of horizontal wells
22  in hydraulic fracturing operations has also been steadily increasing; in 2012, 63.7% of all wells used
23  in hydraulic fracturing operations were horizontal, compared to just 4% in 2003 (DrillingInfo,
24  2014b). See Figure 2-16 for a map presenting the locations of horizontal wells in the United States.

### 6.2.1.4. Well Completion

25  Another important aspect of well construction is how the well is completed into the production
26  zone, because the well's completion is part of the system of barriers and must be intact to provide a
27  fully functioning system.[2] A variety of completion configurations are available. The most common
28  configuration is for casing to extend to the end of the wellbore and be cemented in place (U.S. EPA,
29  2015o; George et al., 2011; Renpu, 2011). Before hydraulic fracturing begins, perforations are made
30  through the casing and cement into the production zone. It is through the perforated casing and
31  cement that hydraulic fracturing is conducted. In some cases, a smaller temporary casing, known as
32  a frac string, is inserted inside the production casing to protect it from the high pressures imposed
33  during hydraulic fracturing operations. Another method of completion is an open hole completion,

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[1] The Well File Review considered any non-horizontal well in which the well bottom was located more than 500 ft (152 m) laterally from the wellhead as being deviated.
[2] Completion is a term used to describe the assembly of equipment at the bottom of the well that is needed to enable production from an oil or gas well. It can also refer to the activities and methods (including hydraulic fracturing) used to prepare a well for production following drilling.

1  where the production casing extends into the production zone and the entire length of the drilled
2  horizontal wellbore through the production zone is left uncased. With open hole completions, the
3  entire production zone can either be fractured all at once in a single stage, or in stages using a
4  casing string set on formation packers, which separate the annulus into stages.[1] A special casing
5  with retractable sleeves is used to fracture each stage separately. Among wells represented in the
6  Well File Review, an estimated 6% of wells (1,500 wells) had open hole completions, 6% of wells
7  (1,500 wells) used formation packers, and the rest were cased and cemented (U.S. EPA, 2015o).

8  In some cases, wells may be recompleted after the initial construction, with re-fracturing if
9  production has decreased (Vincent, 2011). Recompletion also may include additional perforations
10  in the well at a different interval to produce from a new formation, lengthening the wellbore, or
11  drilling new laterals from an existing wellbore.

### 6.2.2.  Evidence of the Existence of Fluid Movement Pathways or of Fluid Movement

12  The following sections describe the pathways for fluid movement that can develop within the
13  production well and wellbore. We also describe the conditions that lead to the development of fluid
14  movement pathways and, where available, evidence that a pathway has allowed fluid movement to
15  occur within the casing or cement, and—in the case of sustained casing pressure (see Section
16  6.2.3)—a combination of factors within the casing and cement. (See Figure 6-3 for an illustration of
17  potential fluid movement pathways related to casing and cement.) We describe available
18  information regarding the rate at which these pathways have been identified in hydraulic fracturing
19  wells or, where such information does not exist, present the results of research on oil and gas
20  production wells in general or on injection wells.[2] Insufficient publicly accessible information exists
21  to determine whether wells intended for hydraulic fracturing are constructed differently from
22  production wells where no fracturing is conducted. However, given the applicability of well
23  construction technology to address the subsurface conditions encountered in hydraulic fracturing
24  operations and production or injection operations in general, this information is considered
25  relevant to the research questions (see Section 6.4).

26  While new wells can be specifically designed to withstand the stresses associated with hydraulic
27  fracturing operations, older wells, which are sometimes used in hydraulic fracturing operations,
28  may not have been designed to the same specifications. Where older wells were not designed
29  specifically to withstand the stresses associated with hydraulic fracturing, their reuse for this
30  purpose could be a concern. Frac strings, which are specialized pieces of casing inserted inside the
31  production casing, may be used to protect older casing during fracturing. However, the effect of
32  fracturing on the cement on the production casing in older wells is unknown. One study on re-
33  fracturing of wells noted that the mechanical integrity of the well was a key factor in determining
34  the success or failure of the fracturing (Vincent, 2011). An estimated 6% of wells (1,400 wells)

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[1] A formation packer is a specialized casing part that has the same inner diameter as the casing but whose outer diameter expands to make contact with the formation and seal the annulus between the casing and formation, preventing migration of fluids.
[2] An injection well is a well into which fluids are being injected (40 CFR 144.3).

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1   represented in the Well File Review were more than 10 years old before they were fractured
2   between 2009 and 2010 (U.S. EPA, 2015o).
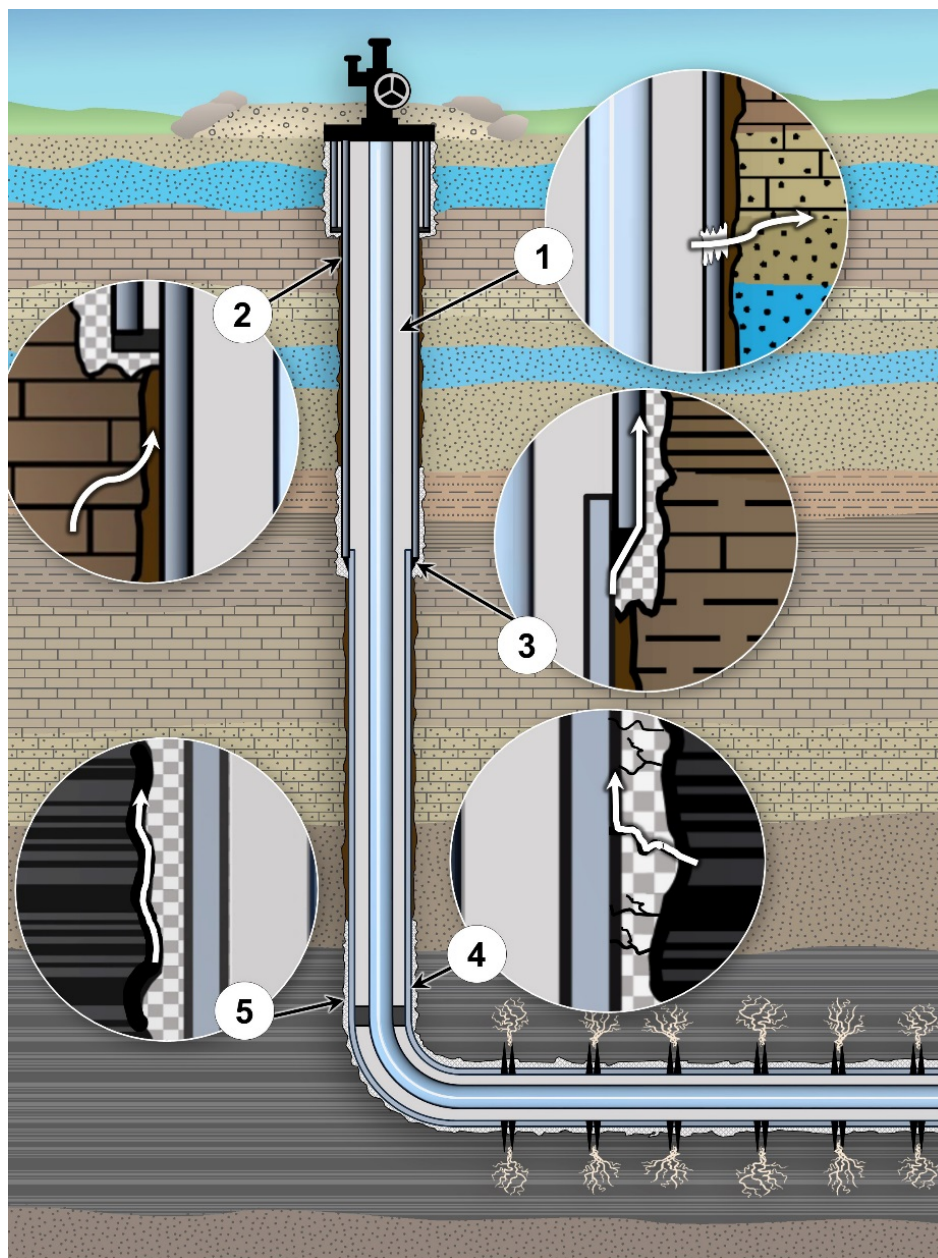


**Figure 6-3. Potential pathways for fluid movement in a cemented wellbore.**

> These pathways include: (1) casing/tubing leak into a permeable formation, (2) migration along an uncemented annulus, (3) migration along microannuli between the casing and cement, (4) migration through poor cement, or (5) migration along microannuli between the cement and formation. Note: the figure is not to scale and is intended to provide a conceptual illustration of pathways that may develop within the well.

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1  Note that there are also potential issues related to *where* older wells are sited. For example, some
2  wells may be in areas with naturally occurring subsurface faults or fractures that could not be
3  detected or fully characterized with the technologies available at the time of construction. See
4  Section 6.3.2.

### 6.2.2.1. Pathways Related to Well Casing

5  High pressures associated with hydraulic fracturing operations can damage the casing and lead to
6  fluid movement that can change the quality of drinking water resources. The casing string through
7  which fracturing fluids are injected is subject to higher pressures during fracturing operations than
8  during other phases in the life of a production well. To withstand the stresses created by the high
9  pressure of hydraulic fracturing, the well and its components must have adequate strength and
10  elasticity. If the casing is compromised or is otherwise not strong enough to withstand these
11  stresses (see Figure 6-2), a casing failure may result. If undetected or not repaired, casing failures
12  will serve as pathways for fracturing fluids to leak out of the casing.[1] Below we present indicators
13  that pathways along the casing are present or allowing fluid movement.

14  Fracturing fluids or fluids naturally present in the subsurface could flow into other zones in the
15  subsurface if inadequate or no cement is present and the pressure in the casing is greater than the
16  formation pressure. As we describe below, pathways for fluid movement associated with well
17  casing may be related to the original design or construction of the well, degradation of the casing
18  over time, or problems that can arise through extended use as the casing succumbs to stresses.

19  Casing failure may also occur if the wellbore passes through a structurally weak geologic zone that
20  fails and deforms the well casing. Such failures are common when drilling through zones containing
21  salt (Renpu, 2011). This type of well damage may also be possible if hydraulic fracturing causes
22  stress failure along a fault. Investigation of a well following a seismic event in England that was
23  attributed to hydraulic fracturing found that the casing had been deformed by the stress from the
24  formation (De Pater and Baisch, 2011). While it is not known if the casing deformation occurred
25  before or after the seismic event, such damage is possible if mechanically weak formations are
26  present. The changes in the pressure field in the portions of the formation near the wellbore during
27  hydraulic fracturing can also cause mechanically weak formations to fail, potentially damaging the
28  well. Palmer et al. (2005) demonstrated through modeling that hydraulic fracturing within coal that
29  had a low unconfined compressive strength could cause shear failure of the coalbeds surrounding
30  the wellbore.

31  Corrosion in uncemented zones is the most common cause of casing failure. This can occur if
32  uncemented sections of the casing are exposed to corrosive substances such as brine or hydrogen
33  sulfide (Renpu, 2011). Corrosion commonly occurs at the collars that connect sections of casing or

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[1] Internal mechanical integrity tests (MITs), such as casing inspection logs or caliper logs, annulus pressure monitoring, and pressure testing, can provide early warning of a problem, such as a leak, within the casing. It is important to note that if a well fails an MIT, this does not mean there is a failure of the well or that drinking water resources are impacted. An MIT failure is a warning that something needs to be addressed, and a loss of integrity is an event that may result in fluid movement from the well if remediation is not performed.

1    at other places where other equipment is attached to the casing. Corrosion at collars may
2    exacerbate problems with loose or poorly designed connections, which are another common cause
3    of casing leaks (King and King, 2013; Brufatto et al., 2003). Watson and Bachu (2009) found that
4    66% of all casing corrosion occurred in uncemented well sections, as shown in Pathway 1 of Figure
5    6-3.

6    Aging and use of the well contribute to casing degradation. Casing corrosion and degradation can
7    occur over time and with extended use. Also, exposure to corrosive chemicals such as hydrogen
8    sulfide, carbonic acid, and brines can accelerate corrosion. Therefore, the potential for fluid
9    migration related to compromised casing tends to be higher in older wells. Ajani and Kelkar (2012)
10   studied wells in Oklahoma and found a correlation between well age and integrity, specifically in
11   wells spaced between 1,000 and 2,000 ft (305 and 610 m) from a well being fractured, the
12   likelihood of impact to the well rose from approximately 20% to 60% as the well's age increased
13   from 200 days to over 600 days. Age was also found to be a factor in well integrity problems in a
14   study of wells in the Gulf of Mexico (Brufatto et al., 2003). The studies mentioned did not look for
15   evidence of such fluid movement pathways, however. Because of this potential for degradation of
16   well components in older wells and the related potential for fluid movement, reentering older wells
17   for re-fracturing may contribute to the development of fluid movement pathways within those
18   wells.

19   As noted above, the casing and cement work together to strengthen the well and provide zonal
20   isolation. Uncemented casing does not necessarily lead to fluid migration. However, migration can
21   occur if the casing in an uncemented zone fails during hydraulic fracturing operations or if the
22   uncemented section is in contact with fluid-containing zones (including the zone being fractured).
23   Sections of well casing that are uncemented may allow fluid migration into the annulus between the
24   casing and formation. Fluid is free to migrate between formations in contact with the uncemented
25   well section in any uncemented annulus without significant hole sloughing or wellbore swelling. If
26   the uncemented section extends through a drinking water resource, fluid migration into the
27   drinking water resource may occur.

28   Other well integrity problems have been found to vary with the well environment, particularly
29   environments with high pressures and temperatures. Wells in high pressure/high temperature
30   environments, wells with thermal cycling, and wells in corrosive environments can have life
31   expectancies of less than 10 years (King and King, 2013).

32   The depth of the surface casing relative to the base of the drinking water resource to be protected is
33   an important factor in protecting the drinking water resource. In a limited risk modeling study of
34   selected injection wells in the Williston Basin, Michie and Koch (1991) found the risk of aquifer
35   contamination from leaks from the inside of the well to the drinking water resource was 7 in
36   1,000,000 injection wells if the surface casing was set deep enough to cover the drinking water
37   resource, and that the risk increased to 6 in 1,000 wells if the surface casing was not set deeper
38   than the bottom of the drinking water resource. An example where surface casing did not extend
39   below drinking water resources comes from an investigation of 14 drinking water wells with
40   alleged water quality problems in the Wind River and Fort Union formations in Wyoming

1   (WYOGCC, 2014). The state found that the surface casing of oil and gas wells was shallower than the
2   depth of 3 of the 14 drinking water wells. Some of the oil and gas wells with shallow surface casing
3   had elevated gas pressures in their annuli (WYOGCC, 2014).

4   During hydraulic fracturing operations near Killdeer, in Dunn County, North Dakota, the
5   production, surface, and conductor casing of the Franchuk 44-20 SWH well ruptured, causing fluids
6   to spill to the surface (Jacob, 2011). The rupture occurred during the 5th of 23 stages of hydraulic
7   fracturing when the pressure spiked to over 8,390 psi (58 MPa). The casing failed, and ruptures
8   were found in two locations along the production casing—one just below the surface and one at
9   about 55 ft (17 m) below ground surface. The surface casing ruptured in three places down to a
10  depth of 188 ft (57 m), and the conductor casing ruptured in one place. Despite a shutdown of the
11  pumps, the pressure was still sufficient to cause fluid to travel through the ruptured casings and to
12  bubble up at the surface. Ultimately, nearly 168,000 gallons (636 m$^3$) of fluids and approximately
13  2,860 tons (2,595 metric tons) of contaminated soil were removed from the site (Jacob, 2011).
14  Sampling of two monitoring wells in the drinking water aquifer identified brine contamination that
15  was consistent with mixing of local ground water with brine from Madison Group formations,
16  which the well had penetrated (U.S. EPA, 2015j). Tert-butyl alcohol (TBA) was also found in the two
17  wells with brine contamination. The TBA was consistent with degradation of tert-butyl
18  hydroperoxide, a component of the fracturing fluid used in the Franchuk well. The rupture
19  (blowout) was the only source consistent with findings of high brine and TBA concentrations in the
20  two wells.[1] For additional information about impacts at the Killdeer site, see Text Box 5-12 in
21  Section 5.5 and Section 6.3.2.2.

22  Inadequate casing or cement can respond poorly when blowout preventers activate. When blowout
23  preventers are activated, they immediately stop the flow in the well, which can create a sudden
24  pressure increase in the well. If the casing or cement are not strong enough to withstand the
25  increased pressure when this occurs, well components can be damaged (The Royal Society and the
26  Royal Academy of Engineering, 2012) and the potential for fluid release and migration in the
27  subsurface increases. Blowouts can also occur during the production phase, and cause spills on the
28  surface that can affect drinking water resources; see Section 7.7.3.2.

29  While well construction and hydraulic fracturing techniques continue to change, the pressure- and
30  temperature-related stresses associated with hydraulic fracturing remain as factors that can affect
31  the integrity of the well casing. Several studies have evaluated the components of a well that can
32  affect well performance and integrity. Ingraffea et al. (2014) conducted a study of well integrity in
33  Pennsylvania production wells to assess overall trends in well integrity. This study identified a
34  significant increase in well integrity problems from 2009 to 2011 rising to 5% to 6% of all wells,
35  followed by a decrease beginning in 2012 to about 2% of all wells, a reduction of approximately 100

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[1] A well blowout is the uncontrolled flow of fluids out of a well. A blowout preventer (BOP) is casinghead equipment that
prevents the uncontrolled flow of oil, gas, and mud from the well by closing around the drill pipe or sealing the hole (Oil
and Gas Mineral Services, 2010). BOPs are typically a temporary component of the well, in place only during drilling and
perhaps through hydraulic fracturing operations.

1 violations among 3,000 wells. The rise in well integrity problems between 2009 and 2011 coincided
2 with an increase in the number of wells in unconventional reservoirs.

3 Emerging isotopic techniques can be used to identify the extent to which stray gas occurring in
4 drinking water resources is linked to casing failure (see Text Box 6-2 for more information on stray
5 gas). Darrah et al. (2014) used hydrocarbon and noble gas isotope data to investigate the source of
6 gas in eight identified "contamination clusters." Seven of these clusters were stripped of
7 atmospheric gases ($Ar^{36}$ and $Ne^{20}$) and were enriched in crustal gases, indicating the gas migrated
8 quickly from depth without equilibrating with intervening formations. The rapid transport was
9 interpreted to mean that the migration did not occur along natural fractures or pathways, which
10 would have allowed equilibration to take place. Possible explanations for the rapid migration
11 include transport up the well and through a leaky casing (Pathway 1 in Figure 6-3) or along
12 uncemented or poorly cemented intervals from shallower depths (Pathways 2 through 5 in Figure
13 6-3). In four Marcellus Shale clusters, gas found in drinking water wells had isotopic signatures and
14 ratios of ethane to methane that were consistent with those in the producing formation. The
15 authors conclude that this suggests that gas migrated along poorly constructed wells from the
16 producing formation, likely with improper, faulty, or failing production casings. In three clusters,
17 the isotopic signatures and ethane to methane ratios were consistent with formations overlying the
18 Marcellus. The authors suggest that this migration occurred from the shallower gas formations
19 along uncemented or improperly cemented wellbores. In another Marcellus cluster in the study,
20 deep gas migration was linked to a subsurface well, likely from a failed well packer.

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**Text Box 6-2. Stray Gas Migration.**

21 Stray gas refers to the phenomenon of natural gas (primarily methane) migrating into shallow drinking water
22 resources, into water wells, or to the surface (e.g., cellars, streams, or springs). Stray gas in the wellhead of a
23 production well is an indicator of an active wellbore pathway. Methane is not a regulated drinking water
24 contaminant, but it can initiate chemical and biological reactions that release or mobilize other contaminants,
25 and gas can accumulate to explosive levels when allowed to exsolve (degas) from ground water in closed
26 environments. Stray gas may originate from conventional and unconventional natural gas reservoirs, as well
27 as from coal mines, landfills, leaking gas wells, leaking gas pipelines, and buried organic matter (Baldassare,
28 2011).

29 Detectable levels of dissolved hydrocarbons (generally methane and/or ethane) exist in most oxygen-poor
30 aquifers, even in the absence of human activity (Gorody, 2012). Pre-drilling studies show that low levels of
31 methane are frequently found in water wells in northern Pennsylvania and New York (Kappel, 2013; Kappel
32 and Nystrom, 2012); one USGS study detected methane in 80% of sampled wells in Pike County, Pennsylvania
33 (Senior, 2014). The origin of methane in ground water can be either thermogenic (produced by high
34 temperatures and pressures in deeper formations, such as the gas found in the Marcellus Shale) or biogenic
35 (produced in shallower formations by bacterial activity in anaerobic conditions). Occurrence of thermogenic
36 methane in shallower formations depends upon the existence of a hydrocarbon source and pathways by
37 which the gas can migrate. Interactions between hydrocarbon production and natural systems must also be
38 considered. For example, Brantley et al. (2014) describe how northern Pennsylvania's glacial history may
39 help explain why stray gas is more common there than in the southern part of the state.

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1   Stray gas migration can be a technically complex phenomenon, in part because there are many potential
2   routes for migration. The lack of detailed monitoring data, including a lack of baseline measurements prior to
3   hydrocarbon production, often further complicates stray gas research. Examining the concentrations and
4   isotopic compositions of methane and higher molecular weight hydrocarbons such as ethane and propane
5   can aid in determining the source of stray gas (Tilley and Muehlenbachs, 2012; Baldassare, 2011; Rowe and
6   Muehlenbachs, 1999). Isotopic composition and methane-to-ethane ratios can help determine whether the
7   gas is thermogenic or biogenic in origin and whether it is derived from shale or other formations (Gorody,
8   2012; Muehlenbachs et al., 2012; Barker and Fritz, 1981). Isotopic analysis can also be used to identify the
9   strata where the gas originated and provide evidence for migration mechanisms (Darrah et al., 2014).

10  However, determining the source of methane does not necessarily establish the migration pathway. Well
11  casing and cementing issues may be an important source of stray gas problems (Jackson et al., 2013b);
12  however, other potential subsurface pathways are also discussed in the literature. Multiple researchers (e.g.,
13  Jackson et al., 2013b; Molofsky et al., 2013; Révész et al., 2012; Osborn et al., 2011) have described biogenic
14  and/or thermogenic methane in ground water supplies in Marcellus gas production areas, although the
15  pathways of migration are generally not apparent. The Osborn et al. (2011) study found that thermogenic
16  methane concentrations in well water increased with proximity to Marcellus Shale production sites, while
17  Molofsky et al. (2013) found the presence of gas to be more closely correlated with topography and elevation.
18  Similarly, Siegel et al. (In Press) found no correlation between methane in ground water and proximity to
19  production wells.

20  The EPA conducted case studies that included investigating stray gas in northeastern Pennsylvania and the
21  Raton Basin of Colorado. In northeastern Pennsylvania, many drinking water wells within the study area
22  were found to have elevated methane concentrations. For some of the wells, the EPA concluded that the
23  methane (both thermogenic and biogenic) was naturally occurring background gas not attributable to gas
24  exploration activities. In others, it appeared that non-background methane had entered the water wells
25  following well drilling and hydraulic fracturing. In most cases, the methane in the wells likely originated from
26  intermediate formations between the production zone and the surface; however, in some cases, the methane
27  appears to have originated from deeper layers such as those where the Marcellus Shale is found (U.S. EPA,
28  2015o). The Raton Basin case study examined the Little Creek Field, where potentially explosive quantities of
29  methane were vented into a number of drinking water wells in 2007. The methane was found to be primarily
30  thermogenic in origin, modified by biologic oxidation (U.S. EPA, 2015l). Secondary biogeochemical changes
31  related to the migration and reaction of methane within the shallow drinking water aquifer were reflected in
32  the characteristics of the Little Creek Field ground water (U.S. EPA, 2015l).

33  While the sources of methane in the two studies could be determined with varying degrees of certainty,
34  attempts to definitively identify the pathways of migration have been inconclusive. In northeastern
35  Pennsylvania, while the sources could not be definitively determined, the Marcellus Shale could not be
36  excluded as a potential source in some wells based on isotopic signatures, methane-to-ethane ratios, and
37  isotope reversal properties (U.S. EPA, 2014i). The Pennsylvania

1    Department of Environmental Protection (PA DEP) cited at least two operators for failure to prevent gas
2    migration at a number of wells within the study area. Evidence cited by the state included isotopic
3    comparison of gas samples from drinking water wells, water bodies, and gas wells; inadequate cement jobs;
4    and sustained casing pressure (although, under Pennsylvania law, oil or gas operators can be cited if they
5    cannot disprove the contamination was caused by their well using pre-drilling samples). A separate study
6    (Ingraffea et al., 2014) showed that wells in this area had higher incidences of well integrity problems relative
7    to wells in other parts of Pennsylvania. In the Raton Basin, the source of methane was identified as the
8    Vermejo coalbeds. Based on modeling of the Raton Basin study area, the migration could be explained either
9    by migration along natural rock features in the area or by migration caused by fracturing around the wellbore
10   (U.S. EPA, 2015l). Because the gas production wells were shut in shortly after the incident began, the wells
11   could not be inspected to determine whether an integrity failure in the wellbore was a likely cause of the
12   migration.[1]

13   These two case studies illustrate the considerations involved with understanding stray gas migration and the
14   difficulty in determining sources and migration pathways. In order to more conclusively determine sources
15   and migration pathways, studies in which data are collected on well integrity and ground water methane
16   concentrations both before and after hydraulic fracturing operations, in addition to the kinds of data
17   summarized above, would be needed.

### 6.2.2.2. Pathways Related to Cement

18   Fluid movement may result from inadequate well design or construction (e.g., cement loss or other
19   problems that arise in cementing of unconventional wells) or degradation of the cement over time
20   (e.g., corrosion or the formation of microannuli), which may, if undetected and not repaired, cause
21   the cement to succumb to the stresses exerted during hydraulic fracturing.[2,3] The well cement must
22   be able to withstand the subsurface conditions and the stresses encountered during hydraulic
23   fracturing operations. In this section, we present indicators that pathways within the cement are
24   present or allowing fluid movement.

25   Uncemented zones can allow fluids or brines to move into drinking water resources. An improper
26   cement job can fail to maintain zonal isolation in several ways. The first is by poor cement
27   placement. If a fluid-containing zone is left uncemented, the open annulus between the formation
28   and casing can act as a pathway for migration of that fluid. Fluids can enter the wellbore along any
29   uncemented section of the wellbore if a sufficient pressure gradient is present. Once the fluids have
30   entered the wellbore, they can travel along the entire uncemented length of the wellbore, unless the
31   wellbore sloughs in around the casing.
32   Because of its low density, gas will migrate up the wellbore if an uncemented wellbore is exposed to
33   a gas-containing formation. Gas may then be able to enter other formations (including drinking

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[1] Shutting in a well refers to sealing off a well by either closing the valves at the wellhead, a downhole safety valve, or a blowout preventer.

[2] External MITs, such as temperature logs, noise logs, oxygen activation logs, and radioactive tracer logs, can indicate improper cementing or degradation of the cement.

[3] Microannuli are very small channels that form in the cement and that may serve as pathways for fluid migration to drinking water resources.

1     water resources) if the wellbore is uncemented and the pressure in the annulus is sufficient to force
2     fluid into the surrounding formation (Watson and Bachu, 2009). The rate at which the gas can move
3     will depend on the difference in pressure between the annulus and the formation (Wojtanowicz,
4     2008). Harrison (1985) also demonstrated that such uncemented annuli could result in the
5     migration of gas into overlying formations open to the annulus.

6     In several cases, poor or failed cement has been linked to stray gas migration (see Text Box 6-2). A
7     Canadian study found that uncemented portions of casing were the most significant contributors to
8     gas migration (Watson and Bachu, 2009). The same study also found that 57% of all casing leaks
9     occurred in uncemented segments. In the study by Darrah et al. (2014) (see Section 6.2.2.1) using
10    isotopic data, four clusters of gas contamination were linked to cement issues. In three clusters in
11    the Marcellus and one in the Barnett, gas found in drinking water wells had isotopic signatures
12    consistent with intermediate formations overlying the producing zone. This suggests that gas
13    migrated from the intermediate units along the well annulus, along uncemented portions of the
14    wellbore or through channels or microannuli. In a study in Utah, Heilweil et al. (2013) used a
15    stream-based methane monitoring method to identify a case of potential gas migration near a well
16    with defective casing or cement.

17    Cementing of the surface casing is the primary aspect of well construction that protects drinking
18    water resources. Most states require the surface casing to be set and cemented from the level of the
19    lowermost drinking water resource to the surface (GWPC, 2014). Most wells—including those used
20    in hydraulic fracturing operations—have such cementing in place. Among the wells represented in
21    the Well File Review, surface casing, which was found to be fully cemented in 93% of wells,
22    extended below the base (i.e., the bottom, deepest, or lowermost part) of the protected ground
23    water resource reported by well operators in an estimated 55% of wells (12,600 wells).[1] In an
24    additional 28% of wells (6,400 wells), the operator-reported protected ground water resources
25    were fully covered by the next cemented casing string. A portion of the annular space between
26    casing and the operator-reported protected ground water resources was uncemented in at least 3%
27    of wells (600 wells) (U.S. EPA, 2015o).[2] Improper placement of cement can lead to cement integrity
28    problems. For example, an improper cement job can be the result of loss of cement during
29    placement into a formation with high porosity or fractures, causing a lack of adequate cement
30    across a water- or brine-bearing zone. Additionally, failure to use cement that is compatible with
31    the anticipated subsurface conditions, failure to remove drilling fluids from the wellbore, and
32    improper centralization of the casing in the wellbore can all lead to the formation of channels (i.e.,
33    small connected voids) in the cement during the cementing process (McDaniel et al., 2014; Sabins,
34    1990). If the channels are small and isolated, they may not lead to fluid migration. However, if they

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[1] In the in the Well File Review, protected ground water resources were reported by well operators. For most wells represented in the Well File Review, protected ground water resources were identified by the well operators from state or federal authorization documents. Other data sources used by well operators included aquifer maps, data from offset production wells, open hole log interpretations done by operators, operator experience, online databases, and references to a general requirement by the oil and gas agency.

[2] The well files representing an estimated 8% of wells in the Well File Review did not have sufficient data to determine whether the operator-reported protected ground water resource was uncemented or cemented. In these cases, there was ambiguity either in the depth of the base or the top of the operator-reported protected ground water resource.

1   are long and connected, extending across multiple formations or connecting to other existing
2   channels or fractures, they can present a pathway for fluid migration. Figure 6-3 shows a variety of
3   pathways for fluid migration that are possible from failed cement jobs.

4   One example of how cement problems associated with hydraulic fracturing contributed to a
5   contamination incident occurred in Bainbridge, Ohio. This incident was particularly well studied by
6   the Ohio Department of Natural Resources (ODNR, 2008) and by an expert panel (Bair et al., 2010).
7   The level of detail available for this case is not typically found in studies of such events but was
8   collected because of the severity of the impacts and the resulting legal action. The English #1 well
9   was drilled to a depth of 3,900 ft (1,189 m) below ground surface (bgs) with the producing
10  formation located between 3,600 and 3,900 ft (1,097 and 1,189 m) bgs. Overlying the producing
11  formation were several uneconomic formations that contained over-pressured gas (i.e., gas at
12  pressures higher than the hydrostatic pressure exerted by the fluids within the well).[1] The original
13  cement design required the cement to be placed 700–800 ft (213–244 m) above the producing
14  formation to seal off these areas. During cementing, however, the spacer fluid failed to return to the
15  surface, indicating lost cement, and the cement did not reach the intended height.[2] Despite the lack
16  of sufficient cement, the operator proceeded with hydraulic fracturing.

17  During the fracturing operation, about 840 gallons (3.2 m³) of fluid flowed up the annulus and out
18  of the well. When the fluid began flowing out of the annulus, the operator immediately ceased
19  operations. About a month later, there was an explosion in a nearby house where methane had
20  entered from an abandoned and unplugged drinking water well that was connected to the cellar
21  (Bair et al., 2010). In addition to the explosion, the over-pressured gas entering the aquifer resulted
22  in the contamination of 26 private drinking water wells with methane.

23  Contamination at Bainbridge was the result of inadequate cement. The ODNR determined that
24  failure to cement the over-pressured gas formations, proceeding with the fracturing operation
25  without verifying there was adequate cement, and the extended period during which the well was
26  shut in all contributed to the contamination of the aquifer with stray gas (ODNR, 2008). Cement
27  logs found the cement top was at 3,640 ft (1,109 m) bgs, leaving the uneconomic gas-producing
28  formations and a portion of the production zone uncemented. Hydraulic fracturing fluids flowing
29  out of the annulus provided an indication that the fracturing had created a path from the producing
30  formation to the well annulus. This pathway may have allowed gas from the producing formation
31  along with gas from the uncemented formation to enter the annulus. Because the well was shut in,
32  the pressure in the annulus could not be relieved, and the gas eventually traveled through natural
33  fractures surrounding the wellbore into local drinking water aquifers. (During the time the well
34  was shut in, natural gas seeped into the well annulus and pressure built up from an initial pressure
35  of 90 psi (0.6 MPa) to 360 psi (2.5 MPa)). From the aquifer, the gas moved into drinking water wells
36  and from one of those wells into a cellar, resulting in the explosive accumulation of gas.

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[1] Hydrostatic pressure is the pressure exerted by a column of fluid at a given depth. Here, it refers to the pressure exerted by a column of drilling mud or cement on the formation at a particular depth.
[2] Spacer fluid is a fluid pumped before the cement to clean drilling mud out of the wellbore.

1   While limited literature is available on construction flaws in wells that have been used in hydraulic
2   fracturing operations, several studies have examined construction flaws in oil and gas wells in
3   general. One study that examined reported drinking water contamination incidents in Texas
4   identified 10 incidents related to drilling and construction activities among 250,000 oil and gas
5   wells (Kell, 2011). The study noted that many of the contamination incidents were associated with
6   wells that were constructed before Texas revised its regulations on cementing in 1969 (it is not
7   clear how old the wells were at the time the contamination occurred). Because this study relied on
8   reported incidents, it is possible that other wells exhibited integrity issues but did not result in
9   contamination of a drinking water well or were not reported. Therefore, this should be considered
10  a low-end estimate of the number of well integrity issues that could be tied directly to drilling and
11  construction activities. It is important to note that the 10 contamination incidents identified were
12  not associated with wells that were hydraulically fractured (Kell, 2011).

13  Several investigators have studied violations information from the PA DEP online violation
14  database to evaluate the rates of and possible factors contributing to well integrity problems,
15  including those related to cement. The results of these studies are summarized in Table 6-1. While
16  all of the studies shown in the table used the same database, their results vary, not only because of
17  the different time frames, but also because they used different definitions of what violations
18  constituted an integrity problem or failure. For example, Considine et al. (2012) considered all
19  events resulting in environmental damage—including effects such as erosion—and found a
20  relatively high violation rate. Davies et al. (2014) and Ingraffea et al. (2014) investigated violations
21  related to well integrity, while Vidic et al. (2013) looked only at well integrity violations that
22  resulted in fluid migration out of the wellbore; these more specific studies found relatively lower
23  violation rates. Olawoyin et al. (2013) performed a statistical analysis that weighted violations
24  based on risk and found that the most risky violations included those involving pits, erosion, waste
25  disposal, and blowout preventers.

**Table 6-1. Results of studies of PA DEP violations data that examined well failure rates.**

| Study | Violations investigated | Data timeframe | Key findings[a] |
|-------|------------------------|----------------|-----------------|

| Study | Violations investigated | Data timeframe | Key findings[a] |
|---|---|---|---|
| Considine et al. (2012) | Violations resulting in environmental damage (3,533 wells studied) | 2008–2011 | Of 845 environmental damage incidents (which resulted in 1,144 violations), approximately 10% were related to casing or cement problems. The overall violation rate dropped from 52.9% of all wells in 2008 to 20.8% of all wells in 2011. |
| Davies et al. (2014) | Failure of one of the barriers preventing fluid migration (8,030 wells studied) | 2005–2013 | Approximately 5% of wells received this type of violation. The incident rate increased to 6.3% when failures noted on forms, but not resulting in violations, were included. |
| Ingraffea et al. (2014) | Violations and inspection records indicating structural integrity loss (3,391 wells studied) | 2000–2012 | Wells in unconventional reservoirs experienced a rate of structural integrity loss of 6.2%, while the rate for conventional wells was 1%. |
| Vidic et al. (2013) | Failure of cement/casing that allowed fluid to leak out (6,466 wells studied) | 2008–2013 | Approximately 3.4% of wells received this type of violation. |
| Olawoyin et al. (2013) | All violations (2,001 wells studied) | 2008–2010 | Analysis of 2,601 violations from 65 operators based on weighted risks found that potentially risky violations increased 342% over the study period, while total violations increased 110%. |

[a] Note: While all of these studies used the same database, their results vary because they studied different time frames and used different definitions of what violations constituted an integrity problem or failure.

1    Because a significant portion of Pennsylvania's recent oil and gas activity is in the Marcellus Shale,
2    many of the wells in these studies were most likely used for hydraulic fracturing. For example,
3    Ingraffea et al. (2014) found that approximately 16% of the oil and gas wells drilled in the state
4    between 2000 and 2012 were completed in unconventional reservoirs, and nearly all of these wells
5    were used for hydraulic fracturing. Wells drilled in unconventional reservoirs experienced higher
6    rates of structural integrity loss than conventional wells drilled during the same time period
7    (Ingraffea et al., 2014). The authors did not compare rates of structural integrity loss in
8    conventional wells that were and were not used for hydraulic fracturing; they assumed that
9    unconventional wells were hydraulically fractured and conventional wells were not.

10    Violation rates resulting in environmental pollution among all wells dropped from 52.9% in 2008
11    to 20.8% in 2011 (Considine et al., 2012), and the drop may be due to a number of factors.
12    Violations related to failure of cement or other well components represented a minority of all well
13    violations. Of 845 events that caused environmental contamination, including but not limited to
14    contamination of drinking water resources, Considine et al. (2012) found that about 10% (85

1    events) were related to casing and cement problems. The rest of the incidents were related to site
2    restoration and spills (the violations noted are confined to those incidents that caused
3    environmental damage i.e., the analysis excluded construction flaws that did not have adverse
4    environmental effects).

5    Another source of information on contamination caused by wells is positive determination letters
6    (PDLs) issued by the PA DEP. PDLs are issued in response to a complaint when the state determines
7    that contamination did occur in proximity to oil and gas activities. The PDLs take into account the
8    impact, timing, well integrity, and formation permeability; however, liability is presumed for wells
9    within a given distance if the oil and gas operator cannot refute that they caused the contamination,
10   based on pre-drilling sampling (Brantley et al., 2014).[1] Brantley et al. (2014) examined these PDLs,
11   and concluded that approximately 20 unconventional gas wells impacted water supplies between
12   2008 and 2012; this equates to 0.1% to 1% of the 6,061 wells spudded between 2008 and 2012 (it
13   is unclear exactly how many PDLs are linked to an individual well). While these oil and gas wells
14   were linked to contamination of wells and springs, the mechanisms for the impacts (including
15   whether fluids may have been spilled at the surface or if there was a pathway through the well or
16   through the subsurface rock formation to the drinking water resource) were not described by
17   Brantley et al. (2014). We did not perform a full and independent review of the PDLs for this
18   assessment.

19   While the studies discussed above present possible explanations for higher violation incidences in
20   hydraulically fractured wells, it should be noted that other explanations that are not specific to
21   hydraulic fracturing are also possible. These could include different inspection protocols and
22   different formation types.

23   Cementing in horizontal wells, which are commonly hydraulically fractured, presents challenges
24   that can contribute to higher rates of integrity issues. The observation by Ingraffea et al. (2014) that
25   wells drilled in unconventional reservoirs experience higher rates of structural integrity loss than
26   conventional wells is supported by conclusions of Sabins (1990), who noted that horizontal wells
27   have more cementing problems because they are more difficult to center properly and can be
28   subject to settling of solids on the bottom of the wellbore. Cementing in horizontal wells presents
29   challenges that can contribute to higher rates of integrity issues.

30   Thermal and cyclic stresses caused by intermittent operation also may stress cement (King and
31   King, 2013; Ali et al., 2009). Increased pressures and cyclic stresses associated with hydraulic
32   fracturing operations can contribute to cement integrity losses and, if undetected, small integrity
33   problems can lead to larger ones. Temperature differences between the (typically warmer)
34   subsurface environment and the (typically cooler) injected fluids, followed by contact with the
35   (typically warmer) flowback water, can lead to contraction of the well materials (both casing and
36   cement), which introduces additional stresses. Similar temperature changes may occur when

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[1] Under Pennsylvania's Oil and Gas Act, operators of oil or gas wells are presumed liable if water supplies within 1,000 ft
(305 m) were impacted within 6 months of drilling, unless the claim is rebutted by the operator; this was expanded to
2,500 ft (762 m) and 12 months in 2012.

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1 multiple fracturing stages are performed. Because the casing and cement have different mechanical
2 properties, they may respond differently to these stress cycles and debond.

3 Several studies illustrate the effects of cyclic stresses. Dusseault et al. (2000) indicate that wells that
4 have undergone several cycles of thermal or pressure changes will almost always show some
5 debonding between cement and casing. Microannuli formed by this debonding can be conduits for
6 gas migration, in particular because the lighter density of gas provides a larger driving force for
7 migration through the microannuli than for heavier liquids.[1] One laboratory study indicated that
8 microannuli on the order of 0.01 in (0.3 mm) could increase effective cement permeability from
9 1 nD ($1 \times 10^{-21}$ m$^2$) in good quality cement up to 1 mD ($1 \times 10^{-15}$ m$^2$) (Bachu and Bennion, 2009).
10 This six-order magnitude increase in permeability shows that even small microannuli can
11 significantly increase the potential for flow through the cement. Typically, these microannuli form
12 at the interface between the casing and cement or between the cement and formation. Debonding
13 and formation of microannuli can occur through intermittent operation, pressure tests, and
14 workover operations (Dusseault et al., 2000).[2] While a small area of debonding may not lead to fluid
15 migration, the microannuli in the cement that result from the debonding can serve as initiation
16 points for fracture propagation if re-pressurized gas enters the microannulus (Dusseault et al.,
17 2000).

18 The Council of Canadian Academies (2014) found that the repetitive pressure surges that occur
19 during the fracturing process would make maintaining an intact cement seal more of a challenge in
20 wells that are hydraulically fractured. Wang and Dahi Taleghani (2014) performed a modeling
21 study that showed that hydraulic fracturing pressures could initiate annular cracks in cement.
22 Another study of well data indicated that cement failure rates are higher in intermediate casings
23 compared to other casings (McDaniel et al., 2014). The failures occurred after drilling and
24 completion of wells, and the authors surmised that the cement failures were most likely due to
25 cyclical pressure stresses caused by drilling. Theoretically, such cyclical pressure events could also
26 be experienced during multiple stage hydraulic fracturing. Mechanical stresses associated with well
27 operation or workovers and pressure tests also may lead to small cracks in the cement, which may
28 provide migration pathways for fluid.

29 Corrosion can lead to cement failure. Cement can fail to maintain integrity as a result of degradation
30 of the cement after the cement is set. Cement degradation can result from attack by corrosive brines
31 or chemicals such as sulfates, sulfides, and carbon dioxide that exist in formation fluids (Renpu,
32 2011). These chemicals can alter the chemical structure of the cement, resulting in increased
33 permeability or reduced strength and leading to loss of cement integrity over time. Additives or
34 specialty cements that can decrease cement susceptibility to specific chemical components are
35 available.

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[1] Microannuli can also form due to an inadequate cement job, e.g., poor mud removal or improper cement placement rate.
[2] A workover refers to any maintenance activity performed on a well that involves ceasing operations and removing the wellhead. Depending on the purpose of the workover and the tools used, workovers may induce pressure changes in the well.

### 6.2.2.3. Sustained Casing Pressure

1  An example of how the issues related to casing and cement discussed in the preceding sections can
2  work together and be difficult to differentiate can be seen in the case of sustained casing pressure.[1]
3  Sustained casing pressure is an indicator that pathways within the well related to the well's casing,
4  cement, or both allowed fluid movement to occur. Sustained casing pressure can result from casing
5  leaks, uncemented intervals, microannuli, or some combination of the three, which can be an
6  indication that a well has lost integrity. Sustained casing pressure can be observed when an annulus
7  (either the annulus between the tubing and production casing or between any two casings) is
8  exposed to a source of nearly continuous elevated pressure. Goodwin and Crook (1992) found that
9  sudden increases in sustained casing pressure occurred in wells that were exposed to high
10 temperatures and pressures. Subsequent logging of these wells showed that the high temperatures
11 and pressures led to shearing of the cement/casing interface and a total loss of the cement bond.

12 Sustained casing pressure occurs more frequently in older wells and horizontal or deviated wells.
13 One study found that sustained casing pressure becomes worse as a well ages. Sustained casing
14 pressure was found in less than 10% of wells that were less than a year old, but was present in up
15 to 50% of 15-year-old wells (Brufatto et al., 2003). While these wells may not have been
16 hydraulically fractured, the study demonstrates that older wells can exhibit more integrity
17 problems. Watson and Bachu (2009) found that a higher portion of deviated wells had sustained
18 casing pressure compared to vertical wells. Increased pressures, cyclic stresses (Syed and Cutler,
19 2010), and difficulty in cementing horizontal wells (Sabins, 1990) also may lead to increased
20 instances of sustained casing pressure in wells where hydraulic fracturing occurs (Muehlenbachs et
21 al., 2012; Rowe and Muehlenbachs, 1999).

22 Sustained casing pressure can be a concern for several reasons. If the pressures are allowed to build
23 up to above the burst pressure of the exterior casing or the collapse pressure of the interior casing,
24 the casing may fail. Increased pressure can also cause gas or liquids to enter lower-pressured
25 formations that are exposed to the annulus either through leaks or uncemented sections.
26 Laboratory experiments by Harrison (1985) demonstrated that over-pressurized gas in the annulus
27 could cause rapid movement of gas into drinking water resources if a permeable pathway exists
28 between the annulus and the ground water. Over-pressurization of the annulus can be avoided by
29 venting the annulus to the atmosphere.

30 In a few cases, sustained casing pressure in wells that have been hydraulically fractured may have
31 been linked to drinking water contamination, although it is challenging to definitively determine
32 the actual cause. In one study in northeastern Pennsylvania, hydrocarbon and isotope
33 concentrations were used to investigate stray gas migration into domestic drinking water (U.S. EPA,

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[1] Sustained casing pressure is pressure in any well annulus that is measurable at the wellhead and rebuilds after it is bled down, not caused solely by temperature fluctuations or imposed by the operator (Skjerven et al., 2011). If the pressure is relieved by venting natural gas from the annulus to the atmosphere, it will build up again once the annulus is closed (i.e., the pressure is sustained). The return of pressure indicates that there is a small leak in a casing or through uncemented or poorly cemented intervals that exposes the annulus to a pressured source of gas that is actively being used. It is possible to have pressure in more than one of the annuli.

1   [2014i](#)). While the composition of the gas in the water wells was consistent with that of the gas
2   found in nearby gas wells with high casing pressures, other possible sources of the gas could not be
3   ruled out. Several gas wells in the study area were cited by the PA DEP for having elevated casing
4   annulus pressures. In another example in Alberta, Canada, 14% of wells drilled since 1971 have
5   shown serious sustained casing flow, defined as more than 10,594 ft$^3$ (300 m$^3$)/day at pressures
6   higher than 0.48 psi/ft (11 kPa/m) times the surface casing depth ([Jackson and Dussealt, 2014](#)).
7   Another study in the same area found gas in nearby drinking water wells had a composition that
8   was consistent with biogenic methane mixing with methane from nearby coalbed methane and
9   deeper natural gas fields ([Tilley and Muehlenbachs, 2012](#)).

10  Adequate well design, detection (i.e., through annulus pressure monitoring), and repair of sustained
11  casing pressure reduce the potential for fluid movement. [Watson and Bachu (2009)](#) found that
12  regulations that required monitoring and repair of sustained casing vent flow or sustained casing
13  pressure had a positive effect on lowering leak rates. The authors also found that wells initially
14  designed for injection experienced sustained casing pressure less often than those that were
15  retrofitted ([Watson and Bachu, 2009](#)).

16  Another study in Mamm Creek, Colorado, obtained similar results. The Mamm Creek field is in an
17  area where lost cement and shallow, gas-containing formations are common. A number of wells in
18  the area have experienced sustained casing pressure, and methane has been found in several
19  drinking water wells along with seeps into local creeks and ponds. In one well, four pressured gas
20  zones were encountered during well drilling and there was a lost cement incident, which resulted
21  in the cement top being more than 4,000 ft (1,219 m) lower than originally intended. Due to high
22  measured bradenhead pressure (661 psi, or 4.6 MPa), cement remediation efforts were
23  implemented ([Crescent, 2011](#); [COGCC, 2004](#)). The operator of this well was later cited by the
24  Colorado Oil and Gas Conservation Commission (COGCC) for causing natural gas and benzene to
25  seep into a nearby creek. The proposed route of contamination was contaminants flowing up the
26  well annulus and then along a fault. The proposed contamination route appeared to be validated
27  because, once remedial cementing was performed on the well, methane and benzene levels in the
28  creek began to drop ([Science Based Solutions LLC, 2014](#)). In response to the incident, the state
29  instituted requirements to identify and cement above the top of the highest gas-containing
30  formation and to monitor casing pressures after cementing.

31  Not every well that shows positive pressure in the annulus poses a potential problem. Sustained
32  pressure is only a problem when it exceeds the ability of the wellbore to contain it or when it
33  indicates downhole communication problems ([TIPRO, 2012](#)). A variety of management options are
34  available for managing such pressure including venting, remedial cementing, and use of kill fluids in
35  the annulus ([TIPRO, 2012](#)).[1]

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[1] A kill fluid is a weighted fluid with a density that is sufficient to overcome the formation pressure and prevent fluids from flowing up the wellbore.

## 6.3. Fluid Migration Associated with Induced Fractures within Subsurface Formations

1  In this section, we discuss potential pathways for fluid movement associated with induced fractures
2  and subsurface geologic formations (outside of the well system described in Section 6.2). We
3  examine the potential for fluid migration into drinking water resources by evaluating the
4  development of migration pathways within subsurface formations, the flow of injected and
5  formation fluids, and important factors that affect these processes.[1]

6  Fluid movement requires both a physical conduit (e.g., the permeable matrix pore volume or a
7  fracture in the rock) and a driving force.[2] In subsurface rock formations, fluid movement is driven
8  by the existence of a hydraulic gradient depending on elevation and pressure, which is also
9  influenced by fluid density, composition, and temperature ([Pinder and Celia, 2006](#)). Pressure
10  differentials in the reservoir and density-driven fluid buoyancy are the key forces governing fluid
11  migration during and after hydraulic fracturing operations. Pressure differentials depend upon the
12  initial conditions within these formations and are directly influenced by pressures that are created
13  by injection or production regimes. Buoyancy depends on density differences among and between
14  gases and liquids, and it causes fluid migration when and where these density differences exist
15  along with a pathway ([Pinder and Gray, 2008](#)).

16  As hydraulic fracturing takes place, injected fluids leaving the well create fractures within the
17  production zone and enter the formation through the newly created fractures. Unintended fluid
18  migration may result from this fracturing process. Migration pathways to drinking water resources
19  could develop as a result of changes in the subsurface flow or pressure regime associated with
20  hydraulic fracturing; via fractures that extend beyond the intended formation or that intersect
21  existing natural faults or fractures; or via fractures that intersect offset wells or other artificial
22  structures ([Jackson et al., 2013c](#)). These subsurface pathways may facilitate the migration of fluids
23  by themselves or in conjunction with the well-based pathways described in Section 6.2. Fluids
24  potentially available for migration include both fluids that are injected into the well (including
25  leakoff) and formation fluids (including brine or natural gas).[3]

---

[1] A subsurface formation (also referred to as a "formation" throughout this section) is a mappable body of rock of distinctive rock type(s), including the rock's pore volume (i.e., the void space within a formation that fluid flow can occur, as opposed to the bulk volume which includes both pore and solid phase volume), with a unique stratigraphic position.
[2] Permeability (i.e., intrinsic or absolute permeability) of formations describes the ability of water to move through the formation matrix, and it depends on the rock's grain size and the connectedness of the void spaces between the grains. Where multiple phases of fluids exist in the pore space, the flow of fluids also depends on relative permeabilities.
[3] Leakoff is the fraction of the injected fluid that infiltrates into the formation and is not recovered during production ([Economides et al., 2007](#)).

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1    The potential for subsurface fluid migration into drinking water resources can be evaluated during
2    two different time periods ([Kim and Moridis, 2015](#)):

    1.  Following the initiation of fractures within the reservoir prior to any production, when the
        injected fluid, pressurizing the formation, flows through the fractures and the fractures
        grow into the reservoir. Fluid leaks off into the formation, allowing the fractures to close
        except where they are held open by the proppant ([Adachi et al., 2007](#)).

    2.  During the production period, after fracturing is completed and pressure in the fractures is
        reduced, and hydrocarbons (along with produced water) flow from the reservoir into the
        well.

10   Note that these two time periods vary in duration. As described in Chapter 2, the first period of
11   fracture creation and propagation (i.e., the hydraulic fracturing itself) is a relatively short-term
12   process, typically lasting 2 to 10 days, depending on the number of stages in the fracture treatment
13   design. On the other hand, operation of the well for production covers a substantially longer period
14   (depending on many factors such as the amount of hydrocarbons in place and economic
15   considerations), and can be as long as 40 or 60 years in onshore tight gas reservoirs ([Ross and King,](#)
16   [2007](#)).

17   The following discussion of potential subsurface fluid migration into drinking water resources
18   focuses primarily on the physical movement of fluids and the factors that affect this movement.
19   Section 6.3.1 describes the basic principles of subsurface fracture creation, geometry, and
20   propagation, to provide context for the discussion of potential fluid migration pathways in Section
21   6.3.2. Geochemical and biogeochemical reactions among injected fluids, formation fluids, subsurface
22   microbes, and rock formations are another important component of subsurface fluid migration and
23   transport. See Chapter 7 for a discussion of the processes that affect pore fluid biogeochemistry and
24   influence the chemical and microbial composition of flowback and produced water.

### 6.3.1.   Overview of Subsurface Fracture Growth

25   Fracture initiation and growth is a highly complex process due to the heterogeneous nature of the
26   subsurface environment. It depends on the geomechanical characteristics of rock formations, fluid
27   properties, pore pressures, and subsurface stress fields. As shown in Figure 6-4, fracture formation
28   is controlled by the three *in situ* principal compressive stresses: the vertical stress ($S_V$), the
29   maximum horizontal stress ($S_H$), and the minimum horizontal stress ($S_h$). During hydraulic
30   fracturing, pressurized fluid injection creates high pore pressures around the well. When the
31   pressure exceeds the local least principal stress and the tensile strength of the rock, failure results
32   and fractures form ([Zoback, 2010](#); [Fjaer et al., 2008](#)).

33   Fractures propagate (increase in length) in the direction of the maximum principal stress, which is
34   perpendicular to the direction of the least principal stress. Deep in the subsurface, the maximum
35   principal stress is generally in the vertical direction because the overburden (the weight of
36   overlying rock) is the largest single stress. Therefore, in deep formations, the local least principal
37   stress is the minimum horizontal stress ($S_h$), and the principal fracture orientation is expected to be
38   vertical. This is the scenario illustrated in Figure 6-4. At shallower depths, where the rock is

1  subjected to less pressure from the overburden, more fracture propagation is expected to be in the
2  horizontal direction. Using tiltmeter data from over 10,000 fractures in various North American
3  reservoirs, Fisher and Warpinski (2012) found that fractures at approximately 4,000 ft (1,220 m)
4  below the surface or deeper are primarily vertical (see below for more information on tiltmeters).
5  Between approximately 4,000 and 2,000 ft (1,220 and 610 m), fracture complexity increases, and
6  fractures at approximately 2,000 ft (610 m) or shallower are primarily horizontal (Fisher and
7  Warpinski, 2012).[1] Horizontal fracturing can also occur in deeper settings in some less-common
8  reservoir environments where the principal stresses have been altered by salt intrusions or similar
9  types of geologic activity (Jones and Britt, 2009).
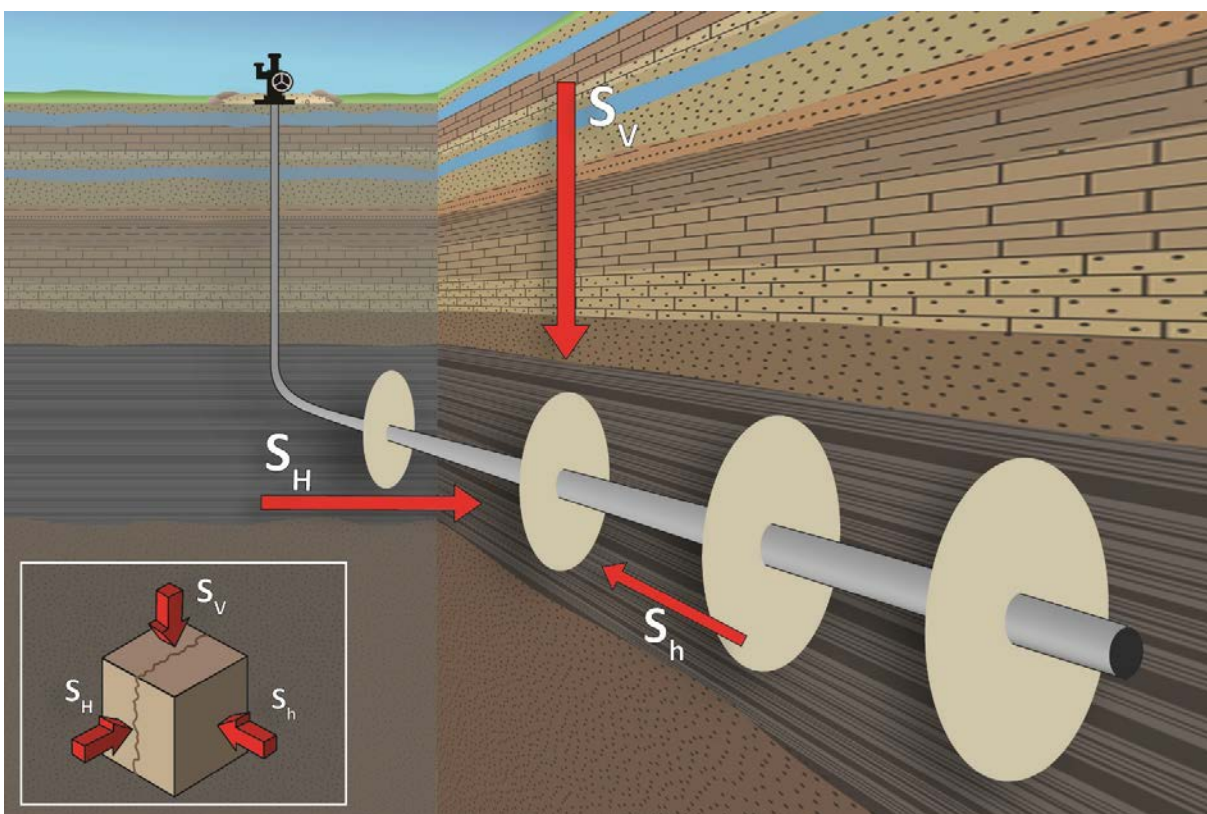


**Figure 6-4. Hydraulic fracture planes (represented as ovals), with respect to the principal subsurface compressive stresses: $S_V$ (the vertical stress), $S_H$ (the maximum horizontal stress), and $S_h$ (the minimum horizontal stress).**

At depths greater than approximately 2,000 ft (610 m), the principal fracture orientation is expected to be vertical, as fractures propagate in the direction of $S_H$.

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[1] Fracture complexity is the ratio of horizontal-to-vertical fracture volume distribution, as defined by Fisher and Warpinski (2012). Fracture complexity is higher in fractures with a larger horizontal component.

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1   In addition to the principal subsurface stresses, other geomechanical and reservoir characteristics
2   and operational factors affect fracture creation, geometry, and propagation.[1] These include initial
3   reservoir pressure and saturation, injected fluid pressure or injection rate, geomechanical rock
4   characteristics, reservoir heterogeneity, tensile strength, fluid type within fractures, and reservoir
5   permeabilities (Kim and Moridis, 2015). Fracture creation is a complex process that involves
6   interactions between multiple properties. For example, as described by Daneshy (2009), fracture
7   height depends on a combination of parameters and processes including the material properties of
8   geologic formations, pore pressures, stress differences in adjacent formations, shear failure
9   (slippage) at the fracture tip, and the reorientation of the fracture as it crosses an interface between
10  formations. Injection rates, the initial water saturation of the formation, and the type of fluid
11  injected also have effects on fracture creation and propagation (Kim and Moridis, 2015, 2013).

12  Numerical modeling techniques have been developed to describe fracture creation and propagation
13  and to provide a better understanding of this complex process (Kim and Moridis, 2013). Modeling
14  hydraulic fracturing in shale or tight gas reservoirs requires integrating the physics of both flow
15  and geomechanics to account for fluid flow, fracture propagation, and dynamic changes in pore
16  volume and permeability. Some important flow and geomechanical parameters included in these
17  types of advanced models are: permeability, porosity, Young's modulus, Poisson's ratio, and tensile
18  strength, as well as heterogeneities associated with these parameters.[2] Some investigations using
19  these models have indicated that the vertical propagation of fractures (due to tensile failure) may
20  be limited by shear failure, which increases the permeability of the formation and leads to greater
21  leakoff. These findings demonstrate that elevated pore pressure can cause shear failure, thus
22  further affecting matrix permeability, flow regimes, and leakoff (Daneshy, 2009). Computational
23  investigations have also indicated that slower injection rates can increase the amount of leakoff
24  (Kim and Moridis, 2013).

25  In addition to their use in research settings, analytical and numerical modeling approaches are used
26  by oil and gas companies to design hydraulic fracturing treatments and predict the extent of
27  fractured areas (Adachi et al., 2007). Specifically, modeling techniques are used to assess the
28  treatment's sensitivity to critical parameters such as injection rate, treatment volumes, fluid
29  viscosity, and leakoff. The industry models range from simpler (typically two-dimensional)
30  theoretical models to computationally more complicated and accurate three-dimensional models.

31  In addition to computational approaches, monitoring of hydraulic fracturing operations can provide
32  insights into fracture development. Monitoring techniques involve both operational monitoring
33  methods and "external" methods that are not directly related to the production operation.
34  Operational monitoring refers to the monitoring of parameters including pressure, flow rate, fluid
35  density, and additive concentrations using surface equipment and/or downhole sensors (Eberhard,
36  2011). This monitoring is conducted to ensure that the operation is proceeding as planned and to

---

[1] Fracture geometry refers to characteristics of the fracture such as height and aperture (width).
[2] Young's modulus, a ratio of stress to strain, is a measure of the rigidity of a material. Poisson's ratio is a ratio of transverse-to-axial (or latitudinal-to-longitudinal) strain, and it characterizes how a material is deformed under pressure. See Zoback (2010) for more information on the geomechanical properties of reservoir rocks.

1  determine if operational parameters need to be adjusted. Interpretation of pressure data can be
2  used to better understand fracture behavior (e.g., Kim and Wang, 2014). Anomalies in operational
3  monitoring data can also indicate whether an unexpected event has occurred, such as
4  communication with another well (see Section 6.3.2.3).

5  The volume of fluid injected is typically monitored to provide information on the volume and extent
6  of fractures created (Flewelling et al., 2013). However, numerical investigations have found that
7  reservoir gas flows into the fractures immediately after they open from hydraulic fracturing, and
8  injection pressurizes both gas and water within the fracture to induce further fracture propagation
9  (Kim and Moridis, 2015). Therefore, the fracture volume can be larger than the injected fluid
10 volume. As a result, simple estimation of fracture volume based on the amount of injected fluid may
11 underestimate the growth of the vertical fractures, and additional information is needed to
12 accurately predict the extent of fracture growth.

13 External monitoring technologies can also be used to collect data on fracture characteristics and
14 extent during hydraulic fracturing and/or production. These monitoring methods can be divided
15 into near-wellbore and far-field techniques. Near-wellbore techniques include the use of tracers,
16 temperature logs, video logs, or caliper logs (Holditch, 2007). However, near-wellbore techniques
17 and logs only provide information for, at most, a distance of two to three wellbore diameters from
18 the well and are, therefore, not suited for tracking fractures for their entire length (Holditch, 2007).
19 Far-field methods, such as microseismic monitoring or tiltmeters, are used if the intent is to
20 estimate fracture growth and height across the entire fractured reservoir area. Microseismic
21 monitoring involves placing one or more geophones in a position to detect the very small amounts
22 of seismic energy generated during subsurface fracturing (Warpinski, 2009).[1] Monitoring these
23 microseismic events gives an idea of the location and size of the fracture network, as well as the
24 orientation and complexity of fracturing (Fisher and Warpinski, 2012). Tiltmeters, which measure
25 extremely small deformations in the earth, can be used to determine the direction and volume of
26 the fractures and, within certain distances from the well, to estimate their dimensions (Lecampion
27 et al., 2005).

### 6.3.2.  Migration of Fluids through Pathways Related to Fractures/Formations

28 As noted above, subsurface migration of fluids requires a pathway, induced or natural, with enough
29 permeability to allow fluids to flow, as well as a hydraulic gradient physically driving the fluid
30 movement. The following subsections describe and evaluate potential pathways for the migration
31 of fracturing fluids, hydrocarbons, or other formation fluids from producing formations to drinking
32 water resources. They also present cases where the existence of these pathways has been
33 documented. As described above, potential subsurface migration pathways for fluid flow out of the
34 production formation are categorized as follows: (1) flow of fluids into the production zone via
35 induced fractures and out of the production zone via flow through the formation, (2) fracture
36 overgrowth out of the production zone, (3) migration via fractures intersecting offset wells and

---

[1] Typical microseismic events associated with hydraulic fracturing have a magnitude on the order of -2.5 (negative two and half) (Warpinski, 2009).

1    other artificial structures, and (4) migration via fractures intersecting other geologic features.
2    Although these four potential pathways are discussed separately here, they may act in combination
3    with each other or in combination with pathways along the well (as discussed in Section 6.2) to
4    affect drinking water resources.

5    In many cases (depending on fracture depth, height, and direction), the distance between the
6    producing formation and the drinking water resource is one of the most important factors affecting
7    the possibility of fluid migration between these formations (Reagan et al., 2015; Jackson et al.,
8    2013c). This distance varies substantially among shale gas plays, coalbed methane plays, and other
9    areas where hydraulic fracturing takes place in the United States (see Table 6-2). Many hydraulic
10   fracturing operations target deep shale zones such as the Marcellus or Haynesville/Bossier, where
11   the vertical distance between the top of the shale formation and the base of drinking water
12   resources may be 1 mile (1.6 km) or greater. This is reflected in the Well File Review, which found
13   that the largest proportion of wells used for hydraulic fracturing—an estimated 6,200 wells
14   (27%)—had 5,000 to 5,999 ft (1,524 to 1,828 m) of measured distance along the wellbore between
15   the induced fractures and the reported base of protected ground water resources (U.S. EPA,
16   2015o).[1] However, as shown in Table 6-2, operations in the Antrim and the New Albany plays take
17   place at relatively shallower depths, with distances of 100 to 1,900 ft (30 to 579 m) between the
18   producing formation and the base of drinking water resources. The Well File Review indicated that
19   20% of wells used for hydraulic fracturing (an estimated 4,600 wells) were located in areas with
20   less than 2,000 ft (610 m) between the fractures and the base of protected ground water resources
21   (U.S. EPA, 2015o). In coalbed methane plays, which are typically shallower than shale gas plays,
22   these separation distances can be even smaller. For example, in the Raton Basin of southern
23   Colorado and northern New Mexico, approximately 10% of coalbed methane wells have less than
24   675 ft (206 m) of separation between the gas wells' perforated intervals and the depth of local
25   water wells. In certain areas of the basin, this distance is less than 100 ft (30 m) (Watts, 2006).

26   Some hydraulic fracturing operations are conducted within formations that contain drinking water
27   resources (see Table 6-2). One example of hydraulic fracturing taking place within a geologic
28   formation that is also used as a drinking water source is in the Wind River Basin in Wyoming
29   (WYOGCC, 2014; Wright et al., 2012). Vertical gas wells in this area target the lower Eocene Wind
30   River Formation and the underlying Paleocene Fort Union Formation, which consist of interbedded
31   layers of sandstones, siltstones, and mudstones. The Wind River Formation also serves as the
32   principal source of domestic, municipal, and agricultural water in this rural area. Hydraulic
33   fracturing in rock formations that meet a state or federal definition of an underground source of
34   drinking water is also known to take place in coalbed methane operations in the Raton Basin (U.S.
35   EPA, 2015l), in the Powder River Basin of Montana and Wyoming (as described in Chapter 7), and
36   in several other coalbed methane plays. In one field in Alberta, Canada, there is evidence that
37   fracturing in the same formation as a drinking water resource (in combination with well integrity

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[1] In the Well File Review, measured depth represents length along the wellbore, which may be a straight vertical distance
below ground or may follow a more complicated path, if the wellbore is not straight and vertical.

1  problems; see Section 6.2.2.2) led to gas migration into water wells (Tilley and Muehlenbachs,
2  2012). However, no information is available on other specific incidents of this type.

**Table 6-2. Comparing the approximate depth and thickness of selected U.S. shale gas plays and coalbed methane basins.**

Shale data are reported in GWPC and ALL Consulting (2009) and NETL (2013); coalbed methane data are reported in ALL Consulting (2004) and U.S. EPA (2004). See Figures 2-2 and 2-4 in Chapter 2 for information on the locations of these basins, plays, and formations.

| Basin/play/formation[a] | Approx. depth (ft [m] below surface) | Approx. net thickness (ft [m]) | Distance between top of production zone and base of treatable water (ft [m]) |
|---|---|---|---|
| *Shale plays* | | | |
| Antrim | 600 to 2,200 [183 to 671] | 70 to 120 [21 to 37] | 300 to 1,900 [91 to 579] |
| Barnett | 6,500 to 8,500 [1,981 to 2,591] | 100 to 600 [30 to 183] | 5,300 to 7,300 [1,615 to 2,225] |
| Eagle Ford | 4,000 to 12,000 [1,219 to 3,658] | 250 [76] | 2,800 to 10,800 [853 to 3,292] |
| Fayetteville | 1,000 to 7,000 [305 to 2,134] | 20 to 200 [6 to 61] | 500 to 6,500 [152 to 1,981] |
| Haynesville-Bossier | 10,500 to 13,500 [3,200 to 4,115] | 200 to 300 [61 to 91] | 10,100 to 13,100 [3,078 to 3,993] |
| Marcellus | 4,000 to 8,500 [1,219 to 2,591] | 50 to 200 [15 to 61] | 2,125 to 7,650 [648 to 2,332] |
| New Albany | 500 to 2,000 [152 to 610] | 50 to 100 [15 to 30] | 100 to 1,600 [30 to 488] |
| Woodford | 6,000 to 11,000 [1,829 to 3,353] | 120 to 220 [37 to 67] | 5,600 to 10,600 [1,707 to 3,231] |
| *Coalbed methane basins* | | | |
| Black Warrior (Upper Pottsville) | 0 to 3,500 [0 to 1,067] | < 1 to > 70 [< 1 to > 21] | As little as zero[b] |
| Powder River (Fort Union) | 450 to >6,500 [137 to 1,981] | 75 [23] | As little as zero[b] |
| Raton (Vermejo and Raton) | < 500 to > 4,100 [< 152 to > 1,250] | 10 to >140 [3 to >43] | As little as zero[b] |
| San Juan (Fruitland) | 550 to 4,000 [168 to 1,219] | 20 to 80 [6 to 24] | As little as zero[b] |

*This document is a draft for review purposes only and does not constitute Agency policy.*

| Basin/play/formation[a] | Approx. depth (ft [m] below surface) | Approx. net thickness (ft [m]) | Distance between top of production zone and base of treatable water (ft [m]) |
|---|---|---|---|

[a] For coalbed methane, values are given for the specific coal units noted in parentheses.

[b] Formation fluids in producing formations meet the definition of drinking water in at least some areas of the basin.

1    The overall frequency of occurrence of hydraulic fracturing in aquifers that meet the definition of
2    drinking water resources across the United States is unknown. Some information, however, that
3    provides insights on the occurrence and geographic distribution of this practice is available.
4    According to the Well File Review, an estimated 0.4% of the 23,200 wells represented in that study
5    had perforations used for hydraulic fracturing that were placed shallower than the base of the
6    protected ground water resources reported by well operators (U.S. EPA, 2015o).[1] An analysis of
7    produced water composition data maintained by the U.S. Geological Survey (USGS) provides insight
8    into the geographic distribution of this practice. The USGS produced water database contains
9    results from analyses of samples of produced water collected from more than 8,500 oil and gas
10    production wells in unconventional formations (coalbed methane, shale gas, tight gas, and tight oil)
11    within the continental United States.[2] Just over 5,000 of these samples, which were obtained from
12    wells located in 37 states, reported total dissolved solids (TDS) concentrations. Because the
13    database does not track whether samples were from wells that were hydraulically fractured, we
14    selected samples from wells that were more likely to have been hydraulically fractured by
15    restricting samples to those collected in 1950 or later and to those that were collected from wells
16    producing from tight gas, tight oil, shale gas, or coalbed methane formations. This yielded 1,650
17    samples from wells located in Alabama, Colorado, North Dakota, Utah, and Wyoming.[3,4] The TDS
18    concentrations among these samples ranged from approximately 90 mg/L to 300,000 mg/L.
19    Samples from approximately 1,200 wells in Alabama, Colorado, Utah, and Wyoming reported TDS
20    concentrations at or below 10,000 mg/L. This analysis, in conjunction with the result from the Well
21    File Review, suggests that, while the overall frequency of occurrence may be low, the activity may
22    be concentrated in some areas of the country.

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[1] The 95% confidence interval reported in the Well File Review indicates that this phenomenon could have occurred in as few as 0.1% of the wells or in as many as 3% of the wells.

[2] We used the USGS Produced Water Geochemical Database Version 2.1 (USGS database v 2.1) for this analysis (http://energy.cr.usgs.gov/prov/prodwat/). The database is comprised of produced water samples compiled by the USGS from 25 individual databases, publications, or reports.

[3] See Chapter 2, Text Box 2-1, which describes how commercial hydraulic fracturing began in the late 1940s.

[4] For this analysis, we assumed that produced water samples collected in 1950 or later from shale gas, tight oil, and tight gas wells were from wells that had been hydraulically fractured. To estimate which coal bed methane wells had been hydraulically fractured, we matched API numbers from coal bed methane wells in the USGS database v 2.1 to the same API numbers in the commercial database DrillingInfo, in which hydraulically fractured wells had been identified by EPA using the assumptions described in Section 2.3.1. Wells with seemingly inaccurate (i.e., less than 12 digit) API numbers were also excluded. Only coalbed methane wells from the USGS database v 2.1 that matched API numbers in the DrillingInfo database were retained for this analysis.

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### 6.3.2.1. Flow of Fluids Out of the Production Zone

1  One potential pathway for fluid migration out of the production formation into drinking water
2  resources is flow of injected fluids (or displacement of formation fluids due to injection) through
3  the formation matrix during or after a hydraulic fracturing treatment. In deep, low-permeability
4  shale and tight gas settings and where induced fractures are contained within the production zone,
5  flow through the production formation has generally been considered an unlikely pathway for
6  migration into drinking water resources (Jackson et al., 2013c). However, there is limited
7  information available on the fate of injected fluids that are not recovered during production (i.e.,
8  leakoff) or displaced formation fluids for cases where hydraulic fracturing takes place within or
9  close to drinking water resources.

10  Leakoff into shale gas formations may be as high as 90% or more of the injected volume (see
11  Section 7.2 and Table 7-2). The actual amount of leakoff depends on the amount of injected fluid,
12  the hydraulic properties of the reservoir (e.g., permeability), the capillary pressure near the
13  fracture faces, and the period of time the well is shut in following hydraulic fracturing before the
14  start of production (Kim et al., 2014; Byrnes, 2011).[1,2] However, despite the potentially large
15  volume of fluid that may be lost into the formation, the flow of this fluid is generally controlled or
16  limited by processes such as imbibition by capillary forces and adsorption onto clay minerals
17  (Dutta et al., 2014; Dehghanpour et al., 2013; Dehghanpour et al., 2012; Roychaudhuri et al., 2011).[3]
18  It has been suggested that these processes can sequester the fluids in the producing formations
19  permanently or for geologic time scales (Engelder, 2012; Byrnes, 2011).

20  A limited number of studies in the literature have evaluated a combination of certain conditions
21  that can facilitate migration of fluids despite these processes. Myers (2012b) suggests that
22  migration of injected and/or formation fluids into the overburden may be possible in cases where
23  there is a significant vertical hydraulic gradient, sufficient permeability, density-driven buoyancy,
24  and the displacement of formation brines by large volumes of injected fluid. Flewelling and Sharma
25  (2014) note that, for migration to occur, an upward hydraulic gradient would be necessary,
26  particularly for brine that is denser than the ground water in the overlying formations; in the case
27  of natural gas, though, buoyancy would provide an upward flux A limited number of studies in the
28  literature have evaluated a combination of certain conditions that can facilitate migration of fluids
29  despite these processes. Myers (2012b) suggests that migration of injected and/or formation fluids
30  into the overburden may be possible in cases where there is a significant vertical hydraulic
31  gradient, sufficient permeability, density-driven buoyancy, and the displacement of formation

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[1] Relative permeability is a dimensionless property allowing for the comparison of the different abilities of fluids to flow in multiphase settings. If a single fluid is present, its relative permeability is equal to 1, but the presence of multiple fluids generally inhibits flow and decreases the relative permeability (Schlumberger, 2014).
[2] Shutting in the well after fracturing allows fluids to move farther into the formation, resulting in a higher gas relative permeability near the fracture surface and improved gas production (Bertoncello et al., 2014).
[3] Imbibition is the displacement of a nonwetting fluid (i.e., gas) by a wetting fluid (typically water). The terms wetting or nonwetting refer to the preferential attraction of a fluid to the surface. In typical reservoirs, water preferentially wets the surface, and gas is nonwetting. Capillary forces arise from the differential attraction between immiscible fluids and solid surfaces; these are the forces responsible for capillary rise in small-diameter tubes and porous materials. These definitions are adapted from Dake (1978).

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1  brines by large volumes of injected fluid. Flewelling and Sharma (2014) note that for migration to
2  occur, an upward hydraulic gradient would be necessary, particularly for brine that is denser than
3  the ground water in the overlying formations; in the case of natural gas, though, buoyancy would
4  provide an upward flux (Vengosh et al., 2014). Some natural conditions could create this upward
5  hydraulic gradient in the absence of any effects from hydraulic fracturing (Flewelling and Sharma,
6  2014). However, these natural mechanisms have been found to cause very low flow rates over very
7  long distances, yielding extremely small vertical fluxes in sedimentary basins—corresponding to
8  some estimated travel times of 100,000 to 100,000,000 years across a 328 ft (100 m) thick layer
9  with about 0.01 nD ($1 \times 10^{-23}$ m²) permeability (Flewelling and Sharma, 2014). Furthermore,
10  fracturing fluid would likely be sequestered in the immediate vicinity of the fracture network due to
11  capillary tension (Engelder, 2012).

12  Over-pressurization of producing formations due to the injection of large amounts of fluid during
13  hydraulic fracturing may support the upward hydraulic gradient for fluid migration (Myers,
14  2012b). Myers' modeling results suggest that significant pressure buildup that occurs at the
15  location of fluid injection may not return to pre-hydraulic fracturing levels for up to a year.
16  However, these findings have been disputed in the literature due to certain suggested limitations of
17  the original study (e.g., extensive simplification of the model, lack of accurate characterization of
18  regional flow, misrepresentation of saturation conditions in shale formations), and they have been
19  found to be physically implausible given the hydrogeologic characteristics of actual sedimentary
20  basins (Cohen et al., 2013; Flewelling et al., 2013; Vidic et al., 2013; Saiers and Barth, 2012). Some
21  researchers have also suggested that pressure perturbations due to hydraulic fracturing operations
22  are localized to the immediate vicinity of the fractures, due to the very low permeabilities of shale
23  formations (Flewelling and Sharma, 2014). However, there are emerging studies indicating that
24  pressure impacts of hydraulic fracturing operations may extend farther than the immediate vicinity
25  and may create risk of induced seismicity (Skoumal et al., 2015). Following hydraulic fracturing
26  operations, a large-scale depressurization would be expected over the longer term due to
27  hydrocarbon production, which may counteract any short-term localized pressure effects of
28  hydraulic fracturing during production and cause fluids to flow primarily toward the fracture
29  network (Flewelling and Sharma, 2014).

30  In responses to these critiques, Myers (2013, 2012a) states that they do not prove his original
31  hypothesis or findings wrong, but instead highlight the need for complex three-dimensional
32  modeling and detailed data collection for improving the understanding of the process and risks to
33  drinking water resources. Myers (2013, 2012a) argues that, given the large volume of hydraulic
34  fracturing operations in formations such as the Marcellus, these formations would have to hold
35  very large volumes of water that would be imbibed into the shale. Furthermore, he notes that
36  migration of these fluids into overlying formations may be facilitated by existing fractures or out-of-
37  zone fracturing (as discussed in the following sections).

### 6.3.2.2.  Fracture Overgrowth out of the Production Zone

38  Fractures that extend out of the intended production zone into another formation or an unintended
39  zone within the same formation could provide a potential fluid migration pathway into drinking

1    water resources ([Jackson et al., 2013c](#)). This migration could occur either through the fractures
2    themselves or in connection with other permeable subsurface features or formations (see Figure
3    6-5). Such "out-of-zone fracturing" is undesirable from a production standpoint and may occur as a
4    result of inadequate reservoir characterization or fracture treatment design ([Eisner et al., 2006](#)).
5    Some researchers have noted that fractures growing out of the targeted production zone could
6    potentially contact other formations, such as higher conductivity sandstones or conventional
7    hydrocarbon reservoirs, which may create an additional pathway for potential migration into a
8    drinking water resource ([Reagan et al., 2015](#)). In addition, fractures growing out of the production
9    zone could potentially intercept natural, preexisting fractures (discussed in Section 6.3.2.4) or
10   active or abandoned wells near the well where hydraulic fracturing is performed (discussed in
11   Section 6.3.2.3).
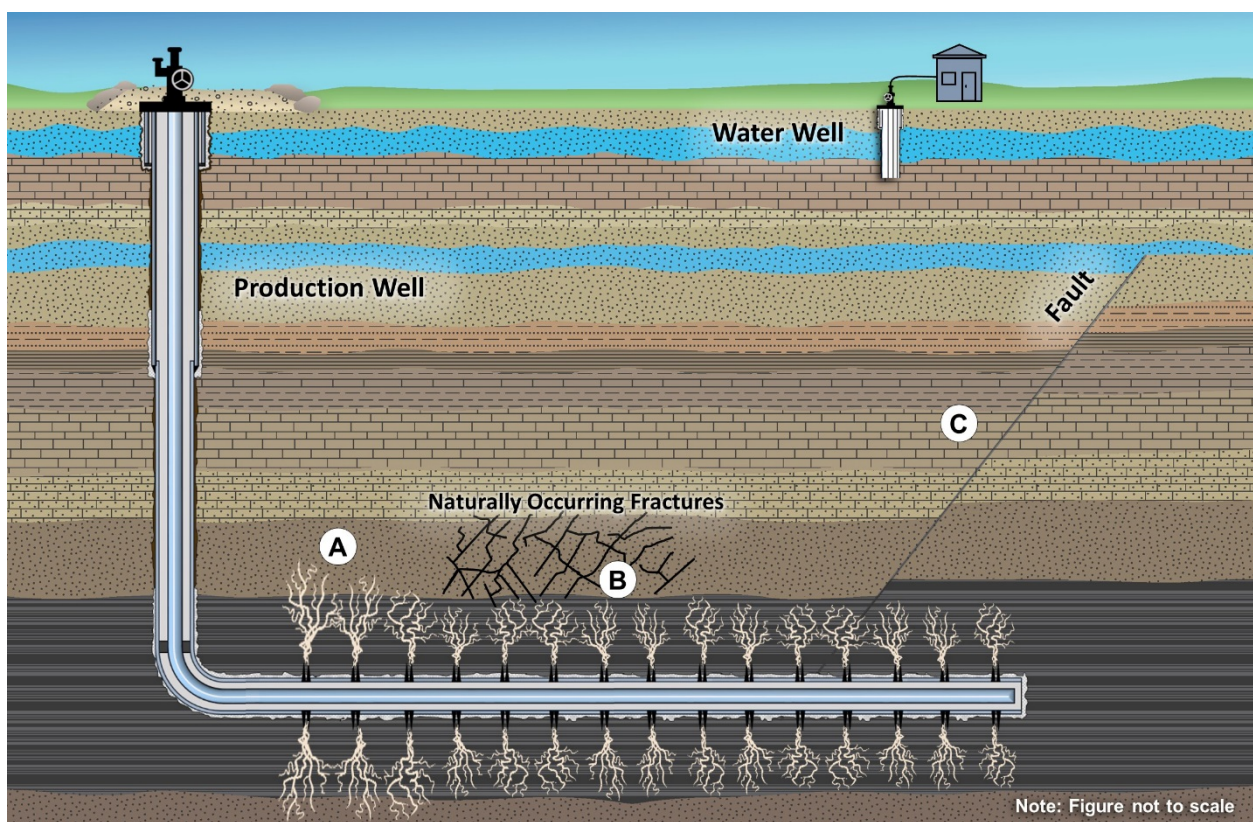


**Figure 6-5. Conceptualized depiction of potential pathways for fluid movement out of the production zone: (a) induced fracture overgrowth into over- or underlying formations; (b) induced fractures intersecting natural fractures; and (c) induced fractures intersecting a transmissive fault.**

Thickness and depth of production formation are important site specific factors for each operation.

12   The fracture's geometry (see Section 6.3.1) affects its potential to extend beyond the intended zone
13   and serve as a pathway to drinking water resources. Vertical heights of fractures created during

1   hydraulic fracturing operations have been measured in several U.S. shale plays, including the
2   Barnett, Woodford, Marcellus, and Eagle Ford, using microseismic and microdeformation field
3   monitoring techniques (Fisher and Warpinski, 2012). These data indicate typical fracture heights
4   extending from tens to hundreds of feet. Davies et al. (2012) analyzed this data set and found that
5   the maximum fracture height was 1,929 ft (588 m) and that 1% of the fractures had a height greater
6   than 1,148 ft (350 m). This may raise some questions about fractures being contained within the
7   producing formation, as some Marcellus fractures were found to extend for at least 1,500 ft
8   (477 m), while the maximum thickness of the formation is generally 350 ft (107 m) or less (MCOR,
9   2012). However, the majority of fractures were found to have heights less than 328 ft (100 m),
10  suggesting limited possibilities for fracture overgrowth exceeding the separation between shale
11  reservoirs and shallow aquifers (Davies et al., 2012). This is consistent with modeling results found
12  by Kim and Moridis (2015) and others, as described below. Where the producing formation is not
13  continuous horizontally, the lateral extent of fractures may also become important. For example, in
14  the Fisher and Warpinski (2012) data set, fractures were found to extend to horizontal lengths
15  greater than 1,000 ft (305 m).
16  Results of National Energy Technology Laboratory (NETL) research in Greene County,
17  Pennsylvania, are generally consistent with those reported in the Fisher and Warpinski (2012) data
18  set. Microseismic monitoring was used at six horizontal Marcellus Shale wells to identify the
19  maximum upward extent of brittle deformation caused by hydraulic fracturing (Hammack et al.,
20  2014). At three of the six wells, fractures extending between 1,000 and 1,900 ft (305 and 579 m)
21  above the Marcellus Shale were identified. Overall, approximately 40% of the microseismic events
22  occurred above the Tully Limestone, the formation overlying the Marcellus Shale that is sometimes
23  referred to as an upper barrier to hydraulic fracture growth. However, all microseismic events were
24  at least 5,000 ft (1,524 m) below drinking water aquifers, as the Marcellus Shale is one of the
25  deepest target formations (see Table 6-2), and no impacts to drinking water resources or another
26  local gas-producing interval were identified. See Text Box 6-3 for more information on the Greene
27  County site.

28  Similarly, in Dunn County, North Dakota, there is evidence of out-of-zone fracturing in the Bakken
29  Shale (U.S. EPA, 2015j). At the Killdeer site (see Section 6.2.2.1 and Chapter 5, Text Box 5-12),
30  fracturing fluids and produced water were released during a rupture of the casing at the Franchuk
31  44-20 SWH well. Water quality characteristics at two monitoring wells located immediately
32  downgradient of the Franchuk well reflected a mixing of local Killdeer Aquifer water with deep
33  formation brine. Ion and isotope ratios used for brine fingerprinting suggest that Madison Group
34  formations (which directly overlie the Bakken in the Williston Basin) were the source of the brine
35  observed in the Killdeer Aquifer, and the authors concluded that this provides evidence for out-of-
36  zone fracturing. Industry experience also indicates that out-of-zone fracturing may be fairly
37  common in the Bakken and that produced water from many Bakken wells has Madison Group
38  chemical signatures (Arkadakskiy and Rostron, 2013b, 2012b; Peterman et al., 2012).

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**Text Box 6-3. Monitoring at the Greene County, Pennsylvania, Hydraulic Fracturing Test Site.**

1  Monitoring performed at the Marcellus Shale test site in Greene County, Pennsylvania, evaluated fracture
2  height growth and zonal isolation during and after hydraulic fracturing operations ([Hammack et al., 2014](#)).
3  The site has six horizontally drilled and two vertical wells that were completed into the Marcellus Shale.
4  Pre-fracturing studies of the site included a 3D seismic survey to identify faults, pressure measurements, and
5  baseline sampling for isotopes; drilling logs were also run. Hydraulic fracturing occurred April 24 to May 6,
6  2012, and June 4 to 11, 2012. Monitoring at the site included the following:

7  • **Microseismic monitoring** was conducted during four of the six hydraulic fracturing jobs on the site,
8  using geophones placed in the two vertical Marcellus Shale wells. These data were used to monitor
9  fracture height growth above the six horizontal Marcellus Shale wells during hydraulic fracturing.

10  • **Pressure and production data** were collected from a set of vertical gas wells completed in Upper
11  Devonian/Lower Mississippian zones 3,800 to 6,100 ft (1,158 to 1,859 m) above the Marcellus. Data were
12  collected during and after the hydraulic fracturing jobs and used to identify any communication between
13  the fractured areas and the Upper Devonian/Lower Mississippian rocks.
14  • **Chemical and isotopic analyses** were conducted on gas and water produced from the Upper
15  Devonian/Lower Mississippian wells. Samples were analyzed for stable isotope signatures of hydrogen,
16  carbon, and strontium and for the presence of perfluorocarbon tracers used in 10 stages of one of the
17  hydraulic fracturing jobs to identify possible gas or fluid migration to overlying zones ([Sharma et al.,](#)
18  [2014a](#); [Sharma et al., 2014b](#)).
19  As of September 2014, no evidence was found of gas or brine migration from the Marcellus Shale ([Hammack](#)
20  [et al., 2014](#)), although longer-term monitoring will be necessary to confirm that no impacts to overlying zones
21  have occurred ([Zhang et al., 2014a](#)).

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22  Extreme vertical fracture growth is generally considered to be limited by layered geological
23  environments and other physical constraints ([Fisher and Warpinski, 2012](#); [Daneshy, 2009](#)). For
24  example, differences in in situ stresses in layers above and below the production zone can restrict
25  fracture height growth in sedimentary basins ([Fisher and Warpinski, 2012](#)). High-permeability
26  layers near hydrocarbon-producing zones can reduce fracture growth by acting as a "thief zone"
27  into which fluids can migrate, or by inducing a large compressive stress that acts on the fracture ([de](#)
28  [Pater and Dong, 2009, as cited in Fisher and Warpinski, 2012](#)). Although these thief zones may
29  prevent fractures from reaching shallower formations or growing to extreme vertical lengths, it is
30  important to note that they do allow fluids to migrate out of the production zone into these
31  receiving formations, which could potentially contain drinking water resources. A volumetric
32  argument has also been used to discuss limits of vertical fracture growth; that is, the volumes of
33  fluid needed to sustain fracture growth beyond a certain height would be unrealistic ([Fisher and](#)
34  [Warpinski, 2012](#)). However, as described in Section 6.3.1, fracture volume can be greater than the
35  volume of injected fluid due to the effects of pressurized water combined with the effects of gas
36  during injection ([Kim and Moridis, 2015](#)). Nevertheless, some numerical investigations suggest
37  that, unless unrealistically high pressures and injection rates are applied to an extremely weak and
38  homogeneous formation that extends up to the near surface, hydraulic fracturing generally induces

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1    stable and finite fracture growth in a Marcellus-type environment and the fractures are unlikely to
2    extend into drinking water resources ([Kim and Moridis, 2015](#)).

3    Modeling studies have identified other factors that affect the containment of fractures within the
4    producing formation. As discussed above, additional numerical analysis of fracture propagation
5    during hydraulic fracturing has demonstrated that contrasts in the geomechanical properties of
6    rock formations can affect fracture height containment ([Gu and Siebrits, 2008](#)) and that geological
7    layers present within shale gas reservoirs can limit vertical fracture propagation ([Kim and Moridis,](#)
8    [2015](#)). Modeling and monitoring studies generally agree that physical constraints on fracture
9    propagation will prevent induced fractures from extending from deep zones directly into drinking
10   water resources ([Kim and Moridis, 2015](#); [Flewelling et al., 2013](#); [Fisher and Warpinski, 2012](#)).

11   Using a numerical simulation, [Reagan et al. (2015)](#) investigated potential short-term migration of
12   gas and water between a shale or tight gas formation and a shallower ground water unit. Migration
13   was assessed immediately after hydraulic fracturing and for up to a 2-year time period during the
14   production stage. The potential migration pathway was assumed to be a permeable fracture or fault
15   connecting the producing formation to the shallower ground water unit. Such a pathway may be
16   either entirely hydraulically induced (due to fracture overgrowth in a case where the separation
17   distance is limited, as discussed below), or may be a smaller induced fracture connecting to a
18   natural, permeable fault or fracture (as discussed in Section 6.3.2.4). For the purposes of this study,
19   the pathway was assumed to be pre-existing, and [Reagan et al. (2015)](#) did not model the fracturing
20   process itself.

21   The subsurface system evaluated in the modeling investigation included a horizontal well used for
22   hydraulic fracturing and gas production, the connecting fracture or fault between the producing
23   formation and the aquifer, and a shallow vertical water well in the aquifer (see Figure 6-5). The
24   parameters and scenarios used in the study are shown in Table 6-3; two vertical separation
25   distances between the producing formation and the aquifer were investigated, along with a range of
26   production zone permeabilities and other variables used to describe four production scenarios. The
27   horizontal well was assigned a constant bottomhole pressure of half the initial pressure of the
28   target reservoir, not accounting for any over-pressurization from hydraulic fracturing. Over-
29   pressurization during hydraulic fracturing may create an additional driving force for upward
30   migration. Results of this investigation, which represents a typical production period, indicate a
31   generally downward water flow within the connecting fracture (from the aquifer through the
32   connecting fracture into the hydraulically induced fractures in the production zone) and some
33   upward migration of gas ([Reagan et al., 2015](#)). In certain cases, gas breakthrough (i.e., the
34   appearance of gas at the base of the drinking water aquifer) was also observed. The key parameter
35   affecting migration of gas into the aquifer was the production regime, particularly whether gas
36   production, driving the fluid migration toward the production well, was occurring in the reservoir.
37   Simulations including a producing gas well showed only a few instances of breakthrough, while
38   simulations without gas production tended to result in breakthrough; these breakthrough times
39   ranged from minutes to 20 days. However, in all cases, the gas escape was limited in duration and
40   scope, because the amount of gas available for immediate migration toward the shallow aquifer was

1 limited to that initially stored in the hydraulically induced fractures after the stimulation process
2 and prior to production. These simulations indicate that the target reservoir may not be able to
3 replenish the gas available for migration in hydraulically induced fractures prior to production.

**Table 6-3. Modeling parameters and scenarios investigated by Reagan et al. (2015).**

This table illustrates the range of parameters included in the Reagan et al. (2015) modeling study. See Figure 6-5, Figure 6-6, and Figure 6-7 for conceptualized illustrations of these scenarios.

| Model parameter or variable | Values investigated in model scenarios |
|---|---|
| ***All scenarios*** | |
| Lateral distance from connecting feature to water well | 328 ft (100 m) |
| Vertical separation distance between producing formation and drinking water aquifer | 656 ft (200 m); 2,625 ft (800 m) |
| Producing formation permeability range | 1 nD ($1 \times 10^{-21}$ m$^2$); 100 nD ($1 \times 10^{-19}$ m$^2$); 1 μD ($1 \times 10^{-18}$ m$^2$) |
| Drinking water aquifer permeability | 0.1 D ($1 \times 10^{-13}$ m$^2$); 1 D ($1 \times 10^{-12}$ m$^2$) |
| Initial conditions | Hydrostatic |
| Production well bottom hole pressure | Half of the initial pressure of the producing formation (not accounting for over-pressurization from hydraulic fracturing) |
| Production regime | Production at both the water well and the gas well; Production at only the water well; Production at only the gas well; No production |
| ***Fracture pathway scenarios*** | |
| Connecting feature permeability | 1 D ($1 \times 10^{-12}$ m$^2$); 10 D ($1 \times 10^{-11}$ m$^2$); 1,000 D ($1 \times 10^{-9}$ m$^2$) |
| ***Offset well pathway scenarios*** | |
| Lateral distance from production well to offset well | 33 ft (10 m) |
| Cement permeability of offset well | 1 μD ($1 \times 10^{-18}$ m$^2$); 1 mD ($1 \times 10^{-15}$ m$^2$); 1 D ($1 \times 10^{-12}$ m$^2$); 1,000 D ($1 \times 10^{-9}$ m$^2$) |

*This document is a draft for review purposes only and does not constitute Agency policy.*

1 Based on the results of the <u>Reagan et al. (2015)</u> study, gas production from the reservoir appears
2 likely to mitigate gas migration, both by reducing the amount of available gas and depressurizing
3 the induced fractures (which counters the buoyancy of any gas that may escape from the
4 production zone into the connecting fracture). Production at the gas well also creates pressure
5 gradients that drive a downward flow of water from the aquifer via the fracture into the producing
6 formation, increasing the amount of water produced at the gas well. Furthermore, the effective
7 permeability of the connecting feature is reduced during water (downward) and gas (upward)
8 counter-flow within the fracture, further retarding the upward movement of gas or allowing gas to
9 dissolve into the downward flow. In contrast, <u>Reagan et al. (2015)</u> found an increased potential for
10 gas release from the producing formation in cases where there is no gas production following
11 hydraulic fracturing. The potential for gas migration during shut-in periods following hydraulic
12 fracturing and prior to production may be more significant, especially when out-of-zone fractures
13 are formed. Without the producing gas well, the gas may rise via buoyancy, with any downward-
14 flowing water from the aquifer displacing the upward-flowing gas.

15 <u>Reagan et al. (2015)</u> also found that the permeability of a connecting fault or fracture may be an
16 important factor for the potential upward migration of gas (although not as significant as the
17 production regime). For the cases where gas escaped from the production zone, the maximum
18 amount of migrating gas depended upon the permeability of the connecting feature: the higher the
19 permeability, the larger the amount. The results also showed that lower permeabilities delay the
20 downward flow of water from the aquifer, allowing the trace amount of gas that entered into the
21 fracture early in the modeled period to reach the aquifer, which was otherwise predicted to
22 dissolve in the water flowing downward in the feature. Similarly, the permeabilities of the target
23 reservoir, fracture volume, and the separation distance were found to affect gas migration, because
24 they affected the initial amount of gas stored in the hydraulically induced fractures. In contrast, the
25 permeability of the drinking water aquifer was not found to be a significant factor in their
26 assessment.

### 6.3.2.3. Migration via Fractures Intersecting with Offset Wells and Other Artificial Structures

27 Another potential pathway for fluid migration is one in which injected fluids or displaced
28 subsurface fluids move through newly created fractures into an offset well or its fracture network,
29 resulting in well communication (<u>Jackson et al., 2013c</u>). This may be a concern, particularly in
30 shallower formations where the local least principal stress is vertical (resulting in more horizontal
31 fracture propagation) and where there are shallow drinking water wells in the same formation.

32 The offset well can be an abandoned, inactive, or active well; if the well has also been used for
33 hydraulic fracturing, the fracture networks of the two wells might intersect. The situation where
34 hydraulic fractures unintentionally propagate into other existing, producing hydraulic fractures is
35 referred to as a "frac hit" and is known to occur in areas with a high density of wells (<u>Jackson et al.,</u>
36 <u>2013a</u>). Figure 6-6 provides a schematic to illustrate fractures that intercept an offset well, and
37 Figure 6-7 depicts how the fracture networks of two wells can intersect.
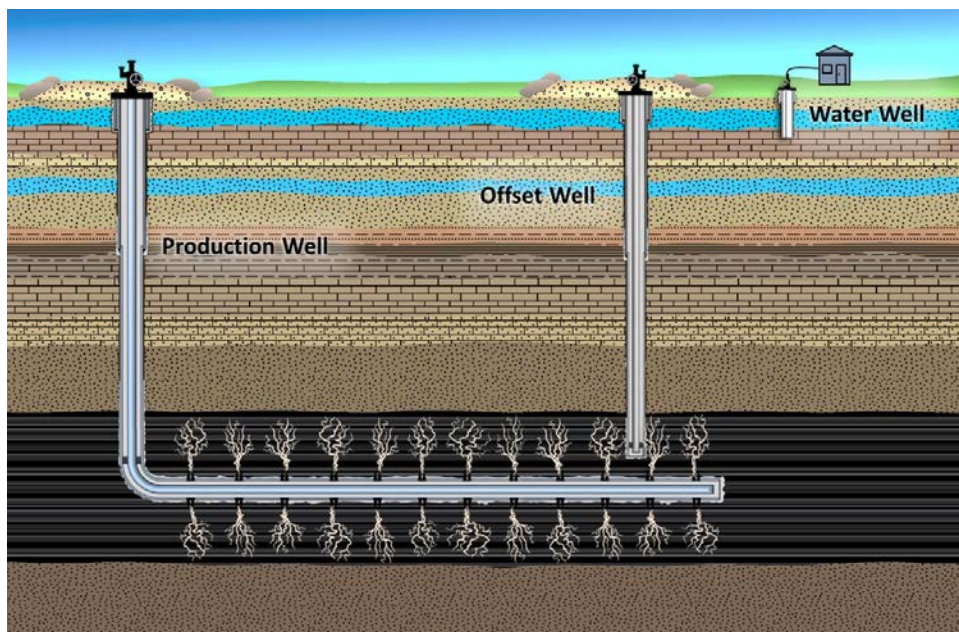
**Figure 6-6. Induced fractures intersecting an offset well (in a production zone, as shown, or in overlying formations into which fracture growth may have occurred).**

This image shows a conceptualized depiction of potential pathways for fluid movement out of the production zone (not to scale).
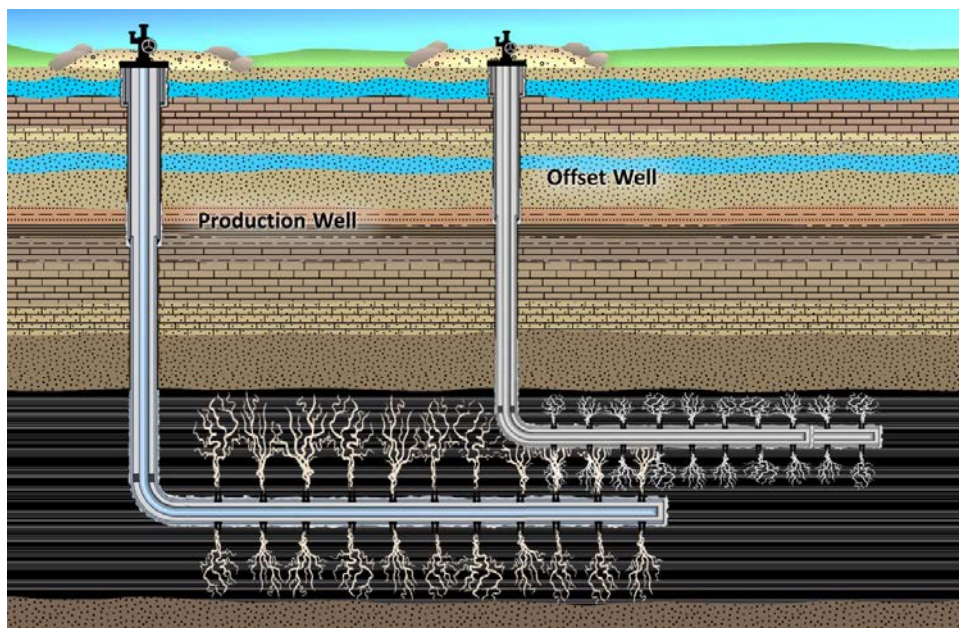


**Figure 6-7. Well communication (a frac hit) via induced fractures intersecting another well or its fracture network.**

This image shows a conceptualized depiction of potential pathways for fluid movement out of the production zone (not to scale).

1   Instances of well communication have been known to occur and are described in the oil and gas
2   literature. For example, an analysis of operator data collected by the New Mexico Oil Conservation
3   Division (NM OCD) in 2013–2014 identified 120 instances of well communication in the San Juan
4   Basin (Vaidyanathan, 2014). In some cases, well communication incidents have led to documented
5   production and/or environmental problems. A study from the Barnett Shale noted two cases of well
6   communication, one with a well 1,100 ft (335 m) away and the other with a well 2,500 ft (762 m)
7   away from the initiating well; ultimately, one of the offset wells had to be re-fractured because the
8   well communication halted production (Craig et al., 2012). In some cases, the fluids that intersect
9   the offset well flow up the wellbore and spill onto the surface. The EPA (2015n) recorded 10
10  incidents in which fluid spills were attributed to well communication events (see Chapter 7 for
11  more information).[1] The subsurface effects of frac hits have not been extensively studied, but these
12  cases demonstrate the possibility of fluid migration via communication with other wells and/or
13  their fracture networks. More generally, well communication events may indicate fracture behavior
14  that was not intended by the treatment design.

15  A well communication event is usually observed at the offset well as a pressure spike, due to the
16  elevated pressure from the originating well, or as an unexpected drop in the production rate (Lawal
17  et al., 2014; Jackson et al., 2013a). Ajani and Kelkar (2012) performed an analysis of frac hits in the
18  Woodford Shale in Oklahoma, studying 179 wells over a 5-year period. The authors used fracturing
19  records from the newly completed wells and compared them to production records from
20  surrounding wells. The authors assumed that sudden changes in production of gas or water
21  coinciding with fracturing at a nearby well were caused by communication between the two wells,
22  and increased water production at the surrounding wells was assumed to be caused by fracturing
23  fluid flowing into these offset wells. The results of the Oklahoma study showed that 24 wells had
24  decreased gas production or increased water production within 60 days of the initial gas
25  production at the nearby fractured well. A total of 38 wells experienced decreased gas or increased
26  water production up to a distance of 7,920 ft (2,414 m), measured as the distance between the
27  midpoints of the laterals; 10 wells saw increased water production from as far away as 8,422 ft
28  (2,567 m). In addition, one well showed a slight increase in gas production rather than a decrease.[2]

29  Other studies of well communication events have relied on similar information. In the NM OCD
30  operator data set, the typical means of detecting a well communication event was through pressure
31  changes at the offset well, production lost at the offset well, or fluids found in the offset well. In
32  some instances, well operators determined that a well was producing fluid from two different
33  formations, while in one instance, the operator identified a potential well communication event due
34  to an increase in production from the offset well (Vaidyanathan, 2014). In another study, Jackson et
35  al. (2013a) found that the decrease in production due to well communication events was much
36  greater in lower permeability reservoirs. The authors note an example where two wells 1,000 ft
37  (305 m) apart communicated, reducing production in the offset well by 64%. These results indicate
38  that the subsurface interactions of well networks or complex hydraulics driven by each well at a

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[1] Line numbers 163, 236, 265, 271, 286, 287, 375, 376, 377, and 380 in Appendix B of U.S. EPA (2015n).
[2] The numbers of wells cited in the study reflect separate analyses, and the numbers cited are not additive.

1  densely populated (with respect to wells) area are important factors to consider for the design of
2  hydraulic fracturing treatments and other aspects of oil and gas production.

3  The key factor affecting the likelihood of a well communication event and the impact of a frac hit is
4  the location of the offset well relative to the well where hydraulic fracturing was conducted (Ajani
5  and Kelkar, 2012). In the Ajani and Kelkar (2012) analysis, the likelihood of a communication event
6  was less than 10% in wells more than 4,000 ft (1,219 m) apart, but rose to nearly 50% in wells less
7  than 1,000 ft (305 m) apart. Well communication was also much more likely with wells drilled from
8  the same pad. The affected wells were found to be in the direction of maximum horizontal stress in
9  the field, which correlates with the expected direction of fracture propagation.

10  Well communication may be more likely to occur where there is less resistance to fracture growth.
11  Such conditions may be related to existing production operations (e.g., where previous
12  hydrocarbon extraction has reduced the pore pressure, changed stress fields, or affected existing
13  fracture networks) or the existence of high-permeability rock units (Jackson et al., 2013a). As Ajani
14  and Kelkar (2012) found in the Woodford Shale, one of the deepest major shale plays (see Table
15  6-2), hydraulic fracturing treatments tend to enter portions of the reservoir that have already been
16  fractured as opposed to entering previously unfractured rocks, ultimately causing interference in
17  offset wells. Mukherjee et al. (2000) described this tendency for asymmetric fracture growth
18  toward depleted areas in low-permeability gas reservoirs due to pore pressure depletion from
19  production at offset wells. The authors note that pore pressure gradients in depleted zones would
20  affect the subsurface stresses. Therefore, depending on the location of the new well with respect to
21  depleted zone(s) and the orientation of the existing induced fractures, the newly created fracture
22  may be asymmetric, with only one wing of the fracture extending into the depleted area and
23  developing significant length and conductivity (Mukherjee et al., 2000). The extent to which the
24  depleted area affects fracturing depends on factors such as cumulative production, pore volume,
25  hydrocarbon saturation, effective permeability, and the original reservoir or pore pressure
26  (Mukherjee et al., 2000). Similarly, high-permeability rock types acting as thief zones may also
27  cause preferential fracturing due to a higher leakoff rate into these layers (Jackson et al., 2013a).

28  In addition to location, the potential for impact on a drinking water resource also depends on the
29  condition of the offset well (see Section 6.2 for information on the integrity of well components). In
30  their analysis, Ajani and Kelkar (2012) found a correlation between well communication and well
31  age: older wells were more likely to be affected. If the cement in the annulus between the casing
32  and the formation is intact and the well components can withstand the stress exerted by the
33  pressure of the fluid, nothing more than an increase in pressure and extra production of fluids may
34  occur during a well communication event. However, if the offset well is not able to withstand the
35  pressure of the fracturing fluid, well components may fail, allowing fluid to migrate out of the well.
36  The highest pressures most wells will face during their life spans occur during fracturing. In some
37  cases, temporary equipment is installed in wells during fracturing to protect the well against the
38  increased pressure. Therefore, many producing wells may not be designed to withstand pressures
39  typical of hydraulic fracturing (Enform, 2013) and may experience problems when fracturing
40  occurs in nearby wells. Depending on the location of the weakest point in the offset well, this could

1  result in fluid being spilled onto the surface, rupturing of cement and/or casing and hydraulic
2  fracturing fluid leaking into subsurface formations, or fluid flowing out through existing flaws in the
3  casing and/or cement (see Chapters 5 and 7 for additional information on how such spills can affect
4  drinking water resources). For example, a documented well communication event near Innisfail,
5  Alberta, Canada (see Text Box 6-4) occurred when several well components failed because they
6  were not rated to handle the increased pressure caused by the well communication (ERCB, 2012).
7  In addition, if the fractures were to intersect an uncemented portion of the wellbore, the fluids
8  could potentially migrate into any formations that are uncemented along the wellbore.

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**Text Box 6-4. Well Communication at a Horizontal Well near Innisfail, Alberta, Canada.**

9   In most cases, well communication during fracturing may only result in a pressure surge accompanied by a
10  drop in gas production and additional flow of produced water or fracturing fluid at an offset well. However, if
11  the offset well is not capable of withstanding the high pressures of fracturing, more significant damage can
12  occur.

13  In January 2012, fracturing at a horizontal well near Innisfail in Alberta, Canada, caused a surface spill of
14  fracturing and formation fluids at a nearby operating vertical oil well. According to the investigation report by
15  the Alberta Energy Resources Conservation Board (ERCB, 2012), pressure began rising at the vertical well
16  less than two hours after fracturing ended at the horizontal well.

17  Several components of the vertical well facility—including surface piping, discharge hoses, fuel gas lines, and
18  the pressure relief valve associated with compression at the well—were not rated to handle the increased
19  pressure and failed. Ultimately, the spill released an estimated 19,816 gallons (75 m$^3$) of fracturing fluid,
20  brine, gas, and oil covering an area of approximately 656 ft by 738 ft (200 m by 225 m).

21  The ERCB determined that the lateral of the horizontal well passed within 423 ft (129 m) of the vertical well
22  at a depth of approximately 6,070 ft (1,850 m) below the surface, in the same formation. The operating
23  company had estimated a fracture half-length of 262 to 295 ft (80 to 90 m) based on a general fracture model
24  for the field. While there were no regulatory requirements for spacing hydraulic fracturing operations in
25  place at the time, the 423 ft (129 m) distance was out of compliance with the company's internal policy to
26  space fractures from adjacent wells at least 1.5 times the predicted half-length. The company also did not
27  notify the operators of the vertical well of the fracturing operations. The incident prompted the ERCB to issue
28  *Bulletin 2012-02—Hydraulic Fracturing: Interwellbore Communication between Energy Wells*, which outlines
29  expectations for avoiding well communication events and preventing adverse effects on offset wells.

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30  In older wells near a hydraulic fracturing operation, plugs and cement may have degraded over
31  time; in some cases, abandoned wells may never have been plugged properly. Before the 1950s,
32  most well plugging efforts were focused on preventing water from the surface from entering oil
33  fields. As a result, many wells from that period were abandoned with little or no cement (NPC,
34  2011b). This can be a significant issue in areas with legacy (i.e., historic) oil and gas exploration and
35  when wells are re-entered and fractured (or re-fractured) to increase production in a reservoir. In
36  one study, 18 of 29 plugged and abandoned wells in Quebec were found to show signs of leakage
37  (Council of Canadian Academies, 2014). Similarly, a PA DEP report cited three cases where natural
38  gas migration had been caused by well communication events with old, abandoned wells (PA DEP,

1    [2009b](#)). The Interstate Oil and Gas Compact Commission ([IOGCC, 2008](#)) estimates that over 1
2    million wells may have been drilled in the United States prior to a formal regulatory system, and the
3    status and location of many of these wells are unknown. Various state programs exist to plug
4    identified orphaned wells, but they face the challenge of identifying and addressing a large number
5    of wells.[1] For example, as of 2000, PA DEP's well plugging program reported that it had
6    documented 44,700 wells that had been plugged and 8,000 that were in need of plugging, and
7    approximately 184,000 additional wells with an unknown location and status ([PA DEP, 2000](#)). A
8    similar evaluation from New York State found that the number of unplugged wells was growing in
9    the state despite an active well plugging program ([Bishop, 2013](#)).

10   The [Reagan et al. (2015)](#) numerical modeling study included an assessment of migration via an
11   offset well as part of its investigation of potential fluid migration from a producing formation into a
12   shallower ground water unit (see Section 6.3.2.2). In the offset well pathway, it was assumed that
13   the hydraulically induced fractures intercepted an older offset well with deteriorated components.
14   (This assessment can also be applicable to cases where potential migration may occur via the
15   production well-related pathways discussed in Section 6.2.) More specifically, this analysis was
16   designed to assess transport through deteriorating cement between the subsurface formations and
17   the outermost casing, through voids resulting from incomplete cement coverage, through breached
18   tubing, or in simpler well installations without multiple casings. The highest permeability value
19   tested for the connecting feature represented a case with an open wellbore. A key assumption for
20   this investigation was that the offset well was already directly connected to a permeable feature in
21   the reservoir or within the overburden. Similar to the cases for permeable faults or fractures
22   discussed in the previous section, the study investigated the effect of multiple well- and formation-
23   related variables on potential fluid migration (see Table 6-3).

24   Based on the simulation results, an offset well pathway may have a greater potential for gas release
25   from the production zone into a shallower ground water unit than the fault/fracture pathway
26   discussed in Section 6.3.2.2 ([Reagan et al., 2015](#)). This difference is primarily due to the total pore
27   volume of the connecting pathway within the offset well; the offset well pathway may have a
28   significantly lower pore volume compared to the fault/fracture pathway, which reduces possible
29   gas storage in the connecting feature and increases the speed of buoyancy-dependent migration.
30   However, as with the fault/fracture scenario, the gas available for migration in this case is still
31   limited to the gas that is initially stored in the hydraulically induced fractures. Therefore, any
32   incidents of gas breakthrough observed in this study were found to be limited in both duration and
33   magnitude.

34   [Reagan et al. (2015)](#) found that production at the gas well (the well used for hydraulic fracturing)
35   also affects the potential upward migration of gas and its arrival times at the drinking water
36   formation due to its effect on the driving forces (e.g., pressure gradient). Similar to the
37   fault/fracture cases described in Section 6.3.2.2, production in the target reservoir appears to
38   mitigate upward gas migration, both by reducing the amount of gas that might otherwise be

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[1] An orphaned well is an inactive oil or gas well with no known (or financially solvent) owner.

1   available for upward migration and creating a pressure gradient toward the production well. Only
2   scenarios without the mitigating feature of gas production result in any upward migration into the
3   aquifer. This assessment also found a generally downward water flow within the connecting well
4   pathway, which is more pronounced when the gas well is operating. The producing formation and
5   aquifer permeabilities appear not to be significant factors for upward gas migration via this
6   pathway. In addition, Reagan et al. (2015) found the permeability of the connecting offset well to be
7   one of the main factors affecting the migration of gas to the aquifer and the water well. Very low
8   permeabilities (less than 1 mD) lead to no migration of gas into the aquifer regardless of the
9   vertical separation distance, whereas larger permeabilities present a greater potential for gas
10  breakthrough.

11  In the same way that fractures can propagate to intersect offset wells, they can also potentially
12  intersect other artificial subsurface structures including mine shafts or solution mining sites. No
13  known incidents of this type of migration have been documented. However, the Bureau of Land
14  Management (BLM) has identified over 28,000 abandoned mines in the United States and is adding
15  new mines to its inventory every year (BLM, 2013a). In addition, the Well File Review identified an
16  estimated 800 cases where wells used for hydraulic fracturing were drilled through mining voids,
17  and an additional 90 cases of drilling through gas storage zones or wastewater disposal zones (U.S.
18  EPA, 2015o). The analysis suggests that emplacing cement within such zones may be challenging,
19  which, in turn, could lead to a loss of zonal isolation (as described in Section 6.2) and create a
20  pathway for fluid migration.

### 6.3.2.4.  Migration via Fractures Intersecting Geologic Features

21  Potential fluid migration via natural fault or fracture zones in conjunction with hydraulic fracturing
22  has been recognized as a potential contamination hazard for several decades (Harrison, 1983).
23  While porous flow in unfractured shale or tight sand formations is assumed to be negligible due to
24  very low formation permeabilities (as discussed in Section 6.3.2.1), the presence of natural
25  "microfractures" within tight sand or shale formations is widely recognized, and these fractures
26  affect fluid flow and production strategies. Naturally occurring permeable faults and larger scale
27  fractures within or between formations may allow for more significant flow pathways for migration
28  of fluids out of the production zone (Jackson et al., 2013c; Myers, 2012a). Figure 6-4 illustrates the
29  concept of induced fractures intersecting with natural faults or fractures extending out of the target
30  reservoir.

31  Natural fracture systems have a strong influence on the success of a fracture treatment, and the
32  topic has been studied extensively from the perspective of optimizing treatment design (e.g., Weng
33  et al., 2011; Dahi Taleghani and Olson, 2009; Vulgamore et al., 2007). Small natural fractures,
34  known as "microfractures," could affect fluid flow patterns near the induced fractures by increasing
35  the effective contact area. Conversely, the natural microfractures could act as capillary traps for the
36  fracturing fluid during treatment (contributing to fluid leakoff) and potentially hinder hydrocarbon
37  flow due to lower gas relative permeabilities (Dahi Taleghani et al., 2013). Rutledge and Phillips
38  (2003) suggested that, for a hydraulic fracturing operation in East Texas, pressurizing existing
39  fractures (rather than creating new hydraulic fractures) may be the primary process that controls

1    enhanced permeability and fracture network conductivity at the site. Ciezobka and Salehi (2013)
2    used microseismic data to investigate the effects of natural fractures in the Marcellus Shale and
3    concluded that fracture treatments are more efficient in areas with clusters or "swarms" of small
4    natural fractures, while areas without these fracture swarms require more thorough stimulation.
5    However, there is very little attention given in the literature to studying unintended fluid migration
6    during hydraulic fracturing operations due to existing microfractures.

7    In some areas, larger-scale geologic features may affect potential fluid flow pathways. As discussed
8    in Text Box 6-2, baseline measurements taken before shale gas development show evidence of
9    thermogenic methane in some shallow aquifers, suggesting that natural subsurface pathways exist
10   and allow for naturally occurring migration of gas over millions of years (Robertson et al., 2012).
11   There is also evidence demonstrating that gas undergoes mixing in subsurface pathways
12   (Baldassare et al., 2014; Molofsky et al., 2013; Fountain and Jacobi, 2000). Warner et al. (2012)
13   compared recent sampling results to data published in the 1980s and found geochemical evidence
14   for migration of fluids through natural pathways between deep underlying formations and shallow
15   aquifers—pathways that the authors suggest could lead to contamination from hydraulic fracturing
16   activities. In northeastern Pennsylvania, there is evidence that brine from deep saline formations
17   has migrated into shallow aquifers over geologic time, preferentially following certain geologic
18   structures (Llewellyn, 2014). As described in Chapter 7, karst features (created by the dissolution
19   of soluble rock) can also serve as a potential pathway of fluid movement on a faster time scale.

20   Monitoring data show that the presence of natural faults and fractures can affect both the height
21   and width of hydraulic fractures. When faults are present, relatively larger microseismic responses
22   are seen and larger fracture growth can occur, as described below. Concentrated swarms of natural
23   fractures within a shale formation can result in a fracture network with a larger width-to-height
24   ratio (i.e., a shorter and wider network) than would be expected in a zone with a low degree of
25   natural fracturing (Ciezobka and Salehi, 2013).

26   A few studies have used monitoring data to specifically investigate the effect of natural faults and
27   fractures on the vertical extent of induced fractures. A statistical analysis of microseismic data by
28   Shapiro et al. (2011) found that fault rupture from hydraulic fracturing is limited by the extent of
29   the stimulated rock volume and is unlikely to extend beyond the fracture network. (However, as
30   demonstrated by microseismic data presented by Vulgamore et al. (2007), in some settings the
31   fracture network can extend laterally for thousands of feet.) In the Fisher and Warpinski (2012)
32   data set (see Section 6.3.2.2), the greatest fracture heights occurred when the hydraulic fractures
33   intersected pre-existing faults. Similarly, Hammack et al. (2014) reported that fracture growth seen
34   above the Marcellus Shale is consistent with the inferred extent of pre-existing faults at the Greene
35   County, Pennsylvania, research site (see Section 6.3.2.2 and Text Box 6-3). The authors suggested
36   that clusters of microseismic events may have occurred where preexisting small faults or natural
37   fractures were present above the Marcellus Shale. At a site in Ohio, Skoumal et al. (2015) found that
38   hydraulic fracturing induced a rupture along a pre-existing fault approximately 0.6 miles (1 km)
39   from the hydraulic fracturing operation. Using a new monitoring method known as tomographic
40   fracturing imaging, Lacazette and Geiser (2013) also found vertical hydraulic fracturing fluid

1 movement from a production well into a natural fracture network for distances of up to 0.6 miles
2 (1 km). However, Davies et al. (2013) questioned whether this technique actually measures
3 hydraulic fracturing fluid movement.

4 Modeling studies have also investigated whether hydraulic fracturing operations are likely to
5 reactivate faults and create a potential fluid migration pathway into shallow aquifers. Myers
6 (2012a, 2012b) found that a highly conductive fault could result in rapid (<1 year) fluid migration
7 from a deep shale zone to the surface (as described in Section 6.3.2.1). Other researchers reject the
8 notion that open, permeable faults would coexist with hydrocarbon accumulation (Flewelling et al.,
9 2013), although it is unclear whether the existence of faults in low permeability reservoirs would
10 affect the accumulation of hydrocarbons because, under natural conditions, the flow of gas may be
11 limited due to capillary tension. Results from another recent modeling study suggest that, under
12 specific circumstances, interaction with a conductive fault could result in fluid migration to the
13 surface only on longer (ca. 1,000 year) time scales (Gassiat et al., 2013). Rutqvist et al. (2013) found
14 that, while somewhat larger microseismic events are possible in the presence of faults, repeated
15 events and aseismic slip would amount to a total rupture length of 164 ft (50 m) or less along a
16 fault, not far enough to allow fluid migration between a deep gas reservoir and a shallow aquifer. A
17 follow-up study using more sophisticated three-dimensional modeling techniques also found that
18 deep hydraulic fracturing is unlikely to create a direct flow path into a shallow aquifer, even when
19 fracturing fluid is injected directly into a fault (Rutqvist et al., 2015). Similarly, a modeling study
20 that investigated potential fluid migration from hydraulic fracturing in Germany found potential
21 vertical fluid migration up to 164 ft (50 m) in a scenario with high fault zone permeability, although
22 the authors note this is likely an overestimate because their goal was to "assess an upper margin of
23 the risk" associated with fluid transport (Lange et al., 2013). More generally, results from Rutqvist
24 et al. (2013) indicate that fracturing along an initially impermeable fault (as is expected in a shale
25 gas formation) would result in numerous small microseismic events that act to prevent larger
26 events from occurring (and, therefore, prevent the creation of more extensive potential pathways).

27 Other conditions in addition to the physical presence of a pathway would need to exist for fluid
28 migration to a drinking water resource to occur. The modeling study conducted by (Reagan et al.,
29 2015) discussed in Section 6.3.2.2 indicates that, if such a permeable feature exists, the transport of
30 gas and fluid flow would strongly depend upon the production regime and, to a lesser degree, the
31 features' permeability and the separation between the reservoir and the aquifer. In addition, the
32 pressure distribution within the reservoir (e.g., over-pressurized vs. hydrostatic conditions) will
33 affect the fluid flow through fractures/faults. As a result, the presence of multiple natural and well-
34 based factors may increase the potential for fluid migration into drinking water resources. For
35 example, in the Mamm Creek area of Colorado (see Section 6.2.2.2), well integrity and drilling-
36 related problems likely acted in concert with natural fracture systems to result in a gas seep into
37 surface water and shallow ground water (Crescent, 2011).

## 6.4. Synthesis

38 In the injection stage of hydraulic fracturing, operators inject fracturing fluids into a well under
39 high pressure. These fluids flow through the well and into the surrounding formation, where they

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1   increase pore pressure and create fractures in the rock, allowing hydrocarbons to flow through the
2   fractures and up the well.

3   The production well and the surrounding geologic features function as a system that is often
4   designed with multiple elements that can isolate hydrocarbon-bearing zones and water-bearing
5   zones, including drinking water resources, from each other. This physical isolation optimizes oil
6   and gas production and can protect drinking water resources via isolation within the well (by the
7   casing and cement) and the presence of multiple layers of subsurface rock between the target
8   formations where hydraulic fracturing occurs and drinking water aquifers.

## 6.4.1.  Summary of Findings

9    Potential pathways for impacts on drinking water (i.e., the movement of hydrocarbons, formation
10   brines, or other fracturing-related fluids into drinking water resources), may be linked to one or
11   more components of the well and/or features of the subsurface system. If present, these potential
12   pathways can, in combination with the high pressures under which fluids are injected and pressure
13   changes within the subsurface, have an impact on drinking water resources.

14   The potential for these pathways to exist or form has been investigated through modeling studies
15   that simulate subsurface responses to hydraulic fracturing, and demonstrated via case studies and
16   other monitoring efforts. In addition, the development of some of these pathways—and fluid
17   movement along them—has been documented.

18   It is important to note that the development of one pathway within this system does not necessarily
19   result in an impact to a drinking water resource. For example, if cracks were to form in the cement
20   of a well, the vertical distance between the production zone and a drinking water resource (and the
21   multiple layers of rock in between) could isolate and protect the drinking water aquifer if pressures
22   were insufficient to allow fluid movement to the level of the drinking water resource. Conversely, if
23   an undetected fault were present in a rock formation, intact cement within the production well
24   could keep fluids from migrating up along the well to the fault and protect drinking water
25   resources.

### 6.4.1.1.  Fluid Movement via the Well

26   A production well undergoing hydraulic fracturing is subject to higher stresses during the relatively
27   brief hydraulic fracturing phase than during any other period of activity in the life of the well. These
28   higher stresses may contribute to the formation of potential pathways associated with the casing or
29   cement that can result in the unintentional movement of fluids through the production wellbore if
30   the well cannot withstand the stresses experienced during hydraulic fracturing operations (see
31   Section 6.2).

32   Multiple barriers within the well, including casing, cement, and a completion assembly, isolate
33   hydrocarbon-bearing formations from drinking water resources. However, inadequate
34   construction, defects in or degradation of the casing or cement, or the absence of redundancies such
35   as multiple layers of casing, can allow fluid movement, which can then affect the quality of drinking
36   water resources. Ensuring proper well design and mechanical integrity—particularly proper

1     cement placement and quality—are important actions for preventing unintended fluid migration
2     along the wellbore.

### 6.4.1.2. Fluid Movement within Subsurface Geologic Formations

3     Potential subsurface pathways for fluid migration include flow of fluids out of the production zone
4     into formations above or below it, fractures extending out of the production zone or into other
5     induced fracture networks, intersections of fractures with abandoned or active wells, and fractures
6     intersecting with faults or natural fractures (see Section 6.3).

7     Vertical separation between the production zone where hydraulic fracturing operations occur and
8     drinking water resources, and lateral separation between wells undergoing hydraulic fracturing
9     and other wells can reduce the potential for fluid migration that can impact drinking water
10    resources.

11    Well communication incidents or "frac hits" have been reported in New Mexico, Oklahoma, and
12    other locations. While some operators design fracturing treatments to communicate with the
13    fractures of another well and optimize production, unintended communication between two
14    fracture systems can lead to spills in the offset well and is an indicator of hydraulic fracturing
15    treatments extending beyond their planned design. Surface spills from well communication
16    incidents have been documented in the literature, which provides evidence for occurrence of frac
17    hits. Based on the available information, frac hits most commonly occur on multi-well pads and
18    when wells are spaced less than 1,100 ft (335 m) apart, but they have been observed at wells up to
19    8,422 ft (2,567 m) away from a well undergoing hydraulic fracturing.

### 6.4.1.3. Impacts to Drinking Water Resources

20    We identified an impact on drinking water resources associated with hydraulic fracturing
21    operations in Bainbridge, Ohio. Failure to cement over-pressured formations through which the
22    production well passed—and proceeding with the fracturing operation without adequate cement
23    and an extended period during which the well was shut in—led to a buildup of natural gas within
24    the well annulus and high pressures within the well. This ultimately resulted in movement of gas
25    from the production zone into local drinking water aquifers (see Section 6.2.2.2).

26    Casings at a production well near Killdeer, North Dakota, ruptured following a pressure spike
27    during hydraulic fracturing, allowing fluids to escape to the surface. Brine and tert-butyl alcohol
28    were detected in two nearby water wells. Following an analysis of potential sources, the only
29    potential source consistent with the conditions observed in the two impacted wells was the well
30    that ruptured. There is also evidence that out-of-zone fracturing occurred at the well (see Sections
31    6.2.2.1 and 6.3.2.2).

32    There are other cases where hydraulic fracturing could be a contributing cause to impacts on
33    drinking water resources, or where the specific mechanism that led to an impact on a drinking
34    water resource cannot be definitively determined. For example:

35        •    Migration of stray gas into drinking water resources involves many potential routes for
36           migration of natural gas, including poorly constructed casing and naturally existing or

1    induced fractures in subsurface formations. Multiple pathways for fluid movement may be
2    working in concert in northeastern Pennsylvania (possibly due to cement issues or
3    sustained casing pressure) and the Raton Basin in Colorado (where fluid migration may
4    have occurred along natural rock features or faulty well seals). While the sources of
5    methane identified in drinking water wells in each study area could be determined with
6    varying degrees of certainty, attempts to definitively identify the pathways of migration
7    have generally been inconclusive (see Text Box 6-2).

8  •  At the East Mamm Creek drilling area in Colorado, inadequate placement of cement
9      allowed the migration of methane through natural faults and fractures in the area. This
10     case illustrates how construction issues, sustained casing pressure, and the presence of
11     natural faults and fractures, in conjunction with elevated pressures associated with well
12     stimulation, can work together to create a pathway for fluids to migrate toward drinking
13     water resources (see Sections 6.2.2.2 and 6.3.2.4).

14   Additionally, some hydraulic fracturing operations involve the injection of fluids into formations
15   where there is relatively limited vertical separation from drinking water resources. The EPA
16   identified an estimated 4,600 wells that were located in areas with less than 2,000 ft (610 m) of
17   vertical separation between the fractures and the base of protected ground water resources.

18   There are places in the subsurface where oil and gas reservoirs and drinking water resources co-
19   exist in the same formation. Evidence we examined suggests that some hydraulic fracturing for oil
20   and gas occurs within formations where the ground water has a salinity of less than 10,000 mg/L
21   TDS. By definition, this results in the introduction of fracturing fluids into formations that meet the
22   Safe Drinking Water Act (SDWA) salinity-based definition of a source of drinking water and the
23   broader definition of a drinking water resource developed for this assessment. According to the
24   data we examined, these formations are generally in the western United States.

25   The practice of injecting fracturing fluids into a formation that also contains a drinking water
26   resource directly affects the quality of that water, since it is likely some of that fluid remains in the
27   formation following hydraulic fracturing. Hydraulic fracturing in a drinking water resource may be
28   of concern in the short-term (where people are currently using these zones as a drinking water
29   supply) or the long-term (if drought or other conditions necessitate the future use of these zones
30   for drinking water).

31   There are other cases in which production wells associated with hydraulic fracturing are alleged to
32   have caused drinking water contamination. Data limitations in most of those cases (including the
33   unavailability of information in litigation settlements resulting in sealed documents) make it
34   impossible to definitively assess whether or not hydraulic fracturing was a cause of the
35   contamination in these cases.

## 6.4.2.  Factors Affecting Frequency and Severity of Impacts

36   Proper cementing across oil-, gas-, or water-bearing zones prevents the movement of brines, gas, or
37   hydraulic fracturing fluids along the well into drinking water resources. The likelihood of
38   contamination is reduced when the well is fully cemented across these zones; however, this is not

1  the case in all hydraulically fractured wells, either because the cement does not extend completely
2  through the base of the drinking water resource or the cement that is present is not of adequate
3  quality. Fully cemented surface casing that extends through the base of drinking water resources is
4  a key protective component of the well. Most, but not all, wells used in hydraulic fracturing
5  operations have fully cemented surface casing.

6  Deviated and horizontal wells, which are increasingly being used in hydraulic fracturing operations,
7  may exhibit more casing and cement problems compared to vertical wells. Sustained casing
8  pressure—a buildup of pressure within the well annulus that can indicate the presence of small
9  leaks—occurs more frequently in deviated and horizontal wells compared to vertical wells. Cement
10  integrity problems can also arise as a result of challenges in placing cement in these wells, because
11  they are more challenging than vertical wells to center properly.

12  Older wells may exhibit more integrity problems compared to newer wells, which may be an issue
13  if older wells are hydraulically fractured or re-fractured. Degradation of the casing and cement as
14  they age or the cumulative effects of stresses exerted on the well over time may result in changes in
15  well integrity. Integrity problems can also be associated with the inadequate design of wells that
16  were constructed pursuant to older, less stringent requirements. Well components that are subject
17  to corrosive environments, high pressures, or other stressors tend to have more problems than
18  wells without these additional stressors.

19  The extent of subsurface fluid migration within subsurface rock formations and the potential for the
20  development of pathways that can adversely affect drinking water depend on site-specific
21  characteristics. These include the physical separation between the production zone and drinking
22  water resources, the geological and geomechanical characteristics of the formations, hydraulic
23  fracturing operational parameters, and the physical characteristics of any connecting feature (e.g.,
24  abandoned wells, faults, and natural fractures).

25  As noted above, vertical separation between the production zone and drinking water resources
26  protects drinking water. Additionally, the proximity of wells undergoing hydraulic fracturing to
27  other wells increases the potential for the formation of pathways for fluids to move via these wells
28  to drinking water resources. For example, if there is a deficiency in the construction of a nearby
29  well (or degradation of the well components), that well could serve as a pathway for movement of
30  fracturing fluids, methane, or brines that might affect a drinking water resource. If the fractures
31  were to intersect an uncemented portion of a nearby wellbore, the fluids could migrate along that
32  wellbore into any uncemented formations.

33  Fractures created during hydraulic fracturing can extend out of the target production zone. Out-of-
34  zone fracturing could be a concern for fluid migration if the hydraulic fracturing operation is not
35  designed to address site-specific conditions, for example if the production zone is thin and fractures
36  propagate to unintended vertical heights, or if the production zone is not horizontally continuous
37  and fractures extend to unintended horizontal lengths. The presence of natural faults or fractures
38  can affect the extent of hydraulic fractures. When faults are present, relatively larger microseismic
39  responses are seen during hydraulic fracturing, and larger fracture growth can occur than in the

1  absence of natural faults or fractures. However, modeling studies indicate that fluid migration from
2  deep production zones to shallow drinking water resources along natural faults and fractures or
3  offset wells is unlikely. These studies indicate that, in both cases, gas available for migration is
4  limited to the amount that existed in the fractures and pore space of connecting features following
5  hydraulic fracturing prior to production. Following the completion of a hydraulic fracturing
6  treatment, depressurization of the production formation surrounding the fractures due to
7  hydrocarbon production would make upward fluid migration into drinking water resources
8  unlikely to occur.

9  Based on the information presented in this chapter, the increased deployment of hydraulic
10  fracturing associated with oil and gas production activities, including techniques such as horizontal
11  drilling and multi-well pads, may increase the likelihood that these pathways could develop. This, in
12  turn, could lead to increased opportunities for impacts on drinking water resources.

### 6.4.3. Uncertainties

13  Generally, less is known about the occurrence of (or potential for) impacts of injection-related
14  pathways in the subsurface than for other components of the hydraulic fracturing water cycle,
15  which can be observed and measured at the surface. Furthermore, while there is a significant
16  amount of information available on production wells in general, there is little information that is
17  specific to hydraulic fracturing operations and much of this data is not readily accessible, i.e., in a
18  centralized, national database.

### 6.4.3.1. Limited Availability of Information Specific to Hydraulic Fracturing Operations

19  There is extensive information on the design goals for hydraulically fractured oil and gas wells (i.e.,
20  to address the stresses imposed by high-pressure, high-volume injection), including from industry-
21  developed best practices documents. Additionally, based on the long history of oil and gas
22  production activities, we know how production wells are constructed and have performed over
23  time. Over the years, many studies have documented how these wells are constructed, how they
24  perform, and the rates at which they experience problems that can lead to the formation of
25  pathways for fluid movement. However, because we do not know which of these wells were
26  hydraulically fractured, we cannot definitively determine whether the rates at which integrity
27  problems arise (or other data pertaining to oil and gas wells in general) directly correspond to
28  wells used in hydraulic fracturing operations.

29  Because wells that have been hydraulically fractured must withstand many of the same downhole
30  stresses as other production wells, we consider studies of the pathways for impacts to drinking
31  water resources in production wells to be relevant to identifying the potential pathways relevant to
32  hydraulic fracturing operations. However, without specific data on the as-built construction of wells
33  used in hydraulic fracturing operations, we cannot definitively state whether these wells are
34  consistently constructed to meet the stresses they may encounter.

35  There is also, in general, very limited information available on the monitoring and performance of
36  wells used in hydraulic fracturing operations. Published information is sparse regarding
37  mechanical integrity tests (MITs) performed during and after hydraulic fracturing, including MIT

1    results, the frequency at which mechanical integrity issues arise in wells used for hydraulic
2    fracturing, and the degree and speed with which identified issues are addressed. There is also little
3    information available regarding MIT results for the original hydraulic fracturing in wells built for
4    that purpose, for wells that are later re-fractured, or for existing, older wells not initially
5    constructed for hydraulic fracturing but repurposed for that use.

6    There are also a limited number of published monitoring studies or sampling data that provide
7    evidence to assess whether formation brines, injected fluids, or gas move in unintended ways
8    through the subsurface during and after hydraulic fracturing. Subsurface monitoring data (i.e., data
9    that characterize the presence, migration, or transformation of fluids in the subsurface related to
10   hydraulic fracturing operations) are scarce relative to the tens of thousands of oil and gas wells that
11   are estimated to be hydraulically fractured across the country each year (see Chapter 2).

12   Information on fluid movement within the subsurface and the extent of fractures that develop
13   during hydraulic fracturing operations is also limited. For example, limited information is available
14   in the published literature on how flow regimes or other subsurface processes change at sites
15   where hydraulic fracturing is conducted. Instead, much of the available research, and therefore the
16   literature, addresses how hydraulic fracturing and other production technologies perform to
17   optimize hydrocarbon production.

18   These limitations on hydraulic fracturing-specific information make it difficult to provide definitive
19   estimates of the rate at which wells used in hydraulic fracturing operations experience the types of
20   integrity problems that can contribute to fluid movement.

### 6.4.3.2.  Limited Systematic, Accessible Data on Well Performance or Subsurface Movement

21   While the oil and gas industry generates a large amount of information on well performance as part
22   of operations, most of this is proprietary or otherwise not readily available to states or the public in
23   a compiled or summary manner. Therefore, no national or readily accessible way exists to evaluate
24   the design and performance of individual wells or wells in a region, particularly in the context of
25   local geology or the presence of other wells and/or hydraulic fracturing operations. Many states
26   have large amounts of operator-submitted data, but information about construction practices or the
27   performance of individual wells is typically not in a searchable or aggregated form that would
28   enable assessments of well performance under varying settings, conditions, or timeframes.
29   Although it is collected in some cases, there is also no systematic collection, reporting, or publishing
30   of empirical baseline (pre-drilling and/or pre-fracturing) and post-fracturing monitoring data that
31   could indicate the presence or absence of hydraulic fracturing-related fluids in shallow zones and
32   whether or not migration of those fluids has occurred. Ideally, data from ground water monitoring
33   are needed to complement theories and modeling on potential pathways and fluid migration.

34   While some of the types of impacts described above may occur quickly (i.e., on the scale of days or
35   weeks, as with integrity problems or well communication events), other impacts (e.g., in slow-
36   moving, deep ground waters) may only occur or be able to be detected on much longer timescales.
37   Given the surge in the number of modern high-pressure hydraulic fracturing operations dating
38   from the early 2000s, evidence of any fracturing-related fluid migration affecting a drinking water

1  resource (as well as the information necessary to connect specific well operation practices to a
2  drinking water impact) could take years to discover.

3  The limited amount of information hinders our ability to evaluate whether—or how frequently—
4  drinking water impacts are occurring (or the potential for these impacts to occur) or to tie possible
5  impacts to specific well construction, operation, or maintenance practices. This also significantly
6  limits our ability to evaluate the aggregate potential for hydraulic fracturing operations to affect
7  drinking water resources or to identify the potential cause of drinking water contamination or
8  suspected contamination in areas where hydraulic fracturing occurs.

### 6.4.4.  Conclusions

9  Fluids can migrate from the wellbore and surrounding subsurface formations due to inadequate
10  casing or cement, and via natural and man-made faults, fractures, and offset wells or mines (see
11  Text Box 6-5). To prevent fluid migration through the wellbore or through subsurface pathways,
12  wells must have adequate casing and cement, and induced fractures must not intersect existing
13  fractures or permeable zones that lead to drinking water resources. Evidence shows that the quality
14  of drinking water resources may have been affected by hydraulic fracturing fluids escaping the
15  wellbore and surrounding formation in certain areas, although conclusive evidence is currently
16  limited.

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**Text Box 6-5. Research Questions Revisited.**

17  *How effective are current well construction practices at containing fluids—both liquids and gases—*
18  *before, during, and after fracturing?*

19  • Wells that were designed with uncemented intervals of casing across porous or permeable zones, wells in
20    which cementing does not resist formation or operational stresses, and wells in which cementing does
21    not meet design specifications have the potential to promote unintended subsurface fluid movement.
22    Even in optimally designed wells, metal casings and cement can degrade over time, either as a result of
23    aging or of exposure to stresses exerted over years of operations. See Section 6.2.2.2.

24  • We have limited information on the degree to which wells are designed and constructed with the
25    multiple layers of casing that can withstand hydraulic fracturing pressures and contact with injected and
26    produced fluids. We also are lacking information about whether wells have suitable cements that can
27    prevent fluid movement outside the wellbore and between the production zone and drinking water
28    resources. We also do not have information on the degree to which mechanical integrity is verified before
29    or after hydraulic fracturing operations. See Section 6.2.2.1.

30  *Can subsurface migration of fluids—both liquids and gases—to drinking water resources occur and*
31  *what local geologic or artificial features might allow this?*

32  • The presence of artificial penetrations, especially poorly constructed offset wells or undetected
33    abandoned wells, mines, or other subsurface structures, provides pathways that, in the presence of a
34    driving force, could allow for fluid movement to shallow geologic zones such as drinking water resources.
35    See Section 6.3.2.3.

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1  • Intersections of induced fractures with transmissive faults or naturally occurring fractures or
2    porous/permeable rock zones can allow fluids to move out of the targeted fracture areas. However,
3    modeling studies indicate that fluid migration from production zones to drinking water resources along
4    natural faults and fractures is unlikely. See Section 6.3.2.4.

5  • Some hydraulic fracturing operations involve the injection of fluids into formations where there is
6    relatively limited vertical separation from drinking water resources. Other hydraulic fracturing is
7    performed within formations that meet the SDWA or state salinity-based definition of a source of
8    drinking water, in addition to the broader definition of a drinking water resource developed for this
9    assessment. See Section 6.3.2.

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# Chapter 7

# Flowback and Produced Water

# 7. Flowback and Produced Water

## 7.1. Introduction

1   Water is a byproduct of oil and gas production. After hydraulic fracturing is completed, either in its
2   entirety or for a specified stage, the operator reduces the injection pressure. Water is allowed to
3   flow back from the well to prepare for oil or gas production. This return-flow water may contain
4   chemicals injected as part of the hydraulic fracturing fluid, chemicals characteristic of the
5   formation, hydrocarbons, and in-formation reaction and degradation products. Initially this water,
6   called flowback, is mostly fracturing fluid, but as time goes on, it becomes more similar to the
7   formation water. For formations containing saline water (brine), the salinity of the water increases
8   as time passes, marking the increased contact time with the formation and in some cases the flow of
9   formation water itself. This later stage water is called produced water, a term which can also refer
10   to flowback and produced water collectively.

11   Flowback and produced water are stored and accumulated at the surface for eventual reuse or
12   disposal. Typical storage facilities include open air impoundments and closed containers such as
13   those shown in Figure 1-1. Produced water is collected and may be taken to disposal wells,
14   recyclers, wastewater treatment plants, or in some cases the water may be left in pits to evaporate
15   or infiltrate. Flowback and produced water leaks can occur on the well pad as a result of human
16   error, failure of container integrity, equipment failure, communication between wells, pipeline
17   leaks, and blowouts.[1] Above-ground piping systems can connect multiple well pads to
18   impoundments, and piping or impoundments may leak. Much produced water is transported by
19   truck, and pad incidents leading to spills of produced water can occur when trucks are filled.
20   On-road accidents are also possible, some of which could release produced water loads to the
21   environment.

22   Impacts to drinking water resources can occur if spilled flowback or produced water enters surface
23   water bodies or aquifers. Environmental transport of chemical constituents depends on the
24   characteristics of the spill, the fluid (e.g., density, as for highly saline water), the chemicals, and the
25   environment. Attenuation processes (e.g., dilution, biodegradation of organics) in surface water and
26   aquifers tend to reduce concentrations.

27   We begin this chapter with a review of definitions for flowback and produced water. We then
28   discuss typical volumes of flowback and produced water on a per-well basis. This information is
29   aggregated to the state and basin level in Chapter 8. The characteristics of hydraulically fractured
30   shale, tight, and coalbed methane (CBM) formations are described. Spatial and temporal trends on
31   composition of produced water are illustrated with examples from the literature and data compiled
32   for this report. The processes controlling the chemical composition of produced water are
33   described in Appendix E. The potential for impacts on drinking water resources of flowback and
34   produced water are described based on reported spill incidents, contaminant transport principles,
35   and field study examples. The chapter concludes with a discussion of uncertainties and knowledge

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[1] For discussion of well communication, see Chapter 6.

1   gaps, factors that influence the severity of impacts, and a synthesis based on the EPA research
2   questions.

### 7.1.1.  Definitions

3   Multiple definitions exist for the terms flowback and produced water. These differing definitions
4   indicate challenges in determining the distinctions between the two terms or indicate that different
5   usage of the terms routinely occurs among various industry, private, and public groups. However,
6   the majority of produced water definitions are fundamentally similar. The following definition is
7   used in this report: water that flows from oil or gas wells.

8   This definition is similar to the definition used by the American Petroleum Institute (API):
9   "Produced water is any of the many types of water produced from oil and gas wells" (API, 2010b);
10  the definition used by the Department of Energy (DOE): "Produced water is water trapped in
11  underground formations that is brought to the surface along with oil or gas" (DOE, 2004), and a
12  similar definition used by the American Water Works Association (AWWA): "Produced water is the
13  combination of flowback and formation water that returns to the surface along with the oil and
14  natural gas" (AWWA, 2013). Produced water can variously refer to formation water, a mixture of
15  spent hydraulic fracturing fluid and formation water or returned hydraulic fracturing fluid. Thus
16  the term produced water is used when a distinction between fracturing fluid and formation water is
17  not necessary.

18  In general, the term flowback refers either to fluids predominantly containing hydraulic fracturing
19  fluid that returns to the surface or to a process used to prepare the well for production. Because
20  formation water can contact and mix with injection fluids, the distinction between returning
21  hydraulic fracturing fluid and formation water is not clear. In the early stages of operation,
22  however, a higher concentration of chemical additives is expected and later, water that is typical of
23  the formation (Stewart, 2013a). In most cases, a precise distinction between these waters is not
24  determined during operations.

25  Various definitions have been used for the term flowback. The American Petroleum Institute
26  defined flowback as "the fracture fluids that return to the surface after a hydraulic fracture is
27  completed," (API, 2010b) and the American Water Works Association used "fracturing fluids that
28  return to the surface through the wellbore after hydraulic fracturing is complete" (AWWA, 2013).
29  Other definitions include production of hydrocarbons from the well (Barbot et al., 2013; U.S. EPA,
30  2012f), or a time period (USGS, 2014f; Haluszczak et al., 2013; Warner et al., 2013b; Hayes and
31  Severin, 2012a; Hayes, 2009). As mentioned above, flowback can also be defined as a process used
32  to stimulate the well for production by allowing excess liquids and proppant to return to the
33  surface. Because we use existing literature in our review, we do not introduce a preferred definition
34  of flowback, but rather we mention the assumptions used by the author(s) we discuss.

## 7.1.  Volume of Hydraulic Fracturing Flowback and Produced Water

35  The characteristics and volume of flowback and produced water vary by well, formation, and time.
36  This section presents information on flowback and produced water volume over various time
37  scales, and where possible, on a per-well and per-formation basis.

1    The amount of flowback from a well varies and depends on several types of factors, including:
2    production, formation, and operational. Production factors include the amount of fluid injected,
3    type of hydrocarbon produced (gas or liquid), and location within the formation. Formation factors
4    include the formation pressure, interaction between the formation and injected fluid (capillary
5    forces) and reactions within the reservoir. Operational factors include loss of mechanical integrity
6    and subsurface communication between wells (U.S. GAO, 2012; Byrnes, 2011; DOE, 2011a; GWPC
7    and ALL Consulting, 2009; Reynolds and Kiker, 2003). The latter two factors might be indicated by
8    an unexpected increase in water production (Reynolds and Kiker, 2003).

9    The processes that allow gas and liquids to flow are related to the conditions along the faces of
10   fractures. Byrnes (2011) conceptualized fluid flow across the fracture face as being composed of
11   three phases. The first is characterized by forced imbibition of fluid into the reservoir and occurs
12   during and immediately following fracture stimulation. Second is an unforced imbibition following
13   stimulation where the fluid redistributes within the reservoir rock, due to capillary forces, when the
14   well is shut-in. The last phase consists of flow out of the formation when the well is opened and
15   pressure reduced in the borehole and fractures. The purpose of this phase is to recover as much of
16   the injected fluid as possible (Byrnes, 2011) in order to reduce high water saturations at the
17   fracture face and eventually allow higher gas flow rates. The length of the last phase and
18   consequently, the amount of water removed depends on factors such as the amount of injected
19   fluid, the permeability and effective permeability of the reservoir, capillary pressure properties of
20   the reservoir rock, the pressure near the fracture faces, and whether the well is flowing or shut in.[1]
21   The well can be shut in for varying time periods depending on operator scheduling, surface facility
22   construction or hookup, or other reasons.

### 7.1.1.  Flowback of Injected Hydraulic Fracturing Fluid

23   Generally, the fluid that initially returns to the surface has been attributed to a mixture of the
24   injected fracturing fluid, its transformation products, and the natural formation water. In some
25   cases, as shown below, the amount of flowback is greater than the amount of injected hydraulic
26   fracturing fluid and the additional water comes from the formation (Nicot et al., 2014) or an
27   adjacent formation (Arkadakskiy and Rostron, 2013a). Several authors used geochemical analyses
28   to postulate mixing between formation water and injected fluid in the Marcellus Shale (Engle and
29   Rowan, 2014; Barbot et al., 2013; Haluszczak et al., 2013); Rowan et al., 2015). These possible
30   explanations are summarized in a following section (see Section 7.6.4). Salinity increases in
31   flowback from highly saline formations, so it is not possible to specify precisely the amount of
32   injected fluids that return in the flowback (GWPC and ALL Consulting, 2009). Rather, such
33   estimates relate the amount of produced water measured at a given time after fracturing as a
34   percentage of the total amount of injected fluid.

35   Estimates vary but in composite indicate on average that between 5% and 75% (see Table 7-1,
36   Table 7-2, and Table 7-3) of the volume of injected fracturing fluid may flow back to the surface

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[1] When multiple fluids (water, oil, gas) occupy portions of the pore space, the permeability to each fluid depends on the
fraction of the pore space occupied by the fluid and the fluid's properties. As defined by Dake (1978), when this effective
permeability is normalized by the absolute permeability, the resulting relationship is known as the relative permeability.

1  after hydraulic fracturing is complete (U.S. EPA, 2015q; Vengosh et al., 2014; Mantell, 2013b; Vidic
2  et al., 2013; Minnich, 2011; Xu et al., 2011). These data (see Table 7-1) illustrate that the formations
3  differ in their water requirements for hydraulic fracturing and generation of produced water over
4  the short term.[1] Low percentages of flowback are typical, as is the decrease of flowback volume
5  with time as the wells enter the production phase (Gregory et al., 2011; McElreath, 2011; GWPC and
6  ALL Consulting, 2009). Some formations produce higher volumes, as noted for the Barnett Shale in
7  Texas (Nicot et al., 2014) and discussed below.

**Table 7-1. Data from one company's operations indicating approximate total water use and approximate produced water volumes within 10 days after completion of wells (Mantell, 2013b).**

| Formation | Approx. total average water use per well (million gal) | Produced water within the first 10 days after completion | | Produced water as a percentage of average water use per well | |
|---|---|---|---|---|---|
| | | Low estimate (million gal) | High or only estimate (million gal) | Low estimate (% of total water use) | High or only estimate (% of total water use) |
| *Gas shale plays (primarily dry gas)* | | | | | |
| Barnett[a] | 3.4 | 0.3 | 1.0 | 9 | 29 |
| Marcellus[a] | 4.5 | 0.3 | 1.0 | 7 | 22 |
| Haynesville | 5.4 | -- | 0.25 | -- | 5 |
| *Liquid plays (gas, oil, condensate)* | | | | | |
| Mississippi Lime | 2.1 | -- | 1.0 | -- | 48 |
| Cleveland/Tonkaw | 2.7 | 0.3 | 1.0 | 11 | 37 |
| Niobrara | 3.7 | 0.3 | 1.0 | 8 | 27 |
| Utica | 3.8 | 0.3 | 1.0 | 8 | 26 |
| Granite Wash | 4.8 | 0.3 | 1.0 | 6 | 21 |
| Eagle Ford | 4.9 | 0.3 | 1.0 | 6 | 20 |

[a] Mantell (2011) reported produced water for the first 10 days at 500,000 to 600,000 gal for the Barnett, Fayetteville and Marcellus Shales.

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[1] Flowback estimates may be based on specific time periods (e.g., the flowback during the first 10, 15, or 30 days).

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**Table 7-2. Additional short-, medium-, and long-term produced water estimates.**

| Location–formation | Produced water as percentage of injected fluid | Reference | Comment |
|---|---|---|---|
| *Estimates without reference to a specific data set* | | | |
| Unspecified Shale | 5% – 35% | Hayes (2011) | |
| Marcellus Shale | 10% – 25% | Minnich (2011) | Initial flowback |
| ND–Bakken | 25% | EERC (2013) | |
| *Estimates with reference to specific data evaluation* | | | |
| *Short duration* | | | |
| Marcellus Shale | 10% | Clark et al. (2013) | 0 – 10 days |
| TX—Barnett | 20% | Clark et al. (2013) | 0 – 10 days |
| TX—Haynesville | 5% | Clark et al. (2013) | 0 – 10 days |
| AR—Fayetteville | 10% | Clark et al. (2013) | 0 – 10 days |
| *Mid duration* | | | |
| WV—Marcellus | 8% | Hansen et al. (2013) | 30 days |
| Marcellus Shale | 24% | Hayes (2011, 2009) | Average from 19 wells, 90 days |
| *Long duration* | | | |
| TX—Barnett | ~100%[a] | Nicot et al. (2014) | 72 months |
| WV—Marcellus | 10% – 30% | Ziemkiewicz et al. (2014) | Up to 115 months |
| TX—Eagle Ford | <20% | Nicot and Scanlon (2012) | Lifetime |
| *Unspecified duration* | | | |
| PA—Marcellus | 6% | Hansen et al. (2013) | |

[a] Approximate median with large variability: 5th percentile of 20% and 90th percentile of 350%.

**Table 7-3. Flowback and long-term produced water characteristics for wells in unconventional formations (U.S. EPA, 2015e).**

Source: (U.S. EPA, 2015q).

| Resource type | Drill type | Fracturing fluid (million gal) | | | Flowback (percent of fracturing fluid returned) | | |
|---|---|---|---|---|---|---|---|
| | | Median | Range | Number of data points | Median | Range | Number of data points |
| **Shale** | Horizontal | 4.0 | 0.13–15 | 50,053 | 6% | 1% – 50% | 6,488 |
| | Directional | 1.6 | 0.051–12 | 124 | 14% | 4% – 31% | 19 |
| | Vertical | 1.2 | 0.015–22 | 4,152 | 24% | 7% – 75% | 18 |
| **Tight** | Horizontal | 2.2 | 0.042–9.4 | 765 | 7% | 7% – 60% | 39 |
| | Directional | 0.60 | 0.056–4.0 | 693 | 6% | 0% – 60% | 263 |
| | Vertical | 0.31 | 0.019–4.0 | 1,287 | 8% | 1% – 83% | 48 |

*Long-term produced water (gal/day per well)*

| Resource type | Drill type | Median | Range | Number of data points |
|---|---|---|---|---|
| **Shale** | Horizontal | 900 | 0–19,000 | 22,222 |
| | Directional | 480 | 22–8,700 | 695 |
| | Vertical | 380 | 0–4,600 | 12,393 |
| **Tight** | Horizontal | 620 | 0–120,000 | 2,394 |
| | Directional | 750 | 12–1,800 | 3,816 |
| | Vertical | 570 | 0–4,000 | 21,393 |

1    In the following subsections, we first discuss water produced during the flowback period, then
2    longer-term produced water.

### 7.1.1.1.  Produced Water during the Flowback Period

3    Data were collected from six vertical and eight horizontal wells in the Marcellus Shale of
4    Pennsylvania and West Virginia (Hayes, 2009). The author collected samples of flowback after one,

1    five, and 14 days after hydraulic fracturing was completed, as well as a produced water sample 90
2    days after completion of the wells. Both the vertical and horizontal wells showed their largest
3    volume of flowback between one and five days after fracturing, as shown in Figure 7-1. The wells
4    continued to produce water, and at 90 days, samples were available from four each of the
5    horizontal and vertical wells. The vertical wells produced on average 180 bbl/day (7,600 gal/day or
6    29,000 L/day) and the horizontal wells a similar 200 bbl/day (8,400 gal/day or 32,000 L/day).
7    Results from one Marcellus Shale study were fitted to a power curve, as shown in Figure 7-2
8    (Ziemkiewicz et al., 2014). These and the Hayes (2009) data show decreasing flowback with time.
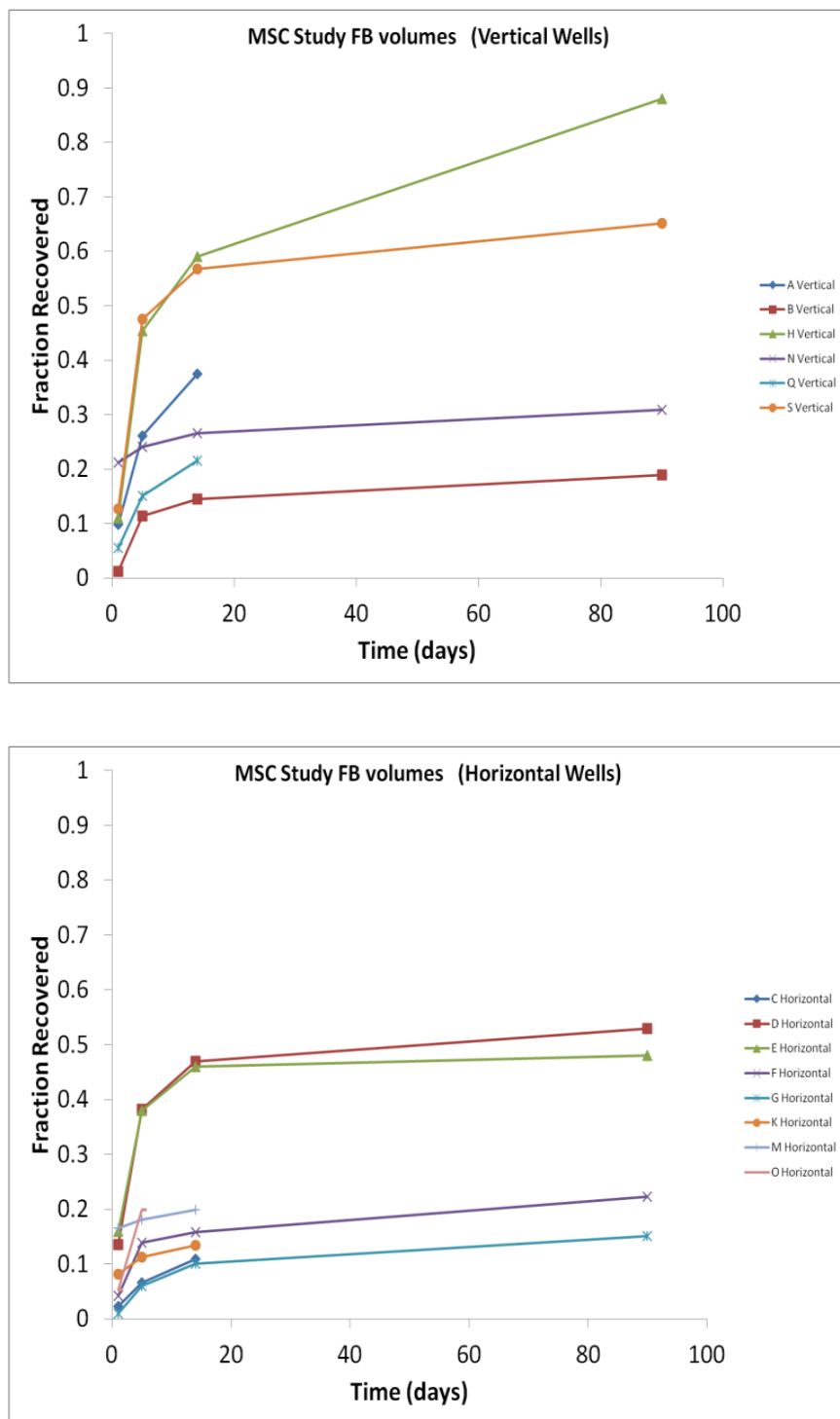
**Figure 7-1. Fraction of injected hydraulic fracturing fluid recovered from six vertical (top) and eight horizontal (bottom) wells completed in the Marcellus Shale.**

Data from Hayes (2009).
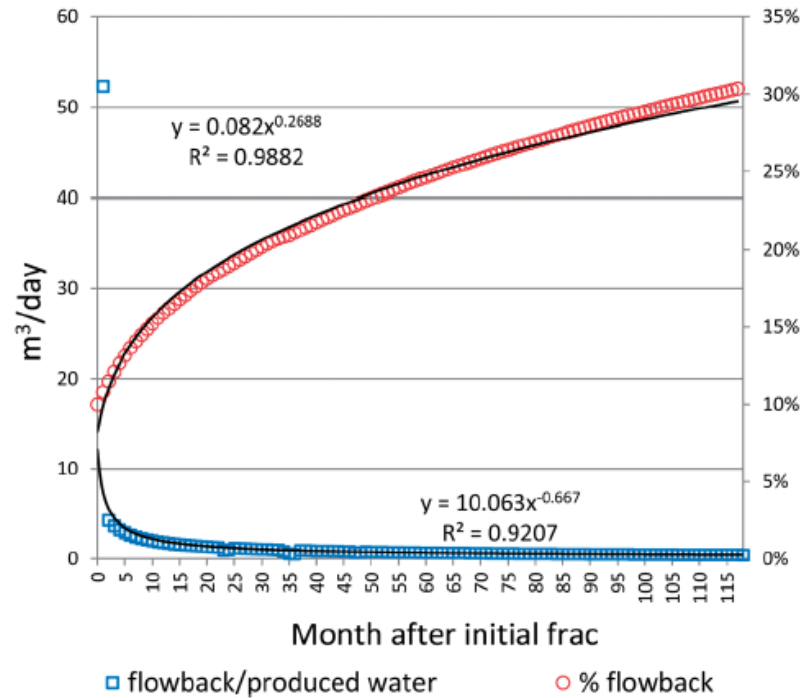
**Figure 7-2. Example of flowback and produced water from the Marcellus Shale, illustrating rapid decline in water production and cumulative return of approximately 30% of the volume of injected fluid.**

Source: Ziemkiewicz et al. (2014). Reprinted with permission from Ziemkiewicz, P; Quaranta, JD; Mccawley, M. (2014). Practical measures for reducing the risk of environmental contamination in shale energy production. Environmental Science: Processes & Impacts 16: 1692-1699. Reproduced by permission of The Royal Society of Chemistry. http://dx.doil.org/10.1039/C3EM00510K.

1  In West Virginia, water recovered at the surface within 30 days following injection or before 50% of
2  the injected fluid volume is returned to the surface is reported as flowback. Data from 271 wells in
3  the Marcellus Shale in West Virginia (Hansen et al., 2013) reveals the variability of recovery from
4  wells in the same formation and that the amount of injected fluid recovered was less than 15%
5  from over 80% of the wells (see Figure 7-3).
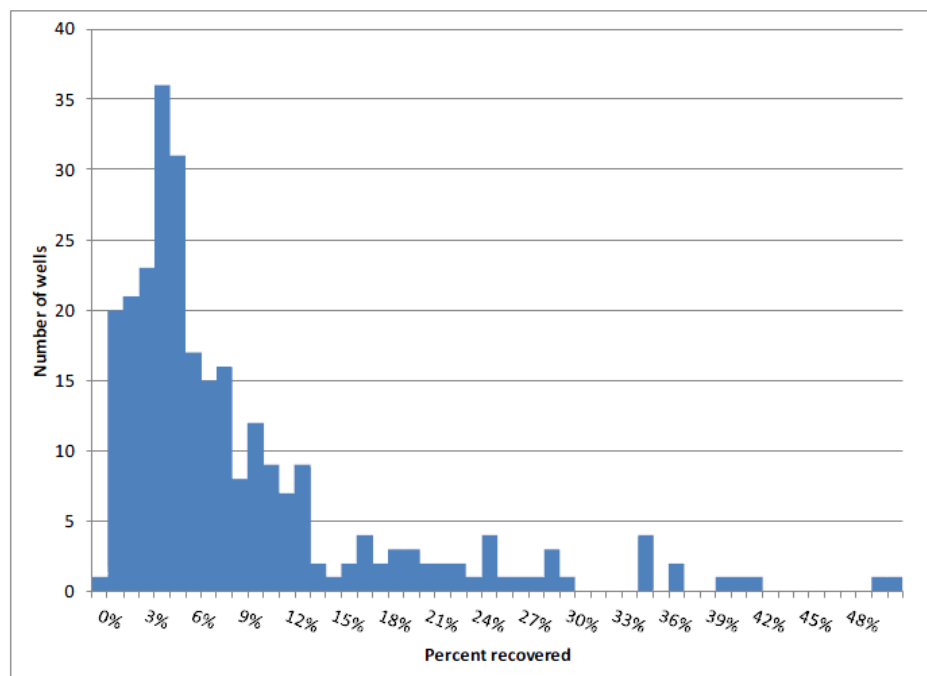
**Figure 7-3. Percent of injected fluid recovered for Marcellus Shale wells in West Virginia (2010–2012).**

Source: Hansen et al. (2013). One data point showing 98% recovery omitted. Reprinted with permission from Hansen, E; Mulvaney, D; Betcher, M. (2013). Water resource reporting and water footprint from Marcellus Shale development in West Virginia and Pennsylvania. Durango, CO: Earthworks Oil & Gas Accountability Project. Copyright 2013. Permission Downstream Strategies, San Jose State University, and Earthworks Oil & Gas Accountability Project.

1   The amount of flowback water produced by wells within the first few days of fracturing varies from
2   formation to formation. Wells in the Mississippi Lime and Permian Basin can produce 10 million gal
3   (37.8 million L) in the first 10 days of production. Wells in the Barnett, Eagle Ford, Granite Wash,
4   Cleveland/Tonkawa Sand, Niobrara, Marcellus, and Utica Shales can produce 300,000 to 1 million
5   gal (1.14 to 3.78 million L) within the first 10 days; while Haynesville wells produce less, about
6   350,000 gal (1.32 million L) (Mantell, 2013b).

## 7.1.2.  Produced Water

7   During oil and gas production, other fluids which contain water are produced with hydrocarbons.
8   Throughout this production phase at oil and certain wet gas production facilities, produced water is
9   stored in tanks and pits that may contain free phase, dissolved phase, and emulsified crude oil in
10  the produced water.[1] This crude oil can be present in the produced water container or pit, because
11  the crude oil is not efficiently separated out by the flow-through process vessels (such as three-
12  phase separators, heater treaters, or gun barrels) and passes through to these containers/pits. The

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[1] Dry natural gas occurs in the absence of liquid hydrocarbons; wet natural gas typically contains less than 85% methane along with ethane and more complex hydrocarbons (Schlumberger, 2014).

1   produced water containers and pits containing oil at production facilities are typically regulated
2   under 40 CFR part 112, produced water containers or pits may also be subject to other applicable
3   state and or local laws, regulations and/or ordinances.

4   Lutz et al. (2013) evaluated data reported to the Pennsylvania Department of Environmental
5   Protection (PA DEP) for the time period January 2000 to December 2011. The data were divided
6   between conventional gas wells that might have been hydraulically fractured and hydraulically
7   fractured completions in the Marcellus Shale. The conventional wells produced less drilling water,
8   less flowback (when fractured), and less brine than the shale wells (Lutz et al., 2013; see Table 1).
9   The average amount of produced water per well was 136,000 gal (514,000 L) for the conventional
10  wells and 1.38 million gal (5.211 million L) for the shale wells. The produced water to gas ratio was
11  1.27 gal (4.8 L) water per MMBtu for the shale wells, which was 2.8 times lower than for
12  conventional wells. Both the produced water and gas produced per well decreased over the four-
13  year period covered by the study. In contrast, conventional oil wells tend to have increased volumes
14  of produced water as they age, and in some cases, older wells may produce five times as much
15  water as new wells (U.S. GAO, 2012).

16  From experience in several shale formations, Mantell (2013b, 2011) characterized the amount of
17  produced water over the long term as high, moderate, or low. Wells in the Barnett Shale,
18  Cleveland/Tonkawa Sand, Mississippi Lime, and the Permian Basin can produce more than
19  1,000 gal (3,800 L) of water per million cubic feet (MMCF) of gas because of formation
20  characteristics. The most productive of these can be as high as 5,000 gal (19,000 L) per MMCF. As a
21  specific example, a high-producing formation in the western United States was described as
22  producing 4,200 gal (16,000 L) per MMCF for the life of the well (McElreath, 2011). The well was
23  fractured and stimulated with about 4 million gal (15 million L) of water and returned
24  60,000 gal (230,000 L) per day in the first 10 days, followed by 8,400 gal (32,000 L) per day in the
25  remainder of the first year.

26  Similarly, produced water from horizontal wells in the Barnett Shale decreased rapidly after the
27  wells began producing gas (Nicot et al., 2014) (see Figure 7-4). The data show a high degree of
28  variability, which was attributed by Nicot et al. (2014) to a few wells with exceptionally high water
29  production. When the produced water data were presented as the percentage of injected fluid, the
30  median exceeded 100% at around 36 months, and the 90th percentile was 350% (see Figure 7-5).
31  This means that roughly 50% of the wells were producing more water than was used in stimulating
32  production. Nicot et al. (2014) noted an inverse relationship between gas and water production but
33  did not identify the source or mechanism for the excess water. Systematic breaching of the
34  underlying karstic Ellenburger Formation was not believed likely; nor was operator efficiency or
35  skill. A number of geologic factors that could impact water migration were identified by (DOE,
36  2011a) in the Barnett Shale, including fracture height, aperture size, and density, fracture
37  mineralization, the presence of karst chimneys underlying parts of the Barnett Shale, and others,
38  but the impact of these on water migration was undetermined.
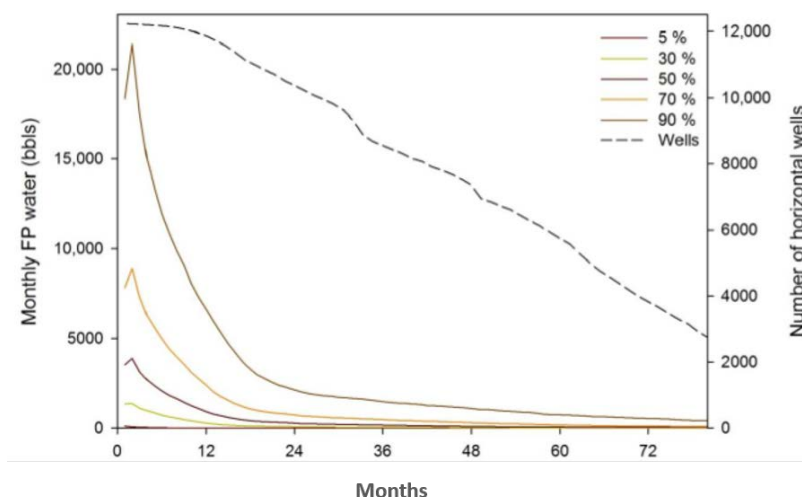
**Figure 7-4. Barnett Shale monthly water-production percentiles (5th, 30th, 50th, 70th, and 90th) and number of wells with data (dashed line).**

Source: Nicot et al. (2014). FP is the amount of water the flows back to the surface, commingled with water from the formation. Reprinted with permission from Nicot, JP; Scanlon, BR; Reedy, RC; Costley, RA. (2014). Source and fate of hydraulic fracturing water in the Barnett Shale: a historical perspective. Environ Sci Technol 48: 2464-2471. Copyright 2014 American Chemical Society.
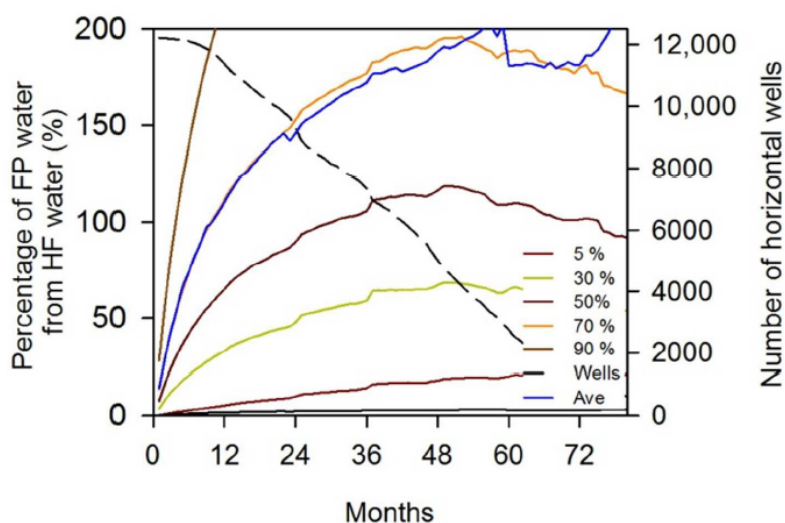


**Figure 7-5. Barnett Shale production data for approximately 72 months.**

Source: Nicot et al. (2014). Flowback and produced water are reported as the percentage of injected fluid. The dashed line shows the number of horizontal wells included. Data for each percentile show declining production with time, but the median production exceeds 100% of the injected fluid. FP is the amount of water the flows back to the surface, commingled with water from the formation. Reprinted with permission from Nicot, JP; Scanlon, BR; Reedy, RC; Costley, RA. (2014). Source and fate of hydraulic fracturing water in the Barnett Shale: a historical perspective. Environ Sci Technol 48: 2464-2471. Copyright 2014 American Chemical Society.

1   The Niobrara, Granite Wash, Eagle Ford, Haynesville, and Fayetteville Shales are relatively dry and
2   produce between 200 and 2,000 gal (760 to 7,600 L) of produced water per MMCF (Mantell, 2013).
3   The Utica and Marcellus Shales are viewed as drier and produce less than 200 gal (760 L)
4   per MMCF. DOE (2011a) concluded that the characteristic small amount of produced water from
5   the Marcellus Shale was due either to its low water saturation or low relative permeability to water
6   (see Chapter 6). For dry formations, low shale permeability and high capillarity cause water to
7   imbibe into the formation, where it is retained permanently (He, 2011). Engelder (2012) estimated
8   that more than half of the fracturing fluid could be captured within the Marcellus if imbibition
9   drove fluid just 5 cm (2 in) deep into reservoir rocks across the fracture surfaces. This estimate is in
10  agreement with the generalized analysis presented by Byrnes (2011), who estimated depths of 5 to
11  15 cm (2 to 6 in).

12  After fracture of coalbeds, water is withdrawn to liberate gas. CBM tends to produce large volumes
13  of water early on: more in fact, than conventional gas-bearing formations (U.S. GAO, 2012). Within
14  producing formations, water production can vary for unknown reasons (U.S. GAO, 2012). Data show
15  that CBM production in the Powder River Basin produces 16 times more water than in the San Juan
16  Basin (U.S. GAO, 2012).

17  The EPA (2015q) reported characteristics of long-term produced water for shale and tight
18  formations (see Table 7-3). For shale, horizontal wells produced more water (900 gal/day) than
19  vertical wells (380 gal/day). Typically, this would be attributed to the longer length of horizontal
20  laterals than vertical wells, but the data were not normalized to these lengths. The formation-level
21  data used to develop Table 7-3 appear in Table E-1 of Appendix E.

22  The EPA (2015q) reported that a general rule of thumb is that flowback occurring in the first 30
23  days of production is roughly equal to the long-term produced water for unconventional
24  formations. As a specific example, from Pennsylvania Marcellus Shale data, the EPA determined that
25  for vertical wells in unconventional formations, 6% of water came from drilling, 35% from
26  flowback, and 59% from long-term produced water; and for horizontal wells the corresponding
27  numbers were 9%, 33%, and 58%. These values deviate from the rule of thumb, because the
28  Marcellus Shale was believed to generate low levels of flowback relative to other formations (U.S.
29  EPA, 2015q).

## 7.2.   Flowback and Produced Water Data Sources

30  Unlike the evaluation of hydraulic fracturing fluid itself where the chemical composition may be
31  disclosed, knowledge concerning flowback and produced water composition comes from
32  measurements made on samples.

33  A number of factors are involved in the proper sampling and analyzing of environmental media
34  (U.S. EPA, 2013e; ATSDR, 2005; U.S. EPA, 1992). There may be significant issues obtaining samples,
35  because the specialized equipment used to contain high-pressure natural gas is not designed for
36  producing environmental samples (Coleman, 2011).

1    To choose the correct analytical methods, it is necessary to have information on:

2    •    Physical state of the sample.

3    •    Identification of analytes of interest.

4    •    Required sensitivity and quantitation limits.

5    •    Analytical objective (i.e., unknown identification, monitoring).

6    •    Required sample containers, preservation, and holding times.

7    Because some components of hydraulic fracturing fluid are proprietary chemicals, and subsurface
8    reaction products may be unknown, prior knowledge of the identity of analytes may not be
9    available. Consequently, studies may be limited in their ability to determine the presence of either
10   unknown or proprietary constituents contained in flowback or produced water simply because of
11   the lack of knowledge of the identities of the constituents.

12   After laboratory analysis, the results are evaluated according to quality criteria. Data may be judged
13   to meet applicable quality criteria as determined by the analytical methods or they may be
14   "flagged." Typically, encountered flags are non-detect, below reporting limit or diluted to meet
15   calibration requirements or because of matrix interference (e.g., Hayes, 2009).[1] For produced
16   water, a primary interference is from high total dissolved solids (TDS). Interferences also arise from
17   agents which cause foaming and alter surface tension (Coleman, 2011). Diluted samples result in
18   higher detection limits, and thus lessen ability to identify lower concentrations in samples.

19   Because of identified limitations in existing methods, the EPA developed new methods for some
20   reported components of hydraulic fracturing fluids, including ethanols and glycols (U.S. EPA,
21   2014k), certain nonionic surfactants (DeArmond and DiGoregorio, 2013a), and acrylamide
22   (DeArmond and DiGoregorio, 2013b).[2,3] Each of these methods are applicable to ground and surface
23   waters, and the last (DeArmond and DiGoregorio, 2013b) to waters with TDS well above 20,000
24   mg/L.

25   Generally, analytical methods are impacted by elevated TDS and chloride concentrations, especially
26   inorganic and wet chemistry methods (Nelson et al., 2014; U.S. EPA, 2014b; Coleman, 2011). Matrix
27   interference impacts standard analysis (EPA Method 8015) for glycols, resulting in high detection
28   limits (10,000 μg/L to 50,000 μg/L) (Coleman, 2011).

29   Produced water levels of naturally occurring radionuclides may be 1,000 to 10,000 times the levels
30   of activity found in typical environmental water samples (U.S. EPA, 2014b). The standard EPA
31   method (Method 900.0) for gross alpha and gross beta involves evaporation of the sample to a layer
32   of residue and analyzing emitted alpha and beta particles. The method has several noted

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[1] Matrix interference occurs when components of the sample other than the analyte of interest have an effect on a measurement (IUPAC, 2014).
[2] The compounds included were: Diethylene Glycol, Triethylene Glycol, Tetraethylene Glycol, 2-Butoxyethanol and 2-Methoxyethanol.
[3] The compounds included were: C12-C16 and C18 alcohol ethoxylates, and alkylphenol ethoxylates.

1　limitations, including known under representation of radium 228, and applicability to drinking
2　water samples with low levels (<500 mg/L) of TDS. As discussed below, produced water can have
3　much higher TDS levels. Because of these limitations, the EPA (2014b) developed an updated
4　method for the detection of gross alpha and gross beta to reduce the matrix interferences, although
5　further improvement is possible.[1]

6　Due to the high ionic strength and dissolved solids concentration of flowback water from shale
7　operations, Nelson et al. (2014) similarly found that traditional wet chemistry techniques (EPA
8　Methods 903.0 and 904.0) inefficiently recover radium from samples, with radium-226 recovery
9　sometimes less than 1% . This concern, which could lead to false negatives, was previously noted by
10　Demorest and Wallace (1992). Nelson and coauthors demonstrated that an accurate assessment of
11　flowback radium levels can be performed through nondestructive high-purity germanium gamma
12　spectroscopy and emanation techniques. Studies reporting radium concentrations obtained directly
13　via wet chemistry techniques or studies reporting third-party radium data via wet chemistry
14　techniques may need to be evaluated appropriately as these techniques may underestimate the
15　total radium loads of produced water (Nelson et al., 2014).

16　Data have been generated from specific produced water studies (e.g., Hayes, 2009) or compilations
17　from various sources, such as the USGS produced water database developed in 2002 (Breit, 2002)
18　and updated in 2014 (Blondes et al., 2014). In this database, data were compiled from a variety of
19　sources, some of which we cite as examples below. The data that appear in this chapter and
20　Appendix E are drawn individually from scientific literature and published reports, where
21　necessary we have filled gaps with data from the USGS database.

## 7.3.　Background on Formation Characteristics

22　Subsurface processes and resulting flowback and produced water composition vary depending
23　upon the mineralogy, geochemistry, and structure of formation solids, as well as, residence time
24　and other factors (Dahm et al., 2011; Blauch et al., 2009). The mineralogy and structure of
25　formation solids are determined initially by deposition, when rock grains settle out of their
26　transporting medium (Marshak, 2004). Generally, shale results from clays deposited in deep,
27　oxygen-poor marine environments, and sandstone results from sand deposited in shallow marine
28　environments (Ali et al., 2010; U.S. EPA, 2004). Coal forms when carbon-rich plant matter collects in
29　shallow peat swamps. In the United States, coal is formed in both freshwater and marine
30　environments (NRC, 2010). In the northern Rocky Mountains, coal formed within freshwater
31　alluvial systems of streams, lakes, and peat swamps. In contrast are parts of the Black Warrior
32　formation, which were deposited in brackish and marine settings (Horsey, 1981).

33　Variation in produced water composition follows, in part, from differences in formations which are
34　related to geologic processes. After deposition, physical, chemical, and biological processes occur as

---

[1] The method developed for determining gross alpha (Th, U, and Po) by liquid scintillation is based on: manganese dioxide coprecipitation followed by group separation of thorium, uranium and polonium on TRU Resin, stripping with ammonium bioxalate, and pulse-shape discrimination liquid scintillation analysis. The average recovery was 74±11% of the known concentration of 230Th with recoveries which ranged from 57% to 104%.

1    sediments and are consolidated and cemented into rocks in a process called diagenesis. These
2    processes, which can also occur in existing sedimentary rocks, are caused by increased pressure,
3    temperature, and reaction with mineral-rich ground water (Rushing et al., 2013; Marshak, 2004).
4    Diagenesis may either decrease or increase porosity and permeability through sediment
5    compaction and mineral precipitation, or through grain and cement dissolution (Ali et al., 2010;
6    Schmidt and McDonald, 1979). Temperature and pressure greatly affect the types and extent of
7    subsurface reactions, influencing the solubility of formation solids, saturation of pore waters, and
8    prevalence of precipitates (Rushing et al., 2013).

## 7.4.    Flowback Composition

9    The composition of returning hydraulic fracturing fluid changes with increasing residence time. In
10   this section, we present several examples from individual wells which demonstrate how
11   concentrations approach apparently asymptotic values during the first few days or weeks after
12   hydraulic fracturing.

### 7.4.1.    General Characteristics

13   Several interacting factors that influence the composition of hydraulic fracturing flowback and
14   produced water are recognized in the scientific literature: (1) the composition of injected hydraulic
15   fracturing fluids, (2) the targeted geological formation and associated hydrocarbon products,
16   (3) the stratigraphic environment, and (4) subsurface processes and residence time (Barbot et al.,
17   2013; Chapman et al., 2012; Dahm et al., 2011; Blauch et al., 2009).

18   By design, hydraulic fracturing exposes fresh, organic- and mineral-rich surfaces. Subsurface
19   interactions between injected hydraulic fracturing fluids, formation solids, and formation waters
20   follow. As residence time increases, allowing in situ interactions between injected fluids, formation
21   fluids, and formation solids, changes in the geochemical content of flowback occur such that it still
22   largely reflects that of injected fluids, while later flowback and produced water reflect that of
23   formation-associated fluid (Rowan et al., 2011).

### 7.4.2.    Temporal Changes in Flowback Composition

24   Ionic loads, metals, naturally occurring radioactive material (NORM), and organics increase in
25   concentration as water production continues (Barbot et al., 2013; Murali Mohan et al., 2013; Rowan
26   et al., 2011). The causes include precipitation and dissolution of salts, carbonates, sulfates, and
27   silicates; pyrite oxidation; leaching and biotransformation of organic compounds; and mobilization
28   of NORM and trace elements. Multiple geochemical studies confirm this trend (Barbot et al., 2013;
29   Haluszczak et al., 2013; Chapman et al., 2012; Davis et al., 2012; Gregory et al., 2011; Blauch et al.,
30   2009).

31   Concurrent precipitation of sulfates (e.g., $BaSO_4$) and carbonates (e.g., $CaCO_3$) alongside decreases
32   in pH, alkalinity, and dissolved carbon load occur over time (Orem et al., 2014; Barbot et al., 2013;
33   Blauch et al., 2009; Brinck and Frost, 2007). Orem and colleagues showed that organics within CBM
34   produced waters also decrease over time, possibly due to the exhausting of coal-associated waters
35   through formation pumping (Orem et al., 2007). Decreases in microbial abundance and diversity
36   also occur over time after hydraulic fracturing (Murali Mohan et al., 2013; Davis et al., 2012).

1  The primary dissolution of native and emplaced salts within the formation and the mobilization of
2  in situ brines constitute the major subsurface processes that control TDS levels in flowback and
3  produced water ([Dresel and Rose, 2010](); [Blauch et al., 2009]()).[1] Leaching of organics appears to be a
4  result of injected and formation fluids associating with shale and coal strata ([Orem et al., 2014]()).

### 7.4.3.  Total Dissolved Solids Enrichment

5  To varying degrees, produced water is enriched in dissolved solids, and the enrichment is
6  dependent upon residence time ([Rowan et al., 2011]()). As an example, TDS concentrations increased
7  until a limit was reached in flowback and produced water samples from four Marcellus Shale gas
8  wells in three southwestern Pennsylvanian counties ([Chapman et al., 2012]()) (see Figure 7-6). As is
9  shown in Figure 7-7, TDS in flowback from both Westmoreland County wells became consistent
10 with TDS concentrations cited for typical seawater (35,000 mg/L) within three days, and became
11 consistent with TDS cited for brines (greater than 50,000 mg/L) within five days ([Chapman et al.,]()
12 [2012]()). TDS concentrations during production exceeded 188,000 mg/L for one well in Greene
13 County. Chapman et al.'s findings are further substantiated by Hayes and colleagues' earlier report
14 of produced water TDS concentrations in 19 Marcellus Shale wells in Pennsylvania and West
15 Virginia ([Hayes, 2009]()). From an initial injected median value of less than 1,000 mg/L, TDS
16 concentrations increased to a median value exceeding 200,000 mg/L within 90 days ([Hayes, 2009]()).
17 In the Marcellus Shale, the cation portion of TDS is typically dominated by sodium and calcium,
18 whereas the anion portion is dominated by chloride ([Chapman et al., 2012](); [Blauch et al., 2009]()). In
19 section 7.6.4, we note that there is disagreement over whether increased salinity in Marcellus Shale
20 produced water is due to dissolution of salts or mixing of formation water with hydraulic fracturing
21 fluid.

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[1] Native salts are formed inside the rock matrix, and can include evaporite minerals such as halite (NaCl), polyhalite ($K_2Ca_2Mg(SO_4)_4 \cdot 2H_2O$), celestite ($SrSO_4$), anhydrite ($CaSO_4$), kieserite ($MgSO_4 \cdot H_2O$), or sylvite (KCl) ([Blauch et al., 2009]()). Hydrologic intrusion emplaces salts within formation pores and fractures ([Blauch et al., 2009]()).

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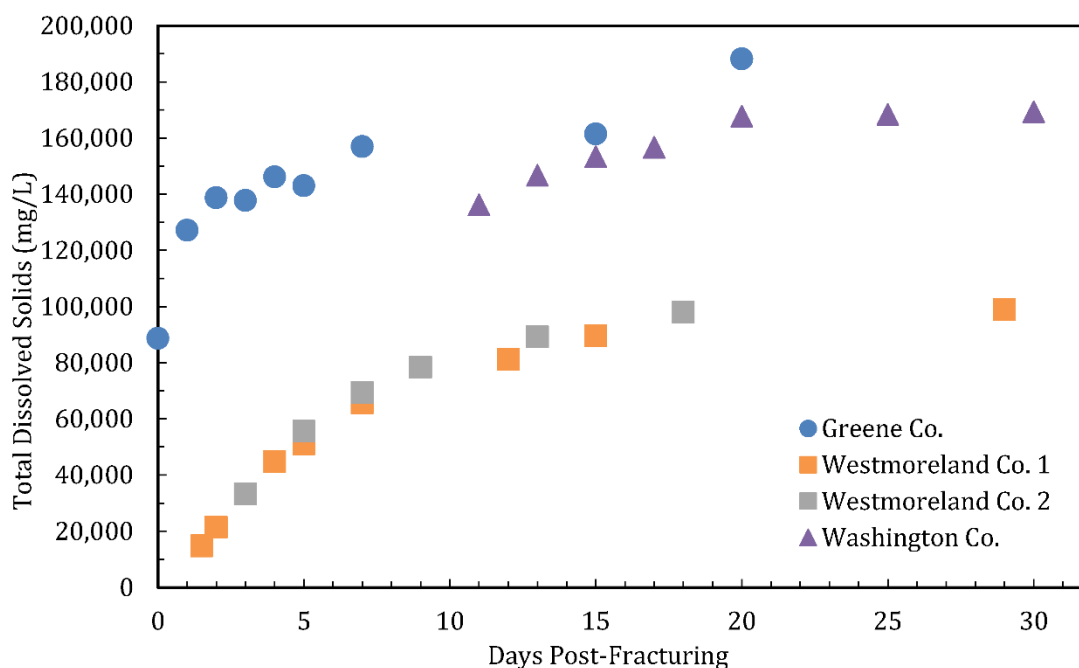
**Figure 7-6. TDS concentrations measured through time for injected fluid (at 0 days), flowback, and produced water samples from four Marcellus Shale gas wells in three southwestern Pennsylvanian counties.**

Data from Chapman et al. (2012).

### 7.4.4.   Radionuclide Enrichment

1   Injected fluids used in hydraulic fracturing typically do not contain radioactive material (Rowan et
2   al., 2011).[1] Shales and sandstones, however, are naturally enriched in various radionuclides, as
3   described below (Sturchio et al., 2001). Radium in pore waters or adsorbed onto clay particles and
4   grain coatings can dissolve and return within flowback (Langmuir and Riese, 1985). Where data are
5   available, radium and TDS produced water concentrations are positively correlated with time
6   passed since hydraulic fracturing (Rowan et al., 2011; Fisher, 1998). Radium remains adsorbed to
7   mineral surfaces in low saline environments, and then desorbs with increased salinity into solution
8   (Sturchio et al., 2001). Over the course of 20 days, Marcellus Shale produced waters from a gas well
9   were enriched almost fourfold in radium and from another gas well were enriched over twofold in
10  TDS concentrations as residence time increased (Chapman et al., 2012; Rowan et al., 2011) (see
11  Figure 7-7).

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[1] Recycling produced water may introduce radioactive material into hydraulic fracturing fluid. See section 8.4.3 and PA DEP (2015b).
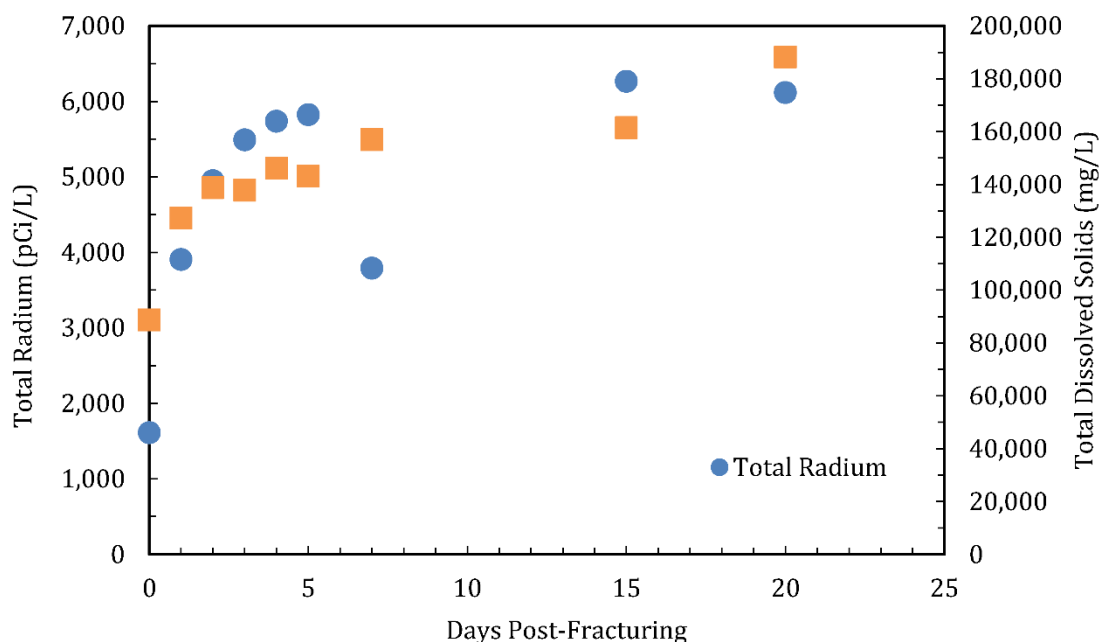
**Figure 7-7. Total radium and TDS concentrations measured through time for injected (day 0), flowback, and produced water samples from mutually exclusive Greene County, PA, Marcellus Shale gas wells.**

Data from Rowan et al. (2011) and Chapman et al. (2012).

### 7.4.5.  Leaching and Biotransformation of Naturally Occurring Organic Compounds

1    Many organics are known to leach naturally into formation water through association with shale,
2    sandstone, and coal strata (Benko and Drewes, 2008; Orem et al., 2007). Orem et al. (2014) show
3    that formation and produced waters from shale plays that were not impacted by production
4    chemicals contain an array of organic compound classes associated with the formation. When
5    unconventional formations are hydraulically fractured, additional organics from the freshly
6    fractured hydrocarbon-bearing formation and the chemical additives contribute to a large increase
7    in flowback and produced water organic loads (Orem et al., 2014).

8    The nature of the in-situ hydrocarbons reflects the formation's thermal maturity and heavily
9    influences the organic content of the produced water.[1] The Marcellus Shale is largely considered a
10   mature formation and therefore consists of wet and dry gas (Barbot et al., 2013; Repetski et al.,
11   2008). Conversely, the Utica Shale is less thermally mature; available hydrocarbon resources
12   consist of oil, condensate, and gas (Repetski et al., 2008). Additionally, some coals within the
13   eastern and west-central regions of the San Juan Basin produce little to no water during production,
14   due to the regional thermal maturity, hydrostratigraphy, and in situ trapping mechanisms (New
15   Mexico Bureau of Mines and Mineral Resources, 1994).

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[1] With increasing subsurface temperature after burial, petroleum source material (kerogen) produces hydrocarbons in a sequence from methane (immature), to oil (more mature), to gas (mature). Gas is produced by thermal cracking of oil (PA DCNR, 2015).

1    Dissolved organic carbon (DOC) concentrations decrease from hydraulic fracturing through
2    flowback in shales and coalbeds (Murali Mohan et al., 2013; Orem et al., 2007). DOC sorption,
3    dilution with injected or formation water, biochemical reactions, and microbial transformation (i.e.,
4    biotransformation in the form of degradation or uptake) may all cause decreased concentrations of
5    DOC during flowback. Organic chemical additives injected during hydraulic fracturing offer a novel
6    carbon and energy source for biotic and abiotic reactions at depth. Injected organics include many
7    sugar-based polymer formulations, most notably of galactose and mannose (i.e., guar gum used as a
8    gelling agent); hydrocarbon distillates used in crosslinkers, friction reducers, and gelling agents;
9    and ethyl and ether glycol formulations used in non-emulsifiers, crosslinkers, friction reducers, and
10   gelling agents. (Wuchter et al., 2013; Arthur et al., 2009b; Hayes, 2009).

11   DOC and chloride concentrations exhibit strongly correlated inverse temporal trends (Barbot et al.,
12   2013; Chapman et al., 2012) for flowback and produced water samples obtained from three
13   Marcellus Shale wells from the same well pad in Greene County, Pennsylvania (Cluff et al., 2014), as
14   shown in Figure 7-8. Chloride concentrations increased five- to six-fold as a function of residence
15   time (i.e., cumulative volumes of produced water). These chloride concentrations followed an
16   increasing linear trend during the first two weeks of flowback (see Figure 7-8a, inset) then began to
17   approach asymptotic levels later in production, indicating that injected fluids had acquired a brine
18   signature as a result of subsurface mixing, fluid-solid interactions, and mineral dissolution
19   processes.

20   DOC concentrations exhibit an inverse trend and decreased through flowback and production
21   (Figure 7-8b) (Cluff et al., 2014). DOC levels decreased approximately twofold between injected
22   fluid and initial flowback samples (Figure 7-8b, inset). DOC concentrations decreased by 11-fold
23   over the study's time frame (nearly 11 months) and leveled off several months after hydraulic
24   fracturing, presumably as a result of in situ attenuation processes (Cluff et al., 2014).

**(a)**



**(b)**
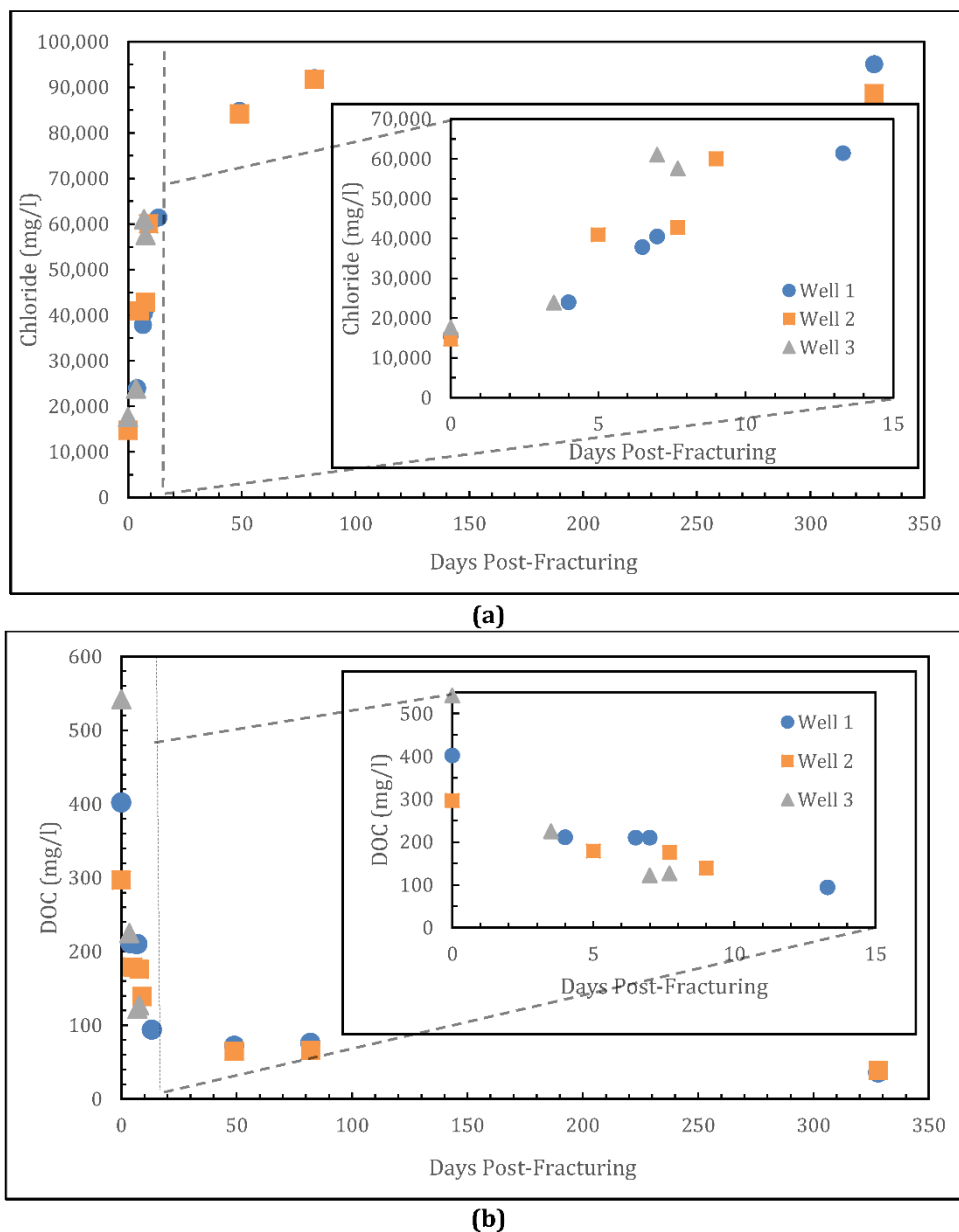
**Figure 7-8. (a) Chloride (Cl) and (b) DOC concentrations measured through time for injected (day 0), flowback, and produced water samples obtained from three Marcellus Shale gas wells from a single well pad in Greene County, PA used for hydraulic fracturing.**

Data from Cluff et al. (2014). Reprinted with permission from Cluff, M; Hartsock, A; Macrae, J; Carter, K; Mouser, PJ. (2014). Temporal changes in microbial ecology and geochemistry in produced water from hydraulically fractured Marcellus Shale Gas Wells. Environ Sci Technol 48: 6508-6517. Copyright 2014 American Chemical Society.

1  [Orem et al. (2014)](#) conducted a temporal study of two coalbed wells over the course of one year.
2  Their results suggest that organic compound concentrations decrease over time. This trend may be
3  due to pumping of water to the surface, which may exhaust coal-associated produced water.
4  Subsequent produced water would not be associated with the coal. This suggests that the early
5  produced water would contain the highest organic load ([Orem et al., 2014](#)).

6  As noted above, most literature addresses general organic parameters such as bulk total organic
7  carbon (TOC) or DOC instead of individual organic compounds ([Sirivedhin and Dallbauman, 2004](#)).
8  Emphasis on the prevalence of bulk organics as opposed to unique organics is due largely to the
9  lack of analytical standards for many compounds and also a lack of knowledge regarding the types
10  of organics to test in produced water samples ([Schlegel et al., 2013](#); [Strong et al., 2013](#)).

## 7.5. Produced Water Composition

11  In this section, we discuss the characteristics of aggregated produced water data without regard for
12  temporal changes. Similarities between conventional and unconventional produced water are
13  noted and the variability between formation types is described. As we discuss below, produced
14  water may contain a range of constituents, but in widely varying amounts. Generally, these may
15  include:

16  • Salts, including those composed from chloride, bromide, sulfate, sodium, magnesium and
17    calcium.

18  • Metals including barium, manganese, iron, and strontium.

19  • Dissolved organics including BTEX and oil and grease.

20  • Radioactive materials including radium (radium-226 and radium-228).

21  • Hydraulic fracturing chemicals and their transformation products.

22  We discuss these groups of chemicals and then conclude by discussing variability within formation
23  types and within production zones.

### 7.5.1. Similarity of Produced Water from Conventional and Unconventional Formations

24  Unconventional produced water is reported to be similar to conventional produced waters in terms
25  of TDS, pH, alkalinity, oil and grease, TOC, and other organics and inorganics ([Wilson, 2014](#);
26  [Haluszczak et al., 2013](#); [Alley et al., 2011](#); [Hayes, 2009](#); [Sirivedhin and Dallbauman, 2004](#)). Although
27  salinity varies in shales and tight formations, produced water is typically characterized as saline
28  ([Lee and Neff, 2011](#); [Blauch et al., 2009](#)). Produced water is also enriched in major anions (e.g.,
29  chloride, bicarbonate, sulfate), cations (e.g., sodium, calcium, magnesium), metals (e.g., barium,
30  strontium), naturally occurring radionuclides (e.g., radium-226 and radium-228) ([Chapman et al.,](#)
31  [2012](#); [Rowan et al., 2011](#)), and organics (e.g., hydrocarbons) ([Orem et al., 2007](#); [Sirivedhin and](#)
32  [Dallbauman, 2004](#)).

### 7.5.2. Variability in Produced Water Composition Among Unconventional Formation Types

33  [Alley et al. (2011)](#) compared geochemical parameters of shale gas, tight gas, and CBM produced
34  water. This comparison aggregated data on produced water from original analyses, peer-reviewed

1   literature, and public and confidential government and industry sources and determined the
2   statistical significance of the results. As shown in Table 7-4, Alley et al. (2011) found that of the
3   constituents of interest common to all three types of unconventional produced water (calcium,
4   chloride, potassium, magnesium, manganese, sodium, and zinc):

5     1)  Shale gas produced water had significantly different concentrations from those of CBM;

6     2)  Shale gas produced water constituent concentrations were significantly similar to those of
7         tight gas, except for potassium and magnesium; and

8     3)  Five tight gas produced water constituent concentrations (calcium, chloride, potassium,
9         magnesium, and sodium) were significantly similar to those of CBM (Alley et al., 2011).

10  The degree of variability between produced waters of these three resource types is consistent with
11  the degree of mineralogical and geochemical similarity between shale and sandstone formations,
12  and the lack of the same between shale and coalbed formations (Marshak, 2004).

**Table 7-4. Compiled minimum and maximum concentrations for various geochemical constituents in unconventional shale gas, tight gas, and CBM produced water (Alley et al., 2011).**

Source: (Alley et al., 2011).

| Parameter | Unit | Shale gas[a] | Tight gas[b] | CBM[c] |
|---|---|---|---|---|
| Alkalinity | mg/L | 160–188 | 1,424 | 54.9–9,450 |
| Ammonium-N | mg/L | - | 2.74 | 1.05–59 |
| Bicarbonate | mg/L | ND–4,000 | 10–4,040 | - |
| Conductivity | µS/cm | - | 24,400 | 94.8–145,000 |
| Nitrate | mg/L | ND–2,670 | - | 0.002–18.7 |
| Oil and grease | mg/L | - | 42 | - |
| pH | SU[d] | 1.21–8.36 | 5–8.6 | 6.56–9.87 |
| Phosphate | mg/L | ND–5.3 | - | 0.05–1.5 |
| Sulfate | mg/L | ND–3,663 | 12–48 | 0.01–5,590 |
| Radium-226 | pCi/g | 0.65–1.031 | - | - |
| Aluminum | mg/L | ND–5,290 | - | 0.5–5,290 |
| Arsenic | mg/L | - | 0.17 | 0.0001–0.06 |
| Boron | mg/L | 0.12–24 | - | 0.002–2.4 |
| Barium | mg/L | ND–4,370 | - | 0.01–190 |

*This document is a draft for review purposes only and does not constitute Agency policy.*

| Parameter | Unit | Shale gas[a] | Tight gas[b] | CBM[c] |
|---|---|---|---|---|
| **Bromine** | mg/L | ND–10,600 | - | 0.002–300 |
| **Calcium** | mg/L | 0.65–83,950 | 3–74,185 | 0.8–5,870 |
| **Cadmium** | mg/L | - | 0.37 | 0.0001–0.01 |
| **Chlorine** | mg/L | 48.9–212,700 | 52–216,000 | 0.7–70,100 |
| **Chromium** | mg/L | - | 0.265 | 0.001–0.053 |
| **Copper** | mg/L | ND–15 | 0.539 | ND–0.06 |
| **Fluorine** | mg/L | ND–33 | - | 0.05–15.22 |
| **Iron** | mg/L | ND–2,838 | 0.015 | 0.002–220 |
| **Lithium** | mg/L | ND–611 | - | 0.0002–6.88 |
| **Magnesium** | mg/L | 1.08–25,340 | 2–8,750 | 0.2–1,830 |
| **Manganese** | mg/L | ND–96.5 | 0.525 | 0.002–5.4 |
| **Mercury** | mg/L | - | - | 0.0001–0.0004 |
| **Nickel** | mg/L | - | 0.123 | 0.0003–0.20 |
| **Potassium** | mg/L | 0.21–5,490 | 5–2,500 | 0.3–186 |
| **Sodium** | mg/L | 10.04–204,302 | 648–80,000 | 8.8–34,100 |
| **Strontium** | mg/L | 0.03–1,310 | - | 0.032–565 |
| **Uranium** | mg/L | - | - | 0.002–0.012 |
| **Zinc** | mg/L | ND–20 | 0.076 | 0.00002–0.59 |

-, no value available; ND, non-detect. If no range, but a singular concentration is given, this is the maximum concentration.

[a] *n* = 541. Alley et al. (2011) compiled data from USGS (2006); McIntosh and Walter (2005); McIntosh et al. (2002) and confidential industry documents.

[b] *n* = 137. Alley et al. (2011) compiled data from USGS (2006) and produced water samples presented in Alley et al. (2011).

[c] Alley et al. (2011) compiled data from Montana GWIC (2009); Thordsen et al. (2007); ESN Rocky Mountain (2003); Rice et al. (2000); Rice (1999); Hunter and Moser (1990).

[d] SU = standard units.

1    Shale gas produced water tends to be more acidic, as well as, enriched in strontium, barium, and
2    bromide. CBM produced water is highly alkaline, and it contains relatively low concentrations of
3    TDS (one to two orders of magnitude lower than in shale and sandstone). It also contains lower
4    levels of sulfate, calcium, magnesium, DOC, sodium, bicarbonate, and oil and grease than typically

1    observed in shale and sandstone produced waters ([Alley et al., 2011](#); [Dahm et al., 2011](#); [Benko and](#)
2    [Drewes, 2008](#); [Van Voast, 2003](#)).[1]

### 7.5.3.  General Water Quality Parameters

3    For this assessment, the EPA identified data characterizing the content of unconventional flowback
4    and produced water in a total of 12 shale and tight formations and CBM basins. These formations
5    and basins span 18 states. Note that in this subsection we treat all fluids as produced. As a
6    consequence, the variability of reported concentrations is likely higher than if the data could be
7    standardized to a specific point on the flowback-to-produced water continuum.

8    For most formations, the amount of general water quality parameter data is variable (see Table E-2
9    of Appendix E). Average pH levels range from 5.87 to 8.19, with typically lower values for shales.
10   Larger variations in average specific conductivity are seen among unconventional formations and
11   range from 213 microsiemens (μS)/cm in the Bakken Shale to 184,800 μS/cm in Devonian
12   sandstones (see Table E-2 of Appendix E). Shale and tight formation produced waters are enriched
13   in suspended solids, as reported concentrations for total suspended solids and turbidity exceed
14   those of coalbeds by one to two orders of magnitude.

15   Of the data presented in Table E-3 of Appendix E, differences are evident between the Black
16   Warrior and the three western formations (Powder River, Raton, and San Juan). The Black Warrior
17   is higher in average chloride, specific conductivity, TDS, TOC and total suspended solids; and lower
18   in alkalinity and bicarbonate than the other three. These differences are due to the saline or
19   brackish conditions during deposition in the Black Warrior that contrast to the freshwater
20   conditions for the western basins.

21   The average dissolved oxygen (DO) concentrations of CBM produced water range from 0.39-1.07
22   mg/L (see Table E-3 Appendix E). By comparison, well-oxygenated surface water can contain up to
23   10 mg/L DO at 15 °C ([U.S. EPA, 2012a](#)). Thus, coalbed produced water is either hypoxic (less than 2
24   mg/L DO) or anoxic (less than 0.5 mg/L DO) and could contribute to aquatic organism stress ([USGS,](#)
25   [2010](#); [NSTC, 2000](#)).

### 7.5.4.  Salinity and Inorganics

26   The TDS profile of unconventional produced water is dominated by sodium and chloride, with large
27   contributions to the profile from mono- and divalent cations ([Sun et al., 2013](#); [Guerra et al., 2011](#)).
28   In order of relative abundance, the following inorganic ions are typically found in highly saline
29   conventional produced water: sodium, chloride, calcium, magnesium, potassium, sulfate, bromide,
30   strontium, bicarbonate, and iodide ([Lee and Neff, 2011](#)). Shale and sandstone produced water tend
31   to be characterized as sodium-chloride-calcium water types, whereas CBM produced water tends to
32   be characterized as sodium chloride or sodium bicarbonate water types ([Dahm et al., 2011](#)).
33   Elevated levels of bromide, sulfate, and bicarbonate are also present ([Sun et al., 2013](#)). Elevated
34   strontium and barium levels are characteristic of Marcellus Shale flowback and produced water

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[1] Several had low representation in the [Alley et al. (2011)](#) data set, including the Appalachian Basin (western New York and western Pennsylvania), West Virginia, eastern Kentucky, eastern Tennessee, and northeastern Alabama.

1   ([Barbot et al., 2013](#); [Haluszczak et al., 2013](#); [Chapman et al., 2012](#)). Data representing shales and
2   tight formations are presented in Table E-4 of Appendix E.

3   [Rowan et al. (2015)](#) acknowledge that the origin of saline water produced from the Marcellus Shale
4   is a matter of debate. One idea is that injected fluid returns at higher salinity, because of dissolving
5   halite and other minerals found in shale ([Blauch et al., 2009](#)). Blauch and colleagues hypothesized
6   that salt layers bearing barium, calcium, iron, potassium, magnesium, sodium, and strontium likely
7   dissolve and contribute to flowback and produced water salinity ([Blauch et al., 2009](#)). However,
8   actual mixing of formation water and fracturing fluid was postulated by [Haluszczak et al. (2013)](#)
9   from arguments based on the near-neutral pH and low levels of chloride and sulfate in the
10  Marcellus data from [Hayes (2009)](#), as well as, the relationship between chloride and bromide. Engle
11  and Rowan determined that water chemistry during the first 90 days of production is controlled by
12  mixing of injected and formation waters and stimulation of bacterial sulfate reduction ([Engle and](#)
13  [Rowan, 2014](#); [Haluszczak et al., 2013](#)). [Rowan et al. (2015)](#) argue, based on an observed shift to
14  isotopically heavier water, that produced water actually contains formation water.[1] Alternately,
15  [Barbot et al. (2013)](#) concluded from analysis of Marcellus Shale produced water that mixing (with
16  formation water) alone could not explain the observed patterns in chloride concentrations.

17  Marcellus Shale produced water salinities range from less than 1,500 mg/L to over 300,000 mg/L,
18  as shown by [Rowan et al. (2011)](#). By comparison, the average salinity concentration for seawater is
19  35,000 mg/L. The TDS concentration of CBM produced water can be as low as 500 mg/L ranging to
20  nearly 50,000 mg/L ([Dahm et al., 2011](#); [Benko and Drewes, 2008](#); [Van Voast, 2003](#)). Lower
21  dissolved solids are expected from CBM produced water, in part, because some coals developed in
22  fresh water environments ([Bouska, 1981](#)). [Dahm et al. (2011)](#) report TDS concentrations from a
23  composite CBM produced water database (*n* = 3,255) for western basins that often are less than
24  5,000 mg/L (85% of samples). In other cases, as for the Black Warrior basin, TDS can be higher
25  along with concentrations of species that contribute to TDS (See Table E-5 Appendix E), such as
26  calcium, chloride, and sodium.

### 7.5.5. Metals

27  The metals content of unconventional produced water varies by well and site lithology, but is
28  typically dominated by the same metals that are associated with conventional produced water.
29  Unconventional produced water may also contain low levels of heavy metals (e.g., chromium,
30  copper, nickel, zinc, cadmium, lead, arsenic, and mercury) ([Hayes, 2009](#)). Data illustrating metal
31  concentrations in produced water appear in Tables E-6 and E-7 of Appendix E.

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[1] The produced water becomes isotopically heavier because of increased prevalence of Oxygen-18 in the water, compared to the more prevalent Oxygen-16.

### 7.5.6. Naturally Occurring Radioactive Material (NORM) and Technologically Enhanced Naturally Occurring Radioactive Material (TENORM)

#### 7.5.6.1. Formation Solids Levels of NORM

1  Elevated uranium levels in formation solids have been used to identify potential areas of natural
2  gas production for decades (Fertl and Chilingar, 1988). Marine black shales are estimated to contain
3  an average of 15–60 ppm uranium depending on depositional conditions (Fertl and Chilingar,
4  1988). Shales that bear significant levels of uranium include the Barnett in Texas, the Woodford in
5  Oklahoma, the New Albany in the Illinois Basin, the Chattanooga Shale in the southeastern United
6  States, and a group of black shales in Kansas and Oklahoma (Swanson, 1955).

#### 7.5.6.2. Produced Water Levels of TENORM

7  When exposed to the environment or concentrated NORM is termed technologically-enhanced
8  naturally-occurring radioactive material (TENORM).[1] Radioactive materials commonly present in
9  shale and sandstone sedimentary environments include uranium, thorium, radium, and their decay
10  products. These are present in most unconventional produced water, but particularly so in
11  Marcellus Shale produced water (Rowan et al., 2011; Fisher, 1998). Low levels of uranium and
12  thorium return during flowback, typically in the concentrated form of mineral phases or organic
13  matter, due to insolubility under prevailing reducing conditions encountered within shale
14  formations (Nelson et al., 2014; Sturchio et al., 2001).

15  Conversely, radium, a decay product of uranium and thorium, is known to be relatively soluble
16  within the redox range encountered in subsurface environments (Sturchio et al., 2001; Langmuir
17  and Riese, 1985). Dissolved radium primarily occurs as $Ra^{2+}$, but it complexes with carbonate,
18  chloride, and sulfate ions as well (Sturchio et al., 2001; Langmuir and Riese, 1985). $Ra^{2+}$ can also
19  substitute for various cations (e.g., $Ba^{2+}$, $Ca^{2+}$, and $Sr^{2+}$) during mineral precipitation, as is
20  sometimes the case with barite or anhydrite precipitation (Rowan et al., 2011).

21  Data from the Marcellus Shale show that radium and TDS produced water concentrations are
22  positively correlated (Rowan et al., 2011; Fisher, 1998). This pattern is expected for other
23  formations because radium remains adsorbed to mineral surfaces in low salinity environments,
24  then desorbs as solution salinity increases (Sturchio et al., 2001). Controlling for this TDS
25  dependence, Marcellus Shale produced water contains statistically more radium than
26  non-Marcellus Shale produced water, with a median total radium content of 2,460 picocuries per
27  liter (pCi/L) (*n* = 52) compared to 1,011 pCi/L (*n* = 91), respectively (Rowan et al., 2011). Radium
28  levels in Marcellus produced water are at several thousand picocuries per liter, with maximum
29  concentrations of total radium (radium-226 and radium-228), radium-226 and radium-228
30  reported at approximately 18,000, 9,000, and 1,300 pCi/L, respectively (Rowan et al., 2011) (see
31  Table E-8 in Appendix E). Data from the Pennsylvania TENORM produced water study (PA DEP,

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[1] The U.S. EPA Office of Radiation (http://www.epa.gov/radiation/tenorm/) states that technologically enhanced naturally occurring radioactive material (TENORM) is produced when activities such as uranium mining, or sewage sludge treatment, concentrate or expose radioactive materials that occur naturally in ores, soils, water, or other natural materials. Formation water containing radioactive materials would contain NORM, because they are not exposed; produced water would contain TENORM because it has been exposed to the environment.

1   2015) showed similar elevated levels, and consistently showed higher medians in unconventional
2   compared to conventional formations (Table E-8 in Appendix E).

### 7.5.7.  Organics

3   The organic content of produced water varies by well and lithology, but consists of certain naturally
4   occurring and injected organic compounds. These organics may be dissolved in water or, for the
5   case of oil production, in the form of a separate or emulsified phase. Produced water organics can
6   contain any of the following: (1) volatile organic compounds (VOCs) such as benzene and toluene,
7   (2) semi-volatile organic compounds (SVOCs) such as phenols; and/or (3) non-VOCs such as
8   macromolecular natural organic matter (Orem et al., 2014; Hayes, 2009; Benko and Drewes, 2008;
9   Orem et al., 2007; Sirivedhin and Dallbauman, 2004). Table 7-5 presents data from naturally
10  occurring organic chemicals in produced water.

**Table 7-5. Concentration ranges (mg/L) of several classes of naturally occurring organic chemicals in conventional produced water worldwide (reported in Neff, 2002).**

Source: (Neff, 2002)

| Chemical class | Concentration range (mg/L) |
|---|---|
| TOC | ≤0.1–>11,000 |
| Total organic acids | ≤0.001–10,000 |
| Total saturated hydrocarbons | 17–30 |
| Total benzene, toluene, ethylbenzene, and xylenes (BTEX) | 0.068–578 |
| Total PAH | 0.040–3 |
| Total steranes/triterpanes | 0.140–0.175 |
| Ketones | 1–2 |
| Total phenols (primarily $C_0$–$C_5$ phenols) | 0.400–23 |

11  Several classes of naturally occurring organic chemicals are present in conventional and
12  unconventional produced waters, with large concentration ranges (Lee and Neff, 2011). In addition
13  to data on total organic carbon (TOC) and dissolved organic carbon (DOC) as indicators of the
14  presence of organics, specifically identified organics include saturated hydrocarbons, BTEX, and
15  polycyclic aromatic hydrocarbon (PAHs) (see Table E-9 of Appendix E). Data are lacking on the
16  presence and concentration of many other types of organic chemicals that might be present in
17  produced water, because of their use in hydraulic fracturing fluid. There are a number of reasons
18  for this difference, some of which could be related to analytical limitations, limited focus of
19  produced water studies, and undocumented subsurface reactions.

### 7.5.8.  Reactions within Formations

20  The introduction of hydraulic fracturing fluids into the target formation induces a number of
21  changes to formation solids and fluids that influence the chemical evolution and composition of

1  flowback and produced water. These changes can result from physical processes (e.g., rock
2  fracturing and fluid mixing) and geochemical processes (e.g., introducing novel, oxygenated fluids)
3  that mobilize trace or major constituents into solution.

4  The creation of fractures exposes new formation surfaces to interactions involving hydraulic
5  fracturing fluids and existing formation fluids. Formations targeted for unconventional
6  development are composed of detrital, cement, and organic fractions. For example, elements
7  potentially available for mobilization when exposed via fracturing include calcium, magnesium,
8  manganese, and strontium in cement fractions, and silver, chromium, copper, molybdenum,
9  niobium, vanadium, and zinc in organic fractions. The storage or release of these elements in newly
10 exposed surfaces is variable and not well studied, in part due to the vast number of possible
11 interactions occurring continuously in the environment at the rock surface (Vine and Tourtelot,
12 1970).

13 Contact with and physical mixing of hydraulic fracturing fluids with existing formation brines also
14 influences the geochemical evolution of produced water. For instance, Marcellus Shale brines have
15 high concentrations of bromide, calcium, chloride, magnesium, sodium, and strontium (Engle and
16 Rowan, 2014). Hydraulic fracturing fluid contains elevated levels of DOC, alkalinity, and sulfate
17 (Engle and Rowan, 2014). Consequently, flowback acquires a geochemical signature reflecting both
18 injected and formation fluids. Produced water containing bothAlthough some constituents of
19 hydraulic fracturing fluids are known to readily degrade in the environment, little is known
20 regarding how the subsurface degradation proceeds or how the constituents interact within a
21 complex matrix of organics (Mouser et al., In Press).

## 7.6.   Spatial Trends

22 As was reported for the volume of produced water (see Section 7.2.2), the composition of produced
23 water varies spatially on a regional to local scale according to the geographic and stratigraphic
24 locations of each well within a hydraulically fractured formation (Bibby et al., 2013; Lee and Neff,
25 2011). Spatial variability of produced water content occurs (1) between plays of different rock
26 sources (e.g., coal vs. sandstone), (2) between plays of the same rock type (e.g., Barnett Shale vs.
27 Bakken Shale), and (3) within formations of the same source rock (e.g., northeastern vs.
28 southwestern Marcellus Shale) (Barbot et al., 2013; Alley et al., 2011; Breit, 2002).

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

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

*This document is a draft for review purposes only and does not constitute Agency policy.*

1    High TDS (more than 200,000 ppm) is common throughout the central portion of the United States
2    in various basins. Low TDS (<10,000 ppm) was found in the basins of the Rocky Mountains, and
3    sometimes in Texas and California. In other areas, there was a mixture of mid-range, which in the
4    case of Illinois was correlated to the depth of producing zones (Breit, 2002).

5    Data further illustrating variability within shale, tight-gas and coalbed formations at both the
6    formation and local scales are presented and discussed in Section E-3 of Appendix E.

## 7.7.    Spill Impacts on Drinking Water Resources

7    Surface spills of flowback and produced water from unconventional oil and gas production have
8    occurred across the country and in some cases have caused impacts to drinking water resources, as
9    described in this section. Released fluids, if not contained on-site, may flow into nearby surface
10   waters or infiltrate into ground water via soil. In this section, we first briefly describe the potential
11   for spills from produced water handling equipment. Next, we address individually-reported spill
12   events. These have originated from pipeline leaks, well blowouts, well communication events, and
13   leaking pits and impoundments. We then summarize several studies of aggregated spill data, most
14   of which are based on state agency spill reports. The section concludes with discussion of two cases.

### 7.7.1.   Produced Water Management and Spill Potential

15   Produced water is typically transported from the wellhead through a series of pipes or flowlines to
16   on-site storage or treatment units (GWPC and IOGCC, 2014). Faulty connections at either end of the
17   transfer process or leaks or ruptures in the lines carrying the fluid can result in surface spills.

18   Recovered fluids may be transferred to surface impoundments for long-term storage and
19   evaporation. Surface impoundments are typically uncovered earthen pits that may or may not be
20   lined. Recovered fluids may overflow from surface impoundments due to improper pit design and
21   weather events.

22   Produced water that is to be treated or disposed of off-site is typically stored in storage tanks or
23   impoundments until it can be loaded into transport trucks for removal (Gilmore et al., 2013). Tank
24   storage systems are typically closed loop systems in which produced water is transported from the
25   wellhead to aboveground storage tanks through interconnecting pipelines (GWPC and IOGCC,
26   2014). Failure of connections and lines during the transfer process or the failure of a storage tank
27   can result in a surface release of fluids.

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

### 7.7.2. Spills of Hydraulic Fracturing Flowback and Produced Water from Unconventional Oil and Gas Production

#### 7.7.2.1. Pipeline Leaks

1  In some locations, pipelines are used to transport produced water. Aggregated information on
2  pipeline leaks from the whole country is not available. This section, rather, contains examples of
3  incidents that have occurred. A leak was detailed in a field report from PA DEP (2009a), which
4  described a leak from a 90-degree bend in an overland pipe carrying a mixture of flowback and
5  freshwater between two impoundments. Along a 0.4 mi (0.6 km) length of the impacted stream,
6  168 fish and 6 salamanders were killed; beyond a confluence at 0.6 km with a creek no additional
7  dead fish were found. The release was estimated at 250 bbl (11,000 gal or 40,000 L). In response to
8  the incident, the pipeline was shut off, a dam was constructed for recovering the water, water was
9  vacuumed from the stream, and the stream was flushed with fresh water (PA DEP, 2009a).

10 In January 2015, 70,000 barrels (2,940,000 gal or 11,130,000 L) of produced water containing
11 petroleum hydrocarbons (North Dakota Department of Health, 2015) were released from a broken
12 pipeline that crosses Blacktail Creek in Williams County, ND. The response included placing
13 adsorbent booms in the creek, excavating contaminated soil, removing oil-coated ice, and removing
14 produced water from the creek. The electrical conductivity and chloride concentration in water
15 along the creek, the Little Muddy River, and Missouri River were found to be elevated above
16 background levels, as were samples taken from ground water recovery trenches.

17 More incidents from North Dakota are documented at the North Dakota Department of Health
18 (NDDOH) Environmental Health web site (see http://www.ndhealth.gov/EHS/Spills/). For the
19 period from November, 2012 to November 2013, NDDOH reported 552 releases of produced water
20 which were retained within the boundaries of the production or exploration facility and 104 which
21 were not (see http://www.ndhealth.gov/ehs/foia/spills/ChartWebPageOG_20121101_
22 20131111.pdf).

#### 7.7.2.2. Well Blowouts

23 Fingerprinting of water from two monitoring wells in Killdeer, ND, was used to determine that
24 brine contamination in the two wells resulted from a well blowout during a hydraulic fracturing
25 operation. Although the target formation was the Bakken Shale, data indicated that the residual
26 signatures of the brine were characteristic of the overlying Madison limestone formation (U.S. EPA,
27 2015j). Prior research into out-of-zone hydraulic fracturing of the Bakken formation indicated that
28 a large number of hydraulically fractured wells contain water that is external to the Bakken Zone
29 (Arkadakskiy and Rostron, 2013a; Arkadakskiy and Rostron, 2012a; Peterman et al., 2012). The
30 Bakken wells that contained external water were found to all contain water from the Mississippian
31 Lodgepole formation (part of the Madison Group). The average volume of external water was 34%
32 and the external water volume ranged from 10% to 100% (Arkadakskiy and Rostron, 2013a).
33 Another example of a well blowout associated with a hydraulic fracturing operation occurred in
34 Clearfield County, PA. The well blew out, resulting in an uncontrolled flow of approximately
35 35,000 gal (132,000 L) of brine and fracturing fluid, along with an unquantified amount of gas;
36 some of the fluids reportedly reached a nearby stream (Barnes, 2010). The blowout occurred while

1  the company was drilling out the plugs used to isolate one fracture stage from another. An
2  independent investigation found that the primary cause of the incident was that the only blowout
3  preventer on the well had not been properly tested. In addition, the company did not have certified
4  well control experts on hand or a written pressure control procedure (Vittitow, 2010). In North
5  Dakota, a blowout preventer failed, causing a release of between 50 and 70 barrels per day (2,100
6  gal/day or 7,900 L/day and 2,940 gal/day or 11,000 L/day) of flowback and oil (Reuters, 2014). A
7  3-ft berm was placed around the well for containment. Frozen droplets of oil and water sprayed on
8  a nearby frozen creek. Liquid flowing from the well was collected and trucked offsite. Multiple well
9  communication events reported by the media have also led to flowback and produced water spills
10 ranging from around 700 to 35,000 gal (2,600 L to 130,000 L) (Vaidyanathan, 2013a).

### 7.7.2.3.  Leaks from Pits and Impoundments

11 Leaks of flowback and produced water from on-site pits and impoundments have caused releases
12 as large as 57,000 gal (220,000 L) and have caused surface and ground water impacts
13 (Vaidyanathan, 2013b; PBFC 2011; PADEP 2010). VOCs have been measured in groundwater near
14 the Duncan Oil Field in New Mexico downgradient of an unlined pit storing produced water (Sumi,
15 2004; Eiceman, 1986). Aspects of environmental transport from unlined pits are discussed below in
16 Section 7.8.5.

17 Two of the EPA's retrospective case studies found potential impacts from produced water
18 impoundments. In the southwest Pennsylvania case study (U.S. EPA, 2015k), elevated chloride
19 concentrations and their timing relative to historical data suggested a recent ground water impact
20 to a private water well occurred near an impoundment. The water quality trends suggested that the
21 chloride anomaly was related to the impoundment, but site-specific data were not available to
22 provide definitive assessment of the causes(s) and the longevity of the impact. Evaluation of other
23 water quality parameters did not provide clear evidence of flowback or produced water impacts.

24 In the Wise County, TX case study (U.S. EPA, 2015m), impacts to two water wells were attributed to
25 brine, but the data collected for the study were not sufficient to distinguish among four possible
26 sources, one of which was leaks from reserve pits and/or impoundments. The others were: brine
27 migrating from underlying formations along wellbores, brine migrating from underlying formations
28 along natural fractures, and brine migrating from a nearby brine injection well. Alternate sources
29 for the impacts were considered, including road salting, landfill leachate, septic tanks, and animal
30 wastes, but evaluation of data showed that these were not likely. A third well experienced similar
31 impacts, but a landfill leachate source could not be ruled out in that case. Ricther and Kreitler
32 (1993) reviewed sources of salinity to ground water resources by evaluating reviewing major
33 sources, which included natural saline ground water, halite dissolution, sea-water intrusion, oil-
34 field brine, agriculture, saline seeps and road salt. For each source Ricther and Kreitler (1993)
35 provided a state-by-state review of the potential occurrence, which can be used as a general guide
36 to potential sources of salt at a specific area of interest.

### 7.7.2.4.  Data Compilation Studies

37 Environmental impacts from hydraulic fracturing-related fluids have been explored to a limited
38 extent in recent scientific literature (Brantley et al., 2014; Farag and Harper, 2014; Gross et al.,

1   2013; Olmstead et al., 2013; Papoulias and Velasco, 2013; Vidic et al., 2013; Considine et al., 2012;
2   Rozell and Reaven, 2012).

3   From an Oklahoma Corporation Commission database of almost 13,000 releases reported from
4   1993 to 2003, Fisher and Sublette (2005) determined that the primary origins of produced water
5   releases were leaks from lines, tanks, wellheads, with lesser numbers of releases from surface
6   equipment, and pits. The most common cause was overflows followed by illegal activity, storms, fire
7   accidents and corrosion. For these types of releases, the median release volume ranged from 20 bbl
8   (840 gal or 3,180 L) to 60 bbl (2,500 gal or 9,500 L), and the maximums from 200 bbl (8,400 gal or
9   31,800 L) to 2,800 bbl (118,000 gal or 445,200 L).

10   As noted in Text Box 5-14 of Chapter 5, U.S. EPA (2015n) characterized hydraulic fracturing-related
11   spills. Of the spills related to hydraulic fracturing activities (457 spills), 225 (49%) were spills of
12   flowback and produced water. These spills were characterized with respect to volumes, spilled
13   materials, sources, causes, environmental receptors, containment, and response. Most of the
14   produced water spills in the EPA study occurred in Colorado (48%) and Pennsylvania (21%).
15   Flowback and produced water constitute 84% (approximately 2.0 million gal or 7.6 million L) of the
16   total volume of hydraulic fracturing-related spills as calculated from Appendix B of U.S. EPA
17   (2015n).[1] Flowback and produced water spills were characterized by numerous low-volume spills;
18   half of the spills with reported volumes were less than 1,000 gal (3,800 L), and few spills exceeded
19   10,000 gal (38,000 L). Of the volume of spilled flowback and produced water, 16% was recovered
20   for on-site use or disposal, 76% was reported as unrecovered, and 8% was unknown. The potential
21   impact of the unknown and unrecovered volume on drinking water resources is unknown.

22   Known sources for flowback and produced water spills include storage containers (e.g., pits,
23   impoundments, or tanks), wells or wellheads, hoses or lines, and equipment. Storage containers
24   accounted for 58% of flowback and produced water spills. The fewest spills occurred from wells
25   and wellheads, but these spills had the greatest spill volumes compared to all other sources.

26   The causes of these spills were human error (38%), equipment failure (17%), failures of container
27   integrity (13%), miscellaneous causes (e.g., well communication, well blowout), and unknown
28   causes. Most of the volume spilled (74%), however, came from spills caused by a failure of
29   container integrity.

30   In some of the cases, spills reached environmental receptors: soil (141 spills), surface water (17
31   spills), and ground water (1 spill); of these spills, 13 reached both soil and surface water.
32   Consequently 146 unique produced water spills reached environmental receptors, accounting for
33   65% of the 225 cases and accounting for approximately 422,000 gal (1.60 million L) of flowback
34   and produced water. Spills with known volumes that reached a surface water body ranged from

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[1] Chemicals and products, fracturing fluid, fracturing water, equipment fluids, hydrocarbons, and unknown fluids
constitute the additional 16% (approximately 360,000 gal or 1.4 million L) of the total volume of hydraulic
fracturing-related spills as calculated from Appendix B of U.S. EPA (2015d).

1　less than 170 gal (640 L) to almost 74,000 gal (280,000 L). In 30 cases, it is unknown whether a spill
2　of flowback and produced water reached an environmental receptor of any type.

3　Gross et al. (2013) analyzed the Colorado Oil and Gas Conservation Commission's database for
4　ground water BTEX concentrations linked to hydraulic fracturing-related surface spills between
5　July 2010 and July 2011 in Weld County, CO. Only spills with an impact on ground water were
6　included in the study. The 77 reported spills accounted for less than 0.5% of nearly 18,000 active
7　wells. Forty-six of the 77 spills consisted of produced water and oil. Of the remaining spills, 23
8　consisted of only oil and 8 consisted of only produced water. Thus the results that follow include
9　cases with no produced water spill. From these composited spills, benzene concentrations in 90%
10　of the ground water samples exceeded 5 µg/L, the U.S. drinking water standard. Additionally, 30%
11　of toluene, 12% of ethylbenzene, and 8% of xylene sample concentrations exceeded 1 mg/L, 0.7
12　mg/L and 10 mg/L, respectively (Gross et al., 2013).

13　Based on five spills for which volumes were reported, the average volume of a produced water spill
14　was 294 gal (1,110 L), ranging from 42 (160 L) to 1,176 gal (4,450 L) (Gross et al., 2013). Spill areas
15　averaged 2,120 ft$^2$, with an average depth of 7 ft. Tank battery systems and production facilities
16　were the biggest volume sources of spills with ground water impacts. Equipment failure was the
17　most common cause of spills with ground water impacts. Shallow ground water within the study
18　area (Niobrara Shale within the Denver-Julesburg Basin) is the main source of water for residents
19　due to limited surface water availability. Of the 77 reported spills, secondary containment was
20　absent from 51 of them (Gross et al., 2013).

21　As noted from the Colorado (Gross et al., 2013) and Oklahoma (Fisher and Sublette, 2005) studies,
22　oil releases may occur alongside produced water spills. Review of recent oil field incidents in North
23　Dakota also shows incidents with both produced water and oil releases
24　(http://www.ndhealth.gov/EHS/Spills/). Oil releases are characterized by a number of features
25　including their unique hydrocarbon composition and physical properties. Impacts can include:
26　surface runoff, infiltration into soils, formation of sheens and oil slicks on surface waters,
27　evaporation, oxidation, biodegradation, emulsion formation, and particle deposition (U.S. EPA,
28　1999).

29　A statistical analysis of oil and gas violations in Pennsylvania found that violations regarding
30　structurally unsound impoundments or inadequate freeboard (vertical distance from the surface
31　water level to the overflow elevation) were the second most frequent type of violation with 439
32　instances in the period from 2008 to 2010 (Olawoyin et al., 2013). In a study of pits and
33　impoundments in West Virginia, Ziemkiewicz et al. (2014) found common problems of slope
34　stability and liner deficiencies. Construction quality control and quality assurance were often
35　inadequate; the authors found a lack of field compaction testing, use of improper types of soil,
36　excessive slope lengths, buried debris, and insufficient erosion control (Ziemkiewicz et al., 2014).

37　Brantley et al. (2014) reviewed PA DEP's online oil and gas compliance database for notices of
38　violation issued to companies developing unconventional gas resources. Between May 2009 and
39　April 2013, 8 spills of flowback and produced water ranging from more than 4,000 gal (15,000 L) to

1    more than 57,000 gal (220,000 L) reached surface water resources. The spills typically resulted in
2    local impacts to environmental receptors and required remediation and monitoring. However, the
3    study indicated the likelihood of a leak or spill of hydraulic fracturing-related fluids was low (less
4    than 1%, based on 32 large spills out of more than 4,000 complete wells). Due to lack of data,
5    specific impacts to the eight receiving surface waters were not discussed, other than noting the
6    produced water had contacted the surface water.

7    The Brantley et al. (2014) analysis of the PA DEP positive determination letters written in response
8    to water-user complaints illustrates the importance of pre-drilling sampling, as it is one criterion
9    that allows operators to refute the presumption that drilling caused water supply impacts (see
10   Chapter 6). The importance of this pre-drilling sampling and analysis is highlighted by naturally
11   occurring exceedances of EPA secondary MCLs for manganese and iron in private wells in
12   Pennsylvania (Boyer et al., 2011; Williams et al., 1998). Boyer et al. (2011) state that more than
13   40% of private water wells in Pennsylvania fail to meet federal drinking water standards. Boyer et
14   al. (2011) analyzed pre-drilling samples from private water wells in northeastern and
15   southwestern Pennsylvania and showed that 20% (of 222 wells) failed the drinking water standard
16   for iron and 27% (of 203 wells) failed for manganese.[1] Williams et al. (1998), in their evaluation of
17   over 200 wells in Bradford, Tioga, and Potter counties in northeastern Pennsylvania, indicate about
18   50% of the wells exceeded secondary MCLs for iron and manganese.[2] According to Boyer et al.
19   (2011), higher concentrations of these constituents tend to be associated with the sodium chloride
20   (Na-Cl) type ground water often found in valleys in zones of more restricted ground water flow
21   (portions of aquifers with low permeability). Saline water can be found at shallow depths in these
22   areas (Williams et al., 1998).

23   As an example of another set of criteria for assessing sites potentially contaminated by hydraulic
24   fracturing activities, the EPA (2012f) developed an approach to study sites where the impacts to
25   drinking water resources and the potential sources of the impacts are unknown, but may have been
26   the object of water-user complaints. The approach is based on a tiered scheme where results from
27   each tier are used to refine activities in higher tiers. The four tiers were as follows:

28   •  Verify potential issue:

29   o  Evaluate existing data and information from operators, private citizens, state and
30      local agencies, and tribes (as appropriate).

31   o  Conduct site visits.

32   o  Interview stakeholders and interested parties.

33   •  Determine approach for detailed investigations:

34   o  Conduct initial sampling of water wells, taps, surface water and soils.

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[1] Percentage of other parameters failing standards: 17% of 233 wells for pH, 3% of 233 wells for TDS, <1% of 226 wells
for chloride, 1% of 218 wells for Barium, <1% of 177 wells for sulfate, 33% of 125 wells for coliforms, 4% of 115 wells for
arsenic, 8% of 122 wells for fecal coliforms, 32% of 102 samples for turbidity.
[2] Naturally occurring constituents occasionally exceeding EPA primary MCLs in this area include barium, combined
radium-226 and radium-228, and arsenic.

1       o    Identify potential evidence of drinking water contamination.

2       o    Develop conceptual site model describing possible sources and pathways of the
3            reported or potential contamination.

4       o    Develop, calibrate, and test fate and transport model(s).

5    •    Conduct detailed investigations to detect and evaluate potential sources of contamination:

6       o    Conduct additional sampling of soils, aquifer, surface water, and produced water
7            pits/tanks where present.

8       o    Conduct additional testing, including further water testing with new monitoring
9            points, soil gas surveys, geophysical testing, well mechanical integrity testing, and
10           stable isotope analyses.

11      o    Refine conceptual site model and further test exposure scenarios.

12      o    Refine fate and transport model(s) based on new data.

13   •    Determine the source(s) of any impacts to drinking water resources:

14      o    Develop multiple lines of evidence to determine the source(s) of impacts to
15          drinking water resources.

16      o    Exclude possible sources and pathways of the reported contamination.

17      o    Assess uncertainties associated with conclusions regarding the source(s) of
18          impacts.

19 This tiered assessment strategy provides an outline for collecting data and evaluating lines of
20 evidence for determining whether impacts have occurred.

### 7.7.3. Case Studies of Potentially Impacted Sites

### *7.7.3.1. Flowback and Produced Water Release from an Illegal Discharge Impacts Surface and Ground Water in Lycoming County, Pennsylvania*

21 An estimated 6,300 gal to more than 57,000 gal (24,000 to 220,000 L) of Marcellus Shale produced
22 water was illegally discharged at XTO Energy Inc.'s Marquardt pad and flowed into the
23 Susquehanna River watershed in November 2010 (U.S. EPA, 2013g).[1] Overland and subsurface flow
24 of released fluids affected proximal surface water, a subsurface spring, and soil. No impacts to
25 drinking water wells and springs within 1 mile of the release were observed at the last sampling
26 date (17 days post-spill). However, residual, soil-associated produced water constituents could
27 reach drinking water resources in the future through surface runoff or infiltration to the ground
28 water (Science Applications International Corporation, 2010). The release, which occurred at XTO's
29 Marquardt 8537H well pad in Penn Township, Lycoming County, PA, was discovered after a routine
30 inspection by the Pennsylvania Department of Environmental Protection. Subsequent investigation

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[1] Violations associated with this incident can be found at the Pennsylvania Department of Environmental Protection's Oil and Gas Compliance Report database found at http://www.portal.state.pa.us/portal/ server.pt/community/oil_and_gas_compliance_report/20299 under the following inspection IDs: 1928978, 1928992, and 1929005.

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1    revealed that flowback and produced water had been discharging into surface waters for over
2    two months after the fluid was released from multiple tanks with open or missing valves on
3    November 16, 2010 (U.S. EPA, 2013g).

4    Geochemical characterization of this produced water indicated concentrations of barium, chloride,
5    iron, manganese, and TDS above Pennsylvania's surface water quality standards, and above the
6    statewide health standards for medium-specific concentrations (SHS MSCs) for ground water use in
7    residential and nonresidential settings (Science Applications International Corporation, 2010). The
8    produced water also contained elevated levels of bromide, calcium, sodium, and strontium, which
9    lack state surface water quality standards and SHS MSCs (Science Applications International
10   Corporation, 2010).

11   Post-spill surface water delineations indicated that released fluids migrated to an unnamed stream
12   known as Tributary 19617. The released fluids migrated approximately 1,400 ft (427 m) overland
13   to a depression which contains a natural fracture with a hydrological connection to a spring
14   (Schmidley and Smith, 2011). The distance from the depression to the stream is approximately
15   600 ft (183 m). Released fluids also drained through surface soils into ground water, which was
16   then released in seeps to the spring and stream; elevated levels of barium, bromide, calcium,
17   chloride, sodium, strontium, and TDS resulted (U.S. EPA, 2013g). Elevated levels of these
18   constituents, particularly barium, bromide, and strontium, were indicative of Marcellus Shale
19   flowback and produced water that had mixed with surface water (Brantley et al., 2014). Barium and
20   chloride were the only dissolved constituents detected in the stream that exceeded state surface
21   water quality standards; the remaining constituents lack established state surface water quality
22   standards.

23   Results from XTO's temporal study of surface water quality confirmed impacts to the stream from
24   produced water. Surface water quality was characterized at the confluence of the stream and
25   spring, and at the stream's upstream and downstream segments, for 65 days post-spill (Science
26   Applications International Corporation, 2010). Downstream barium and bromide levels were one to
27   two orders of magnitude greater than upstream levels through this period. In addition, stream
28   strontium levels were two to three orders of magnitude greater than upstream levels at this time.
29   Chloride was initially detected in the stream with concentrations exceeding state water quality
30   standards (Schmidley and Smith, 2011). Average chloride concentrations for stream samples were
31   two orders of magnitude greater than upstream concentrations (PA DEP, 2011c). By January 2011,
32   stream chloride concentrations had dropped below the limit established by Pennsylvania's surface
33   water quality standards.

34   Delineation of chloride concentrations within on-site soil indicated soil impacts due to overland
35   flow of flowback and produced water (Science Applications International Corporation, 2010). Five
36   hundred tons of affected soil was consequently excavated for off-site disposal. Chloride
37   concentrations decreased with increased distance from the spill site but remained elevated above
38   background levels even at distances of a few thousand feet (Science Applications International
39   Corporation, 2010). Produced water constituents that were present in soil at concentrations above
40   background levels (i.e., barium, sodium, strontium) could be available for long-term runoff and

1  infiltration. For instance, the continued presence of chloride in affected soils is likely due to
2  precipitated chloride salts in soil pores from residual produced water (Science Applications
3  International Corporation, 2010), which may leach later. Near-term sampling (up to 17 days after
4  the spill) found no elevated constituents indicative of runoff or infiltration of produced water when
5  XTO sampled 14 drinking water wells and springs within one mile of the well pad. XTO was ordered
6  to create a storm water collection system for off-site disposal of impacted storm water and to
7  establish on-site water monitoring wells to track long-term ground water quality between the well
8  pad and the stream (Schmidley and Smith, 2011). Other cases of illegal dumping have been
9  reported (Caniglia, 2014; U.S. EPA, 2013g; Paterra, 2011).

### 7.7.3.2.  *Flowback Fluid Reaches Towanda Creek Due to Well Blowout in Bradford County, Pennsylvania, Causing Short-Term Impacts*

10  The Chesapeake Energy ATGAS 2H well, located in Leroy Township, Bradford County, PA,
11  experienced a wellhead flange failure on April 19, 2011, during hydraulic fracturing operations.
12  Approximately ten thousand gallons (38,000 L) of flowback fluids spilled into an unnamed tributary
13  of Towanda Creek, a state-designated trout stock fishery and a tributary of the Susquehanna River
14  (USGS, 2013b; SAIC and GES, 2011). Chesapeake conducted post-spill surface and ground water
15  monitoring (SAIC and GES, 2011). In addition, the EPA, PADEP, and Chesapeake collected split
16  samples from seven private wells within the vicinity of the blowout. The EPA requested that the
17  Agency for Toxic Substances and Disease Registry (ATSDR) evaluate the environmental data
18  collected from seven private wells to determine whether harmful health effects would be expected
19  from consuming or using the well water. Data from Pre-blowout private well samples, collected
20  approximately six months prior to drilling activity at the site, were included in the evaluation.

21  Between the pre- and post-blowout samples, ATSDR (2013) determined that there was factor of ten
22  increases in some analyte concentrations (methane, barium, calcium, chloride, magnesium,
23  manganese, potassium, and sodium) and a factor of 7 increase in iron concentration in one well
24  (RW04) near the site. Other wells showed elevated levels of certain analytes.[1] ATSDR concluded
25  that although the available data *suggested* that the ground water near this site is impacted by gas
26  activities, the data for RWO4 did not conclusively indicate an impact. ATSDR (2013) concluded that
27  further evaluation is needed to characterize any relationship between the drinking water wells and
28  aquifers as a result of changes in site conditions. Further sampling would be required to determine
29  current impacts, trends, and chronic exposures to ground water constituents related to natural gas
30  activities.

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[1] Elevated sodium levels were detected in 6 wells, levels in 5 of them (RW02, RW03, RW05, RW06, and RWO7) may be of concern to sensitive subpopulations; while the last (RW04) would exceed the dietary guideline for both sensitive and the general population. ATSDR judged that elevated lithium concentration in two wells (RW04 and RW06) could be a concern to individuals undergoing lithium therapy. One well (RW02) showed elevated arsenic concentrations, but these were similar in the pre- and post-blowout samples. Gross alpha radiation levels were above the EPA maximum contaminant level in one well (RW03), and ATSDR did not expect adverse health effects from drinking this water. ASTDR did not expect adverse health effects for the user of five private wells (RW01, RW03, RW05, RW06 (excepting for possible lithium impacts) and RW07).

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

### 7.7.4.  Roadway Transport of Produced Water

8    Accidents during transportation of hydraulic fracturing produced water are a possible mechanism
9    leading to potential impacts to drinking water. Nationwide data are not available, however, on the
10   number of such accidents that result in impacts. An estimate of releases from truck transport of
11   produced water could be made as follows:

$$Total\ number\ of\ truckloads = \frac{Total\ number\ of\ wells\ \times Produced\ water\ volume\ per\ well}{Produced\ water\ volume\ per\ truck}$$

12   Then the total distance traveled by all trucks is given by:

$$Total\ distance\ traveled = Total\ number\ of\ truckloads\ \times Distance\ per\ truck$$

13   The number of crashes impacting drinking water resources can be estimated from:

$$\begin{aligned}Total\ crashes\ impacting\ drinking\ water\ resources \\ = Fraction\ of\ crashes\ releasing\ waste\ that\ impacts\ drinking\ water\ resources \\ \times Fraction\ of\ all\ crashes\ releasing\ waste\ \times Crashes\ per\ distance\end{aligned}$$

14   Estimates of all but one of the quantities in these calculations can be made from various literature
15   sources, which are described in Appendix E. A key parameter is the number of crashes of trucks per
16   distance traveled. In 2012, the U.S. Department of Transportation (DOT) estimated that the number
17   of crashes per 100 million highway miles driven of a type of large truck was 110, which is a
18   relatively small number. A key parameter that is unknown is the number of crashes which impact
19   drinking water resources, so definitive estimates of impacts to drinking water resources cannot be
20   made. Alternatively, as an upper bound on drinking water resource impacts, the number of crashes
21   which release waste can be estimated. Based on various assumptions and scenarios presented in
22   Appendix E, the number of crashes with releases is bounded by the low tens of events. At 20 m³ per
23   truckload, the volumes are low relative to the typical volume of produced water.

24   Several limitations are inherent in this analysis, including differing rural road accident rates and
25   highway rates, differing produced water endpoints, and differing amounts of produced water
26   transported. Further, the estimates present an upper bound on impacts, because not all releases
27   would reach or impact drinking water resources.

### 7.7.5. Studies of Environmental Transport of Released Produced Water

1    In this subsection, we describe transport study results that illustrate how produced waters have
2    been shown to be transported from historical disposal practices and spills. Over the history of oil
3    production in the U.S., produced water disposal methods have evolved from land application to
4    storage in unlined ponds to deep well injection (Whittemore, 2007), although some unlined pits
5    continue in use where allowed by states. The changes in practice occurred because of pollution
6    impacts: first to surface waters and aquatic biota, and then to ground water from disposal ponds.
7    Evaluation of sites contaminated by these historic practices sheds light on the potential for
8    transport of released produced water, as discussed below.

9    Impacts to ground water might occur following a spill on land. When the liquid is highly saline, its
10   migration is affected by its high density and viscosity compared with that of fresh water. When
11   spilled flowback or produced water flows over land, a fraction of the liquid is subject to infiltration.
12   The fraction depends on the rate of release, surface cover (i.e., pavement, cracked pavement,
13   vegetation, bare soil, etc.), slope of the land surface, subsurface permeability, and the moisture
14   content in the subsurface.

15   The potential for impacts from produced water spills depends on the distance from the source to
16   receptor; the distance depends on local topography. One study investigated receptor distances on a
17   formation basis (Entrekin et al., 2011). The distance between gas wells and drainage ways was
18   determined to average 273 m (890 ft) for the Marcellus Shale and 353 m (1160 ft) for the
19   Fayetteville Shale (Entrekin et al., 2011). Some wells were much closer, being as close as 1 m (3.28
20   ft). For one location in each formation a separate analysis gave a mean estimate of 153 m (500 ft)
21   for the Marcellus Shale and 130 m (430 ft) for the Fayetteville Shale. The average distance to public
22   drinking water intakes was 15 km (9.32 mi). The average distance to public water supply wells was
23   37 km (23.0 mi) for the Marcellus Shale and 123 km (76.4 mi) for the Fayetteville Shale. As the
24   density of gas development increases the number of gas wells located close to drainage ways and
25   public water supply wells may also increase.

26   For example, Whittemore (2007) described a site with relatively little infiltration due to moderate
27   to low permeability of silty clay soil and low permeability of underlying shale units. Thus, most of
28   the historically surface-disposed produced water at the site flowed into surface drainages.
29   Observed historic levels of chloride in receiving waters resulted from the relative balance of
30   produced water releases and precipitation runoff, with high concentrations corresponding to low
31   stream flows. Persistent surface water chloride contamination was attributed to slow flushing and
32   discharge of contaminated ground water.

33   Transport from the land surface to the water table is further characterized in general by flow
34   through variably water-saturated media, preferential flow paths, fractures in clays, and
35   macropores. Preferential flow paths along microscale heterogeneities are known to exist and
36   dominate transport even after cycles of repeated drying and rewetting. The effect of flowback on
37   transport of colloids has recently been evaluated in laboratory sand columns. The authors found
38   that flowback increased the mobility of colloidal particles, which potentially serve as a source of
39   aquifer contamination (Sang et al., 2014).

1    In another study, Otton et al. (2007) reported on a site in Oklahoma where two abandoned pits
2    were major sources for releases of produced water and oil. Saline water from the pits flowed
3    through thin soils and readily percolated into underlying permeable bedrock. Deeper, less
4    permeable bedrock was contaminated by salt water later in the history of the site, presumably due
5    to fractures (Otton et al., 2007). The mechanisms proposed were lateral movement through
6    permeable sand bodies, vertical movement along shale fractures, and possibly increased
7    permeability from clay flocculation and shrinkage due to the presence of highly saline water (Otton
8    et al., 2007).

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

17   Generally, the deeper that brine can move into an aquifer, as impacted by the volume and timing of
18   the release, the longer the duration of contamination (Whittemore, 2007). Kharaka et al. (2007)
19   reported on studies at a site in Oklahoma with one abandoned and two active unlined brine pits.
20   Produced water from these pits penetrated 3- to 7-m thick shale and siltstone units, creating three
21   plumes of high-salinity water (5,000 to 30,000 mg/L TDS). The impact of these plumes on the
22   receiving water body (Skiatook Lake) was judged to be minimal, although the estimate was based
23   on a number of notably uncertain transport quantities (Otton et al., 2007).

24   Chloride impacts from produced water spills were studied through scenario modeling releases of
25   100 bbl (4,200 gal or 15,900 L) and 10,000 bbl (420,000 gal or 1.59 million L) (API, 2005). The
26   scenarios included transport through a homogeneous or heterogeneous unsaturated zone using the
27   HYDRUS-1D model (Šimunek et al., 1998) and mixing within the top portion of a shallow aquifer
28   using a specially developed spreadsheet model. The results of the scenario modeling indicated that
29   ground water quality is unlikely to be impaired for spills with small soil penetration depths, which
30   correspond to spills distributed over large areas. Large spills of 100,000 bbl (4.2 million gal or 15.9
31   million L) over sandy unsaturated zones were found to have a high potential to impact ground
32   water quality (API, 2005). Spills of less than 100 bbl (420 gal or 1,590 L) were not modeled and
33   were presumed to have low impacts based on the results from the larger spills. The results were
34   constrained by the underlying assumptions of HYDRUS-1D—that there were no preferential flow
35   paths, including fractured systems, systems with macropores, or fine scale heterogeneities. More
36   rapid and spatially extensive transport could occur in these settings.

### 7.7.6.  Coalbed Methane

37   A CBM produced water impoundment in the Powder River Basin of Wyoming was studied for its
38   impact on ground water (Healy et al., 2011; Healy et al., 2008). Infiltration of water from the
39   impoundment was found to create a perched water mound in the unsaturated zone above bedrock

1   in a location with historically little recharge. The subsurface sediments were found to be highly
2   heterogeneous both physically and chemically, which increased the complexity of studying the site.
3   Elevated concentrations of TDS, chloride, nitrate, and selenium were found at the site. For example,
4   TDS exceeded 100,000 mg/L in one lysimeter sample, while the concentration was 2,275 mg/L in a
5   composite CBM produced water sample (Healy et al., 2008). Most of the solutes found in the ground
6   water mound did not originate with the CBM produced water, but rather were the consequence of
7   dissolution of previously existing salts and minerals. The mechanisms were thought to be gypsum
8   dissolution, cation exchange, and pyrite dissolution. Data from other sites indicated that the study
9   site's ground water chemistry may not be typical and that the same phenomena may not occur at
10  other sites in the basin (Healy et al., 2011).

### 7.7.7.  Transport Properties

11  The identified constituents of flowback and produced water include inorganic chemicals in the form
12  of cations and anions (including various types of metals, metalloids, and non-metals, and
13  radioactive materials, among others) and organic chemicals, including identified compounds in
14  various classes, and unidentified materials measured as TOC and DOC. Environmental transport of
15  these chemicals depends on properties of the chemical and properties of the environment, and is
16  extensively discussed in Section 5.8.3. In this section we discuss the characteristics of transport for
17  inorganics and note that some inorganics may move with the water, while many of the others are
18  influenced by reactions. For organic chemicals identified in produced water, we discuss EPI Suite™
19  estimates of the main transport parameters identified in Chapter 5, while noting the influences of
20  salinity and temperature on these properties.

21  Transport of inorganic chemicals depends on the nature of ground water and vadose zone flow, and
22  potential reactions among the inorganic chemical, solid surfaces, and geochemistry of the water.
23  Some inorganic anions (i.e., chloride and bromide) move with their carrier liquid and are mostly
24  impacted by physical transport mechanisms: flow of water and dispersion. In addition to the flow-
25  related processes, transport of most inorganics is driven by three mechanisms related to
26  partitioning to the solid phase: adsorption, absorption, and precipitation. The effects of these
27  mechanisms depend on both chemical and environmental characteristics, including the surface
28  reactivity, solubility, and redox sensitivity of the contaminant; and the type and abundance of
29  reactive mineral phases, and the ground-water chemistry (U.S. EPA, 2007). Through the use of
30  transport models, the effects of physical transport mechanisms and chemical processes are
31  integrated. Examples of transport models for reactive metals include the Geochemist's Workbench
32  (Bethke, 2014) and Hydrus (Šimunek et al., 1998).

33  Properties of organic chemicals which tend to affect the likelihood that a chemical will reach and
34  impact drinking water resources if spilled include high chemical mobility in water, low volatility,
35  and high persistence in water. Using the EPA chemical database EPI Suite™, we were able to obtain
36  actual or estimated physicochemical properties for 86 of the 134 organic chemicals identified in
37  produced water and listed in Appendix A. A portion of these, 66, are used in the chemical mixing
38  stage (see Appendix Table C-8). EPI Suite™ results were generated for solubility, octanol water
39  partition coefficient ($K_{ow}$), and the Henry's constant (see Figure 7-9). The log $K_{ow}$ values are of the

1   identified organic chemicals skewed positively, indicating some of the chemicals have low mobility
2   which may result in less extensive dissolved contaminant plumes in ground water. These
3   compounds, however, have a higher tendency to sorb to particulate or colloidal materials and hence
4   be transported in systems where particle transport is dominant, for example: colloid transport in
5   ground water and sediment transport in surface water. The log Henry constant values tend to be
6   below 0 indicating that at equilibrium the concentration in air is less than the concentration in
7   water. This behavior is reflected in the log solubility plot, as the solubilities skew conversely toward
8   high values.

9   The EPI Suite™ results are constrained by their applicability to one temperature (25 °C), and
10  salinity (low). Temperature changes impact Henry's constant, $K_{ow}$, and solubility, and depend on the
11  characteristics of the chemical and ions present (Borrirukwisitsak et al., 2012; Schwarzenbach et
12  al., 2002). In some cases the effect changes exponentially with salinity (Schwarzenbach et al., 2002).
13  Therefore, property values that depart from the EPI Suite™ values are expected for produced water
14  at elevated temperature and salinity.
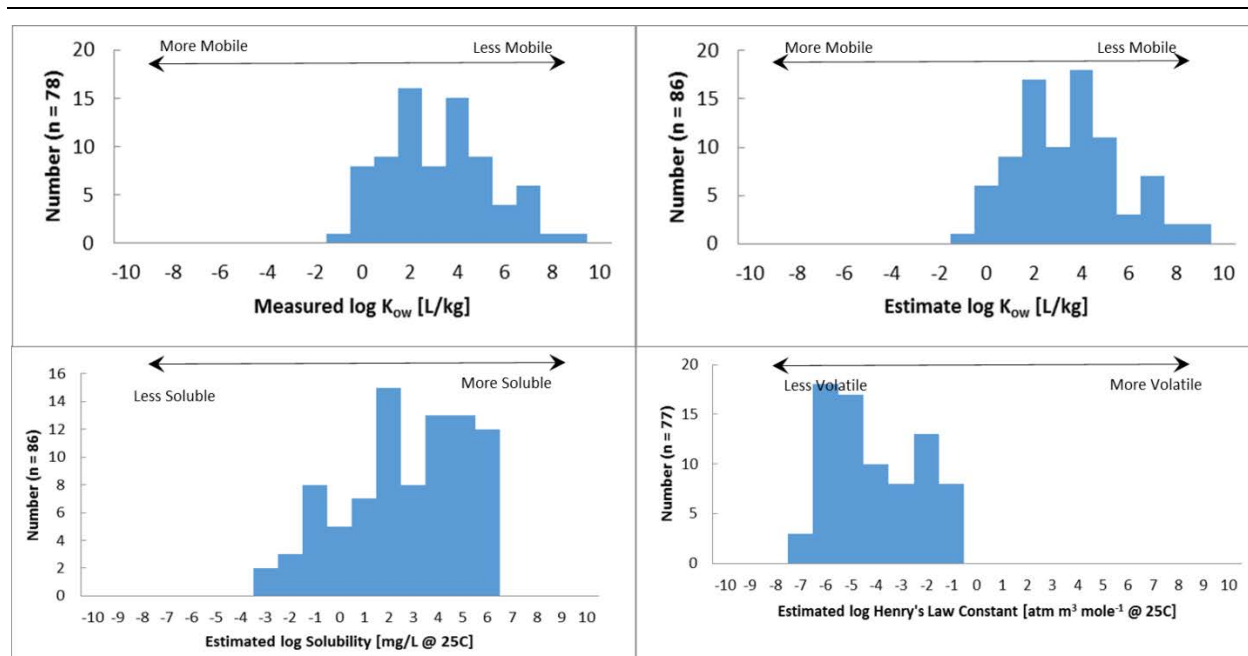


**Figure 7-9. Histograms of physicochemical properties of 86 organic chemicals identified in produced water (physicochemical properties estimated by EPI Suite™).**

## 7.8.   Synthesis

15  After hydraulic fracturing is completed, the operator reduces injection pressure and water is
16  allowed to flow back from the well to prepare for oil or gas production. The flowback water may
17  contain fracturing fluid, fluid from the surrounding formation, and hydrocarbons. Initially this
18  flowback is mostly fracturing fluid, but as time passes, the produced water becomes more similar to
19  the water in the formation. This water is stored at the surface for eventual reuse or disposal.

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1  Impacts to drinking water from flowback and produced water can occur if spilled flowback or
2  produced water enters surface water bodies or aquifers.

### 7.8.1.  Summary of Findings

3  The volume and composition of flowback and produced water vary geographically, both within and
4  between different formations with time and with other site-specific factors. High initial rates of
5  flowback decrease as time continues. The amount of fracturing fluid returned to the surface varies,
6  and typically averages 10% to 25%. In most cases, lower flow rates of produced water continue
7  throughout gas production.[1] In some formations (i.e., the Barnett Shale), the ultimate volume of
8  produced water can exceed the volume of hydraulic fracturing fluid because of inflow of water.

9  The composition of flowback changes with time as the hydraulic fracturing fluid contacts the
10 formation and mixes with the formation water. At the same time, reactions occur between the
11 constituents of the fracturing fluid and the formation. Although varying within and between
12 formations, shale and tight gas produced water typically contains high levels of TDS (salinity) and
13 associated ionic constituents (bromide, calcium, chloride, iron, potassium, manganese, and sodium).
14 Produced water may contain toxic materials, including barium, cadmium, chromium, lead, mercury,
15 nitrate, selenium, and BTEX. CBM produced water can have lower levels of salinity if its coal source
16 was deposited under freshwater conditions.

17 Flowback and produced water spills are known to have occurred across the country. The causes
18 identified for these spills are container and equipment failures, human error, well communication,
19 blowouts, pipeline leaks, and illegal dumping. Spills due to trucking accidents are possible, but
20 accident rates in the United States suggest only a small number of such releases occur.

21 USGS studies of impacts of produced water disposal in unlined pits document the potential for
22 surface releases (in these cases over multiple years) that have led to ground water impacts.
23 Contaminant plumes can be traced to high TDS water disposed of in the pits, or geochemical
24 reaction between infiltrating low-TDS water, in the case of CBM produced water, releasing existing
25 minerals from the unsaturated zone to ground water.

### 7.8.2.  Factors Affecting the Frequency or Severity of Impacts

26 The potential of spills of flowback and produced water to affect drinking water resources depends
27 upon the release volume, duration, and composition. Larger spills of greater duration are more
28 likely to reach a nearby drinking water resource than are smaller spills. The composition of the
29 spilled fluid will also impact the severity of a spill, as certain constituents are more likely to affect
30 the quality of a drinking water resource. Low-volume and short-duration spills are less likely to
31 cause impacts, (see Section 7.7.5).

32 Potential impacts to water resources from hydraulic fracturing-related spills are expected to be
33 affected by watershed and water body characteristics. For example, overland flow is affected by
34 surface topography and surface cover. Infiltration of spilled produced water reduces the amount of

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[1] Note that increasing produced water flow rates are indicative of water breakthrough and declining oil production.

1    water threatening surface water bodies. However, infiltration through soil may lead to ground
2    water impacts. Produced water with lower constituent concentrations may produce lesser impacts,
3    but the USGS studies of CBM produced water impoundments described in section 7.7.5 showed
4    impacts caused by CBM water mobilizing existing minerals. The USGS studies on historical disposal
5    of saline produced water in unlined pits detected ground water plumes discharging into surface
6    water bodies. The potential impact to drinking water in such cases depends on the location of
7    drinking water wells and the size of any surface water supply reservoir.

### 7.8.3.  Uncertainties

8    We first discuss data gaps in our overall knowledge of flowback and produced water; closing such
9    gaps would enable us to better predict impacts on drinking water resources. Second, we present
10   uncertainties that, based on site-specific conditions, also affect our ability to determine potential
11   impacts.

### 7.8.3.1.  *General Data Gaps*

12   Knowledge of volume and some compositional aspects of flowback and produced water are known
13   from published sources. Most of the available data on TENORM has come from the Marcellus Shale,
14   where these are typically high in comparison to the limited data available from other formations.
15   Only a few studies have attempted to sample and characterize the organic fraction of flowback and
16   produced water; some data are available for shale and CBM, but none are available for tight
17   formations. The reported organic chemical data from flowback likely does not capture the full range
18   of chemicals that may be present, either as original components of the hydraulic fracturing solution
19   or transformation products generated in the subsurface.

20   Characterization of produced water organics is limited by several factors. Development or use of
21   proper analytical procedures depends upon knowing the identities of injected chemicals. Because
22   the formulation of hydraulic fracturing fluids can contain proprietary chemicals, the exact
23   formulations are not available. In addition, subsurface transformations yield degradation products,
24   which themselves must have appropriate analytical methods. Further difficulties are due to matrix
25   interference from high-TDS produced water. These problems result in the need to develop new
26   methods for analyzing both organics and inorganics in produced water.

27   Nationwide data on spills of flowback and produced water are limited in two primary ways: the
28   completeness of reported data cannot be determined, and individual states' reporting requirements
29   differ (U.S. EPA, 2015n). Despite various studies, the total number of spills occurring in the United
30   States, their release volumes and associated concentrations, can only be roughly estimated because
31   of underlying data limitations.

### 7.8.3.2.  *Uncertainties at Individual Spill Sites*

32   Spills of flowback and produced water present many uncertainties that, in combination, limit our
33   ability to predict impacts on drinking water resources. Spills vary in volume, duration, and
34   composition. The spilled liquid could be fracturing fluid mixed with formation water in a
35   proportion that depends on the time that has passed since fracturing. As described in Section 7.7,
36   spills may originate from blowouts, well communication, aboveground or underground pipeline

1  breaks, leaking impoundments, failed containers, human error (including illegal activities, failure to
2  detect spills, and failure to report spills) or unknown causes. The difference between these causes
3  affect the size and location of the spill. In addition, the factors governing transport of spilled fluid to
4  a potential receptor vary by site: the presence and quality of secondary or emergency containment
5  and spill response; the rate of overland flow and infiltration; the distance to a surface water body or
6  drinking water well; and transport and fate processes. Impacts to drinking water resources from
7  spills of flowback and produced water depend on environmental transport parameters, which can
8  in principle be determined but are unlikely to be known or adequately specified in advance of a
9  spill.

10  Because some constituents of flowback and produced water are constituents of natural waters (e.g.,
11  bromide in coastal surface waters) or can be released into the environment by other pollution
12  events (e.g., benzene from gasoline releases, or bromide from coal mine drainage), baseline
13  sampling prior to impacts is one way to increase certainty of an impact determination. Further
14  sampling and investigation may be used to develop the linkage between a release and a
15  documented drinking water impact. Produced water spill response typically includes delineation of
16  the extent of oiled soils, sheens on water surfaces, and the extent of saline water. Extensive
17  characterization of produced water is typically not part of spill response, and therefore the
18  chemicals, and their concentrations, potentially impacting drinking water resources are not usually
19  known.

### 7.8.4. Conclusions

20  Flowback and produced water has the potential to affect the quality of drinking water resources if it
21  enters into a surface or ground water body used as a drinking water resource. This can occur
22  through spills at well pads, or during transport of flowback. Specific impacts depend upon the spill
23  itself, the environmental conditions surrounding the spill, water body and watershed
24  characteristics, and the composition of the spilled fluid. Flowback and produced water may contain
25  toxic constituents and can potentially render water unpalatable or unsafe to drink. Conclusive
26  determination of impacts to water resources depends on commitment of resources to the
27  implementation of sampling, analysis and evaluation strategies

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**Text Box 7-1. Research Questions Revisited.**

28  ***What is currently known about the frequency, severity, and causes of spills of flowback and produced***
29  ***water?***

30  • Surface spills of flowback and produced water from unconventional oil and gas production have occurred
31     across the country. Some produced water spills have affected drinking water resources, including a few
32     private drinking water wells. The majority of flowback and produced water spills are under 1,000
33     gallons. The causes identified for these are container and equipment failures, human error, well
34     communication, blowouts, pipeline leaks, and illegal dumping.

35  ***What is the composition of hydraulic fracturing flowback and produced water, and what factors might***
36  ***influence this composition?***

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1 • The composition of flowback and produced water must be determined through sampling and analysis,
2 both of which have limitations. The former due to the ability to access through production equipment and
3 the latter due to issues with identifying target analytes in advance of analysis and the existence of
4 appropriate analytical methods.

5 • The geochemical content of water flowing back initially reflects injected fluids. After initial flowback,
6 returning fluid geochemistry shifts to reflect the geochemistry of formation waters and formation solids.

7 • According to available literature and data, conventional and unconventional flowback and produced
8 water content are often similar with respect to the occurrence and concentration of many constituents.

9 • The least statistical variability in produced water content is exhibited between shale gas and tight gas
10 produced water, and the most statistical variability is exhibited between shale gas and coalbed methane
11 produced water.

12 • Much produced water is generally characterized as saline (with the exception of most coalbed methane
13 produced water) and enriched in major anions, cations, metals, naturally occurring radionuclides, and
14 organics.

15 • Shale and coalbed produced water is enriched in benzene. Benzene is a constituent of concern in
16 Marcellus Shale, Raton CBM, and San Juan CBM produced water. Shale produced water is more likely to
17 contain elevated average total BTEX levels than other unconventional produced water.

18 • Typically, unconventional produced water contains low levels of heavy metals. Elevated strontium and
19 barium levels, however, are characteristic of Marcellus Shale flowback and produced water. CBM and, in
20 particular, shale produced water are likely to contain NORM levels of concern.

21 • Composition data were limited. Most of the available data on produced water content were for shale
22 formations and CBM basins, while little data were available for sandstone formations. Additionally, the
23 majority of data were for inorganics, and little data were available for organics. Many more organic
24 chemicals have been reported to have been used in hydraulic fracturing fluid than have been identified in
25 produced water. The difference may be due to analytical limitations, limited study scopes, and
26 undocumented subsurface reactions.

27 • Hydraulic fracturing flowback and produced water composition is influenced by the composition of
28 injected hydraulic fracturing fluids, the targeted geological formation and associated hydrocarbon
29 products, the stratigraphic environment, and subsurface processes and residence time.

30 • Spatial variability of produced water content occurs between plays of different rock sources (e.g., coal vs.
31 sandstone), between plays of the same rock type (e.g., Barnett Shale vs. Bakken Shale), and within
32 formations of the same source rock (e.g., northeastern vs. southwestern Marcellus Shale).

33 ***What are the chemical and  physical properties of hydraulic fracturing flowback and produced water***
34 ***constituents?***

35 • The identified constituents of flowback and produced water include inorganic chemicals (cations and
36 anions in the form of metals, metalloids, non-metals, and radioactive materials), organic chemicals and
37 compounds, and unidentified materials measured as TOC (total organic carbon) and DOC (dissolved
38 organic carbon). Some constituents are readily transported with water (i.e., chloride and bromide), while
39 others depend strongly on the geochemical conditions in the receiving water body (i.e., radium and
40 barium), and assessment of their transport is based on site-specific factors. Using the EPA chemical

database EPI Suite, we were able to obtain actual or estimated physicochemical properties for 86 (64%) of the 134 chemicals identified in produced water.

- As in the case of chemicals in hydraulic fracturing fluid, chemical properties that affect the likelihood of an organic chemical in produced water reaching and impact drinking water resources in the short-term, include: high chemical mobility in water, low volatility, and high persistence in water. In general, EPI suite results suggest that organic chemicals in produced water tend toward lower mobility in water. Consequently these chemicals could remain in soils or sediments nearby spills. Low mobility may result in smaller dissolved contaminant plumes in ground water. Although these compounds are more likely to be transported associated with sediments in surface water or small particles in ground water. Organic chemical properties vary with salinity and the effects depend on the nature of the chemical.

***If spills occur, how might hydraulic fracturing flowback and produced water contaminate drinking water resources?***

- Spills of flowback or produced water might impact drinking water resources if the spill or release is of sufficient volume and duration, to reach the resource at a sufficient concentration. Impacts to in-use drinking water depend on proximity to sources. Scientific literature and published reports have shown that produced water spills have impacted drinking water resources.

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# Chapter 8

# Wastewater Treatment and Waste Disposal

# 8. Wastewater Treatment and Waste Disposal

## 8.1. Introduction

1  Hydraulic fracturing used for the development of oil and gas resources results in the production of
2  wastewater containing a range of problematic or potentially problematic constituents (see Chapter
3  7) and requiring management. For the purposes of this assessment, hydraulic fracturing
4  wastewater encompasses flowback and produced water (often referred to together as produced
5  water) that is managed using any of a number of practices, including treatment and discharge,
6  reuse, or injection into Class IID wells regulated under the Underground Injection Control (UIC)
7  program under the Safe Drinking Water Act (SDWA) (see also Chapter 1).[1] In this chapter, the term
8  "wastewater" is generally used. In limited cases where more specific information is provided about
9  a wastewater (e.g., a source indicates that the wastewater is flowback), that information is also
10  noted.

11  Although wells producing from either unconventional or conventional oil and gas reservoirs
12  generate produced water during the course of their productive lifespan, wells conducting modern
13  high-volume hydraulic fracturing can generate a large volume of flowback water in the period
14  immediately after fracturing. Stakeholders reported to the U.S. Government Accountability Office
15  that flowback volumes could be 420,000 gal to 2.52 Mgal (10,000 to 60,000 bbl or 1.59 million to
16  9.54 million L) per well per hydraulic fracture (U.S. GAO, 2012) (see Chapter 7.1.1 for more
17  information on produced water volumes per well in various geologic basins and plays). This
18  necessitates having a wastewater management strategy in place at the beginning of activities at the
19  well. Selection of management choices may depend upon the quality and volume of the fluids,
20  logistics, and economics.

21  Treatment and disposal strategies vary throughout the United States and may include underground
22  injection, on-site or offsite treatment for reuse in subsequent hydraulic fracturing operations, reuse
23  without treatment, or other uses. In some areas, wastewater may be applied to the land (e.g., for
24  irrigation) or held in pits for evaporation. The large majority of wastewater generated from all oil
25  and gas operations in the United States is disposed of via Class IID wells (Clark and Veil, 2009). As
26  hydraulic fracturing activity matures, costs of different disposal practices may change in various
27  regions due to factors such as regulations, available infrastructure, feasibility and cost of reuse
28  practices, and other concerns that are difficult to anticipate and quantify at the time of this
29  assessment.

30  Over the past decade, the rapid increase in modern hydraulic fracturing activities has led to the
31  need to manage the associated wastewater. There has been a shift towards reuse in areas where

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

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1   there are relatively few Class IID wells (e.g., the Marcellus Shale region) and indications of interest
2   in reuse in areas where access to water for fracturing is limited (e.g., Anadarko Basin in TX and OK).
3   The term reuse is sometimes used to imply no treatment or basic treatment (e.g., media filtration)
4   for the removal of constituents other than total dissolved solids (TDS), while recycling is sometimes
5   used to convey more extensive treatment (e.g., reverse osmosis (RO)) to remove TDS (Slutz et al.,
6   2012). In this document, the term "reuse" will be used to indicate use of wastewater for subsequent
7   hydraulic fracturing, without regard to the level of treatment.

8   This chapter provides follow-on to Chapter 7, which discusses the composition and per-well
9   volumes of produced water and the processes involved in its generation. In this chapter,
10  discussions are included on management practices for hydraulic fracturing wastewaters, available
11  wastewater production information, and estimated aggregate volumes of wastewater generated for
12  several states with active hydraulic fracturing (Section 8.2). As a complement to information on the
13  composition of wastewaters in Chapter 7, issues of concern associated with wastewater
14  constituents are also presented (Section 8.3). Management methods that are used in 2014-2015 or
15  have been used in recent years are described (Section 8.4). Information is then presented on the
16  types and effectiveness of treatment processes that are suitable for removal of constituents of
17  concern in hydraulic fracturing wastewaters, either in centralized waste treatment facilities (CWTs)
18  or on-site treatment; examples of CWTs are presented (Section 8.5 and Appendix F). With the
19  background provided in the earlier sections of the chapter, documented and potential impacts on
20  drinking water resources are discussed (Section 8.6), and a final synthesis discussion is then
21  provided (Section 8.7).

22  This chapter makes use of background information collected by the EPA's Office of Water (OW) as
23  part of its development of proposed effluent limitations guidelines and standards for wastewater
24  from unconventional oil and gas resources (U.S. EPA, 2015q). These are defined by guidelines and
25  standards as resources in low permeability formations including oil and gas shales, tight oil, and
26  low permeability sandstones and carbonates. Coalbed methane is beyond the scope of those
27  standards. In this chapter we consider wastewater generated by hydraulic fracturing of those
28  unconventional oil and gas resources included in the background research done by OW in addition
29  to wastewater generated by hydraulic fracturing in coalbed methane plays and conventional
30  reservoirs.

## 8.2.  Volumes of Hydraulic Fracturing Wastewater

31  This section of Chapter 8 provides a general overview of aggregate wastewater quantities
32  generated in the course of hydraulic fracturing and subsequent oil and gas production, including
33  estimates at regional and state levels. It also discusses methodologies used to produce these
34  estimates and the challenges associated with the preparation and use of available estimates.
35  (Chapter 7 provides a more in-depth discussion of the processes affecting produced water volumes
36  and presents some typical per-well values and temporal patterns.) Wastewater volumes most likely
37  will vary in the future as the amount and locations of hydraulic fracturing activities change and as
38  existing wells age and move into later phases of production. The volumes and management of

1   wastewater are important factors affecting the potential for wastewater to affect drinking water
2   resources.

3   The volume of wastewater generated is generally tied to the volume of oil and gas production; as
4   operators increase hydrocarbon production, there will be a corresponding increase in wastewater
5   volumes to be managed. For example, data from the Pennsylvania Department of Environmental
6   Protection (PA DEP) (PA DEP, 2015a) (see Figure 8-1) show trends in volumes of wastewater
7   compared to gas produced from wells in the Marcellus Shale in Pennsylvania. Although the data
8   show some variation, they demonstrate a general correlation between wastewater and produced
9   gas.
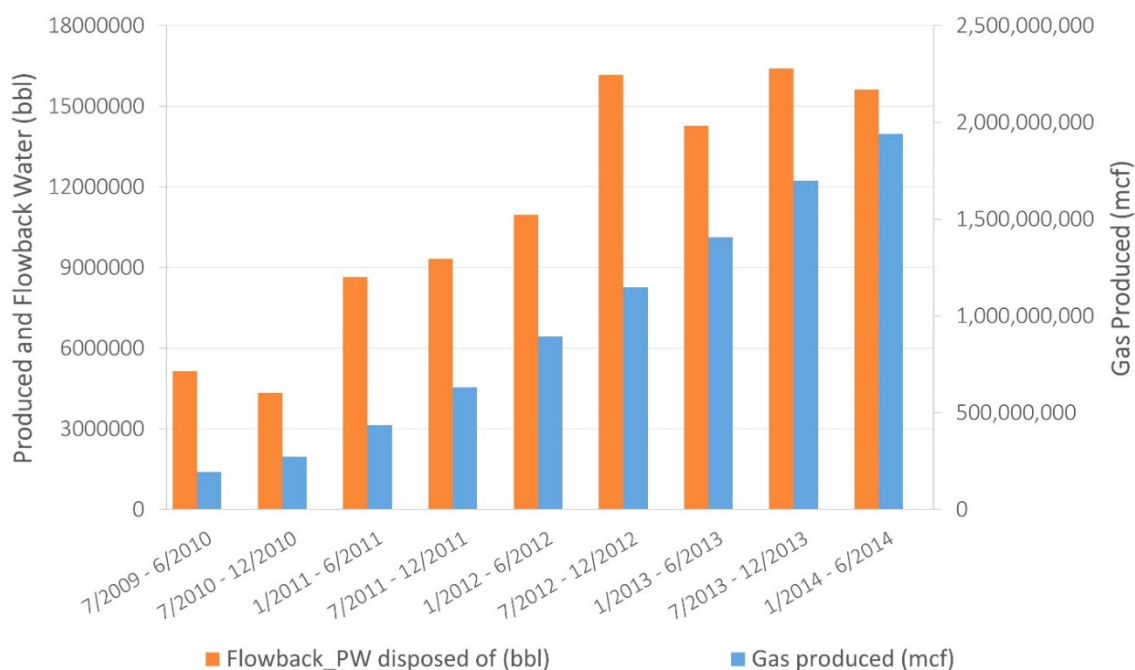


**Figure 8-1. Produced and flowback water volumes and produced gas volumes from unconventional wells in Pennsylvania from July of 2009 through June of 2014.**

Source: PA DEP (2015a).

10  Information presented in Chapter 7 highlights the initial rapid recovery of fluid in the first weeks
11  after fracturing (see Figure 7-2), with a subsequent substantial reduction in the volume of water
12  flowing through the well to the surface. This is followed by recovery of produced water during the
13  longer-term productive phase of the well's life. One source suggests that, as a general rule of thumb,
14  the amount of flowback produced in the days or weeks after hydraulic fracturing is roughly
15  comparable to the amount of long-term produced water generated over a span of years, which may
16  vary considerably among wells (IHS, 2013). Thus, on a local level, operators can anticipate a
17  relatively large volume of wastewater in the weeks following fracturing, with slower subsequent
18  production of wastewater. Wells also generate some amount of drilling-fluid waste. Compared to

1   produced water, however, drilling wastewater can constitute a relatively small portion of the total
2   wastewater produced (e.g., <10% in Pennsylvania during 2004-2013) (U.S. EPA, 2015q) and is not
3   discussed further in this assessment.

4   In addition to variation in per-well wastewater volumes, aggregate wastewater production for an
5   area or region will vary from year to year with hydraulic fracturing activity. For instance, the
6   average annual volume of wastewater generated by all gas production (both shale gas and
7   conventional) in Pennsylvania quadrupled from the 2001-2006 period to the 2008-2011 period.
8   During the latter period, wastewater volume averaged 1.1 billion gal (26 million bbl or 4.2 billion L)
9   per year (Wilson and Vanbriesen, 2012).

### 8.2.1.  National Level Estimate

10   Clark and Veil (2009) estimated that in 2007, the approximately one million active oil and gas wells
11   in the United States generated approximately 2.4 billion GPD (57.4 million bbl/day; 9.1 billion
12   L/day) of wastewater; no newer comprehensive national-level estimate exists in the literature as of
13   April 2015. Note that this estimate is not limited to wastewater from hydraulic fracturing
14   operations. This national-level estimate is reported to represent total oil and gas wastewater (from
15   conventional and unconventional resources, and from wells hydraulically fractured and wells not
16   hydraulically fractured), but the authors note that it does not include the flowback water
17   component. Although Clark and Veil (2009) conducted a state-by-state analysis, the report may
18   have underestimated production due to significant data limitations: 1) data based on a timeframe
19   preceding the dramatic increase in hydraulic fracturing activity seen in more recent years; 2)
20   estimates based on data that were collected and maintained in a variety of ways, making data
21   synthesis difficult; and 3) incomplete data (U.S. GAO, 2012).

### 8.2.2.  Regional/State and Formation Level Estimates

22   The amount of wastewater generated in a given region varies widely depending upon the volume of
23   wastewater generated per well and the number of wells in the area. The factors influencing
24   wastewater production are discussed in Chapter 7, which also discusses differences among
25   formations in volumes recovered during flowback and long-term water production. Table 7-2
26   provides rates of produced water generation for a number of formations in the United States.

27   At an aggregate level, wastewater volumes and geographic variability have been assessed for oil
28   and gas production in several studies. A 2011 study by the Bureau of Land Management (BLM)
29   (Guerra et al., 2011) states that more than 80% of oil and gas wastewater is generated in the
30   western United States, including volumes from both conventional and unconventional resources.
31   The BLM report notes substantial contributions from coalbed methane (CBM) wells, in particular
32   those in the Powder River Basin (Wyoming). The authors state that Wyoming produces the second
33   highest volume of water among the western United States. Guerra et al. (2011) also highlight the
34   large portion of wells and wastewater associated with Texas (44% of U.S. produced water volume).
35   Although the authors do not identify all wastewater contributions from production involving
36   hydraulic fracturing, the regions with established oil and gas production are likely to have methods
37   and infrastructure available for management of hydraulic fracturing wastewater. Figure 8-2
38   summarizes the findings for these western states, demonstrating the wide variability in wastewater

1  from state to state (likely reflecting differences in formation geology and oil and gas production
2  activity).
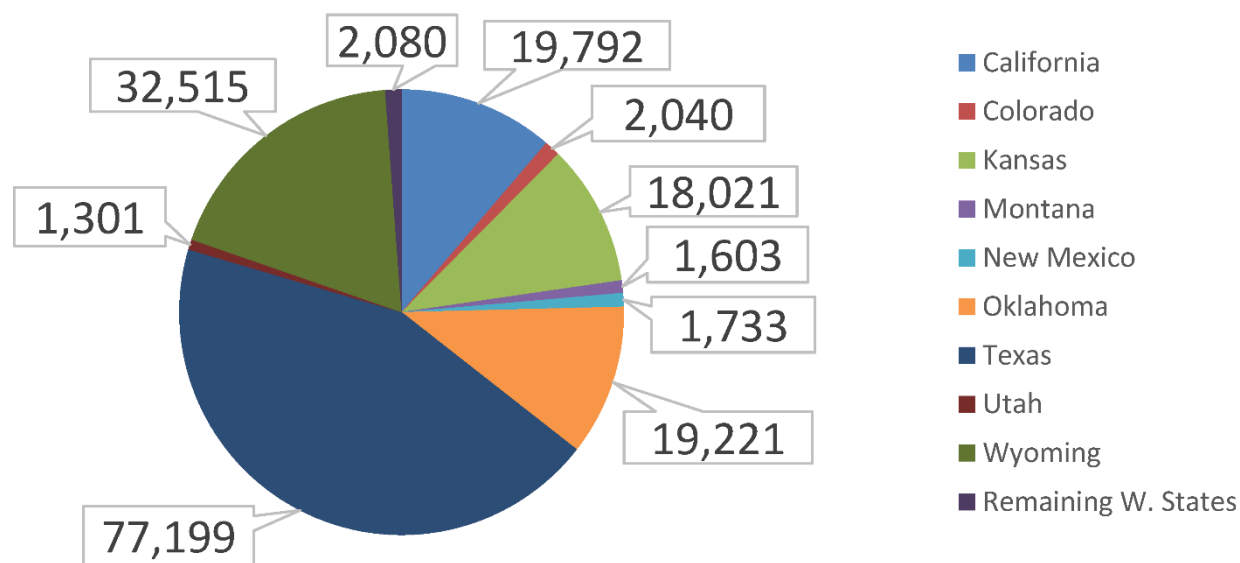


**Figure 8-2. Wastewater quantities in the western United States (billions of gallons per year).**

Source: Guerra et al. (2011).

3   Table 8-1 presents estimates of the numbers of wells and volumes of hydraulic fracturing
4   wastewater generated in North Dakota, Ohio, Pennsylvania, Texas, and West Virginia. The data
5   shown in this table come from publicly available state websites and databases; data for West
6   Virginia reference a report by Hansen et al. (2013) that compiled available flowback data from West
7   Virginia. The reported volumes have been summed by year. These states are represented in Table
8   8-1 because the produced water volumes were readily identifiable as associated with hydraulic
9   fracturing activity. Differences in the years presented for the states are due to differences in data
10  availability from the state agency databases. However, the increases in the numbers of wells
11  producing wastewater and the volumes of wastewater produced are generally consistent with the
12  timing of the expansion of high-volume hydraulic fracturing and track with the increase in
13  horizontal wells seen in Figure 2-12.

14  Several states with mature oil and gas industries (California, Colorado, New Mexico Utah,
15  Wyoming) make produced water volumes publicly available by well as part of their oil and gas
16  production data, but they do not directly indicate which wells have been hydraulically fractured.
17  Some states (Colorado, Utah, and Wyoming) specify the producing formation along with volumes of
18  hydrocarbons and produced water. New Mexico provides data for separate basins as well as for the
19  entire state. Determining volumes of hydraulic fracturing wastewater for these states is challenging
20  because there is a possibility of either inadvertently including wastewater from wells not
21  hydraulically fractured or of missing volumes that should be included. Appendix Table F-1 provides

1    estimates of wastewater volumes in these states in regions where hydraulic fracturing activity is
2    taking place along with notes on data limitations.

3    The data in Table 8-1 and Appendix Table F-1 illustrate the challenges both for compiling a national
4    estimate of hydraulic fracturing wastewater as well as comparing wastewater production among
5    states due to dissimilar data types, presentation, and availability.

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

| State | Basin | Principal Lithologies | Data Type | 2000 | 2004 | 2008 | 2010 | 2011 | 2012 | 2013 | 2014 | Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| North Dakota | Williston | Shale | Produced water | 2 | 3 | 130 | 790 | 1,900 | 4,500 | 8,500 | 9,700 | From North Dakota Oil and Gas Commission website.[a] Data provided for six members of the Bakken Shale. Produced water includes flowback (reports are submitted within 30 days of well completion.) |
| | | | *Wells* | *161* | *152* | *844* | *2,083* | *3,303* | *5,036* | *6,913* | *8,039* | |
| Ohio | Appalachian | Shale | Primarily flowback water | - | - | - | - | 3 | 29 | 110 | - | Data from Ohio DNR Division of Oil and Gas.[b] Utica data for 2011 and 2012. Utica and Marcellus data for 2013. Brine is noted by agency to be mostly flowback. |
| | | | *Wells* | - | - | - | - | *9* | *86* | *400* | - | |
| Pennsylvania | Appalachian | Shale | Flowback water | - | - | - | 92 | 340 | 410 | 350 | 210 | Waste data from PA DEP.[c] 2nd half of 2010 and first half of 2014. Data described as unconventional as determined by formation. Separate codes are provided by PA DEP for flowback and produced water. |
| | | | *Wells* | - | - | - | *334* | *1,564* | *1,622* | *1,465* | *895* | |
| | | | Produced water | - | - | - | 90 | 400 | 730 | 930 | 440 | |
| | | | *Wells* | - | - | - | *1, 035* | *1,826* | *3,665* | *4,761* | *4,889* | |

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| State | Basin | Principal Lithologies | Data Type | 2000 | 2004 | 2008 | 2010 | 2011 | 2012 | 2013 | 2014 | Comments |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Pennsylvania cont. | Appalachian cont. | Shale, cont. | Flowback and produced water | - | - | - | 180 | 740 | 1,100 | 1,300 | 650 | |
| | | | *Wells* | - | - | - | *1,232* | *2,434* | *4,039* | *5,015* | *5,150* | |
| Texas | Unspecified (entire state) | Shale, Sandstone | Flowback water - injected volumes | - | - | - | - | 490 | 2,200 | 3,100 | 2,000 | Waste injection data from Texas Railroad Commission.[d] Monthly totals are provided for entire state. Oct - Dec for 2011, full years for 2012 and 2013, and Jan - Oct for 2014 |
| West Virginia | Appalachian | Shale | Flowback water - Estimated total disposed | - | - | - | 120 | 110 | 59 | - | - | Estimates from Hansen et al. (2013). |

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

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

[c] Pennsylvania Department of Environmental Protection. PA DEP Oil and Gas Reporting Web site. https://www.paoilandgasreporting.state.pa.us/publicreports/Modules/Welcome/Agreement.aspx

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

### 8.2.3. Estimation Methodologies and Challenges

1  Compiling and comparing data regarding wastewater production at the wide array of oil and gas
2  locations in the United States presents challenges, and various approaches are used to estimate
3  wastewater volumes, both at the state and national level. Data from state agency web sites and
4  databases can be a ready source of information, whether publicly available and downloadable or
5  provided directly by agencies upon request. However, due to sometimes significant differences in
6  the types of data collected, mechanisms, formats, and definitions used, data cannot always be
7  directly compared from state to state and can be difficult to aggregate at a national level. The
8  inconsistences encountered in data searches for this assessment agree with recent conclusions by
9  Malone et al. (2015), who noted inconsistences among 10 states with unconventional oil and gas
10  activity in the accessibility, usability, completeness, accuracy, and cost of various types of data (e.g.,
11  wells drilled, production, waste, Class IID wells).

12  One challenge associated with using state production data to estimate the volume of wastewater
13  nationally or regionally is the lack of consistency in data collection (U.S. GAO, 2012). Some states do
14  not include a listing of wastewater (usually listed as produced water volumes) in their publicly
15  available oil and gas production reports, while others do. State tracking of wastewater volumes may
16  or may not include information that helps in determining whether the producing well was
17  hydraulically fractured (e.g., an indicator of resource type or formation). It also might not be clear
18  whether volumes listed as produced water include the flowback component. Some states (e.g.,
19  Colorado) include information on disposal and management methods along with production data,
20  and others do not.

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

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

## 8.3.  Wastewater Characteristics

1  Along with wastewater volume, wastewater characteristics are important for understanding the
2  potential impacts of management and disposal of hydraulic fracturing wastewater on drinking
3  water resources. Chapter 7 provides in-depth detail on produced water chemistry. This section
4  provides brief highlights of the important features of wastewater composition as well as the
5  characteristics of the residuals produced during wastewater treatment.

### 8.3.1.  Wastewater

6  This section briefly discusses why the composition of hydraulic fracturing wastewaters needs to be
7  considered when planning for wastewater management, especially if treatment or reuse are
8  planned. Concerns associated with selected constituents are presented; treatment considerations
9  associated with various wastewater constituents are included in Section 8.5.

#### *8.3.1.1.  Total Dissolved Solids and Inorganics*

10  Wastewaters are generally high in total dissolved solids (TDS), especially waters from shale and
11  tight sandstone formations, with values ranging from less than 1,000 mg/L to hundreds of
12  thousands of mg/L (see Section 7.6.4 and Table 7-4). The TDS in wastewaters from shale
13  formations is typically dominated by sodium and chloride and may also include elevated
14  concentrations of bromide, bicarbonate, sulfate, calcium, magnesium, barium, strontium, radium,
15  organics, and heavy metals (Chapman et al., 2012; Rowan et al., 2011; Blauch et al., 2009; Orem et
16  al., 2007; Sirivedhin and Dallbauman, 2004). Within each play, the minimum and maximum values
17  shown in Table 7-4 suggest spatial variation that may need to be accommodated when considering
18  management strategies such as reuse or treatment. In contrast to shales and sandstones, TDS
19  values for wastewater from CBM formations are generally lower, with concentrations ranging from
20  approximately 250 mg/L to 39,000 mg/L (Benko and Drewes, 2008; Van Voast, 2003) (see
21  Appendix Table E-3). This results in fewer treatment challenges and a wider array of management
22  options.

23  Although TDS has a secondary maximum contaminant level (MCL) (secondary MCLs are non-
24  mandatory water quality standards) of 500 mg/L for aesthetic purposes, it is not considered a
25  health-based contaminant and is therefore not regulated under the EPA's National Primary
26  Drinking Water Regulations, although other standards may apply. For example, a maximum
27  concentration of 500 mg/L has been used by the state of Pennsylvania for some industrial
28  wastewater discharges. Constituents commonly found in TDS from hydraulic fracturing
29  wastewaters may have potential impacts on health or create burdens on downstream drinking
30  water treatment plants if discharged at high concentrations to drinking water resources. Bromide,
31  for example, can contribute to the increased formation of disinfection by-products (DBPs) during
32  drinking water treatment (Hammer and VanBriesen, 2012); see Section 8.6.1.

33  Metals (e.g., barium, cadmium, chromium, lead, copper, manganese, nickel, thallium, and zinc)
34  present in TDS can be toxic to humans and aquatic life at certain concentrations. Health effects of
35  these metals can include kidney damage, liver damage, skin conditions, high blood pressure, and
36  developmental problems (U.S. EPA, 2015i). To ensure safe drinking water, the EPA has established

1   primary MCLs for a number of these constituents. MCLs and action levels for these metals vary from
2   0.002 mg/L for thallium to 1.3 mg/L for copper (U.S. EPA, 2015i). Cadmium has been found in
3   produced water from tight gas formations at concentrations as high as 0.37 mg/L (the MCL is 0.005
4   mg/L), and chromium has been found at concentrations up to 0.265 mg/L (the MCL is 0.1 mg/L)
5   (see Table 7-4).

6   Other constituents of concern among dissolved solids are chloride, sulfate, barium, and boron.
7   Elevated concentrations of chloride and sulfate are of concern because of drinking water aesthetics,
8   and the EPA has established secondary MCLs for both chloride and sulfate of 250 mg/L (U.S. EPA,
9   2015i; Hammer and VanBriesen, 2012). Barium has a primary MCL of 2 mg/L and has been found in
10  some shale gas produced waters at concentrations in the thousands of mg/L (see Table 7-4). Boron
11  is not regulated under the National Primary Drinking Water Regulations, but internal plant
12  specifications for one CWT (e.g., the Pinedale Anticline Facility) and waste discharge requirements
13  (WDR) permit for another (e.g., San Ardo Water Reclamation Facility) limit boron effluent
14  concentrations to 0.75 mg/L (Shafer, 2011; Webb et al., 2009).

### 8.3.1.2. Organics

15  Less information is available about organic constituents in hydraulic fracturing wastewaters than
16  about inorganic constituents, but there are several studies that include some analyses of organic
17  constituents. The organic content in flowback waters can vary based on the chemical additives used
18  and the formation but generally consists of polymers, oil and grease, volatile organic compounds
19  (VOCs), and semi-volatile organic compounds (SVOCs) (Walsh, 2013; Hayes, 2009). Examples of
20  other constituents detected include alcohols, naphthalene, acetone, and carbon disulfide (U.S. EPA,
21  2015i) (see Appendix Table E-10). Wastewater associated with CBM wells may have high
22  concentrations of aromatic and halogenated organic contaminants that that may require treatment
23  depending on how the wastewater will be managed or disposed of (Pashin et al., 2014; Sirivedhin
24  and Dallbauman, 2004). Concentrations of BTEX (benzene, toluene, ethylbenzene, and xylenes), in
25  CBM produced waters are, however, lower than in shale produced waters (see Appendix Table E-9).

26  Certain organic compounds are of concern in drinking water because they can cause damage to the
27  nervous system, kidneys, and/or liver and can increase the risk of cancer if ingested over a period
28  of time (U.S. EPA, 2006). Some organics in chemical additives are known carcinogens, including 2-
29  butoxyethanol (2BE), naphthalene, benzene, and polyacrylamide (Hammer and VanBriesen, 2012).
30  Many organics are regulated for drinking water under the National Primary Drinking Water
31  Regulations. Section 8.6.4 provides further discussion of documented or potential situations in
32  which organic constituents have or might reach drinking water resources.

### 8.3.1.3. Radionuclides

33  Radionuclides are constituents of concern in some hydraulic fracturing wastewaters, with most
34  available data obtained for the Marcellus Shale in Pennsylvania (see Appendix Table E-8). Results
35  from a USGS report (Rowan et al., 2011) indicate that radium-226 and radium-228 are the
36  predominant radionuclides in Marcellus Shale wastewater, and they account for most of the gross
37  alpha and gross beta activity in the waters studied. There are limited data on radionuclides in
38  wastewater from formations other than the Marcellus Shale, but information on the naturally

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1 occurring radioactive material (NORM) in the formations themselves, in particular uranium and
2 thorium, may suggest the potential for high levels of radionuclides in produced water, especially
3 where TDS concentrations are also high. Sections 7.5.4 and 7.6.6 provide further information on
4 radionuclides in formations and in produced waters.

5 The primary radioactive contaminants found in hydraulic fracturing wastewaters (radium, gross
6 alpha radiation, and gross beta radiation) can increase the risk of cancer if consumed at elevated
7 levels over time (U.S. EPA, 2015i). Therefore, the EPA has established drinking water MCLs for
8 combined radium (radium-226 plus radium-228), gross alpha, and gross beta of 5 pCi/L, 15 pCi/L,
9 and 4 millirems/year, respectively (see Section 8.6.2).

### 8.3.2.  Constituents in Residuals

10 Depending on the water being treated and treatment processes used, treatment residuals may
11 consist of sludges, spent media (used filter materials), or brines. Residuals can include constituents
12 such as total suspended solids (TSS), TDS, metals, radionuclides, and organics. The treatment
13 process tends to concentrate wastewater constituents in the residuals. As an illustration of the
14 degree of concentration that can take place, processes such as electrodialysis and mechanical vapor
15 recompression have been found to yield residuals streams with TDS concentrations in excess of
16 150,000 mg/L, from treating waters with influent TDS concentrations of approximately 50,000 –
17 70,000 mg/L (Hayes et al., 2014; Peraki and Ghazanfari, 2014).

18 Also, technologically enhanced naturally occurring radioactive material (TENORM) in wastewaters
19 may cause residual wastes to have elevated gamma radiation emissions (Kappel et al., 2013).[1] One
20 study calculated that typical solids produced by precipitation processes designed to remove barium
21 and strontium from Marcellus Shale wastewater would contain between 2,571 and 18,087 pCi/g of
22 radium in the barium sulfate precipitate (Zhang et al., 2014b). Another similar study using mass
23 balances calculated that sludge from a sulfate precipitation process would average a radium
24 concentration of 213 pCi/g in sludge (Silva et al., 2012). Silva et al. (2012) estimated a radium-226
25 concentration of 58 pCi/g in sludge from lime softening processes, a level that would necessitate
26 disposal of low level radioactive waste.

## 8.4.  Wastewater Management Practices

27 Operators have several strategies for management of hydraulic fracturing wastewaters (see Figure
28 8-3), with the most common choice being disposal via Class IID wells (Clark et al., 2013; Hammer
29 and VanBriesen, 2012). Other practices include reuse in subsequent hydraulic fracturing operations
30 (with varying levels of treatment), treatment at a CWT (often followed by reuse), evaporation (in
31 arid regions), or in some cases, depending on state and local requirements, various other
32 wastewater management strategies (e.g., irrigation, which involves no discharge to waters of the
33 U.S.). The management methods shown in Figure 8-3 represent various strategies, not all of which
34 will happen together.

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[1] Technologically enhanced naturally occurring radioactive materials (TENORM) are radionuclides that have been concentrated or enhanced as the result of human activity.

1   At one time, treatment of unconventional oil and gas wastewaters at publicly owned treatment
2   works (POTWs) was a common practice for wastewater management in the Marcellus region (Lutz
3   et al., 2013). However, this practice has been essentially discontinued following a request from PA
4   DEP that, by May 19, 2011, oil and gas operators stop sending Marcellus Shale wastewater to 15
5   POTWs and CWTs that discharged to surface waters (U.S. EPA, 2015h).

6   Each of these wastewater management strategies may potentially lead to an impact on drinking
7   water resources during some phase of their execution. Such impacts may include accidental
8   releases during transport (see Chapter 7), discharges of treated wastewaters from CWTs or POTWs
9   where treatment for certain constituents has been inadequate, migration of constituents in
10  wastewaters that have been applied to land, leakage from on-site storage pits (see Chapter 7),
11  inappropriate management of residuals (e.g., leaching from landfills or land application), or
12  accumulation of constituents in sediments near outfalls of CWTs or POTWs that have treated
13  hydraulic fracturing wastewater.

14  A reliable census of nationwide wastewater management practices is difficult to assemble due to a
15  lack of consistent and comparable data among states, but Table 8-2 illustrates the variability in the
16  primary wastewater management methods using available qualitative and quantitative sources.
17  Disposal via underground injection predominates in most regions. Reuse is most prevalent in the
18  Appalachian Basin in Pennsylvania. Moderate reuse occurs in the Arkoma (OK, AR) and Anadarko
19  (OK, TX) basins, and use of CWTs occurs predominantly in Pennsylvania.
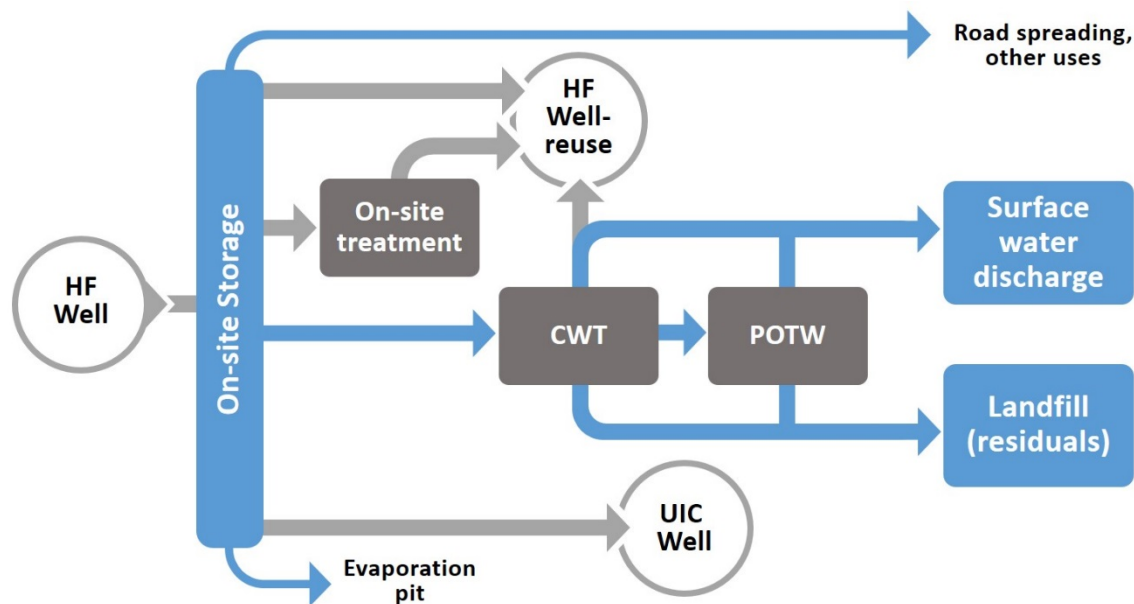


**Figure 8-3. Schematic of wastewater management strategies.**

**Table 8-2. Hydraulic fracturing wastewater management practices in recent years.**

Source: (U.S. EPA, 2015q).

| Basin | Formation | Resource type | Reuse | Injection for disposal | CWT facilities | Notes | Available data[b] |
|-------|-----------|---------------|-------|------------------------|----------------|-------|-------------------|
| **Michigan** | Antrim | Shale Gas | | XXX | | | Qualitative |
| **Appalachian** | Marcellus/Utica (PA) | Shale Gas | XXX | XX | XX | | Quantitative |
| | Marcellus/Utica (WV) | Shale Gas/Oil | XXX | XX | X | Limited Class IID wells in east | Quantitative |
| | Marcellus/Utica (OH) | Shale Gas/Oil | XX | XXX | X | | Mixed |
| **Anadarko** | Granite Wash | Tight Gas | XX | XXX | X[a] | | Mixed |
| | Mississippi Lime | Tight Oil | X | XXX | | Reuse limited but is being evaluated | Qualitative |
| | Woodford; Cana; Caney | Shale Gas/Oil | X | XXX | X[a] | | Qualitative |
| **Arkoma** | Fayetteville | Shale Gas | XX | XX | X[a] | Few existing Class IID wells; new CWT facilities are under construction | Mixed |
| **Fort Worth** | Barnett | Shale Gas | X | XXX | X[a] | Reuse not typically effective due to high TDS early in flowback and abundance of Class IID wells | Mixed |
| **Permian** | Avalon/Bone Springs, Wolfcamp, Spraberry | Shale/tight Oil/gas | X | XXX | X[a] | | Mixed |
| **TX-LA-MS Salt** | Haynesville | Tight Gas | X | XXX | | Reuse not typically cost effective due to high TDS early in flowback and abundance of Class IID wells | Mixed |

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| Basin | Formation | Resource type | Reuse | Injection for disposal | CWT facilities | Notes | Available data[b] |
|---|---|---|---|---|---|---|---|
| **West Gulf** | Eagle Ford, Pearsall | Shale Gas/Oil | X | XXX | X | | Mixed |
| **Denver Julesburg** | Niobrara | Shale Gas/Oil | X | XXX | X | | Mixed |
| **Piceance; Green River** | Mesaverde/Lance | Tight Gas | X | XX | X | Also managed through evaporation to atmosphere in ponds in this region | Qualitative |
| **Williston** | Bakken | Shale Oil | X | XXX | | Reuse limited but is being evaluated | Mixed |

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

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

XXX—The majority (≥ 50%) of wastewater is managed with this management practice.

XX—A moderate portion (≥ 10% and < 50%) of wastewater is managed with this management practice.

X—This management practice has been documented in this location but for a small (< 10%) or unknown percent of wastewater.

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1   Management choices are affected by cost and a number of other factors, including the chemical
2   properties of the wastewater; the volume, duration, and flow rate of the water generated; the
3   logistical feasibility of various options; the availability of necessary infrastructure; federal, state,
4   and local regulations; and operator discretion (U.S. GAO, 2012; NPC, 2011a). The economics (such
5   as transport, storage, and disposal costs) and availability of various treatment and disposal
6   methods are of primary importance (U.S. GAO, 2012). For example, as of early 2015, Pennsylvania
7   has nine operating Class IID wells within the state, whereas Texas has nearly 7,900 (U.S. EPA,
8   2015q).

9   The availability and use of management strategies may change in a region over time as oil and gas
10  development increases or decreases, changing the volumes of wastewater that need to be handled
11  on a local, state, and regional level (see Text Box 8-1 for more information on hydraulic fracturing
12  wastewater management in Pennsylvania). Figure 8-4 illustrates shifting wastewater management
13  practices in Pennsylvania over the last several years as shale gas development has proceeded in the
14  Marcellus Shale. On-site reuse (labeled as "Reuse HF" in Figure 8-4) has grown. Also, most CWT
15  management of Marcellus wastewater in recent years has been at zero-discharge facilities (i.e., for
16  reuse) (an estimated 80% in 2012 and 90% in 2013) (PA DEP, 2015a). Combined with the volumes
17  managed via on-site reuse, Pennsylvania reuse rates are approximately 85% to 80%. In contrast,
18  wastewater disposal data for Colorado (see Figure 8-5) show a steady use of injection wells
19  (injected on lease) since 2000, and an apparent decrease in the use of onsite pits (state data were
20  filtered for formations indicated in the literature to be targets for hydraulic fracturing).

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

1  Gross natural gas withdrawals from shale formations in the United States increased 518% between 2007 and
2  2012 (EIA, 2014c). This production increase has led to larger volumes of wastewater that require appropriate
3  management (Vidic et al., 2013; Gregory et al., 2011; Kargbo et al., 2010). The rapid increase in wastewater
4  generated from oil and gas wells used for hydraulic fracturing has led to many changes in the wastewater
5  disposal practices in the oil and gas industry. Changes have been most evident in Pennsylvania, which has
6  experienced more than a 1,400% increase in natural gas production since 2000 (EIA, 2014c).

7  Lutz et al. (2013) estimated that total wastewater generation in the Marcellus region increased 570%
8  between 2004 and 2013 and concluded that this increase has created stress on the existing wastewater
9  disposal infrastructure. In 2010, in response to concerns over elevated TDS in the Monongahela River basin
10  and studies linking high TDS (and in particular high bromide levels) to elevated DBP levels in drinking water
11  systems (PA DEP, 2011a), PA DEP amended Chapter 95 Wastewater Treatment Requirements under the
12  Clean Streams Law for new discharges of TDS in wastewaters. This regulation is also known informally as the
13  2010 TDS regulation. The regulation disallowed any new indirect discharges (i.e., discharges to POTWs) of
14  hydraulic fracturing waste and set limits of treated discharges from CWTs of 500 mg/L TDS, 250 mg/L
15  chloride, 10 mg/L barium, and 10 mg/L strontium. Existing discharges were exempt.

16  In April 2011, PA DEP requested that oil and gas well operators transporting unconventional wastewater to
17  the eight CWTs and seven POTWs that were exempt from the 2010 TDS regulation voluntarily stop
18  discharging to these facilities. Follow-up letters from PA DEP to the owners of the wells specified that the role
19  of bromides from Marcellus Shale wastewaters in the formation of total trihalomethanes (TTHM) was of
20  concern (PA DEP, 2011a).

21  Between early 2011 and late 2011, although reported wastewater flows more than doubled, Marcellus
22  drilling companies in Pennsylvania reduced their wastewater flows to CWTs that were exempt from the 2010
23  TDS regulation by 98%, and discharge to POTWs was 'virtually eliminated' (Hammer and VanBriesen, 2012).

24  Along with the decreased discharges from POTWs, there has been increased reuse of wastewater in the
25  Marcellus Shale region. From 2008-2011, reuse of Marcellus wastewater has increased, POTW treatment
26  volumes have decreased, tracking of wastewater has improved, and wastewater transportation distances
27  have decreased (Rahm et al., 2013). Maloney and Yoxtheimer (2012) analyzed data from 2011 and found that
28  reuse of flowback water increased to 90% by volume. Disposed flowback water comprised 8% of the total
29  volume. Brine water, which was defined as formation water, was reused (58%), disposed via injection well
30  (27%), or sent to industrial waste treatment plants (14%). Of all the fluid wastes in the analysis, brine water
31  was most likely to be transported to other states (28%). They also concluded that wastewater disposal to
32  municipal sewage treatment plants declined nearly 100% from the first half of 2011 to the second half.
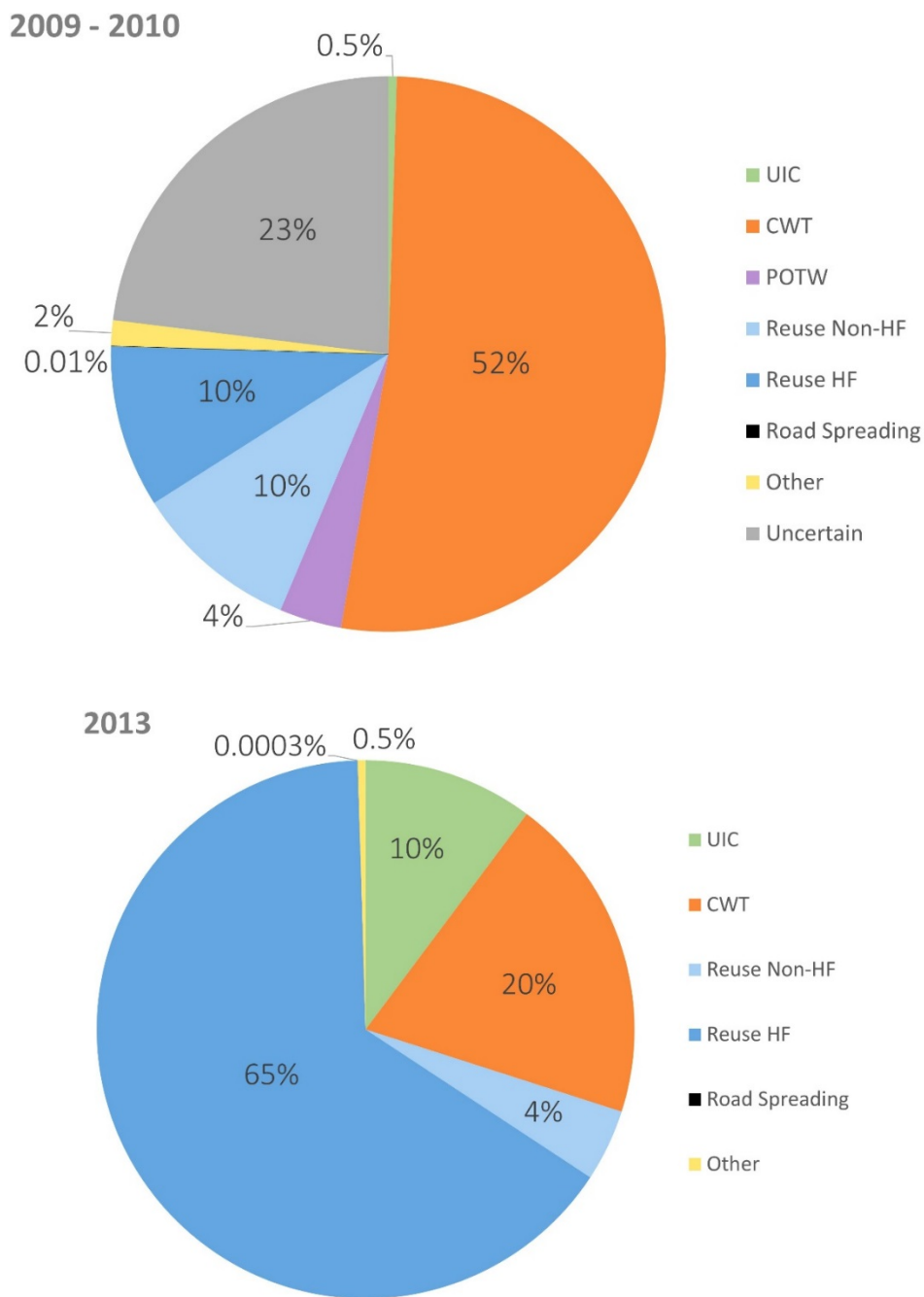
**Figure 8-4. Percentages of Marcellus Shale wastewater managed via various practices for (top) the second half of 2009 and first half of 2010 (total estimated volume of 216 Mgal), and (bottom) 2013 (total estimated volume of 1.3 billion gallons).**

"Reuse HF" indicates on-site reuse. Source: Waste data from PA DEP (2015a).
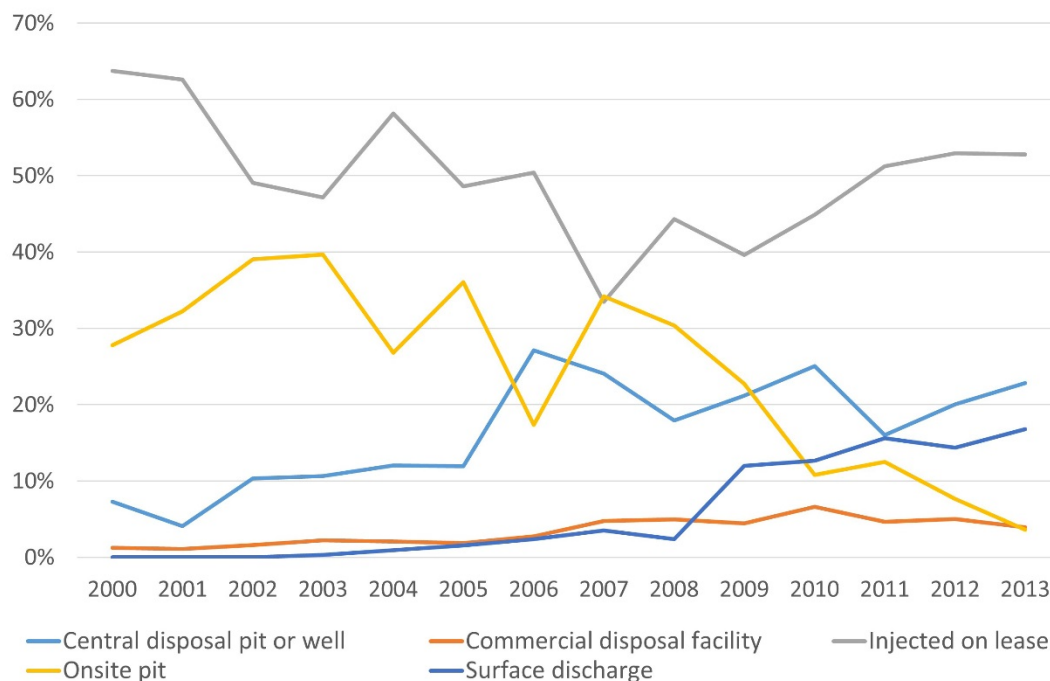
**Figure 8-5. Management of wastewater in Colorado in regions where hydraulic fracturing is being performed.**

Source: Production data from COGCC (2015).

1    Regulations also affect management options and vary geographically. At the Federal level, existing
2    oil and gas effluent limitations guidelines and standards (ELGs) can be found under 40 CFR Part
3    435. These ELGs apply to conventional and unconventional oil and gas extraction facilities in
4    various subcategories (e.g., Offshore, Onshore, Stripper Wells), with the exception of CBM
5    discharges, which are not subject to the existing regulations. Subpart C, the Onshore subcategory,
6    prohibits the discharge of wastewater pollutants to waters of the U.S. from onshore oil and gas
7    extraction facilities. This "zero-discharge standard" means that oil and gas produced water
8    pollutants cannot be directly discharged to surface waters. Operators have met these regulations
9    through underground injection, reuse, or transfer of produced water to POTWs and/or CWTs. West
10   of the 98th meridian (the arid western portion of the continental United States), discharges of
11   wastewater from onshore oil and gas extraction facilities may be permitted for direct discharge to
12   waters of the U.S. if the produced water has a use in agriculture or wildlife propagation when
13   discharged into navigable waters. Definitions in 40 CFR 435.51(c) explain that the term "use in
14   agricultural or wildlife propagation" means that (1) the produced water is of good enough quality
15   to be used for wildlife or livestock watering or other agricultural uses; and (2) the produced water
16   is actually put to such use during periods of discharge. The regulations at 40 CFR 435.52 specify
17   that the only allowable discharge is produced water, with an oil and grease concentration not
18   exceeding 35 milligrams per liter (mg/L). The regulations prohibit the discharge of waste
19   pollutants into navigable waters from any source (other than produced water) associated with

1   production, field exploration, drilling, well completion, or well treatment (i.e., drilling muds, drill
2   cuttings, produced sands).

3   Unpermitted discharges of wastes related to hydraulic fracturing have been described in a number
4   of instances. In Pennsylvania, discharges of brine into a storm drain that discharges to a tributary of
5   the Mahoning River in Ohio. Analyses of the brine and drill cuttings that were discharged indicated
6   the presence of contaminants, including benzene and toluene (U.S. Department of Justice, 2014). In
7   California, an oil production company periodically discharged hydraulic fracturing wastewaters to
8   an unlined sump for 12 days. It was concluded by the prosecution that the discharge posed a threat
9   to groundwater quality (Bacher, 2013). These unauthorized discharges represent both documented
10  and potential impacts on drinking water resources. However, data do not exist to evaluate whether
11  such episodes are uncommon or whether they happen on a more frequent basis and remain largely
12  undetected.

13  The following section provides an overview of hydraulic fracturing wastewater management
14  methods, with some discussion of the geographic and temporal variations in practices. Discussion is
15  provided on common treatment and disposal methods including on-site storage, underground
16  injection, CWTs, reuse of hydraulic fracturing fluids, and evaporation methods. This section also
17  provides discussion on past treatment of hydraulic fracturing wastewater at POTWs. Other
18  management practices are also covered. Brief descriptions of treatment technologies applicable to
19  hydraulic fracturing wastewater are available in Appendix F.

### 8.4.1.  Underground Injection

20  Oil and gas wastewater may be disposed of via Class II injection wells regulated under the
21  Underground Injection Control (UIC) Program under the Safe Drinking Water Act (SDWA)[1]. Class II
22  wells include those used for enhanced oil recovery (IIR), disposal (IID), and hydrocarbon storage
23  (IIH). Nationwide, injection wells dispose of a large fraction of wastewater from the oil and gas
24  industry, including wastewater associated with hydraulic fracturing. A 2009 study notes that the oil
25  and gas industry in the United States generated about 882 billion gal (21 billion bbl or 3.34 trillion
26  L) of produced water in 2007 (Clark and Veil, 2009). More than 98% of this volume was managed
27  via some form of underground injection, with 40% injected into Class II wells. However, a good
28  national estimate of the amount of hydraulic fracturing wastewater injected into Class II wells is
29  difficult to develop due to lack of available on data injection volumes specific to hydraulic fracturing
30  operations that are compiled and able to be compared among states. Also, wastewater management
31  methods are not well tracked in all states. Regional numbers of Class IID wells and generally low
32  reuse rates (see Section 8.4.3), however, are consistent with Class IID wells being a primary means
33  of wastewater management in many areas with hydraulic fracturing activity.

34  This assessment does not address whether there are documented or potential impacts on drinking
35  water resources associated with the injection of hydraulic fracturing wastewaters into Class IID
36  wells. However, should the feasibility of managing hydraulic fracturing wastewater via

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[1] States may be given federal approval to run a UIC program under Section 1422 or 1425 of SDWA.

1  underground injection be limited in any way or become less economically advantageous, operators
2  will likely adjust their wastewater management programs to favor other local practices such as
3  treatment and discharge or reuse. Any new wastewater management decisions would then have to
4  be evaluated in terms of potential impacts on drinking water resources.

5  The decision to inject hydraulic fracturing wastewater into Class IID wells depends, in part, on cost
6  and on the proximity of the production well to the disposal well (and, therefore, transportation
7  costs). For oil and gas producers, underground injection is usually the least expensive management
8  strategy unless significant trucking is needed to transport the wastewater to a disposal well (U.S.
9  GAO, 2012).

10  Class IID wells are not distributed uniformly among states due to differences in geology (including
11  depth and permeability of formations), permitting, and historical demand for disposal of oil and gas
12  wastewater. Table 8-3 shows the numbers of active Class IID wells across the United States, with
13  the total count at a little over 27,000. The greatest numbers of wells are found in Texas, Oklahoma,
14  and Kansas. For example, Texas has nearly 7,900 Class IID wells, with an estimated daily disposal
15  volume of approximately 400 million gal per day (MGD) (1.5 billion L/day) (see Table 8-3). This
16  large disposal capacity in Texas is consistent with the availability of formations with suitable
17  geology and the demand for wastewater disposal associated with a mature and active oil and gas
18  industry. In contrast, Class IID wells are a relatively small portion of Marcellus wastewater
19  management in Pennsylvania (about 10% in 2013 and the first half of 2014) (PA DEP, 2015a)
20  because the state has nine injection wells as of early 2015. Wastewater is generally transported out
21  of state when being managed through injection into Class IID wells. The local availability of Class
22  IID wells and the capacity to accept large volumes of wastewater may begin to be affected by recent
23  state actions concerning seismic activity associated with injection (U.S. EPA, 2014f).

**Table 8-3. Distribution of active Class IID wells across the United States.**

   *Source:* U.S. EPA (2015q)*.*

| State | Nearby basins with hydraulic fracturing | Number of active Class IID wells (2012-2014) | Average disposal rate per well (GPD/well)[a] | Total state disposal rate (MGD) |
|---|---|---|---|---|

| State | Nearby basins with hydraulic fracturing | Number of active Class IID wells (2012-2014) | Average disposal rate per well (GPD/well)[a] | Total state disposal rate (MGD) |
|---|---|---|---|---|
| **AK** | North Slope | 45 | 182,000 | 8.2 |
| **OH** | Appalachian | 188 | 8,900 | 1.7 |
| **WV** | | 66 | 7,180 | 0.47 |
| **PA** | | 9 | 6,380 | 0.057 |
| **NY** | | 10[b] | 3,530 | 0.035 |
| **VA** | | 12 | 17,500 | 0.21 |
| **TN** | | 0 | 0 | 0 |
| **MD** | | 0 | 0 | 0 |
| **NC** | Multiple basins | 0 | 0 | 0 |
| **KS** | Cherokee, Anadarko, Arkoma | 5,516 | 20,900 | 120 |
| **OK** | | 4,622[c] | 35,900 | 170 |
| **AR** | | 611[d] | 30,900 | 19 |
| **MO** | | 11 | 1,270 | 0.014 |
| **CO** | Denver-Julesburg, Green River, Piceance, Uinta | 294 | 50,200 | 15 |
| **WY** | | 330 | --[e] | --[e] |
| **UT** | | 109 | 74,400 | 8.1 |
| **NE** | | 113 | 18,100 | 2.0 |
| **TX** | Fort Worth, Western Gulf, Permian, San Juan, Raton | 7,876 | 54,200 | 430 |
| **NM** | | 736 | 48,600 | 36 |
| **IN** | Illinois | 183 | 3,580 | 0.66 |
| **IL** | | 1,054 | --[e] | --[e] |
| **KY** | | 58 | 1,750 | 0.10 |
| **MI** | Michigan | 779[c] | 16,600 | 13 |
| **CA** | San Joaquin | 826 | 77,800 | 64 |

| State | Nearby basins with hydraulic fracturing | Number of active Class IID wells (2012-2014) | Average disposal rate per well (GPD/well)[a] | Total state disposal rate (MGD) |
|---|---|---|---|---|
| LA | TX-LA-MS Salt | 2,448 | 42,100 | 100 |
| MS | | 499 | 69,500 | 35 |
| AL | | 85 | 44,200 | 3.8 |
| ND | Williston | 395 | 31,600 | 12 |
| MT | | 199 | 31,100 | 6.2 |
| SD | | 21 | 10,200 | 0.21 |
| All other states (NV, FL, OR, IA, and WA)[f] | | 42 | 89,400 | 3.8 |
| Total (not including missing states) | | 27,137 | 40,400 | 1,040 |

[a] Typical injection volumes per well are based on historical annual volumes for injection for disposal divided by the number of active Class IID wells during the same year (primarily data from 2007 to 2013).

[b] These wells are not currently permitted to accept unconventional oil and gas extraction wastewater.

[c] With the exception of Oklahoma and Michigan, wells on tribal lands have not been intentionally included. Wells on tribal lands may be counted if state databases contained them.

[d] Only 24 of the 611 active Class II wells in Arkansas are in the northern half of the state, close to the Fayetteville formation.

[e] Disposal rates and/or number of Class IID wells is unknown.

[f] These are states that have minimal oil and gas activity. The number of wells shown for these states may include all types of Class II wells (e.g., Class II enhanced recovery wells) and therefore is an upper estimate. All other states not listed in this table have minimal oil and gas activity and no active Class IID wells.

## 8.4.2. Centralized Waste Treatment Facilities

1   A CWT facility is generally defined as a facility that accepts industrial materials (hazardous, non-
2   hazardous, solid, or liquid) generated at another facility (off-site) for treatment and/or recovery
3   (EPA, 2000). (A POTW treats local municipal wastewater.) As a group, CWTs that accept oil and gas
4   wastewater offer a wide variety of treatment capabilities and configurations. The fate of treated
5   effluent at CWTs also varies, and can include the following: reuse in fracturing operations, direct
6   discharge (to a receiving water under a National Pollution Discharge Elimination System (NPDES)
7   permit), indirect discharge (to a POTW), or a combination of these. Zero discharge facilities do not
8   discharge to either surface water or a POTW; effluent is generally used for reuse, although
9   evaporation or land application may also be done. Some CWTs may be configured so that they only
10  partially treat the waste stream if allowed by the end use (a reuse application that does not require
11  TDS removal). Potential impacts on drinking water resources associated with treatment in CWTs
12  will depend upon whether the CWT treats adequately for constituents of concern prior to discharge
13  to surface water or to a POTW, and whether treatment residuals are managed appropriately.

14  Clean Water Act (CWA) regulations only apply to facilities that discharge treated wastewater to
15  surface waters or POTWs. For zero-discharge facilities, Pennsylvania and Texas have adopted

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1    regulations to control permitting. PA DEP issues permits (General Permit WMGR123) that allow
2    zero-discharge CWTs to treat and release water back to oil and gas industries for reuse (see the
3    Eureka Resources Facility in Williamsport, PA listed in Table 8-7 as an example of a zero-discharge
4    facility[1]). The Texas Railroad Commission (TXRRC) regulates and categorizes wastewater recycling
5    facilities into different categories: off-lease commercial recycling facilities (capable of being moved
6    from one location to another) and stationary commercial recycling facilities. The Texas regulations
7    also promote oil and gas wastewater treatment for reuse and water sharing (see
8    http://www.rrc.state.tx.us/rules/rule.php).

9    Wastewater from hydraulically fractured wells can be transported by truck or pipeline to and from
10   a CWT (Easton, 2014); this may present a vulnerability for spills or leaks (see Chapter 7). The
11   treated wastewater from CWTs may be integrated with other sources of water (for example, treated
12   municipal wastewater, storm water drainage, or other treated industrial waste streams) for reuse
13   applications (Easton, 2014).

### 8.4.2.1.  Numbers and Locations of CWTs

14   Although there are CWTs serving hydraulic fracturing operations throughout the country, including
15   the Barnett and Fayetteville shale plays plus oil fields in Texas and Wyoming, historically the
16   majority have served Marcellus Shale operations. This is likely because the low availability of
17   injection wells (Boschee, 2014) in Pennsylvania necessitates other forms of management. An EPA
18   study (U.S. EPA, 2015q) identified 73 CWT facilities that have either accepted or plan to accept
19   hydraulic fracturing wastewater (see Table 8-4). Of these, 39 are located in Pennsylvania. Most of
20   these are zero-discharge facilities; they do not discharge to surface waters or POTWs, and they
21   often do not include TDS removal. According to EPA research (U.S. EPA, 2015q), the number of CWT
22   facilities serving operators in the Marcellus and Utica Shales has increased since the mid-2000s as
23   the number of wells drilled in the Marcellus and Utica Shales has increased, growing from roughly
24   five CWTs in 2004 to over 40 in 2013. A similar trend has been noted for the Fayetteville Shale
25   region in Arkansas, where there are fewer Class IID wells available relative to the rest of the state
26   (U.S. EPA, 2015q).

27   In other regions, a small number of newer facilities have emerged in the last several years, most
28   often with TDS removal capabilities. In Texas, for example, two zero-discharge facilities are
29   available to treat wastewater from the Eagle Ford (beginning in 2011 and 2013), both equipped
30   with TDS removal, and one zero-discharge facility with TDS removal is located in the Barnett Shale
31   region (operational beginning in 2008). In Wyoming, the four facilities in the region of the
32   Mesaverde/Lance formations (operations beginning between 2006 and 2012; two zero-discharge
33   and two with multiple discharge options) are all capable of TDS removal (U.S. EPA, 2015q).

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

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**Table 8-4. Number, by state, of CWT facilities that have accepted or plan to accept wastewater from hydraulic fracturing activities.**

Source: U.S. EPA (2015q).

| State | Formation(s) served where hydraulic fracturing occurs | Zero-discharge CWT facilities[a] | | CWT facilities that discharge to surface water or POTW[a] | | Discharging CWT facilities with multiple discharge options[a] | | Total known facilities |
|---|---|---|---|---|---|---|---|---|
| | | Non-TDS removal treatment | TDS removal treatment | Non-TDS removal treatment | TDS removal treatment | Non-TDS removal treatment | TDS removal treatment | |
| AR | Fayetteville | 2 | 0 | 0 | 0 | 0 | 1 | 3 |
| CO | Niobrara, Piceance Basin | 3 (1) | 0 | 0 | 0 | 0 | 0 | 3 |
| ND | Bakken | 0 | 1 (1) | 0 | 0 | 0 | 0 | 1 |
| OH | Utica, Marcellus | 10 (7) | 0 | 1 | 0 | 0 | 0 | 11 |
| OK | Woodford | 2 | 0 | 0 | 0 | 0 | 0 | 2 |
| PA | Utica, Marcellus | 23 | 7 (3) | 6 | 0 | 0 | 3 (1) | 39 |
| TX | Eagle Ford, Barnett, Granite Wash | 1 | 3 | 0 | 0 | 0 | 0 | 4 |
| WV | Marcellus, Utica | 4 (2) | 0 | 0 | 0 | 1 | 1 | 6 |
| WY | Mesaverde, Lance | 0 | 2 | 0 | 0 | 0 | 2 | 4 |
| **Total** | | 45 | 13 | 7 | 0 | 1 | 7 | 73 |

[a] Number of facilities also includes facilities that have not yet opened but are under construction, pending permit approval, or in the planning stages. Facilities that are not accepting process wastewater from hydraulic fracturing activities but plan to in the future are noted parenthetically.

1  Because few states maintain a comprehensive list of CWT facilities and the count provided by the
2  EPA (U.S. EPA, 2015q) includes facilities that plan to accept unconventional oil and gas
3  wastewaters, the data in Table 8-4 do not precisely reflect the number of facilities currently
4  available for handling hydraulic fracturing wastewaters. Additional discussion of CWTs in
5  unconventional oil and gas fields are reviewed in the literature for areas including the Barnett
6  (Hayes and Severin, 2012b) and the Fayetteville (Veil, 2011) as well as other oil fields in Texas and
7  Wyoming (Boschee, 2014, 2012). In addition, news releases and company announcements indicate
8  that new wastewater treatment facilities are being planned (Greenhunter, 2014; Geiver, 2013;
9  Purestream, 2013; Alanco, 2012; Sionix, 2011).

1   Based on oil and gas waste disposal information available from PA DEP ([PA DEP, 2015a](#)) dating
2   back to 2009, the estimated volumes of Marcellus wastewater sent to CWTs range from
3   approximately 113 Mgal (428 million L) in the latter half of 2009 and first half of 2010, to about
4   183 Mgal (693 million L) in 2011, and about 252 Mgal (954 million L) in 2013. These constitute
5   about 52% of the total wastewater volume in 2009-2010, about 25% in 2011, and 20% in 2013,
6   indicating that although total amounts of wastewater have increased (see Table 8-1), the
7   percentage managed through CWTs has decreased.

8   Among the Marcellus wastewater sent to CWTs, an estimated 35% was sent to zero-discharge
9   facilities in Pennsylvania (those with general permits) in the latter half of 2010, and 42% was sent
10  to facilities with NPDES permits (indicating that they can discharge to surface waters). About 23%
11  went to CWTs whose permit types were more difficult to ascertain, generally outside of
12  Pennsylvania. By 2013, the portion sent to zero-discharge facilities had risen to 90%, with about
13  5% sent to CWTs with NPDES permits and 5% sent to CWTs whose discharge permit type is not
14  clear. The high percentage sent to zero-discharge CWTs is consistent with the concerted focus on
15  reuse in Pennsylvania, although CWTs with NPDES permits also often provide treated wastewater
16  for reuse, further limiting discharges to surface waters. The waste records do not indicate if a CWT
17  has more than one permit type.

### 8.4.2.2. Residuals Management

18  Certain treatment processes at CWTs produce liquid or solids residuals as a by-product of that
19  process. The residuals produced depend on the constituents in the treated water and the treatment
20  process used. Residuals can consist of sludges (from precipitation, filtration, settling units, and
21  biological processes); spent media (media requiring replacement or regeneration from filtration,
22  adsorption, or ion exchange processes); concentrated brines (from membrane processes and some
23  evaporation processes); and regeneration and cleaning chemicals (from ion exchange, adsorption,
24  and membrane processes) ([Fakhru'l-Razi et al., 2009](#)). Residuals from CWTs can constitute a
25  considerable fraction of solid waste in an oil or gas production area. [Chiado (2014)](#) found that solid
26  wastes from hydraulic fracturing in the Marcellus accounted for 5% of the weight of waste
27  deposited in landfills in the area, with some area landfills reaching as high as 60% landfill mass
28  coming from hydraulic fracturing activities.

29  *Management of Solid Residuals*

30  CWTs may apply additional treatment to solid residuals including thickening, stabilization (e.g.,
31  anaerobic digestion), and dewatering processes prior to disposal. The solid residuals are then
32  typically sent to a landfill, land applied, or incinerated ([Morillon et al., 2002](#)). Pollutants may
33  accumulate in sludge, which may limit land application as a disposal option. For example, wastes
34  containing TENORMs can be problematic due to the possibility of radon emissions from the landfill
35  ([Walter et al., 2012](#)). In some states, many landfills that are specifically permitted to accept
36  TENORM have criteria written into their permits, including gamma exposure rate (radiation) levels
37  and radioactivity concentration limits. Most non-hazardous landfills have limits on maximum
38  radiation that can be accepted. For example, Pennsylvania requires alarms to be set at all municipal
39  landfills, with a trigger set at 10 μR/hr above background radiation (Pa Code Title 25, Ch. 273.223

1    c). Texas sets a radioactivity limit, requiring that any waste disposed by burial contains less than 30
2    pCi/g radium or 150 pCi/g of other radionuclides (TX Code Ch 4 Section F Section 4.620). Some
3    states have volumetric limitations on TENORM in their permits (e.g., Colorado).

4    Solid residual wastes have the potential to impact the quality of drinking water resources if
5    contaminants leach to groundwater or surface water. In a recent study by PA DEP, radium was
6    detected in leachate from 34 of 51 landfills, with radium-226 concentrations ranging from 54 to
7    416 pCi/L, and radium-228 ranging from 2.5 to 1,100 pCi/L (PA DEP, 2015b). Countess et al. (2014)
8    studied the potential for barium, calcium, sodium, and strontium to leach from sludges generated at
9    a CWT handling hydraulic fracturing wastewaters in Pennsylvania. Tests used various strong acid
10   solutions (to simulate the worst case scenario) and weak acid digestions (to simulate
11   environmental conditions). The extent of leaching varied by constituent and by fluid type; the data
12   illustrate the possibility of leaching of these constituents from landfills.

13   *Management of Liquid Residuals*

14   Practices for management of liquid residual streams are generally the same as for untreated
15   hydraulic fracturing wastewaters, although the reduced volumes tend to lower costs (Hammer and
16   VanBriesen, 2012). Concentrations of contaminants, however, will be higher. Liquids mixed with
17   other wastes can be disposed of in landfills if the liquid concentration is low enough. If the liquid is
18   not injected into a disposal well, treatment to remove salts would be required for surface water
19   discharge to meet NPDES permit requirements and protect the water quality for downstream users
20   (e.g., drinking water utilities) (see Section 8.6). Because some constituents of concentrated
21   residuals can pass through or impact municipal wastewater treatment processes (Linarić et al.,
22   2013; Hammer and VanBriesen, 2012), these residuals may not be appropriate for discharge to a
23   POTW. Elevated salt concentrations, in particular, can reduce or inhibit microbiological treatment
24   at municipal wastewater systems such as activated sludge treatment (Linarić et al., 2013).

### 8.4.3.  Water Reuse for Hydraulic Fracturing

25   Water reuse in hydraulic fracturing operations has increased in recent years, with wastewaters
26   being used to formulate hydraulic fracturing fluids for subsequent fracturing jobs (Boschee, 2014,
27   2012; Gregory et al., 2011; Rassenfoss, 2011). Wastewater may be reused after some form of
28   treatment (sometimes only settling), depending on the reuse water quality requirements, and it
29   may be supplied for use in hydraulic fracturing through various routes. Reused water is discussed
30   in Chapter 4 of this report (Water Acquisition) as well as in this chapter, though in a different
31   context. The water reuse rate described in this chapter is the amount or percentage of generated
32   wastewater that is managed by being provided to operators for use in additional hydraulic
33   fracturing operations. In contrast, Chapter 4 discusses reused wastewater as a source water and as
34   one part of the base fluid for new fracturing fluid.

35   Hydraulic fracturing wastewater reuse reduces costs associated with other forms of wastewater
36   management, and the economic benefits and feasibility of reuse can be expected to figure into
37   ongoing wastewater management decisions. However, although reuse minimizes other forms of
38   wastewater management on a local and short-term basis (e.g., those involving direct or indirect

1   discharge to surface waters), reuse can result in the accumulation of dissolved solids (e.g., salts and
2   TENORMs) as the process returns water to the subsurface. For example, data from a PA DEP study
3   (PA DEP, 2015b) suggests that hydraulic fracturing fluids that include reused wastewater already
4   contain radium-226 and radium-228. Eventually, wastewaters with a component that has been
5   reused more than once will need to be definitively managed, either through treatment or injection.
6   Residuals from treatment will also require proper management to avoid potential impacts on water
7   resources (see Section 8.4.2.2) (Kappel et al., 2013).

### 8.4.3.1.  *Factors in Considering Reuse*

8   In making the decision whether to manage wastewater via reuse, operators have several factors to
9   consider (Slutz et al., 2012; NPC, 2011a):

10  • Wastewater generation rates compared to water demand for future fracturing operations,

11  • Wastewater quality and treatment requirements for use in future operations,

12  • The costs and benefits of wastewater management for reuse compared with other
13    management strategies,

14  • Available infrastructure and treatment technologies, and

15  • Regulatory considerations.

16  Among these factors, costs may be the most significant driver, weighing the costs of transportation
17  from the generating well to the treatment facility and to the new well against the costs for transport
18  to alternative locations (a disposal well or CWT). Trucking large quantities of water can be
19  relatively expensive (from $0.50 to $8.00 per barrel), rendering on-site treatment technologies and
20  reuse potentially economically competitive in some settings (Dahm and Chapman, 2014; Guerra et
21  al., 2011). Also, logistics, including proximity of the water sources for aggregation, may be a factor
22  in implementing reuse. For example, Boschee (2014) notes that in the Permian Basin, older
23  conventional wells are linked by pipelines to a central disposal facility, facilitating movement of
24  treated water to areas where it is needed for reuse.

25  Regulatory factors may facilitate reuse. In 2013, the Texas Railroad Commission adopted rules
26  intended to encourage statewide water conservation. These rules facilitate reuse by eliminating the
27  need for a permit when operators reuse on their own lease or transfer the fluids to another
28  operator for use in hydraulic fracturing (Rushton and Castaneda, 2014). Data for the years after
29  2013 will allow evaluation of whether reuse increases.

30  Recommended compositional ranges for base fluid may shift in the future as fracturing fluid
31  technology continues to develop. Development of fracturing mixture additives that are brine-
32  tolerant have allowed for the use of high TDS wastewaters (up to tens of thousands of mg/L) for
33  reuse in fracturing (Tiemann et al., 2014; GTI, 2012; Minnich, 2011). Some new fracturing fluid
34  systems are claimed to be able to tolerate salt concentrations exceeding 300,000 mg/L (Boschee,
35  2014). This greater flexibility in acceptable water chemistry can facilitate reuse both logistically
36  and economically by reducing treatment needs. Additional discussion of the water quality feasible
37  for reuse and examples of recommended constituent concentrations are included in Appendix F.

1  Reuse rates may also fluctuate with changes in the supply and demand of treated wastewater and
2  the availability of fresh water. Flowback may be preferable to later-stage produced water for reuse
3  because it is typically generated in larger quantities from a single location as opposed to water
4  produced later on, which is generated in smaller volumes over time from many different locations.
5  Flowback water also tends to have lower TDS concentrations than later-stage produced water; in
6  the Marcellus, TDS has been shown to increase from tens of thousands to about 100,000 mg/L
7  during the first 30 days (Barbot et al., 2013; Maloney and Yoxtheimer, 2012) (see Chapter 7). The
8  changing production rate and quality of wastewaters generated in a region as more wells go into
9  production need to be taken into account, as well as possible decreases in the demand for reused
10 water as plays mature (Lutz et al., 2013; Hayes and Severin, 2012b; Slutz et al., 2012).

### 8.4.3.2. Reuse Rates

11 Reliable information on reuse practices throughout the United States is hampered by a limited
12 amount of data that are available and represent different regions of the country. In Table 8-5,
13 estimates have been compiled from various literature sources. Reuse rates are highest in the
14 Appalachian Basin, associated primarily with the Marcellus Shale. Documentation of reuse practices
15 is also more readily available for that region than for other parts of the country.

16 A number of studies have estimated reuse rates for Marcellus wastewater. Although the reported
17 values can differ substantially (see Table 8-5), the data point to a steep increase in reuse since
18 2008, with rates increasing from 0% to 10% in 2008 to upwards of 90% in 2013. As an example, an
19 analysis of waste disposal information from the PA DEP for Marcellus wells in Pennsylvania
20 (Hansen et al., 2013) reports an increase in reuse from 9% (7.17 million gal or 27.1 million L) of
21 total wastewater volumes in 2008 to 56% (343.79 million gal or 1.3014 billion L) in 2011. During
22 that same timeframe, the authors report that disposal via brine/industrial waste treatment plants
23 increased from 32% in 2008 to 70% in 2009, and then declined to 30% in 2011. Because some
24 industrial waste treatment plants can treat wastewater for reuse, some of the volumes indicated by
25 Hansen et al. (2013) as managed by this route may have ultimately been used for fracturing,
26 meaning that the 56% value for 2011 is most likely an underestimate.

**Table 8-5. Estimated percentages of reuse of hydraulic fracturing wastewater.**

| Play or Basin | Source and Year | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 |
|---|---|---|---|---|---|---|---|
| *East Coast* | | | | | | | |
| Marcellus, PA | Rahm et al. (2013) | 9 | 8 | 25 – 48 | 67 - 80 | | |
| Marcellus, PA | Ma et al. (2014) | | 15 - 20 | | | | 90 |
| Marcellus, PA | Shaffer et al. (2013) | | | | | 90 | |
| Marcellus, WV | Hansen et al. (2013) | | | 88 | 73 | 65 (partial year) | |
| Marcellus, PA | Hansen et al. (2013) | 9 | 6 | 20 | 56 | | |
| Marcellus, PA | Maloney and Yoxtheimer (2012) | | | | 71.6 | | |
| Marcellus, PA | Tiemann et al. (2014) | | | | 72 | 87 | |

| Play or Basin | Source and Year | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 |
|---|---|---|---|---|---|---|---|
| Marcellus, PA | Rassenfoss (2011) | | | ~67 overall (general estimate) 96 (one specific company) | | | |
| Marcellus, PA | Wendel (2011) | | | 75-85 | 90 | | |
| Marcellus, PA | Lutz et al. (2013) | 13 (prior to 2011) | | | 56 | | |
| Marcellus, PA (SW region) | Rahm et al. (2013) | ~10 | ~15 | ~25-45 | ~70-80 | | |
| Marcellus, PA (NE region) | Rahm et al. (2013) | 0 | 0 | ~55-70 | ~90-100 | | |
| Marcellus, PA | Rahm and Riha (2014) | | | | 55-80 (general estimate – appears to cover recent years) | | |
| *Gulf Coast & Midcontinent* | | | | | | | |
| Fayetteville | Veil (2011) | | | 20 (single company target) | | | |
| West Permian | Nicot et al. (2012) | | | | 0 | | |
| Midland Permian | Nicot et al. (2012) | | | | 2 | | |
| Anadarko | Nicot et al. (2012) | | | | 20 | | |
| Barnett | Nicot et al. (2012) | | | | 5 | | |
| Barnett | Rahm and Riha (2014) | | | | 5 (general estimate – appears to cover recent years) | | |
| Eagle Ford | Nicot and Scanlon (2012) | | | | 0 | 20 (estimate based on interviews) | |
| East Texas | Nicot and Scanlon (2012) | | | | 5 | | |
| Haynesville | Argonne National Laboratory (2014) | | | | | | 0 |
| Haynesville | Rahm and Riha (2014) | | | | 5 (general estimate – appears to cover recent years) | | |
| *West Coast & Upper Plains* | | | | | | | |
| Bakken | Argonne National Laboratory (2014) | | | | | | 0 |

1    According to Maloney and Yoxtheimer (2012), about 331 million gal (7.9 million bbl or 1.25 billion
2    L) of flowback and about 381 million gal (about 9.1 million bbl or 1.4 billion L) of produced water
3    (excluding flowback) were generated in the Marcellus in 2011. For flowback and produced water

1  combined, about 72% was reused. Of the flowback, 90% was managed through reuse (other than
2  road spreading). Of produced brine water, 55.7% was reused (with 11.6% treated in CWTs and
3  27.8% injected into Class IID wells in Ohio). Reuse is higher in the northeastern part of the
4  Marcellus; in the southwestern portion, easier access to Class IID wells in Ohio makes disposal by
5  injection more feasible (Rahm et al., 2013).

6  Data from Marcellus wastewater management reports submitted to PA DEP (PA DEP, 2015a) were
7  compiled for this assessment; the data suggest that rates of reuse for hydraulic fracturing (as
8  indicated by a waste disposal method of either "Reuse Other than Road Spreading" or a zero-
9  discharge CWT) increased from about 28% in the second half of 2010 to about 60% in 2011, 83% in
10  2013, and 89% in the first half of 2014. These values may be underestimates because wastewater
11  treated at facilities with NPDES permits can be provided to operators for reuse, and the permit
12  types for some facilities could not be determined. Among the forms of reuse, on-site reuse ("Reuse
13  Other than Road Spreading") has risen steadily over the past few years, from about 8% in the
14  second half of 2010 to about 48% in 2011, 62% in 2012, and nearly 70% in the first half of 2014.

15  Outside of the Marcellus region, a lower percentage of wastewater from hydraulic fracturing
16  operations is reused. According to published literature, in Texas in 2011, 0% to 5% of wastewater
17  was reused in most basins, with the exception of the Anadarko Basin (20%) (Nicot and Scanlon,
18  2012); see Table 8-5. Ma et al. (2014) note that only a small amount of reuse is occurring in the
19  Barnett Shale. Reuse has not yet been pursued aggressively in New Mexico or in the Bakken (North
20  Dakota) (Argonne National Laboratory, 2014; LeBas et al., 2013). Other sources, however, indicate
21  growing interest in reuse, as evidenced in specialized conferences (e.g., "Produced Water Reuse
22  Initiative 2014" on produced water reuse in Rocky Mountain oil and shale gas plays), and available
23  state-developed information on reuse (e.g., fact sheet by the Colorado Oil and Gas Conservation
24  Commission) (Colorado Division of Water Resources; Colorado Water Conservation Board;
25  Colorado Oil and Gas Conservation Commission, 2014). The fact sheet discusses piping and trucking
26  wastewater to CWTs in the Piceance Basin to treat for reuse.

### 8.4.4. Evaporation

27  In drier climates of the western United States, natural evaporation may be an option for treatment
28  of hydraulic fracturing wastewater (see Figure 8-6). Production data from the California
29  Department of Conservation's Division of Oil, Gas, and Geothermal Resources (DOGGR) (California
30  Department of Conservation, 2015), for example, lists "evaporation-percolation" as the
31  management method for 23% to 30% of the wastewater from Kern County over the last few years.
32  However, data on volumes of wastewater managed are not readily available for all states where this
33  practice is employed.

34  Evaporation is a simple water management strategy that consists of transporting wastewater to a
35  pond or pit with a large surface area and allowing passive evaporation of the water from the surface
36  (Clark and Veil, 2009). The rate of evaporation depends on the quality of the wastewater as well as
37  the size, depth, and location of the pond. Evaporation also depends on local humidity, temperature,
38  and wind (NETL, 2014). The residual brine or solid can be disposed of in an underground injection
39  well or landfill (see Section 8.4.3.2 for more details). In colder, dry climates, a freeze-thaw

1   evaporation method has been used to purify water from oil and gas wastewater ([Boysen et al.,
2   1999](#)).



**Figure 8-6. Lined evaporation pit in the Battle Creek Field (Montana).**

Source: [DOE (2006)](#). Permission from ALL Consulting.

3   Alternatively, operators may transport wastewater by truck to an off-site commercial facility.
4   Commercial evaporation facilities exist in Colorado, Utah, New Mexico, and Wyoming ([NETL, 2014](#);
5   [DOE, 2004](#)). [Nowak and Bradish (2010)](#) described the design, construction, and operation of two
6   large commercial evaporation facilities in Southern Cross, Wyoming and Danish Flats, Utah. Each
7   facility includes 14,000-gal (53,000 L) three-stage concrete receiving tanks, a sludge pond, and a
8   series of five-acre (20,234 m$^2$) evaporation ponds connected by gravity or force-main underground
9   piping. The Wyoming facility, which opened in 2008, consists of two ponds with a total capacity of
10  approximately 84 million gal (2 million bbl or 320 million L). The Utah facility, open since 2009,
11  consists of 13 ponds with a total capacity of 218.4 million gal (5.2 million bbl or 826.7 million L).
12  Each facility receives 420,000 to 1.47 million gal (10,000 to 35,000 bbl or 1.6 million to 5.56 million
13  L) per day of wastewater from oil and gas production companies in the area. Evaporation pits are
14  subject to state regulatory agency approval and must meet state standards for water quality and
15  quantity ([Boysen et al., 2002](#)). Impacts on drinking water resources from evaporation pits might
16  arise if a pit is breached due to extreme weather or other factors affecting infrastructure and if
17  leaking wastewater reaches a surface water body; such events as related specifically to evaporation
18  pits appear not to have not been evaluated in the literature, and their prevalence is unknown.

### 8.4.5. Publicly Owned Treatment Works

1  Prior to the development of unconventional resources, POTWs were used to treat wastewater and
2  other wastes from conventional oil and gas operations in some eastern states. Although this is not a
3  common treatment method for oil and gas wastes in the United States, the small number of
4  injection wells for waste disposal in Pennsylvania drove the need for disposal alternatives (Wilson
5  and Vanbriesen, 2012). When development of the Marcellus Shale began, POTWs continued to be
6  used to treat wastewater, including wastes originating from new unconventional oil and gas wells
7  (Kappel et al., 2013; Soeder and Kappel, 2009). However, unconventional wastewater from the
8  Marcellus region is difficult to treat at POTWs due to elevated concentrations of halides, heavy
9  metals, organic compounds, radionuclides, and salts (Lutz et al., 2013; Schmidt, 2013). Most of
10  these constituents have the potential to pass through the unit treatment processes commonly used
11  in POTWs and can be discharged into receiving waters (Cusick, 2013; Kappel et al., 2013). In
12  addition, research has found that sudden, extreme salt fluctuations can disturb POTW biological
13  treatment processes (Linarić et al., 2013; Lefebvre and Moletta, 2006). In order to meet NPDES
14  requirements, POTWs used to blend the hydraulic fracturing wastewater with incoming municipal
15  wastewater. For example, Ferrar et al. (2013) note that, per PA DEP requirements, one facility could
16  only accept up to 1% of their influent volume from unconventional oil and gas wastewater per day.

17  The annual reported volume of oil and gas produced wastewater treated at POTWs in the Marcellus
18  Shale region peaked in 2008 and has since declined to virtually zero (see Figure 8-7). This decline
19  has been attributed to stricter discharge limits for TDS for POTWs in Pennsylvania and widespread
20  voluntary compliance on behalf of oil and gas operators with the May 2011 request from PA DEP to
21  cease sending Marcellus Shale wastewater to 15 treatment plants (including both POTWs and
22  CWTs) by May 19, 2011 (Rahm et al., 2013). To comply with the request, the oil and gas industry in
23  Pennsylvania accelerated the transition of wastewater deliveries from POTWs to CWTs for better
24  removal of metals and suspended solids (Schmidt, 2013). However, treated effluent from CWTs
25  may be delivered to POTWs for additional treatment assuming treatment processes at POTWs are
26  not adversely affected and the POTWs can continue to meet NPDES discharge limits (Hammer and
27  VanBriesen, 2012). General Pretreatment Regulations and State and local regulations typically
28  govern the pre-treated water volumes and qualities that can be accepted by the POTW.

29  Although operators stopped sending Marcellus Shale wastewater to POTWs in May of 2011,
30  conventionally produced wastes have continued to be processed at POTWs in Pennsylvania,
31  although at small volumes (29 Mgal and 20 Mgal for the years 2010 and 2011, respectively) (Wilson
32  and Vanbriesen, 2012).
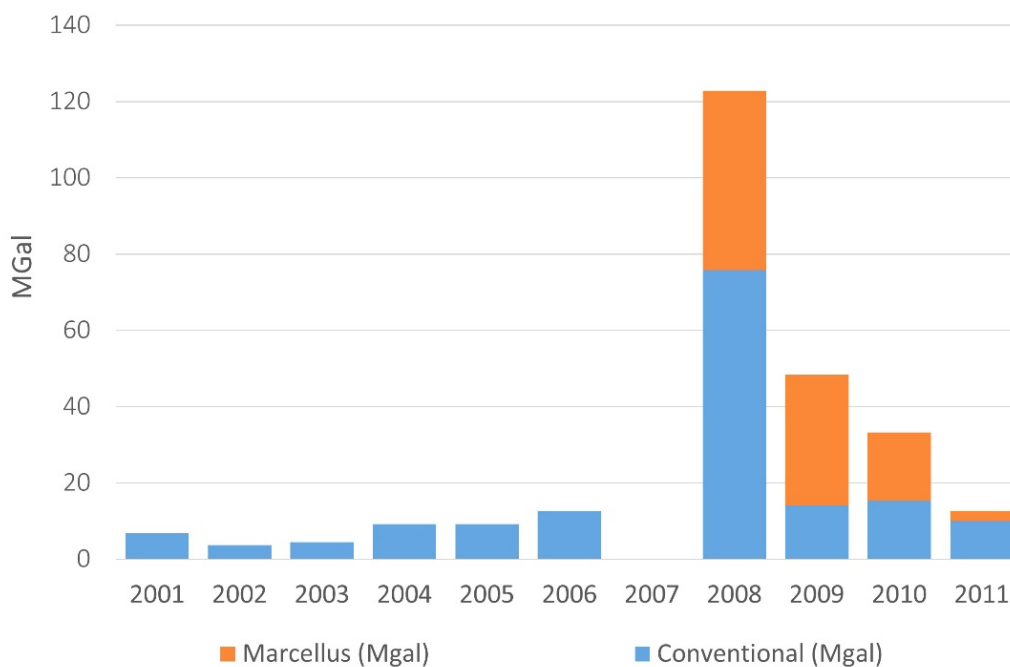
**Figure 8-7. Oil and gas wastewater volumes discharged to POTWs from 2001-2011 in the Marcellus Shale.**

Source: Lutz et al. (2013).

1    At least one study has evaluated POTW effluent chemistry before and after the cessation of
2    treatment of hydraulic fracturing wastewater. Ferrar et al. (2013) collected effluent samples from
3    two POTWs and one CWT facility in Pennsylvania before and after the 2011 PA DEP request.
4    Results from POTW effluent samples collected while the facilities were still treating Marcellus Shale
5    wastewater showed that concentrations of several analytes (barium, manganese, strontium, TDS,
6    and chloride) were greater than various drinking water and surface water criteria (i.e., EPA MCLs
7    and secondary MCLs for drinking water, surface water quality standards for aquatic life, and/or
8    surface water standards for human consumption of aquatic organisms). Results for effluent samples
9    collected after the POTWs stopped receiving Marcellus wastewater showed a statistically
10   significant decrease in the concentrations of several of these constituents. In particular, one of the
11   two POTWs showed a decrease in average barium concentration from 5.99 mg/L to 0.141 mg/L, a
12   decrease in the average strontium concentration from 48.3 mg/L to 0.236 mg/L, and a decrease in
13   the average bromide concentration from 20.9 mg/L to <0.016 mg/L. Influent concentrations at the
14   other POTW were lower (0.55 mg/L for barium, 1.63 mg/L for strontium, and 0.60 for bromide),
15   but significant decreases in these constituents were also seen in the effluents (0.036 mg/L barium,
16   0.228 mg/L strontium, and 0.119 for bromide); this POTW had continued to accept conventional oil
17   and gas wastewater. The authors conclude that the decreases in the concentrations of the various
18   constituents indicate that the elevated concentrations in the first samplings can be attributed to the
19   contribution of wastewater from unconventional natural gas development.

### 8.4.6.  Other Management Practices and Issues

1    Additional strategies for wastewater management in some states include discharging to surface
2    waters and land application. Wastewater from CBM fracturing and production, in particular,
3    generally has lower TDS concentrations than wastewater from other types of unconventional plays
4    and lends itself more readily to beneficial use. Below is a discussion of these other management
5    practices.

#### *8.4.6.1.  Land Application, Including Road Spreading*

6    Land application has been done using brines from conventional oil and gas production. Road
7    spreading can be done for dust control or de-icing. Although recent data are not available, an
8    American Petroleum Institute (API) survey estimated that approximately 75.6 million gal (1.8
9    million bbl or 286 million L) of wastewater was used for road spreading in 1995 (API, 2000). The
10   API estimate does not specifically identify hydraulic fracturing wastewater. There is no current
11   nationwide estimate of the extent of road spreading using hydraulic fracturing wastewater.

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

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

29   Potential impacts on drinking water resources from road spreading, have been noted by Tiemann et
30   al. (2014) and Hammer and VanBriesen (2012). These include potential effects of runoff on surface
31   water, or migration of brines to groundwater. Snowmelt may carry salts or other chemicals from
32   the application site, with the possibility of transport increasing if application rates are high or rain
33   occurs soon after application (Hammer and VanBriesen, 2012). Research on the impacts of
34   conventional road salt application has documented long-term salinization of both surface water and
35   ground water in the northern United States; by the 1990s, 24% of public supply wells in the Chicago
36   area had chloride concentrations exceeding 100 mg/L (Kelly, 2008; Kaushal et al., 2005). When
37   conventional oil field brine was used in a controlled road spreading experiment, elevated chloride
38   concentrations were detected in shallow ground water (531 ppm in winter and 1,360 ppm in

1  summer (Bair and Digel, 1990). The amount of salt contributed to drinking water resources due to
2  road application of hydraulic fracturing wastewaters has not been quantified.

3  In managing solid wastes from oil and gas production, a study on land application of oilfield scales
4  and sludges suggested that radium in samples became more mobile after incubation with soil under
5  moist conditions, due to microbial processes and interactions with the soil and water (Matthews et
6  al., 2006). Overall, potential effects from land application on drinking water resources are not well
7  understood.

8  Additionally, drill cuttings must be managed; in some places they are left in the reserve pit (pit for
9  waste storage), allowed to dry and, buried on-site (Kappel et al., 2013). More, commonly, however,
10 drill cuttings are disposed of in landfills (Chiado, 2014); about half of Marcellus drill cuttings are
11 disposed of in Pennsylvania, while the rest are trucked to Ohio or West Virginia (Maloney and
12 Yoxtheimer, 2012).

### 8.4.6.2. *Management of Coalbed Methane Wastewater*
13 Wastewater from CBM wells can be managed like other hydraulic fracturing wastewater discussed
14 above. However, the wastewater from CBM wells can also be of higher average quality (e.g., lower
15 TDS content) than wastewater from other hydraulically fractured wells, which makes it more
16 suitable for certain management practices and uses. A number of management strategies have been
17 proposed or implemented, with varying degrees of treatment required depending on the quality of
18 the wastewater and the requirements of the intended use (Hulme, 2005; DOE, 2003, 2002).
19 Although specific volumes managed through the various practices below are not well documented,
20 qualitative information and considerations for feasibility are available and presented below.

21 CBM wastewater quality, which can range from an average of nearly 1,000 mg/L TDS in the Powder
22 River Basin to an average of about 4,700 mg/L in the San Juan Basin (see Appendix Table E-3),
23 plays a large role in how the wastewater is managed. In basins with higher TDS such as the San
24 Juan, Uinta, and Piceance, nearly all the wastewater is disposed of in injection wells. Wastewater
25 may also be injected for aquifer storage and recovery, with the intention of later recovering the
26 water for some other use (DOE, 2003).

27 Discharge to rivers and streams for wildlife, livestock, and agricultural use, a management option
28 governed by the CWA, may be permitted in some cases. To be discharged, the wastewater must
29 meet technology-based limitations established by the permit authority and any applicable water
30 quality water quality standards. Direct discharge to streams (with or without treatment) is possible
31 where wastewater is of higher quality. This is a more common method of wastewater management
32 in basins such as the Raton Basin in Colorado and the Tongue River drainage of the Powder River
33 Basin in Montana (NRC, 2010).

34 Agricultural uses include livestock watering, crop irrigation, and commercial fisheries. Livestock
35 watering with CBM wastewater is a common practice, and irrigation is an area of active research
36 (e.g., Engle et al., 2011; NRC, 2010). Irrigation with treated CBM wastewater would be most suitable
37 on coarse-textured soils, for cultivation of salt-tolerant crops (DOE, 2003). NRC (2010) remarks
38 that "use of CBM produced water for irrigation appears practical and sustainable," provided that

1    appropriate measures are taken such as selective application, dilution or blending, appropriate
2    timing, and rehabilitation of soils. Approximately 13% of CBM wastewater in the Powder River
3    Basin in Wyoming, and 26% to 30% in Montana, is used for irrigation (NRC, 2010).

4    As noted above, a degree of treatment is needed for some uses. Plumlee et al. (2014) examined the
5    feasibility, treatment requirements, and cost of several hypothetical uses for CBM wastewater. In
6    several cases, costs for these uses were projected to be comparable to or less than estimated
7    disposal costs. In one case study CBM wastewater for stream augmentation or crop irrigation was
8    estimated to cost between $0.26 and $0.27 a barrel and disposal costs ranged from $0.01 per barrel
9    (pipeline collection system with impoundment) to $2.00 per barrel (hauling for disposal or
10   treatment).

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

### 8.4.6.3.  *Other Documented Uses of Hydraulic Fracturing Wastewater*
16   Uses of wastewater from shales or other hydraulically fractured formations face many of the same
17   possibilities and limitations as those associated with wastewater from CBM operations. The biggest
18   difference is in the quality of the water. Wastewaters vary widely in water quality, with TDS values
19   from shale sand tight sand formations ranging from less than 1,000 mg/L TDS to hundreds of
20   thousands of mg/L TDS (DOE, 2006). Wastewaters on the lower end of the TDS spectrum could be
21   reused in many of the same ways as CBM wastewaters, depending on the concentrations of
22   potentially harmful constituents and applicable federal, state, and local regulations. High TDS
23   wastewaters have more limited uses, and pre-treatment may be necessary (Shaffer et al., 2013;
24   Guerra et al., 2011; DOE, 2006).

25   Documented potential uses for wastewater in the western United States include livestock watering,
26   irrigation, supplementing stream flow, fire protection, road spreading, and industrial uses, with
27   each having their own water quality requirements and applicability (Guerra et al., 2011). Guerra et
28   al. summarized the least conservative TDS standards for five possible uses in the western United
29   States that include 500 mg/L for drinking water (the secondary MCL), 625 mg/L for groundwater
30   recharge, 1,000 mg/L for surface water discharge, 1,920 mg/L for irrigation, and 10,000 mg/L for
31   livestock watering. The authors estimated that wastewater from 88% of unconventional wells in
32   the western United States could be used for livestock watering without treatment for TDS removal
33   based on a maximum TDS concentration of 10,000 mg/L. Wastewater from 10% of unconventional
34   wells were estimated to meet the criterion of 1,000 mg/L TDS for surface water discharge (Guerra
35   et al., 2011).

36   A 2006 Department of Energy (DOE) study points out that the quality necessary for use in
37   agriculture depends on the plant or animal species involved. Other important factors include the
38   sodium adsorption ratio and concentrations of TDS, calcium, magnesium, and other constituents

1   (DOE, 2006). The authors note that in the Bighorn Basin in Wyoming, low salinity wastewater is
2   used for agriculture and livestock watering after minimal treatment to remove oil and grease (DOE,
3   2006).

## 8.5.  Summary and Analysis of Wastewater Treatment

4   A variety of individual treatment techniques and combinations of techniques may be employed for
5   removal of hydraulic fracturing wastewater constituents of concern. These include methods
6   commonly employed in municipal wastewater treatment as well as more advanced processes such
7   as desalination. Treatment technologies are selected based on the water quality of the wastewater
8   to be treated and the effluent concentration required for the intended management method(s) (i.e.,
9   reuse, discharge to POTW, and discharge to surface water body). For example, if reuse is planned,
10  the level of treatment will depend on the water quality needed to formulate the new fracturing
11  fluid.

12  This section discusses treatment technologies that are most effective for removing specific
13  hydraulic fracturing wastewater constituents. It provides information on the unit processes
14  appropriate for treating different types of constituents as well as challenges associated with their
15  use. Considerations when designing a treatment system are also discussed for both centralized and
16  on-site (i.e., mobile) facilities.

### 8.5.1.  Overview of Treatment Processes for Hydraulic Fracturing Wastewater

17  This section provides a brief overview of the treatment technologies used to treat hydraulic
18  fracturing wastewater; Appendix F provides more in-depth descriptions of these technologies.

19  The most basic treatment need for oil and gas wastewaters, including those from hydraulic
20  fracturing operations is separation to remove suspended solids and oil and grease, done using basic
21  separation technologies (e.g., hydrocyclones, dissolved air or induced gas flotation, media filtration,
22  or biological aerated filters). Other treatment processes that may be used include media filtration
23  after chemical precipitation for hardness and metals (Boschee, 2014), adsorption technologies,
24  including ion exchange (organics, heavy metals, and some anions) (Igunnu and Chen, 2014), a
25  variety of membrane processes (microfiltration, ultrafiltration, nanofiltration, RO), and distillation
26  technologies. In particular, advanced processes such as RO or distillation methods (e.g., mechanical
27  vapor recompression (MVR)) are needed for significant reduction in TDS (Drewes et al., 2009; LEau
28  LLC, 2008; Hamieh and Beckman, 2006). An emerging technology is electrocoagulation, which has
29  been used in mobile treatment systems to treat hydraulic fracturing wastewaters (Halliburton,
30  2014; Igunnu and Chen, 2014). Removal efficiencies for hydraulic fracturing wastewater
31  constituents by treatment technology are provided in Appendix F.

### 8.5.2.  Treatment of Hydraulic Fracturing Waste Constituents of Concern

32  The constituents prevalent in hydraulic fracturing wastewater include suspended solids, TDS,
33  anions (e.g., chloride, bromide, and sulfate), metals, radionuclides, and organic compounds (see
34  Section 8.3 and Chapter 7). If the end use of the wastewater necessitates treatment, a variety of
35  technologies can be employed. This section discusses effective unit processes for removing these
36  constituents and provides examples of treatment processes being used in the field as well as

1    emerging technologies. Table 8-6 provides an overview of influent and effluent results at various
2    CWTs for the constituents of concern listed in this section and the specific technology(ies) used to
3    remove them.

**Table 8-6. Studies of removal efficiencies and influent/effluent data for various processes and facilities.**

| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)** | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)** | **Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)** | **9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)** | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)** |
| TSS | Results not reported. | 90% Inf. = 1,272 mg/L Eff. = 9 mg/L Chemical oxidation, coagulation, and clarification | No influent data. Eff.: <4 mg/L Meets NPDES Permit Settling, biological treatment, and induced gas flotation | >90% Inf.: 35 to 114 mg/L Eff.: <3 to 3 mg/L 100 micron mesh bag filter | Results not reported. |
| TDS | >99% Inf. = 8,000 to 15,000 mg/L Eff. = 41 mg/L RO | 99.7% Inf. = 49,550 mg/L Eff. = 171 mg/L MVR (3 units in parallel) | Results not reported. MVR | 98% Inf.: 22,350 to 37,600 mg/L Eff.: 9 to 400 mg/L Thermal distillation | 97% Inf. = 7,000 mg/L Eff. = 180 mg/L Ion exchange softening and double-pass RO |

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| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming ([Shafer, 2011](#))** | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas ([Hayes et al., 2014](#))** | **Judsonia, Sunnydale, Arkansas ([U.S. EPA, 2015f](#))** | **9-month study treating Marcellus Shale waste using thermal distillation ([Boschee, 2014](#); [Bruff and Jikich, 2011](#))** | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) ([Dahm and Chapman, 2014](#); [Webb et al., 2009](#))** |
| Anions | Chloride: >99%<br>Inf. = 3,600 to 6,750 mg/L<br>Eff. = 18 mg/L<br><br>Sulfate: 99%<br>Inf. = 10 to 100 mg/L<br>Eff. = non-detect<br><br>RO | Sulfate: 98%<br>Inf. = 309 mg/L<br>Eff. = 6 mg/L<br><br>Chemical oxidation, coagulation, clarification, and MVR | Sulfate:<br>No influent data.<br>Eff.: 12 mg/L<br><br>Meets NPDES Permit<br><br>MVR | Bromide: >99%<br>Inf.: 101 to 162.5 mg/L<br>Eff.: <0.1 to 1.6 mg/L<br><br>Chloride: 98%<br>Inf.: 9,760 to 16,240 mg/L<br>Eff.: 2.9 to 184.2 mg/L<br><br>Sulfate: 93%<br>Inf.: 20.4 to <100 mg/L<br>Eff.: <1 to 2.2 mg/L | Chloride: >99%<br>Inf. = 3,400 mg/L<br>Eff. = 11 mg/L<br><br>Double-pass RO<br><br>Sulfate: 6%<br>Inf. = 133 mg/L<br>Eff. = 125 mg/L |
| | | | | Fluoride: 96%<br>Inf.: <2 to <20 mg/L<br>Eff.: <0.2 to 0.42 mg/L<br><br>Thermal distillation | Sulfuric acid is added after RO to neutralize the pH so no sulfate removal is expected. |

| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming** (**Shafer, 2011**) | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas** (**Hayes et al., 2014**) | **Judsonia, Sunnydale, Arkansas** (**U.S. EPA, 2015f**) | **9-month study treating Marcellus Shale waste using thermal distillation** (**Boschee, 2014**; **Bruff and Jikich, 2011**) | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas)** (**Dahm and Chapman, 2014**; **Webb et al., 2009**) |
| Metals | Boron: 99%<br>Inf. = 15 to 30 mg/L<br>Eff. = non-detect<br><br>Ion exchange | Iron: >99%<br>Inf. = 28 mg/L<br>Eff. = 0.1 mg/L<br><br>For iron, 90% attributed to chemical oxidation, coagulation, and clarification<br><br>Boron: 98%<br>Inf. = 17 mg/L<br>Eff. = 0.4 mg/L<br><br>Barium: >99%<br>Inf. = 15 mg/L<br>Eff. = 0.1 mg/L<br><br>Calcium: >99%<br>Inf. = 2,916 mg/L<br>Eff. = 3.2 mg/L | Cobalt:<br>No influent data.<br>Eff.: <0.007 mg/L<br><br>Tin:<br>No influent data.<br>Eff.: <0.1 mg/L<br><br>Arsenic:<br>No influent data.<br>Eff.: <0.001 mg/L<br><br>Cadmium:<br>No influent data.<br>Eff.: <0.0001 mg/L<br><br>Chromium:<br>No influent data.<br>Eff.: <0.007 mg/L<br><br>Copper:<br>No influent data.<br>Eff.: <0029 mg/L | Copper: >99%<br>Inf. = <0.2 to <1.0 mg/L<br>Eff. = <0.02 to <0.08 mg/L<br><br>Zinc: inf below detect<br>Inf. = <0.2 to <1.0 mg/L<br>Eff. = <0.02 to 0.05 mg/L<br><br>Barium: >99%<br>Inf. = 260.5 to 405.5 mg/L<br>Eff. = <0.1 to 4.54 mg/L<br><br>Strontium: 98%<br>Inf. = 233 to 379 mg/L<br>Eff. = 0.026 to 3.93 mg/L<br><br>Iron:<br>Inf. = 13.9 to 22.9 mg/L<br>Eff. = <0.02 to 0.06 mg/L<br><br>Manganese: 98%<br>Inf. = 2 to 2.9 mg/L<br>Eff. = <0.02 to 0.04 mg/L | Sodium: 98%<br>Inf. = 2,300 mg/L<br>Eff. = 50 mg/L<br><br>Boron: >99%<br>Inf. = 26 mg/L<br>Eff. = 0.1 mg/L<br><br>RO with elevated influent pH |

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| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)** | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)** | **Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)** | **9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)** | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)** |
| Metals, cont. | | Magnesium: >99% Inf. = 316 mg/L Eff. = 0.4 mg/L  Sodium: >99% Inf. = 10,741 mg/L Eff. = 14.3 mg/L  Strontium: >99% Inf. = 505 mg/L Eff. = 0.5 mg/L  MVR | Lead: No influent data. Eff.: <0.001 mg/L  Mercury: No influent data. Eff.: <0.005 mg/L  Nickel: No influent data. Eff.: 0.002 mg/L  Silver: No influent data. Eff.: <0.0002 mg/L  Zinc: No influent data. Eff.: 0.02 mg/L  Cyanide: No influent data. Eff.: <0.01 mg/L  Meets NPDES permit except for TMDLs for hexavalent chromium and mercury | Boron: 97% Inf. = <1 to 3.12 mg/L Eff. = 0.02 to 0.06 mg/L  Calcium: 98% Inf. = 1,175 to 1,933 mg/L Eff. = 0.36 to 22.2 mg/L  Lithium: 99% Inf. = 9.1 to 14.3 mg/L Eff. = non-detect to 0.13 mg/L  Magnesium: 98% Inf. = 109.8 to 176.8 mg/L Eff. = <0.1 to 2.04 mg/L  Sodium: 98% Inf. = 4,712 to 7,781 mg/L Eff. = 0.37 to 87.9 mg/L  Arsenic: 82% Inf. = <0.01 to 0.028 mg/L Eff. = <0.005 mg/L  Titanium: 86% Inf. = <0.01 to 0.037 mg/L Eff. = <0.005 mg/L | |

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| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)** | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)** | **Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)** | **9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)** | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)** |
| Metals, cont. | | | Settling, biological treatment, induced gas flotation, and MVR | Thermal distillation | |
| Radio-nuclides | Results not reported. | Results not reported. | Not regulated under permit – believed to be absent. | Radium-226: 97% - 99% Inf. = 130 to 162 pCi/L Eff. = 0.224 to 2.87 pCi/L<br><br>Radium-228: 97% - 99% Inf. = 45 to 85.5 pCi/L Eff. = 0.259 to 1.32 pCi/L<br><br>Gross Alpha: 97% - >99% Inf. = 161 to 664 pCi/L Eff. = 0.841 to 6.49 pCi/L<br><br>Gross Beta: 98% - >99% Inf. = 79.7 to 847 pCi/L Eff. = 0.259 to 1.57 pCi/L<br><br>Thorium 232: 71% - 90% Inf. = 0.055 to 0.114 pCi/L Eff. = 0.011 to 0.016 pCi/L<br><br>Thermal distillation | Results not reported. |

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| Constituents of concern | Location and results | | | | |
|---|---|---|---|---|---|
| | **Pinedale Anticline Water Reclamation Facility, Wyoming (Shafer, 2011)** | **Maggie Spain Water-Recycling Facility, Barnett Shale, Texas (Hayes et al., 2014)** | **Judsonia, Sunnydale, Arkansas (U.S. EPA, 2015f)** | **9-month study treating Marcellus Shale waste using thermal distillation (Boschee, 2014; Bruff and Jikich, 2011)** | **San Ardo Water Reclamation Facility, San Ardo, California (Conventional oil and gas) (Dahm and Chapman, 2014; Webb et al., 2009)** |
| Organics | O&G: 99%<br>Inf. = 50 to 2,400 mg/L<br>Eff. = non-detect<br><br>BTEX: 99%<br>Inf. = 28 to 80 mg/L<br>Eff. = non-detect<br><br>Gasoline range organics:<br>RO: 99%<br>Inf. = 88 to 420 mg/L<br>Eff. = non-detect<br><br>Diesel range organics: 99%<br>Inf. = 77 to 1,100 mg/L<br>Eff. = non-detect<br><br>Methanol: 99%<br>Inf. = 40 to 1,500 mg/L<br>Eff. = non-detect<br><br>Oil-water separator, anaerobic and aerobic biological treatment, coagulation, flocculation, flotation, sand filtration, membrane bioreactor, and ultrafiltration | TPH: >80%<br>Inf. = 388 mg/L<br>Eff. = 4.6 mg/L<br><br>BTEX: 94%<br>Inf. = 3.3 mg/L<br>Eff. = 0.2 mg/L<br><br>TOC: 48%<br>Inf. = 42 mg/L<br>Eff. = 22 mg/L<br><br>MVR | Biochemical oxygen demand:<br>No influent data.<br>Eff.: <2 mg/L<br><br>O&G:<br>No influent data.<br>Eff.: <5 mg/L<br><br>Benzo (k) fluoranthene:<br>No influent data.<br>Eff.: <0.005 mg/L<br><br>Bis (2-Ethylhexyl) Phthalate:<br>No influent data.<br>Eff.: <0.001 mg/L<br><br>Butyl benzyl phthalate:<br>No influent data.<br>Eff.: <0.001 mg/L<br><br>Meets NPDES permit<br><br>Settling, biological treatment, induced gas flotation, and MVR | Acetone: 93%<br>Inf. = 8.71 to 13.8 mg/L<br>Eff. = 0.524 to 0.949 mg/L<br><br>Toluene: >80%<br>Inf. = 0.0083 to 0.0015 mg/L<br>Eff. = non-detect to 0.0013 mg/L<br><br>Methane: >99%<br>Inf. = 0.748 to 5.49 mg/L<br>Eff. = non-detect to 0.0013 mg/L<br><br>DRO: 0 to 82%<br>Inf. = 4 to 7.1 mg/L<br>Eff. = 0.99 to 4.9 mg/L<br><br>O&G: No removal<br><br>Thermal distillation | Results not reported. |

### 8.5.2.1. Total Suspended Solids

1   The reduction of TSS is typically required for reuse. Hydraulic fracturing wastewaters containing
2   suspended solids can plug the well and damage equipment if reused for other fracking operations
3   (Tiemann et al., 2014; Hammer and VanBriesen, 2012). For treated water that is discharged, the
4   EPA has a secondary treatment standard for POTWs that limits TSS in the effluent to 30 mg/L (30-
5   day average). In addition, most advanced treatment technologies require the removal of TSS prior
6   to treatment to avoid operational problems such as membrane fouling/scaling and to extend the life
7   of the treatment unit. TSS can be removed by several processes, such as coagulation, flocculation,
8   sedimentation, and filtration (including microfiltration and media and bag and/or cartridge
9   filtration), and with hydrocyclones, dissolved air flotation, freeze-thaw evaporation,
10  electrocoagulation, and biological aerated filters (Boschee, 2014; Igunnu and Chen, 2014; Drewes et
11  al., 2009; Fakhru'l-Razi et al., 2009) (see Appendix F).

12  Technologies that remove TSS have been employed in the Marcellus Shale (sedimentation and
13  filtration) (Mantell, 2013a); Utica Shale (chemical precipitation and filtration) (Mantell, 2013a);
14  Barnett Shale (chemical precipitation and inclined plate clarifier, >90% removal) (Hayes et al.,
15  2014); and Utah (electrocoagulation, 90% removal) (Halliburton, 2014). Details of examples of
16  operating treatment facilities are provided in Table 8-6.

### 8.5.2.2. Total Dissolved Solids

17  The TDS concentration of hydraulic fracturing wastewater is a key treatment consideration, with
18  the TDS removal needed dependent upon the intended use of the treatment effluent. POTW
19  treatment and basic treatment processes at a CWT (i.e., chemical precipitation, sedimentation, and
20  filtration) are not reliable methods for removing TDS. Reduction requires more advanced treatment
21  processes such as RO, nanofiltration, thermal distillation (including MVR), evaporation, and/or
22  crystallization (Olsson et al., 2013; Boschee, 2012; Drewes et al., 2009). RO and thermal distillation
23  processes can treat waste streams with TDS concentrations up to 45,000 mg/L and more than
24  100,000 mg/L, respectively (Tiemann et al., 2014). As noted in section 8.5.1, pretreatment (e.g.,
25  chemical precipitation, flotation, etc.) is typically needed to remove constituents that may cause
26  fouling or scaling or to remove specific constituents not removed by a particular advanced process.
27  Extremely high TDS waters may require a series of advanced treatment processes to remove TDS to
28  desired levels. However, the cost of treating high-TDS waters may preclude facilities from choosing
29  treatment if other options such as deep well injection are available and more cost-effective
30  (Tiemann et al., 2014). Emerging technologies such as membrane distillation and forward osmosis
31  are also showing promise for TDS removal and require less energy compared to other desalination
32  processes (Shaffer et al., 2013).

33  Examples of facilities with advanced technologies and their effectiveness in reducing TDS
34  concentrations in hydraulic fracturing wastewaters from conventional and unconventional
35  resources are summarized in Table 8-6.

### 8.5.2.3. Anions

36  Although chemical precipitation processes can reduce concentrations of multivalent anions such as
37  sulfate, monovalent anions (e.g., bromide and chloride) are not removed by basic treatment

1   processes and require more advanced treatment such as RO, nanofiltration, thermal distillation
2   (including MVR), evaporation, and/or crystallization (Hammer and VanBriesen, 2012).

3   Ion exchange and adsorption are effective treatment processes for removing fluoride but not
4   typically the anions of concern in hydraulic fracturing wastewaters (bromide, chloride, sulfate)
5   (Drewes et al., 2009). Emerging technologies applicable to TDS will typically remove anions.
6   However, issues discussed above, such as the potential for scaling, still apply.

### 8.5.2.4.  Metals and Metalloids

7   Chemical precipitation, including lime softening and chemical oxidation, is effective at removing
8   metals (e.g., sodium sulfate reacts with metals to form solid precipitates such as barium sulfate)
9   (Drewes et al., 2009; Fakhru'l-Razi et al., 2009). However, as mentioned in Section 8.5.2.3, chemical
10  precipitation does not adequately remove monovalent ions (e.g., sodium, potassium), and the
11  produced solid residuals from this process typically require further treatment, such as de-watering
12  (Duraisamy et al., 2013; Hammer and VanBriesen, 2012). Media filtration can remove metals if
13  coagulation/oxidation is implemented prior to filtration (Duraisamy et al., 2013). Advanced
14  treatment processes such as distillation (with pH adjustment to prevent scaling), evaporation, RO,
15  and nanofiltration can remove dissolved metals and metalloids (Hayes et al., 2014; Igunnu and
16  Chen, 2014; Bruff and Jikich, 2011; Drewes et al., 2009). However, if metal oxides are present or
17  formed during treatment, they must be removed prior to RO and nanofiltration processes to
18  prevent membrane fouling (Drewes et al., 2009). Also, boron is not easily removed by RO, achieving
19  less than 50% rejection (the percentage of a constituent captured and thus removed by the
20  membrane) at neutral pH (rejection is greater at higher pH values) (Drewes et al., 2009). Ion
21  exchange can be used to remove other metals such as calcium, magnesium, barium, strontium, and
22  certain oxidized heavy metals such as chromate and selenate (Drewes et al., 2009). Adsorption can
23  remove metals but is typically used as a polishing step to prolong the replacement/regeneration of
24  the adsorptive media (Igunnu and Chen, 2014).

25  The literature provides examples of facilities able to reduce metal and metalloid concentrations in
26  conventional and unconventional hydraulic fracturing wastewaters, some of which are provided in
27  Table 8-6. The facilities in Table 8-6 have achieved removals of 98%–99% for a number of metals.
28  Other work demonstrating effective removal includes a 99% reduction in barium using chemical
29  precipitation (Marcellus Shale region) (Warner et al., 2013a) and over 90% boron removal with RO
30  (at pH of 10.8) at two California facilities (Webb et al., 2009; Kennedy/Jenks Consultants, 2002).
31  However, influent concentration must be considered together with removal efficiency to determine
32  whether effluent quality meets the requirements dictated by end use or by regulations. In the case
33  of the facility described by Kennedy/Jenks Consultants (2002) the boron effluent concentration of
34  1.9 mg/L (average influent concentration of 16.5 mg/L) was not low enough to meet California's
35  action level of 1 mg/L.

36  Newer treatment methods for metals removal include electrocoagulation (Halliburton, 2014;
37  Gomes et al., 2009) and electrodialysis (Banasiak and Schäfer, 2009). Testing of electrocoagulation
38  has been performed in the Green River Basin (Halliburton, 2014) and the Eagle Ford Shale (Gomes
39  et al., 2009). While showing promising results in some trials, results of these early studies have

1    illustrated challenges, with removal efficiencies affected by factors such as pH and salt content.
2    Membrane distillation has also shown promise in removing heavy metals and boron in wastewaters
3    (Camacho et al., 2013).

### 8.5.2.5. Radionuclides

4    Several processes (e.g., RO, nanofiltration, and thermal distillation) are effective for removing
5    radionuclides (Drewes et al., 2009). Ion exchange can be used to treat for specific radionuclides
6    such as radium (Drewes et al., 2009). Chemical precipitation of radium with barium sulfate has also
7    been shown to be a very effective method for removing radium (Zhang et al., 2014b).

8    Data on radionuclide removals achieved in active treatment plants are scarce. The literature does
9    provide some data from the Marcellus Shale region on use of distillation and chemical precipitation
10   (co-precipitation of radium with barium sulfate). The nine-month pilot-scale study conducted by
11   Bruff and Jikich (2011) showed that distillation treatment could achieve high removal efficiencies
12   for radionuclides (see Table 8-6), and Warner et al. (2013b) reported that a CWT achieved over
13   99% removal of radium via co-precipitation of radium with barium sulfate. However, in both
14   studies, radionuclides were detected in effluent samples, and the CWT was discharging to a surface
15   water body during this time (Warner et al., 2013b; Bruff and Jikich, 2011); see Section 8.6.2.
16   Effluent from distillation treatment was found to contain up to 6.49 pCi/L for gross alpha (from 249
17   pCi/L prior to distillation) (Bruff and Jikich, 2011). Between 2010 and 2012, samples of wastewater
18   effluent from a western PA CWT contained a mean radium level of 4 pCi/L (Warner et al., 2013a).

### 8.5.2.6. Organics

19   Because hydraulic fracturing wastewaters can contain various types of organic compounds that
20   each have different properties, specific treatment processes or series of processes are used to
21   target the various classes of organic contaminants. Effectiveness of treatment depends on the
22   specific organic compound and the technology employed (see Appendix F). It should be noted that
23   in many studies, rather than testing for several organic constituents, researchers often measure
24   organics in terms of biochemical oxygen demand and/or chemical oxygen demand, which are an
25   indirect measure of the amount of organic compounds in the water. Organic compounds may also
26   be measured and/or reported in groupings such as total petroleum hydrocarbons (TPH) (which
27   include gasoline range organics (GROs) and diesel range organics (DROs)), oil and grease, VOCs
28   (which include BTEX), and SVOCs.

29   Based on examples found in literature, facilities have demonstrated the capability to treat for
30   organic compounds in hydraulic fracturing wastewaters using a single process or a series of
31   processes (Hayes et al., 2014; Bruff and Jikich, 2011; Shafer, 2011) (see Table 8-6). The processes
32   can include anaerobic and aerobic biological treatment, coagulation, flocculation, flotation,
33   filtration, bioremediation, ultrafiltration, MVR, and dewvaporation. Forward osmosis is an
34   emerging technology that may be promising for organics removal in hydraulic fracturing
35   wastewaters because it is capable of rejecting the same organic contaminants as commercially-
36   available pressure-driven processes (Drewes et al., 2009).

### *8.5.2.7.  Estimated Treatment Removal Efficiencies*

1   There are relatively few studies that have evaluated the ability of individual treatment processes to
2   remove constituents from hydraulic fracturing wastewater and present the resulting water quality.
3   Furthermore, although a specific technology may demonstrate a high removal percentage for a
4   particular constituent, if the influent concentration of that constituent is extremely high, the
5   constituent concentration in the treated water may still exceed permit limits and/or disposal
6   requirements. Appendix Table F-4 presents the results of simple calculations pairing average
7   hydraulic fracturing wastewater concentrations from Chapter 7 with treatment process removal
8   efficiencies reported in the literature in Table F-2.

9   As an example, radium in wastewater from the Marcellus Shale and Upper Devonian sandstones can
10   be in the thousands of pCi/L. With a 95% removal rate, chemical precipitation may result in effluent
11   that still exceeds 100 pCi/L. Distillation and reverse osmosis might produce effluent with
12   concentrations in the tens of pCi/L. A radium concentration of 120 pCi/L, however, could be
13   reduced to less than 5 pCi/L by RO or distillation. Wastewater with barium concentrations in the
14   range of 140 – 160 mg/L (e.g., the Cotton Valley and Mesaverde tight sands) might be reduced to
15   concentrations under 5 mg/L by distillation and roughly 11-13 mg/L by RO. Barium concentrations
16   in the thousands of mg/L would be substantially reduced by any of several processes but might still
17   be relatively high and could exceed 100 m g/L. Table F-4 also illustrates the potential for achieving
18   low concentrations of organic compounds in wastewater treated with freeze-thaw evaporation or
19   advanced oxidation and precipitation.

20   This analysis is intended to highlight the potential impacts of influent concentration on treatment
21   outcome and to illustrate the relative capabilities of various treatment processes for an example set
22   of constituents. Removal efficiencies would differ and likely be greater with a full set of
23   pretreatment and treatment processes that would be seen in a CWT (see Table 8-6).

### 8.5.3.  Design of Treatment Trains for CWTs

24   Based on the chemical composition of the hydraulic fracturing wastewater and the desired effluent
25   water quality, a series of treatment technologies will most likely be necessary. The possible
26   combinations of unit processes to formulate treatment trains are extensive. One report identified
27   41 different treatment unit processes that have been used in the treatment of oil and gas
28   wastewater and 19 unique treatment trains (combinations of unit processes) (Drewes et al., 2009).
29   Fakhru'l-Razi et al. (2009) also provide examples of process flow diagrams that have been used in
30   pilot-scale and commercial applications for treating oil and gas wastewater. Figure 8-8 shows the
31   treatment train for the Pinedale Anticline Facility, which includes pretreatment for dispersed oil,
32   VOCs, and heavy metals and advanced treatment for removal of TDS, dissolved organics, and boron.
33   This CWT can either discharge to surface water or provide the treated wastewater to operators for
34   reuse.
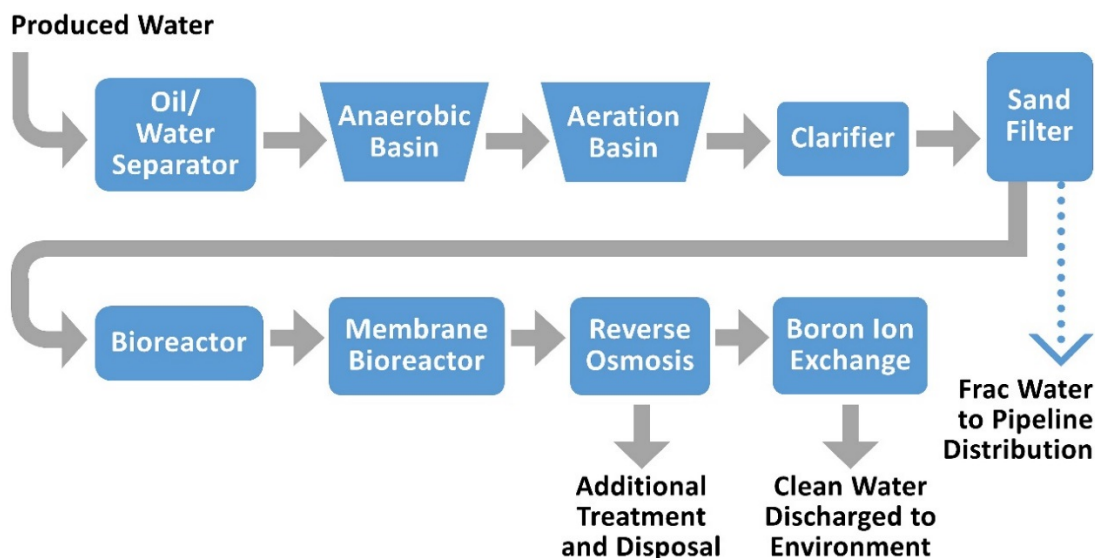
**Figure 8-8. Full discharge water process used in the Pinedale Anticline field.**

Source: Boschee (2012).

1    Table 8-7 provides information on some CWTs in locations across the country and the processes
2    they employ. The table also notes for each facility whether data are readily available on effluent
3    quality. Comprehensive and systematic data on influent and effluent quality from a range of CWTs
4    that treat to a variety of water quality levels is difficult to procure, rendering it challenging to
5    understand removal efficiencies and resulting effluent quality, especially when a facility offers a
6    range of water quality products (e.g., for reuse vs. discharge). For those facilities with NPDES
7    permits, discharge monitoring report (DMR) data may be available for some constituents, although
8    if the facility does not discharge regularly, these data will be sporadic.

9    CWTs such as the Judsonia Central Water Treatment Facility in Arkansas, the Casella-Altela
10   Regional Environmental Services, and Clarion Altela Environmental Services (see Table 8-7)
11   facilities have NPDES permits and use MVR or thermal distillation for TDS removal. As of March
12   2015, the Pinedale Anticline Facility and the Judsonia Facility appear to be the only CWTs in Table
13   8-7 discharging to a surface water body.

**Table 8-7. Examples of centralized waste treatment facilities.**

| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Pinedale Anticline Water Reclamation Facility[a] | WY | Oil/water separation, biological treatment, aeration, clarification, sand filtration, bioreactor, membrane bioreactor, RO, and ion exchange | No - However, facility is permitted to discharge under 40 CFR 435 Subpart E (WY0054224). Facility is permitted to discharge up to 25% of its effluent stream | Yes | Yes, RO (Boschee, 2014, 2012) | The treatment plant produces treated water for reuse and for discharge to surface water. The website indicates the facility is in operation and is recycling to support drilling operations and is discharging to the New Fork River (http://hswater.squarespace.com/pinedale-anticline/). | Yes – DMR data available on Wyoming DEQ website. Some information can also be obtained from Shafer (2011). |
| SEECO – Judsonia Water Reuse Recycling Facility | AR | Settling, biological treatment, induced gas flotation, and MVR | Yes - AR0052051 | Yes | Yes, MVR | The treatment plant provides treated water for reuse and for discharge to surface water. Based on DMR data from late 2014-early 2015, the system is discharging treated water to a surface water body, though intermittently. | DMR data available |

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| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Eureka Resources – Williamsport 2nd Street Facility | PA | Settling, oil/water separation, chemical precipitation, clarification, MVR. Can treat with or without TDS removal. | No - However, future plans to install RO for direct discharge capability | Yes | Yes, MVR | Per Ertel et al. (2013), the facility provides treatment wastewater for reuse and indirect discharge.<br><br>The facility treats entirely or almost entirely hydraulic fracturing wastewater. | No |
| Standing Stone Facility, Bradford County | PA | Settling, oil/water separation, chemical precipitation, clarification, MVR, crystallization | Yes - PA0232351 | Yes | Yes, MVR, crystallizer | The facility can provide treated wastewater for reuse and also has received an NPDES permit for direct discharge.<br><br>The facility treats hydraulic fracturing wastewater. | No |

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| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Wellington Water Works | CO | Dissolved air flotation, pre-filtration, microfiltration with ceramic membranes, activated carbon adsorption. Water is pumped to an aquifer storage and recovery well. Water is then extracted and treated with RO (Alzahrani et al., 2013). | Permit number issued by CO (61879) | Yes | Yes, RO but only after the water is sent to an aquifer storage and recovery well | Per Stewart (2013b), the facility is providing treated wastewater for reuse, for agricultural use, to a shallow well to augment the municipal drinking water supply, and for discharge to the Colorado River. | No |
| Casella Altela Regional Environmental Services (CARES) McKean Facility | McKean County, PA | Pretreatment system (not defined in literature) and thermal distillation | Yes – PA0102288 | Yes | Yes – thermal distillation | The treatment plant is capable of reuse and recycle for fracturing operations and surface water discharge of excess water. However, the facility's website indicates it is only treating water for reuse/recycle as of early 2015 (http://caresforwater.com/location/cares-mckean). | No - just NPDES discharge requirements |

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| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Clarion Altela Environmental Services (CAES) Facility | Clarion County, PA | Pretreatment system (not defined in literature) and thermal distillation | Yes – PA0103632 | Yes | Yes – thermal distillation | The treatment plant capable of reuse and recycle for fracturing operations and surface water discharge of excess water. However, the facility's website indicates it is only treating water for reuse/recycle as of early 2015 (http://caeswater.com/technology/). | No – just NPDES discharge requirements |
| Terraqua Resource Management (aka. Water Tower Square Gas Well Wastewater Processing Facility) | Lycoming County, PA | Equalization tanks, oil-water separation via chemical addition (sulfuric acid, emulsion breaker), pH adjustment, coagulation, flocculation, inclined plate clarifier, sand filtration | Yes – PA0233650 Permit pending approval for discharge to stream (as of 4/17/2009) | Yes | No – However, TARM recognizes that they can't discharge, until they install TDS treatment | According to its website (last updated 2012), the facility reuses/recycles treated water for fracturing operations (http://www.tarmsolutions.com/solutions/). | No |
| Maggie Spain Water-Recycling Facility | Decatur, TX | Settling, flash mixer with lime and polymer addition, inclined plate clarifier, surge tank, MVR | No | Yes | Yes – MVR | A 17-month pilot study using a commercial-scale mobile treatment facility was concluded in 2011. The status is unclear as of early 2015. | Yes – Some information can be obtained from Hayes et al. (2014). |

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| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Fountain Quail/NAC Services - Kenedy | Kenedy, TX | Oil-water separator, coagulation, flocculation, sedimentation, filtration, MVR. | No | Yes | Yes – MVR | According to its website, the facility reuses/recycles treated water for fracturing operations (http://www.aqua-pure.com/operations/shale/ford/ford.html). | No |
| Purestream - Gonzales facility | Gonzales, TX | Induced gas flotation and MVR | No | Yes | Yes - MVR | Per Dahm and Chapman (2014) commercial operations deployed March 2014 for reuse/recycle for fracturing operations. | No |
| LINN Energy Fyre Ranch - Granite Wash | Wheeler County, TX | Induced gas flotation and MVR | No | Yes | Yes - MVR | AVARA system installed for reuse/recycle in June 2014. http://purestream.com/index.php/water-management/vapor-recompression/photos-and-videos | No |

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| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Fluid Recovery Service Josephine Facility[b] | PA | Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier | Expired - PA0095273 | No | No | The facility claims to have stopped accepting Marcellus wastewater September 30, 2011 (Ferrar et al., 2013). It treats conventional oil and gas wastewater. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less. | Yes – Some effluent results obtained from Ferrar et al. (2013) and Warner et al. (2013a). Also minimal DMR data from the EPA. |
| Fluid Recovery Service Franklin Facility[b] | PA | Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier | Expired - PA0101508 | No | No | This facility is not accepting wastewater from hydraulic fracturing operations as of January 2015. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less. | Minimal DMR data from the EPA. |

| Facility | State | Description of Unit Processes | Does CWT have a NPDES permit for discharge? | Does CWT provide effluent for reuse? | Does CWT have advanced process for TDS removal? | What is the status of the facility as of January 2015? | Are effluent quality data available through literature search? |
|---|---|---|---|---|---|---|---|
| Hart Resources-Creekside Facility[b] | PA | Oil-water separator, aeration, chemical precipitation with sodium sulfate, lime, and a polymer, inclined plate clarifier | Expired - PA0095443 | No | No | This facility is not accepting wastewater from hydraulic fracturing operations as of January 2015. The facility will be upgrading to include evaporative technology that will enable it to attain monthly average TDS levels of 500 mg/L or less. | Minimal DMR data from the EPA. |

[a] For Pinedale Anticline Water Reclamation Facility, surface water discharges are permitted under 40 CFR 435 Subpart E (beneficial use subcategory agricultural and wildlife water) not 40 CFR 437 (the discharge permit for CWTs). For the purposes of this assessment, this facility is included with CWTs.

[b] As of May 15, 2013, these facilities are under an administrative order (AO). According to the AO, these facilities must comply with a monthly effluent limit for TDS not to exceed 500 mg/L. This will allow them to treat high-saline wastewaters typical of unconventional oil and gas operations. To meet the requirements of the AO, they have applied to PADEP for a NPDES permit and are planning to install treatment for TDS.

## 8.6. Potential Impacts on Drinking Water Resources

1  Several articles have noted potential effects of hydraulic fracturing wastewater on water resources
2  (Vengosh et al., 2014; Olmstead et al., 2013; Rahm et al., 2013; States et al., 2013; Vidic et al., 2013;
3  Rozell and Reaven, 2012; Entrekin et al., 2011), with one study using probability modeling
4  indicating that water pollution risk associated with gas extraction in the Marcellus Shale is highest
5  for the wastewater disposal aspects of the operation (Rozell and Reaven, 2012). Whether drinking
6  water resources are affected by hydraulic fracturing wastewater depends at least in part upon the
7  characteristics of the wastewater, the form of discharge or other management practice, and the
8  processes used if the wastewater is treated. Other site-specific factors (e.g., size of receiving water
9  and volume of wastewater) determine the magnitude and nature of potential effects, but a thorough
10  exploration of local factors is beyond the scope of this assessment. The majority of hydraulic
11  fracturing wastewater is either injected into a disposal well or, in the case of the Marcellus region,
12  reused for other hydraulic fracturing jobs. Potential impacts on drinking water resources may occur
13  on a local level through several routes: treated wastewater may be discharged directly from
14  centralized waste treatment facilities (CWTs) or indirectly from publicly owned treatment works
15  (POTWs) that receive CWT effluent; sediments in water bodies receiving effluent may accumulate
16  contaminants; spills or leaks may be associated with on-site storage or transportation (see Chapter
17  7); and in previous years, hydraulic fracturing wastewater treated at POTWs was discharged to
18  surface waters.

19  It has been suggested that the most significant effects of hydraulic fracturing on surface water
20  quality are related to discharges of partially treated wastewater, although these effects vary
21  according to region (Kuwayama et al., 2015). A recent study (Bowen et al., 2015) concluded that
22  there is currently no clear evidence of national-level trends in surface water quality (as measured
23  by specific conductivity and chloride) in areas where unconventional oil and gas production is
24  taking place. These authors note that available national level databases have limitations for
25  assessing this question.

26  Pits and impoundments associated with waste management may have impacts on drinking water
27  resources and are discussed in Chapter 7. In addition, unauthorized discharge of wastewater is a
28  potential mechanism for impacts on drinking water resources. Descriptions of several incidents and
29  resulting legal actions have been publicly reported. However, such events are not generally
30  described in the scientific literature, and the prevalence of this type of activity is unclear.

31  Important considerations regarding the potential impact of hydraulic fracturing wastewater on a
32  receiving waterbody include whether constituents in the wastewater are known to have health
33  effects, if they are regulated drinking water contaminants, or if they may give rise to regulated
34  compounds. For some classes of constituents, such as disinfection by-product (DBP) precursors,
35  considerable research exists. For others, information is limited regarding their concentrations in
36  effluents and whether they are likely to affect drinking water at intakes. The following subsections
37  identify several classes of constituents known to occur in hydraulic fracturing wastewater, discuss
38  whether potential impacts are likely, and provide specific examples of information gaps.

### 8.6.1. Bromide and Chloride

1  Bromide and chloride are two constituents commonly found in high-total dissolved solids (TDS)
2  hydraulic fracturing wastewater. As noted in section 8.3.1.1, chloride is a regulated contaminant
3  with a secondary MCL standard of 250 mg/L. Bromide is not regulated but is of concern due to its
4  role in the formation of DBPs (Parker et al., 2014; Krasner, 2009) (see Appendix F for information
5  on DBP formation). High-TDS wastewaters from the Marcellus Shale can be of concern because the
6  limited availability of underground injection for disposal can result in a higher rate of discharge of
7  treated wastewater to surface waters compared to other parts of the country. In response to
8  concerns in part over bromide in discharges, operators in Pennsylvania have discontinued the
9  practice of sending wastewater from hydraulic fracturing operations to POTWs (States et al., 2013).
10  Also, CWTs have been shifting towards treatment of those wastewaters for reuse rather than
11  discharging to surface water bodies (Hammer and VanBriesen, 2012).

12  States et al. (2013) found a strong correlation between bromide concentrations in source water
13  from the Allegheny River in Pennsylvania and the percentage of brominated trihalomethanes in
14  finished drinking water. The authors noted that source water containing 50 μg/L bromide resulted
15  in treated water with approximately 62% of its finished water total trihalomethanes consisting of
16  bromoform, dibromochloromethane, and bromodichloromethane. Source water containing 150
17  μg/L bromide resulted in finished water TTHMs composed of approximately 83% brominated
18  species. Allegheny River bromide concentrations measured during the study ranged from less than
19  25 μg/L to 299 μg/L, with the highest bromide concentrations measured under low-flow
20  conditions. Industrial wastewater sites accounted for approximately 50% of the increase in
21  bromide load as water moved downriver.

22  In addition, a related constituent, iodide can be a constituent in hydraulic fracturing wastewater
23  (see Chapter 7). Although its effects have not been as well documented as those associated with
24  bromide (Xu et al., 2008), iodide raises some of the same concerns (such as DBP formation) as
25  bromide does (Parker et al., 2014; Krasner, 2009). Iodinated DBPs are not regulated by the EPA as
26  of early 2015.

27  As discussed in Section 8.5, removal of dissolved solids, including chloride and bromide, requires
28  advanced treatment processes such as reverse osmosis (RO), distillation, evaporation, or
29  crystallization. Unless the treatment plant receiving the high-TDS wastewater employs processes
30  specifically designed to remove these constituents, effluent discharge may contain high levels of
31  bromide and chloride. Drinking water treatment plants with intakes downstream of these
32  discharges may receive water with correspondingly higher levels of bromide and chloride and may
33  have difficulty complying with Safe Drinking Water Act (SDWA) regulations related to DBPs.

### 8.6.1.1. Effects on Receiving Streams

34  Studies show that discharges from oil and gas wastewater treatment facilities can elevate TDS,
35  bromide, and chloride levels in receiving waters (States et al., 2013; Wilson and Van Briesen, 2013).
36  Wilson and Van Briesen (2013) measured bromide, chloride, and other constituents at water
37  intakes downstream of wastewater discharges for three years in the Monongahela River in western
38  Pennsylvania. By evaluating water chemistry data in the context of flow measurements, the authors

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1    attributed an overall decrease in bromide concentrations from 2010 to 2012 to a decrease in
2    bromide loading; they note that this is likely to be associated with a decrease in management of
3    fossil fuel wastewater at treatment plants that discharge to surface water.

4    Although treatment plant effluents will be diluted upon reaching the receiving water, the dilution
5    may not be adequate to avoid water quality problems if there are existing pollutant loads in the
6    waterbody from contributors such as acid mine drainage or power plant effluents (Ferrar et al.,
7    2013). Warner et al. (2013a) evaluated effluent from the Josephine Brine Treatment Facility (which
8    treated both conventional and unconventional oil and gas wastewater at the time of the study) and
9    concluded that even a 500 to 3,000-fold dilution of the wastewater would not reduce bromide
10   levels to background. In addition, downstream levels of chloride in the receiving stream were
11   elevated, with a downstream value of 88 mg/L as compared to an upstream value of 18 mg/L.

12   A study by Hladik et al. (2014) focused on sampling at sites downstream and near the outfalls of
13   plants that treated oil and gas wastewater in Pennsylvania. The authors documented brominated
14   and iodinated DBPs (e.g., dibromochloronitromethane (DBNM); dibromoiodomethane) at the
15   outfalls of CWTs and POTWs and noted that this DBP signature was different than for those plants
16   that did not accept oil and gas wastewater. For example, concentrations of
17   dibromochloronitromethane ranged from 0.26 to 8.7 µg/L, and dibromoiodomethane was
18   measured at 0.98 and 1.3 µg/L; neither compound was detected at an upstream site or at the outfall
19   of the POTW not accepting oil and gas wastewater. These brominated and iodinated compounds are
20   considered more toxic than other types of DBPs (Richardson et al., 2007). Hladik et al. note that
21   these elevated DBP levels could contribute to DBPs at downstream drinking water intakes and can
22   also be an indicator of the potential for more highly brominated and iodinated DPBs forming in
23   drinking treatment plants downstream of these discharges (Hladik et al., 2014). The sites studied
24   by Hladik et al. (2014) received wastewater from both conventional and unconventional oil and gas
25   development.

26   Research suggests that a relatively small portion of hydraulic fracturing wastewater effluent can
27   notably affect DBP formation. In laboratory studies, Parker et al. (2014) diluted hydraulic fracturing
28   wastewater from the Marcellus and Fayetteville shales with Allegheny and Ohio River waters and
29   then disinfected the mixtures. In chlorinated samples containing as little as 0.01% hydraulic
30   fracturing wastewater, the THM composition shifted significantly away from chloroform species to
31   a greater representation of brominated and iodinated species.

32   Elevated concentrations of bromide in effluents can place a burden on downstream drinking water
33   treatment systems. States et al. (2013) studied influent and finished water at the Pittsburgh Water
34   and Sewer Authority (PWSA) drinking water system, concluding that elevated bromide in the
35   source water led to elevated total trihalomethanes (TTHM) formation in the treated drinking water.
36   The authors also noted a substantial increase in the percentage of brominated TTHMs (States et al.,
37   2013), as discussed above. The utility modified their treatment process and proposed
38   improvements to their storage facilities to address the elevated TTHM levels in the distribution
39   system (Chester Engineers, 2012).

### 8.6.1.2. Modeling

1   The EPA's contaminant modeling shows that that the strategies most likely to reduce bromide
2   impacts on downstream users include reducing effluent concentrations (e.g., discharging flowback
3   versus produced water), discharging during higher stream flow periods, and using a pulsing or
4   intermittent discharge. Weaver et al. (In Press) developed a computer model to estimate river and
5   stream bromide concentrations after treated water discharges. The model utilizes existing data for
6   bromide concentrations in produced water, flowback, and mixtures, combined with existing stream
7   flow data from USGS stations in Pennsylvania. The model parameters include steady state versus
8   transient inputs to receiving waters, high and low streamflow months, varying effluent
9   concentration and types (produced, flowback, and mixed). For steady-state scenarios in the model,
10  bromide concentrations are lowest under high flow conditions with lower concentrations of
11  effluent (flowback and mixed water).

12  A source apportionment study conducted by the EPA considered the relative contributions of
13  bromide, chloride, nitrate, and sulfate from CWTs primarily treating hydraulic fracturing
14  wastewater to the Allegheny River Basin and to water at two downstream public water system
15  intakes on the Allegheny River (U.S. EPA, 2015p). The Allegheny River and its tributaries receive
16  runoff and discharges containing an array of contaminants, including these anions. Contaminant
17  sources include discharges from CWTs for oil and gas wastewater, runoff from acid mine drainage
18  and mining operations, discharges from coal-fired electric power stations, industrial wastewater
19  treatment plant effluents, and POTW discharges. The Allegheny River is also the water supply for
20  thirteen public water systems that serve over 500,000 people in western Pennsylvania.

21  In Pennsylvania, wastewater produced from hydraulic fracturing of the Marcellus formation has
22  been mostly diverted from CWTs and POTWs that discharge to public waters in the state (Hammer
23  and VanBriesen, 2012). Wastewater produced from hydraulic fracturing of non-Marcellus
24  formations, however, continues to be sent to surface-discharging facilities on the Allegheny River.

25  The source apportionment study considered contributions of bromide, chloride, nitrate and sulfate
26  to public water supplies from CWTs and other upriver sources by: developing chemical source
27  profiles, or fingerprints, for discharges upstream of the public water system intakes; characterizing
28  water quality in the river upstream and downstream of the CWTs, electric generating stations, and
29  industrial facilities; characterizing the water quality at the public water system intakes; and
30  analyzing the sampling data collected with the EPA Positive Matrix Factorization (PMF) receptor
31  model in order to quantify relative contributions of contaminant sources to the anions found at the
32  public water system intakes. The study focused on low-flow conditions.

33  CWTs and coal-fired power plants with flue gas desulfurization were found to contribute bromide
34  to the two public water supply intakes. Although acid mine drainage also contributed bromide, its
35  contribution was minor (9% at one intake) compared to the contributions from the CWTs (89%
36  and 37% at the two intakes) and coal-fired power plants (50-59% at one intake). The CWTs, coal-
37  fired power plants, and acid mine drainage combined accounted for 88–89% of the bromide at one
38  intake and 96% of the bromide at the other intake.

### *8.6.1.3. Summary*

1   Most drinking water treatment plants are not designed to address high concentrations of TDS
2   (including bromide and iodide), limiting their options for restricting the formation of brominated
3   and iodinated DBPs. Tighter restrictions on TDS in effluent from POTWs and CWTs have led to a
4   reduction in in-stream bromide concentrations. Advanced treatment processes at CWTs such as
5   reverse osmosis, distillation, evaporation, or crystallization can reduce chloride, bromide, and
6   iodide in surface waters. Strategies such as reducing effluent concentrations, discharging during
7   higher stream flow periods, and utilizing a pulsing or intermittent discharge could also reduce the
8   potential impact of elevated TDS on drinking water treatment plants.

## 8.6.2. Radionuclides

9   Potential impacts on drinking water resources from technologically enhanced naturally occurring
10  radioactive material (TENORMs) associated with hydraulic fracturing wastewater may arise from a
11  number of sources, including: treated wastewater that does not have adequately reduced
12  radionuclide concentrations, accumulation of radionuclides in surface water sediments
13  downstream of wastewater treatment plant discharge points, migration from soils that have
14  accumulated radionuclides from previous activities such as pits or land application, and inadequate
15  management of treatment plant solids that have accumulated radionuclides (such as filter cake).

16  In Pennsylvania between 2007 and 2010, TENORM-bearing produced wastewaters were sent to
17  POTWs, which are generally not required to monitor for radioactivity (Resnikoff et al., 2010).
18  Although the practice of management of Marcellus waters via POTWs has declined, there is still
19  potential for input of radionuclides to surface waters via discharge of CWT effluents either directly
20  to surface waters or indirectly through discharge to POTWs.

21  Data regarding TENORM content in oil and gas wastes that are treated and discharged to surface
22  waters are limited. However, a recent study by the Pennsylvania Department of Environmental
23  Protection (PA DEP) (PA DEP, 2015b) provides information that helps fill this data gap. The study
24  began in 2013 and examined radionuclide (radium-226, radium-228, K-40, gross alpha, gross beta)
25  levels at 29 wastewater plants in Pennsylvania that cover a range of both sources and treatment
26  plant types, including POTWs, CWTs that treat oil and gas wastewaters and can discharge to surface
27  water or a POTW, and zero liquid discharge facilities treating oil and gas wastewater. Four of the 10
28  discharging CWTs sampled during the study discharged to surface water under a National Pollution
29  Discharge Elimination System (NDPES) permit, and the others discharged to POTWs. Six of the
30  POTWs in the study received effluent from a CWT along with municipal wastewater. The CWTs in
31  the study are not described as receiving exclusively Marcellus wastewater, but the study itself was
32  motivated by concerns over an increase in radionuclides in oil and gas wastes observed during the
33  expansion of Marcellus Shale production.

34  The POTWs receiving influent from CWTs treating oil and gas wastewater (along with municipal
35  wastewater influent) had average effluent radium-226 concentrations of 103 pCi/L (unfiltered) and
36  129 pCi/L (filtered) (filtration is used to remove very fine particulates from the water). Those
37  POTWs not receiving influent from CWTs treating oil and gas wastewater effluent had higher
38  average radium-226 values in unfiltered samples (145 pCi/L) and lower values for filtered samples

1   (47 pCi/L). For perspective, the maximum contaminant level (MCL) for radium-226 plus radium-
2   228 is 5 pCi/L. For reference, radium-226 in river water generally ranges from 0.014 pCi/L to 0.54
3   pCi/L (0.5 to 20 mBq/L) (IAEA, 2014). The results of the POTW sampling are inconclusive as to
4   whether the effluents from POTWs receiving CWT-treated oil and gas wastewater are routinely
5   higher than the effluents from those without this type of influent.

6   For the CWTs in the PA DEP study, average radium-226 content in the effluents was an order of
7   magnitude higher than in effluents from the POTWs (1,840 pCi/L unfiltered, 2,100 pCi/L, filtered).
8   The effluent averages were similar to averages for the influent concentrations, although median
9   concentrations in the effluents were much lower than in the influents. Effluent from zero-discharge
10  facilities averaged 2,610 pCi/L radium-226 and 295 pCi/L radium-228, although these effluents
11  would most likely be reused as fracturing fluid (PA DEP, 2015b). The authors do note a potential for
12  environmental effects from spills of influent or effluent from zero-discharge facilities.

13  Warner et al. (2013a) noted that if the activities of radium-226 and radium-228 in Marcellus brine
14  influent at the CWT they studied are similar to those reported by other researchers (Rowan et al.,
15  2011), then the CWT achieved a 1,000-fold reduction in radium content via a process of radium
16  coprecipitation with barium sulfate. The detection of radium in effluents from this CWT (mean
17  values of 4 pCi/L of radium-226 and 2 pCi/L of radium-288) even with what may be high treatment
18  removal efficiency underscores the fact that effluent concentrations depend not only upon the
19  treatment processes used but also the influent concentration.

20  An additional concern related to evaluation of radionuclide concentrations in wastewater is that the
21  high TDS content of hydraulic fracturing wastewater can result in poor recovery of chemical
22  constituents when using wet chemical techniques, leading to underestimations of constituent
23  concentrations. In particular, recovery for radium may be as low as <1% (Nelson et al., 2014).
24  Underestimation of radium content may lead to failure in identifying an impact or potential impact
25  on drinking water resources.

26  In addition to concerns over of the potential for TENORM in discharges to surface waters, there are
27  may be a legacy of accumulation of radionuclides in surface water sediments. Studies of effluent,
28  stream water, and stream sediment associated with a CWT in western Pennsylvania that has
29  treated both conventional and unconventional oil and gas wastewaters indicate that radium-226
30  levels in stream sediment samples at the point of discharge are approximately 200 times greater
31  than upstream and background sediments. This indicates the potential for accumulation of
32  contaminants in localized areas of wastewater discharge facilities (Warner et al., 2013a). Although
33  the CWT in question also accepted conventional oil and gas wastewater, Warner et al. (2013a)
34  observed that the radium-228/radium-226 ratio in the river sediments near the discharge (0.22-
35  0.27) is consistent with ratios in Marcellus wastewater. The authors interpret this as an indication
36  that the radium accumulated in the sediments originated from the discharge of treated
37  unconventional oil and gas wastewater. Another study, however, did not find elevated levels of
38  alkali earth metals (including radium) in sediments just downstream of the discharge points of five
39  POTWs that had previously treated Marcellus wastewater (Skalak et al., 2014). Accumulation of
40  contaminants in sediment may be dependent on treatment processes and their removal rates for

1  various constituents as well as stream chemistry and hydrologic characteristics. Contamination
2  with radium-226 would be potentially be long lived; the half-life of radium-226 is approximately
3  1,600 years, while the half-life of radium-228 is 5.76 years.

4  The recent PA DEP study (PA DEP, 2015b) found that the radium-226 content in sediments near the
5  discharge points for POTWs receiving treated oil and gas effluent from CWTs (along with their
6  municipal wastewater influent) exceeds typical background soil levels of approximately 1 to 2
7  pCi/g of radium-226 and radium-228. The authors conclude that wastewater effluent is the most
8  likely source for the radium in these samples. Results indicate an average of 9.00 pCi/g radium-226
9  and 3.52 pCi/g radium-228 in sediments near outfalls of POTWs. Sediments at 4 CWTs receiving oil
10  and gas wastewater and that discharge to surface water have much higher average concentrations
11  of 84.2 pCi/g for radium-226 and 19.8 pCi/g for radium-228. However, the concentrations of
12  radium in the sediments does not correlate with concentrations of radium in the effluents
13  suggesting that sorption over time affects the concentration of radium in the sediments (PA DEP,
14  2015b).

15  The association of radium with sediments near discharge points is attributed to adsorption of
16  radium to the sediments, a process governed by factors such as the salinity of the water and
17  sediment characteristics. In particular, radium has a high affinity for iron and manganese
18  (hydr)oxides in sediment. Increased salinity promotes desorption of radium from sediments, while
19  lower salinity promotes adsorption, with radium adsorbing particularly strongly to sediments high
20  in iron and manganese (hydro)oxides (Porcelli et al., 2014; Gonneea et al., 2008). Warner et al.
21  (2013a) speculate that the discharge of saline CWT effluent into less saline stream water facilitates
22  sorption of radium onto streambed sediments. The long-term fate of radium sorbed to sediments
23  depends upon changes in water salinity and the sediment properties, including any redox processes
24  that may affect iron and manganese minerals in the sediments.

25  Other solids may contain radionuclides; filter cake samples from treatment at POTWs were found
26  by PA DEP (2015b) to have radium contents greater than typical soil concentrations, and they
27  exhibited a large variation. Filter cake from CWTs had radium concentrations higher than in POTW
28  filter cake. The authors conclude that although the risk to workers and the public from handling
29  and temporary storage of these materials is minimal, there may be environmental risks from spills
30  or long term disposal. There could be impacts on surface waters through spills or effects on ground
31  waters from landfill leachate.

32  Radionuclide accumulation in CWTs or POTWs may continue to affect the plant even after
33  discontinuing treatment of high radionuclide wastewater. Radium can adsorb onto scales in pipes
34  and tanks and will also co-precipitate calcium, barium, and strontium in sulfate minerals (USGS,
35  2014e). Pipe scale in oil and gas production facilities can have radium concentrations as high as
36  154,000 pCi/g, although concentrations of less than 13,500 pCi/g are more common (Schubert et
37  al., 2014). A similar issue, the potential for accumulation and possible release of radionuclides and
38  other trace inorganic constituents in water distribution systems has gained attention, with the
39  potential for drinking water concentrations to exceed drinking water standards (Water Research
40  Foundation, 2010). Scale eventually removed from pipes or other equipment may end up in

1   landfills and then leach into groundwater or run off to a surface water body (USGS, 2013c).
2   Although barium sulfate phases are relatively insoluble, one study demonstrates that barium
3   sulfate scales that were buried in soil could be reduced by microbially mediated processes, allowing
4   release of co-precipitated elements such as radium due to leaching by rainwater (Swann et al.,
5   2004). Monitoring would be needed in order to ascertain the potential for accumulation and release
6   of radionuclides from systems that have treated or continue to treat hydraulic fracturing
7   wastewaters with elevated TENORM concentrations.

8   Accumulation of radionuclides (potassium, thorium, bismuth, radium, and lead) has been evaluated
9   in two pits in Texas that have stored fluids associated with hydraulic fracturing (Rich and Crosby,
10  2013). Gamma radiation in these pits has been found to vary from 8 to 23 pCi/g, with beta radiation
11  varying from 6 to 1329 pCi/g (Rich and Crosby, 2013). Although the study sample size was small,
12  the results suggest that radionuclides associated with sediments from some reserve pits could have
13  potential impacts on surface waters or ground waters. This could happen through migration of
14  affected sediments or soils to surface waters or through leaching to ground water.

15  Salt and radionuclide accumulation can occur near road spreading sites; one study in Pennsylvania
16  found a 20% increase in radium concentrations in soils near roads where wastewaters from
17  conventional operations had been spread for de-icing (Skalak et al., 2014). Accumulation of
18  radionuclides in soils near roads presents a vehicle for potential impacts on drinking water
19  resources. The extent to which hydraulic fracturing wastewater contributes to this depends upon
20  state-level regulations regarding whether hydraulic fracturing wastewater can be used for road
21  spreading.

22  Effluents and receiving waters can be monitored for radionuclides. Research suggests that radium-
23  226 and radium-228 are the predominant radionuclides in Marcellus Shale wastewater, and they
24  account for most of the gross alpha and gross beta activity in the waters studied (Rowan et al.,
25  2011). Gross alpha and gross beta measurement may therefore serve as an effective screening
26  mechanism for overall radionuclide concentrations in hydraulic fracturing wastewater. This in turn
27  can help in evaluating management strategies. Portable gamma spectrometers allow rapid
28  screening of wastewater effluents. Sediments may also be measured for radionuclide
29  concentrations at discharge points.

### 8.6.3. Metals

30  Given the presence in hydraulic fracturing wastewaters of some heavy metals, as well as barium
31  and strontium concentrations that can reach hundreds or even thousands of mg/L (see Table 7-10),
32  surface waters may be impacted if discharges from CTWs or POTWs indirectly receiving oil and gas
33  wastewater via CWTs are not managed appropriately. Spills may also affect surface waters.

34  Common treatment processes, such as coagulation, are effective at removing many metals (see
35  Section 8.5.2.4). A request by the EPA for effluent sampling from seven facilities in Pennsylvania
36  treating oil and gas wastewaters revealed low to modest concentrations of copper (0-50 μg/L), zinc
37  (14 – 256 μg/L), and nickel (8 – 22 μg/L) (U.S. EPA, 2015d, e). However, metals such as barium and
38  strontium have been found to range from low to elevated in some CWT effluents. For the year 2011,

1  for example, effluent from a Pennsylvania CWT had average barium levels ranging from 9 to 98
2  mg/L (PA DEP, 2015a). That facility was operating with a barium removal stage and was treating
3  both conventional and hydraulic fracturing wastewater, although effluent concentrations dropped
4  after May, 2011. The facility is scheduled to upgrade its TDS removal capabilities.

5  Data collected by the EPA between October 2011 and February 2013 at seven Pennsylvania
6  facilities indicate effluent barium concentrations ranging from 0.35 to 25 mg/L (median of 3.5 mg/L
7  and average of 6.7 mg/L). Strontium concentrations ranged from 0.36 to 546 mg/L (median of 297
8  mg/L and mean of 236 mg/L (U.S. EPA, 2015e). A December 2010 effluent sampling effort in at a
9  discharging CWT in Pennsylvania reported average barium and strontium concentrations of 27
10  mg/L and nearly 3,000 mg/L, respectively (eight samples from one plant) (Volz et al., 2011). The
11  facility treats conventional oil and gas wastewaters, and it also received Marcellus wastewater until
12  September, 2011.

13  Limited data are available on metal concentrations in wastewater and treated effluents that are
14  directly discharged; additional information would be needed to assess whether there will be
15  downstream effects on drinking water utilities. NPDES discharge permits, which restrict TDS
16  discharge concentrations, will likely reduce metal effluent concentrations due to the additional
17  treatment necessary to minimize TDS.

### 8.6.4. Volatile Organic Compounds

18  Benzene is a common constituent in hydraulic fracturing wastewater, and it is of concern due to
19  recognized human health effects. A wide range of concentrations of BTEX compounds occurs in
20  wastewater from the Barnett and Marcellus shales. Natural gas formations generally produce more
21  BTEX than oil formations (Veil et al., 2004). Generally, lower concentrations of BTEX occur in
22  wastewater from coalbed methane (CBM) production (see Appendix Table E-9). Processes such as
23  aeration or dissolved air flotation can remove volatile organic compounds (VOCs) during treatment,
24  but if treatment is not adequate, the VOCs may reach water resources. The average concentration of
25  benzene in a December 2010 sampling effort was 12 μg/L in the discharge of a Pennsylvania CWT
26  (Volz et al., 2011). The facility was receiving wastewater from both conventional and
27  unconventional operations at that time. Ferrar et al. (2013) measured mean concentrations of
28  benzene, toluene, ethylbenzene, and xylene in effluents from the same facility, and mean
29  concentrations among the four compounds ranged from about 2 to 46 μg/L. Concentrations were
30  lower for samples taken after May 19, 2011 than before, and the effect was considered statistically
31  significant. The treatment processes at this facility do not include aeration.

32  Leakage from pits or spills creates another potential route of entry to drinking water resources.
33  VOCs have been measured in groundwater near the Duncan Oil Field in New Mexico, downgradient
34  of an unlined pit storing oil and gas wastewater (Sumi, 2004; Eiceman, 1986). VOCs and oil were
35  also found in groundwater about 213 feet (65m) downgradient from an unlined pit in Oklahoma
36  (Kharaka et al., 2002).

### 8.6.5. Semi-Volatile Organic Compounds

1  Little is known about the fate of the SVOC, 2-butoxyethanol (2-BE) (an antifoaming and anti-
2  corrosion agent used in slick-water) (Volz et al., 2011) or its potential impact on surface waters,
3  drinking water resources, or drinking water systems. This compound is very soluble in water and is
4  subject to biodegradation, with a half-life estimation of 1-4 weeks in the environment (Wess et al.,
5  1998). The EPA has not classified 2-BE (or other glycol ethers) for carcinogenicity. 2-BE was
6  detected in the discharge of a Pennsylvania CWT at concentrations of 59 mg/L (Volz et al., 2011).
7  Ferrar et al. (2013) detected 2-BE in the effluents from a CWT in western Pennsylvania at average
8  concentrations of 34 – 45 mg/L; the latter value was measured when the CWT was receiving only
9  conventional oil and gas wastewater. Data are lacking on 2-BE concentrations in surface waters that
10  receive treated effluents from hydraulic fracturing wastewater treatment systems.

11  Polycyclic aromatic hydrocarbons are another common group of semi-volatile organic compounds
12  (SVOCs) in oil and gas wastewater. They have been detected in soils 164 feet (50 m) downgradient
13  of an unlined pit in New Mexico (Sumi, 2004; Eiceman, 1986). PAHs were also found in birds in
14  wetlands fed by oil and gas wastewater discharges in Wyoming (Ramirez, 2002).

### 8.6.6. Oil and Grease

15  Oil and gas wastewater often contains oil and grease from the formation or from oil-based drilling
16  fluids. Typically, oil and grease are separated from the wastewater before discharge either by a heat
17  treatment or by allowing gravity separation followed by skimming. If these processes are
18  inefficient, oil and grease may be integrated with the discharge to surface waters. For example, in
19  some cases, oil and grease are allowed to separate in pits, and water is then withdrawn from the
20  lower part of the pit with a standpipe. If the oil layer is allowed to drop to the level of the standpipe
21  or if the water is agitated, oil and grease may be discharged along with the water. Oil and grease are
22  also often dispersed in wastewater in the form of small droplets that are 4 to 6 microns in diameter.
23  These droplets can be difficult to remove using typical oil/water separators (Veil et al., 2004). In a
24  study by the U.S. Fish and Wildlife Service regarding permitted oil and gas discharges between
25  1996 and 1999 from Wyoming oil fields, 15% of the 62 discharges to Wyoming wetlands reviewed
26  showed visible oil sheens in the receiving water and 10 of the sites sampled exceeded discharge
27  limits of 10 mg/L of oil and grease (Ramirez, 2002).

## 8.7. Synthesis

28  Hydraulic fracturing operations produce fluids during the flowback and production phases
29  (collectively called wastewater) of a production well, along with liquid and solid treatment
30  residuals from treatment processes. A variety of management strategies may be considered, with
31  cost frequently a driving factor. Available information suggests that Class IID wells regulated under
32  the Underground Injection Control (UIC) Program are the most frequently used wastewater
33  management practice, but reuse, discharge after treatment, and various other uses are also
34  employed.

### 8.7.1.  Summary of Findings

1   Hundreds of billions of gallons of wastewater are generated annually in the United States by the oil
2   and gas industry, although national level estimates are difficult to reliably obtain. It is also difficult
3   to produce a nationwide estimate of the amount of wastewater that is attributable specifically to
4   hydraulic fracturing because some states do not specifically identify wastewater from hydraulic
5   fracturing operations in their available wastewater data.

6   The total amount of wastewater produced in an area corresponds generally to oil and gas
7   production and, therefore, may increase if hydrocarbon production increases in a region.
8   Geographically, a large portion of oil and gas wastewater in the United States is reported to be
9   generated in the western part of the country, including contributions from both conventional and
10  unconventional resources. For some states, estimates of hydraulic fracturing wastewater volumes
11  can be made using publicly available production or waste data. Annual estimates compiled in this
12  way range from hundreds of millions to billions of gallons of wastewater generated per state per
13  year. Direct comparisons among these state data are problematic, however, because of a great deal
14  of variability in state data collection, including differences in the years for which data are available,
15  and challenges in definitively identifying wells that have been hydraulically fractured (to
16  distinguish hydraulic fracturing wastewater from that generated from wells that are not
17  hydraulically fractured). Within a given state, however, estimated volumes in areas where
18  hydraulic fracturing is practiced extensively have generally increased over the last several years,
19  along with numbers of wells contributing to total wastewater volumes. For example, the data made
20  available by PA DEP illustrate that the total volume of wastewater generated correlates generally
21  with a significant increase in volume of hydrocarbon production and with the number of
22  production wells. As hydraulic fracturing activities increase and the number of wells increases, the
23  amount of hydraulic fracturing wastewater generated is likely to increase.

#### 8.7.1.1.  *Wastewater Management Practices*

24  Hydraulic fracturing wastewater is managed in a variety of ways, including disposal through Class
25  IID wells; minimal treatment and reuse in subsequent fracturing operations; more complete
26  treatment followed by discharge, disposal, or reuse; evaporation; and other uses such as irrigation
27  (when the wastewater quality is adequate). Unauthorized discharges of hydraulic fracturing
28  wastewaters have been documented; such discharges could potentially impact drinking water
29  resources, but estimates of the frequency of occurrence cannot be developed with the available
30  data.

31  As of 2015, available information suggests that wastewater management practices involve
32  extensive use of Class II wells to manage wastewater from most of the major unconventional plays
33  in the United States, with the notable exception of the Marcellus Shale region in Pennsylvania. More
34  than 98% of wastewater in the oil and gas industry is estimated to be injected into Class II wells
35  annually (including wells for enhanced oil recovery and disposal) (Clark and Veil, 2009). Based on
36  data compiled from 2012 and 2014, there are about 25,000 Class IID wells in the United States (U.S.
37  EPA, 2015q). In particular, large numbers of active injection wells are found in Texas (7,876 or

1  29%), Kansas (5,516 or 20%), Oklahoma (4,622 or 17%), Louisiana (2,448 or 9%), and Illinois
2  (1,054 or 4%).

3  Use of Class IID wells is likely driven by the availability of Class IID wells within reasonable
4  transportation distance and the cost of transporting (and injecting) the wastewaters. In the oil and
5  gas industry, Class IID wells have generally been the most economically favorable wastewater
6  management practice (U.S. GAO, 2012). In Pennsylvania, there are only nine Class IID wells as of
7  February 2015, and a significant growth of gas production using hydraulic fracturing in the
8  Marcellus is generating increasing amounts of wastewater. Treatment and reuse are becoming
9  increasingly popular in the Marcellus Shale region and are in more widespread use in comparison
10  to other oil and gas producing parts of the country.

11  Reuse of hydraulic fracturing wastewater to formulate fracturing fluid in subsequent hydraulic
12  fracturing jobs varies considerably on a national level, and reliable estimates are not available for
13  all areas. As of 2014–2015, the greatest amount of reuse occurs in Pennsylvania, where the scarcity
14  of Class IID wells to receive Marcellus wastewater drives this practice. Recent estimates of
15  wastewater reuse in Pennsylvania range as high as 90% or more. Waste disposal data from the PA
16  DEP (2015a) indicate that much of the reuse happens on-site. Operators also report some reuse of
17  wastewater in other regions such as the Haynesville Shale, the Fayetteville Shale, the Barnett Shale,
18  and the Eagle Ford Shale, although at much lower volume percentages (about 5 – 20%) compared
19  with practices in the Marcellus Shale region. Increased reuse and recycling of hydraulic fracturing
20  wastewaters has the added benefit of providing an additional water supply for hydraulic fracturing
21  fluid formulation in areas where water scarcity is a concern. If, however, hydraulic fracturing
22  activity slows, demand for wastewater for reuse will also decrease, and other forms of wastewater
23  management will be needed.

24  The decision to reuse/recycle depends upon several factors, including the volume and rate of
25  production of the wastewater and whether these are suitable for water needs for ongoing
26  fracturing activities in the area. The composition of the water, in particular the TSS and TDS
27  content, and whether the water quality can be accommodated in the fracturing practices in an area
28  can also influence reuse, including decisions about what type of pretreatment or treatment may be
29  needed to make reuse or recycling feasible.

30  Treatment facilities (either centralized waste treatment facilities (CWTs) or systems designed for
31  on-site use) can be permitted to treat oil and gas wastewaters. Treatment can be followed by
32  discharge to a surface water body or to a POTW, or the treated effluent may be used for reuse. Most
33  CWTs treating hydraulic fracturing wastewater are located in Pennsylvania (39 facilities), and a
34  number of CWTs (11) are located in Ohio. More are under construction or pending approval. Most
35  are "zero-discharge" and do not have the treatment capacity to reduce TDS; their effluent is reused
36  for hydraulic fracturing. Specialized on-site, mobile, or semi-mobile treatment facilities can be used
37  by operators to handle wastewater without the expense of long-distance transportation and can be
38  customized to produce an effluent that meets the water quality needs of the intended disposal or
39  reuse plans.

1   Treatment of hydraulic fracturing wastewaters by publicly owned treatment works (POTWs) was
2   previously practiced in Pennsylvania. POTWs are not designed for the high TDS content of
3   Marcellus wastewaters, and stricter discharge limits for TDS in Pennsylvania, as well as a positive
4   response to a request from Pennsylvania DEP that operators stop sending Marcellus wastewater to
5   POTWs and some CWTs, led to the practice being discontinued in 2011. (Some POTWs in
6   Pennsylvania still accept oil and gas wastewaters from conventional operations, including
7   conventional wells that have undergone hydraulic fracturing.)

8   Management plans will necessarily need to change with time as hydraulic fracturing activities in a
9   region change. The volumes of wastewater also change during the life of a well. The chemical
10  composition of the wastewater changes during the transition from the flowback period and into the
11  production phase. In addition, the demand for reused water to support ongoing fracturing activities
12  will change. Taken in aggregate, these factors may influence costs and choices associated with
13  hydraulic fracturing wastewater management, especially if Class IID wells are limited in a particular
14  area for any reason.

### 8.7.1.2.  Treatment and Discharge

15  One of the most frequently cited concerns regarding hydraulic fracturing wastewater, especially
16  from shale plays and tight sand plays, is the high TDS content, which poses challenges for
17  treatment, discharge, and reuse. Conventional treatment processes such as sedimentation, filtration
18  methods, flotation, chemical precipitation and ion exchange can remove constituents such as oil and
19  grease, major cations, metals, and TSS. Because these processes do not remove monovalent ions
20  (e.g., chloride, bromide, sodium), reducing TDS in these high-salinity wastewaters requires more
21  advanced processes such as reverse osmosis (RO), electrodialysis, and distillation methods.
22  Distillation methods appear to be the approach of choice for newer CWT facilities that are designed
23  to lower TDS. RO, while highly effective, does have limits to TDS concentrations (less than
24  approximately 40,000 mg/L) that it can treat (Shaffer et al., 2013; Younos and Tulou, 2005).

25  Hydraulic fracturing wastewater discharged from treatment facilities without advanced TDS
26  removal processes has been shown to cause elevated TDS, bromide, and chloride levels in receiving
27  waters in Pennsylvania. Existing literature indicates that bromide and chloride are important
28  wastewater constituents with regard to potential burdens on downstream drinking water
29  treatment facilities. Bromide in particular is of concern due to the formation of disinfection by-
30  products (DBPs) during disinfection. Some types of DBPs are regulated under SDWA's Stage 1 and
31  Stage 2 DBP Rules, but a subset of DBPs, including a number of chlorinated, brominated,
32  nitrogenous, and iodinated DBPs, are not regulated. Brominated DBPs (and iodinated DBPs) are
33  more toxic than other species of DBPs. Modeling suggests that very small percentages of hydraulic
34  fracturing wastewater in a river used as a source for drinking water treatment plants may cause a
35  notable increase in DBP formation.

36  Radionuclides (in particular radium-226 and radium-228) in some hydraulic fracturing
37  wastewaters pose concerns for the quality of discharges if they are not adequately treated. Possible
38  elevated radionuclide content in treatment residuals is also a consideration. In Marcellus Shale gas
39  production wastewater, radium-226, radium-228, gross alpha, and gross beta are most cited as the

1  radioactive constituents of concern, and concentrations can range up to thousands of pCi/L. Fewer
2  data exist on uranium content in wastewaters, and data are also limited on radionuclide
3  concentrations in wastewaters from other unconventional plays. A confounding issue in evaluating
4  radium concentrations is underestimation when using traditional wet chemical methods with high-
5  TDS waters. A variety of treatment processes can be used for removal of radium, ranging from
6  conventional methods such as chemical precipitation and filtration to more advanced and costly
7  techniques, such as reverse osmosis or distillation (including mechanical vapor recompression).
8  Whether the effluent from such treatment contains elevated radium, however, will depend upon
9  influent concentrations as well as treatment removal efficiency.

10  Other potential effects on drinking water resources may result from discharges or spills of
11  hydraulic fracturing wastewaters containing elevated concentrations of barium and other metals.
12  Again, the management strategy and treatment choices will affect the likelihood of such impacts.

### 8.7.2.  Factors Affecting the Frequency or Severity of Impacts

13  On a regional scale, potential effects on drinking water resources from hydraulic fracturing
14  wastewater will depend upon the mix of wastewater management strategies used, and potential
15  impacts may change through time if the quantity of hydraulic fracturing wastewater changes and
16  strategies to manage the wastewater change. For example, if use of Class IID wells becomes
17  restricted in parts of the country where they are currently commonly used, the emphasis may shift,
18  at least locally, from use of Class IID wells and towards the use of treatment and either discharge or
19  reuse. Although reuse delays the discharge of wastewater by directing it to ongoing fracturing
20  activities, reuse may ultimately concentrate constituents such as radionuclides (depending upon
21  the ratio of recycled to new water). If a stream of wastewater or portion of wastewater has been
22  used for more than one hydraulic fracturing event and is eventually intended for disposal, the
23  method of disposal will need to be appropriate for the quality of the wastewater.

24  Potential effects on drinking water resources from hydraulic fracturing wastewaters that undergo
25  treatment depend upon the quality and quantity of discharges to receiving waters (discharge could
26  occur directly after treatment at a CWT or indirectly after discharge to a POTW). Hydraulic
27  fracturing wastewater management can consider appropriate levels of treatment and blending so
28  that the resulting TDS content in a receiving water will not result in formation of DBPs during
29  subsequent drinking water treatment and will not impair biological treatment processes.

30  The volumes of discharges relative to the receiving water body size are important local factors to
31  consider in evaluating whether elevated concentrations can be anticipated at downstream drinking
32  water intakes. Small drinking water systems drawing water from smaller streams in affected areas
33  would likely face greater challenges in dealing with high bromide and chloride levels in source
34  waters. Furthermore, other potential impacts on surface water and shallow ground water may exist
35  due to spills of either untreated wastewater or effluent from zero-discharge CWTs, and there will be
36  site-specific factors such as distance to a water body or depth to the water table to consider (see
37  Chapter 5).

1    Results from existing literature and recent PA DEP data suggest that cumulative impacts from
2    radionuclides may occur in sediments at or near discharge points from facilities that treat and
3    discharge oil and gas wastewater (or have done so in the past). There may be consequences for
4    downstream drinking water systems if the sediments are disturbed or entrained due to dredging or
5    flood events. Similarly, some organic constituents may not be removed during treatment, and
6    potential effects on receiving waters and sediments will depend upon the properties of the specific
7    constituents, their concentrations, and the treatment used.

8    The possibility of radionuclides affecting receiving waters and sediments will depend upon the
9    technologically enhanced naturally occurring radioactive material (TENORM) content of the
10   wastewater and the treatment processes used. Although radionuclide contamination at drinking
11   water intakes due to treated hydraulic fracturing fluid has not been detected, a recent PA DEP study
12   (PA DEP, 2015b) has revealed radium in effluents from both CWTs handling oil and gas wastewater
13   and POTWs receiving effluent from such facilities. The concentrations in the CWT effluents were
14   considerably higher than in the POTW effluents. The site selection criteria for this study included
15   some Pennsylvania wastewater facilities whose influents include wastewater from unconventional
16   operations or where radioactivity was measured in the influent, sludges, or effluents (CWTs may
17   also receive conventional wastewater). In regions where unconventional plays are known to be
18   enriched in radionuclides, analysis of TENORMs in untreated hydraulic fracturing wastewaters,
19   selection of appropriate treatment processes, and monitoring of TENORMs in treatment effluent
20   and receiving waters could help address potential impacts on drinking water resources. Gross alpha
21   and gross beta measurements or gamma spectroscopic analyses could be used as initial screening
22   methods for radionuclides. Enrichment of TENORMs in waste products from treatment processes
23   also requires appropriate management to reduce potential impacts.

24   Other management strategies such as irrigation, road spreading, and evaporation are less
25   frequently employed for hydraulic fracturing wastewaters. Irrigation or land application may have
26   potential effects on surface waters depending upon the constituents in the wastewater (e.g., salts
27   and radionuclides), the distance from the site of application to a receiving water, and whether
28   stormwater management measures exist that mitigate runoff. Distance to the water table,
29   precipitation, and the hydrogeologic properties of the soil and sediment will influence whether
30   migration of these constituents results in contamination of shallow ground water.

### 8.7.3. Uncertainties

31   A full understanding of the practices being used for management of hydraulic fracturing
32   wastewaters is limited by a lack of available data in a number of areas. It is difficult to assemble a
33   complete, national- or regional-level picture of wastewater generation and management practices
34   because the tracking and availability of data vary from state to state. Although some states provide
35   well-organized and relatively thorough data, not all states make such information available, and it
36   can be difficult to identify wastewater volumes specifically associated with hydraulic fracturing
37   activity (as compared to all oil and gas production activities). Such data would be needed to place
38   hydraulic fracturing wastewater in the broader context of all oil and gas wastewaters. Data are also

1    generally difficult to locate for production volumes, chemical composition, masses, and
2    management and disposal strategies for residuals.

3    Among management practices, up-to-date information on the volumes of hydraulic fracturing
4    wastewaters disposed of via underground injection in different states are not uniformly available.
5    Without this information, it is difficult to assess whether disposal well capacity will become an
6    issue in areas where hydraulic fracturing activity is expected to increase.

7    Assessment of the potential effects of hydraulic fracturing on drinking water resources is also
8    limited by relatively few data on effluent quality from CWTs that receive oil and gas wastewaters
9    (including those associated with hydraulic fracturing) and POTWs that receive CWT effluents. If a
10   CWT can discharge to surface water (e.g., the CWT has a NPDES permit), some monitoring data may
11   be available that will provide information on effluent quality, but the list of monitored constituents
12   may be limited.

13   In evaluating the treatment effectiveness of full scale facilities, relatively few data exist on the
14   quality of both influents and effluents from treatment facilities, although some manufacturers of
15   patented CWT systems publicize information on treatment effectiveness. A better understanding of
16   the pollutant removal capabilities of facilities would be helped by influent and effluent sampling,
17   timed so that effluent samples are representative of influent samples to the degree possible. There
18   are limited analyses of influent and effluent samples for a wide range of constituents associated
19   with hydraulic fracturing fluids and wastewaters (e.g., major cations and anions, radionuclides,
20   metals, VOCs, SVOCs, diesel range organics (DROs), and total petroleum hydrocarbons (TPH)).
21   Analyses are needed in which the methods are appropriate for the TDS content of the sample.
22   Radium in particular needs to be analyzed using a method suitable for high-salt samples, otherwise
23   concentrations may be underestimated. Continued work towards ensuring that analytical methods
24   exist for the highly complex matrixes often encountered with oil and gas wastewater would provide
25   better certainty in the results of chemical analyses.

26   Monitoring of surface waters, even screening with a simple TDS proxy such as conductivity, would
27   be needed to help assess how often hydraulic fracturing activities (including spills or discharges of
28   wastewater) affect receiving waters; such data are lacking except for some studies in the Marcellus
29   Shale region. Existing data are also limited regarding legacy effects, such as accumulation of
30   contaminants in sediments at discharge points, soil accumulation due to application of de-icing
31   brines or salts from wastewater treatment, and handling of waste water treatment residuals.

### 8.7.4. Conclusions

32   Oil and gas operations in the United States generate billions of gallons of wastewater daily; this
33   includes wastewater associated with hydraulic fracturing activities, although what portion of this
34   oil and gas wastewater is attributable to hydraulic fracturing operations is difficult to estimate due
35   to lack of consistent data regarding wastewater volumes. Available information indicates that the
36   majority of this water is injected into Class IID wells regulated under the Underground Injection
37   Control (UIC) program, although in some areas of the country, wastewater is reused (either with
38   our without treatment) for new hydraulic fracturing jobs. In the Marcellus Shale region in

1    Pennsylvania, the majority of wastewater is currently reused. Wastewater may also be treated in a
2    CWT and discharged to a surface water body or to a POTW, or in certain settings, used for various
3    other uses (e.g., irrigation) if water quality allows. Impacts on drinking water resources may result
4    from inadequate treatment prior to discharge or spills. Particular constituents of concern in
5    wastewater from hydraulic fracturing, especially in the Marcellus Shale region, include bromide and
6    radionuclides. There is limited information regarding the influents and effluents from facilities that
7    treat wastewater from hydraulic fracturing operations.

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8    **Text Box 8-2. Research Questions Revisited.**

9    **What are the common treatment and disposal methods for hydraulic fracturing wastewater, and**
10   **where are these methods practiced?**

11   •   The majority of hydraulic fracturing wastewater in the United States is disposed of via underground
12       injection wells. As of 2014-2015, most states where hydraulic fracturing occurs have an adequate
13       number of Class IID injection wells regulated under the Underground Injection Control (UIC) program.
14       The Marcellus Shale region, especially the northeastern region, is an exception. Wastewater treatment for
15       reuse is increasing in the Marcellus shale region and may continue to increase in western shale plays as
16       the practice becomes encouraged and economically favorable.

17   **How effective are conventional POTWs and commercial treatment systems in removing organic and**
18   **inorganic contaminants of concern in hydraulic fracturing wastewater?**

19   •   Publicly owned treatment works (POTWs) using basic treatment processes cannot effectively reduce
20       elevated total dissolved solids (TDS) concentrations in hydraulic fracturing wastewater. Centralized
21       waste treatment facilities (CWTs) with advanced treatment processes can remove TDS constituents with
22       removal efficiencies ranging from 97% to over 99% as demonstrated at facilities that use treatment
23       processes such as mechanical vapor recompression, distillation, and reverse osmosis (see Table 8-6).
24       Advanced treatment processes have been shown to remove certain contaminants found in hydraulic
25       fracturing wastewater (see Table 8-6). Indirect discharge, where wastewater is pretreated by a CWT and
26       sent to a POTW, may be an effective option for hydraulic fracturing wastewater treatment (with
27       restrictions on contaminant concentrations in the pretreated wastewater that is sent to a POTW). This
28       option would require careful planning to ensure that the pretreated wastewater blended with POTW
29       influent is of appropriate quality and quantity to prevent deleterious effects on biological processes in the
30       POTW or the pass-through of contaminants.

31   •   Facilities that treat wastewater for reuse and employ only basic treatment are unable to remove all
32       contaminants in hydraulic fracturing wastewater. Depending on the water quality requirements for a
33       particular site, these lower quality treated waters may be of adequate quality for reuse on subsequent
34       hydraulic fracturing operations (and will be less costly). Some organic compounds (BTEX, some alcohols,
35       2-butoxyethanol) may not be removed by the processes employed in CWTs if they don't include specific
36       processes that target these compounds (e.g., distillation, advanced oxidation, adsorption).

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1  **What are the potential impacts on drinking water treatment facilities from surface water disposal of**
2  **treated hydraulic fracturing wastewater?**

3  •  Inadequate bromide and iodide removal from treated hydraulic fracturing wastewater has the greatest
4     potential to affect surface water quality and place a burden on downstream drinking water treatment
5     facilities that use chlorine-based disinfection due to the formation of DBPs. Radionuclides, metals, and
6     trace organic compounds in effluents from CWTs may also be of concern if present in treated wastewater
7     or if they accumulate in sediments downstream of discharge points. These constituents have reached
8     drinking water resources via some discharges, although sampling data for effluents and receiving waters
9     are limited. As of 2014-2015, there is no evidence that these contaminants have affected drinking water
10    facilities, but data are lacking for concentrations of these constituents at drinking water intakes in
11    regions with hydraulic fracturing.

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# Chapter 9

# Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

# 9. Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

## 9.1. Introduction

1  Chapters 4 through 8 of this assessment each present a stage of the hydraulic fracturing water cycle
2  and the mechanisms by which activities in those stages produce potential impacts on drinking
3  water resources. In contrast, this chapter presents and integrates what is known about chemicals
4  *across* stages of the hydraulic fracturing water cycle (i.e., used in hydraulic fracturing fluids and
5  detected in hydraulic fracturing wastewater). The discussion is focused on available information
6  about (1) chronic toxicity values—specifically, the available noncancer oral reference values (RfVs)
7  and cancer oral slope factors (OSFs)—of chemicals that could occur in drinking water resources;
8  and (2) properties of chemicals that could affect their occurrence in drinking water resources (see
9  Chapters 5 and 7).[1,2] To the extent that information was available to do so, knowledge of
10  toxicological and chemical properties was combined to illustrate an approach that may provide
11  preliminary insights about the relative hazard potential that chemicals could pose to drinking water
12  resources.

13  Risk assessment and risk management decisions will be informed by the scientific information on
14  the toxicity of chemicals in hydraulic fracturing fluid and wastewater, which recent authors note is
15  incomplete (Goldstein et al., 2014). The U.S. House of Representatives' Committee on Energy and
16  Commerce Minority Staff released a report in 2011 noting that more than 650 products (i.e.,
17  chemical mixtures) used in hydraulic fracturing contain 29 chemicals that are either known or
18  possible human carcinogens or are currently regulated under the Safe Drinking Water Act (House
19  of Representatives, 2011). However, that report did not characterize the potential toxicity of many
20  of the other compounds known to occur in hydraulic fracturing fluids or wastewater. More recently,
21  Kahrilas et al. (2015) reviewed the toxicity and physicochemical properties of biocides used in
22  hydraulic fracturing. Stringfellow et al. (2014) examined the toxicity and physicochemical
23  properties of several classes of chemicals that are reportedly used in hydraulic fracturing; however,
24  this study only reported acute toxicity (from lethal doses), which may differ from the effects of low-
25  dose, chronic exposure to these chemicals. Wattenberg et al. (In Press) assessed the acute and
26  chronic toxicity data that was available for 168 chemicals from the FracFocus database that had at
27  least 25 reports of use in North Dakota. The authors found that 113 of these chemicals had some
28  health hazard data available, but determined that there were significant data gaps, particularly with
29  regards to what is known about the potential chronic toxicity of these chemicals. Overall, available

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[1] A reference value (RfV) is an estimate of an exposure for a given duration to the human population (including susceptible subgroups) that is likely to be without an appreciable risk of adverse health effects over a lifetime. RfV is a generic term not specific to a given route of exposure. In the context of this chapter, the term RfV refers to reference values for noncancer effects occurring via the oral route of exposure and for chronic durations, except where noted. Source: IRIS Glossary (U.S. EPA, 2011d).
[2] An oral slope factor (OSF) is an upper-bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent. This estimate, usually expressed in units of proportion (of a population) affected per mg/kg day, is generally reserved for use in the low dose region of the dose response relationship, that is, for exposures corresponding to risks less than 1 in 100.

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1   information indicates that there may be hundreds of chemicals associated with the hydraulic
2   fracturing water cycle for which toxicological data is limited or unavailable.

3   Furthermore, the potential public health impact of hydraulic fracturing processes is not well
4   understood (Finkel et al., 2013; Colborn et al., 2011). Potential public health implications are
5   highlighted in the recent studies by McKenzie et al. (2014) and Kassotis et al. (2014), but as of early
6   2015, there is a lack of published, peer-reviewed epidemiological or toxicological studies that have
7   examined health effects resulting from water contamination due to hydraulic fracturing. However,
8   numerous authors have noted that with the recent increase in hydraulic fracturing operations there
9   may be an increasing potential for significant public health and environmental impacts via ground
10  and surface water contamination (Goldstein et al., 2014; Finkel et al., 2013; Korfmacher et al., 2013;
11  Weinhold, 2012).

12  This chapter provides a compilation of the chemicals used or released during the fracturing
13  process, and information about their potential health effects. The data are presented in this chapter
14  as follows.

15  Section 9.2 discusses how ten information sources, including the EPA's analysis of the FracFocus
16  database (U.S. EPA, 2015a), were used to create a list of chemicals used in or detected in various
17  stages of the hydraulic fracturing water cycle. This chemical list was initially presented in the EPA's
18  2012 interim progress report (U.S. EPA, 2012f), and has been updated in this assessment with
19  additional chemicals from FracFocus. The consolidated chemical list includes chemicals that are
20  reportedly added to hydraulic fracturing fluids in the chemical mixing stage, as well as fracturing
21  fluid chemicals, formation chemicals, or their reaction products that may be carried in flowback or
22  produced water. Although over half of the chemicals cited on this list are listed in the EPA
23  FracFocus database, this chapter is not meant to be interpreted as a hazard evaluation of the
24  chemicals listed in the EPA FracFocus database alone.

25  Section 9.3 provides an overview of the methods that were used for gathering information on
26  toxicity and physicochemical properties for all chemicals that were identified in Section 9.2, and
27  outlines the number of chemicals that had available data on these properties. For toxicological data,
28  the primary focus is on peer-reviewed, selected chronic oral RfVs and OSFs. This section also
29  discusses additional potential sources of toxicity information: estimates of toxicity predicted using
30  Quantitative Structure Activity Relationship (QSAR) modeling, or toxicological information
31  available on the EPA's Aggregated Computational Toxicology Resource (ACToR) database. This
32  chapter is focused on potential human health hazards of chemicals for the oral route of exposure
33  (drinking water); therefore, the toxicological properties and physicochemical ranking metrics
34  described herein (see below) do not necessarily apply to other routes of exposure, such as
35  inhalation or dermal exposure. In addition, this analysis is focused on individual chemicals rather
36  than mixtures of chemicals used as additives. Furthermore, the propensity for a chemical to pose a
37  physical hazard (e.g. the flammability and explosiveness of stray gas methane) are not considered
38  here.

1  Many chemicals reported in hydraulic fracturing were identified as being of interest in previous
2  chapters of this report. This includes the most frequently used chemicals in hydraulic fracturing
3  fluid (Chapter 5), the most and least mobile chemicals in hydraulic fracturing fluid (Chapter 5), and
4  inorganic chemicals and pesticides that may be detected in flowback and produced water (Chapter
5  7). The available selected chronic oral RfVs and OSFs for these chemicals are summarized in Section
6  9.4.

7  Section 9.5 describes the hazard identification and hazard evaluation of chemicals for which data
8  was available on toxicity, occurrence, and physicochemical properties.[1,2] For hazard identification,
9  the selected chronic oral RfVs and OSFs and health effects for these chemicals are presented and
10  summarized. To illustrate one approach to integrate toxicity, occurrence and physicochemical
11  properties data to generate a hazard potential score, a multicriteria decision analysis (MCDA)
12  framework was developed. In this context, occurrence and physicochemical property data were
13  used as metrics to estimate the likelihood that a chemical could impact drinking water resources.
14  Chemicals considered in these hazard evaluations include a subset of chemicals from the FracFocus
15  database, as well as a subset of chemicals that have been detected in flowback and produced water.

16  In general, characterizing chemicals and their properties on a national scale is challenging and that
17  the use and occurrence of chemicals is likely to differ between geological basins and possibly on a
18  well-to-well basis (see Chapters 5 and 7). Therefore, for the protection of public health at the
19  community level, chemical hazard evaluations may be most useful to conduct on a regional or site-
20  specific scale. This level of analysis is outside the scope of this report; however, the methods of
21  hazard evaluation presented here can also be applied on a regional or site-specific scale in order to
22  identify chemicals that may present the greatest potential human health hazard.

## 9.2. Identification of Chemicals Associated with the Hydraulic Fracturing Water Cycle

23  As the initial step towards developing a hazard evaluation, the EPA compiled a list of chemicals that
24  are used in or released by hydraulic fracturing operations across the country (U.S. EPA, 2012f). Ten
25  sources of information (described in Appendix A) were used to develop this list. This consolidated
26  list was used to compile two sublists: (1) a list of chemicals known to be used in hydraulic
27  fracturing fluids, and (2) a list of chemicals that are reported to have been detected in hydraulic
28  fracturing flowback and produced water. It is likely that, as industry practices change, chemicals
29  may be used or detected that are not included on these lists. In addition, those chemicals that are
30  considered proprietary and identified as confidential business information (CBI) by well operators
31  are not listed or considered.

---

[1] Hazard identification is a process for determining if a chemical or a microbe can cause adverse health effects in humans and what those effects might be. See Terms of Environment at:
http://iaspub.epa.gov/sor_internet/registry/termreg/searchandretrieve/termsandacronyms/search.do.
[2] Hazard evaluation is a component of risk assessment that involves gathering and evaluating data on the types of health injuries or diseases (e.g., cancer) that may be produced by a chemical and on the conditions of exposure under which such health effects are produced. See Terms of Environment at:
http://iaspub.epa.gov/sor_internet/registry/termreg/searchandretrieve/termsandacronyms/search.do.

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1   In total, the EPA identified 1,173 chemicals as being used in hydraulic fracturing fluid and/or
2   detected in flowback and produced water. The complete list of chemicals and associated data is
3   available in Appendices A and B.[1]

### 9.2.1.  Chemicals Used in Hydraulic Fracturing Fluids

4   Of the 1,173 total chemicals, the EPA identified 1,076 chemicals as being used in hydraulic
5   fracturing fluids. Of these, 692 chemicals were listed in the FracFocus database, and therefore had
6   information available in order to calculate their nationwide frequency of use ([U.S. EPA, 2015a](#)).[2]
7   Frequency of use for individual chemicals ranged from low (481 chemicals on the list were used in
8   less than 1% of wells nationwide) to very high (methanol was used in 73% of wells nationwide).
9   Furthermore, only 32 chemicals (excluding water, quartz, and sodium chloride) were reported in at
10  least 10% of the disclosures nationwide (see Section 5.4 and Table 5-2). As noted previously, the
11  FracFocus database does not list or consider those chemicals identified as CBI. The EPA determined
12  that approximately 70% of the disclosures in the FracFocus database contain at least one CBI
13  chemical, and for those disclosures, the average number of CBI chemicals per disclosure was five
14  (see Section 5.4, Text Box 5-3). Additionally, as noted previously, approximately 35% of FracFocus
15  ingredient records were not able to be assigned standardized chemical names. These ingredient
16  records were excluded from the EPA's analysis (see Section 5.10).

### 9.2.2.  Chemicals Detected in Flowback and Produced Water

17  Of the 1,173 total chemicals, 134 were identified as having been detected in flowback or produced
18  water. Included among these chemicals are naturally occurring organic compounds, metals,
19  radionuclides, and pesticides. As reported in Chapter 7, concentration data in flowback or produced
20  water are available for 75 of these 134 chemicals (see Appendix E), including inorganic
21  contributors to salinity (Tables E-4 and E-5), metals (Tables E-6 and E-7), radioactive constituents
22  (Table E-8), and organic constituents (Tables E-9, E-10, and E-11). For these chemicals with
23  concentration data, the measured concentrations spanned several orders of magnitude. For
24  instance, for organic chemicals in produced water from the Marcellus shale formation (Table E-10),
25  average or median measured concentrations ranged from 2.7 µg/L for N-nitrosodiphenylamine to
26  400 µg/L for carbon disulfide. According to the sources listed in Appendix A, 37 of the total 134
27  chemicals in flowback and produced water were also identified as being used in hydraulic
28  fracturing fluid.

## 9.3.  Toxicological and Physicochemical Properties of Hydraulic Fracturing Chemicals

29  Toxicological and physicochemical data were collected as available for each of the chemicals
30  identified in Appendix A. The criteria used to identify and select toxicity values, RfVs and OSFs
31  (Section 9.3.1), and the method used to generate physicochemical property data (Section 9.3.2) are
32  discussed below. A summary of the available data for these chemicals follows in Section 9.3.3. Other

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[1] The list of 1,173 chemicals was finalized as of this 2015 draft assessment. There may be chemicals present in flowback and produced water that are not included on this list.
[2] The FracFocus frequency of use data presented in this chapter is based on 35,957 well disclosures.

1  possible sources of toxicological information, including QSAR-approaches and the EPA's ACToR
2  database, are discussed in Section 9.3.4.

### 9.3.1.   Selection of Toxicity Values: Reference Values (RfVs) and Oral Slope Factors (OSFs)

3  Toxicity information spans a wide range with respect to extent, quality and reliability. Toxicological
4  data may include assessments from various sources including state, national, international, private
5  and academic organizations as well as toxicity information which has not been formalized into an
6  assessment and may be found in the scientific literature and databases including results from
7  guideline tests, high throughput screening assays, alternative assays, and QSAR models. The
8  sources of toxicity values – specifically, chronic oral RfVs and OSFs – selected for the purposes of
9  this chapter are based on criteria developed specifically for this report. For many of the chemicals
10  used in hydraulic fracturing or found in flowback or produced water there may be relevant
11  information, including cancer and noncancer-related information, from one or more sources that
12  were not evaluated in this chapter.

13  The sources of RfVs and OSFs selected for the purposes of this chapter met the following key
14  criteria: 1) the body or organization generating or producing the peer-reviewed RfVs, peer-
15  reviewed OSFs, or peer-reviewed qualitative assessment must be a governmental or
16  intergovernmental body; 2) the data source must include peer-reviewed RfVs, peer-reviewed OSFs,
17  or peer reviewed qualitative assessments; 3) the RfVs, OSFs, or qualitative assessments must be
18  based on peer-reviewed scientific data; 4) the RfVs, OSFs, or qualitative assessments must be
19  focused on protection of the general public; and 5) the body generating the RfVs, OSFs, or
20  qualitative assessments must be free of conflicts of interest with respect to the chemicals for which
21  it derives reference values or qualitative assessments. More detail on these criteria for selection
22  and inclusion of data sources, as well as the full list of data sources that were considered for this
23  study, are available in Appendix G.

24  RfVs and OSFs available from the EPA IRIS, the EPA PPRTV program, ATSDR, and the EPA HHBP
25  program all met the criteria for selection and inclusion as a data source (see Table 9-1). An attempt
26  was made to identify and acquire RfVs and OSFs from all 50 states, but only the peer-reviewed state
27  RfVs and OSFs from California met the stringent selection criteria and were included because of the
28  state's extensive peer review process.[1] One international source for RfVs, the World Health
29  Organization's (WHO) International Programme on Chemical Safety (IPCS) Concise International
30  Chemical Assessment Documents (CICAD), also met the selection criteria. The International Agency
31  for Research on Cancer (IARC) and U.S. National Toxicology Program (NTP) Report on Carcinogens
32  also met the criteria and were used as additional sources for qualitative cancer classifications.

### Table 9-1. Sources of selected toxicityRfVs and OSFs.

| Source | Website |
|--------|---------|

[1] State RfVs and OSFs are also publicly available from Alabama, Texas, Hawaii, and Florida, but they did not meet the criteria for consideration as sources for RfVs and OSFs in this report. See Appendix G for details.

| Source | Website |
|---|---|
| EPA Integrated Risk Information System (IRIS) | http://cfpub.epa.gov/ncea/iris/index.cfm?fuseaction=iris.showSubstanceList |
| Human Health Benchmarks for Pesticides (HHBP) | http://iaspub.epa.gov/apex/pesticides/f?p=HHBP:home |
| EPA Provisional Peer-Reviewed Toxicity Value (PPRTV) database | http://hhpprtv.ornl.gov/index.html |
| Agency for Toxic Substances and Disease Registry (ATSDR) Minimum Risk Levels | http://www.atsdr.cdc.gov/toxprofiles/index.asp#bookmark05 |
| State of California Toxicity Criteria Database | http://www.oehha.org/tcdb/index.asp |
| International Programme on Chemical Safety (IPCS) Concise International Chemical Assessment Documents (CICAD) | http://www.who.int/ipcs/publications/cicad/en/ |

1  EPA generally applies federal RfVs and OSFs for use in human health risk assessments. Therefore,
2  for the purpose of hazard evaluation and making comparisons between chemicals in this chapter,
3  only federal chronic oral RfVs and OSFs from the EPA IRIS, the EPA PPRTV program, ATSDR, and
4  the EPA HHBP program were used. Furthermore, when a chemical had an RfV and/or OSF from
5  more than one federal source, a modification of the EPA Office of Solid Waste and Emergency
6  Response (OSWER) Directive 9285.7-53 tiered hierarchy of toxicity values was applied to
7  determine which value to use. A single RfV and/or OSF was selected from the sources in this order:
8  IRIS, HHBP, PPRTV, and ATSDR.[1] The RfVs considered from these sources included noncancer
9  reference doses (RfDs) from the IRIS, PPRTV, and HHBP programs, and oral minimum risk levels
10 (MRLs) from ATSDR.[2,3]

11 Because there are relatively few OSFs available compared to RfVs, OSFs were excluded from
12 discussion in this chapter; however, all available OSFs are reported in Appendix G. The EPA
13 drinking water maximum contaminant levels (MCLs) were also excluded from this analysis because
14 they are treatment-based. MCLs are set as close to maximum containment level goal (MCLG) values
15 as feasible. However, MCL and MCLGs values are still reported in Appendix G for the sake of
16 completeness.

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[1] The OSWER hierarchy indicates that sources should be used in this order: IRIS, PPRTV, and then other values. In this report, this hierarchy was followed, but HHBP values were used in lieu of an IRIS value for a few chemicals. See Appendix G for details.

[2] A RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. For the complete definition see Appendix G.

[3] An MRL is an estimate of daily human exposure to a hazardous substance at or below which the substance is unlikely to pose a measurable risk of harmful (adverse), noncancerous effects. MRLs are calculated for a route of exposure (inhalation or oral) over a specified time period (acute, intermediate, or chronic). MRLs should not be used as predictors of harmful (adverse) health effects. Chronic MRL: Duration of exposure is 365 days or longer.

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### 9.3.2. Physicochemical Properties

1  As presented in Chapter 5, EPI Suite™ software was used to generate data on the physicochemical
2  properties of chemicals on the consolidated list. EPI Suite™ provides an estimation of
3  physicochemical properties based upon chemical structure, and will additionally provide
4  empirically measured values for these properties when they are available for a given chemical. For
5  more details on the software and on the use of physicochemical properties for fate and transport
6  estimation, see Chapter 5.

### 9.3.3. Summary of Selected Toxicological and Physicochemical Property Data for Hydraulic Fracturing Chemicals

7  Figure 9-1 summarizes the availability of selected RfVs and OSFs and physicochemical data for the
8  1,173 hydraulic fracturing chemicals identified by the EPA.



**Figure 9-1. Overall representation of the selected RfVs and OSFs, occurrence data, and physicochemical data available for the 1,173 hydraulic fracturing chemicals identified by the EPA.**

9  Of the 1,173 chemicals identified by the EPA, only 147 (13%) have federal, or state, or international
10  chronic oral RfVs and/or OSFs from sources listed in Table 9-1. Therefore, chronic RfVs and/or
11  OSFs from the selected sources are lacking for 87% of chemicals that the EPA has identified as
12  associated with hydraulic fracturing. All available chronic RfVs and OSFs from the sources listed in
13  Table 9-1 are tabulated in Appendix G. Chronic RfVs and OSFs for chemicals used in hydraulic
14  fracturing fluids are listed in Tables G-1a through G-1c, and chronic RfVs and OSFs for chemicals

1   reported in hydraulic fracturing flowback and produced water are listed in Tables G-2a through G-
2   2c.

3   From the U.S. federal sources that were considered here, the availability of chronic RfVs and OSFs
4   can be summarized as follows. Of the 1,173 chemicals on the consolidated list, a total of
5   126 chemicals have federal chronic RfVs and/or OSFs. Of these 126 chemicals, 119 have federal
6   chronic RfVs, and 29 have federal OSFs (see Figure 9-1). 22 chemicals have both a federal chronic
7   RfV and a federal OSF, while 7 have a federal OSF only.

8   Overall, when chemicals in hydraulic fracturing fluid and chemicals in flowback are considered
9   separately, the availability of chronic RfVs and OSFs can be summarized as follows:

10  •   For the 1,076 chemicals used in hydraulic fracturing fluid, chronic RfVs from all of the
11      selected federal, state, and international sources were available for 90 chemicals (8.4%).
12      From the federal sources alone, chronic RfVs were available for 73 (6.8%), and OSFs were
13      available for 15 (1.4%).

14  •   For the 134 chemicals reported in flowback and produced water, chronic RfVs from all of
15      the selected federal, state, and international sources were available for 83 chemicals (62%).
16      From the federal sources alone, chronic RfVs were available for 70 chemicals (52%), and
17      OSFs were available for 20 (15%).

18  The IRIS database was the most abundant source of the federal chronic RfVs and OSFs. IRIS had
19  available RfDs for 77 of the total 1,173 chemicals, and OSFs for 27 chemicals. Of the other federal
20  data sources, the PPRTV database had RfDs for 33 chemicals, and OSFs for 2 chemicals; the HHBP
21  database had RfDs for 11 chemicals, but did not have available OSFs for any of the chemicals; and
22  the ATSDR database had chronic oral MRLs for 27 chemicals.

23  In addition to these chronic values, many of the chemicals also have less-than-chronic federal oral
24  RfVs. Subchronic or acute federal RfVs were identified for 91 chemicals on the consolidated list,
25  including 55 chemicals used in hydraulic fracturing fluid (Table G-1d), and 56 chemicals reported in
26  flowback or produced water (Table G-2d). There were 8 chemicals that had less-than-chronic RfVs
27  but lacked a chronic RfV. All of these less-than-chronic RfVs were found on the PPRTV or ATSDR
28  databases; the IRIS database did not have less-than-chronic RfVs for any of these chemicals. These
29  values are not discussed in this report, but are provided in Appendix G as supporting information.

30  From the total list of 1,173 chemicals associated with hydraulic fracturing, EPI Suite™ was able to
31  generate data on physicochemical properties for 515 (44%) of the chemicals (see Appendix A). The
32  remaining 658 chemicals lacked the structural information necessary to generate an estimate.

### 9.3.4.  Additional Sources of Toxicity Information

33  Because the majority of chemicals identified in this report do not have RfVs and/or OSFs from the
34  selected sources, it is likely that risk assessors at the local and regional level may turn to alternative
35  sources of toxicity information. This section discusses other publicly accessible sources of
36  toxicological data that are lower on the continuum of quality and reliability in comparison to the

1    selected  RfVs and OSFs described above. Because the quality of these data is unknown for most
2    chemicals, values from these data sources are not included in the hazard evaluation in this report.

### 9.3.4.1.  Estimated Toxicity Using Quantitative Structure Activity Relationships (QSAR)

3    One potential source of toxicological information is QSAR software, which is able to provide
4    estimates or predictions of toxicity based on chemical structure. QSAR models for toxicity have
5    been used and evaluated in a number of previous studies published in the peer reviewed literature
6    (Rupp et al., 2010; Venkatapathy et al., 2004; Moudgal et al., 2003). A key advantage to QSAR
7    models is that they are able to rapidly and inexpensively estimate toxicity values for chemicals.
8    Compared to toxicological studies involving animals or in vitro methods, which have monetary,
9    time, and ethical considerations associated with them, the QSAR method requires only information
10   on chemical structure in order to generate a toxicity estimation. These values may be of lower
11   quality and less reliable than values generated using traditional toxicological methods. However,
12   because they increase the available pool of toxicity information, QSAR estimates may potentially be
13   a useful resource for risk assessors that are faced with evaluating potential exposures to data-poor
14   chemicals.

### 9.3.4.2.  Chemical Data Available from ACToR

15   An additional tool for obtaining toxicological information is the ACToR database.[1] ACToR is a large
16   data warehouse developed by the EPA to gather and house large and disparate amounts of public
17   data on chemicals including chemical identity, structure, physicochemical properties, in vitro assay
18   results, and in vitro toxicology data (Judson et al., 2009). ACToR contains data on over 500,000
19   chemicals from over 2,500 sources, covering many domains including hazard, exposure, risk
20   assessment, risk management, and use. Data in ACToR is organized on several levels of "assays" and
21   "assay categories". The information available in ACToR ranges from the federal RfVs and OSFs
22   discussed in Section 9.3.1, which have undergone extensive peer review, to other toxicity values
23   and study and test results that have undergone little to no peer review.

24   The ACToR database was searched for information related to the total list of 1,173 chemicals
25   associated with hydraulic fracturing.[2] For the purposes of this chapter, the database was first
26   searched for all of the assays and assay categories that had data on these chemicals. This initial
27   search was then filtered to only include the assay categories that are most relevant to toxicity via
28   the oral route of exposure (drinking water). These assay categories were assigned into the
29   following nine data classes: carcinogenicity, dose response values, drinking water criteria,
30   genotoxicity/mutagenicity, hazard identification, LOAEL/NOAEL, RfD, slope factor, and water
31   quality criteria. The type of data and examples of the data sources included in these data classes can
32   be found in the ACToR database documentation.

33   When all assays and assay categories were considered, it was found that all but 28 of the total 1,173
34   chemicals had available data on ACToR. When only the relevant assays and assay categories were
35   considered, 642 (55%) of the chemicals were found to have data on ACToR. The fraction of

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[1] The ACToR database is available at: http://actor.epa.gov.
[2] The ACToR database was queried for the total list of 1,173 chemicals on April 1, 2015.

1  chemicals that had at least one data point in each of the nine ACToR data classes is shown in Figure
2  9-2. As can be seen in Figure 9-2, about half of the chemicals had some information on water quality
3  criteria, while fewer chemicals had information on the other classes of data.

**Fraction of Chemicals with Data by Class**
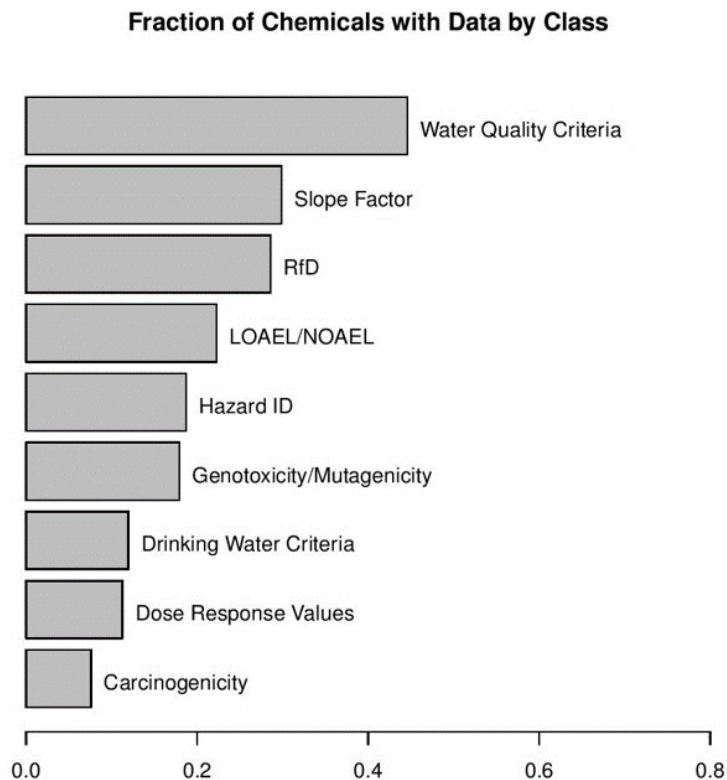


**Figure 9-2. Fraction of chemicals with at least one data point in each ACToR data class.**

4  Focusing on the 1,026 chemicals that lacked a chronic RfV and/or OSF from the selected sources
5  described in Section 9.3.1, 497 (48%) of these chemicals had available data on ACToR. Because
6  ACToR has a significant amount of data on potential chemical hazards, including for some data-poor
7  chemicals, ACToR might help to fill data gaps in the ongoing effort to understand potential hazards
8  of hydraulic fracturing chemicals. Since the quality of the non-peer reviewed values is not known,
9  these data are not considered in the hazard evaluation.

## 9.4.  Hazard Identification of Reported Hydraulic Fracturing Chemicals

10  This section focuses on chemicals that were identified as being of particular interest in previous
11  chapters of this report, or which otherwise may be of particular interest to risk assessors. Federal
12  RfVs are identified for these chemicals as available.

### 9.4.1.  Selection of Additional Chemicals for Hazard Identification

1  Four subsets of chemicals were identified as being of interest in Chapter 5 (Chemical Mixing) and
2  Chapter 7 (Flowback and Produced water):

3     1.  Chapter 5: The most frequently used chemicals in hydraulic fracturing fluid, defined as
4         chemicals being reported to the FracFocus database in at least 10% of well disclosures ([U.S.](#)
5         [EPA, 2015a](#)).

6     2.  Chapter 5: The top 20 most and least mobile chemicals from the EPA's analysis of the
7         FracFocus database ([U.S. EPA, 2015a](#)), as determined based on the octanol-water partition
8         coefficient ($K_{ow}$) from EPI Suite™.

9     3.  Chapter 7: Inorganic chemicals that may be returned to the surface in flowback and
10        produced water. This includes metals, inorganic ions, and naturally occurring radioactive
11        material (NORM).

12    4.  Pesticides occurring in flowback and produced water.

13 The hazard identification for these four subsets of chemicals is presented below.

### 9.4.2.  Hazard Identification Results

#### 9.4.2.1.  *Most Frequently Used Chemicals in Hydraulic Fracturing Fluid (FracFocus)*

14 Chapter 5 listed 35 chemicals that are reported to the FracFocus database in at least 10% of well
15 disclosures ([U.S. EPA, 2015a](#)) (Table 5-2). For 32 of these chemicals (water, quartz, and
16 sodium chloride were excluded from this analysis), only 7 chemicals (22%) have a federal chronic
17 RfV, as shown in Table 9-2. None of these 32 chemicals have available OSFs for cancer. For this
18 subset of chemicals, methanol was reported to be the most frequently used chemical in the
19 FracFocus analysis, followed by hydrotreated light petroleum distillates and hydrochloric acid, all
20 of which were reported in greater than 60% of disclosures. Ethylene glycol, isopropanol, and
21 peroxydisulfuric acid-diammonium salt are the only 3 additional chemicals to have been used in
22 greater than 40% of disclosures.

**Table 9-2. List of the most frequently used chemicals in hydraulic fracturing fluids, with their respective federal chronic RfVs where available.**

> Chemicals are ordered in the table, from high to low, based on their frequency of use from FracFocus. Includes all chemicals reported to FracFocus in at least 10% of well disclosures, excluding water, quartz, and sodium chloride.

| | | RfV | |
| | | Chronic RfD (mg/kg-day) | |
| Chemical | CASRN | | Source |
|---|---|---|---|
| **Methanol** | 67-56-1 | 2 | IRIS |
| **Distillates, petroleum, hydrotreated light** | 64742-47-8 | -- | -- |
| **Hydrochloric acid** | 7647-01-0 | -- | -- |

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| Chemical | CASRN | RfV Chronic RfD (mg/kg-day) | Source |
|---|---|---|---|
| Ethylene glycol | 107-21-1 | 2 | IRIS |
| Isopropanol | 67-63-0 | -- | |
| Peroxydisulfuric acid, diammonium salt | 7727-54-0 | -- | -- |
| Guar gum | 9000-30-0 | -- | -- |
| Sodium hydroxide | 1310-73-2 | -- | -- |
| Propargyl alcohol | 107-19-7 | 0.002 | IRIS |
| Glutaraldehyde | 111-30-8 | -- | -- |
| Ethanol | 64-17-5 | -- | -- |
| Potassium hydroxide | 1310-58-3 | -- | -- |
| Acetic acid | 64-19-7 | -- | -- |
| Citric acid | 77-92-9 | -- | -- |
| 2-Butoxyethanol | 111-76-2 | 0.1 | IRIS |
| Solvent naphtha, petroleum, heavy arom. | 64742-94-5 | -- | -- |
| Naphthalene | 91-20-3 | 0.02 | IRIS |
| 2,2-Dibromo-3-nitrilopropionamide | 10222-01-2 | -- | -- |
| Choline chloride | 67-48-1 | -- | -- |
| Phenolic resin | 9003-35-4 | -- | -- |
| Methenamine | 100-97-0 | -- | -- |
| Carbonic acid, dipotassium salt | 584-08-7 | -- | -- |
| 1,2,4-Trimethylbenzene | 95-63-6 | -- | -- |
| Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides | 68424-85-1 | -- | -- |
| Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy (mixture) | 127087-87-0 | -- | -- |
| Formic acid | 64-18-6 | 0.9 | PPRTV |
| Sodium chlorite | 7758-19-2 | 0.03 | IRIS |
| Nonyl phenol ethoxylate | 9016-45-9 | -- | -- |
| Tetrakis(hydroxymethyl)phosphonium sulfate | 55566-30-8 | -- | -- |
| Polyethylene glycol | 25322-68-3 | -- | -- |
| Ammonium chloride | 12125-02-9 | -- | -- |
| Sodium persulfate | 7775-27-1 | -- | -- |

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### 9.4.2.2. *Most and Least Mobile Chemicals Used in Hydraulic Fracturing Fluid (FracFocus)*

1  Chapter 5 lists the 20 most mobile chemicals (Table 5-7) and 20 least mobile chemicals (Table 5-8)
2  from the EPA's analysis of the FracFocus database (U.S. EPA, 2015a). For these lists, mobility was
3  determined based on $K_{ow}$. For the 20 most mobile chemicals, no federal chronic RfVs or OSFs were
4  available (see Table 9-3). Similarly, for the 20 least mobile chemicals, only one chemical—di(2-
5  ethylhexyl) phthalate—had a federal chronic RfV available (see Table 9-4).

**Table 9-3. List of the 20 most mobile chemicals used in hydraulic fracturing fluid, with their respective federal chronic RfVs where available.**

Chemicals are ordered in the table by lowest estimated log $K_{ow}$. None of these chemicals had federal chronic RfVs available.

| Chemical | CASRN | Log $K_{ow}$ (unitless) | RfV Chronic RfD (mg/kg-day) | Source |
|---|---|---|---|---|
| 1,2-Ethanediaminium, N,N'-bis[2-[bis(2-hydroxyethyl)methylammonio]ethyl]-N,N'-bis(2-hydroxyethyl)-N,N'-dimethyl-, tetrachloride | 138879-94-4 | -23.19 | -- | -- |
| Phosphonic acid, [[(phosphonomethyl)imino] bis[2,1-ethanediylnitrilobis (methylene)]]tetrakis- | 15827-60-8 | -9.72 | -- | -- |
| Phosphonic acid, [[(phosphonomethyl)imino] bis[2,1-ethanediylnitrilobis (methylene)]]tetrakis-, sodium salt | 22042-96-2 | -9.72 | -- | -- |
| Phosphonic acid, [[(phosphonomethyl)imino] bis[2,1-ethanediylnitrilobis (methylene)]]tetrakis-, ammonium salt (1:x) | 70714-66-8 | -9.72 | -- | -- |
| Phosphonic acid, (((2-[(2-hydroxyethyl)(phosphono methyl)amino)ethyl)imino]bis (methylene))bis-, compd. with 2-aminoethanol | 129828-36-0 | -6.73 | -- | -- |
| 2-Hydroxy-N,N-bis(2-hydroxyethyl)-N-methylethanaminium chloride | 7006-59-9 | -6.7 | -- | -- |
| N-(3-Chloroallyl)hexaminium chloride | 4080-31-3 | -5.92 | -- | -- |
| 3,5,7-Triazatricyclo(3.3.1.1(superscript 3,7))decane, 1-(3-chloro-2-propenyl)-, chloride, (Z)- | 51229-78-8 | -5.92 | -- | -- |
| (2,3-dihydroxypropyl)trimethylammonium chloride | 34004-36-9 | -5.8 | -- | -- |
| Phosphonic acid, [[(phosphonomethyl)imino] bis[6,1-hexanediylnitrilobis(methylene)]]tetrakis- | 34690-00-1 | -5.79 | -- | -- |

| Chemical | CASRN | Log $K_{ow}$ (unitless) | RfV Chronic RfD (mg/kg-day) | Source |
|---|---|---|---|---|
| [Nitrilotris(methylene)]tris-phosphonic acid pentasodium salt | 2235-43-0 | -5.45 | -- | -- |
| Aminotrimethylene phosphonic acid | 6419-19-8 | -5.45 | -- | -- |
| Choline chloride | 67-48-1 | -5.16 | -- | -- |
| Choline bicarbonate | 78-73-9 | -5.16 | -- | -- |
| alpha-Lactose monohydrate | 5989-81-1 | -5.12 | -- | -- |
| Lactose | 63-42-3 | -5.12 | -- | -- |
| Tetrakis(hydroxymethyl)phosphonium sulfate | 55566-30-8 | -5.03 | -- | -- |
| Disodium ethylenediaminediacetate | 38011-25-5 | -4.76 | -- | -- |
| Nitrilotriacetamide | 4862-18-4 | -4.75 | -- | -- |
| 1,3,5-Triazine-1,3,5(2H,4H,6H)-triethanol | 4719-04-4 | -4.67 | -- | -- |

**Table 9-4. List of the 20 least mobile chemicals used in hydraulic fracturing fluid, with their respective federal chronic RfVs where available.**

Chemicals are ordered in the table by highest estimated log $K_{ow}$.

| Chemical | CASRN | Log $K_{ow}$ (unitless) | RfV Chronic RfD (mg/kg-day) | Source |
|---|---|---|---|---|

| Chemical | CASRN | Log $K_{ow}$ (unitless) | RfV Chronic RfD (mg/kg-day) | Source |
|---|---|---|---|---|
| Sorbitan, tri-(9Z)-9-octadecenoate | 26266-58-0 | 22.56 | -- | -- |
| Fatty acids, C18-unsatd., dimers | 61788-89-4 | 14.6 | -- | -- |
| Sorbitan sesquioleate | 8007-43-0 | 14.32 | -- | -- |
| Tributyltetradecylphosphonium chloride | 81741-28-8 | 11.22 | -- | -- |
| Sodium bis(tridecyl) sulfobutanedioate | 2673-22-5 | 11.15 | -- | -- |
| 1-Eicosene | 3452-07-1 | 10.03 | -- | -- |
| D&C Red 28 | 18472-87-2 | 9.62 | -- | -- |
| C.I. Solvent Red 26 | 4477-79-6 | 9.27 | -- | -- |
| 1-Octadecene | 112-88-9 | 9.04 | -- | -- |
| Alkenes, C>10 .alpha.- | 64743-02-8 | 8.55 | -- | -- |
| Dioctyl phthalate | 117-84-0 | 8.54 | -- | -- |
| Benzene, C10-16-alkyl derivs. | 68648-87-3 | 8.43 | -- | -- |
| Di(2-ethylhexyl) phthalate | 117-81-7 | 8.39 | 0.02 | IRIS |
| 1-Octadecanamine, N,N-dimethyl- | 124-28-7 | 8.39 | -- | -- |
| N,N-dimethyloctadecylamine hydrochloride | 1613-17-8 | 8.39 | -- | -- |
| Butyryl trihexyl citrate | 82469-79-2 | 8.21 | -- | -- |
| 1-Hexadecene | 629-73-2 | 8.06 | -- | -- |
| Benzo(g,h,i)perylene | 191-24-2 | 7.98 | -- | -- |
| Dodecylbenzene | 123-01-3 | 7.94 | -- | -- |
| Isopropanolamine dodecylbenzene | 42504-46-1 | 7.94 | -- | -- |

### 9.4.2.3. Flowback and Produced Water: Inorganics and NORM

1  In addition to a number of volatile and semi-volatile organic compounds presented below, Chapter
2  7 also discusses the appearance of inorganic constituents such as metals, inorganic ions, and
3  naturally occurring radioactive material (NORM) in flowback and produced water. A number of
4  metals detected in flowback and produced water that appear on the EPA's consolidated list and are
5  noted in Chapter 7 have federal RfVs and/or OSFs listed in Appendix G (Table G-2). These metals
6  and inorganic ions include: iron, boron, chromium, zinc, arsenic, manganese, cadmium, and
7  strontium. These metals have oral RfVs based on a number of health effects including:
8  neurotoxicity, developmental and liver toxicity, hyperpigmentation and keratosis of the skin, and
9  decrements in blood copper status and enzyme activity. Chromium (VI) is classified as a known

1   human carcinogen by IARC and NTP, while arsenic is classified as known human carcinogen by the

2   EPA, IARC, and NTP. Radionuclides, such as radium-226, radium-228, and uranium-238, which are

3   naturally occurring in the formation may also return to the surface within produced water. Each of

4   these radionuclides is classified as a known human carcinogen by the EPA and IARC.

### 9.4.2.4.  *Flowback and Produced Water: Pesticides*

5   Lastly, it should be noted that a number of pesticides appear in the tables presented in Appendix G.

6   These chemicals were reported as having been detected in analyses of hydraulic fracturing

7   flowback and produced waters by several of the 10 sources cited in Appendix A; however, there is

8   much uncertainty about why they were detected. They could have migrated to the shale formation

9   or to the rock surrounding the shale formation, or they could have migrated into source waters

10  used by the hydraulic fracturing operation. It is also possible that these are laboratory

11  contaminants.

## 9.5.  Hazard Identification and Hazard Evaluation of Selected Subsets of Hydraulic Fracturing Chemicals

12  As described in Section 9.4, the majority of chemicals identified in the previous chapters of this

13  report do not have RfVs and/or OSFs from the sources meeting the criteria described in Section

14  9.3.1. This lack of data creates a challenge for hazard evaluation, because the potential human

15  health effects of these chemicals are difficult to determine. On the other hand, other chemicals

16  identified by the EPA have more data available, including chronic RfVs, data on occurrence, and

17  data on physicochemical properties. This section focuses on the hazard evaluation of these subsets

18  of chemicals that had data available.

19  When considering the hazard evaluation of chemicals in drinking water, it is important to

20  remember that toxicity is contingent upon exposure. All chemicals, including pure water, may be

21  toxic if they are ingested in large enough quantities. Therefore, in addition to data on health effects,

22  hazard evaluations must also consider data on potential chemical exposure. In the context of the

23  hazard evaluation presented in this section, chemical occurrence and physicochemical property

24  data were used as metrics to estimate the likelihood that the chemical could reach and impact

25  drinking water resources.

26  For the selected subsets of chemicals that had data available, this section discusses the known

27  toxicological properties based on selected RfVs (hazard identification), and then illustrates one

28  possible method for combining toxicity and exposure potential information for a more data-

29  informed hazard evaluation. Additionally, this section presents a summary of chemicals that have

30  occurrence data across multiple stages of the hydraulic fracturing water cycle.

### 9.5.1.  Selection of Chemicals for Hazard Evaluation

31  From the overall list of 1,173 chemicals identified in this assessment, subsets of chemicals were

32  selected for hazard evaluation if they met the following criteria:

33      1.  Had a federal chronic oral RfV;

1    2.  Had available data on frequency of use (in hydraulic fracturing fluids) or measured
2        concentrations (in flowback and produced water); and

3    3.  Had available data on physicochemical properties.

4  These criteria were selected for hazard evaluation for the following reasons:

5    1.  Federal RfVs generally undergo more extensive independent peer review compared to
6        other sources of RfVs. Additionally, as described above, there are many more chemicals
7        with federal chronic RfVs than chemicals with federal OSFs. Therefore, although OSFs are
8        discussed in the hazard evaluation, chronic RfVs were selected for illustrative purposes of
9        making comparisons between chemicals.

10   2.  Data on frequency of use (in hydraulic fracturing fluids) or measured concentration (in
11       flowback or produced water) provide a metric to help assess the likelihood of chemical
12       occurrence in the hydraulic fracturing water cycle. Chemicals that are used more
13       frequently in hydraulic fracturing fluid have a greater likelihood of accidental release or
14       dissemination due to the fact that they are present at a greater number of wells
15       nationwide. Likewise, chemicals that occur at higher concentrations in flowback or
16       produced water may result in greater exposures. Frequency of detection in flowback or
17       produced water would also be a useful metric for this evaluation, but this information was
18       not available for these chemicals.

19   3.  Information on physicochemical properties enables the estimation of chemical persistence
20       and mobility in the environment. This is discussed in more detail in Section 9.5.2 below.

21 For chemicals that are used in hydraulic fracturing fluids, the FracFocus database was the only
22 source with reliable information on the frequency of use (U.S. EPA, 2015a). For chemicals found in
23 flowback or produced water, data on measured concentration were only available for the 75
24 chemicals presented in Appendix E. Therefore, hazard evaluations were only conducted on
25 chemicals included in these two data sources. While the other data sources listed in Appendix A
26 provide useful information on the diversity of chemicals that may occur in the hydraulic fracturing
27 water cycle, hazard evaluation could not be conducted on these sources in the absence of data on
28 frequency of use or measured concentration.

29 Overall, 37 chemicals used in hydraulic fracturing fluid and 23 chemicals detected in flowback and
30 produced water met the selection criteria for hazard evaluation (see Figure 9-3).
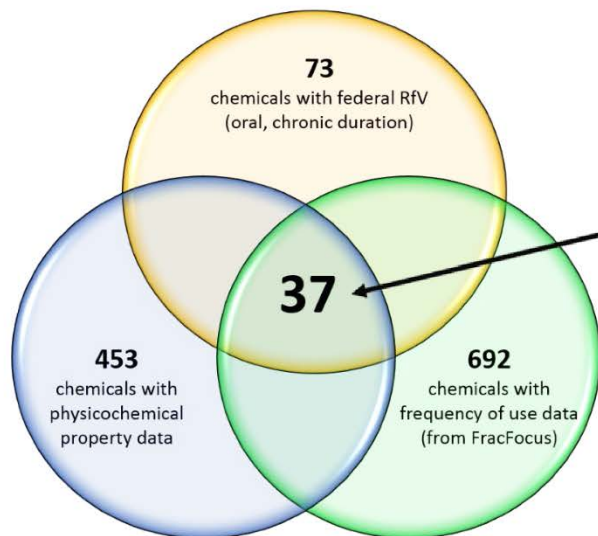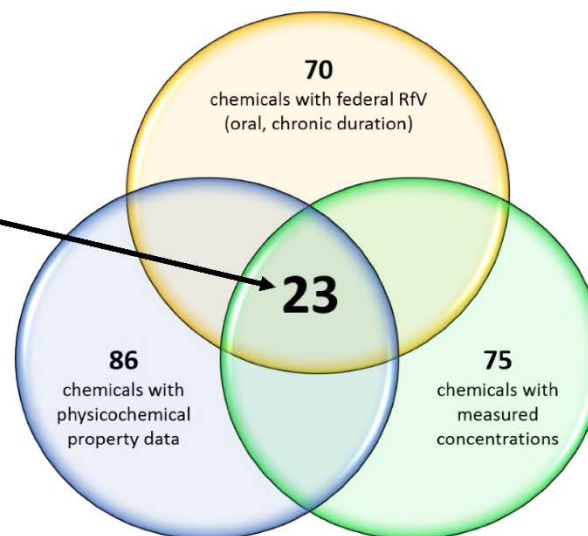
**Figure 9-3. The two subsets of chemicals selected for hazard evaluation included 37 chemicals used in hydraulic fracturing fluid, and 23 chemicals detected in flowback or produced water.**

### 9.5.2. Multi Criteria Decision Analysis (MCDA) Framework for Hazard Evaluation: Integrating Toxicity, Occurrence, and Physicochemical Data

1  Integration or combining of various types of data may provide insights on those chemicals that may
2  be of greater concern than other chemicals to drinking water resources. For the purpose of this
3  chapter, a structured but flexible Multi Criteria Decision Analysis (MCDA) approach was developed
4  to integrate factors related to hydraulic fracturing such as chemical toxicity, occurrence, and
5  physicochemical data. The approach described here is for illustrative purposes only, in order to
6  demonstrate how combining of information may be informative. Alternative frameworks may be
7  considered by risk assessors for similar analyses.

8  In this illustration, a MCDA framework was developed and applied to each list of chemicals
9  identified in Section 9.5.1 and depicted in Figure 9-3 (37 chemicals used in hydraulic fracturing
10 fluids, and 23 chemicals detected in flowback or produced water). The MCDA framework serves to
11 place the toxicity of these chemicals in the context of factors that may increase the likelihood of
12 impacting drinking water resources. In essence, this analysis serves to illustrate the circumstances
13 under which drinking water resources may be affected.

14 MCDA is a well-established analysis tool that is used to transparently integrate multiple lines of
15 evidence to support decision-making. For example, MCDA has been adapted as a method of
16 selecting an optimal cleanup plan for a contaminated site (Linkov et al., 2011), and as a method of
17 integrating chemical hazard data across multiple studies (Hristozov et al., 2014). The MCDA
18 framework employed here is based on the method by Mitchell et al. (2013b), who developed a
19 protocol for ranking chemical exposure potential by integrating data on physicochemical properties
20 and commercial use. This method is similar to approaches used by the petroleum industry to
21 quantitatively rank the potential hazards of hydraulic fracturing chemicals (see Section 5.9).
22 Moreover, the underlying philosophy of this approach is similar to that of the EPA's Design for the
23 Environment (DfE) Program. The DfE's Alternatives Assessment Criteria for Hazard Evaluation (U.S.
24 EPA, 2011a) was developed as a tool for evaluating and differentiating among chemical hazards
25 based on toxicity and physicochemical properties. Recently, this criteria and framework have been
26 applied in the Alternatives Assessment for the Flame Retardant Decabromodiphenyl Ether
27 (DecaBDE) and Flame Retardant Alternatives for Hexabromocyclododecane (HBCD) (U.S. EPA,
28 2014a, d). Aspects of MCDA methods and the DfE's Program for Alternatives Assessment are
29 evident in the National Research Council (NRC)'s "A Framework to Guide Selection of Chemical
30 Alternatives" document (NRC, 2014).

31 The methodology used to illustrate a hazard evaluation MCDA for hydraulic fracturing is outlined
32 below, and schematic of the model is shown in Figure 9-4. Under the MCDA framework, each
33 chemical was assigned three scores:

34      1.  A toxicity score;

35      2.  An occurrence score; and

36      3.  A physicochemical properties score.

1  The three normalized scores were summed to develop a total composite hazard potential score for
2  each chemical. These scores serve as a relative ranking and a means of making comparisons across
3  chemicals. These scores are not intended to define whether or not a chemical will present a human
4  health hazard, or indicate that one chemical is safer than another. Rather, the scores serve as a
5  qualitative metric to identify chemicals that may be more likely to present an impact to drinking
6  water resources, given available data on chemical properties and occurrence.
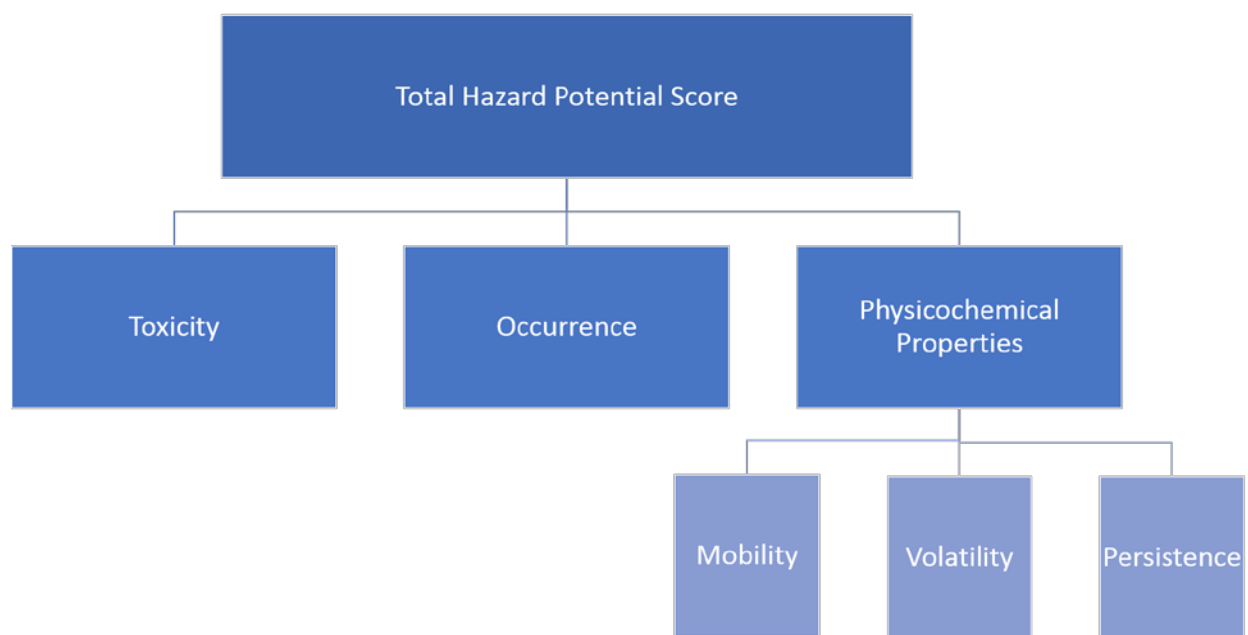


**Figure 9-4. Overview of the MCDA framework applied to the hazard evaluations.**

### 9.5.2.1.  Toxicity Score

7  The toxicity score was based upon the federal chronic RfV, which was determined from peer
8  reviewed sources as described in Section 9.3.1. Within each dataset (chemicals used in hydraulic
9  fracturing fluids, or chemicals detected in flowback and produced water), toxicity was ranked based
10  on quartiles, with each chemical assigned a toxicity score of 1 to 4 (see thresholds outlined in Table
11  9-5). Note that chemicals in the lowest quartile received the highest toxicity score as these
12  chemicals have lower RfVs than for other chemicals.

### 9.5.2.2.  Occurrence Score

13  This score was based on the frequency or concentration at which chemicals were reported within
14  the hydraulic fracturing water cycle. For chemicals used in hydraulic fracturing fluids, the
15  occurrence score was based on the nationwide number of well disclosures for each chemical from
16  the FracFocus database. For chemicals that were detected in hydraulic fracturing flowback and
17  produced water, the occurrence score was based on the average or median measured concentration
18  reported in Appendix E. If the measured concentration of a chemical was reported by multiple

1 studies in Appendix E, the highest of these reported average or median concentrations was used for
2 this calculation. Note that these two metrics of chemical occurrence—frequency of use, and
3 concentration—cannot be directly compared to one another. Therefore, FracFocus chemicals and
4 flowback and produced water chemicals were considered separately for this MCDA hazard
5 evaluation. Within each dataset (chemicals used in hydraulic fracturing fluids, or chemicals
6 detected in flowback and produced water), chemical occurrence was ranked based on quartiles,
7 with each chemical assigned an occurrence score of 1 to 4, as shown in Table 9-5.

### 9.5.2.3.　Physicochemical Properties Score

8 This score was based upon inherent physicochemical properties which affect the likelihood that a
9 chemical will reach and impact drinking water resources. The thresholds chosen for ranking
10 physicochemical properties, shown in Table 9-5, are based on previously published thresholds used
11 in the DfE Alternatives Assessment Criteria for Hazard Evaluation (U.S. EPA, 2011a), the EPA Office
12 of Pollution Prevention and Toxics Pollution Prevention (P2) Framework (U.S. EPA, 2005), and
13 Mitchell et al. (2013b). When refining EPI Suite™ physicochemical properties data for input into
14 this MCDA, empirically measured values were always used when available. If multiple estimated
15 values were available, the most conservative value (i.e., the value resulting in the highest score
16 according to Table 9-5) was used.

17 The total physicochemical properties score for each chemical was based upon three subcriteria:
18 mobility in water, volatility, and persistence. Chemical mobility in water was assessed based upon
19 three physicochemical properties: the octanol-water partition coefficient ($K_{ow}$), the organic carbon-
20 water partition coefficient ($K_{oc}$), and aqueous solubility. Chemical volatility was assessed based on
21 the Henry's law constant, which describes partitioning of a chemical between water and air.
22 Chemical persistence was assessed based on estimated half-life in water, which describes how long
23 a chemical will persist in water before it is transformed or degraded. Details on the evaluation and
24 physicochemical score calculation are provided in the Chapter Annex, Section 9.8.1. For each
25 chemical, the mobility score, volatility score, and persistence score (each on a scale of 1 to 4) were
26 summed to calculate a total physicochemical score.

### 9.5.2.4.　Final MCDA Score Calculations

27 Each raw score (toxicity, occurrence, or physicochemical properties), calculated as described above,
28 was standardized by scaling to the highest and lowest raw score within the set of chemicals. The
29 following equation was used:

30 $$S_{x\_final} = (S_x - S_{min}) / (S_{max} - S_{min})$$

31 in which $S_x$ is the raw score for a particular chemical x, $S_{max}$ is the highest observed raw score within
32 the set of chemicals, and $S_{min}$ is the lowest observed raw score within the set of chemicals. $S_{x\_final}$ is
33 the standardized score for chemical x. Each standardized score (toxicity, occurrence, or
34 physicochemical properties) falls on a scale of 0 to 1. These standardized toxicity, occurrence, and
35 physicochemical properties scores were summed to calculate a total hazard potential score for each
36 chemical. The total hazard potential scores fell on a scale of 0 to 3, with higher scores indicating

1    chemicals that are predicted to be more likely to affect drinking water resources. An example of
2    MCDA score calculation can be found in the Annex, Section 9.8.2.

3    In the MCDA approach illustrated in this chapter, each factor (toxicity, occurrence, physicochemical
4    properties) was given equal weight in the calculation of the final hazard potential score. This was
5    done in order to prevent subjectivity and avoid biasing the results based on any individual variable
6    that was considered in this analysis. This approach is adaptable, however. Risk assessors may
7    choose to apply alternative weights that place more or less emphasis on the various factors being
8    considered, in order to reflect expert judgement of a variable's relative importance. This MCDA
9    approach may also be adapted to include other variables of interest, such as carcinogenic potential,
10   which were not considered in the MCDA approach illustrated in this chapter.

**Table 9-5. Thresholds used for developing the toxicity score, occurrence score, and physicochemical properties score in this MCDA framework.**

| Score | 1 | 2 | 3 | 4 |
|---|---|---|---|---|
| *Toxicity Score* | | | | |
| **Chronic RfV (federal)** | >3rd quartile | >2nd quartile to ≤3rd quartile | >1st quartile to ≤2nd quartile | ≤1st quartile |
| *Occurrence Score* | | | | |
| **Percentage of wells nationwide** | <1st quartile | ≥1st quartile to <2nd quartile | ≥2nd quartile to <3rd quartile | ≥3rd quartile |
| **Concentration (μg/L)** | <1st quartile | ≥1st quartile to <2nd quartile | ≥2nd quartile to <3rd quartile | ≥3rd quartile |
| *Physicochemical Properties Score* | | | | |
| *Mobility score:* | | | | |
| **Log $K_{OW}$** | >5 | >3 to 5 | >2 to 3 | ≤2 |
| **Log $K_{OC}$** | >4.4 | >3.4 to 4.4 | >2.4 to 3.4 | ≤2.4 |
| **Aqueous solubility (mg/L)** | <0.1 | ≥0.1 to <100 | ≥100 to <1000 | ≥1000 |
| *Volatility score:* | | | | |
| **Henry's law constant** | >$10^{-1}$ | >$10^{-3}$ to ≤$10^{-1}$ | >$10^{-5}$ to ≤$10^{-3}$ | ≤$10^{-5}$ |
| *Persistence score:* | | | | |
| **Half-life in water (days)** | <16 | ≥16 to <60 | ≥60 to <180 | ≥180 |

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### 9.5.3. Hazard Evaluation Results

1  Discussed below are the results of the hazard evaluations for each subset of chemicals identified in
2  Section 9.5.1. For each subset of chemicals selected for hazard evaluation, the information
3  presented includes: the available federal chronic oral RfV (hazard identification), followed by
4  highlights of MCDA analyses (hazard evaluation).

5  For this MCDA illustration, the calculated toxicity scores, occurrence scores, physicochemical
6  properties scores, and total hazard potential scores are provided for chemicals used in hydraulic
7  fracturing fluids and chemicals detected in flowback/produced water, respectively. These
8  individual scores make it possible to visualize the extent to which the total hazard potential ranking
9  of each chemical is driven by each of the variables considered in the MCDA.

#### 9.5.3.1. Hazard Identification: Chemical Used in Hydraulic Fracturing Fluid

10  As discussed above, a total of 37 chemicals used in hydraulic fracturing fluids were identified for
11  hazard evaluation using the selection criteria described in Section 9.5.1. Some of the chemicals
12  represented include the BTEX chemicals (benzene, toluene, ethylbenzene, xylenes) as well as
13  naphthalene, acrylamide, phenol, 1,2-propylene glycol, ethylene glycol, 2-butoxyethanol, ethyl
14  acetate, and methanol.

15  These chemicals along with their primary noncancer toxicological properties, including the point-
16  of-departure (POD), total product of uncertainty factors applied, the federal chronic RfV, and the
17  health effect basis for the RfV, are shown in Table 9-6.[1,2] As seen in Table 9-6, all of these chemicals
18  had RfDs available from IRIS, PPRTV, or HHBP. These chemicals induce a variety of adverse
19  outcomes including immune system effects, changes in body weight, changes in blood chemistry,
20  cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and developmental
21  toxicity. The RfD values within this suite of chemicals range from 0.001–20 mg/kg-day, with (E)-
22  crotonaldehyde having the lowest RfD (0.001 mg/kg-day) and 1,2-propylene glycol having the
23  highest (20 mg/kg-day).

24  Comparison of RfVs among a set of chemicals requires a more thorough examination. For instance,
25  to derive the final chronic RfD for a given chemical, a number of UFs may be applied to the POD.
26  Briefly, UFs are applied to account for 5 areas of uncertainty: 1) intraspecies variability; 2)

---

[1] The point-of-departure (POD) is the dose-response point that marks the beginning of a low-dose extrapolation. This point can be the lower bound on dose for an estimated incidence or a change in response level from a dose-response model or a NOAEL or LOAEL for an observed incidence, or change in level of response. See http://www.epa.gov/iris/ for more information.

[2] An uncertainty factor is one of several (generally 10-fold) default factors used in operationally deriving the RfV from experimental data. The factors are intended to account for (1) variation in susceptibility among the members of the human population (i.e., inter-individual or intraspecies variability); (2) uncertainty in extrapolating animal data to humans (i.e., interspecies uncertainty); (3) uncertainty in extrapolating from data obtained in a study with less-than-lifetime exposure (i.e., extrapolating from subchronic to chronic exposure); (4) uncertainty in extrapolating from a LOAEL rather than from a NOAEL; and (5) uncertainty associated with extrapolation when the database is incomplete. See the IRIS Glossary at: http://www.epa.gov/iris/ for more information.

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1  interspecies uncertainty; 3) extrapolation from a subchronic study; 4) extrapolating from a NOAEL;
2  and 5) an incomplete database. A UF of 1, 3 ($10^{0.5}$), or 10 can be applied for any of these areas of
3  uncertainty depending upon the amount and/or type data available. The maximum total UF that
4  can be applied is 3,000; RfDs are not derived for chemicals that invoke the application of a total UF
5  >3,000 or involves the application of the full 10-fold UF in four or more areas of uncertainty (U.S.
6  EPA, 2002a). Therefore, those chemicals with a lower total uncertainty factor generally have more
7  reliable and robust health effect information. For example, although (E)-crotonaldehyde has the
8  lowest RfD, chemicals such as acrylamide, benzene, and dichloromethane have RfDs within a factor
9  of 10 (0.002–0.006 mg/kg-day) but with much less uncertainty reflected in their values. All three
10  latter chemicals have large data sets with reproducible effects, and dose estimated based on
11  physiologically based pharmacokinetic models (for acrylamide and dichloromethane) or have
12  available human health effect data (for benzene). Thus, a chemical with a low RfD may reflect high
13  uncertainty in the value and not necessarily be the most toxic.

14  Although only federal RfVs are considered in this hazard evaluation, eight of these chemicals also
15  have federal OSFs. These include acrylamide, benzyl chloride, 1,4-dioxane, 1,3-dichloropropene,
16  benzene, epichlorohydrin, aniline, and dichloromethane. Of these chemicals, acrylamide is the most
17  potent carcinogen. Acrylamide has an OSF of 0.5 per mg/kg-day and is classified as a likely human
18  carcinogen in IRIS (U.S. EPA, 2010). Benzene is the only chemical listed as a known human
19  carcinogen and has a calculated OSF of 0.015 mg/kg-day (U.S. EPA, 2002b). The OSF values for each
20  of these chemicals can be found in Appendix G.

**Table 9-6. Toxicological properties of the 37 chemicals used in hydraulic fracturing fluid that were identified for hazard evaluation and MCDA analysis.**

Chemicals are ranked, from low to high, based on their respective federal chronic RfVs.

| Chemical | CASRN | Point of departure (mg/kg-day) | Total uncertainty factor | RfV Chronic RfD (mg/kg-day) | Noncancer effect | Source |
|---|---|---|---|---|---|---|
| **(E)-Crotonaldehyde** | 123-73-9 | 3.4 | 3000 | 0.001 | Forestomach lesions | PPRTV |
| **Benzyl chloride** | 100-44-7 | 6.4 | 3000 | 0.002 | Cardiotoxicity | PPRTV |
| **Propargyl alcohol** | 107-19-7 | 5 | 3000 | 0.002 | Renal and hepatotoxicity | IRIS |
| **Acrylamide** | 79-06-1 | 0.053 | 30 | 0.002 | Degenerative nerve changes | IRIS |
| **Benzene** | 71-43-2 | 1.2 | 300 | 0.004 | Decreased lymphocyte count in humans | IRIS |
| **Epichlorohydrin** | 106-89-8 | 6.25 | 1000 | 0.006 | Decreased fertility | PPRTV |

| Chemical | CASRN | Point of departure (mg/kg-day) | Total uncertainty factor | RfV Chronic RfD (mg/kg-day) | Noncancer effect | Source |
|---|---|---|---|---|---|---|
| **Dichloromethane** | 75-09-2 | 0.19 | 30 | 0.006 | Hepatic effects | IRIS |
| **Aniline** | 62-53-3 | 7 | 1000 | 0.007 | Splenic effects | PPRTV |
| **2-(Thiocyano methylthio)benzo thiazole** | 21564-17-0 | 3.8 | 300 | 0.01 | Decreased body weight gain; decreased white blood cells (WBC) and plasma alanine aminotransferase (ALT) | HHBP |
| **Furfural** | 98-01-1 | 30 | 3000 | 0.01 | Liver pathology | HHBP |
| **Naphthalene** | 91-20-3 | 71 | 3000 | 0.02 | Decreased mean terminal body weight > 10% | IRIS |
| **2-(2-Butoxyethoxy) ethanol** | 112-34-5 | 81 | 3000 | 0.03 | Changes in red blood cells (RBC) | PPRTV |
| **1,4-Dioxane** | 123-91-1 | 9.6 | 300 | 0.03 | Liver and kidney toxicity | IRIS |
| **Bisphenol A** | 80-05-7 | 50 | 1000 | 0.05 | Reduced mean body weight | IRIS |
| **1,3-Dichloropropene** | 542-75-6 | 3.4 | 100 | 0.03 | Chronic irritation | IRIS |
| **Toluene** | 108-88-3 | 238 | 3000 | 0.08 | Increased absolute kidney weight | IRIS |
| **Ethylenediamine** | 107-15-3 | 9 | 100 | 0.09 | Liver and kidney toxicity | PPRTV |
| **Ethylbenzene** | 100-41-4 | 97.1 | 1000 | 0.1 | Liver and kidney toxicity; histopathology | IRIS |
| **2-Butoxyethanol (EGBE)** | 111-76-2 | 1.4 | 10 | 0.1 | Hemosiderin deposition in liver (inhalation study) | IRIS |
| **Acetophenone** | 98-86-2 | 423 | 3000 | 0.1 | General toxicity; NO LOAEL identified | IRIS |
| **Didecyldimethyl ammonium chloride** | 7173-51-5 | 10 | 100 | 0.1 | Clinical signs; decreased total cholesterol levels | HHBP |

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| Chemical | CASRN | Point of departure (mg/kg-day) | Total uncertainty factor | RfV Chronic RfD (mg/kg-day) | Noncancer effect | Source |
|---|---|---|---|---|---|---|
| **Cumene** | 98-82-8 | 110 | 1000 | 0.1 | Increased average kidney weight in female rats | IRIS |
| **N,N-Dimethylform amide** | 68-12-2 | 96 | 1000 | 0.1 | Increase in ALT and liver weight | PPRTV |
| **1-Butanol** | 71-36-3 | 125 | 1000 | 0.1 | Hypoactivity and ataxia | IRIS |
| **Xylenes** | 1330-20-7 | 179 | 1000 | 0.2 | Decreased body weight; increased mortality | IRIS |
| **Formaldehyde** | 50-00-0 | 15 | 100 | 0.2 | Decreased weight gain | IRIS |
| **Phenol** | 108-95-2 | 93 | 300 | 0.3 | Decreased maternal weight gain; developmental toxicity | IRIS |
| **2-Methyl-1-propanol (Isobutanol)** | 78-83-1 | 316 | 1000 | 0.3 | Hypoactivity and ataxia | IRIS |
| **Acetone** | 67-64-1 | 900 | 1000 | 0.9 | Nephropathy | IRIS |
| **Ethyl acetate** | 141-78-6 | 900 | 1000 | 0.9 | Mortality and body weight loss | IRIS |
| **Formic acid** | 64-18-6 | 277 | 300 | 0.9 | Reproductive effects | PPRTV |
| **Dodecylbenzenesul fonic acid** | 27176-87-0 | 50 | 100 | 0.5 | Decreased pup weight; kidney pathology | HHBP |
| **Ethylene glycol** | 107-21-1 | 200 | 100 | 2 | Kidney toxicity; chronic nephritis | IRIS |
| **Hexanedioic acid** | 124-04-9 | 470 | 300 | 2 | Decreased body weight | PPRTV |
| **Methanol** | 67-56-1 | 43.1 mg/L[a] | 100 | 2 | Extra cervical ribs; developmental toxicity | IRIS |
| **Benzoic acid** | 65-85-0 | 4.4 | 1 | 4 | No adverse effects observed in humans | IRIS |
| **1,2-Propylene glycol** | 57-55-6 | 5200 | 300 | 20 | Reduced red blood cell counts and hyperglycemia | PPRTV |

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| Chemical | CASRN | Point of departure (mg/kg-day) | Total uncertainty factor | RfV | Noncancer effect | Source |
|---|---|---|---|---|---|---|
| | | | | Chronic RfD (mg/kg-day) | | |

[a] POD based on internal methanol blood concentration using a PBPK model.

### 9.5.3.2. MCDA Results: Chemical Used in Hydraulic Fracturing Fluid

1   The hazard potential scores of the selected 37 chemicals used in hydraulic fracturing fluid are
2   presented in Table 9-7.

**Table 9-7. MCDA results for 37 chemicals used in hydraulic fracturing fluid.**

Chemicals are ranked, from high to low, based on total hazard potential score. See section 9.5.2 for details on the calculation.

| Chemical | CASRN | Physicochemical properties score | Occurrence score | Toxicity score | Total hazard potential score |
|---|---|---|---|---|---|
| Propargyl alcohol | 107-19-7 | 1.00 | 1.00 | 1.00 | **3.00** |
| 2-Butoxyethanol (EGBE) | 111-76-2 | 1.00 | 1.00 | 0.67 | **2.67** |
| N,N-Dimethylformamide | 68-12-2 | 1.00 | 1.00 | 0.67 | **2.67** |
| Acrylamide | 79-06-1 | 1.00 | 0.67 | 1.00 | **2.67** |
| Formaldehyde | 50-00-0 | 1.00 | 1.00 | 0.33 | **2.33** |
| Naphthalene | 91-20-3 | 0.67 | 1.00 | 0.67 | **2.33** |
| Benzyl chloride | 100-44-7 | 0.67 | 0.67 | 1.00 | **2.33** |
| 1-Butanol | 71-36-3 | 1.00 | 0.67 | 0.67 | **2.33** |
| Epichlorohydrin | 106-89-8 | 0.67 | 0.67 | 1.00 | **2.33** |
| 2-(2-Butoxyethoxy)ethanol | 112-34-5 | 1.00 | 0.67 | 0.67 | **2.33** |
| Methanol | 67-56-1 | 1.00 | 1.00 | 0.00 | **2.00** |
| Ethylene glycol | 107-21-1 | 1.00 | 1.00 | 0.00 | **2.00** |
| Formic acid | 64-18-6 | 1.00 | 1.00 | 0.00 | **2.00** |
| Didecyldimethylammonium chloride | 7173-51-5 | 0.33 | 1.00 | 0.67 | **2.00** |
| 1,4-Dioxane | 123-91-1 | 1.00 | 0.33 | 0.67 | **2.00** |
| (E)-Crotonaldehyde | 123-73-9 | 0.67 | 0.33 | 1.00 | **2.00** |
| Aniline | 62-53-3 | 1.00 | 0.00 | 1.00 | **2.00** |
| Furfural | 98-01-1 | 1.00 | 0.00 | 1.00 | **2.00** |

| Chemical | CASRN | Physicochemical properties score | Occurrence score | Toxicity score | Total hazard potential score |
|---|---|---|---|---|---|
| 1,2-Propylene glycol | 57-55-6 | 1.00 | 0.67 | 0.00 | **1.67** |
| Hexanedioic acid | 124-04-9 | 1.00 | 0.67 | 0.00 | **1.67** |
| Toluene | 108-88-3 | 0.33 | 0.67 | 0.67 | **1.67** |
| Phenol | 108-95-2 | 1.00 | 0.33 | 0.33 | **1.67** |
| 2-Methyl-1-propanol (Isobutanol) | 78-83-1 | 1.00 | 0.33 | 0.33 | **1.67** |
| Dichloromethane | 75-09-2 | 0.67 | 0.00 | 1.00 | **1.67** |
| Ethylenediamine | 107-15-3 | 1.00 | 0.00 | 0.67 | **1.67** |
| Bisphenol A | 80-05-7 | 1.00 | 0.00 | 0.67 | **1.67** |
| 2-(Thiocyanomethylthio) benzothiazole | 21564-17-0 | 0.67 | 0.00 | 1.00 | **1.67** |
| Benzene | 71-43-2 | 0.67 | 0.00 | 1.00 | **1.67** |
| Dodecylbenzenesulfonic acid | 27176-87-0 | 0.33 | 1.00 | 0.00 | **1.33** |
| Xylenes | 1330-20-7 | 0.33 | 0.67 | 0.33 | **1.33** |
| Ethylbenzene | 100-41-4 | 0.33 | 0.33 | 0.67 | **1.33** |
| Benzoic acid | 65-85-0 | 1.00 | 0.33 | 0.00 | **1.33** |
| Acetophenone | 98-86-2 | 0.67 | 0.00 | 0.67 | **1.33** |
| 1,3-Dichloropropene | 542-75-6 | 0.67 | 0.00 | 0.67 | **1.33** |
| Cumene | 98-82-8 | 0.00 | 0.33 | 0.67 | **1.00** |
| Ethyl acetate | 141-78-6 | 0.67 | 0.33 | 0.00 | **1.00** |
| Acetone | 67-64-1 | 0.67 | 0.33 | 0.00 | **1.00** |

1  Of the chemicals in hydraulic fracturing fluid that were considered in this hazard evaluation,
2  propargyl alcohol received the highest overall hazard potential score. Propargyl alcohol was used in
3  33% of wells in the FracFocus database, making it one of the most widely used chemicals that was
4  considered in this analysis, and it also had one of the lowest RfVs, with an RfD of 0.002 mg/kg-day.
5  It is also hydrophilic and has relatively low volatility, indicating that it is likely to be readily
6  transported in water. Given these properties, propargyl alcohol received the highest overall ranking
7  across all of the metrics that were considered in the hazard evaluation.

8  The other chemicals that fell in the upper quartile in terms of frequency of use received lower
9  hazard potential scores relative to propargyl alcohol, due to lower estimated toxicity and/or
10  physicochemical properties that are less conducive to transport in water. Naphthalene, used in 19%
11  of wells on the FracFocus database, has an RfD of 0.02 mg/kg-day, and is expected to have
12  somewhat lower transport in water relative to other chemicals because it is moderately
13  hydrophobic and moderately volatile. Methanol (RfD of 2 mg/kg-day), ethylene glycol (RfD of 2

1  mg/kg-day), 2-butoxyethanol (RfD of 0.1 mg/kg-day), formic acid (RfD of 0.9 mg/kg-day), N,N-
2  dimethylformamide (RfD of 0.1 mg/kg-day), and formaldehyde (RfD of 0.2 mg/kg-day)—which
3  were used in 73%, 47%, 23%, 11%, 9%, and 7% of wells in the FracFocus database, respectively—
4  are all expected to be highly mobile in water and have low volatility, but have higher RfVs
5  compared to many of the other chemicals in the assessment. Didecyldimethylammonium chloride
6  (RfD of 0.1 mg/kg-day), used in 8% of wells, is expected to have reduced mobility in water due to its
7  more hydrophobic properties.

8  In addition to propargyl alcohol, the other most toxic chemicals (occurring in the lowest quartile of
9  RfVs) received moderate to high hazard potential scores overall. Acrylamide (RfD of 0.002 mg/kg-
10 day) is used in only 1% of wells, but has physicochemical properties that are very conducive to
11 transport in water, and therefore received one of the highest overall hazard potential scores. Benzyl
12 chloride (RfD of 0.002 mg/kg-day) and epichlorohydrin (RfD of 0.006 mg/kg-day) are used in 6%
13 and 1% of wells, respectively, but scored slightly lower than acrylamide with regards to their
14 physicochemical properties. Other chemicals, including (E)-crotonaldehyde (RfD of 0.001 mg/kg-
15 day), benzene (RfD of 0.004 mg/kg-day), dichloromethane (RfD of 0.006 mg/kg-day), aniline (RfD
16 of 0.007 mg/kg-day), furfural (RfD of 0.01 mg/kg-day), and 2-(Thiocyanomethylthio)benzothiazole
17 (RfD of 0.01 mg/kg-day), received lower overall scores because they are used more infrequently
18 (each in less than 0.1% of wells in the FracFocus database).

### 9.5.3.3.  Hazard Identification: Chemicals Detected in Flowback and Produced Water

19 As discussed above, a total of 23 chemicals detected in flowback and produced water were
20 identified for hazard evaluation using the selection criteria described in Section 9.5.1. Of these 23
21 chemicals, 10 chemicals overlap with the hazard evaluation of chemicals used in hydraulic
22 fracturing fluids. Because of this overlap, many of the effects noted in each hazard evaluation are
23 similar.

24 These chemicals, along with their POD, total products of uncertainty factors applied, federal chronic
25 RfVs, and the health effect bases for the RfVs, are shown in Table 9-8. As seen in Table 9-8, all of
26 these chemicals had RfDs available from IRIS, PPRTV, or HHBP. These chemicals induce a variety of
27 adverse outcomes, including immune system effects, changes in body weight, changes in blood
28 chemistry, pulmonary toxicity, neurotoxicity, liver and kidney toxicity, and reproductive and
29 developmental toxicity. The RfD values within this suite of chemicals range from 0.001–0.9 mg/kg-
30 day, with pyridine having the lowest RfD and acetone having the highest RfD. For this subset of
31 chemicals, 88% have an RfD within 2 orders of magnitude of each other and 78% have RfDs within
32 a factor of 10 (range of 0.01–0.1 mg/kg-day). Some of these chemicals include chloroform,
33 naphthalene, 1,4-dioxane, toluene, cumene, and ethylbenzene.

34 Although only federal RfVs are considered in this hazard evaluation, 2 of these chemicals—benzene
35 and 1-4-dioxane—also have federal OSFs . These chemicals are also included in the hazard
36 evaluation of chemicals used in hydraulic fracturing fluids, discussed above. 1,4-dioxane is a more
37 potent carcinogen compared to benzene. The OSF for 1,4-dioxane is 0.1 per mg/kg-day and is
38 classified as likely to be a human carcinogen by IRIS (U.S. EPA, 2013f).

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**Table 9-8. Toxicological properties of the 23 chemicals detected in flowback and produced water that were identified for hazard evaluation and MCDA analysis.**

Chemicals are ranked, from low to high, based on their respective federal chronic RfVs. Chemicals in *italics* are also included in the hazard evaluation of chemicals used in hydraulic fracturing fluids.

| Chemical | CASRN | Point of departure (mg/kg-day) | Total un-certainty factors | RfV Chronic RfD (mg/kg-day) | Non-cancer effect | Source |
|---|---|---|---|---|---|---|
| **Pyridine** | 110-86-1 | 1 | 1000 | 0.001 | Increased liver weight | IRIS |
| **2-Methylnaphthalene** | 91-57-6 | 3.5 | 1000 | 0.004 | Pulmonary alveolar proteinosis | IRIS |
| ***Benzene*** | 71-43-2 | 1.2 | 300 | 0.004 | Decreased lymphocyte count in humans | IRIS |
| **Chloroform** | 67-66-3 | 12.9 | 1000 | 0.01 | Fatty cyst formation in the liver; elevated SGPT(or ALT) | IRIS |
| ***Naphthalene*** | 91-20-3 | 71 | 3,000 | 0.02 | Decreased mean terminal body weight > 10% | IRIS |
| **Di(2-ethylhexyl) phthalate** | 117-81-7 | 19 | 1000 | 0.02 | Increased relative liver weight | IRIS |
| **2,4-Dimethylphenol** | 105-67-9 | 50 | 3000 | 0.02 | Clinical signs; hematological changes | IRIS |
| **Pyrene** | 129-00-0 | 75 | 3000 | 0.03 | Kidney effects | IRIS |
| ***1,4-Dioxane*** | 123-91-1 | 9.6 | 300 | 0.03 | Liver and kidney toxicity | IRIS |
| **Fluorene** | 86-73-7 | 125 | 3000 | 0.04 | Decreased RBC, packed cell volume and hemoglobin | IRIS |
| **Fluoranthene** | 206-44-0 | 125 | 3000 | 0.04 | Nephropathy; increased liver weights; hematological alterations | IRIS |
| **2-Methylphenol** (o-Cresol) | 95-48-7 | 50 | 1000 | 0.05 | Decreased body weights and neurotoxicity | IRIS |
| ***Toluene*** | 108-88-3 | 238 | 3000 | 0.08 | Increased absolute kidney weight | IRIS |
| **Carbon disulfide** | 75-15-0 | 11 | 100 | 0.1 | Fetal toxicity and malformations | IRIS |

| Chemical | CASRN | Point of departure (mg/kg-day) | Total un-certainty factors | RfV Chronic RfD (mg/kg-day) | Non-cancer effect | Source |
|---|---|---|---|---|---|---|
| *Cumene* | 98-82-8 | 110 | 1000 | 0.1 | increased average kidney weight in female rats | IRIS |
| **Benzyl alcohol** | 100-51-6 | 143 | 1000 | 0.1 | Effects on survival, growth, and tissue histopathology | PPRTV |
| **Dibutyl phthalate** | 84-74-2 | 125 | 1000 | 0.1 | Increased mortality | IRIS |
| *Ethylbenzene* | 100-41-4 | 97.1 | 1000 | 0.1 | liver and kidney toxicity; histopathology | IRIS |
| *Acetophenone* | 98-86-2 | 423 | 3000 | 0.1 | General toxicity; no LOAEL identified | IRIS |
| **Diphenylamine** | 122-39-4 | 10 | 100 | 0.1 | Alterations in clinical chemistry; increased kidney. liver, and spleen weights | HHBP |
| *Xylenes* | 1330-20-7 | 179 | 1000 | 0.2 | Decreased body weight; increased mortality | IRIS |
| *Phenol* | 108-95-2 | 93 | 300 | 0.3 | Decreased maternal weight gain; developmental toxicity | IRIS |
| *Acetone* | 67-64-1 | 900 | 1000 | 0.9 | Nephropathy | IRIS |

### 9.5.3.4.  MCDA Results: Flowback and Produced Water

1    The hazard potential scores of the selected 23 chemicals detected in flowback and produced water
2    are presented in Table 9-9.

**Table 9-9. MCDA results for 23 chemicals in hydraulic fracturing flowback and produced water.**

Chemicals are ranked, from high to low, based on total hazard potential score. See Section 9.5.2 for details on the calculation.

| Chemical | CASRN | Physicochemical properties score | Occurrence score | Toxicity score | Total hazard potential score |
|---|---|---|---|---|---|
| **Benzene** | 71-43-2 | 0.75 | 1.00 | 1.00 | **2.75** |
| **Pyridine** | 110-86-1 | 0.75 | 1.00 | 1.00 | **2.75** |
| **Naphthalene** | 91-20-3 | 0.75 | 0.67 | 1.00 | **2.42** |

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| Chemical | CASRN | Physicochemical properties score | Occurrence score | Toxicity score | Total hazard potential score |
|---|---|---|---|---|---|
| 2,4-Dimethylphenol | 105-67-9 | 1.00 | 0.33 | 1.00 | **2.33** |
| 2-Methylnaphthalene | 91-57-6 | 0.25 | 1.00 | 1.00 | **2.25** |
| Chloroform | 67-66-3 | 0.75 | 0.33 | 1.00 | **2.08** |
| 2-Methylphenol | 95-48-7 | 1.00 | 0.33 | 0.67 | **2.00** |
| Benzyl alcohol | 100-51-6 | 1.00 | 0.67 | 0.33 | **2.00** |
| Bis(2-Ethylhexyl) Phthalate | 117-81-7 | 0.25 | 0.67 | 1.00 | **1.92** |
| Carbon Disulfide | 75-15-0 | 0.50 | 1.00 | 0.33 | **1.83** |
| Toluene | 108-88-3 | 0.50 | 1.00 | 0.33 | **1.83** |
| Acetone | 67-64-1 | 0.75 | 0.67 | 0.00 | **1.42** |
| Pyrene | 129-00-0 | 0.75 | 0.00 | 0.67 | **1.42** |
| Di-n-butyl Phthalate | 84-74-2 | 0.75 | 0.33 | 0.33 | **1.42** |
| 1,4-Dioxane | 123-91-1 | 1.00 | 0.00 | 0.67 | **1.67** |
| Fluoranthene | 206-44-0 | 1.00 | 0.00 | 0.67 | **1.67** |
| Xylenes | 1330-20-7 | 0.50 | 1.00 | 0.00 | **1.50** |
| Ethylbenzene | 100-41-4 | 0.50 | 0.33 | 0.33 | **1.17** |
| Phenol | 108-95-2 | 1.00 | 0.67 | 0.00 | **1.67** |
| Diphenylamine | 122-39-4 | 1.00 | 0.00 | 0.33 | **1.33** |
| Isopropylbenzene | 98-82-8 | 0.25 | 0.67 | 0.33 | **1.25** |
| Acetophenone | 98-86-2 | 0.75 | 0.00 | 0.33 | **1.08** |
| Fluorene | 86-73-7 | 0.00 | 0.00 | 0.67 | **0.67** |

1    The highest total hazard potential scores for chemicals in flowback and produced water went to
2    benzene and pyridine, followed closely by naphthalene. These three chemicals all have RfVs that fell
3    in the lowest (most toxic) quartile relative to other chemicals in the hazard evaluation (RfDs of
4    0.004, 0.001, and 0.02 mg/kg-day, respectively). Benzene fell in the upper quartile of observed
5    chemical concentrations (with a maximum reported average concentration of 680 μg/l; Barnett
6    shale produced water, Table E-9), while pyridine and naphthalene fell in the second highest quartile
7    (with maximum reported average concentrations of 413 and 238 μg/l, respectively; Barnett shale
8    produced water, Table E-10). These three chemicals only scored moderately in terms of their
9    physicochemical properties, however, as all three are expected to have somewhat lower transport
10   in water compared to other chemicals in the assessment. 2-Methylnaphthalene also fell in the
11   lowest quartile in terms of toxicity (RfD of 0.004 mg/kg-day) and the highest quartile in terms of
12   concentration (average of 1,362 μg/l; Barnett shale produced water, Table E-10), but received a
13   slightly lower score than these chemicals with regards to physiochemical properties.

1   Other chemicals occurring in the upper quartile of flowback and produced water concentrations
2   include toluene (average of 760 µg/l; Barnett shale produced water, Table E-9), xylenes (average of
3   360 µg/l; Barnett shale produced water, Table E-9), and carbon disulfide (median of 400 µg/l;
4   Marcellus shale produced water, Table E-10). These chemicals all received moderate hazard
5   potential scores, as all have higher RfDs (lower toxicity) relative to many of the other chemicals in
6   the hazard evaluation, and are all expected to have moderate transport in water relative to the
7   other chemicals.

8   Other chemicals with RfVs that fell in the lowest (most toxic) quartile in flowback and produced
9   water include chloroform (RfD of 0.01 mg/kg-day), di(2-ethylhexyl)phthalate (RfD of 0.02 mg/kg-
10  day), and 2,4,-dimethylphenol (RfD of 0.02 mg/kg-day). Of these, di(2-ethylhexyl)phthalate was
11  detected at moderately high concentrations relative to other chemicals in the assessment (average
12  of 210 µg/l; Barnett shale produced water, Table E-10), but is expected to have reduced mobility in
13  water due primarily to its more hydrophobic properties. The rest are expected to have moderate to
14  high transport in water, but were detected at relatively lower average concentrations compared to
15  other chemicals in the assessment.

### 9.5.4.  Summary of Chemicals Detected in Multiple Stages of the Hydraulic Fracturing Water Cycle

16  A number of chemicals with federal chronic RfVs that are used in hydraulic fracturing fluids were
17  also found to be present in flowback and produced water stages of the hydraulic fracturing water
18  cycle. The use of a chemical in hydraulic fracturing fluids, and subsequent presence in later stages
19  of the hydraulic fracturing water cycle, is of particular interest in demonstrating which chemicals in
20  this dataset may be mixed, injected, and then detected downstream in the water cycle. This section
21  focuses on that group of chemicals.

22  Based on the available information in our datasets, 23 chemicals overall had federal chronic RfVs
23  and were identified as being used in hydraulic fracturing fluids and detected in the
24  flowback/produced water stage of the hydraulic fracturing water cycle. These chemicals are shown
25  in Table 9-10. 10 of these chemicals were included in both the hazard evaluation of hydraulic
26  fracturing fluids (see Table 9-6 and Table 9-7) and the flowback and produced water hazard
27  evaluation (see Table 9-8 and Table 9-9). This means that these 10 chemicals had both frequency of
28  use data from FracFocus and a reported measured concentration in flowback and produced water
29  from Chapter 7 (Appendix E). These 10 chemicals included all of the BTEX chemicals, as well as
30  naphthalene, 1,4 dioxane, acetone, acetophenone, cumene, and phenol. The chemicals of this group
31  with the lowest chronic oral RfVs were benzene, naphthalene, and 1,4-dioxane. These chemicals all
32  have RfDs within an order of magnitude of each other and are known or likely human carcinogens.
33  The next chemical of this group—toluene—has an RfD 20 times greater than benzene. Overall,
34  benzene was the most toxic of the chemicals listed in Table 9-10.

**Table 9-10. List of the 23 chemicals with federal chronic RfVs identified to be used in hydraulic fracturing fluids and detected in the flowback/produced water stage of the hydraulic fracturing water cycle.**

| Chemical | CASRN | Used in hydraulic fracturing fluids? | FracFocus frequency of use data? | Detected in flowback or produced water? | Physicochemical properties data available? | In hazard evaluation?[a] |
|---|---|---|---|---|---|---|
| 1,4-Dioxane | 123-91-1 | Y | Y | Y | Y | FF+FB |
| Acetone | 67-64-1 | Y | Y | Y | Y | FF+FB |
| Acetophenone | 98-86-2 | Y | Y | Y | Y | FF+FB |
| Benzene | 71-43-2 | Y | Y | Y | Y | FF+FB |
| Cumene | 98-82-8 | Y | Y | Y | Y | FF+FB |
| Ethylbenzene | 100-41-4 | Y | Y | Y | Y | FF+FB |
| Naphthalene | 91-20-3 | Y | Y | Y | Y | FF+FB |
| Phenol | 108-95-2 | Y | Y | Y | Y | FF+FB |
| Toluene | 108-88-3 | Y | Y | Y | Y | FF+FB |
| Xylenes | 1330-20-7 | Y | Y | Y | Y | FF+FB |
| 1,2-Propylene glycol | 57-55-6 | Y | Y | Y | Y | FF |
| Dichloromethane | 75-09-2 | Y | Y | Y | Y | FF |
| Ethylene glycol | 107-21-1 | Y | Y | Y | Y | FF |
| Formic acid | 64-18-6 | Y | Y | Y | Y | FF |
| Methanol | 67-56-1 | Y | Y | Y | Y | FF |
| Aluminum | 7429-90-5 | Y | Y | Y | -- | No |
| Iron | 7439-89-6 | Y | Y | Y | -- | No |
| Di(2-ethylhexyl) phthalate | 117-81-7 | Y | -- | Y | Y | FB |
| Acrolein | 107-02-8 | Y | -- | Y | Y | No |
| Arsenic | 7440-38-2 | Y | -- | Y | -- | No |
| Chlorine | 7782-50-5 | Y | -- | Y | -- | No |
| Chromium (III) | 16065-83-1 | Y | -- | Y | -- | No |
| Chromium (VI) | 18540-29-9 | Y | -- | Y | -- | No |
| Zinc | 7440-66-6 | Y | -- | Y | -- | No |

[a] FF+FB: chemical in both the hydraulic fracturing fluid and flowback/produced water hazard evaluations; FF or FB: chemical in either the hydraulic fracturing fluid or flowback/produced water hazard evaluations. A dash indicates data for chemical not available.

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1    An additional 6 chemicals were included in either the hazard evaluation of hydraulic fracturing
2    fluids (see Table 9-6 and Table 9-7) or the flowback and produced water hazard evaluation (see
3    Table 9-8 and Table 9-9), but not both. These chemicals were reported to be have been used in
4    hydraulic fracturing fluids and detected in flowback/produced water, but lacked the occurrence
5    data (frequency of use or a measured concentration) to support inclusion in both of these hazard
6    evaluations. The remaining 8 chemicals reported to have been used in hydraulic fracturing fluids or
7    detected in flowback/produced water were not included in either of the hazard evaluations
8    presented above because they lacked one or more of the inclusion criteria. These chemicals include
9    acrolein as well as several metals. Arsenic and acrolein have the lowest RfDs by an order of
10   magnitude and arsenic is classified as a known human carcinogen by the EPA, IARC, and NTP.
11   Chromium (VI) is also classified as a known human carcinogen by IARC and NTP.

## 9.6.  Synthesis

12   The overall objective of this chapter was to identify and provide information on the toxicological
13   properties of chemicals used in hydraulic fracturing and of hydraulic fracturing wastewater
14   constituents, and to evaluate the potential hazard of these chemicals to drinking water resources.
15   Toward this end, the EPA developed a comprehensive list of 1,173 chemicals with reported
16   occurrence in the hydraulic fracturing water cycle, separating them into subsets based on whether
17   they were reported to have been used in hydraulic fracturing fluids or detected in flowback and
18   produced water. First, for each of these chemicals, RfVs and OSFs from selected federal, state, and
19   international sources were collected when available. Second, for subsets of chemicals that were
20   identified as being of interest in previous chapters of this report, federal chronic RfVs were used to
21   conduct an initial identification of the potential human health hazards inherent to these chemicals.
22   Finally, for other subsets of chemicals that had data available, an approach for a more data-
23   informed hazard evaluation was illustrated by integrating data on federal chronic RfVs, occurrence,
24   and physicochemical properties using an MCDA framework.

### 9.6.1.  Summary of Findings

25   Across the industrial landscape, thousands of chemicals are used commercially that lack toxicity
26   data ([Judson et al., 2009](#)). Similarly, major knowledge gaps exist regarding the toxicity of most
27   chemicals used in hydraulic fracturing fluids or detected in flowback/produced water, impeding the
28   assessment of human health risks associated with drinking water resources affected by hydraulic
29   fracturing.

30   Of the 1,076 chemicals used in hydraulic fracturing fluids, chronic RfVs and/or OSFs from all of the
31   selected federal, state, or international sources were available for 90 chemicals (8.4%). From the
32   federal sources alone, chronic oral RfVs were available for 73 chemicals (6.8%), and OSFs were
33   available for 15 (1.4%). Potential hazards associated with these chemicals include carcinogenesis,
34   immune system effects, changes in body weight, changes in blood chemistry, cardiotoxicity,
35   neurotoxicity, liver and kidney toxicity, and reproductive and developmental toxicity.

36   Of the 134 chemicals that are reported to have been detected in hydraulic fracturing flowback or
37   produced water, chronic RfVs and/or OSFs from all of the selected federal, state, or international

1   sources were available for 83 chemicals (62%). From the federal sources alone, chronic RfVs were
2   available for 70 chemicals (52%), and OSFs were available for 20 (15%). Potential hazards
3   associated with these chemicals include carcinogenesis, immune system effects, changes in body
4   weight, changes in blood chemistry, pulmonary toxicity, neurotoxicity, liver and kidney toxicity, and
5   reproductive and developmental toxicity.

6   Of the chemicals included in the hazard evaluations, benzene is the only one of these chemicals with
7   an OSF that is classified as a known human carcinogen, while acrylamide was found to be the most
8   potent likely human carcinogen. Several other chemicals, including 1,4-dioxane, dichloromethane,
9   naphthalene, and ethylbenzene are also classified as possible, probable, or likely human
10  carcinogens.

11  Toxicity information spans a wide range with respect to extent, quality and reliability. The sources
12  of RfVs and OSFs selected for the purposes of this chapter are based on criteria developed
13  specifically for this report. For the total 1,173 chemicals identified on the EPA's list, federal, state,
14  and international chronic RfVs and/or OSFs that met stringent selection criteria were available for
15  147 (13%) of the chemicals. Several of the RfVs from selected sources were derived using UFs of up
16  to several orders of magnitude, indicating uncertainty when comparing chemicals for potential
17  toxicity and identifying the chemicals that may be more likely to present a human health hazard.
18  For many of the chemicals used in hydraulic fracturing or found in flowback or produced water
19  there may be relevant information, including cancer and noncancer-related information, from one
20  or more sources that were not evaluated in this chapter. In instances where toxicity data is not
21  available from selected sources, risk assessors may need to draw from alternative sources of hazard
22  information. The chapter discusses two available resources for consideration when RfVs and/or
23  OSFs are not available: QSAR-predicted toxicity data, and toxicity data from the EPA's ACToR
24  database. Oral toxicity data was available on ACToR for 642 (55%) of the chemicals. The
25  information available in the ACToR data warehouse ranges from the federal RfVs discussed in
26  Section 9.3.1, which have undergone extensive peer review, to RfVs and study and test results that
27  have undergone little to no peer review.

28  When considering the potential impact of chemicals on drinking water resources and human health,
29  it is important to consider exposure as well as toxicological properties. The majority of chemicals
30  identified in this report lacked the necessary data to conduct such an assessment. However,
31  integrating data on toxicity, occurrence, and physicochemical properties using an MCDA framework
32  enabled a more data-informed hazard evaluation on some chemicals. This analysis highlighted
33  several chemicals that may be more likely than others to reach drinking water and create a
34  toxicological hazard. In hydraulic fracturing fluid, an example is propargyl alcohol. It was among the
35  chemicals with the lowest RfVs considered in this hazard evaluation, was used in 33% of wells in
36  the FracFocus database, and is water soluble with low volatility. In flowback and produced water,
37  examples of such chemicals include benzene, pyridine, and naphthalene. These chemicals were also
38  among those with the lowest RfVs considered in this hazard evaluation, are expected to be
39  relatively mobile in water, and were present at relatively high average concentrations in flowback.

### 9.6.2. Factors Affecting the Frequency or Severity of Impacts

1  When assessing chemical hazards, there are multiple pieces of information that could be taken into
2  account. This includes knowledge of the chemicals used at a given well site, the toxicological and
3  physicochemical properties of these chemicals, the amount of fluid being used and recovered, the
4  likelihood of well integrity failures, and the likelihood of spills and other unintentional releases.
5  These topics were previously discussed in Chapters 5 through 8 of this report. Because of the large
6  volumes of fluid being injected, even chemicals representing a small percentage of the total fluid by
7  mass may pose a potential for exposure in the event of a spill or leak.

8  Overall, contamination of drinking water resources depends on site-, chemical-, and fluid-specific
9  factors (Goldstein et al., 2014), and the exact mixture and concentrations of chemicals at a site will
10 depend upon the geology and the company's preferences. Therefore, potential hazard and risk
11 considerations are best made on a site-specific, well-specific basis. While the MCDA results in this
12 chapter illustrate an approach to evaluate the relative hazards of these chemicals at the national
13 level, a site-specific hazard evaluation would be necessary in order to identify chemicals of concern
14 at the local level.

15 For example, consider (E)-crotonaldehyde, which is one of the more toxic chemicals considered in
16 the hazard evaluation of hydraulic fracturing fluids. (E)-crotonaldehyde is reportedly used in only
17 0.06% of wells in the FracFocus database, based on the EPA's analysis. If the FracFocus database
18 represents a fair sample of all of the wells across the country, then the likelihood of (E)-
19 crotonaldehyde contamination on a nationwide scale is limited. However, this in no way diminishes
20 the likelihood of (E)-crotonaldehyde contamination at well sites where this chemical is used.
21 Therefore, potential exposures to more toxic but infrequently used chemicals are more of a local
22 issue, rather than a national one.

23 This is in contrast with methanol, which was reported in 73% of wells in the FracFocus database.
24 Methanol is soluble and relatively mobile in water, but has a higher RfV relative to other chemicals
25 in the hazard evaluation. Therefore, when considering chemical usage on a nationwide basis,
26 methanol may be expected to have a higher exposure potential compared to other chemicals, with a
27 moderate overall hazard potential due to its relatively high RfV.

### 9.6.3. Uncertainties

28 There are several notable uncertainties in the chemical and toxicological data that limit a
29 comprehensive assessment of the potential health impacts of hydraulic fracturing on drinking
30 water resources.

31 For the purposes of this chapter, the lack of RfVs and OSFs from the sources meeting stringent
32 selection criteria is the most significant data gap. For instance, of the 32 chemicals (excluding
33 water, quartz, and sodium chloride) that are used in ≥10% of wells nationwide according to
34 FracFocus, federal chronic RfVs were only available for 7 chemicals. Without these reliable and peer
35 reviewed data, comprehensive hazard evaluation and hazard identification of chemicals is difficult,
36 and the ability to consider the potential cumulative effects of exposure to chemical mixtures in

1   hydraulic fracturing fluid, flowback, or produced water is limited. Consequently, potential impacts
2   on drinking water resources and human health may not be assessed adequately.

3   Another major uncertainty lies in the total list of chemicals that was compiled for this chapter. As
4   discussed in Section 5.1.3, information is lacking on the chemicals that are used in hydraulic
5   fracturing fluid formulation. CBI chemicals, which were present in approximately 70% of well
6   records on the FracFocus database, were excluded from the EPA's analysis. The analysis also
7   excluded ingredient records that were not able to be assigned standardized chemical names, which
8   resulted in approximately 35% of FracFocus ingredient records being excluded from the report.
9   This lack of data limits the ability to more completely assess the impact of chemicals that are
10  potentially used with great frequency. Moreover, there may be a regional bias in the EPA's analysis
11  of FracFocus, as 78% of chemical disclosures in the FracFocus database came from five states, and
12  47% were from Texas. Despite these limitations, the FracFocus database remains the most
13  complete source for tracking hydraulic fracturing chemical usage in the United States, and therefore
14  was the best available source for the hazard evaluation in this chapter. Although the sources used to
15  compile the chemical list represented the best available data at the time of this study, it is possible
16  that some of these chemicals are no longer used at all, and many of these chemicals may only be
17  used infrequently. Therefore, it may be possible that significantly fewer than 1,076 chemicals are
18  currently used in abundance. Consequently, having a better understanding of the chemicals and
19  formulations, including those that are CBI, along with their frequency of use and volumes, would
20  greatly benefit risk assessment and risk management decisions.

21  Additionally, the list of flowback and produced water chemicals identified in this chapter is almost
22  certainly incomplete. Few studies to date have examined the chemical composition of flowback and
23  produced water, and the hazard evaluation in this chapter relied on data from the relatively small
24  number of studies that are presented in Appendix E of this assessment. As discussed in Chapter 7,
25  chemicals and their metabolites may go undetected simply because they were not included in the
26  analytical methodology. Additionally, chemical analysis of flowback and produced water may be
27  challenging, because high levels of dissolved solids in flowback and wastewater can interfere with
28  chemical detection. As a result, it is likely that there are chemicals of concern in flowback and
29  produced water that have not been detected or reported.

30  Finally, when considering the MCDA framework that was used to illustrate an approach for hazard
31  evaluation, it should be noted that the physicochemical variables were chosen specifically to reflect
32  chemical mobility and persistence in water. While this framework draws attention towards those
33  chemicals that are most likely to be carried in water, it does not attempt to address the numerous
34  other physicochemical variables that may affect chemical exposure. For instance, as discussed in
35  Chapter 5, hydrophobic chemicals may act as long-term sources of pollution by sorbing to soils or
36  sediments. Additionally, volatile chemicals that dissipate into the air have the potential to pose air
37  pollution hazards, which are not considered in this drinking water assessment; or could potentially
38  be deposited in bodies of water that are distant from the hydraulic fracturing site. Furthermore, as
39  discussed in Chapter 5, chemical fate and transport will be influenced by environmental and site-
40  specific conditions. The fate of a chemical in a chemical mixture will be also influenced by the other

1    chemicals that are present in the mixture, and the relative concentrations of each. Although the
2    assessment of these various scenarios is outside the scope of this report, the potential hazards
3    associated with hydrophobic or volatile chemicals should not be discounted when interpreting the
4    results of this hazard evaluation. It should be emphasized that the MCDA framework illustrated in
5    this chapter represents just one method that can be used to integrate chemical data for hazard
6    evaluation, and is readily adaptable to include different variables, different weights for the
7    variables, and site-specific considerations.

### 9.6.4.  Conclusions

8    The EPA has identified 1,173 chemicals used or detected in the hydraulic fracturing water cycle.
9    Toxicity-based chronic RfVs and/or OSFs from sources meeting selection criteria are not available
10   for the large majority (87%) of these chemicals. In addition, 56% of these chemicals do not have
11   physicochemical property data. Furthermore, 36% of the chemicals used in hydraulic fracturing
12   fluids lack data on their nationwide frequency of use, and very few studies have analyzed the
13   chemical composition of flowback and produced water. Given the large number of chemicals used
14   or detected in various stages of the hydraulic fracturing water cycle, as well as the large number of
15   hydraulic fracturing wells nationwide, this missing chemical information represents a significant
16   data gap. Because of these large data gaps for drinking water resources, it remains challenging to
17   fully understand the toxicity and potential health impacts for single chemicals as well as mixtures of
18   chemicals associated with hydraulic fracturing processes. This chapter provides an initial overall
19   assessment of the potential human health effects associated with hydraulic fracturing on a
20   nationwide basis. It also provides tools that may support risk assessment and risk management
21   decision making at the local and regional level.

22   The toxicological data, occurrence data, and physicochemical data compiled in this report provide a
23   resource for assessing the potential hazards associated with chemicals in the hydraulic fracturing
24   water cycle. Additionally, the MCDA framework presented herein illustrates one method for
25   integrating these data for hazard evaluation. While the analysis in this chapter is constrained to the
26   assessment of chemicals on a nationwide scale, this approach is readily adaptable for use on a
27   regional or site-specific basis.

28   This collection of data provides a tool to inform decisions about protection of drinking water
29   resources. Agencies may use these results to prioritize chemicals for hazard assessment or for
30   determining future research priorities. Industry may use this information to prioritize chemicals for
31   replacement with less toxic, persistent, and mobile alternatives. A summary of the findings related
32   to the overall objective of this chapter and the research questions is presented in Text Box 9-1.

**Text Box 9-1. Research Questions Revisited.**

1 **What are the toxicological properties of hydraulic fracturing fluid chemical additives?**

2 • In a nationwide assessment, the EPA identified 1,076 chemicals that are used in hydraulic fracturing
3 fluids. This does not include chemicals classified as CBI, which the FracFocus database indicates are used
4 in more than 70% of wells. Chronic RfVs and/or OSFs from selected federal, state, and international
5 sources were available for 90 (8.4%) of these chemicals. From the federal sources alone, chronic RfVs
6 were available for 73 chemicals (6.8%), and OSFs were available for 15 chemicals (1.4%). RfVs and OSFs
7 were not available for the majority of chemicals that are used in hydraulic fracturing fluid, representing a
8 significant data gap with regards to hazard identification. Of the chemicals that have selected RfVs, health
9 effects include the potential for carcinogenesis, immune system effects, changes in body weight, changes
10 in blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and
11 developmental toxicity.

12 • When considering the hazard evaluation of these chemicals on a nationwide scale, chemicals such as
13 propargyl alcohol stand out for their relatively lower RfVs, high frequency of use, and expected transport
14 and mobility in water. However, the FracFocus database indicates that most chemicals are used
15 infrequently on a nationwide scale; therefore, potential exposures to the majority of these chemicals are
16 more likely to be a local issue, rather than a national one. Accordingly, potential hazard and risk
17 considerations for hydraulic fracturing fluid chemical additives are best made on a site-specific, well-
18 specific basis.

19 **What are the toxicological properties of hydraulic fracturing wastewater constituents?**

20 • This assessment identified 134 chemicals that are reported to have been detected in hydraulic fracturing
21 flowback or produced water. These include chemicals that are added to hydraulic fracturing fluids during
22 the chemical mixing stage, as well as naturally occurring organic chemicals, metals, naturally occurring
23 radioactive material, and other subterranean chemicals that may be mobilized by the hydraulic fracturing
24 process. Chronic RfVs and/or OSFs from selected federal, state, and international sources were available
25 for 83 (62%) of these chemicals. From the federal sources alone, chronic RfVs were available for 70
26 chemicals (52%), and OSFs were available for 20 chemicals (15%). Of the chemicals that had selected
27 RfVs, health effects include the potential for carcinogenesis, immune system effects, changes in body
28 weight, changes in blood chemistry, pulmonary toxicity, neurotoxicity, liver and kidney toxicity, and
29 reproductive and developmental toxicity.

30 • In a hazard evaluation of flowback and produced water data, chemicals such as benzene, pyridine, and
31 naphthalene stood out for their relatively lower RfVs, high average concentrations, and expected
32 transport and mobility in water. However, the chemicals present in flowback and produced water are
33 likely to vary on a regional and well-specific basis as a result of geological differences as well as
34 differences between hydraulic fracturing fluid formulations. Therefore, potential hazard and risk
35 considerations are best made on a site-specific basis.

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## 9.8. Annex

### 9.8.1. Calculation of Physicochemical Property Scores (MCDA Hazard Evaluation)

1  Section 9.5.2 describes how physicochemical properties scores were based on three subcriteria:
2  mobility, volatility, and persistence. These subcriteria scores were calculated as follows:

3  *Mobility score:* Chemical mobility in water was assessed based upon three physicochemical
4  properties: the octanol-water partition coefficient ($K_{ow}$), the organic carbon-water partition
5  coefficient ($K_{oc}$), and aqueous solubility. $K_{ow}$ and aqueous solubility were previously discussed in
6  Section 5.8.3. $K_{oc}$ is a partitioning coefficient that measures the amount of chemical that is adsorbed
7  onto soil organic carbon per the amount of chemical that is dissolved in water. Like $K_{ow}$, $K_{oc}$ is
8  typically reported as a base-10 logarithm (log $K_{oc}$). From EPI Suite™, $K_{oc}$ was estimated using the
9  MCl Method. Chemicals with low $K_{ow}$ and $K_{oc}$ values are hydrophilic, and thus are more likely to
10 move with water rather than sorbing to soils or sediments. Chemicals with high aqueous solubility
11 are also more likely to move with water. Therefore, chemicals with low $K_{ow}$, low $K_{oc}$, or high aqueous
12 solubility were ranked as having greater potential to affect drinking water resources. Using the
13 thresholds designated in Table 9-5, each of these properties was assigned a score of 1-4. The
14 highest of these three scores was designated as the mobility score for each chemical.

15 *Volatility score*: Chemical volatility was assessed based on the Henry's law constant, which was
16 previously discussed in Section 5.8.3. Chemicals with low Henry's law constants are less likely to
17 leave water via volatilization, and were therefore ranked as having greater potential to impact
18 drinking water. Using the thresholds designated in Table 9-5, the Henry's law constant for each
19 chemical was assigned a score of 1-4. This value was designated as the volatility score for each
20 chemical.

21 *Persistence score:* Chemical persistence was assessed based on estimated half-life in water, which
22 describes how long a chemical will persist in water before it is transformed or degraded. From EPI
23 Suite™, half-life in water was estimated using the Level III Fugacity model. Chemicals with longer
24 half-lives are more persistent, and were therefore ranked as having greater potential to affect
25 drinking water. Using the thresholds designated in Table 9-5, the half-life of each chemical was
26 assigned a score of 1-4. This value was designated as the persistence score for each chemical.

27 For each chemical, the mobility score, volatility score, and persistence score (each on a scale of 1 to
28 4) were summed to calculate a total physicochemical score. The total scores were then
29 standardized by scaling to the highest and lowest scores observed in the subset of chemicals, using
30 the equation described in Section 9.5.2.4.

### 9.8.2.  Example of MCDA Score Calculation

For an example of how the MCDA scores were calculated, consider benzene. This demonstrates how the MCDA score was calculated for benzene in the hazard evaluation of chemicals used in hydraulic fracturing fluids:

- With regards to toxicity (Appendix G), benzene was found to have a federal RfD of 0.004 mg/kg-day (source: IRIS). Within the entire set of chemicals in this hazard evaluation, federal RfDs ranged from 0.001 mg/kg-day [(E)-crotonaldehyde] to 20 mg/kg-day (1,2-propylene glycol). The RfD of benzene fell in the lowest (most toxic) quartile of these scores, and therefore was given a toxicity score of 4. When the results were standardized to the highest score (4) and lowest score (1) within the set of chemicals, benzene was calculated to have a toxicity score of 1, as follows:

$$1 = (4 - 1) / (4 - 1)$$

- Benzene was used in 0.0056% of wells nationwide (U.S. EPA, 2015a). This usage frequency falls in the lowest quartile of chemicals, and therefore benzene was given an occurrence score of 1. When the results were standardized to the highest score (4) and lowest score (1) within the set of chemicals, benzene was calculated to have an occurrence score of 0, as follows:

$$0 = (1 - 1) / (4 - 1)$$

- Based on physicochemical properties, benzene received a mobility score of 4 (log $K_{ow}$ = 2.13; log $K_{oc}$ = 1.75; solubility = 2000 mg/l), a volatility score of 2 (Henry's law constant = 0.00555), and a persistence score of 2 (half-life in water = 37.5 days). These scores sum to a total physicochemical properties score of 8. Within the entire set of chemicals in this hazard evaluation, several chemicals received total scores of 9, which was the highest observed score. Cumene received a total score of 6, which was the lowest observed score. When the results were standardized to the high score (9) and low score (6) using the equation above, benzene was calculated to have a physicochemical properties score of 0.67 as follows:

$$0.67 = (8 - 6) / (9 - 6)$$

To calculate the total hazard potential score for benzene, the physicochemical properties score, toxicity score, and occurrence score were summed for a total of 1.67. These results can be seen in Table 9-7, which shows the MCDA results for chemicals used in hydraulic fracturing fluid.

# Chapter 10

# Synthesis

# 10.  Synthesis

1  In this assessment, we examined the potential for hydraulic fracturing for oil and natural gas to
2  change the quality or quantity of drinking water resources, and identified factors that affect the
3  frequency or severity of potential impacts. Drinking water resources are defined broadly in this
4  report as any body of ground water or surface water that now serves, or in the future could serve,
5  as a source of drinking water for public or private use. We assessed potential effects on drinking
6  water resources from both routine operations and potential accidents. Impacts were defined as any
7  change in the quality or quantity of drinking water resources. Where possible, we identified the
8  mechanisms responsible or potentially responsible for any impacts. For example, a spill of
9  hydraulic fracturing fluid is a mechanism by which drinking water resources could be impacted.

10  We did this by following water through the hydraulic fracturing water cycle: (1) the withdrawal of
11  ground or surface water needed for hydraulic fracturing fluids; (2) the mixing of water, chemicals,
12  and proppant on the well pad to create the hydraulic fracturing fluid; (3) the injection of hydraulic
13  fracturing fluids into the well to fracture the geologic formation; (4) the management of flowback
14  and produced water, both on the well pad and in transit for reuse, treatment, or disposal; and (5)
15  the reuse, treatment and discharge, or disposal of hydraulic fracturing wastewater.

16  In this chapter, we summarize major findings of the assessment, organized by each stage of the
17  hydraulic fracturing water cycle (Section 10.1); highlight key uncertainties related to these major
18  findings (Section 10.2); and discuss the assessment's overall conclusions (Section 10.3) and
19  potential uses (Section 10.4).

## 10.1. Major Findings

20  From our assessment, we conclude there are above and below ground mechanisms by which
21  hydraulic fracturing activities have the potential to impact drinking water resources. These
22  mechanisms include water withdrawals in times of, or in areas with, low water availability; spills of
23  hydraulic fracturing fluids and produced water; fracturing directly into underground drinking
24  water resources; below ground migration of liquids and gases; and inadequate treatment and
25  discharge of wastewater.

26  We did not find evidence that these mechanisms have led to widespread, systemic impacts on
27  drinking water resources in the United States. Of the potential mechanisms identified in this report,
28  we found specific instances where one or more mechanisms led to impacts on drinking water
29  resources, including contamination of drinking water wells. The number of identified cases,
30  however, was small compared to the number of hydraulically fractured wells.

31  This finding could reflect a rarity of effects on drinking water resources, but may also be due to
32  other limiting factors. These factors include: insufficient pre- and post-fracturing data on the quality
33  of drinking water resources; the paucity of long-term systematic studies; the presence of other
34  sources of contamination precluding a definitive link between hydraulic fracturing activities and an
35  impact; and the inaccessibility of some information on hydraulic fracturing activities and potential
36  impacts.

1   Below, we provide a synopsis of the assessment's key findings, organized according to each stage of
2   the hydraulic fracturing water cycle. We provide answers to the research questions presented in
3   the Study Plan and Chapter 1. Results from Chapter 9 (Hazard Evaluation and Identification of
4   Chemicals across the Hydraulic Fracturing Water Cycle) are included in the Chemical Mixing and
5   the Flowback and Produced Water sections. While some citations are provided here, individual
6   chapters can be consulted for additional detail and citations.

### 10.1.1. Water Acquisition (Chapter 4)

7   Water is a major component of nearly all hydraulic fracturing operations. It typically makes up
8   almost 90% or more of the fluid injected into a well, and each hydraulically fractured well requires
9   thousands to millions of gallons of water. Cumulatively, hydraulic fracturing activities in the United
10  States used on average 44 billion gal of water a year in 2011 and 2012, according to the EPA's
11  analysis of FracFocus 1.0 disclosures. Although this represents less than 1% of total annual water
12  use and consumption at this scale, water withdrawals could potentially impact the quantity and
13  quality of drinking water resources at more local scales.

#### *Research Questions: Water Acquisition*

- ### *What are the types of water used for hydraulic fracturing?*
14  Water for hydraulic fracturing typically comes from surface water, ground water, or reused
15  hydraulic fracturing wastewater. Hydraulic fracturing operations in the eastern United States
16  generally rely on surface water, while operations in the more semi-arid to arid western states
17  generally use mixed supplies of surface and ground water. In the Marcellus Shale in Pennsylvania,
18  for example, most water used for hydraulic fracturing originates from surface water, whereas
19  surface and ground water are used in approximately equal proportions in the Barnett Shale in
20  Texas (see Figure 10-1a,b). In areas that lack available surface water (e.g., western Texas), ground
21  water supplies most of the water needed for hydraulic fracturing.

22  Across the United States, the vast majority of water used in hydraulic fracturing is fresh, although
23  operators also make use of lower-quality water, including reused hydraulic fracturing wastewater.
24  Based on available data, the median reuse of hydraulic fracturing wastewater as a percentage of
25  injected volumes is 5% nationally, with the percentage varying by location.[1] Available data on reuse
26  trends indicate increased reuse of wastewater over time in both Pennsylvania and West Virginia.
27  Reuse as a percentage of injected volumes is lower in other areas, including regions with more
28  water stress, likely because of the availability of disposal wells. For example, reused wastewater is
29  approximately 18% of injected volumes in the Marcellus Shale in Pennsylvania's Susquehanna
30  River Basin, whereas it is approximately 5% in the Barnett Shale in Texas (see Figure 10-1a,b).

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[1] Reused wastewater as a percentage of injected volumes differs from the percentage of wastewater that is managed through reuse, as opposed to other wastewater management options. For example, in the Marcellus Shale in Pennsylvania, approximately 18% of injected water is reused produced water, while approximately 70% or more of wastewater is managed through reuse (see Figure 10-1a).
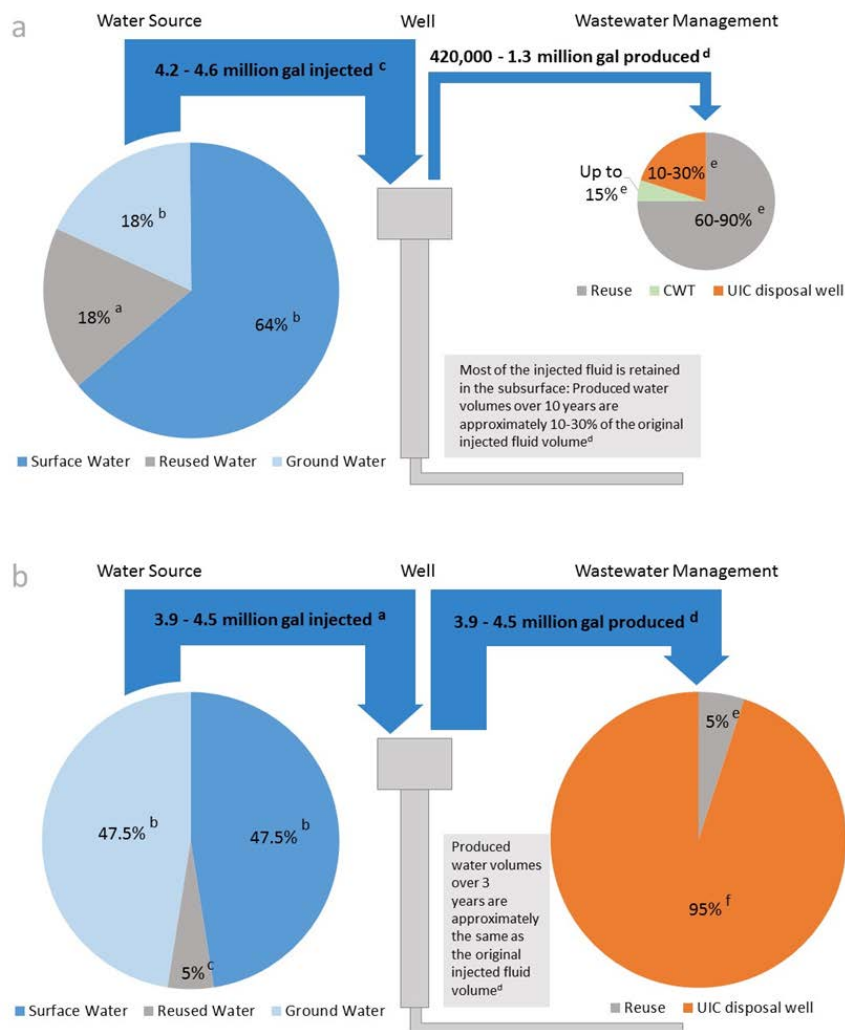
**Figure 10-1. Water budgets representative of practices in the Marcellus Shale in the Susquehanna River Basin in Pennsylvania (a) and the Barnett Shale in Texas (b).**

Pie size and arrow thickness represent the relative volume of water as it flows through the hydraulic fracturing water cycle. Wastewater going to a centralized waste treatment (CWT) facility may be either discharged to surface water or reused. Wastewater going to an underground injection control (UIC) well is disposed of below ground. These examples represent typical water management practices as depicted for the most recent time period reviewed by this assessment. They do not represent any specific well. Sources for 10-1a: (a) Table 4-1 (Hansen et al., 2013); (b) Table 4-3 (U.S. EPA, 2015c); (c) Appendix Table B-5 (Hansen et al., 2013); (d) Table 7-2 (Ziemkiewicz et al., 2014)—Note: produced water value from the West Virginia portion of the Marcellus; it provided the longest-term measurement of produced water volumes; (e) Figure 8-4 (PA DEP, 2015a) and Table 8-5 (Ma et al., 2014; Shaffer et al., 2013). Sources for 10-1b: (a) Appendix Table B-5 (U.S. EPA, 2015a; Nicot et al., 2012; Nicot et al., 2011); (b) Table 4-3 (Nicot et al., 2014); (c) Table 4-1 (Nicot et al., 2012); d: Table 7-2 (Nicot et al., 2014); (e) Table 8-5 (Nicot et al., 2012); (f) Calculated by subtracting reuse values from 100% (see Table 8-5).

- ***How much water is used per well?***

1   The national median volume of water used per hydraulically fractured well is approximately 1.5
2   million gal (5.7 million L), according to the EPA's analysis of FracFocus 1.0 disclosures. This
3   estimate likely represents a wide variety of fractured well types, including vertical wells that
4   generally use much less water per well than horizontal wells. Thus, published estimates for
5   horizontal shale gas wells are typically higher (e.g., approximately 4 million gal (Vengosh et al.,
6   2014)). There is also wide variation within and among states and basins in the median water
7   volumes used per well, from more than 5 million gal (19 million L) in Arkansas, Louisiana and West
8   Virginia to less than 1 million gal (3.8 million L) in California, New Mexico, and Utah, among others.
9   This variation results from several factors, including well length, formation geology, and fracturing
10  fluid formulation.

- ***How might cumulative water withdrawals for hydraulic fracturing affect drinking water
  quantity?***

11  Cumulatively, hydraulic fracturing uses billions of gallons of water each year at the national and
12  state scales, and even in some counties. As noted above, hydraulic fracturing water use and
13  consumption are generally less than 1% of total annual water use and consumption at these scales.
14  However, there are a few counties in the United States where these percentages are higher. For
15  2011 and 2012, annual hydraulic fracturing water use was 10% or more compared to 2010 total
16  annual water use in 6.5% of counties with FracFocus 1.0 disclosures analyzed by the EPA, 30% or
17  more in 2.2% of counties, and 50% or more in 1.0% of counties. Consumption estimates followed
18  the same general pattern. For these counties, hydraulic fracturing is a relatively large user and
19  consumer of water.

20  High fracturing water use or consumption alone does not necessarily result in impacts to drinking
21  water resources. Rather, impacts result from the combination of water use/consumption and water
22  availability at local scales. In our survey of published literature, we did not find a case where
23  hydraulic fracturing water use or consumption alone caused a drinking water well or stream to run
24  dry. This could indicate an absence of effects or a lack of documentation in the literature we
25  reviewed. Additionally, water availability is rarely impacted by just one use or factor alone. In
26  Louisiana, for example, the state requested hydraulic fracturing operations switch from ground to
27  surface water, due to concerns that ground water withdrawals for fracturing could, in combination
28  with other uses, adversely affect drinking water supplies.

29  The potential for impacts to drinking water resources from hydraulic fracturing water withdrawals
30  is highest in areas with relatively high fracturing water use and low water availability. Southern
31  and western Texas are two locations where hydraulic fracturing water use, low water availability,
32  drought, and reliance on declining ground water has the potential to affect the quantity of drinking
33  water resources. Any impacts are likely to be realized locally within these areas. In a detailed case
34  study of southern Texas, Scanlon et al. (2014) observed generally adequate water supplies for
35  hydraulic fracturing, except in specific locations. They found excessive drawdown of local ground
36  water in a small proportion (approximately 6% of the area) of the Eagle Ford Shale. They suggested

1   water management, particularly a shift towards brackish water use, could minimize potential future
2   impacts to fresh water resources.

3   The potential for impacts to drinking water quantity due to hydraulic fracturing water use appears
4   to be lower—but not eliminated—in other areas of the United States. Future problems could arise if
5   hydraulic fracturing increases substantially in areas with low water availability, or in times of water
6   shortages. In detailed case studies in western Colorado and northeastern Pennsylvania, the EPA did
7   not find current impacts, but did conclude that streams could be vulnerable to water withdrawals
8   from hydraulic fracturing. In northeast Pennsylvania, water management, such as minimum stream
9   flow requirements, limits the potential for impacts, especially in small streams. In western North
10  Dakota, ground water is limited, but the industry may have sufficient supplies of surface water from
11  the Missouri River system. These location-specific examples emphasize the need to focus on
12  regional and local dynamics when considering potential impacts of hydraulic fracturing water
13  acquisition on drinking water resources.

- ***What are the possible impacts of water withdrawals for hydraulic fracturing on water quality?***

14  Water withdrawals for hydraulic fracturing, similar to all water withdrawals, have the potential to
15  alter the quality of drinking water resources. Ground water withdrawals exceeding natural
16  recharge rates decrease water storage in aquifers, potentially mobilizing contaminants or allowing
17  the infiltration of lower quality water from the land surface or adjacent formations. Withdrawals
18  could also decrease ground water discharge to streams, potentially affecting surface water quality.
19  Areas with large amounts of sustained ground water pumping are most likely to experience
20  impacts, particularly drought-prone regions with limited ground water recharge.

21  Surface water withdrawals also have the potential to affect water quality. Withdrawals may lower
22  water levels and alter stream flow, potentially decreasing a stream's capacity to dilute
23  contaminants. Case studies by the EPA show that streams can be vulnerable to changes in water
24  quality due to water withdrawals, particularly smaller streams and during periods of low flow.
25  Management of the rate and timing of surface water withdrawals has been shown to help mitigate
26  potential impacts of hydraulic fracturing withdrawals on water quality.

### 10.1.2. Chemical Mixing (Chapter 5)

27  Hydraulic fracturing fluids are developed to perform specific functions, including: create and
28  extend fractures, transport proppant, and place proppant in the fractures. The fluid generally
29  consists of three parts: (1) the base fluid, which is the largest constituent by volume and is typically
30  water; (2) the additives, which can be a single chemical or a mixture of chemicals; and (3) the
31  proppant. Additives are chosen to serve a specific purpose (e.g., adjust pH, increase viscosity, limit
32  bacterial growth). Chemicals generally comprise a small percentage (typically 2% or less) of the
33  overall injected fluid volume. Because over one million gallons of fluid are typically injected per
34  well, thousands of gallons of chemicals can be potentially stored on-site and used during hydraulic
35  fracturing activities.

36  On-site storage, mixing, and pumping of chemicals and hydraulic fracturing fluids have the potential
37  to result in accidental releases, such as spills or leaks. Potential impacts to drinking water resources

1   from spills of hydraulic fracturing fluids and chemicals depend on the characteristics of the spills,

2   and the fate, transport, and the toxicity of chemicals spilled.

### *Research Questions: Chemical Mixing*

- ***What is currently known about the frequency, severity, and causes of spills of hydraulic fracturing fluids and additives?***

3   The frequency of on-site spills from hydraulic fracturing could be estimated for two states, but not

4   for operations nationally or for other areas. Frequency estimates from data and literature ranged

5   from one spill for every 100 wells in Colorado to between approximately 0.4 and 12.2 spills for

6   every 100 wells in Pennsylvania.[1] These estimates include spills of hydraulic fracturing fluids and

7   chemicals, and produced water reported in state databases. Available data generally precluded

8   estimates of hydraulic fracturing fluid and/or chemical spill rates separately from estimates of an

9   overall spill frequency. It is unknown whether these spill estimates are representative of national

10   occurrences. If the estimates are representative, the number of spills nationally could range from

11   approximately 100 to 3,700 spills annually, assuming 25,000 to 30,000 new wells are fractured per

12   year.

13   The EPA characterized volumes and causes of hydraulic fracturing-related spills identified from

14   selected state and industry data sources. The spills occurred between January 2006 and April 2012

15   in 11 states and included 151 cases in which fracturing fluids or chemicals spilled on or near a well

16   pad. Due to the methods used for the EPA's characterization of spills, these cases were likely a

17   subset of all fracturing fluid and chemical spills during the study's time period. The reported

18   volume of fracturing fluids or chemicals spilled ranged from 5 gal to more than 19,000 gal (19 to

19   72,000 L), with a median volume of 420 gal (1,600 L) per spill. Spill causes included equipment

20   failure, human error, failure of container integrity, and other causes (e.g., weather and vandalism).

21   The most common cause was equipment failure, specifically blowout preventer failure, corrosion,

22   and failed valves. More than 30% of the 151 fracturing fluid or chemical spills were from fluid

23   storage units (e.g., tanks, totes, and trailers).

- ***What are the identities and volumes of chemicals used in hydraulic fracturing fluids, and how might this composition vary at a given site and across the country?***

24   In this assessment, we identified a list of 1,076 chemicals used in hydraulic fracturing fluids. This is

25   a cumulative list over multiple wells and years. These chemicals include acids, alcohols, aromatic

26   hydrocarbons, bases, hydrocarbon mixtures, polysaccharides, and surfactants. According to the

27   EPA's analysis of disclosures to FracFocus 1.0, the number of unique chemicals per well ranged

28   from 4 to 28, with a median of 14 unique chemicals per well.

29   Our analysis indicates that chemical use varies and that no single chemical is used at all well sites

30   across the country, although several chemicals are widely used. Methanol, hydrotreated light

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[1] Spill frequency estimates are for a given number of wells over a given period of time. These are not annual estimates nor are they for the lifetime of a well.

1  petroleum distillates, and hydrochloric acid were reported as used in 65% or more of wells,
2  according to FracFocus 1.0 disclosures analyzed by the EPA. Only 32 chemicals, excluding water,
3  quartz, and sodium chloride, were used in more than 10% of wells according to the EPA's analysis
4  of FracFocus disclosures. The composition of hydraulic fracturing fluids varies by state, by well, and
5  within the same service company and geologic formation. This variability likely results from several
6  factors, including the geology of the formation, the availability and cost of different chemicals, and
7  operator preference.

8  Estimates from the EPA's database developed from FracFocus 1.0 suggest median volumes of
9  individual chemicals injected per well range from a few gallons to thousands of gallons, with an
10  overall median of 650 gal (2,500 L) per chemical per well. Based on this overall median and
11  assuming 14 unique chemicals are used per well, an estimated 9,100 gal (34,000 L) of chemicals
12  may be injected per well. Given that the number of chemicals per well ranges from 4 to 28, the
13  estimated volume of chemicals injected per well may range from approximately 2,600 to 18,000 gal
14  (9,800 to 69,000 L).

- ***What are the chemical, physical, and toxicological properties of hydraulic fracturing chemical additives?***

15  Measured or estimated physicochemical properties were obtained for 453 chemicals of the toal
16  1,076 chemicals reported in hydraulic fracturing fluids. We could not estimate physicochemical
17  properties for the inorganic chemicals or mixtures. The 453 chemicals have a wide range of
18  physicochemical properties.

19  Properties affecting the likelihood of a spilled chemical reaching and impacting a drinking water
20  resource, include: mobility, solubility, and volatility. Of the 453 chemicals for which
21  physicochemical properties were available, 18 of the top 20 most mobile ones were reported in the
22  EPA's FracFocus 1.0 database for 2% or less of wells. Choline chloride and tetrakis (hydroxymethyl)
23  phosphonium were exceptions and were reported in 14% and 11% of wells, respectively. These
24  two chemicals appear to be relatively more common, and, if spilled,  would move quickly through
25  the environment with the flow of water. The majority of the 453 chemicals associate strongly with
26  soils and organic materials, suggesting the potential for these chemicals to persist in the
27  environment as long-term contaminants. Many of the 453 chemicals fully dissolve in water, but
28  their aqueous solubility varies greatly. Few of the chemicals volatilize, and thus a large proportion
29  of most hydraulic fracturing chemicals tend to remain in water.

30  Oral reference values and oral slope factors meeting the criteria used in this assessment were not
31  available for the majority of chemicals used in hydraulic fracturing fluids, representing a significant
32  data gap for hazard identification.[1,2] Reference values and oral slope factors are important for

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[1] A reference value is an estimate of an exposure to the human population (including susceptible subgroups) for a given duration that is likely to be without an appreciable risk of adverse health effects over a lifetime. Reference value is a generic term not specific to a given route of exposure.
[2] An oral slope factor is an upper-bound, approximating 95% confidence limit, on the increased cancer risk from a lifetime oral exposure to an agent.

1    understanding the potential human health effects resulting from exposure to a chemical. Chronic
2    oral reference values and/or oral slope factors from selected federal, state, and international
3    sources were available for 90 (8%) of the 1,076 chemicals used in hydraulic fracturing fluids. From
4    U.S. federal sources alone, chronic oral reference values were available for 73 chemicals (7%) of the
5    1,076 chemicals, and oral slope factors were available for 15 chemicals (1%). Of the 32 chemicals
6    reported as used in at least 10% of wells in the EPA's FracFocus database (excluding water, quartz,
7    and sodium chloride), seven (21%) have a federal chronic oral reference value. Oral reference
8    values and oral slope factors are a key component of the risk assessment process, although
9    comprehensive risk assessments that characterize the health risk associated with exposure to these
10   chemicals are not available.

11   Of the chemicals that had values available, the health endpoints associated with those values
12   include the potential for carcinogenesis, immune system effects, changes in body weight, changes in
13   blood chemistry, cardiotoxicity, neurotoxicity, liver and kidney toxicity, and reproductive and
14   developmental toxicity. However, it is important to note that evaluating any potential risk to human
15   populations would require knowledge of the specific chemicals that are present at a particular site,
16   whether or not humans are exposed to those chemicals and, if so, at what levels and for what
17   duration, and the toxicity of the chemicals. Since most chemicals are used infrequently on a
18   nationwide basis, potential exposure is likely to be a local or regional issue, rather than a national
19   issue. Accordingly, consideration of hazards and risks associated with these chemical additives
20   would be most useful on a site-specific basis and is beyond the scope of this assessment.

- ***If spills occur, how might hydraulic fracturing chemical additives contaminate drinking water resources?***

21   There are several mechanisms by which a spill can potentially contaminate drinking water
22   resources. These include overland flow to nearby surface water, soil contamination and eventual
23   transport to surface water, and infiltration and contamination of underlying ground water. Of the
24   151 spills characterized by the EPA, fluids reached surface water in 13 (9% of 151) cases and soil in
25   97 (64%) cases. None of the spills of hydraulic fracturing fluid were reported to have reached
26   ground water. This could be due to an absence of impact; however, it can take several years for
27   spilled fluids to infiltrate soil and leach into ground water. Thus, it may not be immediately
28   apparent whether a spill has reached ground water or not.

29   Based on the relative importance of each of these mechanisms, impacts have the potential to occur
30   quickly, be delayed short or long periods, or have a continual effect over time. In Kentucky, for
31   example, a spill impacted a surface water body relatively quickly when hydraulic fracturing fluid
32   entered a creek, significantly reducing the water's pH and increasing its conductivity (Papoulias
33   and Velasco, 2013).

### 10.1.3. Well Injection (Chapter 6)

34   Hydraulic fracturing fluids are injected into oil or gas wells under high pressures. The fluids flow
35   through the well (commonly thousands of feet below the surface) into the production zone (i.e., the

1   geologic formation being fractured) where the fluid injection pressures are sufficient to create
2   fractures in the rock.

3   There are two major subsurface mechanisms by which the injection of fluid and the creation and
4   propagation of fractures can lead to contamination of drinking water resources: (1) the unintended
5   movement of liquids or gases out of the production well or along the outside of the production well
6   into a drinking water resource via deficiencies in the well's casing or cement, and (2) the
7   unintended movement of liquids or gases from the production zone through subsurface geologic
8   formations into a drinking water resource. Combinations of these two mechanisms are also
9   possible.

### Research Questions: Well Injection

- **How effective are current well construction practices at containing fluids- both liquids and gases - before, during, and after fracturing?**

10   Production wells are constructed to access and convey hydrocarbons from the formations in which
11   they are found to the surface, and to isolate fluid-bearing zones (containing oil, gas, or water) from
12   each other. Typically, multiple casings are emplaced and cemented along the wellbore to protect
13   and isolate the oil and/or natural gas from the formations it must travel through to reach the
14   surface.

15   Below ground drinking water resources are often separated from the production well using casing
16   and cement. Cemented surface casing, in particular, is an important well construction feature for
17   isolating drinking water resources from liquids and gases that may move through the subsurface. A
18   limited risk modeling study of selected injection wells in the Williston Basin in North Dakota
19   suggests that the risk of aquifer contamination from leaks inside the well to the drinking water
20   resource decreases by a factor of approximately one thousand when surface casing extends below
21   the bottom of the drinking water resource ([Michie and Koch, 1991](#)). Most wells used in hydraulic
22   fracturing operations have casing and a layer of cement to protect drinking water resources, but
23   there are exceptions: a survey conducted by the EPA of oil and gas production wells hydraulically
24   fractured by nine oil and gas service companies in 2009 and 2010 estimated that at least 3% of the
25   wells (600 out of 23,000 wells) did not have cement across a portion of the casing installed through
26   the protected ground water resource identified by well operators. The absence of cement does not
27   in and of itself lead to an impact.  However, it does reduce the overall number of casing and cement
28   barriers fluids must travel through to reach ground water resources.

29   Impacts to drinking water resources from subsurface liquid and gas movement may occur if casing
30   or cement are inadequately designed or constructed, or fail. There are several examples of these
31   occurrences in hydraulically fractured wells that have or may have resulted in impacts to drinking
32   water resources. In one example, an inner string of casing burst during hydraulic fracturing, which
33   resulted in a release of fluids on the land surface and possibly into the aquifer near Killdeer, North
34   Dakota. The EPA found that, based on the data analysis performed for the study, the only potential
35   source consistent with conditions observed in two impacted monitoring wells was the blowout that

1    occurred during hydraulic fracturing (U.S. EPA, 2015j). In other examples, inadequately cemented
2    casing has contributed to impacts to drinking water resources. In Bainbridge, Ohio, inadequately
3    cemented casing in a hydraulically fractured well contributed to the buildup of natural gas and high
4    pressures along the outside of a production well. This ultimately resulted in movement of natural
5    gas into local drinking water aquifers (Bair et al., 2010; ODNR, 2008). In the Mamm Creek gas field
6    in Colorado, inadequate cement placement in a production well allowed methane and benzene to
7    migrate along the production well and through natural faults and fractures to drinking water
8    resources (Science Based Solutions LLC, 2014; Crescent, 2011; COGCC, 2004). These cases illustrate
9    how construction issues, sustained casing pressure, and the presence of natural faults and fractures
10   can work together to create pathways for fluids to migrate toward drinking water resources.

11   Fracturing older wells may also increase the potential for impacts to drinking water resources via
12   movement of liquids and gases from the inside of the production well or along the outside of the
13   production well to ground water resources. The EPA estimated that 6% of 23,000 oil and gas
14   production wells were drilled more than 10 years before being hydraulically fractured in 2009 or
15   2010. Although new wells can be designed to withstand the stresses associated with hydraulic
16   fracturing operations, older wells may not have been built or tested to the same specifications and
17   their reuse for this purpose could be of concern. Moreover, aging and use of the well can contribute
18   to casing degradation, which can be accelerated by exposure to corrosive chemicals such as
19   hydrogen sulfide, carbonic acid, and brines.

- ***Can subsurface migration of fluids- both liquids and gases- to drinking water resources occur, and what local geologic or artificial features might allow this?***

20   Physical separation between the production zone and drinking water resources can help protect
21   drinking water. Many hydraulic fracturing operations target deep formations such as the Marcellus
22   Shale or the Haynesville Shale (Louisiana/Texas), where the vertical distance between the base of
23   drinking water resources and the top of the shale formation may be a mile or greater. Numerical
24   modeling and microseismic studies based on a Marcellus Shale-like environment suggest that
25   fractures created during hydraulic fracturing are unlikely to extend upward from these deep
26   formations into shallow drinking water aquifers.

27   Not all hydraulic fracturing is performed in zones that are deep below drinking water resources.
28   For example, operations in the Antrim Shale (Michigan) and the New Albany Shale
29   (Illinois/Indiana/Kentucky) take place at shallower depths (100 to 1,900 ft or 30 to 579 m), with
30   less vertical separation between the formation and drinking water resources (NETL, 2013; GWPC
31   and ALL Consulting, 2009). The EPA's survey of oil and gas production wells hydraulically fractured
32   by nine service companies in 2009 and 2010 estimated that 20% of 23,000 wells had less than
33   2,000 ft (610 m) of measured distance between the point of shallowest hydraulic fracturing and the
34   base of the protected ground water resources reported by well operators.

35   There are also places in the subsurface where oil and gas resources and drinking water resources
36   co-exist in the same formation. Evidence indicates that hydraulic fracturing occurs within these
37   formations. This results in the introduction of fracturing fluids into formations that may currently
38   serve, or in the future could serve, as a source of drinking water for public or private use. According

1   to the data examined, the overall frequency of occurrence of this practice appears to be low, with
2   the activity generally concentrated in some areas in the western United States. The practice of
3   injecting fracturing fluids into a formation that also contains a drinking water resource directly
4   affects the quality of that water, since some of the fluid likely remains in the formation following
5   hydraulic fracturing. Hydraulic fracturing in a drinking water resource is a concern in the short-
6   term (should there be people currently using these zones as a drinking water supply) and the long-
7   term (if drought or other conditions necessitate the future use of these zones for drinking water).

8   Liquid and gas movement from the production zone to underground drinking water resources may
9   also occur via other production wells or injection wells near hydraulic fracturing operations.
10  Fractures created during hydraulic fracturing can intersect nearby wells or their fracture networks,
11  resulting in the flow of fluids into those wells. These well communications, or "frac hits," are more
12  likely to occur if wells are close to each other or on the same well pad. In the Woodford Shale in
13  Oklahoma, the likelihood of well communication was less than 10% between wells more than 4,000
14  ft (1,219 m) apart, but rose to nearly 50% between wells less than 1,000 ft (305 m) apart (Ajani and
15  Kelkar, 2012). If an offset well is not able to withstand the stresses applied during the hydraulic
16  fracturing of a neighboring well, well components may fail, which could result in a release of fluids
17  at the surface from the offset well. The EPA identified incidents in which surface spills of hydraulic
18  fracturing-related fluids were attributed to well communication events.

19  Older or inactive wells—including oil and gas wells, injection wells, or drinking water wells—near
20  the hydraulic fracturing operation may pose an even greater potential for impacts. A study in
21  Oklahoma found that older wells were more likely to be negatively affected by the stresses applied
22  by hydraulic fracturing in neighboring wells (Ajani and Kelkar, 2012). In some cases, inactive wells
23  in the vicinity of hydraulic fracturing activities may not have been plugged properly—many  wells
24  plugged before the 1950s were done so with little or no cement. The Interstate Oil and Gas Compact
25  Commission estimates that over one million wells may have been drilled in the United States prior
26  to a formal regulatory system being in place, and the status and location of many of these wells are
27  unknown (IOGCC, 2008). State programs exist to plug identified inactive wells, and work is on-
28  going to identify and address such wells.

### 10.1.4. Flowback and Produced Water (Chapter 7)

29  Water, of variable quality, is a byproduct of oil and gas production. After hydraulic fracturing, the
30  injection pressure is released and water flows back from the well. Initially this water is similar to
31  the hydraulic fracturing fluid, but as time goes on the composition is affected by the characteristics
32  of the formation and possible reactions between the formation and the fracturing fluid. Water
33  initially produced from the well after hydraulic fracturing is sometimes called flowback in the
34  literature, and the term appears in this assessment. However, hydraulic fracturing fluids and any
35  formation water returning to the surface are often referred to collectively as produced water. This
36  definition of produced water is used in this assessment.

37  The amount of produced water varies, but typically averages 10% to 25% of injected volumes,
38  depending upon the amount of time since fracturing and the particular well (see Figure 10-1a).
39  However, there are exceptions to this, such as in the Barnett Shale in Texas where the total volume

1  of produced water can equal or exceed the injected volume of hydraulic fracturing fluid (see Figure
2  10-1b). Flow rates are generally high initially, and then decrease over time throughout oil or gas
3  production.

4  Impacts on drinking water resources have the potential to occur if produced water is spilled and
5  enters surface water or ground water. Environmental transport of chemical constituents in
6  produced water depends on the characteristics of the spill (e.g., volume and duration), the
7  composition of spilled fluids, and the characteristics of the surrounding environment.

### *Research Questions: Flowback and Produced Water*

- ***What is currently known about the frequency, severity, and causes of spills of flowback and produced water?***

8  Surface spills of produced water from hydraulically fractured wells have occurred. As noted in the
9  Chemical Mixing section above, the frequency of on-site spills from hydraulic fracturing activities
10  could be estimated for two states, but not nationally. Estimates of spill frequencies at hydraulic
11  fracturing sites in Colorado and Pennsylvania, including spills of produced water, ranged from
12  approximately 0.4 to 12.2 spills per 100 wells. Available data generally precluded estimates of
13  produced water spill rates separately from estimates of overall spill frequency. Away from the well,
14  produced water spills from pipelines and truck transport also have the potential to impact drinking
15  water resources.

16  The EPA characterized spill volumes and causes for 225 cases in which produced water spilled on
17  or near a well pad. These spills occurred between January 2006 and April 2012 in 11 states. The
18  median reported volume per produced water spill was 990 gallons (3,750 L), more than double that
19  for spills of hydraulic fracturing fluids and chemicals. The causes of produced water spills were
20  reported as human error, equipment failure, container integrity failure, miscellaneous causes (e.g.,
21  well communication), and unknown causes. Most of the total volume spilled (74%) for all 225 cases
22  combined was caused by a failure of container integrity.

- ***What is the composition of hydraulic fracturing flowback and produced water, and what factors might influence this composition?***

23  A combination of factors influence the composition of produced water, including: the composition
24  of injected hydraulic fracturing fluids, the type of formation fractured, subsurface processes, and
25  residence time. The initial chemical composition of produced water primarily reflects the chemistry
26  of the injected fluids. At later times, the chemical composition of produced water reflects the
27  geochemistry of the fractured formation.

28  Produced water varies in quality from fresh to highly saline, and can contain high levels of major
29  anions and cations, metals, organics, and naturally occurring radionuclides. Produced water from
30  shale and tight gas formations typically contains high levels of total dissolved solids (TDS) and ionic
31  constituents (e.g., bromide, calcium, chloride, iron, potassium, manganese, magnesium, and
32  sodium). Produced water also may contain metals (e.g., barium, cadmium, chromium, lead, and

1  mercury), and organic compounds such as benzene. Produced water from coalbed methane
2  typically has much lower TDS levels compared to other produced water types, particularly if the
3  coalbed was deposited under fresh water conditions.

4  We identified 134 chemicals that have been detected in hydraulic fracturing produced water. These
5  include chemicals added during the chemical mixing stage, as well as naturally occurring organic
6  chemicals and radionuclides, metals, and other constituents of subsurface rock formations
7  mobilized by the hydraulic fracturing process. Data on measured chemical concentrations in
8  produced water were available for 75 of these 134 chemicals.

9  Most of the available data on produced water content are for shale and coalbed methane
10  formations, while less data are available for tight formations, such as sandstones. The composition
11  of produced water must be determined through sampling and analysis, both of which have
12  limitations—the former due to challenges in accessing production equipment, and the latter due to
13  difficulties identifying target analytes before analysis and the lack of appropriate analytical
14  methods. Most current data are for inorganic chemicals, while less data exist for organic chemicals.
15  Many more organic chemicals were reported as used in hydraulic fracturing fluid than have been
16  identified in produced water. The difference may be due to analytical limitations, limited study
17  scopes, and undocumented subsurface reactions.

- ***What are the chemical, physical, and toxicological properties of hydraulic fracturing flowback and produced water constituents?***

18  The identified constituents of produced water include inorganic chemicals (cations and anions, i.e.,
19  metals, metalloids, non-metals, and radioactive materials), organic chemicals and compounds, and
20  unidentified materials measured as total organic carbon and dissolved organic carbon. Some
21  constituents are readily transported with water (i.e., chloride and bromide), while others depend
22  strongly on the geochemical conditions in the receiving water body (i.e., radium and barium), and
23  assessment of their transport is based on site-specific factors. We were able to obtain actual or
24  estimated physicochemical properties for 86 (64%) of the 134 chemicals identified in produced
25  water.

26  As in the case of chemicals in hydraulic fracturing fluid, chemical properties that affect the
27  likelihood of an organic chemical in produced water reaching and impacting drinking water
28  resources include: mobility, solubility, and volatility. In general, physicochemical properties suggest
29  that organic chemicals in produced water tend to be less mobile in the environment. Consequently,
30  if spilled, these chemicals may remain in soils or sediments near spill sites. Low mobility may result
31  in smaller dissolved contaminant plumes in ground water, although these chemicals can be
32  transported with sediments in surface water or small particles in ground water. Organic chemical
33  properties vary with salinity, and effects depend on the nature of the chemical.

34  Oral reference values and/or oral slope factors from selected federal, state, and international
35  sources were available for 83 (62%) of the 134 chemicals detected in produced water. From U.S.
36  federal sources alone, chronic oral reference values were available for 70 (52%) of the 134
37  chemicals, and oral slope factors were available for 20 chemicals (15%). Of the chemicals that had

1  values available, noted health effects include the potential for carcinogenesis, immune system
2  effects, changes in body weight, changes in blood chemistry, pulmonary toxicity, neurotoxicity, liver
3  and kidney toxicity, and reproductive and developmental toxicity. As noted above, evaluating any
4  potential risk to human populations would require knowledge of the specific chemicals that are
5  present at a particular site, whether or not humans are exposed to those chemicals and, if so, at
6  what levels and for what duration, and the toxicity of the chemicals. The chemicals present in
7  produced water can vary based on the formation and specific well, due to differences in fracturing
8  fluid formulation and formation geology. Accordingly, consideration of hazards and risks associated
9  with these chemicals would be most useful on a  site-specific basis and is beyond the scope of this
10 assessment.  .

- ***If spills occur, how might hydraulic fracturing flowback and produced water contaminate drinking water resources?***

11 Impacts to drinking water resources from spills or releases of produced water depend on the
12 volume, timing, and composition of the produced water. Impacts are more likely the greater the
13 volume of the spill, the longer the duration of the release, and the higher the concentration of
14 produced water constituents (i.e., salts, naturally occurring radioactive material, and metals).

15 The EPA characterization of hydraulic fracturing-related spills found that 8% of the 225 produced
16 water spills included in the study reached surface water or ground water. These spills tended to be
17 of greater volume than spills that did not reach a water body. A well blowout in Bradford County,
18 Pennsylvania spilled an estimated 10,000 gal (38,000 L) of produced water into a tributary of
19 Towanda Creek, a state-designated trout fishery. The largest volume spill identified in this
20 assessment occurred in North Dakota, where approximately 2.9 million gal (11 million L) of
21 produced water spilled from a broken pipeline and impacted surface and ground water.

22 Chronic releases can and do occur from produced water disposed in unlined pits or impoundments,
23 and can have long-term impacts. Ground water impacts may persist longer than surface water
24 impacts because of lower flow rates and decreased mixing. Plumes from unlined pits used for
25 produced water have been shown to persist for long periods and extend to nearby surface water
26 bodies.

### 10.1.5. Wastewater Management and Waste Disposal (Chapter 8)

27 Hydraulic fracturing generates large volumes of produced water that require management. In this
28 section we refer to produced water and any other waters generated onsite by the single term
29 "wastewater." Clark and Veil (2009) estimated that in 2007 approximately one million active oil
30 and gas wells in the United States generated 2.4 billion gal per day (9.1 billion L per day) of
31 wastewater. There is currently no reliable way to estimate what fraction of this total volume can be
32 attributed to hydraulically fractured wells. Wastewater volumes in a region can increase sharply as
33 hydraulic fracturing activity increases.

34 Wastewater management and disposal could affect drinking water resources through multiple
35 mechanisms including: inadequate treatment of wastewater prior to discharge to a receiving water,
36 accidental releases during transport or leakage from wastewater storage pits, unpermitted

1  discharges, migration of constituents in wastewaters following land application, inappropriate
2  management of residual materials from treatment, or accumulation of wastewater constituents in
3  sediments near outfalls of centralized waste treatment facilities (CWTs) or publicly owned
4  treatment works (POTWs) that have treated hydraulic fracturing wastewater. The scope of this
5  assessment excludes potential impacts to drinking water from the disposal of hydraulic fracturing
6  wastewater in underground injection control (UIC) wells.

### *Research Questions: Wastewater Management and Waste Disposal*

- ***What are the common treatment and disposal methods for hydraulic fracturing wastewater, and where are these methods practiced?***

7  Hydraulic fracturing wastewater is managed using several options including disposal in UIC wells
8  (also called disposal wells); through evaporation ponds; treatment at CWTs, followed by reuse or
9  by discharge to either surface waters or POTWs; reuse with minimal or no treatment; and land
10  application or road spreading. Treatment of hydraulic fracturing wastewater by POTWs was used in
11  the past in Pennsylvania. This decreased sharply following new state-level requirements and a
12  request by the Pennsylvania Department of Environmental Protection (PA DEP) for well operators
13  to stop sending Marcellus Shale wastewater to POTWs (and 15 CWTs) discharging to surface
14  waters.

15  Wastewater management decisions are generally based on the availability and associated costs
16  (including transportation) of disposal or treatment facilities. A survey of state agencies found that,
17  in 2007, more than 98% of produced water from the oil and gas industry was managed via
18  underground injection ([Clark and Veil, 2009](#)). Available information suggests that disposal wells are
19  also the primary management practice for hydraulic fracturing wastewater in most regions in the
20  United States (e.g., the Barnett Shale; see Figure 10-1b). The Marcellus Shale region is a notable
21  exception, where most wastewater is reused because of the small number of disposal wells in
22  Pennsylvania (see Figure 10-1a). Although this assessment does not address potential effects on
23  drinking water resources from the use of disposal wells, any changes in cost of disposal or
24  availability of disposal wells would likely influence wastewater management decisions.

25  Wastewater from some hydraulic fracturing operations is sent to CWTs, which may discharge
26  treated wastewater to surface waters, POTWs, or back to well operators for reuse in other
27  hydraulic fracturing operations. Available data indicate that the use of CWTs for treating hydraulic
28  fracturing wastewater is greater in the Marcellus Shale region than other parts of the country. Most
29  of the CWTs accepting hydraulic fracturing wastewater in Pennsylvania cannot significantly reduce
30  TDS, and many of these facilities provide treated wastewater to well operators for reuse and do not
31  currently discharge treated wastewater to surface water.

32  Reuse of wastewater for subsequent hydraulic fracturing operations may require no treatment,
33  minimal treatment, or more extensive treatment. Operators reuse a substantial amount (ca. 70–
34  90%) of Marcellus Shale wastewater in Pennsylvania (see Figure 10-1a). Lesser amounts of reuse

1    occur in other areas (e.g., the Barnett Shale; see Figure 10-1b). In certain formations, such as the
2    Bakken Shale in North Dakota, there is currently no indication of appreciable reuse.

3    In some cases, wastewater is used for land applications such as irrigation or road spreading for
4    deicing or dust suppression. Land application has the potential to introduce wastewater
5    constituents to surface water and ground water due to runoff and migration of brines. Studies of
6    road spreading of conventional oil and gas brines have found elevated levels of metals in soils and
7    chloride in ground water.

- ***How effective are conventional POTWs and commercial treatment systems in removing organic and inorganic contaminants of concern in hydraulic fracturing wastewater?***

8    Publicly owned treatment works using basic treatment processes are not designed to effectively
9    reduce TDS concentrations in highly saline hydraulic fracturing wastewater—although specific
10   constituents or constituents groups can be removed (e.g., metals, oil, and grease by chemical
11   precipitation or other processes). In some cases, wastewater treated at CWTs may be sent to a
12   POTW for additional treatment and discharge. It is blended with POTW influent to prevent
13   detrimental effects on biological processes in the POTW that aid in the treatment of wastewater.

14   Centralized waste treatment facilities with advanced wastewater treatment options, such as
15   reverse osmosis, thermal distillation, or mechanical vapor recompression, reduce TDS
16   concentrations and can treat contaminants currently known to be in hydraulic fracturing
17   wastewater. However, there are limited data on the composition of hydraulic fracturing
18   wastewater, particularly for organic constituents. It is unknown whether advanced treatment
19   systems are effective at removing constituents that are generally not tested for.

- ***What are the potential impacts from surface water disposal of treated hydraulic fracturing wastewater on drinking water treatment facilities?***

20   Potential impacts to drinking water resources may occur if hydraulic fracturing wastewater is
21   inadequately treated and discharged to surface water. Inadequately treated hydraulic fracturing
22   wastewater may increase concentrations of TDS, bromide, chloride, and iodide in receiving waters.
23   In particular, bromide and iodide are precursors of disinfection byproducts (DBPs) that can form in
24   the presence of organic carbon in drinking water treatment plants or wastewater treatment plants.
25   Drinking water treatment plants are required to monitor for certain types of DBPs, because some
26   are toxic and can cause cancer.

27   Radionuclides can also be found in inadequately treated hydraulic fracturing wastewater from
28   certain shales, such as the Marcellus. A recent study by the PA DEP (2015b) found elevated radium
29   concentrations in the tens to thousands of picocuries per liter and gross alpha and gross beta in the
30   hundreds to thousands of picocuries per liter in effluent samples from some CWTs receiving oil and
31   gas wastewater. Radium, gross alpha, and gross beta were also detected in effluents from POTWs
32   receiving oil and gas wastewater (mainly as effluent from CWTs), though at lower concentrations
33   than from the CWTs. Research in Pennsylvania also indicates the accumulation of radium in
34   sediments and soils affected by the outfalls of some treatment plants that have handled oil and gas
35   wastewater, including Marcellus Shale wastewater, and other wastewaters (PA DEP, 2015b;

1    [Warner et al., 2013a](#)). Mobilization of radium from sediments and potential impacts on downstream
2    water quality depend upon how strongly the radium has sorbed to sediments. Impacts may also
3    occur if sediment is resuspended (e.g., following storm events). There is no evidence of radionuclide
4    contamination in drinking water intakes due to inadequately treated hydraulic fracturing
5    wastewater.

6    Hydraulic fracturing wastewaters contain other constituents such as barium, boron, and heavy
7    metals. Barium in particular has been documented in some shale gas produced waters. Little data
8    exist on metal and organic compound concentrations in untreated and treated wastewaters in
9    order to evaluate whether treatment is effective, and whether there are potential downstream
10   effects on drinking water resources when wastewater is treated and discharged.

## 10.2. Key Data Limitations and Uncertainties

11   This assessment used available data and literature to examine the potential impacts of hydraulic
12   fracturing for oil and gas on drinking water resources nationally. As part of this effort, we identified
13   data limitations and uncertainties associated with current information on hydraulic fracturing and
14   its potential to affect drinking water resources. In particular, data limitations preclude a
15   determination of the frequency of impacts with any certainty. There is a high degree of uncertainty
16   about whether the relatively few instances of impacts noted in this report are the result of a rarity
17   of effects or a lack of data. These limitations and uncertainties are discussed in brief below.

### 10.2.1. Limitations in monitoring data and chemical information

18   While many activities conducted as part of the hydraulic fracturing water cycle take place above
19   ground, hydraulic fracturing itself occurs below ground and is not directly observable. Additionally,
20   potential mechanisms identified in this assessment may result in impacts to drinking water
21   resources that are below ground (e.g., spilled fluids leaching into ground water). Because of this,
22   monitoring data are needed before, during, and after hydraulic fracturing to characterize the status
23   of the well being fractured and the presence, migration, or transformation of chemicals in the
24   subsurface. These data can include results from mechanical integrity tests performed on
25   hydraulically fractured oil and gas production wells and data on local water quality collected pre-
26   and post-hydraulic fracturing. In particular, baseline data on local water quality is needed to
27   quantify changes to drinking water resources and to provide insights into whether nearby
28   hydraulic fracturing activities may have caused any detected changes. The limited amount of data
29   collected before and during hydraulic fracturing activities reduces the ability to determine whether
30   hydraulic fracturing affected drinking water resources in cases of alleged contamination.

31   Water quality testing for hydraulic fracturing-related chemicals is routinely conducted for a small
32   subset of chemicals reportedly used in hydraulic fracturing fluids or detected in produced water.
33   Public water systems regularly test for selected contaminants under the National Primary Drinking
34   Water Regulations. Approximately 6% of the 1,173 chemicals in Table A-2 and Table A-4 are

1 routinely tested for under these regulations.[1] Private water wells are usually tested less often and
2 for fewer potential contaminants than public water supplies (USGS, 2014c). Since chemical use
3 varies widely across the country, testing for any particular chemical may or may not be appropriate
4 for detecting potential impacts on a drinking water resource from a nearby hydraulic fracturing
5 operation. Furthermore, the concentration, mobility, and detectability (as determined by the lowest
6 concentration that an analytical method is able to determine a chemical's presence) of chemicals
7 used in or produced by hydraulic fracturing operations will affect whether or not it would be
8 identified in a drinking water resource in the event of its release into the environment.

9 Information (identity, frequency of use, physicochemical and toxicological properties, etc.) on the
10 chemicals associated with the hydraulic fracturing water cycle is not complete and limits
11 understanding of potential impacts on drinking water resources. Well operators identified one or
12 more chemicals as confidential in approximately 70% of wells reported to FracFocus 1.0 and
13 analyzed by the EPA (U.S. EPA, 2015a). Additionally, chemicals found in flowback and produced
14 water (see Table A-4) were identified for a limited number of geographic locations and formations.
15 These characterization studies are constrained by available methods for detecting organic and
16 inorganic compounds in flowback and produced water. The identity of hydraulic fracturing-related
17 chemicals is necessary to understand their chemical, physical, and toxicological properties, which
18 determine how they might move through the environment to drinking water resources and any
19 resulting effects. Knowing their identities would also help inform what chemicals to test for in the
20 event of suspected drinking water impacts and, in the case of wastewater, may help predict
21 whether current treatment systems are effective at removing them.

22 Peer reviewed toxicity data for known hydraulic fracturing-related chemicals is very limited. Of the
23 1,173 hydraulic fracturing-related chemicals identified in Appendix A, 147 have chronic oral
24 reference values and/or oral slope factors from the sources that met the selection criteria for
25 inclusion in this assessment. Because the majority of chemicals identified in this report do not have
26 chronic oral reference values and/or oral slope factors, risk assessors at the local and regional level
27 may need to use alternative sources of toxicity information that could introduce greater
28 uncertainties. It also makes an assessment of potential cumulative effects of exposure to chemical
29 mixtures in hydraulic fracturing fluid, flowback, or produced water difficult.

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[1] We identified 73 chemicals that are reported to be used in hydraulic fracturing fluids (see Table A-2) or that have been detected in produced water (see Table A-4) that are tested for as part of the contaminant monitoring conducted for 40 different drinking water standards under the National Primary Drinking Water Regulations (NPDWR). For inorganic chemicals regulated under the NPDWR, we identified the chemical or element itself, its regulated ion (as applicable), or other more complex forms on the list of hydraulic fracturing-related chemicals. For regulated organic chemicals, we identified only the chemical itself on the list of hydraulic fracturing-related chemicals with three exceptions: (1) we identified all four trihalomethanes that comprise total trihalomethanes, (2) we identified two of the five regulated chlorinated/brominated haloacetic acids as their sodium salts, and (3) we identified a subset of polychlorinated biphenyls (PCBs) as Aroclor 1248. Although various forms of petroleum distillates are used in hydraulic fracturing fluids and may contain BTEX or benzo(a)pyrene (the regulated entities that can occur naturally in petroleum distillates), we did not include them in our count of 73 chemicals.

### 10.2.2. Other Contributing Limitations

1   We found other limitations that hamper our ability to assess the potential impacts of hydraulic
2   fracturing on drinking water resources nationally. These include the number and location of
3   hydraulically fractured wells, the location of drinking water resources, and information on industry
4   practices and any changes that may take place in practices in the coming years. Our estimates of the
5   number of fractured wells are based on an evaluation of several commercial and public sources and
6   a number of assumptions. This lack of a definitive well count particularly contributes to
7   uncertainties regarding total water use or total wastewater volume estimates, and would limit any
8   kind of cumulative impact assessment.

9   There are also some fundamental gaps in our understanding of drinking water resources, including
10   where they are located in relation to hydraulic fracturing operations and which might be most
11   vulnerable to impacts from hydraulic fracturing activities. Improving our assessment of potential
12   drinking water impacts requires better information, particularly about private drinking water well
13   locations and the depths of drinking water resources in relation to the hydraulically fractured
14   formations and well construction features (e.g., casing and cement). This information would allow
15   us to better assess whether subsurface drinking water resources are isolated from hydraulically
16   fractured oil and gas production wells.

17   Finally, this assessment summarizes available information on industry practices with respect to the
18   hydraulic fracturing water cycle. While some information on hydraulic fracturing activities is
19   available for many areas of the United States, specific data on water withdrawals for hydraulic
20   fracturing, volumes of flowback and produced water generated, and the disposal or reuse of
21   wastewaters is needed to better characterize potential impacts of hydraulic fracturing on drinking
22   water resources. Additionally, industry practices are rapidly changing (e.g., the number of wells
23   fractured, the location of activities, and the chemicals used), and it is unclear how changes in
24   industry practices could affect potential drinking water impacts in the future. Consideration of
25   future development scenarios was not a part of this assessment, but such an evaluation could help
26   establish potential short- and long-term impacts to drinking water resources and how to assess
27   them.

## 10.3. Conclusions

28   Through this national-level assessment, we have identified potential mechanisms by which
29   hydraulic fracturing could affect drinking water resources. Above ground mechanisms can affect
30   surface and ground water resources and include water withdrawals at times or in locations of low
31   water availability, spills of hydraulic fracturing fluid and chemicals or produced water, and
32   inadequate treatment and discharge of hydraulic fracturing wastewater. Below ground mechanisms
33   include movement of liquids and gases via the production well into underground drinking water
34   resources and movement of liquids and gases from the fracture zone to these resources via
35   pathways in subsurface rock formations.

36   We did not find evidence that these mechanisms have led to widespread, systemic impacts on
37   drinking water resources in the United States. Of the potential mechanisms identified in this report,

1    we found specific instances where one or more of these mechanisms led to impacts on drinking
2    water resources, including contamination of drinking water wells. The cases occurred during both
3    routine activities and accidents and have resulted in impacts to surface or ground water. Spills of
4    hydraulic fracturing fluid and produced water in certain cases have reached drinking water
5    resources, both surface and ground water. Discharge of treated hydraulic fracturing wastewater has
6    increased contaminant concentrations in receiving surface waters. Below ground movement of
7    fluids, including gas, most likely via the production well, have contaminated drinking water
8    resources. In some cases, hydraulic fracturing fluids have also been directly injected into drinking
9    water resources, as defined in this assessment, to produce oil or gas that co-exists in those
10   formations.

11   The number of identified cases where drinking water resources were impacted are small relative to
12   the number of hydraulically fractured wells. This could reflect a rarity of effects on drinking water
13   resources, or may be an underestimate as a result of several factors. There is insufficient pre- and
14   post-hydraulic fracturing data on the quality of drinking water resources. This inhibits a
15   determination of the frequency of impacts. Other limiting factors include the presence of other
16   causes of contamination, the short duration of existing studies, and inaccessible information related
17   to hydraulic fracturing activities.

## 10.4. Use of the Assessment

18   The practice of hydraulic fracturing is simultaneously expanding and changing rapidly. Over 60% of
19   new oil and gas wells are likely to be hydraulically fractured, and this percentage may be over 90%
20   in some locations. Economic forces are likely to cause short term volatility in the number of wells
21   drilled and fractured, yet hydraulic fracturing is expected to continue to expand and drive an
22   increase in domestic oil and gas production in coming decades (EIA, 2014a).[1] As a result, hydraulic
23   fracturing will likely increase in existing locations, while also potentially expanding to new areas.

24   This state-of-the-science assessment contributes to the understanding of the potential impacts of
25   hydraulic fracturing on drinking water resources and the factors that may influence those impacts.
26   The findings in this assessment can be used by federal, state, tribal, and local officials; industry; and
27   the public to better understand and address any vulnerabilities of drinking water resources to
28   hydraulic fracturing activities. This assessment can also be used to help facilitate and inform
29   dialogue among interested stakeholders, and support future efforts, including: providing context to
30   site-specific exposure or risk assessments, local and regional public health assessments, and to
31   assessments of cumulative impacts of hydraulic fracturing on drinking water resources over time
32   or over defined geographic areas of interest.

33   We hope the identification of limitations and uncertainties will promote greater attention to these
34   areas through pre- and post- hydraulic fracturing monitoring programs and by researchers. We also

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[1] In their reference case projections, the U.S. Energy Information Administration (EIA) forecasts that U.S. gas production by 2035 will have increased 50% over 2012 levels. Crude oil production is projected to increase almost 40% above current levels by 2025, before declining in subsequent decades (EIA, 2014a).

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*This document is a draft for review purposes only and does not constitute Agency policy.*

1    hope it will lead to greater dissemination of data in forms accessible by a wide-range of researchers
2    and audiences.

3    Finally, and most importantly, this assessment advances the scientific basis for decisions by federal,
4    state, tribal, and local officials; industry; and the public, on how best to protect drinking water
5    resources now and in the future.

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