Potential Direct Environmental Effects of Well Stimulation

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This section provides an assessment of potential direct environmental effects from the use of well stimulation. Direct environmental effects include potential impacts to water supply, water quality, air quality due to emissions of hazardous air contaminants and climate forcing pollutants, induced seismicity, and other miscellaneous impacts. This assessment considers potential effects from the stimulation process itself, as well as potential effects from transportation of stimulation supplies to the site and disposal of flowback/produced waters following the stimulation. Examples of direct environmental effects of well stimulation reviewed in this assessment are emission of air pollutants from diesel engines operating the pumps injecting the stimulation fluid, and spills of hydraulic fracturing fluid. The approach taken is literature review and data mining to infer potential impacts based on a wide foundation of knowledge and experience for well stimulation operations across the U.S. However, the interpretation of hazards and risks associated with well stimulation techniques, and more broadly oil and gas development is beyond the scope of this document.

Well stimulation technology (WST) can enable new or expanded production of oil. Consequently, indirect effects of well stimulation (such as additional emissions of air pollutants or methane due to expanded production or combustion of oil produced subsequent to stimulation, potential contamination due to leaks or spills that may occur during storage and transportation of oil, and ecological disruption from oil fields under production) can result from oil and gas production that has been enabled by WST. Indirect effects occur with all oil and gas production, whether or not well stimulation techniques have been used, and these will not be comprehensively evaluated in this assessment.

Section 5.1 concerns potential impacts of WST to water resources and reviews the effects on water use and water quality. Section 5.1.1 focuses on issues concerning water supply and demand due to the expected usage of freshwater in well stimulation operations, and compares water demand for stimulation activities in California to elsewhere in the country. Section 5.1.2 describes the typical chemistries of waters used in well stimulation treatments, and identifies potential contaminants that can impact water quality near well stimulation operations. This includes characterization of the injection fluids used in well stimulation in California, and an overview of the constituents typically present in flowback and produced waters from well stimulation operations across the United States (since California-specific data were not available at the time of this assessment). Section 5.1.3 discusses the possible surface and subsurface pathways through which the potential contaminants identified in Section 5.1.2 might be released into surface and groundwater. Section 5.1.4 presents findings from water quality studies that have been conducted for surface and groundwater near sites where WST have been used in the United States and in California, and reviews episodes of known or possible contamination that may have occurred as a result of well stimulation activities.

Section 5.2 concerns potential impacts to the atmosphere in terms of air quality and climate caused by well stimulation operations. Section 5.2.1 provides an overview of possible air quality hazards related to increased well stimulation operations. Studies of air quality effects and emissions of pollutants from oil and gas production operations across the country are reviewed and discussed in the context of the practices common in California. Estimates of pollutant emissions attributable to a stimulation job (from diesel engines, such as trucks and pumping equipment) for practices typical in California are compared to emission estimates for high-volume fracturing practices typical outside of California. Emissions (including fugitive emissions) of volatile organic compounds (VOCs) and pollutants from flares are assessed in the context of current California inventories, the California regulatory context, and general scientific uncertainty.

Section 5.2.2 describes greenhouse gas (GHG) emissions related to well stimulation operations. Overall, oil and gas production operations in California include energy-intensive operations, such as steam generation for enhanced oil recovery, and this section compares energy use and CO_2 emissions from well stimulation operations to overall energy use and CO_2 emissions within the oil and gas production sector. The section also describes methane emissions. Methane can play an important role in total GHG emissions from oil and gas production because methane has a global warming potential (GWP) more than 30 times that of carbon dioxide (on a per mass basis) over 100 years and more than 80 times that of carbon dioxide over 20 years. Methane emissions from oil and gas operations are uncertain, with atmospheric measurements suggesting higher emission rates than standard bottom-up inventories (both nationally and in California). Current California inventories of methane emissions from well stimulation and oil and gas production are discussed in the context of local atmospheric-measurement campaigns.

Section 5.3 evaluates the hazard of induced seismicity due to well stimulation technologies. The processes considered include both the well stimulation itself, and the disposal of wastewater fluids through underground injection following stimulation. The mechanics of induced seismicity were reviewed to provide context for this assessment.

Section 5.4 concerns other impacts of well stimulation operations. The implications of well stimulation for wildlife and ecology are reviewed in Section 5.4.1. The review found no information on the specific impacts of well stimulation, because existing studies focus on the impacts of oil and gas development in general and because wildlife responds to the entire oil field infrastructure and activities. Consequently, this section reviews literature regarding the hazards of oil production development on wildlife in general and makes some inferences regarding the potential hazards of well stimulation. Section 5.4.2 reviews impacts of traffic and noise as a result of well stimulation operations. Well stimulation operations generate noise and lead to an increase in heavy truck traffic for transporting water, chemicals used in fracturing fluids, and equipment needed for well stimulation. Estimates for noise levels and increased truck traffic are provided.

Finally, Section 5.5 provides a summary list of findings from the potential environmental impacts of WST in California. Due to lack of data specific to operations in California, a number of findings are supported by, or partially based on, an analysis and interpretation of information from well stimulation activities elsewhere in the United States.

5.1 Potential Impacts to Water

This section discusses issues related to water usage and water quality that may arise due to the use of WST in unconventional oil production. This assessment considers water demand for well stimulation in California, and several aspects of water quality including a review of potential contaminants that can be present in injection and wastewater fluids from well stimulation operations, potential pathways by which the contaminants can be released into surface and groundwater, and specific cases of known or possible contamination that may have been related to well stimulation in the United States.

Section 5.1.1 examines the water demand for stimulation in California, discusses the water sources, and puts this information into context with other areas across the United States. Section 5.1.2 discusses the chemicals used for well stimulation in California according to an analysis of voluntary disclosures of well stimulation practices reported to the FracFocus Chemical Disclosure Registry (FracFocus). This discussion includes statistical analysis of the usage (Section 5.1.2.1) and the toxicity (Section 5.1.2.2), if toxicity data were available, of any chemicals used in more than 2% of the reported stimulations. Section 5.1.2.3 provides an overview of the amount and typical chemical properties of flowback and produced fluids recovered from well stimulation operations across the U.S, since California-specific data were not available at the time of this assessment. Potential contaminants that can be present in the recovered fluids are discussed — namely those constituents that may be present due to the injection fluids (Section 5.1.2.4) or those that naturally are present in the formation waters such as total dissolved solids (TDS)/salts, trace metals, radioactive elements, and organics (Section 5.1.2.5). A more detailed assessment of the hazards associated with flowback/produced water in California was not conducted due to the lack of data regarding the masses of materials used in well stimulation, recovery factors for flowback waters, and concentrations of potential contaminants in flowback/produced waters.

Section 5.1.3 reviews the documented types of surface and subsurface pathways for potential water contamination associated with well stimulations. Potential surface pathways by which well stimulation could result in water contamination are considered in Section 5.1.3.1. The section opens with consideration of surface spills and leaks, management and disposal of flowback water, and storm runoff and flooding. The discussion of flowback water considers the difficulty of distinguishing between flowback and produced water based on composition. The review of subsurface release pathways in Section 5.1.3.2 focuses on the two components of subsurface pathways that could result in contamination hazards: (1) the possibility of forming permeable pathways that intercept groundwater or surface water resources during the hydraulic fracturing process; and (2) the potential for transport of gas and formation fluids through these pathways into overlying groundwater resources. The former involves the potential failure of confining geological formations that protect groundwater quality, the potential failure of well casing or well cement during fracturing, during production, or at any point during the active or inactive lifetime of the well, and the potential interception of pre-existing pathways (old wells, permeable faults) via induced fractures. The latter includes discussion of the hydrological processes that govern flow and transport through any such induced or intercepted pathway. While actual data are limited, we attempt to evaluate current debate regarding contamination of groundwater resources through subsurface pathways by focusing on existing knowledge and published works in reservoir engineering and related fields.

Section 5.1.4 reviews further information on known or possible contamination cases that have occurred as a result of well stimulation activities. Section 5.1.4.1 considers contamination events that have occurred as a result of legal, accidental, or illegal surface discharges. Section 5.1.4.2 reviews literature regarding groundwater quality near well stimulation sites and possible contamination of groundwater due to fracturing activities. The discussion focuses on the direct effects of constituents that may be present in well stimulation, flowback, and produced fluids identified in Section 5.1.2. It should be noted that the constituents of fluids associated with well stimulation could also have “indirect effects” that can potentially alter water quality due to additional reactions with the surface or groundwater, or with aquifer sediments. For instance, changes could occur in redox conditions resulting from migration of methane into the formation that can trigger microbial methane oxidation and subsequent consumption of oxygen, or due to the introduction of oxygen from the stimulation fluids. Changes to redox can degrade water quality and trigger a host of subsurface geochemical reactions, such as the dissolution of trace metals and radioactive elements into groundwater aquifers. The potential impacts on water quality due to these effects are beyond the scope of this project and have not been investigated for this report.

5.1.1 Quantities and Sources of Water Used for Well Stimulation in California

This section discusses actual total water use for well stimulation activities in California based on the assessments of water use per well for hydraulic fracturing and matrix acidizing from Sections 3.2.3 and 3.4.3, respectively. The available information on the sources of this water is considered, followed by some comparison to water use in other regions across the United States. This section finds that actual water use per well in California is less than in other areas due to a combination of factors described in

Section 3.2.2, including the use of vertical wells with shorter treatment lengths and use of cross-linked gels used in smaller volumes than slickwater. Total water use for well stimulation depends on the level of activity. Assuming a rate of 100-150 well stimulations per month, current total annual water use could be as much as 1.4 million m^3 (1,200 acre-feet). An acre-foot of water is enough to serve two average California households for a year at current water use rates (California Department of Water Resources (DWR) 2012). Farmers in California typically use 3 to 6 acre-feet per year to irrigate one acre of cropland (DWR 2013). While current water demand for WST operations is a small fraction of statewide water use, it can contribute to local constraints on water availability, especially during droughts. The type of impact and its magnitude will depend on local conditions, as well as the where, when, or how much water is used, and thus would require analysis on a site-specific basis.

5.1.1.1 Water Use

It is difficult to accurately estimate the volume of water currently used for hydraulic fracturing and other well stimulation techniques in California due to the lack of comprehensive data. Before 2013, companies engaged in oil and gas production were not required to publish or otherwise disclose information about their water and chemical use. However, some producers voluntarily reported information to state regulators and to the website FracFocus. According to these data, there were 792 reports of hydraulic fracturing in California in 2013 that used a combined total of 300 acre-feet of water. As noted, this estimate is based on voluntary disclosures and may not capture the full extent of hydraulic fracturing activity in California. Further, it does not include water use for matrix acidizing because these data are not included in FracFocus.

An approximate current rate of water use for hydraulic fracturing was determined by estimating the number of hydraulic fracturing operations that take place each month along with an average water use per operation (see Sections 3.2.2 and 3.2.3). A rate of 100 to 150 hydraulically-fractured wells per month was assumed.[1] Voluntary reports in FracFocus for 2011–2013 suggest that the average water use for hydraulic fracturing is 500 m^3 (130,000 gallons) per well, with the 90% confidence interval ranging from 470 to 540 m^3 (120,000 to 140,000 gallons). Based on these estimates, annual water use for hydraulic fracturing is estimated at 560,000 to 970,000 m^3 (150 million to 260 million gallons, or 450 to 780 acre-feet) per year. Assuming a higher average water use of 810 m^3 (210,000 gallons) per well, as estimated from well stimulation notices filed with Division of Oil, Gas and Geothermal Resources (DOGGR) in December 2013 and January 2014, total annual water use could be from 950,000 to 1,400,000 m^3 (770 to 1,160 acre-feet) per year.

[1] This estimate is based on the 76 operations per month implied by FracFocus, DOGGR's GIS well files, and well-record searches in combination, and the 190 hydraulic fracturing notices approved by DOGGR in December 2013.

Water use for matrix acidizing was estimated from information contained in well stimulation permit applications, or notices, filed by operators.[2] DOGGR posted 36 well stimulation notices planning to use acid stimulation on oil wells through mid-January 2014, although 10 of the notices were subsequently withdrawn. All 36 of these notices were filed by Occidental of Elk Hills, Inc. for wells in the Elk Hills oil field in Kern County. The 90% confidence interval for the mean water use, based on 36 notices, is 120–200 m^3 (32,000–53,000 gallons). Assuming a rate of 30 matrix-acidizing stimulations per month over the coming year results in an annual water use of 43,000-72,000 m^3/year (11–19 million gallons, or 35–58 acre-feet per year).

5.1.1.2 Water Sources

According to the well stimulation notices filed through the middle of January 2014, of the 249 planned well stimulation operations, operators plan to use *fresh water* for the majority of treatments (238 of 249), *produced water* for 10 operations, and both fresh and produced water for one operation. The average planned water use is 720 m^3 (210,000 gallons) per well, and the mean or variance does not appear to change depending on the water source (Table 5-1). Furthermore, the notices indicate that most planned hydraulic fracturing activity will occur in Kern County, and most operators plan to purchase water from nearby irrigation districts (Table 5-1). One district, the Belridge Water Storage District, is specified as the water source for 171 of the 213 permits and provides two-thirds of the estimated water supply. In some cases, operators frequently state a primary water source in their water management plans, while noting that water may also be withdrawn from on-site wells.

Table 5-1. Total planned water use for well stimulation by water source, from hydraulic fracturing notices posted in December 2013 through the middle of January 2014.

	Number of operations	Total Water (m^3)	Volume (acrefeet)
"District water"	9	16,000	13
Belridge Water Storage District; own wells	171	130,000	110
Casitas Municipal Water District	3	2,400	2
West Kern Water District*	55	21,000	17
Not specified	11	10,000	8
Total	**249**	**180,000**	**150**

**Note: All 36 permits for matrix acidizing operations filed to date are planned in the Elk Hills field by Occidental corporation, and plan to use water from the West Kern Water District*

[2] *Available through http://maps.conservation.ca.gov/DOGGR/iwst_index.html.*

The Belridge Water Storage District is an irrigation district formed to serve farmers in central Kern County. The primary source of water is from the State Water Project, although limited groundwater supplies are also available. The total planned water use by oil field operators filed during the first one and a half months of the program totals 110 acre-feet (130,000 m^3). This is less than 0.1% of the District's water use in 2012 but about 50% of the District's water used for oil production for that year (Belridge Water Storage District (BWSD) 2013). Deliveries of surface water from the State Water Project can be curtailed or even eliminated during drought years. Indeed, the State Water Project has announced that irrigation districts such as Belridge should plan to expect only 5% of its water allocation this year due to severe drought. The majority of crops within the District's service area are permanent crops that require water every year, increasing competition for limited water resources in the region. As an indication of the constraints on water in the region, Starrh and Starrh Farms, located within the BWSD service area, purchased 1,700 acre-feet of water from a nearby irrigation district at a cost of $1.97 million, or $1,130 per acre-foot (Henry, 2014).

As described above, operators noted that on-site groundwater wells may also be used for water for well stimulation treatments. There is a risk that accessing this water may come at the expense of other users, especially agricultural users in regions adjacent to oil and gas production fields. However, none of the operators specifies when, or under what circumstances, they would switch from purchased canal water to pumping from on-site wells. Groundwater pumping has a number of well-known and possibly detrimental impacts. Despite this, the state does not regulate the quantity of groundwater extracted from wells. Possible impacts of groundwater withdrawals, in addition to competition with agricultural uses, include decreases in river flows, land subsidence, permanent reductions in aquifer storage, increased pumping costs for neighbors, or nearby wells that run dry and need to be re-drilled and deepened. The type of impact and its magnitude will depend on local conditions, as well as the where, when, or how much water is used, and thus would require analysis on a site-specific basis. An additional area of interest and concern is the possible use of produced water for agricultural production. This is briefly addressed in Sections 5.1.3 and 5.1.3.1.4.

5.1.1.3 Comparison of Water Use to Other Regions Across the United States

There are few published estimates in the literature of water use for hydraulic fracturing in unconventional oil deposits. As described in Section 3.2.3, water use per well for hydraulic fracturing to produce oil in California is considerably lower than that reported to produce oil from the Eagle Ford unconventional play in Texas (Nicot and Scanlon, 2012). More generally, much of the published information on water use regards hydraulic fracturing to produce shale gas, which provides another basis for comparison. Average shale-gas water-use intensities of 3,800–23,000 m^3 (1–6 million gallons) per well have been reported in Texas (Nicot and Scanlon, 2012). A study for the US Department of Energy (DOE) reported median volume of fracturing water per well for select shale gas plays of 1,900–3,100 m^3 (2.3–3.8 million gallons; Ground Water Protection Council and ALL Consulting, 2009).

The smaller volume per well in California appears to result from a combination of factors as described in Section 3.2.2, including that vertical wells are predominant as opposed to horizontal wells in the comparison areas (Nicot and Scanlon, 2012; Ground Water Protection Council and ALL Consulting, 2009). Vertical wells presumably have shorter treatment lengths. In addition, gel, mostly cross-linked, is the predominant fracturing fluid. As discussed in Section 2.3.2, gels, particularly cross-linked, are typically used in smaller volumes than slickwater.

5.1.2 Chemistry of Fluids Related to Well Stimulation Operations

This section reviews the chemical compositions of waters related to well stimulation operations – namely the injection fluids (also referred to as well stimulation fluids or fracturing fluids), and the wastewaters recovered from well stimulation operations (i.e. flowback and produced waters). In addition, some contaminants that may be present in the injection and wastewater fluids are identified. This section also provides context for subsequent sections discussing the potential for fluids involved in WST operations to leak into shallow water resources through surface and subsurface pathways.

For the injection fluids (Sections 5.1.2.1 and 5.1.2.2), an evaluation is provided based on an analysis of acute, oral toxicity information for individual constituents that have been used in well stimulation operations in California. The list of chemicals was compiled from disclosures in FracFocus for hydraulic fracturing operations, and from stimulation notices submitted to DOGGR since December 2013 for matrix acidization operations. Both of these sources are dependent on self-reporting and may, therefore, not be comprehensive. The majority of the chemicals applied in California, for which toxicity information is available, are of low toxicity or non-toxic. However, some chemicals of concern were identified, including biocides (e.g. tetrakis(hydroxymethyl)phosphonium sulfate; 2,2-dibromo-3-nitrilopropionamide; and glutaraldehyde), corrosion inhibitors (e.g. propargyl alcohol), and mineral acids (e.g. hydrofluoric acid and hydrochloric acid). Approximately one-third of the chemicals had insufficient available information for evaluation. This toxicological assessment is limited as it considers only one chemical property (i.e. acute mammalian oral toxicity) that may impact human health, and does not consider other effects such as biological responses to acute and chronic exposure, eco-toxicological effects, overall toxicological effects of mixtures of compounds (compared to single-chemical exposure), and potential time-dependent changes in toxicological impacts of fluid constituents, due to their potential degradation or transformations in the environment. Thus further review of the constituents of injection fluids used in well stimulation jobs in California is needed.

For flowback and produced waters, Section 5.1.2.3 first outlines the general characteristics of flowback and produced waters across the U.S., to enable the reader to understand their typical constituents. However, flowback and produced water compositions vary considerably across regions, and their characteristics can change according to the fluids injected during well stimulation, the amount of fluids recovered at the surface, and over the duration of the flowback period. The chemistry of produced waters from unconventional oil production could potentially differ from that of conventional oil production in the same region due to differences in the target formations and interactions of fracturing fluids with formation rocks and water, although this does not generally appear to be the case based on the limited data that is available.

Sections 5.1.2.4 and 5.1.2.5 focus on identifying potential contaminants that could be present in flowback or produced waters, either due to the presence of injection fluids used in fracturing operations or due to dissolved constituents that may be present in formation waters brought up to the surface. Injection fluid constituents typically measured for their residual concentrations in flowback or produced waters include friction reducers, surfactants, PCBs, biocides, alcohols, glycols and organic acids, of which organic chemicals and biocides appear to be of particular concern. Furthermore, formation waters in oil reservoirs can contain naturally occurring dissolved constituents that can potentially degrade water quality, such as some major cations and anions that contribute to salinity and hardness (sodium, calcium, magnesium, chloride), trace elements including heavy metals, radiological material (NORMs), and organics. The list of potential contaminants identified in Section 5.1.2.5 is based on reports of contamination possibly related to well stimulation activities in the United States, but may not necessarily be applicable to California. It was not possible to provide an assessment of problems that may occur in California as there is no publicly available information about the composition of flowback and produced waters from well stimulation operations in California at the time this assessment was conducted.

Ultimately, the constituent concentrations in injection fluids, flowback and produced waters, as well as the specific exposure pathways, will determine potential hazards to human and ecological health. More data on the composition of injection fluids, and flowback and produced waters will enable a more comprehensive evaluation of the hazards to water quality due to fracturing operations in California.

5.1.2.1 Well Stimulation Injection Fluid Composition

As discussed in Section 2.3.2, fracturing fluids contain a series of reagents which serve various functions during the fracturing process. For example, sand is typically used as a “proppant” that ensures that the newly created fractures remain open. Other compounds such as guar gum are added to facilitate efficient delivery of proppant throughout the fracture zone, biocides are added to prevent the growth of bacteria, and other chemicals are added to minimize the mineral deposits (scaling) in the well. Classes of relevant chemicals include gelling and foaming agents, friction reducers, cross-linkers, breakers, pH adjusters/buffers, biocides, corrosion inhibitors, scale inhibitors, iron control chemicals, clay stabilizers, and surfactants (King, 2012; New York State Department of Environmental Conservation, 2011; Stringfellow et al., 2014, US Environmental Protection Agency (US EPA), 2004; Wilson and Schwank, 2013). Lists of common or widely used chemicals have been compiled based on regional or national usage (Stringfellow et al., 2014, US EPA, 2004; Wilson and Schwank, 2013), but no prior investigations have examined chemical use specific to California.

Extensive lists of chemicals frequently used during hydraulic fracturing nationwide are available in the literature (e.g. Stringfellow, et al., 2014). An example short-list of chemicals frequently used during hydraulic fracturing in Michigan is given in Appendix F (Table AF-1.) For this report, a list of constituents used in hydraulic fracturing in California was compiled using information voluntarily disclosed by industry on the FracFocus Chemical Disclosure Registry (http://fracfocus.org/). The FracFocus registry is not easily accessible and data from FracFocus have been compiled by SkyTruth (http://skytruth.org/) and DOGGR into searchable data sets. The data available from SkyTruth for the period between January 2011 (the earliest available) and May 2013 were combined with data compiled by DOGGR (Vincent Agusiegbe, personal communication, see Section 3 for details) for the remainder of 2013 to develop a list of chemicals used in hydraulic fracturing that is specific to California. This list is also presented in Appendix F (Table AF-2). Most of the data included in this analysis are from after April 2012 (see Figure 3-2 for details), which corresponds to an increase in data submissions shortly after a request DOGGR sent to operators in March 2012 asking for voluntary disclosure (Kustic, 2012).

The disclosed list of chemicals compiled by SkyTruth and DOGGR was ranked in terms of their frequency of use in fracturing for on-shore oil production in California; therefore hydraulic fracturing operations applied to natural gas and offshore oil production were not included in this analysis. All chemical used in more than 2% of the wells in California, where hydraulic fracturing was applied and disclosures to FracFocus were made, were included in this analysis. In total, 114 chemicals or chemical mixtures were reported as being used in more than 2% of the wells that have been hydraulically fractured in California. The majority of these additives were identified by Chemical Abstract Service (CAS) number (Table AF-2), but 17 were just identified by common name, group names, or names suggesting mixtures of compounds (Table AF-3). Chemicals can have multiple names, including common names, so CAS numbers are assigned to individual chemicals by the CAS of the American Chemical Society (https://www.cas.org/) to uniquely and definitively identify chemical compounds. Disclosure of chemical usage without reporting CAS numbers has limited value. The 97 chemicals reported with CAS numbers and used in more than 2% of the fracturing operations were further considered. For Table AF-2, chemical names (based on CAS Numbers) were selected from an US EPA report (US EPA, 2012) in order to provide consistency with previous publications.

In addition to hydraulic fracturing, well stimulation techniques also include matrix acidizing (discussed in Sections 2.4). A list of compounds used in matrix acidizing are given in Table AF-4. This list was developed from stimulation notices submitted to DOGGR between December 2013 and mid-January 2014 by operators and others who intended to perform well stimulation operations in the first part of 2014. Submitting a “Notice of Intent” is a new requirement in California as of December 2013, and although the list of compounds in Table AF-4 cannot be considered comprehensive, it is representative of current practices in California. All 70 listed compounds were used in at least 3% of the reported events, with 69 chemicals being applied to 47% or more events.

Comparison of Tables AF-1 and AF-2 suggests that chemicals used for hydraulic fracturing in California differ from chemicals used in other parts of the country. This conclusion is supported by comparison with lists of chemicals reported in the literature (King, 2012; New York State Department of Environmental Conservation, 2011; Stringfellow et al., 2014, US EPA, 2004; Wilson and Schwank, 2013). For example, the use of isothiozolone biocides appears to be more common in California and the use of glutaraldehyde and quaternary ammonia biocides less common in California than elsewhere. Overall, this voluntarily disclosed information of chemicals listed in Table AF-2 is consistent with earlier observations that the large majority of hydraulic fracturing applications in California use a gel-matrix approach and that the use of slick-water applications is less common in California than in other regions of the country. This conclusion is indicated by the high reporting frequency for guar gum and related compounds (used in gel treatments) and the absence of polyacrylamide compounds (used in slick-water treatments) in Table AF-2. The significance of these differences between chemical use in California and other regions of the country needs to be further investigated and confirmed, since chemical usage in industry is an important component of hazard assessment and risk analysis.

5.1.2.2 Preliminary Assessment of Hazards Associated with Well Stimulation Chemicals

Hazards associated with chemicals include physical, health, and environmental hazards (United Nations 2003). Physical hazards include properties such as flammability and oxidizing potential; health hazards include properties such as acute toxicity and skin irritation; and environmental hazards include both narrow and broad effects to environmental systems, particularly effects on aquatic organisms. A complete assessment of hazards associated with chemicals used in well stimulation in California is beyond the scope of this document, so for this report, only acute mammalian toxicity was investigated (see Appendix G). Mammalian toxicity is relevant for the evaluation of chemicals handled during well stimulation operations, especially in the context of the potential exposure of workers and the contamination of drinking water resources. Acute toxicity tests are commonly used as a reference point in both hazard and risk assessment. Examining acute oral toxicity has value for identifying potential chemicals of concern, but it is only the first step in understanding hazards associated with the chemicals used in well stimulation.

Tables 5-2 and 5-3 summarize the number of identified chemicals found in each Global Harmonized System (GHS) category (see Appendix G) for hydraulic fracturing and matrix-acidizing fluids, and provide the number of constituents for which no oral-toxicity information could be located. For this report we also identified compounds with categories above (>5), which may be interpreted as compounds that are non-toxic (Stringfellow et al., 2014). In Tables AF-2 and AF-4, GHS Categories are color-coded. There are no GHS Category 1 compounds (red color) found in the lists of well stimulation chemicals. However, for almost one third of the chemicals reported with CAS numbers, acute oral toxicity data could not be found (Table 5-2). The majority of the chemicals listed are in GHS Category 5 or above, suggesting they have low hazard potential in terms of oral toxicity. Examples of these lower toxicity or non-toxic compounds include guar gum (CAS 9000-30-0), a gelling agent, and ethanol (CAS 64-17-5), which is a common solvent.

Table 5-2. Grouping of chemicals found in hydraulic fracturing fluids in more than 2% of California hydraulic fracturing jobs based on GHS Categories for oral toxicity data (GHS category 1: most toxic; category 5: least toxic).

GHS Category	Number and Percent of Chemicals					
	Oral Rat LD_{50}		Oral Mouse LD_{50}		Oral Rabbit LD_{50}	
	[N]	[%]	[N]	[%]	[N]	[%]
1	0	0%	0	0%	0	0%
2	1	1%	1	1%	0	0%
3	7	7%	2	2%	2	2%
4	15	15%	13	13%	6	6%
5	17	18%	12	12%	2	2%
> 5	25	26%	12	12%	9	9%
No/insufficient data	32	33%	57	59%	78	80%
TOTAL	**97**	**100%**	**97**	**100%**	**97**	**100%**

Table 5-3. Grouping of chemicals found in injection fluids in more than 2% of California matrix acidizing operations based on GHS Categories for oral toxicity data (GHS category 1: most toxic; category 5: least toxic).

GHS Category	Number and Percent of Chemicals					
	Oral Rat LD_{50}		Oral Mouse LD_{50}		Oral Rabbit LD_{50}	
	[N]	[%]	[N]	[%]	[N]	[%]
1	0	0%	0	0%	0	0%
2	1	1%	3	4%	0	0%
3	6	9%	3	4%	0	0%
4	16	23%	9	13%	4	6%
5	15	21%	8	11%	2	3%
> 5	12	17%	10	14%	8	11%
No/insufficient data	20	29%	37	53%	56	80%
TOTAL	**70**	**100%**	**70**	**100%**	**70**	**100%**

For both hydraulic fracturing and matrix acidizing chemicals, oral toxicity data for rats were more readily available than data for mice or rabbits, providing information on 65 of 97 (66%) and 50 of 70 (71%) chemicals applied in hydraulic fracturing and matrix acidizing, respectively (Tables 5-2 and 5-3). Chemicals that had mice or rabbit toxicity data almost always also had data for rats, therefore, rat-based toxicity information provided the most complete basis for a qualitative comparison between oral toxicological effects of constituents found in hydraulic fracturing and matrix-acidizing fluids.

Using acute oral toxicity data for rats, most chemicals (59%) used in hydraulic fracturing are Category 4, 5 and above, and only 8% (representing 8 compounds) are Category 2 or 3. As discussed above, no compounds were in Category 1, the most toxic category. For matrix acidizing fluids, 61% of chemicals are in Categories 4, 5 and >5 and 10% are in Categories 2 and 3. Hence, based on this qualitative analysis, compounds added to injection fluids used in hydraulic fracturing and matrix-acidizing jobs for which oral toxicity data are available are characterized by a similar distribution of oral toxicities for rats.

Although acute oral toxicity data are useful for investigating hazards of industrial chemicals, as discussed above, oral toxicity is only one aspect used during the determination of hazards associated with chemicals. For example, constituents in matrix-acidizing fluids, such as hydrofluoric acid (CAS 7664-39-3), are hazardous, even if there is an absence of data on rat acute oral toxicity (Table AF-4). For example, hydrofluoric acid is toxic when inhaled; it is a contact hazard, causing skin corrosion or chemical burns when in contact with skin; and it is a dermal toxin, due to potential dermal absorption of fluoride.

A more complete analysis of hazards associated with well stimulation chemicals is needed. The total amounts of chemicals used and the concentrations at which they are applied needs to be determined. Other properties that need to be assessed include variables such as corrosivity, ignitability, and chemical reactivity. Future assessments need to evaluate whether the well stimulation chemicals are carcinogens (substances that can cause cancer), endocrine-disrupting compounds (chemicals that may interfere with the body's endocrine system and produce adverse developmental, reproductive, neurological, and immune effects), and bioaccumulable materials (chemicals that increase in concentration in a biological organism over time compared to their concentrations in the environment). Previous studies suggest that some of the compounds listed on Table AF-1 may be endocrine disrupting compounds (Colborn et al., 2010, Kassotis et al., 2013). Chemicals that are endocrine disrupting, carcinogenic, or that bioaccumulate potentially can cause long-term or chronic impacts on ecosystems. Long-term and chronic effects are not necessarily indicated by results of LD_{50} tests as presented in this report.

Potential toxicological hazards may not only involve effects on humans, but also any impacts on aquatic organisms and other receptors. An evaluation of eco-toxicological effects, including the potential impacts of these chemicals on aquatic organisms is needed. Such an analysis will need to consider the large variety of types of toxicity tests applied in this area, which confounds direct comparisons between chemicals. A more complete evaluation of potential eco-toxicological effects of injection fluids applied during well stimulation is needed in the future.

Future analysis should also take into account and evaluate the potential interactive effects between chemicals. The toxic effects of a mixture of two or more compounds can be substantially different from that suggested by simply adding the effects of the single compounds. Overall effects can potentially be smaller or larger, depending on the specific interactions between compounds in the mixture, changes in uptake, etc. To our knowledge, no studies are currently available in the peer-reviewed literature describing the overall toxicological effects of fracturing fluids as mixtures of many different types of compounds; however, a few examples of studies are available where combined effects of small subgroups of fracturing fluid chemicals have been evaluated. For instance, a combination of two biocides (Di-Me Oxazolidine and glutaraldehyde) has been shown to achieve equivalent performance of either alone in fracturing fluids while improving the overall ecotoxicity profile (Enzien et al., 2011). For endocrine-disrupting chemicals acting through a common biological pathway, additive effects of mixtures have been observed, even when individual chemical concentrations were present at levels below an observed effect threshold (Christiansen et al., 2008; Silva et al., 2002; Christiansen et al., 2009).

The use of effluent toxicity tests may be useful for evaluating the effects of mixtures of well stimulation chemicals and associated wastewaters (Riedl et al., 2013). Whole Effluent Toxicity (WET) tests are specifically designed to evaluate the aggregate toxic effects of an aqueous sample without precise information about the chemicals causing that toxicity (US EPA, 2002). For instance, in California, these US EPA methods have been applied in order to evaluate the water quality of agricultural drains in the San Joaquin River and Sacramento River watersheds (Vlaming et al., 2004). Other researchers have evaluated complex mixtures for endocrine disrupting activity using whole water samples (Soto et al., 2003; Zhao et al., 2011). The application of toxicological and eco-toxicological methods for testing mixtures of fracturing fluids is recommended in future studies.

Finally, degradation and transformation reactions affecting fracturing-fluid constituents in the environment need to be considered for future studies. Degradation and transformation reactions could cause either an increase or decrease in toxicological effects. For example, some biocides, such as glutaraldehyde, degrade relatively quickly in the subsurface, leading to lower toxicities in flowback water compared to the injected fluid (Blotevogel et al., 2013). In contrast, a photochemical degradation of polyacrylamide polymers may result in increased environmental hazard, since acrylamide monomer units are more toxic than the parent polymer and acrylamide is a mammalian neurotoxin and a probable carcinogen (Brown et al., 1980).

In summary, numerous chemicals are used for well stimulation in California. A full assessment of the hazards associated with those chemicals is needed. The extensive list of possible WST chemicals provides only part of the information needed to assess risk; additional information on concentrations, synergistic interactions, exposures, and more are also needed to assess risks and environmental impacts from WST. A preliminary assessment, using mammalian acute oral toxicity as a screening criteria, suggests that only a few of the well stimulation chemicals can be considered highly toxic and most compounds are of equivalent toxicity to many commonly used industrial and household chemicals (such as anti-freeze). We note, of course, that many household and industrial chemicals also have potential toxicity under certain circumstance or in different combinations, and we recommend that all such risks be carefully assessed as part of future investigations of risks associated with WST. Numerous compounds can be classified as non-toxic and some are allowed as food additives or are found in food naturally. It is emphasized that mammalian oral toxicity is a very limited screening criteria and that a more complete hazard assessment must include physical, health, and environmental hazards. Other factors that must be considered to fully evaluate hazards associated with these chemicals include eco-toxicological effects, endocrine disruption, bioaccumulation, environmental transformation, and the properties of mixtures of compounds.

5.1.2.3 General Characteristics of Flowback and Produced Waters

After completion of the stimulation process, the pressure in the well is released and the direction of flow is reversed, bringing some of the injected stimulation fluid and formation water to the surface (see Section 2.3 for a description of the hydraulic fracturing process). This fluid is generally classified as either flowback or produced water. Flowback is commonly defined as the return of injected fluids and produced water is water from the formation (US EPA, 2012). The distinction between flowback and produced water during operations is not clear-cut, since mixing occurs in the formation. In practice, the term flowback is used to refer to initial, higher flows in the period immediately after well stimulation and produced water refers to long-term, typically lower flows associated with commercial hydrocarbon production. After the pressure in the well is reduced, flowback water is returned to the surface at high rates for up to several weeks, and this flow is, initially, predominantly fluids that were injected, but over time the fraction of the fluid that represents formation water increases (Barbot, et al. 2013; Clark et al., 2013; Haluszczak et al. 2013: King 2012). Produced water flows to the surface, along with the gas or oil, throughout the production life of the well and originates from water naturally trapped in the geologic formation (King 2012).

Flowback fluids consist of (1) fracturing/injection fluids pumped into the well previously, which include water and the additives described in Section 5.1.2.1, (2) new compounds that may have formed due to chemical reactions between additives, (3) dissolved substances from waters naturally present in the target geological formation, (4) substances that have become mobilized from the target geological formation due to the interaction of fracturing fluids with formation rocks and water, and (5) some oil and/or gas (New York State Department of Environmental Conservation, 2011; Stepan et al., 2010). Thus, the chemistry of flowback waters is generally different from that of the injection fluids, as shown by the example in Table 5-4.

The composition of flowback fluids usually changes over the course of the flowback time-period, gradually evolving from being more similar to the injection fluids to approaching the chemical characteristics of the formation waters. For example, fluid-composition changes were observed in studies conducted in the Marcellus shale (Hayes, 2009; Barbot et al., 2013) and the Bakken (Stepan et al., 2010), indicating concentration increases in the flowback water collected over time for constituents such as TDS (such as shown on Figure 5-1), chloride, and some cations/metals (such as shown on Figures 5-2 and 5-3). In the Marcellus study, water hardness and radioactivity levels were found to increase during the flowback period, but sulfate and alkalinity levels decreased with time.

Table 5-4. An example of differences in the composition of injection fluids and 14-day flowback water collected from seven horizontal wells in the Marcellus shale (Table from Haluszczak et al., 2013, based on data from Hayes, 2009)

	Concentrations in mg/L		
	Injected fluid median, day 0	**Flowback median, day 14**	**Flowback range, day 14**
pH	7.0	6.2	5.8–6.6
Alkalinity as $CaCO_3$	126	71	26–95
Total dissolved solids	735	157000	3010–228,000
Total organic carbon	205	14	1.2–509
Chemical oxygen demand	734	8370	228–128,000
Cl	82	98300	1070–151,000
Br	<10 (<0.2–19)	872	16–1190
SO_4	59	<50*	0.8–89
NH_3-N	16	193	4–359
P	0.36	0.55*	0.04–2.2
Al	0.3*	0.5	0.15–0.91
Ba	0.6	1990	76–13,600
B	0.5	20	2.7–3880
Ca	32	11200	204–14,800
Fe	0.68	47	14–59
K	<50 (3-57)	281	8–1010
Li	0.04	95	4–202
Mg	3.7	875	22–1800
Mn	0.074	5.6	1.2–8.4
Na	80	36400	1100–44,100
Sr	0.82	2330	46–5350
Zn	0.08	0.09	0.07–0.14

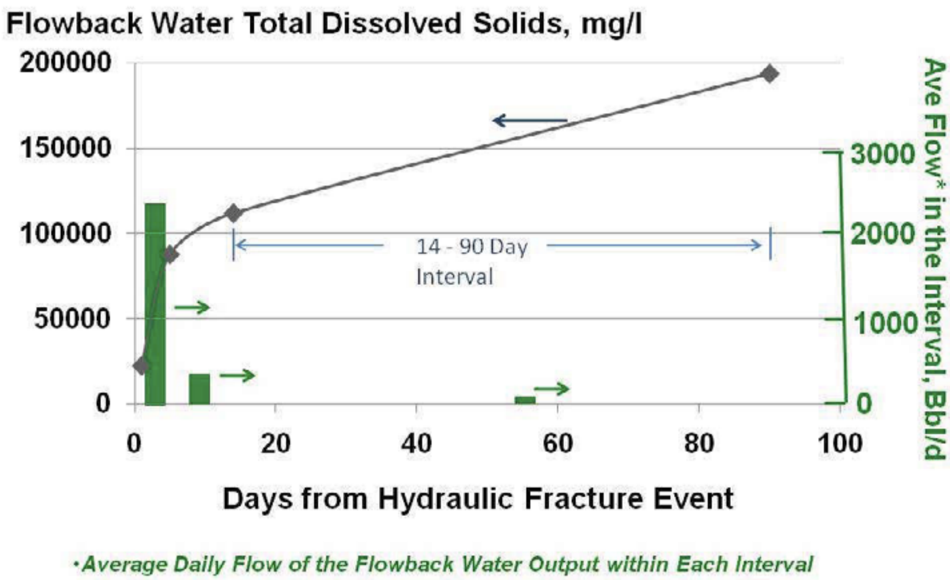

Figure 5-1. TDS content of flowback waters typically increases during the flowback period (Figure from Hayes, 2009 showing data from the Marcellus shale)

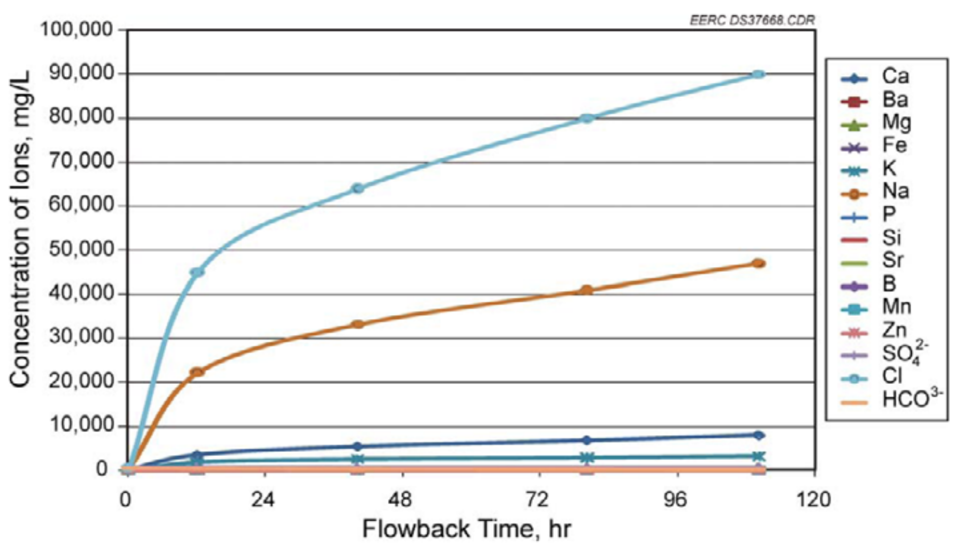

Figure 5-2. Concentrations of some cations (e.g. calcium, potassium, sodium, iron) and anions (e.g. chloride) typically increase during the flowback period in the Bakken shale (Figure from Stepan et al., 2010).

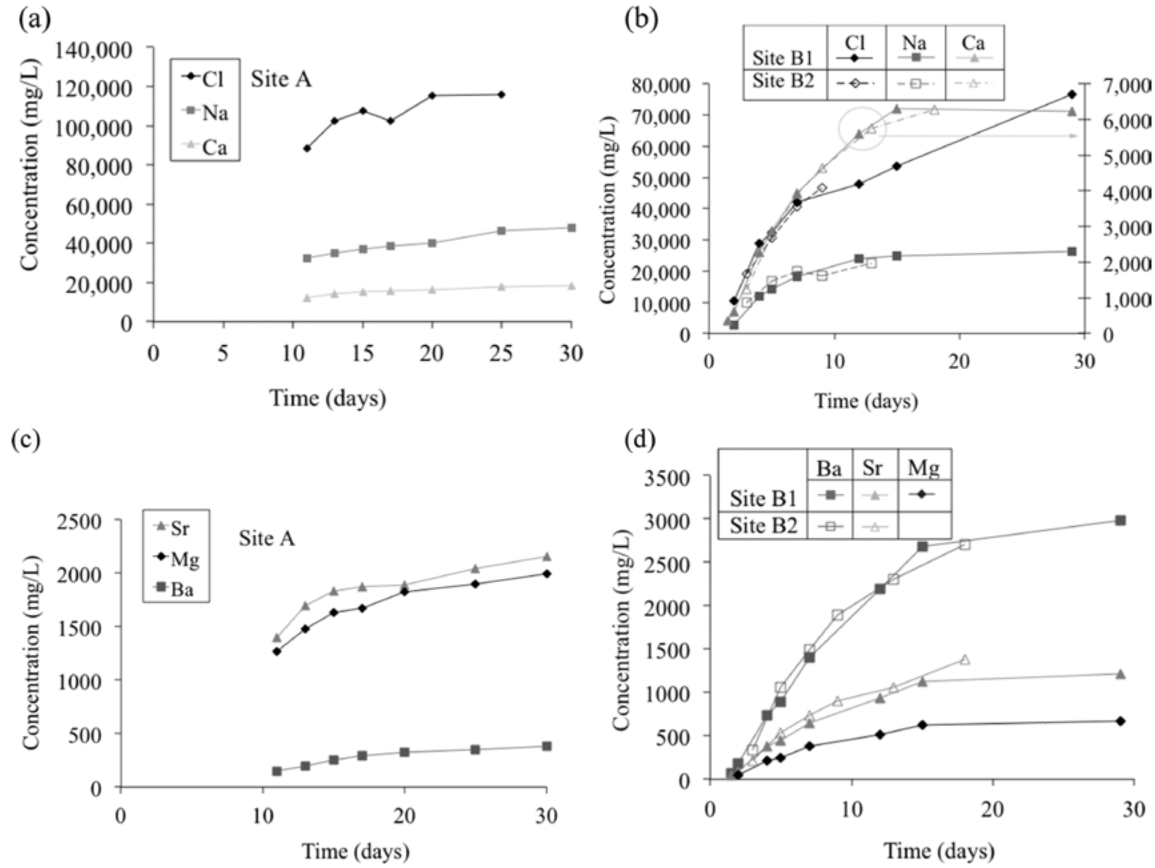

Figure 5-3. Concentrations of some cations (e.g. calcium, sodium, strontium) and anions (e.g. chloride) typically increase during the flowback period in the Marcellus shale (Figure from Barbot et al., 2013). The concentrations of these ions increase over time because the chemistry of the fluid changes from resembling the injection fluids (that are made using waters with low TDS) to formation waters (these typically have high TDS because the waters in most formations are of marine origin).

The duration of the flowback periods can range anywhere from two days to a few weeks, and can vary between producers within a region (e.g., Hayes, 2009; Stepan et al., 2010; Warner et al., 2013; Barbot et al., 2013). Besides variation during the duration of the flowback period, compositions of flowback and produced waters are known to vary geographically, as shown in Table 5-5 (Bibby et al., 2013). The chemical composition of these waters ultimately determines the options available for their treatment, reuse, and disposal, as discussed in Section 5.2.3.1.4.

Once the well is placed into production, the waters recovered from the operations are "operationally defined" as "produced waters." One question that this report addresses is whether produced waters from WST operations in California are different from waters recovered during conventional oil and gas production. It turns out it is difficult to evaluate this question, given the wide variations in the water chemistries of flowback and produced waters, as well as the scarcity of recent data from unconventional and conventional production. A limited number of studies in other regions suggest that the hydraulic fracturing operation has little effect on the eventual produced water chemistry. For example, an industry-sponsored study by the Gas Coalition Institute focusing on fracturing operations in the Marcellus shale (Hayes, 2009) concluded that the general water chemistries of produced water from conventional and unconventional productions are similar. A subsequent study from Pennsylvania State University (Haluszczak et al., 2013) used four different data sources, including the data from the Gas Coalition Institute, and similarly concluded that the general chemistry of later flowback/produced water resembled brines produced from conventional wells, although they also noted that the concentrations of NORMs in the flowback waters (^{226}Ra and ^{228}Ra) were high. Specifically for California, the samples for which data is reported in the "USGS produced water database 2.0" (United States Geological Survey (USGS), 2014), were collected from conventional hydrocarbon wells before 1980. Thus these samples may not be representative of modern produced waters from conventional extraction or of produced water from well stimulation operations. More data is needed on the composition of flowback/produced waters from well stimulation operations in California to assess whether the fluid chemistries would differ significantly from conventional production.

Table 5-5. Comparison of produced water compositions from unconventional and conventional oil and gas operations.

Parameter	Marcellus[a]	Bakken[b]	Conventional Oil[c]	Conventional Oil and Gas (California)[d]
pH	5.1–8.4	5.5–6.5	5.2–8.9	2.6–11.5
Conductivity (mS/cm)		205–221		
Alkalinity (mg/L as CaCO3)	8–577		300–380	
TSS (mg/L)	4–7600			
TDS (mg/L)	680–345000	150000–219000		1000–84891
Chloride (mg/L)	64–196000	90000–130000	36–238534	0–156000
Sulfate (mg/L)	0–1990	300–1000		0–14879
Bicarbonate (mg/L)	0–763	300–1000	8–13686	0–12809
Bromide (mg/L)		300–1000	1–2	1–207
Nitrate (mg/L)	5–802		0–92	0–18
Oil and Grease-HEM (mg/L)	195–36600			
COD (mg/L)	1–1530			
TOC (mg/L)			15–3501	0–2054
Aluminium (mg/L)		ND	0.0–0.1	0–250
Arsenic (mg/L)			0.2–0.9	

Parameter	Marcellus[a]	Bakken[b]	Conventional Oil[c]	Conventional Oil and Gas (California)[d]
Barium (mg/L)	0 - 13800	0 - 25	0.1 - 7.4	0 - 174
Boron (mg/L)		40–192		0 - 602
Calcium (mg/L)	38 - 41000	7540 - 13500	4 - 52920	0 - 13613
Cadmium (mg/L)			0.0 - 0.2	
Chromium (mg/L)			0.1 - 1.0	0 - 200
Copper (mg/L)		ND	0.3 - 2.7	0 - 100
Iron (mg/L)	3 - 321	ND	0.1 - 0.5	0 - 540
Potassium (mg/L)		0 - 5770	2 - 43	0 - 7987
Magnesium (mg/L)	17 - 2550	630 - 1750	2 - 5096	0 - 2260
Manganese (mg/L)		4 - 10	1 - 8	0 - 50
Sodium (mg/L)	69 - 117000	47100 - 74600	405 - 126755	0 - 99920
Nickel (mg/L)			3 - 10	0 - 30
Strontium (mg/L)	1 - 8460	518 - 1010	0 - 2	0 - 600
Zinc (mg/L)		2 - 11	6 - 17	
Ra 226 (pCi/L)	3 - 9280		0 - 10	
Ra 228 (pCi/L)	0 - 1360			
U235 (pCi/L)	0 - 20			
U238 (pCi/L)	0 - 497			
Gross alpha (pCi/L)	37 - 9551			
Gross beta (pCi/L)	75 - 597600			

[a] *Barbot et al. (2013)*

[b] *Stepan et al. (2010)*

[c] *Alley et al. (2011)*

[d] *Compiled for this report from the USGS Produced Water Database 2.0 (USGS, 2014)*

Another open question that remains is the extent of recovery of stimulation fluids during the flowback period. The volume of flowback water recovered may affect the fate of the injected fluid retained in the formation, and the potential for future mobilization of fracturing-fluid constituents in subsurface environments. Recoveries of flowback water will depend on various factors, including how much free water is present in the formation, as well as the rock and fluid properties in the target. A considerable amount of water can be retained in the formation, given that recoveries of fracturing fluids are relatively low - e.g., ranging between 9% and 53% in the Marcellus shale (New York State Department of Environmental Conservation, 2011; Vidic et al., 2013), and between 5% and 41% in the Bakken (Stepan et al., 2010).

Although it is unlikely that retained fracturing fluids will migrate out of the reservoir as discussed below in Section 5.1.3.2, these fluids can potentially interact with formation rocks over time. Hence, the resulting products of these fluid-mineral interactions, which can potentially include environmental contaminants, may appear in produced waters at a later stage. While some consider the environmental risks associated with “trapped chemicals” to be low (King, 2012), more studies are needed on the interactions of injection fluids and their additives with formation rocks and the overall fate of injection fluids in the subsurface environment, in order to determine if these have the potential to alter the chemistry of produced waters over the long term (such as causing slow release of trace metals or radioactive elements).

Available California data do not include specifics on the recovery of fracturing fluids during well stimulation. However, somewhat different recoveries may be expected in California, for two reasons. First, targets in California vary from those in other states geologically. For instance, diatomite, which is one of the main targets for hydraulic fracturing (see Section 3.2.1), has high porosity (as described in Section 4.3.2). Regarding matrix acidizing using mud acid, hydrofluoric and hydrochloric acids are expected to become consumed due to acid-mineral interactions over short penetration depths, while the remaining fluid often migrates over further distances. Assuming the rock is not fractured, acid penetration depths in sandstones have typically been reported to be on the order of 0.3 m (12 inches; Economides and Nolte, 2000) or less than 0.3-0.6 m (1-2 feet; Kalfayan, 2008). However, for high-permeability, high-quartz sands and fractured formations, such as the Monterey Formation (in places), higher than typical volumes of mud acid 3.1 to 3.7 m^3/m (250 or 300 gallons per ft) have been applied to open fracture networks deeper in the formation (Kalfayan, 2008; Rowe et al., 2005).

Second, as described in Section 3.2.4, the predominant fracturing fluid applied in California is a gel, which may vary from the fluids used in the Marcellus and Bakken flowback fraction studies cited. This in turn may affect the penetration depth of injection fluids, the later recovery of fluids, as well as the recovery of specific, individual constituents. Additional information regarding the total estimated volume of recovered fluids in California should become available in the near future, due to new DOGGR reporting requirements (DOGGR, 2013).

5.1.2.4 Fracturing-Fluid Constituents in Flowback and Produced Waters

With respect to fracturing fluid constituents, degradation reactions and interactions with mineral phases within the reservoir may affect their individual recoveries and/or recovery rates. Fracturing fluid constituents that are typically evaluated for their residual concentrations in flowback or produced waters include friction reducers, surfactants, polychlorinated biphenyls (PCBs), biocides, alcohols, glycols, and acids, such as acetic acid (New York State Department of Environmental Conservation, 2011).

For instance, Orem et al. (2014) reported that the general composition of organic substances in produced and formation waters from coalbed methane and gas shale plays across the U.S. were similar. However, the researchers noted that produced water from hydraulic fracturing operations at the Marcellus shale contained a range of additional organic chemicals used as fracturing-fluid constituents, such as solvents, biocides, and scale inhibitors, at levels of 1,000s of μg/L (parts per billion) for individual compounds. Elevated total organic carbon (TOC) concentrations as high as 5,500 mg/L were present in produced waters from hydraulic fracturing operations in the Marcellus shale compared to about 8 mg/L for conventional production. While the concentrations of hydraulic fracturing chemicals and TOC decreased rapidly over the first 20 days of water recovery, some residual organic contaminants remained up to 250 days after hydraulic fracturing. In particular, biocides, which are toxic by necessity, are expected to persist in flowback water, and limit the options for flowback water disposal in the case of high concentrations (Rimassa et al., 2011).

An assessment of fracturing fluids being present in flowback/produced waters in California was not conducted due to the lack of data. Additional information regarding the specific composition of recovered water associated with well stimulation treatments should become available in the near future, due to new California reporting requirements (DOGGR, 2013).

5.1.2.5 Potential Direct Contaminants from Target Formations in Flowback and Produced Waters

The groundwater present in oil and gas reservoirs can contain naturally existing dissolved constituents such as stray gas (e.g., methane), salts, trace metals, NORMs (naturally occurring radioactive materials) and organic compounds that are released into the waters upon their interaction with formation rocks. The amount of dissolved material present in the fluids will depend on several characteristics of the formation, such as its geology, geochemistry, and microbiology. These dissolved constituents can be present in flowback and produced waters recovered at the surface, and can potentially degrade the water quality of shallow groundwater and surface-water resources, if released into those environments.

This section discusses potential contaminants that can be naturally present in the formation, i.e., those substances that have not been added to the injection fluids and include TDS (salts), trace metals, NORMs and organic compounds. Such contaminants could be present at higher levels in flowback and produced waters from unconventional production, as compared to oil and gas conventional production due to differences in the geology of the targets and chemistry of the formation waters. Formation waters are typically high in TDS, and organics, and several studies (particularly in the Marcellus shale) have noted high TDS values present in flowback and produced fluids from stimulation operations. Well stimulation could also lead to the potential release of trace metals due to decrease in pH (that may be relevant in acid stimulation operations) or complexation with organic ligands present in the injection fluids. The source rock (e.g. many shales) may also contain high concentrations of radioactive elements, which may be dissolved in formation waters.

The section outlines some of the problems that have been reported for these potential contaminants in regions where well stimulation has been conducted in the United States (several of these are in the Marcellus shale). Since the composition of flowback and produced waters varies considerably with geography, the relevance of these issues to California is discussed wherever possible. However, it is not currently possible to evaluate whether this list of potential contaminants is complete or even relevant to California since data about the chemistry of flowback and produced waters from stimulation operations in California was not available at the time this assessment was done.

The discussion for this report emphasizes contaminant concerns that are amplified due to the use of well stimulation; the report does not review some of the problems typically associated with conventional oil and gas operations, e.g., contamination by hydrocarbons such as benzene, toluene, ethylbenzene, and xylenes (BTEX), generation of H_2S due to biosouring of wells (Chilingar and Endres, 2005), or contaminants originating from drilling mud, cuttings, and fluids. It should be recognized that contaminants detected in conventional oil and gas operations, although not within the scope of this report, can also be a concern in well stimulation operations.

The interaction of well stimulation fluids with a formation containing unconventional oil can also result in effects such as transformations of the constituents of formation waters and sediments. These include changes to redox conditions, which can occur due to the introduction of oxygen from the stimulation fluids (for instance). Changes to redox can trigger a host of subsurface geochemical reactions such as oxidation of iron, pyrite, or organic matter present in a formation. Introduction of the stimulation fluids to the formation can also lead to changes in microbial communities (Mohan et al., 2013; Struchtemeyer and Elshahed, 2011). The potential changes to flowback fluids due to these effects were not investigated for this report.

5.1.2.5.1 TDS, Salinity and Water Hardness

TDS is defined as the total concentration of solids that will pass through a 0.2 μm filter in solution. Typical TDS values in fresh water are <1,000 mg/L, between 15,000-30,000 mg/L in saline water, between 30,000-40,000 mg/L in seawater, and >40,000 mg/L in brine (wqa.org). Formation waters can contain high TDS concentrations, with salinities far exceeding seawater values, because many shales have marine origins (King, 2012). Thus, flowback and produced waters from well stimulation operations can contain high concentrations of TDS, although the concentrations change during the flowback and production periods. The source of the TDS and salinity in recovered wastewaters could either be salts present in formation brines or salts dissolved from formation rocks (Blauch et al., 2009). Some studies suggest that the TDS in the recovered wastewaters could result from mixing of injection fluids with formation brines (Haluszczak et al., 2013; Engle and Rowan, 2013). But another study of Marcellus shale produced waters found that, while most major cations were correlated with chloride, the variations in their concentrations could not be explained by dilution of existing formation brine with fracturing fluid

(Barbot et al., 2013). A study using strontium isotopes to characterize the signatures of produced waters suggested a basin-wide source of TDS in the Marcellus shale (Chapman et al., 2012).

The most concentrated ions found in flowback and produced waters are typically sodium and chloride (Barbot et al., 2013; Blauch et al., 2009; Haluszczak et al., 2013; Warner et al., 2012). Table 5-6 indicates this is the case in California as well. Magnesium and calcium can also be present at high levels and can contribute to increased water hardness. Typically, sulfate and alkalinity (measured as carbonate or bicarbonate) concentrations were low.

Table 5-6. Average concentrations of major ions and TDS (mg/L) in produced water samples from conventional oil and gas basins in California. Data from the USGS produced water database (USGS, 2014). All samples were collected before 1980.

BASIN	DATA POINTS	pH	BICAR-BONATE	CAL-CIUM	CHLO-RIDE	MAGNE-SIUM	POTAS-SIUM	SODIUM	SULFATE	TDS
COASTAL BASINS	14	7.9	1469	154	5257	82	71	3777	68	11169
LA BASIN	318	7.4	1060	604	16428	300	151	9399	35	27773
SACRAMENTO	12	6.4	372	191	9890	68	26	5980	18	16633
SAN JOAQUIN	344	7.4	1407	764	11121	133	259	5208	88	19570
SANTA MARIA	41	7.4	1354	435	10703	200	118	6047	849	18922
VENTURA	41	7.4	1670	958	13234	167	134	5972	170	26396

TDS can be a concern if present in high concentrations in flowback/produced waters. For example, the TDS content in the Marcellus shale is high, with ranges in flowback waters between 680 and 345,000 mg/L (ppm; Hayes, 2009). Such high TDS values are consistent with waters in the Marcellus being the second saltiest of all basins in the United States (Vidic et al., 2013). One study (Haluszczak et al., 2013) concluded that flowback waters from hydraulic fracturing of Marcellus wells resembled brines produced from conventional gas wells in the region. A study in the Bakken found large differences in the salinities of flowback water, not only between different producers, but also among different wells of a single producer, with values ranging from 60,000 mg/L to over 200,000 mg/L (Stepan et al., 2010).

The TDS values of flowback/produced waters from well stimulation operations in California may be lower than those reported in other regions. Produced waters in California have historically tended to have lower TDS concentrations as shown in Table 5-6. TDS values ranging from 10,000 to 40,000 mg/L have been reported for the formation waters in the San Joaquin and Sacramento basins at depths ranging from ~1,500 to ~3,500 m (Kharaka et al., 1985). A study of produced waters collected from the San Joaquin Basin found TDS of waters produced from depths <1,500 m were <4,000 mg/L (typically

<2,000 mg/L), whereas waters produced from depths >1,500 m were more saline (typically >25,000 mg/L) (Fisher and Boles; 1980). This depth-salinity pattern was found to be consistent with the transition in the basin from nonmarine strata at shallow depths to marine strata at greater depths. Since fracturing operations in California are conducted at shallower depths than in other regions - e.g. more than half the wells that have been stimulated using hydraulic fracturing are within 610 m (2000 feet) of the ground surface (Section 5.1.3.2.1), it is expected that TDS values of target formation waters, and hence of flowback/produced waters will be relatively low (as compared to wastewaters from stimulation operations in other regions in the United States) based on the depth-salinity gradient patterns in the formation, It is also possible that the TDS content of waters recovered from stimulation jobs using gels will be different from TDS values reported for slickwater fracturing. Well stimulation using gels is more common in California (Section 3.2.4).

Contamination by TDS/salty brines has been a problem in some areas where wastewaters recovered from WST operations ultimately ended up in freshwaters (Section 5.2.3.1.4). However, contamination of freshwaters is expected to be less problematic when the recovered fluids are disposed into Class II injection wells or reused in well stimulation operations, as is expected to be the case in California. High TDS values in flowback and produced waters can still be a concern if improperly handled at the surface during management, disposal or reuse.

5.1.2.5.2 Trace Metals

Formation brines can contain high concentrations of trace metals, which may be brought up to the surface in flowback and produced waters. Several studies report measuring high levels of barium, strontium, and iron in the waters recovered from fracturing operations in the Marcellus shale (e.g., Hayes et al., 2009; Barbot et al., 2013; Haluszczak et al., 2013). However, concentrations of trace elements in flowback and produced waters can vary widely across shale plays. For example, barium concentrations in the Fayetteville, Barnett, and Bakken shales can be much lower than elsewhere (Jackson 2013; Stepan et al., 2010). There is no current information available on the trace-element composition of flowback or produced waters recovered from stimulation operations in California.

The Monterey formation is high in trace elements compared to the World Shale Average (WSA) abundance (http://energy.cr.usgs.gov/TraceElements/faq.html). In particular, the lower and middle portions of the Monterey formation consist of different types of lithologies that might be relevant to mobilization of trace metals from the formation, i.e., carbonate-rich, organic-rich shales and phosphatic rock units as discussed in Section 4.3.1. The Monterey formation is also known to have selenium-enriched stratigraphic zones (Issacs 1999). Concentrations of some elements such as chromium, copper, nickel, antimony, selenium, uranium, vanadium, and zinc have been found to be highly correlated with organic carbon content. Other trace metals that were somewhat correlated with organic carbon include As, Ba, Cd, and Mo (Isaacs 1999). Borehole cuttings from the Santa Maria and Santa Barbara areas also had similar patterns showing high trace-element concentrations.

However, the release of trace elements present in the source rock into formation waters is dependent on several factors, such as pH, redox conditions, temperature, and the presence of organics (Kharaka et al., 1985). In general, trace elements can be mobilized as a result of decreases in pH, changes to redox, and the presence of organic ligands that can form complexes with metals (Stumm and Morgan, 1986). Some trace elements (particularly those present as cations) can be potentially mobilized due to decreases in pH, which may be relevant in matrix acidizing jobs. However, some contaminants (e.g., anionic species like arsenate) can be favorably attenuated as a consequence of pH decrease. The injection of biodegradable organic chemicals could result in both pH and redox changes in the subsurface that could alter trace metal mobility. These potential effects are not fully understood.

5.1.2.5.3 Naturally Occurring Radioactive Materials (NORMs)

NORMs include elements such as uranium, radium, and radon gas that are present in low concentrations in ambient soil and groundwater. Formation brines in contact with organic-rich shales can naturally contain high concentrations of radiogenic material. Uranium and thorium are present in many shale source rocks, are typically associated with high organic content (Ferti and Chillinger, 1988), and can decay to ^{226}Ra, ^{228}Ra and radon (Rowan et al., 2011).

The Monterey Formation is approximately six times more enriched in uranium than the WSA values (http://energy.cr.usgs.gov/TraceElements/monterey.html). Uranium concentrations in the Monterey formation rocks range from <2 ppm to more than 1,850 ppm (USGS, 1987). However, the uranium content of California crude oil is not typically high; for example, uranium concentrations in crude oil samples from Tertiary rocks in California ranged from 0.1 to 37.7 ppb (Bell, 1960).

Problems with elevated levels of radium have been noted in oil field equipment that process produced waters from conventional oil and gas production (USGS, 1999), particularly because radium is easily incorporated into barite (barium sulfate) scales, which precipitate when produced waters are brought to the surface. The hazard to operators and to the general public due to radioactive material trapped in scales within oilfield equipment are expected to be low (ALL Consulting, 2008). Moreover, in a survey of oil field equipment conducted by the American Petroleum Institute in 1989, the measurable radioactivity on external surfaces of equipment in California was at or near background level (USGS 1999).

However, flowback and produced waters from some shale formations can potentially contain high levels of NORMs that can be several hundred times U.S. drinking water standards. Several studies have measured high levels of radioactivity in samples collected from the Marcellus shale, which is known to contain radioactive elements (Hill et al., 2004). For example, the highest level of total radium measured in a study of flowback waters from Pennsylvania was 6540 pCi/L (Haluszczak et al., 2012) and uranium concentrations in produced waters from N. Pennsylvania ranged from 0-20 pCi/L for U-235 and 0-297 pCi/L for U-238 (Barbot et al., 2013). Production brine samples from New York showed elevated gross alpha and gross beta results, ranging 14,530 - 123,000 pCi/L, with concentrations of ^{226}Ra ranging from 2,472 to 16,030 pCi/L (NYSDEC, 2009). A study of various samples from the Marcellus shale found radium activities to range from non-detect to 18,000 pCi/L (Rowan et al., 2011). The high concentrations of NORMs found in flowback/produced waters from other shale plays do not imply that a similar situation will occur in California. No information about radioactive element concentrations in flowback or produced waters from stimulation operations in California could be located for this assessment, which is a major data gap in evaluating the hazards associated with WST.

5.1.2.5.4 Organics

Produced waters from oil and gas operations typically contain many organic substances that can originate from sources such as the formation water, formation rocks (e.g. organic-rich shales), oil present in the formation, and (in the case of well stimulation) from chemical additives added to the injection fluids (Orem et al., 2014). Section 5.1.2.1 lists the most commonly disclosed constituents of the stimulation fluids, including organics. The presence of organics in produced waters from conventional oil and gas operations have been extensively described in the literature, including in California (e.g., Fisher and Boles, 1990; Higashi and Jones, 1997). Organic compounds typically found in conventional produced waters include organic acids, polycyclic aromatic hydrocarbons (PAHs), phenols, and volatile organic compounds (VOCs) such as BTEX and naphthalene (Veil et al., 2004).

Very few studies have examined the presence of organics in produced waters from WST operations. Often organics are not measured, since these analyses are expensive and time-consuming. One industry-sponsored study by the Gas Coalition Institute (Hayes, 2009) measured a suite of organics in Marcellus shale flowback waters at the suggestion of the Pennsylvania Department of Environmental Protection, including VOCs, semi-volatile organic compounds (SVOCs), pesticides, and PCBs. The concentrations of most organic constituents were found to be below detection limits, and those VOCs that were measurable were similar to those found in conventional produced waters. The study concluded that it was unnecessary to measure pesticides, PCBs, and a large fraction of VOCs and SVOCs in produced waters from well stimulation. It is worth noting that this study did not measure the non-volatile, polar and water soluble compounds used in well stimulation fluids (Section 5.1.2.1).

As described in Section 5.1.2.4 regarding fracturing fluid in flowback/produced waters, the constituents of injection fluids make up the large fraction of organics additionally present in produced waters from hydraulic fracturing in the Marcellus Shale. These organics are not typically detected in conventional produced waters (Dahm et al., 2012; Orem et al., 2014). No information about organic constituents in flowback or produced waters from stimulation operations in California was identified.

5.1.3 Potential Release Pathways

There are a variety of activities associated with oil and gas development that can potentially release contaminants into surface water and groundwater aquifers. This section provides an overview of surface and subsurface release pathways, emphasizing those pathways and contaminants that are unique to the well stimulation treatments under consideration in this assessment.

In Section 5.1.3.1, surface pathways for water contamination are reviewed. Surface pathways include (1) surface spills and leaks; (2) the management and disposal of flowback/produced water; and (3) stormwater runoff. Produced water and flowback of well stimulation fluids are not managed separately in California, and most flowback/produced water is injected into Class II wells. However, current management practices in California allow for the disposal of flowback/produced water into unlined pits in some areas and reuse for agriculture without prior treatment. A more detailed assessment is needed of disposal and reuse practices to determine if they pose a risk to water resources. Furthermore, there is one documented case of the intentional release of flowback fluids in California, as well as other documented cases of the accidental release of chemicals associated with well stimulation in other states. Detailed assessments are not available as to whether these releases contaminated surface water and/or groundwater aquifers, but this is a potential pathway for surface and groundwater contamination. Furthermore, data on the water quality impacts of well stimulation are limited. Much of the available literature is focused on unconventional natural gas production; far less is available on shale oil production or about well stimulation technologies that may be used to access these resources, e.g., acid fracturing and matrix acidizing.

Section 5.1.3.2 discusses mechanisms for groundwater contamination via migration through subsurface pathways. Potential subsurface pathways include (1) natural and induced high-permeability pathways, the latter possibly created by hydraulically induced fractures propagating outside the target reservoirs; (2) engineered subsurface penetrations such as old wells that have not been properly abandoned and have been intersected by fracturing operations; and (3) direct introduction of contaminants via failing, degraded, or poorly constructed operating wells. Mechanisms of pathway formation, and leakage and transport through these existing, induced, and propagated failures are discussed and, for each pathway, the manner in which contamination may occur and the documentation of such an occurrence are provided, where available.

It is important to note that pathways must first exist (whether natural, preexisting, or induced by operations) before migration can occur. A summary of the literature on the subject, however, suggests that pathway formation via hydraulic fracturing itself is likely to be limited in vertical extent, and documented instances of contamination across the U.S. have been shown to be correlated with nearby operations, but not conclusively linked except in cases of direct injection of contaminants via operator error or well failure (US EPA, 2012).

Well stimulation notices filed to date with DOGGR indicate that much of the current and planned hydraulic fracturing operations in California occur at depths of less than 2000 feet below the ground surface, which is substantially shallower than in other states. Hydraulic fracturing at shallow depths poses a greater risk to water resources because of its proximity to groundwater and the potential for fractures to intersect nearby aquifers. In addition, migration of fracturing fluids via other permeable pathways is also possible. Some studies in other regions across the United States have found a correlation between the location of hydraulically-fractured production wells and elevated concentrations of methane (Osborn et al., 2011; Warner et al., 2012; Warner et al., 2013), arsenic, selenium, strontium (Fontenot et al., 2013) and, to a lesser extent, TDS (Warner et al., 2013). However, there is no consensus as to whether these are naturally occurring, or due to hydraulic fracturing, production well defects, abandoned wells, or a combination of mechanisms. More complete information about the location and quality of groundwater resources relative to the depth at which hydraulic fracturing is occurring in California would make it possible to identify inherently hazardous situations that could and should be avoided.

5.1.3.1 Surface Release Pathways

5.1.3.1.1 Surface Spills and Leaks of Fracturing Fluids

Oil and gas production involves the possibility of surface or groundwater contamination from spills and leaks. Well stimulation, however, raises additional concerns, due to the use of additional chemicals during the stimulation process, the generation of flowback fluids that contain these chemicals, and the increased transportation requirements to haul these materials to the well and disposal sites. Surface release of these chemicals and fluids can run off into surface water bodies and/or seep into groundwater aquifers. In this section, we describe concerns associated with well stimulation chemical usage at the surface and associated transportation concerns. The management and disposal of flowback and produced water are described in Section 5.1.3.1.2.

Well stimulation necessitates the transport and usage of chemicals (see Section 5.1.2). Chemicals needed for well stimulation are typically transported to the site by truck and are stored in the containers in which they were transported. Liquid chemicals and other additives are transported via hose to a blending unit, where they are mixed with the base fluid. Dry additives are poured by hand into the blending unit. This solution is then mixed with a proppant, if necessary, and pumped directly into the well (NYSDEC, 2011). Some of the fluids can be mixed and stored in preparation for the treatment; however, many are added only as the stimulation process is taking place (Cardno ENTRIX, 2012; King, 2012). This ensures that any chemical reactions occur at the appropriate time and in the proper location, and enables operators to ensure that there are no unused mixed fluids for storage or disposal.

These processes can result in chemical releases to the environment. Surface spills and leaks can occur during chemical or fluid transport, pre-stimulation mixing, or as the stimulation process is taking place. In addition, storage containers used for chemicals and well stimulation fluids can leak. These releases can result from tank ruptures, piping failures, blowouts, equipment failures and defects, overfills, fires, vandalism, accidents, or improper operations (NYSDEC, 2011). For example, in September 2009, two pipe failures and a hose rupture in Pennsylvania released 8,000 gallons of a liquid gel mixture during the hydraulic fracturing process, polluting a local creek and wetland (PA Department of Environmental Protection, 2009a; 2009b).

Data on hazardous materials spills are maintained by the California Emergency Management Agency (CEMA). According to California law, any *significant* release or threatened release of hazardous substances must be reported to CEMA.[3,4] According to these data, spills of chemicals typically used in well stimulation fluids, e.g., hydrochloric, hydrofluoric, and sulfuric acids, have occurred at oil and gas operations in California. For example, in February 2012, a storage tank containing 5,500 gallons of hydrochloric acid exploded in the Midway-Sunset Oil Field in Kern County, spreading the acid beyond a secondary containment wall. It is not possible, however, to discern whether stimulation was the intended purpose of this chemical, because acids are used to clear out drilling debris before the well is brought into production and are not uniquely associated with well stimulation.

5.1.3.1.2 Management and Disposal of Flowback/Produced Water

Produced water is generated by both conventional and unconventional oil and gas operations. Flowback fluids, by contrast, are unique to the well stimulation techniques under consideration in this report. In California, produced water and flowback water are managed together. As noted in a recent white paper from DOGGR, "when well stimulation occurs, most of the fluid used in the stimulation is pumped to the surface along with the produced water, making separation of the stimulation fluids from the produced water impossible. The stimulation fluid is then co-disposed with the produced water" (DOGGR, 2013). Given that these fluids are co-mingled, surface release pathways that may be associated with how these fluids are collectively managed are described. Although these fluids are sometimes referred to collectively as "wastewater," this report uses the term "flowback/produced water" in order to avoid confusion. A detailed assessment of the location and method of flowback/produced water disposal for specific wells was beyond the scope of this report. Consequently, we have conducted a review of the hazards associated with wastewater management and disposal in general.

[3] *Hazardous materials are defined as "any material that, because of its quantity, concentration, or physical or chemical characteristics, poses a significant present or potential hazard to human health and safety or to the environment if released into the workplace or the environment" (HSC 25501).*

[4] *Spills on highways must be reported to the California Highway Patrol, who then notifies CEMA.*

5.1.3.1.3 Onsite Storage and Transport of Flowback/Produced Water

Once at the surface, flowback/produced water may be temporarily stored at the well site in pits, embankments, or tanks, and then transported to a disposal site. Currently, oil and gas operations in California typically use closed loop systems to re-inject produced water onsite or transport these fluids through a pipeline network to a nearby disposal site. In some areas, however, trucks may be used. The onsite storage and transportation of flowback can result in the accidental releases of flowback fluids from spills and leaks that can reach surface water and groundwater aquifers. Spills or leaks can occur as a result of tank ruptures, piping and equipment failures, surface impoundment failures, overfills, vandalism, accidents (including vehicle collisions), fires, drilling, and production equipment defects, or improper operations (New York State Department of Environmental Conservation (NYSDEC), 2011).

There is evidence that surface spills of flowback/produced water affect surface water and groundwater aquifers. For example, in 2007, flowback fluids overflowed retention pits in Knox County, KY, releasing the fluid directly into Acorn Fork. The incident killed or displaced all fish, invertebrates, and other biota for months over a 2.7 km (1.7 miles) section of the creek. Papoulias and Velasco (2013) found that fish exposed to Acorn Creek waters showed signs of stress and higher incidence of gill lesions, consistent with exposure to low pH and toxic concentrations of heavy metals. Further, they found that the release degraded water quality sufficiently to have adverse impacts on the health and survival of *Chrosomus cumberlandensis* (Blackside Dace), a federally threatened species. Additionally, in an analysis of surface spills between July 2010 and July 2011 in Weld County, CO, Gross et al. (2013a) found that surface spills of produced water from the fracturing process or crude oil from fractured wells released BTEX to groundwater at levels that exceeded National Primary Drinking Water Maximum Contaminant Levels (MCLs) for each compound. In general, remediation efforts were sufficient to address these spills.

Data on flowback/produced water spills associated with oil and gas operations in California are reported to the DOGGR. According to Title 14, Section 1722 of the California Code of Regulations, “significant” water leaks must be promptly reported to the appropriate DOGGR district office (*California Code of Regulations*, n.d.). The reporting requirements are vague. There is no definition of what constitutes a significant leak, and all spills are likely not reported. According to the available data, between January 2009 and February 2014, 423 surface spills at oil and gas fields in California released nearly 2.8 million gallons of flowback/produced water, or an average of 6,500 gallons per incident. Of these, 34 spills released a total of 88,000 gallons of flowback/produced water into California waterways. Corrosion and sensor failures that cause tanks to overflow are the most common causes of these spills. As described previously, surface spills also have the potential to intercept groundwater aquifers, although lack of data on underlying groundwater quality before and after spills, and/or lack of data on the chemical composition of the spills, and varying conditions across the state limit the ability to evaluate general potential impacts in California.

5.1.3.1.4 Flowback/Produced Water Disposal

Problems with disposal of wastewaters recovered from well stimulation operations have been noted in some regions where flowback and produced waters ultimately ended up in fresh surface waters. For example, in the early development of the Marcellus region (2008-2009), flowback and produced waters were legally discharged into public twastewater treatment plants (WWTPs) that were not equipped to handle the high TDS content of these fluids, which resulted in increased loading of salts to Pennsylvania rivers (Brantley et al., 2014; Vidic et al., 2013; Kargbo et al., 2010). Bromide was also found to be a contaminant of concern due to the presence of carcinogenic disinfection byproducts in the WWTPs, formed from the reaction of elevated levels of bromine present in flowback/produced waters with organics (Ferrar et al., 2013). State regulators in Pennsylvania subsequently discouraged the practice of discharging waters recovered from fracturing operations into WWTPs, due to the many concerns about water quality degradation. There is some evidence that produced water is being discharged into municipal WWTPs but an assessment of this practice is beyond the scope of this report.

In California, disposal of flowback/produced water is typically done by one of three other methods: injection in Class II wells; reuse and recycling for oil and gas production or other beneficial uses; and percolation in unlined surface impoundments. Disposing of oil and gas flowback/produced water introduces surface release pathways that are unique to the disposal method under consideration. Each is described in more detail below.

Class II Wells

The majority of flowback/produced water from oil and gas operations in California is injected into Class II wells (Kiparsky and Hein, 2013). Injection wells are classified according to the location and type of fluid injected. According to the US EPA, Class II wells are used to inject brines and other fluids associated with oil and gas production. Class II well types include saltwater disposal wells, enhanced recovery wells (e.g., water flooding), and hydrocarbon storage wells (US EPA, 2014). Of the more than 30,000 Class II wells in California, about 95% are used for enhanced oil production and ~3% are used for disposal. More than 80% of Class II wells are located in District 4, representing Kern, Inyo, and Tulare Counties (Walker, 2011). There are a few documented cases of contamination associated with injection in Class II wells, as discussed in Section 5.2.3.2.2 of this report. However, as described below, groundwater contamination incidents in Ohio declined after injection in Class II wells replaced earthen pit disposal (Kell, 2011).

Reuse

While injection is the primary mechanism for managing flowback/produced water from oil and gas operations in California and in the rest of the United States (Guerra, Dahm, and Dundorf, 2011), flowback/produced water may also be reused for oil and gas operations (e.g., hydraulic fracturing) or other beneficial purposes (e.g., for irrigation, livestock watering, and some industrial uses). Produced water/flowback may be treated prior to reuse or simply blended with fresh water to bring the levels of TDS and other constituents down to an acceptable range (Veil, 2010).

As described previously, well stimulation notices filed with DOGGR since December 2013 indicate that oil and gas operators are currently using fresh water for well stimulation, and thus flowback/produced water is not being used for subsequent treatments (DOGGR, 2014). Flowback/produced water, however, is used to supplement irrigation water in California in some places. For example, in October 2011, the Central Valley Regional Water Quality Control Board (CVRWQCB) issued a general waiver to allow a discharger to pipe oilfield wastewater to an existing irrigation reservoir, where the water is mixed with groundwater (7% oilfield wastewater/93% groundwater) to irrigate 120 acres of citrus trees in Kern County (CVRWQCB 2011). Also in Kern County, produced water/flowback is treated and delivered by pipeline to a reservoir, where it is blended with surface water and groundwater. The blended water is then used to irrigate farmland throughout the Cawelo Water District service area during the irrigation season and is used to recharge groundwater during the nonirrigation season (CVRWQCB 2012). The Tulare Basin Plan notes that produced water "is used extensively to supplement agricultural irrigation supply in the Kern River sub-basin" (CVRWQCB 2004).

The use of produced water can potentially provide a new source of water supply, e.g., to farmers in San Joaquin Valley, where water resources are extremely constrained. The use of produced water comingled with flowback fluids, however, raises a set of unique concerns that are not yet well understood, especially when the mixture is not treated prior to reuse. In particular, the toxicity, persistence, and mobility of stimulation chemicals and constituents in the flowback/produced water, resulting from degradation of those chemicals and the interaction of the stimulation fluid with the formation, have not yet been evaluated.

Surface Impoundments

In some areas, wastewater from oil and gas operations is also disposed of via percolation in unlined surface impoundments—also sometimes referred to as sumps or pits. Sumps are primarily regulated by the state's nine Regional Water Quality Control Boards.[5] Each regional board is required to formulate and adopt water quality control plans, or basin plans, for all areas within the region. The plans establish water-quality objectives to protect beneficial uses and policies to implement the objectives.

Much of the state's oil production occurs within the jurisdiction of the CVRWQCB and is covered within the Tulare Basin Plan. The Tulare Basin Plan notes that hundreds of sumps are in use in the region to separate oil from wastewater and to dispose of oil

[5] *Local Air Districts also regulate some aspects of oilfield sumps, e.g., VOC emissions.*

field wastewater via percolation. Disposal of oil field wastewater in sumps overlying groundwater with existing and future beneficial uses is permitted if the electrical conductivity (EC) (represents salinity) of the wastewater is less than or equal to 1,000 micromhos per centimeter (μmhos/cm), and a maximum of 200 milligrams per liter (mg/L) chlorides, and 1 mg/L boron, with no other testing required for, or limits on, other contaminants. Oil field wastewater that exceeds these specified salinity limits may be discharged in "unlined sumps, stream channels, or surface water if the discharger successfully demonstrates to the Regional Water Board in a public hearing that the proposed discharge will not substantially affect water quality nor cause a violation of water quality objectives."

There is evidence of groundwater contamination associated with disposal of flowback/ produced water in unlined surface impoundments in other parts of the United States. Kell (2011) reviewed incidents of groundwater contamination caused by oil field activities in Texas between 1993 and 2008 and in Ohio between 1983 and 2007. Of the 211 incidents in Texas over the 16-year study period, more than 35% (or 75 incidents) were associated with waste management and disposal activities. Fifty-seven of these incidents were associated with produced water disposal pits, which were banned in 1969 and closed no later than 1984. Of the 185 groundwater contamination incidents in Ohio over the 25-year period, 5% (or 10 incidents) were associated with the failure of unlined pits. Like Texas, earthen pits are no longer in use in Ohio, and no incidents have been reported since the mid-1980s. Kell (2011) further notes that while there are cases of groundwater contamination incidents associated with Class II injection operations, "documented groundwater contamination incidents dropped significantly after subsurface injection replaced earthen pit disposal as the primary method of produced water management." While these studies and others linking unlined surface impoundments to groundwater contamination do not specify whether well stimulation fluids were the cause of the contamination, they are illustrative of the hazards of this disposal method.

A case in Pavillion, WY, raises additional concerns about the use of surface impoundments to contain flowback and produced water. The Pavillion gas field is located in central Wyoming in the Wind River Basin, the upper portion of which serves as the primary source of drinking water for the area. Oil and gas exploration began in the area in the 1950s and increased dramatically between 1997 and 2006. In 2008, domestic well owners began complaining about taste and odor problems, and residents believed these issues to be linked to nearby natural gas activities. In response to complaints from local residents, the US EPA initiated an investigation, collecting water samples from residential, stock, shallow monitoring, deep monitoring, and two municipal wells. According to the US EPA draft report, released in 2011, high concentrations of hydraulic fracturing chemicals found in shallow monitoring wells near surface pits "indicate that pits represent a source of shallow ground water contamination in the area" (Digiulio et al., 2011). At least 33 surface pits were used to store/dispose of drilling muds, flowback, and produced water in the area. These findings were not contested by Encana Oil and Gas, the company responsible for the natural gas wells, or other the stakeholders (Folger, Tiemann, and Bearden, 2012). There was, however, considerable controversy about US EPA's other findings, i.e., the presence of hydraulic fracturing chemicals in deep wells and thermogenic methane in monitoring and domestic wells, as discussed in Section 5.1.3.2.3.

Illegal Discharges

Illegal waste discharges may result in the release of contaminants to surface water and groundwater aquifers. Kiparsky and Hein (2013) note that lax enforcement of regulations and insignificant penalties can incentivize illegal dumping when the punishments are less costly than proper disposal or reuse. In July 2013, for example, the Central Valley Regional Water Quality Board (CVRWQB) issued a fine to Vintage Production California LLC in the amount of $60,000 for periodically discharging saline water, formation fluids, and hydraulic fracturing fluid to an unlined sump for 12 days. The sump was located next to a newly drilled oil well near the City of Shafter in Kern County. Discharge of high-salinity water into an unlined sump is prohibited in areas with good-quality groundwater, and the Board's Executive Officer noted that there is concern "that similar discharges may have occurred elsewhere throughout the Central Valley" (CVRQCB, 2013). In response, the CVRQCB issued an Order in November 2013 seeking information from oil and gas operators about the discharge of drilling fluids and well completion fluids since January 2012. This information will help the Board identify the characteristics and volumes of waste discharged to land and to evaluate the potential impacts or threatened impacts to water quality posed by the discharge of these fluids to land.

5.1.3.1.5 Stormwater Runoff, Including Floods

Stormwater runoff carries substances that can be harmful to water quality and ecosystem health from the land surface into local waterways. While runoff is a natural occurrence, disturbances to the land surface can increase its timing, volume, and composition. For example, a one-acre construction site with no runoff controls can contribute 35-45 tons of sediment each year, compared to less than 1 ton of sediment per year from forest land (US EPA, 2007a).

There is some evidence that oil and gas operations exacerbate stormwater runoff impacts to water resources. However, it is not clear the degree to which impacts are more generally associated with oil and gas activities or specific to the well stimulation treatments under consideration in this report. Olmstead et al. (2013) found that shale gas operations increased total suspended solid (TSS) concentrations in downstream surface water bodies. The particular mechanism by which this occurred, e.g., precipitation events or initial construction activities, could not be determined.

While limited studies that examine runoff associated with well stimulation activities are available, there are likely to be some impacts that are unique to well stimulation treatments. Specifically, runoff from well pads can pick up spilled chemicals used during well stimulation as well as residual process and flowback fluids that may be located onsite. Additionally, precipitation events and flooding may damage storage and disposal sites or cause them to overflow, washing these materials into waterways. For example, major flooding in 2013 damaged oil and gas operations in northeast Colorado, spilling an estimated 48,000 gallons of oil and 43,000 gallons of produced water (COGCC, 2013). Furthermore, the additional truck traffic associated with transporting materials, equipment, and flowback/produced water can increase wear and erosion on local roads and/or result in the development of new paved and unpaved roads with impacts to surface runoff.

Stormwater discharge is regulated by state and local governments. The National Pollution Discharge Elimination System (NPDES) program regulates stormwater runoff at the federal level. States can receive primacy to administer their own permitting program and can implement stronger requirements, if desired.[6] At the federal level, oil and gas operations have been afforded special protections and are exempt from some provisions in the Clean Water Act. Consequently, oil and gas operators are not required to obtain a stormwater permit unless, over the course of operation, the facility generates stormwater discharge containing a reportable quantity of oil or hazardous substances or if the facility violates a water-quality standard (40 CFR 122.26(c)(1)(iii)).[7] In 2005, the definition of oil and gas exploration and production was broadened to include construction and related activities, although regulations still require well pads larger than one acre to apply for an NPDES stormwater permit. A 2005 study on the surface water impacts of natural gas drilling noted the difficulty of monitoring and suggested that few facilities were monitoring in a way that would allow them to determine whether a NPDES permit was required (US EPA, 2007b).

5.1.3.2 Subsurface Release Pathways

The consideration of potential subsurface contamination pathways is organized into three parts. The first part regards the formation of high permeability pathways by hydraulic fracturing, which regards the extent and permeability of induced fractures and the possibility of connection to overlying aquifers. By definition, a hydrocarbon reservoir is likely to be capped or bounded by low-permeability layers. Thus, contaminant migration requires a pathway, whether natural or induced. The second part addresses issues with wells (drilling, completions, and failures) that may create opportunities for hydrocarbons or fracturing fluids to enter groundwater aquifers. The third part discusses transport processes that could occur within permeable pathways, if they exist, and the evidence for such migration.

[6] *California's Regional Water Quality Control Boards have authority to issue stormwater permits, where they are required.*

[7] *This requirement will not be met by sediment discharges alone.*

In a recent progress report concerning ongoing US EPA studies of the potential impacts of hydraulic fracturing on groundwater resources (US EPA, 2012), the authors state clearly that data concerning hydraulic fracture communication outside of target reservoirs is currently very limited, with few peer-reviewed studies in place to confirm or deny the possibility of contaminant transport due to fracturing operations. In response, the US EPA is performing case studies at multiple locations in the US where contamination of water resources has been alleged in areas with historic oil and gas operation (Texas, Pennsylvania, Colorado, North Dakota). Water sampling from wells and surface water sampling and monitoring is under way to understand the impact of prior drilling, current drilling, and fracturing work, and in the case of North Dakota, the aftermath of a well blowout accident. In conjunction with field studies, modeling studies are being performed to assess the mechanics of fracture propagation and flow processes governing various gas and fluid leakage scenarios (US EPA, 2012). In addition, the US EPA report describes out a number of hypothetical leakage scenarios, but these scenarios have yet to be evaluated. Beyond a literature review, the data and modeling results for the transport studies have not been released to the public or to the general scientific community. Therefore, an independent examination of the literature is required.

5.1.3.2.1 Formation of High Permeability Pathways

One possible concern about hydraulic fracturing operations, in particular those using high-volume injections, is the degree to which induced fractures may extend beyond the target formation to connect to higher permeable aquifers, or to natural or man-made pathways such as faults, natural fractures, or abandoned wells. The current state of understanding about the formation of such permeable pathways due to hydraulic fracturing is surrounded by some controversy, due to concerns about groundwater contamination above hydraulically fractured reservoirs. The bulk of previous published work in the area of hydraulic fracturing has been in the form of data and literature reviews, geomechanical modeling studies, and analysis of existing microseismic data to assess fracture formation and propagation, all with a focus on the creation of permeability for oil and gas production. Basic theoretical and geomechanical work spans decades, with early work on fracture propagation such as that by Hubbert and Willis (1972) and Nordgren (1972), and work on fracture width evolution by Perkins and Kern (1961). The review by Adachi et al. (2007) summarizes much of the early work with a focus on numerical simulation. However, the latest and most relevant published work, directly addressing concerns about possible leakage of gas and fracturing fluids, has occurred since 2011, with multiple papers creating a vigorous debate about the nature and extent of artificial fractures and the processes creating them.

Myers (2012) discusses transport in porous media and in fractures and pathways driven by both natural advection and fracturing-related pressure increases within porous media in the Marcellus shale. Using a simplified flow simulation, the work determined that pressure increases are localized, subside in a year or less, and that the injection-stimulated systems could re-equilibrate in pressure with a year or less. The simulation operates under the assumption of the existence of out-of-formation fracturing or connectivity to permeable faults, which, if present, could drive fluids or gas into overlying formations on decadal scales or more quickly. Supporting this assumption is a published letter by Warner at al. (2012b) stating that microseismic monitoring indicates fracture propagation is more likely in the vertical direction, increasing the possibility of fractures reaching upward toward more permeable formations, or into pathways that are inferred in a previously published geochemical study. However, Myers' (2012) simulation work lacks key coupled hydrological processes, particularly the properties of unsaturated shales (Vidic et al., 2013), and did not include coupled geomechanical modeling.

A later study by Flewelling and coauthors (Flewelling et al., 2013) developed a novel relationship between injected fluid volumes and maximum possible fracture height, calibrated via a dataset of the observed extent of microseismicity during well stimulation operations. The study capped potential vertical fracture propagation at 600 m (2,000 ft) or less. Additional limitations created by injected volumes, combined with the observation that shallow formations are more likely to fracture horizontally rather than vertically, led to the authors' lack of concern about the possibility of fracturing at depths greater than 150 m (500 ft) intercepting shallow groundwater resources, thus disputing Myers' underlying assumptions. In a similar vein, work by Fisher and Warpinski (2012) and a review by Davies and collaborators (Davies et al., 2012) attempted to demonstrate that fracture propagation is inherently limited.

Specifically, Fisher and Warpinski (2012) compare fracture extent as mapped by microseismic data to water well depths for active shale production regions in the Barnett, Woodford, Eagle Ford, and Marcellus formations. They find that vertical fracture extent for deep hydraulic fracturing operations does not bring the fractures in close contact with shallow aquifers, and uses mineback data (artificial fractures excavated and examined *in situ*) and experience to posit fracture growth-limiting mechanisms that would lead to well-contained fractured reservoirs. This work also indicates the likelihood of fractures in shallower formations (<1,200 m or 3,900 ft) having a greater horizontal component (due to decreasing vertical normal stress at shallower depths), with the consequence of reduced likelihood of extended vertical propagation toward shallow aquifers.

Davies et al. (2012) also argues that the height of artificial fractures is limited. The study reviews data on both natural and stimulated fractures, comparing the mechanisms proposed for formation of natural "pipes" and "chimneys" (clusters of large fractures/faults extending hundreds or thousands of meters, typical in sub-seafloor environments) with fractures artificially created for stimulation purposes, or as a result of production accidents or blowouts. Using a variety of datasets, they plotted frequency versus fracture height for natural and artificial fractures, estimating the probabilities that induced fractures could reach specific heights. They find that the majority of artificial fractures (with data focused on the Barnett Shale) range from <100 m (330 ft) to ~600 m (2,000 ft) in height, with approximately a 1% probability of a fracture exceeding 350 m (1,100 ft). They correlate this data with previous studies suggesting that artificial fracture growth is also limited by fracturing fluid volume, similar to the findings of Flewelling et al. (2013). The limited scale of induced fractures is compared with larger, more extensive natural fractures systems, or "pipes," that are created by processes that involve much larger fluid volumes, overpressures, longer time frames, and other factors such as erosion or collapse of surrounding strata. Thus, a minimum separation of 600 m (2,000 ft) between shale reservoirs and overlying groundwater resources is suggested for thigh-volume fracturing operations, although local geology must always be evaluated.

In California, an industry study (Cardno ENTRIX, 2012) evaluated the effects of ten years of hydraulic fracturing and gas production from a Los Angeles Basin oil and gas field. Microseismic monitoring indicates that fractures were contained within the reservoir zone, extending to within no more than 2,350 m (7,700 ft) of the base of the fresh-water zone. However, microseismic inversion depends on an initial velocity model, and thus the characterization of hydraulic fractures via this method can result in some inaccuracy or ambiguity (Johnston and Shrallow, 2011).

Recent studies include coupled flow-geomechanical modeling to increase the fundamental understanding of how fractures form and propagate during injection and pressurization (Kim and Moridis, 2012). A coupled flow-geomechanical simulator (Kim and Moridis, 2013) has been developed using the established TOUGH+ subsurface flow and transport simulator (Moridis and Freeman, 2013) and validated against analytical solutions for poromechanical effects, static fractures, and fracture propagation. The initial work looked at fracture development versus injection rate, and found that shear failure can limit the extent of fracture propagation. Later work using full 3D domains suggests possible inconsistencies between fracture volume and the volume of injected water, resulting from the difference between the propagation of the water front (a flow process) and the propagation of the fractures themselves (a geomechanical process), with the net result that injected fluid volume may underestimate fracturing extent. However, the work also suggests inherent physical limitations to the extent of fracture propagation, for example, the presence of overlying confining formations may slow or stop fracture growth, thus containing fractures within the shale reservoir (Kim et al., 2014).

Application of this work to California requires an understanding of the depth of hydraulic fracturing operations relative to groundwater aquifers. Data regarding hydraulic fracturing depth are available only from the well stimulation notices. Data regarding the true vertical depth of some wells hydraulically fractured are available from FracFocus, and the measured depths of some wells hydraulically fractured are available in DOGGR's GIS well data files. The proportion of wells at various depth-levels is shown on Figure 5-4. A large fraction of the depths are less than 610 m (2,000 ft). The distribution of depths for the hydraulic fracturing intervals is necessarily shallower than the well depth distributions.

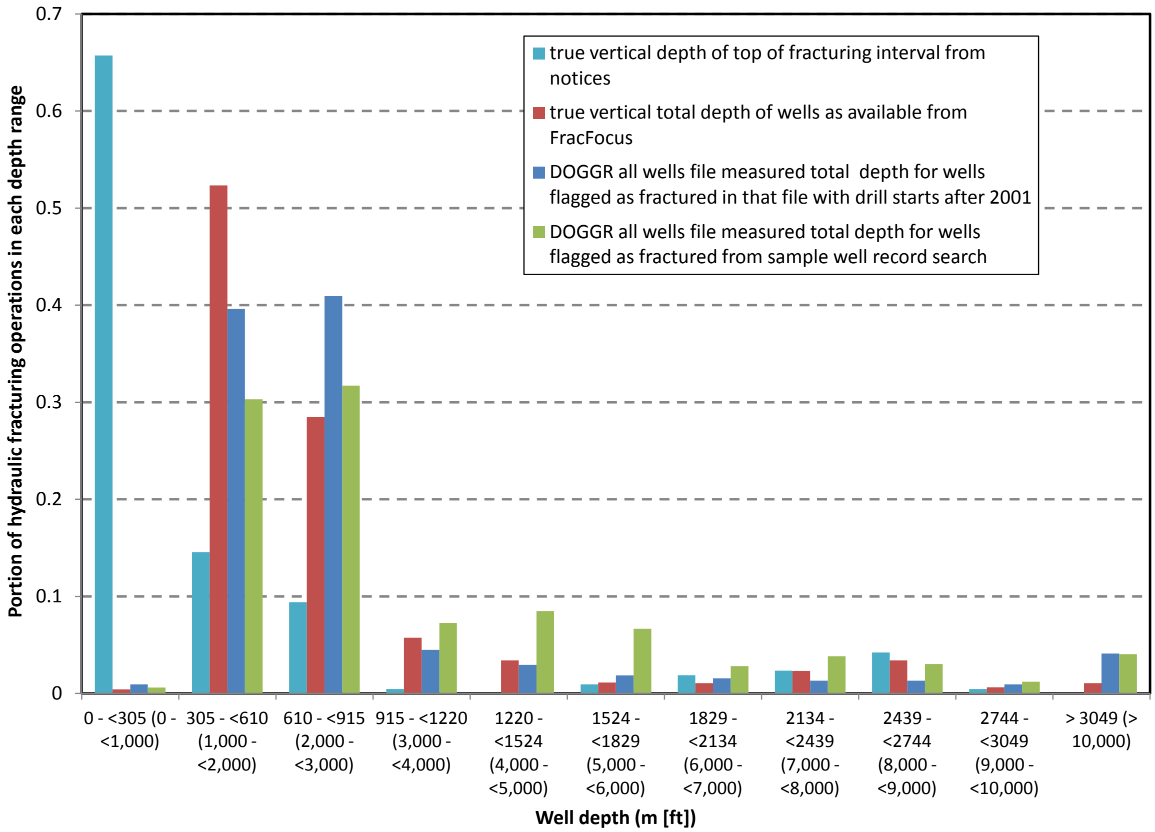

Figure 5-4. Portion of hydraulic fracturing operations vs. depth range (DOGGR data is only for wells drilled after 2001).

Figure 5-5 indicates the depth of the shallowest well hydraulically fractured in each field. The shallowest well depth in a number of fields is less than 610 m (2,000 ft) and in even more fields is less than 1,220 m (4,000 ft). This suggests that the separation between some fracturing intervals and groundwater is less than the suggested 600 m separation based on Flewelling et al. (2013) and Davies et al. (2012), which would imply that the likelihood of propagation of fractures into groundwater aquifers may be higher in California. However, it is important to remember that this depth-separation suggestion was based on high-volume hydraulic fracturing conducted in deep shale reservoirs, meaning the subsurface stress conditions as well as the WST operations are quite different from the situation in California. For example, fractures may primarily propagate horizontally at shallower depth and, due to relatively smaller fracturing fluid volumes, the height distribution of fractures in California may also be smaller than that used as the basis for the depth-separation suggestion (Fisher and Warpinski, 2012).

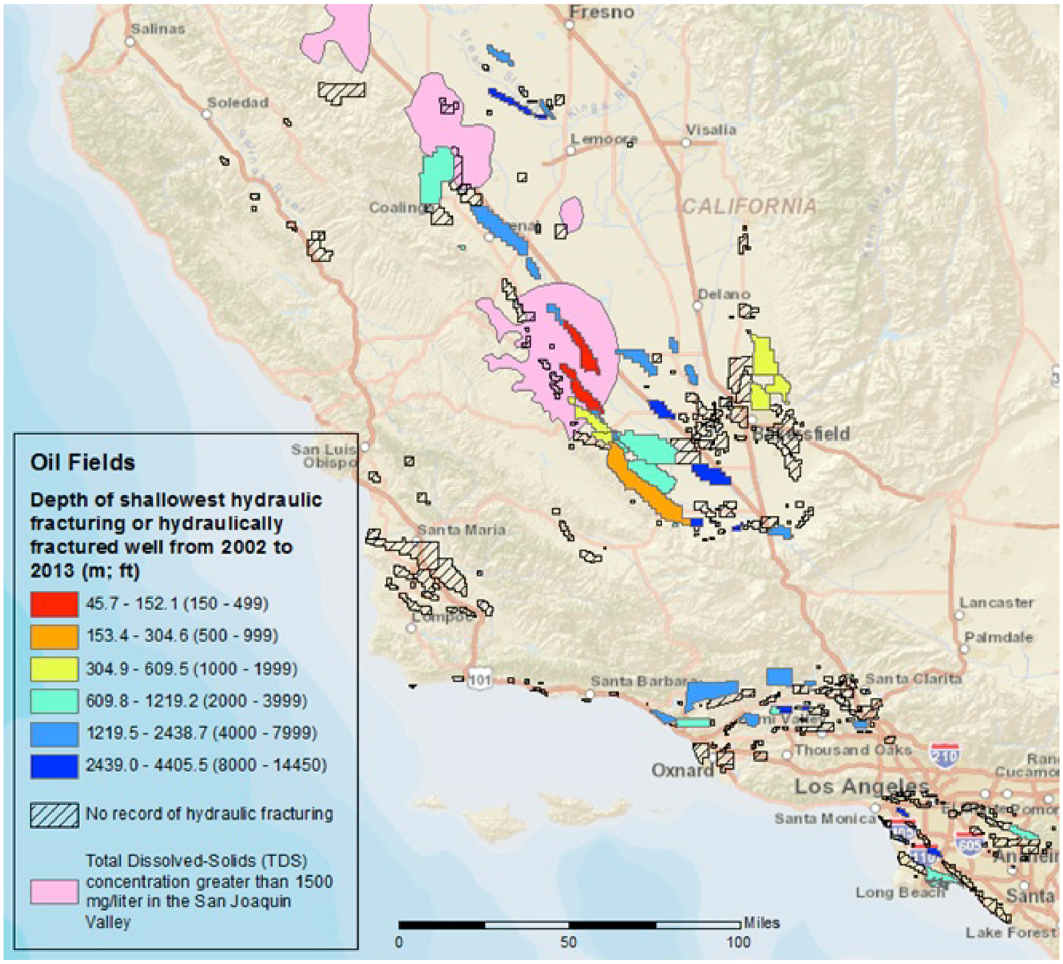

Figure 5-5. A map showing the shallowest hydraulic fracturing depth from the well stimulation notices or hydraulically fractured well depth in each field (measured depth from DOGGR for wells drilled after 2001 or true vertical depth from FracFocus). Pink areas show regions in the San Joaquin Valley where the shallowest groundwater has total dissolved solids above California's short-term secondary maximum contaminant level for drinking water of 1,500 mg/L. Note oil fields colored orange and yellow in the San Joaquin Valley, indicating shallow hydraulic fracturing, located in areas with better groundwater quality. Data from DOGGR 2014(a), DOGGR 2014(b), FracFocus (2013), and Bertoldi et al. (1991).

Also highlighted in Figure 5-5 are regions with relatively poor groundwater resources containing > 1,500 mg/L of total dissolved solids (TDS), from Bertoldi et al. (1991). California has established secondary maximum contaminant levels (SMCLs) for various constituents in drinking water. The SMCLs are based on odor and taste rather than health thresholds. The recommended SMCL for TDS is 500 mg/L and the upper SMCL is 1,000 mg/L. There is also a short-term SMCL of 1,500 mg/L allowed in rare circumstances

(State Water Resources Control Board, 2010). The proportion of fracturing operations vs. depths are re-plotted in Figure 5-6, for operations that occur in regions with groundwater containing < 1,500 mg/L of total dissolved solids (TDS). Comparison of Figure 5-4 with Figure 5-6 suggests that the majority of shallow operations occur in regions where groundwater is of poor quality. However, the well records search results indicate up to 3% of hydraulically fractured wells are shallower than 610 m (2,000 ft) in regions with good to reasonable groundwater quality. This percentage equates to tens of shallow wells being fractured per year given estimates of the number of wells hydraulically fractured. It is unknown if the fracturing intervals in these wells are sufficiently deep to preclude induced fractures propagating into potable groundwater. Because most of the wells are vertical, the fractured interval must be shallower than the total vertical well depth. Page (1973) indicates the base of water with < 2,000 mg/L TDS is up to a thousand meters (thousands of feet) deep in some areas. So the separation between hydraulically fractured well intervals and groundwater may be much less than 600 m (2,000 ft) at tens of wells per year. For instance the shallowest hydraulically fractured well in the Kern River Field, which is shown in yellow just north of Bakersfield in Figure 5-5, is less than 610 m (2,000 ft) deep. Groundwater with less 2,000 mg/L TDS is implied to that depth and deeper by Page (1973). A portion of the produced water from this field is used for irrigation (Coburn and Gillespie, 2002). While the water is treated, it is not known if the treatment would remove all the fracturing fluid constituents that might be of concern.

Further, it is not clear that 1,500 mg/L TDS is the threshold for groundwater to be considered with regard to protection from entry of well stimulation fluids. DOGGR has historically protected groundwater with < 3,000 mg/L and the federal definition of underground sources of drinking water requiring protection is < 10,000 mg/L (Walker 2011). The number of hydraulically fractured intervals within a few hundred meters of groundwater with concentrations less than 20,000 mg/L is necessarily larger than the tens of wells per year mentioned above with regard to groundwater with < 1,500 mg/L TDS. The potential for hydraulic fractures to intercept groundwater resources in cases of shallow well stimulation warrants more careful investigation and monitoring.

For fracturing intervals more than 600 m (2,000 ft) from overlying aquifers, our review of the existing literature suggests that creation of permeable pathways connected to these aquifers solely through hydraulic fracturing operations seems unlikely in most cases, considering the relative separation of the formations and the difficulty of creating very long, extensive fractures or fracture networks. A review by Jackson et al. (2013b) comes to the same conclusion, noting that no "out-of-zone" fracturing has been documented, while at the same time reiterating that studies are limited and data are sparse. However, fracturing that creates connectivity to preexisting pathways, whether naturally formed (pre-existing permeable fractures or faults) or artificial (abandoned, degraded, poorly constructed, or failing wells) cannot be discounted, nor can we ignore the possibility of human error in the drilling, completion, and stimulation processes. Of particular concern is the creation of connections to abandoned or degraded wells, since the existence of such pre-existing pathways is likely in regions targeted for WST operations, and such wells provide a clear case of a potentially permeable feature that connects to shallow formations. Additional research is required to better quantify this hazard (Jackson et al., 2013b).

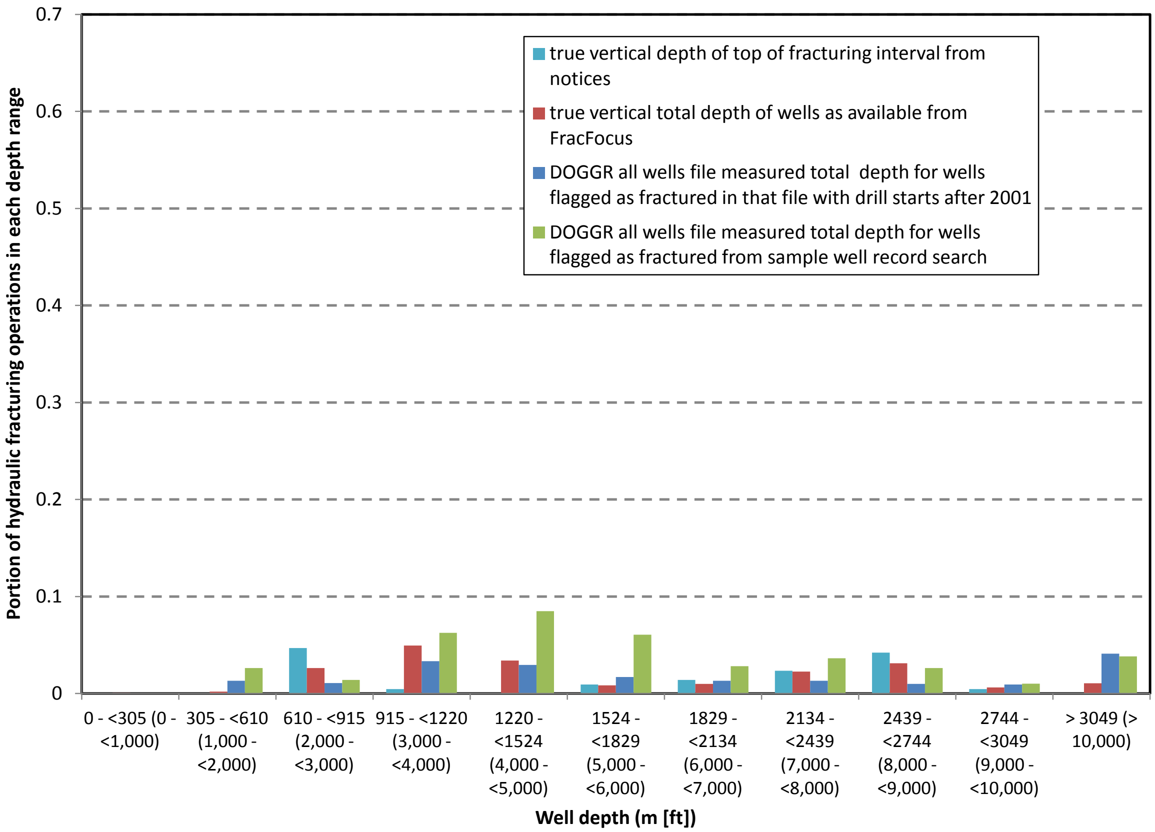

Figure 5-6. Portion of hydraulic fracturing operations vs. depth range, for locations where the overlying groundwater has TDS of less than 1,500 mg/L (DOGGR data is only for wells drilled after 2001).

5.1.3.2.2 Leakage from Wells During Injection, Production, or Stimulation

The possibility of operating wells serving as leakage pathways for subsurface migration has been known for a long time. Papers by Harrison (1983; 1985) indicate that overpressured annuli are a likely mechanism for contamination of groundwater with produced gas or other formation fluids, even for wells using a surface casing to protect shallow aquifers and particularly if the surface casing does not extend to a sufficient depth below the aquifer. Failures in well barriers (cement and casing strings) may allow intrusion of gas and fluids from producing formations below the casing shoe or shallower gas and fluid-bearing formations intersected by the wellbore to lower-pressure annuli, resulting in annular gas flow or sustained casing pressure (SCP) and a pathway for gas migration to the surface (Brufatto et al., 2003; Watson and Bachu 2009) of two overlying aquifers. Multiple factors over the operating life of a well may lead to barrier failure including improper or inadequate cementing, poor mud displacement, and fractures in the cement due to hydraulic and mechanic stresses during construction or associated with tectonic activity and subsidence events; radial cracking of the cement due to thermal and pressure fluctuations in the casings during stimulation and production; and general degradation of the well structure due to age (Bonnet and Parfitis, 1996; Dusseault et al., 2000; Brufatto et al., 2003; Watson and Bachu 2009; Carey et al., 2012). Corrosive subsurface environments (e.g., H_2S, CO_2) also pose a threat to cement and casings throughout the life of a well and after abandonment/plugging, particularly if cement is already impaired (Brufatto et al., 2003; Chilingar and Endres 2004;). The most important mechanism leading to gas and fluid migration, however, is poor well construction or exposed or uncemented casing (Watson and Bachu, 2009).

Unconventional wells may be subject to greater stresses due to mechanical stresses induced in high pressure stimulation and lateral drilling practices. The casing and cement of the vertical section of the well, from the surface casing down to the production zone, is subject to hydraulic and mechanical stress during drilling and operations (see Section 2.2.1.2). For wells used in hydraulic fracturing operations, the high levels of fluid pressure imposed also need to be taken into account during casing selection and well design. During hydraulic fracturing operations, there has been concern that the expansion and contraction of the steel casing during the multiple stages of high-pressure injection may result in radial fracture and/or shear failure at the steel-concrete or concrete-rock interfaces (Carey et al., 2012). This expansion and contraction of the casing, not typically present in conventional oil and gas operations, could lead to separation between the casing and the cement. These processes could create gaps or channels that would serve as conduits between the various strata through which the well penetrates. Current cementing technology may not be sufficient to control for such defects. However, with current practice, the fracturing fluid is pumped down a tubing string within the innermost casing, such that the casing and surrounding cement are not experiencing the high injection pressure associated with the fracturing operation. Monitoring of the annulus between the tubing string and production can identify problems or failures that lead to high casing pressure.

Watson and Bachu (2009) also noted that deviated wellbores, defined as “any well with total depth greater than true vertical depth”, show a higher occurrence of sustained casing pressure and gas migration than vertical wells, likely due to centralization and cementing challenges increasing the likelihood of gaps, bonding problems, or thin regions in the cement. Creation of such annular permeable pathways may create connectivity to higher formations. In a review of the regulatory record, Vidic et al. (2013) noted a 3.4% rate of cement and casing problems in Pennsylvania wells based on filed notices of violation. Pennsylvania inspection records, however, show a large number of wells with indications of cement/casing impairments for which violations were never noted suggesting that the actual rate of occurrence could be higher than that reported by Vidic et al. (2013).

Human error during the well-completion and hydraulic fracturing process must also be considered. A 2011 incident in Alberta, Canada (Energy Resources Conservation Board (ERCB), 2012) involved inadvertent fracturing of an overlying formation and injection of fluids into water-bearing strata below an aquifer. Misreading of well fluid pressures resulted in the perforation gun being fired at 136 m depth (446 ft) rather than the specified 1,486 m (4,875 ft), with subsequent pressurization creating a fractured interval above the base of groundwater protection. Immediate flowback of fracturing fluids recovered most of the injected volume, and monitoring wells were installed at the perforation depth and at 81 m (266 ft) in an overlying sandstone aquifer at a distance of 50 m (164 ft). A hydraulic connection between the fractured interval and the overlying aquifer was not observed, and a surface gas release nearby was not linked to the injected fluids. The ERCB finding states that the incident presented “insignificant” risk to drinking water resources, but criticized the onsite crew’s risk management, noting there were multiple opportunities to recognize abnormal well behavior before the misplaced perforation.

Well integrity must also be evaluated for Class II deep injection wells, which have become the method of choice for the disposal of flowback and produced fluids. The regulatory review by Kell (2011) reviewed incidents of contamination associated with deep injection in Ohio and Texas. The injection process was implicated in six contamination incidents in Texas (none in Ohio); however, properly permitted Class II injection wells still have a significantly better record of protecting groundwater resources than older methods of earthen pit disposal (see Section 5.2.3.1.3). In California, a 2011 report studied the over-24,000 active and 6,900 inactive injection wells in the state and found that, while procedures were in place to protect fresh-water resources, other water resources (with higher levels of dissolved components, but not considered saline) may be at risk due to deficiencies in required well-construction practices (Walker, 2011). Zonal isolation of saline formations via cement placement is not mandated, nor is the isolation of hydrocarbon-containing zones, thus leading to potential migration of fluids into overlying groundwater resources. Depending on the target formation, injection pressures must be monitored and maintained at or below levels appropriate for the geology. In addition, operators are required to perform mechanical integrity tests on Class II wells every five years.

An earlier US Government Accountability Office report (US GAO, 1989) regarding Class II wells across the United States found that, although the total extent of drinking water contamination was unknown, several cases of contamination had been documented. In one-third of the cases, the contamination was caused by communication between injection wells and improperly plugged (abandoned) oil and gas wells nearby, causing injected brines to migrate vertically through the abandoned wellbores. Injection wells built and operating prior to 1976 are exempt from Underground Injection Control (IUC) program permitting requirements (40 CFR 144.31, 146.24) which mandate an area search for abandoned wells within a quarter mile of a new proposed injection wellbores. The GAO report notes that 70% of the injection wells studied were grandfathered and as such the presence of nearby degraded wells was discovered only after contamination had occurred.

Although the work was not specific to hydraulic fracturing, the hazards of degraded wells and well failure are highlighted in a review paper by Chilingar and Endres (2004). They document multiple incidents in which oilfield gas reached the surface through degraded, abandoned, and leaking wellbores. The paper highlights a 1985 incident where well corrosion at shallow depths led to casing failure of a producing well and the migration of gas via faults and other pathways, creating a gas pocket in a permeable collecting zone below a populated area in Los Angeles. Methane accumulated underneath a department store until overpressurization drove gas into the building's basement, resulting in an explosion. A vent well was used to reduce the hazard, but failure of the vent well resulted in another release of gas in 1989, although this was detected before another explosion could occur. While these incidents are not related directly to fracturing operations, they show that cement and casing impairments in modern wells and inadequately cemented abandoned wells may provide pathways for vertical migration of formation gas and fluids.

5.1.3.2.3 Mechanisms of Leakage via Transport Through Subsurface Pathways

To reiterate, contaminant migration requires a pathway, whether natural or induced. If such pathways have been created through hydraulic fracturing operations, whether the result is a direct fracturing into overlying aquifers or a connection to a preexisting pathway for fluid flow outside of the reservoir, reservoir and fracturing fluids may migrate through the subsurface. Data concerning such contamination mechanisms are currently very limited, with few peer-reviewed studies in place and ongoing US EPA assessments as yet unpublished (US EPA, 2012; Jackson et al., 2013b). Transport through preexisting pathways has occurred in conventional oil and gas operations (see previous section), but whether hydraulic fracturing is likely to enhance the problem remains to be determined. In a manner similar to the issue of fracturing and fracture propagation, the core literature consists of a few groups of competing and contentious studies, none of which provides direct evidence of fracturing leading to contaminating groundwater.

Although mechanisms for transport through fractures and faults have been proposed, few conclusions can yet be made about the conditions under which liquid or gas release can occur. Overburden thickness, formation permeabilities, production strategies (assuming no drilling or casing incidents), and other site-specific factors may all regulate the probability of contaminant migration. The study by Myers (2012), mentioned previously, attempted to model flow through artificially created pathways, but did so using a highly simplified flow and hydrologic model (Vidic, 2013). A more recent modeling study by Kissinger et al. (2013) performs porous-media modeling of liquid and gas migration through specific, previously characterized fractured systems. The study, although limited to one set of geological models (and thus to one set of subsurface geometries), does highlight factors that may increase or decrease the risk of contamination. Fluid migration resulting from a two-week fracturing-related overpressure is shown to drive fracturing fluids only a limited distance from the fractured reservoir, even when high-permeability pathways are assumed. Long-term tracer transport and transport of methane to overlying aquifers are shown to be a function of pathway porosity, permeability, and irreducible gas saturation, but only under the assumption of a continuous permeable pathway from the reservoir to the aquifer. Factors such as production strategy or ranges of overburden thickness are not evaluated, but Kissinger et al. (2013) suggest that transport of liquids, fracturing fluids, or gas is not an inevitable outcome of fracturing into connected pathways, and that further evaluation of a range of geological systems is warranted.

Several studies have noted the presence of methane in groundwaters near hydraulic fracturing operations, and have tried to determine the source and pathways for methane migration based on the chemical and isotopic composition of the gas. Methane found in groundwater can either be formed as a result of thermogenic processes at depth or microbial processes in shallower horizons. Biogenic methane typically consists of pure methane and carbon dioxide, whereas thermogenic methane, such as that found in shale gas, will also contain higher-chain hydrocarbons (ethane, propane, butane, and pentane). Biogenic and thermogenic methane are also isotopically different, with the former having a lower ratio of carbon-13 to carbon-12 isotope (more negative δC13 values (-64‰)) than the latter (-50‰) (Osborn et al., 2011a). Earlier studies (Révész et al., 2010) show the presence of both biogenic and thermogenic methane, as well as some ethane, in well water near Marcellus gas production, but variation over time (i.e., before and after production activities commenced) had not been established. However, correlations have been shown to exist between gas reservoir locations and gas production activity and the presence of methane in groundwater and surface water.

The most recent controversies began with the research of Osborn et al. (2011a) that performed geochemical studies of sampled water from 60 drinking-water wells in a gas producing region of northeastern Pennsylvania. They noted that methane concentrations in wells increased with increasing proximity to gas wells, compared to neighboring wells away from production activity. Isotopic ratios of the sampled gas, as well as the presence of longer-chain hydrocarbons, indicated a thermogenic source for the gas, along with matching the geochemistry of gas from nearby production wells. However, evidence of contamination from brines or fracturing fluids was not found in the sampling. This result highlights an important issue, specifically that liquid and gas transport do not necessarily occur together, and that gas migration and liquid migration within the subsurface may occur at different rates and timescales. The absence of brine migration led to the conclusion that methane transport via liquid migration is unlikely to be the source, but rather that leakage and migration of gas through any number of possible permeable pathways (well casings, artificial fractures, or enlarged fractures due to hydraulic fracturing) could have provided the pathway for the contamination. The paper notes the existence of a preexisting fracture network within the overlying formation, combined with numerous undocumented, uncased abandoned wells that could serve as conduits for gas migration. In response, letters by Davies (2011) and Schon (2011) state that leakage through well casings is a better explanation than any fracturing-related process, referencing PA Department of Environmental Protection reports that document specific casing-failure incidents. The responders also promote the hypothesis that the high methane concentrations may be pre-existing, noting that such processes are already documented and well-understood (Dyck and Dunn, 1986) for oil and gas producing formations, and that a lack of evidence for fracturing fluids in the contaminated water supports ongoing natural processes. Further discussion, in a letter to *PNAS* by R. B. Jackson and colleagues, and a follow-up paper (Osborn et al., 2011b; Jackson et al., 2011; 2013a), counters those conclusions, pointing out that methane contamination has indeed occurred, but that natural migration pathways or abandoned wells are the less likely scenarios as, although abandoned wells are common in Pennsylvania, few abandoned wells are known in the area of this particular study. Jackson, however, agrees that casing leakage from poor well construction is a plausible mechanism, while still maintaining that, since it is neither proven nor disproven, hydraulic fracturing operations could be involved in the subsurface processes. A key conclusion of this series of studies is that there is a strong correlation between gas well location and the appearance of stray gas contamination.

Another sampling study by Jackson et al. (2013a) found ethane and propane, as well as methane, in water wells near Marcellus production locations, and also noted isotopic compositions that suggest a "Marcellus-like" origin for the thermogenic component of the methane. The concentration of methane was again correlated most strongly to distance from production activities, as was the ratio of longer-chain hydrocarbons to methane. The authors propose leakage caused by well casing and cementation problems as the most plausible mechanism, noting the number of violations recorded for well-construction issues in nearby production operations. In contrast, another isotopic study by Molofsky and colleagues (Molofsky et al., 2013) states that the isotopic ratios of methane found in Pennsylvania wells are more consistent with samples of shallower Upper Devonian gas rather than Marcellus formation gas, thus casting doubt on the source of the dissolved gas and the existence of connecting pathways.

Geochemical evidence for natural migration of fluids has been published by Warner et al. (2012b), who revisited northeastern Pennsylvania and collected new water samples for comparison to older data published in the 1980s. The study indicated that elevated salinity levels in the region may predate shale gas production in the area, and that geochemical signatures matching that of the Marcellus fluids led to the conclusion that natural permeable pathways may have already existed between the shale and overlying formations. These natural permeability pathways could create contamination hazards if oil and gas operations occur near the zones of enhanced connectivity. In response, Engelder (2012), disputes this possibility, noting that recent drilling data for hundreds of wells suggests the saturation of water in the pore space is typically in the range of 13% to 33%, which is near or below the irreducible water saturation for the shale. Such low saturations would result in capillary binding of the water, restriction of brine migration, and the possible sequestration of fracturing fluids left in the formation, as the aqueous phase would be drawn into the pore space of the shale and rendered immobile. This capillary seal would be expected to trap both gas and liquids within the Marcellus, and this concept is supported by differences in the isotopic signature of Marcellus gas and gas that exists in the overlying formations (see also Molofsky et al., 2013). The previously referenced work by Flewelling et al. (2013) also addresses this issue of formation isolation, pointing out that the occurrence of permeable pathways overlying significant hydrocarbon accumulations is inherently contradictory. Therefore, their work finds that some mechanism that activates pre-existing, but impermeable features or creates new pathways is necessary to allow liquid and gas migration, while noting the potential constraints to vertical fracture propagation mentioned in the previous section. A further response by Warner et al. (2012b) maintains that there are insufficient data to support the capillary binding hypothesis, and that recent production data counter the notion that the shale has little mobile brine—in fact, the opposite has been true for some production wells—but also concedes that mechanisms for rapid brine transport are neither indicated nor understood.

For Marcellus production in Pennsylvania, an extensive review by Brantley et al. (2014) assesses both the scientific literature and the regulatory record, in an attempt to establish a relationship between production activities, known production problems and violations, and the existence of subsurface migration pathways. The paper states up front that fracturing fluids or flowback have never been conclusively tied to a water-contamination incident, and that distinguishing common tracers is challenging, because background concentrations are spatially and temporally variable. The true processes are clouded by lack of information about drilling and production incidents, unreleased water quality data, the sparseness of available data, and lack of knowledge of pre-existing contaminants. Attempts to perform mathematical risk assessments of contamination through all mechanisms have primarily highlighted the lack of knowledge (Rozell and Reaven, 2012) with envelopes of uncertainty spanning orders of magnitude, although when risk is formally assessed, the consequences of wastewater disposal (i.e., potentially large spills) generated more concern than that for subsurface leakage and migration. However, over a thousand complaints about water quality issues have been recorded in areas near Marcellus gas production. The review delves into the regulatory record and finds numerous Notices of Violation, particularly for well-construction problems, in the regions of Pennsylvania where contamination is suspected. While postglacial processes and bedrock fracturing may make the gas-producing regions of the state more susceptible to gas and fluid migration even without stimulative fracturing, there is also the presence of thousands of pre-Marcellus wells, with 200,000 dating from before formal record-keeping began and 100,000 that are essentially unknown (noted in a companion study by Vidic et al., 2013). These potential hazards were highlighted by a 2012 incident in which fracturing operations intercepted an old offset well, resulting in a blowout and the release of gas, but not of fracturing or formation fluids, through the compromised abandoned well. This is consistent with the previous conclusion that care must be taken to avoid situations where hydraulic fracturing creates connectivity to abandoned or degraded wells.

The literature, particularly peer-reviewed literature, is heavily weighted toward regions where public concern over new stimulation technologies has been strongest—currently, regions overlying the Marcellus. In California, there is a history of oil and gas production (Chilingar and Endres, 2005), including the use of hydraulic fracturing technologies, but at present, there is no comprehensive source of information on well stimulation activities (Section 3.2.2). Recently, an industry study (Cardno ENTRIX, 2012) reviewed ten years of hydraulic fracturing and gas production from the Inglewood field, a Los Angeles Basin oil and gas field. The Inglewood field is located in a populated area, and underlies a fresh-water formation that, while not used for drinking water and while likely not connected to nearby drinking water resources, is still regulated and monitored for water quality. Microseismic monitoring indicates that fractures were contained within the hydrocarbon reservoir zone, extending to within no more than 2,350 m (7,700 ft) of the base of the fresh-water zone. The 2011-2012 study showed no impacts to groundwater quality, either through migration of fracturing fluids, formation fluids, or methane gas, even though the formation includes faults and fractures connecting shallow formations to deeper formations. No evidence was found of well-casing failure, when wells have been constructed to industry standards, and thus no direct contamination occurred via stimulation or production activities. However, the review of Chilingar and Endres (2005) documents a history of gas-transport incidents associated with other conventional oil and gas production in the L.A. Basin. The paper documents multiple cases of gas leakage from active oil fields and natural gas storage fields in the Los Angeles Basin and elsewhere, with the most common issue being gas migration through faulted and fractured rocks penetrated by abandoned and leaking wellbores, many of which predate modern well-casing practice and are undocumented or hidden by more recent urban development. These features led to several documented cases of methane from oil and gas operations traveling and reaching near-surface formations or reaching the surface—between leaking wells and near-surface formations, through near-surface faults, and between pressurized gas-storage reservoirs and abandoned wells. While stimulation technologies are not implicated in these events (with the possible exception of water-flooding procedures creating increased pressures that drive transport), they illustrate the real possibility of flow through permeable pathways if such pathways, natural and/or induced, exist and are allowed to communicate with hydrocarbon reservoirs.

It is clear that methane appears in groundwater near hydraulic fracturing operations for shale gas operations, but studies have essentially established only correlation, not causation of leakage pathways. Thus, additional research is required to better quantify this hazard (Jackson et al., 2013b), with a focus on (1) establishing background values of various contaminants, (2) field experiments and monitoring, and (3) better modeling studies to elucidate possible transport mechanisms. In this regard, additional studies are under way to identify tracer materials that could be useful for the monitoring of the migration of fracturing fluids in the subsurface, as well as fracturing fluid-shale interactions. For instance, nanoparticles are currently tested, which could be added as nonreactive tracers to fracturing fluids in the future (Maguire-Boyle et al., 2014). The analysis of strontium (Sr) isotope ratios has been proposed as a useful approach to evaluate fluid-rock interactions (Chapman et al., 2012). In either case, these tracers could provide relevant tools for elucidating open questions regarding potential contaminant pathways related to well stimulation applications in the future.

5.1.4 Case Studies of Surface and Groundwater Contamination

This section examines evidence of the contamination of surface water and groundwater aquifers from well stimulation treatments, and discusses the findings from groundwater quality studies that have been conducted at sites located near well stimulation operations. No reports of water contamination resulting from well stimulation in California were found, although only one study for a site in California was identified in the Inglewood oil field. While limited information is currently available in peer-reviewed literature, two studies provide evidence of surface water contamination. Reports from state agencies provide additional evidence of contamination. Based on the limited data that are available, it appears that groundwater quality near hydraulic fracturing has not been significantly impacted due to well stimulation treatments, although two reported instances of potential groundwater contamination by hydraulic fracturing fluid were identified. Neither of these studies was documented in peer-reviewed literature, and the findings from one (at Pavillion, Wyoming) have been questioned in subsequent studies conducted at the site. Elevated levels of some contaminants that could have been brought up from the target formation, such as methane, TDS, and some trace metals, have been observed in the groundwater near some hydraulic fracturing sites in the United States. However, the sources of these contaminants are in dispute (as described in Section 5.1 .3), and cannot be directly linked to well stimulation treatments.

The potential impacts of well stimulation on surface water and groundwater quality are ultimately dependent on reliable and current baseline data describing water characteristics prior to drilling operations (or if not possible, for representative background sites), and on comprehensive monitoring conducted during and after well stimulation. It should be noted that water quality data near well stimulation sites are sparse, and an absence of studies (or data) neither supports nor refutes evidence of problems. Proper pre-drilling baseline and post-stimulation monitoring data are essential to evaluating the impacts of well stimulation on nearby groundwater. Efforts should be made to collect such data in the future, and the findings from water quality monitoring should be included in reporting requirements for operators.

5.1.4.1 Surface

There are no reports of surface water contamination associated with well stimulation in California, although there are documented cases in other parts of the U.S. For example, in 2007, flowback fluids overflowed retention pits in Knox County, KY, killing or displacing all fish, invertebrates, and other biota for months over a 2.7 km section of the creek. Papoulias and Velasco (2013) found that fish exposed to Acorn Creek waters showed signs of stress and higher incidence of gill lesions, consistent with exposure to low pH and toxic concentrations of heavy metals.

In another study, (Kassotis, Tillit, Davis, Hormann, and Nagel, 2013) examined the presence of known or suspected endocrine-disrupting chemicals used for well stimulation in surface and ground water samples in drilling-dense areas of Garifeld County, Colorado. Nineteen surface water samples were collected from five distinct sites that contained from 43 to 136 natural gas wells within one mile and had a spill or incident related to natural gas drilling processes within the past six years. Additional samples were collected from the Colorado River, which captures drainage from this region, and from nearby reference sites. The study found that most water samples exhibited greater estrogenic, antiestrogenic, and/or antiandrogenic activities than water samples from nearby references sites with limited or no drilling activity.

Additional surface water contamination incidents have been reported, although these are not captured in peer-reviewed studies. Some of these incidents were reported to the appropriate local and/or state agencies, while others may not have been reported. For example, in 2009, a fish kill event in an unnamed tributary to Brush Run in Hopewell Township was reported to the Pennsylvania Department of Environmental Protection. Responders found an overland pipe transporting flowback fluid had failed, releasing about 250 barrels into the tributary (Pennsylvania Department of Environmental Protection, 2010a). Also in Pennsylvania, a wastewater pit overflowed its embankment, polluting a tributary of Dunkle Run. While the company cleaned up the spill once it was discovered, it failed to report the incident to the Pennsylvania Department of Environmental Protection (Pennsylvania Department of Environmental Protection, 2010b).

5.1.4.2 Subsurface

5.1.4.2.1 General Findings from Groundwater Quality Studies in the United States

Typically, monitoring studies sample the natural groundwater in wells in the vicinity of well stimulation operations and draw conclusions based on a comparison of pre-drilling baseline data (if available) and post-drilling monitoring. If pre-drilling baseline data were not available, some studies collected groundwater samples at nearby background sites that had comparable geology and geochemistry, but were relatively unimpacted by well stimulation operations. The list of parameters measured in the groundwater quality studies varied according to the topic under investigation, and included subsets of the following:

- Acidity (pH), alkalinity
- Dissolved gases: Methane, carbon dioxide, oxygen
- General water quality parameters: Total Dissolved Solids (TDS), Total Suspended Solids (TSS), specific conductance, turbidity, Total Organic Carbon (TOC), Dissolved Organic Carbon (DOC)
- Major cations: Sodium, potassium, magnesium, calcium, ammonium
- Major anions: Chloride, bromide, nitrate, nitrite, phosphate, fluoride, cyanide

- Trace metals: Ag (silver), Al (aluminium), As (arsenic), Ba (barium), Be (beryllium), B (boron), Cd (cadmium), Cr (chromium), Co (cobalt), Cu (copper), Fe (iron), Li (lithium), Mn (manganese), Hg (mercury), Mo (molybdenum), Ni (nickel), Pb (lead), Se (selenium), Sb (antimony), Sn (tin), Sr (strontium), Ti (titanium), Th (thorium), U (uranium), Zn (zinc).

- NORM (Naturally occurring radioactive material): Gross alpha, gross beta, ^{226}Ra, ^{228}Ra, Radon, Uranium

- Organics: Oil and grease, Volatile organic compounds (VOCs), Semi-volatile Organic Compounsd (SVOCs), pesticides, Polychlorinated Biphenyls (PCBs)

- Stable isotopes: dC13 (carbon), dO18 (oxygen), dD (hydrogen)

- Selected constituents of injection/fracturing fluids

A limited number of studies have investigated groundwater quality in the vicinity of hydraulic fracturing in several regions, including the Marcellus shale, PA (e.g. Boyer et al., 2011, Brantley et al., 2014 and references therein), the Fayetteville shale, AK (Warner et al., 2013), and one study in California in the Inglewood oil field (Cardno ENTRIX 2012). Most studies comparing baseline trends to post-stimulation measurements did not determine any statistically significant changes in the water quality of nearby groundwater wells resulting from well stimulation operations. Studies reporting elevated levels of some contaminants that were detected in groundwater situated near fracturing operations are discussed below. However, none of the studies could directly link the elevated levels of measured contaminants to the use of well stimulation technologies.

An extensive review of groundwater-contamination claims and existing data can be found in the report of Kell (2011) for the Ground Water Protection Council. The report focuses on Ohio and Texas groundwater-investigation findings from 1983 through 2008, and notes that the literature provides no conclusive documentation of groundwater contamination resulting from the hydraulic fracturing process itself. The study area and time period included development of 16,000 horizontal shale gas wells with multistage fracturing operations in Texas, and one horizontal shale gas well in Ohio. However, the report notes that there is evidence of groundwater contamination due to improper storage of flowback and produced fluids in surface containment pits (as discussed in Section 5.1.3.1.4), a practice that has mostly been replaced by disposal via Class II injection wells that have a significantly better record of protecting groundwater resources than earthen pit disposal. Sections 5.1.3.2.1 and 5.1.3.2.3 discuss the report’s findings on abandoned wells being a leakage pathway. The report concludes that, although no documented links have been found implicating fracturing operations in contamination incidents, a regulatory focus on activities that could be linked to contamination is critical, along with documentation of hydraulic fracturing operations such that regulators can determine which processes put groundwater at risk.

5.1.4.2.2 Detection of Well Stimulation Fluids in Groundwater

Very limited information is currently available in peer-reviewed literature about the detection of fracturing-fluid additives in groundwater. Two reported instances of potential groundwater contamination due to subsurface leakage of hydraulic fracturing fluid within the United States were identified, neither of which has been documented in a peer-reviewed publication (Brantley et al., 2014, Vidic et al., 2013). The first study is a US EPA investigation in Pavillion, WY, where surface storage and disposal of flowback/produced waters was implicated in contamination of shallow surface water (as discussed in Section 5.1.3.1.4). Initial results published in a draft report (DiGiulio et al., 2011) suggested that groundwater wells had been contaminated with various fracturing-fluid chemicals, as well as methane, via flow from the stimulated reservoir to groundwater. However, a follow-up study by the USGS involving resampling of the wells could not confirm these findings (Wright et al., 2012). The US EPA is no longer working on this study, but the State of Wyoming is continuing to investigate these data, with a report due in September 2014.

The second reported incident of contamination is based on a U.S. EPA study focusing on operations in Ripley, WA. In this case, a gel used as a constituent in fracturing fluids was reported to have contaminated a local water well located <330 m (1,000 ft) from a vertical gas well (US EPA, 1987). Contaminant transport could have either occurred through four abandoned wells located near the vertical gas well during the fracturing process, or by contamination from the flush fluid (not used in hydraulic fracturing) used to remove loose rock cuttings prior to cementing (Brantley et al., 2014).

Kassotis et al. (2013) evaluated the potential of elevated activities of endocrine disrupting chemicals in surface and groundwater systems close to natural gas extraction sites utilizing hydraulic fracturing. Surface and groundwater samples were collected in a drilling-dense region in Garfield County, CO, and analyzed for estrogen- and androgen-receptor activities using reporter gene assays in human cell lines. Based on a comparison with reference control sites, the authors concluded that these data suggest elevated endocrine-disrupting chemical activity in surface and groundwaters close to unconventional natural gas drilling operations. However, potential contaminant pathways were not discussed in this publication and are currently unknown.

5.1.4.2.3 Detection of Direct Contaminants from Target Formations in Groundwater

A number of studies have monitored the groundwater in the vicinity of hydraulic fracturing operations for contaminants other than those present in fracturing fluids, such as methane, TDS (including chloride and bromide), heavy metals, NORMs, and organics. None of these studies definitively traced the source of or migration pathway for these contaminants to application of hydraulic fracturing, as discussed in Section 5.1.3.2.3. The contaminants could have either been naturally present in the formation or could have migrated along alternate pathways, unrelated to well stimulation, into the groundwater.

Elevated methane in groundwater near hydraulic fracturing operations was a particular focus of many of the studies. Leakage of fugitive methane into groundwater wells situated near hydraulic fracturing sites is a public concern due to fire and explosion hazard. The US Department of the Interior recommends a warning at dissolved methane levels of 10 mg/L (ppm) and requires action at concentrations greater than 28 mg/L (ppm).

Regions where shale gas production is feasible tend to have naturally high methane concentrations, and have been sites for previous natural gas extraction activities. For example, concentrations as high as 45 to 68.5 mg/L (ppm) have been observed in New York, West Virginia, and Pennsylvania groundwaters (Vidic et al., 2013). A survey of methane concentrations in Southern California, which was carried out following the Ross Department Store explosion, identified eight high-risk areas where methane could pose a safety problem (Geoscience Analytical, 1986). These include the Salt Lake Oil field in Los Angeles, the Newport Oil field, the Santa Fe Springs Oil field; the Rideout Heights area of the Whittier Oil Field; the Los Angeles City Oil field; the Brea-Olinda Oil field; the Summerland Oil field; and the Huntington Beach Oil field. Comprehensive baseline measurements collected before drilling can help determine whether high methane levels detected in wells, post-production, are a result of well stimulation.

As extensively discussed in Section 5.1.3.2.3, some studies have found high concentrations of thermogenic methane in drinking-water wells in Pennysylvania, particularly those within a 1 km radius of hydraulic fracturing operations (Osborne et al., 2011; Jackson et al., 2013a), although the source of the methane detected in those studies is under debate. Another study measuring pre-drilling and post-stimulation methane concentrations in 48 water wells in Pennsylvania located within 760 m (2,500 ft) of Marcellus shale gas wells found no differences in methane levels before and after drilling, except in one well where drilling had been completed nearby (Boyer et al., 2011).

Several studies also focused on measurements of TDS in groundwater, particularly due to the high levels of TDS present in flowback and produced fluids recovered from some shale plays. As discussed in Section 5.1.3.2.3, high salinities detected in some shallow Marcellus groundwater wells could have resulted from migration of brines from deeper formations through natural pathways that were unrelated to hydraulic fracturing (Warner, 2012). A study of 100 groundwater wells located in aquifers overlying the Barnett shale found that that TDS concentrations exceeded the US EPA Maximum Contaminant Level (MCL) of 500 mg/L in 50 out of 91 samples located within 3 km of gas wells, and that the maximum values of TDS near the wells were over three times higher than the maximum value from background reference wells unimpacted by fracturing. However, the study was conducted in aquifers that naturally have high levels of TDS. TDS concentrations in 7 out of 9 samples collected from the background wells also exceeded the MCL, and the average TDS values near the hydraulic fracturing sites were similar to historical data for the region (Fontenot et al., 2013). Monitoring for TDS in the Inglewood oil field near Los Angeles (Cardno Entrix, 2012) found no significant differences in pre-drilling and post-stimulation TDS values; TDS values ranged from 510 to 2,500 mg/L in shallow wells and 1,400 to 3,900 mg/L in deep wells.

Fontenot et al. (2013) also reported that the heavy metals arsenic, barium, selenium, and strontium were found to be present at much higher levels in groundwater wells located < 3 km from production wells in the Barnett shale, when compared to background or historical concentrations. Although the trace elements of concern were known to be naturally present in the formation at low levels, the authors suggest further investigation to determine if the high concentrations detected in the groundwater were a result of fracturing operations. The study did not investigate the complex biogeochemistry that can lead to mobilization of trace elements such as arsenic, but suggested some possible mechanisms by which the development of wells for oil and gas (and indirectly) well stimulation could cause release of trace metals into the groundwater. These include lowering of the water table due to excessive water withdrawals, and mechanical disturbances due to drilling that could loosen iron oxides (potentially mobilizing arsenic and selenium) or sulfate/carbonate scales (potentially mobilizing barium and strontium) from the casings of private wells.

The only study that has identified trace-element concentrations in groundwaters near well stimulation operations in California was conducted in the Inglewood oil field (Cardno Entrix, 2012). Arsenic was the only trace element that exceeded drinking water standards in that study. However, arsenic is naturally present at high levels in Southern California, and concentrations were high in the monitoring wells even before drilling. Information on background levels of trace metals in California is available as part of the USGS Groundwater Ambient Monitoring and Assessment (GAMA) program. High levels of some trace elements such as arsenic, boron, molybdenum, chromium, and selenium have been measured in shallow groundwaters in several regions in California (e.g. USGS, 2006; USGS, 2009). These data should be considered in future investigations that attempt to determine the impact of well stimulation on groundwater quality in California.

In general, there have been no reports of high levels of NORMs found in groundwater near well stimulation operations. It should also be noted that uranium concentrations in some California groundwaters have historically been high. For example, high levels of uranium, frequently exceeding US EPA MCLs, have been noted in the Central Valley and are correlated with high bicarbonate concentrations in the groundwater (Jurgens et al., 2005). Radium levels in California groundwaters are typically low (Ruberu et al., 2005).

A couple of studies have reported measuring some organic constituents in groundwaters near well stimulation operations. These include the US EPA investigation in Pavillion, WY, where glycols and alcohols were detected (DiGuilio et al., 2011) and a study in the Barnett shale, where methanol and ethanol were detected in 29% of samples in private drinking-water wells (Fontenot et al., 2013). However, the presence of organics could not be linked to fracturing operations in either case.

Several articles note that there is a clear need for future studies and the monitoring of multiple water-quality parameters, to ensure that groundwater resources near well stimulation operations are not impacted by well stimulation and related activities

(Brantley et al., 2014; Jackson, 2013; Bibby et al., 2013; Vidic et al., 2013). Future conclusions regarding potential impacts of well stimulation on groundwater quality are ultimately dependent on reliable and current baseline data describing groundwater characteristics prior to drilling operations, or if not possible, for representative background sites. The USGS is currently conducting a broad, US-wide water-quality study, which includes the following objectives (besides others): (1) determine current baseline concentrations of major ions in surface water and groundwater in areas of unconventional oil and gas production; (2) evaluate potential changes in water quality over time; and (3) identify spatial and temporal data gaps where further information is needed to evaluate existing water quality and water-quality trends (Susong et al., 2012). Specifically for California, the USGS has published a series of reports describing groundwater quality for a number of CA basins, such as the Monterey Bay and Salinas Valley basins, and the Los Angeles, Southern San Joaquin and Central Coast basins, as part of the GAMA Priority Basin Project (e.g., USGS, 2011). However, these studies may not be formulated to provide baseline data specific to the question of groundwater contamination due to well stimulation.

5.1.4.2.4 Groundwater Monitoring for Well Stimulation Operations in California

DOGGR requires the reporting of constituents in injected and recovered fluids including chemical composition and radiological information (DOGGR Interim Well Stimulation Reporting Requirements Instructions, 2013). In addition, all well stimulation notices submitted to DOGGR as of January 1, 2014, must include a monitoring plan, regardless of the specific groundwater quality in close proximity to the wells (Vincent Agusiegbe, DOGGR, personal communication). Most of the notices submitted and received in December and subsequently approved did not have monitoring plans because groundwater in those oil fields was exempt from beneficial use.

A monitoring plan, which was approved as part of a well stimulation permit in the Rose oil field as of January 2014, included pre-stimulation and semi-annual measurements of temperature, pH, electrical conductivity, BTEX, and TDS at locations where the property owner requested it for up to two years after the well stimulation was concluded. While the operators claim that this list of constituents provides an appropriate evaluation of the potential impacts of well stimulation, many published scientific studies suggest the need for more comprehensive monitoring plans (e.g., Jackson, 2013; Bibby et al., 2013; Vidic et al., 2013). This would include measuring a larger set of parameters (such as those mentioned above) that are based upon an evaluation of regional geology and typical groundwater and formation water chemistries at more locations than just those requested by the landowners (both monitoring wells near the oil fields and residential wells can be used to collect the data).

5.2 Potential Impacts to Air Quality and Climate

The following sections address air quality and climate impacts. Although air quality and climate impacts are treated separately below, certain aspects of the analysis and literature review in each section may inform discussion in the other section. For example, many processes that lead to emissions of local pollutants also lead to emissions of greenhouse gases. Diesel fuel combustion leads to emissions of nitrogen oxides (NO_x) and particulate matter (PM) but also carbon dioxide. Processes in WST that lead to emissions of volatile organic compounds (VOC) also frequently lead to emissions of methane (methane is a potent greenhouse gas). Some methods to control emissions for air quality also control greenhouse gas emissions, for example reduced emission, or "green," completions control both VOC emissions and methane emissions. In both the Air Quality and Climate Impacts sections comparison are made between "bottom-up" inventories, in which all known sources of emissions are summed to generate a total emission estimate, to "top-down" emission estimates, in which ambient measurements of pollutants or greenhouse gases are used to characterize likely emissions.

5.2.1 Air Quality

This section evaluates the pollutant emission and potential air quality impacts related to well stimulation in California. Most well stimulation activity in California occurs in the San Joaquin Valley, an air basin that is designated as a non-attainment area for ozone and particulate matter (PM) standards, thus marginal changes to air quality may be relevant.

Ideally, one would connect emissions of pollutants directly to air quality impacts through the use of an air quality model that could account for dispersion and chemical transformation of the emitted pollutants as they travel through an air basin. Use of an air quality model is out of the scope of this report. Air quality impacts are instead evaluated by comparing estimates of emissions related to WST to estimates of total emissions from oil and gas processes or other sectors. If emissions are much smaller than emissions from other sectors we assume the air quality impacts are small.

Well stimulation activities that lead to emissions include the use of diesel engines, flaring or venting of gas and the volatilization of chemicals in flowback water. This section presents separate emission estimates for each of the above activities and describes how those emissions compare to emissions from other relevant sectors.

It is reported in this section that emissions of NO_x and $PM_{2.5}$ in California from diesel equipment used for WST, both for on-road trucks and off-road equipment such as pumps, produces negligible emissions compared to other related sectors. Furthermore, emissions from both diesel off-road equipment and on-road trucks could be controlled if the use of diesel engines with NO_x and $PM_{2.5}$ exhaust controls were mandated.

Emissions from flaring in California are uncertain because of variability in flare combustion conditions and to a lack of information regarding the frequency of flare-use during WST operations. However, current California Air Resource Board inventories of pollutant emissions from all flaring suggest that flares as a whole emit less than 0.1% all VOCs and are not a major regional air quality hazard.

Emissions from venting of gases during completion and from volatilization of flowback water constituents have not been measured in California but might be bracketed. The California Air Resources Board (CARB) has conducted a "bottom-up" VOC emission inventory by adding up all known sources of emissions. It is unknown whether these sources included emissions from WST-related produced or flowback water. However, the sum of the emissions in the inventory matches well with "top-down" measurements taken from the air in the San Joaquin Valley (Gentner et al., 2014). This agreement between "bottom-up" and "top-down" estimates of VOC emissions from oil and gas production indicates that California's inventory probably included all major sources. The CARB inventory suggests that venting and VOC emissions from flowback water are small compared to other production related sources of VOC emissions. Emissions from venting during WST could be controlled by requiring reduced emission ("green") completions. Requiring tighter vapor controls on temporary tanks that hold the flowback water could control emissions from fluids produced during WST.

Oil and gas production operations are a major (~10%) source of total anthropogenic ozone precursor emissions in the San Joaquin Valley. Although the marginal emissions from WST alone are small, the potential increase to VOC emissions due to additional oil and gas production activities enabled by WST could potentially impact ozone air quality in the San Joaquin Valley.

5.2.1.1 Air Quality Overview

Most of the WST activity in California takes place in the San Joaquin Valley, a region of California that is designated as a non-attainment area for ozone and particulate matter (PM) standards. Since the region is not currently meeting national standards, any marginal increase to emissions can present a challenge for local regulators in that they will have to find an equal source of emission reductions from a different sector just to maintain the current pollution levels, let alone reduce total emissions so that air quality measurements fall below the standards.

The air quality of a region is characterized by measurements of specific pollutants, including $PM_{2.5}$ and ozone, from central monitors in that region. Before the pollutants emitted from WST encounter and are measured by the monitors, they are dispersed by wind and may undergo chemical transformation in the atmosphere. The manner in which the same emissions will affect air quality will differ depending on the meteorological conditions and the other pollution already present in the atmosphere (the chemical transformations depend on total pollution levels).

There are several methods one might employ to evaluate how WST emissions impact air quality. One could try to determine the impact of WST emissions through analyzing air quality measurements, comparing air quality on days with high WST activity to days with low WST activity. However, the day-to-day variation in WST emissions is unknown and the variability in meteorology and atmospheric chemistry between days would likely overwhelm any signal that might exist otherwise. Instead of depending only on measurements, air quality models are often used to describe how pollutants are dispersed through the atmosphere and chemically transformed. The models connect the pollutant emissions to their air quality impacts. It is out of the scope of this report to develop air quality modeling of WST related emissions. Instead, we can compare WST related emissions to emissions from other sectors. If the magnitude of emissions is much smaller than other known sectors we can assume the air quality impacts from those emissions are much smaller as well. In this report, emissions from WST activities are evaluated by comparison to the sum of total emissions from the oil and gas production and processing sector.

To estimate emissions we would, ideally, have direct measurements of emissions from a representative sample of WST activities in California. There are, however, very few measurements of emissions from WST in California, let alone a representative sample, as WST emissions can vary over wells, reservoirs, operator practices, control technologies and other factors. An additional challenge is that independent scientists have limited access to well pads to conduct emission measurements. In fact, emissions from oil and gas production activities are the subject of numerous scientific studies, reviewed below, and continue to be a source of uncertainty regarding total environmental impacts from oil and gas production.

Given these limitations we estimate emissions from WST in California using a “bottom-up” approach, meaning we break WST into a series of processes and estimate emissions for each process separately. The estimates for each process are based on estimates of activity (for example gallons of fuel-used) and emissions per activity (for example pollutant emissions per gallon of fuel). In some cases, information regarding an activity is limited, and we can develop only a qualitative emissions estimate based on available literature. We compare emissions estimates to the CARB emission inventory, which lists emission estimates by air basin for thousands of separate source-types, including more than 200 oil and gas source-types. Although the total emission estimates from the CARB inventory are publically available, CARB releases relatively little detail on the methodology used to create these emission estimates. In the sections below we note when we believe our comparisons to CARB’s emission inventory may be uncertain due to a lack of knowledge of particular details of CARB’s underlying methodology for inventory development.

The separate WST processes we evaluate include: (1) Bringing supplies to the well pad, including fluids; (2) Pumping the fluid into the well; (3) Venting of gases from the well during WST or completion; (4) Flaring of gases produced during WST or completion; and (5) Evaporation of chemicals from liquids produced during WST and completion.

Each of those practices releases a different set of pollutants. The diesel equipment used to pump the fluid into the well and the diesel trucks used to bring supplies to the well are primarily a concern because of nitrogen oxides (NO_x) and particulate matter (PM) emissions. Increased chronic exposure to particulate matter is associated with increased rates of premature mortality. NO_x is of concern for multiple reasons. NO_x emissions can lead to ozone formation, ozone, a key constituent of photochemical smog, is an irritant and also associated with negative short and long term health impacts. NO_x can also undergo chemical transformation in the atmosphere and condense to a solid form adding to the total $PM_{2.5}$ burden.

Venting of gases from the well and evaporation of chemicals from flowback or produced liquids are of concern due to emissions of VOCs. There are thousands of potential chemicals that fall under the VOC category. Some VOCs are carcinogens or endocrine disruptors and directly hazardous to humans. Many VOCs can react in the atmosphere to increase ozone formation. Some VOCs are transformed to form PM. If gases are flared, instead of vented, then most, but not all, of the VOCs are burned. The combustion during flaring may cause $PM_{2.5}$ and NO_x emissions.

Below we review studies of air quality near oil and gas production operations across the US. This literature review provides context for the range of concerns related to air quality and oil and gas production, including concerns related to WST. However, it should be stressed that environmental impacts from WST operations vary greatly by region and in response to local regulations. Many of the following sections include explicit description of known differences between WST processes observed in other regions and the WST processes that are observed in California. There are also certain studies performed outside of California, for example studies measuring ambient levels of certain toxic VOCs, that have not been replicated inside California. In those cases we report the concerns found in other regions with the caveat that it is unknown whether these same issues are relevant or not to California. The sections following the literature review present emissions estimates for each of the five WST processes listed above. Available technology or practices that could be used to control emissions is mentioned at the end of each emission estimate section.

5.2.1.2 Air Quality Literature Review

The review here focuses on studies in which an attempt has been made to link air quality measurements directly to oil and gas production activities. The studies reviewed below present air quality measurements taken far away from the actual well pads and present a variety of methods to attribute measured pollution levels to oil and gas production sources. These types of studies fall in the general category of “top-down” measurement studies. In some cases top-down measurement studies can be used as a way to ensure the assumptions about activity and emission rates in a parallel bottom-up inventory are correct. Other studies reviewed below use other methods to link air quality to oil and gas emissions, for example, air quality modeling. Together the studies represent the range of concerns related to air quality from oil and gas production and provide examples of techniques used to measure the air quality effects. The studies also show the difficulty and uncertainty inherent in characterizing total air quality impacts from oil and gas production, and show the limits to which air quality impacts from separate processes within oil and gas production, for example WST, can be measured.

A common technique for calculating the air-quality impacts of oil and gas operations is to measure both total pollution and the relative abundance of chemicals associated only with oil and gas production operations. For example, C_2 – C_6 alkanes, like propane (C_3H_8), are emitted from oil and gas operations but not other activities. Measurements in spring 2002 presented by Katzentstein et al. (2003) indicated that oil and gas operations were responsible for “major quantities” of VOC emissions across regions in Texas, Kansas and Oklahoma. Katzentstein et al. (2003) also found evidence that the oil and gas emissions led to surface ozone formation.

In Colorado, Gilman et al. (2013) found that VOC emissions related to oil and gas operations were important sources of ozone precursors (during winter 2011). They used the ratio of propane (associated with oil and gas operations) to ethyne (not associated with oil and gas operations) to distinguish between “urban emissions” and those related to oil and gas operations. Compared to ambient measurements in U.S. cities, including Pasadena, CA, the propane-to-ethyne ratio in Northeastern Colorado was often one to two orders of magnitude larger, indicating the presence of emissions from oil and gas operations. Also in Colorado, in locations with both gas development and residential areas, Colborn et al. (2014) found the presence, in ambient air samples, of potentially health-damaging VOCs, including methylene chloride, various endocrine disruptors, and harmful levels of polycyclic aromatic hydrocarbons (PAHs) associated with oil and gas production. The VOCs were highest during the drilling phase and did not increase during hydraulic fracturing. However, venting and condensate tank flashing emissions accounted for 95% of all VOC emissions in Weld County in Colorado (Bar-Ilan et al., 2008; Bar-Ilan et al., 2008; Pétron et al., 2012). Pétron et al. (2012; 2014) found higher VOC emissions from oil and gas operations than listed in a standard bottom-up inventory in Colorado. Venting and condensate tank flashing emissions accounted for a lower fraction of VOC emissions in other regions of the United States. For example, Zavala-Araiza et al. (2014) report that condensate tanks account for close to 50% of total VOC emissions in the Barnett Shale.

Olaguer (2012) modeled near-source air-quality effects in the Barnett Shale, finding that emissions of NO_x associated with compressor engines and flaring can increase peak 1 hr ozone by 3 ppb 2 km and and farther downwind of the source. Olaguer (2012) states: “Major metropolitan areas in or near shale formations will be hard pressed to demonstrate future attainment of the federal ozone standard, unless significant controls are placed on emissions from increased oil and gas exploration and production.” Formaldehyde emissions from flares and compressors were also found to be of concern.

A few papers and public reports examine links between oil and gas production and related air pollution and health effects. For example, McKenzie et al. (2012) found residents living in Wyoming within 0.5 miles of wells were at greater cancer risk due to exposure to benzene and other emissions than residents living farther away from production activity. In contrast, Bunch et al. (2014), in an industry-funded study, examined ambient VOC measurements in regions around the Barnett Shale, and found little evidence of toxic health effects linked to increased gas production activity. Pinto (2006) described high winter ozone episodes in Wyoming associated with oil and gas production and, separately, a report from the Wyoming Department of Health describes the association between observed high ozone levels and increased respiratory health clinic visits (Pride et al., 2013).

In California, Gentner et al. (2014) found that oil and gas operations in the San Joaquin Valley were responsible for about 8% of the anthropogenic precursors to ozone, consistent with the ~10% of total anthropogenic reactive organic gas that is attributed to oil and gas operations in the CARB inventory (CARB, 2009). In the San Joaquin Valley, ozone sensitivity varies by location and characteristic wind direction, and some urban locations would likely see higher sensitivity to increased VOC emissions than rural areas (Jin et al., 2013).

We can conclude from this literature review that in other regions of the country expanded oil and gas production has caused air-quality hazards. Specifically, high measurements of ozone (NO_x and VOC emissions are precursors to ozone) and emissions of toxic VOCs are of concern in multiple regions around the United States. A major challenge revealed in the literature lies in attributing emissions to specific processes within oil and gas operations. Many of the observations used to evaluate air-quality impacts of oil and gas operations are taken as ambient measurements and can be attributed generally to oil and gas sources based on their chemical characteristics, but cannot be attributed specifically to well stimulation processes versus general production processes. An additional challenge is the lack of peer-reviewed literature analyzing emissions of toxic VOCs from oil and gas operations in California. In fact, Allen (2014) points out, in a review of the air quality impacts of natural gas production and use, that in general, "data are sparse on toxic air pollutant impacts of natural gas production…" While we point to concerns related to emissions of toxic VOCs from oil and gas production in other regions, we are not able to make a definitive statement about how relevant those concerns are for operations in California.

Previous sections of this report describe a number of important differences between well stimulation employed in California compared with other regions. Section 5.1.1 indicates less fluid is used per stimulation operation, and Section 5.1.3.2.1 finds that most hydraulic fracturing in California occurs in relatively shallow wells. Because of these differences, published emission estimates from regions in Colorado or Texas, for example, should not be directly applied to California.

5.2.1.3 Qualitative Discussion of Enhanced Emissions Due to Well Stimulation

As described in the introduction, the processes within WST that can lead to significant emissions include trucking supplies to the well pad, pumping the fluid into the well, venting or flaring of gases from the well during WST or completion, and evaporation of chemicals from liquids produced during WST and completion.

In California, a high emission scenario would occur at a deep oil field remote from pipeline infrastructure. In this situation, infrastructure would not be available to transport fluids, oil, or natural gas by pipeline. All materials and fluids would need to be trucked to and from the site, and any methane produced during well completion would need to be flared. Additionally, the field would have some properties similar to those of unconventional plays outside of California, such as the Marcellus shale, and require massive amounts of fluid, $\sim 10^5$ bbl per well stimulation, and use $\sim 10^4$ gallons of diesel fuel per well stimulation to power the pumps (Rodriguez and Ouyang, 2013).

The assumption in the scenario is that the diesel pumps would be at least five years old and the trucks used to bring fluids and supplies would be older than 2007 model year (older diesel engines can emit an order of magnitude more $PM_{2.5}$ and NO_x per gallon of fuel burned compared to the newest engines that have post-combustion controls). Produced fluids would be stored in temporary open-air ponds or tanks before disposal or treatment, allowing dissolved VOC to evaporate.

In this scenario, high levels of uncontrolled diesel combustion, uncontrolled flaring combustion, and potential evaporation and venting of VOCs could lead to high emissions of a number of key pollutants that may cause air-pollution problems, such as described in the literature.

In contrast to the above, a lower emission scenario potentially more common in California, and perhaps representative of current well stimulation in California's South Belridge oil field, will lead to smaller amounts of emissions. In this scenario, pipelines deliver the fluid for well stimulation, removing the burden of trucking the fluid to each well (although other supplies, such as sand, must be trucked to the site if needed). Significantly less fluid, $\sim 10^3$ bbl per well, is needed for well stimulation compared with practices in other regions. Infrastructure exists to pipe away associated gas, and gas produced during completion, so that, ideally, flaring or venting is not performed. Although not required in the San Joaquin Valley, produced fluids would not be allowed to equilibrate with the atmosphere before disposal to a different well or removal to a water treatment facility. Finally, and also not required, newer diesel equipment (trucks and pumps) would be employed to significantly reduce the emissions per gallon of fuel burned.

The comparison of the two scenarios above demonstrates how important local conditions and practices are in determining the amount of emissions related to well stimulation. When evaluating the air pollution hazards of well stimulation, these questions should be asked explicitly: How much fluid will be needed? Will the fluid be delivered by truck or pipeline? How much fuel will be used for pumping during well stimulation? How will fluids recovered from the well be stored and disposed? Will flaring occur? Will direct venting occur? These questions are considered quantitatively below.

5.2.1.4 Quantitative Discussion of Enhanced Emissions Due to Well Stimulation

This discussion focuses on emissions from three broad categories: (1) exhaust from diesel engines including diesel-powered pumping associated with well stimulation and diesel trucks used to bring and remove supplies and waste (fluids, sand, chemicals, equipment); (2) flaring and venting of gases produced during completion, well workovers, or other practices associated with well stimulation; and (3) evaporative emissions from fluids recovered from the well and fugitive emissions throughout the process.

The type of pollution varies by activity type. Diesel engines are associated with NO_x and $PM_{2.5}$ emissions. Flaring is associated with NO_x, PM, and VOC emissions. Evaporative and general fugitive emissions are a concern due to VOC content within the gas or liquids.

5.2.1.4.1 Exhaust from Diesel Pumps

For emissions from diesel trucks and diesel pumps we focus on NO_x and $PM_{2.5}$ emissions. We use a fuel-use-based approach to estimate total emissions per activity as the product of fuel use and an emission factor (mass emitted per mass of fuel used).

We base our estimates of fuel-use for pumping during well stimulation on published estimates of fuel-use in locations outside of California. To adapt the values to California we assume pumping related fuel-use scales linearly with the total fluid volume pumped and then scale fuel-use based on reported fluid volumes in California. One caveat to this approach is that there are other factors that could affect fuel used for pumping than total fluid volume, for example, the pressure to which the fluid was pumped. There is not enough information to characterize these other factors.

In addition to the total fuel used, we need to estimate the emission rate, mass emitted per mass of fuel used, in order to generate emission estimates. We base emission rates on published emission rate estimates and briefly describe here the regulatory framework that controls emissions from diesel engines.

From a regulatory standpoint, diesel equipment is divided into on-road and off-road categories, and the emissions of pollutants per fuel burned vary with the equipment category along with the specific piece of equipment. In California, on-road vehicles (such as the trucks used to deliver fluids, sand, and other supplies) must meet more stringent emission requirements than off-road vehicles. The most dramatic difference is that most on-road heavy-duty trucks must be equipped with some form of post-combustion particle-control device that removes most of the $PM_{2.5}$ emissions compared to an uncontrolled vehicle. Similar regulations regarding $PM_{2.5}$ and NO_x emissions from off-road diesel equipment and NO_x emissions from on-road vehicles will be phased in slowly over the next 10–15 years. Thus, over the next few years, $PM_{2.5}$ emissions from on-road trucks will be significantly lower (on a per mass of fuel basis) compared with $PM_{2.5}$ emissions from diesel pumps (off-road equipment).

Emission standards for on-road equipment and mobile off-road equipment are regulated by CARB. The emission rate from diesel pumping equipment is 28.0 grams of NO_x/kg fuel (0.028 lb emitted/lb fuel) and 1.5 grams $PM_{2.5}$/kg fuel (0.0015 lb emitted/lb fuel) according to Rodriguez and Ouyang (2013). This NO_x emission factor is similar to other previous measurements and analysis of off-road equipment, specifically Tier 2 rated off-road equipment (Abolhasani et al., 2008; Frey et al., 2008; Millstein and Harley, 2009). The $PM_{2.5}$ emission factor is roughly half of the US EPA's estimate reported in Millstein and Harley (2009) and based on the US EPA's NONROAD model. The difference for the $PM_{2.5}$ emission factor between the US EPA and Rodriguez and Ouyang (2013) is related to assumptions regarding the age of the equipment.

Total emissions are calculated as the product of fuel use × emission rate, where, fuel use is a function of the total volume of fluid pumped, and the pressure at which the fuel is pumped. Equipment other than the pumps only contributes a small portion of total fuel-use during the pumping phase (Rodriguez and Ouyang, 2013).

Rodriguez and Ouyang (2013) reported on stimulations in the Marcellus and Eagle Ford shales. The stimulations studied typically pumped about 135,000 bbl of fluid using about 21,000 gallons of diesel fuel over a two-day period. This fuel-use resulted in ~1900 kg (4200 lb) of NO_x emissions and 100 kg (220 lb) of $PM_{2.5}$ emissions over the period. These estimates are within the range of estimates presented by Litovitz et al. (2013) using similar methodology: 3800–4600 kg (8,400 – 10,100 lb) NO_x and 87–130 kg (192 – 287 lb) $PM_{2.5}$ per total well-site development. This estimate includes emissions from hydraulic fracturing itself, but also other activities.

For California, the emissions estimate is based on pumping 175,000 gallons (5,550 bbl) of fluid based on the average water volumes discussed in Section 3.2.3. This volume is between the average from FracFocus and the average from the well stimulation notices. Lacking any information on actual fuel use, the total amount of fluid pumped is used to estimate total emissions from the pumping phase: 5,550 bbl/135,000 bbl = 1/24 of the emissions compared to the high-emissions case. Note that other aspects of the pumping process, such as the pressure used to pump the fluid, may affect fuel use, but in this example only change due to fluid volume is considered. Total pumping emissions are 1/24th the amount estimated for the prior Marcellus and Eagle Ford example. This is approximately 80 kg (176 lb) of NO_x emissions and 4 kg (9 lb) of $PM_{2.5}$ over a 1-day period.

Assuming a rate of 125 hydraulic fracturing operations per month (the center of the 100–150 operations per month estimate used in Section 5.1.1) allows a daily emission estimate. This rate implies the equivalent of four stimulations per day, which would emit an estimated 320 kg (704 lb) of NO_x and 16 kg (35 lb) of $PM_{2.5}$. For reference, CARB estimates ~16 metric tons (18 short tons) and 0.5 metric tons (0.6 short tons) NO_x and $PM_{2.5}$ emissions per day, respectively, in the San Joaquin Valley in 2008 from off-road diesel engines for use in oil drilling, workovers, and pumping (CARB, 2009).

We conclude, based on the above estimates, that pumping for well stimulation is not currently a major air pollution hazard in California as the estimates for pumping here are only 2-3% of the total emissions from off-road diesel engines associated with oil and gas production in the San Joaquin Valley as estimated by CARB. However, if well stimulation requires significantly more fluid in the future, such as due to a transition to using slickwater, then emissions from diesel pumps could become more important. Technology exists to control emissions from diesel pumping. The use of US EPA classified "Tier 4" nonroad diesel engines would reduce emissions from diesel equipment by 90% compared to diesel equipment from the 1990s.

5.2.1.4.2 Exhaust from Diesel Trucking Activity

To estimate emissions from diesel trucking activity we use similar methods to those used to estimate pumping related emissions. Again we use a fuel-use-based approach to estimate total emissions per activity as the product of fuel use and an emission factor. To estimate fuel-use we reference published estimates of fuel economy for bulk tankers, include the fluid volume estimates from the above example, and chose an arbitrary delivery distance of 100 miles. Emission rates for on-road heavy-duty diesel trucks are based on previous literature.

Heavy-duty trucks, such as bulk tankers and tractor-trailers, consume 6.5 gallons per 10^3 short ton-miles (14 kg of fuel per 10^3 metric ton-kilometer; Davis et al., 2013). Applying emission factors of ~30 g NO_x/kg fuel and ~1 g $PM_{2.5}$ / kg fuel (Dallmann and Harley, 2010) results in freight emission factors of about 686 g NO_x and 23 g $PM_{2.5}$ per 10^3 metric ton-miles.

In the Marcellus and Eagle Ford example based on Rodriguez and Ouyang (2013), 135,000 bbl of fluid were delivered, although the delivery method and distance are not specified. The 135,000 bbl delivered would weigh about 16,000 metric tons. If delivered by truck, and over a hypothetical distance of 100 miles that would be 1.6×10^6 metric ton-miles. Multiplying by the above emission factors yields about 1.1 metric tons (1.2 short tons) NO_x and 37 kg (81 lbs) $PM_{2.5}$ per well stimulation operation.

As described in the California example above, the fluid volume commonly used in California is 1/24th the fluid used in the Marcellus and Eagle Ford example from Rodriguez and Ouyang (2013). Consequently, emissions from delivering the fluid in California would equal about 46 kg (101 lb) NO_x and 1.5 kg (3.3 lb) $PM_{2.5}$ per well stimulation. These emissions may be lower as fluid is often delivered by pipeline close to the well, according to many of the hydraulic fracturing notices.

Based on the data in FracFocus, a reasonable estimate of the average mass of proppant used per hydraulic fracture operation is about 100 metric tons (110 short tons). Transporting this proppant for 100 miles by truck emits 7 kg NO_x and 0.2 kg $PM_{2.5}$ per well stimulation.

Summing the emissions from fluid and sand delivery gives emissions of 53 kg NO_x and 1.7 kg $PM_{2.5}$. Based on the 125 hydraulic fracture operations per month used in the pumping emission estimate, the annual trucking emission estimate is about 79 metric tons

(87 short tons) NO_x and 2.6 metric tons (2.9 short tons) $PM_{2.5}$ per year for sand and fluid deliveries. CARB (2009) estimates on-road diesel trucks emit about 75,000 metric tons (82,000 short tons) of NO_x per year and 2,700 metric tons (3,000 short tons) of $PM_{2.5}$ per year in the San Joaquin Valley. Emissions from delivering supplies are small in comparison.

As with pumping emissions, the above estimates indicate that delivery of supplies, particularly fluids and sand for well stimulation, is not a major contributor to NO_x and $PM_{2.5}$ emissions in California. The basic point being, given current activity levels, the trucking activity required for WST is negligible compared with the trucking activity for other sectors in the San Joaquin Valley. However, if future well stimulation techniques require significantly more fluid, then emissions from the diesel trucks that might deliver the supplies could become more important. As with diesel pumping equipment, technology to control emissions from trucks is available. In fact, most trucks built since 2010 are required to include exhaust controls for $PM_{2.5}$ and NO_x emissions. Simply requiring that the newest trucks be employed could largely reduce emissions from diesel trucking activity associated with WST.

5.2.1.4.3 Emissions from Flares

Information on the number of flares used for well stimulation or completion operations is not available. The combustion conditions and efficiency of the flares are also unavailable. Thus, emissions of CO, NO_x or $PM_{2.5}$ from flaring cannot be estimated quantitatively. Instead, in this section we review basic information about flaring in California and report the results of emission estimates developed by the State.

Emissions during flaring consist of unburned VOC, partially combusted VOCs, $PM_{2.5}$, carbon monoxide (CO), and NO_x produced during combustion. The characteristics of emissions from flares vary across uses and conditions. For example Torres et al. (2012) found combustion efficiency of flares varied under various operating conditions, although this study focused on large industrial flares, which would not necessarily represent the types of flares found in use during well completion activities. Note there are very few studies of flaring efficiency relevant to oil production in the United States. Allen et al. (2013) report 99.5% efficiency for a production flare in the Marcellus Shale, but this does not provide enough data to derive general conclusions about flaring efficiency related to WST. In the San Joaquin Valley, permanent facilities are required to obtain permits in order to operate a flare. Well drilling, completion, and stimulation are considered temporary, however, and would not need to file for a specific permit (personal communication, Mike Oldershaw, San Joaquin Valley Air Pollution Control District (SJVAPCD)).

The official CARB inventory shows emissions from flares, as a percent of total oil and gas emissions in the San Joaquin Valley, are equal to 0.3, 3.6, 1.0, 5.7, 1.5, and 1.6% for VOCs, CO, NO_x, SOx, PM, respectively (CARB, 2009). It is unclear, however, if the reported emissions account for flaring during completion stages. It is also unclear how the volume of gas flared during completion stages would compare to the volume of gas that was assumed flared in CARB's above emission estimates.

CARB's oil and gas survey (Detwiler, 2013) estimates annual total carbon dioxide emissions from flaring in California of 242,454 metric tons (267,260 short tons). A first-order estimate of unburned VOC emissions can be derived from this estimate. Given the lack of relevant peer-reviewed studies addressing flaring efficiency, we assume a standard efficiency of 98% for flaring during production, and a "generic" composition of 80% methane, 15% ethane, and 5% propane (Shires et al., 2009), to find total annual flaring is estimated to emit 335 metric tons (369 short tons) of non-methane hydrocarbons or only 82 metric tons (90 short tons) of VOCs (reactive organic carbon such as propane; methane and ethane are not considered reactive, or precursors to ozone).

This total would not include VOC produced from combustion itself. It is also unclear if this total includes flaring operations during completion or well stimulation operations, or is limited to more permanent flares. However, CARB estimates a total of 369 short tons of anthropogenic reactive VOC emitted per day, or about 135,000 short tons per year in the San Joaquin Valley. VOC emissions from flaring would need to be more than an order of magnitude larger than the oil and gas survey estimated to be of consequence to VOC atmospheric concentrations at the regional level.

In conclusion, although emissions from flaring are uncertain, the two inventories described above indicate that current flaring associated with WST is likely to have a negligible effect on air quality in the San Joaquin Valley. Control technology is not available to control flaring emissions; however, alternate practices can reduce the use of flares, for example requiring reduced emission completions, or "green completions."

5.2.1.4.4 Fugitive and Evaporative Emissions

A representative set of direct measurements of fugitive and evaporative VOC emissions from oil and gas processes in California is not available. In this section we review published emission inventories. We compare a top-down measurement study of VOC emissions from oil and gas production in the San Joaquin Valley to CARB's bottom-up inventory.

As mentioned in the literature review, Gentner et al. (2014) found that ambient VOC measurements from a field campaign in the San Joaquin Valley indicated that oil and gas operations were responsible for 8% of organic compound ozone precursor emissions, consistent with CARB's estimates. Note that estimates from both CARB and Gentner et al. (2014) take into account, to a certain extent, the reactivity of the organic compounds. For example, Gentner et al. (2014) indicate that while petroleum operations comprised 22% of anthropogenic non-methane organic carbon at Bakersfield, petroleum operations account for 8% of anthropogenic ozone precursor emissions. CARB's inventory attributes 60% of all oil and gas-related VOC emissions in the San Joaquin Valley to the top five oil and gas source types, all of which are related to production, as opposed to processing and marketing. The top five sources are listed as "(1) tertiary oil production - cyclic wells, (2) tertiary oil production - steam drive wells, (3) i.c. reciprocating engines, (4) fugitive losses – fittings, and (5) fugitive losses – valves." None of the top five source types is related to evaporative sources; they are instead related to combustion sources and to fugitive emissions during production processes.

Based on CARB's estimate, vented and evaporative emissions from liquids related to WST are not a major source of VOC emissions. The agreement between CARB's bottom-up estimate of emissions from oil and gas operations in the San Joaquin Valley and the top-down estimate for the same sector reported by Gentner et al. (2014) indicate that it is unlikely that there is a large unknown source of VOC emissions. However, one potential problem with CARB's estimate is that it is unclear if VOC emissions during fracturing and completion are incorporated into the inventory, as they are not considered stationary sources by the SJVAPCD. It should be noted that while initial measurements and inventories in the San Joaquin Valley are in agreement, Zavala-Araiza et al. (2014) report that emission inventories for production processes in the Barnett Shale region are not in agreement with atmospheric measurements and Allen et al (2013) find that emissions from liquid unloadings may be poorly represented and potentially underestimated.

In conclusion, evaporative VOC emissions directly from WST have not been directly measured but current California inventory indicates they are unlikely to cause significant impacts to ozone air quality. Technology exists that could control evaporative VOC emissions, such as requiring vapor controls on temporary tanks in which WST flowback water is stored. Additionally, requiring green completions could control vented VOC emissions related to WST. It is important to note that, in the San Joaquin Valley, the oil and gas industry contributes ~8% of anthropogenic ozone precursors, thus any marginal increase to total oil and gas production could potentially lead to increased ozone levels.

5.2.1.5 Air Quality Conclusions

Estimated marginal emissions of NO_x, $PM_{2.5}$, and VOCs directly from activities directly related to WST appear small compared to oil and gas production emissions in total in the San Joaquin Valley where the vast majority of hydraulic fracturing takes place. However, the San Joaquin Valley is often out of compliance with respect to air quality standards and as a result, possible emission reductions remain relevant.

Three major sources of air pollutants include the use of diesel engines, flaring of gas and the volatilization of flowback water. The first, diesel engines (used for transport and pumping of estimated fluid volumes required for WST) emit a small portion of total emissions of nitrogen oxides (NO_x), particulate matter ($PM_{2.5}$), and VOCs associated with other oil and gas production operations as a whole.

Emissions from flaring in California are uncertain because of variability in flare combustion conditions and to a lack of information regarding the frequency of flare-use during WST operations. However, current CARB inventories of pollutant emissions from all flaring suggest that flares as a whole emit less than 0.1% of the VOCs and are not a major regional air quality hazard.

Emissions from volatilization of flowback water constituents have not been measured but might be bracketed. CARB has conducted a "bottom-up" VOC emission inventory by adding up all known sources of emissions. It is unknown whether these sources included emissions from WST-related produced or flowback water. However, the sum of the emissions in the inventory matches well with "top-down" measurements taken from the air in the San Joaquin Valley (Gentner et al., 2014). This agreement between "bottom-up" and "top-down" estimates of VOC emissions from oil and gas production indicates California's inventory probably included all major sources.

The inventory indicates that VOC emissions from oil and gas evaporative sources, such from flowback water, might occur from stimulation fluids produced back after the application of WST, are small compared to other emission sources in the oil and gas development process. Data suggest that emissions from oil and gas production and upstream processing in general contribute to ~10% of anthropogenic VOC ozone precursor emissions in the San Joaquin Valley. Although the marginal emissions from WST alone are small, the potential increase to VOC emissions due to other oil and gas production activities enabled by WST may impact ozone air quality in the San Joaquin Valley.

Emissions from diesel equipment and diesel trucks can be controlled through use of the cleanest engines, such as US EPA classified Tier 4 engines for off-road equipment or on-road truck engines that meet 2010 engine standards. Requiring reduced emission completions can control emissions from flaring and venting related to WST. Emissions from evaporative sources related to WST could be limited by requiring vapor controls on the temporary tanks to which flowback water is stored.

As described above, some of the potential air-quality impacts can be addressed by regulation and largely avoided. Most WST takes place in the San Joaquin Valley. WST is subject to a variety of regulatory processes in the San Joaquin Valley. For example, there are requirements on emissions from individual pieces of equipment, and new drilling operations must meet New Source Review and other regulations. Evaluation of opportunities to reduce emissions of pollutants from WST and other production-related operations would benefit from independent, on-the-ground studies of emissions from individual processes within petroleum production in the San Joaquin Valley.

If practices in California changed, for example if more fluid was used in WST or production moved to remote locations, emissions from activities directly related to WST could become important if left uncontrolled.

5.2.2 Climate Impacts

This section presents estimates of GHG emissions associated with WST. GHG emissions in California occur in a context of needing to reduce total emissions to 2005 levels under AB32 and a Governor's executive order that requires 80% emission cuts by 2050. The oil and gas enterprise worldwide is responsible for a large fraction of the total GHGs emitted to the atmosphere. By far the largest factor in these emissions is burning the fuel, not producing it. Nevertheless oil and gas production produces GHGs and in California these are subject to control under the state's climate laws.

GHG emissions from WST come from fuel-use associated with pumping and supply delivery and also from fugitive methane emissions. We find that CO_2 emissions from fuel-use directly related to WST are negligible.

Fugitive methane emissions in this case include vented and leaked methane during WST and also methane that is emitted from flowback water. Methane emissions from oil and gas operations are uncertain and are currently a major research topic. Because of the uncertainty regarding methane emissions and because methane is a potent greenhouse gas it is a focus of this section.

We review measurement studies and current inventory estimates of methane emissions from oil and gas production in California. A number of measurement studies in California suggest higher methane emissions from oil and gas production activities than is listed in the State inventory. However, even if accepting the higher rate of emissions indicated by the measurement studies, the marginal fugitive methane emissions from the direct application of WST to oil wells are likely to be small compared to the total greenhouse gas emission impacts from current energy–intense oil and gas production in California. Methane emissions related to WST could be controlled by requiring reduced emission, "green," completions and by requiring tighter vapor controls on temporary tanks that hold flowback water.

5.2.2.1 GHG Emissions in California

According to California's official GHG emission inventory, oil and gas extraction processes account for ~16 million tons of CO_2eq emissions, or 3.5% of California's total GHG emissions (CARB 2013). In California, Assembly Bill 32 requires reductions of total GHG emissions to below a cap in 2020 and a Governor's executive order requires 80% emission cuts by 2050, thus increases to emissions from sectors accounting for only a few percent of total emissions may become important if state total emissions are close to the cap.

The marginal GHG emissions from WST are small, of course, compared to the emissions from burning the fuel that is produced from stimulated wells. However, this section focuses on the marginal emissions from WST and not on emissions from combustion of the produced fuel. Emissions of GHG from well stimulation come from fuel combustion and fugitive methane emissions (methane that is vented or leaked from wells or equipment). Emissions of CO_2 are tied directly to fuel-use or flaring.

Per ton emitted, methane is considered to cause much more warming than CO_2 (Myhre et al., 2013). Because methane is such a potent greenhouse gas, small leaks of methane can be important sources of greenhouse gas emissions. Much of this section focuses on methane emissions during WST and oil and gas production in general for this reason.

Another reason that methane emissions are a focus of this section is that methane emissions from oil and gas operations are uncertain. Methane emissions from oil and gas operations are currently a major research topic. Methane leak rates are likely not normally distributed, but heavily skewed so that a few locations may have high leak rates compared to an average location (Brandt et al., 2014). Because of the uncertainty regarding the distribution of methane emissions among locations and across geographies, and the lack of easy access to production locations, most field campaigns designed to measure methane leakage and venting from specific processes during production are unable to capture a representative sample. One potential solution that would allow for a field campaign to derive a representative sample would be for a regulatory agency to compel companies to allow independent researchers access to production areas, for example see the City of Fort Worth Natural Gas Air Quality Study (Eastern Research Group; Sage Environmental 2011). "Bottom-up" estimates of methane emissions, based on counting equipment and processes and applying an average emission factor to each type of equipment and process, commonly produce estimates of total methane emissions that are significantly lower than "top-down" regional measurement campaigns (Brandt et al., 2014).

In this section, a short literature review describes top-down and bottom-up estimates of methane emissions. This is followed by assessments of GHG emissions from three processes that occur during WST: (1) diesel fuel-use for pumping and supply delivery; (2) emissions from flaring; and (3) emissions of fugitive methane.

Some of the same processes associated with VOC emissions from venting, flaring and evaporative sources, described earlier in Sections 5.2.1.4.3 and 5.2.1.4.4, will also lead to methane emissions. As published literature does not provide specific enough information to develop fugitive methane emission estimates from WST, our approach to evaluate fugitive methane emissions will be to compare current California bottom-up inventories of methane emissions from oil and gas production to top-down methane measurements and attempt to bracket total emissions. The total methane emission estimates will then be put in context of total GHG emissions estimates from all oil and gas production activities. Carbon dioxide emissions will be related directly to fuel use, and thus be related directly to processes used to calculate NO_x and $PM_{2.5}$ emissions in the air-quality section.

5.2.2.2 Methane Emissions Literature Overview

In the case of California, top-down measurement studies indicate higher oil and gas emissions than bottom-up inventories. For example, Wennberg et al. (2012) and Peischl et al. (2013), using aircraft measurements, find high emissions of methane from the overall oil and gas system in Southern California compared with bottom-up inventories. Peischl et al. (2013) estimate emissions separately for the oil and gas production and processing sector and the gas transmission and distribution sector, and report higher emissions of methane compared to the bottom-up inventories for both of those sectors. Jeong et al. (2014) compare bottom-up and top-down inventories in California and found that if the emissions rates estimated for the oil and gas industry in southern California by Peischl et al. (2013) were extended across the state, state total methane emissions from oil and associated gas production would be equal to 1% of CA total CO_2eq (100 yr), roughly five times the official state inventory estimate for oil and associated gas production. Gentner et al. (2014) also found qualitative evidence that dairies were responsible for the majority of methane emissions in the San Joaquin Valley, but did not provide quantitative estimates of methane emissions. Jeong et al. (2014) estimated that even accounting for the higher emission inventory for oil and gas production based on ambient measurements, dairies still emit eight times the methane in the San Joaquin Valley compared with oil and gas production, and so the higher California emission estimates are not in conflict with work by Gentner et al. (2014).

Outside of California, Pétron et al. (2012) find that approximately 4% of total methane production is emitted to the atmosphere, approximately two times the methane emissions estimate from a standard bottom-up inventory in Colorado. Karion et al. (2013), using airborne measurements of methane from a large field in Uintah County, Utah, find high emissions: 6.2%–11.7% (1σ) of production. The Petron et al. (2012) emission estimate does not distinguish between emissions by well type (petroleum or natural gas) and should not be directly compared to national inventories that estimate emissions from petroleum production and natural gas production separately (see for example, Brandt et al. 2014.) The Karion et al. study focuses on the gas-bearing portion of the basin, and so should not suffer from this problem. Maps in the Karion et al. paper show clear divergence between oil and gas regions of the Uintah, and a flight path that would isolate the gas wells.

From a national perspective, a report by the US Government Accountability Office (GAO) indicates that up to 5% of total gas production can be vented and flared, and in some cases a majority of the venting and flaring activity occurred during completion (US GAO, 2010). Similarly, Howarth (2011) estimated that up to 3.2% of lifetime gas production is emitted as methane during the flowback period following stimulation in shale gas. However, note that Cathles et al (2012) and O'Sullivan and Paltsev (2012) contest some of the methodology employed by Howarth et al., (2011) and an analysis by the Department of Energy and Climate Change in the United Kingdom (MacKay and Stone 2013) concluded that the result presented in Howarth et al (2011) is an outlier (six times greater than the next highest estimate).

Allen et al. (2013) estimates, based on measurements of a sample of individual completion events and other activities, that only 0.42% of national gross gas production is leaked or vented to the atmosphere. The 0.42% rate includes only production operations and not gathering, processing, and other sectors. However, the work by Allen et al. (2013) depends on measurements of a small sample of facilities, and it is unclear whether the sample is representative of oil and gas operations at large.

The range of emission values reported in the studies above indicates there is a high degree of variability and uncertainty in emissions from oil and gas production. Estimates of methane leakage from oil and gas production across the country are highly variable and depend on specific features of the fields being measured, and are often not directly comparable. The topic of methane emissions from oil and gas production is an active area of research. The limited number of studies in California indicate that current bottom-up inventories have smaller estimates of methane emissions compared to top-down estimates in Southern California.

5.2.2.3 Assessment of Emissions from Diesel Fuel Use Related to Well Stimulation

The approach to estimate CO_2-related emissions is to base the estimates on the amount of diesel fuel use estimated under the scenarios presented in the air-quality section. In the high-fluid-volume scenario, $\sim 10^4$ gallons of diesel fuel were used for pumping during well stimulation. At about 10 kg CO_2/gallon diesel, that is about 100 metric tons CO_2 (110 short tons) per well stimulation event. Carbon dioxide emissions from delivery of supplies are similar in magnitude (see Sections 5.2.1.4.1 and 5.2.1.4.2), so the total emissions in the high-volume scenario are about a couple hundred metric or short tons per well stimulation event from both fluid delivery and pumping.

The low-emissions scenario, based on hydraulic fracturing fluid volumes in California and following the methods in the air-quality section, results in an estimate of about 3 metric tons CO_2 (3.3 short tons) per well stimulation event or about 4,500 metric tons (5,000 short tons) per year, based on the estimate of 125 operations per month.

For perspective, the California Air Resources Board estimated 13 million metric tons per year of direct CO_2 (14.3 million short tons) emissions from steam generators, turbines, and combined heat and power production within the oil and gas industry in California (Detwiler, 2013). The same report estimated that only 45 thousand metric tons CO_2 (50 thousand short tons) were emitted from all water and other non-crude oil pumps. Consequently, a drastic change in well stimulation activity or volume would be needed to materially impact the CO_2 emissions from the oil and gas industry. We conclude that GHG emissions from diesel fuel use during WST are negligible.

5.2.2.4 Emissions of CO_2, CH_4, and N_2O from Flaring

There is little available information regarding flaring in California beyond what is reported in official state inventories, thus in this section we review the state's inventories to generate conclusions. CARB's oil and gas survey (Detwiler, 2013) reported 196 flare "units," accounting for 242×10^3, 812, and 0.4 metric tons of CO_2, CH_4, and N_2O, respectively (267×10^3, 895, 0.44 short tons, respectively). Note that it is unclear what portion of those emissions is related directly to well stimulation or well-completion activities, or even if well stimulation and completion activities were incorporated in that total. As described in Section 5.3.1.4.3 regarding air quality and flaring, in the San Joaquin Valley, where much of California's well stimulation activity takes place, drilling, fracturing, and well completion are considered temporary activities, and thus operators are not required to obtain permits for flaring.

There is uncertainty about the efficiency of flares. Without site-specific information, the standard efficiency assumption for flares, as defined by the American Petroleum Institute (Shires et al., 2009), is that 98% of the gas is combusted, leaving 2% vented. Due to the high GWP of methane, a reduction in average efficiency of a group of flares from 98% to 97% could have a significant impact of total GHG emissions. However, as with fuel use, the relatively low baseline of GHG emissions from flaring from all oil and gas production and processing (as opposed to only well stimulation-related activities) in California suggests that even doubling or tripling the activity of flaring in the State would have only a marginal effect on total GHG emissions from the oil and gas production sector. Current use of flaring in WST causes negligible GHG emissions and could be controlled by requiring green completions.

5.2.2.5 Fugitive Methane Emissions

In this section, we review bottom-up estimates of fugitive methane emissions and compare them to top-down studies. The discrepancy between the bottom-up inventories and top-down measurements of methane emissions (top-down measurements indicate higher methane emissions) from oil and gas operations indicates the high level of uncertainty regarding methane emissions from the sector as a whole. Below we describe what the implications are for GHG emissions if the top-down estimates are correct. Note that additional uncertainty exists when attempting to estimate emissions from a process, such as well stimulation, within the larger group of production activities.

Table 5-7 shows methane and CO_2 emission estimates for oil and gas production from CARB's bottom-up survey. This again shows that CO_2 emissions (primarily due to steam generation for enhanced oil recovery, which is not evaluated in this report) are dominant over methane emissions. Even after increasing the oil and associated gas production methane emissions by a factor of five, as suggested by Jeong et al. (2014), direct CO_2 emissions still dominate total GHG emissions from oil and gas production. Quantifying the portion of fugitive emissions from production processes attributable to well stimulation is not possible without more detailed information on the well stimulation activities. To conclude, the marginal methane emissions from WST are uncertain, but likely much smaller than the direct CO_2 emissions from oil and gas extraction. The marginal fugitive methane emissions from WST could be controlled through the requirement of green completions and by requiring vapor controls for flowback water.

Table 5-7. Estimated greenhouse gas emissions from oil and gas production in 2007 (Detwiler, 2013).

Process	Constituent	Total Statewide, 10^6 metric (short) tons CO_{2e}
Venting (from well workovers)	CH_4	0.07 (0.08)
Venting (from well completions)	CH_4	NOT ESTIMATED
Oil and Associated Gas Production Total	CH_4	1.07 (1.18)
Oil and Associated Gas production and processing total	CH_4	2.1 (2.31)
Oil and Gas CO_2 + CH_4 total (mostly generating steam)	CO_2 + CH_4	18.6 (20.5)

5.2.2.6 Climate Impact Conclusions

Fugitive methane emissions from the direct application of WST to oil wells are likely to be small compared to the total GHG emissions from oil and gas production in California. This is because current California oil and gas operations are energy intensive. However, all GHG emissions are relevant under California's climate laws and many emissions sources can be addressed successfully with best available control technology and good practice.

Fugitive methane emissions for oil and gas production are uncertain and are currently an active area of scientific research. A number of measurement studies in California suggest higher methane emissions from oil and gas production activities than is listed in the State inventory. However, even if accepting the higher rate of emissions indicated by the measurement studies, methane emissions from oil and gas production are still likely to be small compared to direct CO_2 emissions associated with oil and gas production. Additionally, methane emissions directly related to WST are likely to account for only a small portion of total production related methane emissions.

Methane emissions related to WST can be addressed successfully with best controls, such as requiring reduced emission, or "green," completions and requiring vapor controls on temporary tanks in which flowback water is stored. For example, Allen et al. (2013) reported low leakage rates from well completions after some of the controls listed above were implemented compared to uncontrolled processes and ICF International (ICF 2014) analyzed the costs and viability of methane reduction opportunities in the U.S. oil and natural gas industries. We note that while green completions will be required nationally for gas wells starting in 2015, they will not be required for wells that produce oil or oil and associated gas, such as most of the wells in the San Joaquin Valley. Other emissions such as CO_2 from diesel fuel used for pumping fluid or delivering supplies were found to be negligible.

While other regions are currently using WST for the production of petroleum (e.g., the Bakken formation of North Dakota) or gas (e.g., the Barnett shale of Texas), emissions from these regions may not be representative of emissions from California-specific applications of WST. For example, the volume of fluid used for WST operations in California is typically lower than operations in other shale plays, potentially leading to lower evaporative emissions of methane from flowback fluid.

5.3 Potential Seismic Impacts

Induced seismicity is a term used to describe seismic events caused by human activities. These include injection of fluids into the subsurface, when elevated fluid pore pressures can lower the frictional strengths of faults and fractures leading to seismic rupture. Induced seismicity can produce felt or even damaging ground motions when large volumes of water are injected over long time periods into zones in or near potentially active earthquake sources. The relatively small fluid volumes and short time durations involved in most hydraulic fracturing operations themselves are generally not sufficient to create pore pressure perturbations of large enough spatial extent to generate induced seismicity of concern. Current hydraulic fracturing activity is not considered to pose a significant seismic hazard in California. To date, only one felt earthquake attributed to hydraulic fracturing in a California oil or gas field has been documented, and that was an anomalous slow-slip event that radiated much lower energy at much lower dominant frequencies than normal earthquakes of similar size.

In contrast to hydraulic fracturing, earthquakes as large as magnitude 5.7 have been linked to injection of large volumes of wastewater into deep disposal wells in the eastern and central United States. To date, compared to some other states, water disposal wells in California have been relatively shallow and volumes disposed per well relatively small. There are no published reports of induced seismicity caused by wastewater disposal related to oil and gas operations in California, and at present the seismic hazard posed by wastewater injection is likely to be low. However, possible correlations between seismicity and wastewater injection in California have not yet been studied in detail. Injection of much larger volumes of produced water from increased WST activity and the subsequent increase in oil and gas production could increase the hazard, particularly in areas of high naturally-occurring seismicity. Therefore, given the active tectonic setting of California, it will be important to carry out quantitative assessments of induced seismic hazard and risk. The chance of inducing larger, hazardous earthquakes would most likely be reduced by following protocols similar to those that have been developed for other types of injection operations, such as enhanced geothermal. Even though hydraulic fracturing itself rarely induces felt earthquakes, application of similar protocols could protect against potential worst-case outcomes resulting from these operations as well.

5.3.1 Overview of Seismic Impacts

Earthquakes attributed to human activity are termed induced seismicity, and have been observed for many years. Activities that can induce earthquakes include underground mining, reservoir impoundment, and the injection and withdrawal of fluids as part of energy production activities (see National Research Council (NRC), 2013). Note that some authors distinguish between “induced” and “triggered” events according to various criteria (e.g. McGarr et al., 2002; Baisch et al., 2009). In this report we do not make this distinction, but refer to all earthquakes that occur as a consequence of human activities as induced seismicity. With respect to seismicity related to well stimulation for oil and gas recovery, we will address the effect of fluid injection during the initial hydraulic fracturing stimulation and flowback periods as well as the impact of waste fluid disposal during the entire period of stimulation and subsequent production.

An earthquake is a seismic event that involves sudden slippage along a fault or fracture in the Earth. This process occurs naturally as a result of stresses that build up owing to deformation within the Earth’s crust and interior. The size, or magnitude, of an earthquake depends primarily on the surface area of the fault patch that slips and the amount of stress relieved. Earthquake sizes range over many orders of magnitude. There are many more small than large events; roughly, a decrease of one unit in the magnitude scale corresponds roughly to a ten-fold increase in the number of events. As a result, the vast majority of earthquakes can only be detected by sensitive instruments. If, however, the slip area is sufficient to generate an earthquake larger than magnitude 2 to 3 the amount of energy released during the event can generate seismic waves sufficient to produce ground motions that can be felt by humans and in some cases cause structural damage (usually above magnitude 4). Over 1 million natural earthquakes of magnitude 2 or more occur worldwide every year (NRC, 2013).

The mechanism that explains how well stimulation activities can cause earthquakes - i.e., reduction in the forces holding a fault together due to increased fluid pressure in the fault - is fairly well understood (Hubbert and Rubey, 1959). However, applying this knowledge in a predictive sense is difficult because of uncertainties in in situ rock material properties and stress conditions and complexities in well stimulation procedures and the resulting pressure perturbations. Assessing the seismic hazard in a local area due, for example, to fluid injection requires knowledge of pre-existing faults, the state of stress on those faults, rock properties, and subsurface fluid pressures. As in seismic hazard in general, an important part of the hazard assessment procedure is to properly characterize the uncertainties in these input parameters, which are usually large.

To date, the largest observed event caused by hydraulic fracture stimulation itself is the magnitude 3.6 earthquake that occurred in the Horn River Basin in 2011 (see Table 5-8). The lower magnitudes of events associated with hydraulic fracturing relative to those induced by wastewater disposal are generally attributed to the short durations, smaller volumes and smaller pressure disturbances involved in hydraulic fracturing, compared with the longer time periods and much higher volumes of wastewater injection. None of the events related to hydraulic fracturing reported in the literature has occurred in California and (with the possible exception of one paper that discusses a highly anomalous event) we have found no published study that addressed this topic in California. If hydraulic fracturing operations carried out in California to date have, in fact, not caused normal seismic events above magnitude 2, one possible explanation is the small injected volumes employed so far (Section 3.2.3). A shift to larger volumes, perhaps also combined with a shift to deeper stimulation, could increase the probability of such events occurring, and hence increase the hazard.

The largest observed earthquake suspected to be related to wastewater disposal in the US to date is the 2011 magnitude 5.7 event near Prague, Oklahoma (Keranen et al., 2013; Sumy et al., 2014), although the cause of this event is still under debate (Keller and Holland, 2013; McGarr, 2014). The typical wastewater volumes injected per well in California are generally less than those related to shale hydraulic fracturing operations in other parts of the country where induced events have occurred. For example, to date typical California volumes are about four times less than in the Barnett shale in Texas. This would suggest that at the present time the potential seismic hazard from wastewater disposal in California is low compared with other regions in the US. Expanded hydraulic fracturing activity would, of course, require disposal of larger volumes of fluid, which could potentially increase the hazard.

5.3.2 Mechanics of Earthquakes Induced by Fluid Injection

This section summarizes the physical mechanisms responsible for earthquakes induced by fluid injection and the geological and tectonic conditions that influence their occurrence. The characteristics of pore pressure perturbations and induced seismicity resulting from both well stimulation and wastewater disposal and their potential impact on seismic hazard are discussed in Section 5.4.4.

During fluid injection there can be two types of rock failure, tensile and shear. Below we describe these two types of failure in the context of injection operations related to hydraulic fracturing stimulation.

5.3.2.1 Tensile Fracturing

The primary objective of stimulation is to inject fluid into the earth to create a new fracture (a hydraulic fracture) that connects the pores and existing fractures in the surrounding rock with the well, thus forming a permeability pathway that enables the oil and/or gas (and water) in the pores and fractures to be recovered. Hydraulic fractures are created by the rock failing in tension when the fluid pressure exceeds the in situ minimum principal stress (see Section 5.4.2.3 below). In this type of failure the walls of the fracture move apart perpendicular to the fracture plane. These large-scale hydraulic fractures form slowly (hours) and can extend hundreds of meters away from the well. Although the physical processes at the crack tip are not yet fully understood, it appears that the amount of seismic energy radiated as it propagates is small and difficult to detect. Therefore, hydraulic fracture growth is responsible for little if any of the seismicity recorded in the field, and it probably makes little or no contribution to seismic hazard.

5.3.2.2 Shear Failure on Pre-Existing Faults and Fractures

Shear failure on existing faults and fractures can occur during both stimulation and wastewater disposal. During stimulation shear events serve to enhance the permeability of small, existing fractures and faults and to link them up to create conductive networks connected to the main hydraulic fracture. Shear slip is the type of failure that occurs in most natural tectonic earthquakes, and it is shear events on larger faults that can produce perceptible or damaging ground motions at the Earth's surface.

During a shear event the two faces of the fault slip in opposite directions to each other parallel to the fault surface. The conditions for the initiation of shear slip are governed by the balance between the shear stress applied parallel to the fault surface, the cohesion across the fault and the frictional resistance to sliding (shear strength). Stress is the force applied per unit area. Assuming that the cohesion is negligible, these conditions are summarized in the Coulomb criterion,

$$\tau=\mu(\sigma - p),$$

in which an applied shear stress (τ) is balanced by the shear strength, which is the product of the coefficient of friction (μ) and the difference between normal stress (σ) and pore-fluid pressure (p). Shear stress is directed along the fault plane, while normal stress is directed perpendicular to the plane. Nearly all rocks have μ values between 0.6 and 1.0. The quantity (σ-p) is called the effective stress. Effective stress represents the difference between the normal stress, which pushes the two sides of the fault together and increases the frictional strength, and the fluid pressure within the fault, which has the opposite effect. The Coulomb criterion states that slip will occur when the shear stress (τ) exceeds the strength of the fracture (right hand side of the equation). So failure can be instigated by decreasing the effective stress either by decreasing the normal stress (σ) which holds a fracture closed, or by increasing the fluid pressure in the fracture thus pushing the sides of the fracture apart, or by simply increasing the shear stress itself.

5.3.2.3 State of Stress

To assess when a fault will slip according to the Coulomb criterion, it is necessary to know the local state of effective stress, also called in *situ* stress. The *in situ* effective stress state is fully described by pore pressure and three orthogonally directed principal stresses, which are related to the normal and shear stress on a fault by the fault orientation. Within the Earth, the load of the overburden at a given depth usually leads to a compressional state, with one principal stress oriented vertically (σ_v) and having a magnitude equal to the weight per unit area of the overlying rock. This simplifies the problem of determining the complete stress state to estimation of the minimum (σ_h) and maximum (σ_H) horizontal stresses and the azimuth of one of them. However, determining the in situ stress state is still a challenging problem because often only approximate stress directions and the type of stress regime — normal, strike-slip or thrust faulting — are known (e.g., Heidbach et al., 2008). Stress parameters are inferred from available, often sparse measurements in the region, such as earthquake focal mechanisms, wellbore breakouts and drilling-induced fractures (Zoback and Zoback, 1980; Heidbach et al., 2008). In principle, the relative magnitudes of the principal stresses and the stress azimuths enable identification of the faults that are most favorably oriented for slip and calculation of the normal and shear stress acting on them. However, the scarcity of stress measurements usually permits estimation of resolved stresses acting on faults only with significant uncertainty (e.g. NRC, 2013).

In contrast, Townend and Zoback (2000) proposed that, in general, the ambient pore fluid pressure is near-hydrostatic throughout the brittle, upper crust of the Earth in the interiors of tectonic plates. In this case, pore pressures can be estimated relatively reliably just from the thickness of the overburden. Townend and Zoback (2000) used deep crustal permeability data over nine orders of magnitude acquired from six different regions to suggest that faults within the brittle crust are constantly in a state of critical stress; i.e., an incremental increase in shear stress or increase in pore pressure can lead to rupture. However, the difficulty in accurately estimating the shear and normal stress components often prevents precise determination of how near a particular fault is to failure. Exceptions to commonly assumed hydrostatic pressures occur in some deep basins, such as the Raton Basin in Colorado, where Nelson et al. (2013) showed using drillstem tests that deep formations are underpressured. If the crust within these basins is also critically stressed, then an increment in pore pressure less than that required to reach hydrostatic could bring favorably-oriented faults to failure.

5.3.3 Earthquake Measurements

5.3.3.1 Earthquake Recording and Analysis

Seismic waves radiated by earthquakes are recorded by networks of seismometers placed on the Earth surface or deployed in boreholes. Seismic recordings are used to analyze earthquake source parameters, including location in space and time, magnitude, source type and the direction and amount of fault slip, as well as to understand the properties of the rock layers along the propagation path between the earthquake and seismometer. Record fidelity is commonly referred to as “signal-to-noise,” the ratio of signal amplitude to background noise. Placing seismometers in boreholes greatly enhances signal-to-noise, often enabling recording of very small earthquakes (magnitude less than zero).

Earthquake detectability, the minimum magnitude that can be detected at a given location, depends upon the spacing of seismic recording stations within the region. Detectability is usually stated in terms of a threshold magnitude above which a particular earthquake catalog is considered complete. As shown in Figure 5-7, the present completeness threshold is less than magnitude 1 in large areas of California, and less than magnitude 2 over most of the state. This is significantly better than in most other regions of the US, where the completeness threshold provided by the USGS’s Advanced National Seismic System (ANSS) backbone monitoring array and regional networks is generally about magnitude

2.5 or greater (see the Figure on p.131 of NRC, 2013). Temporary arrays of seismometers are often installed at sites of particular interest to increase detectability and improve signal-to-noise in order to enable detailed analyses of the spatial and temporal distributions and mechanisms of microearthquakes (e.g., Frohlich et al., 2011).

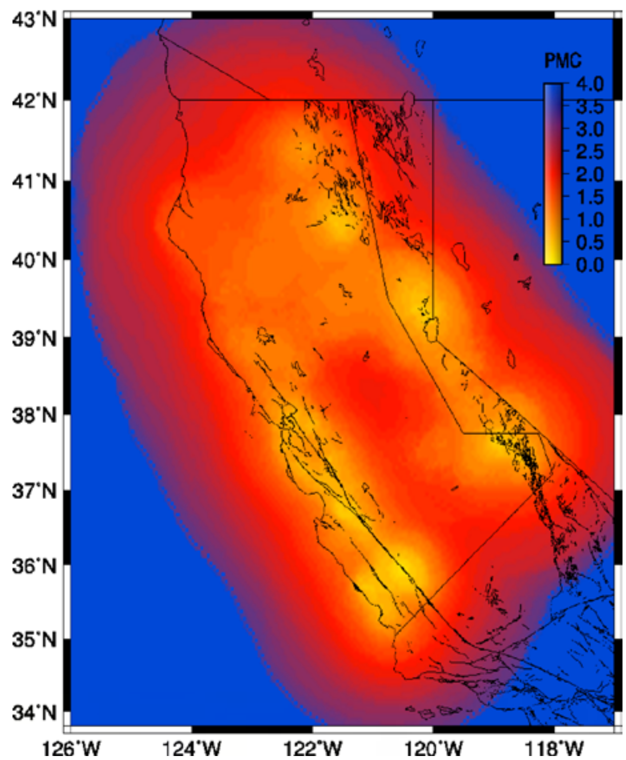

Figure 5-7. Earthquake detectability in California. The map shows the distribution of earthquake magnitudes that can be detected with 99% probability by the USGS ANSS network currently deployed in California (from Bachmann, 2011).

5.3.3.2 Earthquake Magnitude

The size of an earthquake is most commonly expressed as a magnitude, which is a measure of the amount of energy released by slip on the fault. In general terms, the magnitude depends on the size of the area on the fault that undergoes slip. Several magnitude scales are in common use, most of which (e.g. local magnitudes, M_L, and body-wave magnitudes, m_b) are defined based on trace amplitude or signal duration measured on recorded seismograms. However, the moment magnitude (M_w) scale is preferred by most seismologists because M_w is calculated from seismic moment (Hanks and Kanamori, 1979), a more fundamental measure of earthquake size (and energy) that is directly proportional to the product of slip and slipped area. To give an idea of how magnitude relates to slip area, M_w4.5 and M_w3.5 earthquakes rupture fault areas of about 2.5 and

0.2 km^2, respectively. In the remainder of this report we use moment magnitudes when they have been reported and published magnitudes otherwise. In cases when the scale is not specified magnitudes are denoted by "M".

5.3.4 Earthquakes Induced by Subsurface Fluid Injection

In this section, the two fluid injection activities associated with well stimulation that have been observed to induce earthquakes are discussed in terms of their spatial and temporal effects on the distributions of fluid pore pressures.

Fluids are injected into the subsurface for both hydraulic fracturing and wastewater disposal. If elevated pore pressures produced by either hydraulic fracturing or wastewater injection reach nearby faults or fractures, the resulting decrease in effective stress on the fault/fracture planes can lead to shear slip according to the Coulomb failure mechanism discussed in Section 5.3.2.2. Therefore, in both activities the aim should be to prevent the pressure perturbation from reaching larger faults capable of generating significant seismic events, both to minimize the seismic hazard and, in the case of stimulation, to prevent break out and subsequent leakage from the hydrocarbon reservoir. In general, induced seismicity related to well stimulation is dominated by pore-pressure perturbations, not changes in principal stress (NRC, 2013).

The probability of inducing seismic events is determined by the scale of the injection operation, the spatial extent of the affected subsurface volume, ambient stress conditions, and the presence of faults well-oriented for slip. The primary factors affecting the magnitude and extent of a pore-pressure perturbation include the rate and pressure of fluid injection, the total volume injected, and the hydraulic diffusivity (a measure of how fast a pore-pressure perturbation propagates in a saturated rock). At early stages the size of the pressure perturbation depends on the reservoir's hydraulic diffusivity and the duration of the injection, while the maximum pore pressure depends on the product of injection rate and duration divided by permeability (NRC, 2013). At the later stages of wastewater injection the induced pore-pressure field does not depend on the injection rate or permeability, but becomes proportional to the total volume of fluid injected.

Beginning with the earthquakes induced by fluid injection at the Rocky Mountain Arsenal in the 1960s (Healy et al., 1968), the cases of injection into deep disposal wells discussed below indicate that reactivation of basement faults is the predominant cause of lager magnitude induced earthquakes, including the largest events observed to date. This is because the higher stresses at basement depths and the brittle rheologies of crystalline basement rocks mean that favorably-aligned faults are more likely to reactivate under increased pore pressure. This can occur even when the faults lie below the injection interval as a result of hydraulic communication with the injection zone (Justinic et al., 2013). Although the matrix permeability of basement rock is generally very low, critically stressed faults and fractures in this part of the brittle crust can serve as high permeability channels (Townend and Zoback, 2000). This was shown to be the case during an enhanced geothermal system (EGS) stimulation, in which hydraulic shearing of basement rocks resulted in migration of microseismicity consistent with a basement hydraulic diffusivity equivalent to sandstone (Fehler et al., 1998; Shapiro et al., 2003).

5.3.4.1 Spatial and Temporal Characteristics

The volume of the subsurface affected by pore-pressure perturbations directly related to hydraulic fracturing treatments are usually largely confined within at most a few hundred meters of the injection interval, as evidenced by observed microseismicity. Davies et al. (2013) suggest possible fluid pathways that may explain how pore pressure reactivates faults in the vicinities of stimulation zones. Induced shear events are mainly caused by fluids "leaking off" into preexisting fractures intersected by the hydraulic fracture. Shear failure may also occur on nearby, favorably oriented fractures isolated from the pressure perturbation due to perturbation of the local stress field near the tip of the propagating hydraulic fracture (e.g. Rutledge and Phillips, 2003).

In contrast, wastewater disposal operations have been shown to generate overpressure fields of much larger extent. For example, at the Rock Mountain Arsenal, CO significant earthquakes caused by fluid injection occurred 10 km away from the well (Healy et al., 1968; Herrmann et al., 1981; Nicholson and Wesson, 1990). Hydrologic modeling of injection into the deep well at the site indicated that the seismicity front tracked a critical pressure surface of 3.2 MPa (Hsieh and Bredehoeft 1981).

The time delay between cessation of injection and the occurrence of larger (M>2) magnitude seismicity can be quite long. For hydraulic fracturing cases, the longest time delay observed so far is almost 24 hours at the Horn River Basin, BC site (BC Oil and Gas Commission, 2012). The 2011 M2.3 earthquake in Blackpool, UK, occurred about 10 hours after injection ceased at the Preese Hall 1 stimulation well (de Pater and Baisch, 2011). In wastewater disposal cases, much longer time delays are sometimes observed. For example, at the Rocky Mountain Arsenal an M_W4.3 earthquake occurred 15 years after the injection stopped (Herrmann et al., 1981).

These spatial and temporal observations are critical for understanding the causal relationships between injection activities and induced seismicity. Overall there is a lower potential seismic hazard from short-duration hydraulic fracture operations, because of the relatively small volumes of rock that experience elevated pressures, than from disposal of large volumes of wastewater into a single formation over time periods of months to years (NRC, 2013).

5.3.4.2 Maximum Magnitude

McGarr (2014) proposed estimating upper bounds on induced earthquake magnitudes based on net total injected fluid volume, observing that such a relationship is found to be valid for the largest induced earthquakes that have been attributed to fluid injection. Shapiro et al. (2011) proposed a similar approach to estimating maximum magnitude, based on the dimensions of the overpressurized zone deduced from observed microseismicity. Brodsky and Lajoie (2014) concluded that induced seismicity rates associated with the Salton Sea geothermal field correlate with net injected volume rate, which lends support to the proposed general dependence of induced seismicity on net injected volume. However, the approaches proposed by both McGarr (2014) and Shapiro et al. (2011) appear to imply that fault rupture induced by the injection occurs only within the volume of pore-pressure increase. While both are based on observations, the alternative, and perhaps more likely, hypothesis is that a rupture that initiates on a fault patch within the overpressured volume can continue to propagate beyond its boundaries, in which case the possible maximum magnitude is determined by the size of the entire fault. Indeed, McGarr (2014) does not regard that his relationship determines an absolute physical limit on event size.

5.3.5 Observations of Induced Seismicity Related to Well Stimulation

The vast majority of earthquakes induced by fluid injection in general do not exceed ~M1 (e.g. Davies et al., 2013; Ellsworth, 2013). However, larger magnitude earthquakes (M>2) have resulted from both wastewater injection and hydraulic fracturing. Table 5-8 summarizes observations of seismicity M>1.5 that have been reported and then investigated due to their correlation in space and time with wastewater injection or hydraulic fracturing activity. The table also includes observations of wastewater injection induced seismicity not related to well stimulation activities because the underlying physical mechanism of induced seismicity from wastewater injection is the same regardless of the source of wastewater; these observations are denoted with a single asterisk in the 'Proximate Activity' column. Where a series of earthquakes occurred, only the largest magnitude is reported.

After first summarizing criteria for classifying an event as induced, we discuss three cases of induced seismicity that resulted from hydraulic fracturing. Then we discuss four cases of seismicity generally accepted as being attributable to wastewater disposal as well as three cases in which the available evidence could not rule out a natural explanation. The seven additional examples of induced seismicity caused by fluid injection not related to well stimulation are listed in Table 5-8 for completeness, but are not discussed further here.

5.3.5.1 Criteria for Classifying an Earthquake as Induced

The following criteria proposed by Davis and Frohlich (1993) have been commonly used to determine whether an earthquake sequence was induced by fluid injection or occurred naturally:

- Are these events the first known earthquakes of this character in the region?
- Is there a clear correlation between injection and seismicity?
- Are epicenters near wells (within 5 km)?

- Do some earthquakes occur at or near injection depths?
- If not, are there known geologic structures that may channel flow to sites of earthquakes?
- Are changes in fluid pressure at well bottoms sufficient to encourage seismicity?
- Are changes in fluid pressure at hypocentral locations sufficient to encourage seismicity?

These criteria provide a basic foundation for establishing whether or not a given sequence has been induced, and have enabled a clear link between seismicity and injection operations to be established in some of the cases listed in Table 5-8. However, used alone, they have proven inadequate to establish conclusively that other sequences were induced. It is often very difficult to prove causality for the following reasons: (1) In some of the cases – including some of those for which the evidence from in-depth scientific study is generally regarded as being conclusive – there is no clear temporal and/or spatial correlation between injection and the occurrence of specific earthquakes, the largest events having occurred several years after the beginning (e.g. Prague OK) or cessation (e.g. Ashtabula OH) of injection, or up to ~10 km from the injection well (e.g. Rocky Mountain Arsenal and Paradox Valley in Colorado); (2) Often regional seismic network coverage is too sparse to locate the earthquakes with sufficient accuracy - particularly in depth - to investigate in detail their relationship to the injection well; (3) Even if detailed scientific studies are carried out, they are often hampered by lack of densely-sampled volume and pressure data and adequate site characterization. In particular, subsurface pressure measurements are rarely available; (4) While it is relatively straightforward to apply the first criterion to initially identify suspected cases in regions of low naturally-occurring seismicity such as the central and eastern US, discrimination is much more difficult in active tectonic regions like California, where the rate of naturally-occurring seismicity is much higher.

Table 5-8. Observations of seismicity (M>1.5) correlated with hydraulic fracturing and wastewater injection.

Site/Location	Country	Date	Magnitude	Proximate Activity	Induced?	Reference(s)
Rocky Mountain Arsenal, CO	USA	09 Aug 1967	4.85 M_w	Wastewater injection*	Induced	Healy et al., 1968; Herrmann et al., 1981
Matsushiro	Japan	25 Jan 1970	2.8	Wastewater injection*	Induced	Ohtake, 1974
Rangely, CO	USA	1962 – 1975	3.1 M_L	Wastewater injection*	Induced	Nicholson and Wesson, 1990

Site/Location	Country	Date	Magnitude	Proximate Activity	Induced?	Reference(s)
Love County, OK	USA	1977 – 1979	1.9	Hydraulic Fracturing	Unclear	Nicholson and Wesson, 1990
Perry, OH	USA	1983 – 1987	2.7	Wastewater injection*	Induced	Nicholson and Wesson, 1990
El Dorado, AR	USA	09 Dec 1989	3.0	Wastewater injection*	Induced	Cox, 1991
Paradox Valley, CO	USA	24 Jan	4.0 M_w	Wastewater injection*	Induced	Block et al., 2014
Ashtabula, OH	USA	26 Jan 2001	4.3 m_b	Wastewater injection*	Induced	Seeber et al., 2004
Dallas/Fort Worth, TX	USA	16 May 2009	3.3 m_b	Wastewater injection	Induced	Frohlich et al., 2011
Cleburne, TX	USA	09 Jun 2009	2.8 m_b	Wastewater injection	Unclear	Justinic et al., 2013
Garvin County, OK	USA	18 Jan 2011	2.9 M_L	Hydraulic fracturing	Induced	Holland, 2013
Guy-Greenbrier, AR	USA	27 Feb 2011	4.7	Wastewater injection	Induced	Horton, 2012
Blackpool	UK	01 Apr 2011	2.3 M_L	Hydraulic fracturing	Induced	de Pater and Baisch, 2011; Green et al., 2012
Prague, OK	USA	05 Nov 2011	5.7 M_w	Wastewater injection	Induced	Keranen et al., 2013
Youngstown, OH	USA	31 Dec 2011	3.9 M_w	Wastewater injection	Induced	Kim, 2013
Horn River Basin, BC	CAN	19 May 2011	3.6 M_L	Hydraulic fracturing	Induced	BC Oil and Gas Commission, 2012
Raton Basin, CO	USA	23 Aug 2011	5.3 M_w	Wastewater injection	Induced	(see McGarr, 2014)
Timpson, TX	USA	17 May 2012	4.8 M_w	Wastewater injection	Unclear	Frohlich et al., 2014

** Disposed fluids did not result from hydraulic fracturing*

5.3.5.2 Observations of Induced Seismicity Attributed to Hydraulic Fracturing

Several series of M>2 earthquakes have been linked to hydraulic fracturing treatments through detailed scientific investigation. These include the sequences on April 2011, M_L2.3 in Blackpool, UK (de Pater and Baisch, 2011; Green et al., 2012); January 2011, M_L2.9 in Garvin County, OK (Holland, 2013); and May 2011, M_L3.6 in the Horn River Basin, British Columbia (BC Oil and Gas Commission, 2012).

The M_L2.3 earthquake near Blackpool, UK, was attributed to hydraulic fracturing in the Preese Hall 1 well (de Pater and Baisch, 2011). A second M_L1.5 event occurred near the same well in May 2011. Both events occurred at about 2 km (1.2 mi) depth. These events are believed to have resulted from hydraulic connection out to distances further than anticipated, facilitated by bedding planes. Prior to August 2012, this event was the only documented observation of hydraulic fracturing-induced seismicity of magnitude greater than 1.

In January 2011, a sequence of earthquakes (maximum M_L2.8) occurred in close proximity to a hydraulic fracturing treatment operation in the Eola Field, Oklahoma. Initial reporting of the events (Holland, 2011) could not establish a conclusive link to well stimulation. Only after the operator released detailed production data, including underground pressure and injection rate, were the events clearly identified as having been induced (Holland, 2013). This clarification was made possible in part because the earthquake activity ceased during a two-day break in well stimulation due to bad weather and then began again when stimulation resumed.

The largest magnitude earthquakes observed to result from hydraulic fracturing (maximum M_L3.6) occurred in the Horn River Basin in British Columbia, Canada between April 2009 and December 2011 (BC Oil and Gas Commission, 2012). Twenty earthquakes in this series were above M_L3. Although the regional earthquake recording system is unable to detect $M<2$ earthquakes, all seismic events detected in the Horn River Basin occurred during or between hydraulic fracturing treatments. There are numerous north-south trending sub-parallel faults in the region. Average total fluid volume injected per well was 61,612 m^3 (16,276,000 gal) with an average injection rate of 18,720 m^3/day (4,945,000 gal/day).

Nicholson and Wesson (1990) reported on two earthquake series in Oklahoma that occurred in June 1978 and May 1979. The largest of these was M1.9. In each case, nearby hydraulic-fracturing operations correlated with the seismic events, but a lack of local seismic recording resulted in large location uncertainties and prevented a clear determination that the events were induced.

5.3.5.3 Observations of Induced Seismicity Attributed to Water Disposal

There are many cases in which disposal of wastewater related to hydraulic fracturing via Class II wells is the most likely explanation of seismicity. These include seismic events in Dallas-Fort Worth, TX, Guy, AR, Youngstown, OH, Prague, OH, and Raton Basin, CO. In other cases (Cleburne, TX; Timpson, TX), wastewater injection represents one possible explanation, but it was impossible to rule out that the earthquakes were of natural origin.

Texas, like many states east of the Rocky Mountains, had a low rate of natural seismicity before well stimulation began in the Barnett Shale. For example, there were no local felt earthquakes in Dallas-Fort Worth (DFW), TX between 1850 and 2008. Beginning in October 2008, seven weeks after Chesapeake Oil and Gas Company began injecting wastewater in a disposal well in the DFW area, felt earthquakes (m_b2.5 – 3.3) began to be reported by the public. This prompted researchers to deploy a local seismometer array in the area. The local array enabled a reduction in the location uncertainty of the 11 recorded earthquakes from ±10 km (6 mi) to ±200 m (0.125 mi) (Frohlich et al., 2010). These events were all located within 1 km of a northeast-trending normal fault, favorably oriented in the N40 – 47°E regional stress field, and 200 m north (on average) of a water disposal well. Brine-injection volumes for this well averaged 950 – 1,310 m^3/day (252,000 – 346,500 gal/day) during the period covered by the temporary array, which is a typical rate for disposal wells in this and neighboring counties. The depth of wastewater injection in this well (3.1 – 4.1 km; 10,100 – 14,400 ft) was ~1 km (3,300 ft) above the average depth of recorded seismicity (4.4 – 4.8 km; 14,400 – 15,700 ft). Felt seismicity (M>2) continues to occur in the DFW area more than two years after injection ceased in the disposal well.

The following month, in Cleburne, TX, about 50 km southwest of DFW, another series of earthquakes occurred (maximum magnitude m_b2.8) in another area of prior quiescence that contained nearby active water-disposal wells (Justinic et al., 2013). Continuous injection began two years prior to the onset of seismicity. The lack of fluid-pressure data barred detailed understanding of how seismicity correlated with injection, and ultimately prevented positive identification of this series as natural or induced.

On May 17, 2012, a third case of potentially induced seismicity in Texas occurred near Timpson (Frohlich et al., 2014). Epicenters of the earthquake series (maximum M_W4.8) lie along a mapped basement fault about 6 km long. Four active water disposal wells lie within about 3 km of the epicenters and near the largest magnitude event. Total injected volumes for the two largest volume wells were 1,050,000 m^3 and 2,900,000 m^3 (277 billion gallons and 766 billion gallons), with average injection rates exceeding 16,000 m^3/mo (420,000 gallons/mo). The injection interval for all four wells was 1.8 – 1.9 km (5,900 – 6,200 ft), and the top of the basement is at a depth of approximately 5 km (16,000 ft). The five largest earthquakes occurred between depths of 2.75 and 4.5 km (9,000 and 14,800 ft). Although the evidence favors the conclusion that these events were induced, Frohlich et al., (2014) could not rule out the possibility that they occurred naturally.

In central Arkansas, disposal of wastewater from hydraulic fracturing operations in the Fayetteville Shale has been correlated with 224 earthquakes of magnitude M>2.5 that occurred between 2007 and 2011. The largest event, M4.7, occurred on February 27, 2011 (Horton, 2012). In an area of generally diffuse seismicity, 98% of the recent earthquakes occurred within 6 km (3.7 mi.) of three Class II disposal wells. One injection well appears to intersect the Guy-Greenbrier fault within the basement, which is suitably oriented for slip within the regional tectonic stress field (Horton, 2012).

The largest earthquake suspected of being related to injection of wastewater from well stimulation was an M_W5.7 event that occurred within a region of previously sparse seismicity near Prague, OK on November 6, 2011 (Keranen et al., 2013; Sumy et al., 2014). This event is the second largest earthquake instrumentally recorded in the eastern US, and it destroyed 14 homes and injured two people. The hypocenter was located on the previously mapped NNE-SSW-striking Wilzetta fault system and was followed two days later by an M_w5.0 about 2 km to the west. Sumy et al. (2014) proposed that the M_w5.7 mainshock was triggered by an M_w5.0 foreshock that occurred the previous day approximately 2 km from two active wastewater injection wells located within the Wilzetta North oilfield. One well injected into the previously depleted Hunton Limestone reservoir, while the other injected into two deeper formations. The zone of well-located aftershocks of this event extends along the strike of the fault to within about 200 m of these wells. Although injection into the first well began in 1993, the cumulative rate of injection was increased by starting injection into the second, deeper well in December 2005, accompanied by a tenfold increase in wellhead pressure; pressures at both wells averaged approximately 3.5 MPa (508 psi) between 2006 and December 2010, falling to 1.8 MPa (261 psi) in 2011. Keranen et al. also note that local earthquake activity began with an M_w4.1 earthquake a few km from the 2011 mainshock in 2010, during the period of near-peak wellhead pressures, but they do not mention microseismicity before or after this event .

Keranen et al. (2013) concluded that the November 5, 2011 M_w5 event was likely induced by a progressive buildup of overpressure in the effectively sealed reservoir compartment and on its bounding faults (part of the Wilzetta fault system) after the original fluid volume capacity of the depleted reservoir had been exceeded as a result of injection. However, this explanation apparently does not take into account injection into the deeper formations, which are separated from the reservoir by a (presumably relatively low-permeability) shale layer. An alternative explanation might be that the triggering mechanism involved only the more recent injection into the deeper formations, the lowest of which directly overlays basement. McGarr (2014) proposed that the M_w5.7 mainshock was induced directly by injection of much larger volumes into three wells located 10 to 12 km southeast of the epicenter. However, if, as asserted by Keranen et al., the faults of the Wilzetta system form barriers to lateral (SE-NW) flow that compartmentalize the oilfield then it would not be expected that the wells discussed by McGarr would be in hydraulic communication with the westernmost fault of the system on which the earthquake apparently occurred. The occurrence of these events close to several high-volume injection wells strongly suggests that they were likely induced. However, the six-year delay between the significant increase in injection rate and pressure in the Wilzetta North wells and the conflicting hypotheses regarding the source and magnitude of the pressure perturbation mean that natural causes, as proposed by Keller and Holland (2013), cannot at present be ruled out.

During a 14-month period in Youngstown, OH, an area of relatively low historic seismicity, 167 earthquakes (M≤3.9) were recorded in proximity to ongoing wastewater injection (Kim, 2013). Earthquake depths were in the range 3.5–4.0 km and located along basement faults. Given that relatively small fluid volumes (~700 m^3; ~180,000 gallons) were injected prior to the onset of seismicity, there is believed to be a near-direct hydraulic connection to a pre-existing fault. Periods of high and low seismicity tracked maximum and minimum injection rates and pressures. The total injected volume over this period was 78,798 m^3 (20,816,000 gallons) with an average injection volume of 350 m^3/day (1,150 gallons/day) at a pressure of 17.2 MPa (2,490 psi).

Induced seismicity ($M_W \leq 5.3$) near Raton Basin, CO, is believed to have been caused by injection of 7.8 million m^3 (2.1 billion gallons) of wastewater near the southwestern extension of the local fault zone (Rubenstein, et al., 2014, submitted; manuscript referenced in McGarr, 2014). Since this study has not yet been published we are unable to report its conclusions.

5.3.6 Factors Affecting the Potential for Induced Seismicity in California

All of the US cases of induced seismicity related to fluid injection listed in Table 5-8 occurred within the stable continental interior, where tectonic deformation rates are low. California, on the other hand, is situated within an active tectonic plate margin, where the relatively rapid shear stressing rate on the numerous active faults result in much higher seismicity rates, as can be seen in Figure 5-9. If, as discussed in Section 5.3.4 and 5.3.2.3, the Earth's upper crust is generally in a critically-stressed state, then the high loading rates would imply that a relatively high proportion of faults in California will be close to failure at any given time, and hence susceptible to earthquakes triggered by small effective or shear stress perturbations. The abundance of faults large enough to generate M5 and greater earthquakes would suggest that there exists the potential for inducing earthquakes in California at least as large as those observed to date in the mid-continent, and also raises the question of whether earthquakes induced by stress perturbations at typical oil reservoir depths (<5 km) could trigger large (magnitude >6) tectonic earthquakes. However, whereas earthquakes in intraplate regions are often observed to nucleate within the upper few km, suggesting that the seismogenic crust in these regions extends to shallow depths (e.g. Adams et al., 1991; McGarr et al., 2002), evidence suggests that within active plate boundaries like California large earthquakes tend to nucleate at the base of the seismogenic crust at depths on the order of 10 km or greater (e.g. Mori and Abercrombie, 1997; Sibson, 1982). (One notable exception to this was the 1992 M_W7.3 Landers earthquake under the Mojave desert, which nucleated at a depth of 3-6 km.) Mori and Abercrombie (1997) (see also Scholz, 2002) proposed that the upper crust in active regions is more heterogeneous and the prevailing stresses are lower, so that earthquake ruptures that nucleate there are more likely to be arrested before they can grow into large events. Therefore, according to this argument, induced earthquakes in intraplate regions nucleate at or near the top of the seismogenic crust and are more likely to grow into larger events given a sufficiently large fault, but the magnitudes of earthquakes induced at reservoir depths in California are likely to be limited, perhaps below $M_W \sim 5.5$.

Assessment of the potential for induced seismicity, and hence the possible increase in seismic hazard, in California requires data on present and possible future WST activities, and the locations and characteristics of faults and in situ stresses in relation to those activities. Details of WST activities are described in Chapter 2. In the following sections, available fault and stress data are first summarized, followed by a discussion of recorded seismicity and its relation to current and likely future locations of injection activity.

5.3.6.1 California Faults and Tectonic Stress Field

Unlike the central and eastern US, a large number of active faults have been mapped and characterized in California. Figures 5-8 through 5-10 show the surface traces of active faults in California south of latitude 37° contained in the Uniform California Earthquake Rupture Forecast, Version 3 (UCERF3) Fault Model 3.1, prepared by the Working Group on California Earthquake Probabilities (Field et al., 2014). This database contains characterizations, including geometry and average slip rates, of faults known of believed to be active during the Quaternary (the last 2.6 million years). While particular attention should paid to these faults in assessing the potential for induced seismicity (and in siting WST activities), inactive local faults that are suitably oriented for slip in the prevailing in *situ* stress field need to be taken into account (see Section 5.3.6 and 5.34.2.3). The possible presence of unmapped faults, such as the basement faults activated in some of the recent cases of induced seismicity discussed above, also need to be considered. These may be detectable in seismic data acquired during exploration and reservoir characterization, or may be illuminated by microseismicity recorded during early stages of injection.

The most recent published stress data for California are contained in the World Stress Map catalog compiled by Heidbach et al. (2008). Figure 5-8 shows only the highest quality (quality A in the catalog) stress measurements for the southern part of California. These point measurements of the orientation of the tectonic stress field, and in some cases the magnitudes of principal stress components, are derived from observations of wellbore breakouts, earthquake focal mechanisms, tiltmeter monitoring of hydraulic fractures, and geological strain indicators

Although there are a large number of stress measurements in California compared with other regions of the US, the catalog provides only a sparse sampling of the stress field. While overall trends appear relatively uniform, for example a NW-SE maximum horizontal stress direction in the southern San Joaquin and Santa Maria Basins, significant variations are evident. This is to be expected because stress states at the local scale are complicated by heterogeneous distributions of fractures and fracture orientations and are influenced by changes in lithology and rock material properties (e.g. Finkbeiner et al., 1997). Ideally, stress measurements at a given injection site are needed to assess the potential for induced seismicity. To achieve this, it may be possible to employ other measurement techniques in addition to using borehole data. For example, in a hydraulic fracturing experiment in the Monterey formation, Shemeta et al., 1994 studied the geometry of the vertical fracture using continuously recorded microseismic data, regional stress information, and well logs. They found that the microseismic and well data were consistent with both the regional tectonic stress field and fracture orientations observed in core samples and microscanner and televiewer logs. The results of this study suggest that observations of the natural fracture system can be used as indicators for the orientations of induced fractures and hence of the in situ stress.

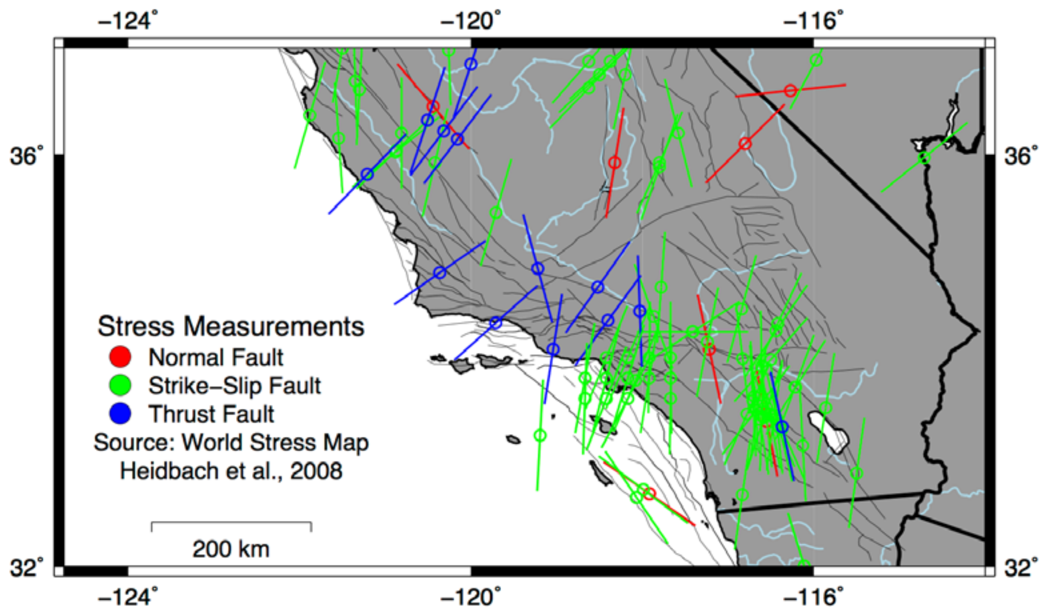

Figure 5-8. Highest quality stress measurements for California from the World Stress Map (Heidbach et al., 2008), plotted with mapped faults from UCERF3 FM 3.1 (Field et al., 2014). Stress measurements show orientation of the maximum horizontal compressive stress direction, color-coded according to stress regime.

5.3.6.2 Naturally-Occurring and Induced Seismicity in California

The generally low-magnitude detection threshold in California discussed in Section 5.3.3.1 means that Californian earthquake catalogs provide a relatively high-resolution picture of seismicity in the state as a whole. Figure 5-9 shows high-precision, relocated epicenters (Hauksson et al., 2012;) of southern California earthquakes recorded between 1981 and 2011, contained in the Southern California Earthquake Data Center catalog (SCEDC, 2013). Intense seismicity occurs along the major fault systems like the San Andreas and Eastern California Shear Zone, and includes relatively frequent (10s to 100s of years), large (M_w>6) earthquakes. Large events accompanied by aftershock sequences have also occurred during this 30-year time period along the western slopes of the Central Valley near Coalinga (1983), near Northridge north of Los Angeles (1994), and along the coast near San Simeon (2003). Elsewhere, lower-magnitude seismicity is generally more diffuse.

In addition to the Los Angeles basin, areas of the southern San Joaquin, Ventura, Santa Clarita and Santa Maria basins, where active water disposal wells are concentrated at present (Figure 5-10), have relatively high rates of seismicity in the 2-5 magnitude range. While undoubtedly most of these earthquakes are naturally-occurring, detailed study of the seismicity in relation to fluid injection will be needed to assess the likelihood that a proportion of the events in these areas are induced. There are numerous published studies of induced seismicity associated with production from geothermal fields in California (e.g. Eberhart-Phillips and Oppenheimer, 1984; Majer et al., 2007; Kaven et al., 2014; Brodsky and Lajoie, 2013). However, while microseismic monitoring is routinely used to monitor hydraulic fracturing operations (e.g. Murer et al., 2012), no systematic study to examine possible correlations of significant (M>2) seismicity with well stimulation or other fluid injection operations at oil and gas fields in California has yet been completed or published.

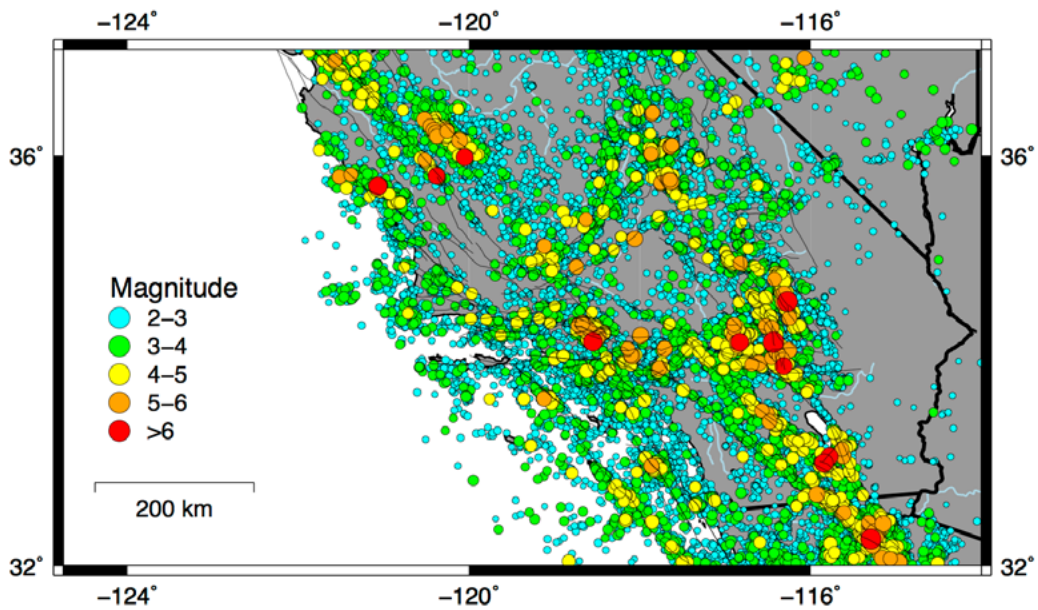

Figure 5-9. High-precision earthquake locations 1981-2011 from Hauksson et al. (2011). Faults as in Figure 5-8.

To our knowledge, in only one published paper (Kanamori and Hauksson, 1992) was a California earthquake greater than magnitude 2 linked to oilfield fluid injection. In that case, the authors attributed the occurrence of a very shallow M_L3.5 slow-slip event to hydraulic fracture injection at the Orcutt oilfield in the Santa Maria basin. This event was anomalous in that it radiated much lower energy at much lower dominant frequencies than normal earthquakes of similar size. One reason for the lack of progress on this front to date is that unlike stable plate interiors, where identification of anthropogenic seismicity is relatively easy, one of the major challenges in tectonically-active regions is the problem mentioned previously of discriminating between induced and naturally-occurring events (e.g. Brodsky and Lajoie, 2013). The University of Southern California Induced Seismicity Consortium is currently carrying out a study of spatial and temporal variations in seismicity statistics in relation to active oilfields in the southern San Joaquin basin. Preliminary results reported by Aminzadeh and Gobel (2013) suggest that systematic differences in earthquake frequency-magnitude distributions and other characteristics may be a promising tool for identifying induced seismicity.

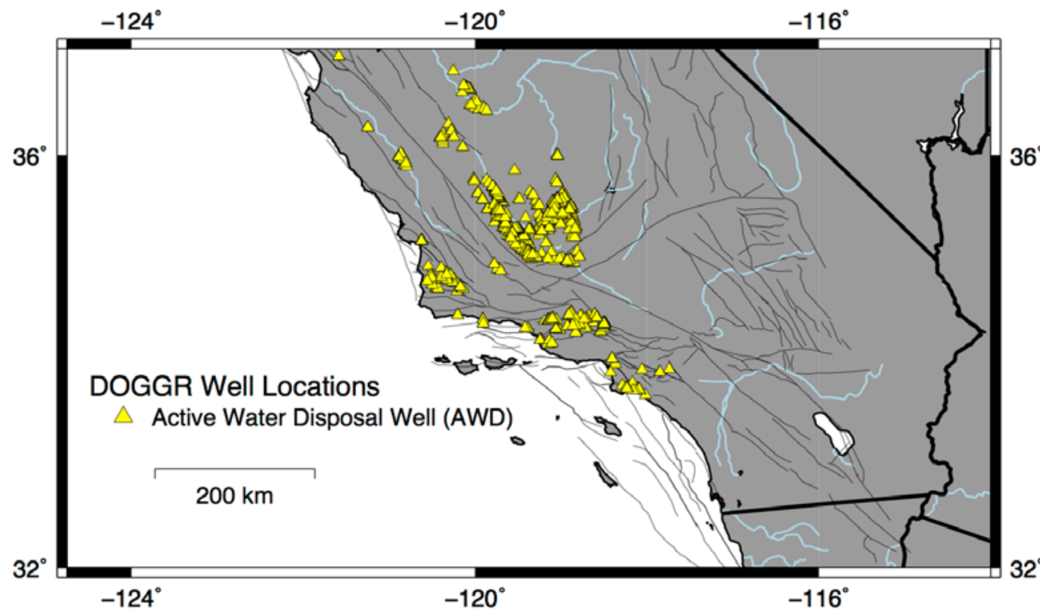

Figure 5-10. Locations of 1509 active water disposal wells from DOGGR (2014b). Faults as in Figure 5-8.

5.3.6.3 Wastewater Disposal Activity in California

With the exception of the San Joaquin Valley, presently active wastewater disposal wells shown in Figure 5-10 are in general situated within a few km of mapped Quaternary active surface faults. Wells along the western margin of the southern San Joaquin Valley are more than 10 km (~6 mi.) away from the San Andreas fault, but several of the southernmost wells are within a few km of the historically active (M_w7.3) White Wolf fault. The crystalline basement under the western margin is 10-12 km deep. The basement surface slopes upward to outcrop at the Sierra front, and in the vicinity of Bakersfield it is at a depth of about 2-3 km, much closer to reservoir depths. In this respect the setting towards the eastern Valley margin appears more similar to that in the midcontinent than in other oil-producing basins in California, although, as discussed previously, the shallow basement may not be capable of nucleating large (M>6) tectonic earthquakes. Within and on the margins of other currently producing basins the structure is generally much more complex, and basement depths are highly variable. Injection depths are available for roughly twenty percent (20%) of the ~1500 active water disposal wells in the DOGGR (2014b) database. Of these, 21 wells in their current configurations have the deepest injection interval at a depth greater than 1.8 km (6,000 ft) (DOGGR, 2014c).

Currently, the total disposal volume per well in California is generally less than in other regions where well stimulation is taking place. According to DOGGR (2010) (the most recent annual report available), total annual wastewater disposed in 2009 for Kern County was approximately 79.4 million m^3 (2.1 billion gal) into 611 active wells. This indicates an average disposal rate of about 360 m^3 (95,000 gallons) per well per day. This is one-fourth of the average 2008 water disposal rate per well of 1,430 m^3 (378,000 gallons) per day in Tarrant and Johnson Counties, Texas, where the Dallas-Fort Worth events occurred (Frohlich et al., 2010).

In-depth analyses are required to examine relationships, in any, of past and current wastewater disposal to seismicity and possible surface and basement fault sources. The results of the analyses will provide a foundation for assessing the potential for induced seismicity as a result of disposing of substantially larger volumes of wastewater, and perhaps also from carrying out hydraulic fracturing in the significantly deeper Monterey source formations. This assessment will form the basis for quantitative seismic hazard analyses at basin scale utilizing the approaches outlined below. In other areas in the US where stimulation-related induced seismicity has occurred, access to accurate, finely sampled (volume per day) injection rate data was a critical piece of information required to demonstrate a causal link.

5.3.7 Induced Seismic Hazard and Risk Assessment

Seismic hazard is defined as the annual probability that a specific level of ground shaking will occur at a particular location. Seismic risk is the probability of a consequence, such as deaths and injuries or a particular degree of building damage, resulting from the shaking. Risk, therefore, combines the hazard with the vulnerability of the population and built infrastructure to shaking, so that for the same hazard the risk is higher in densely populated areas. Seismic hazard maps are developed for California by the USGS and California Geological survey as part of the National Seismic Hazards Mapping Project (http://earthquake.usgs.gov/hazards/index.php). Of the areas in which water disposal wells are currently active (Figure 5-10), seismic hazard from naturally-occurring earthquakes is high in the Los Angeles and Ventura Basins and the Santa Clarita Valley, moderate in the Santa Maria Basin and moderate to high along the western and southern flanks of the southern San Joaquin Valley. The hazard decreases towards the center of the Valley and is relatively low in the Bakersfield area.

Rigorous assessment of the incremental hazard and risk from induced seismicity will be needed both for regulatory purposes and, in the worst-case scenario, for determining liability. In addition to the probability of damage and casualties dealt with in conventional seismic risk analysis, the risk of public nuisance from small, shallow events that occur relatively frequently has also to be considered. Approaches to assess induced seismicity risk can be developed by adapting standard probabilistic seismic hazard assessment (PSHA) and probabilistic seismic risk assessment (PSRA) methods, such as that used by the USGS and CGS. The standard methods cannot be applied directly, however, because (except on a few of the best-characterized fault segments of the San Andreas system) conventional PSHA is based only on mean long-term (100s to 1000s of years) earthquake occurrence rates; i.e. earthquake occurrence is assumed to be time-independent. Induced seismicity, on the other hand, is strongly time- and space-dependent because it is driven by the evolution of the pore pressure field, which must therefore be built in to the calculation of earthquake frequencies and spatial distributions. There is also the problem of discriminating induced from naturally-occurring events.

Developing a rigorous PSHA method for short- and long-term hazards from induced seismicity presents a significant challenge. In particular, no satisfactory method of calculating the hazard in the planning and regulatory phases of a project is available at the present time. This is largely because, whereas in conventional PSHA earthquake frequency-magnitude statistics for a given region are derived from the record of past earthquakes, obviously no such record can exist prior to injection. Using seismicity observed at an assumed “analog” site as a proxy (e.g. Cladouhos, 2012) would not appear to be a satisfactory approach because induced seismicity is in general highly dependent on site-specific subsurface structure and rock properties. Physics-based approaches to generate simulated catalogs of induced seismicity at a given site for prescribed sets of injection parameters are under development (e.g. Foxall et al., 2013). Such approaches rely on adequate characterization of the site geology, hydrogeology, stress and material properties, which are inevitably subject to significant uncertainties (see Chapter 4, Section 5.3 and Section 5.3.6). However, large uncertainties in input parameters are inherent in PSHA in general, and techniques for propagating them to provide rigorous estimates of the uncertainty in the final hazard have been developed.

There has been more progress in developing methods for short-term hazard forecasting based on automated, near real-time empirical analysis of microseismicity recorded by a locally-deployed seismic network once injection is underway (e.g. Bachmann et al., 2011; Mena et al., 2013; Shapiro et al., 2007). Continuously-updated hazard assessments can form the input to a real-time mitigation procedure (Bachmann et al., 2011; Mena et al., 2013), as outlined in the following section (5.3.7.1). Using two different time-dependent empirical models, Bachmann et al. (2011) and Mena et al. (2013) were able to obtain acceptable overall fits of forecast to observed seismicity rates induced by the 2006 EGS injection in Basel, Switzerland over time periods ranging from 6 hours to 2 weeks. However, the forecast occurrence probability of the largest event (M_L3.4), which occurred after well shut-in, was only 15%, and the probability of exceeding the maximum observed ground motion was calculated as 5%. The performance of the method could probably be improved by incorporating a more physically-based dependence on injection rate or pressure (C. Bachmann, personal communication, 2014).

5.3.7.1 Protocols for Evaluating and Reducing the Risk from Induced Seismicity

The issue of induced seismicity is not new or unique to the oil and gas industry. The geothermal industry has had projects not only delayed, but cancelled due to induced seismicity (Majer et al 2007). In 2004 the US DOE and the IEA started an effort to develop protocols and best practices to guide all stakeholders (operators, public, regulators, policy makers) to aid the geothermal industry to advance in a cost effective and safe manner. These protocols/best practices (Majer et al., 2009, 2012, Majer et al., 2014) were jointly developed by researchers, industry and geotechnical engineers. They were not intended to be a universally applicable approach to induced seismicity management, but rather a suggested methodology to observe, evaluate, understand and manage induced seismicity at a geothermal project. It is not a “one size fits all” approach, and stakeholders should tailor their actions to project-specific needs and circumstances.

The oil and gas industry outside of California, especially in the midcontinent, is now facing the same issues with induced seismicity that the geothermal industry faced in the early 2000’s, including public resistance, felt seismicity that is being attributed (rightly or wrongly) to oil and gas operations, and potential regulatory requirements. Therefore, based on the experiences in the geothermal industry, similar protocols and best practices are beginning to be developed by oil and gas companies (mainly in the midcontinent) to implement practices and tools for dealing with induced seismicity issues. Two examples of such protocols are the ones being developed by the Oklahoma Geological Survey and also by an industry consortium of companies in the American Exploration and Production Council (AXPC), a national trade association representing 34 of America’s independent natural gas and oil exploration companies (personal communication, Austin Holland, Oklahoma Geological Survey; Hal McCartney AXPC). Another example of a “protocol” that resembles the geothermal protocol is Zoback (2012), which describes similar steps and could also be used as a guide for oil and gas companies.

Most protocols are a “common sense” approach but guided by the best available science. They are not regulatory documents; consequently the protocols are intended to be living documents and evolve as needed. As new knowledge and experience is gained the protocols should be updated and refined to match “accepted” practices. In the geothermal and other protocols, there is series of recommended steps to address the hazard and risks associated with induced seismicity. Not all steps may be needed and the order of steps may vary. How the protocol is implemented will depend upon such factors as project location, past seismicity, community acceptance needs, current monitoring of seismicity, geologic conditions, past experiences with induced seismicity, and proximity to sensitive facilities. As an example, the geothermal protocol has the following steps for addressing induced seismicity issues as they relate to the whole project. All of the protocols have varying degrees of the following steps. (For details of the protocols, refer to the published editions referenced above.)

1. Perform a preliminary screening evaluation. (Does the project pass basic hazard criteria, i.e. proximity to known active faults, past induced seismicity, near population centers, amount of injection and time of injection, public acceptance issues etc.)

2. Implement an outreach and communication program. (Keep the community informed and educated on anticipated hazards and risk. An important step is gaining acceptance by non-industry stakeholders and promoting safety, the protocols outline the suggested steps a developer should follow to address induced seismicity issues)

3. Review and select criteria for ground vibration and noise. (Which communities, types of structures, etc. will be affected by any induced seismicity; this will inform criteria for setting maximum event sizes)

4. Establish seismic monitoring. (What has been the past seismicity in the area. Also allow data to be collected to develop an understanding of the origin (in space and in time) of any seismicity in the area and help determine if it is induced or natural)

5. Quantify the hazard from natural and induced seismic events. (How big an event is expected and what are the seismicity rates and magnitude distributions. For induced seismicity this may be difficult with a limited amount of geologic and site condition knowledge)

6. Characterize the risk of induced seismic events. (Given information in steps 3,4, and 5) perform a risk analysis. As discussed in Section 5.4.7 this is a challenging problem for induced seismicity, but at least bounds on risk should be estimated.)

7. Develop risk-based mitigation plan. (Such as a stop light procedure as described below, appropriate insurance coverage, etc.)

Figure 5-11 shows an example implementation strategy for the oil and gas induced seismicity protocol that the AXPC is considering. This step-wise approach will depend on specifics of the site and activity. This is a proposed draft that was shown at the KCC/KGS/KDHE Induced Seismicity State Task Force Meeting in Wichita, KS April 16, 2013.

The success of developing specific induced seismicity protocols for WST has yet to be evaluated in the midcontinent, let alone California. In terms of how such risk-reduction protocols may be defined and implemented for WST in California, one would expect a strong similarity to the response of the California geothermal industry. Many geothermal operators in the western US are successfully implementing either all or parts of the geothermal protocol. In addition, the BLM is using the geothermal protocol to develop its own criteria for geothermal permitting on BLM land in the U.S. as a whole.

Current real-time induced seismicity monitoring and mitigation strategies used by most enhanced geothermal system (EGS) operators employ a traffic light system (see Step 7 in the sample protocol above), originally developed for the Berlin geothermal project in El Salavador (Bommer et al., 2006). The traffic light system may incorporate up to four stages of response to seismicity as it occurs, and is generally based on some combination of maximum observed magnitude, measured peak ground velocity and public response.

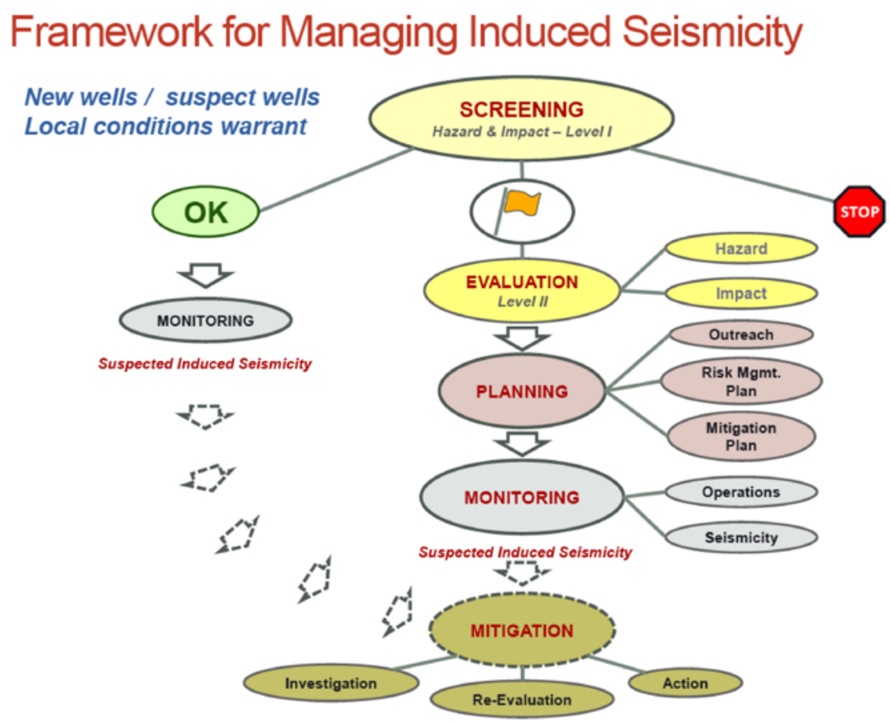

Figure 5-11. This example represents the collective thoughts of subject matter experts drawn from AXPC member companies and other Oil and Gas Industry companies. The subject matter experts include geologists, geophysicists, hydrologists, and regulatory specialists. This is a proposed draft that was shown at the KCC/KGS/KDHE Induced Seismicity State Task Force Meeting in Wichita, KS April 16, 2013. This presentation does not represent the views of any specific trade association or company.

Based on these criteria the injection will be either: 1) continued as planned (green); 2) continued but without increasing the rate (yellow); 3) stopped and pressure bleed-off initiated (orange); or 4) stopped with bleed-off to minimum wellhead pressure (red). Exact definition of these criteria is usually somewhat ad hoc and depends on the project scenario. The traffic light procedure implemented at the Basel EGS project was not successful in preventing the occurrence of the M_L3.4 earthquake on the same day that shut down the project, even though the orange stage was triggered after an M_L2.7 and the well eventually shut down. The traffic light system implemented at the St Gallen, Switzerland EGS project was also unsuccessful in preventing a strongly-felt earthquake that caused minor damage, but the circumstances in that case were highly unusual. The EGS community is currently beginning development of traffic light methods that employ near-real time hazard updating like that reported by Bachmann et al. (2011) and Mena et al. (2013). These will provide input for truly predictive, risk-based decision making based on the evolving seismicity and state of the reservoir.

5.3.8 Summary of Induced Seismicity Hazard Assessment

The severity of ground shaking generated by an earthquake depends on it's magnitude, the proximity of the surface site to the earthquake source, the geology along the seismic wave propagation path, and the local soil or rock conditions. For example, an m^3 earthquake that would likely not be felt if it occurred at a normal seismogenic depth for California would most likely be strongly felt if it occurred nearby at the relatively shallow depths at which most fluid injections take place.

The underlying general mechanism for how well stimulation activities induce seismic events is fairly well understood. However, applying this knowledge in a predictive sense to assess seismic hazard is difficult because of complexities in geology, subsurface fluid flow and well stimulation technology. Advances in coupled hydro-geomechanical modeling and simulation of fluid injection and hydraulic fracturing are beginning to explain how they affect fracture propagation and fault rupture (e.g. Rutqvist et al., 2013). Assessing the seismic hazard in a local area due to, for example fluid injection, requires knowledge of pre-existing faults, the state of stress on those faults, the evolving subsurface pressure field, and fault and rock properties, but many of these parameters will be known only with large uncertainties. However, seismic hazard assessment in general is invariably subject to considerable uncertainty, and an important and mature part of the analysis procedure is to properly characterize the uncertainties in the input parameter and then propagate them through the calculation to provide rigorous uncertainty bounds on the final hazard estimates.

To date, the maximum observed magnitude caused by hydraulic fracturing is M_L3.6 (BC Oil and Gas Commission, 2012). The largest earthquake suspected of being related to wastewater disposal is M_W5.7 (Keranen et al., 2013; Sumy et al., 2014), but the causal mechanism of this event is still the subject of active research and the possibility that it was a natural tectonic earthquake cannot, at present, confidently be ruled out. Overall, the likelihood of such an event occurring in the US as a whole is extremely low, given the current scale of well stimulation activities and the small handful of cases of significant induced seismicity experienced to date.

Hydraulic fracturing as it is carried out at the present time in California is not considered to pose a high seismic hazard. Apart from one highly anomalous event reported by Kanamori and Hauksson (1992), there have been no other published reports of felt seismicity related to either hydraulic fracturing or wastewater disposal in California. However, in many areas of California discriminating small induced events in the 2-4 magnitude range from frequently occurring natural events poses a significant challenge, and systematic studies have begun only recently. The duration and extent of the pressure disturbance from hydraulic fracturing in general are relatively small and, based on experience elsewhere, appear unlikely to generate larger felt or damaging events. The lack of reported felt seismicity for hydraulic fracturing in California is consistent with the relatively shallow injection depths (Section 5.2.3.2.1) and small injection volumes

(Section 3.2.3) currently employed in California operations. A shift to deeper stimulation, particularly if combined with increased injection volumes, could increase the seismic hazard to some degree.

The total volume of wastewater injected in California is much larger than the volume used for well stimulation. However, because present injection volumes are relatively small and injection intervals are shallow compared to other areas of the United States, the seismic hazard related to current wastewater injection is also likely to be relatively low. For example, California's disposed water volume per wastewater-injection well is about four times less and 1.5–2.7 km (4,700–7,700 ft) shallower than disposal into the wells in the vicinity of the Barnett Shale where induced events have occurred (Frohlich et al., 2010). However, further studies of the relationship, if any, between wastewater injection, seismicity and faulting in California will be need to establish this with confidence and to provide a better idea of incremental hazard levels due to induced seismicity.

The results from these studies can then be used as the initial basis for assessing the hazard that would result from increase well stimulation activity. WST applied at the scale presently employed in other regions of the US currently requires the disposal of much larger volumes of both flowback water from the stimulations themselves and produced water resulting from increased and expanded production, which could increase the hazard. Given the high rate of tectonic activity and the large number of active and potentially active faults in most of the areas that might be considered for unconventional recovery, it will be very important to carry out formal, probabilistic assessments of the potential incremental hazard and risk that could result from induced seismicity in those areas.

5.4 Other Potential Impacts

This section briefly addresses miscellaneous other possible impacts of WST.

5.4.1 Wildlife and Vegetation

While the impacts of oil and gas production on wildlife and vegetation are well-documented, the direct impacts of well stimulation are not. It is difficult to parse out direct and indirect impacts from the extant literature. This report outlines the most well-documented impacts of oil and gas production on wildlife and vegetation, examines the aspects that are most likely to be exacerbated by well stimulation, and discusses the native species in California most likely to be impacted by well stimulation.

5.4.1.1 General Effects of Oil and Gas Production on Wildlife and Vegetation

Oil and gas production has been shown to have numerous negative effects on wild animal and plant populations (for a review of unconventional oil and gas production impacts on wildlife impacts, see Northrup and Wittemyer, 2013). The footprint of well pads and support infrastructure such as upgrading facilities, roads, seismic lines, and power lines cause habitat loss (Jones and Pejchar 2013). Unpermitted activities can also cause habitat loss, as in one case in Pennsylvania when a company illegally constructed a wastewater pit in a wetland, important habitat for native wildlife (Department of Environmental Protection, Commonwealth of Pennsylvania). However, the area directly occupied by oil and gas production infrastructure is small compared to the area that is fragmented by the web of seismic lines, power lines, and roads that connect well pads (McDonald et al., 2009). Habitat fragmentation associated with oil and gas development impacts wildlife populations in a number of ways. It can reduce the size of home ranges for territorial animals and force them to travel longer distances to avoid interaction with human-built features (Webb et al., 2011a; 2011b), reduce patch sizes below what is needed by an animal that requires a large area for foraging (Linke et al., 2005), and act as barriers to dispersal (Dyer et al., 2002). Fragmentation also increases the proportion of disturbed edge habitat to interior habitat; some species are more vulnerable to predation or to be killed by humans along edge habitat (Moseley et al., 2009; Nielsen et al., 2006). In addition to habitat loss and fragmentation, noise pollution from oil and gas production have been shown to cause changes in the behavior of local wildlife that contribute to population declines (Bayne et al., 2008; Blickley et al., 2012; Francis et al., 2012). Vehicle collisions kill animals (Nielsen et al., 2006). Organisms sometimes die after drinking from or immersing themselves in wastewater (Ramirez 2010; Timoney and Ronconi 2010). Accidental spills of oil or wastewater can also cause mortality of plants and animals (Brody et al., 2012). In one case, an intentional application of wastewater following hydraulic fracturing caused tree mortality in an eastern forest (Adams 2011). The disturbances caused by oil and gas production promotes colonization by invasive species (Bergquist et al., 2007, Fiehler and Cypher 2011). Organisms that specialize in habitat near human disturbances are often invasive species that can inhabit a wide array of habitats, tolerate human disturbance, and displace native species (Coffin 2007; Belnap 2003; Jones et al., 2014).

All of the above impacts have been specifically documented in areas where well stimulation is commonly applied. However, to our knowledge, no study has attempted to parse out the direct impacts of well stimulation from the impacts of activities that precede and follow well stimulation. As a result, it is not possible to say what proportion of impacts on wildlife and vegetation are directly attributable to the process of well stimulation as opposed to the indirect impacts associated with all oil and gas production activities.

5.4.1.2 Potential Direct Effects of Well Stimulation on Wildlife and Vegetation

This section of the report focuses on wildlife and vegetation impacts of hydraulic fracturing, as it is the most commonly documented form of well stimulation in California. Hydraulic fracturing can affect wildlife and vegetation via direct and indirect pathways. Important direct factors are the possibility of increased toxicity of wastewater, water resource depletion, truck traffic, and noise. Indirectly, hydraulic fracturing can affect biota by increasing the intensity of oil and gas production in existing fields, or, probably to a lesser extent in California, by extending the range of oil and gas production into new areas. However, indirect impacts are beyond the scope of this report.

As mentioned above, the authors of this report could not find any studies that isolated the impacts of well stimulation from the effects of oil and gas production in general. Nonetheless, it is reasonable to infer that certain activities that are associated with well stimulation have the potential to exacerbate the known impacts of oil and gas production on wildlife and vegetation.

5.4.1.2.1 Wastewater Toxicity

One report found that nine chemicals used in hydraulic fracturing are regulated under the Safe Drinking Water Act for their risks to human health (United States House of Representatives Committee on Energy and Commerce 2011); however, the toxicity, concentrations of these chemicals in flowback and produced water, and the likelihood of releases to the environment are unknown (US EPA, 2012). Potential routes of environmental exposure to hydraulic fracturing chemicals include surface spills (discussed in Section 5.1.3.1.1 of this report) and wildlife drinking from or immersing themselves in surface storage ponds (Ramirez 2010; Timoney and Ronconi 2010). Bamberger and Oswald (2012) document a number of observations of harm to livestock, domestic animals, and wildlife that correlated with surface spills or intentional surface applications of wastewater from hydraulically fractured wells; however, these case studies were not controlled, replicated experiments, nor did they distinguish hydraulic fracturing flowback from produced water, so they cannot be taken as definitive evidence of direct harm from hydraulic fracturing operations. As detailed in Section 5.1.3.1.4 of this report, under certain circumstances wastewater can legally be disposed of via unlined sumps or discharged to a stream. As a result it is possible for flora and fauna to come in contact with flowback either in sumps or surface waterways. No studies were found that document whether these practices are either benign or harmful. In sum, it is established that oil and gas production yields wastewater that can at times be fatal to plants and animals, and it is possible under current regulations for wildlife and vegetation to come in contact with flowback in California. However, additional research is necessary to determine the extent to which wastewater toxicity is altered by the inclusion of hydraulic fracturing chemicals.

5.4.1.2.2 Water Depletion

Water use for hydraulic fracturing is discussed in detail in Section 5.1.1 of this report. While the quantity of water used for hydraulic fracturing is a small proportion of freshwater used in the state, it could be an important fraction of water in a given area, especially during periods of drought. If water is sourced from local water districts, it will come out of the overall regional or statewide allotment of water for agriculture, industrial and domestic use in the state. However, unlike water used in other applications, water injected for oil and gas production can effectively leave the water cycle if it is disposed of in a Class II well. Water for oil and gas production can also be sourced from local wells. To date, hydraulic fracturing notices state they are using fresh water from local water districts, with well water as a backup water source. Rapid withdrawal from aquifers can lower the water table, diminish stream recharge, and affect groundwater quality (US EPA, 2011).

5.4.1.2.3 Truck Traffic

Vehicles impact natural habitats by striking and killing animals (Fahrig and Rytwinski 2009), acting as vectors for invasive species (Ansong and Pickering 2013), and causing noise (Blickley et al., 2012; Forman and Deblinger 2014). Road mortality is noted as a major factor affecting the conservation status of three state and federally listed special status species in California: the San Joaquin kit fox, the blunt-nosed leopard lizard, and the California tiger salamander (Williams et al., 1998; Bolster 2010). The San Joaquin kit fox and blunt-nosed leopard lizard ranges overlap with oil fields in the San Joaquin Valley (Williams et al., 1998), while California tiger salamanders can be found in oil fields in Santa Barbara County (US Fish and Wildlife Service (FWS) 2000).

The proppant, and occasionally water, required for hydraulic fracturing is transported via trucks. Section 5.4.2 of this report discusses the amount of truck traffic associated with hydraulic fracturing in the state. However, there is insufficient data to quantify the impact to wildlife and plant populations caused by truck transport associated specifically with hydraulic fracturing.

5.4.1.2.4 Noise from Well Stimulation

As discussed in Section 5.4.2 of this report, there is only one reported measurement of noise during hydraulic fracturing in California. Noise levels of 68.9 and 68.4 decibels (dBA) were measured 1.8 m (5 ft) above the ground 33m (100 ft) and 66 m (200 ft) away from a high-volume hydraulic fracturing operation in the Inglewood Field (Cardno ENTRIX, 2012). These levels are substantially lower than those found to disturb wildlife and ecosystem processes in Blickley et al., 2012 and Francis et al., 2012, but difficult to compare to the noise levels measured in Bayne et al., 2008, which were noted as averaging 48 db(A) (SD 6) at an average distance of 242 m (SD 86). Regardless of the noise levels, well stimulation would increase the duration of noise generation at a well site. Unfortunately, there is insufficient data on typical noise levels associated with well stimulation in California and the behavioral responses of local species to reach any conclusions on how noise from well stimulation affects native fauna in the state.

5.4.1.3 Wildlife and Vegetation Most Likely to be Affected by Well Stimulation

A substantial number of native organisms, including endangered and threatened species, live on existing oil fields in California (Table 5-9), where they could be impacted by the direct effects of well stimulation.

It has been documented that species specializing in saltbush scrub habitat such as Le Conte’s thrashers, San Joaquin antelope squirrel, short-nosed kangaroo rats, and San Joaquin kit foxes occur in oil and gas fields with a low density of well pads and a corresponding low level of human disturbance (fewer than 50 wells in a 36 hectare area, and less than 70% of area disturbed) (Fiehler and Cypher 2011). In plots with higher levels of disturbance, none of these specialist species was found.

Table 5-9. List of special status species inhabiting oil fields in California. Key: CT = listed as threatened by the state of California, CE = listed as endangered by the state of California, FT = listed as threatened by the United States federal government, FE = listed as endangered by the United States federal government. The year the species was listed is given in parenthesis.

Species	**Status**	**Reference**
Bakersfield cactus *(Opuntia basilaris var. treleasei)*	CE (1990) FE (1990)	Williams et al., 1998
Blunt-nosed leopard lizard *(Gambelia silus)*	CE (1971) FE (1967)	Williams et al., 1998
California Condor *(Gymnogyps californianus)*	SE (1971) FE (1967)	US FWS 2005
California Tiger Salamander *(Ambystoma californiense)* 3 Distinct Vertebrate Population Segments: Central CA ………………….. Santa Barbara ……………… Sonoma County …………….	ST (2010) FT (2004 FE (2000) FE (2000)	US FWS 2000
Giant kangaroo rat *(Dipodomys ingens)*	CE (1980) FE (1987)	Williams et al., 1998
Kern mallow *(Eremalche kernensis)*	FE (1990)	Williams et al., 1998
San Joaquin antelope squirrel *(Ammospermophilus nelsoni)*	CT (1980)	Williams et al., 1998
San Joaquin kit fox *(Vulpes macrotis mutica)*	CT (1971) FE (1967)	Williams et al., 1998, Cypher et al., 2000, Fiehler and Cypher 2011
San Joaquin woollythreads *(Monolopia congdonii)*	FE (1990)	US FWS 2001

The California condor forages in the Sespe Oil Field in the Los Padres National Forest. Condors have died or been injured by landing on power poles and colliding with power lines (Mee et al., 2007a). Despite US Forest Service guidelines that well pads be maintained free of debris, oil operations are nonetheless potential sources of microtrash that can cause mortality in condors (Mee et al., 2007b). Power poles, power lines, and microtrash can increase with the intensity of oil production activities; however, they are not uniquely associated with well stimulation, but rather with oil and gas production activity as a whole.

5.4.2 Traffic and Noise

Well stimulation operations generate noise and lead to an increase in heavy truck traffic for transporting water, proppant, chemicals, and equipment. Well stimulation as practiced in California typically requires about a hundred to two hundred heavy truck trips per vertical well and two hundred to four hundred trips per horizontal well, counting two trips for each truck traveling to the site. This is one-third to three-quarters of the heavy truck traffic required for well pad construction and drilling. Noise generation during hydraulic fracturing could not be quantified because of the limited data and estimates, and disagreement between those that are available.

For vertical wells, NYDEC (2011) indicates light truck traffic increases about one-eighth due to hydraulic fracturing relative to that related to pad construction and drilling. Hydraulic fracturing in California is predominantly performed in vertical and near-vertical wells, as discussed in Section 3.2.3.

NYDEC (2011) assesses vehicle trips associated with gas-well development using hydraulic fracturing. Most of the heavy truck trips for hydraulic fracturing are for hauling water and proppant to the site and hauling flowback fluid away. The remaining trips are for equipment and chemical delivery. The analysis estimates one-fifth as many truck trips for produced water disposal as for water supply for a horizontal well, and one-half as many for a vertical well.

The typical tractor-trailer and bulk tanker has an 18.2 to 24.5 metric ton (20 to 27 short ton) capacity (Davis et al., 2013). Consequently, delivery of the average proppant mass of 100 metric tons for hydraulic fracturing in California discussed in Section 5.2.1.4.2 requires 4 to 6 truck trips (all trips are one-way in this discussion, so the values should be doubled to calculate trips to and from the site). Section 5.2.1.4.2 used an estimate of 662 m^3 (175,000 gallons) of water per hydraulic fracturing treatment per well in California. This has a mass of 662 metric tons (728 short tons) and so requires 27 to 36 truck trips.

Flowback water volume may be higher in California than in New York, due to use of hydraulic fracturing in migrated oil reservoirs rather than shale gas reservoirs, as discussed in Section 5.1.2. Assuming the volume of flowback fluid is the same as water used in the hydraulic fracturing results in a total of 58 to 78 heavy truck trips per vertical well that is not near pipelines for water delivery and flowback removal.

For wells in fields with available pipelines, NYDEC (2011) estimated heavy truck traffic would be reduced by 72% for water delivery and 38% for flowback fluid disposal. Applying these reductions to the California case with equal water supply and flowback fluid volumes results in a total of 28 to 38 heavy truck trips per vertical well near pipelines for water and proppant delivery and flowback removal.

Hydraulic fracturing also entails heavy truck trips to transport fluid storage tanks, chemicals, and other equipment, such as the pumps. NYDEC (2011) indicates the number of trips for fluid storage tanks and chemicals is the same as for water and proppant supply for a vertical well far from pipelines. NYDEC (2011) estimates five trips for other equipment, resulting in 36 to 46 trips for fluid tanks, chemicals and other equipment in the California case.

For the California case of a vertical well far from water supply and disposal pipelines, the total of all the heavy truck trips is 94 to 124. For the case of a vertical well near such pipelines, the total is 64 to 84 trips.

NYDEC (2011) estimates 171 to 164 heavy truck trips related to pad construction and drilling of a vertical well far and near pipelines, respectively. The difference is due to fewer trucks for pad construction in the near pipeline case.

NYDEC (2011) does not state the well depth for this estimate, but maps the depth and thickness of the Marcellus Shale, the main unconventional gas resource considered. The top of the Marcellus shale resource is between 1,830 to 2,130 m (6,000 and 7,000 ft) in depth with a thickness less than 100 m (330 ft). The median depth appears to be shallower than the midpoint of this range. Consequently the depth of unconventional gas wells in the Marcellus may be similar to hydraulically fractured wells in California currently, as represented in Section 5.1.3.1. This suggests the drilling rig size may be similar in California, and so the estimated heavy truck traffic related to drilling may be similar.

The analysis above indicates that, on average, hydraulic fracturing increases heavy truck traffic relative to that related to pad construction and drilling by about one-third to one-half for vertical wells near water supply and disposal infrastructure, and one-half to three-quarters for wells far from pipelines.

NYDEC (2011) indicates heavy truck traffic for hydraulic fracturing of horizontal wells would be larger, by up to 2.5 times, than heavy truck traffic for other activities. However, this study assumed 18,000 m^3 (5 million gallons) of water use. The average water volume used to fracture horizontal wells in California is smaller by an order of magnitude, and about two times the volume used above in the vertical well estimate. NYDEC (2011) indicates about twice the heavy truck trips related to pad construction and drilling for a horizontal versus a vertical well. This suggests that, on average, hydraulic fracturing of horizontal wells in California requires about twice the heavy truck trips as for vertical wells.

Only one set of measurements of noise at one site during hydraulic fracturing was identified. Noise levels of 68.9 and 68.4 decibels (dBA) were measured 1.8 m (5 ft) above the ground 33 m (100 ft) and 66 m (200 ft) away from a high-volume hydraulic fracturing operation in the Inglewood Field (Cardno ENTRIX, 2012). For comparison, this is nearly as loud as a typical home vacuum cleaner (Cardno ENTRIX, 2012). The measured noise level is substantially less than 85 to 90 dBA estimate at 76 m (250 ft) in NYDEC (2011). These levels are loud enough to potentially damage hearing. The reason for the difference in noise levels measured by ENTRIX (2012) and reported by NYDEC (2011) is not known. The Inglewood Field operates under an allowable noise limit set by local regulation due to the proximity of the surrounding urban land use (Cardno ENTRIX, 2012). This may have resulted in the deployment of noise mitigation measures that are not typical of other contexts. Consequently it is not clear how representative the identified measurements are with regard to hydraulic fracturing in other settings.

5.5 Conclusions

The main conclusions regarding the potential direct environmental effects from the use of well stimulation are given below. These are organized by technical subject matter. The relevant section numbers for each topic are provided.

Water Quantity and Sources (Section 5.1.1)

1. Water use for typical WST operations in California is much lower than for hydraulic fracturing in unconventional plays outside of California. Given the relatively low average volumes of water for each hydraulic fracturing event, the total water demand for hydraulic fracturing relative to total water supply or compared to other major water uses in the California economy is low in average water years, but can be sufficiently large locally in constrained years or specific watersheds to potentially have an impact.

2. Water use for hydraulic fracturing could substantially increase in California if operators switch from low-volume fracturing with gel to slickwater.

Water Quality: Injection-Fluid Composition and Toxicology (Sections 5.1.2.1 and 5.1.2.2)

3. A list of chemicals used for hydraulic fracturing in the United States and in California was developed from disclosures in FracFocus, but the list is incomplete, to an unknown degree, because of incomplete disclosure in that data source. For matrix acidization, a list of chemicals used was developed from stimulation notices, which did not indicate any undisclosed chemicals. Toxicological data were gathered from various sources for the chemicals on these lists, but such data were available for just a majority of chemicals for oral toxicity.

4. A number of stimulation-fluid constituents are known toxicants to mice and rats during single-component exposure, implying a general possibility of hazardous effects on humans. However, most of the chemicals applied in California for which toxicity information was available are considered to show low toxicity. In any case, the individual constituent concentrations in injection fluids, flowback and produced waters, as well as the specific exposure pathways, will ultimately determine effective doses and potential hazards.

5. In California, injection fluids applied to “general” hydraulic fracturing and matrix-acidizing jobs are different in their overall chemical composition, because of the unique technical needs for each type of application. However, a first, qualitative analysis of oral toxicity during single-component exposure in rats suggests that toxic effects of the chemicals used are fairly comparable between these fluids.

6. During this review, a series of data gaps have been identified in the literature regarding the potential toxicological impacts of fracturing/injection fluids. These include gaps in the following areas: (1) biological responses to acute exposure to many of the stimulation chemicals; (2) biological responses to chronic exposure to stimulation-fluid chemicals, such as carcinogens, endocrine disrupting compounds, and bioaccumulable materials; (3) eco-toxicological effects of fluid constituents on aquatic organisms; (4) overall toxicological effects of fluids as a mixture of compounds (compared to single-chemical exposure); and (5) potential time-dependent changes in toxicological impacts of fluid constituents, due to their potential degradation or transformation in the environment.

Water Quality: Flowback and Produced Water (Sections 5.2.2.3, 5.2.2.4, and 5.2.2.5)

7. Flowback and produced waters exhibit a range of compositions that depend on regional geology, fluids injected, and time at which samples were collected. In general, for oil-bearing shales such as the Monterey, flowback/produced waters would contain oil and gas, dissolved constituents from the formation (major cations/anions, trace elements, NORMs, organics), and potentially constituents of injection fluids and their reaction products.

8. The recovery of wastewaters from well stimulation varies widely, with values between 5 and 53% within the United States. In California, somewhat different recoveries compared to national averages may be expected, for the following two reasons: (1) targets in California differ from those in other states in terms of their local geology and rock types, with often higher permeability zones consisting of moderately brittle rocks; and (2) predominantly cross-linked gel is used for hydraulic fracturing in California as compared to a variety of fluids elsewhere.

9. Fracturing-fluid constituents typically evaluated for their residual concentrations in flowback or produced waters include friction reducers, surfactants, PCBs, biocides, alcohols, glycols and acids, such as acetic acid. Organic chemicals and biocides appear to be of particular concern.

10. While a detailed evaluation proves difficult, current literature suggests that the general composition of produced waters from well stimulation operations is similar to produced waters recovered during conventional oil and gas production.

Water Quality: Potential Impacts to Surface and Groundwater (Sections 5.1.3 and 5.1.4)

11. In California, flowback and produced waters from well stimulation are managed together. Current management practices in California allow for the disposal of oil and gas wastewater, including the co-mingled well stimulation fluids, into unlined pits in some areas and reuse for agriculture without prior treatment. A detailed assessment is needed to ascertain the wastewater disposal practices in the areas where well stimulation is occurring, to determine if they pose a risk to surface water and groundwater aquifers.

12. There are reports of surface water and groundwater contamination in regions where hydraulic fracturing activities are occurring in the United States. In California, there are documented cases of the intentional release of flowback fluids into unlined pits, as well as the accidental release of hazardous chemicals associated with well stimulation. Detailed assessments are not available as to whether these releases contaminated surface water and/or groundwater aquifers.

13. Potential contaminant of concern in flowback/produced waters include methane, TDS (salts), trace metals, NORMs, and some organics. However, at this time, it is not possible to evaluate whether this list of contaminants is relevant to California, since there is very limited information regarding the concentrations of these substances in flowback/produced waters from well stimulation operations in California. Some data may become available in 2014 as operators report the composition of waters recovered from well stimulation operations to DOGGR.

14. There are no recorded instances of subsurface release of hydraulic fracturing fluid into potable groundwater in California, but a lack of studies and consistent and transparent data collection and reporting makes it difficult to evaluate the extent to which this may have occurred. California needs to develop an accurate understanding about the location, depth, and quality of groundwater in oil and gas producing regions in order to evaluate the risks of WST operations to groundwater. This information on groundwater must be integrated with additional geophysical information to map the actual extent of hydraulic fractures to assess whether and where water contamination from WST activities have been or will be a problem.

15. Geomechanical modeling studies conducted for high-volume fracturing operations in the Barnett Shale have indicated that fracturing directly from shale formations into groundwater is unlikely for formations more than 600 m (1,970 ft) below the base of groundwater, but fracture connections to pre-existing permeable pathways (e.g., abandoned or degraded wells) have been discussed as possible migration mechanisms.

16. Most hydraulic fracturing occurs at a depth of less than 610 m (2,000 ft) in California. Much of this occurs in areas with poor groundwater quality, but tens of hydraulic fracturing operations per year in this depth range may occur in areas with higher quality groundwater. Data are not available to assess if changes to groundwater quality as a result of shallow hydraulic fracturing have occurred.

17. In general, monitoring efforts near well stimulation operations in the United States have not been extensive, and data on concentrations of potential contaminants in groundwater are sparse and not easily available to the public. A lack of baseline data on groundwater quality is a major impediment in identifying or clearly assessing the key water-related risks associated with hydraulic fracturing and other WST.

18. The few studies that have monitored groundwater near well stimulation operations in the United States have so far not observed significant impacts to water quality. Elevated levels of constituents detected in a few studies could not be definitively linked to the fracturing operations. For example, even though high levels of methane and TDS have been detected in groundwater near hydraulic fracturing operations outside of California, it has not been demonstrated whether this is solely due to hydraulic fracturing, natural processes, pre-existing pathways, or a combination of mechanisms. Further research is needed to understand the mechanisms by which contamination could occur.

19. Existing wells are often considered as the highest concern for subsurface migration of WST and subsurface fluids (including injection fluids, flowback/produced waters, formation brines and gas present in the target or other subsurface formations). Understanding this potential hazard is critical for the protection of groundwater resources. In particular, the locations and condition of preexisting wells near WST operations in California should be determined to assess potential hazards. Continued monitoring and data collection are warranted to avoid potential risks.

20. Proper well construction is critical for the protection of groundwater resources, and for preventing subsurface release or migration of reservoir or fracturing fluids. Well construction standards should be enforced for WST operations in California.

Air Quality (Section 5.2.1)

21. Estimated marginal emissions of NO_x, $PM_{2.5}$, VOCs directly from activities directly related to WST appear small compared to oil and gas production emissions in total in the San Joaquin Valley where the vast majority of hydraulic fracturing takes place. However, the San Joaquin Valley is often out of compliance with respect to air quality standards and as a result, possible emission reductions remain relevant.

22. Three major sources of air pollutants include the use of diesel engines, flaring of gas and the volatilization of flowback water. The first, diesel engines (used for transport and pumping of estimated fluid volumes required for WST) emit a small portion of total emissions nitrogen oxides (NO_x), particulate matter ($PM_{2.5}$), and volatile organic compounds (VOC) associated with other oil and gas production operations as a whole.

23. Emissions from flaring in California are uncertain because of variability in flare combustion conditions and to a lack of information regarding the frequency of flare-use during WST operations. However, current California Air Resource Board inventories of pollutant emissions from all flaring suggest that flares as a whole emit less than 0.1% of the VOCs and are not a major regional air quality hazard.

24. Emissions from volatilization of flowback water constituents have not been measured but might be bracketed. The California Air Resource Board has conducted a “bottom-up” VOC emission inventory by adding up all known sources of emissions. It is unknown whether these sources included emissions from WST-related produced or flowback water. However, the sum of the emissions in the inventory matches well with “top-down” measurements taken from the air in the San Joaquin Valley (Gentner et al., 2014). This agreement between “bottom-up” and “top-down” estimates of VOC emissions from oil and gas production indicates California’s inventory probably included all major sources.

25. The inventory indicates that VOC emissions from oil and gas evaporative sources, such from flowback water, might occur from stimulation fluids produced back after the application of WST, are small compared to other emission sources in the oil and gas development process. Data suggest that emissions from oil and gas production and upstream processing in general contribute to ~10% of anthropogenic VOC ozone precursor emissions in the San Joaquin Valley.

26. Some of the potential air-quality impacts can be addressed by regulation and largely avoided. Emissions from diesel equipment and diesel trucks can be controlled through use of the cleanest engines, such as US EPA classified tier 4 engines for off-road equipment or on-road truck engines that meet 2010 engine standards. Requiring reduced emission completions can control emissions from flaring and venting related to WST. Emissions from evaporative sources related to WST could be limited by requiring vapor controls on the temporary tanks to which flowback water is stored.

27. If practices in California were to change, for example if more fluid was used in WST or production was moved to remote locations, emissions from activities directly related to WST could become important if left uncontrolled.

Climate Impacts (Section 5.2.2)

28. Fugitive methane emissions from the direct application of WST to oil wells are likely to be small compared to the total greenhouse gas emissions from oil and gas production in California. This is because current California oil and gas operations are energy intensive. However, all greenhouse gas emissions are relevant under California’s climate laws and many emissions sources can be addressed successfully with best available control technology and good practice.

29. Fugitive methane emissions for oil and gas production are uncertain and are currently an active area of scientific research. A number of measurement studies in California suggest higher methane emissions from oil and gas production activities than is listed in the State inventory. However, even if accepting the higher rate of emissions indicated by the measurement studies, methane emissions

from oil and gas production are still likely to be small compared to direct CO_2 emissions associated with oil and gas production. Additionally, methane emissions directly related to WST are likely to account for only a small portion of total production related methane emissions.

30. Methane emissions related to WST can be addressed successfully with best controls, such as requiring reduced emission, or "green," completions and requiring vapor controls on temporary tanks in which flowback water is stored. We note that while green completions will be required nationally for gas wells starting in 2015, they will not be required for wells that produce oil or oil and gas, such as most of the wells in the San Joaquin Valley.

31. Other emissions such as CO_2 from diesel fuel used for pumping fluid or delivering supplies was found to be negligible.

32. While other regions are currently using WST for the production of petroleum (e.g., the Bakken formation of North Dakota) or gas (e.g., the Barnett shale of Texas), emissions from these regions may not be representative of emissions from California-specific application of WST. For example, the volume of fluid used for WST operations in California is typically lower than operations in other shale plays, potentially leading to lower evaporative emissions of methane from flow back fluid.

Induced Seismicity (Section 5.3)

33. The general underlying mechanism for inducing seismic events as a result of well stimulation technologies is well established (i.e., reduction in effective stress due to increased pore pressure) (NRC, 2013).

34. Hydraulic fracturing does not pose a high seismic hazard in California. The duration and extent of pressure increases from hydraulic fracturing is believed to be relatively small. In California in particular, most hydraulic fracturing is shallow and uses a small injection volume. A shift to deeper stimulation, particularly combined with a shift to larger volumes, would increase the hazard. Protocols and best practices developed for other water/wastewater injection activities to limit induced seismicity should be followed.

35. At present, the seismic hazard due to the disposal by injection of flowback water is relatively low in California. While the total volume of wastewater injected is about two orders of magnitude larger than the total volume used for stimulation in WST operation typical for California, the total wastewater injection volumes are generally smaller than in other parts of the country. In addition, injection is relatively shallow, and injection rates are relatively small. Disposal of produced water from oil and gas production in deep injection wells has caused felt seismic

events in several states. However, to date, no felt seismic events have been observed in this state as a result of produced water disposal from oil and gas production. If future WST practices in California result in expanded oil and gas production, the seismic hazard due to produced water disposal could increase, in particular when injecting larger volumes into deeper formations, and in areas of higher seismic risk.

Wildlife and Vegetation (Section 5.4.1)

36. No studies that specifically evaluated impacts of well stimulation on wildlife and vegetation were identified. One reason for this lack of information is that well stimulation occurs alongside other oil and gas production activities, and so its direct effects cannot be readily separated in the field from the overall effects of oil and gas production.

37. Co-management of flowback and produced water creates the possibility that wildlife could be exposed to stimulation-fluid constituents in waters discharged into pits for disposal or used for irrigation.

38. Increased truck traffic related to well stimulation can increase wildlife road mortality. For instance, road mortality is a major factor affecting the San Joaquin kit fox, blunt-nosed leopard lizard, and the California tiger salamander, whose ranges overlap with oil fields.

Traffic and Noise (Section 5.4.2)

39. Well stimulation as practiced in California typically requires about a hundred to two hundred heavy truck trips per vertical well and two hundred to four hundred trips per horizontal well, counting two trips for each truck traveling to the site. This is one-third to three-quarters of the heavy truck traffic required for well pad construction and drilling.

40. Noise generation during hydraulic fracturing could not be quantified because of the limited data and estimates, and disagreement between those that are available.

Indirect Impacts of WST-enabled Increases and Expansion in Production (Entire Section)

41. The primary impacts of WST on California's environment will be indirect impacts due to WST-enabled expansion in the footprint of oil and gas production by way of increased intensity of production in established fields, and potentially by expansion of oil and gas production into new areas. Impacts of WST-enabled production will vary depending on whether expanded production occurs in existing rural or urban fields or in green fields, as well as on the nature of the ecosystems, wildlife, geology, and groundwater in the vicinity.

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