(e.g., salts, metals, metalloids, and radionuclides) will not degrade. In addition, pits or impoundments that have reached the end of their useful life have accumulated residuals. Practices used to decommission these pits include draining and leveling the pit in place or land farming the residual materials into the ground (Rich and Crosby, 2013), although more information is needed on the potential for these practices to affect water resources.

A particular concern for the management of residual wastes is TENORM that originates from the geologic formation and was present in the produced water (SAFER PA, 2015). Studies have found TENORM in solid residuals at oil and gas operations including the filter cake (PA DEP, 2015b), filter socks (Harto et al., 2014), and pit sludges (Rich and Crosby, 2013). Researchers have assessed Marcellus produced water samples, finding that many with low barium and high radium-226 levels would generate sludges that exceed the maximum acceptable radium-226 activity for nonhazardous landfill disposal in Pennsylvania (Silva et al., 2014b; Silva et al., 2014a). In scales that build up on hydraulic fracturing and treatment equipment and sludges that accumulate in tanks and pits, radium can coprecipitate with barium, strontium, or calcium sulfates (Smith et al., 1999). (See Section 8.5.2 for additional discussion of TENORM associated with residuals.)

The accumulation of TENORM in the solids generated can limit or preclude landfills as a disposal option. Walter et al. (2012) point out that wastes containing TENORM can be problematic due to the possibility of radon emissions from the landfill. Regulatory limits on permissible radionuclide levels accepted at non-hazardous landfills vary by state (Silva et al., 2014a).[1] Some states have volumetric limitations on TENORM in their landfill permits (e.g., Colorado). Also, some states write criteria, such as gamma exposure rates (radiation) and radioactivity concentration limits, into permits for many landfills that are permitted to accept TENORM. Silva et al. (2014a) note that there are 50 nonhazardous (RCRA-D) disposal facilities in Pennsylvania, but no TENORM disposal facilities. Texas and other states have disposal facilities for TENORM.

#### 8.4.7.2 Liquid Residuals

Liquid residuals include concentrated brines (from membrane or evaporation processes) and regeneration or cleaning chemicals (from ion exchange, adsorption, and membrane processes) (Fakhru'l-Razi et al., 2009). Practices for managing liquid residual streams from treatment processes are generally the same as for untreated hydraulic fracturing wastewaters, although the treated volumes are smaller, resulting in lower costs (Hammer and VanBriesen, 2012). Concentrations of contaminants in liquid residuals, however, will be higher. The most common disposal method is injection into disposal wells.

If the liquid is not injected into a disposal well, treatment to remove salts would be required for surface water discharge to meet NPDES permit requirements and protect the water quality for downstream users such as drinking water utilities (Section 8.5). Because some constituents of concentrated liquid residual waste streams can pass through or impact municipal wastewater treatment processes (Linarić et al., 2013; Hammer and VanBriesen, 2012), these residuals would

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[1] Examples of permissible radionuclide levels at non-hazardous landfills: Pennsylvania requires alarms to be set at all municipal landfills, with a trigger set at 10 μR/hr above background radiation. Texas sets a radioactivity limit, requiring that any waste disposed by burial contains less than 30 pCi/g radium or 150 pCi/g of other radionuclides.

not be appropriate for discharge to a POTW. Elevated salt concentrations, in particular, can have detrimental effects on microbiological treatment at municipal wastewater systems, such as activated sludge treatment (Linarić et al., 2013).

Liquid residuals can also be mixed with a solidifying agent such as Portland cement and then disposed of in landfills, or they can undergo advanced treatment processes to generate products such as road salt or industrial chemicals (SAFER PA, 2015).

### 8.4.7.3 Potential Impacts from Solid and Liquid Residuals

Residual wastes have the potential to impact the quality of drinking water resources if contaminants leach to groundwater or reach surface water. In a recent study by PA DEP, radium was detected in leachate from 34 of 51 landfills that accept waste from the oil and gas industry (Marcellus in particular). Radium-226 concentrations ranged from 54 to 416 pCi/L, and radium-228 ranged from 2.5 to 1,100 pCi/L (PA DEP, 2015b). (See also Section 8.5.2 and see Chapter 9 for health effects associated with radium). Countess et al. (2014) studied the potential for a wide array of elements to leach from sludges generated at a CWT handling hydraulic fracturing wastewater in Pennsylvania. Tests used strong acid solutions (to simulate the worst case scenario) and weak acid digestions (to simulate environmental conditions). The data illustrate the possibility of leaching of these constituents from landfills. The extent of leaching varied by constituent and by fluid type, but the authors concluded that boron, bromide, calcium, magnesium, manganese, silicon, sodium, and strontium had high potential to migrate from the residual solids, with bromide and sodium having the highest leaching potential (Countess et al., 2014). (See also Section 5.8 in Chapter 5 for discussion of the processes governing the movement of constituents in the subsurface.)

In another study assessing the leaching behavior of residuals from hydraulic fracturing operations, Sharma et al. (2015) found that alkali metals, alkaline earth metals, and bromide had the highest leaching potential of the constituents tested. The authors also found that disposing of hydraulic fracturing residuals along with other solids (e.g., at a municipal landfill) produces a greater leaching potential than if the residuals are disposed of by burying or land disposal designed for solely the hydraulic fracturing residuals. This is due to the more acidic leachate formed at the co-disposal locations (Sharma et al., 2015).

Sang et al. (2014) studied the potential for hydraulic fracturing fluid to mobilize colloidal particles in the soil. The study used microspheres and sand particles as surrogates for contaminant particles. The authors note that the chemistry of hydraulic fracturing fluid favors transport of colloids and mineral particles through rock cracks, and they found that infiltration of flowback fluid can transport existing pollutants such as heavy metals, radionuclides, and pathogens, in unsaturated soils (Sang et al., 2014). Heavy metals can also move through soil. Although not specific to hydraulic fracturing wastes, Camobreco et al. (1996) report high levels of heavy metal transport in soil columns, with 12% recovery for lead, 15% for copper, 23% for zinc, and 30% for cadmium (Camobreco et al., 1996).

Residuals, whether liquid or solid, are the most concentrated wastes and waste streams associated with hydraulic fracturing operations. Contaminants in the produced water will accumulate in the sludges in storage tanks/pits, in scale on the equipment, and in treatment facilities. Proper management and disposal of these highly concentrated wastes is critical to minimize the potential for impacts on water resources.

## 8.5 Potential Impacts of Hydraulic Fracturing Wastewater Constituents on Drinking Water Resources

The previous section discussed the potential impacts of specific wastewater management strategies on drinking water resources. The severity of impacts, however, depends largely on the constituents in the wastewater, the concentrations of those constituents, and their health and ecological effects. This section will discuss the potential impacts of several specific types of hydraulic fracturing wastewater constituents on drinking water resources.

The impacts or potential impacts discussed in the literature are heavily focused on discharges from CWTs, including treated wastewater that is discharged indirectly through POTWs. Available evidence suggests that the effects of hydraulic fracturing on surface water quality are related to discharges of partially treated wastewater (Kuwayama et al., 2015a). Other avenues of contamination for both surface water and groundwater include leaks from pits and impoundments, landfill leachate, and leaching from contaminated sediments and other improperly managed solid wastes.

As noted, an important consideration regarding the potential impacts of hydraulic fracturing wastewater on receiving water is whether there are constituents of concern known to have health effects or that can give rise to compounds with health effects. See Chapter 9 for discussion of the health effects of wastewater constituents. For some classes of constituents, such as DBP precursors, considerable research exists regarding concentrations in the waste stream and impacts on downstream drinking water treatment plants or the finished drinking water after treatment. For other constituents, information is limited, especially within the context of hydraulic fracturing activities. There may also be unknown constituents because some ingredients in the original hydraulic fracturing fluids are claimed to be CBI. The following subsections identify several classes of constituents known to occur in hydraulic fracturing wastewater, discuss whether potential impacts are likely, and detail information gaps.

### 8.5.1 Bromide, Iodide, and Chloride

Halides, including bromide, chloride, and iodide, are commonly found in high-TDS hydraulic fracturing wastewater. As noted in Section 8.3.1.1, chloride is a regulated contaminant with a secondary MCL of 250 mg/L. Bromide and iodide are not regulated, but are of concern due to their role in the formation of DBPs (Parker et al., 2014; Krasner, 2009). (See Appendix F for information on DBP formation.) High-TDS wastewaters from the Marcellus Shale have been the focus of concern due to the state's history of treating these wastewaters at POTWs (without pretreatment) and at CWTs without TDS removal capabilities (Text Box 8-1). Discontinuing the practice of sending shale gas wastewater to POTWs without pretreatment (States et al., 2013), and compliance with the new EPA pretreatment standards for discharges of unconventional oil and gas wastewaters helps mitigate this problem. This section describes the role of some constituents in high-TDS fluids in the formation of DBPs and provides more details on the effects on surface waters as observed in Pennsylvania. The lessons learned and steps taken in the Marcellus region can provide valuable knowledge for operators and state agencies in other parts of the country that treat and discharge high-bromide and high-iodide wastewaters.

#### 8.5.1.1 Influence of Bromide and Iodide on Formation of Disinfection Byproducts

Disinfection byproducts (DBPs) are formed when organic material comes in contact with disinfectants (e.g., chlorine, chloramine, chlorine dioxide, or ozone). Of particular concern are DBPs formed in the presence of halides (e.g., bromide or iodide). The type of DBP formed depends on the organic precursors in the source water and the disinfectant used. Regulated DBPs include total trihalomethanes (TTHM), five haloacetic acids (HAA5), bromate, and chlorite. There are, however, many additional DBPs that are not regulated and may in fact be of greater concern than the regulated species. Brominated forms of DBPs, for example, are considered to be more toxic and carcinogenic than chlorinated species (McGuire et al., 2014; Parker et al., 2014; States et al., 2013; Krasner, 2009; Richardson et al., 2007). Another halide, iodide is also found in some hydraulic fracturing wastewater (Chapter 7), and although its effects have not been as well documented as those associated with bromide, iodide raises some of the same concerns regarding formation of toxic DBPs as bromide (Xu et al., 2008).

Studies have found that elevated bromide levels in water correlate with increased DBP formation in the drinking water that is delivered to customers (also called "finished drinking water") (Obolensky and Singer, 2008; Matamoros et al., 2007; Hua et al., 2006; Yang and Shang, 2004). Harkness et al. (2015) studied the chemical composition of flowback, produced waters, treated wastewaters, instream flows downstream from discharges, and accidental spill sites. The study found high concentrations of bromide and iodide in the flowback and produced waters, concluding that the elevated levels of these constituents could promote the formation of toxic brominated and iodinated DBPs in downstream drinking water systems (Harkness et al., 2015).

In terms of the resulting DBP formation, laboratory experiments using hydraulic fracturing wastewater from the Marcellus and Fayetteville shales and river water from the Allegheny and Ohio rivers suggest that a relatively small portion of hydraulic fracturing wastewater can notably affect DBP formation (Parker et al., 2014). In particular, trihalomethanes (THM; a category of DBPs) were shown to shift towards greater brominated and iodinated species with a little as 0.01% hydraulic fracturing wastewater in disinfected samples. Modeling work by Landis et al. (2016) evaluated the impact of CWT discharges on DBP formation at a drinking water system and suggested that although only a 3% increase in overall TTHM formation was predicted, the model predicted a decrease in chlorinated THM and a substantial shift toward a higher percentage of the more-toxic brominated THMs (Landis et al., 2016).

States et al. (2013) found a strong correlation between bromide concentrations in source water from the Allegheny River in Pennsylvania and the percentage of brominated THMs in finished water at a drinking water facility using Allegheny source water. Bromide concentrations in the river water measured during the study ranged from less than 25 µg/L to 299 µg/L. The authors noted that source water containing 50 µg/L of bromide resulted in treated drinking water with approximately

62% of total THMs consisting of brominated species. When the source water contained 150 μg/L bromide, the brominated THM percentage was 83% (States et al., 2013).

Pope et al. (2007) reported that increased bromide levels are the second best indicator of DBP formation, with pH being the first. Furthermore, bromine (which may be formed from bromide in the water during disinfection) reacts as much as ten times faster and more efficiently with DBP precursors than chlorine (Westerhoff et al., 2004). These studies show that increased bromide concentration in a drinking water resource shifts the DBP formation towards more-toxic brominated forms.

If disinfection is accomplished using ozonation instead of or in addition to chloramination or chlorination, bromide and iodide in the source water can form two additional constituents: bromate and iodate. Iodate, although formed during disinfection by ozonation, is not considered a DBP and is non-toxic (Allard et al., 2013). Bromate, however, is a DBP of concern and has an MCL of 0.010 mg/L (U.S. EPA, 1998).

Another category of DBP that is not regulated is the nitrogenous DBPs, including nitrosamines. Data are lacking on the formation of nitrogenous DBPs specifically linked to hydraulic fracturing wastewater, but their formation is possible. During chloramination, bromide can enhance the formation of the nitrosamine N-nitrosodimethylamine (NDMA) in waters containing the precursor dimethylamine (DMA) (Le Roux et al., 2012; Luh and Mariñas, 2012). As with some other non-regulated DBPs, nitrogenous DBPs may be more toxic than the regulated ones (Harkness et al., 2015; McGuire et al., 2014; Parker et al., 2014).

As discussed in Section 8.4 and Text Box 8-3, removal of dissolved solids, including chloride and bromide, requires advanced treatment processes such as reverse osmosis (RO), distillation, evaporation, or crystallization. Unless the treatment plant receiving the high-TDS wastewater employs processes specifically designed to remove these constituents, effluent discharge may contain high levels of bromide and chloride. Drinking water systems with intakes downstream of these discharges may receive water with correspondingly higher levels of bromide and chloride and may have difficulty complying with SDWA regulations related to DBPs.

### 8.5.1.2 Effects on Receiving Waters

Studies show that discharges from oil and gas wastewater treatment facilities can elevate TDS, bromide, and chloride levels in receiving waters, and potential impacts may be detectable far downstream (> 1km) of an outfall (States et al., 2013; Warner et al., 2013a; Wilson and Van Briesen, 2013). The work by Landis et al. (2016) in the Allegheny River mentioned above is consistent with these findings. The authors studied the impacts of a CWT accepting oil and gas wastewater on water quality at a downstream drinking water intake. They found that compared to data from upstream (background) locations, bromide concentrations at the intake were increased by 53% at low streamflow and 22% during high streamflow.[1]

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[1] Background samples are those taken from locations upstream of, and therefore unaffected by, permitted facilities.

Elevated TDS, chloride, and bromide can serve as indicators of potential influence from hydraulic fracturing wastewater in surface water and can also raise concerns about DBP formation in downstream drinking water systems. Elevation of bromide has been shown to place a burden on downstream drinking water systems. The Pittsburgh Water and Sewer authority (PWSA) drinking water system concluded that elevated bromide in their source water led to elevated TTHMs in their finished drinking water, with a substantial increase in the percentage of brominated TTHMs (States et al., 2013). The utility modified their treatment process and proposed improvements to their storage facilities to address the elevated TTHM levels in the distribution system (Chester Engineers, 2012).

Conversely, changes in regional wastewater handling that reduce bromide discharges can be reflected in receiving waters. A three-year study at water intakes downstream of wastewater discharges on the Monongahela River in western Pennsylvania evaluated water chemistry in the context of flow measurements. The authors concluded that an overall decrease in bromide concentrations at drinking water intakes from 2010 to 2012 was likely associated with shale gas operators voluntarily ceasing the practice of sending high-bromide wastewaters to treatment facilities that discharge to surface waters without adequate TDS removal (Wilson and Van Briesen, 2013).

Elevated TDS and halides need to be interpreted in the context of other inputs into a watershed. An EPA source apportionment study of the Allegheny River in Pennsylvania (U.S. EPA, 2015o) found that CWTs accounted for almost 90% of the bromide at one drinking water treatment plant intake and 37% of the bromide at another intake. Other sources include coal-fired power plants and acid mine drainage. Furthermore, although effluent is diluted when discharged to a water body, this may not always be sufficient to avoid water quality problems if there are existing pollutant loads in the waterbody from other contributors (e.g., such as acid mine drainage or power plant effluent) (Ferrar et al., 2013). Warner et al. (2013a) evaluated effluent from the Josephine Brine Treatment Facility, which treated both conventional and unconventional (as defined by PA DEP) oil and gas wastewater at the time of the study. The authors concluded that even a 500 to 3,000-fold dilution of the wastewater would not reduce bromide levels to background. Modeling by Weaver et al. (2016) suggests that bromide levels in receiving streams can be improved by reducing concentrations in the effluent, discharging during periods of high streamflow, and discharging intermittently (pulsing). (See Appendix F for additional description of modeling studies.)

In addition to concerns about formation of DBPs within downstream drinking water systems, treatment at the upstream CWTs and POTWs themselves can also produce DBPs if the facilities disinfect prior to discharge. The DBPs may then be released into receiving waters and increase concerns about the total loads of brominated and iodinated DBPs at downstream drinking water systems (Hladik et al., 2014). A study by Hladik et al. (2014) documented brominated and iodinated DBPs at the outfalls of CWTs and POTWs treating both conventional and unconventional wastewater and noted that this DBP signature was different than for those plants that did not accept oil and gas wastewater.

#### 8.5.1.3 Other Constituents That Can Affect Downstream DBP Formation

In addition to halogens, organic matter and ammonium can also be present in hydraulic fracturing wastewater (Chapter 7; Appendix E) and can have an influence on the formation of DBPs at downstream drinking water systems (Harkness et al., 2015). Experimental work by Parker et al. (2014) found that a mixture of river water with 1-2% flowback by volume could contribute to DBP formation due to the higher dissolved organic carbon content of the flowback. Harkness et al. (2015) studied the chemical composition of water associated with oil and gas production and found high concentrations of ammonium in the flowback and produced waters. Elevated levels of ammonium can convert chlorine to chloramines at downstream drinking water treatment plants. This could have an impact on the plant's disinfection practices because chloramines are a weaker disinfectant than chlorine (Harkness et al., 2015; Parker et al., 2014).

#### 8.5.1.4 Mitigating Impacts from TDS and Halides on Drinking Water Utilities

High bromide concentrations and low flow conditions in waterways have been shown to increase DBP formation in downstream drinking water systems (States et al., 2013). Most drinking water treatment plants are not designed to address high concentrations of TDS (including bromide and iodide), limiting their options for restricting the formation of brominated and iodinated DBPs when these halides are present.

To mitigate these impacts, one strategy that was implemented in Pennsylvania was to disallow influent of high-TDS wastewaters to POTWs and CWTs that discharged to streams and were not designed to treat for TDS. Wilson and Van Briesen (2013) showed that this strategy was effective for reducing bromide concentrations at drinking water utilities downstream from POTWs and CWTs that had formerly accepted hydraulic fracturing wastewaters (States et al., 2013; Warner et al., 2013a; Wilson and Van Briesen, 2013). Alternatively, advanced treatment processes such as reverse osmosis, distillation, evaporation, and crystallization, can be employed to reduce constituents that contribute to high TDS (e.g., such as chloride, bromide, and iodide), reducing impacts on surface waters and, subsequently, downstream drinking water utilities. Strategies such as discharging during higher streamflow periods and using a pulsing or intermittent discharge could also reduce the frequency and severity of potential impacts on drinking water systems from elevated TDS.

### 8.5.2 Radionuclides

Potential impacts on drinking water resources from TENORM associated with hydraulic fracturing wastewater can arise through a number of pathways, including: treated wastewater in which radionuclides were not adequately removed; accumulation of radionuclides in surface water sediments downstream of wastewater treatment plant discharge points; migration or mobilization from soils that have accumulated radionuclides from previous activities such as pits or land application; and inadequate management of treatment plant solids (such as filter cake), landfill leachate, or sediments in pits or tanks that have accumulated radionuclides.

An additional concern is the potential for underestimation of radium concentrations in hydraulic fracturing wastewater due to the high TDS content. When using wet chemical techniques, high TDS concentrations can result in poor recovery of some chemical constituents. For radium, recovery may be as low as <1% in a high-salt matrix (Nelson et al., 2014). This may lead to the inability to identify an impact on drinking water resources or an underestimation of the severity of an impact. Research suggests that spectroscopic methods are more appropriate for analysis of radium in high-TDS wastewaters (Nelson et al., 2014).

A recent study by the PA DEP (PA DEP, 2015b) provides information that helps fill a general data gap regarding TENORM content in oil and gas wastes that are treated and discharged to surface waters. The study, although not exclusive to Marcellus wastes, was motivated by concerns over an increase in radionuclides in oil and gas wastes observed during the expansion of Marcellus Shale production. The study began in 2013 and quantified radionuclide (radium-226, radium-228, K-40, gross alpha, and gross beta) levels at CWTs, POTWs, well sites, and landfills and discussed human health and environmental implications. Other relevant studies addressing radionuclides focus on CWTs that have handled Marcellus wastewater, evaluation of solids in storage pits, and analysis of scale on pipes and tanks.

#### 8.5.2.1 Effluent from POTWs

In Pennsylvania between 2007 and 2010, TENORM-bearing wastewaters were sent to POTWs, which are generally not required to monitor for radioactivity (Resnikoff et al., 2010). Although management of Marcellus wastewaters via POTWs has declined, there is still potential for input of radionuclides to surface waters via discharge of CWT effluent either directly to surface water or indirectly through discharge to POTWs. The potential for TENORM to pass through treatment at POTWs is one of the concerns addressed in the EPA's recently promulgated pretreatment standards for unconventional oil and gas wastewaters that discharge to POTWs.

Six of the POTWs in the PA DEP TENORM study received effluent from a CWT along with municipal wastewater. Note that the CWTs in the study are not described as receiving exclusively Marcellus wastewater. The POTWs that receive both CWT effluent and municipal waste had radium in their effluent (overall average effluent radium-226 concentration of 103 pCi/L, with a range of <35 to 340 pCi/L). Those POTWs receiving only municipal wastewater also contained radium, with an average effluent radium-226 concentration of 145 pCi/L.[1] These concentrations are many times higher than the MCL for radium (5 pCi/L) and are also orders of magnitude higher than typical background values; radium-226 in river water generally ranges from 0.014 pCi/L to 0.54 pCi/L (IAEA, 2014).[2]

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[1] These values are for unfiltered samples. In filtered samples, the POTWs that receive both CWT effluent and municipal waste had higher average radium-226 values than those for POTWs only treating municipal waste (497 pCi/L vs. 85 piCi/L). Filtered samples are passed through a filter to remove fine particles; concentrations of constituents in filtered samples are often lower than in unfiltered samples. However, liquid samples in this study were filtered after preservation with acid. Therefore, the difference between unfiltered and filtered samples may not be reliable.

[2] A confounding issue for this study is that it was not clear why the radium-226 concentrations were comparable or higher for those POTWs not receiving oil and gas CWT effluent. However, sample sizes were small and possible alternative sources for the radium were not discussed. The report also did not describe how it was verified that the POTWs did not receive contributions from oil and gas wastewater.

### 8.5.2.2 Effluent from CWTs

Four of the ten CWTs sampled during the PA DEP TENORM study (PA DEP, 2015b) discharged to surface water under a National Pollution Discharge Elimination System (NDPES) permit, and the other six discharged to POTWs. The average radium-226 concentration in the effluent from the CWTs (1,840 pCi/L for unfiltered samples) was an order of magnitude higher than in effluent from the POTWs. Samples of treated wastewater from zero-discharge facilities contained higher concentrations, averaging 2,610 pCi/L radium-226 and 295 pCi/L radium-228 (PA DEP, 2015b). The treated wastewater from these zero-discharge facilities will likely be reused for subsequent hydraulic fracturing jobs, postponing the need for disposal, but reuse could result in overall increases in some constituents of concern due to repeated passage through the subsurface. In addition, there is also a potential for impacts on drinking water resources from spills and leaks associated with wastewater storage and handling at these facilities.

Sampling done at the Josephine Brine Treatment Plant in western PA from 2010 – 2012 (Warner et al., 2013a) detected radium in the effluent (mean values of 4 pCi/L of radium-226 and 2 pCi/L of radium-228). Treatment at the facility removes radium by coprecipitation with barium sulfate. The authors note that if the activities of radium-226 and radium-228 in Marcellus brine influent at the CWT are similar to those reported by other researchers (Rowan et al., 2011), then the CWT achieved a 1,000-fold reduction in radium content. (This facility also accepted conventional oil and gas wastewater.) The detection of radium in the effluent from this CWT suggests that if the influent concentration is extremely high, radium will still be found in the effluent of a treatment plant even if the treatment process removes a high percentage (see Section 8.4 and Appendix F for additional discussion on constituent removal efficiencies at CWTs).

### 8.5.2.3 Accumulation in Sediments

In addition to concerns about TENORM in discharges to surface waters, studies have shown the potential for a legacy of radionuclide accumulation in surface water sediments. The PA DEP TENORM study (PA DEP, 2015b) found radium in sediments near the outfalls for CWTs (averages of 84.2 pCi/g and 19.8 pCi/g for radium-226 and -228, respectively) and three POTWs receiving treated oil and gas wastewater from CWTs (radium-226 and radium-228 concentrations ranging from 1.8 to 18.2 pCi/g). Typical background soil levels of radium are approximately 1 to 2 pCi/g (PA DEP, 2015b).

Warner et al. (2013a) measured radium-226 levels in stream sediment samples at the point of discharge of a CWT that had treated both conventional oil and gas wastewater and unconventional Marcellus wastewater. They found concentrations approximately 200 times greater than upstream and background sediments. This indicates the potential for accumulation of contaminants in localized areas near wastewater discharge facilities. Although the CWT studied by Warner et al. (2013a) also accepted conventional oil and gas wastewater, the authors observed that the radium-228/radium-226 ratio in the river sediments near the discharge (0.22 – 0.27) is consistent with ratios in Marcellus wastewater. The authors indicate that the radium likely accumulated in the sediments, originating from the discharge of treated unconventional Marcellus oil and gas wastewater. Accumulation of TENORM can also occur in sediments receiving discharged effluent from landfills that accept oil and gas wastes. In the PA DEP TENORM study (PA DEP, 2015b), samples of impacted soils were collected at three landfill outfalls. Radium-226 and -228 were detected in all samples (2.82 to 4.46 pCi/g and 0.979 to 2.53 pCi/g, respectively).

A study by Skalak et al. (2014), on the other hand, did not find elevated levels of alkali earth metals (including radium) in sediments just downstream of the discharge points of five POTWs that had previously treated Marcellus wastewater. These inconsistencies among studies suggest that accumulation of contaminants in sediment may depend on treatment processes and their removal rates for each constituent as well as stream chemistry and hydrologic characteristics. Contamination with radium-226 would potentially be long lived because of the long half-life of radium.[1]

The association of radium with sediments near discharge points is attributed to adsorption of radium to the sediments, a process governed by factors such as the salinity of the water and sediment characteristics. Increased salinity promotes desorption of radium from sediments, while lower salinity promotes adsorption, with radium adsorbing particularly strongly to sediments high in iron and manganese (hydr)oxides (Porcelli et al., 2014; Gonneea et al., 2008). Warner et al. (2013a) speculate that when saline CWT effluent is discharged into stream water, the lower salinity of the stream environment facilitates sorption of radium onto streambed sediments. The long-term fate of radium sorbed to sediments depends upon changes in water salinity and the sediment properties, including any reduction/oxidation chemical reactions that affect iron and manganese minerals in the sediments. Additionally, the sediment may be physically transported downstream due to high flows or if sediment is disturbed and resuspended.

### 8.5.2.4 Pits and Tanks

Where pits or impoundments are used, radionuclides may accumulate in the bottom sludges and can also be found in soils once the pit is closed and leveled. A study of three centralized wastewater storage impoundments in southwestern Pennsylvania (Zhang et al., 2015a) showed that radium-226 accumulated in various components of the bottom solids, including through coprecipitation with barium sulfate. Sludge from one pit showed a substantial increase in radium-226 between sampling events 2.5 years apart (from 8.8 pCi/g to 872 pCi/g). The authors attributed the steep increase to enrichment in radium during cycles of wastewater reuse. In Texas, accumulation of radionuclides (potassium, thorium, bismuth, radium, and lead) was documented for two pits that stored fluids associated with hydraulic fracturing (Rich and Crosby, 2013). One pit was decommissioned and used as farmland, and the other was active at the time of sampling. Analyses of soil and sludge samples detected a number of radionuclides, including radium-226, radium-228, thorium-228, strontium-90, and potassium-40 (radium-226 was only found at the former pit). Rich and Crosby (2013) note a total beta radiation value of 1,329 pCi/L in one sample from the active pit. They note that this value exceeded regulatory guidelines even though the values for individual

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[1] The half-life of radium-226 is approximately 1,600 years, while the half-life of radium-228 is 5.76 years. The half-life is the time it takes for half of the nuclei in a sample of a radioactive element to decay. After two half-lives, one fourth of the original sample will be left, and after three half-lives there will be one eighth of the original sample remaining, and so forth.

radionuclides did not exceed regulatory guidelines, suggesting that using a single radionuclide (i.e., radium) as an indication of exposure can underestimate total radioactivity.

Although the sample sizes were small for both the Zhang et al. (2015a) and the Rich and Crosby (2013) studies, the results suggest that radionuclides associated with sediments from some pits could have potential impacts on surface water or groundwater. These studies illustrate the need for appropriate management where wastes have high TENORM content. Rich and Crosby (2013) note that pits are often found in agricultural regions. If pit solids that are incorporated into soils (e.g., during draining and leveling or during land application) contain radionuclides, they may reach surface water in runoff or leach from the solids and migrate to groundwater. In active pits, Rich and Crosby (2013) note that TENORM in the contents may be deposited onto crops and soil through aerosolization or breaching. The Pennsylvania study (Zhang et al., 2015a) suggests that landfill leachate may be affected by receiving sludges from impoundments that store produced water and will need to be managed appropriately.

With radium-226 values of 121 pCi/g and 872 pCi/g, sludges from the pits studied by Zhang et al. (2015a) exceeded the limit for disposal as a nonhazardous solid in a municipal or industrial solid waste landfill but would meet the radium-226 limits for disposal in a hazardous waste landfill. There are currently no federal requirements to test solid residuals for radionuclides before disposal. At landfills studied in the PA DEP TENORM report (PA DEP, 2015b), seven samples of treated effluent from nine facilities that accept oil and gas waste had radium-226 values ranging from 105 pCi/L to 378 pCi/L and radium-228 values ranging from <6 pCi/L to 1,100 pCi/L. Untreated effluent from the nine landfills had radium-226 contents ranging from 70 to <139 pCi/L. The study authors conclude that there is "limited potential" for environmental impacts from spills or discharges of leachate from these facilities.

Where wastewater is stored in tanks, TENORM concentrations can increase through radioactive ingrowth.[1] Radium-226 and radium-228 are generally considered the radionuclides of greatest concern in wastewaters and are the most frequently measured. But recent research indicates that in closed environments such as tanks, where the radium decay product radon cannot escape, total radioactivity may increase due to ingrowth of other decay products of radium such as Pb-210, Po-210, and Th-228 (Nelson et al., 2015). Experimental work by Nelson et al. found that concentrations of these decay products in Marcellus produced water that was stored in a sealed drum started growing immediately. Concentrations started at zero and reached 10.49 pCi/L for Po-210 and 155 pCi/L for Th-228 over the first 50 and 66 days of storage, respectively. The authors note that these decay products are not soluble, would be associated primarily with particulates, and could be bioavailable. This study demonstrates that analyzing for radium will not provide a complete indication of sample radioactivity if the water is stored in a closed environment and that subsequent management decisions would need to take into account possible increases in radioactivity due to ingrowth.

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[1] The ingrowth, or growth within a sample, of radioactive daughter products from radionuclides initially present in the sample can cause greater radioactivity than that resulting from the parent radionuclides alone.

### 8.5.2.5 Other Solids

Other solid wastes associated with unconventional oil and gas production that may contain radionuclides include solid residuals from POTWs and CWTs and scale in oil and gas equipment. Filter cake samples from POTWs were found by PA DEP (2015b) to have highly variable radium-226 concentrations, with an average of 16 pCi/g, while typical soil concentrations in Pennsylvania have been found to be less than 2.5 pCi/g (Greeman et al., 1999). Filter cake from CWTs had an average radium-226 concentration of 111 pCi/g. The authors conclude that there could be impacts on surface waters through spills or effects on groundwater from landfill leachate containing contaminants originating in residuals sent to landfills.

Accumulation of TENORM-bearing scale in CWTs or POTWs may continue to affect the treatment plant even after discontinuing treatment of wastewaters containing high radionuclide concentrations. Radium can adsorb onto scales in pipes and tanks and can also be removed from water by coprecipitation if sulfate or carbonate is added to hydraulic fracturing wastewater to precipitate calcium, barium, or strontium (Kappel et al., 2013; USGS, 2013a). Pipe scale in oil and gas production facilities has been found to have radium concentrations as high as 154,000 pCi/g, although concentrations of less than about 13,500 pCi/g are more common (Schubert et al., 2014). A similar issue, the potential for accumulation and possible release of radionuclides and other trace inorganic constituents in water distribution systems, has gained attention, with the potential for drinking water concentrations to exceed drinking water standards (Water Research Foundation, 2010). Scale eventually removed from pipes or other oil and gas equipment can end up in landfills and then leach into groundwater or run off to surface water (USGS, 2013a). Also, laboratory research suggests that radium in land-applied barium sulfate scales from conventional oil and gas operations may become mobilized by microbial processes, rendering the radium more mobile and bioavailable (Matthews et al., 2006; Swann et al., 2004); see discussion in Section 8.4.6.1. Monitoring would be needed in order to ascertain the potential for accumulation and release of radionuclides from systems that have treated or continue to treat hydraulic fracturing wastewaters with elevated TENORM concentrations.

### 8.5.2.6 Road Spreading

Salt and radionuclide accumulation can occur near road spreading sites; one study in Pennsylvania describes a roughly 20% increase in average radium-226 concentrations in soils near five roads where wastewaters from conventional operations had been spread for de-icing (Skalak et al., 2014). However, the standard deviation for the samples was large (24 pCi/g), and background concentrations were approximately 1 pCi/g. Should significant accumulation of radionuclides in soils near roads occur, it would present a vehicle for potential impacts on drinking water resources. The frequency with which hydraulic fracturing wastewater contributes to this type of impact depends on state-level regulations dictating whether the wastewater can be used for road spreading.

### 8.5.2.7 Potential for Monitoring

Effluent from treatment plants (e.g., CWTs, POTWs) and receiving waters can be monitored for radionuclides. Research suggests that radium-226 and radium-228 are the predominant radionuclides in Marcellus Shale wastewater, and they account for most of the gross alpha and gross beta activity in the waters studied (Rowan et al., 2011). Gross alpha and gross beta measurements may, therefore, serve as an effective screening mechanism for the presence of radionuclides in hydraulic fracturing wastewater. This in turn can help in evaluating management strategies. Portable gamma spectrometers allow rapid screening of wastewater effluent. Sediments can also be measured for radionuclide concentrations at discharge points. If an accurate assessment of total radioactivity is needed rather than a screening, measuring radium content may not be adequate depending upon how wastewater has been stored. Analyses of other radionuclides such as Pb-210, Po-210, and Th-228 may be warranted, especially if the wastewater has been stored in closed loop systems.

### 8.5.3 Metals

Given the presence in hydraulic fracturing wastewaters of some heavy metals, as well as barium and strontium concentrations that can reach hundreds or even thousands of milligrams per liter (Table 7-5), surface waters may be impacted if discharges from CTWs or POTWs indirectly receiving oil and gas wastewater via CWTs are not managed appropriately or if spills occur.

Common treatment processes, such as chemical precipitation, are effective at removing many metals (Section 8.4). Effluent sampling results collected between October 2011 and February 2013 from seven facilities in Pennsylvania treating oil and gas wastewaters were requested by the EPA. The results revealed low to modest concentrations of copper (0 – 50 μg/L), zinc (14 – 256 μg/L), and nickel (8 – 22 μg/L) (U.S. EPA, 2015f, g). However, metals such as barium and strontium were found to range from low to elevated concentrations in the effluent for some of the facilities. The data showed effluent barium concentrations ranging from 0.35 to 25 mg/L (median of 3.5 mg/L and average of 6.7 mg/L). For results that were greater than 2 mg/L, the drinking water MCL for barium was exceeded. Strontium concentrations ranged from 0.36 to 546 mg/L (median of 297 mg/L and mean of 236 mg/L) (U.S. EPA, 2015g). (See Chapter 9 for information on health effects for barium and strontium.)

Volz et al. (2011) discussed a December 2010 effluent sampling effort at a Pennsylvania CWT that had been treating both conventional and Marcellus wastewater; they measured average barium and strontium concentrations of 27 mg/L and nearly 3,000 mg/L, respectively (eight samples from the one plant) (Volz et al., 2011). NPDES compliance data submitted for 2011 shows that effluent from the same CWT had average barium effluent levels ranging from 26 to 98 mg/L in the months prior to PA DEP's April 2011 request to cease sending hydraulic fracturing wastewater to this and other facilities exempt from the 2010 TDS regulation (U.S. EPA, 2015f, g). After May, 2011, barium effluent concentrations dropped to average values of 9 to 22 mg/L. The facility is scheduled to upgrade its TDS removal capabilities, which should help decrease concentrations of metals in the effluent.

Limited data are available on metal concentrations in wastewater and treated effluent that are directly discharged; additional information would be needed to assess whether there could be downstream effects on drinking water utilities. NPDES discharge permits, which restrict TDS discharge concentrations, would likely reduce metal effluent concentrations due to the additional treatment necessary to minimize TDS.

### 8.5.4 Volatile Organic Compounds

Benzene is a common constituent in hydraulic fracturing wastewater, and it is of concern due to recognized human health effects. A wide range of concentrations of BTEX compounds occurs in wastewater from the Barnett and Marcellus shales. Natural gas formations generally produce more BTEX than oil formations (Veil et al., 2004), and lower concentrations of BTEX naturally occur in wastewater from CBM production (Appendix Table E-9). The organic chemistry of Marcellus wastewater has been found by Akob et al. (2016) to be more variable than that of inorganic constituents, indicating the need to consider the concentrations of organic compounds when planning wastewater management.

Processes such as air stripping or dissolved air flotation can remove VOCs during treatment, but if treatment is not adequate prior to discharge, the VOCs may reach water resources. For example, the average benzene concentration measured in the discharge from a Pennsylvania CWT in December 2010 was 12 µg/L (Volz et al., 2011) exceeding the MCL for benzene of 5 µg/L.[1] The facility was receiving wastewater from both conventional and unconventional operations at that time. Ferrar et al. (2013) measured for BTEX in effluent from the same facility, and mean concentrations among the four compounds ranged from approximately 2 to 46 µg/L. Concentrations were lower for samples taken after May 19, 2011 (when Marcellus operators voluntarily stopped sending wastewater to POTWs and CWTs exempt from the 2010 TDS regulation), and the difference between pre and post May 2011 sampling was considered statistically significant.

Spills and leakage from pits creates another potential route of entry to drinking water resources, as described in Section 8.4.5. Akob et al. (2016) documented the microbial degradation of organic compounds in Marcellus produced water and note that more research is needed to evaluate how this could mitigate the migration of organic constituents in the event of spills or leaks.

### 8.5.5 Semi-Volatile Organic Compounds

Little is known about the fate of the SVOC, 2-butoxyethanol (2-BE) (an antifoaming and anti-corrosion agent used in slick-water) (Volz et al., 2011) or its potential impact on surface waters, drinking water resources, or drinking water systems. This compound is very soluble in water and is subject to biodegradation, with an estimated half-life of approximately 1-4 weeks in the environment (Wess et al., 1998). It is classified by the EPA's Integrated Risk Information System (IRIS) as not likely to be carcinogenic to humans, and the International Agency for Research on Cancer (IARC) classifies it as having insufficient evidence to determine carcinogenicity (see Chapter 9 for more information). 2-BE was detected in the discharge of a Pennsylvania CWT at

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[1] Among the BTEX compounds, the MCL for benzene is the lowest at 5 µg/L; the MCL for ethylbenzene is 700 µg/L, the MCL for toluene is 1,000 µg/L, and the MCL for xylenes is 10,000 µg/L.

concentrations of 59 mg/L (Volz et al., 2011). Ferrar et al. (2013) detected 2-BE in the effluent from a CWT in western Pennsylvania at average concentrations of 34 – 45 mg/L; the latter value was measured when the CWT was receiving only conventional oil and gas wastewater. Data are lacking on 2-BE concentrations in surface waters that receive treated effluent from hydraulic fracturing wastewater treatment systems.

Polyaromatic hydrocarbons (PAHs; a group of SVOCs) have been found in hydraulic fracturing wastewater (Section 7.3.4.7, Table 7-6). PAHs detected in an unlined pit containing oil and gas wastewater near the Duncan Oil Field in New Mexico were also detected in soils 82 ft (25 m) downgradient at concentrations ranging from 2,000 to 4,900 µg/kg and 164 ft (50 m) downgradient, with concentrations ranging from 22 to 370 µg/kg (Sumi, 2004; Eiceman, 1986).

### 8.5.6 Oil and Grease

Oil and grease in oil and gas wastewater can come from the formation or from oil-based drilling fluids. Typically, oil and grease are separated from the wastewater before discharge either by heat treatment or by gravity separation followed by skimming. If these processes are inefficient, oil and grease can be integrated with the discharge to surface waters. For example, in some cases, oil and grease are allowed to separate in pits, and water is then withdrawn from the lower part of the pit. If the oil layer is allowed to drop to the level of the standpipe or if the water is agitated, oil and grease may be discharged along with the water. Oil and grease are also often dispersed in wastewater in the form of small droplets that are 4 to 6 microns in diameter. These droplets can be difficult to remove using typical oil/water separators (Veil et al., 2004).

A study was conducted in Wyoming by the U.S. Fish and Wildlife Service from 1996 to 1999 of sixty five oil and gas sites that discharge to ephemeral streams and subsequently to wetlands. Fifteen percent of the wetlands receiving wastewater contained oil-stained vegetation and had a visible oil sheen on the sediments. In addition, ten of twelve sites that were randomly selected for water sample collection (from oil field separator or skim pit effluent) exceeded the discharge limit of 10 mg/L for oil and grease with one site as high as 54 mg/L (Ramirez, 2002).

## 8.6 Synthesis

A variety of strategies may be considered for the management of hydraulic fracturing wastewater. Important factors for planning management include cost, logistics, wastewater composition, wastewater volumes, and regulations. Available information suggests that Class IID wells regulated under the UIC Program are the most frequently used wastewater management practice, but reuse, sending to a CWT, and various other methods are also employed.

### 8.6.1 Summary of Findings

#### 8.6.1.1 Wastewater Volumes

The most current national estimate of the total wastewater volume generated in the oil and gas industry (both onshore and offshore) was 889.59 billion gal (21.18 billion bbls or 3.37 trillion L) in 2012, although this estimate is subject to a number of uncertainties (Veil, 2015). The total amount of wastewater generated may increase if hydrocarbon production increases in a region, although Veil (2015) suggests that this trend may not hold true at the national level. Geographically, a large portion of onshore oil and gas wastewater in the United States is reported to be generated in the western part of the country, consistent with the areas where most oil and gas wells are located and most production takes place.

Obtaining reliable national estimates of the amount of wastewater attributable to hydraulic fracturing is a challenge. State data collection efforts vary, and in many states, production data do not identify which wells have been hydraulically fractured. However, annual estimates compiled from those states where hydraulic fracturing wastewater is identified range from hundreds of millions to billions of gallons of wastewater generated each year. Data from individual states indicate that along with an increase in the numbers of hydraulically fractured wells, associated wastewater volumes have generally increased over the last several years into 2014. However, while there is a general correlation between unconventional oil and gas production and wastewater volume, the relationship is complicated by several factors such as timing of drilling and production. More complete and comparable estimates of local, state, and regional wastewater volumes would facilitate wastewater management on the part of operators as well as planning on the part of agencies that oversee wastewater management.

#### 8.6.1.2 Wastewater Management Practices

Hydraulic fracturing wastewater is managed in a variety of ways, including disposal via Class IID wells; minimal treatment and reuse (in subsequent fracturing operations); more complete treatment followed by reuse; sending to CWTs for treatment followed by direct discharge or transfer to POTWs; evaporation; and other uses such as agriculture and wildlife use (allowed only in the arid west when the wastewater is of good enough quality for such uses). All of these strategies have the potential to affect drinking water resources. Wastewater management practices continue to shift in response to evolving understanding of environmental concerns, emplacement of new regulatory controls, changes in costs, and changes in technology and operator practices. Unauthorized discharges of hydraulic fracturing wastewater have also been documented, and such discharges can potentially impact drinking water resources.

As of 2015, available information suggests that Class IID disposal wells are a primary wastewater management practice for operators in most of the major unconventional reservoirs in the United States, with the notable exception of the Marcellus Shale region in Pennsylvania. Class IID wells tend to be economically favorable, especially if they are located within a reasonable transportation distance from well sites (U.S. GAO, 2012). In particular, large numbers of active injection wells are found in Texas (7,876), Kansas (5,516), Oklahoma (3,837), Louisiana (2,448), and Illinois (1,054) (U.S. EPA, 2016d).

Pennsylvania is somewhat unique in having only nine Class IID wells (as of February 2015), along with having experienced significant growth of shale gas production in the Marcellus and corresponding production of large volumes of wastewater. Operators producing from unconventional formations (as defined by PA DEP) have managed their wastewater through the use of POTWs (a practice that is subject to recently promulgated regulations), CWTs, extensive reuse for hydraulic fracturing operations, and hauling to disposal wells (to a lesser degree). The wastewater management history in Pennsylvania provides an example of evolving strategies to manage the treatment, discharge, storage, and reuse of hydraulic fracturing wastewaters that are high in constituents of concern (e.g., bromide, TDS, and TENORM).

Reuse of hydraulic fracturing wastewater to formulate fluid for subsequent hydraulic fracturing jobs is most prevalent in Pennsylvania (as high as 90%), with much of the reuse happening on-site (PA DEP, 2015b). Reuse is practiced in other regions as well (e.g., Haynesville Shale, the Fayetteville Shale, the Barnett Shale, and the Eagle Ford Shale), but at much lower rates (about 5 – 20%). Reliable estimates are not available for all areas of the United States because waste management practices are not consistently reported across all states. If hydraulic fracturing activity slows, demand for wastewater for reuse will also likely decrease, and other forms of wastewater management will be needed. Potential impacts associated with reuse center on concerns over the storage of untreated or minimally treated wastewater on-site or transport to CWTs.

Treatment of hydraulic fracturing wastewater may be done at CWTs or using mobile or semi-mobile systems designed for on-site use. Treatment at a CWT may be followed by direct discharge by the CWT to surface water, indirect discharge to a POTW in accordance with recently promulgated regulations, or reuse. Most CWTs treating hydraulic fracturing wastewater are located in Pennsylvania (about 40 facilities), with a limited number in other states. CWTs vary widely in treatment capabilities, ranging from producing high-quality effluent to minimal treatment for reuse.

Other wastewater management practices, such as evaporation and agricultural uses, represent a smaller fraction of wastewater management nationally. These practices can, however, be locally significant. Although specific instances of contamination were not identified for this assessment, these practices could lead to impacts on drinking water resources if facilities are not properly constructed and maintained or if water quality is not adequately characterized to ensure that management is appropriate.

### 8.6.1.3 Treatment and Discharge

Wastewater that is treated and subsequently discharged by CWTs can result in impacts due to inadequate treatment. A frequently cited concern is the high TDS content in wastewaters from unconventional formations, which poses challenges for treatment, discharge, and reuse. Treatment processes such as sedimentation, filtration, flotation, and chemical precipitation are capable of removing constituents such as oil and grease, major cations, metals, and TSS. They do not, however, adequately reduce TDS in high-salinity wastewaters. More advanced processes such as reverse osmosis (RO) or distillation are needed if TDS removal is required (Shaffer et al., 2013; Younos and Tulou, 2005). Most available information on treatment of hydraulic fracturing wastewater is based on practices used in Pennsylvania because that is where most data have been collected.

Hydraulic fracturing wastewater discharged from treatment facilities without advanced TDS removal processes has resulted in elevated TDS concentrations (including bromide, iodide, and chloride levels) in receiving waters. Impacts from these discharges is due largely to the role of bromide and iodide in DBP formation at downstream drinking water systems, potentially causing higher levels of harmful DBPs in finished drinking water.[1] Modeling suggests that very small percentages of hydraulic fracturing wastewater added to a river used as a source for drinking water systems could cause a notable increase in DBP formation.

Radionuclides (i.e., TENORM), which are present in some hydraulic fracturing wastewaters, can cause impacts if the wastewater is discharged without adequate treatment. TENORMs have been measured in effluent from wastewater treatment facilities receiving Marcellus wastewater (which includes effluent sent for reuse and not discharged to surface water). Radium-226, radium-228, gross alpha, and gross beta are most cited as the radioactive constituents of concern, likely due to the availability of test methods for these constituents in wastewater. Radium concentrations can range up to thousands or tens of thousands of pCi/L. Fewer data are available on concentrations of uranium and other radionuclides in hydraulic fracturing wastewaters. Also, fewer data exist on radionuclide concentrations in wastewaters from unconventional formations other than the Marcellus, limiting our ability to assess potential impacts from TENORM on a nationwide basis.

Other constituents posing health or environmental concerns that can be discharged in inadequately treated hydraulic fracturing wastewater include organic compounds, barium, strontium, and other metals. Chemicals used in the fracturing fluid or their degradation products could also be present. A variety of treatment processes can be used for removal of these contaminants, from commonly used methods such as chemical precipitation and filtration to more advanced and more costly techniques, such as reverse osmosis, distillation, and mechanical vapor recompression.

#### 8.6.1.4 Storage and Disposal Pits and Impoundments

Regardless of the wastewater management practices used, some type of temporary storage of fluids is generally required. Storage can be in the form of tanks as well as pits and/or impoundments. Pits encompass a variety of structures, from on-site pits for storage at the well site to larger, centralized facilities (typically referred to as "impoundments" or "ponds"). Some states allow evaporation pit facilities or percolation pits as a means of wastewater disposal. The locations and number of pits are not well documented in most states, nor are pits associated with hydraulic fracturing operations necessarily identified, despite efforts by the U.S. EPA (U.S. EPA, 2003b) and environmental groups such as SkyTruth to identify pits in use. Information that is typically available on state websites includes permitted centralized commercial evaporation facilities (COWDFs) most commonly used in the western United States.

Impacts on both groundwater and surface water resources due to inadequate pit capacities, overfilling, and leaks have been documented. In extreme precipitation events, pits can be overtopped. Leaks can occur if liners are compromised or were not used. With an increased emphasis on reuse in some regions, the need for temporary storage of high-TDS wastewater increases the potential for leaks and spills from pits and during fluid handling.

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[1] Some types of DBPs are regulated under SDWA's Stage 1 and Stage 2 DBP Rules, but a subset of DBPs, including a number of chlorinated, brominated, nitrogenous, and iodinated DBPs, are not regulated. Brominated and iodinated DBPs are more toxic than other species of DBPs.

Unlined pits, in particular, provide a pathway for contaminants to reach groundwater, and impacts on groundwater from historic and current uses of unlined pits in the oil and gas industry have been documented. The resulting contamination can be long-lasting. States have taken measures to phase out the use of unlined disposal and storage pits, but unlined pits that are still in use can provide an ongoing potential source of contamination for groundwater (Grinberg, 2014).

#### 8.6.1.5 Residuals

Solid and liquid residuals associated with hydraulic fracturing wastewater (treatment residuals from CWTs, sludges from tanks and pits, and pipe scale) could have impacts on drinking water resources if not managed and disposed of properly. Liquid residuals are inappropriate for surface water discharge or discharge to a POTW due to high concentrations of salts and other contaminants; they are commonly disposed of in an injection well. Solid residuals may leach a number of constituents, such as alkali metals, alkaline earth metals, and bromide. They can also contain TENORM if radionuclides are present in the wastewater being treated. Given that residuals are commonly disposed of in landfills, TENORM can be problematic due to the possibility of radon emissions and radioactivity in the landfill leachate. Solids from pits or tanks can also contain TENORM if the wastewater contains radionuclides, and one study has shown the potential for radioactivity to increase in the closed environment of tanks.

### 8.6.2 Factors Affecting the Frequency or Severity of Impacts

The frequency and severity of impacts on drinking water resources from hydraulic fracturing wastewater will depend upon the wastewater composition and volumes, and the mix of wastewater management strategies used.[1] The types of potential impacts (along with frequency and severity) may shift in time as management practices change in response to evolving environmental, regulatory, economic, or logistical drivers. The frequency and severity of impacts can also depend on the size and initial quality of the drinking water resource and its proximity to wastewater management operations.

#### 8.6.2.1 Role of Changing Wastewater Management Practices

The most common disposal option for hydraulic fracturing wastewater is injection into Class II disposal wells. If this option becomes restricted in a given location, the wastewater management options could shift, at least locally, towards other options such as sending wastewater to CWTs for treatment and either discharge or reuse. Although reuse avoids the immediate need to discharge wastewater by directing it to ongoing hydraulic fracturing activities, the practice could concentrate radionuclides or other constituents as fluid moves through cycles of reuse. Whether such concentrations would be significant depends on the ratio of recycled to “fresh” water when the wastewater gets reused. Alternatively, wastewater might need to be transported to more distant

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[1] Both national and state regulations affect the wastewater management practices used. At a national level, although the EPA’s oil and gas ELG regulations generally prohibit the direct discharge of oil and gas wastewater to waters of the U.S., treatment and discharge of hydraulic fracturing wastewater can occur under certain limited circumstances, such as under an exemption authorizing discharge for agricultural and wildlife use in the arid west, or by Centralized Waste Treatment facilities. For additional information on national regulations relevant to hydraulic fracturing wastewater management, see Text Box 8-2.

Class IID wells. This option, while attractive from the perspective of limited disposal impacts, could increase the frequency of impacts from spills and leaks during transportation (see Chapter 7 for discussion of roadway transport of produced water).

### 8.6.2.2 Treatment and Discharge

Both the frequency and severity of potential impacts on drinking water resources from treated hydraulic fracturing wastewater depend on the influent concentrations of the constituents in the wastewater and the type and adequacy of the treatment processes employed. If treatment and/or blending is inadequate, the resulting quality in a receiving water could, for example, influence formation of DBPs during subsequent drinking water treatment, impair biological treatment processes, and release TENORMs into receiving waters.

The volume of treated effluent discharged relative to the size of the receiving water body is an important local factor affecting the frequency and severity of potential impacts. Because of dilution effects, drinking water systems drawing from smaller rivers will likely face greater challenges in dealing with contaminants in their source water than systems drawing from larger rivers receiving the same volume of effluent. Seasonal changes in streamflow will also affect frequency and severity by affecting the degree of dilution. Existing loadings of pollutants from other sources in a watershed can increase the frequency and severity of potential impacts if the additional contributions from hydraulic fracturing wastewater cause concentrations to exceed thresholds.

Direct discharges of wastewaters with lower TDS concentrations to ephemeral streams are allowed in parts of the country where the wastewater is considered to be “of good enough quality” for livestock watering and wildlife use, and the discharges may constitute a large portion of streamflow. Permits authorizing such discharges may only require monitoring for a limited set of constituents. In particular, they may not necessarily require monitoring for specific constituents associated with hydraulic fracturing. The potential for water quality impacts from such discharges depends upon whether chemicals used for fracturing fluid or maintenance (or their degradation products) are present and at what concentrations. Long-term discharges to these ephemeral streams could result in ongoing impacts if there are unrecognized or unaddressed water quality issues.

Concerns about radionuclides in hydraulic fracturing wastewater have received considerable public attention, especially in the Marcellus region. The severity and frequency of impacts on receiving waters and sediments from TENORM depends upon the TENORM content in the wastewater (highest in regions with NORM-rich formations), temporal variability in the wastewater composition, and the treatment processes used. There are insufficient data to indicate whether radionuclides from these wastewaters have reached drinking water intakes. However, data do suggest that radionuclides can accumulate in sediments at or near discharge points from facilities that treat and discharge oil and gas wastewater. A recent PA DEP study (PA DEP, 2015b) reported radium in the effluent of both CWTs handling oil and gas wastewater and POTWs receiving effluent from such facilities.

Analysis of TENORM concentrations in hydraulic fracturing wastewaters prior to treatment, selection of appropriate treatment processes that adequately address the TENORM levels, and monitoring of TENORM in the treated effluent and receiving waters could help address the frequency and severity of potential impacts on drinking water resources in these areas. However, a confounding issue is underestimation of radium concentrations when using traditional wet chemical methods with high-TDS waters. This could consequently cause underestimation of frequency or severity of impacts. Newer studies have begun to use gamma spectroscopy for better recovery, which could help with more accurate assessment of frequency and severity of impacts (Nelson et al., 2014).

Accumulation of other contaminants such as organic compounds or metals in sediments at or near discharge points is also possible. If the sediments are disturbed or entrained due to dredging or flood events, contaminated sediments could be transported downstream closer to drinking water systems. The fate of such sediments and likelihood of mobilization of contaminants originating from hydraulic fracturing wastewaters have not been explored. The frequency and severity of impacts related to contaminated sediments would depend on a number of site-specific factors such as concentrations in the sediments, effluent quality, volume from the discharging facility, stream water quality, and stream hydrodynamics.

### 8.6.2.3 Storage and Disposal Pits and Impoundments

Tanks, pits, and impoundments, ever-present at oil and gas operations and CWTs, provide an opportunity for impacts on drinking water resources. Tanks are generally regarded as being safer than pits in terms of containment, although recent research has shown the potential for an increase in radioactivity in tank sediment if the wastewater contains TENORM. For pits and impoundments, the likelihood and severity of impacts due to spills and leaks depends in part on state construction and maintenance requirements for pits and how well these are observed. Frequency and severity of impacts will be lessened by attention to design standards, competent construction, and operational practices.

Liners, in particular, are an important measure to protect groundwater resources and are a common aspect of pit construction requirements. Liner specifications address materials, thickness, and leak detection. If a liner is compromised or nonexistent, the severity of impacts on groundwater will be affected by the volume leaked, the composition of the water in the pit, the depth to the water table, soil permeability, and the capacity of the soil to retain certain pollutants as the water percolates through. If substantial sediment has built up in the bottom of the pit, then in the event of a liner breach, contaminants may leach if the sediments permit water to pass through and into the soil. The fate and transport of wastewater contaminants in the subsurface is governed by a complex set of physical, chemical, and biological processes that dictate interactions between wastewater constituents and soil minerals, degradation or transformation of wastewater constituents, and possible mobilization of constituents in the soil under a pit (see Section 5.8 in Chapter 5 for a thorough discussion of processes affecting movement of constituents in the subsurface). Duration of use is also a consideration; the longer a pit with a faulty or nonexistent liner receives wastewater, the more severe the ultimate impact could be on underlying sediment and groundwater.

In the event of overtopping of a pit due to overfilling or extreme weather, the severity of impacts on surface water or groundwater will depend on the volume that overflows, wastewater composition, distance to surface water (if wastewater flows over land), depth to the water table, and soil properties (if the overflow infiltrates into the soil). If the overflow reaches a stream or river, the size of the spill relative to stream size and flow rate could also affect the severity of the impact. The combined factors that can contribute to overflows include capacity of the pit, the volume of fluid stored in the pit (i.e., freeboard) at the start of the precipitation event, and failure to monitor/ reduce pit fluid levels if needed.

As with concerns over discharges, the potential for impacts will be tied to other, existing stresses within a watershed. If the surface water is already receiving pollutant loadings from other sources, then an additional contribution from a pit-related leak or spill may not be as readily accommodated without causing water quality impairment.

#### 8.6.2.4 Other Management Practices and Management of Residuals

Other management strategies such as irrigation, road spreading, and evaporation are less frequently employed for hydraulic fracturing wastewaters. The severity of impacts on surface waters from irrigation and road spreading will depend on the constituents in the wastewater (e.g., salts, radionuclides, and chemicals used in hydraulic fracturing), the distance to a receiving water, and whether stormwater management measures exist to mitigate runoff. The factors influencing whether constituents will migrate to shallow groundwater include depth to the water table, precipitation, soil permeability, and the soil's ability to retain pollutants that can adsorb to particles. If irrigation and road spreading are long-term management practices, the frequency of impacts will likely be proportional to the frequency with which the practices are employed.

Liquid and solid residuals generated from the treatment, storage, and handling of hydraulic fracturing wastewater have highly concentrated waste constituents. This could increase the potential severity of impacts due to spills that reach surface water resources or leach to groundwater. Potential impacts from management of residuals can be lessened in frequency and severity through careful handling, adequate characterization (including TENORM content), and selecting an appropriate disposal method, including use of a landfill that can accept TENORM waste if needed.

### 8.6.3 Uncertainties

A full understanding of hydraulic fracturing wastewater management is limited by a lack of available data in several areas. First, it is difficult to assemble a complete national- or regional-level picture of wastewater volumes and the management practices used because the tracking and availability of data vary from state to state. Although some states provide well-organized and relatively thorough data, not all states collect or make such information available. It can be difficult to identify wastewater volumes specifically associated with hydraulic fracturing (as compared to all oil and gas production activities). Such data would be needed to place hydraulic fracturing wastewater in the broader context of all oil and gas wastewaters. It is also generally difficult to determine whether hydraulic fracturing wastewater is being injected under a given disposal well permit because the permit rarely identifies which production wells are contributing to the wastewater stream. Data are also generally difficult to locate for wastewater production volumes, the chemical composition and concentrations in wastewater, and the management and disposal strategies for residuals.

Up-to-date information on the volume of hydraulic fracturing wastewater disposed of via underground injection by state is not uniformly available. Without this information, it is difficult to assess whether disposal well capacity will become an issue in areas where hydraulic fracturing activity is expected to increase or where use of disposal wells may become restricted locally or regionally.

For CWTs permitted to discharge to surface water, the ability to assess the potential effects of these discharges on drinking water resources is limited by the lack of effluent water quality data. Some monitoring data are required by the permit, but the list of monitored constituents may be limited. Selection of the appropriate water quality parameters to be monitored is critical to ensure that potentially problematic constituents are identified (e.g., chemicals associated with hydraulic fracturing fluids, maintenance chemicals, and degradation products of those chemicals). Some chemicals used in fracturing fluids are not disclosed, and analytical methods are lacking for some chemicals of concern and their degradation products.

Pollutant removal capabilities of the treatment facilities would also be valuable information to have, but this would require well-coordinated collection of both influent and effluent samples; this type of data is even less commonly available. In addition, the use of inappropriate analytical methods for the high TDS wastewater associated with hydraulic fracturing operations can complicate the use of available data. Methods used should be suitable for the highly complex matrix of contaminants encountered with oil and gas wastewater to have confidence in the results of chemical analyses.

Monitoring of surface waters downgradient of discharges, such as screening with a TDS proxy (i.e., conductivity), would also help assess the frequency of impacts on receiving waters by hydraulic fracturing activities (including spills and discharges of wastewater). Such data can also give an estimation of the severity of those impacts. Other than a few studies in the Marcellus Shale region, these types of water quality data are lacking. Existing data are also limited regarding legacy effects, such as accumulation of contaminants in sediments at discharge points, soil accumulation due to application of de-icing brines or salts from wastewater treatment, and handling of wastewater treatment residuals.

Assessing longer-term impacts on surface water quality from hydraulic fracturing activities in general is severely hampered by inadequate data. Bowen et al. (2015) state that available national-level databases are inadequate for addressing the question of whether there is evidence of national-level trends in surface water quality (as measured by specific conductivity and chloride) in areas where unconventional oil and gas production is taking place. Work by the Northeast-Midwest Institute and the USGS (Betanzo et al., 2016) was undertaken to explore the types and amounts of data needed to assess whether shale gas development activities contaminate surface water or groundwater in the Susquehanna River Basin. The focus was on longer-term cumulative impacts because detection of such impacts requires water quality monitoring. Detection of impacts (in either surface water or groundwater) requires a systematic monitoring approach that includes sampling at appropriately selected locations at an adequate frequency and duration and for a suite of water quality parameters to detect changes over time. Comparison sites without hydraulic fracturing activity are needed as well. The authors concluded that the data necessary to detect changes in surface water or groundwater due to hydraulic fracturing activities do not currently exist for the Susquehanna River Basin.

### 8.6.4 Conclusions

Oil and gas operations in the United States generated an estimated 2.43 billion gal of wastewater per day (about 60 million bbls/day) in 2012 (Veil, 2015). This includes wastewater associated with hydraulic fracturing activities, although what portion of this oil and gas wastewater is attributable to hydraulic fracturing operations is difficult to estimate. Available information indicates that the majority of hydraulic fracturing wastewater is injected into Class IID wells regulated under the UIC Program. In the Marcellus Shale region in Pennsylvania, this option is limited, and the majority of wastewater is reused (either with or without treatment) for new hydraulic fracturing jobs. Hydraulic fracturing wastewater may also be treated at a CWT and discharged by the CWT to surface water or to a POTW. In the western United States, wastewater is used in other ways (e.g., livestock watering) if water quality allows. Wastewater is also sent to evaporation ponds for disposal or stored on-site or in centralized pits or impoundments prior to final disposal or reuse.

Impacts on drinking water resources have resulted from discharges of inadequately treated wastewater and from leaks, spills, and percolation associated with pits. Other mechanisms for impacts include improper handling of treatment residuals or pit and tank sludges as well as leaching and runoff associated with other wastewater management practices. The impacts related to pits and residuals/sludges affect both surface water and groundwater; unlined pits or those with compromised liners present a particular concern (see Chapter 7 for additional discussion of spills). The constituents that have received the greatest attention in the literature include TDS, DBP precursors (especially bromide), and radium, although hydraulic fracturing wastewater can contain elevated concentrations of a number of organic and inorganic constituents of concern. Regardless of the management option utilized, if the wastewater is not thoroughly characterized or sampling is not conducted for analytes of concern, the severity and frequency of the impacts will be unknown or unquantified. The nature and volume of wastewater generated through hydraulic fracturing activities necessitate careful consideration of handling, treatment, and ultimate reuse or disposal to ensure that water resources are not adversely impacted. There is also a need for reliable and consistent waste generation data collection and reporting, improved efforts to characterize wastewater quality (both treated and untreated), and systematic monitoring efforts to be able to detect impacts on drinking water resources.

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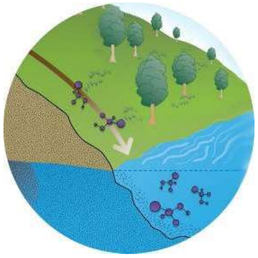

# Chapter 9. Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

## Abstract

This chapter identifies chemicals associated with hydraulic fracturing and provides an overview of the potential human health effects associated with these chemicals, as well as variables that could affect chemical occurrence in drinking water. The EPA has identified 1,606 chemicals associated with hydraulic fracturing, including 1,084 chemicals that are used in hydraulic fracturing fluid and 599 chemicals that have been detected in produced water. There is some uncertainty surrounding this chemical list, as it does not include a subset of chemicals that are classified as confidential business information, and because understanding of produced water composition is constrained by limitations of analytical chemistry as well as site-specific variations in the geochemistry of hydraulically fractured rock formations.

The EPA used selected federal, state, and international sources of toxicological data to identify toxicity values that can be used to support risk assessment for these chemicals, including chronic oral reference values (RfVs) for noncancer effects and oral slope factors (OSFs) for cancer. Chronic oral RfVs or OSFs were available for 173 (11%) of the total 1,606 chemicals. Health effects associated with chronic oral exposure to these chemicals include carcinogenicity, neurotoxicity, immune system effects, changes in body weight, changes in blood chemistry, liver and kidney toxicity, and reproductive and developmental toxicity.

For the majority of chemicals that lack chronic oral RfVs or OSFs, risk assessors will have to turn towards other sources of toxicological information that may have greater uncertainty than RfVs and OSFs, including quantitative structure-activity relationship (QSAR) models or additional data from the EPA's Aggregated Computational Toxicology Resource (ACToR) database. To understand whether specific chemicals can affect human health through their presence in drinking water, data on chemical concentrations in drinking water are needed. In the absence of these data, a preliminary analysis of relative hazard potential for drinking water resources can be conducted using the multi-criteria decision analysis (MCDA) approach outlined in this chapter. The MCDA combines data on toxicity, occurrence, and physicochemical properties for selected subsets of chemicals and was used in this chapter to highlight several chemicals that may be more likely than others to reach drinking water resources and present a health hazard.

Overall, while evidence suggests that hydraulic fracturing has the potential to impact human health, the actual human health implications are not well understood or well documented. Given that chemicals in hydraulic fracturing fluids and produced water are likely to vary on a regional basis and even between individual wells, the materials presented in this chapter are best applied for risk assessment and risk management decision-making at the local level.

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# 9. Identification and Hazard Evaluation of Chemicals across the Hydraulic Fracturing Water Cycle

## 9.1 Introduction

In this chapter, we present and integrate what is known about chemicals in the hydraulic fracturing water cycle, and provide an initial assessment of the potential for these chemicals to impact human health. The discussion is focused on the availability of toxicity values and qualitative assessments that can be used to inform the risk assessment of these chemicals for oral exposure via drinking water—in particular, the available noncancer oral reference values (RfVs), cancer oral slope factors (OSFs), and qualitative cancer classifications.[1,2,3] Public health impacts will depend upon both the inherent toxicity of these chemicals and the potential for human exposure. We highlight several field studies that have detected hydraulic fracturing-related chemicals in drinking water resources, and discuss properties of chemicals related to environmental fate and transport that could affect their potential impact on drinking water resources. To the extent information was available to do so, knowledge of toxicological and chemical properties was combined to illustrate a preliminary analysis of the relative hazard that these chemicals could pose to drinking water resources. The data are presented in this chapter as follows:

Section 9.2 provides a brief background on public health concerns surrounding hydraulic fracturing and unconventional oil and gas extraction, which have been highlighted in several recent studies.

Section 9.3 discusses how information sources were used to create a list of chemicals used in or detected in various stages of the hydraulic fracturing water cycle. The consolidated chemical list includes chemicals reportedly added to hydraulic fracturing fluids in the chemical mixing stage, as well as fracturing fluid chemicals, formation chemicals, or their reaction products that may be carried in produced water.

Section 9.4 provides an overview of the methods that were used for gathering information on toxicity and physicochemical properties for all chemicals identified in Section 9.3, and outlines the number of chemicals that had available data on these properties. For toxicological data, the primary focus is on chronic oral RfVs, OSFs, and qualitative cancer classifications from selected data sources that met the EPA's criteria for inclusion in this assessment. This section also discusses other

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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 (U.S. EPA, 2011f). In the context of this report, the term RfV refers to reference values for non-cancer effects occurring via the oral route of exposure and for chronic durations, except where noted.

[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 (U.S. EPA, 2011f).

[3] Qualitative cancer classifications are a system used for the hazard identification of probable carcinogens, in which human data, animal data, and other supporting evidence are combined to characterize the weight of evidence (WOE) regarding the potential of an agent to cause cancer in humans.

potential sources of toxicity information: the use of quantitative structure-activity relationship (QSAR) modeling to estimate chemical toxicity, as well as the availability of toxicological information on the EPA's Aggregated Computational Toxicology Resource (ACToR) database. A brief description of other potential tools and approaches that may be used by stakeholders for site-specific evaluation of chemical hazards, but are not used in this report, is provided in Appendix G.

Section 9.5 describes the potential hazards of subsets of chemicals identified as being of interest in previous chapters of this report. This includes chemicals in hydraulic fracturing fluid (Chapter 5); organic chemicals, inorganic chemicals, and pesticides detected in produced water (Chapter 7); stray gas, such as methane (Chapter 6); and disinfection byproducts (DBPs) formed from constituents of hydraulic fracturing fluid wastewaters (Chapter 8). We discuss instances in which these chemicals have been detected in drinking water resources in areas of hydraulic fracturing activity, and provide an overview of the available toxicological information for these chemicals.

Section 9.6 uses a multi-criteria decision analysis (MCDA) framework to provide a preliminary analysis of the potential hazards of chemicals used in hydraulic fracturing fluids or detected in produced water. The MCDA framework is used to integrate data on chemical toxicity, occurrence, and physicochemical properties. In this context, occurrence and physicochemical properties are used as metrics to estimate the likelihood that a chemical will reach and impact drinking water resources. Chemicals considered in these hazard evaluations include a subset of chemicals identified in the EPA FracFocus 1.0 project database, as well as a subset of organic chemicals that have been detected in produced water.

This chapter is not a human health risk assessment. As shown in Text Box 9-1, risk assessment consists of four basic steps: hazard identification, dose-response assessment, exposure assessment, and risk characterization. This chapter provides an overview of hazard identification and dose-response assessment for these chemicals, but lacks information to fully characterize exposure and risk. In Section 9.5, we highlight instances in which these chemicals have been detected in drinking water resources, but these data are only available for a small number of chemicals. The MCDA approach in Section 9.6 provides a method for integrating data on toxicity and exposure potential, but should be considered only as a preliminary analysis, and should not be used in place of local data on chemical exposure.

This chapter is focused on potential human health hazards of chemicals for the oral route of exposure (drinking water); therefore, the toxicological properties and physicochemical ranking metrics described herein do not necessarily apply to other routes of exposure that may occur with these chemicals, such as inhalation or dermal exposure. We additionally note that this analysis is focused on individual chemicals, rather than mixtures of chemicals used as additives.

In general, characterizing chemicals and their properties on a national scale is challenging and the use and occurrence of chemicals is likely to differ between geological basins and possibly on a well-to-well basis (Chapters 5 and 7). Therefore, for the protection of human health at the local level, chemical hazard evaluations are best conducted on a regional or site-specific scale. This level of analysis is outside the scope of this report; however, the methods of hazard evaluation presented here can also be applied on a regional or site-specific scale in order to identify chemicals that may present a potential human health hazard.

**Text Box 9-1. Applying Toxicological Data for Human Health Risk Assessment.**

Understanding potential human health impacts requires knowledge not only of the inherent toxicity of the chemicals found in contaminated environmental media, but also of the potential for exposure to these chemicals. The process of evaluating the nature and probability of such impacts is known as human health risk assessment. Overall, human health risk assessment includes four basic steps (U.S. EPA, 2016a):

**1. Hazard identification:** Examining whether a chemical has the potential to cause harm to humans and/or ecological systems, and if so, under what circumstances.

**2. Dose-response assessment:** Examining the numerical relationship between exposure and effects.

**3. Exposure assessment**: Examining what is known about the frequency, timing, and levels of contact with a chemical.

**4. Risk characterization:** Examining how well the data support conclusions about the nature and extent of risk from exposure to a chemical. Information from the hazard identification, dose-response assessment, and exposure assessment are summarized and integrated into quantitative and qualitative expressions of risk.

The RfVs and OSFs compiled by the EPA in this study pertain to the first two steps of human health risk assessment: identifying chemicals that have the potential to affect human health (hazard identification), and characterizing the exposure levels at which those effects occur (dose-response assessment). These toxicity values may be used in combination with site-specific chemical exposure information (exposure assessment) in order to evaluate potential human health risks (risk characterization). Qualitative cancer classifications characterize the weight of evidence regarding the potential for a chemical to cause cancer, and therefore provide additional information that can be used for hazard identification.

Toxicity information spans a wide range with respect to extent, quality and reliability. The RfVs, OSFs, and qualitative cancer classifications compiled in this study are those identified by the EPA as being of the highest quality and reliability, per the criteria discussed in this chapter. The QSAR-based toxicity estimates discussed in this chapter are considered to be lower on the continuum of quality and reliability, but may provide useful information pertaining to hazard identification and dose-response assessment when a chemical does not have an RfV or OSF available. The EPA's ACToR database provides an aggregation of a wide range of toxicological data that may also be useful for supporting the risk assessment of these chemicals. This chapter provides information on whether a chemical has data available from ACToR; however, it is beyond the scope of this report to evaluate the quality and reliability of data for these chemicals within ACToR, or to provide guidance on how the data within ACToR should be used to support human health risk assessment.

## 9.2 Overview: Hydraulic Fracturing and Potential Impacts on Human Health

As discussed in the previous chapters of this assessment, a variety of chemicals are associated with the hydraulic fracturing water cycle. Chemicals are added to hydraulic fracturing fluids at the chemical mixing stage (Chapter 5), and then injected into the well (Chapter 6). These chemical additives may return to the surface in produced water, along with chemicals from the formation (Chapter 7). The chemicals in produced water may persist in wastewater effluents, with some constituents contributing to the formation of disinfection byproducts in treated wastewater (Chapter 8). Through events such as large volume spills (Figure 9-1), mechanical integrity failures, hydraulic fracturing directly into groundwater resources, or discharge of inadequately treated hydraulic fracturing wastewater, there are specific instances in which these chemicals have been demonstrated to enter drinking water resources. Thus, there is potential for human exposure to these chemicals, and the potential for adverse human health effects resulting from exposure.

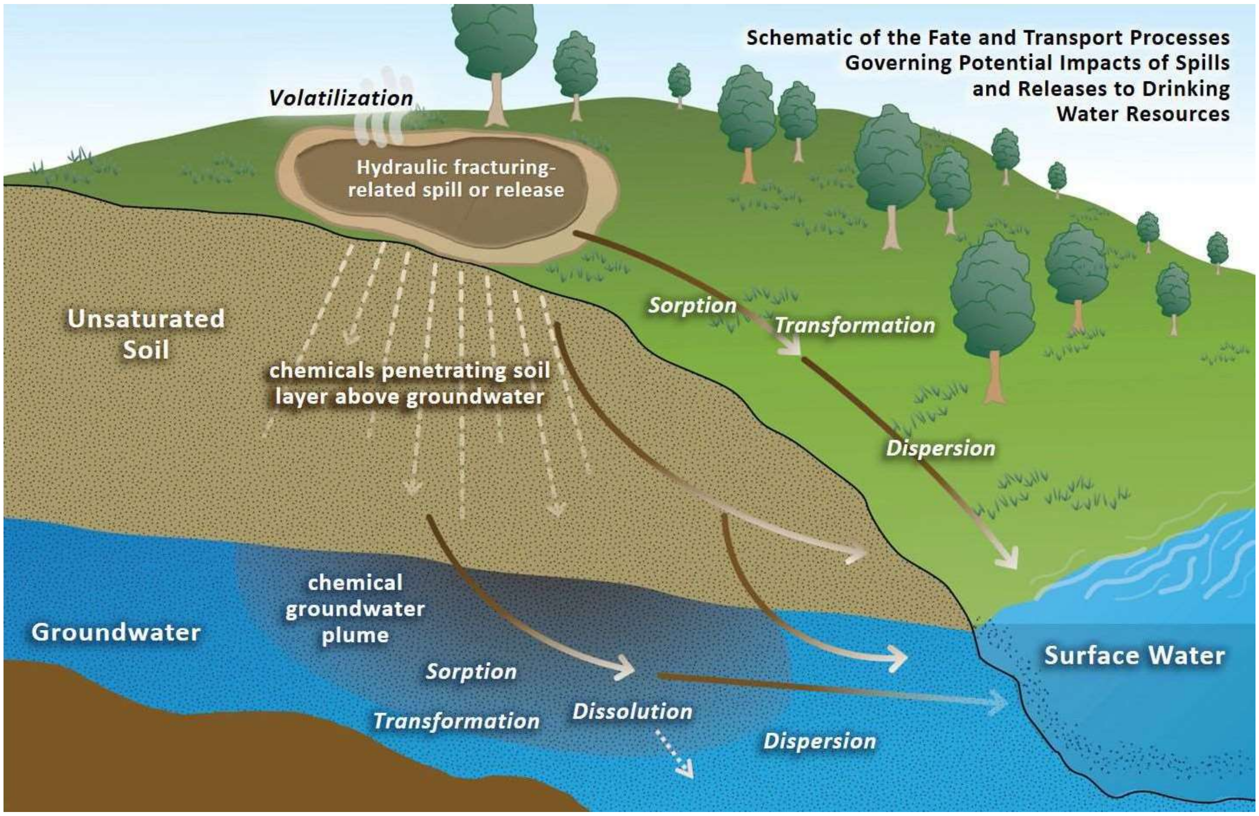

**Figure 9-1. Fate and transport schematic for a hydraulic fracturing-related spill or release.**

Multiple authors have noted with the recent increase in hydraulic fracturing operations there may be an increasing potential for significant public health and environmental impacts (Goldstein et al., 2014; Finkel et al., 2013; Korfmacher et al., 2013; Weinhold, 2012). These concerns have been highlighted in several recent studies. An epidemiological study in Colorado demonstrated residential proximity of pregnant mothers to natural gas wells is associated with an increased incidence of congenital heart defects, and, to a lesser extent, neural tube malformations (Mckenzie et al., 2014). A similar study in Pennsylvania found pregnant mothers living closer to unconventional natural gas wells were more likely to have infants that were small for gestational age, with lower birth weights compared to infants from mothers living farther from wells (Stacy et al., 2015). Residential proximity to natural gas wells in the Marcellus Shale is associated with an increase the number of self-reported health symptoms, particularly upper respiratory and dermal symptoms (Rabinowitz et al., 2015), chronic rhinosinusitis, migraine headache, and fatigue symptoms (Tustin et al., 2016). Laboratory studies have found that endocrine disrupting activity measured using in vitro bioassays may be elevated in surface and groundwater at known hydraulic fracturing spill sites (Kassotis et al., 2014) and in surface water downstream from a hydraulic fracturing wastewater injection facility (Kassotis et al., 2016). Although none of these studies demonstrate a direct effect of hydraulic fracturing activity on human health, and none of the epidemiological studies provided measures of individual or population level exposures or differentiated between drinking water contamination and other potential routes of exposure (e.g., air pollution), all are suggestive of a relationship between unconventional oil and gas development and adverse health outcomes.

Previous chapters of this report have identified cases in which contamination of drinking water resources could clearly be linked to hydraulic fracturing activity. For example, equipment failure and human error have led to spills of hydraulic fracturing fluids across the country and have affected the quality of drinking water resources (U.S. EPA, 2015m; Brantley et al., 2014; COGCC, 2014; Gradient, 2013). Other studies highlighted in previous chapters provide indirect evidence hydraulic fracturing activity has contaminated surface water or groundwater. For example, two recent studies in the Marcellus Shale detected known hydraulic fracturing-related chemicals in nearby groundwater wells, and used multiple lines of evidence to link the origin of these chemicals to hydraulic fracturing activity (Drollette et al., 2015; Llewellyn et al., 2015).

There have also been documented impacts on ecological receptors. In Knox County, Kentucky, retention pits holding hydraulic fracturing flowback fluids overflowed into Acorn Fork Creek during the development of four natural gas wells, causing a decrease in pH and increase in conductivity.[1] Organics and metals including iron and aluminum formed precipitates in the stream, and fish and aquatic invertebrates were killed or displaced in a 2.7 km length of the stream affected by the release (Papoulias and Velasco, 2013). A field report from the Pennsylvania Department of Environmental Protection (PADEP) described a leak in an overland pipe carrying a mixture of flowback and freshwater between two impoundments that impacted a 0.6 km length of a stream, in which 168 fish and 6 salamanders were killed (PA DEP, 2009b).

In some instances, chemical concentrations in drinking water resources impacted or potentially impacted by hydraulic fracturing activity exceeded their respective primary or secondary maximum contaminant level (MCL), or health advisory levels provided by the EPA's National Primary Drinking Water Regulations (NPDWRs) and Drinking Water Standards and Health Advisories (DWSHA) tables (U.S. EPA, 2012b), indicating that these chemicals are present at levels that may impact human health.[2] Examples will be discussed in Section 9.5. These studies generally did not indicate the contaminated water was used directly for human consumption, so it is not clear that people are being exposed to these chemicals at these levels. Nevertheless, these studies indicate that hydraulic fracturing activity may contribute to the entry of chemicals into drinking water resources at potentially harmful levels.

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[1] "Flowback" refers to fluids containing predominantly hydraulic fracturing fluid that return from a well to the surface. Flowback is a type of produced water. See Chapter 7 for more details.

[2] Maximum contaminant level (MCL): The highest level of a contaminant that is allowed in drinking water. MCLs are set as close to the maximum contaminant level goal (MCLG) as feasible using the best available analytical and treatment technologies and taking cost into consideration. MCLs are enforceable standards. The MCLG is a non-enforceable health benchmark goal which is set at a level at which no known or anticipated adverse effect on the health of persons is expected to occur and which allows an adequate margin of safety (U.S. EPA, 2012b).

Risk assessment and risk management decisions will be informed by scientific information on the toxicity of chemicals in hydraulic fracturing fluid and wastewater. The U.S. House of Representatives' Committee on Energy and Commerce Minority Staff released a report in 2011 noting that more than 650 products (i.e., chemical mixtures) used in hydraulic fracturing contain 29 chemicals that are either known or possible human carcinogens or are currently regulated under the Safe Drinking Water Act (House of Representatives, 2011). More recently, several studies have performed a reconnaissance of toxicity and/or physicochemical property data for specific subsets of chemicals used in hydraulic fracturing fluids (Elliott et al., 2016; Wattenberg et al., 2015; Stringfellow et al., 2014; Colborn et al., 2011), and have provided discussion on the hazards inherent to these chemicals. In all cases, authors reported toxicity data was not available for many of the chemicals assessed in these studies, with some studies indicating significant data gaps. For instance, Wattenberg et al. (2015) evaluated 168 chemicals commonly used in hydraulic fracturing fluids in North Dakota, and reported that 59% did not have chronic toxicity data available, and 35% did not have acute toxicity data available. Elliott et al. (2016) performed a systematic evaluation of reproductive and developmental toxicity for 1021 chemicals used in hydraulic fracturing fluids or detected in wastewater, and found this toxicity information was lacking for 76% of these chemicals.

Overall, while combined evidence suggests hydraulic fracturing has the potential to impact human health via contamination of drinking water resources, the actual public health impacts are not well understood and not well documented. Available information indicates there are many chemicals within the hydraulic fracturing water cycle that are known to be hazardous to human health, as well as hundreds of chemicals for which toxicological data is limited or unavailable.

In this chapter, our primary goal is to evaluate the availability of toxicity data for a list of chemicals used in hydraulic fracturing fluids or present in produced water, focusing primarily on toxicity values from sources that meet the criteria for inclusion in this assessment, and to highlight chemicals that may pose human health hazards.

## 9.3 Identification of Chemicals Associated with the Hydraulic Fracturing Water Cycle

As the initial step towards evaluating the hazards of chemicals in the hydraulic fracturing water cycle, the EPA compiled a list of chemicals used in or released by hydraulic fracturing operations across the country.[1] This section describes the compilation of that list. This consolidated list includes a total of 1,606 chemicals, and can be broken down into two sublists: (1) a list of chemicals used in hydraulic fracturing fluids, and (2) a list of chemicals detected in produced water from hydraulically fractured wells (Text Box 9-2).

This list demonstrates the range and variety of chemicals that are associated with the hydraulic fracturing industry. These chemicals should not be considered unique to the hydraulic fracturing

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

industry; many of the chemicals used in hydraulic fracturing fluids are widely used industrial chemicals, and many of the chemicals in produced water are naturally occurring. Although this list represents the best information available to the EPA at the time of the assessment, it should not be considered comprehensive. It is likely that, as industry practices change, chemicals may be used or detected that are not included on these lists. Some additional limitations to this chemical list are described in the subsections below.

**Text Box 9-2. The EPA's List of Chemicals Identified in Hydraulic Fracturing Fluids and/or Produced Water.**

This chemical list progressed through multiple iterations as the EPA's hydraulic fracturing study was developed, culminating in the list of 1,606 chemicals presented in this report.

The first iteration of this chemical list was published in the interim progress report (U.S. EPA, 2012h), and included 1,026 chemicals that were identified from ten sources of information. Seven of these information sources were documents from federal and state governmental units—including the EPA (U.S. EPA, 2011a, e, 2004a; Material Safety Data Sheets), the U.S. House of Representatives (House of Representatives, 2011), the New York State Department of Environmental Conservation (NYSDEC, 2011), and the Pennsylvania Department of Environmental Protection (PA DEP, 2010a)—which obtained data directly from industry. This includes a list of chemicals provided directly to the EPA by nine well operating companies, representing chemicals used in hydraulic fracturing fluids between 2005 and 2009, and a list of chemicals detected by these companies in produced water from 81 wells. The remaining three sources are as follows: a technical report prepared by the Gas Technology Institute for the Marcellus Shale Coalition, which is a drilling industry trade group (Hayes, 2009); a peer-reviewed journal article by Colborn et al. (2011); and the FracFocus Chemical Disclosure Registry, which is a national hydraulic fracturing chemical registry developed by the Ground Water Protection Council and the Interstate Oil and Gas Compact Commission (GWPC, 2012).

In the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d), this chemical list was updated to 1,173 chemicals. The updated chemical list includes the 1,026 chemicals published in the progress report, along with additional chemicals that were identified in the EPA FracFocus 1.0 report (U.S. EPA, 2015a).

For the final version of this assessment, the list has again been updated to include additional chemicals in produced water, which were identified from 18 additional literature sources. The final list includes a total of 1,606 chemicals that have been reported as used in hydraulic fracturing fluids or detected in produced water. The complete list of sources used to compile the final chemical list is provided in Appendix Table H-1. To the extent possible, after chemicals were identified from the sources in Table H-1, the EPA verified the identity of the chemicals used in hydraulic fracturing fluids and detected in produced water of hydraulically fractured wells as described in Appendix Section H.1.

### 9.3.1 Chemicals Used in Hydraulic Fracturing Fluids

Of the 1,606 total chemicals, the EPA identified 1,084 chemicals as being used in hydraulic fracturing fluids. This list was originally introduced in Chapter 5 of this assessment (Section 5.4), which describes some of the chemical classes and their purpose, and identifies the most frequently used chemicals. This list of 1,084 chemicals is shown in Appendix Table H-2.

Although a total of 8 sources were used to identify the list of chemicals used in hydraulic fracturing fluids, only one source—the EPA analyses based on disclosures submitted to FracFocus—had sufficient information for estimating the frequency with which these chemicals were used (Section 5.4, Text Box 5-1).[1] Of the 1,084 chemicals, 688 were identified in the EPA FracFocus 1.0 report (U.S. EPA, 2015a).[2] Frequency of use for individual chemicals ranged from low (480 chemicals on the list were reported in less than 1% of disclosures nationally) to very high (methanol was reported in 73% of disclosures nationally).

As discussed in Chapter 5, this list provides valuable information on the chemicals used in hydraulic fracturing fluids, but should not be considered complete. For example, in the analysis of the disclosures submitted to the FracFocus 1.0 registry, the EPA was only able to assign standardized chemical names to 65% of ingredient records. The remaining 35% of ingredient records did not have valid CASRNs and were excluded from the analysis because they could not be assigned a standardized chemical name (U.S. EPA, 2015a). In a more recent analysis of data reported to the FracFocus registry through April 2015, Konschnik and Dayalu (2016) found that 80% of chemicals had valid CASRN. That analysis identified an additional 263 CASRNs that are not on the EPA's list of chemicals used in hydraulic fracturing fluids (Dayalu and Konschnik, 2016).

Industry use of CBI is another factor that likely limits the completeness of this chemical list and introduces uncertainty. For example, companies submitting to FracFocus 1.0 were not required to disclose chemicals claimed as CBI. EPA determined that approximately 70% of the disclosures submitted to FracFocus 1.0 contain at least one CBI chemical, and for those disclosures, the average number of CBI chemicals per disclosure was five. Overall, 11% of ingredients were reported to FracFocus 1.0 as CBI (U.S. EPA, 2015a). Konschnik and Dayalu (2016) report a 5.6% increase in the number of CBI ingredients, as well as an increase in the number of disclosures reporting the use of at least one CBI ingredient (Section 5.4; Text Box 5-2).

Although FracFocus disclosures do not provide the name or CASRN of CBI chemicals, the chemical family is sometimes provided. The EPA determined that 79% of CBI ingredient records submitted to FracFocus 1.0 had enough information to partially define the chemical and assign it to a chemical family (U.S. EPA, 2015a). This resulted in the designation of 448 standardized chemical families to which these chemicals could be assigned. The most common standardized chemical families for CBI ingredients were oxyalkylated alcohol (4.7% of CBI ingredient records), petroleum distillates (4.0% of CBI ingredient records), and quaternary ammonium compounds (3.6% of CBI ingredient records) (U.S. EPA, 2015a) (Appendix Table B-1). These standardized chemical family designations are not discussed further in this chapter, but may be useful for site-specific risk assessment, as they

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[1] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN. In the interest of including as many chemicals as possible, this analysis includes chemicals that do not have valid concentration data. The 692 chemicals includes 16 chemicals that are listed as being used as proppants.

[2] EPA analyses based on disclosures submitted to FracFocus identified 692 unique CASRN. Of these 692, we determined that 4 chemicals are listed under two different CASRN (indicated in the footnote of Appendix Table H-2). Frequency of use data is therefore available for 688 chemicals that were included on EPA's list of chemicals in hydraulic fracturing fluids.

may provide insight into potential physicochemical properties and toxicity of CBI chemicals used at a particular site.

### 9.3.2 Chemicals Detected in Produced Water

Of the 1,606 total chemicals, the EPA identified 599 as having been detected in produced water. Included among these chemicals are naturally occurring organic compounds, metals, radionuclides, industrial chemicals, and pesticides. These chemicals were originally introduced in Chapter 7 of this assessment, and were compiled from a total of 21 sources. Seventy-seven of the total 599 chemicals in produced water were also identified by at least one of the sources in Appendix H as being used in hydraulic fracturing fluid. However, the EPA used different sets of sources to identify chemicals used in hydraulic fracturing fluids versus those detected in produced water, and there is not a matched comparison between the chemicals used in hydraulic fracturing fluids and returned in produced water at each particular well. Therefore, it is difficult to draw direct comparisons between these two chemical lists, or to use these lists to draw conclusions on the persistence of chemicals in produced water from hydraulically fractured wells. The list of 599 chemicals identified in produced water is shown in Appendix Table H-4.

Although this list provides useful information on the chemical composition of produced water, it is not likely that the data sources were able to capture all of the chemicals present. Chemicals and their metabolites may go undetected in produced water because they were not targeted in the analytical protocols, they were below the limit of detection, or because no standard analytical method exists. Additionally, as discussed in Chapter 7, the composition and concentration of chemicals in produced water will differ depending upon factors like the geology of the formation, the chemicals used for hydraulic fracturing, and the amount of time that has elapsed since hydraulic fracturing. There is therefore expected to be a high degree of local and temporal variation in these chemicals, and there was not sufficient information to determine the frequency with which these chemicals were detected on a national basis.

Concentration data in produced water are available for 175 of these 599 chemicals (Appendix E), including inorganic contributors to salinity (Appendix Tables E-4 and E-5), metals (Appendix Tables E-6 and E-7), radioactive constituents (Appendix Table E-8), and organic constituents (Appendix Tables E-9, E-11, E-12, and E-13). The remaining chemicals were detected in produced water, but concentration was not reported. For these chemicals with concentration data, the measured concentrations spanned several orders of magnitude. For instance, for organic chemicals in produced water from the Marcellus shale formation (Appendix Table E-11), average or median measured concentrations ranged from 2.7 μg/L for N-nitrosodiphenylamine to 400 μg/L for carbon disulfide.

## 9.4 Toxicological and Physicochemical Properties of Hydraulic Fracturing Chemicals

As the next step towards evaluating the hazards of chemicals in the hydraulic fracturing water cycle, toxicological and physicochemical data were collected as available for each of the chemicals identified in Appendix H. This section describes the compilation of these data, and discusses the extent to which toxicological and physicochemical property data are available for this list of chemicals.

The primary focus of the toxicological analysis in this chapter is on the availability of chronic oral RfVs and OSFs from sources that met the EPA's criteria for inclusion in this study. Qualitative cancer classifications were also identified from these sources when available. This is not intended to be an exhaustive compilation of toxicity values for this chemical list. Rather, it is intended to be a reconnaissance of high-quality toxicological information that met the EPA's criteria for inclusion in this study. If a source of RfVs, OSFs, or qualitative cancer classifications was not included here, that only means that it did not meet the criteria for the purposes of the EPA's study, which are described in this chapter in Section 9.4.1.

Section 9.4.1 describes the criteria used to identify and select RfVs, OSFs, and qualitative cancer classifications, and describes the availability of these toxicological data for the chemicals on the EPA's list of hydraulic fracturing-related chemicals. The next two sections describe additional sources of toxicological information, which may be useful for hazard evaluation when chronic oral RfVs and OSFs are not available: Section 9.4.2 describes the use of a QSAR model to estimate chronic oral toxicity, and Section 9.4.3 describes the availability of additional toxicological information on the EPA's ACToR database. Section 9.4.4 describes other available software tools and approaches that may be used by stakeholders for site-specific risk assessment, but are not utilized in this report. Section 9.4.5 discusses the methods used in this report to generate physicochemical property data, and presents the availability of physicochemical property data for the chemicals on the EPA's list. A brief overview of the toxicity values discussed in the chapter is presented in Text Box 9-3.

As a resource that can be used to support risk assessment at hydraulic fracturing sites, all of the selected RfVs, OSFs, qualitative cancer classifications, QSAR-based toxicity estimates, and physicochemical property data described in this chapter will be compiled into an electronic database that will be publicly accessible via the EPA's website. Additionally, the EPA's compilation of toxicity data for this chemical list has been discussed in two recent manuscripts, both of which focused on the list of 1,173 chemicals that was presented in the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d). Yost et al. (2016b) describes the compilation of RfVs and OSFs for the list of 1,173 chemicals. Yost et al. (2016a) describes the use of a QSAR model to estimate toxicity for the list of 1,173 chemicals.

### Text Box 9-3. Toxicity Values for Hydraulic Fracturing-Related Chemicals.

Here we provide a brief description of the toxicity values that are presented in this chapter, and how they should be interpreted and used to evaluate chemical hazards. Formal definitions of these terms are footnoted in the chapter and can also be found in the glossary (Appendix J).

**Reference value (RfV):** RfVs are health-protective values, which describe the dose of a chemical that is likely to be without an appreciable risk of adverse health effects. In general, lower RfVs indicate greater toxicity; however, comparison of RfVs among a set of chemicals requires careful consideration. RfVs are developed by considering the full database of epidemiological and experimental studies available for a particular chemical.

*(Text Box 9-3 is continued on the following page.)*

### Text Box 9-3 (continued). Toxicity Values for Hydraulic Fracturing-Related Chemicals.

These data are used to identify the critical effect, which is the first adverse effect, or its known precursor, that occurs as the dose rate increases (U.S. EPA, 2011f). An RfV is then derived by starting with a quantitative point of departure (POD), which is the toxicological dose-response point that marks the beginning of a low-dose extrapolation for the critical effect, and applying uncertainty factors (UFs) to derive a value for the protection of human health. UFs are applied to account for 5 areas of uncertainty: (1) intraspecies variability; (2) interspecies uncertainty; (3) extrapolation from a subchronic study; (4) extrapolating from a no-observed-adverse-effect level (NOAEL); and (5) deficiencies in the database. A UF of 1, 3, or 10 can be applied for any of these areas of uncertainty depending upon the amount and/or type data available, up to a maximum total UF of 3,000 (U.S. EPA, 2002). Thus, a chemical with a low RfV may reflect high uncertainty in the value, and not necessarily the toxicity of the chemical. Chemicals with a lower total UF generally have more reliable and robust health effect information.

**Oral slope factor (OSF):** An OSF is a measure of the increased cancer risk from a lifetime oral exposure to an agent. Higher OSFs indicate greater carcinogenic potency. As with RfVs, OSFs are developed by considering the full database of epidemiological and experimental studies for a particular chemical, and evaluating the increase in cancer incidence as dose rate increases. OSFs should be considered in conjunction with qualitative cancer classifications, which characterize the weight of evidence regarding the agent's potential to cause cancer in humans.

**No-observed-adverse-effect level (NOAEL):** NOAEL is defined as the highest exposure level at which there are no biologically significant increases in the frequency or severity of adverse effect between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered adverse or precursors of adverse effects (U.S. EPA, 2011f).

**Lowest-observed-adverse-effect level (LOAEL):** LOAEL is defined as the lowest exposure level at which there are biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control group (U.S. EPA, 2011f). Lower LOAELs indicate greater toxicity.

**Maximum contaminant level (MCL):** MCLs are the highest level of a contaminant that is allowed in drinking water. MCLs are set as close to the maximum contaminant level goal (MCLG) as feasible using the best available analytical and treatment technologies and taking cost into consideration. MCLs are enforceable standards. The MCLG is a non-enforceable health benchmark goal which is set at a level at which no known or anticipated adverse effect on the health of persons is expected to occur and which allows an adequate margin of safety (U.S. EPA, 2012b). Whereas RfVs, LOAELs, and NOAELs are expressed in terms of dose (mg/kg-day), MCLs are expressed in terms of the concentration of an agent in water (μg/L).

### 9.4.1 Reference Values (RfVs), Oral Slope Factors (OSFs), and Qualitative Cancer Classifications

For the purpose of this study, the EPA's primary goal was to identify high quality toxicity values that met the criteria for inclusion in this study, and that could be used by stakeholders to support the risk assessment of hydraulic fracturing chemicals (Text Box 9-1). Briefly, the sources of RfVs, OSFs, and qualitative cancer classifications selected by the EPA for the purposes of this chapter met the following key criteria:

1. The body or organization generating or producing the peer-reviewed RfVs, peer-reviewed OSFs, or peer-reviewed qualitative assessment must be a governmental or intergovernmental body.
2. The data source must include peer-reviewed RfVs, peer-reviewed OSFs, or peer reviewed qualitative assessments.
3. The RfVs, OSFs, or qualitative assessments must be based on peer-reviewed scientific data.
4. The RfVs, OSFs, or qualitative assessments must be focused on protection of the general public.
5. The body generating the RfVs, OSFs, or qualitative assessments must be free of conflicts of interest with respect to the chemicals for which it derives reference values or qualitative assessments.

These five criteria were developed by the EPA specifically for the purpose of this assessment, and are similar to the EPA Office of Solid Waste and Emergency Response (OSWER) recommendations for selecting toxicity values in conducting site-specific risk assessments (Regional Tier 3 Toxicity Value Workgroup, 2013; U.S. EPA, 2003a, 1989).[1] The OSWER directives provide recommendations on the appropriate sources of toxicity values and toxicological information that should be considered in risk assessments, and were designed to recognize toxicity values that were developed using the best available scientific information. In addition, these directives outline references to various resources that provide guidance on the approaches and issues considered in deriving toxicity values. This type of information can be especially important in cases in which multiple sources of toxicity values need to be considered or evaluated, or in which a value needs to be developed. More detail on these criteria for selection and inclusion of data sources, as well as the full list of data sources that were considered for this study, are available in Appendix G.

Table 9-1 shows the data sources that met these five criteria for the selection of toxicological information. The federal databases of RfVs or OSFs that met these criteria are the EPA's Integrated Risk Information System (IRIS) database, the EPA's Provisional Peer-Reviewed Toxicity Value (PPRTV) database, the EPA's Human Health Benchmarks for Pesticides (HHBP) database, and the Agency for Toxic Substances and Disease Registry (ATSDR) database. IRIS and PPRTV also provide qualitative cancer classifications. One state source of RfVs and OSFs, the California Environmental Protection Agency (CalEPA) Toxicity Criteria Database, met the criteria for inclusion.[2] One intergovernmental source of RfVs, the World Health Organization (WHO) International Programme on Chemical Safety (IPCS) Concise International Chemical Assessment Documents (CICAD), met the criteria for inclusion. The International Agency for Research on Cancer (IARC) and U.S. National Toxicology Program (NTP) Report on Carcinogens (RoC) also met the criteria and were used as additional sources for qualitative cancer classifications.

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[1] OSWER changed its name to the Office of Land and Emergency Management (OLEM), effective December 15, 2015.

[2] 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.

**Table 9-1. Sources of selected RfVs, OSFs, and qualitative cancer classifications.**

| Type of toxicological Information | Data source | Website |
|---|---|---|
| RfVs, OSFs, and qualitative cancer classifications | EPA Integrated Risk Information System (IRIS) database | http://cfpub.epa.gov/ncea/iris/index.cfm?fuseaction=iris.showSubstanceList |
| RfVs, OSFs, and qualitative cancer classifications | EPA Provisional Peer-Reviewed Toxicity Value (PPRTV) database | http://hhpprtv.ornl.gov/index.html |
| RfVs, OSFs | EPA Human Health Benchmarks for Pesticides (HHBP) database | http://iaspub.epa.gov/apex/pesticides/f?p=HHBP:home |
| RfVs | Agency for Toxic Substances and Disease Registry (ATSDR) Minimum Risk Levels | http://www.atsdr.cdc.gov/toxprofiles/index.asp#bookmark05 |
| RfVs, OSFs | California Environmental Protection Agency (CalEPA) Toxicity Criteria Database | http://oehha.ca.gov/tcdb/index.asp |
| RfVs | World Health Organization (WHO) International Programme on Chemical Safety (IPCS) Concise International Chemical Assessment Documents (CICAD) | http://www.who.int/ipcs/publications/cicad/en/ |
| Qualitative cancer classifications | National Toxicology Program (NTP) 13th Report on Carcinogens (RoC) | https://ntp.niehs.nih.gov/pubhealth/roc/ |
| Qualitative cancer classifications | International Agency for Research on Cancer (IARC) Monographs | http://monographs.iarc.fr/ |

In addition to the sources in Table 9-1, we also consulted the NPDWRs and DWSHA tables (U.S. EPA, 2014a) to determine whether the chemicals on this list are regulated as drinking water contaminants. NPDWRs provide a list of MCLs, which are legally enforceable standards on the concentration of a substance that is allowed in drinking water under the Safe Drinking Water Act. In this chapter, MCL values are referenced as a means of comparison with reported concentration data where appropriate, and are reported in Appendix G and are compiled on the EPA's electronic database for the hydraulic fracturing study.

As noted above, this chapter focuses on the presentation and use of chronic RfVs. Chronic RfVs account for the potential that chemical exposure may be continuous, in low concentration, and over a longer duration. In the absence of reliable information on the potential duration of chemical exposure, this is a conservative assumption for the protection of human health. Chronic RfVs are also lower than less-than-chronic RfVs (e.g., acute, intermediate, or subchronic toxicity values), and are therefore more health protective. For these reasons, chronic RfVs are generally preferred as the default by risk assessors when conducting site-specific risk assessments (U.S. EPA, 1989) and when developing regional screening levels (U.S. EPA, 2016b). In contrast, acute RfVs are more applicable for single exposures and/or exposures of limited frequency to high concentration and shorter durations (e.g., emergencies). However, the availability of less-than chronic RfVs are also presented for the sake of completeness.

Some chemicals had chronic oral RfVs or OSFs available from more than one of the sources in Table 9-1. For these chemicals, we selected a single value for use in this chapter by applying a modification of the EPA OSWER Directives 9285.7-53 and 9285.7-86 tiered hierarchy of toxicity values (U.S. EPA, 2003a). A single RfV and/or OSF was selected from the sources in this order: HHBP (pesticides only), IRIS, PPRTV, ATSDR, and then other available values. The RfVs considered from these sources included chronic oral reference doses (RfDs) from the IRIS, PPRTV, and HHBP programs; chronic oral minimal risk levels (MRLs) from ATSDR; oral maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intakes (TDIs) from CICAD.[1,2,3,4,5]

Of the 1,606 chemicals identified by the EPA, 173 (11%) have federal, state, or international chronic oral RfVs and/or OSFs from sources listed in Table 9-1. Chronic oral RfVs and/or OSFs from the selected sources are lacking for the remaining 1,433 (89%) chemicals that the EPA has identified as associated with hydraulic fracturing. All available chronic oral RfVs and OSFs from the sources listed in Table 9-1 are tabulated in Appendix G. Chronic oral RfVs and OSFs for chemicals used in hydraulic fracturing fluids are listed in Appendix Tables G-1a through G-1c, and chronic oral RfVs and OSFs for chemicals reported in hydraulic fracturing flowback or produced water are listed in Appendix Tables G-2a through G-2c. The EPA's IRIS database was the most abundant source of these toxicity values.

Overall, when chemicals in hydraulic fracturing fluid and chemicals in produced water are considered separately, the availability of chronic RfVs and OSFs can be summarized as follows:

- For the 1,084 chemicals used in hydraulic fracturing fluid, chronic oral RfVs or OSFs from at least one of the selected federal, state, and international sources were available for 98 chemicals (9%). From the US federal sources alone, chronic oral RfVs were available for 81 chemicals (7%), and OSFs were available for 15 chemicals (1%).

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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 that are pesticides.

[2] An 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. It can be derived from a NOAEL, LOAEL, or benchmark dose, with uncertainty factors generally applied to reflect limitations of the data used. Generally used in the EPA's non-cancer health assessments (U.S. EPA, 2011f). This estimate is expressed in terms of mg/kg-day.

[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), non-cancerous 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 (ATSDR, 2016). Chronic MRL: Duration of exposure is 365 days or longer. This estimate is expressed in terms of mg/kg-day.

[4] An MADL is the maximum allowable daily level of a reproductive toxicant at which the chemical would have no observable adverse reproductive effect, assuming exposure at 1,000 times that level (OEHHA, 2012). This estimate is expressed in terms of µg/day.

[5] A TDI is an estimate of the intake of a substance, expressed on a body mass basis, to which an individual in a (sub) population may be exposed daily over its lifetime without appreciable health risk (WHO, 2015). This estimate is expressed in terms of mg/kg-day.

- For the 599 chemicals reported in produced water, chronic oral RfVs or OSFs from at least one of the selected federal, state, and international sources were available for 120 chemicals (20%). From the US federal sources alone, chronic oral RfVs were available for 97 chemicals (16%), and OSFs were available for 30 chemicals (5%).

In addition to these chronic values, some of the chemicals also have less-than-chronic oral RfVs available from the sources listed in Table 9-1. Subchronic, acute, or intermediate oral RfVs were identified for 103 chemicals on the consolidated list, including 60 chemicals used in hydraulic fracturing fluid (Appendix Table G-1d), and 73 chemicals reported in produced water (Appendix Table G-2d). The majority of these chemicals also had chronic oral RfVs available, although there were 10 chemicals that had less-than-chronic oral RfVs but lacked a chronic oral RfV. All of these less-than-chronic RfVs were found on the PPRTV, ATSDR, or HHBP databases. As stated above, chronic values more protective of human health than less-than-chronic values, and are generally preferred for risk assessment. These less-than-chronic values are therefore not discussed further in this report, but are provided in Appendix G as supporting information.

Of the 1,606 chemicals identified by EPA, 207 (13%) had a qualitative cancer classification available from at least one of the sources listed in Table 9-1, which include IRIS, PPRTV, IARC, and RoC. These classifications are based on the weight-of-evidence (WOE) that a chemical causes cancer in humans. Of these 207 chemicals:

- 21 were reported by at least one source to be a known carcinogen in humans.
- 66 were reported by at least one source to be a probable or possible carcinogen in humans. These chemicals have been demonstrated to be carcinogenic in animal models, but have limited or insufficient data to adequately assess carcinogenicity in humans.
- 117 were reported to be not classifiable as to carcinogenicity in humans. These chemicals have been evaluated by at least one of these sources for their potential to cause cancer, but had inadequate evidence from human exposure and animal studies to assess carcinogenic potential.
- 3 were reported as not likely to be a human carcinogen.

The complete list of chemicals with qualitative cancer classifications are shown in Appendix Table G-1e (chemicals in hydraulic fracturing fluids) and Appendix Table G-2e (chemicals in produced water).

### 9.4.2 Estimating Toxicity Using Quantitative Structure Activity Relationship (QSAR) Modeling

Because the majority of chemicals identified in this report do not have RfVs and/or OSFs from the selected sources, it is likely that risk assessors at the local and regional level may turn to alternative sources of toxicological information. One potential resource is QSAR modeling software, which is able to provide estimates or predictions of toxicity based on chemical structure. A key advantage to QSAR models is that they are able to rapidly and inexpensively estimate toxicity values for chemicals. A disadvantage is that QSAR estimates may be of higher uncertainty and less reliable than values generated using traditional toxicological methods. However, because they increase the available pool of toxicity information, QSAR estimates may be a useful resource for risk assessors that are faced with evaluating potential exposures to data-poor chemicals.

A recent study by Yost et al. (2016a) used TOPKAT (Toxicity Prediction by Komputer Assisted Technology) QSAR software to estimate toxicity for the EPA's list of chemicals used in hydraulic fracturing fluids or detected in produced water, and evaluated how effectively these toxicity estimates could be used to rank chemicals based on toxicity. The chemical list examined in this study is the list of 1,173 chemicals published in the external review draft of the EPA's hydraulic fracturing study report (U.S. EPA, 2015d) (Text Box 9-2), so the full list of 1,606 chemicals was not assessed using the QSAR model. TOPKAT is commercially available QSAR software that is able to estimate the rat chronic oral lowest-observed-adverse-effect level (LOAEL), which is the LOAEL measured in a rat model following chronic oral exposure to a chemical.[1]

The authors of this study used TOPKAT to generate rat chronic oral LOAEL estimates for EPA's list of chemicals, and assigned qualitative confidence scores (high, medium, or low) to each estimate based on parameters reported by the model. The authors then examined a list of 48 chemicals that had both a high-confidence TOPKAT LOAEL estimate and a chronic oral reference dose (RfD) from EPA's IRIS database. The authors ranked these 48 chemicals from most toxic to least toxic based on either TOPKAT LOAEL estimate or on IRIS chronic oral RfD, and then used Spearman rank correlation to examine the similarity between these chemical rankings.

Of the 1,173 hydraulic fracturing chemicals, TOPKAT was able to generate toxicity estimates for 515 (44%) of the chemicals, including 453 chemicals that are used in hydraulic fracturing fluids, and 86 chemicals that have been detected in produced water. The authors found a strong and statistically significant correlation between chemical rankings based on high-confidence TOPKAT LOAEL estimates and on IRIS chronic oral RfDs, indicating that high-confidence TOPKAT LOAEL estimates can effectively be used to rank chemicals based on toxicity when experimentally derived toxicity values are not available. Overall, TOPKAT LOAEL estimates were available for 417 chemicals in this study that lack chronic oral RfVs or OSFs from the sources identified by EPA. Of these, 389 were found to be high-confidence estimates.

When available, the high-confidence TOPKAT LOAEL estimates from Yost et al. (2016b) are discussed in this chapter as an additional resource that can be used to rank chemicals based on toxicity. Low- or medium-confidence TOPKAT LOAEL estimates are not shown in this chapter, as the use of these values for chemical ranking has not been validated.

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[1] LOAEL is defined as the lowest exposure level at which there are biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control group following chronic (lifetime) exposure.

### 9.4.3 Chemical Data Available from EPA's Aggregated Computations Toxicology Resource (ACToR) Database

An additional tool for obtaining information focused on toxicology and risk assessment is the EPA's ACToR database.[1] ACToR is a large data warehouse developed by the EPA to consolidate large and disparate amounts of public data on chemicals, including data on chemical identity, structure, physicochemical properties, in vitro assay results, and in vitro toxicology data. The primary goals of ACToR are to make information on chemical health effects and exposure potential readily accessible, to characterize chemical toxicological data gaps, and to provide a resource for model building to address data gaps in environmental risk information (Judson et al., 2012).

ACToR contains data on over 500,000 chemicals from over 2,500 data sources, covering many domains including hazard, exposure, risk assessment, risk management, and use. Data sources and collections in ACToR include the US EPA, National Institutes of Health (NIH), the Centers for Disease Control and Prevention (CDC), US Food and Drug Administration (FDA), State Agencies, the European Chemicals Agency (ECHA), corresponding government agencies in Canada (e.g., Health Canada), Europe and Japan, the World Health Organization (WHO), and non-governmental organizations (NGOs). Data within ACToR ranges from the federal RfVs and OSFs discussed in Section 9.3.1, which have undergone extensive peer review, to other toxicity values and study and test results that have undergone little to no peer review.

ACToR organizes these data into several levels of "assays" and "assay categories," which serve to classify data sets according to the nature of the data. For instance, the "Hazard" assay category includes all data that are associated directly or indirectly with toxicology experiments. The "Risk Management" assay category includes regulatory and non-regulatory risk management benchmarks. Considering the diversity and overlapping nature of the data resources within ACToR, a single data set may fall into multiple assay categories (Judson et al., 2012).

We searched the ACToR database for information related to the list of 1,606 hydraulic fracturing-related chemicals. Specifically, we searched within the "Hazard" and "Risk Management" assay categories of ACToR. Results of the query were then filtered to include the assays that are most relevant to chemical exposure via drinking water. These assays were assigned into the following nine data classes: carcinogenicity, dose response values, drinking water criteria, genotoxicity or mutagenicity, hazard identification, LOAEL/NOAEL, RfV, OSF, and water quality criteria.[2]

Of the 1,606 chemicals, it was found that 735 (46%) have some data available within these data classes on ACToR, with the total number of data points found for individual chemicals ranging from 1 to 243. Figure 9-2 shows the percentage of the total 1,606 chemicals that had data available in each of the nine ACToR data classes, and indicates the fraction of those chemicals that also had a chronic oral RfV or OSF available from at least one of the selected sources in Table 9-1. As can be seen in Figure 9-2, 37% of the chemicals had some information on hazard identification, 25% had

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[1] The ACToR database, including the full list of data collections and assays, is available at: http://actor.epa.gov.

[2] NOAEL is defined as the highest exposure level at which there are no biologically significant increases in the frequency or severity of adverse effect between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered adverse or precursors of adverse effects. Source: U.S. EPA (2011f).

information on carcinogenicity, and 24% had a LOAEL or NOAEL identified. A LOAEL and/or NOAEL identified from a well conducted dose-response study are often considered the minimum data needed for RfV derivation (U.S. EPA, 2002).

Focusing on the 1,433 chemicals that lacked a chronic RfV and/or OSF from the selected sources described in Section 9.3.1, 567 (40%) had available data within at least one of these data classes on ACToR. Thus, ACToR has a significant amount of potentially useful data on chemical hazards, including for some data-poor chemicals, and might help to fill data gaps in the ongoing effort to understand potential hazards of hydraulic fracturing chemicals.

It is outside the scope of this assessment to evaluate the quality and reliability of data within ACToR that has not already undergone peer review. Therefore, with the exception of data from the sources listed in Table 9-1, data from ACToR was not considered in the hazard evaluation presented in this chapter. However, as a potential resource for risk assessors, the tables in this chapter indicate whether a chemical had data available on ACToR.

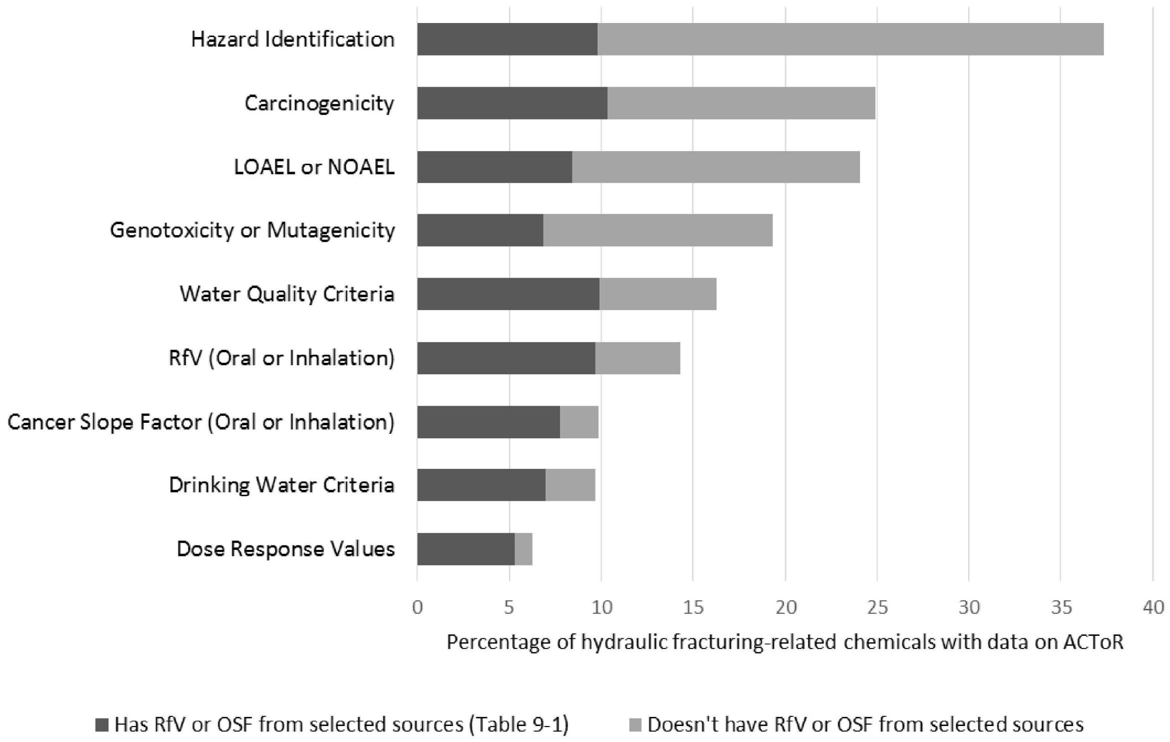


**Figure 9-2. Percentage of hydraulic fracturing-related chemicals (out of 1,606 total) with at least one data point in each ACToR data class.**

### 9.4.4 Additional Tools for Hazard Evaluation

In addition to the methods and approaches utilized in this chapter, there are other potential tools and approaches that could be used by stakeholders to prioritize and estimate toxicity of chemicals that have a limited toxicity database. We briefly describe three such approaches in Appendix G (Section G.4): the Threshold of Toxicological Concern (TTC) approach, the Organisation for

Economic Co-operation and Development (OECD) QSAR Toolbox, and the application of data from high throughput screening (HTS) assays. Toxicity predictions from these additional data sources can be either quantitative or qualitative, and may be used to fill and address gaps related to risk assessment.

Although these additional tools may be potentially useful for the evaluation of chemical hazards, they currently have limited utility in this chapter, and are not discussed further. The TTC approach requires an estimate of human intake, which is challenging for hydraulic fracturing-related chemicals, since the potential for human exposure is generally not clear. The OECD QSAR Toolbox is potentially useful for qualitative assessment, and may be useful for quantitative toxicity assessment as its human health hazard and repeated dose toxicity databases expand. HTS assays are an emerging technology, and the potential application of these data for human health risk assessment is not well understood. These tools would be more appropriately applied by stakeholders on a site-specific basis, as preliminary steps to identify potential chemicals of concern.

### 9.4.5 Physicochemical Properties

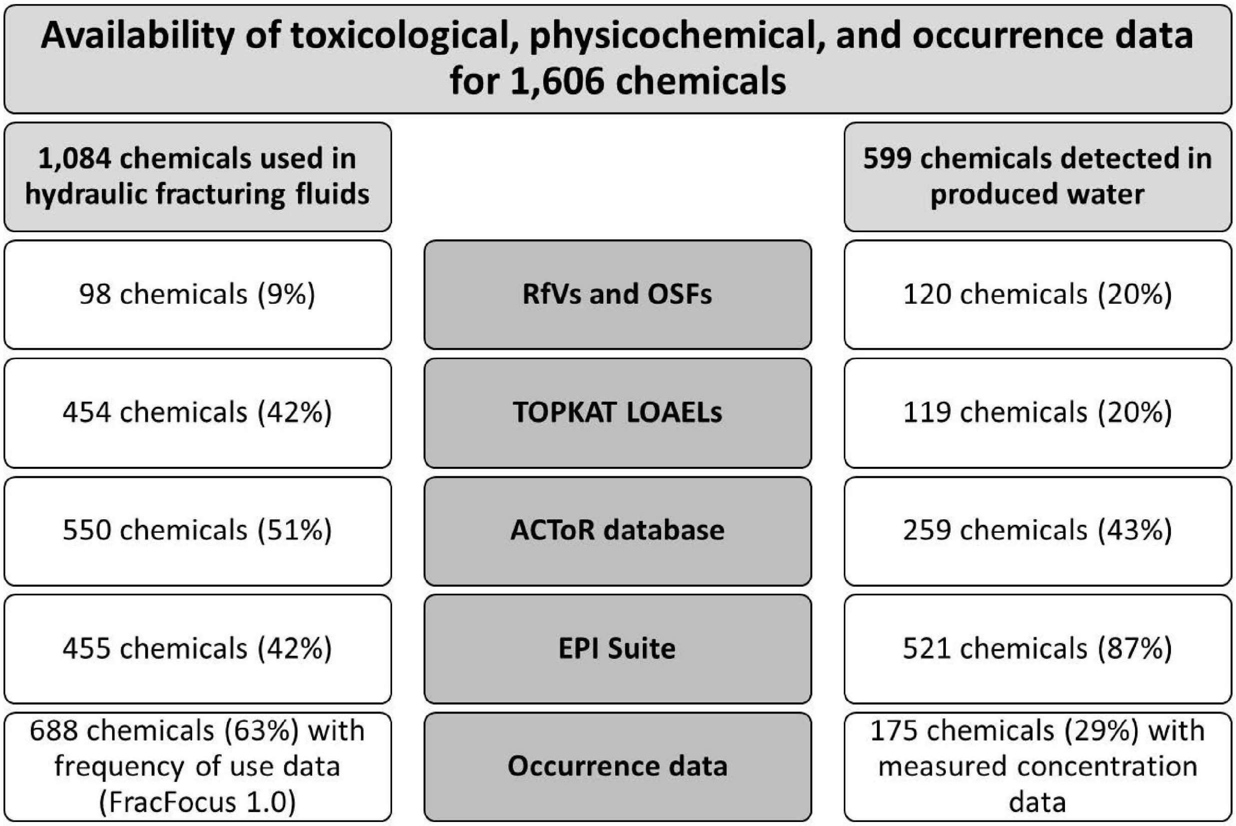
As presented in Chapter 5, EPI Suite™ software was used to generate data on the physicochemical properties of the hydraulic fracturing-related chemicals identified by EPA. EPI Suite provides an estimation of physicochemical properties based upon chemical structure, and will additionally provide experimentally measured values for these properties when they are available for a given chemical. For more details on this software and on the use of physicochemical properties for fate and transport estimation, see Chapter 5.

From the total list of 1,606 chemicals associated with hydraulic fracturing, EPI Suite was able to generate data on physicochemical properties for 917 (57%) of the chemicals (Appendix H). This includes 455 chemicals that are reported in hydraulic fracturing fluids, 521 chemicals that have been reported in produced water, and 59 chemicals that were both used in hydraulic fracturing fluids and reported in produced water. The remaining 689 chemicals on EPA's total list lacked the structural information necessary to generate estimates.

In addition to EPI Suite, two other software programs were consulted to generate physicochemical property data for EPA's list of hydraulic fracturing-related chemicals. QikProp (Schrodinger, 2012) and LeadScope (Leadscope Inc., 2012) are commercial products designed primarily as drug development and screening tools, which are able to estimate properties related to chemical fate and transport as well as pharmacokinetics. Properties generated by QikProp and LeadScope are generally more relevant to drug development than to environmental assessment. The properties generated by QikProp and LeadScope were not used in the analysis presented in this report, but will be compiled on the electronic database for EPA's hydraulic fracturing study.

### 9.4.6 Summary of Available Toxicological and Physicochemical Information for Hydraulic Fracturing Chemicals

Figure 9-3 summarizes the toxicological and physicochemical information that is available for the list of hydraulic fracturing chemicals identified by EPA in this study. This figure also summarizes the availability of data on the occurrence of these chemicals in hydraulic fracturing fluids (frequency of use) or in produced water (measured concentrations).



**Availability of toxicological, physicochemical, and occurrence data for 1,606 chemicals**

| 1,084 chemicals used in hydraulic fracturing fluids | | 599 chemicals detected in produced water |
|---|---|---|
| 98 chemicals (9%) | **RfVs and OSFs** | 120 chemicals (20%) |
| 454 chemicals (42%) | **TOPKAT LOAELs** | 119 chemicals (20%) |
| 550 chemicals (51%) | **ACToR database** | 259 chemicals (43%) |
| 455 chemicals (42%) | **EPI Suite** | 521 chemicals (87%) |
| 688 chemicals (63%) with frequency of use data (FracFocus 1.0) | **Occurrence data** | 175 chemicals (29%) with measured concentration data |

**Figure 9-3. Overall representation of the selected toxicological, physicochemical, and occurrence data available for the 1,606 hydraulic fracturing-related chemicals identified by the EPA.**

Overall, there is a clear paucity of chronic oral RfVs and OSFs for this list of chemicals, indicating that the majority of chemicals associated with hydraulic fracturing activity have not undergone significant toxicological assessment. QSAR-based toxicity estimates (TOPKAT LOAELs) were available for a larger number of these chemicals, and were often available for chemicals that lack chronic oral RfVs and OSFs. EPA's ACToR database offers additional toxicological data that may be useful for the hazard evaluation of these chemicals, although the quality and reliability of the data for these chemicals within ACToR was not evaluated here.

## 9.5 Hazard Identification of Hydraulic Fracturing Chemicals

This section focuses on the hazard identification of subsets of chemicals that were identified as being of particular interest in previous chapters of this report, or which otherwise may be of particular interest to risk assessors. For these chemicals, we summarize what is known about events that may lead to the entry of these chemicals into drinking water resources. We provide examples of recent studies that have reported these chemicals in drinking water resources, including examples in which these chemicals have been reported at concentrations exceeding MCLs. We then summarize the available toxicological data for these chemicals, including chronic oral RfVs, OSFs, cancer classifications, QSAR-based toxicity estimates (TOPKAT LOAELs), and the availability of relevant toxicological information from EPA's ACToR database—and indicate which chemicals are regulated by EPA as drinking water contaminates.

We focused on the following subsets of chemicals:

1. Chemicals used in hydraulic fracturing fluids (Chapter 5)
2. Organic chemicals that may be returned to the surface in produced water, including naturally occurring hydrocarbons such as BTEX (Chapter 7)
3. Inorganic chemicals that may be returned to the surface in produced water, including metals, inorganic ions, and technologically enhanced naturally occurring radioactive material (TENORM) (Chapter 7)
4. Methane in stray gas, which has been reported in drinking water resources in areas of hydraulic fracturing activity (Chapter 6)
5. Disinfection byproducts (DBPs) that may be formed from wastewater constituents (Chapter 8)
6. Banned chemicals reported in produced water, specifically organochlorine pesticides and polychlorinated biphenyls (PCBs).
7. Chemicals on EPA's consolidated list that were reported in both hydraulic fracturing fluids and produced water

The hazard identification for these subsets of chemicals is presented below.

### 9.5.1 Chemicals Used in Hydraulic Fracturing Fluids

Chapter 5 provided an overview of chemicals that are used in hydraulic fracturing fluids. These chemicals have the potential to enter drinking water resources through events such as spills of hydraulic fracturing fluids, injection of hydraulic fracturing fluids directly into groundwater, and leakoff of fluids into the formation. These chemicals may also persist in produced water, and may enter drinking water resources through spills or releases of produced water or inadequately treated wastewater.

Several recent field studies have detected chemicals that are commonly used in hydraulic fracturing fluids in groundwater near hydraulically fractured wells. In some cases, the origin of the chemicals could be clearly linked to hydraulic fracturing activity. For example, in Killdeer, North Dakota (Section 6.2.2.1), evidence strongly suggests a well blowout during hydraulic fracturing led to the contamination of a drinking water aquifer with tert-butyl alcohol, a degradation product of tert-butyl hydroperoxide used in hydraulic fracturing fluids at that site (U.S. EPA, 2015i). In groundwater monitoring wells in the Pavillion Field in Wyoming, Digiulio and Jackson (2016) reported detections of organic chemicals used in hydraulic fracturing fluids at that site, including 2-

butoxyethanol, naphthalene, 1,2,4-trimethylbenzene, diethylene glycol, methanol, ethanol, and isopropanol, likely as a result of shallow hydraulic fracturing in that region.

Other studies provide indirect evidence that chemical contaminants originated from hydraulic fracturing activity. For example, in the Marcellus Shale in Pennsylvania, Llewellyn et al. (2015) detected trace levels of 2-butoxyethanol in water wells near several hydraulically fractured wells, with multiple lines of evidence suggesting that the chemical originated from a surface spill or leak related to hydraulic fracturing activity. In northeastern Pennsylvania, Drollette et al. (2015) found trace concentrations of known constituents of hydraulic fracturing fluid in private residential groundwater wells, including di(2-ethylhexyl) phthalate, with evidence suggesting that the chemicals originated from known surface spills of hydraulic fracturing fluids. In the Barnett Shale, Texas, a survey of water quality in public and residential wells reported chemicals that are known to be used in hydraulic fracturing fluids, including methanol, ethanol, isopropanol, and propargyl alcohol, but it was not clear whether these chemicals originated from hydraulic fracturing activity or from other potential sources (Hildenbrand et al., 2015).

Table 9-2 shows the list of chemicals that were reported in at least 10% of disclosures nationally in the EPA FracFocus 1.0 project database (excluding water, quartz, and sodium chloride), and shows the noncancer toxicity data (chronic oral RfVs and TOPKAT LOAEL estimates) and ACToR data available for these chemicals.[1] Cancer information is provided in Table 9-3. Nine (26%) of these 34 chemicals have a chronic oral RfV available from at least one of the sources in Table 9-1. Chronic oral RfVs ranged from 0.002 mg/kg-day (propargyl alcohol) to 2 mg/kg-day (methanol and ethylene glycol). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, developmental toxicity (extra cervical ribs), reproductive toxicity, neurotoxicity, and decreased terminal body weight. Only one of these chemicals, sodium chlorite, is regulated in drinking water under the NPDWRs.

Of the 25 chemicals that lack chronic oral RfVs, 11 have high-confidence TOPKAT LOAEL estimates available. Of these, methenamine (~14% of disclosures) had the lowest TOPKAT LOAEL estimate, and choline chloride (~15% of disclosures) had the second lowest. All but five of these chemicals had at least some relevant toxicological data available on EPA's ACToR database.

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[1] The analysis of the FracFocus 1.0 project database presented in this chapter did not exclude chemicals that lacked valid concentration data, in order to present a more inclusive analysis of the potential toxicity of chemicals used in hydraulic fracturing fluids. The chemical list and percent disclosures listed for each chemical is therefore slightly different that those shown in Chapter 5 (Table 5-3), which excluded chemicals lacking valid concentration data.

**Table 9-2. 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. Chronic oral RfVs, TOPKAT LOAEL estimates, and availability of ACToR data are shown when available.**

Chemicals are ordered in the table, from high to low, based on their number of disclosures in the EPA FracFocus 1.0 project database. Water, quartz, and sodium chloride are excluded from this analysis. Asterisk (*) indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

| | | | Chronic oral RfV[b] | | | QSAR | ACToR |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **% of Disclosures[a]** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical effect[c]** | **TOPKAT LOAEL[d] (mg/kg)** | **# of data points[e]** |
| Methanol | 67-56-1 | 73% | 2 | IRIS | Extra cervical ribs | | 122 |
| Distillates, petroleum, hydrotreated light | 64742-47-8 | 67% | | | | | 4 |
| Hydrochloric acid | 7647-01-0 | 66% | | | | | 50 |
| Ethylene glycol | 107-21-1 | 47% | 2 | IRIS | Kidney toxicity | 130 | 102 |
| Isopropanol | 67-63-0 | 46% | | | | 81.4 | 26 |
| Diammonium peroxydisulfate | 7727-54-0 | 44% | | | | | 11 |
| Guar gum | 9000-30-0 | 39% | | | | | 2 |
| Sodium hydroxide | 1310-73-2 | 39% | | | | | 26 |
| Propargyl alcohol | 107-19-7 | 33% | 0.002 | IRIS | Renal and hepatotoxicity | | 42 |
| Glutaraldehyde | 111-30-8 | 33% | | | | 398 | 13 |
| Ethanol | 64-17-5 | 31% | | | | 59.2 | 182 |
| Potassium hydroxide | 1310-58-3 | 31% | | | | | 21 |
| Acetic acid | 64-19-7 | 25% | | | | 183 | 35 |
| Citric acid | 77-92-9 | 24% | | | | 55.8 | 25 |
| 2-Butoxyethanol | 111-76-2 | 23% | 0.1 | IRIS | Hemosiderin deposition in the liver | 707 | 44 |
| Solvent naphtha, petroleum, heavy arom. | 64742-94-5 | 21% | | | | | 5 |
| Naphthalene | 91-20-3 | 19% | 0.02 | IRIS | Decreased terminal body weight | 67.5 | 157 |

| | | | **Chronic oral RfV**[b] | | | **QSAR** | **ACToR** |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **% of Disclo-sures**[a] | **RfV (mg/kg -day)** | **Source of RfV** | **Critical effect**[c] | **TOPKAT LOAEL**[d] **(mg/kg)** | **# of data points**[e] |
| 2,2-Dibromo-3-nitrilopropionamide | 10222-01-2 | 16% | | | | 52.4 | |
| Choline chloride | 67-48-1 | 15% | | | | 20.8 | 24 |
| Phenol-formaldehyde resin | 9003-35-4 | 14% | | | | | |
| Carbonic acid, dipotassium salt | 584-08-7 | 14% | | | | 137 | 3 |
| Methenamine | 100-97-0 | 14% | | | | 12.3 | 15 |
| Thiourea, polymer with formaldehyde and 1-phenylethanone | 68527-49-1 | 13% | | | | | |
| 1,2,4-Trimethylbenzene | 95-63-6 | 13% | 0.01 | IRIS | Decreased pain sensitivity | 91.5 | 71 |
| Polyethylene glycol | 25322-68-3 | 13% | | | | | 5 |
| Polyethylene glycol nonylphenyl ether | 9016-45-9 | 13% | | | | | 4 |
| Quaternary ammonium compounds, benzyl-C12-16-alkyldimethyl, chlorides | 68424-85-1 | 12% | 0.44 | HHBP | Decreased body weight and weight gain | | 3 |
| Poly(oxy-1,2-ethanediyl)-nonylphenyl-hydroxy branched | 127087-87-0 | 12% | | | | | |
| Ammonium chloride | 12125-02-9 | 12% | | | | | 18 |
| Formic acid | 64-18-6 | 11% | 0.9 | PPRTV | Reproductive toxicity | | 72 |
| Tetrakis(hydroxy-methyl) phosphonium sulfate | 55566-30-8 | 11% | | | | 148 | 3 |
| Sodium chlorite* | 7758-19-2 | 11% | 0.03 | IRIS | Neuro-developmental effects | | 66 |

| Chemical Name | CASRN | % of Disclo-sures[a] | Chronic oral RfV[b] |  |  | QSAR | ACToR |
|---|---|---|---|---|---|---|---|
|  |  |  | RfV (mg/kg -day) | Source of RfV | Critical effect[c] | TOPKAT LOAEL[d] (mg/kg) | # of data points[e] |
| Alcohols, C12-14, ethoxylated propoxylated | 68439-51-0 | 11% |  |  |  | 1450 |  |
| Sodium persulfate | 7775-27-1 | 10% |  |  |  |  | 6 |

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; HHBP = Human Health Benchmarks for Pesticides; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] The FracFocus frequency of use data presented in this chapter is based on 35,957 FracFocus disclosures that were deduplicated, within the study time period (January 1, 2011 to February 28, 2013), and with ingredients that have a valid CASRN.

[b] Reference value (RfV): 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. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[c] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[d] TOPKAT LOAEL: The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2. Values are rounded to 3 significant figures.

[e] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

Table 9-3 shows the chemicals reported in at least 10% of disclosures nationally in the EPA FracFocus 1.0 project database that are considered by at least one of the sources in Table 9-1 to be known, probable, or possible human carcinogens. Ethanol is classified as a "carcinogenic to humans" (Group 1) by IARC. Naphthalene is classified by IARC as "possibly carcinogenic to humans" (Group 2B), and is classified by RoC as "reasonably anticipated to be a human carcinogen," while IRIS classifies naphthalene as having inadequate data to assess carcinogenic potential. Neither chemical has an available OSF.

### Table 9-3. List of OSFs and qualitative cancer classifications available for all carcinogenic chemicals reported to FracFocus 1.0 from January 1, 2011 to February 28, 2013 in 10% or more disclosures.

Includes all chemicals from Table 9-2 that are classified as known, probable, or possible human carcinogens by at least one of the sources in Table 9-1.

| | | OSF[a] | | Qualitative cancer classification | | | |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **OSF (per mg/kg-day)** | **Source of OSF** | **IRIS[b]** | **PPRTV[c]** | **IARC[d]** | **RoC[e]** |
| Ethanol | 64-17-5 | | | | | 1 | |
| Naphthalene | 91-20-3 | | | "Data are inadequate to assess human carcinogenic potential" | | 2B | RAHC |

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): 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. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

In addition to evaluating chemicals that are frequently used in hydraulic fracturing fluids, we also evaluated the availability of toxicological data for subsets of chemicals that are used less frequently on a national basis (Figure 9-4). For this analysis, we binned the chemicals according to frequency of use as identified from the EPA FracFocus 1.0 project database (>10% of disclosures, 5-10% of disclosures, 1-5% of disclosures, <1% of disclosures, or unknown frequency of use), and evaluated the percentage of chemicals within each bin that have available chronic oral RfVs or OSFs, TOPKAT LOAEL estimates, and relevant data on ACToR. This analysis demonstrates that the availability of chronic oral RfVs and OSFs is low across all of these subsets of chemicals. Proportionately, the availability of chronic oral RfVs, OSFs, and data on ACToR is slightly higher for chemicals that are used in >10% of disclosures, compared to chemicals that are used less frequently.

Of the chemicals on the EPA's list that had frequency of use data available from the EPA FracFocus 1.0 project database, the majority were used in <1% of disclosures (n=480), suggesting that potential exposure to these chemicals is more likely to be a local issue rather than a national issue. Given that the analysis of the EPA FracFocus 1.0 project database presented in this chapter was based on 35,957 disclosures, a chemical used in <1% of wells nationally could still be used in several hundred wells. Chemicals used infrequently on a national basis could still be used more frequently within certain areas or counties, increasing the potential for local exposure to that chemical.

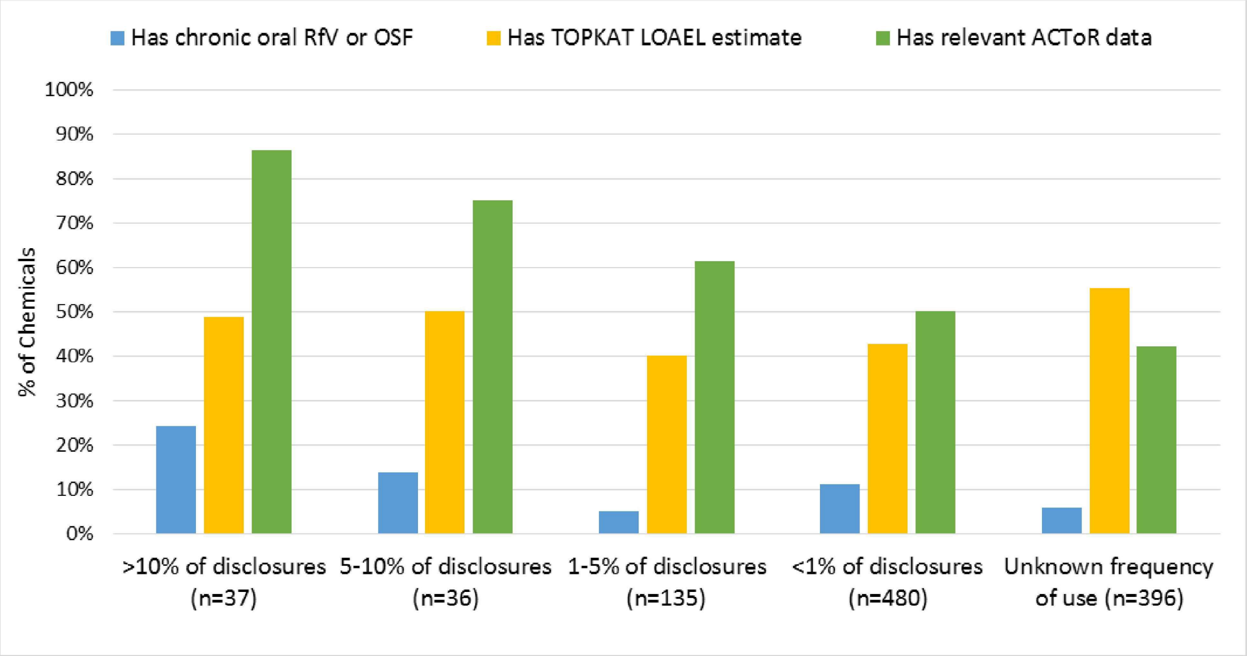

**Figure 9-4. Availability of toxicity data (chronic oral RfVs/OSFs, TOPKAT LOAEL estimates, and relevant data on ACToR) for subsets of chemicals used at various frequencies in hydraulic fracturing fluids, as determined based on the number of disclosures in the EPA FracFocus 1.0 project database.**

As described in Chapter 5, many of the chemicals used in hydraulic fracturing fluids can be classified as chemical mixtures. Among the most common chemical mixtures on EPA's list of chemicals are petroleum distillates (i.e., hydrocarbon solvents), which are complex mixtures of petroleum hydrocarbons.[1] Two of the most frequently used chemicals in Table 9-2 are petroleum distillates. (Petroleum) hydrotreated light distillates is a mixture of hydrocarbons having carbon numbers predominantly in the range of C9 through C16, and was reported as used in 67% of disclosures in the EPA FracFocus 1.0 project database. Heavy aromatic (petroleum) solvent naphtha is a mixture consisting predominantly of aromatic hydrocarbons in carbon fraction range of C9 through C16, and was reported as used in 21% of disclosures in the EPA FracFocus 1.0 project database. These petroleum distillates lack chronic oral RfVs or OSFs, and have little information available in ACToR. However, a methodology that describes the toxicity and derivation of surrogate

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[1] Total petroleum hydrocarbons (TPH) is a term used to describe a large family of several hundred chemical compounds that originally come from crude oil. TPH is a mixture of chemicals, but they are all made mainly from hydrogen and carbon, called hydrocarbons. TPH are divided into groups of petroleum hydrocarbons that act alike in soil or water. These groups are called petroleum hydrocarbon fractions. Each hydrocarbon fraction contains many individual chemicals. Some chemicals that may be found in TPH are hexane, jet fuels, mineral oils, benzene, toluene, xylenes, naphthalene, and fluorene, as well as other petroleum products and gasoline components. Source: ATSDR (2011).

toxicity values for such mixtures was developed by the Total Petroleum Hydrocarbon Criteria Working Group (TPHCWG) (Edwards et al., 1997). This indicator/surrogate approach uses a combination of toxicity data and existing RfVs on individual compounds and fraction-specific mixtures. Examples of compounds present in each fraction include: toluene, ethylbenzene, and styrene (C5-C8) and isopropylbenzene (cumene), naphthalene, fluorene, pyrene, and methylnaphthalene (C9-C16). No data was available for consideration for C>16. Applying their methodology, the TPHCWG derived surrogate aliphatic and aromatic oral toxicity values for fractions in the C5-C8, C9-C16, and C17-C35 ranges. For aromatics, the toxicity ranking was C9-C16 and C17-C35 > C5-C8; and for aliphatics, the toxicity ranking was C9-C16 > C17-C35 > C5-C8. As reviewed by the TPHCWG, compounds above C20 are likely not volatile or soluble in groundwater and will remain at the release site and compounds above C35 are typically not likely to be bioavailable by the oral route of exposure. These surrogate toxicity values are not included in EPA's analysis in this report, but this methodology might be useful for risk assessors at sites where these petroleum distillates are used.

We additionally note that several of the frequently used chemicals in Table 9-2 are designated as being "generally recognized as safe" (GRAS) for use in food additives or food contact substances by the U.S. Food and Drug Administration (FDA). This includes hydrochloric acid, guar gum, sodium hydroxide, sodium chloride, potassium hydroxide, acetic acid, citric acid, choline chloride, carbonic acid dipotassium salt, ammonium chloride, and formic acid. Overall, 103 chemicals on EPA's list of chemicals used in hydraulic fracturing fluids have GRAS designations by the FDA. GRAS chemicals may be used by hydraulic fracturing industry operators in an effort to avoid more hazardous chemicals and minimizes concern in the public perception (Loveless et al., 2011). However, GRAS determinations are often specific to certain conditions as expressed in the FDA GRAS Notification Database and therefore do not indicate that the same chemical is safe for use in hydraulic fracturing fluids. For instance, formic acid is considered GRAS for specific use in paper food packaging materials (U.S. FDA, 2016), but has a chronic oral RfD of 0.9 mg/kg-day based on reproductive effects (U.S. EPA, 2010b). For human health risk assessment in areas of hydraulic fracturing activity, hazard and dose-response relations for these chemicals need to be assessed in the context of the use and levels that are likely to be encountered in an appropriate exposure scenario.

### 9.5.2 Organic Chemicals in Produced Water

Chapter 7 discussed the detection of volatile and semi-volatile organic chemicals in produced water. Many of these chemicals, including the BTEX chemicals and related hydrocarbons, occur naturally in hydrocarbon formations and are characteristic of produced water from oil and gas production wells in both conventional and unconventional reservoirs. Some of these chemicals have anthropogenic origins, such as di(2-ethylhexyl) phthalate, which does not occur naturally but has known use in hydraulic fracturing fluids. Naphthalene is an example of a chemical that may occur naturally in hydrocarbon formations but is also used frequently in hydraulic fracturing fluids (19% of disclosures in the EPA FracFocus 1.0 project database; Table 9-2). These chemicals have the potential to enter drinking water resources through events such as spills of produced water, mechanical integrity failures, infiltration into groundwater from produced water storage pits, and persistence in inadequately treated wastewater.

Several recent field studies have reported these organic constituents in surface water and groundwater in areas of hydraulic fracturing activity. For example, the BTEX chemicals, diesel-range organics, gasoline-range organics, and naphthalene were detected in groundwater monitoring wells in Pavillion Field, Wyoming, likely as a result of legacy contamination from leaking unlined production fluid storage pits (Digiulio and Jackson, 2016). BTEX chemicals were also found to be elevated above their respective MCLs following spills by the oil and gas industry in Colorado, and were reduced to lower concentrations following remediation (Gross et al., 2013). Ferrar et al. (2013) reported mean concentrations of the BTEX chemicals in effluent from a centralized waste treatment (CWT) facility in Pennsylvania ranged from about 2 to 46 μg/L, with significantly lower concentrations observed after oil and gas well operators were asked to stop discharging waste at this facility (Text Box 8-1). In a survey of 500 private and public water supply wells overlying and adjacent to the Barnett Shale in Texas, Hildenbrand et al. (2015) reported that benzene concentrations exceeded their MCL in all 34 wells where benzene was detected, while toluene, ethylbenzene, and xylenes were prevalent at trace levels; the authors noted that BTEX detections occurred at a high rate in an area that houses a large number of underground injection wells for drilling waste disposal, but it was not clear that these chemicals originated from hydraulic fracturing activity or from another potential source.

As there were a large number of organic chemicals identified on EPA's list, this section focuses on the toxicological evaluation of those organic chemicals that had measured concentration data available in Appendix E and had at least some toxicity data available from the sources in Table 9-1, TOPKAT, or ACToR (69 chemicals total).[1] There were an additional 46 organic chemicals that had measured concentration data in Chapter 7 or Appendix E that did not have any toxicity data available. Organic chemicals that lacked concentration data and are not discussed here.

For this subset of 69 organic chemicals, noncancer toxicity values (chronic oral RfVs and high confidence TOPKAT LOAEL estimates) and ACToR data availability are shown in Table 9-4, and cancer information (OSFs and qualitative cancer classifications) are shown in Table 9-5. Chronic oral RfVs were available for 31 of these chemicals, and ranged from 0.001 mg/kg-day (pyridine) to 0.9 mg/kg-day (acetone). Critical effects for these chemicals include kidney/renal toxicity, hepatotoxicity, neurotoxicity, reproductive toxicity (decreased maternal weight gain), developmental toxicity (decreased offspring body weight, fetal toxicity), and decreased terminal body weight. Six of the chemicals in Table 9-4 are regulated as drinking water contaminants under the NPDWRs: the BTEX chemicals (benzene, ethylbenzene, toluene, xylenes), benzo(a)pyrene, and di(2-ethylhexyl) phthalate.

Of the 38 chemicals in Table 9-4 that lack chronic oral RfVs, 10 have high-confidence TOPKAT LOAEL estimates available. Several of these had similarly low LOAEL estimates: benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, dibenz(a,h)anthracene, and N-nitrosodiphenylamine. Notably, 33 of the chemicals

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[1] Note that chemical names presented in this chapter and in Appendix H sometimes differ from the chemical names presented with the concentration data in Appendix E. This is because Appendix E uses the chemical names provided by the original sources of chemical data, while this chapter and Appendix H use chemical names that were verified by EPA during the curation of the chemical list. See Appendix H for details on the curation of the chemical list.

in Table 9-4 were added to EPA's chemical list after the release of the external review draft (Text Box 9-2), and therefore were not included in the QSAR analysis (Section 9.4.2).

**Table 9-4. List of a subset of organic chemicals that have been detected in produced water, with respective chronic oral RfVs, TOPKAT LOAEL estimates, and availability of ACToR data shown when available.**

Includes organic chemicals that were identified on the EPA's list of chemicals in produced water (Appendix H) that have measured concentration data available in Appendix E and have at least some toxicity data available from the sources consulted by the EPA. Chemicals are ordered in the table from most toxic to least toxic based on chronic oral RfV. Chemicals without RfVs were ordered based on TOPKAT LOAEL, and then by number of data points on ACToR. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

| | | Chronic oral RfV[a] | | | QSAR estimate | ACToR |
|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical Effect[b]** | **TOPKAT LOAEL[c] (mg/day)** | **# of data points[d]** |
| Pyridine | 110-86-1 | 0.001 | IRIS | Increased liver weight | 69.5 | 114 |
| Benzidine | 92-87-5 | 0.003 | IRIS | Brain cell vacuolization; liver cell alterations in females | | 127 |
| 2,4-Dichlorophenol | 120-83-2 | 0.003 | IRIS | Decreased delayed hypersensitivity response | | 122 |
| Benzene* | 71-43-2 | 0.004 | IRIS | Decreased lymphocyte count | 77.6 | 238 |
| 2-Methylnaphthalene | 91-57-6 | 0.004 | IRIS | Pulmonary alveolar proteinosis | 103 | 52 |
| 1,3,5-Trimethylbenzene | 108-67-8 | 0.01 | IRIS | Decreased pain sensitivity | 63 | 76 |
| 1,2,4-Trimethylbenzene | 95-63-6 | 0.01 | IRIS | Decreased pain sensitivity | 91.5 | 71 |
| Chloroform | 67-66-3 | 0.01 | IRIS | Moderate/marked fatty cyst formation in the liver and elevated serum glutamic pyruvic transaminase (SGPT) | 47.1 | 221 |
| Naphthalene | 91-20-3 | 0.02 | IRIS | Decreased mean terminal body weight in males | 67.5 | 157 |
| 2,4-Dimethylphenol | 105-67-9 | 0.02 | IRIS | Clinical signs (lethargy, prostration, and ataxia) and hematological changes | 112 | 88 |
| Di(2-ethylhexyl) phthalate* | 117-81-7 | 0.02 | IRIS | Increased relative liver weight | 4040 | 229 |

| | | Chronic oral RfV[a] | | | QSAR estimate | ACToR |
|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical Effect[b]** | **TOPKAT LOAEL[c] (mg/day)** | **# of data points[d]** |
| Pyrene | 129-00-0 | 0.03 | IRIS | Kidney effects (renal tubular pathology, decreased kidney weights) | 36.1 | 129 |
| 1,4-Dioxane | 123-91-1 | 0.03 | IRIS | Liver and kidney toxicity | 207 | 148 |
| Fluoranthene | 206-44-0 | 0.04 | IRIS | Nephropathy, increased liver weights, hematological alterations, and clinical effects | 44.6 | 103 |
| Fluorene | 86-73-7 | 0.04 | IRIS | Decreased RBC, packed cell volume and hemoglobin | 95.1 | 120 |
| m-Cresol | 108-39-4 | 0.05 | IRIS | Decreased body weights and neurotoxicity | 123 | 103 |
| o-Cresol | 95-48-7 | 0.05 | IRIS | Decreased body weights and neurotoxicity | 229 | 94 |
| Toluene* | 108-88-3 | 0.08 | IRIS | Increased kidney weight | 163 | 188 |
| Diphenylamine | 122-39-4 | 0.1 | HHBP | Alterations in clinical chemistry; increased kidney. liver, and spleen weights | 30.8 | 86 |
| Carbon disulfide | 75-15-0 | 0.1 | IRIS | Fetal toxicity/ malformations | 126 | 89 |
| Benzyl alcohol | 100-51-6 | 0.1 | PPRTV | Effects on survival, growth, and tissue histopathology | 210 | 45 |
| Ethylbenzene* | 100-41-4 | 0.1 | IRIS | Liver and kidney toxicity | 226 | 207 |
| Cumene | 98-82-8 | 0.1 | IRIS | Increased average kidney weight in female rats | 246 | 101 |
| Acetophenone | 98-86-2 | 0.1 | IRIS | General toxicity | 274 | 58 |
| Dibutyl phthalate | 84-74-2 | 0.1 | IRIS | Increased mortality | 2090 | 143 |
| Xylenes* | 1330-20-7 | 0.2 | IRIS | Decreased body weight, increased mortality | 110 | 174 |
| Benzyl butyl phthalate | 85-68-7 | 0.2 | IRIS | Significantly increased liver-to-body weight and liver-to-brain weight ratios | | 194 |
| Phenol | 108-95-2 | 0.3 | IRIS | Decreased maternal weight gain | 134 | 170 |

| **Chemical Name** | **CASRN** | **Chronic oral RfV[a]** | | | **QSAR estimate** | **ACToR** |
|---|---|---|---|---|---|---|
| | | **RfV (mg/kg-day)** | **Source of RfV** | **Critical Effect[b]** | **TOPKAT LOAEL[c] (mg/day)** | **# of data points[d]** |
| Biphenyl | 92-52-4 | 0.5 | IRIS | Renal papillary mineralization in male F344 rats | | 103 |
| Caprolactam | 105-60-2 | 0.5 | IRIS | Reduced offspring body weight | | 39 |
| Acetone | 67-64-1 | 0.9 | IRIS | Nephropathy | 119 | 79 |
| Benzo(g,h,i)perylene | 191-24-2 | | | | 29.1 | 68 |
| Indeno(1,2,3-cd)pyrene | 193-39-5 | | | | 38.6 | 111 |
| Dibenz(a,h)anthracene | 53-70-3 | | | | 38.9 | 96 |
| Benzo(b)fluoranthene | 205-99-2 | | | | 39 | 121 |
| Benzo(k)fluoranthene | 207-08-9 | | | | 39 | 118 |
| N-Nitrosodiphenylamine | 86-30-6 | | | | 39.4 | 99 |
| Benzo(a)pyrene* | 50-32-8 | | | | 43 | 184 |
| Phenanthrene | 85-01-8 | | | | 61.3 | 69 |
| p-Cresol | 106-44-5 | | | | 95.5 | 98 |
| Dioctyl phthalate | 117-84-0 | | | | 4740 | 61 |
| Caffeine | 58-08-2 | | | | | 134 |
| Benz(a)anthracene | 56-55-3 | | | | | 122 |
| Chrysene | 218-01-9 | | | | | 114 |
| 2-Mercaptobenzothiazole | 149-30-4 | | | | | 95 |
| 1,2-Diphenylhydrazine | 122-66-7 | | | | | 83 |
| Dimethyl phthalate | 131-11-3 | | | | | 79 |
| N-Nitroso-N-methylethylamine | 10595-95-6 | | | | | 42 |
| 4-(1,1,3,3-Tetramethylbutyl)phenol | 140-66-9 | | | | | 30 |
| p-Tert-butylphenol | 98-54-4 | | | | | 27 |
| 2,6-Di-tert-butylphenol | 128-39-2 | | | | | 22 |
| Dimethylphenol | 1300-71-6 | | | | | 17 |
| 2-Ethylhexyl diphenyl phosphate (Octicizer) | 1241-94-7 | | | | | 14 |
| 2,5-Cyclohexadiene-1,4-dione | 106-51-4 | | | | | 12 |
| Cholesterol | 57-88-5 | | | | | 11 |

| Chemical Name | CASRN | Chronic oral RfV[a] | | | QSAR estimate | ACToR |
|---|---|---|---|---|---|---|
| | | RfV (mg/kg-day) | Source of RfV | Critical Effect[b] | TOPKAT LOAEL[c] (mg/day) | # of data points[d] |
| Benzothiazole | 95-16-9 | | | | | 10 |
| Octadecanoic acid | 57-11-4 | | | | | 9 |
| Butanoic acid, butyl ester | 109-21-7 | | | | | 9 |
| Tetradecanoic acid | 544-63-8 | | | | | 7 |
| Triphenyl phosphate | 115-86-6 | | | | | 7 |
| Dodecanoic acid | 143-07-7 | | | | | 6 |
| Drometrizole | 2440-22-4 | | | | | 6 |
| 3-(4-Methoxyphenyl)-2-ethylhexylester-2-propenoic acid | 5466-77-3 | | | | | 6 |
| 2,6-Bis(dimethylethyl)-2,5-cyclohexadiene-1,4-dione | 719-22-2 | | | | | 3 |
| Diphenylmethane | 101-81-5 | | | | | 3 |
| Isopropyl myristate | 110-27-0 | | | | | 2 |
| 2-[2-[4-(1,1,3,3-tetramethylbutyl)phenoxy]ethoxy]-ethanol | 2315-61-9 | | | | | 2 |
| Sterane | 50-24-8 | | | | | 1 |
| 3-(4-Methoxyphenyl)-2-propenoic acid | 830-09-1 | | | | | 1 |

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; HHBP = Human Health Benchmarks for Pesticides; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): 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. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] TOPKAT LOAEL: The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2. Values are rounded to 3 significant figures.

[d] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

Of the organic chemicals in produced water listed in Table 9-4, 17 have available OSFs and 23 are classified as known, probable, or possible carcinogens (Table 9-5). Benzidine and benzene were both classified as human carcinogens by IRIS, IARC, and RoC, with benzidine being the most potent carcinogen listed in Table 9-5 (OSF of 230 per mg/kg-day). Benzo(a)pyrine is classified as a human carcinogen by IARC, and as a probable human carcinogen by IRIS. The remaining chemicals were classified as probable or possible human carcinogens.

**Table 9-5. List of OSFs and qualitative cancer classifications available for a subset of organic chemicals that have been reported in produced water.**

Includes organic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that have measured concentration data available in Chapter 7 or Appendix E (Table 9-4) and are classified as known, probable, or possible carcinogens. Chemicals that had OSFs available are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

| | | **OSFs**[a] | | **Qualitative Cancer Classifications** | | | |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **OSF (per mg/kg-day)** | **Source of OSF** | **IRIS** [b] | **PPRTV** [c] | **IARC** [d] | **RoC** [e] |
| Benzidine | 92-87-5 | 230 | IRIS | A (Human carcinogen) | | 1 | Known |
| N-Nitroso-N-methylethylamine | 10595-95-6 | 22 | IRIS | B2 (Probable human carcinogen) | | 2B | |
| Benzo(a)pyrene | 50-32-8 | 7.3 | IRIS | B2 (Probable human carcinogen) | | 1 | RAHC |
| Dibenz(a,h)anthracene | 53-70-3 | 4.1 | CalEPA | | | 2A | RAHC |
| Indeno(1,2,3-cd)pyrene | 193-39-5 | 1.2 | CalEPA | | | 2B | RAHC |
| Benzo(b)fluoranthene | 205-99-2 | 1.2 | CalEPA | | | 2B | RAHC |
| Benzo(k)fluoranthene | 207-08-9 | 1.2 | CalEPA | | | 2B | RAHC |
| 1,2-Diphenylhydrazine | 122-66-7 | 0.8 | IRIS | B2 (Probable human carcinogen) | | | RAHC |
| Benz(a)anthracene | 56-55-3 | 0.7 | PPRTV | B2 (Probable human carcinogen) | | 2B | RAHC |
| Chrysene | 218-01-9 | 0.12 | CalEPA | B2 (Probable human carcinogen) | | 2B | |
| 1,4-Dioxane | 123-91-1 | 0.1 | IRIS | "Likely to be carcinogenic to humans" | | 2B | RAHC |

| Chemical Name | CASRN | OSFs[a] | | Qualitative Cancer Classifications | | | |
|---|---|---|---|---|---|---|---|
| | | OSF (per mg/kg-day) | Source of OSF | IRIS [b] | PPRTV [c] | IARC [d] | RoC [e] |
| Benzene | 71-43-2 | 0.015-0.055 | IRIS | A (Human carcinogen) | | 1 | Known |
| Chloroform | 67-66-3 | 0.019 | CalEPA | B2 (Probable human carcinogen) | | 2B | RAHC |
| Di(2-ethylhexyl) phthalate | 117-81-7 | 0.014 | IRIS | B2 (Probable human carcinogen) | | 2B | RAHC |
| Ethylbenzene | 100-41-4 | 0.011 | CalEPA | D (Not classifiable as to human carcinogenicity) | | 2B | |
| Biphenyl | 92-52-4 | 0.008 | IRIS | "Suggestive evidence of carcinogenic potential" | | | |
| N-Nitrosodiphenyl-amine | 86-30-6 | 0.0049 | IRIS | B2 (Probable human carcinogen) | | 3 | |
| Naphthalene | 91-20-3 | | | “Data are inadequate to assess human carcinogenic potential” | | 2B | RAHC |
| Cumene | 98-82-8 | | | D (Not classifiable as to human carcinogenicity) | | 2B | RAHC |
| 2-Mercaptobenzo-thiazole | 149-30-4 | | | | | 2A | |
| m-Cresol | 108-39-4 | | | C (Possible human carcinogen) | | | |
| o-Cresol | 95-48-7 | | | C (Possible human carcinogen) | “Data are inadequate for the assessment of human carcinogenic potential” | | |

| Chemical Name | CASRN | OSFs[a] | | Qualitative Cancer Classifications | | | |
|---|---|---|---|---|---|---|---|
| | | OSF (per mg/kg-day) | Source of OSF | IRIS [b] | PPRTV [c] | IARC [d] | RoC [e] |
| Benzyl butyl phthalate | 85-68-7 | | | C (Possible human carcinogen) | | 3 | |

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; HHBP = Human Health Benchmarks for Pesticides; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): 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. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

### 9.5.3 Inorganic Chemicals and TENORM in Produced Water

Chapter 7 discussed the detection of inorganic constituents such as metals, inorganic ions, and TENORM in produced water. Examples include barium, cadmium, chromium, copper, lead, manganese, nickel, zinc, and radium. In general, these chemicals are naturally occurring, and are characteristic of produced water from both conventional and unconventional reservoirs. These chemicals have the potential to enter drinking water resources through events such as spills of produced water, mechanical integrity failures, infiltration into groundwater from produced water storage pits, and persistence in inadequately treated wastewater.

The entry of inorganic constituents of produced water into drinking water resources has been documented in numerous studies. In Pennsylvania, elevated levels of barium and strontium have been observed in CWT effluent (PA DEP, 2015a), with effluent concentrations dropping after oil and gas well operators were asked to stop discharging waste at this facility (see Text Box 8-1 for details on temporal trends in wastewater management in Pennsylvania). Likewise, effluent concentrations at two publicly owned treatment words (POTWs) that had accepted Marcellus wastewater were found to have lower concentrations of bromide, chloride, barium, strontium, and sulfate after oil and gas well operators were asked to stop discharging waste at this facility in May 2011 (Ferrar et al., 2013). Effluents from POTWs and CWTs that handle Marcellus Shale wastewater have been found to have levels of radium-226 and radium-228 that exceed the MCL for radium and are significantly higher than typical background levels of radium in river water (PA DEP, 2015b). Radium-226 and radium-228 have been demonstrated to accumulate in sediments near the outfalls of CWTs and of POTWs that handle oil and gas wastewater from CWTs (PA DEP, 2015b; Warner et al., 2013a), and in sediments receiving effluent from landfills that accept oil and gas wastes (PA DEP, 2015b). In West Virginia, water samples collected downstream of a hydraulic fracturing wastewater injection facility had elevated specific conductance and total dissolved solids, elevated bromide, chloride, sodium, barium, strontium, and lithium concentrations, and different strontium isotope ratios compared to those found in upstream, background waters (Akob et al., 2016). In a survey of 500 groundwater wells overlying and adjacent to the Barnett Shale in Texas, Hildenbrand et al. (2015) reported a variety of metals and anions that are known produced water constituents at concentrations that sometimes exceeded primary or secondary MCLs, health advisory levels, or other suggested levels as provided in the EPA Drinking Water Standards, although it was not clear that these chemicals originated from nearby hydraulic fracturing activity or from other potential sources.

For the inorganic chemicals that were identified in produced water on EPA's chemical list, noncancer toxicity values (chronic oral RfVs) and ACToR data availability for these chemicals are shown in Table 9-6, and cancer information (OSFs and qualitative cancer classifications) are shown in Table 9-7. As shown in Table 9-6, chronic oral RfVs were available for 26 of these chemicals, ranging from 0.00002 mg/kg-day (phosphorus) to 1.6 mg/kg-day (nitrate). Critical effects for these metals include neurotoxicity, developmental and liver toxicity, hyperpigmentation and keratosis of the skin, and decrements in blood copper status and enzyme activity. Nineteen of the inorganic chemicals in Table 9-6 are regulated as drinking water contaminants under the NPDWR.

All but one of these inorganic chemicals had at least some relevant data available on EPA's ACToR database. However, none of the inorganic chemicals have TOPKAT LOAEL estimates available, as this QSAR model is only able to generate estimates for organic chemicals (Section 9.4.2).

### Table 9-6. List of inorganics and TENORM reported in produced water, and respective chronic oral RfVs and OSFs when available.

Includes inorganic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H). Chemicals are ordered from most toxic to least toxic based on chronic oral RfV. Chemicals without chronic oral RfVs were ordered in terms of the number of data points on ACToR. *Indicates chemicals are regulated as drinking water contaminants under the NPDWR.

| | | Chronic oral RfVs[a] | | | ACToR |
|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical effect[b]** | **# of data points[c]** |
| Phosphorus | 7723-14-0 | 0.00002 | IRIS | Parturition mortality; forelimb hair loss | 113 |
| Vanadium | 7440-62-2 | 0.00007 | PPRTV | Kidney histopathology | 76 |
| Arsenic* | 7440-38-2 | 0.0003 | IRIS | Hyperpigmentation and vascular complications | 243 |

| **Chemical Name** | **CASRN** | **Chronic oral RfVs[a]** | | | **ACToR** |
|---|---|---|---|---|---|
| | | **RfV (mg/kg-day)** | **Source of RfV** | **Critical effect[b]** | **# of data points[c]** |
| Cobalt | 7440-48-4 | 0.0003 | PPRTV | Decreased iodine uptake | 76 |
| Antimony* | 7440-36-0 | 0.0004 | IRIS | Hematological; alterations in glucose and cholesterol | 163 |
| Cadmium* | 7440-43-9 | 0.0005 | IRIS | Proteinuria | 230 |
| Beryllium* | 7440-41-7 | 0.002 | IRIS | Intestinal lesions | 186 |
| Mercury | 7439-97-6 | 0.002 | CICAD | Renal toxicity | 177 |
| Lithium | 7439-93-2 | 0.002 | PPRTV | Adverse effects in multiple organ systems | 43 |
| Chromium (VI) | 18540-29-9 | 0.003 | IRIS | None reported | 120 |
| Selenium* | 7782-49-2 | 0.005 | IRIS | Clinical selenosis | 232 |
| Silver | 7440-22-4 | 0.005 | IRIS | Argyria | 120 |
| Molybdenum | 7439-98-7 | 0.005 | IRIS | Increased uric acid levels | 73 |
| Iodine | 7553-56-2 | 0.01 | CICAD | | 27 |
| Nitrite* | 14797-65-0 | 0.1 | IRIS | Methemoglobinemia | 109 |
| Chlorine | 7782-50-5 | 0.1 | IRIS | No adverse effect level | 116 |
| Manganese | 7439-96-5 | 0.14 | IRIS | Central nervous system (CNS) effects | 128 |
| Barium* | 7440-39-3 | 0.2 | IRIS | Nephropathy | 167 |
| Boron | 7440-42-8 | 0.2 | IRIS | Decreased fetal weight (developmental) | 93 |
| Zinc | 7440-66-6 | 0.3 | IRIS | Decreases in erythrocyte Cu, Zn-superoxide dismutase (ESOD) activity in humans | 163 |
| Lead* | 7439-92-1 | 0.5 μg/day [d] | CalEPA | Reproductive Toxicity | 168 |
| Strontium | 7440-24-6 | 0.6 | IRIS | Rachitic bone | 67 |
| Iron | 7439-89-6 | 0.7 | PPRTV | Adverse gastrointestinal effects | 73 |
| Aluminum | 7429-90-5 | 1 | PPRTV | Neurotoxicity | 88 |
| Chromium (III) | 16065-83-1 | 1.5 | IRIS | No effects observed | 71 |

| Chemical Name | CASRN | Chronic oral RfVs[a] | | | ACToR |
|---|---|---|---|---|---|
| | | RfV (mg/kg-day) | Source of RfV | Critical effect[b] | # of data points[c] |
| Nitrate* | 14797-55-8 | 1.6 | IRIS | Clinical signs of methemoglobinemia in excess of 10% | 130 |
| Nickel | 7440-02-0 | | | | 181 |
| Copper* | 7440-50-8 | | | | 163 |
| Thallium* | 7440-28-0 | | | | 136 |
| Chromium | 7440-47-3 | | | | 125 |
| Uranium-238* | 7440-61-1 | | | | 100 |
| Ammonia | 7664-41-7 | | | | 90 |
| Zirconium | 7440-67-7 | | | | 55 |
| Alpha particle* | 12587-46-1 | | | | 55 |
| Fluoride* | 16984-48-8 | | | | 53 |
| Radium* | 7440-14-4 | | | | 52 |
| Beta particle* | 12587-47-2 | | | | 51 |
| Magnesium | 7439-95-4 | | | | 40 |
| Tin | 7440-31-5 | | | | 40 |
| Chloride | 16887-00-6 | | | | 32 |
| Sodium | 7440-23-5 | | | | 31 |
| Sulfate | 14808-79-8 | | | | 27 |
| Potassium | 7440-09-7 | | | | 25 |
| Titanium | 7440-32-6 | | | | 25 |
| Calcium | 7440-70-2 | | | | 24 |
| Radium-226* | 13982-63-3 | | | | 13 |
| Radium-228* | 15262-20-1 | | | | 11 |
| Sulfide | 18496-25-8 | | | | 11 |
| Caesium | 7440-46-2 | | | | 7 |
| Caesium-137 | 10045-97-3 | | | | 6 |
| Silicon | 7440-21-3 | | | | 5 |
| Rubidium | 7440-17-7 | | | | 5 |

| Chemical Name | CASRN | Chronic oral RfVs[a] | | | ACToR |
|---|---|---|---|---|---|
| | | RfV (mg/kg-day) | Source of RfV | Critical effect[b] | # of data points[c] |
| Bromide | 24959-67-9 | | | | 2 |
| Sulfite | 14265-45-3 | | | | 1 |
| Uranium-235* | 15117-96-1 | | | | 1 |
| Octasulfur | 10544-50-0 | | | | |

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer-Reviewed Toxicity Value; CalEPA = California Environmental Protection Agency; CICAD = Concise International Chemical Assessment Documents; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): 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. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

[d] CalEPA MADLs are in units of µg/day, while all other chronic oral RfVs in this table are in units of mg/kg-day.

OSFs were available for 4 of the inorganic chemicals reported in produced water, and 14 are classified as known or probable carcinogens (Table 9-7). OSFs ranged from 15 per mg/kg-day for cadmium to 0.0085 mg/kg-day for lead. Chromium (VI), arsenic, alpha particle, beta particle, radium-226, and radium-288 are all classified as known human carcinogens by all sources reporting in this table. Beryllium and cadmium are both classified as known human carcinogens by IARC and NTP, and as probable human carcinogens by EPA. Lead, cobalt, nickel, nitrate, and nitrite are classified by these sources as possible or probable human carcinogens.

### Table 9-7. List of qualitative cancer classifications available for inorganics and NORM that were reported in produced water.

Includes inorganic chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that classified as known, probable, or possible carcinogens by at least one of the sources in Table 9-1. Chemicals that had OSFs available are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

| Chemical Name | CASRN | OSF[a] | | Qualitative Cancer Classifications | | | |
|---|---|---|---|---|---|---|---|
| | | OSF (per mg/kg-day) | Source of OSF | IRIS[b] | PPRTV[c] | IARC[d] | RoC[e] |
| Cadmium | 7440-43-9 | 15 | CalEPA | B1 (Probable human carcinogen) | | 1 | Known |
| Arsenic | 7440-38-2 | 1.5 | IRIS | A (Human carcinogen) | | 1 | Known |

| Chemical Name | CASRN | OSF[a] | | Qualitative Cancer Classifications | | | |
|---|---|---|---|---|---|---|---|
| | | OSF (per mg/kg-day) | Source of OSF | IRIS[b] | PPRTV[c] | IARC[d] | RoC[e] |
| Chromium (VI) | 18540-29-9 | 0.5 | CalEPA | Inhaled: A (Human carcinogen)<br>Oral: D (Not classifiable as to human carcinogenicity) | | 1 | Known |
| Lead | 7439-92-1 | 0.0085 | CalEPA | B2 (Probable human carcinogen) | | 2B | RAHC |
| Alpha particle | 12587-46-1 | | | | | 1 | |
| Beryllium | 7440-41-7 | | | B1 (Probable human carcinogen) | | 1 | Known |
| Beta particle | 12587-47-2 | | | | | 1 | |
| Radium | 7440-14-4 | | | | | 1 | |
| Radium-226 | 13982-63-3 | | | | | 1 | |
| Radium-228 | 15262-20-1 | | | | | 1 | |
| Cobalt | 7440-48-4 | | | | Likely to be carcinogenic to humans | 2B | |
| Nickel | 7440-02-0 | | | | | 2B | RAHC |
| Nitrate | 14797-55-8 | | | | | 2A | |
| Nitrite | 14797-65-0 | | | | | 2A | |

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; PPRTV = Provisional Peer Reviewed Toxicity Values; HHBP = Human Health Benchmarks for Pesticides; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): 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. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

### 9.5.4 Organochlorine Pesticides and Polychlorinated Biphenyls (PCBs) in Produced Water

EPA's list of chemicals detected in produced water includes several chemicals that have been banned from commercial use: specifically, organochlorine pesticides and Aroclor 1248, which is a commercial PCB mixture. These chemicals were reported by two of the sources used to compile EPA's chemical list (Appendix H): a technical report prepared by the Gas Technology Institute for the Marcellus Shale Coalition (MSC), which is a drilling industry trade group (Hayes, 2009); and a report by the New York State Department of Environmental Conservation (NYSDEC), which referenced the results of the MSC study (NYSDEC, 2011). These chemicals are listed in Table 9-8 along with their respective noncancer toxicity values (chronic oral RfVs and TOPKAT LOAELs) and availability of relevant toxicological information on ACToR. Cancer information (OSF or qualitative cancer classification) for these chemicals is listed in Table 9-9.

There is uncertainty about why organochlorine pesticides and PCBs were detected, as they are not used in hydraulic fracturing fluids and are not naturally occurring. The MSC study stated the banned substances were detected sporadically and at low concentrations, and suggested they may have originated from laboratory contamination. The NYSDEC report suggested that the banned substances may have been introduced to the shale or the water as a result of drilling or fracturing operations. It is possible that these chemicals were present as legacy contaminants in the source water used for hydraulic fracturing fluid formulation, or were mobilized from the environment near the well. Although these chemicals are notable for their high toxicity, the extent to which these chemicals may be detected in produced water from other hydraulic fracturing sites is not clear.

Chronic oral RfVs for these organochlorine pesticides ranged from 0.000013 mg/kg-day (Heptachlor epoxide) to 0.0005 mg/kg-day (heptachlor), and were all based on liver toxicity. All of these pesticides had TOPKAT LOAEL estimates, and all have relevant data available within EPA's ACToR database.). Heptachlor epoxide, heptachlor, and lindane are regulated as drinking water contaminants under the NPDWR.

**Table 9-8. List of organochlorine pesticides and PCBs that were reported in produced water, and their respective chronic oral RfVs, TOPKAT LOAEL estimates, and availability of data in EPA's ACToR database.**

Includes banned chemicals that were identified on EPA's list of chemicals in produced water (Appendix H). Chemicals are ordered from most toxic to least toxic based on chronic oral RfV. Chemicals without chronic oral RfVs were ordered in terms of the number of data points on ACToR. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

| | | Chronic oral RfV[a] | | | QSAR | ACToR |
|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical effect[b]** | **TOPKAT LOAEL[c] (mg/kg)** | **# of data points[d]** |
| Heptachlor epoxide* | 1024-57-3 | 0.000013 | IRIS | Increased liver-to-body weight ratio in both males and females | 0.595 | 168 |
| Aldrin | 309-00-2 | 0.00003 | IRIS | Liver toxicity | 0.743 | 166 |

| | | Chronic oral RfV[a] | | | QSAR | ACToR |
|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical effect[b]** | **TOPKAT LOAEL[c] (mg/kg)** | **# of data points[d]** |
| Dieldrin | 60-57-1 | 0.00005 | IRIS | Liver lesions | 0.442 | 167 |
| Lindane* | 58-89-9 | 0.0003 | IRIS | Liver and kidney toxicity | 23.9 | 238 |
| Heptachlor* | 76-44-8 | 0.0005 | IRIS | Liver weight increases in males | 0.927 | 203 |
| beta-Hexachlorocyclohexane | 319-85-7 | | | | 23.9 | 88 |
| delta-Hexachlorocyclohexane | 319-86-8 | | | | 23.9 | 22 |
| Aroclor 1248 | 12672-29-6 | | | | 21.87 | 35 |
| p,p'-DDE | 72-55-9 | | | | 14.6 | 103 |
| Endrin aldehyde | 7421-93-4 | | | | 4.09 | 27 |
| Endosulfan I | 959-98-8 | | | | 2.27 | 32 |
| Endosulfan II | 33213-65-9 | | | | 2.27 | 32 |

CASRN = Chemical Abstract Service Registry Number; RfV = Reference value; IRIS = Integrated Risk Information System; QSAR = Quantitative structure-activity relationship; TOPKAT = Toxicity Prediction by Komputer Assisted Technology; ACToR = EPA's Aggregated Computational Toxicology Online Resource

[a] Reference value (RfV): 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. RfVs considered in this analysis include chronic oral reference doses (RfDs) from IRIS, PPRTV, and HHBP; chronic oral minimal risk levels (MRLs) from ATSDR; maximum allowable daily levels (MADLs) from CalEPA; and tolerable daily intake (TDI) from CICAD. See Section 9.4.1.

[b] Critical effect: The first adverse effect, or its known precursor, that occurs to the most sensitive species as the dose rate of an agent increases.

[c] TOPKAT lowest-observed-adverse-effect level (LOAEL): The LOAEL is the lowest exposure level at which there are biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. TOPKAT LOAELs were predicted using a QSAR-based software model, as described in Section 9.4.2.

[d] Indicates the total number of data points available for a chemical in the relevant data classes on EPA's ACToR database, as described in Section 9.4.3.

OSFs were available for 7 of the organochlorine pesticides that are classified as known, probable, or possible human carcinogens (Table 9-9). OSFs ranged from 17 per mg/kg-day (aldrin) to 0.34 per mg/kg-day (p,p'-DDE). Aldrin, dieldrin, heptachlor epoxide, heptachlor, beta-hexachlorocyclohexane, and p,p'-DDE are classified as probable or possible carcinogens. Lindane is classified as a known carcinogen by IARC, and as "reasonably anticipated to be a human carcinogen" by RoC.

**Table 9-9. List of OSFs and qualitative cancer classifications available for organochlorine pesticides reported in produced water.**

Includes banned chemicals that were identified on EPA's list of chemicals in produced water (Appendix H) that are classified as known, probable, or possible carcinogens by at least one of the sources in Table 9-1. Chemicals are ordered in this table from most potent (highest OSF) to least potent (lowest OSF).

| | | OSF[a] | | Qualitative cancer classifications | | | |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **OSF (per mg/kg-day)** | **Source of OSF** | **IRIS[b]** | **PPRTV[c]** | **IARC[d]** | **RoC[e]** |
| Aldrin | 309-00-2 | 17 | IRIS | B2 (Probable human carcinogen) | | 3 | |
| Dieldrin | 60-57-1 | 16 | IRIS | B2 (Probable human carcinogen) | | 3 | |
| Heptachlor epoxide | 1024-57-3 | 9.1 | IRIS | B2 (Probable human carcinogen) | | | |
| Heptachlor | 76-44-8 | 4.5 | IRIS | B2 (Probable human carcinogen) | | 2B | |
| beta-Hexachlorocyclohexane | 319-85-7 | 1.8 | IRIS | C (Possible human carcinogen) | | | |
| Lindane | 58-89-9 | 1.1 | CalEPA | | | 1 | RAHC |
| p,p'-DDE | 72-55-9 | 0.34 | IRIS | B2 (Probable human carcinogen) | | | |

CASRN = Chemical Abstract Service Registry Number; IRIS = Integrated Risk Information System; CalEPA = California Environmental Protection Agency; IARC = International Agency for Research on Cancer Monographs; RoC = National Toxicology Program 13th Report on Carcinogens

[a] Oral slope factor (OSF): 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. OSFs considered in this analysis include values from IRIS, PPRTV, HHBP, and CalEPA. See Section 9.4.1.

[b] IRIS assessments use EPA's 1986, 1996, 1999, or 2005 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[c] PPRTV assessments use EPA's 1999 guidelines to establish descriptors for summarizing the weight of evidence as to whether a contaminant is or may be carcinogenic. See glossary in Appendix G for details.

[d] The IARC summarizes the weight of evidence as to whether a contaminant is or may be carcinogenic using five weight of evidence classifications: Group 1: Carcinogenic to humans; Group 2A: Probably carcinogenic to humans; Group 2B: Possibly carcinogenic to humans; Group 3: Not classifiable as to its carcinogenicity to humans; Group 4: Probably not carcinogenic to humans. See glossary in Appendix G for details.

[e] The listing criteria in the 13th RoC Document are: Known = Known to be a human carcinogen; RAHC = Reasonably anticipated to be a human carcinogen.

### 9.5.5 Methane in Stray Gas

Chapter 6 discussed stray gas as a potential hazard in areas of hydraulic fracturing activity (Text Box 6-3). Stray gas refers to the phenomenon of natural gas (primarily methane, plus lesser amounts of ethane) migrating into shallow groundwater, into water wells, or to the surface (e.g., cellars, streams, or springs). As discussed in Chapter 6, some studies indicate an association between hydraulic fracturing activity and elevated methane concentrations in drinking water, while other studies did not find such a correlation. Potential pathways for migration of stray gas into aquifers include pathways along production wells with casing and/or cement issues, through naturally existing fractures, through induced fractures, or via a route that is some combination of these pathways.

Although ingestion of methane is not considered to be toxic, it has the potential to pose a physical hazard. Methane can accumulate to explosive levels when allowed to exsolve (degas) from groundwater in closed environments. High concentrations of methane may also displace oxygen and act as an asphyxiant (NIOSH, 2000), potentially causing suffocation, loss of consciousness, or symptoms such as headache and nausea. Methane is not a regulated drinking water contaminant. Methane does not have an RfV, OSF, or qualitative cancer classification available from any of the sources consulted by EPA, and did not have a high-confidence TOPKAT LOAEL estimate. Information on methane is available within the ACToR database.

### 9.5.6 Disinfection Byproducts (DBPs) Formed from Wastewater Constituents

Some of the inorganic constituents of hydraulic fracturing produced water, including chloride, bromine, iodine, and ammonium, can contribute to the formation of DBPs during wastewater treatment (Harkness et al., 2015; Parker et al., 2014). The entry of these constituents into drinking water resources—e.g., as a result of wastewater spills or from the discharge of inadequately treated hydraulic fracturing wastewater—can result in DBPs in finished drinking water from downstream drinking water treatment plants (States et al., 2013). DBPs may also be formed when hydraulic fracturing produced water is treated at a centralized or publicly owned treatment works, and may reach drinking water resources when the treated wastewater is discharged to surface water (Hladik et al., 2014). Currently, there are no data available on the concentrations of DBPs in finished drinking water as related to contributions of DBP precursors from hydraulic fracturing wastewater.

Regulated DBPs such as bromate, chlorite, haloacetic acids, and trihalomethanes are a small subset of the full spectrum of DBPs that include other chlorinated and brominated DBPs as well as nitrogenous and iodated DBPs. Long term exposure to these DBPs can result in an increased risk of cancer, anemia, liver and kidney effects, and central nervous system effects. Some of the unregulated DBPs may be more toxic than their regulated counterparts (Harkness et al., 2015; McGuire et al., 2014; Parker et al., 2014). In addition, brominated forms of DBPs are considered to be more cytotoxic, genotoxic, and carcinogenic than chlorinated species based on studies using rodents, various types of human cells, and a salmonella strain containing human P450 genes (McGuire et al., 2014; Parker et al., 2014; States et al., 2013; Krasner, 2009; Richardson et al., 2007). As with brominated DBPs, there is concern that some iodinated forms of DBPs are more cytotoxic and genotoxic than chlorinated species (McGuire et al., 2014; Parker et al., 2014; Krasner, 2009; Richardson et al., 2007), as evidenced by studies involving rodent research and human cell research (Plewa et al., 2010; Plewa and Wagner, 2009; Richardson et al., 2007). The MCLs (mg/L) for the regulated DBPs are: 0.01 for bromate, 1.0 for chlorite, 0.06 for haloacetic acid, and 0.08 for total trihalomethanes.

### 9.5.7 Chemicals Detected in Multiple Stages of the Hydraulic Fracturing Water Cycle

As mentioned in Section 9.3 above, there were a total of 77 chemicals on EPA's list that were identified as being used in hydraulic fracturing fluids and detected in produced water. The presence of these chemicals within both of these stages of the hydraulic fracturing water cycle may indicate that these chemicals persist after they are injected into the well. However, this is not necessarily the case, as some of these chemicals (e.g., BTEX, naphthalene, metals) also occur naturally in oil and gas reservoirs. Additionally, the EPA's list of chemicals used in hydraulic fracturing fluids and list of chemicals in produced water were compiled from different sets of sources, and does not provide a matched comparison between the chemicals used in hydraulic fracturing fluid and the chemicals present in produced water at a particular site. There may have been other chemicals in present in produced water that were not detected by these studies due to limitations of analytical chemistry. Thus, the EPA's composited chemical list cannot reliably be used to draw conclusions on the persistence of hydraulic fracturing chemicals following well injection.

Of the 77 chemicals identified in both hydraulic fracturing fluids and produced water, 45 have a chronic oral RfV or OSF available from at least one of the sources in Table 9-1. These 45 chemicals and their respective toxicity values are shown in Table 9-10, with frequency of use data from the EPA FracFocus 1.0 project database provided when available. Eleven of these chemicals are regulated as drinking water contaminants.

**Table 9-10. List of 45 chemicals on EPA's list that were used in hydraulic fracturing fluids and detected in produced water and have an RfV or OSF available.**

Frequency of use data from the EPA FracFocus 1.0 project database is provided when available. Chemicals with available data from the FracFocus 1.0 project database are ordered from high to low based on frequency of use. Chemicals without frequency of use data are ordered from most toxic to least toxic based on chronic oral RfV. *Indicates chemicals that are regulated as drinking water contaminants under the NPDWRs.

| | | | Chronic oral RfVs[b] | | | OSFs[d] | |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **% of Disclo-sures[a]** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical Effect[c]** | **OSF (per mg/kg-day)** | **Source of OSF** |
| Methanol | 67-56-1 | 73% | 2 | IRIS | Extra cervical ribs | | |
| Ethylene glycol | 107-21-1 | 47% | 2 | IRIS | Kidney toxicity | | |
| Propargyl alcohol | 107-19-7 | 33% | 0.002 | IRIS | Renal and hepatotoxicity | | |
| 2-Butoxyethanol | 111-76-2 | 23% | 0.1 | IRIS | Hemosiderin deposition in liver (inhalation study) | | |
| Naphthalene | 91-20-3 | 19% | 0.02 | IRIS | Decreased mean terminal body weight in males | | |

| | | | Chronic oral RfVs[b] | | | OSFs[d] | |
|---|---|---|---|---|---|---|---|
| **Chemical Name** | **CASRN** | **% of Disclo-sures[a]** | **RfV (mg/kg-day)** | **Source of RfV** | **Critical Effect[c]** | **OSF (per mg/kg-day)** | **Source of OSF** |
| 1,2,4-Trimethylbenzene | 95-63-6 | 13% | 0.01 | IRIS | Decreased pain sensitivity | | |
| Formic acid | 64-18-6 | 11% | 0.9 | PPRTV | Reproductive Toxicity | | |
| N,N-Dimethylformamide | 68-12-2 | 9% | 0.1 | PPRTV | Increase in ALT enzyme and liver weight | | |
| Benzyl chloride | 100-44-7 | 6% | 0.002 | PPRTV | Cardiotoxicity | 0.17 | IRIS |
| 1,2-Propylene glycol | 57-55-6 | 4% | 20 | PPRTV | Reduced RBC counts and hyperglycemia | | |
| Xylenes* | 1330-20-7 | 2% | 0.2 | IRIS | Decreased body weight, increased mortality | | |
| D-Limonene | 5989-27-5 | 2% | 0.1 | CICAD | Increased liver weight | | |
| 1-Butanol | 71-36-3 | 1% | 0.1 | IRIS | Hypoactivity and ataxia | | |
| Toluene* | 108-88-3 | 0.7% | 0.08 | IRIS | Increased kidney weight | | |
| Bis(2-chloroethyl) ether | 111-44-4 | 0.7% | | | | 1.1 | IRIS |
| 2-(2-Butoxyethoxy)ethanol | 112-34-5 | 0.6% | 0.03 | PPRTV | Changes in red blood cells (RBC) | | |
| 1,3,5-Trimethylbenzene | 108-67-8 | 0.5% | 0.01 | IRIS | Decreased pain sensitivity | | |
| Cumene | 98-82-8 | 0.5% | 0.1 | IRIS | Increased average kidney weight in female rats | | |
| Iron | 7439-89-6 | 0.4% | 0.7 | PPRTV | Adverse gastrointestinal effects | | |
| 1,2,3-Trimethylbenzene | 526-73-8 | 0.4% | 0.01 | IRIS | Decreased pain sensitivity | | |

| Chemical Name | CASRN | % of Disclosures[a] | Chronic oral RfVs[b] | | | OSFs[d] | |
|---|---|---|---|---|---|---|---|
| | | | RfV (mg/kg-day) | Source of RfV | Critical Effect[c] | OSF (per mg/kg-day) | Source of OSF |
| Phenol | 108-95-2 | 0.4% | 0.3 | IRIS | Decreased maternal weight gain | | |
| Ethylbenzene* | 100-41-4 | 0.4% | 0.1 | IRIS | Liver and kidney toxicity | 0.011 | CalEPA |
| 1,4-Dioxane | 123-91-1 | 0.3% | 0.03 | IRIS | Liver and kidney toxicity | 0.1 | IRIS |
| Acetone | 67-64-1 | 0.2% | 0.9 | IRIS | Nephropathy | | |
| Boron | 7440-42-8 | 0.05% | 0.2 | IRIS | Decreased fetal weight | | |
| o-Xylene* | 95-47-6 | 0.05% | 0.2 | ATSDR | Neurotoxicity | | |
| Acetophenone | 98-86-2 | 0.04% | 0.1 | IRIS | General toxicity | | |
| Quinoline | 91-22-5 | 0.02% | | | | 3 | IRIS |
| Dichloromethane* | 75-09-2 | 0.02% | 0.006 | IRIS | Hepatic effects (hepatic vacuolation, liver foci) | 0.002 | IRIS |
| Trimethylbenzene | 25551-13-7 | 0.01% | 0.01 | IRIS | Decreased pain sensitivity | | |
| Benzene* | 71-43-2 | 0.01% | 0.004 | IRIS | Decreased lymphocyte count | 0.015-0.055 | IRIS |
| Bisphenol A | 80-05-7 | 0.01% | 0.05 | IRIS | Reduced mean body weight | | |
| Aluminum | 7429-90-5 | 0.003% | 1 | PPRTV | Neurotoxicity | | |
| Hydrazine | 302-01-2 | 0.003% | | | | 3 | IRIS |
| Chlorobenzene* | 108-90-7 | 0.003% | 0.02 | IRIS | Histopathologic changes in liver | | |
| Arsenic* | 7440-38-2 | | 0.0003 | IRIS | Hyperpigmentation and vascular complications | 1.5 | IRIS |
| Acrolein | 107-02-8 | | 0.0005 | IRIS | Decreased survival | | |
| Chromium (VI) | 18540-29-9 | | 0.003 | IRIS | None reported | 0.5 | CalEPA |
| Tributyl phosphate | 126-73-8 | | 0.01 | PPRTV | Occasional salivation | 0.009 | PPRTV |